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Research article

A new direction for tackling phosphorus inefficiency in the UK food system

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ABSTRACT

The UK food system is reliant on imported phosphorus (P) to meet food production demand, though inefficient use and poor stewardship means P is currently accumulating in agricultural soils, wasted or lost with detrimental impacts on aquatic environments. This study presents the results of a detailed P Substance Flow Analysis for the UK food system in 2018, developed in collaboration with industry and government, with the key objective of highlighting priority areas for system interventions to improve the sustainability and resilience of P use in the UK food system. In 2018 the UK food system imported 174.6 Gg P, producing food and exportable commodities containing 74.3 Gg P, a P efficiency of only 43%. Three key system hotspots for P inefficiency were identified: Agricultural soil surplus and accumulation (89.2 Gg P), loss to aquatic environments (26.2 Gg P), and waste disposal to landfill and construction (21.8 Gg P). Greatest soil P accumulation occurred in grassland agriculture (85% of total accumulation), driven by loadings of livestock manures. Waste water treatment (12.5 Gg P) and agriculture (8.38 Gg P) account for most P lost to water, and incineration ashes from food system waste (20.3 Gg P) accounted for nearly all P lost to landfill and construction. New strategies and policy to improve the handling and recovery of P from manures, biosolids and food system waste are therefore necessary to improve system P efficiency and reduce P accumulation and losses, though critically, only if they effectively replace imported mineral P fertilisers.

1. Introduction

Phosphorus (P) is fundamental to our food production systems, being an essential nutrient for healthy and productive crops and livestock. However, ongoing debate about the longevity of mineable P resources (Chowdhury et al., 2017), potential future geopolitical pressures on P supply (Elser et al., 2014) and the negative impact of poor P management on our aquatic environment (Steffen et al., 2015) are growing sustainability issues that require a radical change in how P is managed in local and national food systems (Brownlie et al., 2021). Having no mineable P reserves of its own, the UK relies heavily on imported P to ensure its food system P demand is met, making the UK potentially vulnerable to future P supply pressures. A current dependence on Russian imports of P fertilisers and war in Ukraine has brought this UK vulnerability even more into focus. Furthermore, poor P management and inefficiency, particularly in the waste water treatment and agricultural sectors, mean that elevated P concentrations are currently the most common reason for failure to meet water quality targets for good ecological status in English rivers and lakes (Environment Agency, 2019).

Improving food system P sustainability should be achieved through more sparing and efficient use of P imports (e.g. fertiliser and animal feed), maximizing P recovery and recycling, and minimizing waste and loss (Withers et al., 2018). Identifying hotspots of system inefficiency and loss is therefore critical if the food system is to become more sustainable. Substance Flow Analysis (SFA) is a method that can be used to assess P flows and stocks at local, regional and national scales with the aim of identifying such problem hotspots (Chowdhury et al., 2014). Analysis and evaluation of the SFA then facilitates targeted action for redesigning the system to increase the efficiency of P use, improve

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environmental sustainability and overcome P supply vulnerabilities (Brunner, 2010).

Detailed P budgets for UK agriculture have been published (CAS, 1978; Withers et al., 2001), and more recently, assessments of P in the UK food system using SFA models have been undertaken for the years 2005 (van Dijk et al., 2016) and 2009 (Cooper and Carliell-Marquet, 2013). However, the financial crisis of 2008 led to a 850% price hike in P fertiliser prices in 2009, creating a significant drop in P fertiliser usage rates for that year (Defra, 2019) suggesting agronomic P input flows for the more recent UK model are likely unrepresentative of the true national picture. Furthermore, there are significant differences in data sources and assumptions used to inform P flows between the two previous P SFA models which creates large discrepancies between comparative flows. For example, different data assumptions used to calculate the volume of food and feed imported into the UK suggest a near halving of P imported in these sources between 2005 and 2009 which seems improbable. Additionally, subsequent management interventions that should affect system P dynamics in the UK have occurred over the last decade. These interventions include tighter regulation on the amount of P that can be used in detergents (The Detergents (Amendment) Regulations 2016), continued investment in sewage treatment by the waste water industry (Environment Agency, 2019), and the introduction of initiatives to reduce the impact of agriculture on water quality, such as Catchment Sensitive Farming (CSF) in England (Davey et al., 2020) and equivalent schemes in the devolved administrations (Wales, Northern Ireland and Scotland). Combined, these factors mean that a new assessment of P in the UK food system is needed.

In this paper, building on an SFA model developed for the regional food system in Northern Ireland (Rothwell et al., 2020), we correct previous discrepancies in data sources and assumptions in collaboration with industry and government to provide a comprehensive up-to-date analysis of P flows in the UK food system. Uniquely, we include new detailed analysis of sub-system P flows in the livestock, crop, food, feed, fertiliser, waste management and bioenergy sectors. The key objective is to provide a robust evidence base to underpin new government targets towards a cleaner, healthier and more biodiverse environment, identify key areas of P inefficiency and loss, and highlight priority areas for policy and management interventions that would improve the sustainability and resilience of P use in the UK food system.

2. Methods

2.1. System boundary and data sources

This SFA has been developed for the UK and includes all major flows and stores of materials containing P that are relevant to the national food system. The system boundary is the geographical border of the UK so considers England, Scotland, Wales and Northern Ireland as one single food system. The model has been developed for a single year (2018), which is a typical SFA approach and considered an appropriate timescale for minimizing the impact of data fluctuations in anthropogenic systems (Brunner and Rechberger, 2016). This year had the greatest availability of complete data sets at the start of the analysis; where 2018 data were not available, the most recent years complete data were used. Data for material flows within the food system were taken initially from published national statistics, industry reports or scientific literature. Where gaps in the data remained, expert knowledge was sought from relevant stakeholders. The material P contents and any associated P co-efficients were taken from various published sources or expert opinion where necessary. The flows of P are derived from multiplying the material mass flow by the assigned material P content and are presented as Gg of elemental P per annum (Gg P/a). Full details of flow descriptions, and data sources for P contents and material flows are available in Supplementary Material (Table S1).

2.2. SFA model and data uncertainty

The SFA model was constructed and visualized using the free software STAN which applies the data uncertainty for data reconciliation, error propagation and final model balancing as described fully by Cencic and Rechberger (2008) and Cencic (2016).

Assignment of data uncertainty to the P flow data follows the systematic approach developed by Laner et al. (2016) and Zoboli et al. (2016). Briefly, data quality indicators *Reliability*, *Completeness*, *Composition*, *Temporal correlation, Geographical correlation*, *Further correlation* and *Expert opinion* are assigned an evaluation score which are translated into a final co-efficient of variation for that flow data. A fuller description of the method used is given in Rothwell et al. (2020).

Full details of the uncertainty assigned to different flows, details of flow input data and STAN reconciled output flow data are available in Supplementary Material (Table S1). In this study the mean value change from inputted flow data to the STAN reconciled data was 0.01% which suggests a very high confidence in the input data.

2.3. System description

The current model comprises of 10 processes, 4 sub-processes and 109 flows. The model flows represent the movement (import, export, flow between processes, and losses) of significant P containing materials that make up the UK food system. There are some P flows that have non-food system applications that subsequently become part of the food system and therefore must be considered, thus P used in detergents and for treating drinking water to prevent plumbosolvency that ultimately ends up in the waste water treatment system and potentially recycled back to agricultural land is included in this analysis. The model processes represent the major components of the national food system and are described briefly below:

1) *Animal Husbandry and Aquaculture*, considers all the agricultural livestock used for food production and farmed fish production and all associated inputs and outputs which include grass, fodder, processed feed, food products and manure. Livestock products destined for meat are considered as liveweight as slaughtering is accounted for in *Food, Feed and Fertiliser Processing.*
2) *Crops and Grass* is essentially the farmed soil with all associated P inputs from fertilisers, manures and secondary P sources and all P offtake in crops and grass produced for human and animal consumption, and biofuel and bioenergy production. This process includes a stock that represents the annual accumulation of P in the UK agricultural soil which is essentially the balance between the soil P inputs and the P offtake in crops and grass and P losses to water.
3) *Food, Feed and Fertiliser Processing* manages all the national imports and exports of food, feed and fertiliser associated with the food system, as well as linking UK agricultural production with consumption and waste management. Industries that represent the processing and manufacture of food, feed and fertilisers as well as the retail sector (e.g. supermarkets) in the UK are included in this process.
4) *Consumption* represents all food purchased by the population in both households and the food service sector (which is all food prepared outside the household e.g. restaurants, takeaway, hospitality, schools, hospitals). This process also includes P contained in detergents used by the population and P in drinking water.
5) *Waste Water Treatment* represents the treatment of sewage generated by the 96% of the population connected to and treated by the national sewer network (Defra, 2012), as well as a smaller amount from food processing and industry.

6) *Septic Tanks* handles the sewage from those properties not connected to the national sewer network and utilize small, private waste water treatment facilities (approx. 4% of the population).
7) *Waste Management and Bioenergy* represents the management of domestic, industrial (principally food processing) and agricultural wastes and by-products generated within the food system. In the UK this is usually by the repurposing of useable by-products to other sectors, or for waste - incineration, anaerobic digestion (AD) or composting, with subsequent waste products being recycled to secondary uses or disposed in landfill. This process also includes industries that handle biofuels and bioenergy crops grown specifically for energy production, and subsequent managing of by-products and wastes. The model only considers large commercial AD plants that handle waste collected from domestic and commercial sources. There are many smaller, on farm AD plants that use mainly manures and crops grown locally as feed stock. The products (and P) from these AD plants are usually returned to the land on the same farm, or locally, so represent a local cycle. Therefore omitting these from the model does not alter the national P balance (Rothwell et al., 2020).
8) *Landfill & Construction* is the final disposal destination for those P containing waste materials that are not recycled in the food system. Ashes from incineration can be landfilled or used as aggregates in construction.
9) *Waterbodies* (fresh and coastal) represents the total annual loss of P from our food system to our fresh and coastal waters, predominantly point source P losses from waste water treatment and diffuse P losses from agriculture.
10) *Non-food and Domestic Markets* acts as a sink that handles agricultural and waste management by-products that are used for

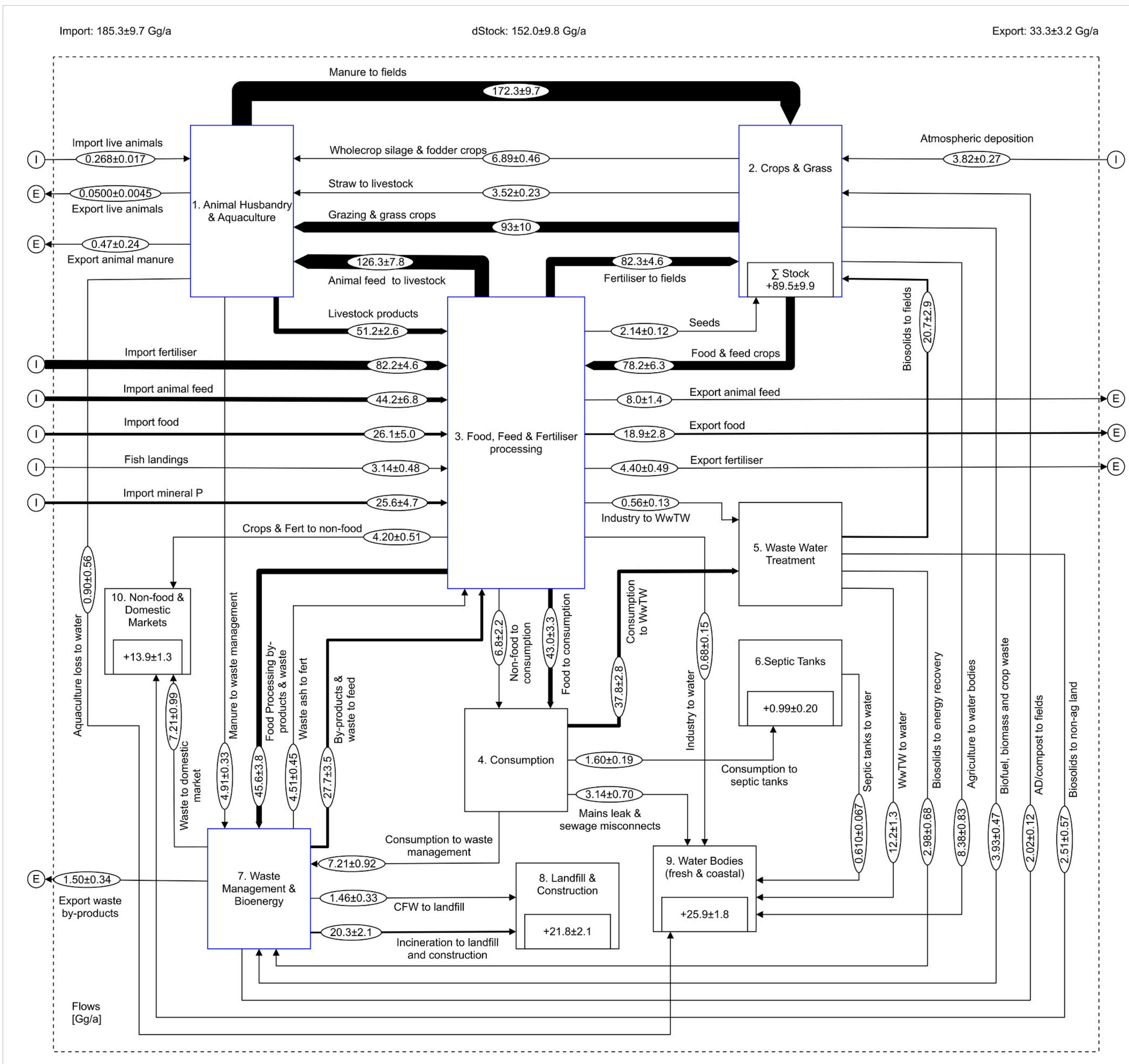

Fig. 1. Phosphorus Substance Flow diagram of the UK food system for 2018, all values are shown as Gg/a ±standard error and have been reconciled by STAN. Values within a process are the annual accumulation within that sector. Line thickness of the flows is proportional to the magnitude of the flow value.

non-food purposes, for example wheat used for starch production and animal slaughter waste used for fertiliser used in household gardens and pet food.

The processes *Animal Husbandry and Aquaculture*, *Crops and Grass*, *Food, Feed and Fertiliser Processing,* and *Waste Management and Bioenergy* processes have been expanded to include detailed sub-processes.

3. Results

3.1. National P SFA

As expected for a country with no minable P reserves of its own, the UK is a net importer of P to support its food system. The UK P SFA (Fig. 1) shows that total imports of P into the UK food system in 2018 were around 185 Gg (2.79 kg P cap^{-1} yr^{-1}). Around 33 Gg of P was exported (0.5 kg P cap^{-1} yr) leaving a national 'stock' of around 152 Gg P which is utilized in the UK food system. This equates to a net national import of 2.29 kg P cap^{-1} yr^{-1} which is slightly higher than the 1.9 kg P cap^{-1} yr^{-1} reported by Cooper and Carliell-Marquet (2013) for the UK in 2009, though this may be explained by the short-term P fertiliser price spike and reduced fertiliser use and import for that year (Defra, 2019). van Dijk et al. (2016) reported 3.32 kg P cap^{-1} yr^{-1} for the UK in 2005, the difference with our figure being accounted for by greater fertiliser (105 Gg P) and food (55 Gg P) import estimates. Phosphorus fertiliser use in the UK has decreased by around a quarter since 2005 owing to improved agricultural practice (see Fig. 6.). However, differences in food P import estimates are likely due to different data sources and methodological assumptions and highlight the challenge of making direct comparisions between different food system SFA models. The UK compares favourably with the rest of Europe, with Ott and Rechberger (2012) reporting an average 4.7 kg P cap^{-1} yr^{-1} for the EU 15 and van Dijk et al. (2016) reporting a value of 4.4 kg P cap^{-1} yr^{-1} for the EU 27.

Mineral inorganic fertilisers were the largest import of P into the UK food system (82.2 Gg P) which represents 44% of all food system P imports, imported animal feed contained 44.2 Gg P (24% of total), food for human consumption was 26.1 Gg (14% of total) and mineral P was 25.6 Gg (14% of total). The mineral P is used to supplement P intake in livestock feed (17.3 Gg P), as food additives in the food processing industry (1.7 Gg P) (which is estimated to be around half total food additive P intake, the rest being already embedded in imported processed food), for treating drinking water to prevent plumbosolvency (3.8 Gg P), and as P in detergents (2.6 Gg). P exported from the food system was mostly in food (18.9 Gg P) along with small amounts of exported animal feed (8.0 Gg P) and fertiliser (4.4 Gg P).

The P efficiency of a food system can be estimated by dividing all food system P imports (i.e. fertiliser, food and feed and associated mineral P additives) by the P contained in food produced for consumption and any exportable commodities such as fertiliser and livestock feed (van Dijk et al., 2016). If the non-food P used for plumbosolvency and detergents are ignored, UK national food system P imports in 2018 were 174.6 Gg P which produced 43 Gg P in food for UK markets and 31.3 Gg P in exported food, feed and fertiliser which represents a national food system P efficiency of 43%. Food system P efficiencies for the EU 27 vary between 19% for Spain and 111% for Slovakia, with an overall EU 27 mean of 51% (van Dijk et al., 2016), so the UK appears to have a lower than average food system P efficiency than other EU countries. By comparison, applying the same food system efficiency approach to data from the US (Suh and Yee, 2011) suggests an efficiency of 36% for 2007.

3.2. Agriculture

3.2.1. Animal husbandry and aquaculture

A temperate, maritime climate means that grassland is the most dominant type of farmland in the UK, particularly in the west of the country, covering 70% of the UK agricultural area. This means that the livestock sector is a large and important component of UK agricultural production. Total livestock feedstuff P inputs in 2018 were around 226 Gg P comprising of feed, grass, wholecrop and fodder and mineral P additives and supplements. The animal husbandry and aquaculture sector output in meat, milk, eggs and wool contained 51.2 Gg P when meat P output is expressed as live weight (LW) representing a P efficiency (converting P intake into P embedded in product) of 23%. If meat output is considered as dressed carcass weight (DCW) P efficiency is 16% (Table S2.) which remains consistent with previous estimates for 2009 by Cooper and Carliell-Marquet (2013) and this low value is typical of food systems with a large ruminant population (Rothwell et al., 2020).

A more detailed breakdown of inputs and outputs by livestock type is shown in the livestock sub-process (Fig. 2). Cattle and poultry were by far the largest consumers of manufactured feed P, eating 50.5 Gg (40% of total) and 46.3 Gg (37% of total) feed P respectively. Cattle were also the largest consumer of grass P eating 71 Gg (76% of total), with sheep accounting for 21.2 Gg (23% of total).

The model assumes a stable livestock population, so P not embedded in the product output is excreted in manure which contains around 177.7 Gg P. The livestock sub-process (Fig. 2), shows that cattle dominate manure P output with their excreta containing 106 Gg P (60% of total). Poultry are the next largest producer of manure P containing 29.5 Gg P (17% of total). Sheep produce 28.3 Gg (16% of total) of excrement P though most of this will be directly deposited during grazing. Pig (10.9 Gg P, 6% of total) and other livestock (2.96 Gg P, 2.5% of total) manure accounts for the remaining P excretion. Nearly all livestock manure P is returned to agricultural land (172.2 Gg) either directly deposited by grazing animals or collected and spread from housed animals, around 5.4 Gg manure P is processed or exported.

Associated P efficiencies by the different livestock types are shown in Table S1. The pig sector has the highest P efficiencies of 29% DCW, and poultry efficiency is only slightly lower at 22% DCW. The ruminant livestock have lower P efficiencies of 15% DCW for cattle and 5% DCW for sheep. This difference is most likely caused by the animal life cycle, for example, broilers produced for meat are continually growing during their typical six-week life cycle (Leinonen et al., 2012), and therefore absorbing much of the consumed feed P in their bones and tissues. Conversely, much of the ruminant population (e.g. dairy cattle) are mature animals that are in P homeostasis (Horst, 1986) and therefore excreting a higher proportion of consumed P.

Aquaculture is only locally important in the UK, mostly represented by the Scottish salmon farming industry and therefore P flows are small compared to other livestock production systems. Feed inputs were 1.83 Gg P, with 0.94 Gg P contained in the fish produced representing a P efficiency of 51%.

3.2.2. Crops and grass (soil system)

Like most countries with intensive agricultural systems, P inputs to the soil are used to maintain crop and grass growth in the UK, mostly as livestock manures and mineral fertilisers. Total elemental P input into the UK agricultural soil surface in 2018 was 283 Gg P. Manure was the largest P input supplying 172 Gg P (61% of total) and mineral fertiliser was the second largest, supplying 82 Gg P (29% of total). Total P offtake in all the crops and grass that were grown was 185 Gg meaning the P uptake efficiency of the UK soil based agricultural system was 65%. A further 8.4 Gg P were estimated to be lost to waterbodies via diffuse pollution from agricultural soil, which represents 32% of total national losses to water (Fig. 1). This represents a total surplus application of 98 Gg P, or 8 kg ha^{-1} across the agricultural area of the UK, excluding rough grazing, which is slightly higher than published national estimates of 7 kg ha^{-1} for 2018 (Defra, 2020). When the 8.4 Gg P diffuse losses to water are accounted for in the soil balance, this means around 90 Gg P actually accumulated in UK agricultural soils in 2018 (7.3 kg P ha^{-1} excluding rough grazing).

The national picture, however, hides large differences in P surplus

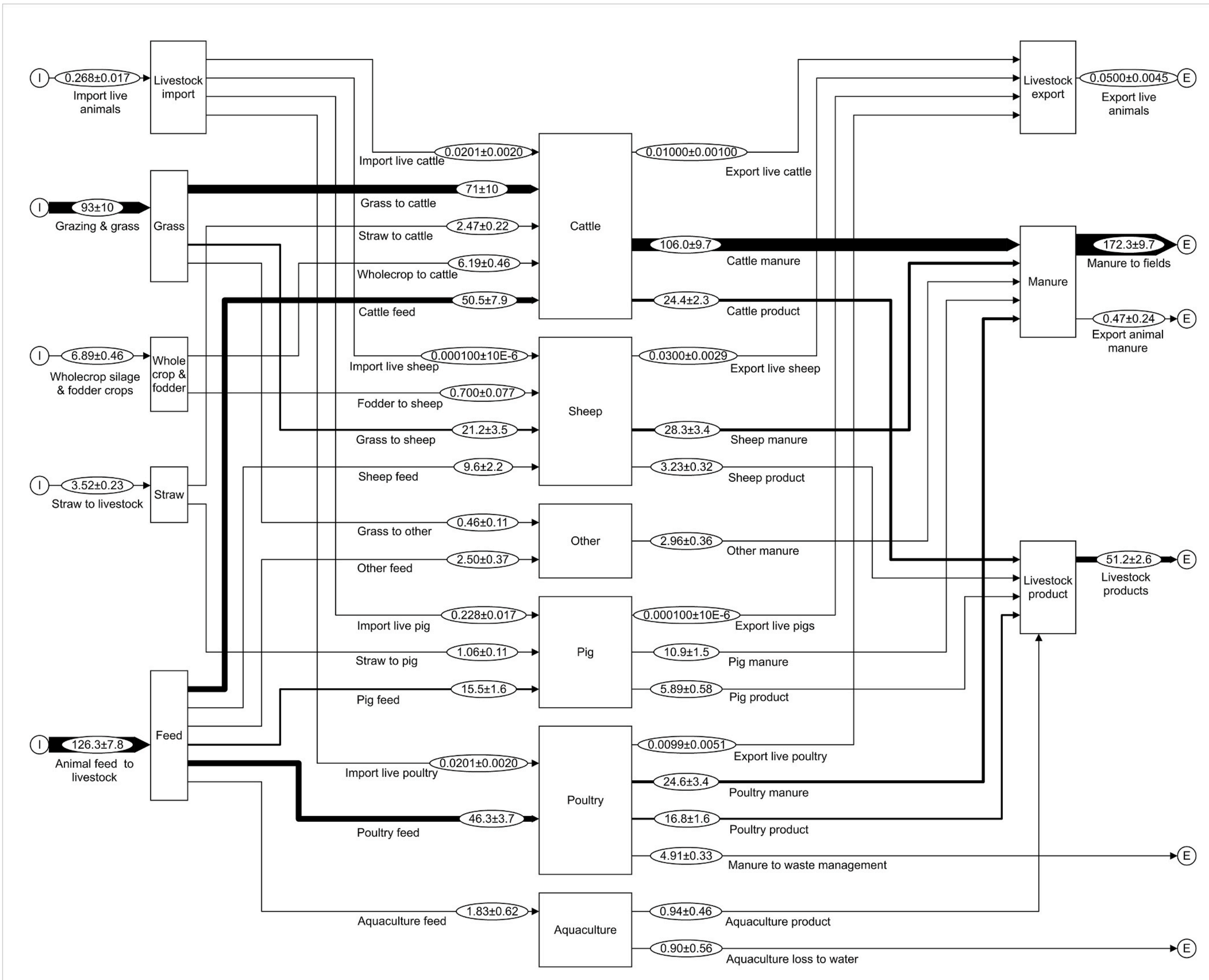

Fig. 2. Phosphorus Substance Flow diagram of the UK *Animal Husbandry & Aquaculture* sub-system for 2018, livestock product is as liveweight for meat animals, all values are shown as Gg/a ±standard error and have been reconciled by STAN. Line thickness of the flows is proportional to the magnitude of the flow value.

and efficiency by different agricultural systems. A breakdown of the soil-based system separated by tillage and grass-based agriculture in the *Crops and Grass* sub-process (Fig. 3) shows that, as might be expected, manure inputs are mostly used on grassland agriculture, principally because that is where the livestock are present. Grassland received 145 Gg of manure P (84% of total manure P) and 26 Gg P from mineral fertiliser (32% of total fertiliser P) and a smaller amount from seeds, food waste and atmospheric deposition, supplying a combined 3.4 Gg P. In total, grassland received 175 Gg P inputs and the grass produced, both grazed and harvested, contained 93 Gg P representing a P efficiency of only 53%. This leaves a surplus annual application above actual requirements of 82 Gg P or 11 kg P ha^{-1} on UK grassland (excluding rough grazing). Estimated diffuse losses to water from grassland were 5.2 Gg P, so when this loss is accounted for, actual P accumulation in grasslands was 76.4 Gg (representing 85% of the UK accumulated soil surplus of 90 Gg P) or 10.4 kg P ha^{-1}.

In contrast, UK tillage agriculture appears more P efficient. These systems received the largest portion of mineral P fertiliser used (56.2 Gg P, 68% of total), 27.1 Gg P from manures and 25.3 Gg P from other sources, total P input to tillage systems was therefore 109 Gg. Total tillage offtake was 92 Gg P giving a P use efficiency of 85% and an annual surplus P application of 16.2 Gg P, or 3.5 kg P ha^{-1} on UK tillage land. Diffuse losses from tillage land were estimated at 3.2 Gg P which means actual soil P accumulation in tillage land for 2018 was 13.1 Gg P or 2.8 kg P ha^{-1}.

3.3. Food, Feed and Fertiliser Processing

The food processing sector is supplied by both UK produced and imported food and the *Food, Feed and Fertiliser* sub-process (Fig. 4), shows in 2018 the UK food processing sector received a total of 133 Gg of P and produced 43 Gg P in food destined for the UK market and 18.9 Gg P in exported foods, representing a P efficiency of 55%. The UK fertiliser industry imported 82.2 Gg P for processing and packaging and utilized an additional 4.51 Gg P from recycled ashes. This produced 82.3 Gg P in fertiliser for use on UK agricultural fields and 4.4 Gg P in fertilisers for export. The animal feed processing sector received and outputted 134.3 Gg P, around 45.2 Gg P of this came from UK grown crops, 44.2 Gg P from imported grains and meals, 27.7 Gg P from UK generated food by-products and wastes and 17.2 Gg from imported mineral P that is either added to manufactured feed or used to supplement P intake on farm. This means 46% of the P consumed by UK livestock is directly imported, 34% is from UK grown crops and 21% from food by-products and waste generated by the UK food processing

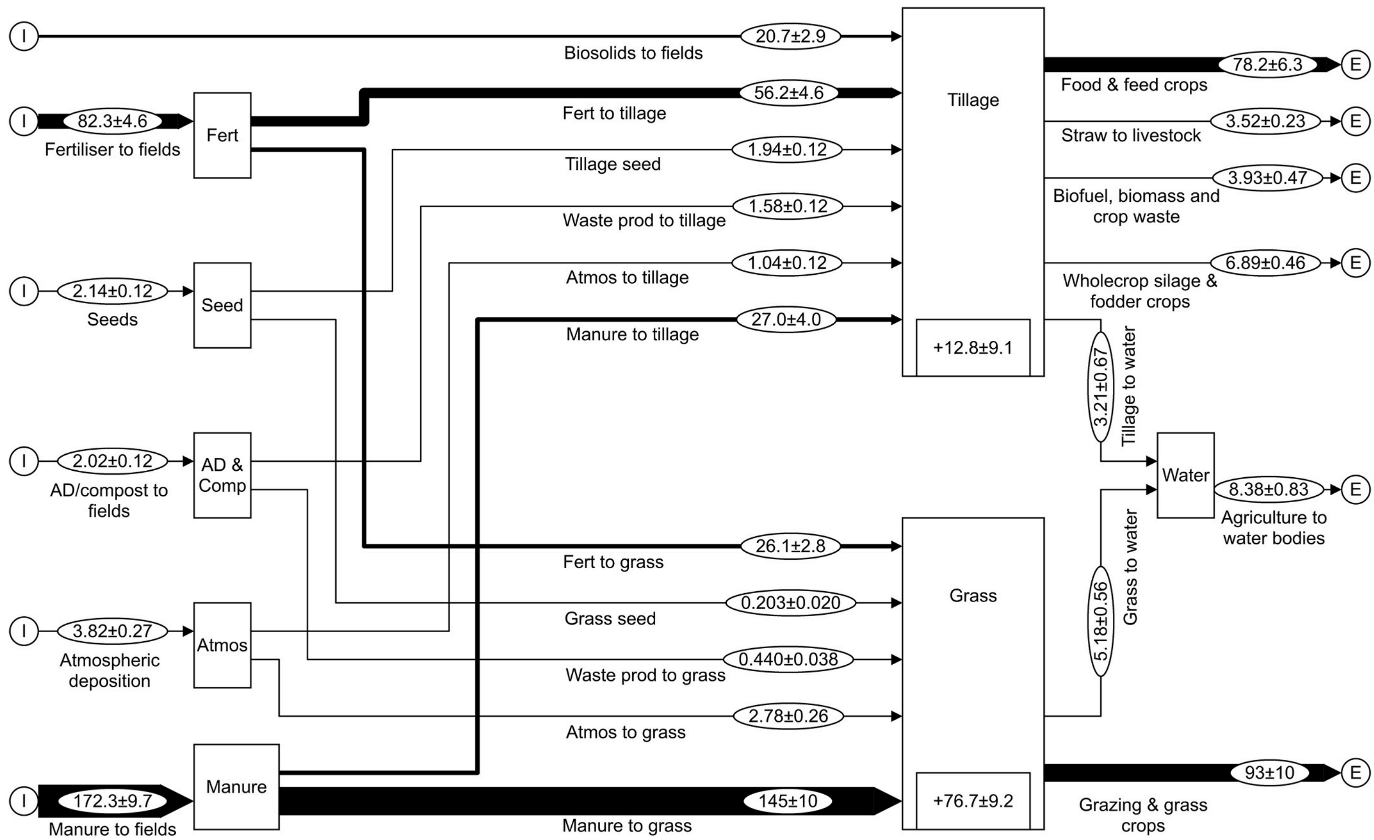

Fig. 3. Phosphorus Substance Flow diagram of the UK agricultural *Crops & Grass* sub-process for 2018, all values are shown as Gg/a ±standard error and have been reconciled by STAN. Values within a process are the annual accumulation within that sector. AD = anaerobic digestion. Line thickness of the flows is proportional to the magnitude of the flow value.

sector.

3.4. Consumption

The SFA estimates that in 2018, purchased food in households and the food service sector contained 43 Gg P (Fig. 1), the equivalent of 1.77 g P cap^{-1} day. Food waste contained around 7.21 Gg P leaving a human P intake of 35.8 Gg P or 1.48 g P cap^{-1} day. The STAN model has increased this from the inputted figure of 1.35 g P cap^{-1} day taken from Forber et al. (2020), making it slightly higher than previous UK estimates of 1.37 g P cap^{-1} day for 1973 (CAS, 1978) 1.21 g P cap^{-1} day in 1987 (Gregory, 1990) and 1.37 P cap^{-1} day in 2009 (Cooper and Carliell-Marquet, 2013). However, estimates using diet survey data are likely to underestimate intake (Forber et al., 2020), meaning the new intake value may be more realistic. The 7.21 Gg P in food waste from households and the food service sector represents 17% of P purchased in food, though household food waste has reduced by 18% since 2007 (WRAP, 2020).

3.5. Waste water treatment

In 2018, human consumption of P in food, P added during drinking water treatment, P in detergents and P from the food processing sector generated around 42 Gg P in waste water. Approximately 96% of the UK population is connected to the national sewer network (Defra, 2012) where effluent is treated at Waste Water Treatment Works (WWTW). However, it is estimated that incorrectly connected domestic waste water systems could divert 5% (2.1 Gg P/a) of household sewage discharge into surface drainage rather than the sewer network (Comber et al., 2013) and would therefore be untreated. WWTW therefore received a total of around 38.4 Gg P coming from human consumption and waste (31.1 Gg P), drinking water (3.56 Gg P), detergents (2.42 Gg P) and a much smaller amount (0.56 Gg P) from food processing. Discharge from WWTW, including that from storm tanks and combined sewer overflows (CSO), is estimated at 12.5 Gg P which represents a removal efficiency of 67%, which is comparable with other UK estimates (Naden et al., 2016). The WWTW P losses represented around 47% of the total estimated loss to water from the UK food system. P removed in biosolids at WWTW is mostly recycled to agricultural land (20.5 Gg P, 79% of total) with much smaller amounts applied to non-agricultural land (2.51 Gg P, 10% of total) or incinerated for energy production and ultimately lost to landfill (2.98 Gg P, 11% of total). Approximately 4% of the population are served by small private waste water treatment systems including septic tanks. The estimated P load and discharge to those are 1.6 and 0.61 Gg P respectively.

Of the total estimated 42 Gg P in waste water generated, 14.9 Gg P (36%) is lost from WWTW, storm flow, CSO, septic tanks and misconnects which represents a significant inefficiency in P recovery. In addition to P losses from WWTW a further 1.2 Gg P is estimated to be lost to the environment from mains water leaking of P treated drinking water (Ascott et al., 2016).

3.6. Waste management and bioenergy

In total, the *Waste Management and Bioenergy* sub-process received 64.6 Gg P in 2018, by far the largest component (45.6 Gg P) of which was food processing by-products and waste (Fig. 5). The most significant use of this is for livestock feed (26.2 Gg P) which represents around 21% of the total livestock P intake from processed feeds. Incineration processes in the UK received a combined 24.9 Gg P, the largest component of this was animal slaughter waste (8.9 Gg P), followed by consumer food waste (4.75 Gg P), poultry manure (4.49 Gg P), biosolids from

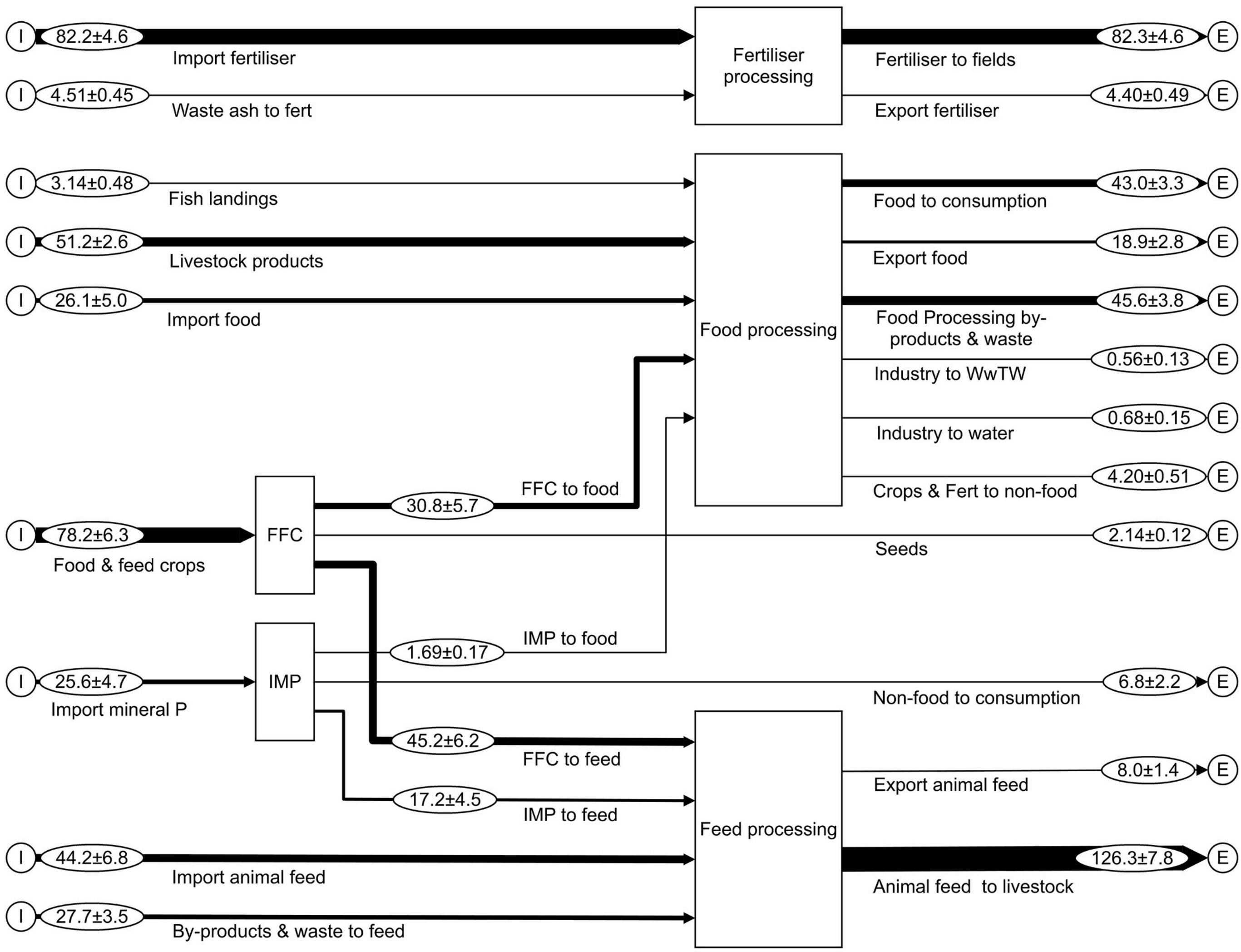

Fig. 4. Phosphorus Substance Flow diagram of the UK *Food, Feed and Fertiliser processing* sub-process for 2018, all values are shown as Gg/a ±standard error and have been reconciled by STAN. FFC = Food and Feed Crops, IMP = Import Mineral Phosphorus. Line thickness of the flows is proportional to the magnitude of the flow value.

WWTW (2.98 Gg P), biomass crops grown specifically for energy production (2.16 Gg P) and food processing waste (1.57 Gg P). Around 4.5 Gg P in the subsequent ashes produced were recycled as fertilisers, the remaining ash assumed to be landfilled (Slorach et al., 2019) or used as construction aggregate (Defra, 2013). Commercial AD and composting accounted for around 2.44 Gg P, most of which is returned to agricultural land (2.02 Gg P). Around 1.46 Gg P from food waste collected from households and the food service sector were directly landfilled, derived from food waste incorrectly disposed in municipal waste collection, or from those regions that do not support separate food waste collection.

Overall, of the 64.6 Gg P by-product and waste P generated by the UK food system, 43 Gg P of was ultimately reused by repurposing or recycling, an efficiency of 66%, and 21.8 Gg P was lost to landfill and construction aggregates. By far the largest P loss to landfill and construction are from incineration ashes (20.3 Gg P).

4. Discussion

This analysis shows that, excluding detergent and plumbosolvency P, the UK imported 174.6 Gg P to meet food system demand in 2018, though a national food system P efficiency of only 43% meant only 74 Gg P ended up in food and exportable commodities. This level of inefficiency is not sustainable if our environment and its biodiversity is to be preserved and vulnerabilities to P supply issues are to be addressed.

The UK national food system requires P to meet food production demand, though theoretically, much of this could be met by what is already in the system. For example, secondary P sources applied to agricultural land (manure, biosolids and food waste) contained a total of 195 Gg P while crops and grass uptake was 185 Gg P. Despite this, we continue to import large amounts of P into the food system due to this inefficiency, meaning that fundamentally there is too much P in the system.

Three significant areas of P inefficiency and loss in the UK food system are identified in this SFA; namely, the agricultural P soil surplus, P loss to water and P sent to landfill and construction. The combined total unused or lost P from these three areas in 2018 was 137.2 Gg P which is equivalent to 74% of the food system P imports. With appropriate intervention and system level change, these three areas represent a substantive opportunity to improve P efficiency and sustainability in the UK food system.

4.1. *Agricultural P surplus*

The over application of P to agricultural soil above actual requirement is the greatest hotspot of inefficiency in the UK food system, amounting to nearly 90 Gg of P accumulating in UK agricultural soils in

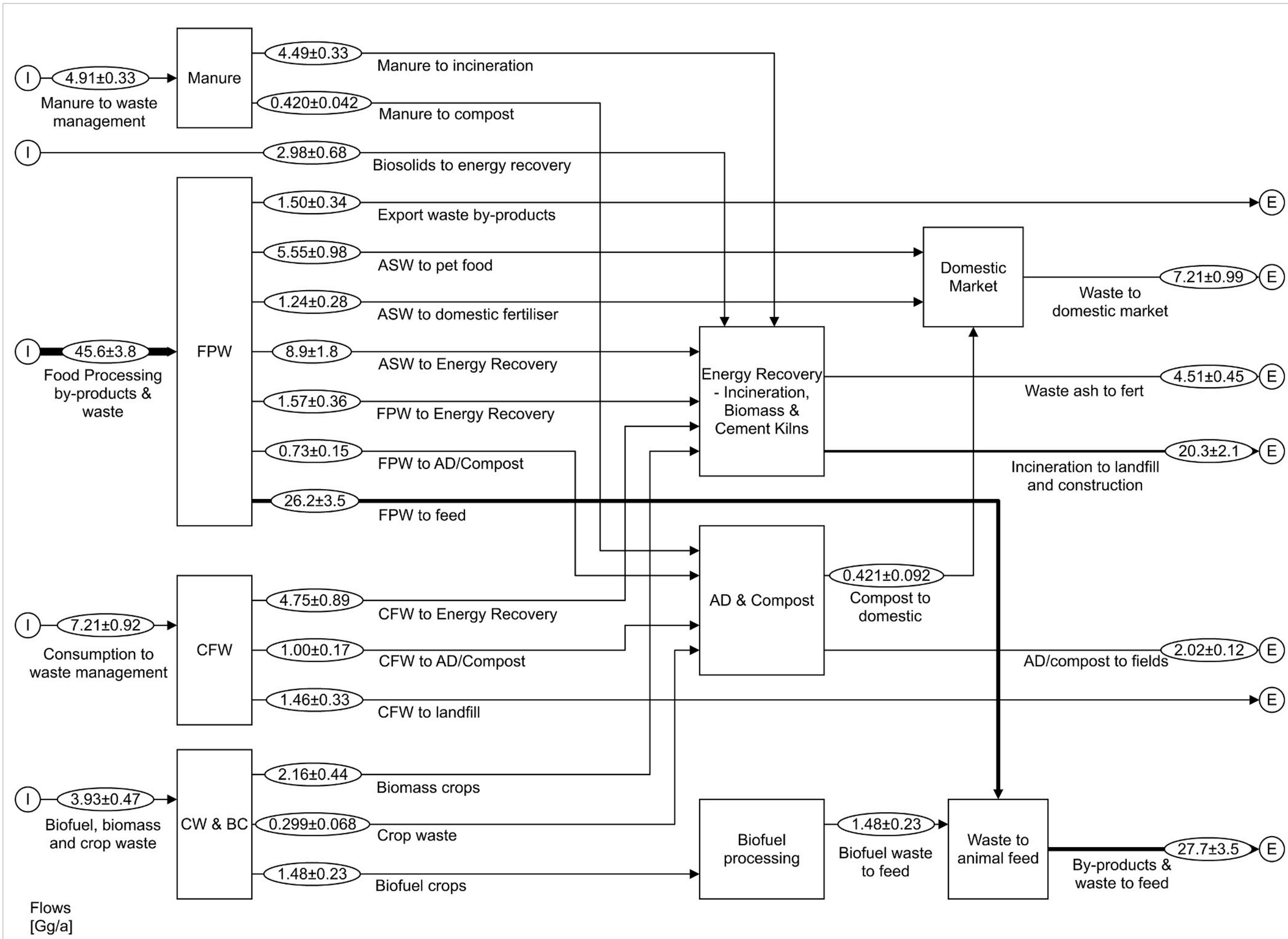

Fig. 5. Phosphorus Substance Flow diagram of the UK *Waste Management & Bioenergy* sub-process for 2018, all values are shown as Gg/a ±standard error and have been reconciled by STAN. FPW = Food Processing Waste, ASW = Animal Slaughter Waste, CFW = Consumption Food Waste, CW & BC = Crop Waste and Biomass/Biofuel Crops, AD = Anaerobic Digestion. Line thickness of the flows is proportional to the magnitude of the flow value.

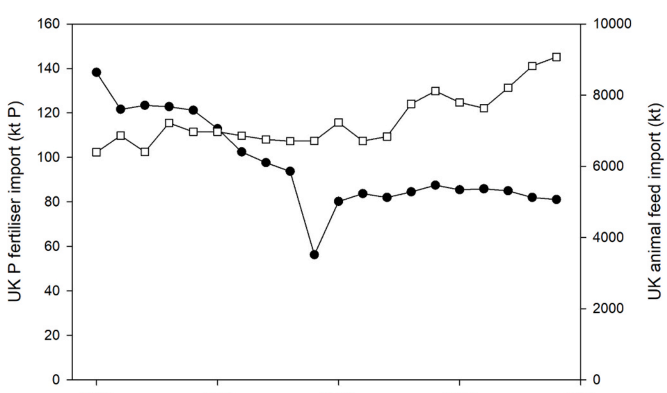

Fig. 6. Annual animal feedstuffs (wheat, soya and maize grain, cereal and legume meals and by-products, and processed feeds, all destined for animal feed) imported into the UK from 2000 to 2019 (open squares) and UK P fertiliser import (filled circles) over the same time period. Wheat and maize feed grain data sourced from AHDB (2020), all other feed data from HMRC (2019) and fertiliser P data from AIC (pers. comm. October 2020).

2018. Not only is this wasteful, but the continued accumulation of P surpluses in landscapes only exacerbates diffuse P losses to water and environmental degradation (Powers et al., 2016). Improved management of P inputs in the UK, particularly within the tillage sector, have seen the national P surplus steadily decline since its peak of ca. 20 kg P ha^{-1} in the 1970s (CAS, 1978; Withers et al., 2001). However, the national P surplus has remained relatively stable at around 6–7 kg P ha^{-1} since 2010 (Defra, 2020).

Agricultural systems or regions with high livestock density usually have a greater soil P surplus (Withers et al., 2020), owing to the P contained in manure being greater than local crop or grass demand (e.g. Nesme et al., 2015). The bulky nature of manure makes it difficult to transport to areas where is could be used efficiently (Bateman et al., 2011; Svanback et al., 2019) and consequently most manure is typically applied locally, often regardless of actual P requirements (Sharpley et al., 1994). The UK national soil P surplus is currently driven primarily by the P inefficiency and surplus in the grassland system (Fig. 3) where manure P input dominates. In England this is happening despite stipulations in the governments 'Rules for farmers and land managers to prevent water pollution' (Defra, 2018) that actual grassland P input requirements must be planned so that fertiliser and manure use does not exceed crop need by soil testing to prevent overapplication.

This local overabundance of P can be partially attributed to the inefficiency of livestock at turning P consumed into P in product (Schipanski and Bennett, 2012), meaning for the UK, 86% of the P fed to livestock ends up in manure that is inefficiently utilized. Grass production *per se* is not necessarily P inefficient, it is the misbalance between input and output that drives the low P efficiency and thus the surplus

(Simpson et al., 2014). P inefficiency in grassland systems is therefore currently constrained by the inherent P biology of livestock. This is not necessarily an issue in itself, and a food system with a significant population of livestock, such as the UK, could support this biological inefficiency, as long as the P excreted in manure can be reused in agricultural production without continuing to contribute to the agricultural P surplus. In fact, if manure P could be managed to effectively replace current mineral fertiliser P use in other sectors, P fertiliser imports could be reduced and the national P surplus would decline. Processes that improve manure handling and distribution (e.g. Hjorth et al., 2010) or, even better, recover fertiliser grade P from manure for integration into fertiliser production (e.g. Dadrasnia et al., 2021; Zangarini et al., 2020) are therefore critical in reducing the national soil P surplus.

On the contrary, steadily increasing animal feed imports in the UK over the last decade are only exacerbating the manure mountain (Fig. 6). In 2005, 46.8 Gg P was estimated to be imported into the UK in animal feed grains, meals and minerals (van Dijk et al., 2016), suggesting a 30% increase to 2018. This SFA shows that 46% of P fed to animals is now directly imported. This reliance on imported P to feed UK livestock is representative of an increasing trend for housed livestock intensification, in particular within the poultry and dairy sectors, which rely considerably on imported high protein crops such as soya (March et al., 2014; Tallentire et al., 2018). At the same time, P fertiliser use and import has been relatively stable in the UK (Fig. 6), meaning our agronomic P oversupply is increasingly coming from manures that are difficult to manage. Targeting improvements in how the P content of livestock manures can be reduced by further manipulating feed P intake (e.g. Ferris et al., 2010; Liu et al., 2019) and breed efficiency (Kasper et al., 2020; Kyriazakis, 2011) are also therefore important. However, where improved manure P management through recovery, redistribution, feed manipulation or improved genetics is not achievable or practical, local reductions or limits on livestock numbers, and therefore imported feed used, will be needed to ensure manure P supply does not exceed grass and crop demand where that P can be realistically used.

4.2. Losses to water

Our analysis estimates around 26 Gg P/a is potentially lost to UK fresh and coastal waters, most significantly from WWTW and agriculture. This analysis brings together estimates of P losses from a range of data sources (see Table S1, supplementary data). Although some of this data is now dated it still represents our best current understanding and critically uses data from processed based models for diffuse agricultural P losses (Davison et al., 2008), providing a significant improvement from previous SFA estimates (Cooper and Carliell-Marquet, 2013; van Dijk et al., 2016). Data for diffuse agricultural P losses for England are the most recent (Elliott, 2019) and account for the impact of diffuse P mitigation measures taken up by the agricultural sector including Catchment Sensitive Farming which has reduced orthophosphate concentrations by around 13% in targeted catchments (Davey et al., 2020). That said, greatest diffuse P losses from agriculture are associated with areas of high livestock intensity (Jarvie et al., 2010; Leip et al., 2015) and, in particular, manure application (Bowes et al., 2015). Furthermore, in the livestock dominated agricultural system of Northern Ireland, there is a demonstrable relationship between P surplus driven by manure input and water quality (Rothwell et al., 2020). Thus, reducing manure-driven soil P surpluses as discussed in section 4.1 are critical to helping address diffuse agricultural P pollution.

Point source losses from waste water management continue to be a major source of inefficiency and cause of widespread river eutrophication (Neal et al., 2010), with 36% of all waste water P generated being lost to water. The total P load to WWTW has reduced by nearly a third since 2009 estimates (Cooper and Carliell-Marquet, 2013). A significant component of this being the reduced contribution of detergent P following the implementation of new regulations (The Detergents (Amendment) Regulations 2016). Comber et al. (2013) estimated the detergent P load to be nearly 10 Gg P (23% of total) in 2011, our new analysis suggests this has reduced to 2.42 Gg P (6.3% of total) in 2018 (Table 1). Continued investment by waste water companies in P removal technology is predicted to further reduce P losses by over a half in England and Wales between 2020 and 2027 which could account for around an additional 5 Gg P/a being recovered and therefore not lost to water (Environment Agency, 2019). However, P recovered from wastewater must be in a form that is agronomically effective if it is to contribute effectively to a circular P economy. Simply removing more P in biosolids and deploying to agricultural land will only increase the national surplus, as with manure P, unless the removed P is an effective replacement for imported mineral P fertiliser. Technologies and strategies for P removal from WWTW are therefore critically important if the P is to be effectively recycled back into the food system (Kirchmann et al., 2017). Most advanced P removal in the UK is currently based on precipitation technology with Al and Fe (Yeoman et al., 1988) which may be cost effective, but P in the resulting biosolids has low bioavailability to crops (O'Connor et al., 2004) and the P is not possible to recover (Environment Agency, 2019). Biological Nutrient Recovery (BNR) offers greater opportunity for P recovery but is again expensive and has not been widely used in the UK (Environment Agency, 2019). Other technologies such as struvite removal have potential to produce effective fertilisers (Talboys et al., 2016) and, although requiring significant treatment infrastructure change, could offer a substantial P fertiliser source for the UK (Kleemann et al., 2015).

4.3. Losses to landfill and construction

Thirdly, the loss of P to landfill and construction aggregates from the UK food system accounted for around 22 Gg in 2018, and our analysis would suggest that co-incinerated ashes are the major source of these P-containing materials (AIC pers. comm. October 2020) accounting for around 20.3 Gg P. Incineration is increasingly being used as a waste management approach to divert biodegradable matter from landfill to generate energy (Foster et al., 2021). However, after incineration, the P remains in the ashes, so if those ashes are landfilled or locked up in construction aggregates, this remains a significant, and possibly increasing, pathway for P loss.

While recycling incineration ashes as agricultural fertilisers is a critical part of a circular nutrient economy, contamination with potentially toxic elements (Bogush et al., 2018), and variable feedstock mean co-incineration ashes are often unsuitable. The 4.5 Gg P in ashes recycled as fertilisers in 2018 (Fig. 5) were mostly from poultry manure, where the known material supply, mono-incineration and therefore composition, aids ease of re-use (AIC pers. comm. October 2020). If co-incineration is to continue as a route for food system waste management then technologies that allow fertiliser grade P recovery from waste ashes are needed (Brandjes, 2019; Leng et al., 2019). Other innovative technologies are being developed that may provide better alternatives for the recapture and circularity of nutrients, particularly P (Huygens and Saveyn, 2018). For example, slaughter waste, which is

Table 1
Details of UK food system waste water treatment phosphorus inputs (not including septic tanks), percentage of load received, discharge and phosphorus removal efficiency for 2018. All data are STAN adjusted and account for estimated losses via miss-connects and mains leaks.

P sources received to WWTW	Gg P	% of load received
Human excreta	31.1	82
Consumer food waste to sink	0.75	1.9
Detergent	2.42	6.3
Plumbosolvency	3.56	9.3
Food processing	0.56	1.5
Total received at WWTW	**38.4**	
Discharge from WWTW inc. storm tanks and CSO	**12.5**	
Removal efficiency %	**67%**	

currently the largest source of wasted food system incineration ashes, could be treated to produce a P fertiliser with similar performance to conventional fertiliser (Darch et al., 2019).

4.4. Policy direction

It is widely acknowledged that there are significant gaps in the governance of P through all sectors of the food system (Rosemarin and Ekane, 2016). In addition, a recent UK House of Commons report detailing the significant contribution of food system P mismanagement, particularly waste water treatment and agriculture, on the health of UK aquatic environments (House of Commons Environmental Audit Committee, 2022) suggests it is therefore timely to introduce new P stewardship options into government policy.

The UK water industry have set ambitious targets for reducing P losses from WWTW by 2027, meaning agriculture could become proportionally more responsible for P pollution (Environment Agency, 2019). Agricultural P pressure on the UK landscape, as this SFA highlights, comes from a system level failure to manage a surplus of around 98 Gg P mostly from livestock manure, driven by external feed P imports for intensive livestock production. Current UK policy focus for controlling agricultural P loss is mostly targeted at reducing the physical transfer of phosphorus to water courses to prevent environmental impact (Davey et al., 2020). However, targeting system level inefficiency and the P surplus at the national, regional and catchment scale could be a longer term solution and should be prioritised while continuing current mitigation measures. Current agricultural guidance in the UK (AHDB, 2017) should in theory minimise P overapplication and surplus from manures by recommending only applying P that is needed, but this is targeted only at the farm scale. This will not address the scale of inefficiency and overabundance of P within the wider UK food system, meaning that P surplus is inevitable in livestock-dominated areas, i.e. it is a system level failure that requires policy-driven system change and governance of P beyond the farm-gate to correct (Withers et al., 2018).

Any new UK policy direction for addressing system P inefficiency and surplus should also enable a reduction in imported, mined primary P resources, thus reducing national food system reliance on, and vulnerability to, external P supply. Policies that promote the physical transfer of manure between sites of production and arable agriculture are being tried in other countries, but struggle with the spatial separation of livestock manure production and arable agriculture (Wei et al., 2021) that is also problematic in the UK (Bateman et al., 2011). Although recycling manure has the added co-benefits of utilising manure nitrogen (N) and carbon (C), it is not practical over distance.

New policy initiatives towards a circular nutrient economy could provide the incentive and infrastructure needed to manage manure and nutrient separation at an appropriate scale to facilitate the recovery of fertiliser grade P (and N) that can be integrated into the UK fertiliser market. Critically, however, recovered P must directly replace imported fertiliser P if it is to address the national P surplus and increase UK food system P efficiency. This would require continued engagement and support from the fertiliser industry and other food system actors (Sarvajayakesavalu et al., 2018) with recovered secondary P being increasingly regarded and used as a viable P resource and substitute for imported fertiliser. Manure P recovery can be economically viable providing it is done at the right scale (Martin-Hernandez et al., 2022) and similar strategies apply to other waste management materials such as food waste (Campos et al., 2019) and biosolids (Cieslik and Konieczka, 2017). However, it is the current lack of policy and governance that is hindering wider deployment of these technologies and the spawning of a circular nutrient economy (Desmidt et al., 2015).

New policy direction that significantly increases the circularity of recovered mineral P has the potential to significantly alter UK food system P dynamics, provide a long-term control over P pollution from UK agriculture and increase resilience to potential future disruptions in P supply. Future work will examine the impact of secondary P cycling policy on the national P surplus, national food system efficiency and potential P losses to water through SFA scenario analysis.

5. Conclusion

Using best available data sources secured by collaboration with industry and government, our SFA has produced a definitive assessment of food system P stocks and flows to help benchmark future progress in meeting environmental policy targets in the UK. Although the UK food system is reliant on imported P, the national food system is only 43% efficient. New policy and actions that target the three key areas of inefficiency (soil accumulation, loss to water, loss to landfill), and in particular address issues around manure P use and management, are needed if the UK food system is to become more sustainable in its future P use and adapt to an uncertain P future. Policies that target the recovery of P from secondary sources (manure, biosolids, food waste) are critical to addressing this inefficiency. However, only by effectively replacing imported mineral fertiliser P, will the national P surplus decline and efficiency improve. Recycling alone without consideration of the efficiency of their use will only add to the national surplus. This SFA has also highlighted significant differences in P efficiency between different sectors of the UK agricultural system. Significant regional differences exist in UK agriculture and land use type, suggesting that regionally appropriate assessment and interventions may also be needed.

Author statement

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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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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.115021.

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