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A wholelife cost and carbon perspective of alternatives to septic tanks utilising nature-based solutions

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ABSTRACT

Septic tank systems (STSs) are widely utilised flowsheets for decentralised wastewater treatment in the UK. With a growing consensus that STSs have a sizeable detrimental impact on the environment, there is a need for rural flowsheets with improved treatment capabilities. This study examines the lifetime cost and carbon emissions of using an enhanced septic tank nature-based solution (EST-NBS) to improve STS compared to a package treatment system (submerged aerated filter (SAF)). The whole-life cost (WLC) of the flowsheets and Scope 2 cradle-to-grave lifetime carbon emissions (LCEs) of the flowsheets were assessed. The EST-NBS flowsheets represent a lower cost improved treatment system than SAFs at population equivalents (PEs) from 5 to 1,000. An STS averages an LCE of over 4,000 kg CO_{2eq} PE^{-1}, with all other considered flowsheets having lower emissions. The EST-NBS flowsheets had lower carbon emissions than SAFs. Even at low populations upgrading from an STS to an EST-NBS is a competitive abatement strategy, with costs of £260 tCO_{2eq}^{-1} emissions avoided, at 1,000 PE an NBS flowsheet has an abatement cost of –£17 tCO_{2eq}^{-1}. This shows the potential of using NBS flowsheets in rural wastewater treatment providing both a carbon and cost incentive against traditional designs.

Key words: LCA, nature-based solutions, septic tanks, WLC

HIGHLIGHTS

- A septic tank system averages the highest lifetime carbon emissions (LCEs) of all the options mentioned here.
- On top of being cost-effective, the nature-based solution (NBS) flowsheets have lower carbon emissions than more intensified SAF at populations from 5 to 1,000 PE.
- There is a potential of using NBS flowsheets in rural wastewater treatment providing both a carbon and cost incentive against traditional designs.

1. INTRODUCTION

Septic tank systems (STSs) represent a quarter of all sewage treatment systems worldwide (UN Habitat and WHO 2021) with similar estimates for STS usage in the US and Europe (Withers *et al.* 2014). For instance, in Scotland alone, around 160,000 properties are connected to STSs (O'Keefe *et al.* 2015), with over 1,200 operated by Scottish Water. STS consists of two primary elements: the septic tank and drainage field or soak away (Withers *et al.* 2012). Septic tanks are simple tanks with or without baffles that function as low-rate anaerobic reactors which have the primary function of solids removal and retention until the collected solids (sludge) is transported away for further treatment at another facility. Recommended practice is to desludge the tanks annually (HM Government 2015) although logistical constraints often mean they are desludged more frequently. Indeed, of Scottish Water operated septic tanks the majority are desludged at least every 6 months (57%), with just 1% desludged less than once a year (Jefferson 2022). A drainage field consists of a series of perforated pipes, atop a bed of media that distributes the septic tank effluent allowing it to slowly percolate through the drainage media and into the soil underneath (Figure 1).

There is growing consensus that the impact of STS on water quality is often understated (Dudley & May 2007) and it is estimated that over 80% of STS in the UK are working inefficiently due to their age (May *et al.* 2015). Expected effluent quality after the septic tanks for a functioning system is a total suspended solids of 80 mg l^{-1},

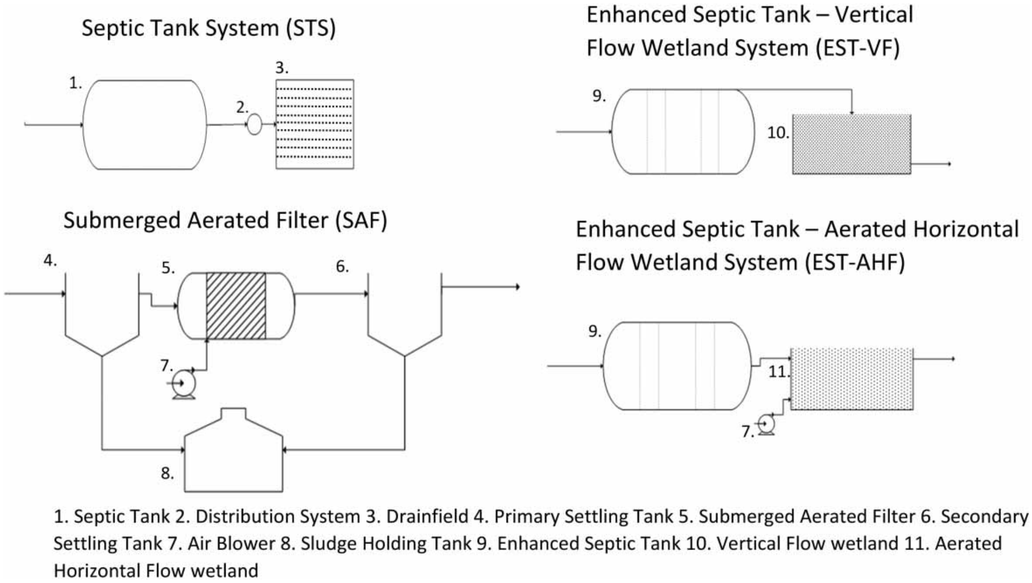

Figure 1 | Illustration of the analysed wastewater treatment flowsheets.

biochemical oxygen demand of 90 mg l^{-1}, with the ammonia remaining unchanged from the feed or slightly increasing with a typical level of around 35 mg l^{-1} (Table 1). Recent findings indicate that in rural communities STS disproportionately affect surface water nitrogen, in comparison to fertiliser usage (Halliday *et al.* 2014). For instance, ammonia concentrations in ground and surface waters have been shown to more than double downstream of STS (Withers *et al.* 2012; Herren *et al.* 2021). In addition, the anaerobic storage of sludge results in release of methane emissions with an estimated rate of 11 gCH_4 PE^{-1} d^{-1} equating to 0.11 tCO_2e yr^{-1} over the 100-year time horizon and 0.34 tCO_2e yr^{-1} over the 20-year time horizon (European Commission 2022). The regular frequency of desludging also adds further emissions and generates a nuisance impact on local residents. Accordingly, there is a growing need to update the flowsheet for onsite treatment beyond STS to one that improves treatment, reduces greenhouse gas emissions and reduces the frequency of desludging visits. The second ambition aligns to pledges made by the water industry to reach net zero either as full net zero emissions by 2040 (Scottish Water 2020) or operational emissions by 2030 (Water UK 2020a). Beyond the main aspirations there is an appetite to deliver co-benefits to extend the value proposition that any new solution can provide in terms of environmental or societal benefits.

1.1. Alternative flowsheets

Two alternative approaches are considered (Figure 1). In the first approach, the primary stage is an enhanced septic tank (EST), which combines concepts from septic tanks, anaerobic ponds, and anaerobic baffled reactors (Brown 2023). The system uses baffles to increase digestion thus extending the time between desludging. The effluent from the septic tank is further treated in an aerobic nature-based solution configured as either a passively aerated vertical flow subsurface wetland (VF) or a forced aerated horizontal flow subsurface wetland (AHF) (Figure 1). Both are established technologies used in wastewater treatment and can deliver considerably enhanced effluent quality (Butterworth *et al.* 2016a). For instance, AHF wetlands have been reported to deliver

Table 1 | Flow and consent target for improved rural wastewater treatment flowsheets

Parameter	Influent loading (g PE^{-1} d^{-1})	Effluent consent concentration (mg l^{-1})	Septic tank effluent (mg l^{-1})
Total suspended solids (TSS)	80	25	80
Biochemical oxygen demand (BOD_5)	60	25	90
Chemical oxygen demand (COD)	150	–	160
Ammonia (NH_4-N)	8	15	35

median effluent qualities of 4 mg l^{-1} TSS, 4 mg l^{-1} BOD, and 0.2 mg l^{-1} NH_4-N when used as a tertiary treatment and 20 mg l^{-1} TSS, 5 mg l^{-1} BOD, and 0.8 mg l^{-1} NH4-N when used as a post primary sedimentation treatment (Butterworth *et al.* 2016b). Produced methane is collected for reuse or flaring to avoid methane emission into the atmosphere. Recent pilot trials of the NBS post an EST revealed the median effluent of the NBS post EST to be 5 m l^{-1} TSS, 11 mg l^{-1} BOD, and 0.5 mg l^{-1} NH_4-N and 12 m l^{-1} TSS, 8 mg l^{-1} BOD, and 3 mg l^{-1} NH_4-N for the VF and AHF, respectively.

The other option is to use a commercially available package treatment process based on a submerged aerated filter (SAF) to provide aerobic treatment (Figure 1). The system consists of four primary elements: a primary settling tank, an aerated filter, secondary settling tank and sludge holding tank (Figure 1). The sludge from the primary and secondary settling tanks are stored in a sludge holding tank which requires desludging similarly to a septic tank. The certified effluent quality of package treatment SAFs of the style used in this study is 16 mg l^{-1} TSS, 11 mg l^{-1} BOD, and 8 mg l^{-1} NH_4 (Tricel Environmental 2022).

The aim of the current paper is to establish the financial and carbon basis for updating from septic tank systems to alternative flowsheets for population equivalents between 5 and 1,000. To achieve this the four systems were designed at different population equivalents and the wholelife cost (WLC) and lifecycle carbon emissions (LCEs) calculated. To aid in this goal the flowsheets will be considered from two case study perspectives: an old site and a new site. In addition, the paper will also consider the potential societal and ecological impacts of these flowsheets to provide an overall picture of the potential for using EST-NBS-based systems.

2. METHODOLOGY

2.1. Design case studies

The first case study considered an existing site in which an STS is being replaced. The site aims to represent a 'normal' site, with dry soil conditions and a soil percolation rate, v_p of 50 mm min^{-1}. The site is situated 40 miles from a large works where desludging tankers and site visits originate from. The site has sufficient topography for gravity feeding of the below ground assets. The second case study considers the construction of a treatment system with equivalent ground and geographical conditions but is a new site. Therefore, the construction of auxiliaries such as security fencing and infrastructure such as roads need to be considered. Both case studies examined 15 population equivalent size flowsheets: 5, 10, 20, 30, 50, 75, 100, 150, 200, 300, 400, 500, 600, 800, and 1,000 PE to assess the impact scale has on the significance of the key design choices. For both case studies, current STS design practice will be considered for a baseline of comparison.

2.2. Flowsheet influent and consent

British Water Code of Practice flows and loads (British Water 2013) were used for the influent loading of the flowsheets (Table 1). The design flow of the flowsheets is 0.375 m^3 d^{-1} PE^{-1} and a peak flow of 0.6 m^3 d^{-1} PE^{-1} equating to three dry weather flows (DWFs) (Supplementary material, Appendix A). The flowsheets for improved rural treatment works were designed to reach a secondary treatment level (Table 1). Given that STS are unconsented systems (Withers *et al.* 2012) the STS flowsheets were not designed to treatment consent but instead government recommendation (Scottish Government 2019) to reflect current practice.

2.3. Flowsheet design

Septic tanks in the STS were assumed to have a minimum hydraulic retention time (HRT) of 12 h at peak flow. The tank is installed below ground as per the manufacturer's recommendations. All submerged tanks in this study were assumed to have a maximum design volume of 65 m^3. The desludging of the septic tanks is assumed to be yearly based on British guidance (HM Government 2015) and Scottish Water operated tanks (Jefferson 2022). The drainage field is sized according to the Scottish Government guidance (2019) giving an area of 12.5 m^2 PE^{-1} for a field with a v_p of 50 mm min^{-1}. The drainage field is assumed to have a maximum length of 20 m (Scottish Government 2019) and be constructed from perforated 13 mm low density polyethylene piping. The pipes are assumed to be installed in 0.8-m-deep and 0.5-m-wide trenches filled with a media of 4–10 mm pea gravel (Scottish Government 2019). The trenches were assumed to be 2 m from any other drainage trenches.

The SAF package treatment plants were selected from available commercial units based on treatment hydraulic loading rates and that certified effluent was below the target range (Premier Tech Aqua 2017; Tricel Environmental 2022). The flowsheets were assumed to be installed as per the manufacturer recommendation (Tricel Environmental 2022). The effluent of the SAF is assumed to be discharged to surface waters.

The EST was designed to have a minimum HRT of 4 h at peak flow, with four internal baffles. This was based on recent low temperature hydrolysis characterisation and pilot-scale reactor testing, which showed baffles were required to produce an extended desludging interval of 7 years rather than simply increasing the overall septic tank volume (Jefferson 2022). EST systems were assumed to be constructed through earthworks, with a lined pit and soft cover made from synthetic rubber, using concrete posts for baffling. The wetlands were designed to current standard design guidance (Dotro *et al.* 2017) to mirror current practice in the UK (Butterworth *et al.* 2016b). The VF wetland area was determined to ensure they had a peak flow hydraulic loading of 0.12 m^3 m^{-2} d^{-1} with a square area and with a treatment depth of 0.8 m and freeboard of 0.3 m, the maximum length of a single VF wetland is 25 m. The media comprised of a main treatment layer of 55 cm of sharp sand and 10 mm shingle mixed at a 2:1 ratio with the final 5 cm layer of 10 mm shingle. The treatment layer sits above a drainage layer of 20–40 mm gravel for the bottom 15 cm and a 5-cm transition layer of 10 mm shingle (Jenkins 2017). Distribution and collection pipes, constructed from PVC piping, are positioned every 1.25 m on the surface and bed of the wetland with 0.5×0.5 m concrete splash plates placed every 1.25 m (Jenkins 2017). The system is dosed 8 times a day with 5 minutes doses delivered by a bell siphon within the EST.

The AHF wetlands design is based on a standard horizontal subsurface flow wetland used for tertiary treatment in the UK but with the adaptation of having a coarse bubble aeration grid installed on the flow of the wetland as is current practice (Butterworth *et al.* 2016b). Accordingly, the wetland was sized with a maximum treatment width of 25 m and organic loading below 15 g_{BOD} m^{-2} d^{-1} with an air flow of 0.26 m^3 PE^{-1} h^{-1} (Butterworth *et al.* 2016b). The AHF was assumed to have a treatment height of 0.6 m and a freeboard of 0.3 m containing 4–10 mm pea gravel as the main treatment media and distribution layers of 100 mm stone for the first 0.5 m from the inlet and outlet, held in place with steel gabions.

Fences were assumed to be 1.8 m steel palisade fence posts with a 3 m wide access gate (Water UK 2020b). Road infrastructure design was outsourced to quoting contractors consisting of two road designs, gravel and bitumen. Gravel roads were used when desludging visits were less than every 3 years (Jefferson 2022) otherwise bitumen roads were assumed. The drainage fields of STS are fenced off as per Scottish Water guidance.

STSs are not monitored beyond desludging. In contrast, the alternative flowsheets were assumed to be monitored monthly with electrical components inspected yearly. Replacement frequencies were assumed to be yearly for the filters, every 3 years for silencers, 5 years for bearings, and 15 years for non-return and pressure relief valves. The operator time is contracted as £35 h^{-1} with inspections lasting 1 h. Desludging was assumed to be conducted by diesel tankers, with the volume of sludge removed assumed to be 33% of the total tank volume for STS, ESTs, and SAFs. Inspection and maintenance vehicles are modelled as diesel vans with travel originating from a large works site.

2.4. Economic assessment

Costs were calculated at June 2021 values, with costs adjusted by the Consumer Price Index (CPI) (ONS 2022). Flowsheets were calculated through component costing, with each element worth more than £50 quoted in triplicate. Where off the shelf products were not available cost curves were used to determine the value of the component. The system lifetime of the project was assumed to be 30 years (Gallagher & Gill 2021).

The net present value (NPV) of operational expenditure was calculated using;

$$\mathrm{NPV} = \frac{R_t}{(1+i)^t}$$

where R_t is the cash flow at time t, t is the year of operation, and i is the discount rate. The discount rate represents the balance of interest and returns on investments against inflation, an average discount rate of 3.5% was used based on UK NET Zero and HM Treasury Greenbook guidance (Water UK 2020a).

2.5. Lifecycle analysis

The scope of the pledges given by the UK water companies for 2030 and 2040 are primarily Scope 2 with Scope 3 emissions only considered when a core activity is outsourced (Water UK 2020a). Scope 2 emissions are defined as both the direct emissions from operation and construction of the flowsheet and indirect emissions from the consumption of electricity, heating or cooling for the operation of the flowsheet but not produced by the flowsheet (Greenhouse Gas Protocol 2014). The lifecycle analysis (LCA) is analysed from cradle to grave, the system

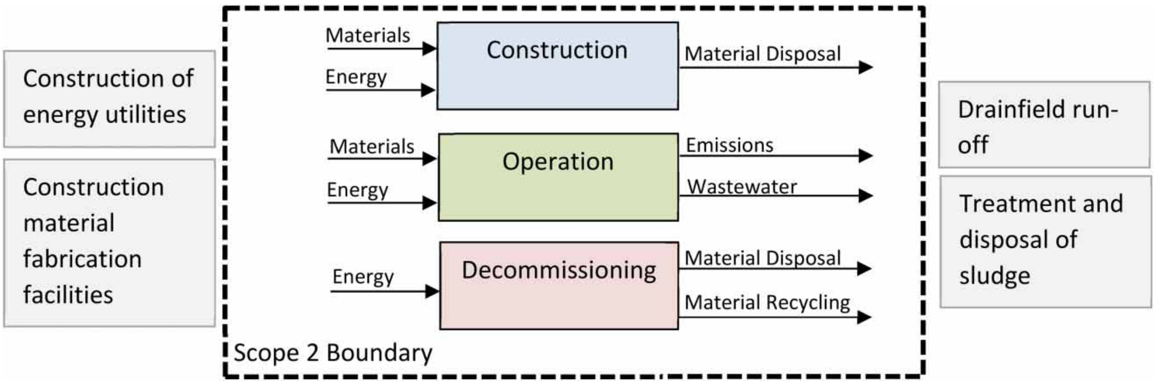

Figure 2 | LCA system boundaries.

boundaries are outlined in Figure 2, (Resende *et al.* 2019). Embedded carbon and end of life calculations were conducted using OpenLCA 1.11.0 (GreenDelta), with the Ecoinvent v3.8 database (Ecoinvent). The lifecycle impact assessment model used for this study was ReCiPe 2016 Midpoint (H).

Flowsheet process emissions are derived by best available data in the literature (Supplementary material, Table A1) and biogas flaring was modelled as flaring of sour waste natural gas with biogenic carbon dioxide release from the Ecoinvent v3.8 database. Desluding tankers were modelled as unladen on the outward journey and the additional sludge weight is assumed to have a density of 1,000 kg m^{-3}. Desludging tankers were modelled as the average carbon emissions per tonne mile of Euro 3 through to six lorries. Three lorry weight classes are considered, 3.5–7.5, 7.5–16, and 16–32 tonnes assuming capacities of 2, 10.5, and 19 m^3, respectively. The carbon emissions of UK grid electricity up to 2040 are based on UK department for Business, Energy and Industrial strategy projections (2019). Post 2040, electrical grid emissions are based on sum of root square error with an exponential decay model, modelled using MATLAB (Supplementary material, Appendix B).

Uncertainty of the capital emissions was calculated using a Monte Carlo method with 1,000 iterations (Heijungs 2020), using the uncertainty within the Ecoinvent database. Due to the large variability in reported process emissions for each system, the uncertainty in process emissions was considered qualitatively.

3. RESULTS AND DISCUSSION

3.1. Existing site study

3.1.1. Wholelife cost

At all examined populations the SAFs have the highest WLC per PE (Figure 3). To illustrate, in the case of 10 PE, the WLC of the SAF was £3,160 PE^{-1} compared to £1,100 PE^{-1} for the STS and £1,650 and £1,750 PE^{-1} for the EST-VF and EST-AHF, respectively. The difference in WLC per PE between the different options decreased as the scale increased due to economies of scale reducing the impact of fixed costs such as operator maintenance time. For example, at a PE of 1,000, the WLC per PE was £748, £439, £675, and £370 PE^{-1} for the SAF, STS, EST-VF, and EST-AHF, respectively.

Comparison across the different alternative options shows that the EST-NBS flowsheets are a lower cost upgrade to STS than SAFs (Figure 3). The WLC per PE decreased for all options but at different rates such that the WLC per PE for the EST-AHF and the STS reached parity at sizes of 150 PE and above. The critical components that delivered the cost saving per PE in the EST-AHF were the operating costs of the AHF compared to the operating components in the STS as the capital cost per capita are approximately equal. Operating costs were 81% higher than those for the STS at small scales (Figure 3) but the difference decreased with increasing scale such that at 1,000 PE the operating costs of the STS and EST-AHF are within 4%. The principle operating cost of the STS is desludging, which decreases from £776 to £226 PE^{-1} as the scale increases from 10 to 1,000 PE. In comparison, the main operating costs for the EST-AHF are the operating and maintenance costs associated with the AHF which decreases from £1,350 to £172 PE^{-1} from 10 to 1,000 PE.

Similarly, the WLC per PE of SAFs and EST-VFs is comparable beyond 150 PE but higher than the STS. For instance, at 1,000 PE the SAF and EST-VF represent an additional 70 and 54% WLC compared to the STS, respectively. However, sub-150 PE, the EST-NBS flowsheets represent a notable saving compared to SAFs and

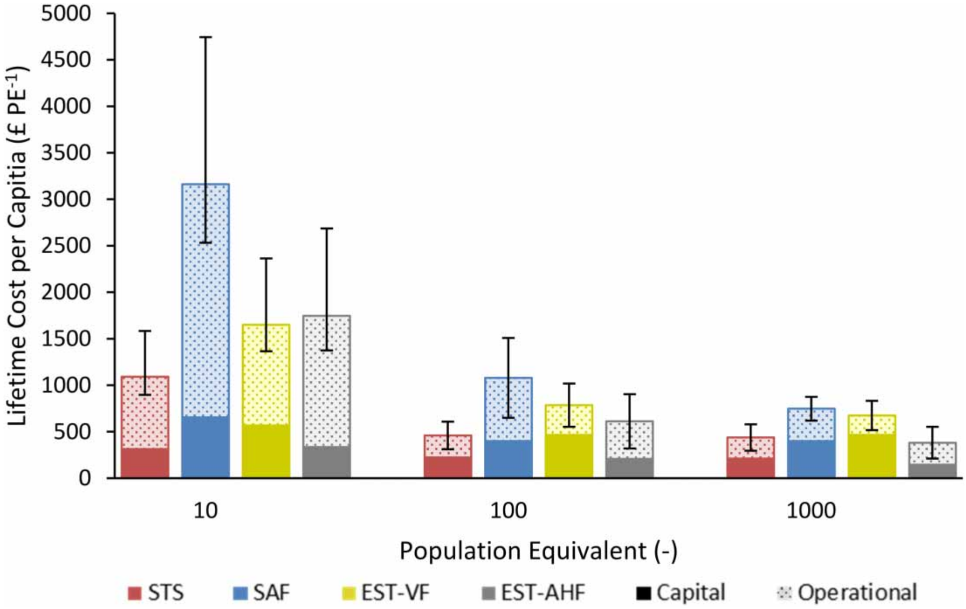

Figure 3 | 30-year WLC per capita for existing site flowsheets at 10; 100; 1,000 PE. Error bars show discount rate from 0 to 6%.

hence offer a lower cost option for enhancing treatment effectiveness compared to an STS. To illustrate, at 100 PE the WLC of an EST-VF and EST-AHF are £786 and £613 PE^{-1} compared to £1,080 PE^{-1} for a SAF. While the EST-VF is a more expensive option than the EST-AHF at larger scales, below 30 PE the EST-VF is the least cost option for improved treatment. This reflects the low operating cost of the VF system. At 10 PE the VF has a lifetime operating cost 25% lower than the AHF, however as the economies of scale reduce the impact of fixed factors such as operator transport and time, the increased capital cost of the VF system means that the EST-AHF is a lower cost alternative at larger scales.

Analysis of the cost breakdown reveals that the operating costs dominate the WLC at smaller scales but become a more minor component as scale increases reaching approximate parity with CAPEX at a scale of 30 PE, 150 PE, 75 and 1,000 PE for the SAF, STS, EST-VF, and EST-AHF, respectively. To illustrate the contribution of operating costs, at a scale of 10 PE, was 79% for the SAF, 71% for the STS, 81% for the EST-AHF, and 66% for the EST-VF (Figure 3). In comparison, at 1,000 PE the OPEX contributes 30% of the WLC per PE for the SAF and 31% for the EST-VF.

The change in WLC per PE can be utilised to derive an economy of scale exponent assuming a standard power law fit. The different options group into two sets with the exponents of 0.61 and 0.66 for the SAF and EST-AHF and 0.80 and 0.82 for the STS and EST-VF. This compares to a to standard value of 0.66 reflecting the dominance of civil structures on the overall costs for the SAF and EST-AHF. Whereas the higher values of the STS and EST-VF indicate towards more modular components similar to the exponents reported for membrane based systems (Jefferson 2022).

3.1.2. Lifetime carbon emissions

A different perspective of the flowsheets is observed when viewed through the lens of the lifetime carbon emissions (LCEs). The EST-NBS options both reduced LCE substantially compared to either the SAF or the STS at all scales of operation (Figure 4). To illustrate with respect to a 10 PE scale, the LCE was 1,740 and 1,870 kg CO_{2eq} PE^{-1} for the EST-VF and EST-AHF, respectively, and this compared to 3,150 and 4,190 kg CO_{2eq} PE^{-1} for the SAF and STS, respectively. Accordingly, adoption of the alternative options reduced the LCE compared to that of the existing STS by between 56 and 90% for the EST-AHF across all scales. The equivalent reduction across all scales was between 60 and 86% for the EST-VF and between 26 and 71% for the SAF. In terms of the EST-NBS options, the VF option has a lower LCE at small scales and reaches parity with AHF option at a scale of 100 PE. Beyond that the EST-AHF provides the lowest LCE as the capital components decreased more significantly with scale than that of the EST-VF. The reduction in capital emissions for an EST-VF is 60 kg CO_{2eq} PE^{-1} across the scale of 10–1,000 PE whereas the equivalent change for the EST-AHF is 160 kg CO_{2eq} PE^{-1} (Figure 4).

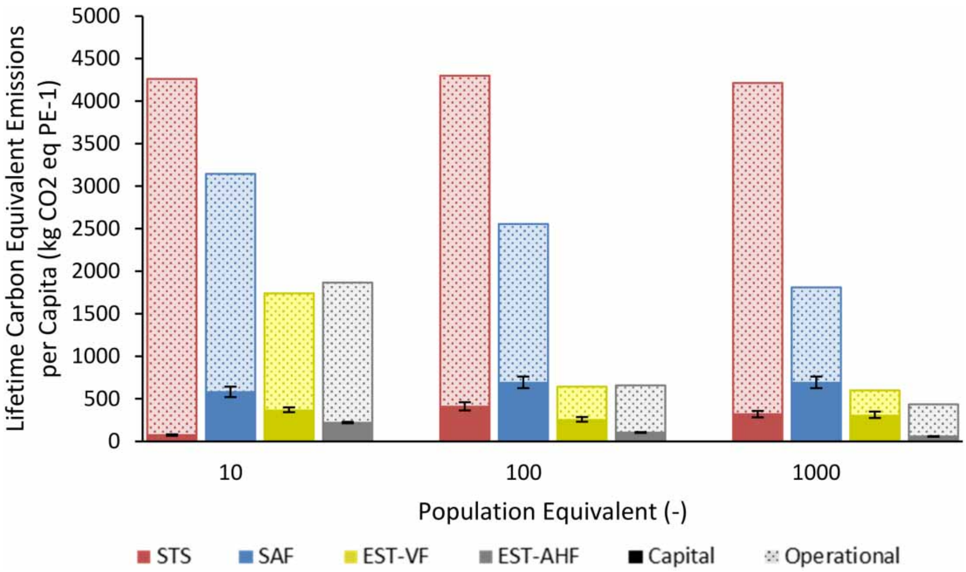

Figure 4 | 30-year LCEs per capita for existing site flowsheets at 10; 100; and 1,000 PE (error bars are 95% confidence intervals from Monte Carlo analysis).

The LCE for the STS has a negligible scale impact ranging from 4,260 kg CO_{2eq} PE^{-1} at small scale to 4,210 kg CO_{2eq} PE^{-1} at larger scales due to the fact that the majority of the LCE is derived from process emissions of methane. This is in line with other studies into the lifetime carbon footprint of septic tank flowsheets which indicated that process emissions account for 90% of the total LCE (Resende *et al.* 2019) compared to 85% in the current study. While the STS directly vents methane into the atmosphere, the other options either prevent its formation by utilising aerobic conditions (SAF) or capture it (EST-VF; EST-AHF). The impact is to reduce the LCE associated with process emissions from 129 kg CO_{2eq} PE^{-1} yr^{-1} for a 100 PE in the case of an STS to 0.98, 0.88, and 0.80 kg CO_{2eq} PE^{-1} yr^{-1} for SAF, EST-VF and EST-AHF at the same scale. Over the 30-year lifetime of the assessment, the capture of methane in the EST-NBS systems avoids 3.8 t CO_{2eq} PE^{-1} compared to the STS, this is the equivalent to 3 flights from London to New York (Ecoinvent database 2022).

With respect to a centralised wastewater treatment, consisting of an activated sludge system and anaerobic digester, the operational emissions per capita are approximately 82 kg CO_{2eq} PE^{-1} yr^{-1} (Piao *et al.* 2016), which is equivalent to a SAF flowsheet and greater than either EST-NBS flowsheet which are between 10 and 54 kg CO_{2eq} PE^{-1} yr^{-1}. The operational emissions of an STS system is greater than that of a centralised works on a per capita basis.

3.1.3. Uncertainty analysis of the LCE

The uncertainty in the capital emission of the LCE is low (Figure 4) with no significant impact on findings. On the other hand, the determination of process emissions remains the greatest area of uncertainty in the estimation of the LCE due to a lack of reported emission rates and the overall consistency of approach. To illustrate, in the case of the SAF a value of 64 gCO_{2-eq} m^{-3} wastewater treated was derived from nitrous oxide emissions for SAFs from UK carbon accounting guidance (UK WIR 2008) and methane emissions from sludge holding tanks of IPCC (IPCC 2014). However, in a recent study from China, emissions ranged from 264 to 443 gCO_{2eq} m^{-3} wastewater treated with an average of 333 gCO_{2eq} m^{-3} wastewater treated (Hua *et al.* 2022). The consequence is a potential increase in process emission of five times, based on average and nine times, based on the maximum level reported which results in an additional 4 kg CO_{2eq} PE^{-1} yr^{-1}, however, this increase does not significantly impact the findings of this study in terms of technology comparison (Figure 5).

In the case of the AHF, there is no reported data on emissions post an anaerobic process. The closest approximated reported case is for treatment of fish farm effluent with an emission rate of 1.62 kg CO_{2eq} PE^{-1} yr^{-1} (Maltais-Landry *et al.* 2009), the methane flux is in line with that reported for domestic AHF systems (Wang *et al.* 2008). Reported data does exist for the vertical flow systems with projected process emissions ranging from 0.16 to 4.6 kg CO_{2eq} PE^{-1} yr^{-1} (Mander *et al.* 2014). Nitrous oxide accounts for 10% of the process

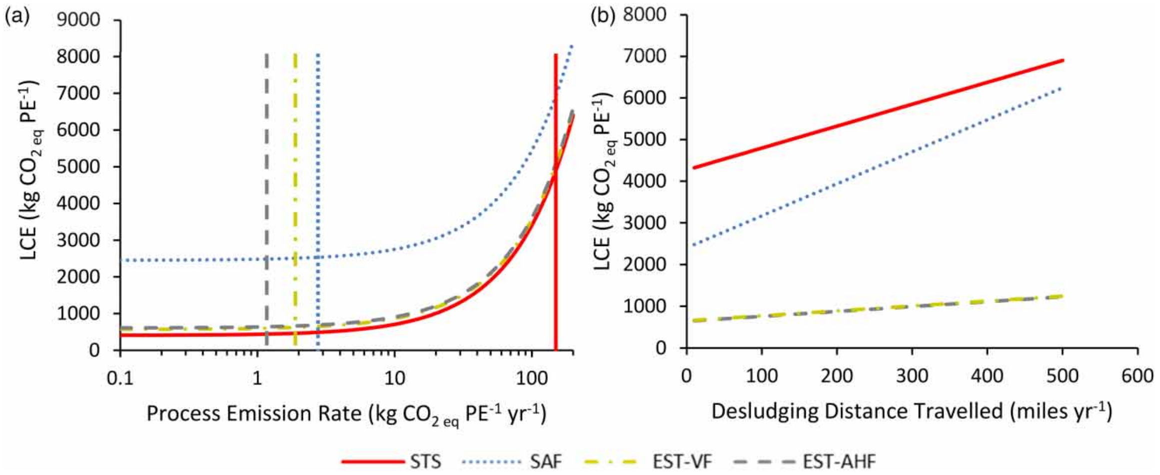

Figure 5 | Impact of (a) process emission rates and (b) desludging miles travelled on LCE for the flowsheets at 100 PE. The vertical lines shown in (a) represent the values used in the current study.

emissions of VF wetlands and 100% of AHF wetlands. This compares to 83% for the SAF revealing that management of the nitrogen treatment pathway is critical in minimising process emissions from such flowsheets. The general indications of how to minimise nitrous oxide emission are linked to avoiding overloading, running as close to steady state as possible and ensuring sufficient oxygen is distributed through the biofilm (Kampschreur *et al.* 2009). Comparing the operation of the NBS wetlands, the AHF runs in a continuous mode opposed to the batch nature of VFs. However, despite the expectation of an AHF having a lower N_2O emission rate than a VF, currently available literature gives a nitrous oxide emission rate for an AHF as 1.12 g N_2O kg $TN^{-1}_{treated}$ (Maltais-Landry *et al.* 2009) compared to 0.18 g N_2O kg $TN^{-1}_{treated}$ for a VF (Mander *et al.* 2014). Measurement and establishing conditions to minimise overall process emissions remains an area of need and a recommendation for future work.

Interestingly, the estimation of methane emissions from septic tanks is consistent across different studies, with numerous studies reporting a value of 11 gCH_4 PE^{-1} d^{-1} (Diaz-Valbuena *et al.* 2011; Truhlar *et al.* 2016; Huynh *et al.* 2021). This equates to an overall emission rate of 112 kg CO_{2eq} PE^{-1} yr^{-1}.

The impact of possible variation in emission rate was established for the four technologies through a sensitivity analysis (Figure 5). The uncertainty of the actual emission rate is unlikely to impact the relative findings of the study. For instance, the other emissions for the SAF mean that considering all improved flowsheets to have the same process emissions would have negligible impact of the study's findings in terms of the SAF (Figure 5). Further, the uncertainty in the process emission between the two NBS options means that they can be effectively viewed as similar. Overall, the EST-NBS systems can be viewed as able to deliver the lowest LCE of the four options considered irrespective of the uncertainty. For instance, for an EST-AHF to have a greater carbon footprint than SAFs this would require an additional 780 kg CO_{2eq} PE^{-1} at 1,000 PE.

The other area of major uncertainty is related to the desludging frequency and distances travelled. The desludging frequency of the packaged treatment systems and the septic tanks is approximately 1 year and this compares to the EST-NBS systems of 7 years. However, the desludging frequency of water utility operated septic tanks is known to vary from 3 to 24 months and is commonly accomplished through an integrated schedule to desludge multiple septic tanks in individual tankering trips. The contribution desludging makes to the operational emissions is 1, 8, 1, and 1% at 100 PE with a distance of 40 miles for the STS, SAF, EST-VF, and EST-AHF, respectively. If both the distance travelled and desludging frequency were double that of the modelled systems (or equivalent) there would be a 13, 30, and 19% increase in the LCE of the STS, SAF, and EST-NBS flowsheets, respectively. This could be the case for particularly remote sites such as flowsheets situated on small islands where tankers are required to travel back to the mainland. However, although the desludging can have a significant impact on the LCE it does not impact the findings with respect to flowsheets with reasonable operational variation (Figure 5(b)). For example, with a travelling distance of 40 miles per trip it would require desludging every 60 days for the EST-NBS flowsheets to have an equivalent LCE to a SAF.

3.1.4. Abatement potential

Over the predicted life span of 30 years, the relative cost of upgrading to an EST-VF at 10 PE is £560 PE^{-1} which produces a lifetime carbon reduction of 2,520 kg CO_{2eq} PE^{-1}, giving an abatement cost of £220 tCO_{2eq}^{-1} over 30 years. Comparatively at 10 PE, the abatement cost of a SAF and EST-AHF is £1,860 and £260 tCO_{2eq}^{-1}. At larger populations, the abatement costs of the alternative options are reduced as the price differential from an improved flowsheet to STS decreases while the carbon reduction increases. For example, at the 1,000 PE scale, the abatement cost of a SAF, EST-VF and EST-AHF are £92, and £65, and –£17 tCO_2 eq^{-1}. The negative number represents a cost-effective investment, providing an operational saving alongside carbon reduction, for example, a reduction in travel both saves money and fuel and the associated carbon emissions. The significances of the estimated abatement costs can be considered by framing the number within the general abatement costs published for selected technologies for the water sector (Table 2). Overall, this demonstrates that the proposed alternatives can provide a cost-effective means of carbon abatement while delivering a higher level of treatment. To illustrate, at 1,000 PE an EST-AHF has a lower lifetime cost and emissions than STS meaning the abatement cost is –£17 tCO_2 eq^{-1}, a more cost-effective abatement technique than the injection of biomethane to the grid.

3.2. New site study

In the case of new sites, additional consideration needs to be given for infrastructure costs associated to road and fencing. For instance, if the site requires regular desludging visits such as with the STS and SAF, a bitumen road is required (Jefferson 2022). If the desludging frequency is less than every 3 years, then gravel road is considered suitable (Jefferson 2022). The impact is seen through the difference in cost and emissions of the two options. Bitumen based roads have a cost of £130 m^{-1} and emissions of 66 gCO_{2eq} m^{-1} $lane^{-1}$ compared to £26 m^{-1} and emissions of 4.9 gCO_{2eq} m^{-1} $lane^{-1}$ for gravel-based roads (Espinoza *et al.* 2019). Consequently, there is a differential cost of £104 m^{-1} and an emission of 61.1 gCO_{2eq} m^{-1} $lane^{-1}$. The exact impact will depend on the specific distances associated with a given site but applies for all options. Consequently, the cost and emissions for the NBS system will always be lower, further reinforcing the positive impact compared to a SAF or STS observed for the existing site case study.

The other major additional cost is associated with fencing which is dependent on the land footprint of the system. The footprint of the different options at 100 PE was 1,260, 50, 589, and 360 m^2 PE^{-1} for the STS, SAF, EST-VF and EST-AHF, respectively. This converts to a fencing cost of £10,000 for STSs and £2,600, £4,100, and £6,000 for SAF, EST-VF and EST-AHF, respectively. The lower costs for the EST-NBS over the STS represent the cost associated with the drainage field. This equates to a 22% increase in the lifetime costs of an STS, however for the improved systems the impact is lower being 3, 8, and 7% for SAF, EST-VF, and EST-AHF, respectively. Overall, fencing does not significantly change the assessment of costing compared to an existing system. For instance, at 100 PE an STS remains the lowest cost system, followed EST-AHF, then EST-VF and finally SAF.

In more rural sites there is the possibility that an electrical connection is not possible. The SAF and EST-AHF are dependent on an electrical connection to operate, compared to an STS and EST-VF which can operate passively. The cost of installing mains electricity connection is £13,000 for a small business and can rise to up to £50,000 (Northern Power Grid 2022). Onsite power generation could also be used but it is likely that a redundancy connection would be required. Therefore, in these scenarios, the adoption of a passive system such as the EST-VF would likely be beneficial.

Table 2 | The carbon reduction and the abatement cost of selected technologies for the water sector (Water UK 2021)

Technology	Details	Abatement cost (£ tCO_{2eq}^{-1})
Transport reduction	Reduction in fleet travelling	−100 to −500
Solar power	Reaching 20% of annual energy consumption via solar power	−100 to −500
Wind power	Reaching 20% of annual energy consumption via wind power	−100 to −500
Biomethane	Grid injection of biomethane	>10
Alternative anaerobic digestion	Upgrading conventional anaerobic digestion to thermal hydrolysis	>100
Natural sequestration	Commitment to plant 11 million trees by 2030	>1,000

3.3. Future outlook

A key to the lower LCE for the EST-NBS options is the fact that no methane is vented into the atmosphere. However, if the gas from the EST systems were vented then the overall LCE would be unfavourable compared to the STS and show the importance of gas management. The retrofitting of flares to STS could provide a low-cost method of reducing the carbon load of STS, however it will not provide the additional treatment capabilities of the improved flowsheets and therefore is not a suitable solution if increased treatment is required.

Instead of flaring, the produced methane could be utilised to provide heating for the EST or used within the local community providing the potential for further carbon offsets. A similar proposal for a sewer mining project to irrigate a park in Australia proposed the use of the gas produced to power several BBQ rings to enable people visiting the park to cook their lunch (Jefferson 2022). While there is currently insufficient knowledge to assess the feasibility of the different options, it provides an illustration of approaches to add additional value from the EST-NBS flowsheets.

Further, the above assessment has been based on standard practice and has assumed a standard wetland design has been used. However, the adoption of the NBS component offers further opportunity concerning co-benefits such as biodiversity gain and social benefits such as improving mental health and resilience associated with people interacting with green infrastructure. Accordingly, the wetlands could be designed to be open to the public or protected using green security fencing solutions such as Hawthorn, which provide an analogous level of security to that of palisade fencing. The costs implication is an increase from £110 m^{-1} for steel palisade fence posts to £140 m^{-1} for Hawthorn. However, hedge rows have the potential for significant carbon storage (Axe *et al.* 2017), coupled with the improved aesthetic value this is an adjustment to flowsheets which warrants further investigation. Taking the concept further, the wetlands can be designed for enhanced aesthetic value with access for the public to spend time by the wetlands (Figure 6). A separation of the anaerobic system from the accessible areas will be required for the use of a flare to be accessible and efforts should be may to limit the visual impact of any flaring. The aesthetic nature of the system has the potential to reposition the treatment works in relation to the local community to add social value and engagement such that low level maintenance could be conducted by members of the local community. While such considerations require development, they offer additional benefits to the basic items considered in this study and hence further enhance the case for adoption.

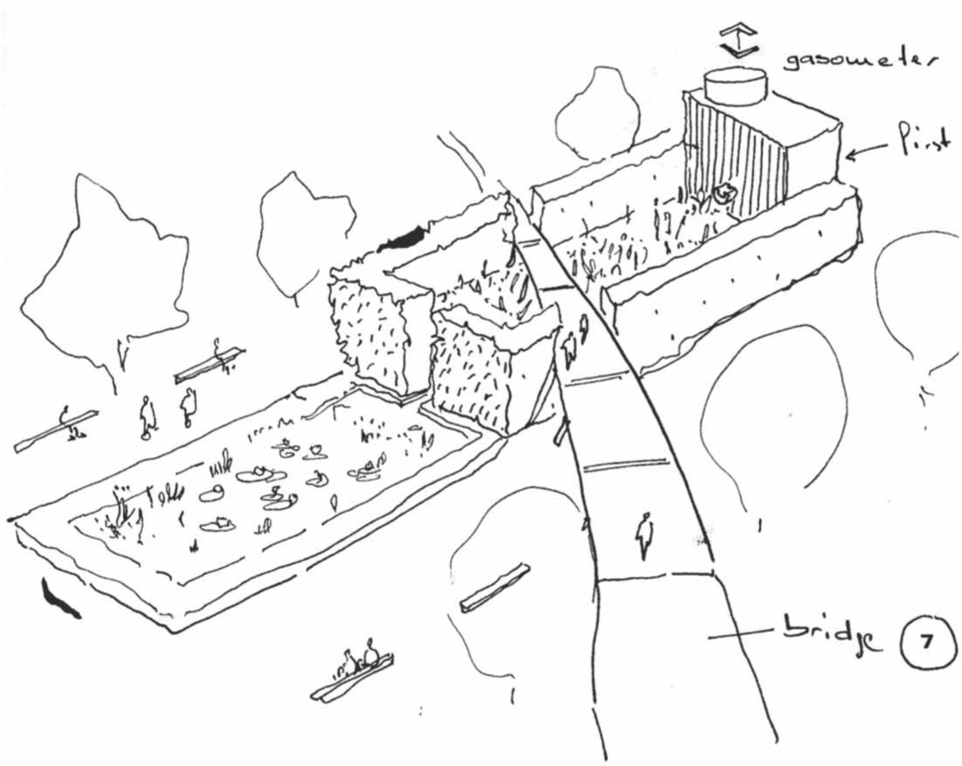

Figure 6 | Treatment garden concept from Professor Anton James, RMIT University (Jefferson 2022).

4. CONCLUSIONS

Nature-based solutions to rural wastewater provide a lower cost method of improving treatment of septic tank systems compared to SAFs at both new and existing sites. In addition to being a cost-effective solution, the NBS flowsheets have lower carbon emissions than more intensified SAF at populations from 5 to 1,000 PE. The work shows the potential that additional focus on rural treatment can have on the carbon emissions of the UK water sector, comparable in cost effectiveness to upgrading large anaerobic digesters and biomethane grid injection. This is coupled with improved water treatment, reducing the eutrophication burden which STS impose. There is scope to expand the flowsheet to provide a social space and societal gain, alongside treatment and carbon reduction.

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DATA AVAILABILITY STATEMENT

All relevant data are available from an online repository or repositories (https://cord.cranfield.ac.uk/articles/software/Data_underpinning_NERC_Research_Translation_Grassland_Management_project/11359613).

CONFLICT OF INTEREST

The authors declare there is no conflict.

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