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Water Resource Balancing: Is a closed loop system possible that enables sustainable rural water treatment?

Published by CREW – Scotland's Centre of Expertise for Waters. CREW connects research and policy, delivering objective and robust research and expert opinion to support the development and implementation of water policy in Scotland. CREW is a partnership between the James Hutton Institute and all Scottish Higher Education Institutes supported by MASTS. The Centre is funded by the Scottish Government.

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Please reference this report as follows: Hough, R.L., Amponsah, N., Troldborg, M., Allan, R. & Stutter, M. 2016. Water Resource Balancing: Is a closed loop system possible that enable sustainable rural supplies? CRW2014_06. Available online at: crew.ac.uk/publications

Dissemination status: Unrestricted

Cover photograph courtesy of: Matteo Tamburini, Richard Allan, The James Hutton Institute
ISBN: 978-0-902701-44-1

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The EU has resource efficiency as a priority going forward. The Roadmap to a Resource Efficient Europe (http://eur-lex.europa.eu/legal-content/EN/TXT/?uri=CELEX:52011DC0571) outlines how we can transform Europe's economy into a sustainable one by 2050. It points to green growth and circular economy principles as strategically important to delivering success. More recently, the Scottish Government published the consultation document "Making Things Last" which had specific focus on potential growth of the circular economy within Scotland (http://www.gov.scot/Resource/0048/00484140.pdf).

Within this context water resources are an enabling factor with waste water being viewed as a potential resource. Water supports most activities while at the same time consuming, producing or carrying resources (energy, nutrients and other components) through the catchment management, drinking water and waste water treatment processes. Water resources and how these are managed are critical to informing the Scottish Government's (as well as UK and EU) growth strategy, resource efficiency plans and developing a closed loop circular economy. The project output has identified optimised and sub-optimal resource recovery systems for small scale supplies in rural communities that require a new approach to sustainable water treatment.

Objectives of research

The aim of this work was to carry out a mass balance of energy, nutrients and other potential resources at a range of scales (single house, small community and large urban scale). The focus of the study was on waste water treatment and resource recovery. The study considered whether a closed loop cycle for water and energy was possible in these situations. In doing so, the project team identified technologies, systems and approaches that may need to be adopted to make this possible. Furthermore the project team considered economic factors for selected parameters and approaches as well as taking into account factors such as environmental impact, scalability and life cycle analysis (LCA).

The objectives were to:

- Identify the resources of interest through a literature review
- Gather and analyse data on the stocks and flows of the identified resources in the chosen process aspects, or scale.
- Determine the inter-play between identified resources that determine economic levels of resource efficiency.
- Report on the mass balance of target resources within managed and natural water systems.
- Identify and quantify optimised and sub-optimal systems making suggestions for improvement and innovation potential. Determine sustainability in managed water systems and natural water systems using selected resources as an initial indicator.

Key findings and recommendations

- In terms of energy efficiency, human health an environmental impacts, the large waste water treatment plant operated by Scottish Water is significantly better than the EU average
- Sludge to land AD is more energy efficient than sludge to AD to land; but more adverse environmental impacts. The level of impacts for either disposal route would reduce if the energy mix moves further towards renewable sources.
- Small rural sites – potential to reduce environmental & human health impacts through biomass production

Willow biomass production irrigated with waste water has fewer adverse impacts on human health or the environment that reed biomass production Recommendations to further explore the possibilities for recycling of the dominant P flows in waste systems (dominantly wastewaters in the context of the data explored here) in Scotland are to:

(i) Examine spatial scales in the linkages for where the dominant P is produced in wastes and where it may be consumed by appropriate application to agricultural soils. The latter requires a spatial approach based on crop P requirements and soil test P data that allows the correct targeting of P according to crop requirements and guarding against over application or application to soils sensitive to P leaching to minimise environmental consequences.

(ii) Develop local budgets for target areas where there seems an optimum balance between wastewater and other P sources and favourable agricultural land. Build on these with details of other factors including costs and other aspects (processing chemicals, fuel etc.) to make a more holistic budget approach of associated resources alongside the P budget (the current study deemed this was not able to be done realistically at a national level but is more feasible at a local level).

(iii) Support the above spatial sources and sinks analyses with an appraisal of the technologies available for P recycling, including how these options may make different P sources more/less favourable, how separation/purification techniques (e.g. the becoming widely adopted struvite precipitation) may alter transportation costs, or remove social sensitivities around the usage of the raw products. There may be local advantages to be made through P recovery at small rural scales through septic tanks in particular. Although this study shows that decentralised wastewater systems have relatively small flows of P compared to centralised systems, it is important that the former are amongst the most concentrated P flows entering rural ecosystems. These concentrated wastewater P flows (10-20 mgP L-1) may be easier technically to strip P from than slightly lower concentration centralised effluents.

1.0 Introduction

The treatment of waste water uses significant amounts of energy, both in terms of consumption (primarily for water pumping, but also to drive various treatment processes) and in terms of the embedded energy within the fabric of the treatment plant and any substrates used as part of the treatment process. Reducing this energy demand is very important if we are to work towards a more sustainable future. Having said this, there is little opportunity to reduce energy consumption within the water treatment process itself without major infrastructural changes which also come with significant energy costs. A more achievable strategy (at least in the short- to medium-term) is to try to offset some of the energy expenditure. For example, production of electricity from renewable sources can be used to offset consumption of electricity from non-renewable sources.

Waste water is a largely untapped source of valuable nutrients. Considering that production of fertilisers for agricultural purposes takes significant levels of energy, it may be possible to use resource recovery as a way to offset electricity consumption. The extent to which we can achieve energy off-setting may depend on the number of resource recovery measures that can be built into the system or supply chain. Consider the below series of charts looking at flows of energy, phosphorus (an important and non-renewable nutrient required by agriculture), and clean water.

Figure 1 depicts the current situation showing the balance between losses and gains. Most of the arrows face downwards, indicating losses of these important resources. Working through the series of charts, you will note that the potential to reduce these losses increases as we increase the number of resource recovery methods used within the supply chain. Currently, these arrows are not quantified and must be viewed as indicative.

This project investigated this concept of resource recovery and evaluated the potential to offset energy inputs. This topic is potentially extremely large, so by necessity we looked at a sub-set of what may be possible, with a focus on systems already being used or demonstrated:

The objectives were to:

1. A national (Scotland) scale mass balance was undertaken to estimate the stocks and flows of phosphorus in an attempt to better quantify the flows depicted in the charts above.
2. Undertake a life cycle analysis of a large urban waste water treatment plant to make the following comparisons:
    a. Environmental performance compared to other similar treatment works across the EU
    b. Environmental performance when sludge is applied direct to land compared with a more complex process chain (sludge > anaerobic digestion > land) to understand more fully if our hypothesis depicted in the charts above is likely
3. Undertake a life cycle analysis of a small rural waste water treatment plant where we are currently undertaking pilot-scale research into sustainability options for using effluent for biomass production

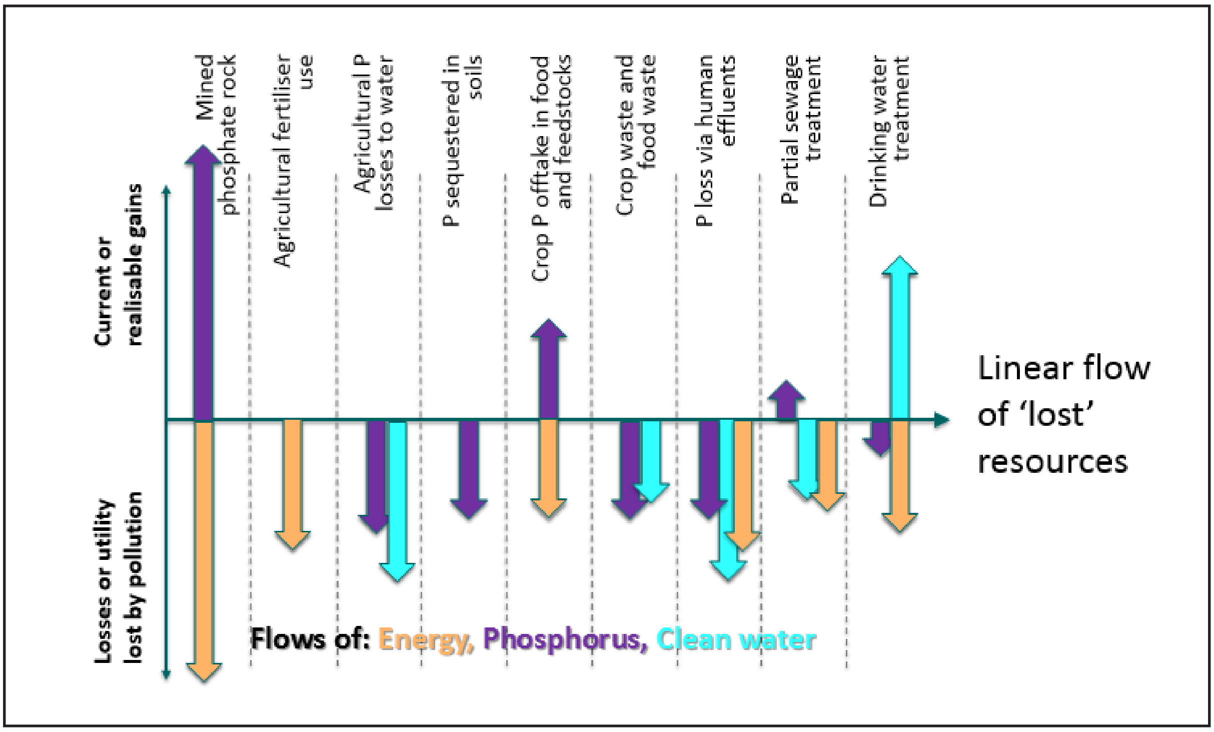

Figure 1. Flows of energy, Phosphorus and clean water

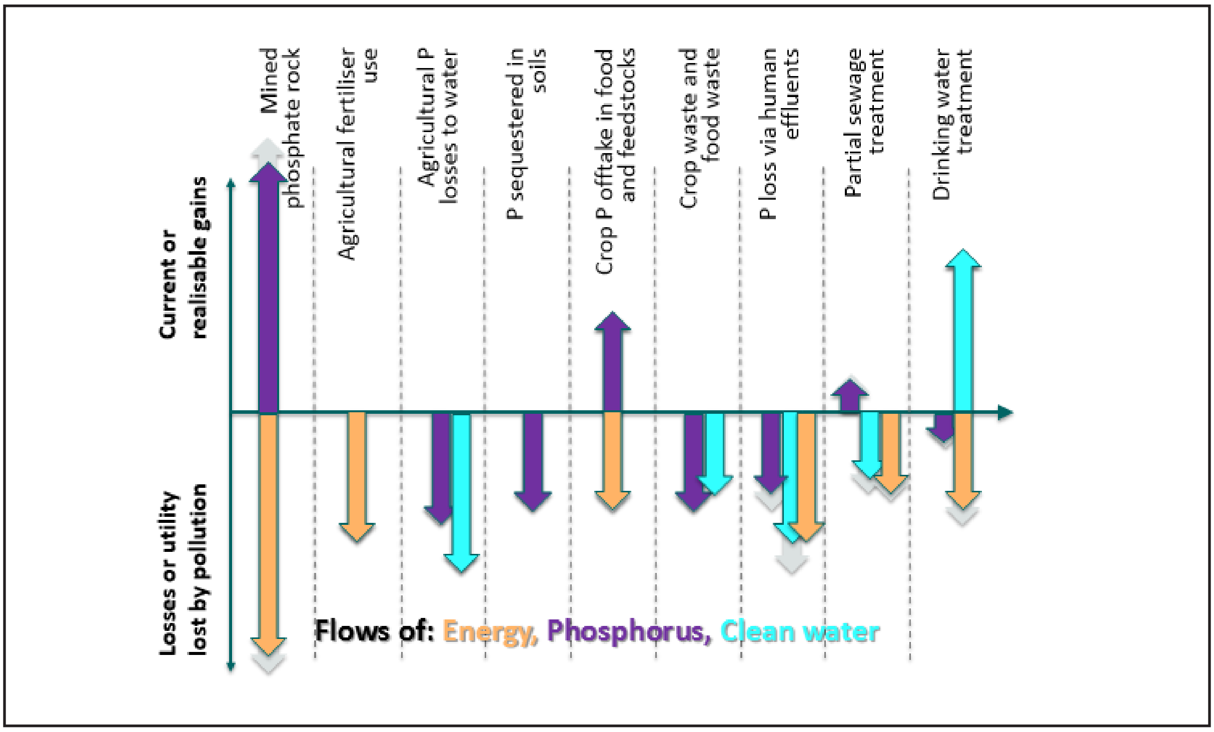

Figure 2. Removing P from dishwasher detergents

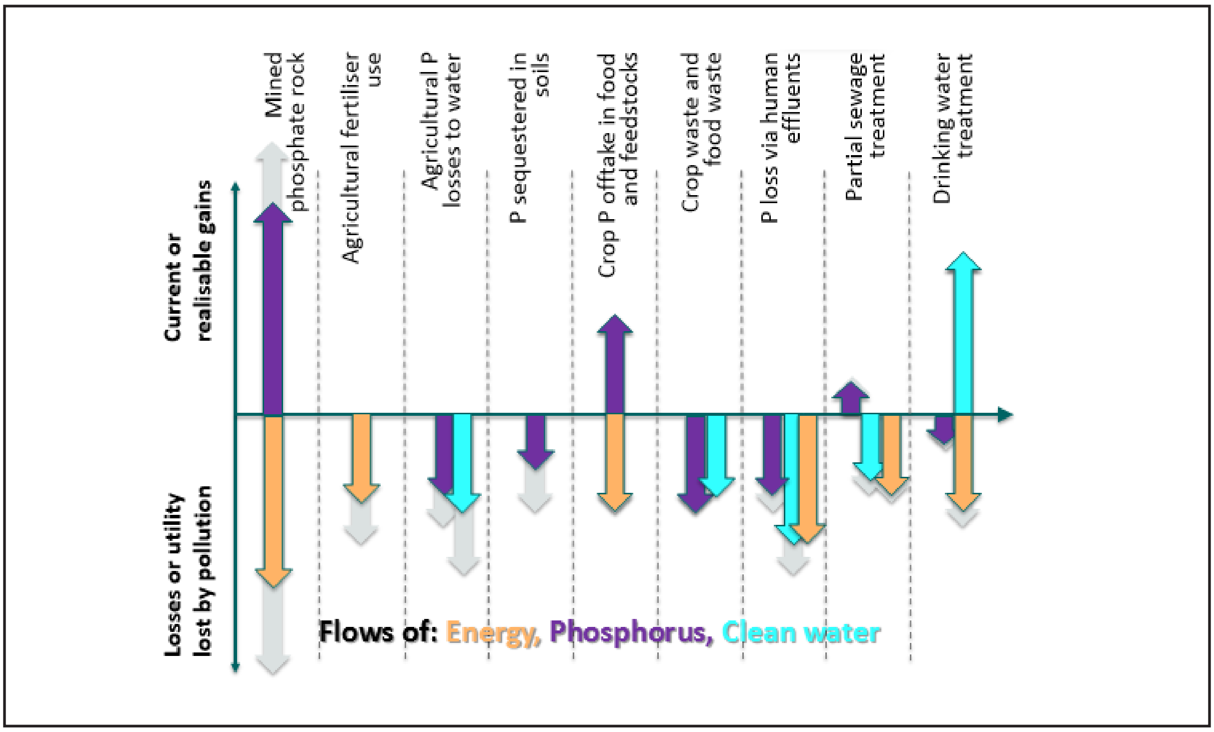

Figure 3. Agri-tech: soil and crop improvements

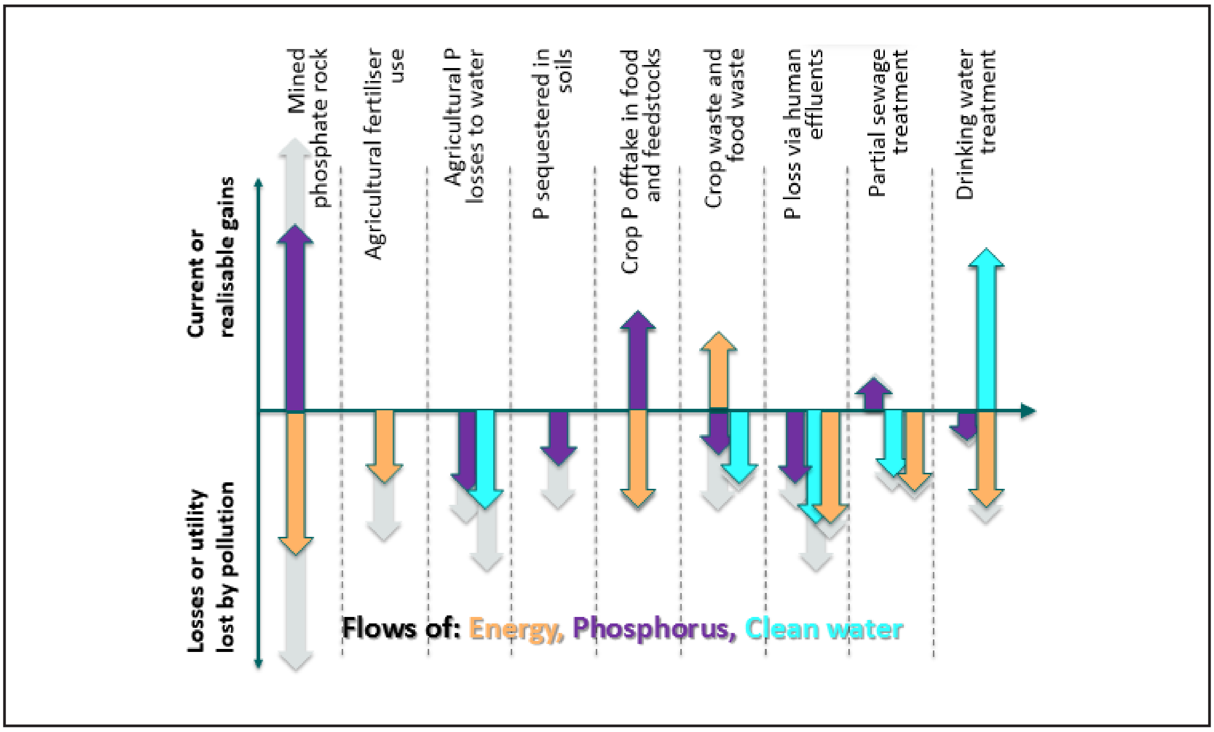

Figure 4. Anaerobic digestion from food and crop waste

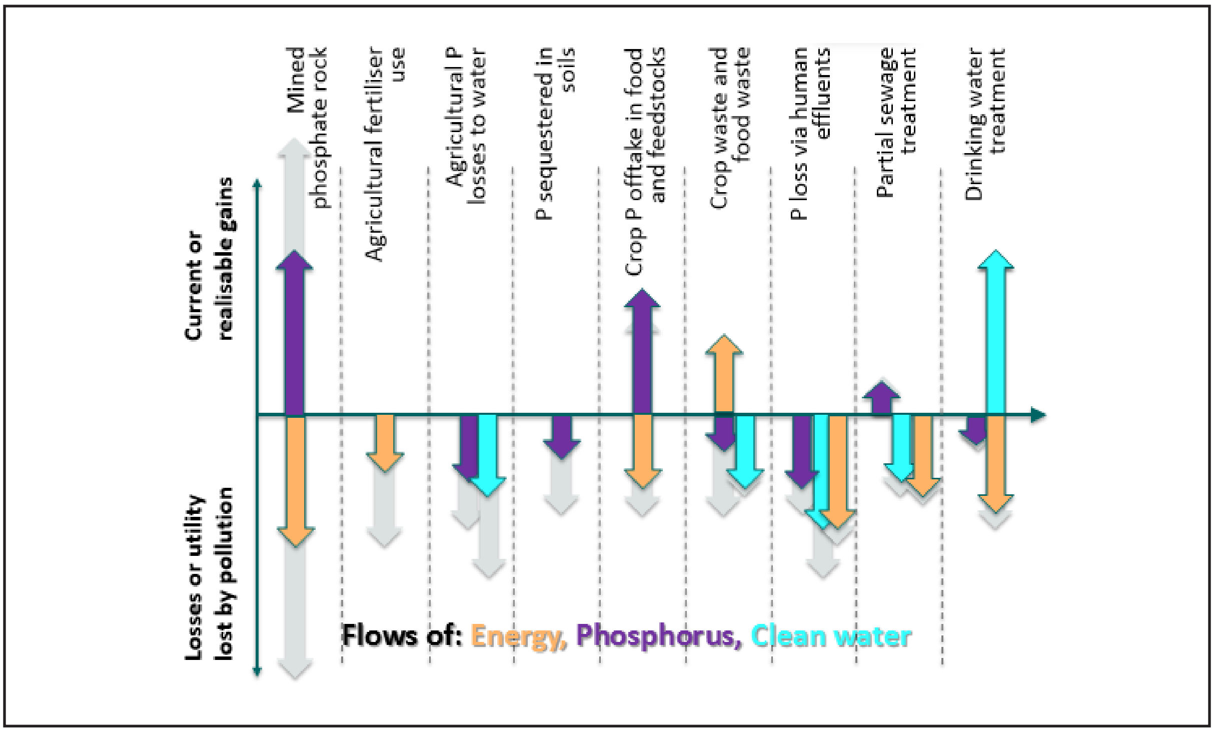

Figure 5. Smart buffer strips

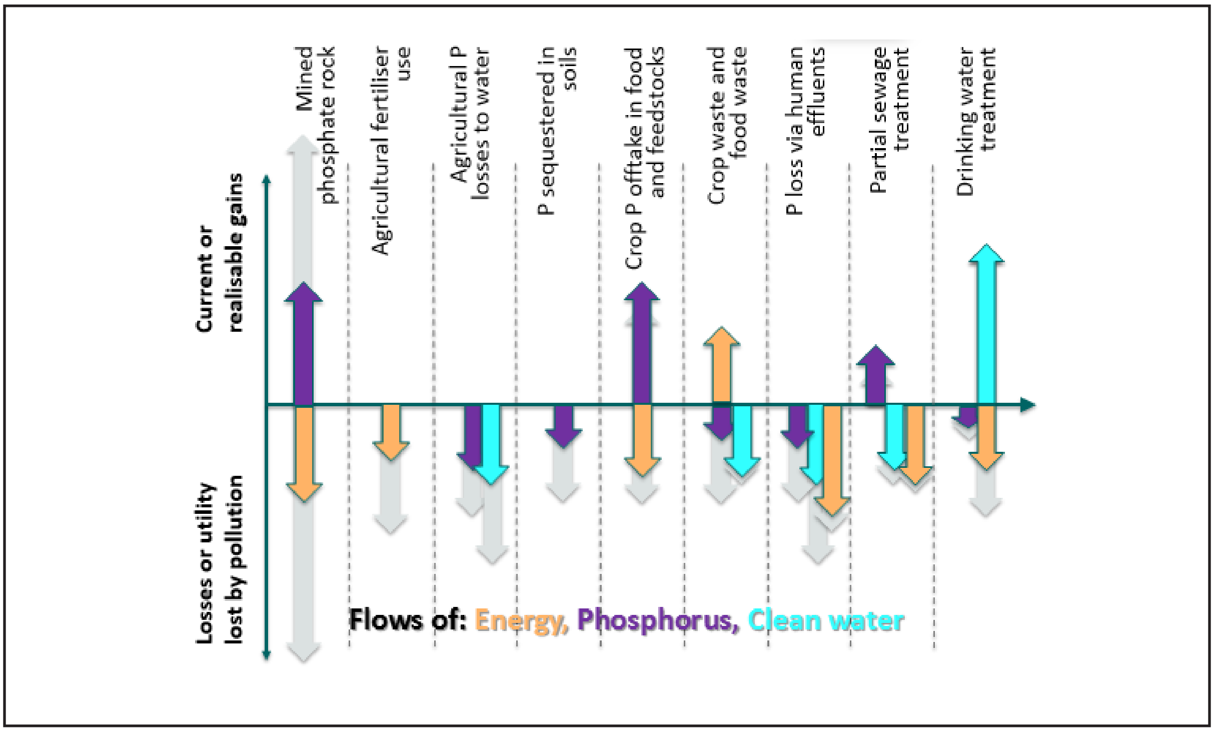

Figure 6. Household P recovery at source

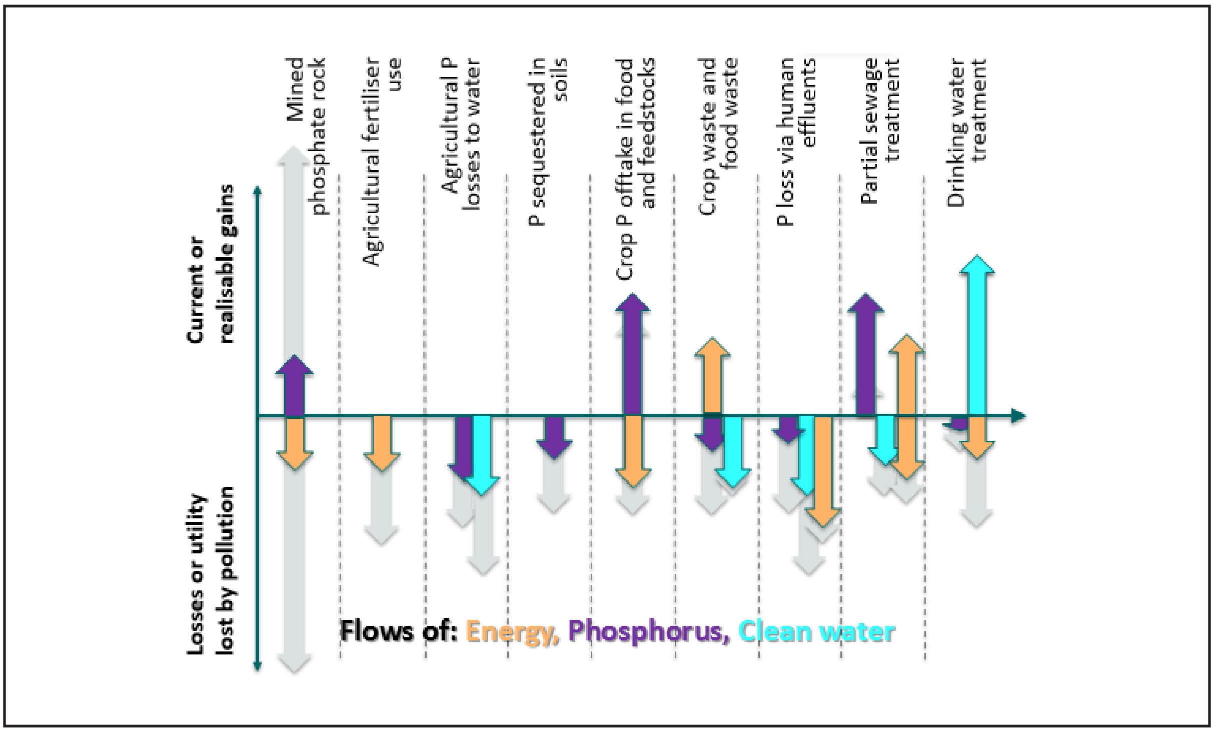

Figure 7. Energy from waste waters

2.0 National-scale mass balance

2.1 Introduction

It is well understood that the non-renewable resource phosphorous (P) provides an important model for resource efficiency studies and 'circular-economy' concepts (e.g. Verstraete et al. 2009; Stutter 2015). The reasons for the focus on this particular element are:

I. the essential nature of the element in the human food chain without which we could not support our agricultural production,
II. reliance on sources of rock phosphate that are of declining quality, economic viability and subject to geopolitics,
III. the inefficiency of capture of P from current human activities, and
IV. the important role of P in eutrophication of fresh waters as a result of poor capture from wastewaters and overuse in agriculture.

As such P makes the ideal model for systems management due to the combined sustainability aspects of improving security of supply by using less more effectively in terms of: inputs, minimising agronomic inefficiencies (e.g. better soil-crop management), recapturing waste flows and removing damaging flows from reaching sensitive ecosystems. In the context of an assessment of sustainable rural communities, it is apparent that whilst the dominant P loss pathways are of an urban nature (e.g. wastewaters from large population centres), the potential for reuse, as partial replacement for chemical fertiliser, is in a rural context.

In terms of the study objectives, whilst we highlight above the case for P recycling, the P budget approach is necessary in answering objective 1, Identify main resources of interest for reuse/recovery/recycling. The methods described below show the process that has been undertaken at a Scotland specific level for the first time, with current (or as recent as possible) data and modelling to answer objective 2, Gather and analyse stocks and flows data on key resources for chosen processes and scales. These data are then used to report on objective 3, Report on the mass balance of target resources within/linking natural and managed water systems.

The focus of this current investigation of P resource efficiency at a Scotland scale is on the internal P cycling processes that cascade from the major inputs such as rock phosphate-derived fertilisers, imported food and feed. The inputs and exports have not been quantified exhaustively, but these would include imports and exports of the fertiliser itself and embedded P in goods such as food, or animals. Nor have associated resources been considered at this national contextual level. For example, the electricity or chemical usages of different treatment and reuse options have not been considered. The primary aim was to evaluate processes particularly in the agri-environmental and water systems areas, looking at both P mass flows and their concentrations. The former highlight the main areas of resource flows, whilst the latter show more or less concentrated flows, which respectively are easier or more difficult for P recovery.

Examples of previous P budget studies include at national level (Austria, Egle et al. 2014; China, Li et al. 2010), sub-national (Phoenix area of Arizona, Metson et al. 2012) and large watershed levels (US Mendota watershed, Bennet et al. 1999). A previous budget at UK level was presented by Cooper and Carliell-Marquet (2013) using data from 2009. Like the present study this earlier UK level study provided an overview of P stocks and flows across the domestic household, waste and agricultural sectors and sought to look at the role of the water industry/sector in recycling opportunities. However, in the context of the overall goals of the current study, these data from the earlier study did not include different scales of decentralized wastewater systems, nor did they use accurate and specific data for Scotland. We took these points as indicating requirement for a first examination of the resource budget for P at a Scotland level as the basis for considerations of local scale resource efficiency options.

2.2 Methods

2.2.1 Budget calculations

Agricultural systems

Fertiliser

The British Survey of Fertiliser Practice (2013) reported both a total usage of chemical fertiliser P in Scotland of 46 ktonnes of P2O5 for 2013 and a breakdown of chemical fertiliser usage per crop type averaged across total crop class areas. The latter were multiplied by the crop areas from Scottish Government's agricultural census taken June 2015. The sum of this across all agricultural land for Scotland of 20.36 ktonnes of P (46.63 ktonnes P2O5) agreed with the value from the 2013 fertiliser usage report. This excluded manures.

Manure

Manure P loads were calculated as: dry matter P content of manure * annual dry matter manure production per head * animal numbers for Scotland.

Manure production was taken from Gerlagh and Gielen (1999), whilst dry matter total P was taken from the British Survey of Fertiliser Practice (2013). Total animal numbers were derived from the 2015 Scottish agricultural census. Total annual manure P production was derived by summing the above calculations for cattle, sheep, pigs and poultry. Other biosolids used as fertiliser inputs were considered separately.

Food and fodder production

Typical yields, P contents and P offtakes for cereals, oil seeds and grass were taken from the Scotland Rural College's advisory Technical Note 668 (SRuC, 2015), as midway in the range stated. Additional P offtake data were derived from industry webpages (e.g. Limagrain for stock feed crops). These were multiplied by the crop areas for Scotland taken from the 2015 census and summed. For grass under 5 years, a P offtake for two cuts of silage (32kg P2O5/ha/year) was applied to half the area and the other half a grazing offtake (3kg P2O5/ha/year that assumed 80% P recycling through manure). For grass over 5 years, an estimate was made of the areas between 80% grazing (at 3kgP2O5/ha/year) and 20% hay production (at 36kg P2O5/ha/year). An estimate of 1.0 kg P2O5/ha/year was applied to extensive grassland.

Food waste

A 2015 UK Parliament briefing paper (http://researchbriefings.files.parliament.uk/documents/SN07045/SN07045.pdf) cites data from Scotland from a 2009 report by Zero Waste Scotland. Zero Waste Scotland (2009) cites 566000 tonnes of food waste from Scottish Households in 2008 and apportions this between 17 food categories with pathways of disposal via (i) the sewer (largely milk and drink down household sinks), (ii) home composting and home animal feed, and (iii) Local Authority collection. However, no figures are given to divide the latter route between landfill and recycling via composting or AD. The AD calculations showed that three AD plants cited as food waste feedstocks consumed

33 tonnes P/year (accounting for 9% of the food waste collected by local authorities). It is a policy aspiration in Scotland to end food and greenwaste going to landfill, but at present a pragmatic routing of that remaining was 41% to landfill and 50% composted and applied to agricultural land. To the latter the amount of 60 tonnesP/year of home composted food waste was added. The P contents of the 17 categories of foods were derived from a USDA National Database (USDA, 2015).

Brewing and distilling industry

Total grain inputs were reported as 40.3 ktonnes for beer making and 1.59 million tonnes for distilling (Zero Waste Scotland, 2015). At a typical fresh mass P content of 8 kg P2O5/tonne of barley grain, this demands 5.70 ktonnes P annually. An expert judgement suggested that the brewing and distilling industries accounted for 50% utilisation of the entire Scottish barley production. This resulted in 3.92 and 1.78 ktonnes P from Scotland's crop and imported grains, respectively. The secondary product waste streams were itemised by the Zero Waste Scotland (2015) report and attributed between animals feeds (spent grain, hops, yeast, trub from brewing and spent draff, pot ale, dark grains from distilling), brewery keiselguhr to landfill, with secondary usage (assumed to be 20% of total) as soil conditioners for hops, keiselguhr and hops. The report noted a discrepancy in distillery pot ale was due to consented discharge to sea. Distillery spent lees was stated as partly disposed via wastewaters, but calculations in the AD sector recognised 153 ktonnes P / year (16%) of lees were currently used for distillery AD plants. Together these secondary waste streams gave 2.97 ktonnes P / year when P contents of 0.5 and 1.8 kgP2O5/tonne fresh mass waste were applied to grains and pot ale, respectively (SRuC, 2015) and this was scaled up according to dry matter ratios for pot ale syrup. The resulting P apportionment in secondary products was 38% to sea > 27% to WWTW > 24% to animal feed > 6% to AD for energy > 5% to soil and <0.01% to landfill.

Wastewater systems

Scottish Water's annual returns for 2014-15 (WICS, 2015) reported 4,956,600 household population (resident and visitors) on mains sewerage services. The difference between this and the total 2011 census population of 338400 people was taken to be those on decentralised wastewater (septic tanks); in broad agreement with an estimated 161,000 septic tanks in Scotland (i.e. one septic tank serving an average household of 2.1 people). This gave a resulting split of 93.6% and 6.4% on sewered and non-sewered wastewater services.

Sewered wastewater systems

According to Scottish water's 2014-15 annual returns (WICS, 2015), 1854 WWTW across Scotland handle an average of 3003 ML/day of sewage. In categorising WWTW we considered the following categories (reported separately by WICS, 2015): (i) 1216 primary treatment works (including septic tanks adopted by the water authority), (ii) 481 secondary process WWTW (combining activated sludge and biological), and (iii) 127 tertiary process WWTW (combining A1, A2, B1 and B2 types). A further category of 36 WWTW connected to sea (preliminary, screened and unscreened) was considered as direct discharge to oceans.

In calculation of the P mass handling of the different WWTW classes, P was considered to have the same distribution as reported mass loads for biological oxygen demand (BOD). The proportion of the total BOD load being handled by each category of WWTW was multiplied by the total population of sewered households and by an average P loading of effluent, comprising: 0.104 kgP/person/year of P dosing of tap water, 0.030 kgP/person/year of food waste down sinks, 0.154 kgP/person/year from household detergents (Richards et al., 2015) and 0.329 and 0.146 kgP/person/year from human urine and faeces (Jonsson et al., 2006). This sum of 0.763 kgP/person/year agrees well with a monitoring study from a Scottish sewer of 0.767 kgP/person/year, reported by Gilmour et al. (2008).

This calculation gave the household domestic P input to the categorised WWTW. From a further set of reported figures (WICS, 2015) on the split of the BOD load between domestic, non-domestic and trade, imported sludge and septic tank effluents a further non-household load was calculated and apportioned to secondary and tertiary WWTW classes.

Non-sewered wastewater systems

As highlighted above, there are 338,400 systems (determined by difference between the total national and sewered population), giving 6.4% on non-sewered wastewater services. A previous report by O'Keefe et al (2014) documented a breakdown of 62,000 septic tanks registrations in Scotland, held by SEPA. These were apportioned according to tank treatment and discharge to either: sea, surface waters, or soakaway. Groupings were applied to the treatment type categories, so that those classified Untreated (1%) and Preliminary (0.2%) were grouped as untreated, Primary treated (91%) was considered to be basic treatment and the sum of those classified Secondary (7%) and Tertiary (0.5%) were considered to be Advanced treatment. The total load of 243 tonnes P /year from decentralised, non-sewered households was split according to these proportions. In addition the exports from the tanks were assigned according to the data in O'Keefe et al (2014) between discharge to sea, inland waters and soakaway soils at percentages of 12, 22, 65%, respectively for untreated effluents, 1, 19, 80% for basic septic tank's effluents and 2, 55, 43%, respectively for advanced treatment septic tanks. Note that the dominant effluent output for basic treated tanks is to soakaways but for advanced treatment tanks this is to surface waters, since this is allowed by the regulator.

Other P inputs to wastewaters

Detergent P loads were derived from a recent study (Richards et al, 2015) in Scotland coupling direct analysis of a range of supermarket-available household detergents and cleaning products with household and behaviour data taken by survey. The study showed that of 0.154 kgP/person/year from total household detergent usage, 0.147 kgP/person/year comprised dishwasher detergents. The export of detergent P was divided between routing to WWTW and septic tanks at a percentage of 93.6% and 6.4%, respectively, according to the population on the sewered network and the difference between the total national population.

Mains tap water P dosing was determined as an average of 1.9 mgP/L of supplied water (UKWIR, 2012). The national P input of 1252 tonnes P/year for water dosing was derived from this concentration multiplied by the 659,190 ML/year mains water supplied according to Scottish water's 2014-15 annual returns (WICS, 2015). The export of this P was apportioned by multiplying the P mass by a split of the supply volume according to:

(i) The total water volume supplied to households was taken as the national population minus the number of people on private water supplies reported as 150000 (http://www.gov.scot/Topics/Environment/Water/17670/pws), then multiplied by 150 L/person/day, resulting in 43% of the stated national supply;

(ii) Subtracting this household volume from the total volume supplied by Scottish Water gave a volume assumed as water supply to trade (57% of supplied volume).

According to WRAP (2011) and using 2006/7 data, the main uses of non-domestic mains water in Scotland were manufacturing (41%), unclassified (16%), energy sector (9%), public administration and defence (9%) and food and accommodation services (7%). The export of the household supply was further divided between routing to WWTW and septic tanks at a percentage of 93.6% and 6.4%, respectively.

Sludge processing

Data comes from two sources in the WICS (2015) reports: 102,385 tonnes of sludge dry matter were generated from 11 out of 21 WWTW within a private financing project set, with a minor amount of 20,200 tonnes sludge dry matter from remaining Scottish water controlled WWTW. These were summed to give the national outputs. In the calculation of P loads a single reported value of 28% dry matter and 20.8 kgP per tonne dry matter were used (Stutter, 2015) as measured at the Cambi-plant in Aberdeen. From this an annual export of 2130 tonnes, P was derived and distributed between advanced processing sludge applied to farmland (45%), incinerated (30%), applied to reclaimed land (16%), conventionally-processed sludge applied to farmland (7%), landfill (1%) and other (1%). It was recently disclosed by SEPA in a parliamentary answer to a public petition (https://www.scottish.parliament.uk/S4_PublicPetitionsCommittee/General%20Documents/20150612_PE1563_A_SEPA.pdf) that 9000 ha (being 0.5%) of agricultural land received an average application rate of 6 tonnes/ha.

The data from the PFI contracted WWTW (Table E3; WICS, 2015) apportioned the sludge to the categories of WWTW, being responsible for 84% of the sludge output. The unidentified remaining 16% was considered to come from secondary process WWTW. Nationally this apportionment indicated that 0%, 34% and 27% of sludge processed came from primary, secondary and tertiary categorised WWTW, respectively, with a further 39% being handled by the single dedicated facility at Dalderse, SW Scotland (largely pumped in from secondary WWTW).

Anaerobic digestion

A national web database of biogas plants for the UK (available at http://www.biogas-info.co.uk/) was queried for plants in Scotland and gave 14 returns together with data on the location feedstock nature and feedstock annual demand. Additionally data was found at http://www.scottishwater.co.uk/business/horizons/horizons-environment/anaerobic-digestion for a further biogas plant operated by Scottish Water at Deerdykes. The feedstock categories comprised: brewery waste, mixed farm feedstock including animal wastes, food wastes and mixed farm plant and vegetables. Three analytical values for P contents of digestate were used firstly two from vegetable processing industry and food waste feedstocks of 16.7 and 9.5 kgP / tonne dry matter, respectively (at 4.8 and 4.2% dry matter) from a study in England with (A Lag Brotons, pers. Comm) and the third directly from one of the Scottish plants processing of 14.7 kgP per tonne dry matter at 5% dry matter content (Stutter, 2015). These P contents were applied to corresponding categories of feedstock across the 15 Scottish biogas reactors and assumed that no P loss occurred during the AD processing. The result was 289 tonnes P / year both as a total input and available output in digestate to support P recycling. The feedstock P inputs were apportioned on the basis of the documented categories as: brewery & distillery waste (54%), crops and fodder (25%), food waste (11%) and farm animal waste (10%). The usage of digestate currently was assumed to be wholly for application to agricultural land.

Losses to surface waters

From agricultural land

These were modelled as two separate pathways with discrete data sources. Firstly, erosion losses of P were considered by loss coefficient methods from data in Balana et al. (2012), which basically extended long established loss coefficients from land classes by adding extreme categories at the low end (for extensive grassland) and high end (for highly eroding activities such as potatoes). The data were derived for and calibrated on the Lunan catchment in NE Scotland in Balana et al. (2012). The areas of crop, fruit and grassland from the 2015 agricultural census for Scotland were then assigned a loss coefficient of P erosion yield and summed for all areas to give an erosion loss of 741 tonnes P/year delivered to surface waters. For dissolved P losses a different modelling approach was used. A relationship was developed between 28 drainflow water total dissolved P values attained from a spatio-temporal study of national farmland in Scotland and the soil agronomic P test status, according to Scotland's modified Morgan's extraction method used for advisory purposes (Stutter, in prep). The equation (including term standard errors) was:

log TDP concentration (µg/L) = 0.638±0.197 + 0.924±0.199* logMMorgan's P ($p<0.001$, $n=28$)

The land areas from the agricultural census were then assigned a soil P status that assumed the median was the advisory guide P value (Scotland Rural College's advisory Technical Notes 663 and 668; SRuC, 2015), but that the upper and lower quartiles of the distribution matched soil test P data measured at 88 fields in Scotland categorised according to land use. This was felt to be a fair distribution of the soil test P nationally for each crop type. These gave an average of soil total dissolved P leaching values of between 0.044mgP/L for grassland to 0.072mgP/L for potatoes. These concentrations were then multiplied by modelled runoff values derived from rainfall estimates of 800, 1000 and 1200 mm/year, for areas growing cereals, intensive grassland and extensive grassland, respectively, and an assumed runoff of 70% of rainfall. The sum of these dissolved P loads across all areas gave a P leaching of 1173 tonnes P/year.

From other land

Of non-agricultural land, only newly planted forestry land use is considered to have P fertiliser applied to any extent and hence be a part of this P budget considering P sources and sinks. For this expert advice was sought from staff in Forestry Commission Scotland (I Cowe, pers. Comm.) and Forest Research (B Raynor, pers. Comm.). It was considered that new forest plantings would often have a single application to aid establishment at application rates of 60 kg/ha P2O5. This would be given commonly as a 1m2 spot treatment for each tree, which for a 2700 tree per ha commercial planting density, would equal 16kg/ha P2O5. An assumption was made that this applied to all area of new plantings, hence this is a maximum usage scenario. However, it was suggested that, whilst this was standard practice for Forestry Commission establishment (Forestry Commission Bulletin 95 (1999), within private sector forestry this may approximate to 50% fertiliser usage.

The final area to apply the 16kg/ha P2O5 fertiliser assumption was taken from National Forest Inventory Woodland Area Statistics: Scotland (2011). This publication showed 192,500 ha of new planting between 1989 and 2009. It was assumed that this was a constant rate of 9625 ha, which resulted in an annual consumption of 0.07 ktonnes P year-1.

From wastewaters

The resulting combined input to national WWTW calculated above was 4897 tonnes P/year. Considering the sludge removal calculation of 2550 tonnes P/year, this leaves 2767 tonnes P/year discharged to waters. At the total sewage volume of 1,096,168 ML annually, this gives an average concentration for effluents discharged to surface waters of 2.1 mgP/L.

The effluent P losses to surface waters from each individual WWTW category were calculated by difference from the inputs and other outputs (i.e. sludge processing). This was then divided out by the volumes into each WWTW category (apportioned from the total stated volume of 3003.3 ML/day nationally of sewage according to the BOD loading proportions) to give the effluent concentrations discharged to surface waters. These were 3.36, 1.78, 0.61 and 3.21 mgP/L for primary, secondary, tertiary WWTW and discharge to sea, respectively. These were in broad agreement with a regulatory dataset of final effluent concentrations of P.

From septic tanks and soakaways

These were calculated previously and were negligible for untreated effluents (0.3 and 0.5 tonnes P / year to sea and inland waters), limited to sea from basic and advanced treatment tanks (3 and 0.4 tonnes P/year, respectively), but of significance for discharges to inland waters of basic and advanced tanks (41 and 10 tonnes P / year, respectively) due partly to the loads, but more to the concentrated nature of the effluent. Richards et al. (2016) measured total P concentrations for septic tank effluent in Scotland as 14.6±1.5 (mean±1s.e.) mgP/L, where advanced septic tanks are reported as 2mgP/L (L. May, pers. Comm) and the latter are unregulated for direct surface discharge.

2.2.2 What is not included?

Food waste from commercial food production and preparation (e.g. food processing) and supply (hospitality and shops) was not included in the literature search and appears in the budget as an unknown flow (?) to the non-domestic effluent. However, using the records of the wastewater system provided through WICS (2015) the transport and fate of P from that source is included in the wastewater P budget. No data could be found for the application of compost onto land (assumed to be onto agricultural land). If this was to include compost applied to gardens this may be quite a considerable P mass flux. Although it would be expected to be small relative to the main P budget components at this scale, it is increasingly important considering budgets at sub-national scale.

A couple of environmental fluxes of P were not derived for this study. One is the flux of P from all of Scotland’s rivers to the sea since the data to make this calculation at an appropriate level of accuracy were not available. A second flux not quantified was that of the P leached from non-agricultural land. This has three components, namely P leached from land where fertiliser is applied in (i) gardens and (ii) forestry land, and also (iii) a background leaching of P from land under semi-natural (or extensive grazing) where fertiliser is not applied. For the later it was assumed that the P leached approximated to the rainfall P loads from the atmosphere. Since this was a closed loop itself with no input from the managed P cycle then this was considered a net zero loss.

2.3 Discussion and recommendations

The main flows around the national P budget are in the agricultural system. This is where the opportunities for recycling of P are possible at, and between, national and local scales. Currently agricultural production (in terms of human food, animal feeds and products) is driven by the input of 20.4 ktonnes P year-1 of chemical fertiliser and 9.1 ktonnes P year-1 of animal manures (the latter input of manures can already be considered to be P recycling).

Compared to this total of 29 ktonnes P year-1 applied to agricultural soils, this study finds only 1.3, 0.3 and an unknown, assumed small, input of ktonnes P year-1 to agricultural soils from sewage sludge, AD and compost, respectively. Hence, this represents <10% of the input of chemical P fertiliser annually. However, one aim of the national scale budget was to test whether the potential for offsetting of chemical P inputs to the Scottish agricultural system was possible in terms of the magnitude of other potential inputs available. If we consider all the P flows through the wastewater system, it is shown that 3.75 ktonnes P year-1 comes through centralised wastewater handling systems from sewered households and a further 1.13 ktonnes P year-1 from non-domestic effluents. In addition a smaller 0.42 ktonnes P year-1 comes through decentralised wastewater systems from households using septic tanks. So, with full recycling and P capture, approximately 40% chemical fertiliser P replacement may be possible if socio-economic constraints in it usage and technical constraints in its capture could be made. This study has highlighted some main system inefficiencies, notably the apparent accumulation of ~10 ktonnes P year-1 in soils and ~5 ktonnes P year-1 into waters. Together these represent ¾ of the bought in rock P and have an approximate resource value of £22 million per year. There are many strands of evidence that elevated P concentrations have serious deleterious impacts on freshwaters and it is likely that the environmental damage from P flows into freshwaters has a financial value in itself. This study has highlighted that P losses to surface waters come 1.91 ktonnes P year-1 (44%) from agricultural land and 2.41 ktonnes P year-1 from water waste streams (53% from WWTW effluents and 3% from non-sewered septic tanks).

The present study did not evaluate the imports and exports at a Scottish level. Whilst it may be concluded that, since the UK has no rock phosphate reserves, all the consumed P is imported, there are additional imports in the form of animal feed and products and processed food. These are not considered here but comprised 18% of the 138 ktonnes year-1 imported P at a UK scale in the study by Cooper and Carliell-Marquet (2013). These authors also found 23.5 ktonnes P year-1 was exported at a UK scale (comprising Crops 47%, fertiliser 23%, animal products 15%, processed food 9%, animal feed 6%). An interesting conclusion of this earlier study was that at a UK level the water industry removed 31.5 ktonnes P year-1 through sludge processing from the total 55 ktonnes P year-1 flow through the UK combined WWTW (i.e. leaving 23.5 ktonnes P year-1 or 43% discharged to waters). Of this sludge 22.5 ktonnes P year-1 was applied to land and represented around 8% of the total UK fertiliser requirement. However, this was apparently being applied to only 1.5% of the agricultural land in England and Wales, indicating that large localised application rates (estimated as >120 kgP/ha) were occurring, which will have serious consequences for P leaching risk from soils to waters.

Recommendations to further explore the possibilities for recycling of the dominant P flows in waste systems (dominantly wastewaters in the context of the data explored here) in Scotland are to:

(i) Examine spatial scales in the linkages for where the dominant P is produced in wastes and where it may be consumed by appropriate application to agricultural soils. The latter requires a spatial approach based on crop P requirements and soil test P data that allows the correct targeting of P according to crop requirements and guarding against over application or application to soils sensitive to P leaching to minimise environmental consequences.

(ii) Develop local budgets for target areas where there seems an optimum balance between wastewater and other P sources and favourable agricultural land. Build on these with details of other factors including costs and other aspects (processing chemicals, fuel etc.) to make a more holistic budget approach of associated resources alongside the P budget (the current study deemed this was not able to be done realistically at a national level but is more feasible at a local level).

(iii) Support the above spatial sources and sinks analyses with an appraisal of the technologies available for P recycling, including how these options may make different P sources more/less favourable, how separation/purification techniques (e.g. the becoming widely adopted struvite precipitation) may alter transportation costs, or remove social sensitivities around the usage of the raw products. There may be local advantages to be made through P recovery at small rural scales through septic tanks in particular. Although this study shows that decentralised wastewater systems have relatively small flows of P compared to centralised systems, it is important that the former are amongst the most concentrated P flows entering rural ecosystems. These concentrated wastewater P flows (10-20 mgP L-1) may be easier technically to strip P from than slightly lower concentration centralised effluents.

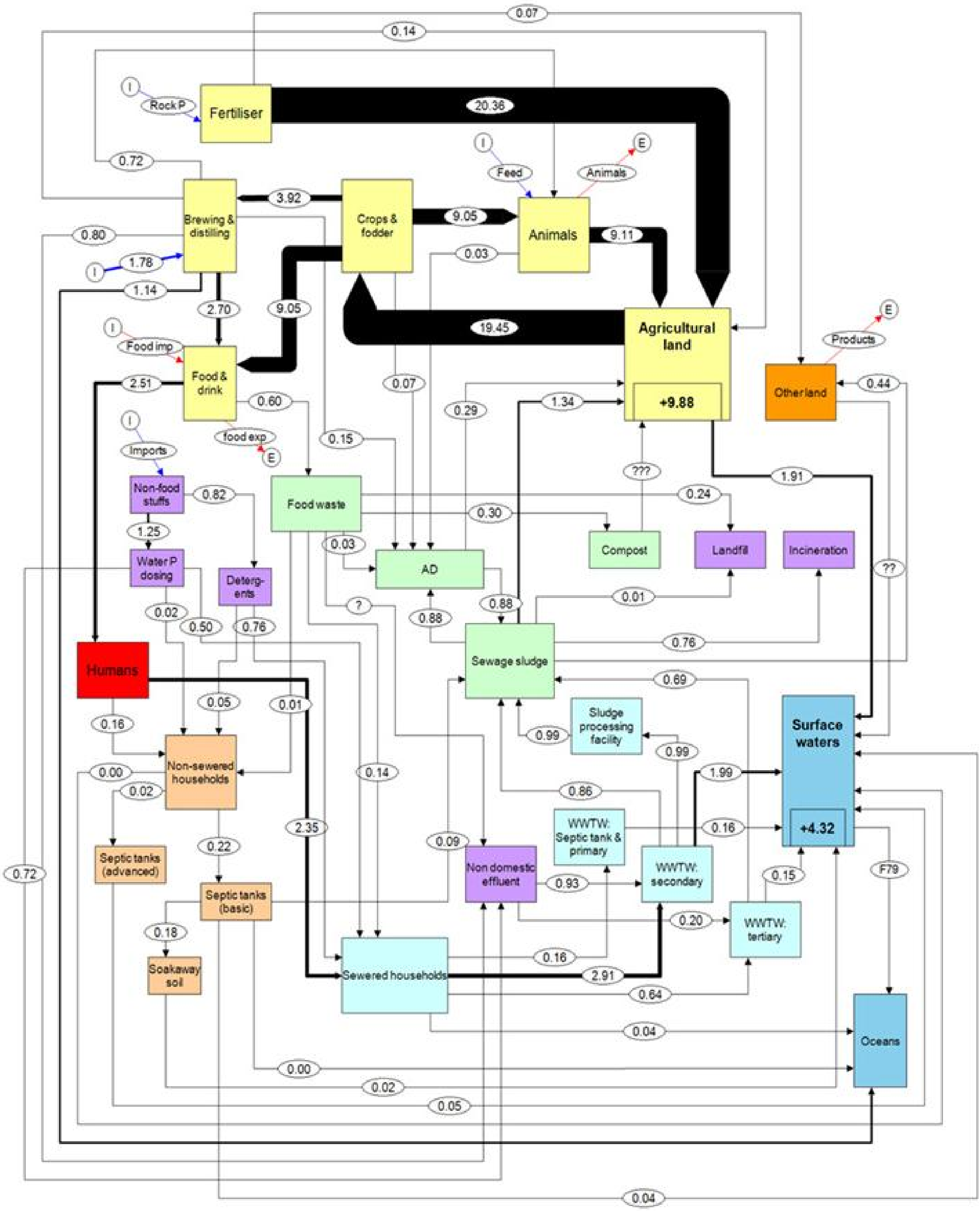

Figure 8. A national resource budget for Scotland using 2015 data derived from literature review, primary data, expert input and modelled estimates. Arrow widths (and values) represent flows (in ktonnes P year-1) between the considered system components (boxes, coloured: yellow, agronomic system; purple, industrial components; green, recycling opportunities; mid blue, surface waters; light blue, centralized wastewater systems; tan, decentralized wastewater systems). Increases in stocks are shown for agricultural land and surface waters. Letters ‘I’ and ‘E’ denote imports and exports, respectively, which have not been quantified.

3.0 Life Cycle Analysis (LCA)

Life cycle analysis (LCA) is a technique for assessing the environmental aspects associated with a product (such as water) over its lifecycle. The primary aim of an LCA is to analyse the contribution of the life cycle stages to the overall environmental load, usually with the aim to prioritise improvements on products or processes. In the case of this project, the aim was to investigate options to increase the environmental sustainability of waste water treatment.

The LCA analyses undertaken in this report were carried out using the SimaPro LCA software and in accordance with the principles detailed in the following ISO standards for LCA:

ISO 14040: Principles and framework
ISO 14041: Goal and Scope definition and inventory analysis
ISO 14042: Life Cycle Impact assessment
ISO 14043: Interpretation

Briefly, the main stages of the LCA are:

1. Define the goal and scope of the study
2. Making a model of the product life cycle with all environmental inflows and outflows – this is usually referred to as the life cycle inventory (LCI)
3. Understanding the environmental relevance of all inflows and outflows – usually referred to as the life cycle impact assessment (LCIA)
4. Interpretation of the study

3.1 LCA of large urban waste water treatment plant

3.1.1 Goal and scope

The objective of this work was to develop a life cycle process diagram for typical wastewater treatment plant (WWTP) technologies currently used in Scotland and use this to identify where the greatest potentials are for environmental improvement and energy savings.

The project initiated a number of research activities to collect data and address the objectives. Including:

- Engaging Scottish Water (SW) in technical and policy discussions concerning wastewater treatment in Scotland.
- Discussions with industry professionals on understanding the processes involved.
- LCA literature and reports review and reliance on Ecoinvent database (LCA database which is a component of the LCA software used, SimaPro).

3.1.2 Baseline model

It was agreed to use a specific large waste water treatment plant in the Central Belt of Scotland as a baseline model. Through discussions with Scottish Water, we replicated the existing waste water treatment plant within the SimaPro software (Figure 9):

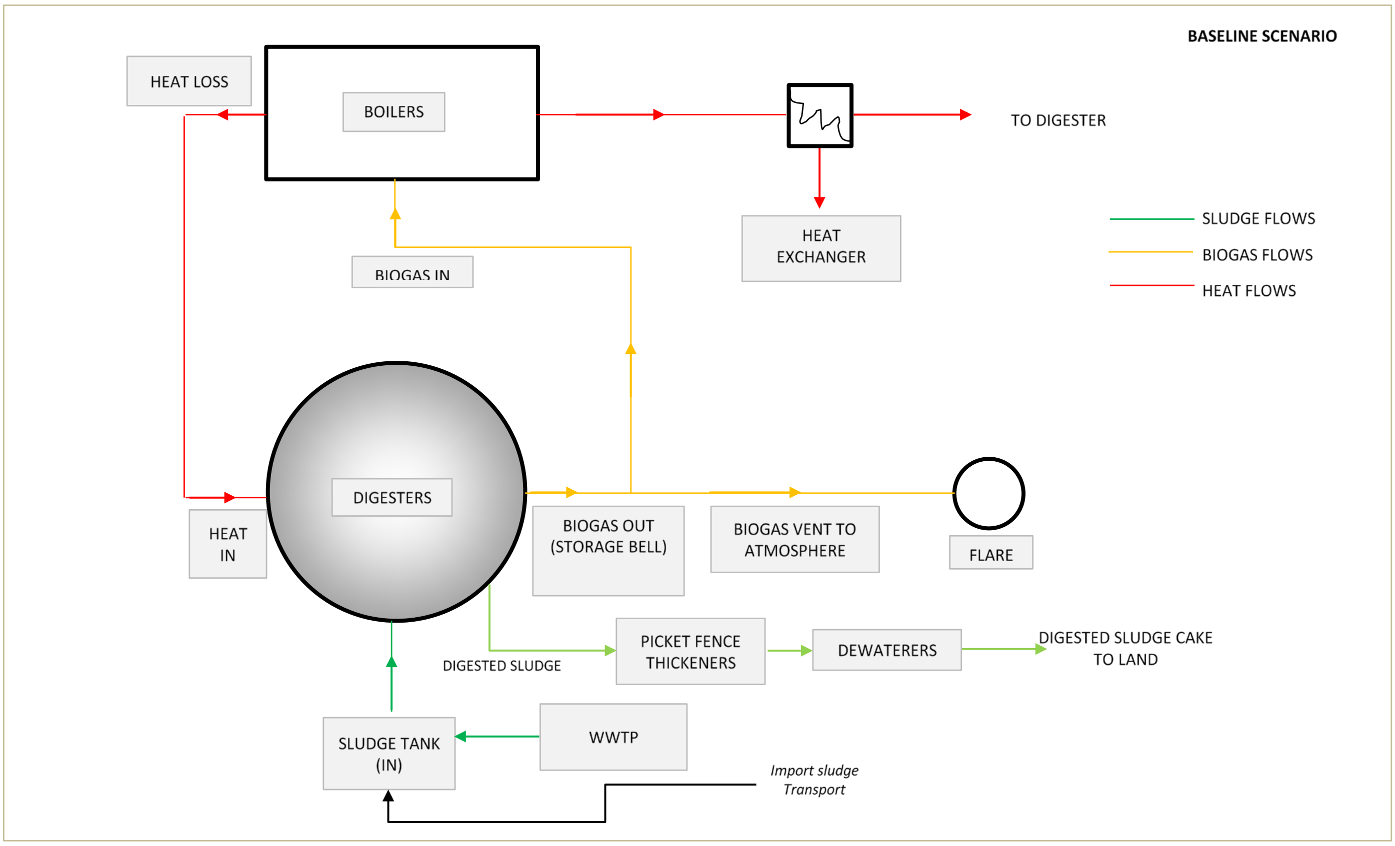

Figure 9. Schematic diagram of the baseline scenario used in the life cycle analysis; specifically, this diagram captures all the major processes and flows occurring at an existing large waste water treatment facility managed by Scottish Water.

We drew our system boundaries around the operation of the treatment plant as it was important to compare the sustainability of the current process of operation to alternatives. As already discussed in the introduction, the aim is to work largely with existing infrastructure and explore sustainability options for this.

All environmental inflows and outflows associated with the system described in Figure 2 were described and quantified using two sources of data: (i) generic figures available in existing life cycle inventories, including ECOINVENT; (ii) site-specific data obtained with permission from Scottish Water. The final model system and associated data base of inflows and outflows is referred to as the life cycle inventory (LCI). The LCI was used to perform the analyses described in the following sections.

3.1.3 Cumulative energy demand (CED) of the Baseline Scenario

The cumulative energy demand is a useful indicator for the energy intensity of a process (Frischknecht et al. 2007). The CED has been defined as "…the entire demand, valued as primary energy, which arises in connection with the production, use and disposal of an economic good" (VDI 1997). The life cycle of a product (such as clean water) can generally be subdivided into the three phases of 'production' (P), 'use' (U) and 'disposal' (D) in which final energies, e.g. electricity or fuel, are engaged (Röhrlich et al., 2000).

Beside the direct energy input for production, use and disposal of a product, production facilities, as well as raw materials, auxiliary materials and consumables are also used. These are products that need energy for their own production process. Furthermore, the final energy applied in the processes is a product of mining, transformation and transport processes which again are performed using machines and consuming different energy carriers and materials.

The question of the energy expenditure for the energy supply is answered by determining the necessary amount of primary energy. The total of all energy inputs, concerning the consumption of primary energy, is called the Cumulative Energy Demand (CED) of a product (Equation 1):

$$CED = CED_P + CED_U + CED_D \qquad [1]$$

The CED is a parameter that forms the basis for further energetic assessment values (e.g. energy pay-back time or amortisation time); as well as overall efficiency of supply.

Despite the apparent simplicity of Equation 1, it should be noted that different final energies (e.g. fossil fuels, electricity, thermal) have different physical qualities and thermodynamic properties. For example, it is inappropriate to directly compare 1 MJ electricity with 1 MJ low-temperature heat. Therefore the question of CED cannot be answered adequately by simply summing up the final energies that are employed in a process. However, it is possible to determine the required primary energies from the different final energies using the overall efficiency of supply, which means that the comparison of process becomes possible. The overall efficiency of supply g describes the relation of the final energy provided and the CED (Equation 2, Equation 3):

$$g_{el} = \frac{W_{el}}{m_{fuel}^{prim}\, H_u^{prim} + \sum_i CED_{plant,i}} \qquad [2]$$

$$g_{fuel} = \frac{m_{fuel}\, H_u}{m_{fuel}^{prim}\, H_u^{prim} + \sum_i CED_{plant,i}} \qquad [3]$$

Where:

g_{el} is the overall efficiency of supply of electricity

g_{fuel} the overall efficiency of supply of fuels

$m_{fuel}^{prim}\, H_u^{prim}$ is the energy of the primary energy carrier

W_{el} is the supplied electricity

$m_{fuel}\, H_u$ is the supplied energy of fuel, and

$CED_{plant,i}$ is the Cumulative Energy Demand for installing, running and disposing of machines, plants and consumables providing the electricity or the fuel

Equations 1 – 3 were applied to the process chain described in Figure 9 in order to estimate CED for the baseline scenario of a large urban waste water treatment plant located in the Central Belt of Scotland. This was implemented using the SimaPro LCA software. The analysis indicated that the vast majority (> 90%) of energy demand in waster water treatment from all energy sources, was for electricity to run the process. This was primarily for the movement of water via pumping. A small proportion of the CED was attribuatble to treatment process materials such as iron and aluminium sulphates and iron (III) cwhloride (Figure 10).

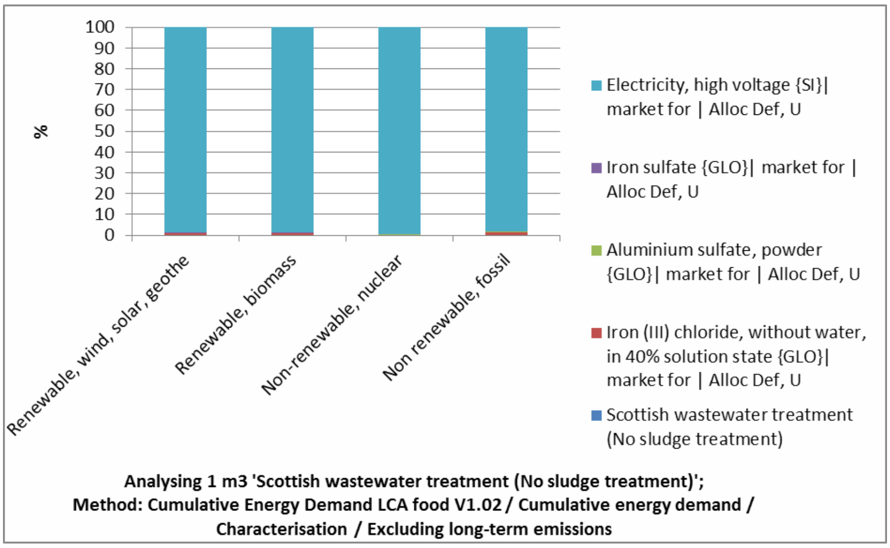

Figure 10 – Cumulative Energy Demand to treat 1 m^3 of waste water using the process chain depicted in Figure 9

3.1.4 CED of sludge handling options

As shown in Figure 7, sludge management and the potential to generate energy from sewage sludge is one approach to reducing losses of both energy and phosphorus from the system. In order to explore this, we manipulated our baseline scenario to compare the lifecycle CED profile for the waste water treatment plant if sludge is applied to land vs. being used to generate electricity and heat via an anaerobic digester (Figure 11). It can be seen that the CED for the biogas scenario tends to be greater from all energy sources primarily due to the additional energy associated with the infrastructure, operation and maintenance, and final decommissioning of the AD plant. As the energy use is dominated by electricity demand (Figure 10), the mix of energy sources will significantly influence the impacts (environmental, human health) that may result from these processes. The current energy mix is dominated by fossil and nuclear sources which are likely to have greater environmental impacts than electricity derived from renewable installations.

3.1.5 Impact analysis of sludge handling options

Figure 12 provides a schematic overview of the impact analysis methodology (Goedkoop & Spriensma, 1999) adopted in this study. The inventory data are used to parameterise environmental models that predict impacts on a series of mid-point categories, such as climate or land-use. These models are then extended up to an end-point level. Where impact category indicators that relate to the same endpoint also share a common unit, these indicators can be aggregated.

The LCIA methodology of Goedkoop & Spriensma (1999) was implemented in the SimaPro software and initially applied to the inventory of the baseline scenario in order to analyse the environmental performance of the waste water treatment plant (Figure 13) and to compare this to the European average environmental performance of waste water treatment (Figure 14).

It can be seen in Figure 13 that for the majority of mid-point impact indicators, lifecycle electricity consumption is the major contributor. Only Eutrophication Potential is primarily attributed to processes and materials used during the waste water treatment process itself.

In comparison to the European Average, the environmental performance of the Scottish Water treatment plant depicted in Figure 9 is significantly improved. Most notably, eutrophication potential is about a fifth of the European average indicating that the processes and materials used in the treatment process are extremely well optimised and the quality of final effluent is generally very high.

The LCIA methodology of Goedkoop & Spriensma (1999) was implemented in the SimaPro software to explore the potential environmental impacts of the two sewage sludge handling scenarios (sludge to land vs. sludge to biogas). The results of these analyses are summarised in Figure 15. In corroboration with Figure 11, the inclusion of anaerobic digestion in the system increases the life cycle impacts on climate change and resources. This is primarily attributable to the additional CED of the construction, operation and maintenance, and decommissioning

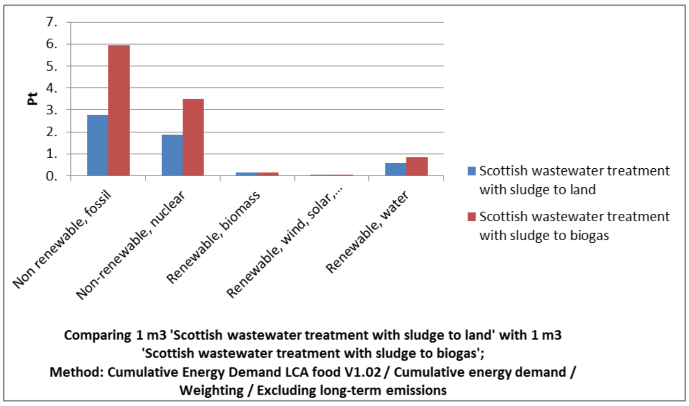

Figure 11 - Cumulative Energy Demand (millipoints; a relative scale enabling meaningful comparison between different systems) to treat 1 m^3 of waste water using the process chain depicted in Figure 9 with either sludge to land or sludge to anaerobic digestion as sludge handling options.

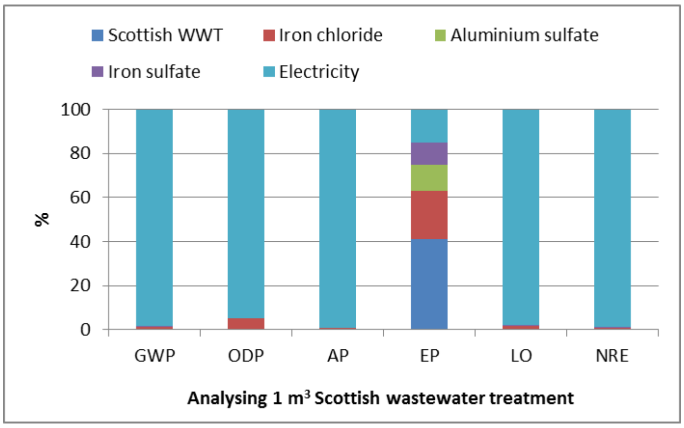

Figure 13 – Environmental performance of the baseline scenario (Figure 2): GWP Global Warming Potential; ODP Ozone Depletion Potential; EP Eutrophication Potential; AP Acidification Potential; LO Land Occupation; NRE Non-Renewable Energy

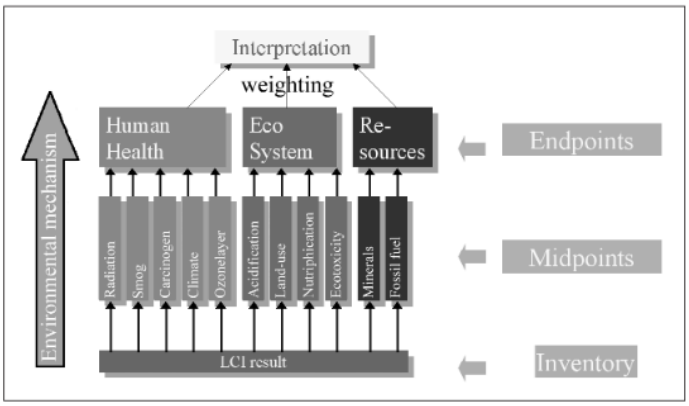

Figure 12 – Schematic overview of life cycle impact assessment (LCIA) approach of Goedkoop & Spriensma (1999)

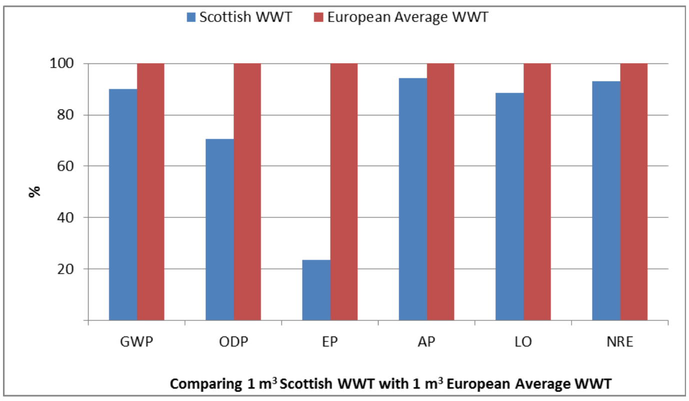

Figure 14 – Comparison of the Baseline Scenario (Figure 2) with European average: GWP Global Warming Potential; ODP Ozone Depletion Potential; EP Eutrophication Potential; AP Acidification Potential; LO Land Occupation; NRE Non-Renewable Energy

of an AD facility. Due to the reliance on electricity as an energy source, these impacts could be reduced if the energy mix moves towards more renewable sources. Conversely, the use of AD reduced end point impacts on human health and ecosystem quality. Primarily due to reduction in the volume of waste material being spread to land, as well as changes to the pathogen and contaminant profile due to the AD process.

3.2 LCA of sustainability options at small rural waste water treatment plant

Establishing renewable energy systems to reduce the greenhouse gas (GHG) emissions is critical to meet the targets set in the Climate Change (Scotland) Act 2009 and practical action needs to be taken towards the delivery of the 2020 Route Map for renewable Energy goals.
Current renewable energy options focus on wind, tidal or wave or hydro power, geothermal or photovoltaic plants, waste conversion and dedicated biomass production. Based on life cycle analyses, biomass production can have high carbon emissions per unit amount of energy produced (gCO2eq kWhe-1). This is in part because dedicated energy crops often need significant inputs of fertiliser to sustain the crop. Options for renewable energy should, by definition, be sustainable. Therefore wherever possible, systems should be developed to maximise multiple benefits and minimise disbenefits.

One way of improving the economics and GHG footprint of biomass production is to irrigate it with wastewater. Raw sewage typically contains around, 40 mg l-1 of total nitrogen and 12 mg l-1 of total phosphorus. The discharge of N and other macronutrients into the environment is reduced by wastewater treatment plants, which frequently rely on high energy treatment systems or chemical additives which have embodied energy costs, contribute to global resource depletion and can generate an additional waste stream.

In Enkoping, a town of 20,000 inhabitants in the centre of Sweden, a 75 ha free draining Willow short rotation coppice plantation treats 11 tonnes of nitrogen and 0.2 tonnes of phosphorus per year from wastewater, producing 10 tonnes of dry matter per hectare per year to be used in local wood combustion power stations. While reducing the fertiliser needs and increasing the economic potential of willow biomass, a disadvantage of this type of system is the risk of nitrous oxide emissions (a GHG) and nitrogen leaching into the deeper soil due to the absence of a fully waterproof soil layer or a liner. Land availability is also critical to the success of such an approach. We are proposing an alternative approach, in which willow coppice is grown in lined constructed wetland beds.

Constructed wetland systems (CWs) are an efficient low energy wastewater treatment system. CWs are particularly implemented, after a primary treatment, to treat wastewater from small rural works or as tertiary effluent treatment at larger works. Reeds are commonly used in this type of system but there is no established market to use reed biomass so it is rarely utilised, frequently not harvested from the beds and thus nutrients taken up during the growing season leach back into the effluent. CWs planted with willows would create a treatment system that has the additional benefit of generating a useful bioenergy crop. Such an approach could be implemented as new systems where land is available or to replace existing reed bed CWs requiring renewal (10–20 year lifespan) but has not been demonstrated in the UK to date.

The James Hutton Institute has been undertaking a study at a small rural waste water treatment plant in Aberdeenshire to inform practices for combined wastewater polishing and biomass production in order to maximise the combined benefits (efficient wastewater treatment and wood fuel production) and minimize external energy inputs (i.e. inorganic fertiliser, chemical additives, electricity) and N2O production.

The James Hutton Institute has been working with Scottish Water, Cardiff University and Craig Thomson (Contractor) to develop a demonstration site to study and evaluate the performance for wastewater polishing and biomass production of a pilot scale CW planted with willows at a rural wastewater treatment site in Aberdeenshire. The existing system comprised a septic tank serving a small village with effluent disposed of via a soak-away. In April 2013, two beds (6 m x 3 m x 0.42 m) were built and lined with a waterproof EPDM membrane. Each bed was filled (from the bottom) with 10 cm of coarse gravel, 20 cm of light expanded clay aggregate (LECA) and 10 cm of pea gravel. Bed A was planted with 120 willows (Salix viminalis x burjatica var. Ashton Stott), Bed B with 72 reeds (Phragmites australis), Figure 16.

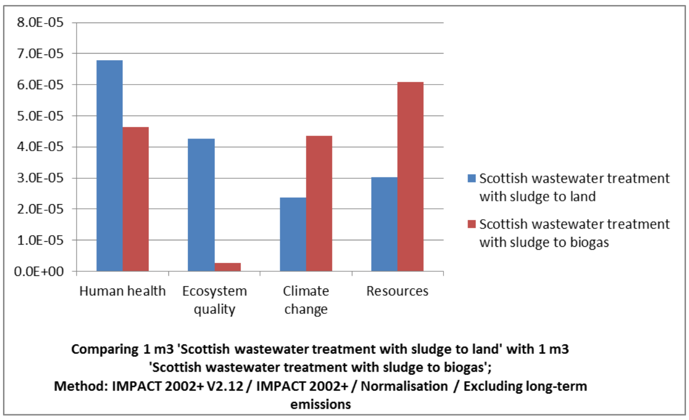

Figure 15 – Endpoint impacts (millipoints; a relative scale enabling meaningful comparison between different systems) comparing treatment and application of 1 m^3 of waste water and associated sludge to land with 1 m^3 of waste water and associated sludge to biogas using the process chain depicted in Figure 9.

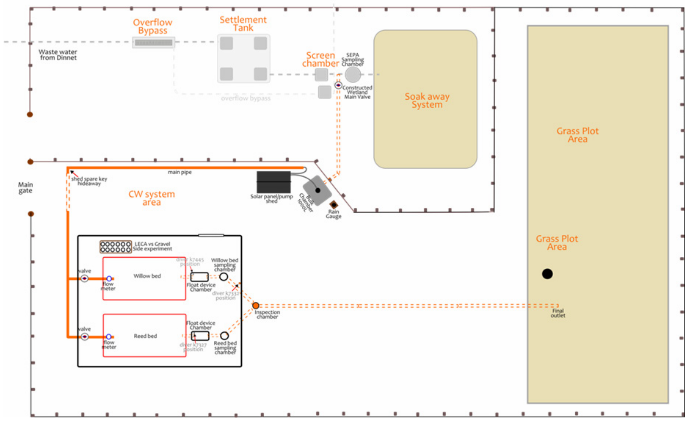

Figure 16 – Schematic overview of small (120 person-equivalent) rural waste water treatment plant and constructed wetland systems developed by the James Hutton Institute

The goal and scope of the life cycle analysis were to Identify and quantify the environmental impacts of the constructed wetland system, and to compare the environmental performance of a willow short rotation coppice to a reed bed. Data for the inventory were derived from three sources:

1. The ECOINVENT inventory
2. Measurements made at the waste water treatment facility and constructed wetlands
3. Data provided by Scottish Water

Because we were specifically comparing the environmental performance of willows vs. reeds, we were able to restrict the system boundaries to only the operational aspects of the life cycle and were able to exclude construction and decommissioning phases (Figure 17). This simplified the analysis significantly. As with the first LCA (Figure 2), the functional unit of analysis was 1 m^{-3} of waste water. The results of the impact analysis are shown in Figure 18 (mid-point impacts) and Figure 19 (end-point impacts).

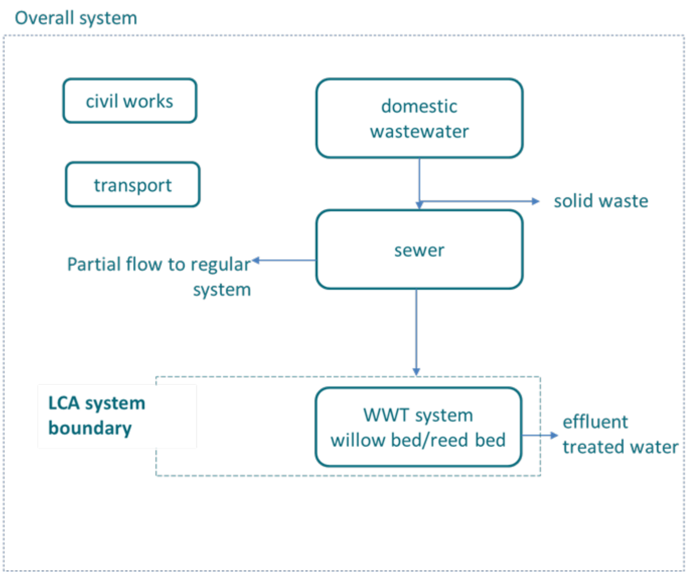

Figure 17 – System boundaries

Apart from global warming potential, production of willow biomass irrigated with waste water reduces mid-point impacts when compared to reed production. The largest reductions in impact are seen for human and ecosystem toxicity, this is primarily due to the fact that the willows are more efficient at taking up contaminants and nutrients from the waste water compared to the reeds.

This is reflected in the end-point impact categories, again showing that willow production reduces impacts on both human health and ecosystems. This highlights the fact that Willow is efficient in nitrogen and phosphorus uptake and the resultant low eutrophication potential has less impact on ecosystem resources.

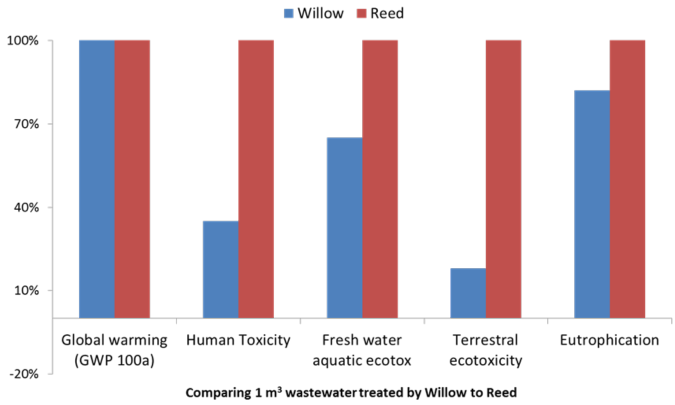

Figure 18 – Mid-point impacts of the process chain depicted in Figure 9 with biomass production as either willows (blue) or reeds (red)

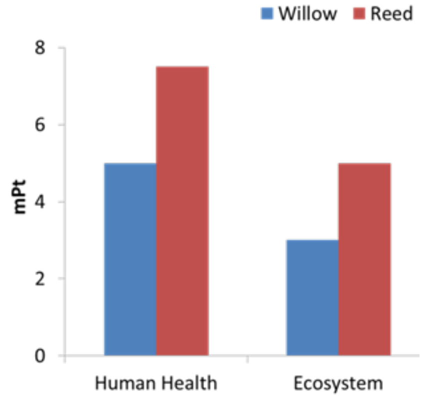

Figure 19 – End point impacts (millipoint scale) of the process chain depicted in Figure 9 with biomass production as either willows (blue) or reeds (red)

4.0 Overall conclusions

Headline findings

- In terms of energy efficiency, human health and environmental impacts, the large waste water treatment plant operated by Scottish Water is significantly better than the EU average
- Sludge to land AD is more energy efficient than sludge to AD to land; but more adverse environmental impacts. The level of impacts for either disposal route would reduce if the energy mix moves further towards renewable sources.
- Small rural sites – potential to reduce environmental & human health impacts through biomass production
- Willow biomass production irrigated with waste water has fewer adverse impacts on human health or the environment that reed biomass production

5.0 References

Balana BB, Lago M, Baggaley N, Castellazzi M, Sample J, Stutter S, Slee B, Vinten A. 2012. Integrating Economic and Biophysical Data in Assessing Cost-Effectiveness of Buffer Strip Placement. Journal of Environmental Quality 41:380-388

Bennet EM, Reed-Andersen T, Houser JN, Gabriel JR, Carpenter SR. 1999. A phosphorus budget for the Lake Mendota watershed. Ecosystems 2: 69-75.

British Survey of Fertiliser Practice, 2013. https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/301474/fertiliseruse-report2013-08apr14.pdf

CooperJ, Carliell-Marquet C. 2013. A substance flow analysis of phosphorus in the UK food production and consumption system. Resources, Conservation and Recycling 74: 82-100. DOI: 10.1016/j.resconrec.2013.03.001

Egle et al. 2014. The Austrian P budget as a basis for resource optimization. Resources, Conservation and Recycling 83: 152–162

Frischknecht, R.; Jungbluth, N.; Althaus, H.J.; Doka, G.; Dones, R.; Hischier, R.; Hellweg, S.; Humbert, S.; Margni, M.; Nemecek, T.; Spielmann, M. 2007. Implementation of Life Cycle Impact Assessment Methods: Data v2.0. ecoinvent report No. 3, Swiss centre for Life Cycle Inventories, Dübendorf, Switzerland.

Gilmour et al. 2008. http://web.sbe.hw.ac.uk/staffprofiles/bdgsa/11th_International_Conference_on_Urban_Drainage_CD/ICUD08/pdfs/740.pdf

Gerlagh, T. and Gielen, D.J. 1999 Matter 2.0 A module characterisation for the agriculture and food sector. European Commission, Brussels, ECN-C—99-048

Goedkoop, M.J. & Spirensma, R. 1999. The eco-indicator 99, a Damage Orientated Approach for LCIA, Ministry VROM, The Hague, 1999.

Jonsson H., Baky A., Jeppsson U, Hellstrom D. and Karrman E. (2006) Composition of urine, faeces, greywater and biowaste for utilisation in the URWARE model. Urban water report.

Results from the June 2015 Agricultural census Scotland.

J O'Keefe, J Akunna, J Olszewska, A Bruce, L May, R Allan 2014. Practical measures for reducing phosphorus and faecal microbial loads from onsite wastewater treatment system discharges to the environment: A review.

Li S, Yuan Z, Bi J, Wu B. 2010. Anthropogenic phosphorus flow analysis of Hefei City, China. Science of the Total Environment 408; 5715-5722.

Metson GSM, Hale RL, Iwaniec DM, Cook EM, Corman JR, Galletti CS, Childers DL. 2012. Phosphorus in Phoenix: a budget and spatial representation of phosphorus in an urban ecosystem. Ecological Applications 22(2): 705–721

National Forest Inventory Woodland Area Statistics: Scotland, May 2011. http://www.forestry.gov.uk/pdf/NFI_Scotland_woodland_area_stats_2010_FINAL.pdf/$FILE/NFI_Scotland_woodland_area_stats_2010_FINAL.pdf

Richards S, Paterson E, Withers PJA, Stutter M. 2015. The contribution of household chemicals to environmental discharges via effluents: Combining chemical and behavioural data. J. Env. Manag. 150, 427-434.

Richards S, Paterson E, Withers PJA, Stutter M (2016). Septic tank discharges as multi-pollutant hotspots in catchments. Sc. Tot. Environ. 542: 854-863.

Röhrlich, M., Mistry, M., Martens, P.N., Buntenbach, S., Ruhrberg, M., Dienhart, M., Briem, S., Quinkertz, R., Alkan, Z. & Kugeler, K. 2000. A method to calculate the Cumulative Energy Demand (CED) of lignite extraction. International Journal of Life Cycle Analysis, 5:369-373.

Stutter M 2015. The composition, leaching and sorption behaviour of some alternative sources of phosphorus for soils. Ambio 44 (suppl 2): S207-S216.

UKWIR, 2012. UK Water Industry Research Limited. Report no. 12/DW/04/12. Alternatives to plumbsolvency control.

USDA, 2015. United States Department of Agriculture Nutrient Data Laboratory, National Nutrient Database for Standard Reference v.2.3.2. https://fnic.nal.usda.gov/food-composition/usda-nutrient-data-laboratory.

Verein Deutscher Ingenieure (VDI). 1997. Cumulative Energy Demand, Terms, Definitions, Methods of Calculation, VDI-Richtlinie 4600

Verstraete W., van der Caveye P., Diamantis V 2009. Maximum use of resources present in domestic 'used water'. Bioresource Technol. 100: 5537-5545.

WICS, 2015. Scottish Water annual returns Tables A, E and commentary notes. http://www.watercommission.co.uk/UserFiles/Documents/WIC2015%20Commentary%20Final%2009Jun15.pdf

WRAP, 2011. Freshwater availability and use in the United Kingdom. Project RSC014-001. Final Report.

Zero Waste Scotland, 2009. The Food we Waste in Scotland. Project EVA077-001, Sept 2009. http://www.zerowastescotland.org.uk/sites/default/files/The%20food%20we%20waste%20in%20Scotland%20-%20full%20report.pdf

Zero Waste Scotland, 2015. Circular Economy Sector Study on Beer, Whisky and Fish, June 2015. Final Report. http://www.zerowastescotland.org.uk/sites/default/files/ZWS645%20Beer%20Whisky%20Fish%20Report_0.pdf

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CREW is a Scottish Government funded partnership between the James Hutton Institute and Scottish Universities.