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Research article

Swedish food system transformations: Rethinking biogas transport logistics to adapt to localized agriculture

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A B S T R A C T

Ensuring future food and energy security will require large changes in consumption and production patterns, including enhanced animal and human excreta recycling. Although these shifts are considered in many scenario studies, their implications on the logistical requirements for effective recycling are rarely analysed. Here we translated two existing stakeholder co-designed food system scenarios for Sweden to 5 × 5 km resolution maps of animals, crops, and humans. We used optimization modelling to identify biogas plant locations to minimize transport costs and maximize nutrient reuse. We then compared scenarios, including full recycling under current landscape configuration, through Life Cycle Assessment. The reduction in meat consumption and imported food in both co-designed scenarios, by definition, led to less nutrients available in manure for recycling back on cropland, and less material available for digestion. Less excreta meant lower national benefits, for example 50% less greenhouse gas emissions savings in the most divergent scenario. However on a per transport basis the benefits of recycling were more important: recycling remained a net financial benefit even if transport costs were to increase. Although fewer biogas plant locations were necessary (184 and 228 for alternative futures, vs 236 under current conditions) to process human and animal excreta, the regional clustering of locations did not change substantially across scenarios. Regions such as Skåne and Västra Götaland consistently required the most biogas plant locations across scenarios. Focusing early construction investments in these regions would be resilient to a large array of food system futures. Our spatially-explicit open access scenario maps can be used to explore logistics for such planning, and explore the impact of landscape configuration on other sustainability priority areas.

1. Introduction

Current food systems are largely unsustainable and require transformation (Foley et al., 2011). There is a growing body of work imagining what such sustainable futures might look like when multiple behavioral and technological changes are combined (Tälle et al., 2019; Willett et al., 2019). Implementing even just one intervention can have such systemic effects that it must be considered in the planning of multiple sustainability spheres. For instance, the climate crisis not only requires agriculture to reduce greenhouse gas emissions (GHG) and adapt to a rapidly changing set of growing conditions, but also contemplate its role in producing bioenergy without disrupting food supply (Scherer and Verburg, 2017). Meat consumption has a very large impact on agricultural greenhouse gas emissions and land requirements (Poore and Nemecek, 2018; Röös et al., 2018). However decreasing animal product consumption also alters options for nutrient recycling as it decreases the amount of manure available for reuse on fields. A systemic approach to planning for a more sustainable food system is thus required to minimize trade-offs and unintended consequences.

A circular economy for organic waste is an imperative part of sustainable food systems because no biological (and thus agricultural) system can be waste/by-product free; that is, although there is ample room and need for waste reduction, humans and ani-

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mals will always excrete and some harvested crop material will not be edible (Jurgilevich et al., 2016). Further, these wastes contain essential plant nutrients such as nitrogen (N), phosphorus (P) and potassium (K) as well as energy. Hence, the safe and effective recycling of these wastes is a vital part of decreasing food systems' dependence on 'fossil' resources and environmental pollution (Schulte-Uebbing and de Vries, 2021; Steffen et al., 2015; Withers et al., 2020).

Current agricultural production relies on the use of N and P fertilizers produced from non-renewable sources and through energy-intensive processes (Dawson and Hilton, 2011). Their widespread use has allowed for a spatial disconnect between rural areas (where food is produced) and urban areas (where food is consumed), and between livestock intensive areas and areas of intensive feed production nationally and globally (Fridman et al., 2021; Jones et al., 2013; Nesme et al., 2018). Valuable nutrients that could be used for food production are often lost from agricultural and urban waste management systems to freshwater and marine environments where they contribute to eutrophication (Glibert, 2020; Ibisch et al., 2016). Unbalanced fertilization moreover promotes the systematic stripping of nutrients other than N and P from soils (Jones et al., 2013). The need to mitigate eutrophication and secure future fertilizer availability have intensified the search for ways to reduce nutrient losses and increase nutrient recycling (Cordell and White, 2014; Galloway et al., 2014) while climate change has made finding alternative energy sources simultaneously an essential imperative.

Although the need for increased effective (human and animal) excreta recycling is clear, how to achieve it in a rapidly changing world remains difficult. The multiple food system changes that are required, and desired, imply that when planning for tomorrow's recycling infrastructure we should not solely rely on today's estimates of waste supply or demand for nutrients. There are diverse efforts to develop predictions of future agricultural land use, accounting for economic and climate drivers, but their resolution is too low to make decisions on specific technologies or logistical requirements (Busch, 2006; Verburg et al., 2019). These spatially explicit representations of the future often focus on only one change variable at a time (e.g., increased mean temperature or flood risk). A majority of these representations of the future are forecasts given current trends (e.g., GHG) and not scenarios developed through other means (e.g., future visioning of desirable futures). Without accounting for a more diverse range of land use changes, which emerge from a patchwork of planned scenarios and reactive adaptations, communities are less likely to foster the transformations they need (Bennett et al., 2021; Lemp et al., 2008).

There are few assessments of how changing food consumption and production patterns will alter how society must spatially plan resource reuse infrastructure, and as such a circular economy for energy and nutrients related to food systems (Tälle et al., 2019). This is surprising as transportation costs are often cited as a major barrier to organic waste reuse (Akram et al., 2019b; Sharpley et al., 2016; Zhang et al., 2021). Actors (e.g., biogas plant operators) that currently use organic waste as feedstock go to considerable lengths to minimize costs associated with acquiring this feedstock and reaching markets with products or residues (Ammenberg and Feiz, 2017; Long et al., 2018). Previous work also shows that coordination among multiple actors, at multiple spatial and organizational scales, is required to minimize these transport costs and maximize benefits (Koppelmäki et al., 2021; Metson et al., 2020). Costs are a function of distance, weight, handling, and actual purchasing cost versus market value of the final product; finding ways to minimize these parameters requires spatially explicit information (Akram et al., 2019b; Hamelin et al., 2019). In other words, there is not only a gap between our current food system and a sustainable one, there is also a gap in the tools required to plan how to get there without missing important interactions among required changes.

The overall objective of this work is to uncover how logistical requirements of excreta recycling vary under current and selected sustainable Swedish food system parameters. We do so by spatializing two visions of Sweden's food system (Karlsson et al., 2018; 2017) and applying an optimization framework to excreta recycling. Our approach looks at minimizing transport costs while also maximizing N, P, K, and energy recycling through biogas plants (Metson et al., 2020). We compare these two alternative scenarios to how full recycling would look like under current conditions (circa 2015). We ask:

1. Where are the supply and demand areas for the nutrients and energy in organic waste located under stakeholder preferred scenarios?
2. How can these supply and demand areas be connected to meet crop nutrient demands and produce biogas?
3. What are the financial and environmental impacts of supporting this optimized nutrient and energy circular economy?

2. Methods

In order to answer our research questions we use four distinct methodological approaches (Fig. 1) in order to:

A. spatialize scenarios,
B. convert to nutrient supply and demand,
C. optimize transport to and from biogas plants, and finally
D. estimate cost and environmental impacts through Life Cycle Assessment.

In approach A we developed algorithms in Python which fulfilled explicit objectives and constraints in two future scenarios (see Section 2.1) while accounting for where people, animals, and crops are currently located. In approach B, data are further combined to create 5×5 km grids (25 km^2 cells) of nutrient supply and demand which can be used the optimization model (approach C). All data and code (from a 25 km^2 abstraction level), including detailed methods documentation, are available on Gitlab (https://gitlab.liu.se/future-land-use/public-map-automation). Approaches B to D match those outlined in Metson et al. (2020) and are briefly explained in the following sections, focusing on updates and modifications.

We summarize results in terms of gridded spatial patterns nationally and then selected values at the regional level in order to better highlight differences among the three scenarios.

2.1. Scenario specifications

There are many approaches to creating future scenarios. In this paper we use two stakeholder co-created future visions of what a Nordic food system should look like in 2030 (Karlsson et al., 2018; 2017). Both forecasts of what future food systems will/could look like, and visions of what the future should like, are useful tools for planning (Iwaniec et al., 2020; Lemp et al., 2008). We have opted to focus on desirable and co-created visions as a way to 1) acknowledge that current systems and trajectories in our food systems are not sustainable (Campbell et al., 2017; Foley et al., 2011), 2) society must work towards what diverse groups of people want, and 3) co-created visions are more salient and useful for change than those created only by academics (Mitchell et al., 2014). For instance, working with stakeholders, on the basis of existing initiatives, scenario work about sustainable food systems in the Stockholm city-region were better able to identify trade-offs amongst priorities (Sellberg et al., 2020). Even if said scenarios may be very

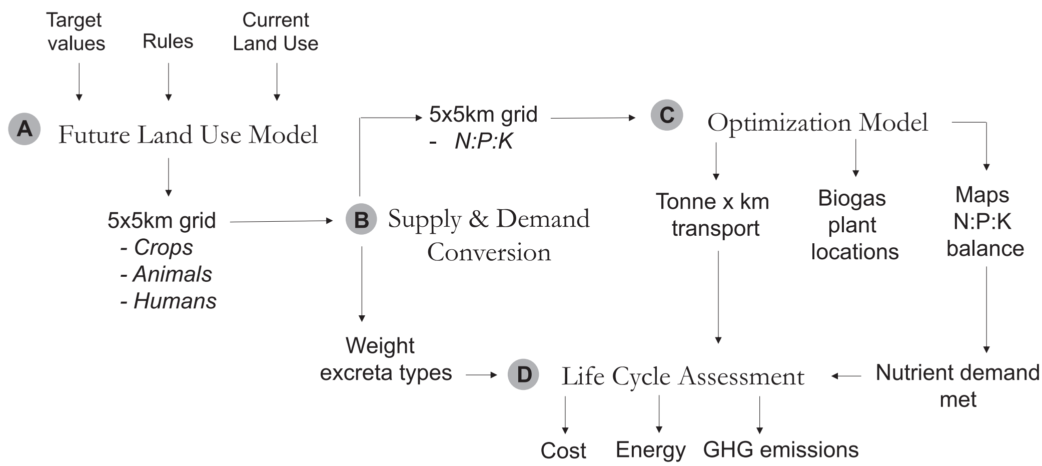

Fig. 1. Main methodological steps overview. Arrows between steps with accompanying text indicate the inputs and outputs of our models and how they interact with one another. Approaches A and B are further detailed in Fig. 4. N:P:K stands for nitrogen, phosphorus, potassium, km stands for kilometer, and GHG stands for greenhouse gas.

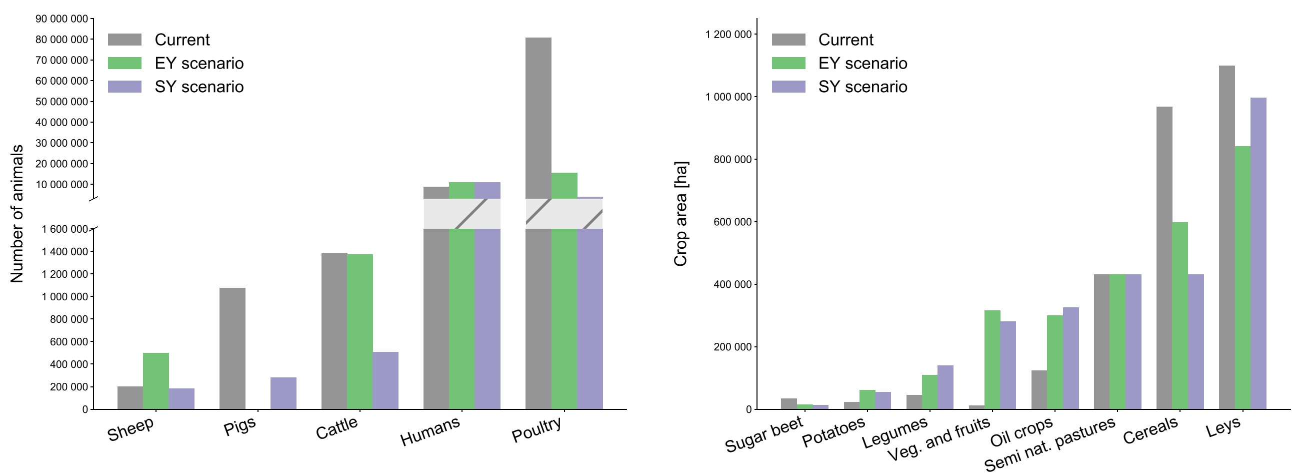

Fig. 2. National number of animals (left panel) and area of crops (right panel) under current (grey) and future scenarios (green as the Efficiency (EY) and purple as the Sufficiency (SY)). The total area under agricultural production expressed in hectares (ha) is the same in all scenarios but the area of different crop categories changes. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

different from current conditions, it is still necessary to overcome barriers that already exist today to enable sustainable transitions (Kabbe, 2018; Sharpley et al., 2016). As such, analyzing these scenarios with regards to transport and resource use logistics is highly relevant.

Karlsson et al. (2018), (2017) scenario creation focused on mitigating climate change and other sustainability issues through better resource use, production, and consumption patterns across Nordic country food systems. Co-developed normative statements about the future were translated to tangible criteria to create numerical scenarios using a mass flow model. In both scenarios key features included:

1. the reduction of animal protein consumption,
2. the preservation of semi-natural pasture lands (i.e., enough grazing animals to support those lands),
3. the production of local food as opposed to importing what can be grown in the country,
4. the use of production methods that are organic/ecological,
5. and the decrease of waste production while increasing reuse of resources.

These features mean that the scenarios are drastically different to current agricultural production in Sweden; they have much fewer animals on the landscape, and a substantial increase in fruit and vegetable production (Fig. 2, Table 1). The two scenarios also differ from one another. The Sufficiency (SY) scenario is the more

Table 1
Total number of animals and hectares of crops for main categories to be placed in Sweden according to three scenarios. The current scenario represents current values circa 2015 data. The Efficiency (EY) and Sufficiency (SY) scenarios for 2030 have been re-scaled from their original source to match the assumption of no new agricultural land conversion and the conservation of all semi-natural pastures.

Scenario	Current	EY	SY
Number of animals per category			
Sheep	200 718	499 261	185 163
Pigs	1 07 5633	0	284 000
Cattle	1 383 287	1 319 699	488 950
Chickens	80 866 633	15 638 000	3 920 000
Humans	8 849 982	11 094 861	11 094 861
Hectares of land per category			
Sugar beet	34 572	15 223	13 523
Potatoes	22 541	62 331	55 369
Legumes	44 973	109 445	140 702
Vegetables and fruits	12 770	315 895	280 607
Oil crops	123 674	302 007	325 748
Semi-natural pastures	431 045	431 045	431 045
Cereals	968 512	598 042	431 435
Ley and pastures in rotation	1 099 958	841 812	996 759

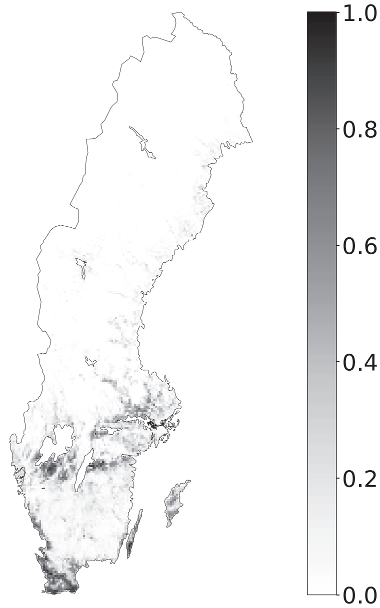

Fig. 3. Agricultural land intensity circa 2015 in Sweden. Darker colors represent a larger area of the 25km^2 cell is made-up of agricultural land. Black (1) would be 100% of the cell made-up of agriculture and white (0) would be none.

restrictive of the two; it only allows animals to be fed grass and by-products, focusing the rest of agricultural land production on crops that can be consumed by people. The Efficiency (EY) scenario relaxes the animal feed constraint and allows for more leys in organic rotation to be fed to animals and some feed production on arable land. EY thus includes more animals than SY and has a different make-up of crops on the landscape. The scenarios, as described Karlsson et al. (2017), do not give any indications on *where* crops, animals, and humans should/could be placed, other than not expanding production (Fig. 3); this is what we will do here.

In order to put the SY and EY scenarios into perspective, we compare them to a said Current scenario which is derived from Swedish 2013–2016 data. We consider Current a scenario, as we model how and where biogas plants could be used to move excreta to meet crop demands (called Biogas Generic scenario in Metson et al. (2020), and see Supplemental Information (SI) for more details on necessary adaptations to the EY and SY scenarios to match assumptions to datasets).

2.2. Land use and nutrient map creation

We converted the EY and SY scenarios to gridded spatial datasets; this process is split into Crops, Animals and Humans in the following subsections. Subsequent to allocating an amount of crops, animals, and people to each grid cell, we further converted these amounts to nutrient supply and demand and conducted a sensitivity analysis. An overview of how processes link together is presented in Fig. 4.

2.2.1. Crops

To allocate future crops to specific grid cells, we combined rules derived from Karlsson et al. (2017) with information on which types and amounts of crops were grown where in the mid 2010's. The first step was to match Jordbruksverket (JBV)—the Swedish Board of Agriculture—crop codes (JBV, 2021) to the crop categories and sub-categories used in the scenarios. For each 25 km^2 cell, we could then determine how much of each crop category was grown in the Current scenario (JBV, 2013). We assumed that when possible, locating future crop categories to where they were currently grown accounted for some climate, machinery, and knowledge requirements for crop types. Still, this assumption was balanced with the objective to increase crop diversity, as well as the following specific EY and SY constraints. These were:

1. No oil crops (rape seed) or legumes are placed above 63 degrees latitude.
2. Oil crops cannot make up more than 17% of the rotating crops (oil crops, cereals, legumes, leys, potatoes, sugar beet, and fallow) in a cell.
3. Legumes can not make up more than 10% of the rotating crops in a cell.
4. Leys must have at least 33% of the total area of rotating crops in a cell.

To do the cell allocations, a combination of placer and allocator algorithms were used. Placers give a visitation order to all the cells in Sweden for each crop category. The more of a crop category a cell has in the Current scenario, or the closer a cell is to a cell with the category, the higher it's visitation priority. Allocators determine what to do in a cell when it is visited (e.g., decrease the area of crop category x by y %). The placers and allocators were first applied to crops that needed to decrease to create free area. This free area was subsequently used by applying a different set of allocators and placers to convert it to the crop categories that increased.

The crop allocation model was run before the animal allocation model, with the exception of a few sub-categories. The placement of animals was dependent on the location of pasture lands and as such needed to be run first. However, many JBV crop codes could not be allocated to sub-categories for cereals, legumes, leys. To circumvent this issue, we specified the sub-category types after animals were placed, allocating animal feed and pasture sub-categories close to where animals were placed.

2.2.2. Animals

A combination of per animal feed requirements (type and amount of different crops in hectares) and current animal locations (JBV, 2016) were used to distribute animals on the landscape. The placer visited cells based on the availability of feed, visiting cells with more feed first. If two cells had the same amount of feed available the number of current animals in a cell was used as a tiebreaker. Unlike the crop allocation process, all existing animals were removed before placing new animals.

After matching JBV animal types to the scenario animal categories and sub-categories, animals in each category were placed in lockstep (each getting one pass over Sweden before moving on to the next, and starting over until target values were reached). The lockstep order placed those categories with larger area needs and most complex rules for placement first. The amount of pasture space needed for each animal was calculated by dividing the total area of semi-natural pasture and rotating pastures by the number of animals in each scenario weighted for the relative importance of these pasture types in each animal sub-category's annual dry-matter intake according to Karlsson et al. (2017). Cows and sheep needed access to both semi-natural pasture and rotating pasture and were placed first and second. Poultry, which did not have pasture as part of their feed composition, were placed based on where current farms were located and placing a cap of 3 000 birds per farm (SVA, 2020), and placed third. Pigs only needed access to rotating pasture and thus were placed fourth.

2.2.3. Humans

Allocating humans to the landscape was done based on 2030 municipal population projects (SCB, 2020) and an allocator and

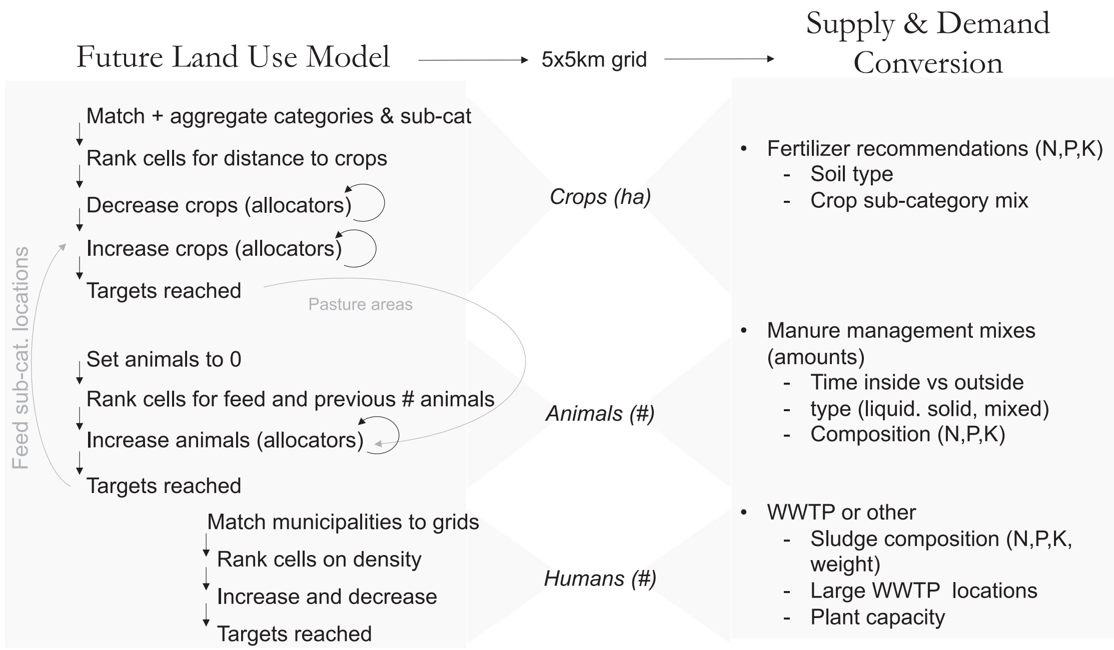

Fig. 4. Major steps in spacializing scenarios and converting them to nitrogen (N), phosphorus (P), potassium (K), and weights for supply and demand in each cell. Sub-cat stands for sub-category, ha stands for hectares, # stands for number of animals or humans, and WWTP stands for wastewater treatement plant.

placers which accounted for existing population densities. Cells were assigned to a municipality and, like with crops, municipalities could fall into an increase or a decrease category (which use different allocators). As was done with crops and animals, increases happened first in areas with more people, but were capped at around a 20% increase in populated cell (also accounting for the amount of non-agricultural land in the cell). Decreases were done along the same principle.

2.2.4. Nutrients in cells

Each cell, depending on the amount and type of crops, animals, and humans, was assigned a N:P:K demand and supply based on methods and data sources described in Metson et al. (2020). A few key distinctions must be made however:

Nutrient demands for each crop sub-category were based on a weighted average of current crop composition. Semi-natural pastures, as well as the pastures in rotation sub-category for leys, were assumed to have a net zero nutrient demand because of grazing animals. The assumption was slightly different for the Current scenario to match Metson et al. (2020), where a cell's crop nutrient demand was decreased (to a minimum of 0) with the amount of nutrients excreted by grazed animals (given the length of time they spend outside). For each cell, the sub-category crop nutrient demands were added together to get a total nutrient demand which accounted for local soil conditions.

The N:P:K supply in animal manure was calculated by multiplying the number of animals in a sub-category by relevant excretion factors (see GitLab). The manure from cows, sheep, and pigs was assumed to only be collectable for the period of time when animals were not out grazing. For humans, the same approach as our previous work was used.

2.2.5. Sensitivity analysis

With a number of intertwining assumptions and rules affecting our spatial conversion of SY and EY scenarios, it was important to test the global sensitivity of our model. We used a variance-based sensitivity analysis with a modified Sobol sequence developed by Saltelli et al. (2010) for input parameter set generation. The use of a global sensitivity analysis allowed us to explore the range of values possible for a selected output variable as well as the relative importance of parameters to said output variable results.

Importantly, we selected parameters (and their ranges) where SY scenario target values could be met. This ensured that the model was solvable and that model runs were comparable to one-another (see documentation online via GitLab for further explanation and results).

2.3. Optimization

All three scenarios make use of the same optimization model, and this model involves the localization and sizing of biogas plants as well as the transportation flows to and from biogas plants. For WWTP plants, only flows from the plants are considered. The optimization model can be seen as a combination of the transportation problem and the facility location problem, but with additional constraints concerning nutrient contents and biogas production technicalities.

The model simultaneously tries to minimize the total cost of transportation (of feedstock) to biogas plants and the cost of transportation (of digestate) from biogas plants to croplands, without over-applying nutrients and respecting a fixed biogas plant capacity. An overview of the model is given in Fig. 5, which also shows how parameters and variables are connected.

2.3.1. Notation

In Table 2 we specify all sets, parameters and variables used in the optimization model.

2.3.2. Mathematical model

$$\min \quad \sum_{i\in S}\sum_{k\in L}\sum_{m\in M}\alpha_{ik}x_{ik}^{m} + \sum_{k\in L}\sum_{j\in D}\beta_{kj}y_{kj} + \sum_{h\in H}\sum_{j\in D}\gamma_{hj}w_{hj} \tag{1}$$

$$\text{s.t.} \quad \sum_{k\in L}x_{ik}^{m} = s_i^{m} \qquad i\in S,\ m\in M \tag{2}$$

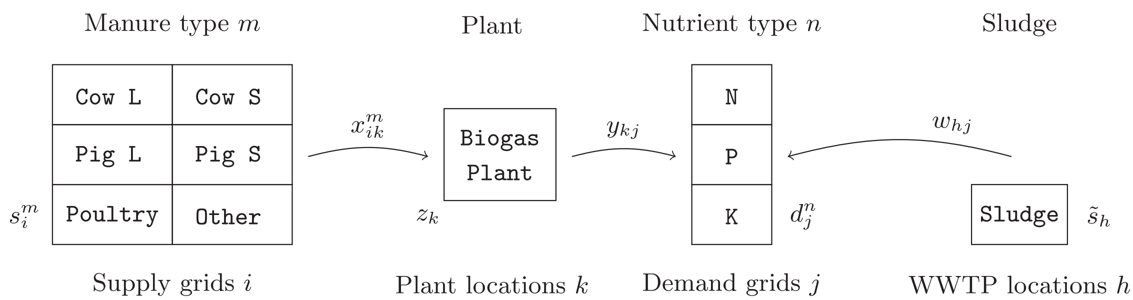

Fig. 5. Overview of the optimization model. Manure categories (expressed as six types, including solid (S) and liquid (L) when relevant) are sent to biogas plants and then back to the fields to satisfy nutrient needs expressed as nitrogen (N), phosphorus (P), and potassium (K). Sludge from wastewater treatment plants (WWTP) are also used to satisfy the nutrient needs.

Table 2
Sets, parameters and variables used in the optimization model. Nutrients nitrogen (N), phosphorus (P) and potassium (K), and six different categories of manure (which come in liquid (L) and solid (S) forms), are considered.

Set	Description	Indices
S	set of supply grids	$i \in \{1, \dots, N_S\}$
D	set of demand grids	$j \in \{1, \dots, N_D\}$
L	set of biogas plant locations	$k \in \{1, \dots, N_L\}$
H	set of sludge plant locations	$h \in \{1, \dots, N_H\}$
N	set of nutrients	$n \in$ { N, P, K}
M	set of manure types	$m \in$ { Cow L, Cow S, Pig L, Pig S, Poultry, Other}
Parameter	**Description**	
α_{ik}, β_{kj}, γ_{hj}	unit transportation cost for manure (per tonne and km), from a supply grid i to plant locations k, from plant locations k to demand grids j, and from sludge locations h to demand grids j, respectively	
s_i^m, $\tilde{s}_h$	supply (in tonnes) of manure type m at supply grid i, and human sludge at location h, respectively	
d_j^n	demand (in tonnes) of nutrient n at demand grid j	
a^n	relaxation factor for nutrient n	
c_i^{mn}, $\tilde{c}_h^n$	concentration of nutrient n in manure type m at supply grid i, and concentration of nutrient n in sludge at location h, respectively	
$\bar{c}_k^n$,	average concentration of nutrient n in manure at location k	
b_{LB}^m, b_{UB}^m	lower and upper bounds, respectively, on the amount of manure of type m allowed at a biogas plant	
Γ	maximum number of biogas plants to build	
$\underline{C}$, $\bar{C}$	biogas plant min and max capacities, respectively	
Variable	**Description**	
x_{ik}^m	amount of manure (tonnes) of type m transported from grid i to plant at location k	
y_{kj}	amount of manure (tonnes) transported from plant at location k to grid j	
w_{hj}	amount of sludge (tonnes) transported from location h to grid j	
z_k	number of plants (integer) built at location k	

$$\sum_{i\in S}\sum_{m\in M} x_{ik}^m \le \bar{C} z_k \qquad k \in L \qquad (3)$$

$$\sum_{i\in S}\sum_{m\in M} x_{ik}^m \ge \underline{C} z_k \qquad k \in L \qquad (4)$$

$$b_{\mathrm{LB}}^m \cdot \sum_{i\in S}\sum_{\tilde{m}\in M} x_{ik}^{\tilde{m}t} \le \sum_{i\in S} x_{ik}^m \qquad k \in L,\ m \in M \qquad (5)$$

$$b_{\mathrm{UB}}^m \cdot \sum_{i\in S}\sum_{\tilde{m}\in M} x_{ik}^{\tilde{m}} \ge \sum_{i\in S} x_{ik}^m \qquad k \in L,\ m \in M \qquad (6)$$

$$\sum_{k\in L} z_k \le \Gamma \qquad (7)$$

$$\sum_{i\in S}\sum_{m\in M} x_{ik}^m = \sum_{j\in D} y_{kj} \qquad k \in L \qquad (8)$$

$$\sum_{j\in D} w_{hj} = \tilde{s}_h \qquad h \in H \qquad (9)$$

$$\sum_{k\in L} \bar{c}_k^n y_{kj} + \sum_{h\in H} \tilde{c}_h^n w_{hj} \le a^n d_j^n \qquad j \in D,\ n \in N \qquad (10)$$

$$x_{ik}^m \ge 0 \qquad i \in S, k \in L,\ m \in M \qquad (11)$$

$$y_{kj} \ge 0 \qquad k \in L,\ j \in D \qquad (12)$$

$$w_{hj} \ge 0 \qquad h \in H,\ j \in D \qquad (13)$$

$$z_k \ge 0,\ \text{integer} \qquad k \in L \qquad (14)$$

The objective function (1) is to minimize the total transportation costs, where α_{ik}, β_{kj}, and γ_{hj} are the unit transportation costs (per tonne and km) for manure sent from supply grid i to biogas plant location k, for manure sent from biogas plant location k to demand grid j, and for sludge sent from WWTP plant location h to demand grid j, respectively. Constraint (2) makes sure that all manure is sent to a biogas plant. Constraint (3) is to ensure that the biogas plant capacity (upper bound) is respected, given by parameter $\bar{C}$, and also makes sure that no manure is sent to

Table 3
Life cycle inventory assumptions used to model biogas production from excreta. Values are largely based on our modeling approach in Metson et al. (2020).

	Anaerobic digestion	Ugrading and compression
Electricity	36[a] MJ/t	1.6[b] MJ/Nm^3 biogas
Heat, own biogas	85[a] MJ/t	0 (water scrubber)
Iron chloride	5 kg/t	0
Biogas flared	1% of production[c]	0
Biogas slippage	1% of production[d]	1%
Methane yields[e]		
- Cow manure (liquid)	213[f] Nm^3 CH_4/t VS	
- Cow manure (solid)	200[f] Nm^3 CH_4/t VS	
- Pig manure (liquid)	268[f] Nm^3 CH_4/t VS	
- Pig manure (solid)	259[f,g] Nm^3 CH_4/t VS	
- Poultry manure	278[f] Nm^3 CH_4/t VS	
- Other manure	250[f] Nm^3 CH_4/t VS	
- Sludge	300[h] Nm^3 CH_4/t VS	

[a] 33–80 (el.) and 70–250 (heat) in Berglund and Börjesson (2003), (2006); Lantz et al. (2009) We have assumed large-scale modern plants which tend to have better performance.
[b] Bauer et al. (2013).
[c] Own biogas is used for heating, therefore we have assumed low flaring rate.
[d] Avfall Sverige (2016).
[e] We applied 90% conversion efficiency.
[f] Jordbruksverket (2014).
[g] Carlsson and Uldal (2009).
[h] 300 (315–400) in Bachmann (2015); Hellstedt et al. (2010).

a biogas plant location which is not used. Constraint (4) is similar to constraint (3), to be used if there is a lower bound on the capacity (given by parameter $\underline{C}$) at a biogas plant. (For example, one might want to avoid using less than 10% of a plant's capacity.) Constraint (5) make sure that a biogas plant get at least b^m_{LB} percentages of manure type m. Similarly, constraint (6) make sure that a biogas plant get at most b^m_{UB} percentages of manure type m.

Further, constraint (7) limits the number of biogas plants that can be built (specified by parameter Γ). Constraint (8) is a balance constraint, ensuring that all manure that arrive also leave each biogas plant. Constraint (9) makes sure that all sludge is sent to demand grids. Constraint (10) ensures that the demand of each nutrient n, at each demand grid j, is respected. Parameter a^n allows to adjust (tighten or loosen) this demand, and the value is set to 1 for N (where demand is higher than supply), 1.25 for P, and 3 for K, where demand is lower than supply. Here, $\bar{c}^n_k$ are average values for nutrient concentration over all supply grids; this is necessary in order to keep the model linear. Constraints (11)–(13) define continuous transportation variables; x^m_{ik} (tonnes of manure of type m sent from grid i to plant at location k), y_{kj} (tonnes of manure sent from plant at location k to grid j), and w_{hj} (tonnes of sludge sent from location h to grid j), respectively. Finally, constraint (14) defines the integer variables z_k (the number of biogas plants built at location k).

2.3.3. Nutrient concentrations at the biogas plants

One issue with the optimization model is that average values for nutrient concentration are used in order to keep all constraints linear. As the nutrient concentration of the manure sent to each plant is not exactly equal to the average values, the acquired solution is not feasible with respect to the upper bounds of N:P:K demands.

But, with a solution at hand, it is straightforward to calculate the concentration levels of each nutrient $n \in N$ for the manure sent to each biogas plant at location k.

$$\hat{c}^n_k = \left(\sum_{i\in S}\sum_{m\in M} c^{mn}_i x^m_{ik}\right) \Big/ \left(\sum_{i\in S}\sum_{m\in M} x^m_{ik}\right) \qquad k \in L,\ n \in N \qquad (*)$$

To handle this issue, without considering a non-linear model, one can adjust the acquired solution somewhat. First solve the optimization problem using average nutrient concentration values, which yield a feasible solution with respect to transportation of manure *to* each plant, as well as plant locations and sizes. (This corresponds to variables x and z.) Second, fix the variable values of x and z and calculate actual concentration levels of each nutrient at each plant using equation $(*)$, and solve the problem again to get feasible transportation of manure *from* each plant (variables y).

2.3.4. Model outputs

Outputs from the optimization model consist of the locations of biogas plants together with a transportation plan which specifies where (to which plant) each supply grid's amount and type of manure (as feedstock) should be sent. The transportation plan also specifies which crop demand grid the excreta from each biogas plant (as digestate) should be sent to. The outputs can then be summarized at different levels. For our purposes we looked at differences among scenarios at the national level for total nutrient demand met and transportation costs. We also looked at cell specific values and aggregate outcomes at the regional level, including the number of biogas plants and the proportion of transports that come from outside of a political boundary. These political boundary aggregations can be particularly helpful in identifying areas that are more variable, or particularly robust, among scenarios.

2.4. Life cycle assessment model

While the optimization results provide locations for biogas plants based on minimized transportation they do not directly estimate the environmental and financial implications of the studied land-use scenarios. Therefore we integrated the results of the optimization with Life Cycle Assessment (LCA) models.

Following the ISO standard for LCA (ISO, 2006a; 2006b) we define the goal, functional unit, and scope of the study. Our goal was to complement the optimization results with an indication of the environmental performance of excreta management through biogas and biofertilizer production under the studied scenarios. Through an attributional approach and using a spreadsheet-based tool which was developed in our previous studies (Feiz et al., 2020; Metson et al., 2020), this could provide a simplified indication of how relevant physical flows—such as transportation or anaerobic digestion—can affect the overall environmental performance of the scenarios (Ekvall, 2019; Finnveden and Potting, 2014).

The functional unit was set as the treatment of the annual amount of excreta generated in Sweden with energy and nutrient recovery via anaerobic digestion (i.e. biogas and biofertilizer production). The amounts of excreta in different scenarios are different, but the treatment of available excreta was assumed to be functionally equivalent across different scenarios. The system boundaries include transportation of excreta to biogas plants production of biogas and digestate (as biofertilizer), upgrading of biogas, transportation of biogas to market and its utilization as a substitute for diesel fuel for transportation, transportation of biofertilizer to farms storage of biofertilizer and finally its land application and utilization as a substitute to mineral fertilizers (Fig. 6). The contribution of the following processes are presented in the LCA results:

1. *Transportation* of excreta and digestate as biofertilizer.
2. *Digestate handling*, including storage and land application (spreading).
3. *Biogas production and distribution*, including related processes such as upgrading.
4. *Substitution of mineral fertilizers*, based on the nutrient content of the spread biofertilizer on farm land.

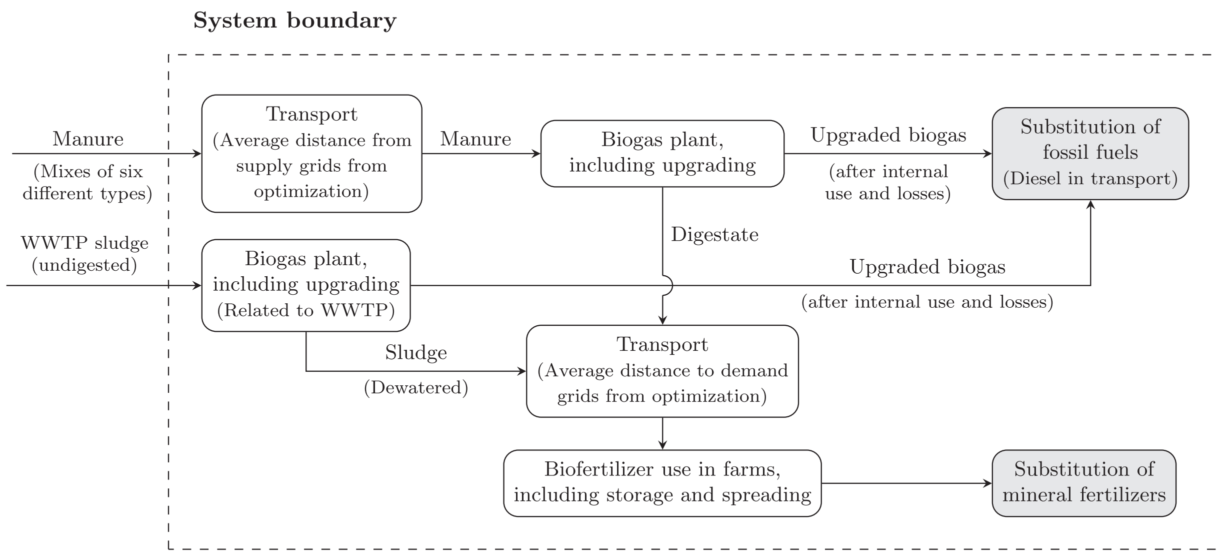

Fig. 6. System overview of the life cycle assessment (LCA) model for the studied scenarios. The included activities are transport, biogas and upgrading plants, storage and spreading of digestate as biofertilizer, use of upgraded biogas as a substitute for diesel fuel. Wastewater treatment plant (WWTP), except biogas production, are assumed to be outside the life cycle system boundary. Life cycle flows such as use of electricity or fuel use in processes are included, but are not shown in the diagram.

5. *Substitution of fossil fuels*, based on the energy content of the upgraded biogas.
6. *Net*, representing sum of all processes.

We compared the scenarios from the perspective of climate change (GWP_{100} model by IPCC, 2013), primary energy use, and cost (operating cost and sales) of transportation, energy, materials, and products. We focused on climate impact due to its relevance to the sustainability discourse and political goals in Sweden (Swedish EPA, 2018). We also included primary energy use which we believe is an important category associated with energy-related studies (Arvidsson and Svanström, 2016). Although cost analysis is not a common part of LCA, we considered it to be a relevant issue to be included. Our cost analysis mainly includes the running costs (considering the activities included in Fig. 6) and does not include investment, governmental support, and administration costs. Therefore, the results are indication of running costs and incomes insofar as they are associated with the cost of transportation and the used resources as well as sales of biogas and biofertilizer.

Most of the underlying assumptions about the life cycle impact of excreta management through biogas production were similar to Metson et al. (2020), with a few updates, mainly regarding transportation costs to better account for differences between long and short distances (Table 6). The underlying assumptions for biogas production were assumed to be the same for all plants and scalable according to the amount of manure received. We constructed “Model Biogas Plants” (average plants) which used average values concerning manure amount and mixture, inbound distance from the supply cells to the biogas plants, outbound distance from the biogas plant to demand cells, and the allowed N:P:K application rates to avoid overfertilization. A similar approach for the sludge was used. After modeling the life cycle impact of each “Model Biogas Plant” we multiplied it by the total number of plants to get the total impact of excreta management in each scenario (Fig. 9 and Table SI-3). Since the total amount of excreta is not the same in the Current, EY, and SY we also provide the LCA results per tonne excreta treated (Figure SI-19).

Since the underlying assumptions for all these model biogas plants are similar we have decided to show one inventory example (Table SI-1) which is based on the Current scenario (model biogas plant for manure). The inventory is created using key assumptions regarding biogas production (Table 3), transportation activities (Table 6), emission factors from storage and spreading of digestate (Table 7), and how the produced digestate and biogas can substitute mineral NPK fertilizers and diesel fuel (Table 4). Estimated values (inputs, outputs) in the inventory are then multiplied by their corresponding life cycle impact (Table 5) or relevant im-

Table 4
Values used to calculate the avoided impacts due to the substitution of mineral fertilizers by digestate, and diesel fuel by biomethane in the LCA model. “GWP”, “PE”, and “Cost” stand for climate impact, primary energy use, and resource cost, respectively.

Item	GWP (kg CO_2eq)	PE (MJeq)	Cost (€)
Mineral N fertilizer (1 kg)	6.7[a]	48[a]	0.9[b]
Mineral P fertilizer (1 kg)	3.2[a]	19[a]	1.7[b]
Mineral K fertilizer (1 kg)	0.9[c]	19[c]	0.6[b]
Diesel fuel (1 MJ)	0.091[a]	1.2[a]	0.03[a]

[a] Börjesson et al. (2010)
[b] Calculated based on Yara N27, P20 and K50 products (Yara, 2018).
[c] Ecoinvent LCA database 3

Table 5
Life cycle impacts used for some of the main inputs to the LCA model. “GWP”, “PE”, and “Cost” stand for climate impact, primary energy use, and resource cost, respectively.

	GWP (grCO_2eq)	PE (MJ)	Cost (€ cent)
Electricity, Swedish mix (1 MJ)	13.1 [a]	1.7 [c]	1.8 [c]
Diesel fuel (1 MJ)	90.5 [b]	1.2 [b]	3.8 [c]
Iron chloride (1 kg)	47 [d]	4.3 [e]	27 [f]
Fresh water for industrial use (1 m3)	121 [g]	0.1 [g]	76 [f]

[a] EcoInvent (2014a); Swedish Energy Agency (2017)
[b] Edwards et al. (2014)
[c] Börjesson et al. (2016); Gode et al. (2011)
[d] Klackenberg (2019)
[e] EcoInvent (2014b)
[f] Personal communication (2017)
[g] Wallén (1999)

Table 6
List of the main assumptions used for modeling the transportation cost and fuel consumption. It is assumed that all transportation are performed by 25–40 t trucks with trailer.

Assumption	Liquid manure or digestate	Solid manure	Compressed biogas
Payload (t)	35	28	7.7 (cylinders)
Fuel consumption (Liter/100 km)[a]	32.0	30.3	45.2
Fuel consumption, idle (Liter/hour)[b]	4.0	4.0	4.0
Loading/unloading (hour)[c]	1.0	1.0	1.0
Cost of loading/unloading (€/hour)[d]	95	95	95
Cost of transport (€/km)[e]	1.3	1.3	1.3
Specific cost at 10 km (€/tkm)[e]	0.35	0.43	1.57
Specific cost at 50 km (€/tkm)[e]	0.13	0.16	0.58
Specific cost at 100 km (€/tkm)[e]	0.10	0.12	0.46
Specific cost at 10 km (€/t)[e]	3.5	4.3	15.7
Specific cost at 50 km (€/t)[e]	6.4	8.0	29.0
Specific cost at 100 km (€/t)[e]	10.0	12.5	45.6

[a] Mårtensson (2018)
[b] Green Truck Partnership (2015)
[c] Trafikverket (2020)
[d] Murto et al. (2013)
[e] Calculated based on other values.

Table 7
List of main assumptions for emissions from storage and spreading of digestate. TN stands for Total Nitrogen, TAN for Total Ammonium Nitrogen, B_0 for biogas yield, TC for Total Carbon, and VS for Volatile Solids. Other than these losses we assume nitrogen, phosphorus, and potassium applied as digestate can substitute mineral fertilizers based on recommended levels. In previous work we have explored this assumption in greater detail (Akram et al., 2019a).

Storage emissions	Unit	Liquid	Solid
Nitrogen, ammonia (NH_3-N)	% of TN	1.0%[a]	10.0%[b]
Nitrogen, nitrous oxide (N_2O-N) (direct)	% of TN	0.0%[c]	1.4%[d]
Nitrogen, nitrous oxide (N_2O-N) (indirect)	% of NH_3-N released	1.0%[a]	1.0%[a]
Methane (CH_4)	% of B_0 (m^3 CH_4/t VS)	3.5%[a]	1.0%[e]
Spreading emissions	Unit	Liquid	Solid
Nitrogen, ammonia (NH_3-N)	% of TAN	7.5%[a]	15.0%[b]
Nitrogen, nitrous oxide (N_2O-N) (direct)	% of TN	1.13%[c]	1.13%[c]
Nitrogen, nitrous oxide (N_2O-N (indirect)	% of NH_3-N released	1.0%[a]	1.0%[a]
Carbon binding in soil	% TC	29.0%[a]	29.0%[a]

[a] Börjesson et al. (2016)
[b] Qvist Frandsen et al. (2011)
[c] IPCC (2006)
[d] Vandré et al. (2013)
[e] Raaholt et al. (2011)

pact factors (e.g. impact characterization factors in IPCC GWP_{100}) to calculate the life cycle impact.

Additional assumptions for transportation and emissions from storage and spreading of digestate are presented in Table 6 and Table 7, respectively.

Biogas utilization is not the focus of our work, therefore we made a common assumption that all biogas plants use their own biogas for heating, and the rest of the biogas (after considering internal use and losses) is upgraded and compressed and transported via trucks to a market located at 50 km distance. We have assumed that the upgraded biogas can be used as a diesel-substitute with 90% efficiency (Delgado and Muncrief, 2015). Similar to Metson et al. (2020), we have assumed that the N:P:K content of the digestate replaces mineral fertilizers. The fossil fuel and mineral fertilizers that are replaced by the produced biogas and biofertilizer are characterized in Table 4.

3. Results

3.1. Change in crop and animal make-up across the country

Future land use under EY and SY showed less animal intensive areas as well as more crop diversity, but followed current patterns (Figs. 7, SI-1 and SI-8). As per the national number differences among scenarios (Fig. 2), the Karlsson et al. (2018) scenarios showed less animals on the landscape overall, and less concentrated production. There were higher animal numbers in the south of Sweden (top panel Fig. 7), but there was a clear decrease in intensity with the alternative futures (Figures SI-2, SI-3), especially the SY scenario (top panel Figs. 7, SI-9, SI-10). Although ley (dotted with semi-natural pastures) in the lower third of Sweden was fairly constant as a dominant across all three scenarios, vegetables and fruits were more present in the northern two thirds for EY and SY (Figs. 7, SI-5, SI-7, SI-12, SI-14). Differences are more pronounced among current, EY, and SY with regards to the second most dominant crop, especially in middle and northern Sweden. Cereals have been replaced by legumes/oil crops in middle Sweden, and potatoes further north (Figs. 7, SI-4, SI-6, SI-11, SI-13). The scenario rule of no oil crops (rape seed) or legumes above 63 degrees is clearly visible in the sharp cut off in the SY scenario second most dominant crop. In summary, the spatial distribution of crops and animals reflects both current patterns and the target values and rules from the future scenarios used in the model.

3.2. Capacity to meet nutrient requirements

Across all three scenarios, more crop demand could be met after the redistribution of excreta as digestate from biogas plants.

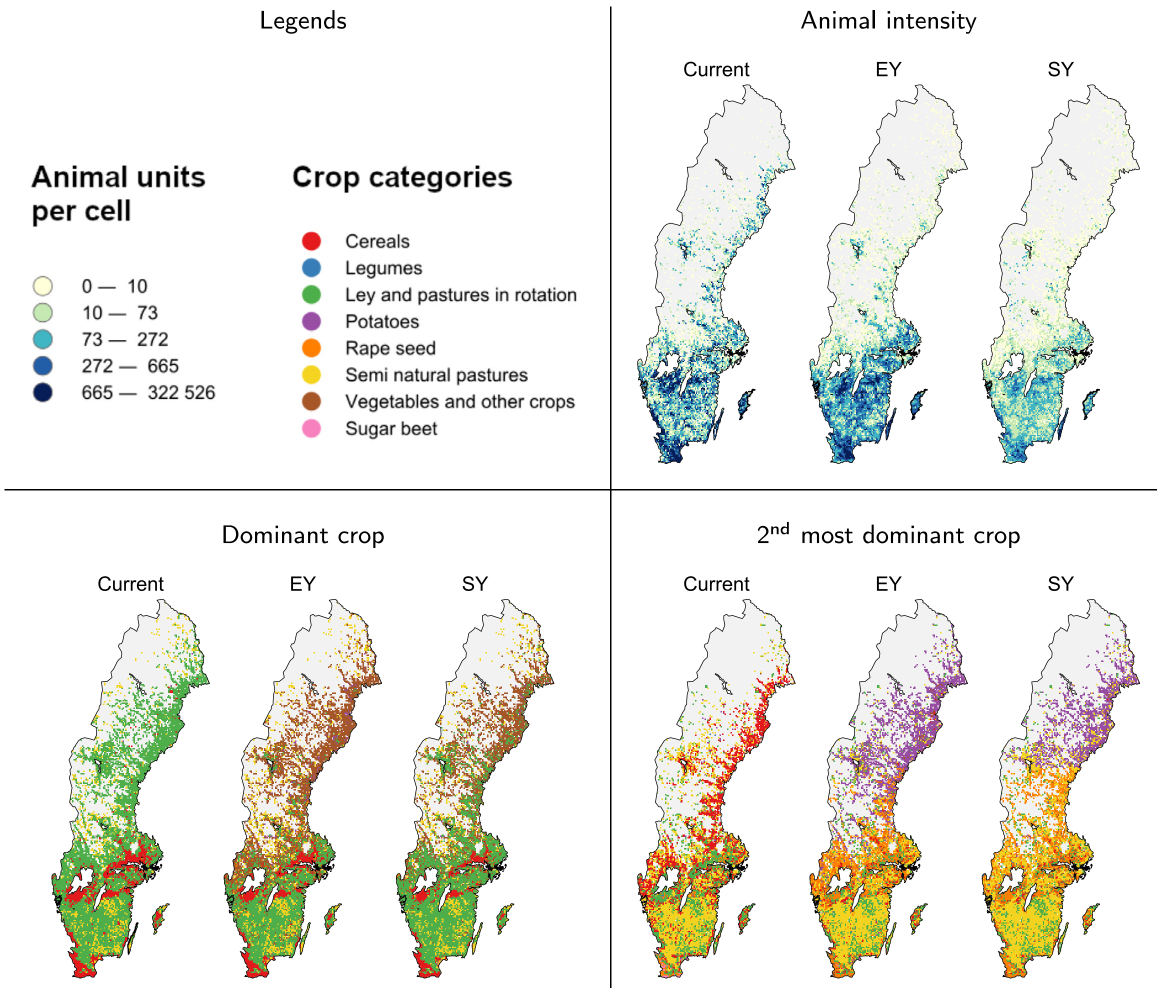

Fig. 7. Location of animals and crops in current (circa 2015, left maps), efficiency (EY, middle maps), sufficiency (SY, right maps). The left top panel shows the legends which are kept constant in all maps. The right top panel shows the intensity of animals in a 25 km^2 grid. Animal units were calculated by multiplying number of animals in a cell by EU conversion rates (European Commission, 2020). Color bins are based on quantiles from the current scenario and kept constant across scenarios so that they can be compared. The left bottom panel shows the crop category that the largest area of all crop categories in a grid. The right bottom panel shows the second most dominant crop category in a grid.

The difference between pre- and post- optimization was not as striking in EY and SY than under Current conditions (comparing Figure SI-16 and Fig. 8). This was expected as there was both less capacity to meet demand nationally (Table SI-2, so less chance of surplus areas) and our land-use model aimed to place animals in a more distributed manner, and closer to feed, across the country. In the Current scenario redistributing excreta through biogas plants meant that 51% more crop P demand could be met, but for SY only 8% more N, and 33% more P crop demand benefited from excreta (Table SI-2). Importantly however, after transport, EY and SY have no surplus 25 km^2 cells: The full national potential of nutrients in excreta could be used after redistribution.

In line with the decrease in available animal manure, the average weight of transport events drastically decreased in the alternate futures we considered, especially SY; however the number of plant locations to process these reduced amounts did not decrease as much (Table SI-2). Under the Current scenario, 236 plant locations were selected to move animal (120) and human (116) excreta. This number decreased to 228 for the EY and 184 for the SY scenarios, and where plants to treat human waste increased slightly in number to accommodate population growth. Although Skåne experienced the largest decreases in available manure across scenarios, it still remained the region with the second most biogas plant locations across all scenarios (Table SI-2, Figure SI-18). Västra Götaland had the most plant locations across all scenarios, and the largest decrease in the nutrient balances between current and alternate futures (larger deficit of all three nutrients, Fig. 8) after optimization. Västerbotten saw the largest decreases in the number of biogas plant locations selected, from 18 (Current) down to 10 (EY) and 9 (SY). Overall, the distribution of where to build biogas plants did not change, even under the SY scenario which had much fewer animals and a different crop composition compared to today; the model suggests only to build fewer or smaller plants (Figure SI-16 and Table SI-2).

Transport cost decreases in EY and SY were not uniform among regions (Table SI-2). At the national level, the most drastic decreases in costs were associated with the redistribution of excreta as digestate back to croplands. Kalmar saw the largest cost reductions for its average biogas plant between the EY and Current scenarios. In the SY scenario, Kalmar also saw large average reductions for transport to and from biogas plants, but it was Skåne that saw the largest reduction in transportation costs out of biogas plants. Interestingly neither of these regions saw the largest reductions in distances or transport weights across scenarios; this highlights that costs are indeed a combination of both parameters. There were also a few instances where the average costs associated with trans-

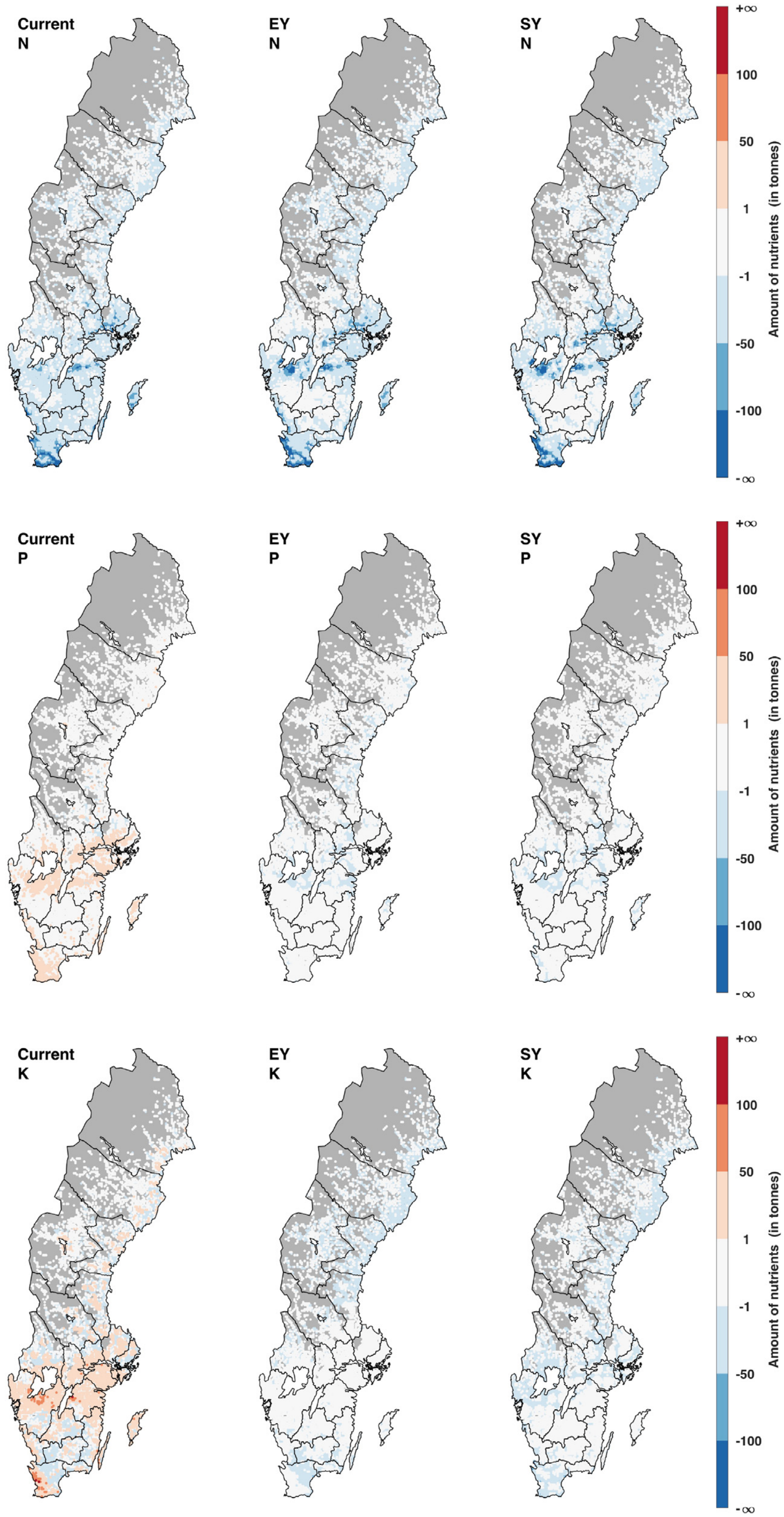

Fig. 8. Nutrient balance maps for Sweden in tonnes of nutrient per 25 km^2 cells across scenarios after transportation by the optimization model. The top row is nitrogen (N), the middle row is phosphorus (P), and the bottom row is potassium (K). The left column is the Current scenario, the middle is the Efficiency scenario (EY), and the right is the Sufficiency scenario (SY). Values are expressed in tonnes of nutrient per 25 km^2 cell. Blue represents a deficit (crop nutrient demand > manure+sludge) and red a surplus, where bold values along the y axis represent the cutoff values (tonnes) for each color bin. Black contours within Sweden represent regional borders. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

ports to biogas plants increased, notably for Västmanland in the EY scenario.

Inter-regional transports remained important across all scenarios, although regional changes, just like with costs, were not always consistent for EY or SY (Table SI-2). In the Current scenario around 20% of animal manure transport events entailed cross-regional movement, while this proportion was higher for transports leaving WWTPs (47%). On average, our two alternative futures saw a larger proportion of transports to biogas plants coming from other regions, and SY saw a decrease in the proportion leaving biogas plants to be used on croplands outside each region. Across both EY and SY, Kalmar saw large decreases in the proportion of transports exported to other regions and Gävleborg saw large increases in the proportion of in-bound transports. Other differences were unique to each scenario (Table SI-2).

For example, although Gotland (an island) had no transports across its borders in the Current scenario, the SY scenario would suggest that 22% of its biogas plant transport inputs would come from manure in other regions.

See Section SI-2 for more details about differences among the scenarios, including nutrient supply and demand (Figure SI-15, and transport costs).

3.3. Costs and benefits related to centralized recycling through biogas plants

All three scenarios explored show a net benefit to recycling excreta through biogas plants (Fig. 9). However, monetary benefits have a relatively smaller margin for EY and SY than for the Current scenario.

There are relatively large benefits for all scenarios with regards to climate impact and primary energy use. Still, the SY scenario would result about 50% less savings than under the Current scenario; 0.7 vs 1.6 million tonnes CO_2 climate savings. The amount of possible fossil fuel and mineral fertilizer substitution, given the different amount of excreta available, among the scenarios drives the above pattern.

The financial net benefits across all three scenarios is quite close to zero. Transportation has a relatively large role in the cost of excreta management in all scenarios. The specific cost of transportation is lower in EY and and even lower in SY (compared to Current), which leads to a noticeable difference in the total net specific cost among the scenarios (Figure SI-19). However, compared to Current, the relative cost of transportation in contrast to the gains that can be achieved through substitution is significantly reduced in EY and SY scenarios. For example, let us consider the ratio of the cost of transportation (of excreta and digestate) to the gains from substitution of mineral fertilizers. In Current, this ratio is 1.4 (i.e., each € gained through substitution of mineral fertilizers corresponds to 1.4 € transportation cost). However, the corresponding values for EY and SY are 1.0 and 0.8 respectively. So, along with the reduction of total amount of excreta that has to be efficiently managed, the relative importance of transportation decreases. For this reason, the SY scenario is less sensitive to variations in transportation costs or the gains through substitution of mineral fertilizers. For example, consider a 20% increase in the both transport costs and the price of mineral fertilizers. In the Current scenario, the increased transportation cost would increase the net cost by 57% which is only partly offset by the 31% reduction of the net cost due to increased gains from the substitution of fertilizers. However, in SY the higher transport cost (17% increased net cost) is almost offset by the additional revenues from the substitution of mineral fertilizer (14% decreased net cost).

In summary, centralized processing of excreta may still have net benefits given a drastically different food system with less animals.

4. Discussion

4.1. Realism

Although our model used current crop distribution and scenario constraints to ensure realism, exploring regional scenarios could further increase realism. Some of the implications of our rule choices may not be adapted to regional realities. For example the large increases in vegetable production in the northern two thirds of Sweden produced by our model (Fig. 7) may not make sense (without careful variety selection and perhaps greenhouses). This placement is likely an artifact of not allowing oil and legume crops in the north, while simultaneously keeping hectares of crop categories that decreased (e.g., leys and cereals) where they were most prevalent under current conditions (which is likely in the south). Incorporating regionally specific constraints, perhaps prioritizing keeping northern agricultural lands similar to today, may create more drastic changes in southern regions but may be more realistic.

A different approach which could also increase realism is to use a forecasting approach, evaluating the land use (and other environmental and social outcomes) impact of specific policy interventions. Agent based models have been used, for instance, to model the outcomes of greening the EU Common Agricultural Policy (CAP). The CAP direct payments to farmers scheme has so far failed to deliver on the majority of the EU's food system sustainability goals (Pe'er et al., 2019; Scown et al., 2020). Looking at Southern Sweden, the AgriPoliS model (Brady et al., 2012) demonstrated how 2013 CAP reforms were unlikely to support environmental improvements (e.g., biodiversity) in part because of this spatial scale disconnect (Hristov et al., 2020). The diversity of options given to farmers means they can minimize their cost (e.g., fallow a field that is not productive) in a way that may not match what should be done to maximize environmental benefits (e.g., fallowing a field in a particular location even if productive). The authors suggest that having member states have more spatially-explicit target measures could improve outcomes (as per the example above specifying where certain actions must be taken, for instance what fields to fallow). Although our scenarios do not give such targeted measures or metrics, examining how close our spatially-explicit futures are to spatial outcomes from forecasting models could further prompt insights into the types of interventions needed to support multiple sustainability objectives.

Modeling desirable outcomes at the national (or EU) level, while also accommodating for regional, farm, and even field level constraints and decisions is challenging in both policy and research endeavors. Often separate models, or adapted versions, need to be used. A pluralistic approach is, and will continue to be, necessary.

4.2. Desirability and necessity

The future scenarios we explored are radically different than today; mounting evidence suggests that those are the radical changes food systems must undergo to sustainability feed the world. One could argue that our land-use allocation assumptions, which focused on keeping crops where they are, is perhaps even too conservative. Recent work in the USA has demonstrated that agricultural landscape complexity, including crop diversification, can increase yields by the same order of magnitude as the effect of precipitation variation or soil characteristics (Nelson and Burchfield, 2021). Supporting high crop yields, while limiting water and air quality impairment and biodiversity losses related to N management, will also require extremely high nutrient use efficiency all over the EU (Schulte-Uebbing and de Vries, 2021). In Sweden, where there has already been extensive work to increase nutrient

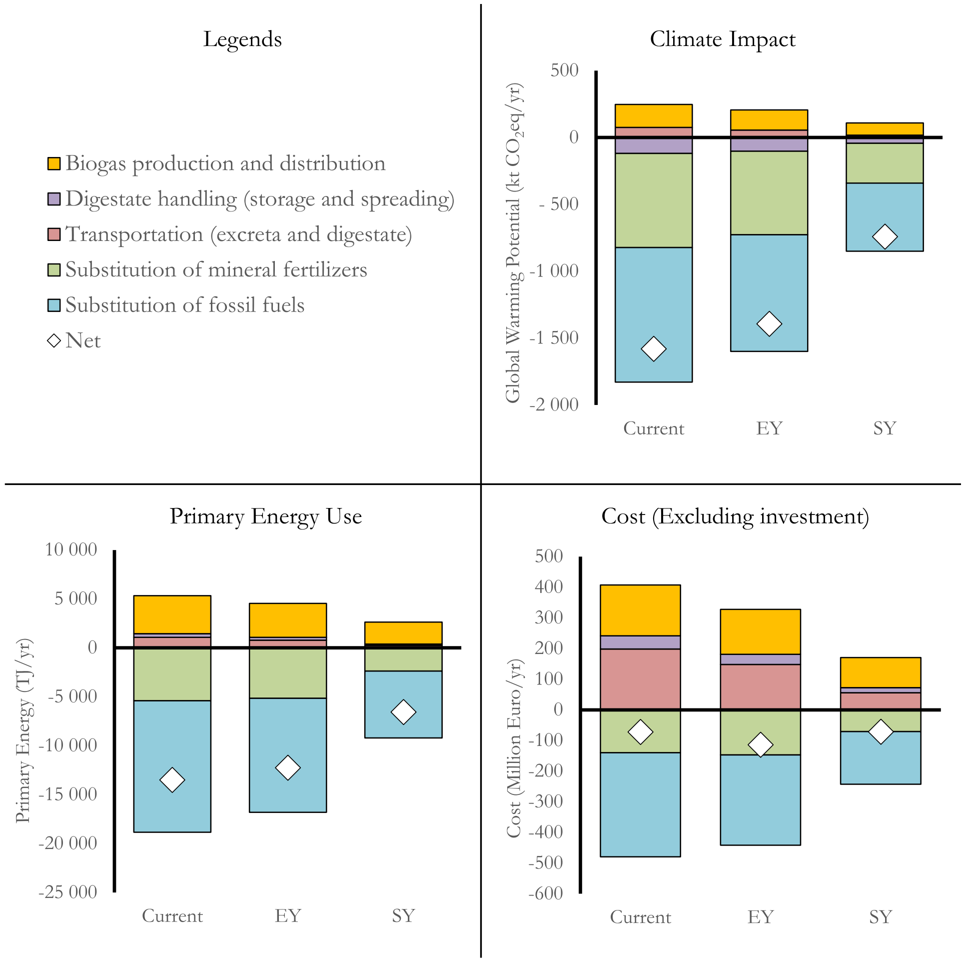

Fig. 9. Life cycle impacts of producing biogas from excreta under the conditions of the studied scenarios (EY for Efficiency Scenario, and SY for Sufficiency Scenario) compared with the current situation (Current). Top-right: climate impact, bottom-left: primary energy use, bottom-right: cost. Positive values indicate *savings* or *revenue*. The diamond shapes represent the *net effect* considering all impacts/costs and savings/revenues. Actual numbers are presented in Table SI-3.

use efficiency, meeting these environmental objectives would mean needing to achieve close to 90% N use efficiency (Schulte-Uebbing and de Vries, 2021). Our results assume that close to 100% nutrient efficiency use takes place, and this is only possible if manure is moved across the landscape. Even with the animal and crop redistribution in the EY and SY models, maximizing the use of N (and thus hopefully minimizing losses) required transport. In other words judicious, and circular, nutrient management will play a central role in required food system changes, making the results of this study important for planning. Even if it may not be very profitable under current economic conditions, changing and diversifying agricultural land use and moving excreta to match crop needs will be essential.

4.3. Resilience

Logistical planning under radically different agricultural systems could prioritize centralized/large scale biogas investments in those areas that are likely to keep relatively higher concentrations of animals and people and simultaneously consider smaller, or more modular, systems that can adapt in other areas. Our spatial interpretation of both EY and SY scenarios resulted in less (and less dense) animal husbandry and larger changes in dominant crops grown in the northern half of the country (Fig. 7). Still, there were fewer changes in terms of the location of biogas plants and the environmental and economic benefits of full recycling (Figures 9, SI-18, and SI-19). Although fewer biogas plants would be needed, our models indicate that the relative distribution among regions is relatively constant across scenarios.

Sweden has yet to achieve its biogas production targets and is planning to build more plants (Klackenberg, 2021; Norstedts Juridiks, 2019). Our results could be used to prioritize building in those locations that the model selected in all three scenarios. In addition, for those regions where few locations and plants are needed (e.g., Blekinge, Gotland, and Västmanland) further exploration is particularly important to make sure local conditions are incorporated into the model and subsequent decisions. Overall, our study shows that investment in biogas and centralized recycling infrastructure seems resilient to a large suite of future food system configurations.

4.4. Model expansions to better account for diverse and cross-level conditions

4.4.1. More substrates

Next steps include explicitly accounting for additional organic waste streams, in particular crop residues, food processing waste and household organic waste. These organic resources become more important in planning a circular economy as the amount of animal manure decreases. The importance of alternatives to animal manure is already clear when comparing the relative importance of human excreta across scenarios; most strikingly human excreta accounts for 64% of total P supply under the SY scenario but only 26% under current conditions. Even under current conditions non-excreta substrates are an important source of green energy and nutrients (Hamelin et al., 2019). Incorporating these other residue streams would also have the benefit of being able to better account for the need for dryer materials (with higher TS) in biogas production.

In particular, some of these substrates may be very relevant in certain regions or localities and therefore need to be included in more realistic local analyses. Expanding our spatial analysis to have a more detailed local view and include a broader set of substrates will increase its practical relevance for stakeholders who are interested in the development of resource efficient energy and nutrient recovery through biogas production.

4.4.2. Diversify technology options

Of particular interest when considering alternative futures is incorporating additional technology options that would change the weight and nutrient composition of recycled fertilizer products. Looking at anaerobic digestion with biogas production makes sense as its capacity to reuse energy and nutrients means that its diverse benefits are not tightly linked to the size of the plant (Arias et al., 2020). Biogas solutions are also highly compatible with other nutrient recovery technologies after digestion (Harder et al., 2019; Johannesdottir et al., 2020). Combining different recovery technologies in different locations and at different scales could alter the cost-benefit ratios of recycling and could result in different supply and demand areas being linked.

Looking at different technologies for different organic waste types is likely the first step. Because there is so much less animal manure in the EY and SY future scenarios, processing the manure through smaller biogas reactors, on-farm or within $25km^2$ cells, to power farm equipment (more in line with Karlsson et al. (2017)) may also be worth further exploration. The cost-benefit analysis would likely need to include investment costs; in places where the weight of digestate makes it prohibitively expensive to transport, perhaps small-scale biogas plant construction is more favorable than the centralized model we employ in our current model. Still, within grid recycling is likely not enough to maximize the reuse of the nutrient value of all excreta. Even with the redistribution of animals closer to pasture and feed, our model outputs showed nutrient imbalances which warrant transport between cells. For instance, Sweden would increase its capacity to meet crop N demand by 9% by moving excreta between cells rather than by simply recycling within each cell under the EY scenario (Table SI-2). The gains are more drastic for P and less so for K, but this is true across all three scenarios, not just EY and SY.

At the same time nutrients from cities could be treated in biogas reactors and then further concentrated to travel where they are most needed; simultaneously tackling concerns about contaminants and opening up possibilities to alter N:P:K ratios and transport costs. In the Swedish context, some relevant technologies include struvite production, P extraction from ash (e.g., EasyMining), hydrothermal carbonization (e.g., C-Green), and various source separation options (e.g., Reko-Lab). Sewage sludge in Sweden contains about 26 g of P per kg (SCB, 2014), while struvite products derived from WWTP digestate typically contain around 120–130 g P per kg (Hall et al., 2020). The order of magnitude difference in concentration could make longer transport distances of struvite to meet crop nutrient demand more economical. Struvite is also poised to be accepted as an input to EU organic agriculture (EGTOP, 2016), and thus including it as an option would further increase the realism of our modeling work. Still, many nutrient recovery technologies focus on one nutrient, often P, while recent work shows that adequate supply of N (Barbieri et al., 2021) and K (Reimer et al., 2020) are of particular concern in organic production systems. Multiple types of organic wastes and multiple processing technologies will be needed to meet diverse growing conditions and crop nutrient demands across the country, and adapted government rules and policies will be needed to facilitate their implementation.

4.4.3. Cross-level land-use allocation rules

Finally, we cannot discount the importance of alternative land use allocation rules and food system features to fulfill social and environmental sustainability goals. The scenarios explored here, and the series of rules we developed to translate these scenarios in to maps, represent only a few possible options for Sweden's future food landscape. Looking not only at national goals, but also at how such plans fit in to EU and global futures, and farm-level and local realities, is necessary for society to create resilient and desirable realities. Other scenario spatial modeling endeavours have for instance looked the effect of farm type on multi functionality in Sweden (Boke Olén et al., 2021). In the USA using optimization modeling to allocate crop types in a watershed could reduce nutrient losses to waterways (Femeena et al., 2018). In other words, a multi-scale and multi-perspective approach is needed (Pereira et al., 2021), and the actual future created will be a combination or patch-work of multiple scenarios and processes (Bennett et al., 2021). In our case, model improvements and extensions could benefit from a focus on sensitive parameters. For example, the feed needs of milking cows, and subsequent characteristics of their manure, had the largest impact on total nutrient supply, and the percentage nutrient demand met in grids, in our sensitivity analysis. This makes sense as milking cows make up the largest share of nutrient supply nationally, and thus exploring alternative values for feeding milking cows will be important. Important additions to model could include land-based aquaculture, agricultural land abandonment to support biodiversity, and urban agriculture.

5. Conclusions

The logistical planning that must be done today to facilitate a biobased circular economy should consider building in resilience for the drastically different land uses, and organic material quantities, required for sustainable food futures. The food system changes required "upstream" of recycling activities to meet sustainable food, energy, and water objectives will change both the amounts and the locations of nutrient supply and demand.

Our model changed total animals more in the south but the make-up of crops and animals more in the north, especially in the SY scenario compared to today. When these changes were translated to nutrient supply and demand however, the need for biogas plants regionally followed similar patterns across all scenarios. In other words, even with drastic reductions in excreta available, and increased crop needs, many optimal locations for biogas plants remained the same. Human excreta became a proportionally more important part of meeting crop demands. As such, how Sweden decides to treat and reuse human excreta will require particular attention.

Nutrient recycling and biogas production were less financially profitable in aggregate for all of Sweden for the EY and SY scenarios compared to the Current. Still the co-produced scenarios, and full recycling under current conditions, all resulted in net benefits financially, and in terms of energy and GHG. In fact, both alternative futures were more beneficial per unit of transport. This means that full recycling of Sweden's excreta would be more resilient to fluctuations in transport or fertilizer prices, even if the country can meet less of its national nutrient and energy demands under more self-sufficient conditions.

Our modeling results do not show consistent regional changes across scenarios and metrics of change. As such, planning will need to be nuanced and explicit in what is prioritized. The challenge ahead, given that new plants and agreements are investment heavy, is to coordinate actions among multiple actors so that those locations for biogas plants that could best serve the largest number of farmers under different scenarios get support now. In this way Sweden would be constructing the future system it wants to see, while remaining resilient to the many faces a desirable future can take.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary material

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.spc.2021.10.019

References

Akram, U., Quttineh, N.-H., Wennergren, U., Tonderski, K., Metson, G.S., 2019. Enhancing nutrient recycling from excreta to meet crop nutrient needs in sweden - a spatial analysis. Sci Rep 9 (10264).
Akram, U., Quttineh, N.-H., Wennergren, U., Tonderski, K., Metson, G.S., 2019. Optimizing nutrient recycling from excreta in Sweden and Pakistan: higher spatial resolution makes transportation more attractive. Frontiers in Sustainable Food Systems 3 (50).
Ammenberg, J., Feiz, R., 2017. Assessment of feedstocks for biogas production, part II'results for strategic decision making. Resour. Conserv. Recycl. 122, 388–404.
Arias, A., Feijoo, G., Moreira, M.T., 2020. What is the best scale for implementing anaerobic digestion according to environmental and economic indicators? J. Water Process Eng. 35, 101235.
Arvidsson, R., Svanström, M., 2016. A framework for energy use indicators and their reporting in life cycle assessment. Integr Environ Assess Manag 12 (3), 429–436.
Avfall Sverige, 2016. Rapportering av data från metanmåtningar enligt Egenkontroll metanutsläpp-frivilligt åtagande 2007 -2015. Technical Report. Avfall Sverige.
Bachmann, N., 2015. Sustainable biogas production in municipal wastewater treatment plants. Technical Report. IEA Bioenergy.
Barbieri, P., Pellerin, S., Seufert, V., Smith, L., Ramankutty, N., Nesme, T., 2021. Global option space for organic agriculture is delimited by nitrogen availability. Nature Food 2 (5), 363–372.
Bauer, F., Hulteberg, C., Persson, T., Tamm, D., 2013. Biogas upgrading-Review of commercial technologies (SGC270). Technical Report. http://www.sgc.se/ckfinder/userfiles/files/SGC270.pdf
Bennett, E.M., Biggs, R., Peterson, G.D., Gordon, L.J., 2021. Patchwork earth: navigating pathways to just, thriving, and sustainable futures. One Earth 4 (2), 172–176.
Berglund, M., Börjesson, P., 2003. Energianalys av biogassystem (Energy systems analysis of biogas systems). Technical Report. Lund University.
Berglund, M., Börjesson, P., 2006. Assessment of energy performance in the life-cycle of biogas production. Biomass Bioenergy 30 (3), 254–266.
Boke Olén, N., Roger, F., Brady, M.V., Larsson, C., Andersson, G.K.S., Ekroos, J., Caplat, P., Smith, H.G., Dånhardt, J., Clough, Y., 2021. Effects of farm type on food production, landscape openness, grassland biodiversity, and greenhouse gas emissions in mixed agricultural-forestry regions. Agric Syst 189, 103071.
Brady, M., Sahrbacher, C., Kellermann, K., Happe, K., 2012. An agent-based approach to modeling impacts of agricultural policy on land use, biodiversity and ecosystem services. Landsc Ecol 27 (9), 1363–1381.
Busch, G., 2006. Future European agricultural landscapes - what can we learn from existing quantitative land use scenario studies? Agriculture, Ecosystems and Environment 114 (1), 121–140.
Börjesson, P., Lantz, M., Andersson, J., Björnsson, L., Möller, B.F., Fröberg, M., Hanarp, P., Hulteberg, C., Iverfeldt, E., Lundgren, J., Röj, A., Svensson, H., Zinn, E., 2016. Methane as Vehicle Fuel—A Well-to-Wheel Analysis (METDRIV). Technical Report.
Börjesson, P., Tufvesson, L., Lantz, M., 2010. Life cycle assessment of biofuels in Sweden. Lund University, Report (70).
Campbell, B.M., Beare, D.J., Bennett, E.M., Hall-Spencer, J.M., Ingram, J.S.I., Jaramillo, F., Ortiz, R., Ramankutty, N., Sayer, J.A., Shindell, D., 2017. Agriculture production as a major driver of the earth system exceeding planetary boundaries. Ecol. Soc. 22 (4), art8.
Carlsson, M., Uldal, M., 2009. Substrathandbok för biogasproduktion. Technical Report. Svenskt Gastekniskt Center.
Cordell, D., White, S., 2014. Life's bottleneck: sustaining the world's phosphorus for a food secure future. Annu Rev Environ Resour 39 (1), 161–188.
Dawson, C.J., Hilton, J., 2011. Fertiliser availability in a resource-limited world: production and recycling of nitrogen and phosphorus. Food Policy 36, S14–S22.
Delgado, O., Muncrief, R., 2015. Assessment of Heavy-Duty Natural Gas Vehicle Emissions: Implications and Policy Recommendations. Technical Report. ICCT. http://www.theicct.org/
EcoInvent, 2014. Electricity, country mix, Sweden low voltage, Allocation. In: Ecoinvent database version 3.
EcoInvent, 2014. Iron(II) chloride, Global, Allocation. In: Ecoinvent database version 3.
Edwards, R., Haas, H., Larivé, J.-F., Lonza, L., Maas, H., Rickeard, D., 2014. Well–to-wheels report version 4. a; JEC WTW analysis. Institute for Energy and Transport, Joint Research Centre, Luxembourg: Publications Office of the European Union 2014.
EGTOP - Expert Group for Technical Advice on Organic Production. 2016. Final Report on Organic Fertilizers and Soil Conditioners (II). European Comission - Agriculture and Rural Development, Brussels, Belgium. Available at: https://ec.europa.eu/info/sites/default/files/food-farming-fisheries/farming/documents/final-report-egtop-fertilizers-ii_en.pdf.
Ekvall, T., 2019. Attributional and consequential life cycle assessment. IntechOpen.
European Commission, 2020. Glossary: Livestock unit (LSU).
Feiz, R., Johansson, M., Lindkvist, E., Moestedt, J., Påledal, S.N., Svensson, N., 2020. Key performance indicators for biogas production'methodological insights on the life-cycle analysis of biogas production from source-separated food waste. Energy 200, 117462.
Femeena, P.V., Sudheer, K.P., Cibin, R., Chaubey, I., 2018. Spatial optimization of cropping pattern for sustainable food and biofuel production with minimal downstream pollution. J. Environ. Manage. 212, 198–209.
Finnveden, G., Potting, J., 2014. Life Cycle Assessment. In: Wexler, P. (Ed.), Encyclopedia of Toxicology (Third Edition). Academic Press, Oxford, pp. 74–77.
Foley, J.A., Ramankutty, N., Brauman, K.A., Cassidy, E.S., Gerber, J.S., Johnston, M., Mueller, N.D., O'Connell, C., Ray, D.K., West, P.C., 2011. Solutions for a cultivated planet. Nature 478 (7369), 337–342.
Fridman, D., Koellner, T., Kissinger, M., 2021. Exploring global interregional food system's sustainability using the functional regions typology. Global Environ. Change 68, 102276.
Galloway, J.N., Winiwarter, W., Leip, A., Leach, A.M., Bleeker, A., Erisman, J.W., 2014. Nitrogen footprints: past, present and future. Environ. Res. Lett. 9 (11), 115003.
Glibert, P.M., 2020. From hogs to HABs: impacts of industrial farming in the US on nitrogen and phosphorus and greenhouse gas pollution. Biogeochemistry 150 (2), 139–180.
Gode, J., Martinsson, F., Hagberg, L., Öman, A., Höglund, J., 2011. Miljöfaktaboken 2011-Uppskattade emissionsfaktorer för bränslen, el, våme och transporter. Technical Report.
Green Truck Partnership, 2015. Technology Study — Engine idle management. Technical Report. Green Truck Partnership.
Hall, R.L., Boisen Staal, L., Macintosh, K.A., McGrath, J.W., Bailey, J., Black, L., Gro Nielsen, U., Reitzel, K., Williams, P.N., 2020. Phosphorus speciation and fertiliser performance characteristics: a comparison of waste recovered struvites from global sources. Geoderma 362, 114096.

Hamelin, L., Borzęcka, M., Kozak, M., Pudełko, R., 2019. A spatial approach to bioeconomy: quantifying the residual biomass potential in the EU-27. Renewable Sustainable Energy Rev. 100, 127–142.
Harder, R., Wielemaker, R., Larsen, T.A., Zeeman, G., Öberg, G., 2019. Recycling nutrients contained in human excreta to agriculture: pathways, processes, and products. Crit Rev Environ Sci Technol 49 (8), 695–743.
Hellstedt, C., Starberg, K., Olsson, L.-E., Hellström, D., Jonsson, L., Mossakowska, A., 2010. Increased biogas production at the Henriksdal Wastewater Treatment Plant (WWTP). Technical Report.
Hristov, J., Clough, Y., Sahlin, U., Smith, H.G., Stjernman, M., Olsson, O., Sahrbacher, A., Brady, M.V., 2020. Impacts of the EU's common agricultural policy "Greening" reform on agricultural development, biodiversity, and ecosystem services. Appl Econ Perspect Policy 42 (4), 716–738.
Ibisch, R., Westphal, K., Borchardt, D., Lukat, E., Boteler, B., Solheim, A.L., Austnes, K., Leujak, W., Schmedtje, U., 2016. European assessment of eutrophication abatement measures across land-based sources, inland, coastal and marine waters. ETC/ICM Report. ETC/ICM, Magdeburg, Germanz.
IPCC, 2006. Chapter 10: Emissions from Livestock and Manure Management. In: 2006 IPCC Guidelines for National Greenhouse Gas Inventories Volume 4 Agriculture, Forestry and Other Land Use, Vol. Volume 4. IPCC.
IPCC, 2013. Climate change: radiative forcing as global warming potential (GWP100), baseline model of 100 years of the IPCC 2013. In: Climate Change 2013. The Physical Science Basis. Working Group I Contribution to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. IPCC.
ISO, 2006. ISO 14040:2006 Environmental Management; Life-Cycle Assessment-Principles and Framework. Technical Report.
ISO, 2006. ISO 14044:2006 Environmental Management; Life-Cycle Assessment-Requirements and Guidelines. Technical Report.
Iwaniec, D.M., Cook, E.M., Davidson, M.J., Berbés-Blázquez, M., Georgescu, M., Krayenhoff, E.S., Middel, A., Sampson, D.A., Grimm, N.B., 2020. The co-production of sustainable future scenarios. Landsc Urban Plan 197, 103744.
JBV, 2013. Kartor och Geografiska informationssystem/jordbruksblock. Data by Jordbruksverket (JBV). https://jordbruksverket.se/e-tjanster-databaser-och-appar/e-tjanster-och-databaser-stod/kartor-och-gis.
JBV, 2016. Kartor och Geografiska informationssystem/Data för produktionsplatser för djurhållning. Data by Jordbruksverket (JBV).
JBV, 2021. Crop codes (Grödkoder). Data by Jordbruksverket (JBV). https://jordbruksverket.se/stod/lantbruk-skogsbruk-och-tradgard/sam-ansokan-och-allmant-om-jordbrukarstoden/grodkoder.
Johannesdottir, S.L., Macura, B., McConville, J., Lorick, D., Haddaway, N.R., Karczmarczyk, A., Ek, F., Piniewski, M., Księżniak, M., Osuch, P., 2020. What evidence exists on ecotechnologies for recycling carbon and nutrients from domestic wastewater? a systematic map. Environmental Evidence 9 (1), 24.
Jones, D.L., Cross, P., Withers, P.J.A., DeLuca, T.H., Robinson, D.A., Quilliam, R.S., Harris, I.M., Chadwick, D.R., Edwards-Jones, G., 2013. REVIEW: Nutrient stripping: the global disparity between food security and soil nutrient stocks. J. Appl. Ecol. 50 (4), 851–862.
JVB, 2014. Beställning av uppdrag kring gaspotentialer för gödselgasstödet—Bilaga 1.
Jurgilevich, A., Birge, T., Kentala-Lehtonen, J., Korhonen-Kurki, K., Pietikäinen, J., Saikku, L., Schösler, H., 2016. Transition towards circular economy in the food system. Sustainability 8 (1), 1–14.
Kabbe, C., 2018. Circular Economy: Bridging the Gap between Phosphorus Recovery and Recycling. In: Phosphorus Recovery and Recycling. Springer, Singapore, pp. 45–57.
Karlsson, J., Carlsson, G., Lindberg, M., Sjunnestrand, T., Röös, E., 2018. Designing a future food vision for the Nordics through a participatory modeling approach. Agronomy for Sustainable Development 38 (6).
Karlsson, J., Röös, E., Sjunnestrand, T., Pira, K., Larsson, M., Andersen, B.H., Sørensen, J., Veistola, T., Rantakokko, J., Manninen, S., Brubæk, S., 2017. Future Nordic Diets: Exploring ways for sustainably feeding the nordics. Technical Report. Nordic Council of Ministers TemaNord 2017:566, Rosendahls, Denmark.
Klackenberg, L., 2019. Hållbarhetskriterier för biodrivmedel. https://www.energigas.se/publikationer/haallbarhetskriterier-foer-biodrivmedel/.
Klackenberg, L., 2021. Biomethane in Sweden- market overview and policies. Technical Report. Swedish Gas Association.
Koppelmäki, K., Helenius, J., Schulte, R.P.O., 2021. Nested circularity in food systems: a nordic case study on connecting biomass, nutrient and energy flows from field scale to continent. Resour. Conserv. Recycl. 164, 105218.
Lantz, M., Ekman, A., Börjesson, P., 2009. Systemoptimerad produktion av fordonsgas-en miljö-och energisystemanalys av söderåsens biogasanläggning, Vol. 69. Lunds universitet. Avdelningen för miljö-och energisystem.
Lemp, J.D., Zhou, B.B., Kockelman, K.M., Parmenter, B.M., 2008. Visioning versus modeling: analyzing the land-use-transportation futures of urban regions. J. Urban Plann. Dev. 134 (3), 97–109.
Long, C.M., Muenich, R.L., Kalcic, M.M., Scavia, D., 2018. Use of manure nutrients from concentrated animal feeding operations. J. Great Lakes Res. 44 (2), 245–252.
Metson, G.S., Feiz, R., Quttineh, N.-H., Tonderski, K., 2020. Optimizing transport to maximize nutrient recycling and green energy recovery. Resources, Conservation & Recycling: X 9–10, 100049.
Mitchell, C., Cordell, D., Fam, D., 2014. Beginning at the end: the outcome spaces framework to guide purposive transdisciplinary research. Futures 65, 86–96.
Murto, M., Gregeby, E., Holmström, D., Castillo, M.d. P., Lagerkvist, A., Horváth, I.S., 2013. Ökad metanproduktion genom outnyttjade substrat och processutveckling. Technical Report. Waste Refinery.
Mårtensson, L., 2018. Emissions from Volvo's trucks.
Nelson, K.S., Burchfield, E.K., 2021. Landscape complexity and US crop production. Nature Food 2 (5), 330–338.
Nesme, T., Metson, G.S., Bennett, E.M., 2018. Global phosphorus flows through agricultural trade. Global Environ. Change.
Norstedts Juridiks, 2019. Mer biogas! För ett hållbart Sverige. Betånkande av Biogasmarknadsutredningen. Statens Offentliga Utredningar. SOU 2019:63. http://www.nj.se/offentligapublikationer
Pe'er, G., Zinngrebe, Y., Moreira, F., Sirami, C., Schindler, S., Müller, R., Bontzorlos, V., Clough, D., Bezák, P., Bonn, A., Hansjürgens, B., Lomba, A., Möckel, S., Passoni, G., Schleyer, C., Schmidt, J., Lakner, S., 2019. A greener path for the EU common agricultural policy. Science 365 (6452), 449–451.
Pereira, L., Kuiper, J.J., Selomane, O., Aguiar, A.P.D., Asrar, G.R., Bennett, E.M., Biggs, R., Calvin, K., Hedden, S., Hsu, A., Jabbour, J., King, N., Köberle, A.C., Lucas, P., Nel, J., Norström, A.V., Peterson, G., Sitas, N., Trisos, C., Vuuren, D.P.v., Vervoort, J., Ward, J., 2021. Advancing a toolkit of diverse futures approaches for global environmental assessments. Ecosystems and People 17 (1), 191–204.
Personal communication, 2017. Personal communication with biogas producing companies.
Poore, J., Nemecek, T., 2018. Reducing food's environmental impacts through producers and consumers. Science 360 (June), 987–992.
Qvist Frandsen, T., Rodhe, L., Baky, A., Edström, M., 2011. Best available technologies for pig manure biogas plants in the Baltic Sea Region. Technical Report. Baltic Sea 2020.
Raaholt, B., Bergström, B., Broberg, A., Holtz, E., Nordberg, U., Castillo, M.D.P., Baky, A., 2011. Bedömning av två tekniker för torkning av lätt nedbrytbart organiskt matavfall [Assessment of two techniques for drying of easily degradable organic bio-waste]. Technical Report. Waste Refinery.
Reimer, M., Möller, K., Hartmann, T.E., 2020. Meta-analysis of nutrient budgets in organic farms across europe. Organic Agriculture.
Röös, E., Carlsson, G., Ferawati, F., Hefni, M., Stephan, A., Tidåker, P., Witthöft, C., 2018. Less meat, more legumes : prospects and challenges in the transition toward sustainable diets in sweden. Renewable Agric. Food Syst 1–14.
Saltelli, A., Annoni, P., Azzini, I., Campolongo, F., Ratto, M., Tarantola, S., 2010. Variance based sensitivity analysis of model output. design and estimator for the total sensitivity index. Comput Phys Commun 181 (2), 259–270.
SCB, 2014. Utsläpp till vatten och slamproduktion PDF. Technical Report. Statistiska centralbyrån (SCB). https://docplayer.se/5057092-Utslapp-till-vatten-och-slamproduktion-2012.html
SCB, 2020. Population size, number of birth, deaths and migration by region, sex and age. Year 2020 - 2070. We have selected the following - Table contents: Population, Region: Municipality, all of them, Sex: both, Age: no selection, Year: 2030, Format: "Kommaavgrånsad utan rubrik".
Scherer, L., Verburg, P.H., 2017. Mapping and linking supply- and demand-side measures in climate-smart agriculture. a review. Agron. Sustainable Dev. 37 (6), 66.
Schulte-Uebbing, L., de Vries, W., 2021. Reconciling food production and environmental boundaries for nitrogen in the european union. Sci. Total Environ. 786, 147427.
Scown, M.W., Brady, M.V., Nicholas, K.A., 2020. Billions in misspent EU agricultural subsidies could support the sustainable development goals. One Earth 3 (2), 237–250.
Sellberg, M.M., Norström, A.V., Peterson, G.D., Gordon, L.J., 2020. Using local initiatives to envision sustainable and resilient food systems in the stockholm city-region. Glob Food Sec 24, 100334.
Sharpley, A., Kleinman, P., Jarvie, H., Flaten, D., 2016. Distant views and local realities: the limits of global assessments to restore the fragmented phosphorus cycle. Agricultural & Environmental Letters 1 (1), 160024.
Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., de Vries, W., de Wit, C.A., 2015. Planetary boundaries: guiding human development on a changing planet. Science 347 (6223), 1259855.
SVA, 2020. Poultry -KRAV certification guidelines. https://www.sva.se/en/animals/poultry/.
Swedish Energy Agency, 2017. Greenhouse gas emissions (Våxthusgasutslåpp). Technical Report. Swedish Energy Agency. http://www.energimyndigheten.se/fornybart/hallbarhetskriterier/drivmedelslagen/vaxthusgasutslapp/
Swedish EPA, 2018. Sweden's Environmental Objectives. Technical Report. Swedish Environmental Protection Agency, Stockholm.
Trafikverket, 2020. Analysmetod och samhållsekonomiska kalkylvården för transportsektorn. Technical Report.
Tälle, M., Wiréhn, L., Ellström, D., Hjerpe, M., Huge-Brodin, M., Jensen, P., Lindström, T., Neset, T.-S., Wennergren, U., Metson, G.S., 2019. Synergies and trade-offs for sustainable food production in sweden: an integrated approach. Sustainability 11 (3), 601.
Vandré, R., Wulf, S., Häußermann, U., Horlacher, D., 2013. N2O emissions from solid manure storage-calculation of a national emission factor. Landtechnik 68 (1), 38–42.
Verburg, P.H., Alexander, P., Evans, T., Magliocca, N.R., Malek, Z., Rounsevell, M.D., van Vliet, J., 2019. Beyond land cover change: towards a new generation of land use models. Curr Opin Environ Sustain 38, 77–85.
Wallén, E., 1999. Livscykelanalys av dricksvatten-en studie av ett vattenverk i Göteborg.
Willett, W., Rockström, J., Loken, B., Springmann, M., Lang, T., Vermeulen, S., Garnett, T., Tilman, D., DeClerck, F., Wood, A., Jonell, M., Clark, M., Gordon, L., Fanzo, J., Hawkes, C., Zurayk, R., Rivera, J.A., Vries, W.D., Sibanda, L., Afshin, A., Chaudhary, A., Herrero, M., Agustina, R., Branca, F., Lartey, A., Fan, S., Crona, B., Fox, E., Bignet, V., Troell, M., Lindahl, T., Singh, S., Cornell, S., Reddy, S., Narain, S.,

Nishtar, S., Murray, C., 2019. Food in the anthropocene: the EAT-lancet commission on healthy diets from sustainable food systems. Lancet 6736 (18), 3–49.
Withers, P.J.A., Forber, K.G., Lyon, C., Rothwell, S., Doody, D.G., Jarvie, H.P., Martin-Ortega, J., Jacobs, B., Cordell, D., Patton, M., Camargo-Valero, M.A., Cassidy, R., 2020. Towards resolving the phosphorus chaos created by food systems. Ambio 49 (5), 1076–1089.
Yara, 2018. Gödslingsråd—Handbok för båsta gödslingsekonomi!. Technical Report. Yara AB, Malmö.
Zhang, T., Hou, Y., Meng, T., Ma, Y., Tan, M., Zhang, F., Oenema, O., 2021. Replacing synthetic fertilizer by manure requires adjusted technology and incentives: A farm survey across China. Resources, Conservation and Recycling 168, 105301.