

Quantifying the energy consumption and greenhouse gas emissions of changing wastewater quality standards

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Abstract

Regulations to ensure adequate wastewater treatment are becoming more stringent as the negative effects of different pollutants on human health and the environment are understood. However, treatment of wastewater to remove pollutants is energy-intensive, so has added significantly to the operation costs of wastewater treatment plants. Analysis from six of the largest wastewater treatment works in England reveals that the energy consumption of these treatment works has doubled in the last five years due to expansions to meet increasingly stringent effluent standards and population growth. This study quantifies the relationship between energy use for wastewater treatment and four measures of pollution in effluents from UK wastewater treatment works (Biochemical Oxygen Demand, Ammoniacal Nitrogen, Chemical Oxygen Demand and Suspended Solids). The linear regression results show that indicators of these pollutants in effluents, together with the extension of plants to improve wastewater treatment, can predict over 95% of energy consumption. Secondly, using scenarios, the energy consumption and greenhouse gas emissions of effluent quality standards are estimated. The study finds that tightening effluent standards to increase water quality could result in a doubling of electricity consumption and an increase of between 1.29 and 2.30 additional MTCO₂ per year from treating wastewater in large works in the UK.

Keywords: wastewater treatment, energy, water quality, water-energy nexus, climate change

Introduction

The discharge of untreated wastewater from urban areas into waterways has significant detrimental effects on ecosystems and potentially also human health. Therefore, regulations to ensure adequate treatment are becoming more stringent as the negative effects of different pollutants on humans and the environment are understood. Water quality measurement technologies are also becoming more cost-effective, which is enhancing understanding of the extent of pollutants in the aquatic environment.

To meet the requirements of the Water Framework Directive (WFD), which is the EU legislation for ground and surface waters, regulation of effluent discharges from waste water treatment works and other industrial plants has become more stringent (Marsh et al., 2002). The WFD's goal is for member states of the European Union to achieve 'good status' of their water bodies by 2015. The UK has made considerable progress to achieving this status, but it is still far from meeting the EU water pollution targets by the second management cycle, which extends to 2021, six years after the initial deadline (Voulvoulis et al., 2017). Further tightening of effluent standards is therefore possible in the future.

Maintaining and improving water quality in the face of population growth and climate change pressures will be an ongoing challenge for the water sector. Climate change has the potential to impact water management objectives by altering water quantity, temperature, quality and freshwater biodiversity (Arnell et al., 2015). As people become wealthier and expectations change, there is likely to be greater demand for environmental improvements. This is likely to be reflected in increased environmental standards over time (Ofwat, 2015). Furthermore, EurEau, the European association of the water service sector, believes that there have been insufficient measures to protect drinking water so far (Barker et al., 2015). This means that further regulation to protect drinking water sources can be expected in the second period of the WFD.

As effluent standards change and become more stringent, higher plant investment and operation costs will be incurred. This is coupled with necessary capacity increases to cater for increasing populations,

and projections of more extreme rainfall events (Burt et al., 2016). More advanced treatment processes can lead to increases in power consumption and may require more chemical loading for treatment, which in turn leads to increases in financial costs and, depending upon the source of electricity generation, greenhouse gas emissions (Gu et al., 2016). According to the WssTP (2011), increasing standards for discharges from wastewater treatment have led to a doubling of energy use in WWT in the UK between 1990 and 2011, and this is projected to double again in the next 15 years. Increasing effluent quality standards are widely believed to be the largest cause of rising carbon emissions in wastewater treatment (Baleta and McDonnell, 2012; Water UK, 2008).

Even though there is recognition that increased energy costs and carbon emissions will be incurred, there is little work that has been undertaken to quantify specific or potential future costs. According to Sadler, Georges and Thornton, (2009) there have been no detailed investigations of potential treatment requirements associated with Water Framework Directive standards. Some theoretical work has been carried out, particularly on carbon footprints (Barber, 2009; Marsh et al., 2002; Sadler et al., 2009), mostly concluding that aeration accounts for the largest contribution to the carbon footprint of wastewater treatment.

In this context, there remains a tension between different pieces of environmental legislation in most countries worldwide. On the one hand, the aim to improve the environment by driving up standards in the wastewater sector, and on the other hand, reducing anthropogenic greenhouse gases that drive climate change targets. In the UK, this tension manifests itself as a tension between EU legislation, including the WFD and the Urban Waste Water Treatment Directive, and the UK Climate Change Act of 2008. This has already led to authors suggesting the need to address these conflicting demands effectively (Baleta and McDonnell, 2012; Parsons and Marcet, 2012). Decarbonization of electricity supply is already occurring which will reduce the carbon footprint of wastewater treatment, but at the moment 75.5% of global electricity supply is still generated from fossil fuels. Even when

decarbonization of the electricity sector is complete, electricity will continue to represent a major operational cost burden for water utilities.

Using operational data from six Water Resource Recovery Facilities (WRRF) in the London region, operated by Thames Water, that have undergone significant expansions to deal with increasingly stringent effluent standards, this study aims to quantify the potential future costs and greenhouse gas emissions of such effluent standard increases in the UK. To address this objective, an empirical model is estimated from observations of the relationship between pollution removal and energy consumption in large WRRFs. The model is then used to test scenarios to understand the potential costs and greenhouse gas implications for the UK. The study addresses four measurements of pollution that are monitored in large WRRFs and by the EU WFD. These are Biochemical Oxygen Demand (BOD), Ammoniacal Nitrogen ($\text{NH}_3\text{-N}$), Suspended Solids (SSs) and Chemical Oxygen Demand (COD). The use of empirical data on actual plant energy use is revealing compared to previous estimates of the costs of achieving improved effluent standards, which have been based on more theoretical estimates and data on idealised plant performance (UKWIR, 2015). Moreover, for the first time we have been able to relate observed energy use with effluent water quality monitoring for four indicators of water quality.

The paper focusses upon total energy use in WRRFs and does not examine power demands at different times of day or how this interacts with variable time-of-day electricity pricing. As water utilities seek to reduce their energy costs, they will increasingly seek to optimise their power demand in response to variable energy prices. The potential benefits of optimising water pumping relative to variable energy prices have already been quantified by Oikonomou et al. (2018), Meschede et al. (2019) and Kernan et al. (2017). However, there is less scope for rescheduling power use in WRRFs. As a first step in understanding the drivers behind increasing energy use in WRRFs and the implications of further increases in effluent standards, in this paper our focus is upon total energy use rather than power demand and energy costs.

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114 **Materials and Methods**

115 **Datasets**

116 The data used for this study comes from Thames Water Ltd. Time series of electricity consumption
117 (kWh/month), wastewater flows (m³/month), and monthly incoming and outgoing concentration (mg/l)
118 for Biochemical Oxygen Demand (BOD), Ammoniacal Nitrogen (NH₃-N), Suspended Solids (SSs) and
119 Chemical Oxygen Demand (COD) were obtained for six large WRRFs in London for the period of January
120 2011 to December 2014. During this period, a series of upgrades were commissioned in four of the
121 plants. Upgrades to the WRRFs to meet more stringent effluent standards included the type of upgrade:
122 inlets to increase the flows passed to treatment (more treatment streams, more/larger pumping
123 stations, as well as inlet works expansions); additional primary settlement tanks (PSTs), additional
124 Activated Sludge Processes (ASPs) and final settlement tanks (FSTs) as well as the percentage of flow
125 these upgrades would treat, and when they were installed.

126 **Methodological approach**

127 Between the periods of 2010-2015 the energy use of all wastewater treatment in Thames Water almost
128 doubled, also adding over 25% to the company's overall energy bill. Previous analysis found that WWT is
129 approximately 50% of total sector electricity consumption(Majid et al., 2020). The increase in WWT
130 energy consumption is almost solely due to new extensions built in five main WRRFs in London to meet
131 higher demands and new standards. Due to data limitations, one of these WRRFs is not included in this
132 analysis, but two other large WRRFs in London are used to complement the dataset. The dots in Figure 1

show the location and relative size of the six WRRFs in London included in the analysis. These WRRFs have similar standards for effluent discharges (there are slight differences depending on the river section on which they are located), which are expected to continue to increase in the future (Defra, 2012, 2008). The four main types of extensions that have been added to the WRRFs to improve the quality of effluent include inlets, PSTs, ASPs and/or FSTs.

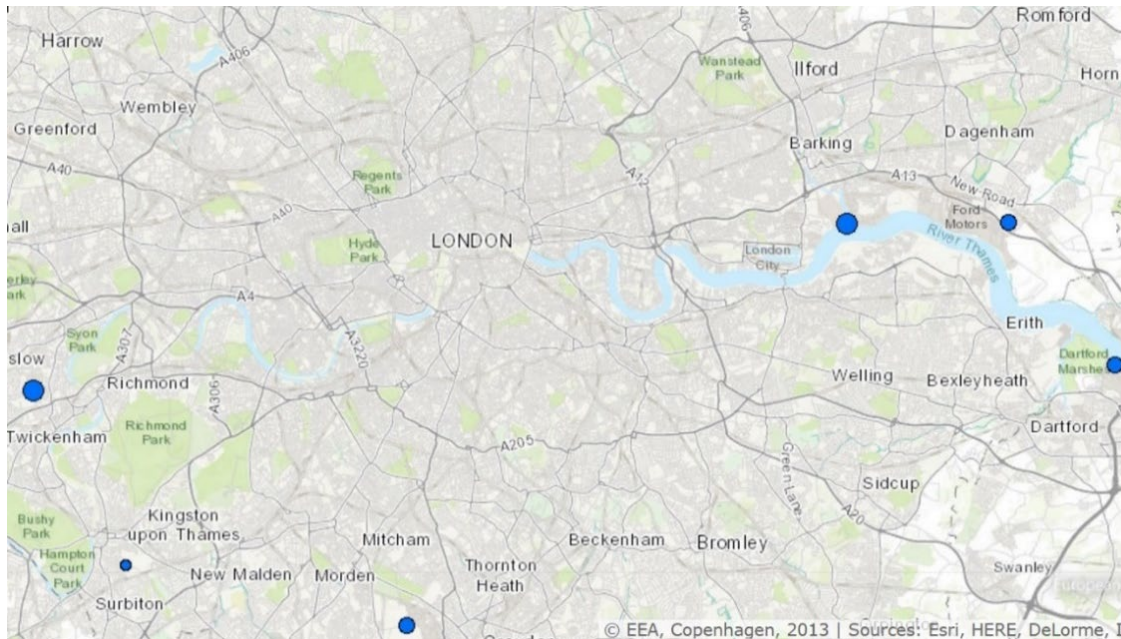


Figure 1: Map showing the location and relative size of the six WRRFs under study (European Environment Agency, 2014)

A robust linear regression analysis was performed to establish whether there was a statistical relationship between electricity consumption at the WRRFs and the extensions installed to meet new standards and the amount of pollution removed. Similar approaches have been used in studies investigating energy consumption at WRRFs (Carlson and Walburger, 2007; Longo et al., 2016), although this approach is novel in the consideration of technology and pollution removed. Extensions are defined as an addition to a plant of the four types described above (inlets, PSTs, ASPs and FSTs) built in a WRRF. In the regression model, the extension installations are included in the month when the extension went online (started treating sewage), and are represented as the amount of additional flow being treated at

the works by this extension, as a percentage of the flow. For example, a new aeration lane may have been installed at one of the works in December 2012 to treat 29% of the flow at the works, so this is represented as 129 from December 2012 onwards, and so on. In this way, we can not only capture the energy implications of installing new technologies at WRRFs but also begin to estimate how much specific amounts of additional treatment might cost in the future.

The amount of pollution removed is represented in the form of four key pollution measures: Biochemical Oxygen Demand (BOD), Ammoniacal Nitrogen ($\text{NH}_3\text{-N}$), Suspended Solids (SSs) and Chemical Oxygen Demand (COD). They are aggregated into a composite pollution load indicator as has been done in previous studies (Feng and Chen, 2016; Longo et al., 2016; Rodriguez-Garcia et al., 2011). This was done because the pollution loads are strongly correlated with each other, thus transforming them into one composite indicator allows the model to be more robust and avoid autocorrelation outputs. Load in the influent to the WRRF is defined as the weight of the relevant pollutant (in kg) calculated from its concentration (in mg/l) multiplied by the incoming wastewater flow (in m^3) at the WRRF when the sample was taken. The conversion to load is important because all the plants under study are preceded by combined sewer systems (conveying both sewage and storm-water) and thus the effect of dilution is relevant.

The elements of the index were weighed according to the relative energy it takes to treat a kg of each, as indicated by wastewater experts in the sector (Thames Water wastewater expert, personal communication, 20th May 2017). Ammoniacal nitrogen was the most significant, as it takes four times the oxygen to treat a kg of ammonia than a kg of BOD. COD was removed as it represents the same measure as BOD (in terms of the energy needed to treat them), so including it would have double counted energy consumption. Finally, SS is the least significant in terms of the energy it takes to remove, as most SS settles out in primary settlement tanks, and little of what is left reaches aeration plants.

Thus, the SS was left at its original measure. Consequently, the Weighed Pollution Load Removal Index was defined as follows:

$$PLR = ((BOD_i - BOD_o).2 + (NH_3i - NH_3o).8 + (SS_i - SS_o)).WWflow \quad (1)$$

where PLR stands for Pollution Load Removal Index in kg, i indicates the incoming pollution concentration in mg/l at the WRRF and o indicates the outgoing concentration in mg/l at the WRRF outlet. The pollution load is then calculated by multiplying the amount of pollution removed by the flow received at the WRRF in m^3 . Thus, the resulting composite indicator reports the average pollution load removed at each WRRF monthly.

In order to improve the robustness of the estimation, the analysis was carried out jointly for all WRRFs, recognising that this implies that the effect of given plant improvements is similar across all WRRFs. Robust regression is applied because the residual distribution of the data was prone to outliers, thus invalidating standard linear regression assumptions. The robust fitting method is less sensitive than ordinary least squares to large changes in small parts of the data and was thus appropriate. After several iterations and variables were considered, the model was established as follows:

$$E = \beta_0 + PLR + Inlet + PST + ASP + FST + \epsilon \quad (2)$$

where β_0 represents the intercept, E is the total energy use in kWh in N plants, PLR is the Pollution Load Removal Index described above; Inlet, PST, ASP and FST represent the type of extension/technology and additional load treated in it at N plants, and ϵ is the error term.

The four types of extension (inlets, PSTs, ASPs and FSTs) were initially considered in the model iterations as equation (2) above shows. However, the installation of inlets, primary settlement tanks and final settlement tanks did not have a significant correlation with energy consumption. Additional aeration

lanes did have a very significant effect on the energy consumption, and thus this variable was used in the final model.

Scenario selection

Once the model was built, three scenarios of more stringent effluent quality standards were established set. Potential energy costs (kWh) and GHG emissions from meeting these standards were estimated to establish their potential impacts on the UK water sector. As the sample of the WRRFs included works that are all the largest denomination in Europe (over 150,000 population equivalent (p.e.) served), the costs were up-scaled to the 101 WRRFs in the UK that are of comparable size. These 101 works serve 59% of the UK population, and have capacity to serve another 20%. Three more stringent effluent standard scenarios were chosen to evaluate how the cost of removing pollutants to a higher level might influence the energy consumption of the treatment work (Table 1). The addition of aeration processes to the WRRF was also tested in the scenarios as their addition was found to significantly impact the energy consumption of the WRRFs. Thus, the proportion of ASP that was added to each scenario can also be seen in Table 1.

Barber (2012) used similar scenarios to test how more stringent standards might influence the electricity consumption and operational greenhouse gas emissions of a theoretical WRRF, but did not include COD. Marsh, Vale and Watson's (2002) set out benchmark values for high quality effluent, which are used in Scenario C. The final scenario standards were derived from the two publications and UK Technical Advisory Group on the Water Framework Directive recommendations (UK TAG, 2013), then discussed with sector experts and summarised as follows:

214 *Table 1: Scenario standards adapted from Barber (2012) and Marsh et al. (2002) and extensions to WRRFs in the scenarios*

Scenario/ Pollutant (mg/l)	Current average standards*	Scenario A	Scenario B	Scenario C
Biochemical Oxygen Demand (BOD)	22	20	10	5
Ammoniacal Nitrogen (NH ₃ -N)	8	6	4	1
Suspended Solids (SSs)	25	15	10	5
Chemical Oxygen Demand (COD)	No current standards	50	30	20
Extensions to WRRFs to meet standards				
New ASPs	NA	ASP for 20% of flow	ASP for 40% of flow	ASP for 60% of flow

215 *may vary slightly between specific WRRFs.

216 The current average standards for BOD and SS are representative of the Tidal Thames WRRF standards,
 217 where most of the sampled works are found, while AmmN standards can be slightly higher. The three
 218 scenario standards were applied to three representative WRRFs of different capacities to evaluate the
 219 energy consumption across sizes. The defined sizes represent a range of medium to large WRRFs such as
 220 the sample represented in our model, of 400ML/day, 600ML/day and 800ML/day.

221 There are 9 resulting scenarios that combine the three pollution standard scenarios and extensions with
 222 the three WRRF sizes. The PLR is recalculated for each scenario using eq. (1) above to obtain the
 223 resulting energy consumption from having to have higher quality outgoing effluent. In this iteration, the
 224 empirical outgoing pollution concentrations are substituted by the scenario figures in Table 1 to

calculate the pollution load that would have to be removed under each scenario. The incoming pollution data remains the same as in the empirical dataset. These figures are then converted to load as in eq. (1) with the scenario WRRF sizes.

Upscaling to implications for the UK

The UK has 101 WRRFs that serve 150,000 people and over, the largest size denominator in Europe for WRRFs. This data is used to make an estimation of the potential UK-wide additional annual electricity costs from meeting higher effluent standards.

The electricity cost estimates are then converted to potential GHG emissions. The current UK electricity conversion factor of 0.412 kgCO₂e per kWh is used. Alternative future emissions factors are also used to calculate the potential additional GHG emissions, due to likely grid decarbonization futures. The Committee on Climate Change sets out scenarios for the Fifth Carbon Budget, including the central scenario (0.18 kgCO₂e/kWh), which is their best assessment of the technologies and behaviours required to meet targets and the barriers scenario (0.25 kgCO₂e/kWh), which represents less favourable conditions (Committee on Climate Change, 2015). This results in three scenarios of grid electricity intensity that are compared to the three effluent standard futures to result in a range of additional emissions. The conversion applied is as follows:

$$MTCO_2 = (101.E_c).G.12/10^9 \quad (3)$$

where E_c is the mean monthly energy consumption (in kWh/month) by scenario, multiplied by G , the GHG conversion factor (in kgCO₂/kWh) and annualized, then converted into MTCO₂, and scaled by the number of similar sized WRRFs in the UK, to obtain the estimated GHG emissions that could result from three scenarios of more stringent effluent quality standards.

Results

Figure 2 shows three timelines: the wastewater flows time series of the WRRFs in the study, the energy consumption over the same period and the extensions that were built in the sample WRRFs over the same period. The third timeline show what type of plant expansions were installed as well as the specific points in time when they came online. Each shape size represents the number of that type of extension built, and the number within the circle represents how many. Each line represents a different type of technological expansion, either new inlets, PSTs, ASPs or FSTs. As can be noted, there is no increase in the average wastewater flow over the sample period, but there is a sharp increase in the overall energy consumption of the WRRFs, which matches when the first extensions come online, and by 2014 the monthly energy consumption of the 6 WRRFs has doubled. The figure illustrates that there is variability in energy use, but notwithstanding the variability, there is a clear signal associated with plant expansion. Furthermore, as the wastewater flows time series on the second y axis shows, there is no change in the average wastewater flows being treated at the WRRFs, pointing to the fact that the increases in energy consumption can be attributed to the plant expansions.

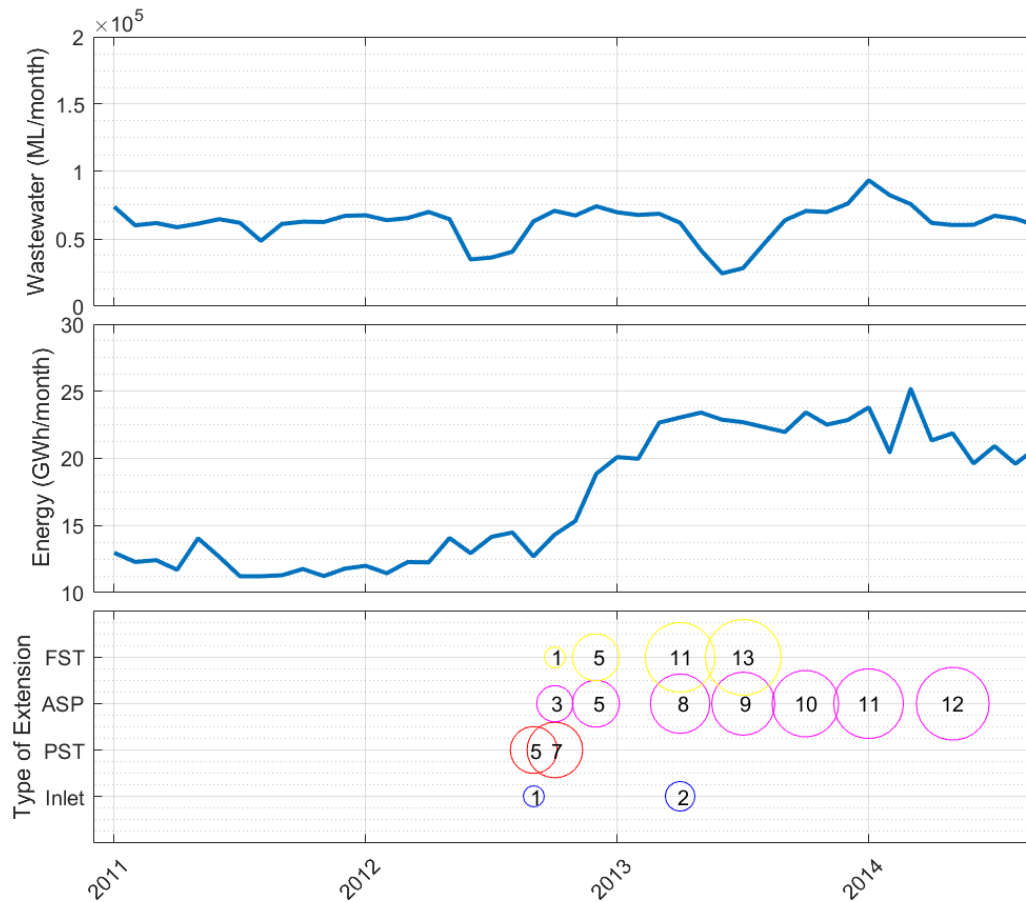


Figure 2: Energy Consumption and Wastewater Flows of WRRFs under study with plant extensions

Figure 3 shows the statistically significant relationship (correlation coefficient 0.856) between energy use and the Pollution Removal Index (PLR). As can be observed, the more pollution is removed the higher the energy consumption. A logarithmic transformation is applied to the scatter plot to account for the range in infrastructure sizes, as has been done previously in studies dealing with large WRRFs (Carlson and Walburger, 2007). The larger energy numbers account for the larger WRRFs and the smaller numbers for the smaller WRRFs, where there is more variance. If the sample contained more WRRFs of the middle sizes the sample points would be more continuous.

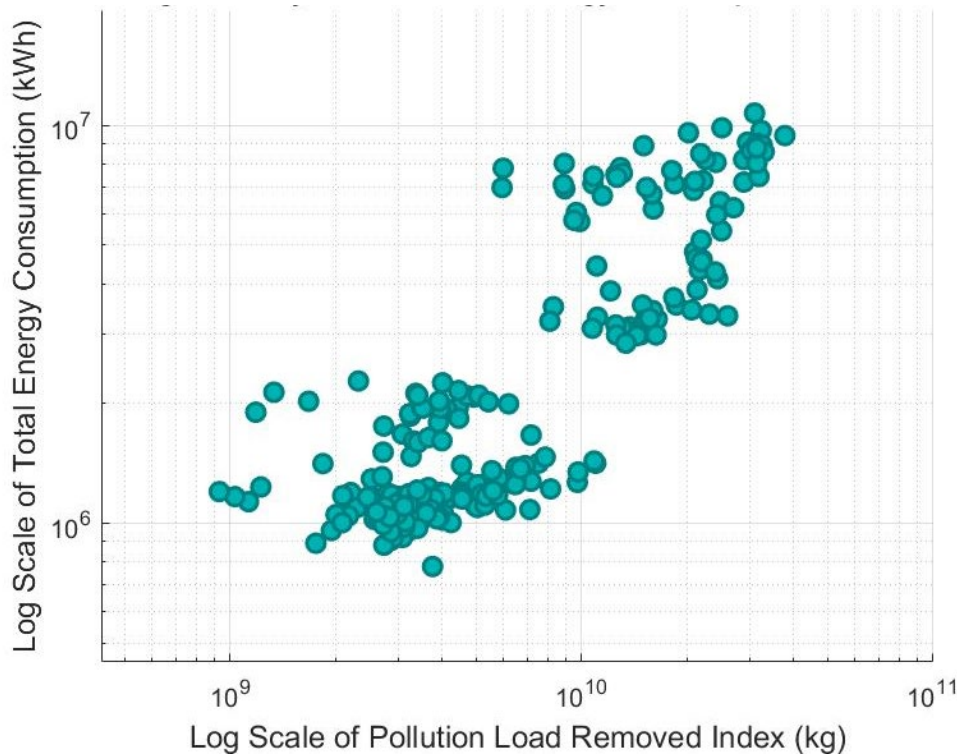


Figure 3: Correlation between average PLR and energy consumption at WRRFs.

Using equations 1 and 2 above for the relationship between energy and PLR and energy and WRRF technological extensions, a final regression model was built to quantify the relationship between pollution removal, specific types of plant extensions and energy consumption. The final model that represents energy use as a function of the removal of four key pollutants, water flow (in the form of load) and the type of technological extension built to treat wastewater to a higher quality is shown in Table 2. The model explains over 95% of the energy use variation as noted by the adjusted R^2 correlation statistic, it is also a highly statistically significant model as denoted by each predictor's p-value and the overall model p-value. The t-statistic values show that the independent variables are a very good fit. As can be seen in the table, the final model shows that the variance in energy consumption is mostly determined by the amount of pollution load removed, as well as the installation of additional activated sludge processes.

Table 2: Robust linear regression model results.

Estimated Coefficients:	Estimate	SE	tStat	pValue
Intercept	-4.06E+06	1.94E+05	-20.91	1.24E-55
Installation of additional ASP	45759	1.88E+03	24.30	3.63E-66
Pollution Removal Index	0.0002	3.97E-06	43.91	1.55E-115

Number of observations: 239; Error degrees of freedom: 236; Root Mean Squared Error: 4.59e+05

R-squared: 0.951; Adjusted R-Squared **0.951**; F-statistic vs. constant model: 2.31e+03;

p-value = **1.32e-155**.

The final linear regression model (robust fit) is thus:

$$\text{Energy Consumption} = \beta_0 + \text{ASP} + \text{PLR} + \epsilon \quad (4)$$

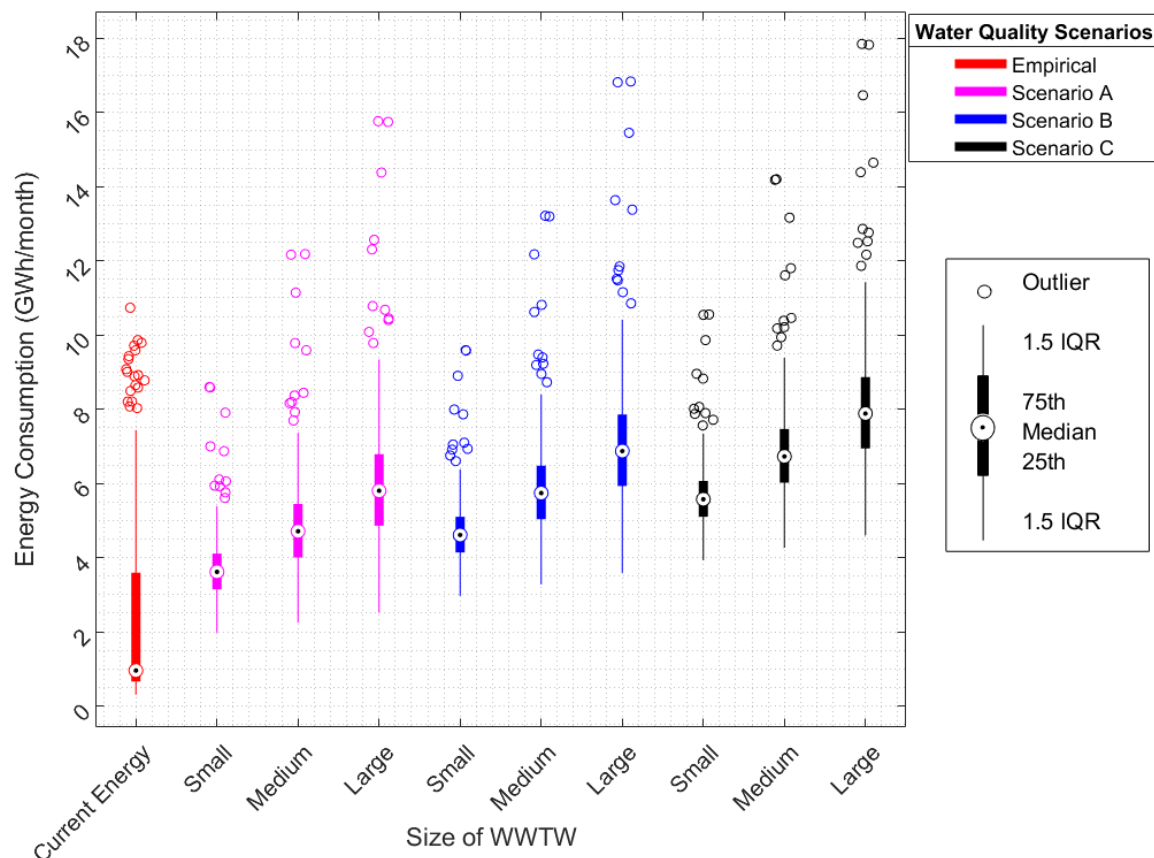
Where β_0 represents the intercept, ASP represents the Activated Sludge Process technology added, PLR represents the Pollution Removal Index and ϵ represents the error term, which in the scenario predictions is represented as the randomized standardized error from the empirical data. The final regression model is used to predict the energy consumption of wastewater treatment in the UK under changing effluent standards for a set of scenarios.

Scenario results

Figure 4 shows the monthly energy consumption for the scenario sample WRRFs under the Scenarios A, B and C of more stringent effluent quality standards for each of the three sizes of works. The boxplots represent the median energy values in GWh per month for each WRRF. The tops and bottoms of each

301 'box' represent the 25th and 75th percentiles of the results, respectively. The dot represents the median,
302 the vertical lines represent the furthest observations within 1.5 interquartile range and observations
303 beyond are outliers. The first boxplot represents current empirical energy consumption. Each set of
304 three boxplots thereafter represents Scenario A, B or C, and the order in which they're shown represent
305 the three sizes of works. For example, ScenA-MedT, indicates effluent standard Scenario A and the
306 middle sized WRRF. As can be seen, the size of a WRRF has a large influence on the energy consumption
307 of the treatment works. However, the more stringent standards in each scenario also result in increasing
308 energy consumption with higher effluent quality requirements. It can also be noted in the boxplot that
309 both in the actual data sample and in the modelled results there are quite a few high-energy
310 consumption outliers due to some months of the sample having particularly large contamination loads
311 to be removed. These are most likely due to drier conditions leading to higher concentrations of
312 pollution in specific months, thus leading to some variability in the energy costs of removing the
313 pollutants. It is also noteworthy that the current distribution is skewed whereas the projections are
314 more symmetrical, apart from the outliers, because linear regression assumes a gaussian distribution in
315 the projections.

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318 *Figure 4: Energy Consumption of each WRRF under scenarios of more stringent effluent standards and three medium-to-large*
 319 *WRRF sizes.*

320

321 Up-scaling Results

322 The electricity consumption of the sample WRRFs in the case study doubled over the period between
 323 2010-2015. The results show that on average energy consumption in large WRRFs for the whole of the
 324 UK could further increase by 3,147, 4,397 and 5,593 GWh per year to meet more stringent effluent
 325 standards (subject to baseline consents), ranging in a percentage increase of between 112% and 211%.
 326 Table 3 shows the absolute increases for energy consumption alongside the increases in GHG emissions.
 327 In terms of additional GHG emissions, the MTCO₂ emitted will depend on the composition of energy

sources in the grid. Using the three scenarios of decarbonisation described above, Table 3 shows the average potential range of additional emissions under each of the more stringent effluent standard scenarios versus the grid decarbonisation scenarios.

Table 3: GHG emissions (MTCO₂/year) and Energy Consumption (GWh/year) from more stringent effluent standards against grid electricity intensity factors

Effluent Standards	Current Standards	Scenario A	Scenario B	Scenario C
Average energy consumption (GWh/year)				
GWh/year	2805.9	5953.3	7203.1	8399.7
Carbon emissions factor	Average GHG emissions (MTCO₂/year)			
Current 0.41 kgCO ₂ e/kWh	1.15	2.43	2.94	3.42
Barriers Scenario 2020 0.25 kgCO ₂ e/kWh	0.70	1.49	1.80	2.10
Central Scenario 2020 0.18 kgCO ₂ e/kWh	0.51	1.07	1.30	1.51

The results show that more decarbonised futures somewhat offset the increase in emissions from more stringent standards. There are 0.51-1.51 annual MTCO₂ emitted under the most decarbonised grid, versus a range of 1.15-3.42 under current emissions factors. However, even with the lower effluent standards and the best mix of energy sources in the grid there could still be over 0.5 MTCO₂ emitted annually from changing effluent standards towards 2020 (depending on the emissions factor). Taking into account that according to the latest UK statistics, the waste management sector (which includes the wastewater sector) emitted just over 18 MTCO₂ of the 495 MTCO₂ emitted in the UK in 2015 (BEIS, 2017), such increases could be very significant.

It is important to know that these calculations would only concern the large WRRFs in the UK, which have the capacity to treat wastewater for the population equivalent of approximately 53 million (EEA, 2015). There are 1896 WRRFs in the UK, and while the remaining 1795 WRRFs are of smaller size with an average size of 25 ML/day (and a combined treatment capacity of 37 million p.e.), meeting more stringent effluent standards at these smaller works will also require larger quantities of energy, especially as smaller works do not benefit from economies of scale. Furthermore, this study only considers increases in the direct operational energy and emissions from WRRFs. Even though studies have found that operational emissions from electricity consumption are the largest and most significant at WRRFs, there are other indirect costs and implications from tighter effluent standards, such as more direct emissions from the processes themselves, that are not included in this study.

Discussion

The final model results show a significant relationship between pollution removed at a WRRF and the energy consumed. This is coupled with the finding that the addition of more treatment in the form of activated sludge processes has a very strong impact on the energy consumption of the works.

We note that alongside increasing nitrogen and phosphorus standards, more stringent standards for other emerging contaminants (ECs), such as metaldehyde, hormones or pharmaceutical products, are to be expected, potentially requiring even more energy intensive technologies to remove difficult pollutants. Current state of the art WRRFs have not been designed for the removal of ECs and additional infrastructure will be needed (Sichel et al., 2011). Technologies can include advanced oxidation processes, ozonation, activated carbon, or membrane technologies, which will all incur significant capital and operational costs. Raghav *et al.* (2013) suggested that removing ECs of concern is technically possible, but extremely energy intensive. Further work is still needed to understand the long-term

environmental and health effects of ECs in order to evaluate whether costly investments and their associated energy and greenhouse gases would be justifiable.

Previous studies have recognised aeration as one of the largest energy consumers within wastewater treatment (Caffoor, 2008; Curtis, 2010; Gao et al., 2014), which has been confirmed in the current study, despite technological innovation in recent years. This highlights the importance of promoting further technological innovation, including energy conservation and wastewater-derived resource recovery options, as well as catchment management strategies that reduce the need for aeration, to reduce water pollution and develop new low-cost treatment solutions. Curtis (2010) presents a comprehensive comparison of alternative strategies and technologies to reduce the need for aeration in WWT. That discussion is expanded below to include newer, state-of-the art technologies and approaches to treat wastewater to high quality standards at lower costs in the following discussion.

A study by the UKWIR (2015), on the cost effectiveness of reducing phosphorus not only in the water sector but also in other sectors, found that measures directed at wastewater treatment works are generally less costly and would achieve a greater improvement in compliance than catchment based measures. Few very innovative nutrient removal technologies with large-scale potential abound. So far, urine separation is one of the few areas of research. However, cultural obstacles are difficult to overcome and, even if they are, urine separation is likely to be seen first in in new-buildings, as it is hard to retrofit into existing wastewater systems. Source separation technologies for wastewater systems are reviewed in depth by Larsen, Udert and Lienert (2013). Water utilities in the UK have also started investing in technologies for advanced pre-treatment which reduces SS/BOD in primary treatment and offloads pressure on ASP systems to reduce energy demand.

Other options include green infrastructure systems, including reed beds, lagoons and wetlands, that are low energy natural systems to pre-treat sewage and runoff at or near the source. Crites, Middlebrooks

and Reed (2010) present an in-depth analysis of the design features and performance for multiple green infrastructure options that provide low cost, low energy use WWT. Chouinard *et al.* (2015) carry out a comparative analysis of cold-climate constructed wetlands (CW) in Canada and China. They report on a Canadian study that used CW to treat effluent, which achieved removals of 34% BOD, 52% of ammonia, 90% of phosphorus and 93% of suspended solids. Some newer CW technologies have also been proven to remove emerging organic contaminants such as pharmaceuticals and pesticides, while also removing over 99% of SSs, BOD and NH_4 (Ávila *et al.*, 2015). The cost of building and operating natural treatment systems is usually lower than conventional treatment options, but land requirements are a limitation as well as capacity, thus so far have limited application for large urban areas such as London.

Anaerobic processes, which are widely used in the wastewater sector in warmer regions, could be further integrated into WWT, for example by substituting primary settlement tanks, which would reduce energy consumption, and provide more biogas. Methanogenic systems and microbial fuel cells have shown potential, although they still perform poorly in colder regions, and are used mostly in small-scale settings (de Mes *et al.*, 2003). Decentralization is also gaining traction in the wastewater sector. Chong *et al.* (2011) compared two types of decentralised systems that produce high quality effluent, on their energy consumption and GHG emissions, with mixed results. Decentralization may provide benefits for future infrastructure but it is hard to implement in already established large wastewater systems. Oxidation and disinfection include UV light and ozone technologies which are high energy, but with potential for low-cost development. Advanced Oxidation Processes (AOPs) are one such area of research, which involve the generation and use of the hydroxyl radical to oxidize compounds that cannot be oxidized by conventional oxidants. Saharan *et al.* (2014) review AOPs in detail as well as their combinations, making recommendations for the most efficient treatment depending on conditions.

They are particularly attractive because solar light can be used in some of the applications (Agulló-Barceló et al., 2013).

Even though the primary function of WRRFs is to remove contaminants from wastewater, they can serve as sources of energy and other materials. Energy recovery from waste is becoming a reality with emerging technologies able to recover significant resources from wastewater (Kretschmer et al., 2016). Gao, Scherson and Wells (2014) discuss the potential of direct energy recovery from nitrogen in WRRFs from different methods such as the CANDO process, which converts ammonia to nitrous oxide gas. Mo and Zhang (2013) systematically review and compare methods for energy generation. Combined heat and power systems have large capital costs, but are appropriate for large WRRFs, and can achieve reductions of up to 26% of electricity consumption (Stillwell et al., 2010). All Sludge Treatment centres in the Thames system already use CHP in some form, which is focused particularly at the larger works. Biosolids incineration is another technique that has been applied in WWT and has been reported to reduce 57% of electricity use in some works in the US (Clayton et al., 2014). However, incineration does present challenges such as the release of persistent environmental pollutants, quality problems and high capital investments (Mo and Zhang, 2013).

On-site effluent hydropower can involve the generation of power from flowing effluent. There are opportunities to develop new or retrofitted generation of renewable energy in the UK's wastewater system (Elías-Maxil et al., 2014). These systems require there to be level differences in the pipes, or between the stream and turbine, as well as enough flow. Other approaches for energy generation include THP (pre-treatment at high pressure/temperature to achieve better digestions and generation), which is being adopted widely in the UK, onsite renewable generation, heat pumps, bioelectrochemical systems, and the use of microalgae. The challenges for energy generation include large capital costs (for

example, for combined heat and power systems), lack of reliability and specific requirements for local conditions.

A major limitation to the uptake of new technological approaches, which has been observed in this study, is the lack of technical and scientific information on the applicability, performance, energy costs and sustainability (e.g. emissions) of many emerging options. Highly urbanised areas such as London are under multiple pressures, and finding the appropriate methods to treat wastewater to higher quality implies tough decisions and significant trade-offs. It is likely that water utilities will have to continue to invest in end-of-pipe solutions, due to land and other constraints; although work with farmers and other strategies to deal with the sources of contaminants is proving key to reduce the inflow of pollution into WRRFs. The continued decarbonisation of the grid will play an essential part in the water sector's energy and greenhouse gas footprint, as end-of-pipe solutions tend to be energy intensive. There may still be an invigoration of technological innovation in wastewater as carbon prices increase, and other developments might occur that are currently unknown. The water sector's continued involvement in the climate change debate will be key to drive innovation in the area.

Finally, some of the limitations of the present study should be highlighted. The results are limited in that the availability and accuracy of the data was not complete, which increase the likelihood of potential model errors and approximations. Only large WRRFs were used, but more complete studies could assess and compare different sized works. Furthermore, the upscaling of the results may have led to some errors and scale effects. The model considers a set of specific scenarios to illustrate potential future costs. However, possible future scenarios other than the scenarios considered could have been used, and even though several were trialled, different or extreme scenarios could yield alternative results. Further research could aim to quantify these relationships under a larger uncertainty space, as well as

under a much larger range of external factors, such as renewable energy uptake or innovative technologies, although new data would be needed to include such aspects.

Conclusion

This study quantified the relationship between the removal of key pollutants in large wastewater treatment works and the associated energy consumption. It also demonstrated that the addition of activated sludge processes increases the energy consumption at the works. The analysis was carried out using six major wastewater treatment works in the UK and used to estimate the energy and greenhouse gas emissions of changing effluent standards on wastewater treatment. The results showed that more stringent effluent standards could result in at least doubling of electricity consumption in large WRRFs in the UK, and the addition of between 1.3 and 2.3 additional MTCO₂ per year for the sector. It is important to note that more stringent standards are essential to improving effluent and environmental quality. Thus, these conclusions show the difficult trade-offs between seemingly complementary environmental policies. The increases could have significant financial and regulatory implications for the water sector, and could impact the UK's emissions reduction targets. It was also found that the decarbonisation of the grid will contribute significantly to offsetting the emissions incurred by more stringent effluent standards. However, more coherence is needed between environmental policies to avoid negative trade-offs and meet national emissions reductions without compromising water quality.

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