water

MDPI

Article

Comparing the Life Cycle Energy Consumption, Global Warming and Eutrophication Potentials of Several Water and Waste Service Options

Xiaobo Xue [1,*], Troy R. Hawkins [2], Mary E. Schoen [3], Jay Garland [4] and Nicholas J. Ashbolt [5]

1 Department of Environmental Health Sciences, State University of New York at Albany, Rensselaer, NY 12144, USA
2 Enviance, 5780 Fleet Street, Suite 200, Carlsbad, CA 92008, USA; trh@alumni.cmu.edu
3 Soller Environmental, Inc., 3022 King Street, Berkeley, CA 94703, USA; mschoen@sollerenvironmental.com
4 U.S. Environmental Protection Agency, Office of Research and Development, National Exposure Research Laboratory, 26 W. Martin Luther King Drive, Cincinnati, OH 45268, USA; Garland.Jay@epa.gov
5 School of Public Health, University of Alberta, Edmonton, AB T6G 2G7, Canada; ashbolt@ualberta.ca
* Correspondence: xxue@albany.edu; Tel.: +1-518-402-0273

Academic Editor: Andreas N. Angelakis
Received: 21 January 2016; Accepted: 1 April 2016; Published: 20 April 2016

Abstract: Managing the water-energy-nutrient nexus for the built environment requires, in part, a full system analysis of energy consumption, global warming and eutrophication potentials of municipal water services. As an example, we evaluated the life cycle energy use, greenhouse gas (GHG) emissions and aqueous nutrient releases of the whole anthropogenic municipal water cycle starting from raw water extraction to wastewater treatment and reuse/discharge for five municipal water and wastewater systems. The assessed options included conventional centralized services and four alternative options following the principles of source-separation and water fit-for-purpose. The comparative life cycle assessment identified that centralized drinking water supply coupled with blackwater energy recovery and on-site greywater treatment and reuse was the most energy- and carbon-efficient water service system evaluated, while the conventional (drinking water and sewerage) centralized system ranked as the most energy- and carbon-intensive system. The electricity generated from blackwater and food residuals co-digestion was estimated to offset at least 40% of life cycle energy consumption for water/waste services. The dry composting toilet option demonstrated the lowest life cycle eutrophication potential. The nutrients in wastewater effluent are the dominating contributors for the eutrophication potential for the assessed system configurations. Among the parameters for which variability and sensitivity were evaluated, the carbon intensity of the local electricity grid and the efficiency of electricity production by the co-digestion with the energy recovery process were the most important for determining the relative global warming potential results.

Keywords: Life cycle assessment; water service; sanitation service; energy; greenhouse gas emission; nutrient

1. Introduction

Satisfying the demand for water and sanitation services currently requires significant amounts of energy to collect, treat, and deliver drinking water, and to collect, treat and dispose of resulting wastewater [1]. Water and wastewater utilities are among the largest consumers of energy in municipalities, regions, and countries [2]. Meanwhile, the effluent from wastewater treatment facilities such as septic systems and secondary treatment facilities contains significant amounts of nitrogen and phosphorus and ranks as a significant contributor for riverine and coastal eutrophication [3].

As major energy users and nutrient releasers, shifting the current municipal water services to a resource recovery–oriented, novel design presents promising opportunities to reduce energy consumption and to mitigate eutrophication [4].

A variety of possible solutions can be defined based on the principle of resource recovery, such as greywater treatment and reuse, blackwater co-digestion with food waste for energy and nutrient recovery, diverted urine and feces for fertilizer and soil conditioner, and rainwater harvesting for local uses. Given that greywater accounts for some 70% of wastewater in a conventional sewer [5], reusing treated greywater within the producing household could dramatically reduce the need for large sewers and drinking water supplies [5]. More efficient energy use and nutrient recovery would also be made possible by separating greywater from energy-concentrated food and other blackwater residuals. Further, the diverted urine and feces could be used as a replacement for energy-intensive synthetic fertilizers in nearby farmlands/gardens, and could potentially offset energy requirements for producing synthetic fertilizers [6–8]. In addition, rainwater harvesting and reuse provide the possibility of providing various water supplies and enhancing system resilience to storm events, drought and water shortage. The technical feasibility of options for providing water-related services incorporating source separation and water fit-for-purpose has been demonstrated in pilot projects in Europe, North America and Oceania [9–16]. The prospects and potential energy impacts of these novel technologies in the U.S. are currently under exploration [17].

Life Cycle Assessment (LCA) is a well-established system assessment method to quantify energy consumption and environmental impacts through the entire life cycle of a product or process. LCA has been widely used to evaluate the system performances of water and wastewater technologies. For example, LCA studies have addressed specific aspects of conventional centralized drinking or wastewater systems, *i.e.*, various options for drinking water supply systems [18–21], centralized and decentralized wastewater treatment [22–24], stormwater management strategies [25], or entire water and wastewater service systems [26]. Several recent review articles summarized the LCA developments in the water management area and emphasized the continuing research needs [24,26,27]. In particular, research needs include (1) a broader system boundary to include the entire water and wastewater services; (2) a transparent life cycle inventory for emerging treatment technologies including resource recovery and fit-for-purpose design options; and (3) a robust inventory assessment with variability and sensitivity analyses.

Due to the promotion of integrated water management, recent publications have investigated the life cycle impacts of the entire water and wastewater services [26]. However, the majority of these studies are focused on the centralized service options [24,27], which are not necessarily the most sustainable solutions. Meanwhile, recent LCA studies have started to pay attention to the resource recovery–orientated wastewater services [28–34] due to their potentials of closing water and nutrient loops. Most of these resource recovery–based LCA studies only focused on wastewater services, and excluded the benefits of saving energy from water services. As an exception, Remy and Jekel [28] included the energy and carbon benefits of saving energy from water services, but their study did not include the inventory of nutrient releases. In addition, the comparative conclusions between the resource recovery-based technologies and centralized design are not consistent. Thibodeau *et al.* [33] suggested that the blackwater source separation showed a higher global warming impact than the centralized design. On the other hand, Tillman *et al.* [34] estimated that blackwater and urine source separation had lower environmental impacts than the existing centralized design. Benetto *et al.* [29] reported that a urine source separation system coupled with greywater treatment required less energy and emitted less greenhouse gas emission than the centralized system. Lehtoranta *et al.* [32] evaluated several on-site wastewater treatment options and recognized that the dry toilet with greywater treatment had the least environmental impact among the selected sanitation services. Lam *et al.* [31] compared several sanitation services and suggested that urine separation with greywater treatment exhibited the lowest environmental impacts. In summary, it is evident that studies providing transparent and comparative energy, GHG and nutrient inventory of the whole water and wastewater service systems are needed. In addition, detailed variability/sensitivity analysis is valuable for interpreting life cycle impacts.

Therefore, the life cycle energy consumption, GHG emissions and nutrient releases of conventional centralized and four resource recovery-based water and waste service systems were estimated, using the Cape Cod region as a case study. Based on local datasets obtained from the town of Falmouth in Cape Cod, when available, a detailed life cycle inventory was compiled for the existing and proposed centralized water and wastewater systems, and for alternative design options including dry composting toilets, urine-diversion toilets, low-flush toilets, greywater treatment and household reuse, rainwater treatment and indoor use, and blackwater and food co-digestion. In addition, the environmental benefits of multiple measures including water demand and supply management, greywater reuse, and adoption of resource recovery–based water system configurations were estimated.

2. Methods

Water and waste systems were investigated using data relevant to townships within Cape Cod, Massachusetts. The Cape Cod region is characterized as having a humid climate, averaging 40 in of annual precipitation based on the last 50 years [35]. Where more specific data were available, the parameters were based on the Falmouth community, which is the second-largest municipality in Cape Cod with a population of 31,500 as of 2011 [36], located in southwestern Cape Cod. The Cape Cod community is evaluating a range of potential water system configurations in order to mitigate the coastal eutrophication, reduce energy consumption, protect public health, and sustain economic development. In order to assist in the evaluation of the technology candidates, we assessed the water system options in a series of work from the environmental, human health [37], resilience [38], and economic [39] and overall metric perspectives [27]. As an integral part of holistic assessment, this analysis followed the LCA standard principle described by the International Organization for Standardization's (ISO) 14040 series to quantify the energy consumption, GHG emissions and nutrient releases of five design options.

Among a range of potential system configurations, we quantified the business as usual (BAU) centralized services and four alternative decentralized systems with the capability of recovering resources. The first alternative system collected household urine and feces in dry composting toilets with greywater diverted to the existing septic system (CT-SS), and use of the current municipal drinking water supply. The second alternative option included urine-diverting toilets with all other wastewater to existing septic systems (UD-SS) to capture urine nutrients. The third alternative option utilized low-flush toilets connected to a blackwater pressure municipal sewer and household greywater treatment and reuse for toilet flushing, washing of clothes and lawn irrigation (BE-GR). This option included community food residuals co-digestion for combined heat and power (CHP) recovery and digestate from co-digestion for alternative fertilizer. The fourth alternative option (BE-GRR) added to the BE-GR innovation rainwater harvesting and treatment with disinfection to supplement the hot water household supply (Figure 1).

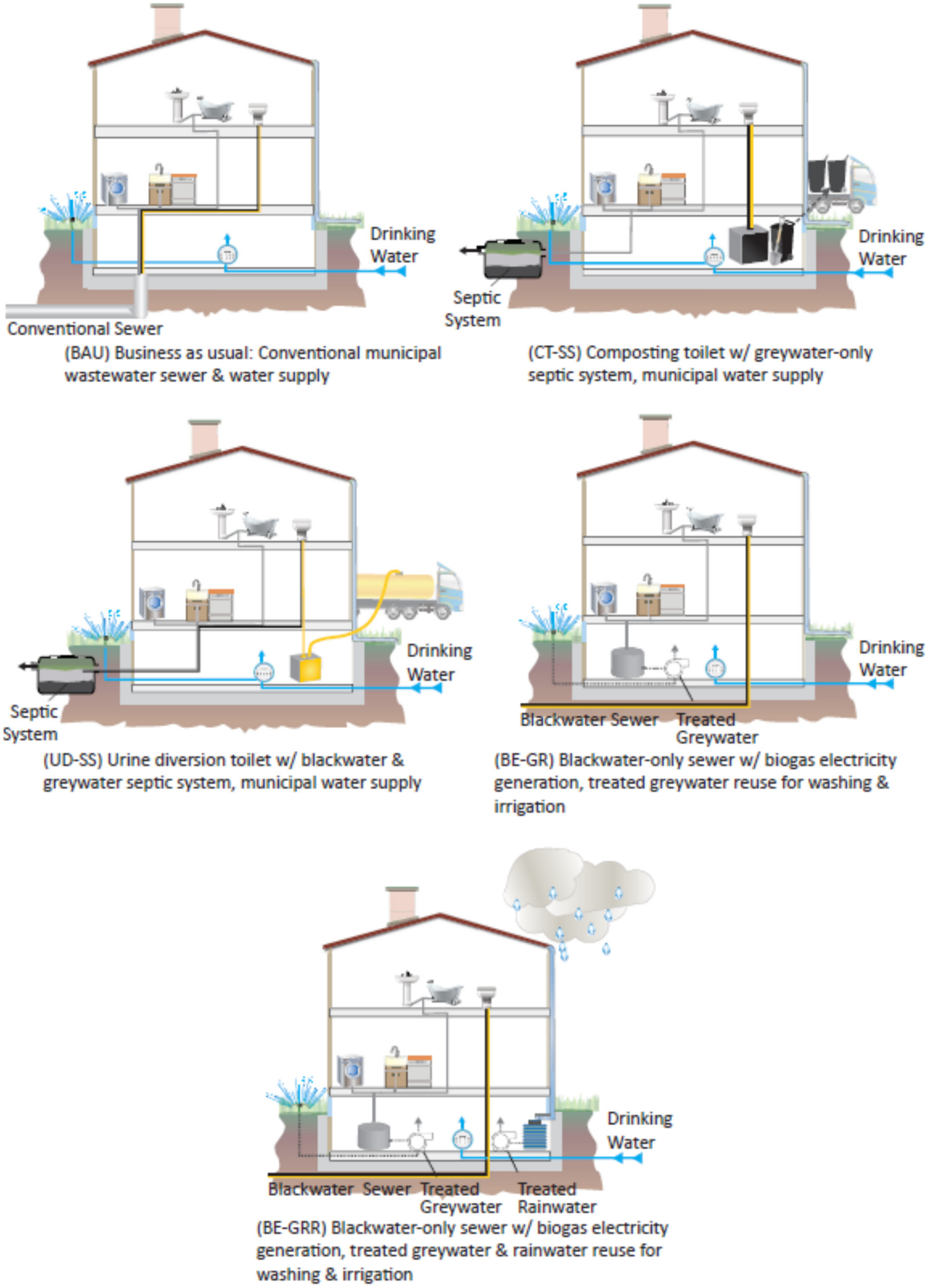

Figure 1. Water system configurations. Reprinted with permission from [37].

2.1. Goal and Scope

The goal of the study was to estimate the energy consumption, global warming potential (GWP) and eutrophication potential (EU) over the life cycles of the selected options, and to compare the energy reduction potentials of multiple strategies. The scope of this study included the material and energy inputs, and associated environmental releases during the construction, operation and maintenance of water service starting from water extraction and ending with wastewater discharge/reuse. The environmental releases focused on the greenhouse gas emissions and nutrient releases. Different lifetimes of the system elements were taken into account in the maintenance stage. The energy and GWP implications of the end-of-life handling of system components were assumed to be negligible, while the management processes associated with conventional wastewater, greywater, blackwater, rainwater, and novel toilets were explicitly modeled.

Aligned with several previous LCA studies, a service-oriented functional unit was used in this study to reflect households' water and sanitation requirements [26]. The functional unit was one household's water and sanitation demands. Figure S1 describes the household's water and sanitation demands. While compost and urine collected from CT-SS and UD-SS options is used as nitrogen and phosphorus fertilizers, blackwater from BE-GR and BE-GRR is co-digested with solid waste for energy supply and residuals for local fertilizer. In order to represent the fertilizer and energy benefits from the alternative systems, system expansion was practiced in order to ensure fair comparison. BAU, which does not supply energy or fertilizer, was augmented with additional grid electricity and synthetic fertilizer, so that each scenario supplied the equivalent amount of electricity, nitrogen and phosphorus fertilizers. Similarly, CT-SS and UD-SS were expanded with additional grid electricity. Figure S2 describes the functional unit and system expansion.

2.2. Life Cycle Inventory and Impact Assessment

The Falmouth community consumes about 4.6 million gallons per day (MGD) of water, approximately 60% of which is extracted from surface sources with the residual from a shallow aquifer [40]. All sources were treated through the conventional processes including alum coagulation, sedimentation, filtration with pH adjustment, and chlorine gas disinfection. The electricity demand for potable water treatment and distribution via 625 km of mains was estimated from local utility datasets [40]. Drinking water loss via the distribution system was considered to vary from 8% to 15%, in accordance with national averages [41,42]. The energy and material needs for water extraction, treatment and distribution were supplied by local utilities.

The national average estimates for energy and chemical inputs required for activated sludge wastewater treatment were utilized to model the life cycle inventory of sewer treatment and discharge [43]. The energy consumption for sewer collection is based on pilot design plans for Cape Cod [44]. The sludge from water and wastewater services is assumed to transport to the local landfill via truck.

Based on consultation with multiple composting toilet manufacturers (such as Sun-mar and Phoenix), the average electrical load for operating a fan for the composting toilets was 5 $W \cdot d^{-1}$. Low-volume flush toilet use for options BE-GR and BE-GRR (Figure 1) was assumed to be 500 $J \cdot flush^{-1}$ (provided for the PropelAir^{TM}) and to be used four times per day for each of the three household members.

For the options utilizing a septic tank (CT-SS, UD-SS, Figure 1), the energy consumption required for pumping and transporting residuals from the household was estimated at 5 $MJ \cdot (year \cdot household)^{-1}$ and 68 $MJ \cdot (year \cdot household)^{-1}$, respectively. The energy estimates for septic tank cleaning and transporting residuals were based on the assumption that a 9500 L vacuum truck was utilized to clean a septic tank every three years [45]. Additionally, for the annual collection and transport of compost (CT-SS) and collected yellow water (UD-SS), the estimated energy consumptions were 440 $MJ \cdot household^{-1}$ for CT-SS and 1280 $MJ \cdot household^{-1}$ for UD-SS. These estimates were based on an adult producing 0.5–2.5 L of urine per adult per day [46] and flushwater volumes of 0.2–0.6 $L \cdot flush^{-1}$ [15,47]. We assumed a 3 m^3 urine storage tank, based on 20 $L \cdot d^{-1}$, 70% of the urine collection rate, and three months of storage time before collection. Further, we assumed that urine and compost were transported to a less-nutrient-sensitive watershed where they could be used as soil amendments.

For options using decentralized greywater treatment (biological treatment, UV disinfection) and household reuse (pumped to the household for toilet flushing/clothes washing), the energy consumption depended on the type of biological filtration process and head height pressure for reuse within the household. We consulted several manufactures and utilized the likely range (1.44–5.4 $MJ \cdot m^{-3}$) for on-site filtration [46]. Small-scale medium-pressure UV lamps were assumed for greywater disinfection, operating at 0.02–0.08 $mJ \cdot m^{-2}$ at 35% of UV light efficiency to reach the water quality requirement for toilet flushing [48,49]. The additional energy required for pumping treated greywater to its destination was estimated based on the flow rate, water pressure, pump and motor efficiencies [50] described in Table S6.

For the rainwater harvesting (BE-GRR, Figure 1) option, we assumed a typical roof area of 90 m^2 with 85% of that area connected to the collection system, and 80% reliability of supply with a 3 m^3 rainwater tank (Table S2). The on-site rain water treatment processes include in-line filtration (nominal 20 μm household water filter), household UV disinfection, and pressure distribution of treated rainwater to the hot water heater system. The energy consumption for infiltration and disinfection processes was calculated based on manufacturing and pilot testing datasets [50,51]. In addition, the energy consumption for redistributing the treated rainwater was estimated according to the pump performance curve of a Grundfos CH2 pump, as adopted for multiple Australian on-site rainwater treatment systems [50,51] (Table S7).

The BE-GR and BE-GRR systems differ from CT-SS and UD-SS in their capability for CHP generation from the source-separated blackwater and food co-digestion components. The energy consumption for blackwater transport within a pressure sewer was estimated from basic design principles [52] and case studies conducted in Europe [10,11]. Several reports commissioned by the U.S. Environmental Protection Agency, Department of Energy, Department of Agriculture, and the State of Massachusetts [53–56] together with the U.S. EPA's Co-digestion Analysis model [57] were used to estimate the availability of additional inputs for the bioreactor sourced from household food waste and restaurant grease traps in Falmouth, and the electricity generation capacity of the associated methanogenic digestion and electricity generation processes. Based on blackwater, household waste, and restaurant grease trap residuals, the total daily digester load was estimated at $1.4 \cdot \times 10^6$ $kg \cdot d^{-1}$ (Table S10). The biogas generation rate was estimated to be 0.75 to 1.12 $m^3 \cdot kg^{-1}$ of volatile solid converted [58]. Approximately 10% of the biogas produced from the co-digestion process was assumed to be used to operate the plant and provide electricity for the associated buildings [55]. We estimated the biogas methane content to be 60%, used $3.5 \cdot \times 10^4$ $kJ \cdot m^{-3}$ as the heat value of methane gas [58,59], and estimated a 55% methane-to-electricity conversion efficiency [59]. To allow for comparison of the systems, the landfilling of restaurant grease trap and household food wastes were included in the functional unit of the BAU, CT-SS, and UD-SS systems. Since the assumptions for collecting and transporting restaurant grease trap and house food wastes were the same for all scenarios, the calculations of collecting and transporting solid waste were not included in the inventory due to the comparison purpose of this study.

The energy consumption for extracting raw materials and manufacturing pipes was estimated from various sources; however, the energy consumption for producing toilets was excluded due to the similarity of material use for the different types of toilets and their negligible influences on comparative life cycle impacts. More detail regarding the material and construction requirements for the centralized water treatment plant, wastewater treatment plant, on-site septic tank-leach fields, greywater storage and treatment, and rainwater storage and treatment systems is provided in the Supporting Information (Table S5), as is information on the life cycle inventories of pipes and water distribution scenarios (Table S11). In order to estimate the total pipe material requirements, a length-weighted average cross-sectional area was calculated to determine an equivalent average pipe diameter and then multiplied by the total length of the distribution network. The calculations of pipe materials included uPVC, PVC, HDPE, galvanized iron, copper, ductile iron cement lined, and cast iron pipes with various diameters (ranging from 0.5 in to 60 in for water pipes, and two to 80 in for wastewater pipes described in Table S12) and fittings following the Plastic Pipe and Fitting Association data provided in the BEES database [60].

The energy used to dig the trenches for water and wastewater infrastructure was estimated according to the evacuation volume and diesel requirement by a John Deere 135G excavator. The associated greenhouse gas emissions were estimated based on diesel consumption and the NONROAD model [61]. The shipping distance for materials from storage to construction sites was assumed to range from 10 to 70 km. Ecoinvent v2.0 was used to estimate background life cycle inventory items for which new unit processes were not developed [62]. Since electricity was determined to be an important background process, both the local Falmouth and U.S. average electricity mix in the year 2010 (Tables S16 and S17) were used to simulate the life cycle GWPs [63]. Average synthetic nitrogen fertilizer was represented by urea (25%), ammonium nitrate (25%), liquid ammonia (25%), and diammonium phosphate (25%). Average phosphorus synthetic fertilizer was represented by diammonium phosphate (50%) and triple superphosphate (50%). The inputs to and releases from the production processes for nitrogen and phosphorus fertilizers were estimated using the ecoinvent processes representing European average production conditions augmented with U.S.-specific emissions data derived using the National Emissions Inventory [64]. The equivalent nutrient values of compost, urine, and methanogenesis digestate were estimated according to their nutrient content and bioavailability as derived from review studies [65,66]. The nutrient contents of digestate used for the BS and RH system elements were derived from a field study, which evaluated the effects of anaerobic digestion on digestate nutrient availability [55]. The equivalent nutrient contents of compost, urine and digestate are summarized in Table S15 of the SI.

The nitrogen and phosphorus outputs from activated sludge treatment systems at nearby towns were used to represent the nutrient discharge profile of the BAU. The volatilization of N_2O in sewer systems was calculated based on the equations derived from Short *et al.* [67]. The nitrogen and phosphorus outputs for the septic elements came from the pilot testing datasets from the Barnstable County Department of Health and the Environment in Cape Cod. The assumptions about volatilization of nitrogen species for storage and transport stages of compost and urine were provided in our prior work [39]. The nutrient releases from digestate applied as an alternative fertilizer for local farmlands were computed based on the literature values [55,68,69].

The lifetime of centralized water and wastewater treatment plants was assumed to be 50 years and the water distribution and wastewater collection network lifetimes were assumed to be 100 years. A functional life span of 15 years was assumed for blackwater transport, greywater, and rainwater pump elements. The life cycle assessment data management and calculations were performed using the open source OpenLCA software package [70]. An advantage of using OpenLCA is that the software is freely and publicly available and the datasets developed for this study can be easily transferred to promote transparency and can be easily updated to incorporate future modeling efforts. With the OpenLCA software package, we created modular unit processes to describe system elements, and connected them with background datasets to represent the full supply chains and life cycle implications and estimated the greenhouse gas and energy use implications of each. A diagram depicting connections between foreground unit processes is included in the SI (Figure S1) and key values are summarized in Table 1 below.

The life cycle energy consumption represents the cumulative energy use, which derives from both fossil energy sources such as coal, natural gas and oil and alternative sources such as biofuel, solar energy, and others. The Tool for the Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI), developed by the U.S. EPA, provides the U.S. characterization factors for global warming emissions and eutrophication-related releases [71]. The global warming potential (GWP) and eutrophication potential (EU) were calculated based on environmental releases and their corresponding characterization factors.

Table 1. Key life cycle inventory input parameters [1].

Input Parameter	Input Statistic	Range for the Sensitivity Analysis		References
	Distribution[2]	Low (5th Percentile)	High (95th Percentile)	
Carbon intensity of national average electricity mix, kg CO_2-eq· (kWh)$^{-1}$	Triangular (0.51, 0.67, 0.84)	0.56	0.80	[59,72]
Carbon intensity of Falmouth average electricity mix, kg CO_2-eq· (kWh)$^{-1}$	Triangular (0.31, 0.41, 0.53)	0.28	0.49	[59,72]
Electricity from co-digestion and CHP processes, kWh· (household· day)$^{-1}$	Triangular (270, 320, 385)	285	370	[46,53–58,60,63]
Flow rate for water distribution system, m^3· (household· day)$^{-1}$	Normal (1.7, 0.2)	1.4	2.0	[1,73]
Pump and motor efficiencies for water distribution system	Triangular (0.65, 0.78, 0.85)	0.69	0.82	[1,73]
Chemical and energy inputs for centralized water treatment plant, MJ· (household· day)$^{-1}$	Normal (0.5, 0.06)	0.40	0.65	[1,73]
Chemical and energy inputs for centralized wastewater treatment plant, MJ· (household· day)$^{-1}$	Normal (0.94, 0.24)	0.75	1.7	[43]
Flow rate for greywater distribution, m^3· (household· day)$^{-1}$	Normal (0.84, 0.08)	0.68	1.0	[50]
Pump and motor efficiencies for greywater distribution	Triangular (0.68, 0.78, 0.85)	0.69	0.82	[50]
Flow rate for blackwater transport m^3· (household· day)$^{-1}$	Normal (0.06, 0.004)	0.029	0.043	[12,15]
Pump and motor efficiencies for blackwater transport	Triangular (0.68, 0.78, 0.85)	0.69	0.82	[12,15]
Flow rate for rainwater transport, m^3· (household· day)$^{-1}$	Normal (0.68, 0.07)	0.54	0.82	[50]
Pump and motor efficiencies for rainwater transport	Triangular (0.68, 0.78, 0.85)	0.69	0.82	[50]
Electricity use for on-site filtration treatment, MJ· m^{-3}	Normal (3.6, 0.99)	1.44	5.4	[46,50,51]
UV dose for on-site UV treatment, MJ· m^{-2}	Triangular (0.02, 0.04, 0.08)	0.03	0.07	[48,49]

[1] Full dataset provided in Table S17 and S18. [2] Parameters are in parentheses, in this order: for the normal distribution, mean and standard deviation; for the triangular distribution, minimum, peak, and maximum.

2.3. Variability and Sensitivity Analysis

Monte Carlo analysis (MCA) was used to quantify the variability and uncertainty of parameters contributing to energy consumption, GWP and EU. Inputs included chemical usage in municipal water treatment, chemical usage in municipal wastewater treatment, pump efficiency, motor efficiency, UV energy intensity, carbon intensity of electricity from the grid, electricity production from co-digestion, and nutrient concentrations in effluents from various system designs (Table S17). When sufficient datasets were available, best-fit probability distributions were simulated for the input parameters. Otherwise, triangle distributions with max, most likely, and min values were assigned based on the available datasets. Anderson Darling sampling methods were used with over 10,000 iterations in a model constructed in @Risk (Palisades Corp. V6.1, Ithaca, NY, USA) using supply chain/life cycle total values for system components of the five system options exported from OpenLCA to Excel. The stochastic distributions of inputs and outputs are explained in detail in Tables S17 and S18. A sensitivity analysis was performed by perturbing each variable one at a time while holding other variables constant at their reference case values to determine its influence on the GWP results.

3. Results

3.1. BAU

From a life cycle energy perspective, the BAU scenario ranked as the most energy-intensive system option, which, on average, was estimated to require the cumulative fossil energy of

1100 MJ· (day· household)$^{-1}$ (Figure 2). For BAU, the equivalent electricity requirement accounted for approximately 40% of the total life cycle energy consumption with water and wastewater services presenting approximately 40% and 60% of the rest of the life cycle energy requirement, respectively. The relative rankings of GWP for the five systems followed those of the energy requirement results. The conventional centralized water and wastewater scenario had the highest median GWP result of 110 kg CO_2-eq· (day· household)$^{-1}$ (Figure 3). Aligned with the life cycle energy and GWP findings, the conventional centralized water and wastewater scenario had the highest EU result of 14 g N-eq· (day· household)$^{-1}$ (Figure 3). The BAU system ranked as the most significant energy consumer and environmental releaser (both global warming and eutrophication potentials), mainly because it had the highest electricity use during treated water distribution and sewage treatment stages, and zero recovery of energy and nutrients.

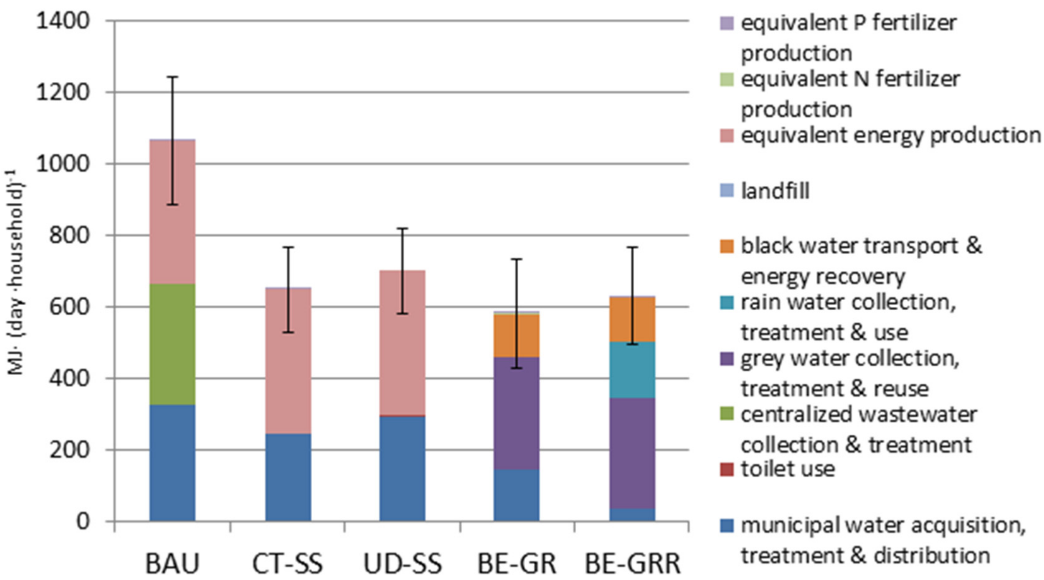

Figure 2. Life cycle energy consumption of water systems, MJ· (day· household)$^{-1}$. The range of variability bar presents the values at the assumed fifth and 95th percentiles.

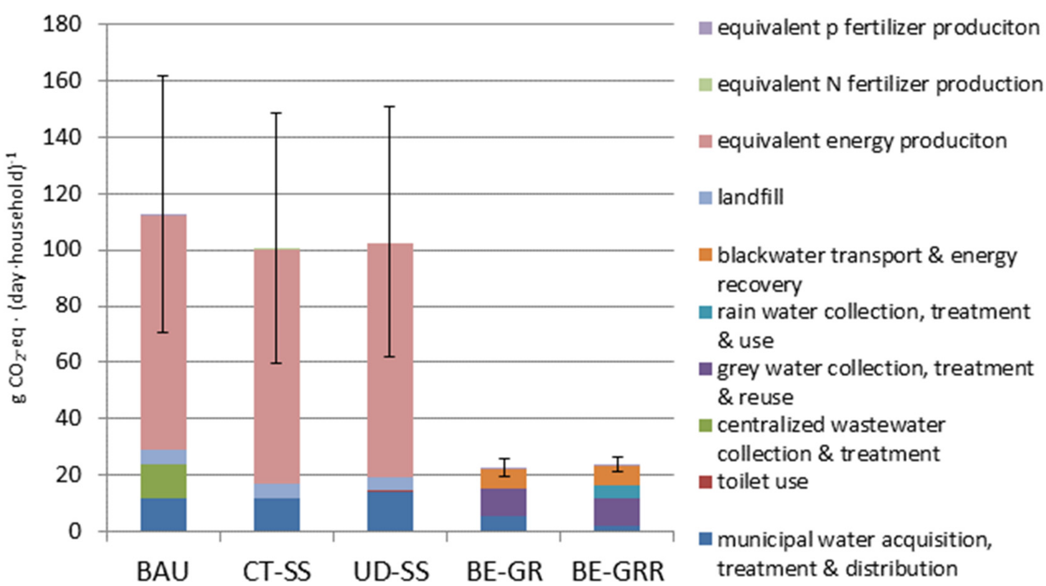

Figure 3. Life cycle global warming potentials of water systems, g CO_2-eq· (day· household)$^{-1}$. The range of variability bar presents the values at fifth and 95th percentiles.

3.2. CT-SS and US-SS

Compared with BAU, the septic tank systems (CT-SS and UD-SS) presented significant life cycle energy reduction, mainly due to the exclusion of the centralized sewer service. Because a urine-diverting toilet demands more water than a composting toilet, UD-SS showed higher life cycle energy and GWP than CT-SS. For the CT-SS and UD-SS systems, the equivalent energy production and municipal drinking water supply together were the dominating contributors, consuming more than 95% of the total life cycle energy (Figure 2). In contrast to BAU, CT-SS and UD-SS had the lowest EU, due to having the lowest nutrient discharge in septic effluent. CT-SS demonstrated a slightly lower EU than UD-SS, because the fecal solids, which contain nitrogen and phosphorus, were excluded from the septic treatment systems for CT-SS (Figure 4).

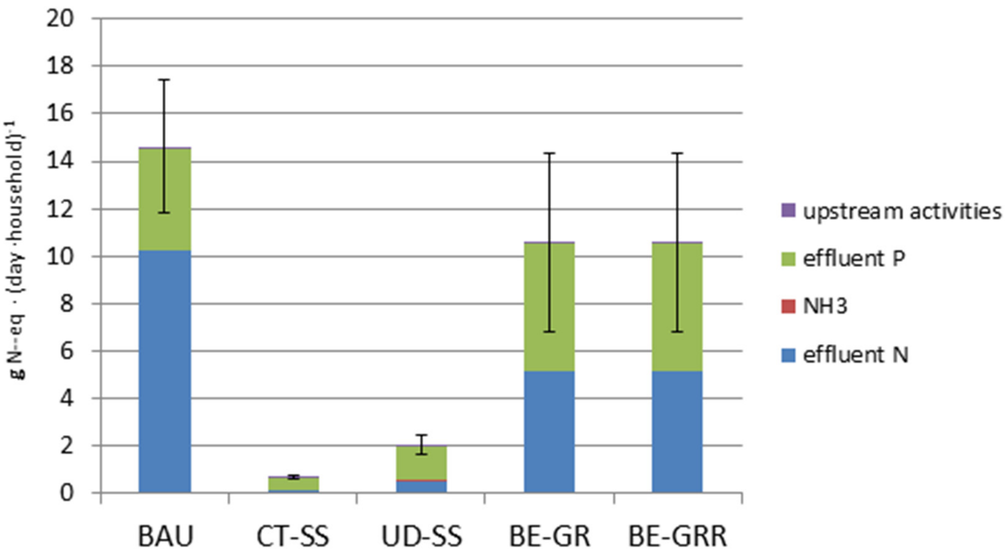

Figure 4. Life cycle eutrophication potentials of water systems, g N-eq· (day· household)$^{-1}$. The range of variability bar presents the values at fifth and 95th percentiles.

3.3. BE-GR and BE-GRR

The blackwater-only energy recovery sewer (BS-GR) system had the lowest net energy requirement, estimated to an average 590 MJ· (day· household)$^{-1}$ (Figure 2). This finding is similar to those previously described in Germany [12,74] and from a theoretical evaluation for the Netherlands [75]. Aligned with energy consumption, the options involving the blackwater sewer with energy recovery were the least global warming–intensive with 23 $kgCO_2$-eq· (day· household)$^{-1}$ for the system including the rainwater element (BE-GRR) and 21 $kgCO_2$-eq· (day· household)$^{-1}$ for the system without it (BE-GR) (Figure 3). Due to the higher energy intensity for the rainwater element compared to the municipal drinking water element, BE-GRR exhibited a slightly higher life cycle energy and GWP than BE-GR. The energy recovered from blackwater and food waste offset approximately half of the total energy consumption associated with water and wastewater conveyance and treatment for the BE-GR and BE-GRR options. For greywater systems (BE-GR and BE-GRR), on-site greywater treatment was the major contributor, resulting in more than 50% of the total energy consumption, and when rainwater harvesting (BE-GRR) was included, it consumed about 25% of the total energy consumption. Additionally, the energy savings derived from substituting synthetic fertilizer with urine were negligible, being approximately 2.5 MJ· (day· household)$^{-1}$ (Figure 2). Although the digestate contains considerable amounts of nitrogen and phosphorous, the eutrophication potentials of BE-GR and BE-GRR remained lower than that of the BAU option (Figure 4).

3.4. Carbon Intensity of Different Treatment Stages

The carbon intensity, defined as the life cycle global warming potential divided by the treated volume, was estimated for various system elements. The most carbon-intensive stage was blackwater transport via pressure sewer, with an average value of 0.060 kg $CO_2 \cdot L^{-1}$ blackwater. The following carbon-intensive stages were rainwater treatment and greywater treatment, with average intensities of 0.040 kg $CO_2 \cdot L^{-1}$ rainwater, and 0.035 kg $CO_2 \cdot L^{-1}$ greywater. In contrast, the lowest carbon intensity occurred with centralized water treatment and distribution, with an average value of 0.018 kg $CO_2 \cdot L^{-1}$. The intensive energy use in transporting blackwater through pressure sewers from households to the community digester caused its high carbon intensity. Compared with municipal centralized water treatment/distribution, or onsite greywater treatment/distribution, on-site rainwater collection/treatment was the most carbon-intensive option for supplying shower water for households due to the energy-intensive characteristics of both pump operation and UV disinfection for rainwater (Figure 3). Previous studies in Australia and the United Kingdom reported that the carbon intensity of rainwater treatment ranges from 0.03 to 0.07 kg $CO_2 \cdot L^{-1}$ treated rainwater, consistent with our results [50–52]. In addition, the relative GWP contributions of infrastructure components to each stage in the water cycle are presented in Figure S2, showing that the infrastructure components account for less than 5% of the total GWP.

The analysis suggested that supply chain activities had minimal contribution to the life cycle EU of water systems (Figure 4). The nutrient discharge from wastewater treatment elements resulted in more than 95% of the total eutrophication potentials for all systems. Phosphorus was estimated to contribute to more than 45% of the total eutrophication potentials for all systems. For the EU of BAU, the nutrient discharge from the secondary wastewater treatment process was the primary contributor for EU. Similarly, the nutrient discharge from the septic system–treated greywater ranked as the largest contributor for CT-SS and UD-SS. In addition, nutrient outputs from digesters dominated the EU of BE-GR and BE-GRR systems.

3.5. Variability, Uncertainty and Sensitivity

The statistical distributions of parameters in this study reflect both naturally occurring variability and uncertainty embedded in our lack of knowledge/modeling. Variability stems from both water and wastewater processes and their supply chain activities. Water- and wastewater process-related variability was caused by various flow rates, different water pressure requirements, and distinct influent quality. The changes in flow rates and water pressure lead to different energy requirements for transporting water and wastewater flows. The fluctuating influent quality could result in distinct energy requirements for infiltration, disinfection, and aeration processes. The natural variability within supply chain activities resulted from various electricity sources and production technologies. Uncertainty is characterized by the lack of confidence/knowledge about parameter values. Due to limited implementation of the CT-SS, UD-SS, BE-GR, and BE-GRR systems in the U.S., uncertainty in results for those options was estimated in comparison with the conventional centralized system.

The variations in the life cycle energy consumption and global warming potentials for each system evaluated (BAU, CT-SS, UD-SS, BE-GR, and BE-GRR) are presented in Figures 2 and 3. The 90% confidence intervals of GWP for BAU, CT-SS and UD-SS were approximately 100 kg CO2-eq$\cdot$(day$\cdot$household)$^{-1}$ (the 5th–95th percentile ranges are given in Figure 3). For the two blackwater sewer systems (BE-GR and BE-GRR), their global warming potentials indicated relatively smaller variations, being approximately a fifth of the range in the BAU.

The sensitivity analysis results are presented as tornado graphs in Figure 5. The median values of GWPs for each system are represented with a vertical line, and the horizontal bars describe the deviation in GWP impact associated with changes in the parameters labeled on the left. It is clear from Figure 5 that changing the amount of electricity produced from the co-digester process, the carbon intensity of the electricity, the chemical and electricity inputs for the centralized wastewater treatment plant, the UV dose, the flow rate, and the pump efficiency may all alter the global warming results of water systems. Among the investigated parameters, the carbon intensity of electricity (the life cycle global warming potential of electricity) and the amount of electricity produced from the co-digester process appear to be the two most influential variables of the systems studied. For BAU, CT-SS, and UD-SS, the equivalent electricity production amounting to the electricity produced from the co-digester and CHP processes was the most influential factor impacting their global warming potentials. The expanded electricity production could result in an increase of GWPs by up to 20 kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$, and a decrease of GWPs by up to 40 kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$, for BAU, CT-SS, and UD-SS systems. The carbon intensity of electricity ranks as the second most influential parameter for the GWP of water systems. The variation of carbon intensity of electricity may result in a variation of up to $+/-30$ kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$ in the GWPs of BAU, CT-SS, and UD-SS systems. Similarly, the carbon intensity of electricity was the most influential variable for BE-GR and BE-GRR systems. The GWPs of BE-GR and BE-GRR varied by approximately $+/-20$ kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$ through varying the carbon intensity of the Falmouth electricity mix. Although utilizing the national electricity mix resulted in increased GWPs of water systems (at least a 30 kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$ increase for BAU, CT-SS, and UD-SS systems, and at least a 10 kg CO_2-eq$\cdot$ (day$\cdot$ household)$^{-1}$ for BE-GR and BE-GRR systems), the rankings of global warming potentials of water systems did not change. In addition, varying the remaining parameters (including the chemical and electricity inputs for the centralized water treatment plant, the chemical and electricity inputs for the centralized wastewater treatment plant, the UV dose, the flow rate, and the pump and motor efficiency) resulted in relatively small variations of the global warming potential for the centralized water system.

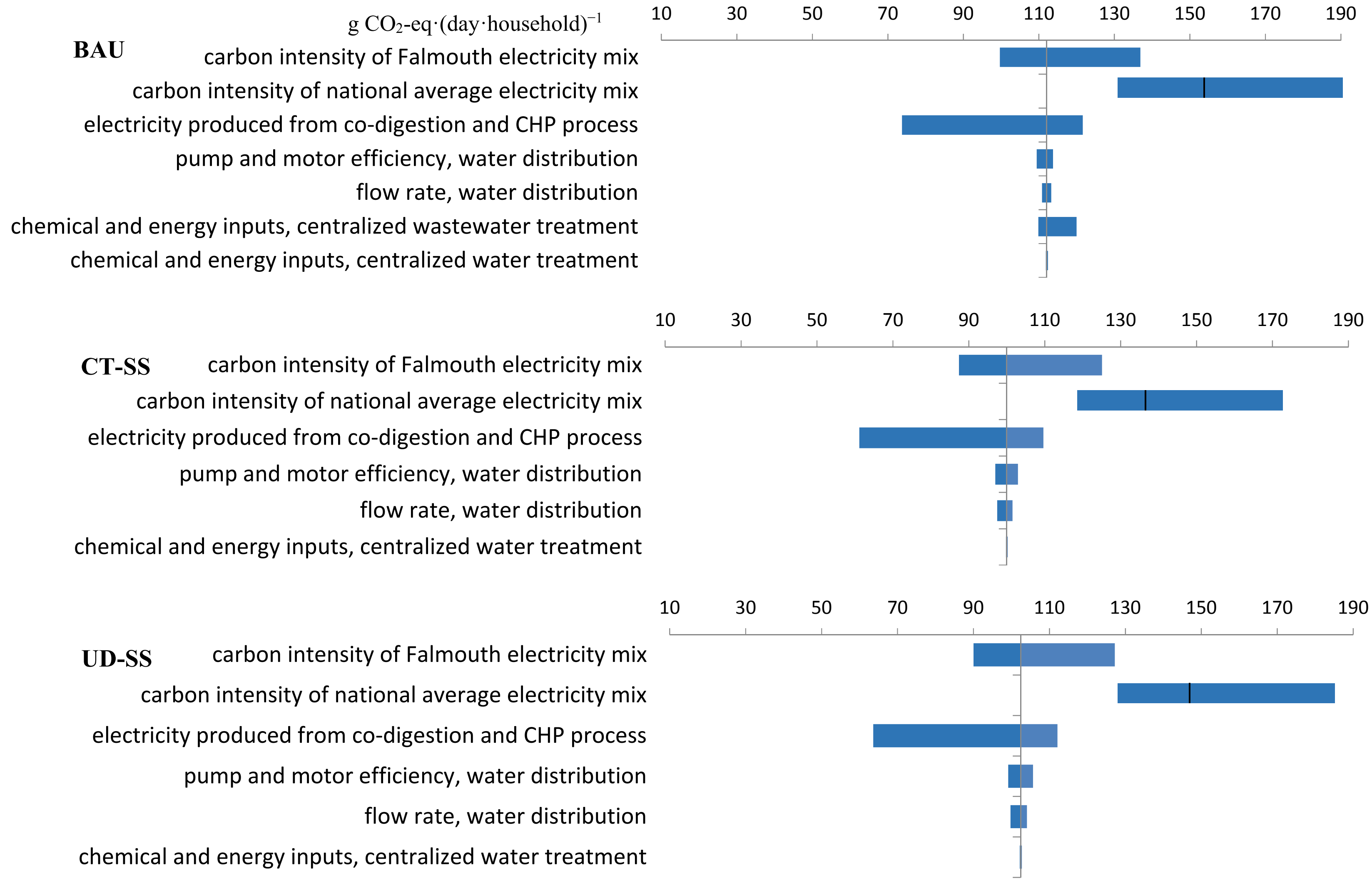

Figure 5. *Cont.*

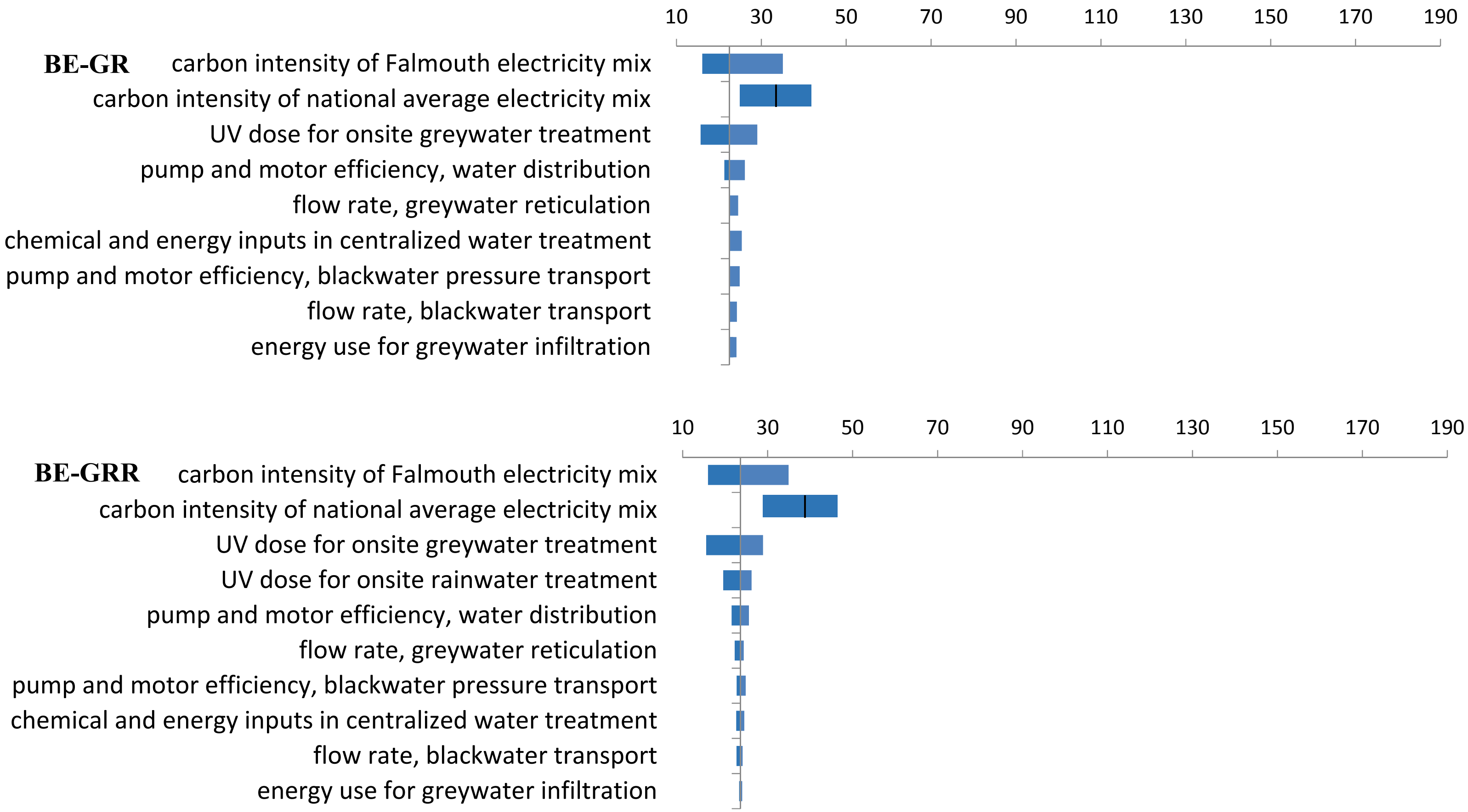

Figure 5. Sensitivity analysis of global warming potential of water systems. Vertical lines represent the base case for each system. The bars demonstrate the variation associated with the following changes in the parameters listed at the left: carbon intensity, electricity produced from co-digestion, pump and motor efficiencies, flow rate, electricity input in water treatment plant, electricity input in wastewater treatment plant, UV dose. The ranges of each variable are identified in Table 1.

3.6. Opportunities to Reduce Life Cycle Energy and Eutrophication Potentials of Water Systems

The GWP and EU mitigation potentials through multiple measures are illustrated in Figure 6, including reducing water leaking through the municipal water supply, conserving water at the household through utilizing water-saving devices, and system shifts from conventional centralized infrastructure (BAU) to alternative systems which are capable of reusing on-site treated greywater (BE-GR, BE-GRR), and producing fertilizers (UD-SS) and also energy (BE-GR, BE-GRR). Among the investigated reduction scenarios, replacing the centralized water and wastewater system (BAU) with the blackwater sewer and household greywater recycling configuration (BE-GR) had the highest potential to reduce energy consumption and GWPs, equivalent to 480 MJ· (household· day)$^{-1}$ and 54 kg CO_2-eq· (household· day)$^{-1}$, respectively. Both blackwater energy recovery and on-site greywater recycling can offset the energy consumed by water and wastewater treatment and distribution processes. The on-site greywater reuse leads to energy and GWP reductions because of (1) decreasing the water treatment and distribution demand from municipal water and rain sources; and (2) offsetting centralized wastewater collection and treatment energy requirements. In addition, the corresponding energy consumption and GWP savings for the BAU system could reach 40 MJ· (household· day)$^{-1}$ as a result of a 5% reduction of household water use by installing water-saving devices (e.g., low-flow shower heads, aerators and toilets), whereas reducing water loss during distribution by 5% only resulted in approximately 10 MJ· (household· day)$^{-1}$ of energy reduction. Indoor water conservation can reduce the needs for both water supply and wastewater treatment, therefore resulting in a larger reduction than controlling water loss during the distribution stage. In addition, among the evaluated options, shifting a centralized system into septic systems coupled with urine-diverting or composting toilets (UD-SS and CT-SS) provides the highest EU reduction potential. Reducing water loss and water use provides a negligible reduction of EU potentials.

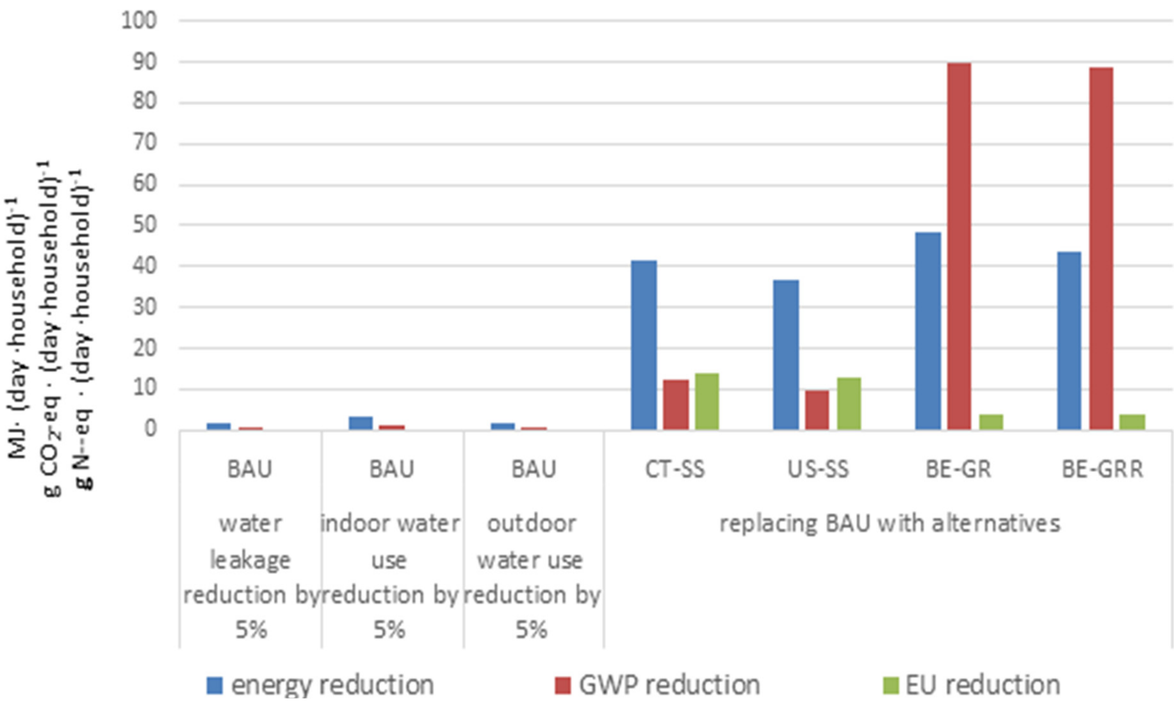

Figure 6. The reduction of life cycle energy consumption, global warming and eutrophication potentials through multiple measures. The units of energy reduction, global warming potential reduction, and eutrophication potential reduction are MJ· (household· day)$^{-1}$, gCO_2-eq· (household· day)$^{-1}$, and g N-eq· (household· day)$^{-1}$, respectively.

4. Discussion

This comparative analysis based on the Cape Cod study suggested that (1) food waste, grease trap and blackwater co-digestion largely offsets the energy required for water and waste services; and (2) on-site greywater treatment and reuse has a significant potential to reduce energy consumption for water and wastewater services. The ability to draw general conclusions based on this work is, to some extent, limited by the use of specific values representing local topography, population density, water resources and quality, and climate conditions. For example, the local Falmouth topography requires pumping wastewater to a centralized wastewater treatment plant at a relatively high elevation and over a long distribution system, which results in a high energy intensity for centralized wastewater in Falmouth. The rainfall quantity and quality also influence the size of rain tanks and the energy consumption for filtration and disinfection processes. Although directly applying the findings of the Falmouth community to other communities will require revisiting the datasets sensitive to local conditions, the transparent life cycle inventory and unit processes created in this study are valuable for future comparison.

The eutrophication potentials of water systems were partially based on the assumption that urine was transported out of Falmouth and digestate was applied locally. Our life cycle energy results suggested that the energy saving of displacing synthetic fertilizer with transporting urine was large enough to transport urine via a diesel truck up to 70–120 km, depending on the truck specifics. Considering the larger volume (about two times' urine volume due to the addition of household/restaurant waste) and lower nutrient content of digestate compared to urine, 5 km of truck transport will offset the energy savings from displacing synthetic fertilizer with digestate. The land area of Falmouth is approximately 120 km^2. Based on the local conditions, it is expected that urine is transported out of Falmouth and digestate is applied locally.

Although energy, GWP and EU focus on limited aspects of system evaluation, they are well-established indicators that have been determined for a large number of products and services, thus making them meaningful measures for system evaluation and comparison. Secondly, the current work compiles a completely transparent life cycle inventory for on-site technology options including greywater, blackwater, and rainwater management within the U.S. context. Additionally, the technology performance of systems designed based on the principles of source separation and water fit-for-purpose is an evolving area of study. This study utilized the best-available data relevant to the assessed systems to quantify the energy and carbon emissions of relatively novel system elements; future experiments and full-scale datasets would make these analyses more precise. In addition, future work needs to synthesize technical, economic, environmental, and social impacts for aiding decision-making [27], as well as to incorporate additional system elements and configurations. These additional configurations may include recycling nutrients and energy from sludge generated from centralized wastewater treatment plants, improving treatment efficiencies through advanced septic system design, wetland systems and others.

This study identified the environmental benefits of multiple measures (Figure 5). The systematic evaluation of the whole anthropogenic water cycle was key in the comparative energy-saving analyses, which considered water supply and demand management measures, novel system configurations with energy and nutrient recovery, and water recycling and reuse schemes. Currently, household water conservation and distribution loss control have gained attention in the U.S. and elsewhere, due to their potentials of improving energy efficiency and climate resiliency [76]. However, the analysis suggests that building and operating water services that include energy recovery and local greywater reuse elements could save up to 14 times more energy than currently advocated water conservation and loss control measures.

With the focus on demonstrating the importance of undertaking a full water service analysis, this analysis does not aim to include all reduction measures. It is important to note that conservation of nutrients and energy from centralized wastewater treatment plants may improve their environmental performances. It is suggested that future studies should evaluate the effectiveness of a full set of reduction strategies when datasets are available.

Despite the GWP and EU savings associated with the CT-SS, UD-SS, BE-GR and BE-GRR options, hurdles related to technology scale-up and cost optimization, training of relevant trades, and public perception issues must be overcome prior to their implementation [77]. It has long been assumed by various agencies that building centralized sewers and drinking water services is the preferred solution to regions with moderate to high housing density. This view is well understood given that provision of centralized sanitation and treated drinking water is considered as one of the most important public health interventions that humans have devised [78]. The human health impacts of the evaluated decentralized systems are not higher than the centralized service option [37]. Moreover, the recent cost analysis [39] suggests that the life cycle costs of the proposed alternative systems (such as composting and urine-diverting toilet options) are cheaper than a conventional centralized sewer. This is especially true in Cape Cod and the surrounding region where the cost estimates for conventional sewers are particularly high [36]. In addition, the conserved fertilizers from the use of source separation technologies may also support the mitigation of phosphorus shortages in the long-term [79]. Overall, given the magnitude of energy savings and additional benefits, systems incorporating energy recovery and those designed under principles of source separation and water fit-for-purpose (illustrated here with treated greywater for non-potable household uses) should be given equal consideration alongside conventional sewers in municipal water infrastructure, nitrogen mitigation, and community redevelopment decision support.

5. Conclusions

The study builds the transparent energy, GHG and nutrient inventory of the whole water and wastewater service systems including both centralized and resource recovery–based systems, from a life cycle perspective. The detailed variability/sensitivity analysis was conducted to understand the key influencing factors and to support future data collection efforts. Based on the comparative analysis, the centralized drinking water supply system coupled with blackwater energy recovery and on-site greywater treatment/reuse was the most energy- and carbon-efficient water service system among the five options. The electricity generated from blackwater co-digestion was estimated to offset at least 40% of the life cycle energy consumption for water/waste services. Composting and urine-diverting toilet options demonstrated the lowest life cycle eutrophication potential. The global warming potential results were most sensitive to the carbon intensity of the electricity grid and electricity production of the co-digestion facility. Transitioning from a conventional centralized system to a system incorporating energy recovery and greywater recycling offered up to a 14-fold reduction in global warming potential compared to the currently advocated water conservation and loss control measures for centralized services for the Falmouth case study. In addition, caution should be taken when the specific datasets and results for Cape Cod are directly applied to other case studies. The transparent life cycle inventory of water and wastewater treatment provided in this study is capable of serving as the foundation for future modifications and comparisons. If available, location-specific electricity datasets should be adapted for other case studies.

Supplementary Materials: The following are available online at www.mdpi.com/2073-4441/8/4/154/s1. The Supplementary Materials contains water balance calculations, system diagrams, energy and GHGs calculation for treatment, distribution, co-digestion and CHP processes, material requirements for pipe and treatment infrastructure, nutrient contents of compost, urine, and digestate, input and output flows utilized to build life cycle inventory, and stochastic distributions of key parameters.

Acknowledgments: This project was supported by the U.S. Environmental Protection Agency Office of Research and Development through the ORISE Post-Doctoral Fellowship Program and other project-related support. This effort was also supported by the Faculty Research Award Program and Individual Development Awards Program of the State University of New York at Albany. The senior author (NA) would like to acknowledge many years of discussions and recent data provided by Greg Peters (University of New South Wales and Chalmers University) and Sven Lundie (University of New South Wales and PE International Hamburg) and Hilder Maingay and Earle Barnhart of Falmouth. The authors also would like to acknowledge Jennifer Cashdollar and Michael Gonzales and Cissy Ma (U.S. Environmental Protection Agency) for their insightful comments and suggestions. The views expressed in this article are those of the authors and do not necessarily reflect the views or policies of the U.S. Environmental Protection Agency. Any mention of specific products or processes does not represent endorsement by the U.S. Environmental Protection Agency.

Author Contributions: The authors have equal intellectual contributions. The authors together have designed the research project, collected datasets, conducted the life cycle and statistical analyses, and written the manuscript.

Conflicts of Interest: The authors declare no conflict of interest.

References

1. United States Goverment Accountability Office. *Amount of Energy Needed to Supply, Use, and Treat Water is Location-Specific and Can be Reduced by Certain Technologies and Approaches*; Unitied States Government Accountability Office: Washington, DC, USA, 2011; p. 10.
2. Olsson, G. *Water and Energy. Threats and Oppurtunities*; IWA Publishing: London, UK, 2012.
3. US EPA. *National Rivers and Streams Assessment 2008–2009. A Collaborative Survey. Draft*; epa/841/d-13/001. US EPA: Washington, DC, USA, 2013.
4. Howe, C.; Mukheibir, P.; Gallet, E. *Institutional Issues for Green-Grey Infrastructure based on Integrated "One Water" Management and Resource Recovery*; Institute for Sustainable Futures, University of Technology: Sydney, Australia, 2013.
5. Burn, L.S.; De Silva, D.; Shipton, R.J. Effect of demand management and system operation on potable water infrastructure costs. *Urban Water* **2002**, *4*, 229–236. [CrossRef]
6. Ishii, S.K.; Boyer, T.H. Life cycle comparison of centralized wastewater treatment and urine source separation with struvite precipitation: Focus on urine nutrient management. *Water Res.* **2015**, *79*, 88–103. [CrossRef] [PubMed]
7. Tervahauta, T.; van der Weijden, R.D.; Flemming, R.L.; Hernández Leal, L.; Zeeman, G.; Buisman, C.J. Calcium phosphate granulation in anaerobic treatment of black water: A new approach to phosphorus recovery. *Water Res.* **2014**, *48*, 632–642. [CrossRef] [PubMed]
8. Villarroel Walker, R.; Beck, M.B.; Hall, J.W.; Dawson, R.J.; Heidrich, O. The energy-water-food nexus: Strategic analysis of technologies for transforming the urban metabolism. *J. Environ. Manag.* **2014**, *141*, 104–115. [CrossRef] [PubMed]
9. Brown, V.; Jackson, D.W.; Khalifé, M. 2009 melbourne metropolitan sewerage strategy: A portfolio of decentralised and on-site concept designs. *Water Sci. Technol.* **2010**, *62*, 510–517. [CrossRef] [PubMed]
10. Chen, R.; Wang, X.C. Cost-benefit evaluation of a decentralized water system for wastewater reuse and environmental protection. *Water Sci. Technol.* **2009**, *59*, 1515–1522. [CrossRef] [PubMed]
11. Gikas, P.; Tchobanoglous, G. The role of satellite and decentralized strategies in water resources management. *J. Environ. Manag.* **2012**, *2009*, 144–152. [CrossRef] [PubMed]
12. Kinstedt, K. *Optimization of the Collection and Transport of Blackwater in Source-Separated Wastewater Systems*; Technical University Hamburg-Harburg: Hamburg, Germary, 2012.
13. Malisie, A.F.; Prihandrijanti, M.; Otterpohl, R. The potential of nutrient reuse from a source-separated domestic wastewater system in Indonesia—Case study: Ecological sanitation pilot plant in Surabaya. *Water Sci. Technol.* **2007**, *56*, 141–148. [CrossRef] [PubMed]
14. Otterpohl, R.; Braun, U.; Oldenburg, M. Innovative technologies for decentralised water-, wastewater and biowaste management in urban and peri-urban areas. *Water Sci. Technol.* **2003**, *48*, 23–32. [PubMed]
15. Peter-Fröhlich, A.; Pawlowski, L.; Bonhomme, A.; Oldenburg, M. EU demonstration project for separate discharge and treatment of urine, faeces and greywater—Part I: Results. *Water Sci. Technol.* **2007**, *56*, 239–249. [CrossRef] [PubMed]
16. Sharma, A.; Burn, S.; Gardner, T.; Gregory, A. Role of decentralised systems in the transition of urban water systems. *Water Sci. Technol. Water Supply* **2010**, *10*, 577–583. [CrossRef]
17. Kiparsky, M.; Sedlak, D.L.; Thompson, B.H.J.; Truffer, B. The innovation deficit in urban water: The need for an integrated perspective on institutions, organizations, and technology. *Environ. Eng. Sci.* **2013**, *30*, 395–408. [CrossRef] [PubMed]

18. Mo, W.; Nasiri, F.; Eckelman, M.J.; Zhang, Q.; Zimmerman, J.B. Measuring the embodied energy in drinking water supply systems: A case study in the great lakes region. *Environ. Sci. Technol.* **2010**, *44*, 9516–9521. [CrossRef] [PubMed]
19. Mo, W.; Zhang, Q.; Mihelcic, J.R.; Hokanson, D.R. Embodied energy comparison of surface water and groundwater supply options. *Water Res.* **2011**, *45*, 5577–5586. [CrossRef] [PubMed]
20. Stokes, J.R.; Horvath, A. Life cycle energy assessment of alternative water supply systems. *Int. J. Life Cycle Assess.* **2006**, *11*, 335–343. [CrossRef]
21. Stokes, J.R.; Horvath, A. Energy and air emission effects of water supply. *Environ. Sci. Technol.* **2009**, *43*, 2680–2687. [CrossRef] [PubMed]
22. Lassaux, S.; Renzoni, R.; Germain, A. A life cycle assessment of water from the pumping station to the wastewater treatment plant. *Int. J. Life Cycle Assess.* **2007**, *12*, 118–126.
23. Rodriguez-Garcia, G.; Molinos-Senante, M.; Hospido, A.; Hernández-Sancho, F.; Moreira, M.T.; Feijoo, G. Environmental and economic profile of six typologies of wastewater treatment plants. *Water Res.* **2011**, *45*, 5997–6060. [CrossRef] [PubMed]
24. Corominas, L.; Foley, J.; Guest, J.S.; Hospido, A.; Larsen, H.F.; Morera, S.; Shaw, A. Life cycle assessment applied to wastewater treatment: State of the art. *Water Res.* **2013**, *47*, 5480–5492. [CrossRef] [PubMed]
25. Wang, R.; Eckelman, M.J.; Zimmerman, J.B. Consequential environmental and economic life cycle assessment of green and gray stormwater infrastructure for combined sewer systems. *Environ. Sci. Technol.* **2013**, *47*, 11189–11198. [CrossRef] [PubMed]
26. Loubet, P.; Roux, P.; Loiseau, E.; Bellon-Maurel, V. Life cycle assessment of urban water systems: A comparative analysis of selected peer-reviewed literature. *Water Res.* **2014**, *67*, 187–202. [CrossRef] [PubMed]
27. Xue, X.; Schoen, M.E.; Ma, C.; Hawkins, T.R.; Ashbolt, N.J.; Cashdollar, J.; Garland, J. Critical insights for a sustainability framework to address integrated community water services: Technical metrics and approaches. *Water Res.* **2015**, *77*, 155–169. [CrossRef] [PubMed]
28. Remy, C.; Jekel, M. Energy analysis of conventional and source-separation systems for urban wastewater management using life cycle assessment. *Water Sci. Technol.* **2012**, *65*, 22–29. [CrossRef] [PubMed]
29. Benetto, E.; Nguyen, D.; Lohmann, T.; Schmitt, B.; Schossler, P. Life cycle assessment of ecological sanitation system for small-scale wastewater treatment. *Sci. Total Environ.* **2009**, *407*, 1506–1516. [CrossRef] [PubMed]
30. Kärrman, E.; Jönsson, H. Normalising impacts in an environmental systems analysis of wastewater systems. *Water Sci. Technol.* **2001**, *43*, 293–300. [PubMed]
31. Lam, L.; Kurisu, K.; Hanaki, K. Comparative environmental impacts of source-seperation systems for domestic wastewater manageemnt in rural China. *J. Clean. Prod.* **2015**, *104*, 185–198. [CrossRef]
32. Lehtoranta, S.; Vilpas, R.; Mattila, T.J. Comparison of carbon footprints and eutrophication impacts of rural on-site wastewater treatment plants in Finland. *J. Clean. Prod.* **2014**, *65*, 439–446. [CrossRef]
33. Thibodeau, C.; Monette, F.; Bulle, C.; Glaus, M. Comparison of black water source-seperation and conventional sanitation systems using life cycle assessment. *J. Clean. Prod.* **2014**, *67*, 45–57. [CrossRef]
34. Tillman, A.; Svingby, M.; Lundstrom, H. Life cycle assessment of municipal waste water systems. *Int. J. Life Cycle Assess.* **1998**, *3*, 145–157. [CrossRef]
35. NOAA. Cape Cod Precipitation Datasets. Available online: http://www.ncdc.noaa.gov/cdo-web/ (accessed on 15 June 2014).
36. Cape Cod Comission. Cape Cod Comission Resource Center. Available online: http://www.capecodcommission.org/index.php?id=62&a=topic&cat=Population (accessed on 15 June 2014).
37. Schoen, M.E.; Xue, X.; Hawkins, T.R.; Ashbolt, N.J. Comparative human health risk analysis of coastal community water and waste service options. *Environ. Sci. Technol.* **2014**, *48*, 9728–9736. [CrossRef] [PubMed]
38. Schoen, M.; Hawkins, T.R.; Xue, X.; Ma, C.; Garland, J.; Ashbolt, N.J. Technologic resilience assessment of coastal community water and wastewater service options. *Sustain. Water Qual. Ecol.* **2015**, *6*, 75–87. [CrossRef]
39. Wood, A.; Blackhurst, M.; Xue, X.; Hawkins, T.R.; Ashbolt, N.J.; Garland, J. Cost-effectiveness of nitrogen mitigation by alternative household wastewater management technologies. *J. Environ. Manag.* **2015**, *150*, 344–354. [CrossRef] [PubMed]
40. Water Department in Town of Falmouth. Water Utility. Available online: http://www.falmouthmass.us/depart.php?depkey=water (accessed on 15 June 2014).

41. US EPA. Indoor Water Use in the United States. Available online: http://www.epa.gov/WaterSense/pubs/indoor.html (accessed on 15 June 2014).
42. US EPA. Water & Energy Efficiency. Available online: http://water.epa.gov/infrastructure/sustain/waterefficiency.cfm (accessed on 15 June 2014).
43. US EPA. *Evaluation of Energy Conservation Measures for Wastewater Treatment Facilities*; US EPA: Washington, DC, USA, 2010.
44. Cape Cod Water Protection Colloborative. Comparison of Costs for Wastewater Management Systems Applicable to Cape Cod. Available online: http://www.ccwpc.org/index.php/component/content/article/36-wastewater-reports/78-comparison-of-costs-for-wastewater-management-systems-applicable-to-cape-cod (accessed on 15 June 2014).
45. Massachusetts Department of Environmental Protection. Septic Systems. Available online: http://www.mass.gov/eea/agencies/massdep/water/wastewater/septic-systems-title-5.html (accessed on 15 June 2014).
46. Memon, F.A.; Zheng, Z.; Bulter, D.; Shirley-Smith, C.; Lui, S.; Makropoulos, C.; Avery, L. Life cycle impact assessment of greywater recycling technologies for new developments. *Envion. Monit. Assess.* **2007**, *129*, 27–35. [CrossRef] [PubMed]
47. Cape Cod Eco-Toilet Center. Urine-diverting, Flush Toilet. Available online: http://capecodecotoiletcenter.com/types-of-eco-toilets/urine-diverting-toilets/ud-flush-toilets/ (accessed on 15 July 2014).
48. The Aquionics UV Product. The aquionics uv lamp energy consumption. 2013. Available online: http://www.aquionics.com/main/ (accessed on 15 July 2014).
49. Trojan UV products. Trojan uv lamp energy consumption. Available online: http://www.trojanuv.com/ (accessed on 15 July 2014).
50. Schulz, M.; Short, M.D.; Peters, G.M. A streamlined sustainability assessment tool for improved decision making in the urban water industry. *Integr. Environ. Assess. Manag.* **2012**, *8*, 183–193. [CrossRef] [PubMed]
51. Hallmann, M.; Grant, T.; Alsop, N. *Life Cycle Assessment and Life Cycle Costing of Water Tanks as a Supplement to Mains Water Supply for Yarra Valley Water*; Center for Design at RMIT University: Melbourne, Australia, 2003.
52. Crites, R.; Tchobanoglous, G. *Small and Decentralized Wastewater Management Systems*; The McGraw-Hill Companies, Inc.: Davis, CA, USA, 1998.
53. USDA. The food availability data system. Available online: http://www.ers.usda.gov/data-products/food-availability-(per-capita)-data-system.aspx (accessed on 15 September 2014).
54. US EPA. *Opportunities for Combined Heat and Power at Wastewater Treatment Facilities: Market Analysis and Lessons from the Field*; US EPA: Washington, DC, USA, 2011.
55. US NREL. *Feasibility Study of Anaerobic Digestion of Food Waste in St. Bernard, Louisiana*; NREL/TP-7A30-57082. US DOE: Washington, DC, USA, 2013.
56. Massachusetts Department of Environmental Protection. Identification, Characterization, and Mapping of Food Waste and Food Waste Generators in Massachusetts. Available online: http://www.mass.gov/dep/recycle/priorities/foodwast.pdf (accessed on 15 June 2014).
57. US EPA. Organics: Co-Digestion Economic Analysis Tool (CoEAT). Available online: http://www.epa.gov/region9/organics/coeat/index.html (accessed on 15 June 2014).
58. Tchobanoglous, G.; Burton, F.L.; Stensel, H.D. *Wastewater Engineering, Treatment and Reuse*; The McGraw-Hill Companies, Inc.: Davis, CA, USA, 2003.
59. US NREL. Coal-Fired Electricity Generation Results—Life Cycle Assessment Harmonization. Available online: http://www.nrel.gov/analysis/sustain_lca_coal.html (accessed on 15 June 2014).
60. NIST. The Buidling for Environmental and Economic Sustainability (BEES) Software and Database. Available online: http://www.nist.gov/el/economics/BEESSoftware.cfm (accessed on 15 June 2014).
61. US EPA. *Nonroad Model (Nonroad Engines, Equipment, and Vehicles)*; US EPA: Washington, DC, USA, 2009.
62. Swiss Centre for Life Cycle Inventories. The Ecoinvent Database. Available online: http://www.ecoinvent.org/database/ (accessed on 15 June 2014).
63. US EIA. Electricity. Available online: http://www.eia.gov/electricity/ (accessed on 15 June 2014).
64. US EPA. *National Emissions Inventory (NEI) Air Pollutant Emissions Trends Data*; US EPA: Washington, DC, USA, 2013.
65. Meinzinger, F.; Oldenburg, M. Characteristics of source-separated household wastewater flows: A statistical assessment. *Water Sci. Technol.* **2009**, *59*, 1785–1791. [CrossRef] [PubMed]

66. Meinzinger, F.; Londong, J.; Otterpohl, R. *Resource Efficiency of Urban Sanitation Systems: A Comparative Assessment using Material and Energy Flow Analysis*; Hamburg University of Technology (TUHH): Hamburg, Germany, 2010.
67. Short, M.D.; Daikeler, A.; Peters, G.M.; Mann, K.; Ashbolt, N.J.; Stuetz, R.M.; Peirson, W.L. Municipal gravity sewers: An unrecognised source of nitrous oxide. *Sci. Total Environ.* **2014**, *468–469*, 211–218. [CrossRef] [PubMed]
68. Xue, X.; Landis, A.E. Eutrophication potential of food consumption patterns. *Environ. Sci. Technol.* **2010**, *44*, 6450–6456. [CrossRef] [PubMed]
69. Xue, X.; Landis, A.E. Evaluating agricultural management practices to improve the environmental footprints of corn-derived bioproducts. *Renew. Energy* **2014**, *66*, 454–460. [CrossRef]
70. GmbH, G. Openlca. Available online: http://www.openlca.org/ (accessed on 15 June 2014).
71. US EPA. *Tool for Reduction and Assessment of Chemical and Other Environmental Impacts (TRACI)*; US EPA: Washington, DC, USA, 2015.
72. US EPA. How Clean is the Electricity I Use?—Power Profiler. Available online: https://oaspub.epa.gov/powpro/ept_pack.charts (assessed on 23 January 2015).
73. Electric Power Research Institute (EPRI). *Water And Sustainability (Volume 4): U.S. Electricity Consumption for Water Supply and Treatment—The Next Half Century*; EPRI: Concord, CA, USA, 2002.
74. Otterpohl, R. Options for alternative types of sewerage and treatment systems directed to improvement of the overall performance. *Water Sci. Technol.* **2002**, *45*, 149–158. [PubMed]
75. Zeeman, G.; Kujawa, K.; de Mes, T.; Hernandez, L.; de Graaff, M.; Abu-Ghunmi, L.; Mels, A.; Meulman, B.; Temmink, H.; Buisman, C.; *et al.* Anaerobic treatment as a core technology for energy, nutrients and water recovery from source-separated domestic waste(water). *Water Sci. Technol.* **2008**, *57*, 1207–1212. [PubMed]
76. Stokes, J.R.; Horvath, A.; Sturm, R. Water loss control using pressure management: Life-cycle energy and air emission effects. *Environ. Sci. Technol.* **2013**, *47*, 10771–10780. [CrossRef] [PubMed]
77. Larsen, T.A.; Alder, A.C.; Eggen, R.I.L.; Maurer, M.; Lienert, J. Source seperation: Will we see a paradigm shift in wastewater handling? *Environ. Sci. Technol.* **2009**, *43*, 6121–6125. [CrossRef] [PubMed]
78. Asano, T.; Levine, A.D. Wastewater reclamation, recycling and reuse: Past, present and future. *Water Sci. Technol.* **1996**, *33*, 1–14. [CrossRef]
79. Cordell, D. The Story of Phosphorus: Sustainability Implications of Global Phosphorus Scarcity for Food Security. Ph.D. Thesis, Linköping University, Linköping, Sweden, 2010.