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Ola Stedje Hanserud

Phosphorus Management in an Environmental Systems Perspective

The role of secondary phosphorus recycling in the case of Norway

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Thesis for the degree of Philosophiae Doctor

Trondheim, February 2018

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Abstract

Crop production depends on fertilizer inputs, including phosphorus (P), to maintain soil fertility over time. The P source of mineral fertilizers is mined phosphate rock, a non-renewable resource that has been applied in excess to agricultural soils in Western Europe for decades. A more sustainable food system is one that uses P more efficiently to reduce the extraction of phosphate rock and to reduce the risk of P losses associated with the accumulation of P in the soil.

The overall aim of this thesis was to contribute to improved P management and P use efficiency in Norwegian crop production by increasing our understanding of the potential for secondary P recycling. The first step was therefore to map flows and stocks of P in and between economic sectors associated with the Norwegian food system. Substance flow analysis (SFA) was employed in this work, including the integration of P plant-availability with SFA to obtain a more realistic picture of the total fertilizer value of organic residues. Furthermore, the total P fertilizer requirement on a national and regional (county) scale was adjusted according to soil P levels. This was done to provide a more correct picture of the theoretical potential of secondary P to cover P fertilizer requirements and replace mineral P in the short term.

The results showed that there is substantial P consumption in agriculture, of the same order of magnitude as the throughput of P in fisheries and aquaculture. At 10.2 kilotonnes P per year, the losses of P from fisheries and aquaculture are also comparable to the net stock soil accumulation, which is 12 kilotonnes P per year. Furthermore, secondary plant-available P has the theoretical potential to satisfy national P fertilizer requirements in both the short and long term. This can, in fact, be achieved by animal manure alone. It was demonstrated that manure P is unequally distributed among Norwegian counties, with livestock-dense counties in the south-west and west of Norway typically displaying great surpluses of manure P after covering internal P fertilizer requirements. In contrast, in counties in the south-east arable crop production dominates and they would have a P deficit without P fertilizer imports. The full potential of manure to replace mineral fertilizer can therefore only be realized if manure P is redistributed to where it is needed.

These findings were followed up by a life cycle assessment (LCA) study that looked at the environmental impacts of redistributing dairy cow manure over 500 km from a county with a P surplus to a county with a net P requirement. We compared several technology options and concluded that the most promising option was pretreatment by anaerobic digestion, followed by solid-liquid separation of the digestate using a decanter centrifuge. This alternative redistributed 71% of P in the cattle manure and did not increase potential environmental impacts compared to conventional cattle manure management.

Secondary P fertilizer has the potential to replace mineral P fertilizer, but the amount of avoided mineral P calculated in LCA depends on the assumptions made in the calculations, which differ between studies. In the last paper, I identify three substitution principles used in the LCA literature and, through a case study, show that they can greatly affect the inventory of avoided mineral P and the final environmental impact results.

In conclusion, this work has shown that organic residues in Norway have a great potential to meet P fertilizer requirements, and that the P redistribution that is necessary to realize this potential does not have to result in increased environmental impacts.

Sammendrag

Planteproduksjon avhenger av tilførsel av næringsstoffer i gjødsel, blant annet fosfor (P), for å vedlikeholde næringsstatusen i jord over tid. Kilden til P i mineralgjødsel er utvunnet fosfatstein, en ikke-fornybar global ressurs som har blitt tilført i overskudd til landbruksjord i Vest-Europa gjennom årtier. I en mer bærekraftig matproduksjon må P brukes mer effektivt for å redusere utvinningen av fosfatstein og redusere risikoen for tap av P som er forbundet med akkumulering av P i jord.

Det overordnede målet med dette doktorgradsarbeidet er å bidra til forbedret P-forvaltning og mer effektiv bruk av P i norsk planteproduksjon for mat og fôr gjennom økt forståelse av potensialet for resirkulering av P. Det første steget i arbeidet var derfor å kartlegge beholdninger og strømmer av P i og mellom de sektorene som kan assosieres med det norske matsystemet. Materialstrømsanalyse (SFA) ble brukt, inkludert en integrering av plantetilgjengelighet av P i ulikt organisk avfall for å få et mer realistisk bilde av gjødselverdien. Videre ble det totale P-gjødselbehovet på nasjonal- og fylkesnivå justert med hensyn på P-nivået i jord. Dette for å få et mer korrekt bilde av det teoretiske potensialet for hvor mye resirkulert (sekundær) P kan dekke av gjødselbehovet og erstatte mineralsk (primær) P på kort sikt.

Resultatene viste at det er et betydelig konsum av P i landbruket, i samme størrelsesorden som omsetningen av P i fiskeri- og akvakultursektoren. Tapene av P fra fiskeri og akvakultur på 10.2 kilotonn P per år er også sammenlignbare med netto akkumulering av P i jord på 12 kilotonn P per år. Videre fant vi at resirkulert plantetilgjengelig P har et teoretisk potensiale til å dekke det nasjonale P-gjødselbehovet på både kort og lang sikt og kan dekkes av P i husdyrgjødsel alene. Fosfor i husdyrgjødsel er imidlertid ujevnt fordelt mellom norske fylker. Typisk har fylkene på Vest- og Sørvestlandet med stor husdyrtetthet også store overskudd av P etter å ha dekket sitt interne P-gjødselbehov, i motsetning til fylkene i Sørøst-Norge, som er dominert av kornproduksjon, og som har et P-gjødselunderskudd uten import av P-gjødsel. Husdyrgjødselens fulle potensiale for å erstatte mineralgjødsel kan dermed bare realiseres om P i husdyrgjødsel omfordeles til der det trenges.

Disse funnene ble fulgt opp av en livsløpsanalyse (LCA) hvor vi så på miljøeffektene av å omfordele storfegjødsel over 500 km fra et fylke med et P-overskudd til et fylke med et netto P-gjødselbehov. Vi sammenlignet ulike alternative teknologier og konkluderte med at den mest lovende løsningen var biogassprosessering etterfulgt av mekanisk separasjon av bioresten med dekantersentrifuge. Dette alternativet omfordelte 71% av P i storfegjødselen og økte ikke de potensielle miljøpåvirkningene sammenlignet med konvensjonell storfegjødselhåndtering.

Sekundær P-gjødsel fra ulikt organisk avfall kan potensielt erstatte mineralsk P-gjødsel, men estimert mengde unngått mineralsk P i LCA er avhengig av antagelsene som blir gjort for utregningen, noe som kan variere mellom studier. I siste artikkel identifiserer jeg tre ulike substitusjonsprinsipper brukt i LCA-litteraturen og viser, gjennom et casestudie, at valget av substitusjonsprinsipp har en betydelig påvirkning på estimert mengde erstattet mineralsk P og de endelige miljøpåvirkningsresultatene.

Som en konklusjon viser dette doktorgradsarbeidet at det ligger et stort potensiale i organisk avfall i Norge for å dekke det nasjonale P-gjødselbehovet, og at nødvendig omfordeling av sekundær P for å realisere dette potensialet ikke trenger å skje på bekostning av store miljøpåvirkninger.

Acknowledgements

First of all, I would like to thank the Norwegian Institute of Bioeconomy Research (NIBIO), for giving me the opportunity to embark on this PhD project, which has been funded as part of the internal research project "Opportunities for sustainable use of phosphorus in food production".

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A bunch of wonderful colleagues at NIBIO that, on a daily basis, provide much needed mental time-outs with discussions on pretty much any more or less meaningful topic, and for being great sources of soil, crop and nutrient related knowledge.

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Last, but not least, I want to thank friends and family, my parents in particular. Their support has meant a lot. Tusen takk.

Ås, September 2017

Ola Stedje Hanserud

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Abbreviations

AD	anaerobic digestion
CC	climate change
FD	fossil resource depletion
ha	hectare
K	potassium
kt	kilotonne = 1000 metric tonnes
LCA	life cycle assessment
ME	marine eutrophication
MFE	mineral fertilizer equivalent
N	nitrogen
P	phosphorus
PMF	particulate matter formation
SFA	substance flow analysis
TA	terrestrial acidification
yr	year

List of publications included in the thesis

Paper I	Hamilton, H. A., E. Brod, **O. S. Hanserud**, E. O. Gracey, M. I. Vestrum, A. Bøen, F. S. Steinhoff, D. B. Müller and H. Brattebø (2016). "Investigating Cross-Sectoral Synergies through Integrated Aquaculture, Fisheries, and Agriculture Phosphorus Assessments: A Case Study of Norway." Journal of Industrial Ecology 20(4): 867-881.
Paper II	**Hanserud, O. S.**, E. Brod, A. F. Øgaard, D. Mueller and H. Brattebø (2016). "A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway." Nutr. Cycl. Agroecosyst. 104(3): 307-320.
Paper III	Hamilton, H. A., E. Brod, **O. Hanserud**, D. B. Müller, H. Brattebø and T. K. Haraldsen (2017). "Recycling potential of secondary phosphorus resources as assessed by integrating substance flow analysis and plant-availability." Science of the Total Environment 575: 1546-1555.
Paper IV	**Hanserud, O. S.**, K.-A. Lyng, J. W. De Vries, A. F. Øgaard and H. Brattebø (2017). "Redistributing phosphorus in animal manure from a livestock-intensive region to an arable region: Exploration of environmental consequences." Sustainability 9(4): 595.
Paper V	**Hanserud, O. S.**, F. Cherubini, A. F. Øgaard, D. B. Mueller and H. Brattebø (2018). "Choice of mineral fertilizer substitution principle strongly influences LCA environmental benefits of nutrient cycling in the agri-food system." Science of the Total Environment 615: 219-227.

The author's contributions

All the papers are co-authored. I am the first author of Papers II, IV and V and the main contributor to these studies.

For Papers II and V, the goal and scope, as well as the methodological concepts, were decided in close dialogue with the co-authors. The detailed model description, data collection, and analysis were carried out by me. I was responsible for the presentation of the results and writing of the papers, while the co-authors contributed through discussions, suggestions, and critical reviews of the manuscripts.

For Paper IV, the research was designed by Dr. Jerke W. De Vries, Prof. Helge Brattebø and myself. The development of the software model was primarily carried out by Kari-Anne Lyng, while I collected the data for the case study and carried out estimations of the inventory feeding into the software model. I was responsible for the presentation of the results and writing of the paper, while the co-authors contributed through discussions, suggestions, and critical reviews of the manuscripts.

I am the third author of Papers I and III, where I was involved in the research design and data collection and gave my input to the analysis and writing through discussions, suggestions, and critical review.

1 Introduction

1.1 Phosphorus as a resource

Crop production depends on several essential factors, including the application of plant nutrients to maintain soil fertility and crop yields at the desired level. Mineral fertilizer is an important source of plant nutrients in modern agricultural production. Of the mineral fertilizer macro-nutrients, phosphorus (P) has received special attention for being non-renewable and potentially scarce, as it is sourced from mined phosphate rock. The biogeochemical flow of phosphorus has crossed the boundary for what constitutes a safe operating space for the planet in terms of how much mineral P fertilizer should be added annually to erodible agricultural soils and the amount of P in freshwater flowing into oceans (Rockstrom et al., 2009; Steffen et al., 2015). The hike in the price of phosphate rock that took place in 2007/2008 caused a particular stir and spurred renewed debate about global P management, the remaining lifetime of phosphate rock reserves, and the possibility of a global P crisis (Cordell et al., 2009; Cordell and White, 2011; Scholz and Wellmer, 2013; Ulrich and Schnug, 2013). One aspect of global P availability is that the supply side is dominated by a few countries that control reserves and/or production (Jasinski, 2017). None of these countries are located in Europe, and in 2013, phosphate rock was included on the European Commission's list of critical commodities (European commission, 2014) based on the evaluated supply risk and its importance to the European economy. The dependence on imports of such a critical input for agricultural production is seen as contributing to food system vulnerability (Cordell and Neset, 2014; HCSS, 2012). During the last 40 years, Western Europe, in particular, has seen high application of mineral P fertilizer, which, together with the application of animal manure, has far exceeded the cumulative crop P uptake in the same period (Sattari et al., 2012). This over-application of P fertilizer to European agricultural soils over time has led to substantial amounts of accumulated soil P, referred to as legacy P. Legacy P can serve as a secondary source of P and potentially substitute mineral P imports (Rowe et al., 2016).

1.2 Phosphorus as a pollution problem

However, over-application of P to agricultural soils and the build-up of high soil P levels is not just inefficient use of the P resource, it is also associated with a higher risk of P losses to water recipients (Smith et al., 1999). Globally, P losses to water recipients are challenging the planet's capacity to handle this input while keeping marine ecosystems stable (Carpenter and Bennett, 2011). In fact, agriculture is the most important contributor of P to surface waters, in particular in developed countries, where point-source losses of P have been significantly reduced (Kleinman et al., 2011). The loss of P to freshwater recipients is a regional and local challenge, causing freshwater eutrophication, but it can also cause anoxic ocean events (Rockstrom et al., 2009). Losses of P from agricultural soils are typically diffuse and mainly occur through runoff and erosion, since P is mainly adsorbed to soil particles and to a much lesser extent dissolved in solution (Sharpley et al., 2013).

However, losses are less a function of fertilizer input and more a function of the level of legacy P, also referred to as soil P level, already in the soil (Bechmann, 2014; Kleinman et al., 2011).

1.3 Phosphorus management for sustainable food systems

It seems intuitive, then, that a more sustainable food system is one that uses P more efficiently as a fertilizer in order to reduce the extraction of primary mineral P reserves and to reduce the losses of P to the environment, where it causes harm. Increasing P efficiency can be understood as achieving the same output in terms of food production using less P input. This can also be specified for subsystems of the food system, such as soil P efficiency for plant production, defined as the ratio of P uptake in harvested crops over the sum of P inputs (Senthilkumar et al., 2012). There are several ways of increasing P use efficiency in the food system. Withers et al. (2015) propose five R strategies (5R) to increase P resource efficiency. They are presented in the perceived ascending order of difficulty of implementation: Realign P inputs – Reduce P losses to water – Recycle P in bioresources – Recover P in wastes – Redefine P in the food chain. Realigning P inputs means matching the inputs of P fertilizer more closely with the requirement for P fertilizer, including taking into account the contribution of legacy P to plant growth. A reduced P fertilizer requirement as an effect of high levels of legacy P has been demonstrated, among others, by Sattari et al. (2012).

Organic residues (also referred to as bioresources or organic waste) in the food system are important potential sources of P that can be recycled back into food production to replace mineral P fertilizer. This input of P can thus be called secondary P, in contrast to the primary P that comes from mined phosphate rock, while organic residues used as fertilizer can be referred to as secondary fertilizer. Of the organic residues in the food system, animal manure commonly constitutes the most important source of secondary P (Cordell et al., 2009). However, the specialization that has taken place in agricultural regions has to a large extent broken the crop-livestock P cycle by geographically segregating intensive livestock production (and manure generation) from areas dominated by crop production (Ashley et al., 2011; Sharpley et al., 2015). Livestock farming imports P through feed crops but does not return P in manure to crop areas where the feed is produced because of the costs associated with manure transport (Nesme et al., 2015). Areas of high livestock density are therefore often associated with accumulation of excess P and high levels of legacy P in agricultural soils, while specialist crop regions depend on mineral P fertilizer to nurture the crops (Nesme et al., 2015). The recovery and redistribution of manure P from areas of P surplus to crop lands with a P deficit would reduce this imbalance and improve regional P use efficiencies (MacDonald et al., 2011).

1.4 Current P management in Norway

Norwegian agriculture displays many of the same characteristics and challenges as other countries in Western Europe as regards P management: high soil P levels in areas specialized in livestock production, lower levels in areas dominated by cereal production, and high overall levels of legacy P in agricultural soils due to decades of P fertilizer over-application (Bechmann, 2014). To reduce P over-application and associated P losses to water bodies, the processing and geographical redistribution of manure P has been discussed, although the costs and energy requirements are seen as barriers (Bechmann and Øgaard, 2010; Knutsen and Magnussen, 2011). It has also been proposed to increase the recycling of P in organic residues, such as sewage sludge and meat bone meal, in the Norwegian food system to substitute the use of mineral P fertilizer, although limited P plant-availability may reduce the substitutability (Bøen and Grønlund, 2008). Nonetheless, the appropriate economic and regulatory incentives for improved utilization of P in the food system are still missing (Bøen and Haraldsen, 2011). There is currently no upper regulatory limit on P fertilizer application to agricultural soils in Norway, although there is a restriction on livestock density equal to 2.5 livestock units per hectare (ha) (Amery and Schoumans, 2014; The Norwegian regulations relating to organic fertiliser, 2003). Since a livestock unit represents 14 kg P (equal to the average annual excretion of a dairy cow), livestock farmers are required to have enough land to accommodate 35 kg P per ha, although the manure does not need to be evenly distributed.

In order to define priorities for how to improve P use efficiency in the Norwegian food system, it is clearly necessary to gain systematic accounts of the prevalence of P in the food system, as also noted by Farestveit et al. (2015).

1.5 Main aim and research questions

The overall aim of this thesis has been to contribute to improved P management and P use efficiency in Norwegian agricultural crop production by improving our understanding of the potentials for secondary P recycling.

In order to make a meaningful contribution to this aim, three main research questions were formulated:

Research question 1:
What are the stock and flow characteristics of P and plant-available P in the Norwegian food system and what theoretical potential does secondary P have to satisfy the P fertilizer requirement in the short and long term and, as such, substitute mineral P fertilizer?

Research question 2:
What are the life cycle environmental impacts of technological options for geographic redistribution of secondary P to increase system-wide P use efficiency in Norway, and what are the critical factors and processes in such a P redistribution?

Research question 3:
How will different substitution principles critically influence the LCA inventory and impact results when analyzing the substitution of mineral fertilizers by organic fertilizers in terms of nutrients?

1.6 Structure of the thesis

This thesis is divided into four main chapters. This Chapter 1 is an introductory chapter containing background and research questions, while Chapter 2 presents the methodology employed, some of the central elements included in the studies, and the case study systems that have been studied.

Chapter 3 goes on to summarize the main findings in the papers, while, in Chapter 4, I discuss the findings in light of the research questions and discuss some implications of the work. The final Chapter 5 concludes the thesis.

Figure 1 below shows how the papers relate to the research questions and the methodology used.

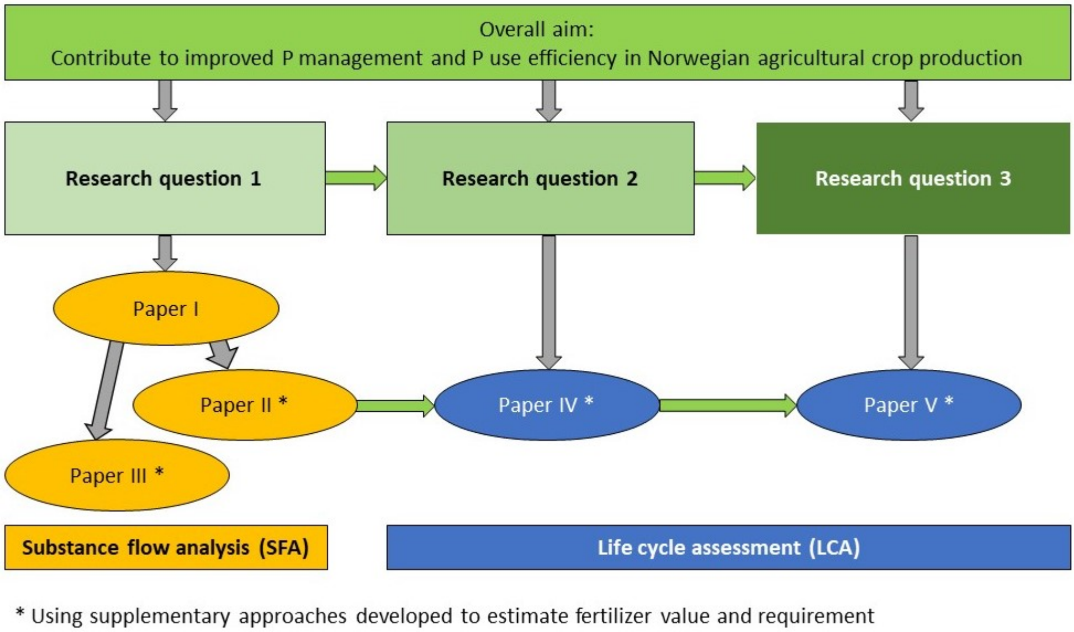


Figure 1. Relating research questions to papers and methodology

2 Research methods

In the papers included in this doctoral work, two main methods were used to answer the research questions, namely Substance Flow Analysis (SFA) and Life Cycle Assessment (LCA). In addition, supplementary approaches for estimating fertilizer requirement and fertilizer value were developed in order to provide necessary data input for the SFA and LCA work. The following subsections will briefly present how the methods were used in the papers, as well as the geographical context and case studies used.

2.1 Papers I-III: SFA and P flow modelling

2.1.1 Introducing SFA

Substance flow analysis (SFA) is a version of material flow analysis (MFA) that focuses on single substances (such as P) instead of a more complex material (such as a food commodity). The terms SFA/MFA are also used interchangeably, and MFA is defined as "*… a systematic assessment of the flows and stocks of materials within a system defined in space and time*" (Brunner and Rechberger, 2004, p.14). The method is based on mass balance of flows inside and across the system boundaries of a defined system and its processes, where inputs of a substance into a process have to equal outputs plus any net stock change. A general MFA procedure is visualized in Figure 2, while Table 1 shows a spreadsheet setup used to determine substance flow rates. A substance flow rate (inflow, outflow or net stock accumulation; "$\dot{X}$" in Table 1) is estimated through the collection of data on material flows (e.g., the amount of barley harvested in Norway in 2009; "$\dot{m}$" in Table 1) and then multiplied by the substance concentration of the materials (e.g., the concentration of P in barley; "c" in Table 1).

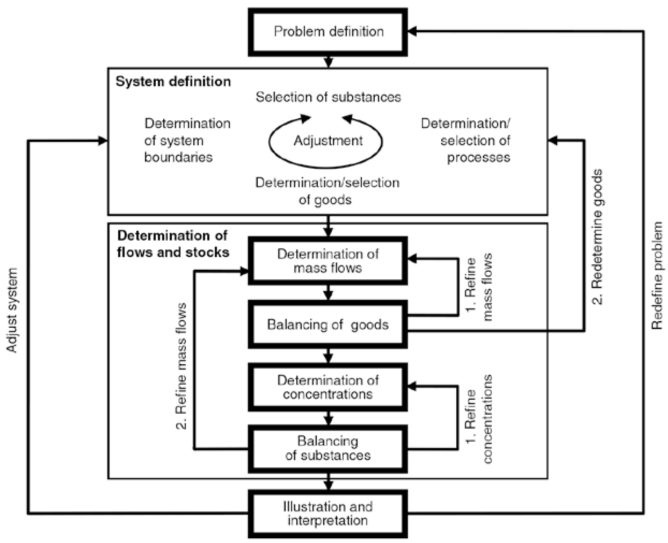


Figure 2. Procedures for MFA (Brunner and Rechberger, 2004)

Table 1. Data spreadsheet for MFA/SFA to determine substance flow rates (Brunner and Rechberger, 2004)

Goods	Flow Rate, t/year	Concentration of Substance $S_1, S_2, S_3, \cdots, S_n$, mg/kg					Substance Flow Rate $S_1, S_2, S_3, \cdots, S_n$, kg/year				
G_1	$\dot{m}_1$	c_{11}	c_{12}	c_{13}	$\cdots$	c_{1n}	$\underline{\dot{X}}_{11}$	$\underline{\dot{X}}_{12}$	$\underline{\dot{X}}_{13}$	$\cdots$	$\underline{\dot{X}}_{1n}$
G_2	$\dot{m}_2$	c_{21}	c_{22}	c_{23}	$\cdots$	c_{2n}	$\underline{\dot{X}}_{21}$	$\underline{\dot{X}}_{22}$	$\underline{\dot{X}}_{23}$	$\cdots$	$\underline{\dot{X}}_{2n}$
G_3	$\dot{m}_3$	c_{31}	c_{32}	c_{33}	$\cdots$	c_{3n}	$\underline{\dot{X}}_{31}$	$\underline{\dot{X}}_{32}$	$\underline{\dot{X}}_{33}$	$\cdots$	$\underline{\dot{X}}_{3n}$
$\vdots$	$\vdots$	$\vdots$	$\vdots$	$\vdots$	$\cdots$	$\vdots$	$\vdots$	$\vdots$	$\vdots$	$\cdots$	$\vdots$
G_k	$\dot{m}_k$	c_{k1}	c_{k2}	c_{k3}	$\cdots$	c_{kn}	$\underline{\dot{X}}_{k1}$	$\underline{\dot{X}}_{k2}$	$\underline{\dot{X}}_{k3}$	$\cdots$	$\underline{\dot{X}}_{kn}$

Note: Substance flow rates are underlined. G = name of good; S = name of substance

2.1.2 Overall system description

SFA was used in Papers I-III to map the flows and stocks of P in the Norwegian food system, including P imports, stock accumulations, P flows between sectors of the food system, P in waste flows, and P losses to the environment. Data were collected for the years 2009–2011 and an annual average was calculated to smooth out variations from year to year. The SFA was quasi-stationary, allowing stocks to change from one year to another through the calculation of net stock changes.

In Paper I, we used SFA to map the P flows in the Norwegian food system. The P flows were determined on a national scale, with the spatial system boundary set to the Norwegian economic zone, including coastal and marine waters for aquaculture and fisheries because of the high importance of these sectors in Norway. In addition to studying past flows (assumed to be representative of the current flows at the time of publication), a scenario for the year 2050 was developed to identify some possible challenges for national P management caused by the anticipated fivefold increase in aquaculture production by 2050 (DKNVS and NTVA, 2012).

In Paper II, national scale flows were disaggregated down to regional scale, subdividing Norway into its 19 counties (Figure 3), in order to see how P flows are distributed geographically and to identify any regional differences. The processes included in the system were reduced to agricultural soil and municipal wastewater treatment. Furthermore, the paper made an early attempt at considering quality aspects of the P flows in terms of P plant-availability, so that flows of total P (as in Paper I) could be further adjusted to represent the flows of P with the same fertilizer effect as mineral P fertilizer (see Section 2.1.3 below for more details).

Paper III built on the national scale P flows arrived at in Paper I, but extended the SFA to integrate P plant-availability (see Section 2.1.3 below for more details). Plant-availability of P was integrated to obtain a better picture of the fertilizer value of secondary resources and their potential to substitute mineral fertilizer, which is commonly overestimated when relying on flows of total P.

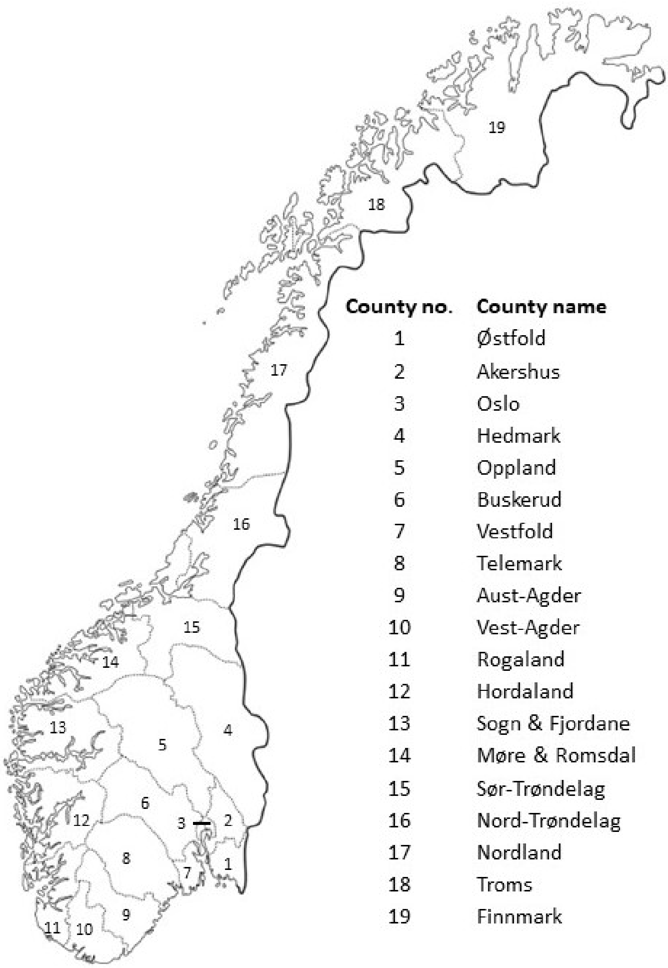


Figure 3. A map of Norway and its 19 counties.

2.1.3 SFA and P plant-availability

An underlying motivation for Papers I-III was to highlight the theoretical potentials for substituting mineral P fertilizer with secondary P in organic residues, thereby highlighting opportunities to improve the P use efficiency of the Norwegian food system. However, some organic waste flows have low degrees of plant-available P, such as chemically precipitated sewage sludge and meat bone meal (see for example Brod et al., 2015; Øgaard and Brod, 2016). The use of total P in SFA for such materials would clearly overestimate their P fertilizer value and the amount of mineral fertilizer that can theoretically be replaced. The inclusion of plant-availability in SFA was operationalized through the term mineral fertilizer equivalent (MFE; also referred to as relative agronomic efficiency (RAE)), which states plant-availability as a relative measure (in %) of the fertilizer effect of a substrate in comparison to mineral fertilizer. It is then assumed that all mineral P fertilizer is plant-available, i.e., with an MFE/RAE of 100%. This term is then also used to quantify mass in Paper V (kg MFE-P) when quantifying the amount of mineral fertilizer equivalent P in a secondary fertilizer with a certain mass total P. Mineral fertilizer equivalent is therefore a particularly useful term when the objective is to estimate the amount of mineral fertilizer that a secondary fertilizer could potentially substitute. As an example, chemically precipitated sewage sludge may have an MFE of about 30% (Øgaard and Brod, 2016), which means that only 30% of the total P in that sludge has the same fertilizer effect as mineral P fertilizer. This, in turn, means that 100 kg total P in chemically precipitated sewage sludge has the same fertilizer effect as 30 kg of mineral P fertilizer (both stated in elemental mass of P). In Paper II, MFE values were found in Norwegian and international literature, while in Paper III they were mainly based on Norwegian pot experiments complemented by international experimental data and other literature (see Paper III for more details). The MFE/RAE concept can be illustrated by a two-pool soil model, where the fraction of applied secondary P with a P fertilizer effect equivalent to mineral P fertilizer enters the readily available P pool, while the remaining P enters a residual pool (Figure 4).

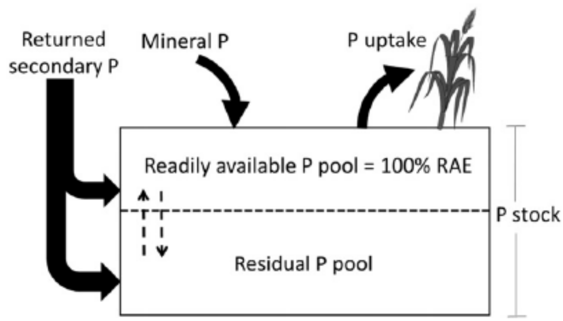


Figure 4. Two-pool P plant-availability with relative agronomic efficiency (RAE)

2.2 Papers IV & V: LCA and environmental impacts of P management

2.2.1 Use of LCA in Papers IV and V

Life cycle assessment (LCA) is defined and described in ISO 14040 and 14044 (2006a, b) as a method for evaluating the potential environmental impacts associated with the life cycle of a product or service. It is further outlined in documents such as the ILCD Handbook (European Commission JRC, 2010) and in Baumann and Tillman (2004). LCA was used in Papers IV and V. The papers share the same starting point, namely that we want to find the best use of animal manure in terms of its nutrient content. Based on this, an input unit-related functional unit (FU) was chosen, supported by Cherubini and Strømman (2011). However, the papers differ in their aim. Paper IV studies the management of dairy cow manure to estimate the life cycle environmental impacts of redistributing manure P using several technology options. Paper V studies conventional dairy cow manure management as a case to examine a more general issue in LCAs on nutrient recycling: the use of different assumptions for mineral fertilizer substitution in the life cycle inventory (LCI) phase of LCA.

2.2.2 The LCA case studies and P redistribution options

The main aim of Paper IV was to estimate the environmental impacts of redistributing manure P from a county with a manure P surplus to a county with a P deficit and a need to import P fertilizer. Based on Paper II, we chose the county with the greatest P surplus as the donor, Rogaland county, and the county with the largest deficit, Akershus county, as the recipient. Hence, we examined the redistribution of manure P from Rogaland county to Akershus county, including some 500 km transport from the south-west to the south-east of Norway (see Figure 3), based on the use of different processing technologies. Rogaland county is an agricultural region with high livestock density and therefore a tendency to very high soil P levels, which is assumed in both Papers IV and V. The application of manure is limited to 35 kg P ha^{-1} (The Norwegian regulations relating to organic fertiliser, 2003). Akershus county is dominated by cereal production, has low livestock density, and is therefore dependent on P fertilizer import.

The FU of the systems studied in both Papers IV and V was set to be the management of 1 tonne of fresh dairy cow manure. In Paper IV, five different technologies for P redistribution were compared to a reference scenario of local application at a hypothetical donor farm (see Table 2 for descriptions). Two of the scenarios included using solid-liquid separation (screw press / decanter centrifuge). We also decided to combine anaerobic digestion (AD) with solid-liquid separation in two scenarios, since it is a national ambition to increase the processing of manure with AD in order to reduce the climate change impact of Norwegian agriculture (Norwegian Ministry of Agriculture and Food, 2009). The biogas produced was assumed upgraded to green gas to substitute the production and use of fossil diesel fuel. The fifth scenario evaluated the environmental consequences of transporting unseparated slurry from donor to recipient.

Table 2. Description of redistribution technology options

Scenario	Description
Ref	Reference scenario. Manure stored in a manure cellar below the animal house and applied locally to grassland on the donor animal farm.
SP	Pre-stored slurry separated by screw press (SP). The resulting solid fraction is stored, hygienized, and transported to a recipient farm in Akershus county, and applied to arable land. Liquid fraction stored and applied locally.
DC	Like the SP scenario, but separation by decanter centrifuge (DC).
AD_SP	Pre-stored slurry digested through anaerobic digestion (AD), then separated by screw press (SP). The digested solid fraction is stored, hygienized, and transported to Akershus county, and applied to arable land. Digested liquid fraction stored and applied locally.
AD_DC	Like the AD_SP press scenario, but separation by decanter centrifuge (DC).
NoSep	No separation of slurry. Slurry stored as in the reference scenario, then hygienized and transported in its entirety to Akershus county, and applied on arable land.

Paper V used the reference scenario from Paper IV as a model for its case study, although it is not explicitly set in the context of Rogaland. The manure management system in both papers ends with the application of cattle slurry on agricultural land, where it substitutes for mineral fertilizer. The system boundary and processes included in Papers IV and V are shown in Figure 5.

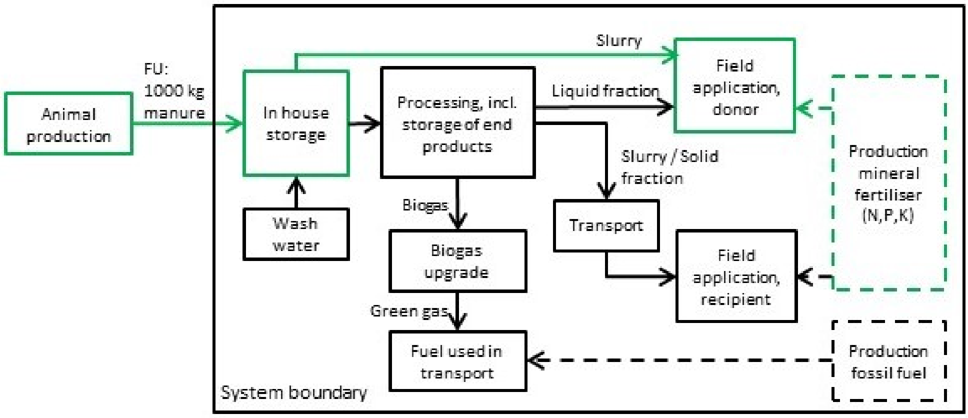


Figure 5. The system boundary for the LCA studies in Papers IV and V. All the processes and flows shown were included in Paper IV, while only those in green were included in Paper V.

2.2.3 Replacement of mineral fertilizer and substitution principles

In Papers IV and V, secondary P is recycled back into food production by applying manure to the field. The management of manure (or any other source of secondary P used as a fertilizer) is a multifunctional system that both provides the function of waste management and produces fertilizer (see for example Ekvall and Finnveden, 2001). By applying secondary P to farmland, we assume that it replaces mineral P fertilizer. To credit this displacement of primary mineral fertilizer, the system boundaries can be expanded to include its (avoided) production and to subtract the associated inventory of emissions and resource and energy use from the rest of the system. This is sometimes referred to as system expansion or the substitution method and is a common approach to credit recycling of materials in multi-functional systems for waste management (Laurent et al., 2014).

Different assumptions can be used to calculate the amount of avoided mineral fertilizer, and Paper V analyses how such assumptions can influence LCA results. Three different substitution principles were identified from the LCA literature: the one-to-one, maintenance, and adjusted maintenance substitution principles (Table 3). A mathematical description of the principles is provided in Paper V.

Table 3. Description of mineral fertilizer substitution principles (abbreviation of principles in parentheses)

Mineral fertilizer substitution principle	Description
One-to-one substitution principle (One-to-one principle)	• The amount of avoided mineral N, P, and K fertilizer equals the amount of MFE-N, -P, and -K in the organic fertilizer in a ratio of 1:1.
Maintenance substitution principle (Maintenance principle)	• A certain crop or crop rotation receiving the organic fertilizer is given. • Applied MFE-N, -P, and -K in the organic fertilizer is compared to the general crop fertilizer requirement for each nutrient. • Any over-application does not substitute mineral fertilizer.
Adjusted maintenance substitution principle (Adjusted principle)	• A certain crop or crop rotation receiving the organic fertilizer is given. • Applied MFE-N, -P, and -K in the organic fertilizer is compared to the crop fertilizer requirement for each nutrient, adjusted for local or regional soil characteristics. • Any over-application does not substitute mineral fertilizer.

2.2.4 Impact categories used in life cycle impact assessment

The environmental impacts in Paper IV were estimated using five ReCiPe impact categories (climate change, marine eutrophication, terrestrial acidification, particulate matter formation, fossil resource depletion) (Goedkoop et al., 2009) in addition to a two-part category used to specifically highlight the consequences in terms of phosphorus of the scenarios – called AMP/POA. Avoided mineral P (AMP) is identical to (negative) depletion of mineral P and is sometimes included in broader impact categories of abiotic or fossil resource depletion in LCA. Emissions of P to water recipients throughout the product life cycle are likewise captured in impact categories of eutrophication, sometimes specified as freshwater eutrophication, since freshwater is where P is usually the limiting factor for algal growth. However, neither depletion of mineral P nor direct emissions of P to water fully captures the potential risk associated with the over-application of nutrients, which is why P over-application (POA) was added as the second part of this impact category. In Paper V, all eighteen ReCiPe categories were included to obtain a richer picture of how avoided mineral fertilizer production influenced impacts.

2.3 P fertilizer requirement

An important part of more efficient use of P in food production is to determine the requirement for P fertilizer and to align fertilizer application with that requirement (Withers et al., 2015). The actual application of P to agricultural land may be far from the recommended amount of P fertilizer, and, in Norway as a whole, P is typically over-applied to agricultural land (Paper I). The required amount of P fertilizer is a function of crop type and expected yield level, as well as the amount of legacy P in the soil, which can be considered a source of secondary P (Rowe et al., 2016). In this PhD work my intention has been to estimate the total P fertilizer requirement on a regional or country scale to help improve P management in both the short run and the longer run, and to include levels of legacy soil P in these estimates.

The relationship between the application of P fertilizer and crop yields follows the law of diminishing returns (Syers et al., 2008). Hence, there is a critical soil P level above which any further P application has limited to no positive effect on crop yields, only increasing fertilizer costs and the potential losses of P to water recipients. Below this critical soil P level, there is a risk of crop yield reduction and loss of income for the farmer. We can call this an economic-environmental optimal soil P level.

In Norway, soil P levels in agricultural soils are generally high (see Paper II), and the same is found for Western Europe as a whole (Sattari et al., 2012). The level of plant-available soil P in Norway is measured using the P-AL extraction method, which extracts P from a sample of soil using ammonium-acetate-lactate and shows the result as mg P-AL per 100 g soil (Egnér et al., 1960). Krogstad et al. (2008) proposed P fertilizer corrections according to P-AL values for grass and cereals in Norway. In order to approach the optimal soil P level, they prescribed reducing P fertilization in cases of high soil P levels (Table 4) in relation to maintenance fertilization.

Maintenance fertilization involves matching P fertilizer input with the crop P offtake, so that the soil P level is maintained at a constant level, which is the prescribed fertilization strategy when soil P is in the optimal range. Maintenance fertilization therefore reflects the long-term perspective, in accordance with Schoumans et al. (2015), and it informs fertilizer regime 1 (FR1) in Paper II and the maintenance substitution principle in Paper V. Adjusting P fertilizer input based on soil P levels is a strategy for reaching the optimal level, and it can therefore be seen as a shorter-term perspective. This adjustment informs fertilizer regime 2 (FR2) in Paper II, referred to as a transition fertilization strategy, as it is a transition to the longer-term soil P state. The analog substitution principle in Paper V is the adjusted maintenance substitution principle. The utilization of legacy P in areas with high soil P levels through reduced P input is in line with the general recommendations of Rowe et al. (2016).

Table 4. Classes of P-AL level and percentage correction of the P requirement for grass, cereals, and oilseed production (Krogstad et al., 2008)

Class	P-AL value (mg per 100g soil)	Name of class	Regression equation for percentage correction (Y) of P requirement
A	1-5	Low	Y=-25*P-AL+125
B	5-7	Medium/ Optimal	Y=0
C1	7-10	Moderate high	Y=-14.28*P-AL+100
C2	10-14	High	Y=-14.28*P-AL+100
D	> 14	Very high	Y=-100

In the proposed set of corrections in Table 4, the general fertilizer requirement for a crop is corrected based on the P-AL value of a soil, given as ranges in five P-AL classes. As farmers sample and analyze their soils to find its P-AL status, the proposed classification facilitates adjustments in P fertilization. An example of how the correction is determined for both high and low soil P levels can be found in the Supplementary Material, Section 4, of Paper V.

The correction of high soil P levels is obviously a dynamic parameter in an iterative procedure: reduced P fertilization leads to a desired reduction in the soil P level over time (MacDonald et al., 2012), and the percentage correction of P fertilizer inputs is subsequently reduced when the soil P level approaches the optimal (see Table 4).

In Paper II, corrections of the P fertilizer requirement were based on a county-based weighted P-AL average. A more detailed description of the method and the collected P-AL data can be found in the Supplementary Material to Paper II. In Paper III, we based the P fertilizer demand on a weighted P-AL average for the country as a whole. In Papers IV and V, typical or plausible P-AL values were used for the given regional geographical context, guided by the collected P-AL data in Paper II.

2.4 Fertilizer value of secondary resources

Mineral fertilizer equivalence (MFE) of P was integrated in the SFA studies in Papers II and III, as mentioned in Section 2.1.3. In Papers IV and V, the fertilizer value of nitrogen (N) and potassium (K) was also included, since N, P, and K are all present in animal manure and can replace mineral N, P, and K fertilizers, respectively. Hence, only focusing on P fertilizer value is an oversimplification that could result in wrong recommendations from an LCA. Cattle manure P was assumed to have an MFE of 100%, based on Brod et al. (2015), and the same was assumed for manure K, based on De Vries et al. (2015). The amount of N (MFE-N) to replace mineral N fertilizer is usually a function of local characteristics surrounding the field application, since losses of N to both the atmosphere and water recipients can be substantial both during and after spreading (Oenema et al., 2007). In addition, part of the organically bound N in organic fertilizers such as animal manure will mineralize during the growing season and become available to plants.

Here, it should be noted that the amount of MFE-N was calculated differently in Papers IV and V. I highlight this, since the amount of MFE-N and the N:P ratio (or more specifically, the MFE-N:MFE-P ratio) of a secondary fertilizer also influence the utilization of secondary P (see Paper V for more details). The amount of MFE-N in secondary fertilizers that could potentially replace mineral N fertilizer may also make a difference to LCA impact results, since the (avoided) production of mineral N fertilizer is highly energy-demanding (Hasler et al., 2015). In Paper IV, the calculation of MFE-N is based on general expected gaseous losses of mineral N to the atmosphere during field application and general mineralization rates of organic N (N_{org}) during the growing season, as described in the online fertilizer handbook from NIBIO (2016). This calculation of MFE-N can be described as follows:

$$MFE\text{-}N = N_{min} \times MFE\ N_{min} + N_{org} \times k_{mineralization} \qquad \text{(Eq. 1)}$$

where N_{min} is the amount of mineral N (often given as the amount of ammonium N (NH_4-N)) in the secondary fertilizer prior to field application (kg N), $MFE\ N_{min}$ is a general mineral fertilizer equivalent of applied N_{min} including expected N losses (% of N_{min}), N_{org} is the amount of organic N in the secondary fertilizer prior to field application (kg N), and $k_{mineralization}$ is a general mineralization factor (% of N_{org}).

The above procedure is the prescribed method for determining the N fertilizer value of organic fertilizers such as animal manure in Norway. However, it has the shortcoming that it does not necessarily comply with the mass balance principle for agricultural soil because losses of N to water recipients are not included, and because the life cycle inventory of N losses to the atmosphere and water recipients in LCA is often calculated independently using emission factors from the literature (Heimersson et al., 2016). In Paper V, we therefore estimated the amount of MFE-N based on the difference between the applied N_{min} (given as NH_4-N), including the mineralization of N_{org} during the growing season, and N losses to the environment, as in Eq. 2. The method is further described in the Supplementary Material of Paper V.

$$MFE\text{-}N = N_{min} + N_{org} \times k_{mineralization} - N_{losses} \qquad \text{(Eq. 2)}$$

3 Main findings

Sections 3.1–3.5 present the main findings of Papers I-V. A short introduction to each paper is given below.

Paper I (Hamilton et al., 2016) presents a national substance flow analysis for phosphorus in the Norwegian food system, including aquaculture and fisheries. The study indicates the major P flows within the current system as well as in a scenario for 2050 based on a fivefold increase in aquaculture production.

Paper II (Hanserud et al., 2016) provides a disaggregated soil P balance down to county level in Norway and demonstrates how the secondary organic resources animal manure and sewage sludge could cover the required P fertilizer in agricultural production given two different fertilizer regimes.

Paper III (Hamilton et al., 2017) integrates different qualities of P in terms of plant-availability with the substance flow analysis methodology to indicate the P fertilizer potential of secondary P resources in the Norwegian food system. This is based on the flows of total P described in Paper I.

Paper IV (Hanserud et al., 2017) explores the environmental consequences of redistributing P in animal manure from a region with high livestock density and a P surplus to a region dominated by arable farming and a need to import P fertilizer.

Paper V (Hanserud et al., 2018) examines the assumptions that are implicitly or explicitly made in LCA studies on organic fertilizer as regards calculating the amount of avoided mineral fertilizer. The assumptions used can be decisive for the resulting impacts and the conclusions drawn.

Each paper, including supplementary materials, offers detailed results and interpretations of them, at different levels of importance and resolution. The selected main findings chosen for presentation below will later be discussed with respect to how they inform the overarching research questions that were introduced in section 1.5, methodological strengths and weaknesses, and implications of this work.

3.1 Paper I

Investigating cross-sectoral synergies through integrated aquaculture, fisheries, and agriculture phosphorus assessments: A case study of Norway

As a starting point for the work in this thesis, this article aimed to quantify the current P flows and stocks in the Norwegian food system, including aquaculture, fisheries, agriculture, food processing, and consumption, as well as the waste management sector. A simplified overview of the resulting flows and stocks is shown in Figure 6, in which the processes of fisheries and aquaculture are merged and several smaller flows are either merged into aggregated flows or not shown for the sake of visual clarity.

At first glance, the most striking feature of the estimated flows and stocks is the predominance of mineral fertilizer production in Norway, with large imports of phosphate rock and equally large exports of P fertilizer products. Barely 10% of the mineral P fertilizer produced is used on Norwegian soils, including urban greening and landfill covers, in addition to agricultural land. Another characteristic of the food system is the large throughput of P in the fisheries and aquaculture sector, which drives P consumption and losses at levels comparable to the agriculture sector. The combined losses of P from fisheries and aquaculture at 10.2 kt P yr^{-1} (dumped fish scrap from fisheries and fish excrements, feed losses, and escaped fish from aquaculture) is of the same order of magnitude as the net stock accumulation of P in agricultural and greening soils at 12 kt P yr^{-1}.

Furthermore, in addition to what can be seen directly from Figure 6, it was found that aquaculture production has a fish feed consumption (17.4 kt P yr^{-1}) of the same order of magnitude as livestock consumption of P in feed, fodder, and grazing (12 kt P yr^{-1}). P in waste flows related to processing, retail and human consumption was relatively small in comparison to agriculture, fisheries, and aquaculture.

In addition to the retrospective P flow analysis for 2009–2011 we developed a scenario to show how a projected fivefold increase in aquaculture production by 2050 could affect P flows upstream and downstream. Here, the fivefold increase in farmed fish also results in a fivefold increase in P in lost feed and fish excrement equal to 45 kt P yr^{-1}, given no change in technology. The amount lost is well above the inputs to agriculture. The projected increase in aquaculture production is made possible by increasing the imports of fish feedstuff to 55 kt P yr^{-1}, compared to 9.4 kt P yr^{-1} for 2009–2011.

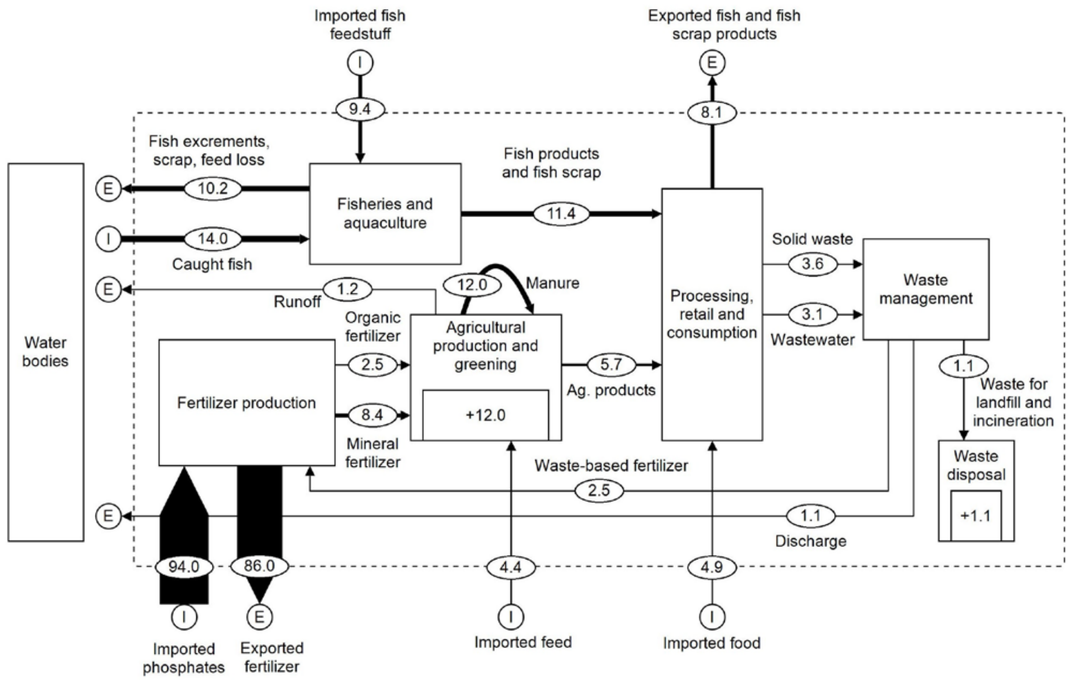


Figure 6. Simplified phosphorus balance of the Norwegian food system presented in Paper I, kt P yr^{-1}, averaged 2009-2011 data. Mass balance inconsistencies are not included. I = import; E = export.

3.2 Paper II

A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway

In this paper, we built on the findings from Paper I, where it was concluded that secondary P is used inefficiently in food production in Norway and, in particular, in livestock production in agriculture. In Paper II, we suggested three possible main causes for this, namely i) geographical segregation between where secondary P is generated and where it is needed; ii) disregard for the levels of plant-available P already in the soil; and iii) the varying plant availability of secondary P. We aimed to estimate the theoretical fertilizer potential of animal manure and sewage sludge to supply the required P fertilizer for crops, using a county-level scale to be able to observe any regional differences. In the status quo soil balance for agricultural soil, i.e., with all the studied inputs (mineral P fertilizer, manure, and sewage sludge) and outputs (harvested crops and P losses), all counties had a positive balance, ranging between 2.7–14.7 kg P ha^{-1}. Hence, more total P was applied to agricultural soil than was removed, and P was accumulated in the soil. However, to be able to study whether the secondary P resources alone could have covered the required P fertilizer, we evaluated two fertilizer regimes (FR) for the same period that differed in their determination of required P fertilizer. FR1 assumed that the required P fertilizer equaled plant P offtake. FR2 adjusted the required P fertilizer according to the level of soil available P, and, because of the high soil P levels in many counties, the effect of this was an overall reduction in required P fertilizer. The fertilizer value of the total amount of manure and sewage sludge was then compared to the total amount of required P fertilizer. FR1 and FR2 yielded a national average P surplus of 1.2 and 6.2 kg P ha^{-1}, respectively, while the results by county are shown in Figure 7.

With FR1, twelve counties had the theoretical potential to cover all the required P fertilizer using plant-available P in manure and sewage sludge and still emerge with a surplus, while seven had a deficit and a need to import P fertilizer to cover crop P removal. With FR2, only three counties (with Oslo counting as one county) had a deficit, while the remaining sixteen had a surplus when the contribution from plant available P in the soil was taken into account.

The findings demonstrate that, by not taking into account the levels of plant available P in the soil, the maintenance fertilization strategy reflected in FR1 underestimates the amount of land (the number of counties) where manure and sewage sludge can supply all the required P fertilizer in the short term. In the same way, FR1 greatly underestimates the surplus fertilization in the short term compared with FR2 where soil P levels are taken into account. FR1 indicates that, also in the long term, counties with a P surplus could export secondary P to cover the required P fertilizer in counties with a deficit, and that the country as a whole would still have a surplus of 1.16 kt P yr^{-1} in manure and sewage sludge.

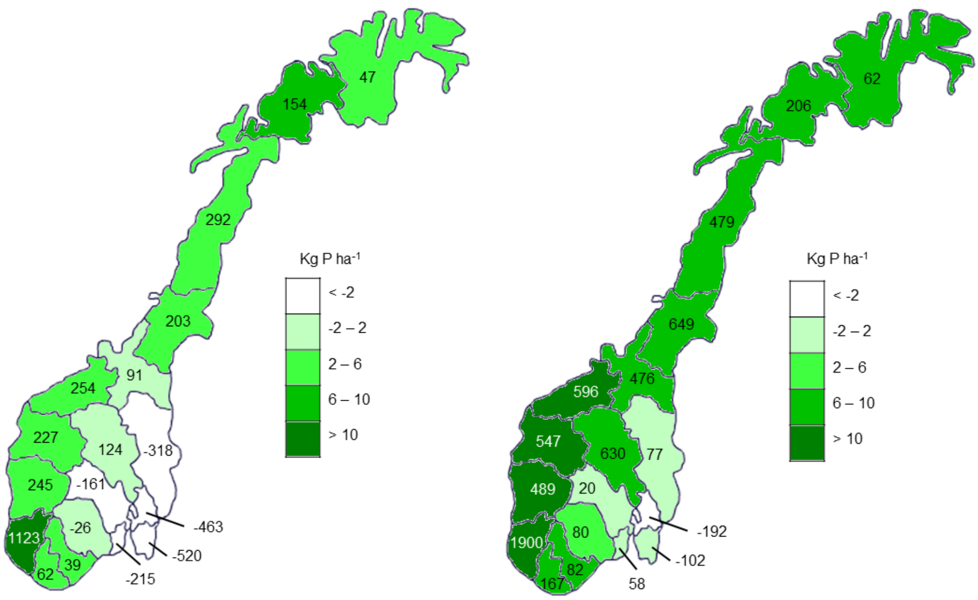


Figure 7. Annual surplus fertilization (tonnes P) in numbers, and surplus fertilization per hectare (kg P ha^{-1}) in color code for FR1 (left) and FR2 (right), 2009-2011.

3.3 Paper III

Recycling potential of secondary phosphorus resources as assessed by integrating substance flow analysis and plant-availability

The flows of total P were adjusted for plant-availability for each type of secondary organic resource, calculated using 95% confidence intervals. The results were therefore stated as a minimum and maximum value given the assigned uncertainty. The adjustment for plant-availability significantly reduced the fertilizer potential of most of the secondary resources, from a total of 28 kt total P yr^{-1} to between 12.7 and 26.3 kt plant-available P yr^{-1} as the minimum and maximum estimates, respectively (Figure 8). Of the secondary resources, manure has the largest recycling potential, equaling 8.7 to 11.4 kt plant-available P yr^{-1}, given the combination of a large amount of total P generated per year and a high MFE of between 76 to 100% (weighted MFE average for all manure types). The P fertilizer requirement at the national level was adjusted based on soil P values and estimated to be 5.8 kt plant-available P yr^{-1}. Another interesting secondary resource is fish sludge, which today is mainly lost directly to water bodies. However, that may change with stricter future regulations concerning such losses and as a result of the development of cost-effective technology to enable the collection and processing of fish sludge. Based on an overview of characteristics of the different secondary P resources, three other resources in addition to manure and fish sludge were also thought to be of particular interest for P recycling in Norway, namely anaerobically digested food waste, sewage sludge, and meat bone meal. The paper discusses the largest barriers to their efficient utilization as secondary P fertilizers in the Norwegian food system. Nonetheless, we note that even the minimum estimate of plant-available P in manure alone has the theoretical potential to satisfy the entire demand for P fertilizer in Norway, and as such replace 100% of the applied mineral P fertilizer.

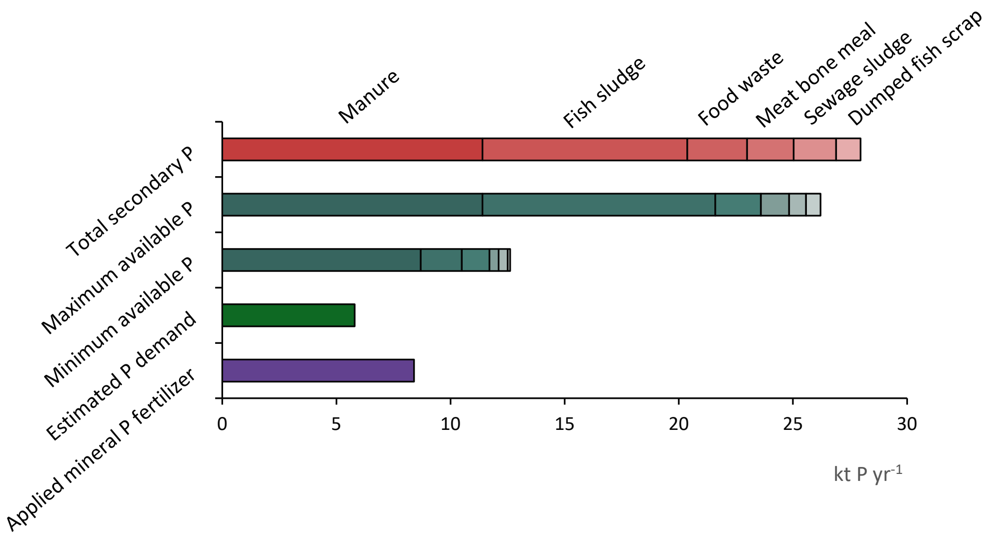


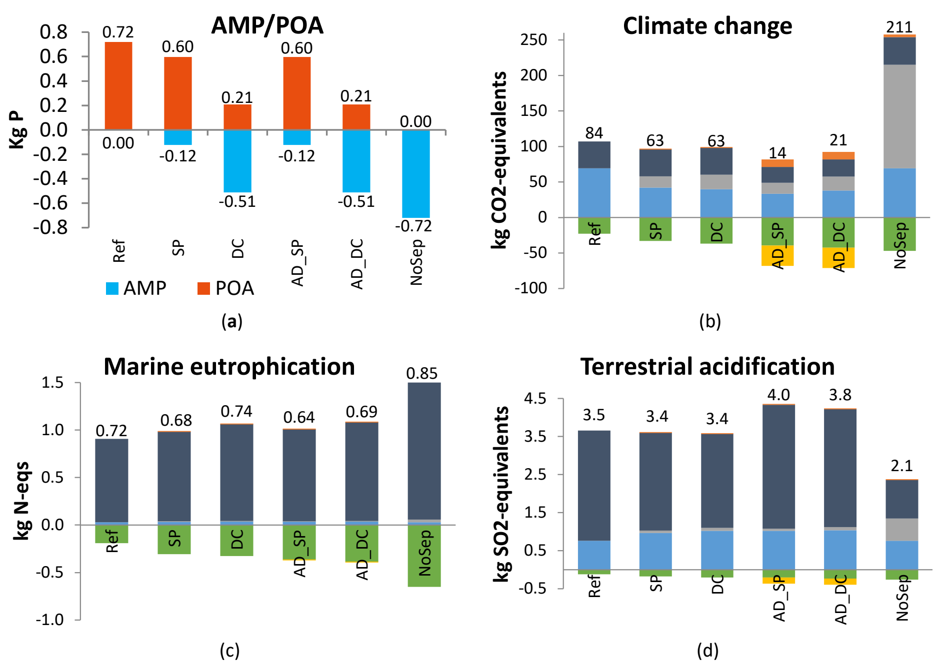
Figure 8. Total secondary P, maximum and minimum P recycling potential of secondary products, estimated P fertilization demand, and mineral P fertilizer applied in 2009-2011.

3.4 Paper IV

Redistributing phosphorus in animal manure from a livestock-intensive region to an arable region: Exploration of environmental consequences

In this paper, we carried out a life cycle assessment to study whether more efficient P use through manure P redistribution comes at the price of increased environmental impacts when compared to a reference system. The paper was motivated by the findings of Paper II, and we examined the redistribution of manure P from Rogaland county to Akershus county, including transport over some 500 km from the south-west to the south-east of Norway, using different processing technologies (Table 2).

Unsurprisingly, the scenario with no separation of slurry before transport (NoSep) redistributed the most manure P (100%) and therefore also substituted the most mineral P. However, this option had by far the highest potential climate change impacts and fossil fuel depletion of the alternative scenarios, because of the contribution from transportation (Figure 9). It is therefore not considered a realistic option for future manure management. The combination of AD with decanter centrifuge (AD_DC) seemed to be the most promising for manure P redistribution, and, compared to the reference, it had similar or lower impacts for all impact categories. The decanter centrifuge separated and redistributed 71% of the P in the slurry, in comparison with 17% for the screw press.



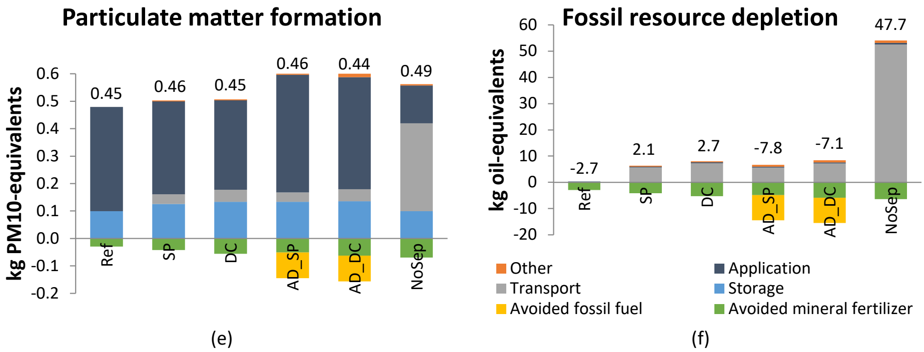


Figure 8. Contribution of the different processes in each scenario to the potential impacts on: (a) avoided mineral P (AMP)/P over application (POA); (b) climate change; (c) marine eutrophication; (d) terrestrial acidification; (e) particulate matter formation; and (f) fossil resource depletion. In (b)–(f): "Other" comprises the processes of separation, anaerobic digestion, biogas upgrading, and hygienization; "Application" comprises donor and recipient field application; "Storage" comprises in-house storage and end-product storage; and the net impact is shown in numbers above/below the bars. The scenarios are shown along the X-axis: Reference (Ref), solid-liquid separation by screw press (SP), solid-liquid separation by decanter centrifuge (DC), pretreatment by anaerobic digestion (AD) followed by SP (AD_SP), pretreatment by AD followed by DC (AD_DC), no separation of slurry before transportation (NoSep).

Furthermore, we wanted to explore the influence of different regional characteristics on impacts, i.e., differences in crop production and soil P level. For this purpose, we compared the reference, where 100% of the FU was applied in the donor region, with the NoSep scenario, where 100% of the FU was applied in the recipient region. We excluded the impacts from transportation and hygienization. The findings indicate that regional differences in typical crop production and soil P level influence the resulting impacts. Cereal production and lower soil P levels in the recipient region utilized the applied nutrients better and had lower overall emissions than intensive grass production and higher soil P levels in the donor region (Figure 10). Except for marine eutrophication, net impacts in the NoSep scenario were a factor of 1.4-2.7 lower than in the reference. This can mostly be explained by lower gaseous N emissions from manure application in cereal production, resulting in a greater amount of N left in the manure to replace mineral N fertilizer.

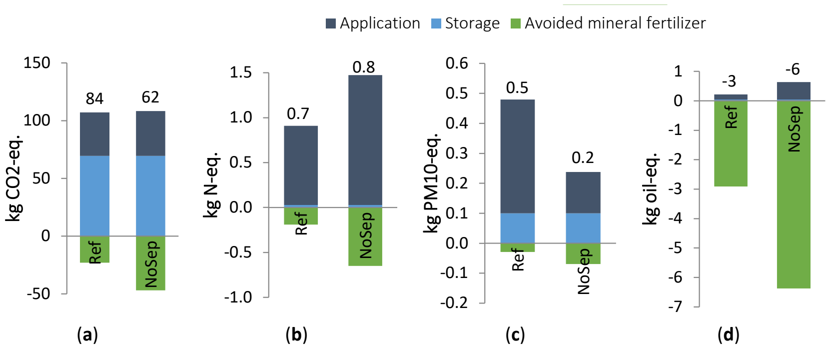


Figure 9. Impact results where the influence of regional differences is isolated – for the following impact categories: (a) climate change; (b) marine eutrophication; (c) particulate matter formation; and (d) fossil resource depletion. The net impacts are shown in numbers (rounded) above the bars and exclude hygienization and transport for the NoSep scenario. The impact category terrestrial acidification showed almost identical results to particulate matter formation and was therefore omitted from this figure for the sake of simplicity.

3.5 Paper V

Choice of mineral fertilizer substitution principle strongly influences LCA environmental benefits of nutrient cycling in the agri-food system

In Paper V, we follow up an observation made in Paper IV that the contribution and importance of avoided mineral fertilizer for net impacts vary quite a lot between different LCA studies involving the recycling of nutrients in organic fertilizer. We suspected that those differences stemmed from the different assumptions used when the inventory of avoided mineral fertilizer is calculated. Based on scientific publications in the field, we identified three mineral fertilizer substitution principles (Table 3).

The inventory of avoided mineral fertilizer varied substantially between the substitution principles (Table 5). The avoided mineral P fertilizer varied by 100% from the one-to-one principle, where all the applied MFE-P replaced mineral P, to the adjusted principle, where none of the applied MFE-P replaced mineral P fertilizer. The variation was also great for avoided mineral K fertilizer, while avoided N fertilizer remained constant across principles since it was under-applied with all principles.

Table 5. Inventory of avoided mineral fertilizers for the different substitution principles. Avoided mineral N, P, and K fertilizer as a percentage of MFE nutrients applied is shown in brackets.

Substitution principle	N (kg N)	P (kg P)	K (kg K)
One-to-one	2.09	0.72 (100%)	5.89 (100%)
Maintenance	2.09	0.61 (85%)	3.45 (59%)
Adjusted	2.09	0.00 (0%)	1.29 (22%)

In the next step, we carried out a life cycle impact assessment (LCIA) for the whole system to enable us to evaluate the importance of the beneficial impact of the avoided mineral fertilizers in relation to the other processes of manure storage and slurry field application. The impact results are shown in Figure 11, and it can be observed that the pattern in Table 5 is repeated in many of the impact categories, where the avoided impact of mineral fertilizer dominates over the impacts from storage and field application.

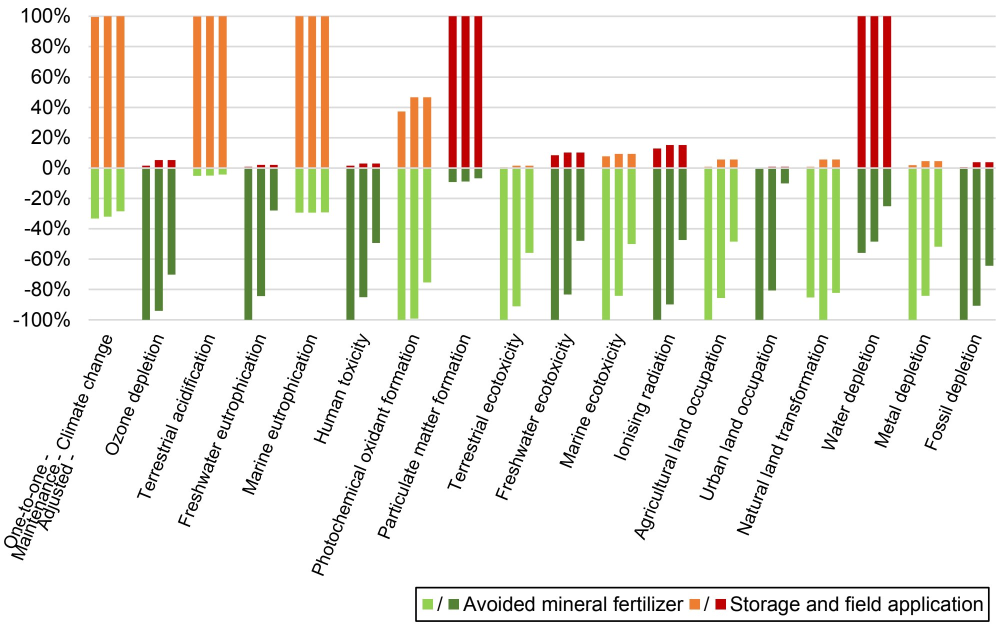


Figure 10. Impacts from avoided mineral fertilizer compared with impacts associated with storage and field application of manure. Impacts are shown as a percentage of the highest absolute value across the three substitution principles for each impact category, presented from left to right: the one-to-one, maintenance, and adjusted principles. Two different colors are used for each of the processes – avoided mineral fertilizer and storage and field application – to increase readability.

Lastly, we wanted to see the effect on the inventory of avoided mineral fertilizer using four different sensitivity scenarios in addition to the case study. They are:

- "No reg": No regulation of application rates, high soil P and K levels as in the case study.
- "Nitrate": Limitation of manure N application rate according to the European Nitrate Directive, high soil P and K levels as in the case study.
- "Case study, low": Assuming low soil P and K levels as opposed to the high soil P and K levels assumed in the case study.
- "Nitrate, low": Like the "Nitrate" scenario, but with low soil P and K levels (see Paper V for more details on the scenarios).

Together with the case study, the scenarios showed that limiting the application rate beyond just applying MFE nutrients according to crop requirement led to a larger application area and better utilization of the MFE nutrients (Figure 12). The application area associated with the case study and the sensitivity scenarios was, in ascending order: No reg (0.0078 ha) – Case study/Case study, low (0.021 ha) – Nitrate/Nitrate, low (0.035 ha). In addition, assuming low soil nutrient levels instead of high soil nutrient levels in the case study showed that the adjusted principle replaced more mineral fertilizer than the maintenance principle. This is because it is recommended to apply more P and K fertilizer than that removed through crop offtake when soil nutrient values are low. For high soil P and K levels, both the one-to-one principle and the maintenance principle overestimate the amount of mineral fertilizer an organic fertilizer can replace, while, with low nutrient soil P and K levels, the maintenance principle would potentially underestimate the amount of avoided mineral fertilizer.

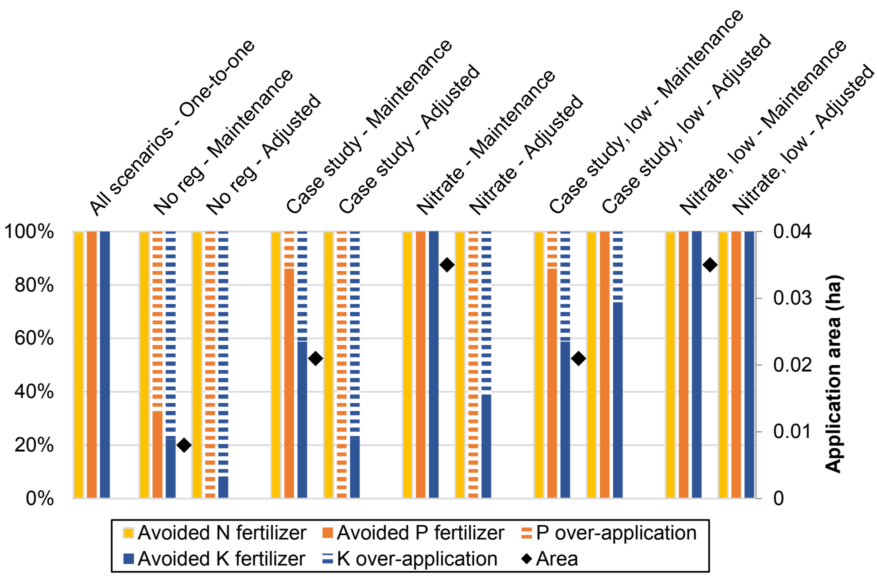


Figure 11. Avoided N, P, and K mineral fertilizer and P and K over-application for the sensitivity scenarios and case study, shown as percentages of the applied MFE nutrients for the three substitution principles. The one-to-one principle is identical across all scenarios and is only shown once. For any given scenario, the application area is the same for the maintenance and adjusted principles. The scenarios are placed in ascending order in terms of application area, first for the scenarios with high soil P and K levels, then for the low soil P and K levels.

4 Discussion

4.1 Main findings in light of the research questions

In the following, I discuss how the findings presented in Chapter 3 help answer the research questions posed in the introductory Chapter 1 and how the findings relate to other literature.

4.1.1 Research question 1

What are the stock and flow characteristics of P and plant-available P in the Norwegian food system and what theoretical potential does secondary P have to satisfy the P fertilizer requirement in the short and long term and, as such, substitute mineral P fertilizer?

The first three papers all contribute to answering this research question by quantifying the stocks and flows of total P and plant-available P in the Norwegian food system. Traits such as great P throughput in aquaculture and fisheries and large P imports and exports through fertilizer production for the global market would make the Norwegian case stand out clearly in a blueprint format for a between-country comparison (Jedelhauser and Binder, 2015).

The findings demonstrate that secondary P has great theoretical potential to cover both short and long-term P requirements for Norwegian crop production. This potential is particularly found in manure and fish sludge, which together constitute around 83% of the estimated amount of plant-available P in secondary resources in Norway. The theoretical potential relies, among other things, on the redistribution of surplus manure P from counties with high livestock densities to counties with a need for P fertilizer import. With such a redistribution, manure alone would be able to satisfy crop P demand in the short and long term.

In a similar way, Metson et al. (2016) estimated that 74% of the fertilizer P demand for U.S. corn production can be met by using secondary P sources from within the same counties as where corn is produced, while the remaining secondary P would need to be transported from other counties. Manure was identified as the largest source of P recycling. Klinglmair et al. (2017) studied the ability of manure, sewage sludge, and composted organic household waste to replace mineral fertilizer P through inter-regional redistribution in Denmark and found that the use of mineral P fertilizer could be reduced by 80%.

In essence, the theoretical potential of P in organic residues to satisfy P fertilizer demand and substitute mineral P fertilizer is an exercise in matching total amounts of secondary plant-available P with total crop P requirements. Some of the organic residues, such as fish sludge and fish scrap, are currently not accessible for recycling as they are lost directly to water recipients, while manure is accessible but requires processing and transport for it to be applied where P fertilizer is needed. These are recycling barriers that mean that the practical substitution potential is lower than the theoretical potential within the current regulatory framework.

A general barrier to P recycling from organic residues is the relatively low cost of mineral P fertilizer compared to the costs of P recycling, and some researchers propose that not even tripling the cost of mineral fertilizer would make secondary P recovery and recycling economically viable (Koppelaar and Weikard, 2013). Instead of waiting for slowly increasing production costs of phosphate rock to eventually spur P recycling on a large scale (Mew, 2016), regulatory measures could be the way forward. As such, the lack of appropriate legislative incentives (e.g., stricter limitation of P fertilizer application) can be considered an important barrier to P recycling and mineral P substitution. Stricter regulation of P fertilizer application would motivate a redistribution of P from areas with P surplus and thereby create opportunities for increased trade in secondary P.

Even within today's regulatory framework, however, there are opportunities for mineral fertilizer substitution that are underexploited. Barriers to this may be uncertainty regarding the fertilizer value of secondary resources, such as manure (Nesme et al., 2011), which may lead to higher mineral fertilizer application than necessary. It is also found to be more challenging for farmers to plan for and use organic fertilizers compared with mineral fertilizer, and this, combined with odor problems from fertilizer application, are important barriers to achieving widespread use of organic fertilizer (Case et al., 2017) and mineral P substitution. Lacking demand for organic fertilizers, the alternative may be non-food uses such as landscaping, which removes P from the food system (see for example Huttunen et al., 2014).

4.1.2 Research question 2

What are the life cycle environmental impacts of technological options for geographic redistribution of secondary P to increase system-wide P use efficiency in Norway, and what are the critical factors and processes in such a P redistribution?

I have attempted to answer this research question in Paper IV. Here, it was found that, of the five technological options studied, anaerobic digestion (AD) of cattle slurry and subsequent solid-liquid separation of the digestate by decanter centrifuge (AD_DC scenario) was the most promising from the perspective of environmental impacts and the amount of P redistributed. This is similar to the findings of ten Hoeve et al. (2014), who found that decanter centrifuge separation could potentially be the best option for P redistribution from pig slurry over 100 km in Denmark.

The transportation of unseparated slurry (NoSep scenario) redistributed the most P, but it also had by far the largest potential impacts on climate change (CC) and fossil resource depletion (FD). Hence, the assessed options for P redistribution imply a trade-off, where we concluded that the transport of bulky untreated manure will be too costly in terms of environmental impacts to legitimize redistribution of the extra P. This complements the picture of unseparated manure being uneconomical to transport over longer distances (Liu et al., 2008; Whalen and Chang, 2001).

A decisive aspect of P redistribution is a situation with a P surplus in one region that motivates transport to another region with a need to import P fertilizer. In Paper IV, this situation of surplus/import need was reflected in very high soil P levels in the donor region and moderately high levels in the recipient region. Assuming optimal soil P levels at both donor and recipient farm in a scenario analysis almost eliminated P over-application at the donor farm and, in that sense, therefore removed the motivation for redistribution. Given this motivation, though, a critical factor for redistribution was the use of solid-liquid separation, which substantially reduced the volume to be transported and therefore the contributions to CC and FD from transport – as opposed to transporting unseparated slurry. The choice of separation technology – screw press or decanter centrifuge – greatly affected the amount of P separated to the solid fraction for transport and redistribution to substitute mineral P, but had less influence on the other impact categories (CC, marine eutrophication (ME), terrestrial acidification (TA), particulate matter formation (PMF), and FD).

Among the most critical parameters in relation to the final impact results were N emissions from application, which dominated the impacts of ME (through losses of nitrate (NO_3)), TA, and PMF (through emissions of ammonia (NH_3)). This is in line with the findings of De Vries et al. (2012). Methane (CH_4) and NH_3 emissions from storage also made important contributions to CC, TA, and PMF. The avoided production and application of mineral fertilizer made important contributions to CC, ME, and FD, but did not dominate impacts in any scenario except for the contribution to the reference impact on FD. This is in contrast to the results of ten Hoeve et al. (2014), and in particular Brockmann et al. (2014), where the benefits of avoided mineral fertilizer had a more prominent influence.

Adding AD as a pretreatment before solid-liquid separation was not important for the redistribution of P as such, but it had a mostly beneficial effect on impacts since it reduced CH_4 emissions from end-product storage and reduced nitrous oxide (N_2O) emissions from field application. The upgrading of produced biogas to green gas and the assumed avoided production of fossil fuel had a particularly beneficial impact on CC and FD. However, the additional mineralization of organic N during AD increased the NH_3 emissions and counteracted the benefits of avoided fossil fuel for TA and PMF.

Finally, the redistribution of P to the recipient region led to avoided mineral P production, but the application of manure on the recipient arable land also had other beneficial effects on the redistribution scenarios (see Figure 10) – in particular in relation to gaseous N emissions. This is explained in Section 3.4.

4.1.3 Research question 3

How will different substitution principles critically influence the LCA inventory and impact results when analyzing the substitution of mineral fertilizers by organic fertilizers in terms of nutrients?

In Paper V, we identified three different principles for the substitution of mineral fertilizer by secondary nutrients, and the findings showed that the choice of principle did have a great influence on the life cycle inventory of avoided mineral fertilizer (Table 5) as well as on the LCIA results (Figure 11). In the following, I will briefly discuss the factors that influence the variation in the inventory of avoided mineral fertilizer, how LCIA results are affected, and, lastly, how we can interpret the estimated mineral fertilizer substitution.

There are three factors that make the identified principles (one-to-one, maintenance, and adjusted maintenance) yield different inventories of avoided mineral fertilizer, namely:

i) application rate,
ii) soil nutrient level, and
iii) nutrient ratios.

The influence of the first two factors is shown in Figure 12, where a linear relationship can be observed between decreasing application rates (increasing application area) and decreasing difference between the principles. The difference, and thus the importance of the principle chosen, also decreased when moving from high to low soil nutrient levels. The third factor, nutrient ratios, has to do with the nutrient ratio in the organic fertilizer compared to the ratio between nutrients required by the crop. For example, organic fertilizers tend to have a lower N:P ratio than that required by crops, which leads to P over-application when fertilizers are applied according to their N content (Withers et al., 2015). This is the reason why the one-to-one principle systematically overestimates the amount of mineral P fertilizer that can be substituted by secondary fertilizers. However, this also means that, if organic residues could be processed to fit the *general* nutrient ratio requirements of local crops, there would be no difference between the one-to-one and the maintenance principle. Similarly, there would be no difference between any of the principles if the resulting secondary fertilizer product was made to fit the crop fertilizer requirement adjusted for soil nutrient levels.

A varying inventory of avoided mineral fertilizer is a necessary, but not sufficient condition for the chosen substitution principle to influence the final LCIA results. In LCIA, the avoided mineral fertilizer also needs to contribute substantially to net impacts relative to other processes in order to make a difference. In the case study in Paper V, twelve out of eighteen impact categories were highly affected by the substitution principle chosen (Figure 11). Had a similar comparative paper been based on a different case study, the influence on the final LCIA results of different substitution principles could have been quite different – even with a similar variation in the inventory of avoided mineral fertilizer.

In Paper V, we estimated the potential amount of mineral fertilizer to be substituted by recycled plant-available nutrients from organic residues, based on an assumption of perfect substitution between the two. The actual displacement of mineral fertilizer is determined in the market as a function of supply and demand dynamics (Geyer et al., 2016). Substitution happens when the farmer goes to the fertilizer market and decides to get (more) secondary fertilizer *and* at the same time get less mineral fertilizer. Depending on how the farmer perceives the fertilizer value and other characteristics of the secondary fertilizer, the actual substitution ratio does not need to be 1:1. A one-to-one substitution ratio between primary and secondary materials can be considered an exception rather than the rule (Geyer et al., 2016). For any substitution principle, the amount of avoided mineral fertilizer estimated in our analysis should therefore be interpreted as a maximum of what we might hope to be displaced by the produced and applied secondary fertilizer.

4.2 Methodological strengths and weaknesses

4.2.1 Strengths of the methods used

Substance flow analysis (SFA) is a powerful analytical tool for investigating flows and stocks of P in an anthropogenic system (Brunner, 2010), and it enables the identification of P "hotspots" (Cordell et al., 2012) in terms of the most important sectors in the food system at the national level, as in Paper I. Another strength of SFA is its ability to quantify flows and stocks on different scales, which enabled the identification of geographical P hotspots in counties in Norway with high amounts of secondary P in the form of manure and sewage sludge production.

Many national SFA studies on P have not addressed the issue of P plant-availability as a barrier to secondary P recycling and they therefore risk overestimating the secondary P fertilizer potential (Paper III). The integration of plant-availability of P with the SFA methodology has resulted in a more realistic estimate of the fertilizer potential of important secondary resources such as manure, fish sludge, meat bone meal, and sewage sludge.

Furthermore, the adjustment of the P fertilizer requirement to soil P levels in both SFA and LCA studies in this work has provided a more realistic approximation of the actual P fertilizer demand for crop production. This is in line with Rowe et al. (2016) and Withers et al. (2015). For SFA, the adjustment is a way of showing that the short-term potential of secondary P to cover P fertilizer demand may be quite different from the longer-term potential.

In the LCA studies, a strength has been the transparent use of parameters and the inclusion of calculation procedures to arrive at the different estimates in the inventory phase. These procedures are thoroughly described in the Supplementary Material supporting the papers. Furthermore, scenario analyses have been used systematically to identify critical processes, parameters, and promising technology options, as shown in Paper IV.

The LCA study in Paper IV accounted for P over-application (POA) used as a separate relevant impact category (together with avoided mineral P (AMP)) in parallel with the midpoint LCA ReCiPe indicators. The strength of the POA-category is the avoidance of uncertain downstream impacts (explained in the following), the indication of potentially problematic soil P accumulation, and its ability to relate directly to the FU. It also reveals inefficient use of P fertilizer. Over-application of P over time causes soil P accumulation that increases the risk of P losses through erosion, but the relationship between a positive soil P balance (i.e., POA) and P losses is not clear in the short term (Bechmann, 2014). The risk of P losses depends more on factors such as soil P status, topography, hydrologic activity, soil management, and the use of cover crops and other mitigation measures against erosion (Kleinman et al., 2011; Maguire et al., 2005). The potential impact on freshwater eutrophication of lost P is also subject to great regional variation (Helmes et al., 2012). Thus, we found it difficult to estimate P losses based on a single event of fertilizer application, and equally difficult to use the freshwater eutrophication category, and we opted for POA as a more appropriate indicator.

In Paper V, we demonstrated the critical importance of using appropriate substitution principles in LCA when studying systems for the recycling of secondary nutrients to displace mineral fertilizer nutrients. The findings and proposed recommendations could lead to greater transparency and comparability in similar studies in the future, and possibly incentivize increased P use efficiency.

4.2.2 Weaknesses and shortcomings

A weakness in Papers II and III is the assumption that the data on soil P levels are representative for all cropland where grass, cereals, oilseeds, green fodder, and silage are produced. These are the crops for which the correction of P requirement is carried out, in accordance with Krogstad et al. (2008). The mentioned crops were grown on 95.5% of the total cultivated land in Norway during the period studied, 2009–2011. I therefore assumed that the vast majority of the soil samples were taken at farms where these crops were grown, since the data on soil P levels are not associated with specific production systems.

Another weakness of Paper II is the equation between the concept of P surplus fertilization in a county and the amount of exportable secondary P surplus from a county. An underlying assumption for equating the two is that all manure P from grazing animals is deposited where P fertilizer is needed. This may be a questionable assumption, since the decision to use a field for grazing may not be motivated by the field's need for fertilizer, and if it does not hold, in whole or in part, the surplus fertilization overestimates the amount of exportable surplus P.

In Paper IV, we employed an input unit-related functional unit (FU) to assess the redistribution of manure P. In the context of bioenergy systems, Cherubini and Strømman (2011) point out that the choice of FU may influence the interpretation of the final results, and that several FUs should be used to show results. The reason for this is that evaluating alternatives from different angles, by using different FUs could highlight characteristics that rank the alternatives differently. In the same way, an alternative FU could have been tested in Paper IV, e.g., the redistribution of 1 kg P. To be able to redistribute 1 kg P the different scenarios would have to scale differently the amount of fresh manure to be managed. This could have affected the ranking of the scenarios in the different impact categories.

In the LCA studies in Papers IV and V, it is a shortcoming that we did not collect information about some substances present in manure that could be beneficial (other macro- and micro-nutrients) as well as unwanted (such as antibiotics and heavy metals). These additional characteristics are not commonly included in LCAs on manure management, but they would have provided a richer picture of the impacts, both beneficial and negative, of manure as fertilizer.

The field application of manure in Paper V highlights a shortcoming concerning the actual boundary conditions of the receiving farmland. The amount of slurry in the FU is rather small and we can safely assume that there is enough available land (0.021 – 0.035 ha) to spread the slurry on farmland. However, this assumption might not hold if, instead, the FU had been set to represent the total farm or regional production of a certain organic fertilizer: we might then have seen that the required application area is greater than the area actually available at the farm or in the region. This is analogous to the assumption highlighted as problematic by Ekvall and Finnveden (2001), namely that there is always a market ready to absorb any increase in recycled material. The lack of integrating limits of receiving compartments – whether a market or agricultural land – is therefore a shortcoming because it inadequately reflects reality. The consequences of surpassing receiving capacity for recycled materials should be dealt with by expanding the system (Ekvall and Finnveden, 2001). In the context of the current case study, the need for more application area than that available at the farm could require an expansion of the system to include the transport of organic fertilizer surplus and application on external farmland (see for example De Vries et al., 2012). The fraction of the total production that needs external application could then also be used for a smaller FU.

4.3 Implications of this work

There are some policy implications to be drawn from the studies that make up this thesis that could improve P management and P use efficiency related to the Norwegian food system over time. I also present two possible avenues for further research that could build on this thesis.

4.3.1 Policy and practice

Decision-support tools for P management

SFA and LCA have proved to be useful system-wide analytical tools for this thesis, and insight based on their use can inform decision makers when prioritizing and designing measures for improving national P management. The use of SFA is crucial in order to map P flows in the food system and to highlight hotspots that show particular promise as low-hanging fruit for P recycling, both sector-wise and geographically. The multi-regional SFA is a first step towards developing an understanding of where, geographically, these hotspots can be found. It gives useful pointers for potential “export”-counties with surpluses of secondary P, but the resolution at the county level is too coarse to form the basis for internal redistribution of secondary P to satisfy intra-regional P fertilizer requirements. Improving the spatial resolution down to farm level should be part of a collaborative effort between different actors, including the research community. Further, given the information about where there is surplus P and where it is actually needed, adjusted for residual soil P levels, the use of LCA will give indications about which technologies and infrastructure should be prioritized to realize the secondary P fertilizer potential, while at the same time minimizing the negative consequences for the environment.

Measures to support improved P management in Norway

With the findings on, in particular, the secondary P fertilizer potential in Norway produced by this research and the research of Helen Hamilton and Eva Brod (2016), there seems to be a good enough basis to start implementing measures to realize this potential. Even though this research has not looked into possible measures and their effectiveness, I will attempt to outline some actions or processes that I believe will take P management in Norway in the right direction. First, it appears that regulations on the application of P fertilizer need to be adapted to provide appropriate disincentives in order to avoid P fertilizer over-application (the sum of mineral and secondary P). An adaptation of the regulations could be combined with positive economic incentives to compensate some of the expected extra costs involved for farmers, especially related to the processing and redistribution of manure P.

Second, national management of P involves a wide range of different actors, and bringing them together will be important for setting a well-grounded and consensus-based agenda for how to proceed to improve national P management. There are examples of organized forums in other countries, termed nutrient platforms or, more specifically, phosphorus platforms, that are well under way with constructive dialogue. Key actors include farmers' organizations, waste and wastewater associations, authorities, industry, and the research community.

Multiple processes need to take place to achieve improved national P management. In addition to revising current legislation, there is also a need for innovation (e.g., the development and design of secondary fertilizer products adapted to farmers' needs and preferences), as well as further research to evaluate new technologies and support P management measures. Some areas of further research based on the current thesis work are outlined in Section 4.3.2 below.

P is part of N-P-K

Phosphorus needs to be looked at in conjunction with other main nutrients in crop production, especially N and K, that are also present in secondary organic resources. For example, the recycling of secondary N and the associated avoided mineral N production can have important environmental benefits for the system in terms of saved energy use. Moreover, new manure management, such as the solid-liquid separation scenarios studied in Paper IV, could reduce the emissions of manure N, increase the amount of plant-available N remaining in the soil, and thereby further reduce the need for mineral N input. In general, focusing only on the P in secondary resources entails a risk of missing out on important environmental benefits associated with also recovering and recycling other nutrients.

4.3.2 Further research

From the current work, I see two different and particularly interesting avenues for further research: i) improve our understanding of the environmental effects of secondary P processing and redistribution on a greater scale; and ii) improve our understanding of substitutability between secondary fertilizer products and mineral fertilizers.

i) The LCA studies in Papers IV and V are based on the management of a functional unit that represents a small fraction of the actual farm production of cattle manure during a year. The challenges concerning the missing integration of boundary conditions are discussed in Section 4.2.2 above. To realize improved P management at the national level, predictions about the system-wide consequences of different management scenarios could serve as important decision-support. This calls for studies that visualize the boundary conditions for required fertilizer nutrients for crop production at a finer spatial resolution (for example at farm or municipal level) combined with management scenarios for different processing technologies and logistical solutions. To this end, a combination of SFA and LCA could be particularly useful in order to study how scenarios affect flows of P and other secondary nutrients, as well as the environmental impacts associated with processing and logistics. This has been done earlier, for example in the context of urban wastewater management (Venkatesh et al., 2009). The continued integration of assessments of plant-availability of secondary nutrients will then be key to identifying how scenarios fulfill the overall goal of redistributing secondary plant-available nutrients from where they are generated to where they are needed.

ii) As discussed in Paper V, substitutability is not only a matter of technical functional equivalence between a secondary product and the primary product it is assumed to replace, but something that takes place in the fertilizer market (Geyer et al., 2016). We still lack knowledge about how potential users of a secondary fertilizer view its quality and applicability on their farms compared to the mineral fertilizer products they are familiar with, also in terms of price. Simply put, we need to know whether there is a market to absorb new secondary fertilizer products and what effect this has on sales of mineral fertilizer. Geyer et al. (2016, p.1011) put it this way: "There is no engineering relationship or law of physics that requires primary production to decrease as recycling increases." Thus, without anything more to base the substitution potential of secondary nutrients on than their fertilizer value, we risk overestimating their potential. The research on substitutability through the market and in the perspective of user preferences can build on the framework for accounting for product displacement in LCA proposed by Vadenbo et al. (2016).

5 Conclusions

The main contributions to existing knowledge are listed below, followed by some concluding remarks.

- This work demonstrates that there are great potentials for improving the management of phosphorus (P) in the agricultural food production system in Norway by using secondary P sources to replace mineral P fertilizer.
- Together with fish sludge, manure was identified as the most important source of secondary P in Norway, and the most readily available.
- The integration of P plant-availability with substance flow analysis (SFA) gives a more realistic picture of the total fertilizer value of secondary P sources such as animal manure, fish sludge, food waste, and sewage sludge at the regional and national level.
- Adjusting the P fertilizer requirement at the regional and national level according to soil P levels provides a more correct picture of the theoretical potential of secondary P to cover P fertilizer requirement and to substitute the use of mineral P fertilizer. Due to overall high soil P levels in Norway, the short-term P fertilizer requirement at the national level is substantially lower than the longer-term requirement. This assumes that P application rates are reduced accordingly in the short term and that soil P levels, as an effect, approach the optimal range over time.
- Manure P alone may be sufficient to satisfy national crop P requirements in the short as well as the long term if redistributed well both intra- and inter-regionally at county level.
- A life cycle assessment (LCA) of the processing and transport of dairy cattle slurry over 500 km indicated that geographical redistribution of secondary P does not necessarily need to imply increased life cycle environmental impacts. It also highlighted opportunities to improve the use efficiency of other nutrients, such as nitrogen.
- It was found that the choice of substitution principle used in LCA to estimate the amount of potentially avoided mineral fertilizer when secondary fertilizers are applied can greatly affect the inventory of avoided mineral fertilizer and the final environmental impact results. It is therefore recommended to at least state and justify the substitution principle used in LCA in order to increase transparency and comparability with other studies.

The overall aim of this work has been to contribute to improved P management and P use efficiency in Norwegian agricultural crop production – through increased understanding of the potentials for secondary P recycling. I believe that important insight has been gained into the great potential that lies in using organic residues as secondary fertilizer to replace mineral P fertilizer, and into the effects of how we estimate this substitution. I hope that this work can serve to motivate and inform both action and further research to help realize the still untapped recycling potential. This, in turn, could over time lead to reduced dependence on imported mineral P fertilizer for domestic food production, a reduction in P losses, and reduced demand for the global, non-renewable, and vital resource that phosphate rock is.

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Paper I

Investigating cross-sectoral synergies through integrated aquaculture, fisheries, and agriculture phosphorus assessments: A case study of Norway

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Investigating Cross-Sectoral Synergies through Integrated Aquaculture, Fisheries, and Agriculture Phosphorus Assessments

A Case Study of Norway

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Supporting information is available on the *JIE* Web site

Summary

Future phosphorus (P) scarcity and eutrophication risks demonstrate the need for systems-wide P assessments. Despite the projected drastic increase in world-wide fish production, P studies have yet to include the aquaculture and fisheries sectors, thus eliminating the possibility of assessing their relative importance and identifying opportunities for recycling. Using Norway as a case, this study presents the results of a current-status integrated fisheries, aquaculture, and agriculture P flow analysis and identifies current sectoral linkages as well as potential cross-sectoral synergies where P use can be optimized. A scenario was developed to shed light on how the projected 2050 fivefold Norwegian aquaculture growth will likely affect P demand and secondary P resources. The results indicate that, contrary to most other countries where agriculture dominates, in Norway, aquaculture and agriculture drive P consumption and losses at similar levels and secondary P recycling, both intra- and cross-sectorally, is far from optimized. The scenario results suggest that the projected aquaculture growth will make the Norwegian aquaculture sector approximately 4 times as P intensive as compared to agriculture, in terms of both imported P and losses. This will create not only future environmental challenges, but also opportunities for cross-sectoral P recycling that could help alleviate the mineral P demands of agriculture. Near-term policy measures should focus on utilizing domestic fish scrap for animal husbandry and/or fish feed production. Long-term efforts should focus on improving technology and environmental systems analysis methods to enable P recovery from aquaculture production and manure distribution in animal husbandry.

Introduction

The linearity of anthropogenic phosphorus (P) flows, from extraction to consumption to waste with limited recycling, makes the current use of P unsustainable. Because P is both a finite resource and a pollutant, the need for improving P management has been deemed as critical (Cordell et al. 2009). In order to do so, a systems-wide understanding is necessary given that P is metabolized across both natural and anthropogenic processes in a variety of sectors. Potentials for reducing P losses and increasing recycling must be identified through a holistic approach, encompassing all processes of the anthropogenic P cycle.

Thus far, data gaps limit our ability to accurately model P flows in the aquaculture, fisheries, and agriculture sectors together on a country scale. The aquaculture and fisheries sectors,

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today, are significant drivers for anthropogenic P use, and this will only increase considering the projected large growth in aquaculture and their dependency on fisheries and agriculture for feed ingredients (Subasinghe 2005; Troell et al. 2009; Abreu et al. 2011). Such increases are likely to shift resource cycles on a global level; however, the implications will be particularly far reaching in countries with large aquaculture sectors. Cordell and colleagues (2013) have noted the particular importance of aquaculture and fisheries and also highlight the problem of data gaps within the aforementioned sectors. Matsubae-Yokoyama and colleagues (2009) have modeled fisheries as a P input to the food system in Japan; however, the implications of the rapidly growing aquaculture sector have not been explored. Because these sectors can be linked, primarily through the production of feed and secondary fertilizers, we use a systems approach and quantify the sources, sinks, and inefficiencies of P metabolism within our case study to identify the most effective strategies for reducing the dependency on finite primary P resources and losses to the environment.

Scope and Research Questions

In order to identify the largest potentials for improving P management, a P balance was conducted on a country that exemplifies the need for integrated fisheries, aquaculture, and agricultural assessments. Norway has large, export-oriented aquaculture and fisheries industries, whereas Norwegian agriculture predominantly serves the national market with a 45% self-sufficiency level in 2010 (NILF 2011). In terms of size, however, Norwegian aquaculture is especially significant. In 2011, Norway was the largest per capita aquaculture producer in the world with 0.23 tonnes of fish and crustaceans per capita and ranked sixth in total production quantities (FAO 2011; SSB 2014). In addition, production in this sector is expected to increase 5 times by 2050, necessitating a corresponding quintupling of fish feed production (DKNVS and NTVA 2012). Because Norwegian fish feed today contains roughly 52% imported agricultural products, it is clear that the agricultural and aquaculture sectors are, to a large degree, interdependent (Pettersen 2013; SSB 2014; EWOS 2010). Sectoral linkages such as this exemplify the need for integrated studies given that P management cannot be optimized if the major drivers are excluded and synergies remain unexplored.

In this article, we aim to, for the first time, integrate the aquaculture, fisheries, and agricultural sectors in a P balance model of Norway in order to address the following questions:

1. What are the current P cycles in aquaculture, fisheries, and agriculture and how are they linked? What is the current state of secondary P recycling (i.e., the use of by-products or waste as secondary P sources)?
2. What are the projected changes and how could this affect P cycles given the current linkages? Specifically, how does this affect waste streams and thus potential future secondary P sources? What challenges could arise?
3. What are the options and opportunities for systems-wide integration of P management?

Methods

To answer the aforementioned questions, a substance flow analysis (SFA) was performed on the Norwegian P system using the free-ware material flow analysis software, STAN, to visualize the results. The system was defined for the economic zone of Norway, including coastal areas where aquaculture production occurs and the marine waters where Norwegian fisheries operate. In an attempt to avoid annual variations, averaged data from 2009 to 2011 were used. The following paragraphs include a further explanation of the key processes and the definitions thereof. For more information regarding the remaining processes, refer to the supporting information available on the Journal's website.

Agricultural and Greening Soil

Agricultural soil refers to permanent grassland that is used for fodder/grazing and arable land that is used for the production of cereals, potatoes, oil seeds, legumes, vegetables, fruits, and fodder. Greening soil includes parks, covering for landfills, gardens, and other areas where mineral and organic fertilizer are utilized, excluding agricultural land. Discharge from soils and the net addition to soil P stocks were included in this process. Forestry soils were omitted because the anthropogenic influence on the P flows in forest systems was determined to be negligible relative to the overall system.

Aquaculture

This process includes all of the fish production taking place within the aquaculture sector, including coastal nets. In Norway, marine aquaculture, or fish farming under controlled conditions inside of cages placed in marine environments, is predominant. Fish farms are typically open-cage systems, where effluent water is continuously exchanged with surrounding waters (Wang et al. 2012; Troell and Norberg 1998). This results in the loss of large amounts of nutrients in a variety of forms. Dissolved inorganic P is released through excretion, particulate organic P is lost through defecation and uneaten feed, and dissolved organic P occurs through the dissolution of particulate P (Wang et al. 2012; Troell and Norberg 1998).

Fisheries and Fish Processing

This process includes all marine fish and shellfish caught by Norwegian vessels and all fish caught by foreign vessels and landed in Norway. Wild fish processing was modeled to include both land-based processing and processing at sea. By-product utilization was included in this analysis including by-products dumped at sea and processed for recycling into new products. Sport fishing was estimated to be negligible.

Fish Feed Production

Fish feed refers to feed for fish farmed in aquaculture for human consumption. Fish feed production includes both domestic and imported raw feed ingredients and prepared imported fish feed. Over time, the composition of fish feed has changed dramatically to favor the use of more plant ingredients owing to the high costs of fish products (Sørensen et al. 2011). In 2013, however, fish feed consisted of 67% plant-based products, including plant protein, rapeseed oil and plant-based binders, and 33% fish-based products, primarily fish oil and fish meal (Ytrestøyl et al. 2014b).

Fertilizer Production

Mineral fertilizer and organic fertilizer production are included in this process. Mineral fertilizer refers to the fertilizer produced from imported phosphates. Organic fertilizer includes domestically produced meat bone meal, sewage sludge, and food waste.

Agriculture

The major animal groups and plant products produced in Norway are included in this process: cereals, potatoes, oil seeds, legumes, vegetables fruits, fodder, dairy and cattle, pigs, sheep and goats, poultry, horses, and fur animals. Imported live animals were estimated to result in small P flows and were therefore not considered. A large amount of sheep, some cattle, horses, and goats graze in forest pastures in the summer. Excreted P in forest pastures was considered to leave the system boundaries, whereas assimilated P in animals during the summer was assumed to be negligible.

Processing, Retail and Consumption

All domestic animal/plant processing and trade is included in this process. Losses during food processing, retail, wholesale, and human consumption are also included. Imported food is assumed to go directly to consumption without an intermediate processing phase.

Data Sources and Quantification

Tables 1and 2 include flow and stock descriptions, equations, and their respective data sources. Data were primarily collected from government statistics, reports, company publications, expert interviews, and scientific publications. Mass balance was utilized in cases of poor data availability. Otherwise, each flow was individually calculated. The net accumulation of P in stocks, including soil stocks, off-site wastewater treatment stocks, and construction and landfill stocks, was calculated by mass balance. Mass balance inconsistencies, designated MBI as seen in figure 1, are marked in red and are given as the total input minus the total output for each process. This was done to allow inconsistencies to be assessed in relation to the size of the relevant process flows.

Data for mineral fertilizer production was obtained from YARA (Nyhus 2013), a Norwegian fertilizer company representing 85% to 90% of the Norwegian market share for mineral fertilizer sales (Mattilsynet 2013). The wastewater calculations were based on a set of assumptions related to the wastewater treatment efficiencies and the part of the P load transported to the wastewater treatment plant. Further information related to this can be found in the Supporting Information on the Web. Additionally, because the process efficiency calculations cannot be calculated through the results presented, detailed explanations and the calculation methods can be found in the Supporting Information on the Web.

Scenario Development

Given the projected growth in aquaculture production, a scenario was developed to shed light on how such production increases could propagate throughout up- and downstream processes. The scenario was based on a fivefold increase in production by 2050, a Norwegian population of 6.6 million, and no growth in the fisheries industry (SSB 2014). Today's technologies were used by scaling up data and holding the transfer coefficients and system structure from the 2009–2011 model constant. Because this estimate does not consider changes within many of the variable drivers, the purpose of this scenario was not to describe the most likely situation in 2050. Rather, the aim was to highlight potential challenges for future P management in Norway. This scenario can be used to inform a more refined analysis needed to direct future policies both within and across the different sectors. The aforementioned mass balance inconsistencies were not visualized for this scenario because the relative error remained the same, given that technologies were held constant.

Results

The P balance, figure 1, indicates the major imports, exports, flows, sinks, and losses of P within the Norwegian system. The results from the 2050 scenario can be seen in figure 2.

Overall, we determined that Norway is a net importer of P (total imported goods – total exported goods), with an average net import of 30,000 tonnes of P per year (P/yr) or 6.2 kilograms of P per capita per year from the recent period 2009–2011. By a substantial margin, the largest P flows were represented by the import of rock phosphates for fertilizer production with the subsequent export of most of the mineral P fertilizers for use in other countries. As shown in figure 1, P flows caused by aquaculture and agriculture drive P consumption and losses at similar levels. Comparing similar flows between the sectors, domestic use of mineral fertilizer based on imported phosphates in soils (8,400 tonnes P/yr) and imported fish feedstuff (9,400 tonnes P/yr) are similar, indicating that both sectors are equally

Table 1 Flow desciptions

Flow origin and destination	*Flow name*	*Flow description*
0,1	Imported rock phosphates	Quantity of P in imported rock phosphate for the production of mineral fertilizer
0,2	Atmospheric deposition	Quantity of P to agricultural land through atmospheric deposition
0,4	Imported husbandry feedstuff	Quantity of P in imported products and phosphates for husbandry feed produced in Norway
0,5	Caught fish	Quantity of P in fish and crustaceans, except mammals, caught by Norwegian vessels
0,6	Imported fish feedstuff	Quantity of P in imported fish feed and feedstuff used for meal and fish feed production. P content based on country of origin
0,8	Imported food	Quantity of P in imported food products, including pet food
1,0	Exported fertilizer	Quantity of P in exported mineral fertilizer
1,2a	Organic fertilizer	Quantify of P in sludge, meat bone meal, and food waste used as fertilizer
1,2b	Mineral fertilizer	Quantity of P in mineral fertilizer applied to agricultural and greening soil
2,0	Runoff	Quantity of P lost from agricultural land to water bodies through the processes of diffusive, point source, and background runoff
2,3	Plant uptake, grazing, and green fodder	Quantity of P taken up by agricultural crops without plant residues, which are assumed to stay on the field. Reference year 1/7–30/6 and the P in green fodder, silage, hay, and pasture. Calculation of P in silage, hay, and pasture is based on: 1 FeM (feed unit) = 1.176 kg DM grass, 2.6 kg P/ton DM grass.
3,0	Manure to forest	Quantity of P excreted in the forest during summer.
3,2a	Manure	Quantity of P excreted by animals in 1 year. Corrected for P excreted by animals grazing in the forest. All manure is assumed to be returned to agricultural land.
3,2b	Seeds and planting potatoes	Quantity of P applied to agricultural land in cereal seeds and planting potatoes. P in grass, vegetable, and fruit seeds is assumed to be negligible. Reference year 1/7–30/6.
3,4	Plant products for husbandry feed	Quantity of P in domestically produced cereals, oil seeds, and legumes for feed
3,8a	Plant products for human consumption	Quantity of P in harvested crops for human consumption. All vegetables, potatoes, and fruits produced are assumed to be used for human consumption.
3,8b	Husbandry products	Quantity of P in milk and eggs produced and living animals leaving the farm to be slaughtered
4,3	Husbandry and fur feed	Quantity of P in husbandry and fur feed
4,8	Pet food	Quantity of P in pet food
5,0	Dumped scrap	Quantity of P in fish scrap that is discarded at sea
5,6	Wild whole fish and scrap	Quantity of P in wild fish and fish scrap from fisheries used for fish feed production
5,8	Processed wild fish, crustaceans, and scrap	Quantity of P in caught fish, fish scraps, and crustaceans
6,0	Exported wild fish feedstuff	Quantity of P in exported fish feed and feedstuff from fisheries fish
6,4	Fish meal and silage	Quantity of P in fish meal and silage used for husbandry feed
6,7	Fish feed	Quantity of P in fish feed used in aquaculture
7,0a	Escaped fish	Quantity of P in escaped and other lost fish from aquaculture
7,0b	Fish excrements and feed losses	Quantity of P in lost fish feed, feces, and excretion from aquaculture including onshore hatcheries
7,8	Farmed fish	Quantity of P in farmed fish from aquaculture, including dead fish
8,0a	Exported fish products	Quantity of P in exported wild fish products, farmed fish products, and crustaceans from Norway
8,0b	Exported fish scrap products	Quantity of P in exported wild fish scrap and farmed fish scrap
8,0c	Exported food	Quantity of P in exported domestically produced meat, milk, eggs, cereals, vegetables, potatoes, and fruits
8,6	Salmon scrap	Quantity of P in salmon scrap used for fish meal and silage for husbandry feed
8,9a	Processing and retail waste	Quantity of P in food processing (animal slaughtering and vegetables), wholesale, and retail waste

(*Continued*)

Table 1 Continued

Flow origin and destination	Flow name	Flow description
8,9b	Wastewater	Quantity of P from human waste (urine + feces) and municipal wastewater, including wastewater to separate wastewater treatment for <50 person equivalents
8,9c	Municipal solid waste	Quantity of P in food waste generated by households and the service sector
9,0a	Discharge	Quantity of P discharged to water from wastewater treatment
9,0b	Exported meat and bone meal	Quantity of P in exported waste, including meat bone meal, food waste, and fish silage for export
9,1	Waste-based fertilizer	Quantity of P in fertilizer and soil amendment products derived from sludge, meat bone meal, and food waste
9,4	Meat bone meal for husbandry feed	Quantity of P in meat bone meal used for husbandry feed
9,10	Waste for incineration, landfill, and construction	Quantity of P in waste incinerated or directly landfilled

Note: Flow origin and destination refers to the process number in which the flow originates and ends. For example, 6,7 refers to the flow originating from process 6 and ending in process 7.
P = phosphorous; FeM = feed unit for milk production; kg P/tonne = kilograms of phosphorous per tonne; DM = dry matter.

Table 2 Flow origin and destination, flow equations, and sources for both organic matter and P contents (Pc)

Flow origin and destination	Equation	Material quantity sources	P contents sources
0,1	Personal communication with O. Nyhus (2013)	1	
0,2	Agricultural area × rate of atmospheric P deposition per area	2	3
0,4	Imported products × Pc + (Husbandry feed × Pc – raw products used for husbandry feed × Pc)	4	5, 6, 7
0,5	All fish and crustaceans caught by Norwegian vessels × Pc + Fish caught in rivers × Pc	2	8,9
0,6	Imported meal × Pc + Imported fish feed × Pc + Imported fish for fish feed production × Pc + Imported milled fish products for feed × Pc + Imported agriculture products × Pc	2, 10, 11	2, 12, 13
0,8	Sum of imported food products × Pc	2	6, 14, 15, 16
1,0	Personal communication with O. Nyhus (2013)	1, 25	
1,2a	See 9,1	2, 17, 18, 19, 20, 21, 22	23, 24
1,2b	Mineral fertilizer applied to agriculture soil × Pc + Educated estimate	1, 25	
2,0	Agricultural area × rate of diffusive, point source, and background runoff	26	
2,3	Cereal, potato, oil seed, legumes, and vegetable and fruit yields × Pc + Green fodder silage, hay, and pasture × Pc	2, 27, 28	6, 15, 29, 30
3,0	Number of animals in the forest × time in the forest × P excreted per animal	2, 33	33, 34
3,2a	Number of animals × P excreted per animal – number of animals in the forest × time in the forest × P excreted per animal	2, 33	33, 34
3,2b	Cereal, oil seed, legume seeds, and planting potatoes × Pc	27, 31	6
3,4	Cereals, oil seeds, and legumes produced used for husbandry feed production × Pc	4	6
3,8a	Cereals, vegetables, potatoes, and fruit produced used for human consumption × Pc	2, 27	6
3,8b	Cow milk, goat milk, and egg × Pc + (Number of slaughtered animals × slaughter weight/dressing percentage) × Pc + (Number of fur produced × weight fur animals) × Pc	2, 17, 18, 19 28, 32, 35,36	6, 15, 16
4,8	Pet food × Pc	7, 38, 39	50
4,3	Husbandry feed × Pc + fur feed × Pc	4, 7, 38,39	40, 51
5,0	Dumped fish scrap × Pc	2	8, 9
5,6	Fish for fish feed production × Pc + Fish scrap for fish feed production × Pc	2, 41, 42, 43	8, 9, 44

(*Continued*)

Table 2 Continued

Flow origin and destination	*Equation*	*Material quantity sources*	*P contents sources*
5,8	(Landed fish and shellfish for human consumption × Pc – Fish scrap × Pc) + (Fish caught in rivers × Pc)	2, 41, 42, 43	8, 9, 13, 44
6,0	Wild fish meal × Pc + Whole forage fish × Pc + Fish feed × Pc	2	8, 12, 13
6,4	Wild fish meal for husbandry and fur feed × Pc + Wild fish silage for husbandry and fur feed × Pc + Salmon meal for husbandry and fur feed × Pc + Salmon silage for husbandry and fur feed × Pc	2, 4, 41, 42, 43	8, 9, 11, 12, 13,
6,7	Fish feed for aquaculture × Pc	13	13
7,0a	Total amount of escaped fish × average estimated weight × Pc	2, 42, 43	13
7,0b	Total amount of fish feed fed × fish feed loss rate × Pc + Total amount of P in fish feed eaten × ratio lost owing to excretion and feces	2, 41, 42, 43	45, 46, 47, 48
7,8	Total farmed fish produced × Pc + Total quantity of dead fish × Pc	2	13
8,0a	Total exported farmed salmon, herring, pelagic fish, and white fish products × Pc + Total exported crustaceans × Pc	2	9, 13, 44, 47
8,0b	Total exported farmed salmon, herring, white fish, and pelagic fish silage × Pc + Total exported fish scraps × Pc	41, 42, 43	8, 9, 13
8,0c	Amount of exported domestically produced food × Pc	2	6, 14, 15, 16
8,6	Salmon scrap and silage for husbandry × Pc	4, 41, 42, 43	13
8,9a	Quantity of meat bone meal × Pc + Vegetable processing waste × Pc + Wholesale food waste × Pc + Retail food waste × Pc	17, 18, 19, 20,52	6,23
8,9b	Quantity of discharged P to water / (1 – treatment efficiency) + Population connected to separate wastewater treatment × Quantity of P generated per capita – Quantity of P in septic tank content	22	
8,9c	Quantity of organic household waste × Pc + Quantity of organic service waste × Pc	2	24
9,0a	Quantity of discharged P to water	22	
9,0b	Quantity of exported meat bone meal × Pc + Quantity of exported kitchen and canteen waste × Pc + Quantity of exported mixed household waste × Fraction of organic waste in mixed waste × Pc	2, 17, 18, 19, 20, 21, 50	23, 24
9,1	Wastewater – discharge × Fraction of sludge to agriculture, greening, and other use + Quantity of meat bone meal for agriculture and greening × Pc + Quantity of biologically treated municipal waste × Pc	2, 17, 18, 19, 20, 22	23, 24
9,4	Quantity of meat bone meal for husbandry feed × Pc	17, 18, 19, 20	23
9,10	Wastewater – discharge × Fraction of sludge to landfill + Quantity of meat bone meal to incineration × Pc + Quantity of household waste to incineration × Fraction of organic waste in mixed waste × Pc + Quantity of landfilled organic waste × Pc	2, 17, 18, 19, 20, 22	23, 24

Sources: [1](Nyhus 2013); [2](SSB 2014); [3](Oredalen 2000); [4](Norwegian Agriculture Authority 2011); [5](Lentner and Wink 1981); [6](Norwegian Food Composition Database 2014); [7](Mattilsynet 2010aa); [8](Hjerne and Hansson 2002); [9](Czamanski et al. 2011); [10](Norwegian Seafood Federation 2013); [11](Ytrestøyl et al. 2014a); [12](FAO Fisheries Department 1986); [13](Ytrestøyl et al. 2011); [14](USDA and ARS 2013); [15](Antikainen et al. 2005); [16](IFP 2006); [17](Animalia 2010); [18](Animalia 2011); [19](Animalia 2012); [20](Viste 2010); [21](Ahmed 2013); [22](Berge and Mellem 2012); [23](Norsk Protein 2013); [24](Møller et al. 2012); [25](Mattilsynet 2013); [26](Eggestad et al. 2001); [27](Breen 2013); [28](Norwegian Agricultural Economics Research Institute 2011); [29](Bakken et al. 2014); [30](Johansen et al. 2003); [31](Glorvigen 2013); [32](Heimberg 2014); [33](Karlengen et al. 2012); [34](Poulsen 2012); [35](Avdem 2013); [36](Bryhn 2013); [37](IFP 2006); [38](Mattilsynet 2009); [39](Mattilsynet 2011); [40](Ahlstrøm 2013); [41](RUBIN 2009); [42](RUBIN 2010); [43](RUBIN 2011); [44](National Institute of Nutrition and Seafood Research 2007); [45](Lall 1991); [46](Wang et al. 2012); [47](Reid et al. 2009); [48](Bergheim and Braaten 2007); [49](Rørtveit and Nerland 2013); [50](Raadal et al. 2008); [51](Mattilsynet 2010bb); [52](Hanssen and Schakenda 2011)

reliant on imports for production. Additionally, total fish feed consumption (12,000 tonnes P/yr) and husbandry feed, fodder, and grazing in agriculture (17,400 tonnes P/yr) are in the same order of magnitude. In terms of losses and potentially available secondary P sources, fish excrements and feed losses are very comparable to the net accumulation of P in soil stocks for plant production, with 9,000 tonnes P/yr in aquaculture losses versus 12,000 tonnes P/yr net addition to stock in soils (see figure 2).

Although aquaculture and agriculture drive P consumption and losses at comparable levels, we determined that their P efficiencies vary substantially. We estimated that plant production has an efficiency of 70% and animal husbandry has an efficiency of 10%, showing that trophic factors play a major role in the losses of P. In comparison with aquaculture, we estimated a 31% fish production efficiency, making aquaculture more efficient than animal production, but with fish

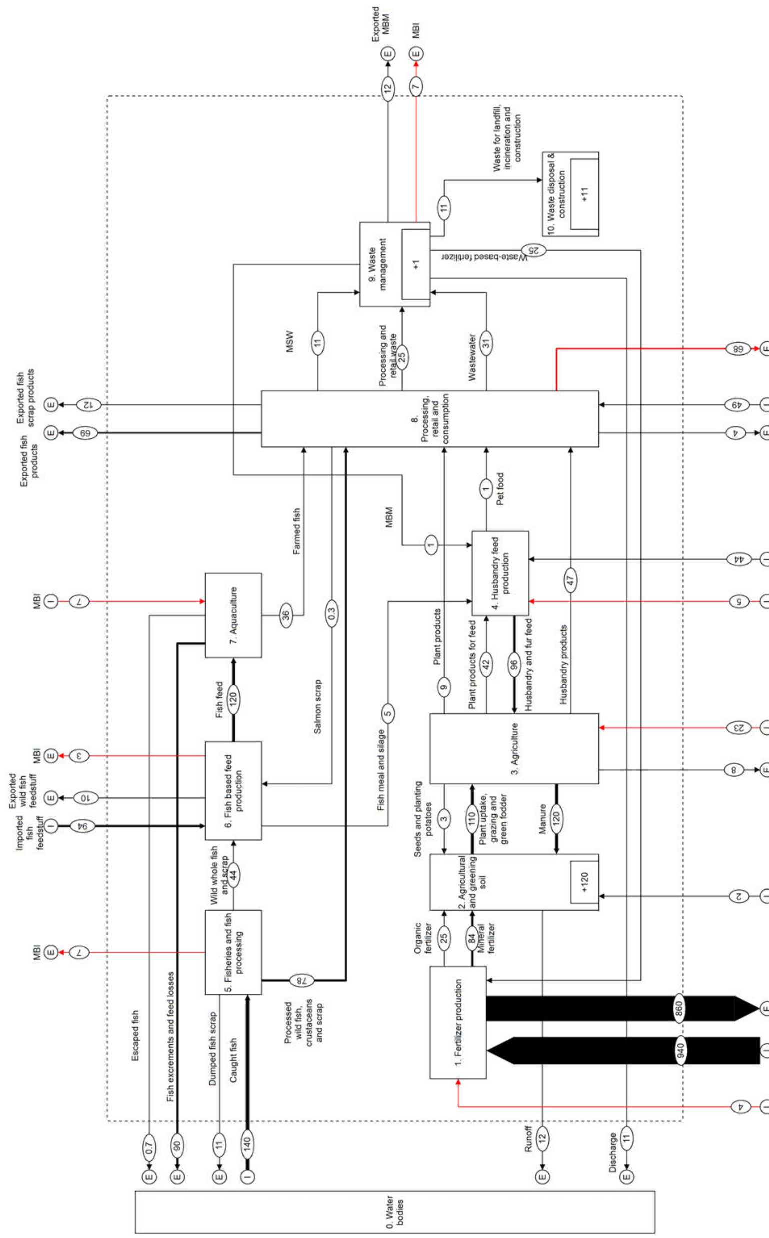

Figure 1 Norwegian phosphorus balance, 100 tonnes P, averaged 2009–2011 data. MBI = mass balance inconsistencies; MBM = meat bone meal; MSW = municipal solid waste; I = import; E = export.

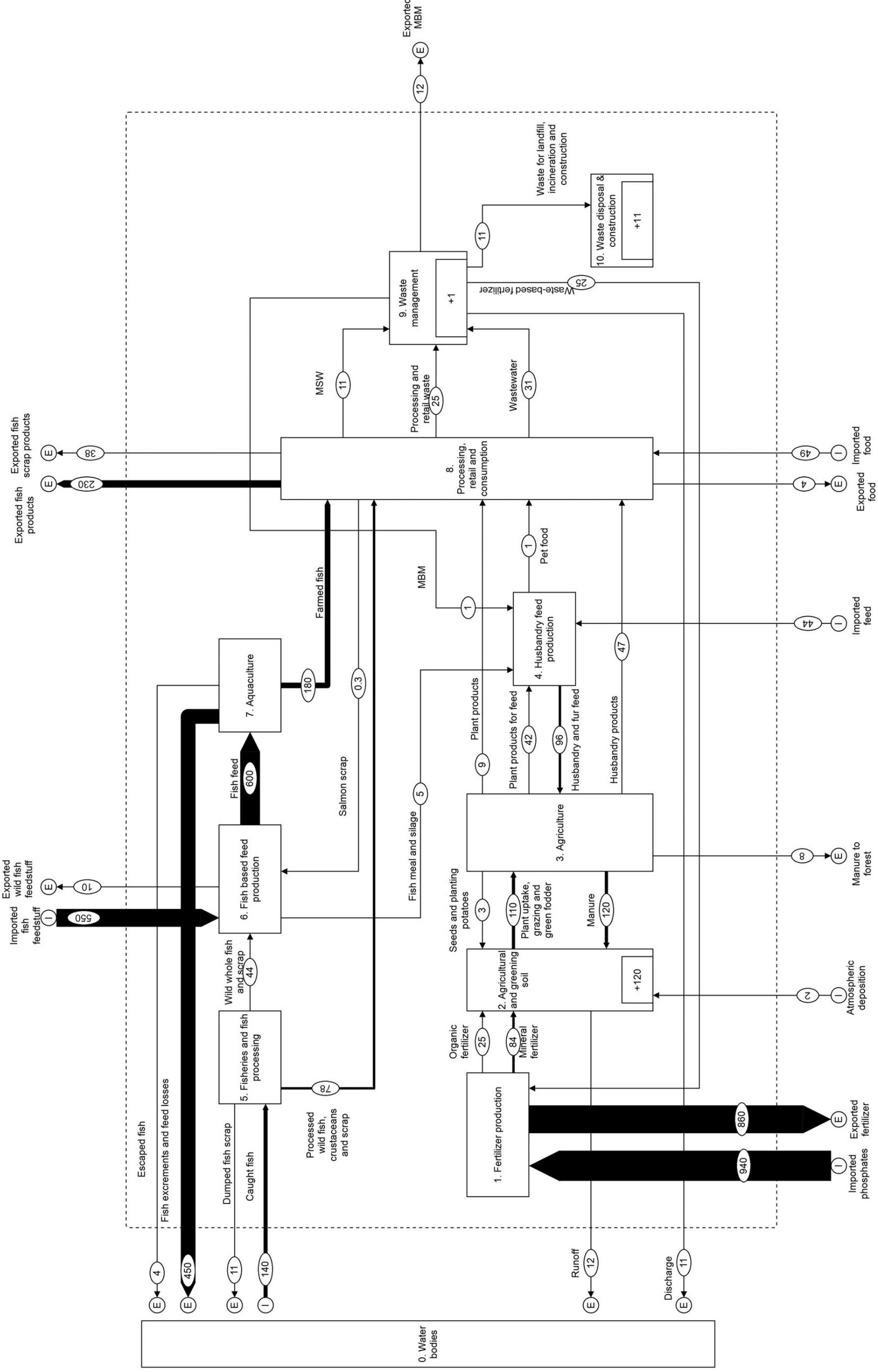

Figure 2 Aquaculture 2050 scenario, 100 tonnes P. MBM = meat bone meal; MSW = municipal solid waste; I = import; E = export.

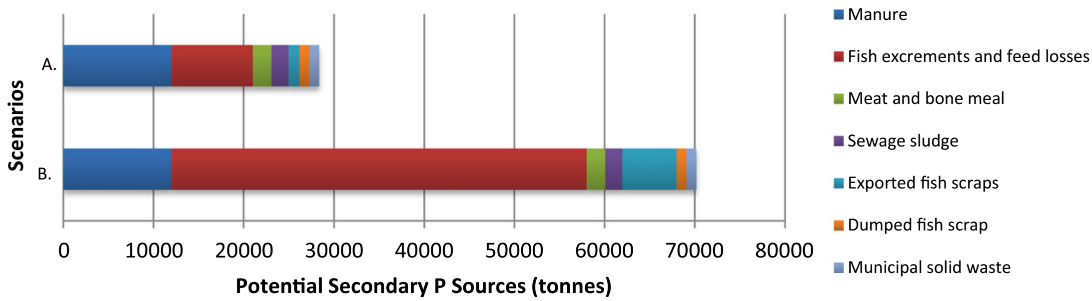

Figure 3 Potential secondary P sources for A. 2009–2011 and B. 2050 scenario in tonnes P.

excrements and feed losses entering water bodies versus P collected in manure.

With respect to fisheries, we determined that the amount of P in landed fish, or fish that were brought onto land from fishery vessels (14,000 tonnes P/yr), is approximately 4 times larger than the P in farmed fish from aquaculture (3,600 tonnes P/yr). We estimated an efficiency of approximately 92% for fisheries. Driving the high efficiency is the lack of feed input and the relatively high utilization of fish scraps from land-based processing. Compared to agriculture products, the P in landed fish far exceeds the P in both plant and husbandry products, with 9,800 tonnes P/yr combined. In terms of existing cross-sectoral synergies between aquaculture, agriculture, and fisheries, the only link found was the use of a small amount of fish meal and silage (500 tonnes P/yr) for the production of husbandry feed.

Waste flows downstream from human consumption were relatively small compared to agriculture, aquaculture, and fisheries. The largest flows include processing and retail waste (2,500 tonnes P/yr) and wastewater (3,100 tonnes P/yr). Waste that was not returned to agricultural and greening soils was accumulated in landfills and construction sites (1,100 tonnes P/yr).

The 2050 scenario reveals the future significance of the aquaculture sector (figure 2). With fish production at 5 million tonnes P/yr and technologies held constant, lost P to marine waters reaches 45,000 tonnes P/yr through fish excrements and feed losses. Compared to the 2009–2011 model, this far exceeds the mineral fertilizer demands of agriculture. Additionally, the aforementioned P losses approach orders of magnitude comparable to imported rock P for fertilizer production, 94,000 versus 45000 tonnes P/yr, respectively. In terms of fish feed needed to sustain the fivefold increase, the 2050 scenario highlights the drastic increase in reliance on imported fish feedstuff, from 9,400 to 55,000 tonnes P/yr. Overall, our results indicate that, with current technologies, such growth could lead to a dramatic increase in P losses to marine waters as well as the dependency on imported P in fish feedstuff. In terms of P recycling, however, the overall amount of potential secondary P sources drastically increases from the base-case scenario to the 2050 scenario (figure 3).

Landed scraps from fisheries and aquaculture are, in general, well utilized either domestically or internationally through the export of fish scraps predominantly for fish feed production (Olafsen et al. 2013). We determined that the only unutilized waste in fisheries, within the system boundaries, is dumped fish scraps, 1,100 tonnes P, mainly from offshore onboard fish processing. This represents a potential secondary source of P if collected and brought to land. In aquaculture, secondary P in fish scraps are also well utilized; however, fish excrements and feed losses represent a substantial amount of unutilized secondary P. Additionally, in order to reduce the dependency on external sources of P, exported fish scraps, 1,200 tonnes P (both wild and farmed), could be considered a source of secondary P if utilized domestically. Compared to agriculture, we determined that 50% of waste flows from agriculture and waste treatment get returned to agricultural land, not including urban greening. This fraction includes all manure produced, 50% of sludge produced, 20% of the meat bone meal produced, and 3% of generated food waste. Further information related to the aforementioned calculations can be found in the Supporting Information on the Web. Treated wastewater effluents were not included as a viable source of secondary P and were thus not included in figure 3. Technological upper limits for P removal during wastewater treatment, a lack of further treatment requirements, and direct dumping make this an unviable source of secondary P.

Discussion

Data Quality

The largest mass balance inconsistencies were found in the process "processing, retail and consumption." Because each flow in this process was separately calculated, errors could be the result of excluded hidden waste flows. Fish domestically consumed, for example, was calculated using data for purchased seafood in Norway. This masks waste flows between the producer and the consumer, and therefore large inconsistencies could be losses from wasted food owing to transportation and expiration. Further inconsistencies could be owing to similar issues related to unaccounted flows for food processing wastes. Poor data availability within food processing and postconsumer wastes weakened the robustness of this portion of the model. Wastewater calculations, for example, were based off of a set of detailed assumptions, found in the Supporting Information on the Web, regarding treatment type and treatment efficiency. Mass balance calculated stocks include a large amount of uncertainty given that errors and missing flows could be masked through the balance. For most flows within aquaculture, fisheries, and agriculture, the data were relatively robust given that data were primarily sourced from governmental reports and national statistics. Fish excretion and feed loss calculations, however, were based on mass balance principles. Though this approach is relatively uncertain, the values found were comparable to an independent study conducted by Wang and colleagues (2012) for 2009. Wang and colleagues found that 9,400 tonnes P/yr was discharged to the environment from Norwegian aquaculture as compared to 9,000 tonnes P/yr found in this study. The lower value could be owing to our use of a higher P content of salmon, meaning that more P was assimilated in the biomass resulting in fewer losses to the environment.

Current Recycling, Inefficiencies, and Potential Improvements

Unlike most other countries where agriculture is clearly the dominating sector, in Norway, the P flows in agriculture, aquaculture, and fisheries are of similar magnitude. Therefore, within these three sectors, there is a substantial opportunity to utilize secondary P sources and reduce the overall domestic demand for mineral P. The following paragraphs discuss the current state of recycling within the different sectors and highlight inefficiencies and areas for substantial improvements.

Aquaculture

In Norway, marine water bodies represent the second largest sink for P. Norwegian aquaculture contributes three P emission streams to the marine environment. Particulate organic P (POP) from fish feces and excess feed represent the largest output of P in volume. Dissolved organic (DOP) and inorganic P (DIP) contribute significantly less P by volume, but are more readily bioavailable than POP. DIP is especially bioavailable and readily taken up by marine phytoplankton in close proximity to the net. Multiple environmental factors determine the dominant phytoplankton species that grow, but studies in Norwegian waters have shown that commercially important kelp and seaweed species achieve higher growth rates close to aquaculture pens (Handå et al. 2013). Multitrophic aquaculture seeks to take advantage of waste nutrient streams by generating valuable biomass from multiple species. Efforts in Norway have focused on kelp/seaweed combined with suspended bivalves to take up DIP/DOP and benthic invertebrates, such as sea urchins, for POP (Bellona 2013).

Currently, for Norwegian sea-based aquaculture, technology limits the ability to utilize this secondary form of P; however, recovery systems are being conducted on a pilot scale. With the development and implementation of technologies such as integrated multitrophic aquaculture, this sector represents a large potential source of secondary P (Wang et al. 2012). Harvested biomass can potentially be used to displace marine ingredients in fish feed, thus directly closing the loop between losses and feed production. However, similar to manure, P recovery from aquaculture is limited by spatial distribution. Fish farms are distributed along the coast of Norway and recovered P must be processed and transported to, for example, fish-feed–producing factories or other areas with P deficits. This could exasperate energy systems and potentially increase costs beyond economic viability.

Land-based aquaculture systems (i.e., controlled on-land recirculating systems used for fish farming) also represent a potential solution for collecting secondary P (Tal et al. 2009). Land-based recirculating tanks result in significantly smaller P losses and allow for easier removal and collection of excreted P, feces, and excess feed. Nonetheless, though land-based systems may be preferable from a P perspective, there are several barriers related to increased costs, energy demand, land demand, and fresh water demand (Aspass et al. 2014). Ocean-based closed containment systems could present a means of reducing pressure on the aforementioned resources while aiding nutrient recovery; however, studies have shown that these technologies are in their infancy and potentially result in animal health issues (Chadwick et al. 2010). The aforementioned barriers must be overcome in order to consider these systems viable solutions.

Fisheries

Secondary P from fisheries is primarily comprised of the scraps and by-products from seafood processing. Seafood processing of fillets from whitefish and herring leave by-products rich in P owing to most of the P being found in the bones. The efficiency of by-product recycling is high in Norway, except for the offshore whitefish fleet, which currently dumps approximately 90% of by-products (Olafsen et al. 2013). Opportunities for improvements in secondary P recycling for fisheries are twofold.

The first way to improve secondary P recycling is to bring all by-products and by-catch to land. The ban on dumping of commercially important species has recently been extended to include all Norwegian by-catch with a few notable exceptions, including fish that are no longer fit for human consumption owing to damage (Gullestad 2015). The ban has been estimated to have reduced dumping of commercially important species to between 2% and 8% of catch (Valdemarsen and Nakken 2002). However, difficulty in enforcement, combined with minimal incentive to land commercially unimportant catch, adds a high degree of uncertainty to dumping estimates. By-products from fish processing can be legally discarded, and this practice is common for large vessels operating far from land. A common platform for the processing of fish by-products and unwanted by-catch onboard fishing vessels is needed to utilize this resource.

The second option to improve secondary P recycling is to process more fish in Norway. The current trend in the Norwegian seafood industry is to outsource fish processing in order to reduce costs (Henriksen 2013). Even exported partially processed whitefish, with the guts and head removed, still contain a large percentage of P in the backbone and pin bones (Opstvedt and Mjelde 1994; Albrektsen et al. 2014). Once the fish leaves Norway, this export becomes a lost source of secondary

P, as seen from a Norwegian resource perspective. Measures to promote a larger degree of processing in Norway have the potential to increase secondary P availability and other important by-product streams.

The Use of Manure and Waste Products in Agriculture

In terms of ability to recycle secondary P, the agricultural and postconsumer waste subsystems have much larger practical capabilities owing to the presence of P in solid and collectable forms, such as manure and sewage sludge. This is in contrast to fish scraps that are dumped offshore or as soluble and particle P lost in fjord systems. Despite the relatively large returns of these products back to agricultural soils, P in manure and sewage sludge are not being recycled efficiently, as shown by the P accumulation in soils. The reason for this, however, is different for each secondary product.

Animal husbandry, and thus manure, is unequally distributed throughout the country, making the distance between the point of manure generation and crop and cereal production needing fertilization potentially vast. The high water content of manure, and thus large weight loads coupled with long transportation distances, puts a strain on costs and energy demand. This limits the ability to transport the secondary P, despite its high plant availability, to agricultural areas with cereal production, for example, in the southeastern part of the country (Knutsen and Magnussen 2011). The spatial distribution of manure issue is not specific to Norway. Whalen and Chang (2001) showed that, in Canada, it is uneconomical to transport manure long distances, and therefore manure is applied mostly to the land surrounding animal husbandry. This combined with the fact that manure is applied based on crop nitrogen requirements resulting in the application of excess P, the accumulation of P in cultivated soils in Canada is the result of long-term manure application. This is likely to be the case in Norway as well.

For postconsumption P flows, such as wastewater and municipal solid waste, the variable plant availability of P introduces barriers. Waste treatment technologies are chosen to meet a variety of criteria, including cost, energy use, and treatment efficiency, and not necessarily P recovery for recycling. In Norway, chemical precipitation is widely used to treat wastewater, despite it drastically decreasing P plant availability (Morse et al. 1998; Vogelsang et al. 2006). This is owing to the need for cost-/space-effective indoor treatment plants that can handle cold winters. In addition, sewage sludge raises concerns surrounding contaminants, such as organic pollutants and pathogens, that could be harmful to human health and affect long-term soil quality. This reduces farmers' willingness to accept them as appropriate substitutes (Refsgaard et al. 2004). The fraction of wastewater sludge that is applied to agricultural soils, however, is used for its soil amendment properties, rather than its P contents, and is thus not primarily used to displace mineral P fertilizer (Refsgaard et al. 2004).

It is likely that P accumulation in agricultural soil is primarily owing to ineffective manure and sludge application: Because of the spatial imbalance between manure generation and crop production areas where mineral P is applied, manure is often applied where P is in excess. Additionally, because the plant availability of P in sludge is poor, it is common practice to apply both mineral fertilizer and sludge to agricultural soils. Therefore, even though the return of secondary agricultural P products to land is relatively high, the reuse or actual plant uptake of secondary P is low and the soil remains a large sink of P.

Options for improving P recycling in agriculture are primarily technology based. Improving manure distribution would require the development and utilization of technologies that can reduce the weight content of manure, thus allowing lower transportation costs. In terms of sewage sludge, despite the challenges related to the Norwegian climate, a shift toward biological wastewater treatment would increase the plant availability of P (Morse et al. 1998). Given that P from the waste sector is easily collectable and regionally concentrated, the distribution of this P source would be manageable. However, this is only viable if barriers related to energy and costs are overcome.

Integrating Aquaculture, Agriculture, and Fisheries for the Utilization of Phosphorous

Scenario: Future Challenges

The future fivefold aquaculture growth, which will result in an inevitable increase in P losses to water bodies, is likely to pose many environmental challenges. With 5 times more P reaching fjord systems, the risk of eutrophication in coastal marine waters drastically increases. The high flux rate of fjord systems currently limits eutrophication concerns, given that the strong currents quickly exchange nutrients with coastal waters (Skogen et al. 2009). However, with drastic increases in P waste flows, concentrations can potentially reach levels that exceed the flux capacity of the fjords. This will have to be considered carefully by public authorities when realizing growth. Additionally, future strains on P resources abroad may cause problems in Norway because of the increased demand for imported feedstuff. Norway does not have feed sources of their own, and feedstuff needs will therefore either have to be sourced from other countries or from the ocean through fisheries. Given that it is highly likely that the imported agricultural feedstuff products are produced using mineral fertilizers, increased fish feed production means that Norway will also indirectly increase its mineral P footprint unless P losses along the production chain are recovered and recycled.

Scenario: Future Opportunities

From a Norwegian standpoint, if P sources are utilized efficiently, the fivefold growth could also represent an opportunity for moving Norwegian agriculture toward becoming import rock P independent. Currently, cross-sectoral utilization of secondary P is hardly explored and is certainly not optimized. The only link between aquaculture, fisheries, and agriculture is the fish waste products that are used as inputs to husbandry and fur animal feed. This represents only a small fraction of the P needed in agricultural feed and approximately one third of the total excess aquaculture fish scrap, with the remainder being exported. With the goal of domestically sourcing P for food production, it is imperative to map and quantify secondary P sources and how they could be optimally utilized within and between sectors.

As discussed, in agriculture, the utilization of returned secondary P is poorly managed and, in order to improve this, related to logistics, energy use and P quality will have to be overcome. Potential improvements that can be realized without substantial technological breakthroughs, however, include optimizing aquaculture and fishery wastes for feed production. Owing to the fact fish feed cannot be sourced from the same species of fish, secondary P from salmon, which comprises over 90% of Norwegian aquaculture, cannot be used to feed salmon (Norwegian Directorate of Fisheries 2014). It can, however, be used for nonsalmon species or as a feed component in animal husbandry. This potential will increase as aquaculture production increases, representing a growing opportunity for displacing imported P in agriculture feed.

Another important step for optimizing fish scrap P recycling is by reducing the amount of dumped fish scrap. Technology and financial incentives currently limit the ability to collect and reuse this waste. However, the prices of by-products are expected to rise with the prices of vital nutrients, such as omega 3-fatty acids, thereby improving the feasibility of utilizing this waste. Marine by-products are especially valuable in Norway because 99.9% of the marine landed catch is nonsalmon species (Norwegian Directorate of Fisheries 2013). This means that the by-products can be used in aquaculture, thus reducing the P burden from agriculture-based P products abroad and the mineral P, which is likely used to produce them.

With secondary P from aquaculture likely reaching orders of magnitude that are comparable to imported rock P for the production of fertilizers, this represents a substantial opportunity to displace not only domestic mineral P needs for fertilizer production, but also global, given that approximately 90% of Norway's produced fertilizers are exported. The P burden would effectively be shifted from mineral P to sea-based P. There are a substantial amount of hurdles to overcome to realize this opportunity. Obstacles related to P quality, salt concentrations, cost, energy needs, and the technologies for recovery will require a substantial amount of research before cross-sectoral synergies can be realized, but if solved, could provide partial solutions to reducing the dual problem of eutrophication and mineral P dependency.

Future Research

In order to identify more in-depth P management solutions, future research is needed. Through this study, it was shown that several methodological adaptations of the model could help to provide a more holistic basis for decision making. In the choice of technological systems, it is clear that energy and other critical materials are closely coupled to P. A good example of this is the increased P demand through the production of bioenergy from crops. To produce this type of energy, biomass inputs require P, indicating that increased bioenergy production would also increase P requirements. Because of these types of relationships, it would be beneficial to study P in combination with other critical materials/nutrients. A multilayer material flow analysis approach could include societally important materials and energy and would preferably lead to overall, not merely P-specific, improvements in resource management.

Further, concerns surrounding technology, plant availability, and practical/economic feasibility for recycling P make the inclusion of a "feasibility assessment" of great value. Such a study would highlight more practical issues regarding recycling given that often quality issues, such as plant availability and/or the presence of heavy metals, play a major role in the ability and willingness to reuse P within agricultural systems. If P is looked at only through an elemental point of view, information regarding the existence of contaminants that reduce the usability of P is masked. Other feasibility elements include the spatial and temporal distribution of P. Conventional SFA methods portray P in an aggregate form and do not include resolution regarding physical location, a factor that plays a key role in the feasibility of reusing P. This would better be examined by spatially explicit SFA modeling.

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Table S1. Assumptions utilized for wastewater treatment calculations (Sources: [1](Berge & Mellem, 2012) [2]Own assumptions)

Table S2. Calculation for the percentage of meat and bone meal returned to agriculture: DM = dry matter; MBM = meat and bone meal

Table S3. Calculation for the percentage of sludge returned to agriculture

Table S4. Calculation for the percentage of food waste returned to agriculture; MSW =municipal solid waste

Paper II

Erratum to: A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway

A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway

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The supplementary material can be downloaded from

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ERRATUM

Erratum to: A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway

Ola Stedje Hanserud · Eva Brod · Anne Falk Øgaard · Daniel B. Müller · Helge Brattebø

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In the original publication, the units written into Fig. 2b, d, e were incorrectly published, although the units given in the captions were correct. The revised version of the figures are given (Fig. 2).

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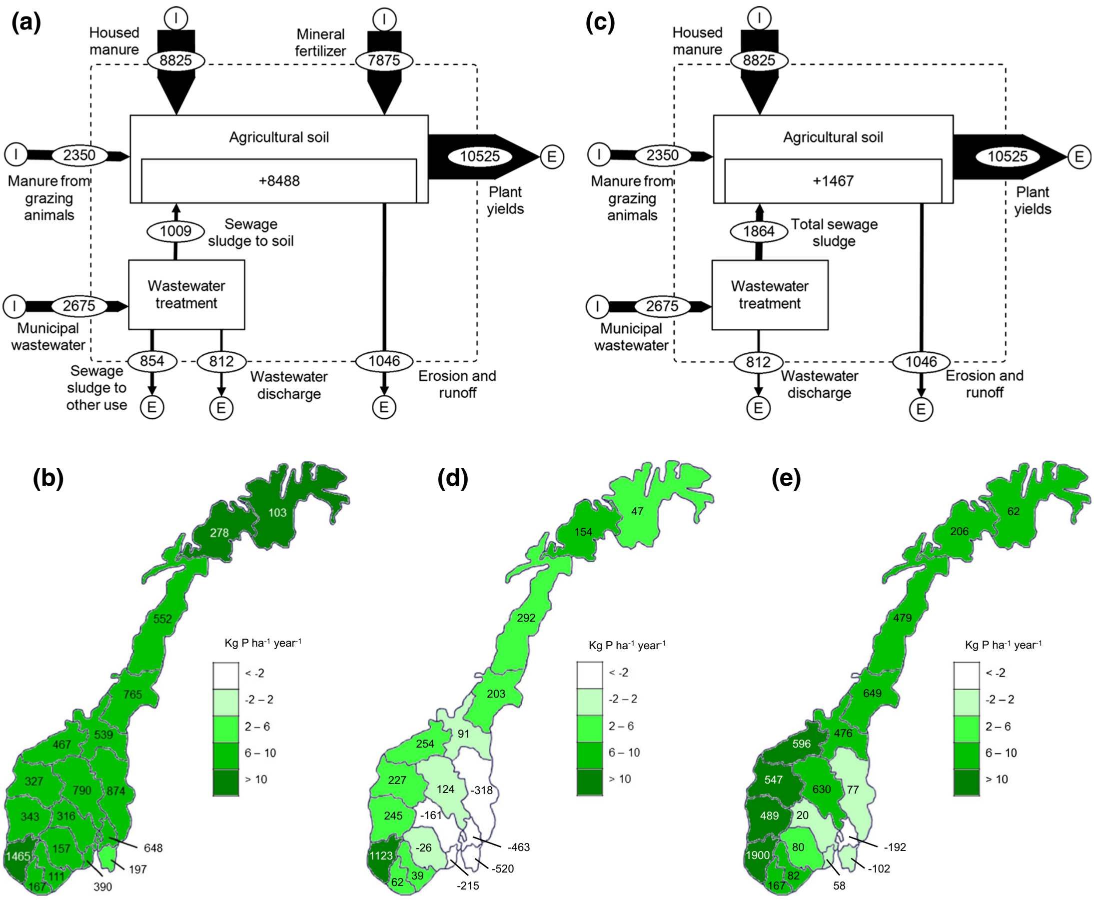

Fig. 2 **a** FR0: annual P balance for agricultural soil in Norway (tonnes P year^{-1}), 2009–2011. **b** FR0: annual net stock change (tonnes P year^{-1}) and net stock change per hectare (kg P ha^{-1} year^{-1}), 2009–2011. **c** FR1 and FR2: annual P balance for agricultural soil in Norway (tonnes P year^{-1}), 2009–2011. **d** FR1: annual surplus fertilization (tonnes P year^{-1}) and surplus fertilization per hectare (kg P ha^{-1} year^{-1}), 2009–2011. **e** FR2: annual surplus fertilization (tonnes P year^{-1}) and surplus fertilization per hectare (kg P ha^{-1} year^{-1}), 2009–2011

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PERSPECTIVE

A multi-regional soil phosphorus balance for exploring secondary fertilizer potential: the case of Norway

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Abstract Phosphate rock is a non-renewable source of phosphorus (P) in mineral fertilizer and many countries need to use P fertilizer more efficiently in food production. This study explored the theoretical fertilizer potential of the P-rich bioresources animal manure and sewage sludge to supply the required P fertilizer for crops. We used Norway as a case study and employed multi-regional substance flow analysis with averaged annual data for the period 2009–2011. In a status quo soil balance for agricultural soil, all counties had a positive balance with a national average of 8.5 (range between counties of 2.7–14.7) kg P ha^{-1}. In addition, two fertilizer regimes (FR) were evaluated for the period; FR1 omitted mineral P fertilizer from the balance and assumed bioresource addition matched plant P offtake regardless of soil available P, while FR2 omitted fertilizer from the balance and adjusted bioresource inputs according to whether soil available P was above (adjusted downwards) or below (adjusted upwards) the optimum soil P level. FR1 and FR2 gave a national average P surplus of 1.2 (range −7.0 to 11.2) and 6.2 (range −2.5 to 19.0) kg P ha^{-1}, respectively. The secondary P fertilizer potential of bioresources for meeting P requirements was found to be underestimated in the short term by not taking into account the actual plant-available soil P level. Our conclusion was that the P fertilizer values of manure and sludge have the theoretical potential to meet the P fertilizer requirements of all Norwegian crops assessed in both the short-term and long-term perspective.

Keywords P plant availability · Soil P balance · P-AL · P fertilizer potential · Secondary P · SFA

Electronic supplementary material The online version of this article (doi:10.1007/s10705-015-9721-6) contains supplementary material, which is available to authorized users.

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Introduction

Sound management of phosphorus (P) as an essential plant nutrient is key to maintaining or increasing crop yield (Syers et al. 2008), minimizing consumption of non-renewable phosphate rock (Cordell et al. 2009) and minimizing P losses causing eutrophication of water recipients (Smith et al. 1999). Today, food production in many countries is highly dependent on imports of primary P in mineral P fertilizer. This dependency could be reduced if secondary P in available bioresources within a country or a region were to be used more efficiently.

Geographical segregation of animal husbandry and arable farming is a source of differences in regional soil P balances, which tend to be significantly more positive in animal-dense areas than in arable-dominated areas (Senthilkumar et al. 2012). Human settlements are often unevenly distributed and are becoming increasingly urbanized. Human excreta and wastewater are viewed globally as an important renewable and easily accessible source of recycled P, and urban centres are becoming P hotspots (Cordell et al. 2009). However, both animal manure and human excreta are bulky materials and costly to transport, and national-scale analysis of material flows may therefore overestimate the feasibility of secondary P recycling from such flows (Senthilkumar et al. 2012). Multi-regional scale studies are able to give a first impression of the geographical distribution of materials within a country and create an understanding of where P-rich bioresources are generated and where P fertilizer is needed, as described by Bateman et al. (2011) for manure in England.

Past over-application of P fertilizer has resulted in a great build-up of P, including plant-available P, in European agricultural soils (Schoumans et al. 2010; Van Dijk et al. Accepted). Application of P fertilizer to crops follows the law of diminishing returns (Syers et al. 2008). Above a certain soil P level, further application of P fertilizer has limited or no effect on yields and is therefore inefficient use of a limited resource. High P accumulation in soil is also associated with increased losses of P in runoff and erosion risking eutrophication in surface waters (Smith et al. 1999). Consequently, P-rich soil is a source of P that should be tapped into with both the resource and pollution perspective in mind. Sattari et al. (2012) showed that the projected global P fertilizer demand up to 2050 could be decreased substantially by including past build-up of soil P (residual P or legacy P) as a resource. Re-aligning the inputs of P to match crop requirements is seen as an important step towards increased P efficiency (Withers et al. 2015).

Ultimately, the use of total P content in material flows can overestimate the fertilizer value of secondary P in bioresources. For example, the use of chemical precipitation in wastewater treatment plants results in a sewage sludge in which P is mainly present in aluminium/iron-bound form with low plant availability (Frossard et al. 1994; Krogstad et al. 2005). Although other factors such as soil type and content of available P in the soil also influence the plant availability of P in sludge (Krogstad et al. 2005), quantification of the plant-available P in bioresources could give a good indication of secondary P fertilizer potential.

Thus, there are three main causes of ineffective use of secondary P: (1) Geographical segregation between where secondary P is generated and where it is needed; (2) disregard of the existing plant-available soil P; and (3) the chemical form and plant availability of secondary P affecting its fertilizer value.

The main objective of this study was to explore the theoretical secondary fertilizer potential contained within P-rich bioresources, using Norway as a case study. We hypothesized that the overall net demand for mineral P fertilizers in Norwegian agriculture is close to zero if the secondary P in existing bioresources (animal manure and sewage sludge) is utilized to its theoretical potential. To examine how that potential differed geographically across the country, we disaggregated material flows down to regional county level. The theoretical fertilizer potential in animal manure and sewage sludge was explored by quantifying plant-available P and assuming a regional soil P balance without the use of mineral P fertilizer. Moreover, we used a measure for the level of plant-available P in Norwegian agricultural soils to estimate regional P fertilizer requirements, and compared those with values obtained applying a simplified strategy of maintenance fertilization that assumes optimal soil P levels.

Materials and methods

System definition

We used substance flow analysis (SFA) (see e.g. Brunner and Rechberger 2004) to develop a multi-regional soil P balance for the 19 counties in Norway, looking at the major flows of P into and out of agricultural soil. Thus, the system boundary was set around agricultural soil in each county, including permanent pasture used for fodder production and grazing, but excluding uncultivated land[1] used for grazing, such as forest, mountain and coastal terrain. Outdoor horticulture was not included in the study due to poor availability of regional statistical data, but the amount of P in horticultural produce (including greenhouse horticulture) has been estimated to be roughly 1 % of P in total plant yields on a national scale. Greenhouse horticulture was considered outside the system boundary of agricultural soil and with negligible P flows to agricultural soil. All input flows to agricultural soil were considered to be exogenously determined except the input flows from the wastewater treatment process. This process was included in the system in order to explore how changes in sewage sludge distribution can affect inputs to agricultural soil and the soil P balance. The counties of Oslo and Akershus are often treated as one statistical entity and thus were also treated as one entity and county (Oslo and Akershus) in this study, resulting in 18 independent systems to be quantified (Fig. 1). Each flow was independently calculated and a multi-year average was produced for the period 2009–2011 in an attempt to avoid annual variations. A visualization of the system was generated by the material flow analysis freeware STAN (Fig. 2a). Some bioresources containing P were not included in the analysis, either because of lack of regional-scale data or because their use as a fertilizer in agriculture in the study period was considered to be insignificant. Meat and bone meal (MBM) produced from slaughter waste is a P-rich commercial product sold domestically and exported abroad as both fertilizer and a feed ingredient for pet and fur animals. Around 85 % of the MBM in Norway is produced in three processing plants (Viste, personal communication), and it is consequently not generated in all counties. The relevance of MBM as a potential fertilizer in the future is entirely dependent on market developments. MBM used as fertilizer was, on average for 2009–2011, in the order of 1–2 % of the total national P input to agricultural soil according to our calculations, and the proportion has since decreased further. Therefore we opted to omit MBM as a fertilizer input in the present study.

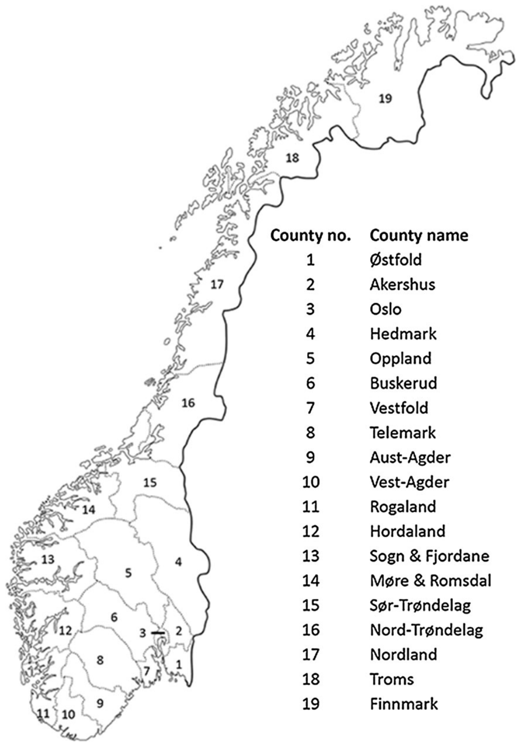

Fig. 1 Map showing the 19 counties in Norway. Data for counties 2 and 3 were combined in this study

Processes

Agricultural soil is defined as soil where crops are grown for human and animal consumption and that receives different materials containing P as a fertilizer or soil amendment. Agricultural soil includes permanent pastures where animals graze and deposit P-rich manure, and these areas may also be fertilized by mineral P fertilizer. Outputs of P from soil are harvested plant yields and diffuse losses through erosion and run-off. Plant residues were assumed here to be returned to soil and therefore not considered an output flow.

Wastewater treatment encompasses all treatment of collected municipal wastewater in wastewater treatment plants (WWTP) with a capacity >50 person equivalents.[2] In 2011, 83 % of the Norwegian

[1] In Norwegian: utmarksbeite.

[2] Statistics on wastewater treatment distinguish between WWTPs with capacity over and under 50 person equivalents

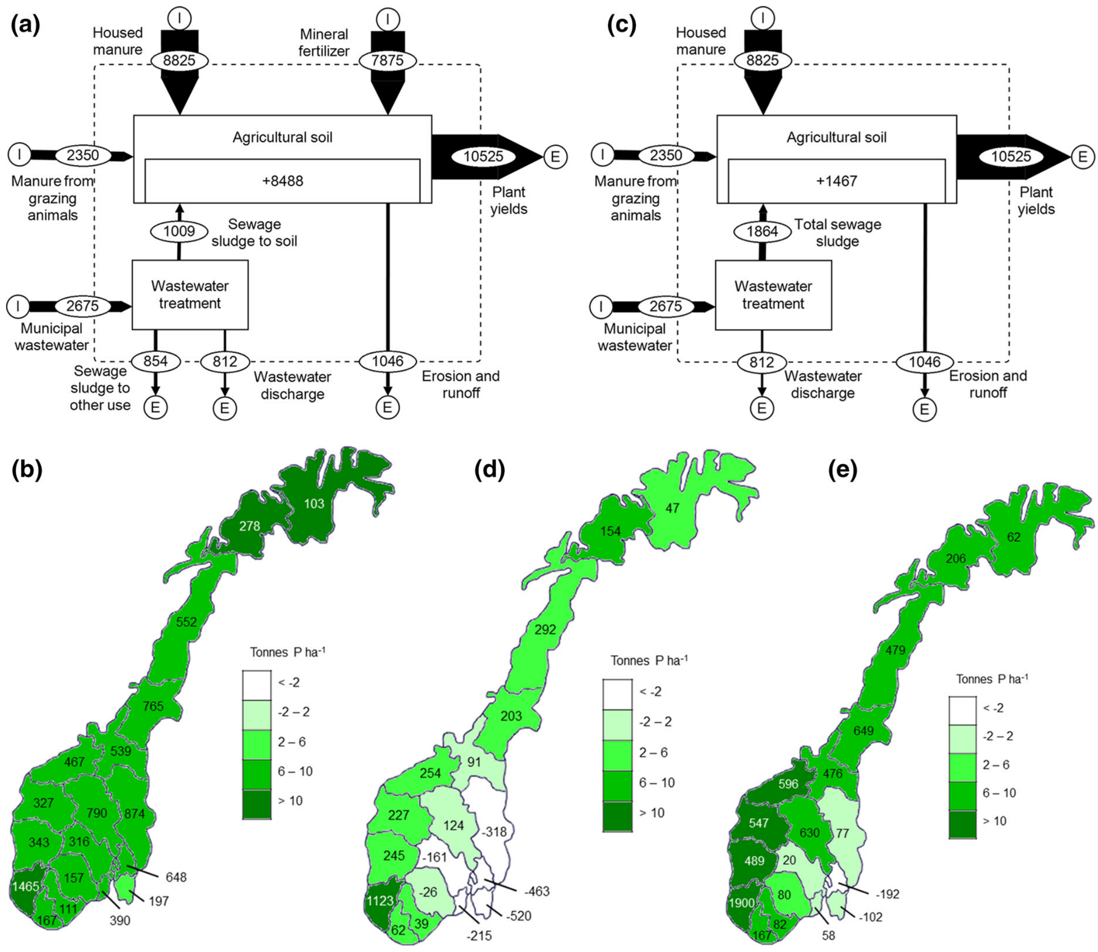

Fig. 2 **a** FR0: annual P balance for agricultural soil in Norway (tonnes P year^{-1}), 2009–2011. **b** FR0: annual net stock change (tonnes P year^{-1}) and net stock change per hectare (kg P ha^{-1} year^{-1}), 2009–2011. **c** FR1 and FR2: annual P balance for agricultural soil in Norway (tonnes P year^{-1}), 2009–2011. **d** FR1: annual surplus fertilization (tonnes P year^{-1}) and surplus fertilization per hectare (kg P ha^{-1} year^{-1}), 2009–2011. **e** FR2: annual surplus fertilization (tonnes P year^{-1}) and surplus fertilization per hectare (kg P ha^{-1} year^{-1}), 2009–2011

population was connected to a wastewater treatment plant with a treatment capacity of more than 50 person equivalents (Berge and Mellem 2012). In addition to sewered sanitary wastewater from households and other public and private buildings, municipal wastewater also includes wastewater from industrial processes, as well as septic tank contents emptied by tanker trucks. The treatment process produces effluent wastewater discharged to a water recipient and sewage sludge distributed for different uses. In 2011, 56 % of the sludge (measured as dry matter) was applied to agricultural land, 25 % to greening, 14 % as cover for landfill and 2 % was landfilled (Berge and Mellem 2012). Greening comprises use of sludge on urban green areas and roadside areas, for land restoration and as input in the production of soil products.

Footnote 2 continued
(pe). A pe is defined in Norway as the amount of organic matter degraded biologically over 5 days with a biochemical oxygen demand of 60 g O_2 per day (The Norwegian regulations relating to pollution control 2004).

Flows

Flow descriptions, equations and their respective data sources are shown in Tables 1 and 2. *Mineral fertilizer* (MF) is a commercial product and was quantified based on trade statistics on county level for the total sale of MF. According to our rough estimates, the amount of mineral P fertilizer not used in agriculture is approximately 5–10 % of the total amount of MF sold on national level, but breaking this down to county level would be difficult. We concluded that the regional statistics at hand provided a good enough approximation of the use of MF in agriculture. *Housed manure* from confined animals included the major animal husbandry groups: cattle, pigs, poultry, sheep and goats. We assumed that all of the housed manure was applied to agricultural soil within the county of origin and that inter-regional trade in manure was insignificant for the study period. A survey in 2000 showed that 7 % of the farms spreading manure on their land receive manure from others, while 11 % of farms sell or give away manure to others (Statistics Norway 2001). However, according to Gundersen (personal communication), most of the trade in manure is between neighbouring farms. This supports our assumption on lack of inter-regional trade. For *manure from grazing animals,* only cattle and sheep were considered for permanent pasture, depositing manure directly on the soil. The estimation of P in manure, housed and from grazing, is described in detail in the appendix (Online Resource 1). We assumed that the P in manure is as available to plants as P in MF (see for example Oenema et al. 2012; Smith and van Dijk 1987). For *sewage sludge,* we calculated the total amount of P as well as the amount of P that can replace MF, which hereafter is used interchangeably with the term plant-available P. The method used for estimating plant-available P in sewage sludge is described in detail in the appendix (Online Resource 1), and was based on statistics for wastewater treatment and literature on mineral fertilizer equivalency (MFE) of P in sewage sludge from the common treatment processes in Norway (see e.g. Øgaard 2013). The method considers the influence of a specific mix of wastewater treatment methods within a county on both the amount of P retained in sludge and its plant availability. The diffuse losses of P from soils through *erosion and runoff* were calculated by Eggestad (personal communication) based on statistics for production subsidy applications and a method described by Eggestad et al. (2001), where the loss of P is proportional to the loss of soil and determined by e.g. soil erodibility, topography and land use. The output flow of *plant yield* was based on statistics for the nine dominant crops in Norway, which together covered 98 % of all cultivated area in Norway in the period 2009–2011 (Statistics Norway 2014): wheat, barley, oats, rye and triticale, oilseeds, potato, green fodder and silage, peas and grass. To account not only for the amount of harvested grass but also the amount of grass eaten by grazing animals on agricultural land, we used a national total amount of grass and pasture yield and distributed this between counties based on grass area and a productivity factor to account for regional differences in yield per hectare. The method for estimating P in grass yield per county is further described in the appendix (Online Resource 1).

Net stock change

Net stock change (ΔS) was calculated for the process ‘agricultural soil’ to indicate an addition (positive ΔS) or withdrawal (negative ΔS) of net amounts of P from the stock of soil P. The net stock change, also called the soil balance, was calculated by subtracting the sum of the outputs from the sum of the inputs as shown in Eq. 1, where i and j denote the different inputs and outputs, respectively. For the process of wastewater treatment, we assumed that there was no stock accumulation over time.

$$\sum_i Input\ i - \sum_j Output\ j = \Delta S \qquad (1)$$

Fertilizer regimes

In order to test the hypothesis and explore the research questions formulated at the start of the study, we chose to examine three fertilizer regimes (FR) for the period 2009–2011 with different soil P balances and/or fertilization strategies. These FRs only describe different perspectives on the specified period and therefore must not be confused with scenarios intended to describe the future. Nevertheless, we later discuss the possible implications of the results for future fertilization strategies.

Table 1 Description of the P flows quantified at the regional scale in Norway

Flow name	Flow description
Mineral fertilizer	The quantity of P in mineral fertilizer products used for crop production
Housed manure	The quantity of P in housed animal manure from cattle, pigs, poultry, sheep and goats
Manure from grazing animals	The quantity of P in manure from grazing animals deposited directly onto agricultural soil
Municipal wastewater	The quantity of P in collected untreated municipal wastewater
Sewage sludge to soil	The total quantity of P and the quantity of plant-available P in sewage sludge applied to agricultural soil
Sewage sludge to other use	The total quantity of P and the quantity of plant-available P in sewage sludge used elsewhere than on agricultural soil
Wastewater discharge	The quantity of P in wastewater treatment plant effluents discharged to water recipients
Erosion and run-off	The quantity of P in diffuse losses from agricultural soil
Plant yields	The quantity of P in harvested wheat, barley, oats, rye and triticale, oilseeds, potato, green fodder and silage, peas and grass, including the grass grazed by animals

- FR0: Status quo soil P balance
- FR1: Soil P balance without MF, maintenance fertilization strategy
- FR2: Soil P balance without MF, transition fertilization strategy

FR0 describes the annual status quo soil P balance, based on statistics for all described input and output flows of P for agricultural soils. Annual net agricultural soil accumulation (net stock change) was quantified in terms of the total amount of P according to Eq. 1. An estimate of the amount of plant-available P in sludge was also included, to show the status quo fertilizer value of sludge applied in agriculture.

FR1 In this fertilizer regime, we wanted to see whether plant-available P in manure and sewage sludge generated in a county, i.e. the total secondary P fertilizer potential, would be sufficient alone to provide the amount of P fertilizer required according to a maintenance fertilization strategy. Mineral fertilizer was therefore omitted as an input in this regime. In a maintenance fertilization strategy the required P fertilizer input equals the amount of P removed from the soil through plant yields. This is a simplified fertilizer regime in that it implicitly assumes optimal levels of soil P (see FR2). As an optimal soil P level is the goal in the long term, this fertilizer regime also represents the long-term equilibrium fertilization strategy. The calculated difference between the total P fertilizer potential and the fertilizer requirement was called surplus fertilization, and was calculated as shown in Eq. 2. The total theoretical fertilizer potential in sewage sludge was considered to be the plant-available P in all sewage sludge produced in a county, i.e. the combined flow of sewage sludge to soil and sewage sludge to other use. This combined flow was called *total sewage sludge*. We omitted P losses through erosion and runoff from the calculation of surplus fertilization, since such losses are usually not taken into consideration in fertilization planning in Norway. Phosphorus losses from arable land in Norway are mainly caused by erosion (Ulén et al. 2012), which means that P is lost with the soil to which it is bound and therefore does not change the concentration of plant-available P in the remaining soil. Fertilization planning is based on concentrations of plant-available P in soil. Furthermore, in the short term the P losses by erosion are expected to be low compared with the total P stock in soil.

$$\begin{aligned} \textit{Surplus fertilization} &= \textit{Housed manure} + \textit{manure from grazing animals} \\ &\quad + \textit{plant available P in total sewage sludge} - \textit{fertilizer requirement} \end{aligned} \tag{2}$$

Table 2 Methods used to calculate the P flows at the regional scale

Flow name	Equation	Material quantity sources	P content sources*
Mineral fertilizer	Mineral fertilizer applied to agricultural soil × Pc	1, 2, 3	1, 2, 3
Housed manure	Number of animals × P excreted per animal—number of animals grazing on uncultivated land × time grazing × P excreted per animal—number of animals grazing on agricultural soil × time grazing × P excreted per animal	4, 5, 6; Time grazing ag. soil: 7	5
Manure from grazing animals	Number of animals grazing on agricultural soil × time grazing × P excreted per animal	4, 6; Time grazing: 7	5
Municipal wastewater	Quantity of discharged P to water/(1—treatment effect)	8, 9, 10	
Sewage sludge to soil	Total quantity of P: (Municipal wastewater—quantity discharged P to water) × fraction of sludge to agriculture	8, 9, 10; Plant avail. P: 8–12	
	Quantity of plant-available P: total quantity of P × weighted average share of plant-available P (see Online Resource 1 for method)		
Sewage sludge to other use	Total quantity of P: (Municipal wastewater—quantity discharged P to water) × (1—fraction of sludge to agriculture)	8, 9, 10; Plant avail. P: 8–12	
	Quantity of plant-available P: Total quantity of P × weighted average share of plant-available P		
Wastewater discharge	Quantity of discharged P to water	8, 9, 10	
Erosion and run-off	Eggestad, personal communication		
Plant yields	Cereal, potato, oil seed, legume, green fodder and silage yields × Pc + grass yields × Pc × area factor × productivity factor	4; Oilseeds and legumes: 13; Grass: 14, 15	16; Grass: 17

Pc = P concentration; [1,2,3] (Norwegian Food Safety Authority 2010, 2011, 2012); [4] (Statistics Norway 2014); [5] (Karlengen et al. 2012); [6] (Norwegian Agriculture Agency 2014); [7] (Bjørlo, personal communication); [8] (Berge and Mellem 2010); [9] (Berge and Mellem 2011); [10] (Berge and Mellem 2012); [11] (Øgaard 2013); [12] (Krogstad et al. 2005); [13] (Breen, personal communication), [14] (Norwegian Agricultural Economics Research Institute 2014); [15] (Bakken et al. 2014); [16] (Antikainen et al. 2005); [17] (Johansen et al. 2003)

* Parameters used for P content in animal manure and plant yields are given in Online Resource 2

FR2 was similar to FR1 except one significant difference: the amount of P fertilizer required for producing grass, cereal, green fodder and silage, and oilseeds (98.4 % of the total plant P yield) was adjusted to account for the existing level of plant-available soil P in the calculation of fertilization surplus or shortage. The adjustment was made to approach, over a series of years, the level of plant-available soil P viewed as optimal in Norwegian fertilizer planning, regarding both yield and the risk of diffuse P losses. The reference for the adjustment was maintenance fertilization, and the fertilization strategy followed during the adjustment phase is termed *transition fertilization*. For P-deficient soils, the amount of P applied in fertilizer should exceed the amount of P removed through plant harvest, while in soils with high levels of plant-available soil P the fertilizer P amount should be lower than crop P removal. At high levels of plant-available soil P, the release of P from the soil stock covers part or all of the crop's P requirement (Krogstad et al. 2008). In Norway, plant-available P in soil is estimated by P-AL (mg per 100 g soil) extracted by the ammonium-acetate-lactate method (0.1 M ammonium lactate and 0.4 M acetic acid, pH 3.75) according to Egnér et al. (1960). Table 3 shows the different classes of P-AL

Table 3 Classes of P-AL level and percentage correction of P requirement for grass, cereals and oilseed production (Krogstad et al. 2008)

Class	P-AL value (mg per 100 g soil)	Name of class	Mean P-AL class value*	Regression equation for percentage correction (Y) of P requirement	Mean percentage correction (Y) of P requirement*
A	1–5	Low	3	Y = −25 * P-AL + 125	50
B	5–7	Medium/optimal	6	Y = 0	0
C1	7–10	Moderate high	8.5	Y = −14.28 * P-AL + 100	−21.38
C2	10–14	High	12	Y = −14.28 * P-AL + 100	−71.36
D	>14	Very high	–	Y = −100	−100

* Columns added by us

level in soil and the recommended correction of P fertilizer requirement as a percentage of maintenance fertilization amount, as described by Krogstad et al. (2008). The recommendations bear a resemblance to the system used in the UK (Tóth et al. 2014). A P-AL level of 5–7 mg/100 g soil is considered optimal (Krogstad et al. 2008) and no correction should be made to the maintenance fertilization. P-AL measurements for each county for the period 2001–2011 were obtained from the soil database administered by the Norwegian Institute for Agricultural and Environmental Research (Bioforsk) (Grønlund, personal communication), which records P-AL data on farm level. Norwegian regulations require fertilizer plans to be based on soil analyses no older than 8 years. Hence, data from a time span of 10 years should represent the majority of Norwegian agricultural soils, assuming that all data have been submitted to the database. For each county, the P-AL data were distributed between P-AL classes. Based on this distribution and a mean percentage correction of P requirement for each class, we calculated a correction (%) of the P requirement for grass, cereals, green fodder and silage, and oilseeds in each county. A further description of the method can be found in the appendix (Online Resource 1). The correction was multiplied by the plant P yield for the respective crop to get an adjusted fertilization requirement, which was then added to the non-adjusted P requirement for the other crops (1.6 % of total plant P yield) to obtain a corrected total fertilizer requirement. The corrected total fertilizer requirement was balanced against the same inputs as in FR1 for the surplus fertilization calculation (Eq. 2). Although the fertilizer requirement was adjusted, the system flows stayed unchanged from FR1 and the soil P balance was therefore identical to that in FR1.

Uncertainties

Plant P uptake from sewage sludge varies with the soil type to which it is applied and the type of sludge produced at a specific WWTP (Krogstad et al. 2005; Øgaard 2013). Krogstad et al. (2005) found higher plant P uptake in a clay soil compared to a moraine soil, indicating lower P sorption capacity in the clay soil. As soil type affects plant P uptake from both sewage sludge and mineral fertilizer, the effect on the relative difference in uptake reflected in the MFE can be expected to be small. Øgaard (2013) found plant P uptake to be significantly different when equal amounts of P in chemically precipitated sludge from different WWTPs were applied to soil. This variation is reflected in the MFE range given for chemically and chemical-biologically treated sludge in Online Resource 1. We believe that the MFE values used in this study are good enough approximations for plant-available P in sludge, given the prevailing treatment technologies in the study period. Any long-term release of plant-available P from sludge beyond the year of application was assumed to be detected in P-AL measurements and would subsequently affect the P fertilization requirement. The calculation of fertilization adjustment in FR2 relied on the assumption of representativeness of the recorded soil samples for a county. This was considered to be satisfactory for all counties but one, as discussed in the appendix (Online Resource 1). The use of a mean value for the different P-AL classes (Table 3) is a simplification associated with some uncertainty, since the measurements within each class may be skewed towards the upper or the lower limit of the class in a specific county. This simplification was made in order to use the same percentage correction values for all counties. Lastly, uncertainty in the statistical data was expected to be low. The main source of data was Statistics Norway, and we used a bottom-up approach to estimate the majority of the flows.

Results

FR0

The soil P balance (Table 4) showed a positive net stock change and thus an annual surplus application of P to agricultural soil in all counties for the period 2009–2011. The net stock change varied from 2.7 kg P ha^{-1} in Østfold to 14.7 kg P ha^{-1} in Rogaland, with a national average of 8.5 kg P ha^{-1}. The national average soil P balance was very close to the 8.6 kg P ha^{-1} estimated for the EU15 countries as a whole by Ott and Rechberger (2012), but somewhat less than e.g. the 13 kg P ha^{-1} estimated for Finland (Antikainen et al. 2005). The aggregated national flows and stock changes for the system are shown in Fig. 2a and a county-wise distribution of the net stock change is visualized on a map in Fig. 2b. Rogaland stands out, with a particularly high surplus due to the high amount of animal manure P, both housed and from grazing, in combination with MF. In most counties, P in sewage sludge contributed only a small part of the total P input to agricultural soil (≤13 %), but in the populous Oslo and Akershus region the sludge contribution was 35 % of the total input.

Table 4 FR0: Soil P balance

County	Inputs					Outputs		ΔS	Area[1]	ΔS/area
	MF	HM	MGA	SS	SSp	Yield	Loss			
Østfold	749	359	32	58	15	945	57	197	73,739	2.7
Oslo/Akershus	793	223	42	559	140	898	72	648	77,795	8.3
Hedmark	1215	718	138	40	10	1203	34	874	105,306	8.3
Oppland	671	946	249	42	11	1094	25	790	102,217	7.7
Buskerud	505	234	71	35	9	494	35	316	51,621	6.1
Vestfold	563	201	30	115	29	475	43	390	41,053	9.5
Telemark	194	139	42	16	4	225	9	157	24,966	6.3
Aust-Agder	91	83	29	1	0	85	7	111	11,108	10.0
Vest-Agder	116	168	61	26	6	186	18	167	18,965	8.8
Rogaland	503	1619	569	49	17	1115	161	1465	99,945	14.7
Hordaland	203	420	172	9	3	368	92	343	41,456	8.3
Sogn and Fjordane	192	536	139	4	1	452	93	327	44,584	7.3
Møre and Romsdal	324	634	164	0	0	556	99	467	56,310	8.3
Sør-Trøndelag	503	696	175	50	18	807	78	539	74,373	7.2
Nord-Trøndelag	683	976	185	4	2	981	103	765	87,183	8.8
Nordland	351	571	177	0	0	461	86	552	57,302	9.6
Troms	157	228	55	0	0	135	28	278	25,195	11.0
Finnmark	63	72	19	0	0	46	5	103	9519	10.8
Total	7875	8825	2350	1009	265	10,525	1046	8488	1,002,635	8.5

All numbers in tonnes P per year averaged for the period 2009–2011, except area in hectares (ha) and ΔS/area given as kg P ha^{-1} year^{-1}

MF, Mineral fertilizer; HM, Housed manure; MGA, Manure from grazing animals; SS, Sewage sludge to soil; SSp, Sewage sludge to soil, plant-available P; Yield, Plant yields; Loss, Erosion and run-off; ΔS, Net stock change; Area, Total agricultural area

[1] Statistics Norway 2014

Table 5 FR1 and FR2: Soil P balance and surplus fertilization

County	FR1/FR2							FR1			FR2			
	Inputs				Outputs		ΔS	FReq	SF	SF/area	Correction	FReq_c	SF	SF/area
	HM	MGA	TSS	TSSp	Yield	Loss								
Østfold	359	32	134	34	945	57	−477	945	−520	−7.0	−45.0	528	−102	−1.4
Oslo/Akershus	223	42	675	170	898	72	−29	898	−463	−5.9	−30.6	627	−192	−2.5
Hedmark	718	138	114	29	1203	34	−267	1203	−318	−3.0	−34.6	808	77	0.7
Oppland	946	249	91	23	1094	25	167	1094	124	1.2	−46.8	588	630	6.2
Buskerud	234	71	110	28	494	35	−114	494	−161	−3.1	−37.2	313	20	0.4
Vestfold	201	30	121	30	475	43	−167	475	−215	−5.2	−59.8	203	58	1.4
Telemark	139	42	72	18	225	9	19	225	−26	−1.0	−47.2	120	80	3.2
Aust-Agder	83	29	49	12	85	7	68	85	39	3.5	−52.1	42	82	7.4
Vest-Agder	168	61	73	18	186	18	99	186	62	3.3	−57.1	80	167	8.8
Rogaland	1619	569	140	50	1115	161	1053	1115	1123	11.2	−70.3	338	1900	19.0
Hordaland	420	172	61	22	368	92	192	368	245	5.9	−66.2	125	489	11.8
Sogn and Fjordane	536	139	10	3	452	93	141	452	227	5.1	−71.0	132	547	12.3
Møre and Romsdal	634	164	34	12	556	99	177	556	254	4.5	−61.7	214	596	10.6
Sør-Trøndelag	696	175	76	27	807	78	62	807	91	1.2	−47.8	422	476	6.4
Nord-Trøndelag	976	185	65	23	981	103	142	981	203	2.3	−46.3	535	649	7.4
Nordland	571	177	15	5	461	86	216	461	292	5.1	−40.5	275	479	8.4
Troms	228	55	17	6	135	28	137	135	154	6.1	−38.8	83	206	8.2
Finnmark	72	19	6	2	46	5	46	46	47	5.0	−32.6	31	62	6.5
Total	8825	2350	1864	511	10,525	1046	1467	10,525	1161	1.2	−48.1	5462	6224	6.2

All numbers in tonnes P per year averaged for the period 2009–2011, except SF/area given as kg P ha^{-1} $year^{-1}$ and Correction in %

MF, Mineral fertilizer; HM, Housed manure; MGA, Manure from grazing animals; TSS, Total sewage sludge, TSSp, Total sewage sludge, plant-available P; Yield, Plant yields; Loss, Erosion and runoff; ΔS, Net stock change; FReq, Fertilizer requirement; SF, Surplus fertilization; Area, Total agricultural area; Correction, Weighted average percentage correction of P requirement for grass, cereals, green fodder and silage, and oilseeds; FReq_c, Fertilizer requirement corrected for P-AL in soil

FR1

With manure and sewage sludge as the only P inputs, the regional surplus fertilization ranged from −7.0 kg P ha^{-1} in Østfold to 11.2 kg P ha^{-1} in Rogaland (Table 5), the national average being 1.2 kg P ha^{-1}. The segregation of animal husbandry and cereal farming has an obvious impact on the regional differences. The south-western and western counties of Rogaland, Hordaland and Sogn and Fjordane have animal densities of 1.0–1.7 manure animal units (MAU)[3] ha^{-1}, while the south-eastern counties of Østfold and Oslo and Akershus, which tend to specialize in cereal production, have animal densities of 0.3–0.4 MAU ha^{-1} (Bechmann 2005). According to the surplus fertilization data for the maintenance fertilization strategy (Table 5), 12 counties had the theoretical potential to replace the P in harvested crops by plant-available P in manure and sludge, including diversion of sewage sludge from other uses to agricultural soil. The remaining six counties (Oslo and Akershus counting as one) with negative surplus fertilization would have needed to import P fertilizer to compensate for plant P removal. The aggregated national flows and net stock change

[3] One MAU represents around 14 kg P (The Norwegian regulations relating to organic fertiliser 2003).

for the system are shown in Fig. 2c, and the county-wise distribution of the annual surplus fertilization is visualized on a map in Fig. 2d. Plant-available P in sewage sludge constituted only a minor part of the total secondary P fertilizer potential (≤12 %) in all counties except Oslo and Akershus, where sewage sludge contributed 39 % of the total potential.

FR2

When the level of plant-available soil P was taken into account, the fertilizer requirement decreased substantially in all counties (Table 5). On the national scale, the total fertilizer requirement of 5462 tonnes P in FR2 was a 48 % reduction from FR1. This reflects overall high levels of plant-available soil P in Norway, measured as P-AL. The calculation of the weighted average percentage correction of P requirement for grass, cereals, green fodder and silage, and oilseeds showed that P fertilization for these crops could have been reduced by 31–71 % relative to maintenance fertilization for the different counties in the period 2009–2011. As these crops constitute 98.4 % of total plant P yield, the overall reduction in fertilizer requirement would be in the same range. Consequently, the surplus fertilization for the period increased dramatically from FR1, ranging from -2.5 kg ha^{-1} in Oslo and Akershus to 19 kg ha^{-1} in Rogaland. The number of counties self-sufficient in P fertilizer increased from 12 in FR1 to 16 in FR2. The aggregated national flows and stock changes for the system are identical to those in Fig. 2c, while the county-wise distribution of the annual surplus fertilization is visualized on a map in Fig. 2e.

Discussion

Short-term and long-term fertilization strategy

The results strongly suggest that too much P fertilizer was applied to Norwegian agricultural soil in the period 2009–2011, particularly according to the transition fertilization strategy in FR2 compared with the maintenance fertilization in FR1. We have reason to believe that the application of P fertilizer has not changed substantially since 2009–2011. In the short and medium term, a transition fertilization strategy should therefore be followed to reduce P fertilization in line with the recommended corrections given in Krogstad et al. (2008) and incorporated into FR2. Once the optimal P-AL level of 5–7 in agricultural soil is reached, the long-term fertilization strategy should be maintenance fertilization in the direction described in FR1. The earlier build-up of legacy soil P can contribute P to crops over several decades. Refsgaard et al. (2013) concluded that reducing soil P-AL value from 20 to 10 at an annual cereal yield of 4 tonnes per ha would in theory take 34 years. The transition period will vary between counties depending on P-AL level and crop removal assuming that the recommended fertilization corrections are otherwise followed. One of the main reasons why the recommended fertilization corrections are not followed by many farmers today may be that the actual fertilizer value of bioresources such as animal manure and sewage sludge is unknown to the farmer or disregarded (see for example Johnston and Dawson 2005; Nesme et al. 2011; Refsgaard et al. 2004) and therefore they are not used to replace mineral P fertilizer. In addition, P-free mineral fertilizer may cost more than a complete NPK fertilizer on the Norwegian market. Another important factor is the lack of regulatory and economic incentives for farmers in livestock-dense areas to transport surplus manure P over greater distances (Knutsen and Magnussen 2011). This also applies to distribution of manure between fields operated by the same farmer, as the proportion of rented land and transport distances for manure are increasing with structural changes to larger farms (Bergslid and Solemdal 2014). Fields close to manure storage facilities tend to receive more manure than fields further away.

Theory versus P redistribution feasibility

In FR1 and FR2, we assumed that all P in manure and sewage sludge generated in a county could be used within that county where P fertilizer is needed. This requires a redistribution of secondary P fertilizer between farms and between municipalities,[4] where distances may be great, meaning that this is a costly endeavour, especially for bulky animal manure (Liu et al. 2008). Redistribution of secondary P fertilizer is expected to depend on economic incentives, technology, regulatory framework, institutional ownership

[4] The lowest political administrative level in Norway—a county is made up of municipalities.

and social acceptance of the use of secondary P fertilizer, in order for this theoretical potential to be fully explored (Cordell et al. 2009; Koppelaar and Weikard 2013). By not considering the challenges with P redistribution within and between regions in a country, the recycling potential may be overestimated (Senthilkumar et al. 2012). The feasibility of how and when such redistribution may take place was not examined in this study. Hence, the surplus fertilization indicating the amount of secondary P which may be exported from a county must be considered a theoretical quantity on an aggregated level, delineating what can be achieved. The drivers of redistribution will in effect decide how fast a county can move from its current P management practice into de facto transition fertilization. Nevertheless, the overall consequence of realizing the full theoretical potential in all counties is a national surplus of secondary P fertilizer in Norway as a whole, both during the transition fertilization phase and in the long term with maintenance fertilization (see SF totals in Table 5). This surplus could either be stored in a P 'bank' for later use or exported to other countries. The share of the surplus that could be absorbed by greening or horticulture is considered to be minor.

Expanded wastewater potential

In this study we only considered the amount of P in sewage sludge that can replace mineral fertilizer P, given existing technology for wastewater treatment. However, we expect wastewater treatment processes in the future to be able to recover and recycle a greater part of the P in the form of various wastewater-based fertilizer products. This expectation is based on an increased awareness surrounding P as a valuable resource [for example the inclusion of phosphate rock on the list of critical raw materials in the EU (European commission 2014)] and national efforts to reduce losses of P to waterways in compliance with the EU Water Framework Directive. In addition to P recovery from sewage sludge, there are options to source-separate sanitary wastewater, which would allow P-rich fractions such as urine or blackwater to be separately treated in systems designed for resource reuse (Langergraber and Muellegger 2005; Udert and Wächter 2012). The factors for P recovery and recycling from wastewater used in this study thus need to be revisited at a later date.

Relative regional importance of manure versus sludge

Given that there are limited resources among relevant actors to help increase recycling of P from bioresources regionally and nationally in the years to come, the results (see FR1) suggest that priority should be given to recycling and redistribution of P in animal manure in all counties. However, in Oslo and Akershus the combination of a greater population density and agricultural activity dominated by cereal production has made the P fertilizer potential in wastewater almost equally interesting. From this, we concluded that efforts to recycle secondary P fertilizer from bioresources should be informed by their relative regional importance. There will also be important insights to be gained from further disaggregating regional data to see how bioresources vary in relative importance on a smaller scale. Several cities outside Oslo and Akershus are experiencing increased urbanization and may become regional hotspots for secondary P from wastewater and organic household waste (Cordell et al. 2012), even though animal manure dominates the county as a whole.

Conclusions

This study explored the theoretical potential of the bioresources animal manure and sewage sludge to supply the P fertilizer requirement of crops in Norway. It was found that if P in these resources were to be well redistributed within and between counties, Norway as a whole could be self-sufficient in P fertilizer for all crops assessed in both in the short and the long term. Taking the recorded levels of plant-available soil P into account substantially decreased the amount of P fertilizer required compared with a maintenance fertilization strategy assuming optimal soil P levels. Maintenance fertilization and an optimal soil P level are the goal in the long run, but overestimate the P fertilizer requirement in Norway in the short term. Similarly, the maintenance fertilization strategy underestimates the potential of bioresources to supply the crop P fertilizer requirement in the short term in regions with high levels of plant-available soil P.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.

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Paper III

Recycling potential of secondary phosphorus resources as assessed by integrating substance flow analysis and plant-availability

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Recycling potential of secondary phosphorus resources as assessed by integrating substance flow analysis and plant-availability

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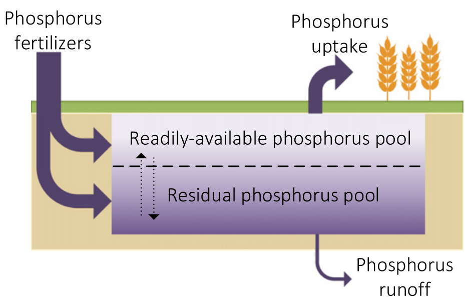
HIGHLIGHTS

- Plant availability of secondary phosphorus resources vary.
- Phosphorus plant availability-extended substance flow analysis method is developed.
- Relative agronomic efficiency and substance flow analysis are integrated.
- Provides national level assessment of secondary P resource quality and quantity.
- Norwegian case highlights manure's large resource potential.
- Imports of food and feedstuff are reason for large secondary P surpluses.
- Secondary phosphorus needs regulation to avoid soil accumulation.

GRAPHICAL ABSTRACT



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ABSTRACT

The plant-availability of phosphorus (P) plays a central role in the ability of secondary P resources to replace mineral fertilizer. This is because secondary P plant-availability varies, often with large fractions of residual P that has no immediate fertilization effect. Therefore, if low quality secondary P fertilizers are applied, they will accumulate in soils that, in the long run, may increase the risk of P runoff and eutrophication. Substance flow analyses (SFA), used to identify potentials for improved P management, have not considered this well-known quality barrier. We, therefore, argue that traditional SFA over-estimates the fertilizer potential of secondary P resources. Using Norway as a case, we present a plant-availability extended SFA methodology that integrates SFA and the concept of relative agronomic efficiency. To account for the plant-available soil P stock and long-term soil interactions, we adjust the Norwegian P fertilization demand based on soil P values. We found that, while the method has uncertainties particularly for long-term estimations, it more realistically estimates secondary P fertilizer potentials and is adaptable to other countries. For Norway, we found the overall secondary P fertilizer potential reduced by 6–55% when considering plant-availability. The most important secondary resource was manure, which had the highest P plant-availability and quantities large enough (10.9 kt plant-available P/yr) to meet Norway's entire P fertilization demand (5.8 kt plant-available P/yr). However, barriers related to its transportability need to be overcome to efficiently use this resource. Fish sludge was also an important product, with 6.1 kt plant-available P/yr but with uncertain plant-availability data. We argue that high quality secondary P resources can theoretically meet Norway's P fertilization demand and, therefore, make Norway mineral P independent. However, it is important that their use is carefully regulated based on plant-availability to eliminate the soil accumulation of both available and residual P.

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1. Introduction

Phosphorus (P) is an essential nutrient for food production and, currently, P fertilizer demands in developed countries are primarily met through the mining of limited, non-renewable phosphate rocks. Overall, food production accounts for above 90% of the global mineral P demand, highlighting the vast importance of fertilizer consumption in the anthropogenic P cycle (Brunner, 2010). Moreover, agricultural P use is inefficient implying minimum recycling, large losses and soil accumulation (Cordell et al., 2009). Therefore, in order to sufficiently address P management and avoid potential P scarcity, one of the primary research strategies must focus on reducing the dependency on mineral fertilizer through the effective recycling of secondary P resources as fertilizer (Brunner, 2010).

Substance flow analysis (SFA) is often used to describe the anthropogenic P cycle, estimate recycling potentials and identify opportunities for improving P management through reducing mineral P consumption (Matsubae-Yokoyama et al., 2009; Ott and Rechberger, 2012; Suh and Yee, 2011; Seyhan, 2009; Antikainen et al., 2005; Senthilkumar et al., 2014; Cordell et al., 2013; Cooper and Carliell-marquet, 2013; Hamilton et al., 2015; Klinglmair et al., 2015; van Dijk et al., 2016). Strategies for realizing the latter are largely centered on replacing mineral fertilizers by recovering and returning secondary P resources, such as sewage sludge, to agricultural soils (Senthilkumar et al., 2014; Hamilton et al., 2015; Klinglmair et al., 2015; van Dijk et al., 2016). Senthilkumar et al. (2014), for example, estimated that France could displace 21% of their mineral fertilizer demand if they utilized secondary P in municipal waste and sewage sludge. These studies, however, neglect one of the greatest barriers to secondary fertilizer recycling: the plant-availability of P in secondary P resources.

Secondary P forms vary considerably, often with large fractions of P bound in complex and slowly soluble compounds (hereafter referred to as residual P) that are not immediately available to plants. Chemically precipitated sewage sludge from wastewater treatment, for example, has relatively low P fertilization effects due to the high percentage of P that is adsorbed to Fe-/Al-hydr(oxides) or present as Fe-/Al-phosphates (Krogstad et al., 2005). Therefore, it is important to emphasize that the return of secondary P resources to agricultural land is not equal to efficiently recycling P. If applied, resources with high fractions of residual P can accumulate in non-labile P soil pools (Brod et al., 2015a), which can pose an environmental risk following soil P erosion to surface waters and subsequent eutrophication (Brod et al., 2015a; Rekolainen et al., 2006; Ekholm et al., 2005). While accumulated residual soil P can, over time, contribute to the readily-available P pool through soil transfer mechanisms, the rate of release is often too slow to meet the critical P amount needed by the crop to achieve optimal yields (Syers et al., 2008).

Due to the above, we hypothesize that the use of SFA without accounting for plant availability over-estimates the potential of secondary P resources to replace mineral fertilizers. Studies have highlighted the need for exploring the quality of secondary P resources (Matsubae-Yokoyama et al., 2009; Hanserud et al., 2016). Ringeval et al. (2014), for example, recognized that secondary P resources, such as sewage sludge, have both a plant-available and plant-unavailable fraction. Nonetheless, SFA has not been adapted to include these aspects. Such advancements are important in order to i) identify suitable replacements for mineral P fertilizer that do not contribute to soil P accumulation and ii) obtain more realistic estimates of their potential at a systems level. Low quality secondary P resources can then be evaluated for use in other secondary P applications, e.g. as raw ingredients for animal husbandry feed or fish feed, or for alternative treatments that increase P availability.

Here, we use the concept of relative agronomic P efficiency (RAE) [also called mineral fertilizer equivalents (MFE)] to integrate plant availability into P SFA's. RAE is an established measure for the P fertilization effects of secondary P resources compared to mineral P application and is determined by growth experiments. We, therefore, integrate the results from secondary P resource growth experiments for manure, fish sludge, food waste, meat bone meal, sewage sludge and fish scrap with SFA and apply this adapted method to Norway. In addition, we account for the plant-available soil P and long-term soil interactions by adjusting the Norwegian P fertilization demand through the use of soil P values. We answer the following research questions:

i) Which secondary P resources have the highest potential, based on plant-availability extended SFA, for providing immediate substitutes for mineral fertilizer? What is their potential at a national level?
ii) Which secondary P resources should be avoided as secondary fertilizers due to their low plant-availability and, thus, large risk for soil accumulation if applied?
iii) What is the Norwegian P fertilization demand after accounting for the plant-available soil P stock? Can the use of high quality secondary P resources meet the Norwegian P fertilization demand?
iv) What are the strengths and weaknesses of the developed plant-availability extended SFA approach?
v) What are the policy implications of the plant-availability extended P SFA for Norway? What are the additional barriers to overcome for efficiently using secondary P resources as fertilizer in Norway?

2. Methods

In this study, we combined the concept of RAE, a measure of the fertilization effects of secondary P resources, with SFA in order to identify and assess the magnitude of secondary P resources that, based on plant-availability, best serve as immediate replacements for mineral fertilizer. This was done by expanding upon an existing SFA of P flows in Norway (Hamilton et al., 2015) by multiplying total P flows for secondary P resources by their corresponding RAE values to obtain the total plant-available secondary P potential. In addition, we accounted for the existing plant-available soil P stock, which serves as an additional P source for plants, through the use of soil P values. The following paragraphs detail each of these concepts, our related assumptions and how we combined these to develop our plant-availability extended SFA methodology.

2.1. Substance flow analysis

SFA tracks the flows and stocks of a given chemical element or compound throughout a defined system as it obeys mass balance (Brunner and Rechberger, 2004). In the case of P, this method serves as a powerful tool for quantifying the losses, accumulation, inefficiencies and drivers of P in order to identify potential future challenges, scarcity and environmental pollution (Brunner, 2010). Here, we build upon previous research (Hamilton et al., 2015) and use an SFA of P flows in Norway as a case. We maintained the same time frame (2009–2011) and system boundaries, defined as the economic zone of Norway including water bodies where Norwegian aquaculture and fisheries operate, as the original study. Based on the results, here, we determined the most important potential secondary P resources in Norway for further consideration as secondary P fertilizer. We excluded: i) resources that were better utilized at a higher trophic level, e.g. animal husbandry feed and ii) P losses that were a result of technological treatment limitations, such as the P remaining in treated wastewater that was discharged to water bodies. For further information regarding the methods and results from this study, refer to Hamilton et al. (2015).

2.2. Technology assumptions

We considered the treatment technologies applied to secondary P resources in 2009–2011. Exceptions were i) fish excrements/feed losses generated by marine aquaculture (later referred to as fish sludge), which were not collected in 2009–2011, ii) fish scrap where losses entered directly to water bodies untreated and iii) food waste that was incinerated or exported. For these secondary P resources, we assumed treatment technologies based on available data and the most likely pathways for secondary fertilizer recycling in Norway. For fish sludge, we obtained an RAE value for only one resource treated via reactor composting (Brod et al., 2015a) and, due to data limitations, applied this to all generated fish sludge. Also due to data limitations, we assumed the plant-availability of P in fish scrap to be similar to that of meat bone meal, as P in fish scrap is also primarily sourced from the bones (Adler et al., 2014; Trøite, 2007). For food waste treatment, we assumed 50% composting and 50% anaerobic digestion.

2.3. Relative agronomic efficiency (RAE)

In order to quantify the plant-available fraction of P in secondary P resources, we applied the method of RAE. This relative measure was chosen over absolute metrics for fertilization effects, such as P use efficiency (PUE) and P solubility. RAE and PUE differ in that RAE uses water-soluble mineral fertilizer as the reference, i.e. mineral fertilizer is defined as 100% RAE, while PUE strongly depends on local soil conditions and is typically in the range 10–25% of fertilizer P (Kratz et al., 2010). Calculated RAE values (Table 1) were multiplied by total P flows, as calculated by SFA, to obtain the flows of plant-available P within the Norwegian P system.

Fig. 1 provides a schematic illustrating how RAE is defined. The soil stock is divided into two pools: a readily-available P pool and a residual P pool. The readily-available P pool represents the P that is immediately available to plants while the residual P will only be made available to plants through microbial or chemical processes (the upward dotted arrow). For optimal yields, plants require a critical amount of P in the readily-available P pool. Therefore, for most soils, readily-plant available P must be added, as the rate of transfer from the residual pool to the readily-available pool is insufficient to reach this critical value at the time needed by the crop (Syers et al., 2008). The downward dotted arrow represents the conversion of available P to residual P over time. As shown in Fig. 1, mineral fertilizer is used as a benchmark, as all water-soluble mineral P will be readily-available for plants (RAE = 100%, per definition). On the basis of P plant uptake, RAE estimates the fraction of the secondary P fertilizers that will enter the readily-available soil P pool and, thus, can substitute water-soluble mineral P fertilizer during the first growing season. The remaining P in secondary P resources enters the residual P soil pool. Therefore, the higher the fraction of secondary P that enters the readily-available P pool, the better it serves as a replacement for mineral P fertilizer and the less it will contribute to accumulation of P in the soil (Ekholm et al., 2005). It is important to note that RAE values above 100% can be observed. This is due to the increased flow from the residual P stock to the readily-available P stock relative to mineral fertilizer (indicated in Fig. 1 as the upward dotted arrow), e.g. if low molecular organic acids in organic fertilizers replace phosphate on soil particles (Øgaard, 1996). In addition, we underline that residual P that is not available to plants might still be utilized by algae and, thus, can contribute to eutrophication (Rekolainen et al., 2006; Ekholm et al., 2005; Ekholm and Krogerus, 2003).

RAE values were primarily sourced from Norwegian pot experiments of one-year duration (Table 1). Using a nutrient-deficient sand-peat mixture as experimental soil and ryegrass (*Lolium multiflorum*) as the experimental crop, these experiments compared the fertilization effects of secondary P resources to i) no P treatment and ii) mineral P fertilizer. All other nutrients were applied in sufficient amounts. For more detailed information regarding the methods, refer to Brod et al. (2015a, 2015b). Norwegian values were supplemented with values from relevant international studies on pot and field experiments. All obtained values were weighted equally. We assume that the obtained RAE values are representative of Norwegian conditions. For secondary P resources with no available RAE values, we used similar resources as proxies. It is important to note, however, that these resources do not represent large amounts of P. For horse manure, we assumed an RAE of 80% due to a lack of data. Mechanically treated sludge was assumed to have 100% RAE based on Hanserud et al. (2016).

To express the uncertainty of RAE values of secondary P resources, we calculated 95% confidence intervals of all collected RAE values based on the assumption that the RAE values are normally distributed (Table 1). For sheep and goat manure, for which we had only one observation, we applied an uncertainty of ±25% to the given value to produce uncertainty ranges. For horse manure, dumped fish scrap and mechanically treated sewage sludge, where there were no observations, we applied an uncertainty of ±25% to their assumed RAE values.

Table 1
Secondary P resources and their respective relative agronomic efficiency (RAE) ranges using 95% confidence intervals. References included. RAE = relative agronomic efficiency; n = number of observations.

Secondary P resource	Category/treatment	RAE (%)	n	Reference
Manure	Cattle manure	[82; 112]	15	Brod et al. (2015b), During et al. (1973), Goss and Stewart (1977), Larsen (1981), Motavalli et al. (1989), Smith and Van Dijk (1987), Leytem and Westermann (2005), Delin and Nyberg (2015)
	Sheep and goat manure	[75; 125]	1	McAuliffe et al. (1949)
	Pig manure	[77; 123]	6	Smith and Van Dijk (1987), Leytem and Westermann (2005), Tunney and Pommel (1987)
	Poultry manure	[63; 73]	4	Brod et al. (2015b), Smith and Van Dijk (1987), Delin and Nyberg (2015)
	Fox and mink manure	[29; 94]	2	Delin and Nyberg (2015), Ylivainio et al. (2008)
	Horse manure	[55; 105]		Own assumption
Fish sludge	Reactor-composted	[21; 115]	3	Brod et al. (2015b), Brod et al. (2016)
Food waste	Compost	[39; 65]	10	Brod et al. 2015b, Sinaj et al. (2002)
	Digestate	[55; 86]	4	Brod et al. (2015b)
Meat bone meal	Treated with heat and pressure	[19; 60]	5	Brod et al. (2015b), Delin and Nyberg (2015), Ylivainio et al. (2008), Bøen et al. (2006)
Sewage sludge	Chemical or chemical-biological treatment	[20; 37]	9	Øgaard and Brod (2016)
	Biological treatment	[75; 125]	1	Krogstad et al. (2005)
	Mechanical treatment	[75; 125]		Own assumption
Dumped fish scrap[a]		[19; 60]		Own assumption

[a] Assumed to be equal to meat bone meal.

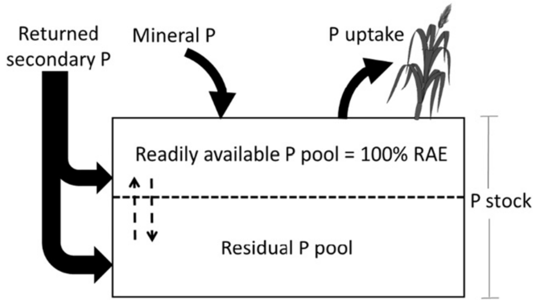

Fig. 1. Conceptual drawing of relative agronomic efficiency (RAE).

2.4. P fertilization demand

The total potential of Norwegian secondary P resources based on plant-availability was compared with the domestic demand for P fertilization. The demand for P fertilization, however, depends on the strategies and policies surrounding the management of the existing soil P stock, as plant-available soil P can be used as a P source by plants. The soil stock accumulated as a result of the over-application of P to agricultural soils over the past decades. Today, new fertilizer recommendations (2008) aim at reducing these stocks with the goal of, over time, reaching a critical soil P level that is considered optimal in Norwegian fertilizer planning to both optimize yield and reduce the risk of surface runoff (Krogstad et al., 2008). Therefore, calculations for the P fertilization demand need to account for the targeted stock changes. In this study, we use the new fertilizer recommendations as a reference and develop a national fertilization correction factor that accounts for this additional P source. It is important to note that, in this study, we accept the methods for which the fertilization recommendations were based on. There are other methods for estimating plant-available soil P, e.g. water extractable P (Pote et al., 1996) or Olsen P (bicarbonate extraction) (Horta and Torrent, 2007), which we do not consider.

In Norway, plant-available P in soils is estimated as P extractable in 0.1 M ammonium lactate and 0.4 M acetic acid adjusted to pH 3.75 (P-AL) (Egnér et al., 1960). Norwegian fertilization recommendations advise P fertilizer application to be reduced to below the amount of P removed by plants in areas with high P-AL values (>7 mg P-AL 100 g^{-1} soil). This is in order to, over time, reduce soil P stocks to reach the optimal soil P concentration, which is considered to be zero balance where P uptake equals P fertilization (Krogstad et al., 2008). While Norwegian plant-available soil P values vary geographically both above (this is primarily the case) and below the optimal soil P concentration, we have applied a weighted average national correction factor (Hanserud et al., 2016). This factor corrects for the fertilization effects from soil P for the following crops: grass, cereals, green fodder and silage, and oilseeds, which together account for 95.5% of the agricultural land use (Statistics Norway, 2014). For the remaining crops including vegetables, fruits, peas and potatoes, we assumed no correction in P demand. The weighted correction factor was based on Hanserud et al. (2016) and was calculated by combining country wide, farm-level P-AL data available in the soil database administered by the Norwegian Institute of Bioeconomy Research with percentage correction of P requirement recommended for different classes of P-AL levels (Krogstad et al., 2008). A central assumption made is that the P-AL data in the database are representative for all soils used for grass, cereals, green fodder and silage, and oilseed production and that the methods used to develop the fertilization recommendations are correct. It is also important to note that this is a moving correction factor. The study presented here is static and, thus, analyzes the situation today. Long-term changes in the soil P pool values due to i) lowered fertilizer application compared to plant uptake, ii) changes in farming practices, iii) soil conditions, iv) transfer of P from the residual to the available pool and v) transfer of P from the readily-available pool to the residual pool could require an upward or downward adjustment of the correction factor.

2.5. Scenario

We applied the calculated RAE values with the aforementioned treatment technology and plant-availability assumptions to the 2009–2011 system. The system structure was held constant and, thus, no changes were made regarding fertilizer application or the return of secondary P resources as compared to Hamilton et al. (2015). However, we included a fertilizer market that includes all primary and secondary P resources, in order to enable the visualization of the theoretical recycling potential and their subsequent uses. The presented RAE corrected system, using average RAE values, does not allow for mass balance consistency because the RAE values only consider a fraction of the P contained in the flows. We, therefore, included the total P content in parenthesis for flows of returned secondary P resources in order to enable mass balance consistency for the plant production process. In addition, we applied the fertilization demand correction factor to the system and compared applied P with the calculated demand for plant-available P based on the existing soil P stock in 2009–2011.

3. Results

A systems-level comparison of the total P and plant-available P (midpoint RAE value presented) balances for Norway is shown in Fig. 2. The system consists of 6 processes: plant production, animal husbandry, aquaculture/fisheries, food processing, human consumption and waste management. P flows begin with the application of fertilizer (both primary and secondary) to agricultural soils. Plants uptake P (11 kt/yr) to produce feed for animal husbandry and crops for human consumption and surplus P is accumulated in the soil stock (12 kt P/yr). Animal husbandry and aquaculture import feedstuff (4.4 and 9.4 kt P/yr, respectively) and fisheries supply aquaculture with caught fish (14 kt P/yr) for feed purposes. Animal, fish and plant products are processed into products for export (8 kt P/yr) and domestic consumption (3.5 kt P/yr). In addition, Norway imports substantial amounts of food for human consumption (4.9 kt P/yr) and wastes produced from domestic consumption are treated in the waste management process. These background flows were the same for both total P and plant-available P (in grey).

Secondary P resources (currently exploited and potentially exploitable resources) were corrected based on plant-availability with total P flows highlighted in red (Fig. 2 top) and plant-available P flows highlighted in blue (Fig. 2 bottom). These resources are sent to a hypothetical fertilizer market to allow for the visualization of utilized (applied secondary products) versus unutilized resources (losses) and the calculation of secondary plant-available P efficiency. From this, we found that the Norwegian system poorly utilizes plant-available P, as shown by the losses from the fertilizer market out of the system. Between 1.8 and 10.3 kt plant-available P/yr of fish sludge and 0.1–0.6 kt plant-available P/yr of fish scraps was lost to water bodies from aquaculture. In addition, between 0.5 and 1.3 kt plant-available P/yr was incinerated, landfilled or exported. Furthermore, we found that P was heavily over-applied to agricultural soils. We estimated that during the 2009–2011 period, between 11.8 and 15.0 kt plant-available P/yr was over-applied to agricultural soils and included with these resources was an additional 2.8 kt of residual P/yr. This contributed to the large net accumulation of stock (12 kt P/yr), owing to the over-application of both plant-available and residual secondary P.

When comparing the two balances (Fig. 2 top and bottom), we found that considering plant-availability significantly reduced the recycling potential of most secondary P resources. Nonetheless, there is still substantial opportunity to substitute mineral P fertilizer, with

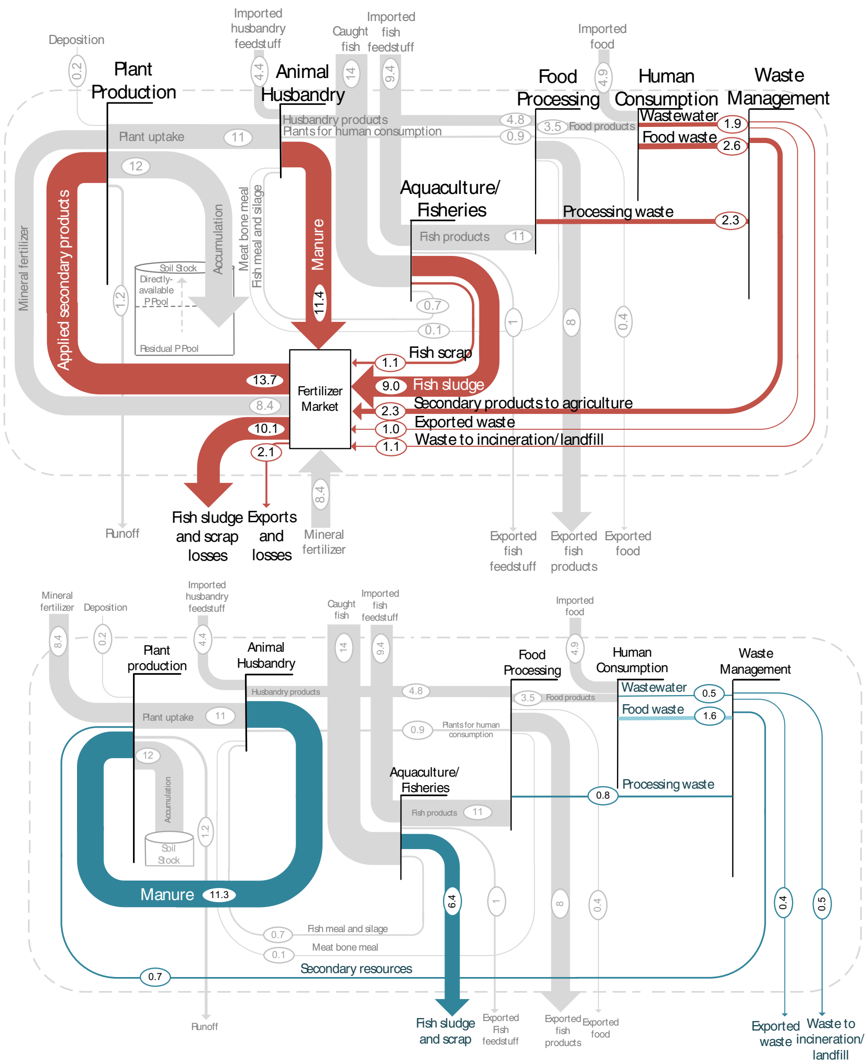

Fig. 2. Norwegian phosphorus balance (top), kt P/yr, averaged 2009–2011, red flows highlighting the secondary P resources considered; Norwegian phosphorus balance corrected for RAE (bottom), RAE midpoint values presented, kt plant-available P/yr, averaged 2009–2011, blue flows highlighting the resources that RAE were applied to.

manure showing, by far, the largest potential. P in manure is not only highly plant-available, with between 76 and 100% RAE (weighted RAE average for all manure types), but also present in large quantities nationally, with between 8.7 and 11.4 kt plant-available P/yr (Fig. 2, bottom). Therefore, even when considering the minimum plant-availability of P in manure, this resource alone could potentially meet the fertilization demand of Norway (estimated at 5.8 kt plant-available P/yr), while avoiding the accumulation of residual P (Fig. 3). Furthermore, we found that, while fish sludge is a secondary P resource present in large amounts (9 kt total P/yr), the low number of RAE observations that produced a wide confidence interval (between 21 and 115% RAE) requires more research before concluding on its ability to immediately replace mineral fertilizer. In addition, we found that food waste represents another resource that has a relatively high plant-availability of P with 41–75% RAE (Table 1), however, in comparatively small amounts of 1.2 and 2.0 kt plant-available P/yr. It was determined that chemically and chemical-biologically treated sewage sludge, meat bone meal and fish scrap represent relatively poor immediate substitutes for P in mineral fertilizer. Chemically and chemical-biologically treated sewage sludge had particularly high fractions of residual P of 63–80%. Similarly, meat bone meal and fish scrap contained high amounts of residual P with between 40 and 81%.

Overall, we found that the plant-available fertilizer recycling potential of Norwegian secondary P resources ranged from 12.7 to 26.3 kt plant-available P/year (Fig. 3), which was a reduction of between 6 and 55% as compared to total P. Correcting for the existing plant-available soil P resulted in a Norwegian fertilization demand of 5.8 kt plant-available P/yr, which was a reduction of 48% for the P fertilization demand of grass, cereals, green fodder and silage, and oilseeds. Therefore, plant-available P in secondary P resources is much larger than the demand for plant-available P according to the fertilizer recommendations.

4. Discussion

The RAE-extended SFA method provides a first approximation for assessing the plant-availability of P in regional systems, thereby enabling the evaluation of fertilization strategies at the regional/national levels. The inclusion of plant-availability of P within SFA is an important advancement for P assessments, as it provides a more realistic estimation of the secondary P fertilizer recycling potential, highlights the most promising secondary P resources in terms of amounts and mineral P substitutability and corrects for long-term soil interactions. In traditional SFA, there is a risk of focusing on secondary P resources that are significant in terms of total P amounts but not from a plant-availability perspective. A lack of consideration for the plant-availability of recycled secondary P resources i) increases the risk for environmental impacts and ii) cheats the farmers who purchase those resources with the purpose of having a positive fertilizer effect.

We, therefore, argue for i) secondary P resources to be evaluated based on their quantity and quality, ii) P treatment technology choices to be based on their ability to produce high quality secondary P resources and iii) systems evaluation for optimal use of primary and secondary P resources. Possibilities for using secondary P in other applications, such as ingredients in animal and fish feed, were not evaluated in this study due to the vast number of additional food and safety barriers associated with using waste resources as feed (Adler et al., 2014). In the future, however, these potentials should be explored in order to fully optimize the use of secondary P resources. These steps are essential for finding viable substitutes for mineral P, closing P resource loops (both residual and plant-available), reducing the risk for environmental degradation and, thus, are central for long-term P management.

4.1. Method

4.1.1. Strengths and limitations of plant-availability and P fertilization demand estimates

4.1.1.1. Technology. As technologies evolve over time, especially if technology choices are based on secondary P plant-availability, the RAE values will need to be adjusted. A potential shift towards technologies that produce high quality secondary P resources could be, for example, advanced P recovery from sewage sludge. Currently, the widespread use of chemical precipitation in Norwegian wastewater treatment plants removes 90 to 95% of the P contents in wastewater but leaves it in a mostly unavailable form (Krogstad et al., 2005; Øgaard and Brod, 2016; Berge and Mellem, 2012). However, a range of studies examine the development of alternative solutions for technically advanced P recovery from sewage sludge (Schoumans et al., 2015). Nanzer et al. (2014), for example, found that P fertilizer from sewage sludge ash prepared with $MgCl_2$ as chemical reactant during thermal treatment had RAE of 71 and 88% after application to an acidic and a neutral soil. This is clearly higher than current RAE of Norwegian sewage sludge treated chemically or chemical-biologically (20–37%) shown in Table 1.

4.1.1.2. Time aspect. RAE studies conducted over 1-year time scales neglect the long-term fertilization effects of secondary P resources. These effects have been shown to be potentially important; Bøen and Haraldsen (2013), for example, found that the P fertilization effects of meat bone meal and biosolids were higher than the unfertilized control the third year after application to a silty loam, while there were no differences between the fertilizer treatments and the unfertilized control the second year after application. Here, the long-term fertilization effects of secondary P resources were approximated through the use of the weighted correction factor derived from soil P tests (error related to this approach covered later). Increased flow from the residual soil stock (i.e. where the residual P from secondary P resources would accumulate) to the readily-available soil P stock would be reflected by the soil P tests. This would, therefore, result in a lowered demand for plant-available P, allowing for reduced plant-available soil P application. While we cannot comment on local soil variability, the developed moving correction factor would be able to account for changes in the plant-available soil P stock that take place over time and at a national level. Further research, however, is needed to better understand the long-term stock dynamics of P in different soil types.

4.1.1.3. Variability of RAE values. The RAE values for specific secondary P resources, such as meat bone meal and fish sludge, were highly uncertain, as shown by the presented confidence intervals (Table 1). This was due to either i) the statistical treatment of data with few observations or ii) the RAE of many secondary P resources being dependent on the pH of the target soil. Regarding i), this is particularly important to keep in mind when interpreting the results for fish sludge. The RAE range for fish sludge was 21–115%, which was primarily due to the low number of observations (3 RAE values). While fish sludge RAE values of over 100% are theoretically possible (Ogaard, 1996), it is unlikely that they would be to the extent presented here. In terms of ii), several secondary P resources contain P in the form of stable Ca-phosphates (Brod et al., 2015a). The solubility of stable Ca-phosphate decreases with increasing soil pH (Lindsay, 1979), thus, explaining the wide confidence interval for the RAE of meat bone meal (Table 1), in which P is mainly present as apatite (Brod et al., 2015a). The P fertilization effect for fish sludge has also been shown to be dependent on soil

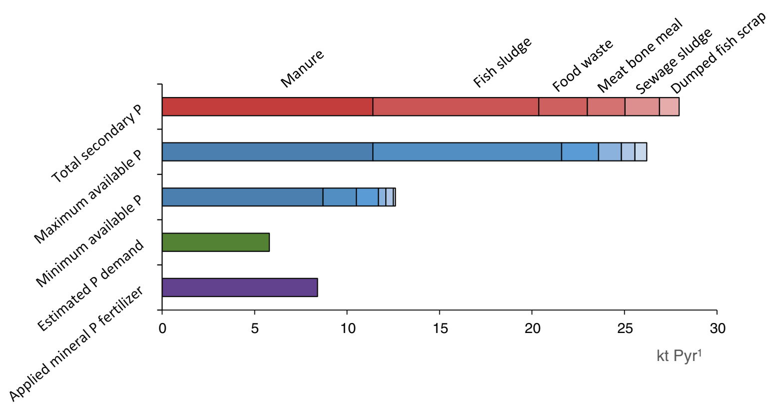

Fig. 3. Total secondary P, maximum and minimum P recycling potential of secondary P resources, estimated P fertilization demand and mineral P fertilizer applied in 2009–2011.

pH (Brod et al., 2015a), owing to the wide range of observed RAE values. Therefore, the RAE values presented serve only as indicative values at a national level, thus, further highlighting their limitations on local soil conditions.

4.1.1.4. Data availability. The validity of the horse manure, fish scrap and mechanically treated sewage sludge assumptions on P plant-availability is unknown due to a lack of experimental studies on these particular secondary P resources. However, the P amounts in these resources were insignificant relative to the overall system and, therefore, the aforementioned assumptions are unlikely to have significantly influenced the results.

4.1.1.5. P fertilization demand. The estimated P fertilization demand value presented in this study is applicable for today's situation. Therefore, as changes in the soil stock occur over time and fertilization recommendations/policies change, the correction factor will have to be adjusted to account for the new plant-available soil P value. This allows for the plant-availability P explicit methodology to account for the long-term fertilization effects of secondary P resources and changes within the soil stock due to soil interactions. Nonetheless, this method relies, amongst others, on the assumptions that i) the extraction methods and subjectivity of the fertilization recommendations are accurate and ii) the soil P level data that forms the basis for calculating the correction factor are representative. Errors related to both assumptions were considered out of the scope of this study, as the Norwegian fertilization recommendations were not questioned. However, potential error related to i) the Norwegian soil P testing techniques could be due to the dissolution of calcium phosphates when using the extractants 0.1 M ammonium lactate and 0.4 M acetic acid adjusted to pH 3.75. Calcium phosphates, that are otherwise unavailable to plants, would be measured as available and, thus, artificially inflate the plant-available P values. Nonetheless, we do not correct for these sources of errors. For ii), we assume that the measurements behind the data are representative for all cropland at the studied scale (national in this case) for which the P requirement is corrected. As there are variations in soil characteristics on any scale, even within a farm, that soil measurements are unable to capture, the correction factor only serves as an indicative value. Hanserud et al. (2016) evaluated the measurements at a regional scale and found that all 19 Norwegian regions (counties) were covered satisfactorily - except one that represents only 2.5% of the national agricultural area. A more detailed description of the method and its uncertainties can be found in Hanserud et al. (2016).

For a detailed discussion of traditional SFA methodology as it was applied to P flows in Norway, refer to Hamilton et al. (2015).

4.1.2. Applicability of the method

As SFA studies on P are widespread, this approach can be adapted to other countries or regions by supplementing or replacing the above RAE values with region and resource specific values. To account for the demand for fertilizer P and long-term fertilization effects, it is also important to account for the plant-available soil P stock through soil P testing. Because the uncertainties associated with this method will decrease with improved data availability, more research oriented towards developing and testing the plant-availability of P in secondary P resources is needed in order to apply this on a large scale.

4.2. P recycling potential in Norway

While this study focused on assessing secondary P resources in terms of quality (i.e. P plant-availability) and quantity (i.e. plant-available P amounts at a national level), several other factors play a role in determining the suitability and/or feasibility of secondary P fertilizer use. In Table 2, we present an overview of Norwegian secondary P resource characteristics, both qualitative and quantitative, for comparison. The characteristics included were: i) 'P plant-availability', where plant-available resources were defined to have high RAE values, ii) 'transportability', where transportable resources were defined to include resources that have high dry matter contents, iii) 'nitrogen to phosphorus ratio', where crop demands defined resources with favorable ratios, iv) 'quantity', where resources with large quantities were defined to include resources that, at a national level, can fulfill the P fertilization demand and v) 'accessibility', where accessible resources were defined to include resources that are non-dispersed and collected centrally.

Based on the aforementioned criteria, we chose the resources with the highest potential as secondary P fertilizers and, in the following paragraphs, we discuss the largest barriers for efficiently utilizing them within the Norwegian food system.

4.2.1. Manure

We have shown that, in Norway, manure alone has a large enough potential, both in terms of amounts and plant-availability, to meet the entire P fertilization demand, with approximately 40% still remaining. Therefore, in theory, Norway could be fully mineral fertilizer independent and, even more, export the remaining secondary P from manure and other resources to countries that produce the animal and fish feed that Norway imports. However, despite manure's large potential, in Norway, it is not efficiently used as a secondary P resource, with the primary barrier being the spatial distribution between areas with P surpluses (areas with intensive animal husbandry) and areas with P deficits (crop production areas) (Klinglmair et al., 2015; Hanserud et al., 2016; Senthilkumar et al., 2012). Large discrepancies in regional P balances due to a lack of manure management are an international challenge, particularly for countries with intensive animal husbandry, including Norway (Hamilton et al., 2015; Hanserud et al., 2016), the United States (Kleinman et al., 2011), the United Kingdom (Kleinman et al., 2015), Northern Ireland (Kleinman et al., 2015) and Canada (Whalen and Chang, 2001), to name a few.

For this to be overcome, measures can target i) reducing animal density or ii) improving the cost and energy effectiveness of transporting manure. The latter can be accomplished by implementing new technologies that reduces manure's water content or extract P (as well as other essential nutrients), as for example suggested by Achat et al. (2014). One way of incentivizing this technological development could be to implement stricter regulations for manure use, as the current practices will not change within the regulatory frameworks. This is especially the case when the low cost of mineral fertilizer provides little economic incentive for efficiently managing manure.

Table 2
Secondary P resource characteristics that determine the suitability/feasibility of their use as mineral fertilizer substitutes in Norway. Fields are marked with a rating scheme: (+) denotes resources that exhibit the corresponding characteristic; (−) denotes resources that do not exhibit the corresponding characteristic; (*) represents resources where the characteristic is uncertain/variable.

	Manure	Fish sludge	Sewage sludge	Meat bone meal	Fish scrap	Composted food waste	Anaerobically digested food waste
P plant availability	+	*	−	−	*	−	+
Transportability	−	−	+	+	+	+	−
Nitrogen to phosphorus ratio	−	−	−	−	−	−	+
Quantity	+	+	−	−	−	−	−
Accessibility	+	−	+	+	−	+	+

In Norway, current regulations do not require the efficient utilization of manure P despite the manure spreading area being based on the manure's P content: In order to manage livestock density, the minimum farm size has to be ≥0.4 ha per livestock unit (a measure of an animal's P excretion), which correlates to an upper limit of 35 kg P/ha (Amery and Schoumans, 2014; Lovdata, 2003). In addition, farmers are required to make fertilization plans for their agricultural areas based on the expected crop yield and soil P levels in order to receive production subsidies. However, this plan is not further controlled to ensure that it is being followed accordingly (Delin and Nyberg, 2015). Furthermore, there are currently no obligations to limit the over-application of P from both mineral P fertilizer and secondary sources (Lovdata, 2003). Measures for addressing this could, therefore, include government subsidies or the implementation of stricter limitations on total P application (mineral fertilizer and secondary P resources) per soil area with an adaptation to soil P levels, as done in e.g. Sweden and the Netherlands (Kleinman et al., 2015; Smit et al., 2015; The Swedish Board of Agriculture, 2012).

4.2.2. Fish sludge

Our results highlight that, in Norway, fish sludge represents a large fraction of the P recycling potential (Fig. 3) and, therefore, improvements in fish sludge handling in marine aquaculture are crucial for sustainable P management. Moreover, the importance of P management within aquaculture is ever-increasing due to the sector's rapid growth and anticipated quintupling in salmon production by 2050 (DKNVS and NTVA, 2012). This expected expansion will heavily depend on imported plant-based fish feed ingredients that have high P fertilizer requirements abroad and, thus, potential resource limitations. Therefore, such growth will likely shift P cycles also on a global scale (Hamilton et al., 2015).

Overall, the P from aquaculture waste is lost directly to water bodies. Therefore, an essential first step for utilizing this P is to develop technologies that collect and recycle P in fish sludge from Norwegian offshore and onshore aquaculture pens. Options for incentivizing this technological development could be in form of government regulations that forbid P losses to fjord systems. Industries would then have to develop, mature and implement technologies that recover P and produce transportable, fish sludge-based fertilizer resources that can be used in Norway and abroad. Today, there are only few technologies in place to recover P from aquaculture. However, ongoing pilot projects for integrated multi-trophic aquaculture, that harness excess nutrients through the growth of macroalgae in the proximity of fish farms (Wang et al., 2014), might provide a large-scale solution for recovering P. Additional potential future technologies include (semi-) closed land-based systems, where the fish sludge can be collected and e.g. be treated by anaerobic digestion to produce biogas (Aspaas et al., 2014; Tal et al., 2009). It is important to note that harnessed aquaculture P can also be used for a number of purposes including fish feed ingredients, biofuels feedstock and pharmaceuticals.

4.2.3 Anaerobically digested food waste

Food waste-based anaerobic digestate represents a secondary P resource with a relatively high P plant-availability and, in addition, a favorable ratio of nutrients (P:nitrogen (N): potassium (K)) for plant uptake (Haraldsen et al., 2011). While we found that there are comparatively insignificant levels of P in food waste-based anaerobic digestate at a national level, it could represent an immediate replacement for mineral fertilizer, if accordingly promoted by regulations on the use secondary P resources as fertilizer. Currently, regulations do not consider nutrient concentrations in secondary P resources as compared to the concentration of heavy metals and dry matter (Lovdata, 2003). As a result, current regulations restrict the use of nutrient-rich resources with low contents of organic carbon, such as food waste-based anaerobic digestate. This barrier will need to be overcome to exploit the potential of this high quality secondary P fertilizer.

4.2.4 Sewage sludge

In this study, we found that 55% of Norwegian sewage sludge is currently returned to agricultural land, primarily based on its positive effects as soil conditioner and as liming material (Refsgaard et al., 2004). With this current practice, P tends to be heavily over-applied as compared to the plants needs (Krogstad et al., 2005), as the P contents of sewage sludge, both residual and plant-available, are usually not accounted for in fertilization plans. However, even if the plant-available P fraction was accounted for in fertilization plans, the application of sewage sludge as a P fertilizer would still result in a large accumulation of residual P with associated environmental risks, unless new technologies for sewage sludge treatment are applied.

4.2.5 Meat bone meal

Meat bone meal is another example of a secondary P resource that is applied to agricultural soil with low P utilization. In Europe, the use of meat bone meal as a fertilizer was banned in 2002 due to the risk of bovine spongiform encephalopathy (BSE) or “mad cow disease”. However, the European Union (EU) lifted these restrictions in 2006 and meat bone meal can now be used as a fertilizer in all EU countries, unless it is suspected to be infected or based on specified risk material (Ylivainio et al., 2008). Nonetheless, according to van Dijk et al., the meat bone meal P potential is hardly utilized, as most meat bone meal is incinerated - amounting to 20% of the total P losses from all sectors in the EU-27 combined [244 kt P/yr] (van Dijk et al., 2016). In Norway, however, the meat bone meal that is applied to agriculture soils is done so inefficiently. The N:P ratio in meat bone meal (1.5–2 (Brod et al., 2014)) is low compared to the ratio required by agricultural crops (7.5–8 (Bioforsk, 2012)), resulting in the considerable over-application of P since meat bone meal is usually applied as N fertilizer. Thus, meat bone meal would only be efficiently utilized as an alternative P fertilizer if it was applied based on the crops' P needs and if N was applied in addition for example as mineral N fertilizer. However, the application of meat bone meal based on plant-available P would still result in the additional application of the accompanied residual P, unless appropriate technologies are implemented to increase plant-availability of P as e.g. chemical extraction of P from bone ash for the production of soluble P fertilizers (Krupa-Żuczek et al., 2008).

5. Conclusions

The inclusion of plant-availability within SFA is an important methodological advancement that allows for more accurate estimations of secondary P recycling potentials. Such knowledge is essential for policy-making, as a shift in focus from total P to plant-available P would result in a different solution space for developing and implementing efficient P recycling solutions. This was exemplified by our Norwegian case that showed that i) plant-available P in manure was enough to cover Norway's entire P fertilization demand and, therefore, developing systems to transport manure could lead to full independence from imported mineral P and ii) for P recycling to be effective, it is essential to, first, develop methods for collecting secondary P and, second, evaluate secondary P treatment technologies based on their ability to produce high quality products. Current Norwegian policies based on total P have led to suboptimal results, as shown by the yearly large accumulation of P in soils. This is due to, amongst others, a lack of understanding regarding the fertilization effects of secondary P resources; such knowledge gaps have led to the disregard of secondary P's ability to replace primary P and, thus, secondary P management being based on pollution risks rather than fertilization effects.

Author contributions

The manuscript was written through contributions of all authors. All authors have given approval to the final version of the manuscript.

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Abbreviations

MFE	mineral fertilizer equivalents
P	phosphorus
SFA	substance flow analysis
RAE	relative agronomic efficiency

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Paper IV

Redistributing phosphorus in animal manure from a livestock-intensive region to an arable region: Exploration of environmental consequences

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Article

Redistributing Phosphorus in Animal Manure from a Livestock-Intensive Region to an Arable Region: Exploration of Environmental Consequences

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Abstract: Specialized agricultural production between regions has led to large regional differences in soil phosphorus (P) over time. Redistribution of surplus manure P from high livestock density regions to regions with arable farming can improve agricultural P use efficiency. In this paper, the central research question was whether more efficient P use through manure P redistribution comes at a price of increased environmental impacts when compared to a reference system. Secondly, we wanted to explore the influence on impacts of regions with different characteristics. For this purpose, a life cycle assessment was performed and two regions in Norway were used as a case study. Several technology options for redistribution were examined in a set of scenarios, including solid–liquid separation, with and without anaerobic digestion of manure before separation. The most promising scenario in terms of environmental impacts was anaerobic digestion with subsequent decanter centrifuge separation of the digestate. This scenario showed that redistribution can be done with net environmental impacts being similar to or lower than the reference situation, including transport. The findings emphasize the need to use explicit regional characteristics of the donor and recipient regions to study the impacts of geographical redistribution of surplus P in organic fertilizer residues.

Keywords: life cycle assessment (LCA); manure management; phosphorus; nutrient recycling; nutrient redistribution

1. Introduction

Animal manure is a key component in cycling phosphorus (P) between animals and crops. Manure is also one of the main inputs of P to agricultural soils [1]. However, the P cycle between animals and crops has largely been broken by regional specialization in livestock production or arable farming [2,3]. Areas with high livestock density generally have high levels of soil P, as application of P-rich animal manure often exceeds crop P requirements, and the resulting soil P accumulation increases the risk of P losses to waterways through erosion and run-off [4]. Partial segregation of livestock and arable production is prevalent in, amongst others, Western and Northwestern

European countries such as France [5], the UK [6], and Norway [7]. Soil P accumulation due to high input of manure P is a current challenge in many Western European countries [8], and substantial soil P accumulation in agricultural production systems in general is found both in- and outside of Europe [9]. Specialist arable farming regions have to import mineral P fertilizer to compensate for P exports with crop products and lack of animal manure to maintain soil fertility. Mineral P fertilizer comes from mined non-renewable phosphate rock, of which around 80% is used as mineral fertilizer globally [10]. In order to reduce consumption of phosphate rock, reduce soil P accumulation and associated risk of P loss, and achieve healthier global P stewardship, more efficient use of P in agriculture is needed [9,11].

Geographical redistribution of surplus manure P from livestock-intensive regions to arable regions is considered crucial for improving P use efficiency in agriculture [12]. Hanserud et al. [7] showed that manure P alone could potentially replace all mineral P fertilizer in Norway if redistributed well within and between counties. However, manure management affects both the environment and human health in various negative ways, and the geographical context within which it occurs has a great influence on the environmental effects. The most important impacts are global warming (mostly through emissions of methane (CH^4) and nitrous oxide (N_2O)), acidification of soils and particulate matter formation (emissions of ammonia (NH_3)), and marine and freshwater eutrophication (losses of nitrate (NO_3^-) and phosphate (PO_4^{3-}) to water) [13,14]. Particulate matter formation in the air can cause human respiratory health problems. Manure management also contributes to depletion of fossil resources through its use of fossil fuel, but may delay potential depletion of phosphate rock by substituting for mineral P fertilizer.

The life cycle assessment (LCA) methodology has been used in a few recent studies to evaluate the environmental impacts from manure management that includes nutrient redistribution [15–17]. These studies have to varying degrees included characteristics of the donor and recipient region that influence the impacts of redistribution. However, resource use, emissions, and yields may vary greatly between agriculture regions, also within the same country. Agri-food systems should therefore be modelled with a high level of geographical explicitness to enable a fair comparison between systems [18].

In the present study, our main objective was to estimate the potential life cycle environmental impacts for systems that redistribute manure P between two regions with different characteristics—a donor region with a manure P generation surplus, and a recipient region with a P deficit and P fertilizer import requirement. A second objective was to study the influence that regional differences may have on environmental impacts in such redistribution systems. For this generic purpose, we chose to examine a case study with two regions in Norway where a high degree of agricultural specialization is present: one region with a relatively high livestock density and one region dominated by cereal crop production. The central research question this paper attempts to answer is whether more efficient P use in agriculture through manure P redistribution comes at a price of increased environmental impacts for the manure management system as a whole compared to a reference system.

2. Materials and Methods

2.1. LCA Approach and Functional Unit

Life cycle assessment is defined and described in ISO 14040 and 14044 [19,20] as a method for evaluating the potential environmental impacts associated with the life cycle of a product or service. It is further outlined in documents such as the ILCD Handbook [21].

The LCA was performed with the use of the software SimaPro 8.1.1. The function of the system studied here was set as management of manure from dairy cows on a donor farm with surplus manure P for redistribution. As we aimed to compare the best uses of a given biomass, an input-related functional unit (FU) was used [22]. Thus, the FU chosen was management of one ton of fresh dairy cow manure, serving as the starting point for redistribution of manure P, organized in a set of scenarios.

2.2. Geographical Scope and Technology Choice

Within the geographical setting of Norway, in a previous study we characterized all 19 counties in Norway in terms of their agricultural soil P balance [7]. That study identified the county of Rogaland in south-west Norway as having a particularly high surplus of manure P and it was therefore chosen as the donor region of P for redistribution in this study (Figure 1). The county of Akershus is one of three counties, all in the south-east, that require P fertilizer imports. Akershus was chosen as the recipient region in this study (Figure 1). Hanserud et al. [7] showed that even if manure P were distributed well within Rogaland to cover internal P fertilizer requirements, there would still be a substantial surplus of P to export. The FU in the present study represented this surplus. Data on typical crops, soils, and agricultural practices in the donor and recipient counties (Table 1) were used to estimate region-based nutrient requirements and emissions from fertilizer application.

As there are currently no incentives for treatment of manure and trade in manure nutrients between farms and regions in Norway, various redistribution scenarios had to be constructed hypothetically for the analysis. These scenarios were based on technologies that are already in use, or planned/likely to be used in the future. The cost of transporting untreated, bulky manure slurry is prohibitively high [23] and manure P therefore requires processing to become more transportable for redistribution between geographical areas. Mechanical solid–liquid separation is currently the most commonly applied processing method to enable manure redistribution [17]. Such separation concentrates a proportion of dry matter (DM), P, and other nutrients in a more transportable solid fraction, while most of the volume and the rest of the DM and nutrients are left in a liquid fraction to be spread locally [24]. Solid–liquid separation of slurry by screw press is a likely solution for the small farming units characteristic of Norway. Use of a decanter centrifuge was also included, to compare the impacts of two different separation technologies. Separation by screw press is the cheaper alternative, but diverts less DM and nutrients, P in particular, into the solid fraction than the more costly decanter centrifuge [24].

The effect of including anaerobic digestion (AD) of manure as a pre-separation step was also studied, because of the likely future increase in use of AD on Norwegian livestock farms. In 2009, the Norwegian government signalled an ambition to process 30% of all housed animal manure in Norway by AD to produce biogas (i.e., green energy), as a measure to reduce the greenhouse gas emissions from the agricultural sector [25]. Anaerobic digestion is not in itself a technology to redistribute nutrients and needs to be combined with other technologies, such as solid–liquid separation.

Figure 1. Location of the donor county Rogaland and the recipient county Akershus in southern Norway.

Table 1. Assumptions on crop yields, fertilizer requirements and application practices on donor and recipient farms.

Parameter	Donor Farm, Rogaland	Recipient Farm, Akershus
Soil P level	Very high	Moderately high
Main crop [a]	Grass	Cereals
Yield	10,000 kg DM grass/ha; 3 cuts	4000 kg DM spring wheat/ha
Fertilizer requirement [b,c]	270 kg N, 0 kg P (30 kg P), 168 kg K per ha	105 kg N, 10 kg P (14 kg P), 50 kg K per ha
Time of manure fertilizer application	85% (80–90%) within growing season, 15% (10–20%) in autumn [d]	100% in spring
Type of application	Liquid application with broadcast spreader, surface spreading in moderate weather conditions (sun and wind); mineral fertilizer: broadcast spreading	Solid fractions: solid manure spreader, incorporation within 3 h. Slurry: broadcast spreading, incorporation within 3 h; mineral fertilizer: broadcast spreading

DM = dry matter; N = nitrogen; K = potassium; ha = hectare; [a] [26]; [b] Based on [27] (without adjustment for soil P level in brackets); [c] For calculation of P fertilizer requirement, see Section 2 of the Supplementary Materials; [d] [28].

2.3. Scenarios

Five scenarios were developed to provide a basis for comparing alternative P redistribution strategies to a reference situation of no P redistribution. These were:

Ref:	Reference scenario. Manure stored in house in a manure cellar and applied locally to grassland on the donor animal farm.
SP:	In-house pre-stored slurry separated by screw press (SP). The resulting solid fraction is stored, hygienized, and transported to a recipient farm in Akershus county and applied to arable land. Liquid fraction stored and applied locally.
DC:	As the SP scenario, but separation by decanter centrifuge (DC).
AD_SP:	In-house pre-stored slurry digested through anaerobic digestion (AD), then separated by screw press (SP). The digested solid fraction is stored, hygienized, and transported to Akershus county and applied to arable land. Digested liquid fraction stored and applied locally.
AD_DC:	As the AD_SP press scenario, but separation by decanter centrifuge (DC).
NoSep:	No separation of slurry. Slurry stored as in the reference scenario, then hygienized and transported in its entirety to Akershus county and applied on arable land.

The NoSep scenario is the extreme version of redistributing manure P. Transport of unseparated slurry with its high water content is unlikely to ever take place over long distances because of high expected transport costs, but was included here as a scenario to compare the effect of no separation.

2.4. System Boundary

The system boundary and the main processes involved are shown graphically in Figure 2. The system starts with the generation of cattle manure, which is stored in house in a manure cellar. During the in-house storage (called pre-storage in the scenarios involving AD and/or solid–liquid separation) wash water is added, increasing the mass of the FU and turning the manure into a more liquid slurry. The subsequent processing involved in each scenario is presented in Section 2.3 above, and a graphical break down of the processing is shown in Figure 3. Further details on each process are provided in Sections 2.5.2–2.5.6. The alternative scenarios entail use of different technologies and capital goods. Production of capital goods was included for equipment for manure/fertilizer field application, but not for the AD reactor, the outside storage facilities or the manure separation machinery. Brogaard et al. [29] found that the construction of an AD plant for the annual treatment of 80,000 tons mixed waste (75% manure) contributed very little towards the overall life cycle environmental impact of the plant. A similar conclusion was reached by Mezzullo et al. [30] for a small-scale farm-based AD plant fed cattle waste. We therefore decided to leave AD plant construction outside the system boundary. We made a similar assumption of negligible life cycle impacts for the construction of outside storage facilities and separation machinery. Hygienization is required before application of slurry or slurry products on land other than that owned or rented by the donor farm [31]. A hygienization step was therefore included in the process chain after storage of the products to be transported to the recipient farm. Application of manure products was assumed to replace use of mineral fertilizer components in all scenarios, according to plant nutrient requirement for typical crop yields in the two regions. Manure nitrogen (N), P, and potassium (K) replaced production and field application of mineral N, P, and K fertilizer components, respectively. Production of the final compound fertilizer was not included in the analysis. In the scenarios including anaerobic digestion, the produced biogas was assumed upgraded to green gas (also called biomethane) to replace fossil fuel (diesel) for public transport purposes. Upgrading biogas to green gas is likely to take place in the donor region where an existing network for distribution of natural gas is considered for transport of farm-produced biogas to a central upgrading facility [32].

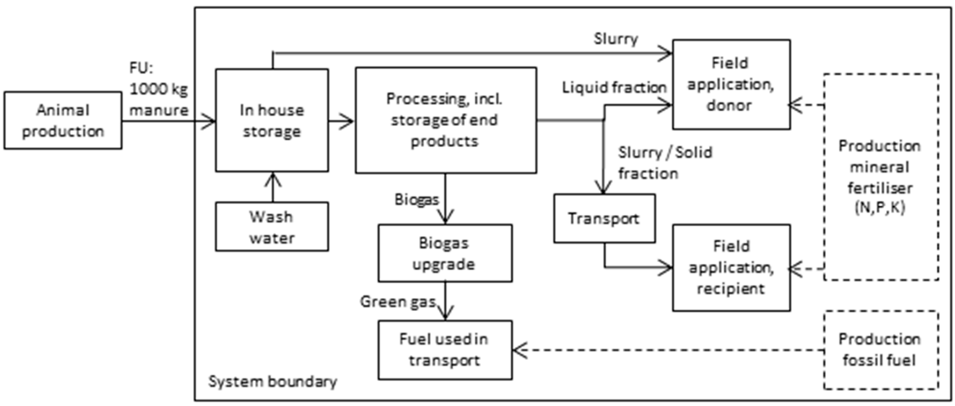


Figure 2. System boundary, main processes (boxes), and flows (arrows) included in the LCA. Dotted boxes and arrows indicate avoided processes and flows, respectively.

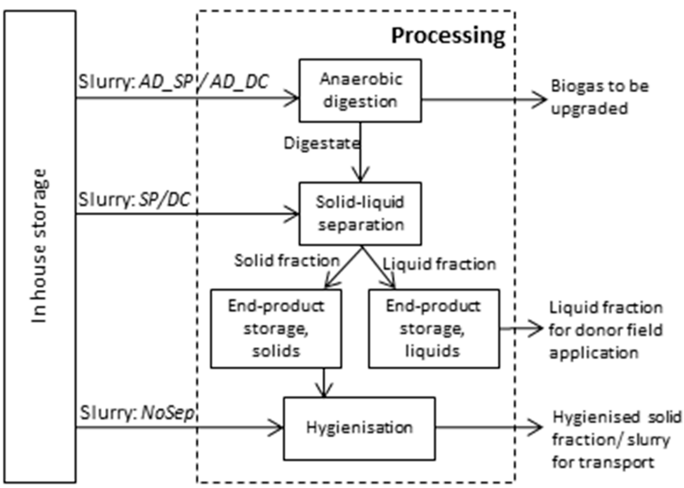


Figure 3. An overview of the processes included in the main process of "Processing" in Figure 2, indicated by the dotted line. The figure shows the processes (boxes) used under the different scenarios (in italics) to process slurry from in house storage and how the processes are connected by flows (arrows).

2.5. Life Cycle Data Inventory and Assumptions

Data on process emissions and resource use were as much as possible collected in the literature to reflect Norwegian conditions and presented in the subsections below. Data from ecoinvent database 3.1 (“allocation recycled content”) were used for processes such as transport, energy use, and spreading of fertilizer [33]. Details on assumptions and calculations can be found in Sections 2–5 of the Supplementary Materials.

2.5.1. Manure Characteristics

The chemical composition of the different intermediate and end-products through the life cycle stages is shown in Table 2. The characteristics of the fresh manure were based on a dairy cow with annual milk production of 7000 kg, excreting 1.64 tons manure with a DM content of 10.4% per month [34]. Of the DM, 88% was assumed to be organic material [35]. The content of total-N, ammonium N (NH_4-N), and P was set at 6.2, 3.6, and 0.72 kg/ton fresh manure, respectively [34], while the K content was set at 3.4 kg K/ton stored slurry (equal to 5.9 kg K/ton fresh manure), in accordance with Daugstad et al. [36]. An amount of 1.2 tons of wash water per cow per month was assumed added to the manure storage in house [37], turning manure into slurry and increasing the mass after excretion by 73%.

Table 2. Chemical composition of manure and manure products. All numbers in kg.

Scenario	Stage/Manure Product	Mass	DM	OM	Tot-N	NH_4-N	P	K
All	After animal	1000	104.0	91.5	6.2	3.6	0.7	5.9
Ref, NoSep	After in house storage—3 months	1723	94.8	82.4	5.9	3.3	0.7	5.9
SP, DC, AD_SP/DC	After in house storage—1 month	1727	99.4	86.9	5.9	3.3	0.7	5.9
	After separation							
SP	Liquid	1537	62.6	54.8	5.1	3.0	0.6	5.2
	Solid	190	36.8	32.2	0.9	0.4	0.1	0.6
DC	Liquid	1485	38.8	33.9	4.3	2.8	0.2	4.9
	Solid	242	60.6	53.0	1.7	0.5	0.5	0.9
	After AD of stored slurry							
AD_SP/DC	Digestate	1698	69.9	57.4	5.9	4.2	0.7	5.9
	After separation following AD							
AD_SP	Liquid	1511	44.0	36.2	5.1	3.8	0.6	5.2
	Solid	187	25.9	21.3	0.9	0.5	0.1	0.6
AD_DC	Liquid	1460	27.3	22.4	4.3	3.6	0.2	4.9
	Solid	238	42.7	35.0	1.7	0.7	0.5	0.9
	After end-product storage							
SP	Liquid	1532	57.2	49.3	5.0	2.9	0.6	5.2
	Solid	188	34.5	29.9	0.9	0.3	0.1	0.6
DC	Liquid	1482	35.4	30.5	4.2	2.8	0.2	4.9
	Solid	238	56.9	49.3	1.6	0.5	0.5	0.9
AD_SP	Liquid_dig	1507	40.4	32.6	5.0	3.7	0.6	5.2
	Solid_dig	185	24.4	19.8	0.9	0.4	0.1	0.6
AD_DC	Liquid_dig	1458	25.0	20.2	4.2	3.5	0.2	4.9
	Solid_dig	235	40.2	32.6	1.6	0.6	0.5	0.9

OM = organic matter; Tot-N = total nitrogen; NH_4-N = ammonium nitrogen; SP = screw press; DC = decanter centrifuge; Liquid = liquid fraction from separation; Solid = solid fraction from separation; AD = anaerobic digestion; AD_SP/DC = AD_SP and AD_DC; dig = separated fraction of digestate after AD.

2.5.2. In-House Storage

Around 76% of dairy cattle manure in Norway is managed in liquid systems [38] (p. 161) and stored in a manure cellar under the animal house, and such a system was assumed in this study. For the reference and NoSep scenarios, we assumed an average of three months of storage before further handling, while the other scenarios had one month of pre-storage in the manure cellar before further processing. In the absence of better data, we assumed the same NH_3 volatilization rate for the two storage periods. The emissions factors for CH_4-C were based on the Tier 2 approach described in IPCC [39], which states the CH_4-C emissions as a percentage of the OM entering storage (Equation 1). We assumed a maximum methane producing capacity (B_0) for dairy cattle in Norway of 0.23 m3 CH_4/kg OM, as suggested by Morken et al. [35]. As methane conversion factor (MCF), we used the factors given in IPCC [39] for pit storage below animal houses in cool climates (≤10 °C) for >1 month for the 3 month storage (MCF of 17%) and <1 month for the 1 month pre-storage (MCF of 3%). The degradation of OM for the three month storage was set to 10% of OM [40], while the one month pre-storage was assumed to be half of this, i.e., 5%.

$$\text{Emission factor } \mathrm{CH_4\_C}\ (\mathrm{kg/kg\ OM}) = \mathrm{B_0} \times 0.67 \times (\mathrm{MCF}/100\,\%)/1.34, \quad (1)$$

where OM is organic material in manure entering storage (kg), also termed volatile solids (VS), B_0 is maximum methane producing capacity for cattle manure (m^3 CH_4/kg OM), 0.67 is a conversion factor from m^3 to kg CH_4 (kg CH_4/m^3 CH_4), *MCF* is methane conversion factor given type of storage (%) and 1.34 is a conversion factor from CH_4 to CH_4-C (kg CH_4/kg CH_4-C). Table 3 summarizes the emission factors used for the inventory analysis of the manure management system.

Table 3. Emission factors used for the life cycle phases in the LCA

Emission Factor	Unit	In House Storage	End-Product Storage				Field Application			
			LF	SF	LF_{dig}	SF_{dig}	Slurry, LF, LF_{dig}; Grass Land	SF, SF_{dig}; Arable Land	Slurry; Arable Land	Mineral Fertilizer
NH_3-N	% of NH_4-N	7 [a]	1.7 [d]	5 [d]	1.7 [d]	5 [d]	29 [h]	4 [h]	10 [h]	1% N [i]
N_2O-N	% of tot-N	0.1 [b]	0.5 [e]	2 [e]	0.5 [e]	2 [e]	1.25 [b]/0.63 [g]	1.25 [b]/0.63 [g]	1.25 [b]	1.25 [b]
NO_3-N	% of tot-N	-	-	-	-	-	12.8 [j]	23.3 [j]	23.3 [j]	12.8/23.3 [j]
CH_4-C_{long}	% of OM	2 [c]	0.4 [f]	0.12 [f]	0.06 [g]	0.02 [g]	-	-	-	-
CH_4-C_{short}	% of OM	0.35 [c]	-	-	-	-	-	-	-	-
MFE N_{min}	% of NH_4-N	-	-	-	-	-	34.5/54 [k,l]	65 [k]	73 [k]	100
MFE N_{org}	% of N_{org}	-	-	-	-	-	10.2 [k]	10 [k]	10 [k]	-

'-' = not included, LF = liquid fraction from separation, SF = solid fraction from separation, LF_{dig} = liquid fraction from separated digestate, SF_{dig} = solid fraction from separated digestate, OM = organic material, CH4-C_{long} = methane emissions from long-term storage (3 months), CH_4-C_{short} = methane emissions from short-term storage (one month), MFE N_{min} = mineral fertilizer equivalent of applied mineral nitrogen, MFE N_{org} = mineral fertilizer equivalent of applied organic nitrogen. [a] [41]; [b] [42] (Tables 4.12 and 4.17); [c] Based on [35,39]; [d] [43], unit is in % of tot-N for SF and SF_{dig}; [e] [39]; [f] Based on [35]; [g] Based on [44]; [h] [45]; [i] [38]; [j] Based on [46,47]; [k] [27]; [l] 34% for slurry and 54% for LF and LF_{dig}.

2.5.3. Processing

Separation efficiency for the screw press and decanter centrifuge is shown in Table 4. We assumed the same separation efficiency for DM and OM. In the absence of consistent data on the separation of K, we assumed that it was similar to that of NH_4-N [24]. The NH_4-N separation efficiency for the screw press was set equal to mass separation. Separation efficiency for digestate and slurry was not found to be significantly different in a statistical two-sided T-test of the data provided by Hjorth et al. [24] and was therefore assumed to be equal. Furthermore, we assumed that the emissions to water and air during separation and hygienization were negligible. Electricity used in the different processes was assumed to be the NordEl electricity mix, because of the common Nordic electricity market. For anaerobic digestion, we used the BioValueChain model described in Lyng et al. [48] to estimate biogas yield and subsequent conversion to green gas (bio-methane) in an upgrading step. Monodigestion of cattle manure was assumed to take place in a mesophilic digester at 37–40 °C. The model assumed 75 kWh electricity use/ton DM into the reactor and 250 kWh/ton DM heat use [48]. The energy carrier for heat was assumed to be wood chips. The model uses a potential biogas yield of 260 Nm^3/ton DM with a CH_4 content of 65%, with a realistic output of 70% of the potential yield. Mineralization of organic nitrogen (N_{org}) to mineral nitrogen (N_{min}), given as NH_4-N, during digestion was set equal to degradation of OM, which was calculated to be 34% [49]. We assumed that all biogas was sent to upgrade and that installation of 10 km polyethylene pipe was necessary to connect to existing natural gas pipe infrastructure and upgrade facilities [50]. For the upgrading process, PSA technology was assumed, with a methane loss of 1.5% of the biogas methane to be upgraded. The energy requirement for hygienization of the manure was set at 24 kWh electricity/ton substrate for thermal treatment at 70 °C degrees for 1 h [51].

Table 4. Separation efficiency (% of substrate left in solid fraction) for screw press and decanter centrifuge and separation electricity demand.

Separation Technology	Mass	DM	OM	Tot-N	NH_4-N	P	K	Electricity Demand (kWh/ton) [b]
Screw press	11 [a]	37 [a]	37	15 [a]	11	17 [a]	11	1.1
Decanter centrifuge	14 [a]	61 [a]	61	28 [a]	16 [a]	71 [a]	16	4.3

[a] Hjorth et al. [24]; [b] Møller et al. [52].

2.5.4. End-Product Storage

The total storage time (in house storage plus end-product storage) for all scenarios was set to be similar, so that the timing of field application was not affected by the chosen scenario. The liquid fraction was assumed to be stored in a closed outdoor storage tank and the solid fraction in an open solid manure storage. For emissions of CH_4, an MCF of 3.5% and 1% was used for liquid and solid storage, respectively [35]. Sommer et al. [44] reported a 90% reduction in CH_4 emissions from storage of digested slurry compared with non-digested and we assumed the same reduction for storage of digested solid and liquid fractions (see Section 3 of the Supplementary Materials for calculation). Emissions of N_2O-N from liquid fractions were set to 0.5% of N based on IPCC [39] as a conservative estimate.

2.5.5. Transport

Transport of manure nutrients from the donor farm in Rogaland to the recipient farm in Akershus was assumed to take place by road (Lorry 16–32 metric ton, EURO4 RER) over an average distance of 500 km. Emissions from transport related to spreading of manure products were included in the ecoinvent background data for field application.

2.5.6. Field Application

Livestock farms in Norway are required to have sufficient spreading area so as not to exceed 35 kg manure P/ha/year [31], and this determined the necessary spreading area for the reference scenario. For the alternative scenarios, we assumed that the spreading area in hectares at the donor farm was the same as in the reference. The rate of manure product application on the recipient farm was assumed to be according to the level of available P in soil and crop P requirements to ensure good use of the transported manure P.

Emissions from field application and the calculation of mineral fertilizer substitution were both affected by the assumptions made on the type and timing of application (Table 1). Direct emission of N_2O for undigested fractions was assumed in line with IPCC [42], while digested liquid and solid fractions were assumed to have 50% lower emissions after spreading on the field according to Sommer et al. [44]. Indirect N_2O-N emissions were set to 1% of NH_3-N emissions and 0.75% of NO_3-N emissions to water [53]. Losses of ammonia during spreading were based on Morken and Nesheim

[45], and the emission factor used for Rogaland was an average of emissions in spring, summer, and autumn weighted by the amount spread in each season. Emission factors for NO_3 to water recipients were calculated from FracLEACH factors in representative small catchment areas in the donor and recipient region [47] (see Section 4 in the Supplementary Materials for details). Losses of P to water through erosion and runoff occur on both the donor and recipient farms, but were not estimated in this study. According to Bechmann [26] there is no clear relationship between soil P balance and P losses. Losses of P from agricultural areas are instead influenced by a range of factors, such as soil P status, tillage practices, and transport processes that connect a field with surface waters [4]. The soil P balance correlates better with available soil P status over time than for a shorter period [26], and the effect of changes during one single year (as in this case study) is therefore difficult to assess without assuming a trend over time.

2.5.7. Manure Fertilizer Value and Mineral Fertilizer Substitution

Substitution of mineral fertilizer components was calculated based on the limiting factor for plant growth, being either nutrients applied or fertilizer required. The amounts of N, P, and K required in the donor and recipient region are shown in Table 1. For P, we adjusted the requirement based on plant-available soil P values in the donor and recipient regions [54,55]. We then used a mineral fertilizer equivalence (MFE; used to compare fertilizer values of secondary products with mineral fertilizer) of 100% of the total P content in the different manure products, based on Brod et al. [56]. The MFE of K was assumed to be equal to that of P. The MFE for N was calculated according to the Norwegian fertilization handbook [27], which subtracts expected N losses from the MFE-N value depending on factors such as field application method, time from application to soil incorporation, weather conditions during application, and in which season the application is done. More information on the calculation of MFE-N can be found in Section 5 of the Supplementary Materials. For the avoided production of mineral N, P, and K fertilizer, we used the ecoinvent database for the production of ammonium nitrate (NH_4NO_3), triple superphosphate ($Ca(H_2PO_4)^2$), and potassium chloride (KCl), respectively.

2.6. Impact Assessment

The environmental impact categories considered were climate change (CC, expressed in kg CO_2-equivalents (eq.)), marine eutrophication (ME, expressed in kg N-eq.), terrestrial acidification (TA, expressed in kg SO_2-eq.), particulate matter formation (PMF, expressed in kg PM10-eq.), and fossil resource depletion (FD, expressed in kg oil-eq.). For CC, the IPCC 2013 characterization factors for a 100-year perspective were applied, as implemented in SimaPro 8.1.1. These characterization factors have been changed from earlier IPCC values and methane now has a characterization factor of 30.5 kg CO_2-eq., biogenic methane 27.75 kg CO_2-eq. and N_2O 265 CO_2-eq. For the categories ME, TA, PMF, and FD, the ReCiPe midpoint hierarchist perspective impact assessment method was used [57]. In addition, we calculated the potential amount of avoided mineral P fertilizer per scenario and the P over application per scenario, both given in kg P. The amount of P that did not substitute mineral P was applied in excess (over application). Therefore, the sum of the absolute values of the two indicators would be constant across scenarios and equal the total amount of P in the FU. Over application of manure P is possible because the allowed application rate does not take into account the actual P fertilizer requirement of the receiving soil.

2.7. The Effect of Regional Differences

To study the isolated net contribution to impacts from regional differences between the donor and the recipient region, we chose to look at the reference and the NoSep scenarios. The two scenarios spread the same amount of unseparated and undigested slurry on the field, assuming hygienization does not alter the fertilizer value of the transported slurry. To fully see the net influence of the regional differences, we excluded the contribution from hygienization and transport in the NoSep scenario.

2.8. Sensitivity Analysis

Sensitivity analysis was carried out according to the tiered approach suggested by Clavreul et al. [58], where we included the two first steps. The proposed first step—contribution analysis—was included in the interpretation of the results in Section 3.1. The second step—sensitivity analysis—was subdivided into perturbation analysis and scenario analysis. For the perturbation analysis we selected 23 parameters, which were all increased by 10% (complete overview in Supplementary Materials). The analyzed parameters were limited to those expected to influence the results the most, such as the parameters for manure characterization (e.g., the concentration of nitrogen in fresh manure) and field emissions of NH_3 and NO_3. The result was given as a sensitivity ratio (SR), described by [58] as the ratio between the relative change in result and parameter (Equation (2)). An SR of 0.1 would mean that a 50% increase in the parameter yields a 5% increase in the result. Only parameters with SR greater than 0.1 as an absolute value are presented, and we selected the reference and AD_SP scenarios for the perturbation analysis. In the scenario analysis, we explored the effect on all scenarios of (i) applying manure products at the recipient farm according to N content instead of P content, as N-based fertilizer application is more common practice, and (ii) optimal soil P levels at both the donor and recipient farm, which implies balanced fertilization (P fertilizer application equals removal of P in crop yields). In addition, in a third scenario analysis we explored the effect on the net life cycle climate change impact for the Ref, DC, AD_DC, and NoSep scenarios of 0–1500 km transport distance and transport by lorry, train (freight train (CH), electricity, Alloc Rec, U) and ship (freight, sea, transoceanic ship (GLO), processing, Alloc Rec, U). This was done by subtracting the contribution from lorry transport from the total life cycle climate change impact for the four scenarios at zero km and then adding the climate change impact of the different transport modes for a distance of 0–1500 km.

$$\text{Sensitivity ratio (SR)} = \frac{R_{\Delta}}{R_{init}} / \frac{P_{\Delta}}{P_{init}}, \tag{2}$$

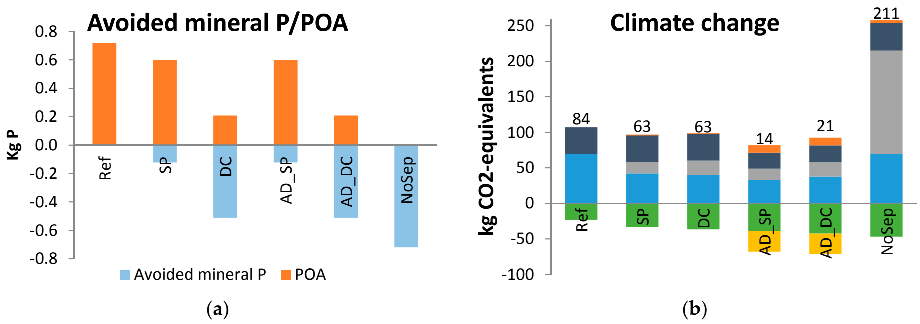
where R_{init} is the initial result value, $R\Delta$ is the change in result value, P_{init} is the initial parameter value, and $P\Delta$ is the change in parameter value.

3. Results

The following sub-sections present the results from the impact assessment and the uncertainty analysis. Background data as well as additional information of under- or over-application of plant-available nutrients are provided in the Supplementary Materials.

3.1. Impact Assessment Results

The contribution of the different life cycle processes to the environmental impact categories for the different scenarios are shown in Figure 4.



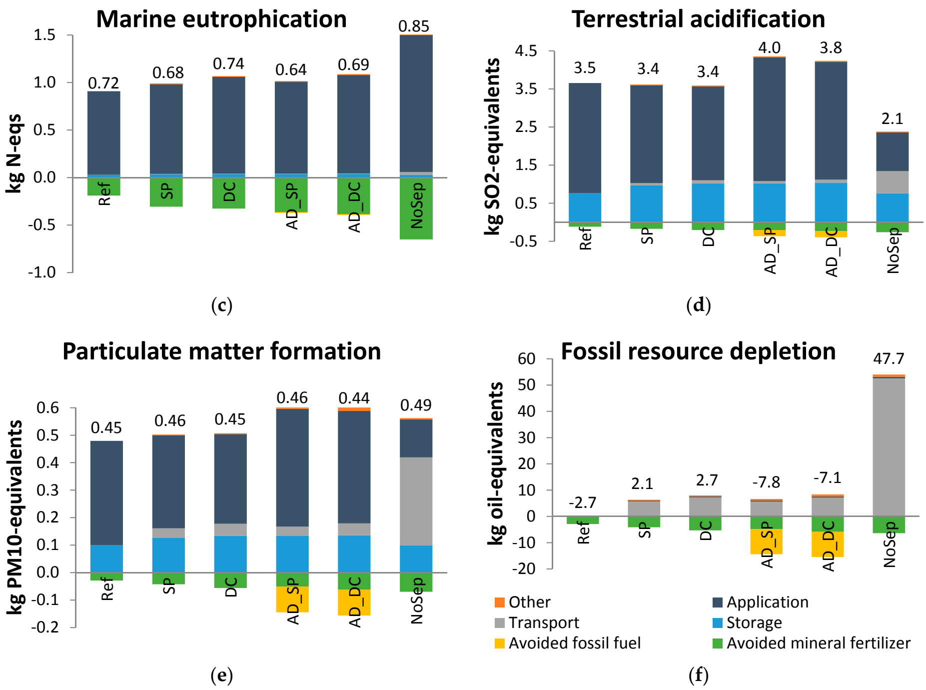


Figure 4. Contribution of the different processes in each scenario to the potential impacts on: (**a**) avoided mineral P/P over application (POA); (**b**) climate change; (**c**) marine eutrophication; (**d**) terrestrial acidification; (**e**) particulate matter formation; and (**f**) fossil resource depletion. In Figure 4b–f: "Other" contains the processes of separation, anaerobic digestion, biogas upgrading, and hygienization; "Application" contains donor and recipient field application; "Storage" contains in house storage and end-product storage; and the net impact is shown in numbers above the bars.

Of the scenarios evaluated, the NoSep scenario redistributed the highest amount of manure P to the recipient farm and therefore gave the highest amount of avoided mineral P fertilizer. However, the NoSep scenario also had by far the highest potential net impacts on climate change and fossil resource depletion, 150% and 1700% higher, respectively, than the second worst scenario in these categories. For both impact categories, the higher impact can be attributed to transport, showing the environmental cost of transporting a great amount of water in unseparated slurry.

The DC and AD_DC scenarios employing a decanter centrifuge separated 71% of the manure P into a transportable solid fraction, compared with 17% of the manure P with screw press separation in the SP and AD_SP scenarios. This was based on the separation efficiency values given in Table 4. The two centrifuge scenarios therefore replaced a higher amount of mineral P fertilizer in the recipient region. The P in the locally applied manure products did not replace any mineral P, since the mineral P fertilizer requirement for the donor grassland was zero due to high levels of available soil P.

The scenarios that included anaerobic digestion (AD_SP and AD_DC) performed better than the non-AD separation scenarios (SP and DC) for climate change and fossil resource depletion, mostly because of the ability to replace fossil fuel with the upgraded biogas. The net climate change impact was on average 73% lower for the AD scenarios, while for fossil resource depletion the average net impact for AD_SP/AD_DC was 409% lower than for SP/DC. However, the AD scenarios had, on average, a 15% higher impact for terrestrial acidification compared with the non-AD scenarios due to a higher contribution from field application of liquids. This can be explained by mineralization of organic N during the AD process giving more NH_4-N in the digestate and its separated fractions to volatilise as NH_3 from storage and field application. This effect has earlier been pointed out by Amon et al. [59].

All scenarios had quite similar net impacts on marine eutrophication and particulate matter formation. For marine eutrophication, the redistribution of manure N from a farming area with low NO_3 losses to an area with higher NO_3 losses led to increased direct emissions for the redistribution scenarios. Separation and redistribution had, at the same time, two positive effects on the manure N fertilizer value: (i) the liquid fraction applied locally had lower viscosity than before separation, thus infiltrating faster into the ground after surface spreading, losing less N to the air and having more N available to plants; and (ii) the N in the redistributed products was incorporated into the arable soil shortly after application, which also reduced the losses of NH_3 to the air and therefore increased the amount of N available to plants.

The reference scenario performed similarly to or slightly worse than the non-AD separation scenarios SP and DC for all impact categories except fossil resource depletion. For fossil resource depletion, the reference net impact was 4.8–5.4 kg oil-eq. lower because it did not include external transportation. The negative net impact for the reference was because the benefit of the avoided production and application of mineral fertilizer more than outweighed the fossil fuel used for spreading the slurry. For climate change, the impact of the SP and DC scenarios was 25% lower than for the reference. This was due to the lower CH_4 emissions for the short in-house storage period in SP/DC (see Table 3) combined with benefits from greater amounts of replaced mineral fertilizer. Comparing the reference to the AD scenarios AD_SP and AD_DC, the reference had similar or greater impacts for all impact categories except for terrestrial acidification, where the reference impact was 8–11% lower.

The processes of separation, anaerobic digestion, upgrading and hygienization had little or negligible influence on any impact category.

3.2. Isolation of the Effect of Regional Differences

The influence of regional differences on impacts is shown in Figure 5, where the contribution from hygienization and transport is excluded for the NoSep scenario. The characteristics of the recipient region in the NoSep scenario gave lower impacts (27–113% reduction) in all categories relative to the reference scenario except for marine eutrophication where the net impact was 14% higher. The reduced impacts in the recipient region were either caused by a greater amount of avoided mineral fertilizer, lower emissions from slurry application, or a combination of the two, while the higher eutrophication impact is explained by higher rates of NO_3 losses from arable land than from grass land.

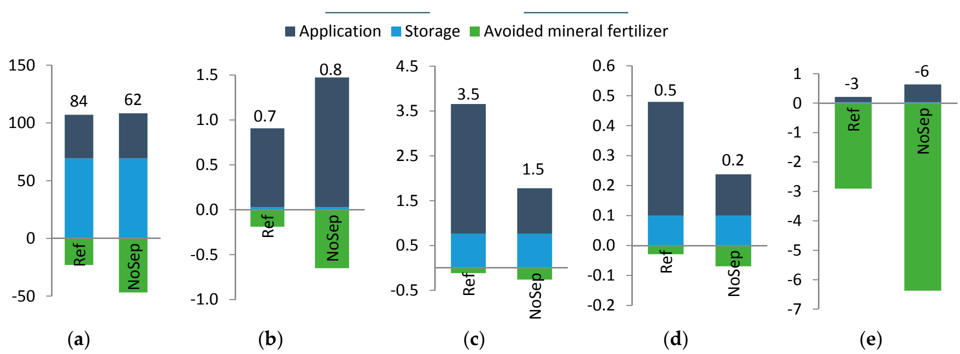


Figure 5. Impact results where the influence of regional differences is isolated for the following impact categories: (**a**) climate change (in kg CO_2-eq.); (**b**) marine eutrophication (in kg N-eq.); (**c**) terrestrial acidification (in kg SO_2-eq.); (**d**) particulate matter formation (in kg PM10-eq.); and (**e**) fossil resource depletion (in kg oil-eq.). The net impacts are shown in numbers (rounded) above the bars and exclude hygienization and transport for the NoSep scenario.

3.3. Sensitivity Analysis

3.3.1. Perturbation Analysis

For the reference scenario, three parameters had a sensitivity ratio (SR) of one or higher (Table 5). The effect on marine eutrophication of a variation in the factor for NO_3 emissions from application on grassland was the greatest, with an SR of 1.1. That meant that if the emission factor for NO_3 from application on grassland had been 10% larger, for example, the net impact for marine eutrophication for the reference had increased by 11%. For the AD_SP scenario, six parameters had an SR ≥1 for one or more impact categories (Table 5). Changing the content of total N in the raw manure had a particularly great effect on both marine eutrophication and climate change, with an SR of 1.1 and 1.3, respectively. Changing the DM content of manure had contrasting effects on the two scenarios. For the reference scenario, increasing the DM content led to an increase in climate change, as this increased the amount of OM to be converted to CH_4 emissions. For AD_SP, the effect described above was outweighed by the increased amount of biogas produced replacing fossil fuel, thus giving an SR of −0.2. Overall, parameters determining the composition of manure and slurry dominated the presented parameters for both scenarios. This shows that the impacts in the model are, to varying extents, sensitive to changes in manure composition in particular and that sensitivity varies between scenarios.

Table 5. Sensitivity ratio (SR) results from the perturbation analysis for the reference and AD_SP scenarios. Only parameters with an absolute SR value ≥0.1 for at least one impact category are shown, and values ≥0.5 are shown in bold.

Parameter	Impact Category				
	CC	ME	TA	PMF	FD
Reference scenario					
DM content manure	0.8	-	-	-	-
OM share of DM in manure	0.8	-	-	-	-
Tot-N content manure	0.3	1.0	0.3	-	0.3
NH_4-N content manure	−0.1	-	0.4	1.0	0.4
P content manure	-	-	0.5	-	0.3
NH_3 emission application on grass	-	0.2	-	0.8	-
NO_3 emission application on grass	-	1.1	-	-	-
CH_4 emission long storage manure cellar	0.8	-	-	-	-
AD_SP scenario					
DM content manure	−0.2	-	-	−0.2	1.2
OM share of DM in manure	1.2	-	−0.2	−0.2	−0.1
Tot-N content manure	1.3	1.1	0.4	0.4	0.3
NH_4-N content manure	−0.9	−0.1	0.6	0.7	0.2
P content manure	−0.2	-	-	-	0.1
Amount of manure per cow	−0.4	-	-	-	0.3
Amount of wash water per cow	0.5	-	-	-	−0.3
NH_3 emission application on grass	-	0.2	0.8	0.9	-
NO_3 emission application on grass	0.1	1.0	-	-	-
NO_3 emission application on arable land	-	0.3	-	-	-
Separation efficiency mass	1.2	-	-	-	−0.8
Separation efficiency Tot-N	0.4	0.1	-	-	-

"-" = absolute SR value <0.1; CC = climate change; ME = marine eutrophication; TA = terrestrial acidification; PMF = particulate matter formation; FD = fossil resource depletion.

3.3.2. Scenario Analysis of Basis for Fertilizer Application on Arable Land

Changing from a P-based to an N-based application of fertilizer on arable land did not change the ranking of the scenarios for any impact category. N-based manure application in the recipient arable region produced only minor changes in most impacts except for P rock depletion, where it reduced the amount of avoided mineral P by 82% for scenarios DC and AD_DC, by 59% for scenarios SP and AD_SP, and by 51% for the NoSep scenario (Figure 4).

3.3.3. Scenario Analysis of Soil P Level

Assuming optimal soil P levels and balanced fertilization at both the donor and recipient farm mostly affected fossil resource depletion and avoided mineral P/P over application as more mineral P was avoided in the donor region. For fossil resource depletion the net impact for the reference and the SP scenario was reduced by 37% and 43%, respectively. A change in the ranking of scenarios only happened for avoided mineral P/P over application, where all scenarios but the reference replaced the maximum amount of mineral P fertilizer. In the reference, 15% of the applied P was still over applied. This could happen because the maximum allowed manure P applied per hectare exceeds the donor P fertilizer requirement even at optimal soil P levels.

3.3.4. Scenario Analysis of Transport Distance and Mode

Varying the transport distance and mode affected the total life cycle climate change impact for the four selected scenarios as shown in Figure 6. All the three scenarios involving transport (NoSep, DC, and AD_DC) started off with a lower impact than the reference scenario at zero km, where the contribution from transport was subtracted from total life cycle climate change impact of the scenarios. The differing slopes of the lines for the same transport mode across scenarios reflect the different masses transported. Transport by lorry had, in addition, approximately one order of magnitude higher impact on a per ton kilometre basis than that of freight train and freight ship. The small solid fraction in the AD_DC scenario could be transported more than 1500 km by any transport mode without reaching the net impact of the Reference scenario. On the other extreme, the bulky slurry in the NoSep scenario could only be transported 65 km by lorry before the scenario reached the same potential impact as the Reference, while freight ship would increase that distance to 945 km.

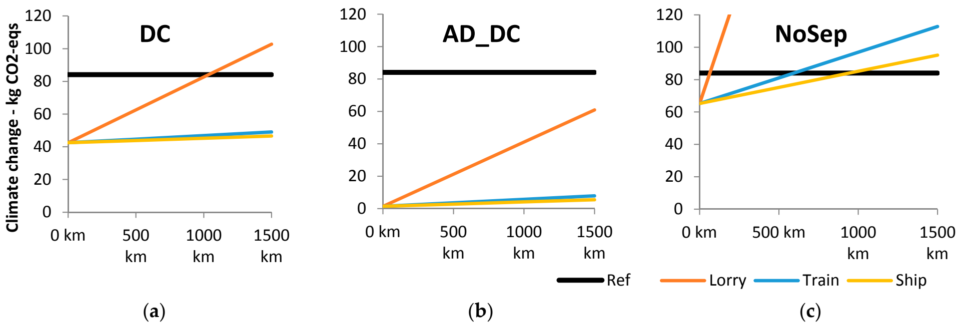


Figure 6. Sensitivity of the life cycle climate change net impact of varying transport distance and transport modes between the donor and recipient farm per FU. The following scenarios were compared against the reference (Ref) scenario: (**a**) the DC scenario; (**b**) the AD_DC scenario; (**c**) the NoSep scenario. In NoSep, a higher volume was transported than in the two other redistribution scenarios and the increase in net impact per additional kilometre was therefore greater for the NoSep scenario.

4. Discussion

4.1. The Environmental Impact of Manure P Redistribution

In this study, the main objective was to explore the potential environmental impacts involved in redistributing manure P from a region of P surplus to a region with a need to import P fertilizer. The findings demonstrate that increased P use efficiency through geographic redistribution of manure P does not need to come at the cost of increased environmental impacts compared to business as usual. Combining anaerobic digestion with decanter centrifuge separation of the digestate seemed particularly promising. Despite the long transport distance, scenarios including solid–liquid separation mostly had a similar or lower potential impact on the environment than the reference scenario. ten Hoeve et al. [17] reported similar findings for solid–liquid separation (without AD) of pig slurry and redistribution of manure nutrients over 100 km in Denmark and also identified centrifuge separation as potentially the most environmentally beneficial option. The lower potential impact on fossil resource depletion for the AD scenarios than for the reference scenario contrasts De Vries et al. [16], who found that processing cattle manure with AD actually increased the potential for fossil depletion by 19%. The difference may be explained by our manure processing system (excluding AD) being simpler and requiring two- to eight-fold less energy per ton substrate, by the Nordic electricity mix consisting of more renewable energy such as hydropower, and by the upgraded biogas replacing fossil diesel, considered the best use of biogas in environmental terms in Norway [48]. Substitution of diesel fuel was considered realistic for this case study, but would overestimate the benefit from biogas production in regions where the distance to upgrading facilities may be too long to be economically viable.

4.2. The Influence of Regional Characteristics and Transport

The second objective of this paper was to explore whether characteristics of the donor and recipient region influenced the net impacts of the scenarios, in line with recent recommendations for improving LCA studies of agri-food systems [18]. Applying the slurry in the recipient region (NoSep scenario) gave a clear reduction in net impacts compared to slurry application in the donor region (Reference scenario) (Figure 5). For the other scenarios where manure products were applied in both regions, the use of processing technologies determined the mix of donor/recipient application in each scenario. Regional differences also motivated the identification of a recipient region for surplus P from the donor region through differing soil P levels. Assuming optimal soil P test values in both regions (Section 3.2.3) practically eliminated the problem of P over-application in the donor region and therefore also removed the motivation to redistribute manure P in the first place.

Transport mode and distance was another factor thought to influence environmental impacts of nutrient redistribution, tested in Section 3.2.4. The results showed that the NoSep scenario most likely could benefit considerably from transport by train or ship in terms of potential climate change impact. For the other redistribution scenarios—represented by the DC and AD_DC scenarios—the mode of transportation had less of an impact on net climate change for the distance used in the case study when compared to the contribution from the other life cycle processes (Figure 4). However, this very simple indication of the effects of transportation contains two erroneous underlying assumptions: the estimations assume that the recipient region may be down to zero kilometres away from the donor region, which is impossible, and it is also unlikely that either train or ship go all the way from farm gate to farm gate.

The influence on impacts of regional characteristics support the previously mentioned recommendations [18], and imply that future LCA studies on geographical redistribution of secondary nutrients need to specify the characteristics of both the donor and the recipient region. However, the transport sensitivity indicates that, unless a greater fraction of the FU is to be transported (as in the NoSep scenario), or the distance is >>500 km, the transport of manure products does not dominate potential impacts on climate change.

4.3. Assumptions for Mineral Fertilizer Substitution

Other studies have identified avoided mineral fertilizer as a dominant and beneficial contribution to impacts on climate change and fossil resource depletion in particular [15,48], but that was not the case in the current study. The varying importance of mineral fertilizer substitution seems to originate from different assumptions regarding how manure nutrients replace mineral fertilizer nutrients. Both Brockmann et al. [15] and Lyng et al. [48] assumed that plant available manure nutrients replace the equivalent amount of mineral fertilizer nutrients. In this study, we took a more conservative approach to mineral fertilizer substitution by relating it to fertilizer nutrient requirements. Hence, any over-application of a nutrient did not replace the corresponding mineral nutrient. The long-term effect of soil P accumulation is accounted for through soil P tests, on which any necessary corrections to P fertilization are based [54]. Knowing more about the sensitivity of the results to different assumptions regarding mineral fertilizer substitution might make studies easier to compare and should be looked into in future research.

The emissions associated with production of mineral fertilizer vary depending on the production technology used [60]. We used ecoinvent data for average European fertilizer production in this study. However, according to Refsgaard et al. [61], the mineral fertilizer produced in Norway is manufactured using the best available technology in Europe. Emissions from mineral fertilizer production and thus the benefits from mineral fertilizer substitution may therefore have been overestimated in the present study. This is presumably most relevant for the impact categories reflecting energy use, such as climate change and fossil resource depletion, but is not expected to change the ranking between the scenarios. A breakdown of contributions to climate change in this study showed that approximately 60% of the avoided emissions (measured in CO_2-eq.) from replacing mineral fertilizer came from its production, while the remaining 40% came from emissions related to field application.

4.4. Parameter Uncertainties

There are uncertainties surrounding several of the parameters used in this study, including e.g., emissions of CH_4 and N_2O from storage. According to Rodhe et al. [62], there were negligible emissions of N_2O from slurry stored outdoors under cover in Sweden, a similarly cold climate to Norway. Moreover, Dinuccio et al. [63] observed no N_2O emissions from storage of untreated cattle slurry or its separated solid and liquid fractions at 5 and 25 °C. This could mean overestimation of the climate change contribution from the fractions stored outdoors in this study, since N_2O was the most important greenhouse gas emitted from this process in term of CO_2-eq. Dinuccio and colleagues also found that CH_4 emissions from storage of cattle slurry were lower at 25 °C than at 5 °C. The lower emissions at 25 °C were explained by higher water loss over time and, thus, an increased concentration of inhibitory substances for methanogenesis [63]. In contrast, Sommer et al. [44] found that CH_4 emissions were positively correlated with OM and temperature, with a transfer of slurry from in-house storage to outdoor storage in a colder environment resulting in a modelled reduction in CH_4 emissions from cattle slurry. However, the perturbation analysis performed in the present study showed that the model was rather insensitive to variation in most factors for storage emissions except the rate of CH_4 emissions during in-house storage in the reference scenario (Table 5).

4.5. The Studied Case and the European Perspective

In the present study, we employed region-specific parameters to determine field emissions and manure fertilizer values, which make the results less directly transferable to geographical settings different from the case study. The possibility of reducing applicability of results by specifying the conditions surrounding slurry field application was also previously noted by the authors of [17]. However, we believe that the case study regions represent the larger scale variation in national agricultural P balances between the EU member states [64]. Most western European countries have positive agricultural P balances—caused e.g., by application of manure P from intensified livestock production—and consequently high levels of accumulated soil P. Many central and eastern European countries have an agricultural P deficit on national level [64]. The need for P redistribution is clearly present. The inclusion of specific regional parameters in future LCA studies is necessary to determine the most environmentally beneficial redistribution solutions on a case-to-case basis as potential donor and recipient regions necessarily reflect different farming systems. The greater perspective on nutrient imbalances between regions has also motivated thoughts on the long term structure of agricultural production: the Food and Agriculture Organization of the United Nations has stated that livestock production should be located within economic reach of arable land to receive the waste produced and so avoid problems of nutrient loading [65]. Better co-location of animal and crop farming would obviously reduce transport-related emissions associated with manure P redistribution, but this study indicates that other processes in the value chain may be more important for environmental impacts. Our results are as such in line with the findings of Willeghems et al. [66].

4.6. Limitations and Further Research

The influence of capital goods were assumed to be negligible for the results and, therefore, not looked into. However, the study could have included capital goods if only to rule out any notable contribution. After the common manure cellar storage, every scenario used a unique combination of capital goods, and it is plausible that a certain combination may in fact contribute somewhat to impacts. If time and resources allow, we recommend that this be looked into in future research comparing alternatives for manure management. Furthermore, although manure P redistribution can be environmentally beneficial, this may be regarded a necessary but not sufficient condition to ensure implementation of redistribution systems. A central enabling factor will be social acceptance, apart from regulatory and/or economic incentives, and in this study we assumed for simplicity that the transported manure products were directly acceptable and usable at the recipient farm. In reality, these manure products may have to be processed further to meet the needs of receiving farmers. Aspects such as compatibility with existing spreading equipment and an N-P-K nutrient balance to match crop fertilizer requirements will have to be addressed. Future LCA studies on manure P redistribution could therefore include the life cycle environmental impacts of such additional processing.

5. Conclusions

The purpose of this study was to estimate the potential environmental impacts of redistributing manure P from a livestock-intense region with P surplus to an arable farming region with a need for P fertilizer, exemplified by two regions in Norway. The performed life cycle assessment (LCA) indicates that such redistribution can be done without increasing most impacts when compared to a reference scenario with no redistribution. Anaerobic digestion of cattle slurry with subsequent solid–liquid separation of the digestate by decanter centrifuge was the most promising scenario studied. The result is specific for the case study and not directly transferable to other geographical settings, but the overall challenge of agricultural specialization and associated P use inefficiency is relevant for many areas. We show that different regional characteristics do affect impacts related to field application and substitution of mineral fertilizer and we expect the same for other cases where P redistribution is considered. Different characteristics between agricultural regions are what motivate P redistribution in the first place, and this study reemphasizes the need to include region-specific parameters in LCA studies on nutrient redistribution.

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Paper V

Choice of mineral fertilizer substitution principle strongly influences LCA environmental benefits of nutrient cycling in the agri-food system

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Choice of mineral fertilizer substitution principle strongly influences LCA environmental benefits of nutrient cycling in the agri-food system

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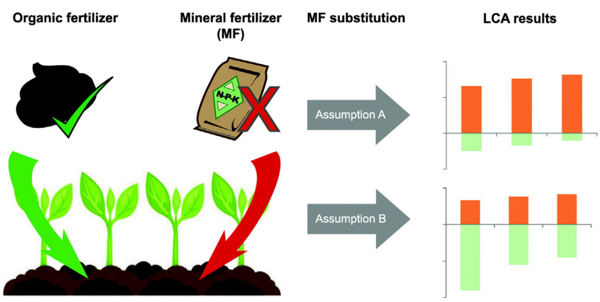
HIGHLIGHTS

- Methods for mineral fertilizer substitution in LCA need more clarity.
- Three substitution principles used in LCA are identified from literature.
- The importance of these substitution principles is tested in a case study.
- Choice of principle greatly affects environmental impacts.
- A set of recommendations for LCA practitioners is proposed.

GRAPHICAL ABSTRACT



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ABSTRACT

Increased nutrient cycling in the agri-food system is a way to achieve a healthier nutrient stewardship and more sustainable food production. In life cycle assessment (LCA) studies, use of recycled fertilizer products is often credited by the substitution method, which subtracts the environmental burdens associated with avoided production of mineral fertilizer from the system under study. The environmental benefits from avoided fertilizer production can make an important contribution to the results, but different calculation principles and often implicit assumptions are used to estimate the amount of avoided mineral fertilizer. This may hinder comparisons between studies. The present study therefore examines how the choice of substitution principles influences LCA results. Three different substitution principles, called one-to-one, maintenance, and adjusted maintenance, are identified, and we test the importance of these in a case study on cattle slurry management. We show that the inventory of avoided mineral fertilizer varies greatly when the different principles are applied, with strong influences on two-thirds of LCA impact categories. With the one-to-one principle, there is a risk of systematically over-estimating the environmental benefits from nutrient cycling. In a sensitivity analysis we show that the difference between the principles is closely related to the application rate and levels of residual nutrients in the soil. We recommend that LCA practitioners first and foremost state and justify the substitution method they use, in order to increase transparency and comparability with other studies.

Abbreviations: FU, functional unit; ha, hectare; K, potassium; LCA, life cycle assessment; LCI, life cycle inventory analysis; LCIA, life cycle impact assessment; MFE, mineral fertilizer equivalent; MFE-N, -P, and -K, mineral fertilizer equivalent of nitrogen, phosphorus, and potassium in organic fertilizer; N, nitrogen; P, phosphorus.

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1. Introduction

Increased recycling of nutrients is key to achieving improved nutrient use efficiency in agri-food systems and more sustainable food production (Sutton et al., 2013). Use of the life cycle assessment (LCA) methodology can provide policy-relevant information about the environmental effects of nutrient recycling options (Notarnicola et al., 2017). Waste handling and nutrient recycling is an example of a multi-functional system in LCA, fulfilling more than one function (Ekvall and Finnveden, 2001): providing both the function of waste management and the function of fertilizer production. In studies on multi-functional systems for waste handling and recycling, it is common to use the substitution method to credit the recycling of materials (Laurent et al., 2014). The same method is often used in studies on recycling of nutrients in organic residues (Hélias and Brockmann, 2014). The substitution method implies subtracting the environmental burdens of the avoided production from the system (Heijungs and Guinée, 2007) and is also referred to as the avoided burdens method and the equivalent to systems expansion (Vadenbo et al., 2016). There is still an ongoing debate concerning the recommended way to allocate burdens in multi-functional systems (see e.g. Ekvall and Finnveden, 2001; Heijungs and Guinée, 2007; Pelletier et al., 2015; Weidema and Schmidt, 2010). However, our intention in this paper is not primarily to contribute to that discourse. Instead, our point of departure is that the substitution method will most probably continue to be a frequently used method in LCA studies involving recycling of materials, including nutrients in organic residues. According to Vadenbo et al. (2016), the benefit obtained from avoided processes frequently dominates the results of LCA studies of resource recovery in waste management, and thus assumptions affecting substitution are a crucial step in the life cycle inventory analysis (LCI) of the LCA performance (Ekvall, 1999; Heijungs and Guinée, 2007). Brockmann et al. (2014) examined the influence on LCA results of different ways of calculating the nitrogen fertilizer value of pig slurry and found that it affected the net impact in some impact categories by a factor of around 1.5–3, while other categories were unaffected. However, given a certain fertilizer value, there is a need to make another, and potentially equally important, set of assumptions regarding how those plant-available nutrients replace the production and field application of mineral fertilizer. An example of such a set of assumptions is that all the plant-available nutrients in the organic fertilizer displace their mineral fertilizer counterparts in a ratio of one to one. Although easy to apply, this assumption of one-to-one displacement does not consider agronomic reality, as it ignores unbalanced nutrient composition and any over-application of nutrients, and therefore risks overestimating nutrient recycling benefits. It has been suggested that studies showing differing beneficial impacts of avoided mineral fertilizer may use very different substitution assumptions (Hanserud et al., 2017), one-to-one displacement being one. Vadenbo et al. (2016) exemplified a proposed framework for substitutability with the field application of an organic fertilizer to displace mineral fertilizer, but did not fully cover the different perspectives on fertilizer requirement that could be decisive for substitutability of organic fertilizers in particular. They also stopped after the LCI phase of the LCA. The overall aim of the present study is to demonstrate how choice of substitution principle affects the results in LCA studies that include mineral fertilizer substitution. To the best of our knowledge, no previous study deals systematically and explicitly with this issue. In this paper, we first identify three different substitution principles for mineral fertilizer substitution found in published LCA studies on organic fertilizers and nutrient recycling. Studies that employ the different substitution principles we describe also differ in terms of types of organic residues studied and in scope. Therefore, we compare how these principles affect inventory modelling and environmental impacts in a case study. We employ a case study on conventional cattle slurry management in southern Norway, since it is a simple and well-defined system with few disturbing factors for the analysis of nutrient recycling, and because we have access to data of good quality, which is important for such a case study demonstration of substitution principles.

2. Material and methods

2.1. Principles for mineral fertilizer substitution

Here we describe three different substitution principles identified in the LCA literature (Table 1), where each principle includes a set of assumptions. These assumptions are made, explicitly or implicitly, by the LCA practitioner to determine the amount of avoided mineral fertilizer in the LCI phase of the LCA due to field application of organic fertilizer. We base the substitution of mineral fertilizer on a mineral fertilizer equivalent (MFE) value of the organic fertilizer for the nutrients nitrogen (N), phosphorus (P), and potassium (K). MFE is the fraction of the total content of N, P, and K in the organic fertilizer with the same fertilizer effect as mineral fertilizer N, P, and K, respectively. Thus, e.g. 1 kg of MFE-N in the organic fertilizer is assumed to be perfectly substitutable with 1 kg of mineral N fertilizer, both expressed as mass of elemental N. The MFE values of N, P, and K in organic fertilizer can be either technically or institutionally determined (see Vadenbo et al., 2016). The three substitution principles described below differ in the way the amount of avoided mineral fertilizer is determined based on estimated MFE values. Moreover, inputs of fertilizer, crop outputs, and associated environmental burdens need to be related to a functional unit (FU) in LCA that defines the function of the system under study (ISO, 2006a). In studies including nutrient recycling, a typical FU may be input-related, since the function of the system is often to find the best use of a certain nutrient-containing substrate (Cherubini and Strømman, 2011). We therefore base the following descriptions on the use of an input-related FU equal to management of a certain mass of substrate (management of 1 tonne of fresh dairy cow manure is used in the case study).

2.1.1. One-to-one substitution principle

This is the most straight-forward and seemingly most commonly employed principle in LCA studies using the substitution method for

Table 1
Brief description of three mineral fertilizer substitution principles identified in the literature, with increasing complexity from top to bottom (abbreviated name of principle in brackets).

Mineral fertilizer substitution principle	Description	Literature references
One-to-one substitution principle (one-to-one principle)	• The amount of avoided mineral N, P, and K fertilizer equals the amount of MFE-N, -P, and -K in the organic fertilizer in a ratio of 1:1.	De Vries et al. (2012b), Bernstad and la Cour Jansen (2011), Lantz and Börjesson (2014), Mezzullo et al. (2013), Lyng et al. (2015), Brockmann et al. (2014)
Maintenance substitution principle (maintenance principle)	• A certain crop or crop rotation receiving the organic fertilizer is given. • Applied MFE-N, -P, and -K in the organic fertilizer is compared with the general crop fertilizer requirement for each nutrient. • Any over-application does not substitute mineral fertilizer.	De Vries et al. (2012a), Hamelin et al. (2014), Hamelin et al. (2011), Tonini et al. (2012)
Adjusted maintenance substitution principle (adjusted principle)	• A certain crop or crop rotation receiving the organic fertilizer is given. • Applied MFE-N, -P, and -K in the organic fertilizer is compared with the crop fertilizer requirement for each nutrient, adjusted for local or regional soil characteristics. • Any over-application does not substitute mineral fertilizer.	ten Hoeve et al. (2014), Croxatto Vega et al. (2014), Hanserud et al. (2017)

nutrient recycling. Under this principle, the content of MFE-N, -P, and -K in organic fertilizer products is assumed to replace the equivalent amount of mineral fertilizer N, P, and K, respectively. Studies employing this principle include De Vries et al. (2012b), Bernstad and la Cour Jansen (2011), Lantz and Börjesson (2014), Mezzullo et al. (2013), Lyng et al. (2015), and Brockmann et al. (2014). With the one-to-one principle, it is assumed that the nutrient value of a secondary fertilizer is independent of the fertilizer requirement of any crop and receiving soil. The principle can be described with the following equation:

$$AMF_X = MFE\text{-}X \tag{1}$$

where AMF_X is the amount of avoided mineral fertilizer of nutrient X (kg), and *MFE-X* is the amount of mineral fertilizer equivalent of nutrient X (kg) according to the chosen FU. A rougher version of the one-to-one principle would be to assume 1:1 substitution between the total amount of an applied nutrient and the corresponding mineral fertilizer nutrient, without considering the MFE fraction of the applied nutrient. We do not include this version in the remainder of this paper, as consideration of an MFE fraction seems more common in papers on nutrient recycling. However, in LCA studies on solid waste management with material recovery, Laurent et al. (2014) noted in their review that most studies assume a 1:1 substitution ratio without considering lower quality in the recovered materials.

2.1.2. Maintenance substitution principle

This principle assumes a typical crop grown on the farmland receiving the organic fertilizer, and the amount of avoided mineral fertilizer is determined based on the general fertilizer recommendation for this crop. An average annual fertilizer recommendation for a typical crop rotation can also be used, as done by Hamelin et al. (2011). The general fertilizer recommendation is usually proposed with mineral fertilizer in mind, based on empirical results. With this, common losses to the environment are accounted for in the fertilizer recommendations. An alternative to general fertilizer recommendations is to assume application of fertilizer equal to the amount of nutrients removed from the land with the harvested crop and lost to the environment. In both cases, the input into the field should match the output from the field, so that soil nutrient level is maintained constant. Studies that employ this principle include De Vries et al. (2012a), Hamelin et al. (2014), Hamelin et al. (2011), and Tonini et al. (2012). Since the ratio between nutrients in the organic fertilizer often does not match the necessary nutrient ratio required by the crop (Brod et al., 2015), the maintenance principle allows the LCA analyst to correct the inventory for over- and under-application of single nutrients. Any applied amount exceeding the fertilizer requirement (over-application) should not replace mineral fertilizer (Hamelin et al., 2011), since the farmer would not apply more mineral fertilizer than required. The application area needed for the FU may be determined based on the nutrient content in the organic fertilizer, and either: i) the fertilizer requirement per hectare (ha) or ii) maximum application rate given in the current regulatory framework. In the case of (i), the area is typically based on the content of MFE-N in the organic fertilizer and the N fertilizer requirement per area unit for the crop (Hansen et al., 2006). An example of (ii) is the maximum application rate of 170 kg total N in animal manure per hectare (ha) stated in the European Nitrate Directive (European Commission, 2017). Regardless of whether the application area is based on fertilizer requirement or limited by regulations, the area dictates the application rate of the remaining nutrients, typically MFE-P and MFE-K. If the application in the receiving region is limited by regulation, both methods should be used and the larger area chosen, so the application area complies with both. To describe the maintenance principle mathematically, we start by determining the necessary application area according to i), based on N content:

$$A_N = MFE\text{-}N/\alpha_N \tag{2}$$

where A_N is the area (ha) needed to spread the nutrients in the FU based on its MFE-N content, *MFE-N* is the amount of MFE-N (kg) in the FU, and α_N is the general requirement of N fertilizer per hectare (kg/ha) for a given crop. The application area according to (ii) could be expressed in the following general way:

$$A_{Regulation} = X/Limit_X \tag{3}$$

where $A_{Regulation}$ is the area (ha) needed to spread the nutrients in the FU according to regulation, X is the amount of the nutrient X (kg) in the FU (either MFE-X or total amount of nutrient X), and $Limit_X$ is the application rate limit (kg/ha) for nutrient X (either MFE-X or total amount of X). The final application area, A (ha), is the largest area of the two methods:

$$A = \max(A_N; A_{Regulation}) \tag{4}$$

Next, the actual amount of fertilizer nutrient X required, Req_X (kg X), is equal to the necessary application area, A, multiplied by the general fertilizer requirement of nutrient X per ha for a given crop, α_X (kg/ha):

$$Req_X = A \times \alpha_X \tag{5}$$

The avoided mineral fertilizer per FU is equal to the lowest value – either the applied MFE nutrient or the fertilizer requirement (Eq. (6)). Eqs. (5) and (6) must be calculated separately for each of the nutrients N, P, and K.

$$AMF_X = \min(MFE\text{-}X; Req_X) \tag{6}$$

2.1.3. Adjusted maintenance substitution principle

This principle adjusts the general fertilizer requirement used in the maintenance principle to account for the influence of local or regional soil characteristics. This adjustment is particularly relevant for P and K, as existing levels of residual P and K in the soil may greatly affect the recommended fertilizer application rate (Sattari et al., 2012; van der Bom et al., 2017). In many parts of the world, but particularly in western Europe, residual P has accumulated in agricultural soils over time due to inputs of mineral P fertilizer and manure exceeding crop uptake (Sattari et al., 2012). The residual P can in turn serve as a source of P and reduce the need for P fertilizer (Rowe et al., 2016). The adjustment factor β_X for fertilizer nutrient X is dimensionless and provides the adjusted fertilizer requirement when multiplied by the general fertilizer requirement (Eq. (7)). There are different ways of determining the adjustment factor, and according to Tóth et al. (2014) the UK and the Hungarian approach are distinctly different as regards P fertilization. The UK approach uses only soil P level to adjust the fertilizer requirement, while the Hungarian approach includes additional soil characteristics such as soil texture and $CaCO_3$ content. For this reason, we do not specify how β_X should be determined, but provide an example of its use in the case study. Studies using the adjusted maintenance substitution principle in LCA on organic fertilizers include ten Hoeve et al. (2014), Croxatto Vega et al. (2014), and Hanserud et al. (2017), which all reduce the fertilizer requirement for P according to high soil P levels. The necessary application area is determined as in Eqs. (2)–(4), while the adjusted fertilizer requirement is calculated as:

$$Req_X Adj = A \times \alpha_X \times \beta_X = Req_X \times \beta_X \tag{7}$$

where $Req_X Adj$ is the adjusted fertilizer requirement for nutrient X (kg) per FU. The avoided mineral fertilizer per FU in the adjusted principle is equal to the lowest value – either the applied MFE nutrient or the adjusted fertilizer requirement:

$$AMF_X = \min(MFE\text{-}X; Req_X Adj) \tag{8}$$

2.2. Case study description

2.2.1. LCA approach, goal, and scope

LCA is a standardized methodology described in ISO 14040 and 14044 (2006a,b) and further described in documents such as the ILCD handbook (European Commission JRC, 2010). In order to illustrate the influence of the different substitution principles on the inventory and environmental impacts, we perform an LCA on the management of dairy cow manure, representing an organic fertilizer. By studying the entire manure management system, it is possible to evaluate the importance of mineral fertilizer substitution impacts relative to total system impacts. Here we study a fairly conventional manure management system, in order to focus on mineral fertilizer substitution without introducing additional beneficial co-products like biogas from anaerobic digestion. The amount of avoided mineral fertilizer is calculated individually for the one-to-one, maintenance, and adjusted maintenance substitution principles, in order to compare the resulting inventories and final LCIA results. Because the production of biomass as a by-product is often the driver of subsequent management, an input-related FU (Cherubini and Strømman, 2011), set as management of 1 tonne of fresh dairy cow manure, is used in the case study.

2.2.2. Geographical scope and system boundary

The case is located within the geographical context of southern Norway, in a livestock-intensive region with grass as the dominant crop, and is largely based on the reference scenario described in Hanserud et al. (2017). The soil test P level is assumed to be very high (Hanserud et al., 2017) and soil K levels are assumed to be at a level where the general recommendation on K application is reduced due to K contribution from soil. The system boundary and the processes included are shown in Fig. 1. The system starts with generation of fresh dairy cow manure that enters in-house storage of manure. During storage of manure, gaseous emissions to the atmosphere occur and wash water is conducted to the storage facility, turning the manure into a slurry and increasing the volume to be further managed. The stored slurry is eventually applied to farmland as an organic fertilizer, inducing losses of N to both atmosphere and water/soil recipients through runoff and leaching. The application rate is limited by the Norwegian regulation on organic fertilizer, which permits a maximum application of 35 kg P in manure per ha on livestock farms (The Norwegian regulations relating to organic fertiliser, 2003). The nutrients contained in the slurry contribute to displacing production and field application of mineral fertilizer, with the amount of mineral fertilizer displaced depending on the substitution principle employed. To account for the beneficial impact of avoided mineral fertilizer production, the system is expanded here to include the production of separate mineral N, P, and K fertilizer, as in Hansen et al. (2006). Materials and energy used in construction of the manure storage facility are omitted, as they are expected to contribute little to the overall life cycle impact of the FU (see e.g., Brogaard et al., 2015). However, capital goods associated with slurry spreading on the field are included in the ecoinvent process of broadcast spreading.

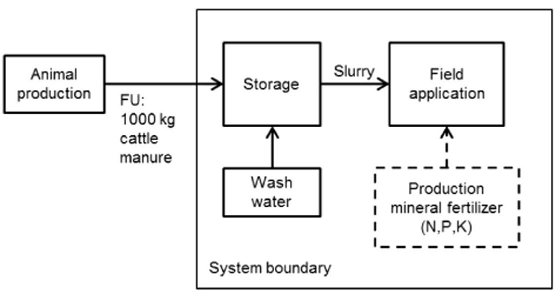

Fig. 1. System outline for processing and recycling of nutrients in dairy cow manure. Boxes represent processes, arrows represent flows between processes, and the dotted box and arrow represent avoided production and avoided flow of products, respectively.

2.2.3. Inventory analysis

A number of assumptions made for the case study can affect emissions and resource use throughout the system. Characterization of the dairy cattle manure, emission factors for the processes, and chemical composition of the FU are provided in the Supplementary material to this paper. The following subsections provide background information and assumptions relevant to the processes included in the system.

2.2.3.1. Storage. Slurry is assumed to be stored for an average of 3 months in a manure cellar under the animal house, during which emissions of ammonia (NH_3), nitrous oxide (N_2O), and methane (CH_4) occur while organic matter (OM) is degraded, reducing the amount of dry matter (DM) and the total slurry mass. The degradation of OM is assumed to be 10% of the OM entering storage (Wesnæs et al., 2009).

2.2.3.2. Field application. Stored slurry is assumed to be applied on local grassland by broadcasting during the growing season (Gundersen and Heldal, 2015). The emissions and resource use associated with slurry broadcasting (combustion of diesel fuel, tire abrasion, etc., excluding emissions from the slurry) is a function of application area in ecoinvent (Wernet et al., 2016). It is therefore relevant for the maintenance and adjusted principles, where area can be calculated, but not for the one-to-one principle, where a spreading area is not possible to determine. However, this does not prevent calculation of emissions based on slurry characteristics, spreading technology used, and characteristics of the receiving farmland, which are identical for all three principles. Because of the P application limit, the slurry is applied based on its MFE-P content, requiring a greater area than application based on its MFE-N content and the N fertilizer requirement.

2.2.3.3. Fertilizer value of slurry. The amount of MFE-N is calculated here by a mass balance-based method described by Brockmann et al. (2014), where the amount of plant-available N is the difference between the applied mineral N (NH_4-N), including the mineralization of organic N during the first growing season, and N losses to the environment. Details of the calculation can be found in the Supplementary material. The MFE value for P in dairy cow slurry is assumed to be 100% (Brod et al., 2015) and the same is assumed for K (De Vries et al., 2015). Losses of N, P, and K to water recipients are assumed to be similar for manure and mineral fertilizer, and are therefore not subtracted from the MFE values.

2.2.3.4. Fertilizer requirement. The maintenance and adjusted principles require information about a typical crop and an expected crop yield to determine fertilizer requirement. Here we assume intensive grass production with annual expected yield of 10,000 kg DM grass per ha, harvested over three cuts. The general fertilizer requirement, α_X, is then determined based on the Norwegian fertilizer handbook (NIBIO, 2016), corresponding to 270 kg N/ha (α_N), 30 kg P/ha (α_P), and 168 kg K/ha (α_K). For the adjusted principle, the adjustment factor, β_X, for P and K is calculated based on information about soil type and soil nutrient levels, giving $\beta_P = 0$, and a $\beta_K = 0.375$. Information on soil characteristics and calculations can be found in the Supplementary material.

2.2.4. Impact assessment

For impact assessment, we use the LCA software SimaPro 8.1.0.60 and the impact assessment methodology of ReCiPe 2008 (Goedkoop et al., 2009). Midpoint impacts for the 18 ReCiPe impact categories are calculated based on the hierarchical 100-year perspective. A smaller selection of around 3–6 impact categories is often used for systems involving recycling of nutrients in agri-food systems, but by choosing all 18

ReCiPe impact categories we get a better overall picture of the distribution of relative dominance of mineral fertilizer substitution compared with the contribution from the other system processes.

2.2.5. Sensitivity analysis

In addition to the case study, we examine the effect of the following four sensitivity scenarios on amount of avoided mineral fertilizer:

- "No reg": no regulation of application rates, high soil P and K levels as in the case study. Application area is determined based on MFE-N content in the slurry and N fertilizer requirement. Although resulting application rates do not correspond to good agronomic practice for organic fertilizers with low nutrient concentrations, such as cattle slurry, this may be more relevant for other products with a higher dry matter content and higher nutrient concentrations.
- "Nitrate": limitation of manure N application rate according to the European Nitrate Directive, high soil P and K levels as in the case study. The directive limits the application of total N in manure to 170 kg N per ha and applies to farmland in Norway located within catchment areas draining to vulnerable coastal zones.
- "Case study, low": assuming low soil P and K levels as opposed to the high soil P and K levels assumed in the case study. The same application rate limitation as in the case study is assumed. With the Norwegian fertilizer recommendations, low soil P and K values would lead to a higher application than the general fertilizer recommendations, and therefore $\beta > 1$. An adjustment factor of $\beta_P = \beta_K = 1.25$ reflects plausible low soil nutrient levels in Norway, although this would be unusual and not found in a typical livestock farming region.
- "Nitrate, low": this reflects a situation where the EU Nitrate Directive applies, as in the "Nitrate" scenario, but with low soil P and K levels, as in the "Case study, low" scenario.

3. Results

3.1. Inventory of avoided mineral fertilizer

The inventory of avoided mineral fertilizer for the three substitution principles (Table 2) reveals notable differences between the principles as regards P and K. The amount of avoided mineral N fertilizer does not vary, since MFE-N is utilized optimally with the maintenance and adjusted principles and is therefore equal to the one-to-one principle. Employing the maintenance principle reduces the avoided P mineral fertilizer to 86% of the maximum amount in the one-to-one principle and the K mineral fertilizer to 59%. The adjustment for high soil P and K levels in the adjusted principle reduces the amount of avoided mineral P fertilizer to 0% and the K mineral fertilizer to 22%, compared with the one-to-one principle. The application area required for the maintenance and adjusted principles is 0.021 ha (see Supplementary material for calculation of application area and avoided mineral fertilizers). Thus the results show that employing the adjusted principle instead of the maintenance principle represents a greater change in this case study than moving from the one-to-one principle to the maintenance principle.

Table 2
Inventory of avoided mineral fertilizers for the different substitution principles. Avoided mineral N, P, and K fertilizer as a percentage of MFE nutrients applied is given in brackets.

Substitution principle	N (kg N)	P (kg P)	K (kg K)
One-to-one	2.09 (100%)	0.72 (100%)	5.89 (100%)
Maintenance	2.09 (100%)	0.62 (86%)	3.46 (59%)
Adjusted	2.09 (100%)	0.00 (0%)	1.30 (22%)

3.2. Relative importance of avoided mineral fertilizer

The results of the LCIA for the whole cattle slurry management system show that the choice of substitution principle has a great influence on the impact in 12 out of the 18 ReCiPe impact categories included (Fig. 2). In these 12 categories, the negative (beneficial) impact of avoided mineral fertilizer dominates the combined positive impact from storage and field application, and there is a marked difference between the greatest avoided impact (one-to-one principle) and the smallest avoided impact (adjusted principle), reflecting the inventory in Table 2. In one more category, "Natural land transformation", avoided mineral fertilizer dominates storage and field application impacts, but here the impact does not vary greatly between the principles. In key impact categories within agri-food systems, such as "Climate change", "Terrestrial acidification", and "Marine eutrophication", the different substitution principles have little or no influence on the impact. In these categories, emissions and resource use associated with storage and field application dominate impacts from the avoided mineral fertilizer. The smaller positive impact for the one-to-one principle in several impact categories compared with the maintenance and adjusted principles is because the one-to-one principle excludes emissions and resource use associated with machinery for field application (operation of tractor and slurry spreader). Table A4 in the Supplementary material shows impact values for all impact categories in their respective units, as well as the separate contributions from storage and field application.

3.3. Sensitivity analysis

The amounts of avoided N, P, and K mineral fertilizer calculated for the substitution principles in four different sensitivity scenarios are compared to the results in the case study in Fig. 3. With the one-to-one principle, 100% of the MFE nutrients in the FU are assumed to replace their mineral fertilizer counterparts. Where avoided mineral fertilizer is <100%, the required amount of a nutrient is less than the applied amount, thus representing a situation of over-application. The remaining percentage to reach 100% is then the percentage of nutrient over-application. In the settings with high soil P and K levels (the "No reg" and "Nitrate" scenarios plus the case study), 0% of the applied MFE-P leads to avoided mineral P fertilizer and 100% is over-applied, while the corresponding ranges for K are 8–37% avoided mineral K fertilizer and 63–92% K over-application.

The results demonstrate three central points. First, by restricting the application rate, the necessary application area needs to increase compared with a scenario with no regulation. The fertilizer requirement increases with the application area, as does the amount of avoided mineral fertilizer, up to the point where requirement matches application. The nitrate scenario requires the largest application area, as the EU Nitrate Directive sets a stricter limitation on the cattle slurry application rate than the Norwegian manure P application limit. Second, when low soil nutrient levels instead of high levels are assumed, the fertilizer requirement according to the adjusted principle is greater than for the maintenance principle. Hence, the maintenance principle underestimates the substitution potential compared with the adjusted principle when soil nutrient levels are low, as seen in the "Case study, low" scenario.

Third, the overall context (crop production, soil characteristics, relevant regulations on application) assumed for field application of the FU affects the relative difference between the substitution principles. At one extreme is a setting where no restriction on application rate leads to FU application on a very small area, and great overestimation of avoided mineral fertilizer with the one-to-one principle compared with the maintenance and adjusted principles. At the other extreme is the "Nitrate, low" scenario, where the FU is spread on a much larger area that also requires additional fertilizer due to its low soil P and K levels. The results then show no difference between the three principles

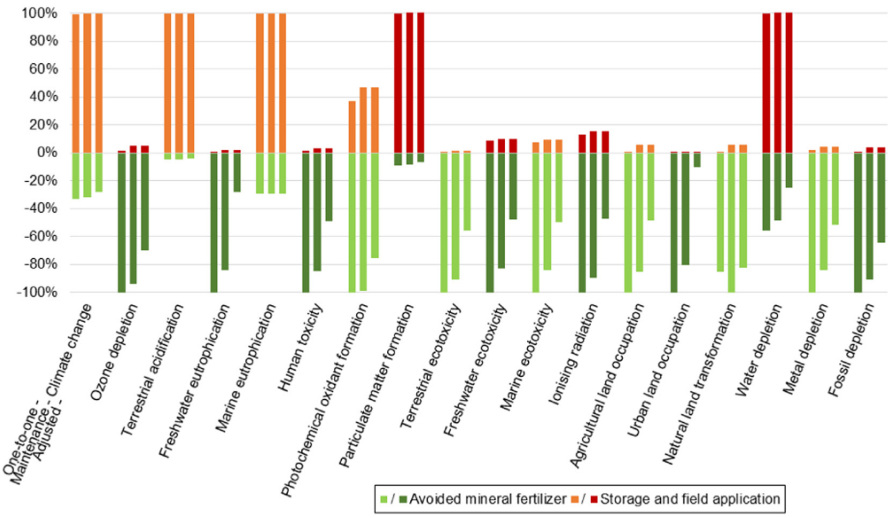

Fig. 2. Impacts from avoided mineral fertilizer compared with impacts associated with storage and field application of manure. Impacts are given as percentage of the highest absolute value across the three substitution principles for each impact category, presented from left to right: one-to-one, maintenance, and adjusted principles. Two different colors are used for each of the processes avoided mineral fertilizer and storage and field application to increase readability. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

when it comes to avoided mineral fertilizer, as all the applied plant-available nutrients are needed.

4. Discussion

4.1. Does choice of substitution principle matter?

The results from the case study demonstrate that the substitution principle used can have a great influence on the inventory of avoided mineral fertilizers and on the system-wide LCIA impacts. This resonates well with Lyng et al. (2015), who in a study on biogas value chains found that the avoided burdens associated with the co-products biogas and digestate significantly influenced the results. They reasoned that assumptions regarding substitution were of particular importance for the results. A consequence of the variability in results between the principles is that comparisons between studies are hindered if the substitution principle used is not stated explicitly and if the effect it may have on results is not considered. The negative consequences for comparability of not stating key assumptions are also discussed by e.g., Heijungs and Guinée (2007).

Our sensitivity analysis shows that the regulatory (e.g., restriction of application rate) and agronomic (e.g., soil P levels) context can affect the differences in the results obtained with the three substitution principles, and therefore the importance of choice of principle. The findings

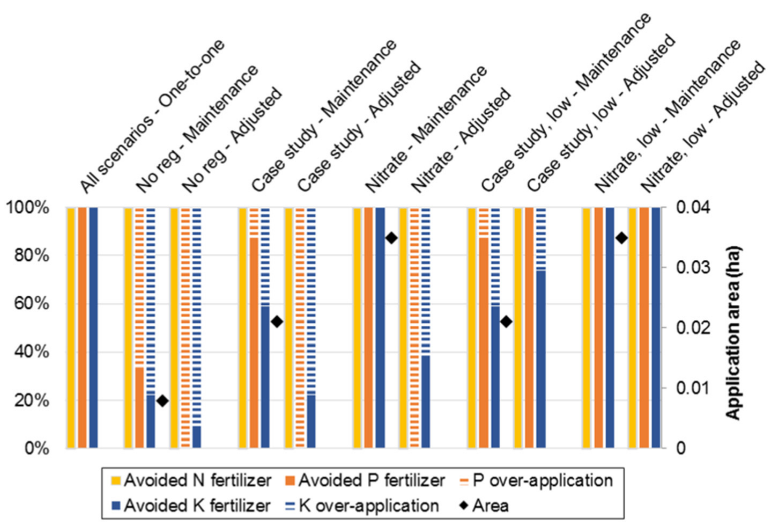

Fig. 3. Avoided N, P, and K mineral fertilizer and P and K over-application for the sensitivity scenarios and case study, given in percentages of the applied MFE nutrients for the three substitution principles. The one-to-one principle is identical across all scenarios and is only shown once. For any given scenario, the application area is the same for the maintenance and adjusted principles. The scenarios are placed in ascending order in terms of application area, first for the scenarios with high soil P and K levels, then for the low soil P and K levels.

in the current paper, based on a case on cattle slurry management, will also be relevant for LCA studies with different scopes (e.g., other organic fertilizers). As long as secondary nutrients are assumed applied to soils to replace mineral fertilizers there are assumptions to be made for determining the inventory of avoided mineral fertilizer that can affect final results. However, as case studies involving nutrient recycling differ in so many aspects, the sensitivity scenarios we use in this study should not be used uncritically as guidance and justification for a particular substitution principle.

4.2. What is the problem with the one-to-one substitution principle?

The one-to-one principle is easier and faster for the LCA practitioner to use in making the inventory rather than having to collect information on representative crop/soil characteristics and carry out calculations similar to those made in our case study. However, the one-to-one principle has shortcomings that largely derive from mismatches between the nutrient composition of the organic fertilizer and the required fertilizer nutrient composition for the crop. Organic fertilizers tend to contain more P relative to N (lower N:P ratio) compared with crop demand, leading to P over-application as application is often based on N content (Brod et al., 2015; Withers et al., 2015). The one-to-one principle therefore commonly overestimates the fertilizer value of organic fertilizers compared with the maintenance and adjusted principles. This overestimation may be further accentuated by adjusting fertilizer requirement for soil characteristics, and an overestimated substitution ratio may have important effects on the final LCA results (Laurent et al., 2014).

By using the one-to-one principle, it seems that the LCA practitioner implicitly assumes that all applied MFE nutrients will be taken up by plants in the long run and replace mineral fertilizer. Any possible surplus application of a nutrient in one year may then replace mineral fertilizer in the following year, and ultimately all the applied MFE nutrients in the FU replace mineral fertilizer nutrients at some point. This would hold if there were a feedback mechanism involved at the level of the farmer, who in the following years would adjust fertilizer application to account for this residual effect. However, this is often not the case, because of uncertainty about the residual fertilizer value after the first year (Nesme et al., 2011; Sandars et al., 2003). Even if there were a feedback mechanism, it would be inconsistent to assume adjustment for residual nutrients in the years following application of the FU while ignoring any residual nutrients from previous years.

The ability to punish nutrient over-application by making it visible in the results is provided in the maintenance and adjusted principles, but not in the one-to-one principle. Excessive inputs of N and P in food production and subsequent large losses of P and reactive N to the environment are challenging the resilience of water recipients (Carpenter and Bennett, 2011; Galloway and Cowling, 2002). Therefore, exposing nutrient over-application in the results reveals inefficient nutrient use and the potential risk of nutrient losses to the environment that this represents.

4.3. Calculation of avoided mineral fertilizer may (dis)incentivize efficient nutrient use

The way in which avoided mineral fertilizer is calculated in LCA studies may ultimately affect resource use efficiency through influencing policy. For example, in the western European context of generally high amounts of accumulated residual P (Sattari et al., 2012), the one-to-one principle gives maximum credit for secondary P application, irrespective of actual fertilizer requirements. As P over-application remains invisible, the one-to-one principle does not provide any incentive for improved nutrient use efficiency. Employing the adjusted principle in particular would give field application less credit and reward measures to utilize the nutrients more efficiently. Such measures could include changes in nutrient composition of organic fertilizers through processing, or geographical redistribution of organic fertilizers to soils that have a greater P requirement.

4.4. Limitations

The proposed principles estimate substitution based on functionality and do not constitute a more complete framework for substitution like that put forward by Vadenbo et al. (2016), where market response and user preferences for secondary products are included to determine substitutability for recycled products. By not considering displaced primary products as an effect of demand and supply dynamics, we may still be overestimating the environmental benefits of avoided production (Geyer et al., 2016). However, information on market dynamics and user preferences is rare, as also noted by Vadenbo et al. (2016), because e.g., LCAs are often performed for new processes and promising fertilizer products that may still only be used on a very limited scale. Additionally, animal manure, the most important input of fertilizer nutrients together with mineral fertilizers, is commonly applied on farmland managed by the livestock farmer, thus escaping any market dynamics. Assuming that LCAs continue to be performed without economic information on substitutability, this study shows that overestimations of substitutability may also occur at the level of functionality.

In this study we do not consider the effect of different mineral fertilizer products to be displaced by organic fertilizers. In a study on the impact of different mineral fertilizer products, Hasler et al. (2015) showed that choosing the right sources of mineral N fertilizer, in particular, can reduce cradle-to-field emissions by 20% compared with the worst alternatives.

4.5. Recommendations for future LCA studies with mineral fertilizer substitution

The three principles require different levels of information, which may not be easily available or even relevant to collect. For example, if the organic fertilizer is traded on a market where potential end-users cover a large geographical area with diverging dominant crops and soil types, it may be meaningless to gather information beyond fertilizer characteristics such as MFE values. This would justify use of the one-to-one principle. Based on availability of information and the findings in this study, we make the following recommendations for LCA practitioners:

1. State and justify the method/principle used to calculate avoided mineral fertilizer. This will increase transparency and improve comparability with other studies.
2. Use of the one-to-one principle carries the risk of overestimating the benefit of avoided mineral fertilizer. If avoided mineral fertilizer contributes substantially to, or dominates, the net impacts of the LCIA results, a sensitivity analysis should be run to see whether smaller amounts of avoided mineral fertilizer affect the conclusions.
3. In general, of the three principles, the adjusted maintenance substitution principle gives an estimate of the fertilizer requirement that is closest to reality, and therefore a better approximation of the amount of avoided mineral fertilizer. We recommend using this principle whenever possible.

5. Conclusions

This paper examines whether different assumptions used to determine avoided mineral fertilizer affect the results in LCAs that include nutrient cycling. There is currently a lack of transparency concerning the influence of such assumptions in different studies. We demonstrate through a case study that assumptions, here identified and described as three substitution principles, can have a great effect on the inventory of avoided mineral fertilizer and on impact assessment results. Employing the one-to-one substitution principle runs the risk of overestimating the benefits from nutrient cycling. In contrast, using a principle where the avoided mineral fertilizer is based on fertilizer requirement adjusted for soil nutrient levels can reveal nutrient over-application and has particular potential to incentivize improved nutrient use efficiency. In order to increase transparency and comparability between studies, our most important recommendation to LCA practitioners is simply to state and justify the principle used.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2017.09.215.

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