11981 words

# Energy performance factors in wastewater treatment plants: a review

Bruno J. Cardoso [a*], Eugénio Rodrigues [a], Adélio R. Gaspar [a], Álvaro Gomes [b]

[a] University of Coimbra, ADAI, Department of Mechanical Engineering,
Rua Luís Reis Santos, Pólo II, 3030-788 Coimbra, Portugal
[b] University of Coimbra, INESC Coimbra, Department of Electrical and Computer Engineering,
Rua Sílvio Lima, Pólo II, 3030-290 Coimbra, Portugal
[*] Corresponding author: bruno.cardoso@adai.pt

## Abstract

Given their increasing number and the implementation of more energy-intensive treatment methods, wastewater treatment plants (WWTPs) expect to increase energy consumption. In addition, climate change presents new challenges to the operation of these facilities, thus being critical to understand how to improve their energy performance and environmental sustainability while ensuring the quality of service provided. This paper reviews the latest publications on the energy performance of municipal WWTPs, particularly on the different phases of the treatment process and the impacting factors. The contextual and underlying factors that influence energy performance were identified, categorized, and analyzed through a broad survey. The most significant factors are the plant size, load factor (plant capacity utilization), and dilution factor. The implementation of anaerobic-anoxic-oxic systems is considered suitable in sensitive areas requiring high pollutant and nutrient removal rates, presenting, in some cases, moderate energy consumption (0.267 kWh/m$^3$), being similar to the conventional activated sludge (0.269 kWh/m$^3$). In more stringent effluent quality requirements, such as wastewater reuse, membrane bioreactors are advised, despite higher consumption (0.33 kWh/m$^3$). Energy improvements can also be achieved by implementing automatization, inverters, and strategies that increase flexibility and adaptability in the operational process. Lastly, given the multidimensional characteristics of the WWTPs assessment, further improvements may be identified if the energy performance of these plants

is compared using holistic and multi-criteria approaches, integrating multiple inputs and outputs simultaneously.

**Keywords:** Wastewater treatment plant; Energy performance; Energy efficiency; Benchmarking; Water; Performance factors

# 1 Introduction

Design and management of wastewater treatment plants (WWTPs) are driven by effluent quality standards defined by national regulations. Until recently, the environmental impacts resulting from energy consumption have been secondary (Rojas and Zhelev, 2012), even though the water sector is energy-intensive, representing nearly 44 % of the municipality's energy costs (Copeland and Carter, 2017).

In 2014, the global energy demand for the water sector was estimated at 120 million tonnes of oil equivalent, which corresponds to 1395.6 terawatt-hours, and projections estimate this value will double before 2040 (IEA, 2016). These projections assume: (i) a rise in municipal and industrial water withdrawals triggered by higher demand from the residential, commercial, and agricultural sectors provoked by population growth and urbanization (Bodík and Kubaská, 2013); (ii) a rise in nontraditional water supply sources, such as desalination and wastewater reuse, aggravated by the uneven distribution of water and stress over water resources caused by climate change (Wakeel et al., 2016); (iii) a rise in the contaminant load (Hernández-Sancho et al., 2011a), and lastly, (iv) stringent regulatory and environmental standards for effluent quality and wastewater reuse (Hernández-Sancho et al., 2011a).

This increase in energy consumption may lead to a considerable boost in $CO_2$ emissions in the water sector when fossil fuels are used in power plants (Hao et al., 2015) and to substantial growth in operational costs due to energy being a major cost contributor (Molinos-Senante et al., 2014b). Therefore, the trend will likely deplete natural resources, increase environmental risks, and severe financial challenges (Ananda, 2018).

In highly urbanized and industrialized countries, population growth has generated more wastewater, which requires adequate treatment before being safely discharged into the receiving water bodies (Chatzisymeon, 2015). To maintain the ecological status of water resources, the population connected to wastewater collection and treatment has increased (Di Fraia et al., 2018), leading to countless urban WWTPs operating throughout the world (Hernández-Chover et al., 2018). The number of WWTPs will continue to grow to ensure availability and sustainable management of water and sanitation for all (Goal 6 of the United Nations' 2030 Agenda). Additionally, more advanced and energy-intensive treatment processes will be adopted in the following decades (Di Fraia et al., 2018).

The operation of WWTPs poses critical challenges in terms of economic and environmental sustainability. For example, a rise in resource consumption, greenhouse gas (GHG) emissions onsite and offsite, and energy costs are some challenges (Lorenzo-Toja et al., 2015). So, to ensure sustainable development, the negative impacts of these facilities must not exceed their benefits (Jiang et al., 2020). A progressive change in attitudes has undeniably led to a new paradigm that seeks to maximize recovery of energy and resources through the design and management of these facilities, thus pursuing a transition to neutral or even positive energy plants (Maktabifard et al., 2018).

Due to the economic, social, and administrative pressures, it is essential to evaluate and improve the EE of existing facilities (Longo et al., 2016) to guarantee the sector's long-term sustainability. Managers and operators must minimize operating costs and increase environmental sustainability while ensuring the quality of their provided service (Molinos-Senante et al., 2014b). The performance analysis of European WWTPs uncovered a significant energy efficiency (EE) potential (Torregrossa et al., 2017) and energy cost savings of up to 25 % without decreasing effluent quality (Castellet and Molinos-Senante, 2016).

This article reviews the latest scientific publications to: (1) determine which of the treatment phases has the most impact on energy consumption and may improve EE; (2) identify the contextual and underlying factors that affect energy performance; (3) compare performance across various countries, different levels of treatment and secondary treatment technologies, and (4) understand how EE and long-term sustainability may be achieved in municipal WWTPs.

By reviewing the latest findings and technological advancements in wastewater treatment, this work contributes to the development and adoption of sustainability strategies by focusing on implementing measures to improve EE and reduce the environmental impacts of WWTPs while safeguarding treatment efficiency.

# 2 Material and Methods

The literature review characterizes the energy consumed in the treatment process, the different specific energy consumption (SEC) values worldwide, and how various contextual or underlying factors influence energy consumption and the performance of the facilities. From the studies and benchmarking processes, it was possible to draw up conclusions and recommendations. The methodology is illustrated in Figure 1.

Initially, peer-reviewed publications indexed in Science Direct and Google Scholar websites were collected. A combination of keywords in the title was used, which included “WWTP”, “wastewater”, “sewage”, “energy”, “efficiency”, “benchmarking”, “intensity”, “assessment”, “performance”, “consumption” and “analysis”. These publications were filtered by date to obtain the most recent studies. Additional filtering criteria were employed: (a) publications from 2020 that include the most recent results; (b) publications between 2017 and 2019 with at least one citation per year; and (c) publications before 2017 with at least ten citations in total. Therefore 296 publications were gathered.

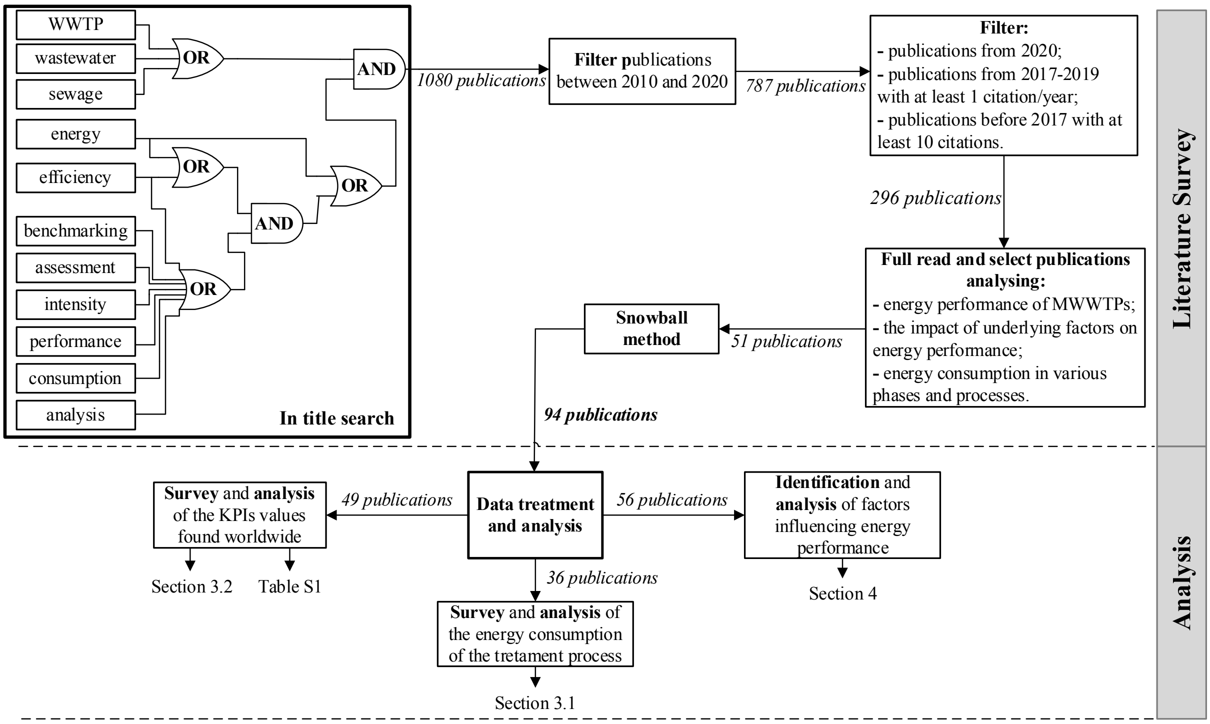


**Figure 1** - Overall procedure framework.

From the above sample, the selection was conducted according to the following criteria: (a) publications analyzing or comparing the energy performance of municipal WWTPs, either by calculating key performance indicators (KPIs) that relate energy consumption to the volume of treated wastewater; or to the population equivalent (PE); or to the pollutant load removed; or by applying other benchmarking methodologies such as data envelopment analysis (DEA); (b) publications assessing the impact of underlying factors, such as size, type of secondary treatment technology, or capacity load rate; and lastly, (c) publications which have examined the energy consumption of the various phases, stages, and equipment comprising the wastewater treatment process.

A sample of 51 publications was obtained. This sample presents an adequate geographical distribution of WWTPs, covering all continents, despite some more represented countries. During the detailed analysis, new studies were collected from the list of references. The publications were only added when the study presented an added value or analyzed countries not earlier included, thus improving the sample's heterogeneity and guaranteeing the broadest geographical coverage (Figure 2). A final sample of 94 publications was obtained.

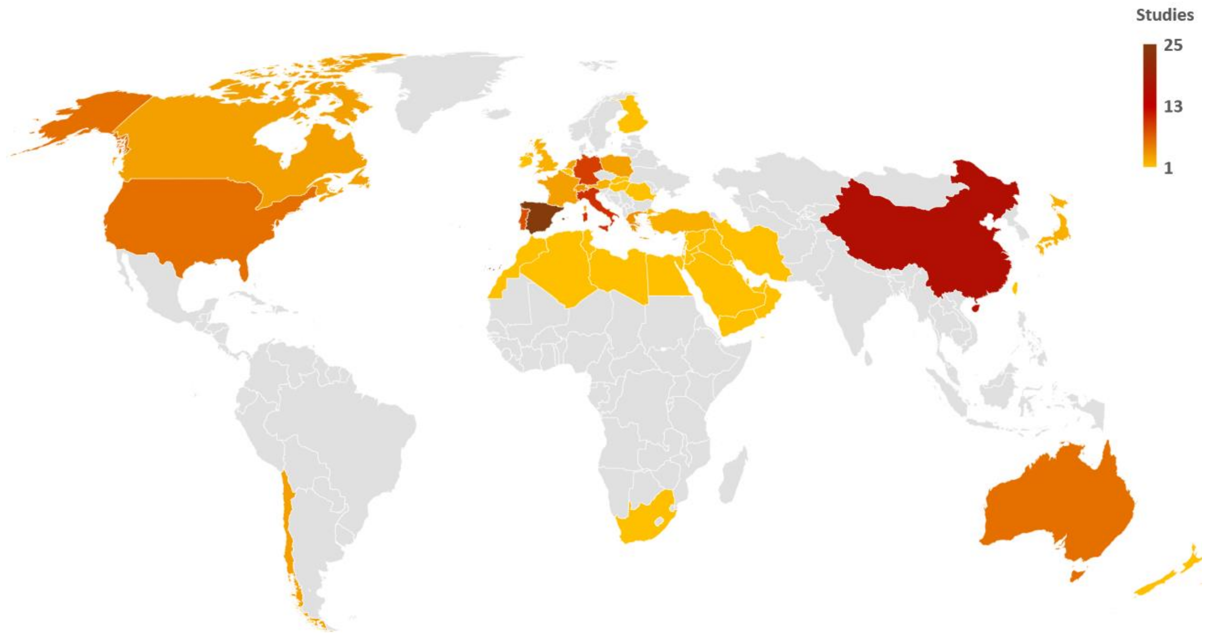


Figure 2 - Geographical coverage of the sample.

In the second stage, the methodologies employed in the studies, the calculated SEC values, and the conclusions, among others, were extracted from the publications. Then, the values of the several simple KPIs were surveyed, tabulated (Table S1), and analyzed (Section 3.2). Afterward, the survey and analysis of the energy consumption of the various phases and stages of treatment were performed (Section 3.1). Finally, the identification, categorization, and critical analysis of the factors influencing energy performance were carried out (Section 4).

# 3 Wastewater operation performance

Energy has an essential role in water and wastewater services, generally accounting for the second-largest share of operating costs, often only surpassed by personnel costs (Copeland and Carter, 2017). Nevertheless, energy costs vary significantly (Table 1), ranging from 9 %

(Lindtner et al., 2008) to the majority of the operating costs (Tsagarakis et al., 2000), as operating and other main costs depend on context and country. Moreover, energy costs depend on several internal factors (*e.g.*, onsite energy production from biogas or renewable sources) and external factors (*e.g.*, energy tariffs) (Lindtner et al., 2008).

Given the impact on operating costs, energy savings are particularly relevant as it contributes to decreasing energy costs while reducing GHG emissions (Rodriguez-Garcia et al., 2011). Furthermore, appropriate guidelines and strategies can be drawn from identifying the most significant energy consumers (Silva and Rosa, 2015), thus reducing the dependence on purchased energy and reaching financial sustainability (Maktabifard et al., 2018). Alternatively, energy costs could be further reduced by implementing energy production solutions through biogas (de Haas et al., 2015) or renewable energy solutions, such as photovoltaic systems (Yang et al., 2020).

**Table 1 -** Percentage of total operating costs related to energy costs.

| Publication | Country | Operating costs | Size |
|---|---|---|---|
| (Tsagarakis et al., 2000) | Greece | 50 % | < 10,000 PE |
| (Lindtner et al., 2008) | Austria | 9 % | > 100,000 PE |
| (Molinos-Senante et al., 2010) | Spain | 18 % | 1,000,000 – 8,000,000 $m^3$/year |
| (Rodriguez-Garcia et al., 2011) | Spain | 26 % | > 4000 PE |
| (Silva et al., 2012) | Portugal | 25 % | 360 – 54,500 $m^3$/day |
| (Vera et al., 2013) | Chile | 25 % | 1000 – 500,000 PE |
| (Molinos-Senante et al., 2014b) | Spain | 22 % | 3000 – 3,000,000 $m^3$/year |
| (Haslinger et al., 2016) | Austria | 11 % | > 10,000 PE |
| (Haslinger et al., 2016) | Austria | 17 % | < 10,000 PE |
| (Castellet and Molinos-Senante, 2016) | Spain | 22 % | Unspecified |

## 3.1 Energy in wastewater treatment phases

Generally, wastewater treatment consists of up to five main stages: preliminary treatment, primary treatment, secondary treatment, tertiary treatment, and sludge treatment (Silva et al., 2015). In the preliminary treatment stage, wastewater may undergo several steps, such as screening, grit removal, and grease removal. Typically, these steps constitute a small portion of the total electricity consumption, varying between 0.022 kWh/$m^3$ and 0.042 kWh/$m^3$ (Longo et al., 2016), and may reach 11 % of the total electricity consumed depending on the installation and the intensity of the treatment (Mamais et al., 2015).

In most cases, the primary treatment stage is a simple separation step in circular settling tanks equipped with mechanical scrappers (Longo et al., 2016). As such, this stage may be affected by design and operation and is typically less energy-intensive than the others (Wakeel et al., 2016). The primary settling stage, for example, consumes between 0.043 Wh/$m^3$ and 0.07 Wh/$m^3$ (Longo et al., 2016).

The secondary treatment usually represents the largest portion of energy consumption due to the aeration of wastewater in the biological process (IEA, 2016). This requirement is mainly satisfied by injecting oxygen into the reactors through a mechanical or surface aeration system (a mixture of air and water on the surface) or through a diffuse aeration system (using blowers) (Mamais et al., 2015). Typically, these systems have high installed power and work almost continuously, making them significant energy consumers (Belloir et al., 2015). However, the total amount depends on the level of contamination and type of technology (Wakeel et al., 2016). For example, aeration systems consume between 0.18 kWh/$m^3$ and 0.8 kWh/$m^3$ (Longo et al., 2016), representing, on average, 40 % and 75 % of the total energy consumed in large and small plants, respectively (Mamais et al., 2015).

Table 2 summarizes the energy impact of aeration and other primary consumers on total plant energy consumption found in the literature. A non-neglectable portion of energy consumption belongs to submersible mixers and sludge recirculation pumps, which work for extended periods despite their low power. In some cases, these submersible mixers — used to mix the biological process — may represent 10.5 % to 14.8 % of the plant's energy consumption. Although not necessarily excessive or indicative of energy waste, the energy consumption can be reduced simply and inexpensively by adapting and optimizing the operating profile of some equipment without affecting the treatment process (Cardoso et al., 2020). However, this consumption is repeatedly overlooked in small WWTPs (Foladori et al., 2015).

Given that a great part of the energy consumed is usually related to the secondary treatment and aeration, this should be the main object of study to find possible improvements and EE measures (*e.g.*, control optimization of the biological treatment process).

**Table 2 -** Main energy consumers in WWTPs (percentage of total plant energy consumption).

| Publication | Country | Aeration | Sludge treatment | Pumping | Others | Remarks |
|---|---|---|---|---|---|---|
| (Aymerich et al., 2015) | Spain | 42 % | 14 % | 20 % | 24 % | 1 WWTP; advanced biological removal; 115,000 PE. |
| (Mizuta and Shimada, 2010) | Japan | 48 % | 29 % | 15 % | 8 % | 4 WWTPs; conventional activated sludge (CAS); 48,790 – 59,171 $m^3$/day. |
| (Panepinto et al., 2016) | Italy | 50 % | 29 % | - | 21 % | 1 WWTP; tertiary treatment; 2,700,000 PE. |
| (Tao and Chengwen, 2012) | China | 52 % | 9 % | 18 % | 21 % | 1 WWTP; representing several types and scales. |
| (Henriques and Catarino, 2017) | Portugal | 53 % | - | 12 % | 35 % | 14 WWTPs; several types and scales. |
| (Zaborowska et al., 2017) | Poland | 53 % | - | 30 % | 17 % | 1 WWTP; biological nutrient removal; 200,000 PE. |
| (Cardoso et al., 2020) | Portugal | 54 % | 13 % | - | 33 % | 2 WWTPs; CAS; 19,300 – 21,000 PE. |
| (Gu et al., 2017) | - | 60 % | 12 % | 12 % | 16 % | Representative of CAS system. |
| (Mamais et al., 2015) | Greece | 66 % | 8 % | - | 26 % | 4 WWTPs; 13,500 – 3,900,000 PE. |
| (Marner et al., 2016) | Germany | 67 % | 11 % | 5 % | 17 % | Several types and scales. |

Suspended solids, pathogens (viruses and bacteria), and pharmaceuticals are typically removed in the tertiary treatment stage (Zeng et al., 2017). The excess of nutrients (nitrogen and phosphorous) may also be removed through chemical methods. However, these are not addressed in this study. This stage includes processes that might be quite energy-intensive, such as ultraviolet radiation (UV), ozone disinfection, filtration, and other advanced techniques to disinfect water or remove non-biodegradable organic matter (Silva et al., 2015). For instance, UV systems may consume between 0.045 kWh/$m^3$ and 0.11 kWh/$m^3$ (Longo et al., 2016).

Together with aeration and pumping, the sludge treatment stage typically demands the highest amount of electricity (Maktabifard et al., 2018). Indeed, this stage may represent 8 % (Mamais et al., 2015), 15 % (IEA, 2016), or 35 % (Di Fraia et al., 2018) of total energy consumption. The sludge treatment stage mainly comprises thickening, stabilization, and dewatering processes (Scholz, 2016). Typically, thickening and dewatering have a marginal share in energy consumption (Vaccari et al., 2018), while stabilization is more energy-demanding due to aerobic stabilization (Longo et al., 2016). Compared to aerobic stabilization, anaerobic digestion is a more energy-efficient option and the most common method applied (Scholz, 2016). However, its viability depends on the facility size, as energy production improves the plants' self-sufficiency and reduces energy costs (Longo et al., 2016).

## 3.2 Key performance indicators

Five energy-related KPIs were identified in the literature for the several types of treatment typologies and countries: (i) per volume of treated wastewater (kWh/$m^3$), (ii) per population equivalent (kWh/PE), (iii) per Chemical Oxygen Demand removed (kWh/kg COD rem), (iv) per five-day Biological Demand Removed (kWh/kg BOD rem), and (v) per mass of nitrogen removed (kWh/kg N rem). These are summarized in Table S1 in the Supplementary

Material. The two first indicators are the most common KPIs used in the scientific literature (Longo et al., 2016).

Figure 3 shows the average values (diamond) and range of values (bars) of SEC for the various countries addressed in the literature. The acronyms for the countries in the figures include numbers to indicate the respective study (identical to the one used in Table S1 in the Supplementary Material).

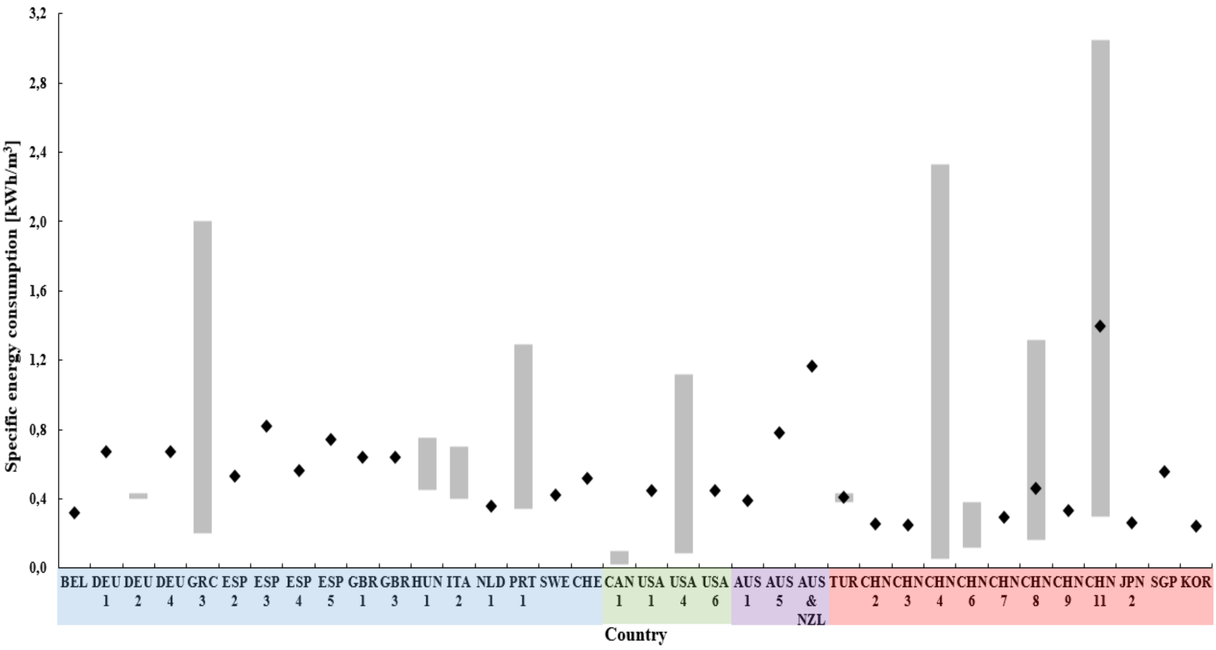


**Figure 3 -** Specific energy consumption of municipal WWTPs in several countries.

A difference in SEC between countries is observable, particularly China and Japan, which have lower values than Spain, Germany, the United Kingdom, Switzerland, and the United States of America (Figure 3). This difference is partly justified by disparities in target effluents and the dominant level of treatment. The primary treatment, the dominant process in Asia, is much less demanding and less energy-intensive than the secondary and tertiary treatment levels, widely found in OECD countries (IEA, 2016).

The large variations of SEC exhibited in some countries (*e.g.*, China) cannot be solely attributed to the size of the plants. For example, the CHN6 study presents a much larger variation in sizes than the CHN4 and CHN11. However, the CHN6 also has a smaller SEC variation and a higher SEC minimum than the CHN4. Therefore, if distinct plant sizes were the only justification for the SEC variations, then the CHN6 study would have to present a larger variation of SEC values and a smaller minimum value than the one found in CHN4.

Figure 4 depicts that energy intensity (EI) increases with the treatment level of wastewater. The primary treatment stage (0.1 kWh/m$^3$ to 0.37 kWh/m$^3$) has a much lower SEC than the other treatment levels, namely secondary (0.272 kWh/m$^3$ to 1.27 kWh/m$^3$) and tertiary (0.23 kWh/m$^3$ to 10.55 kWh/m$^3$). This trend is also observed within each country. For example, primary treatment in the Middle East and North Africa presents values ranging between 0.1 kWh/m$^3$ and 0.3 kWh/m$^3$, while values for secondary treatment are between 0.272 kWh/m$^3$ and 0.59 kWh/m$^3$. In Australia, primary treatment values (0.1 kWh/m$^3$ to 0.37 kWh/m$^3$) are also lower than those found for other treatment stages, namely the mean value for the secondary treatment (0.837 kWh/m$^3$) and the range of values for the tertiary treatment (0.23 kWh/m$^3$ to 10.55 kWh/m$^3$).

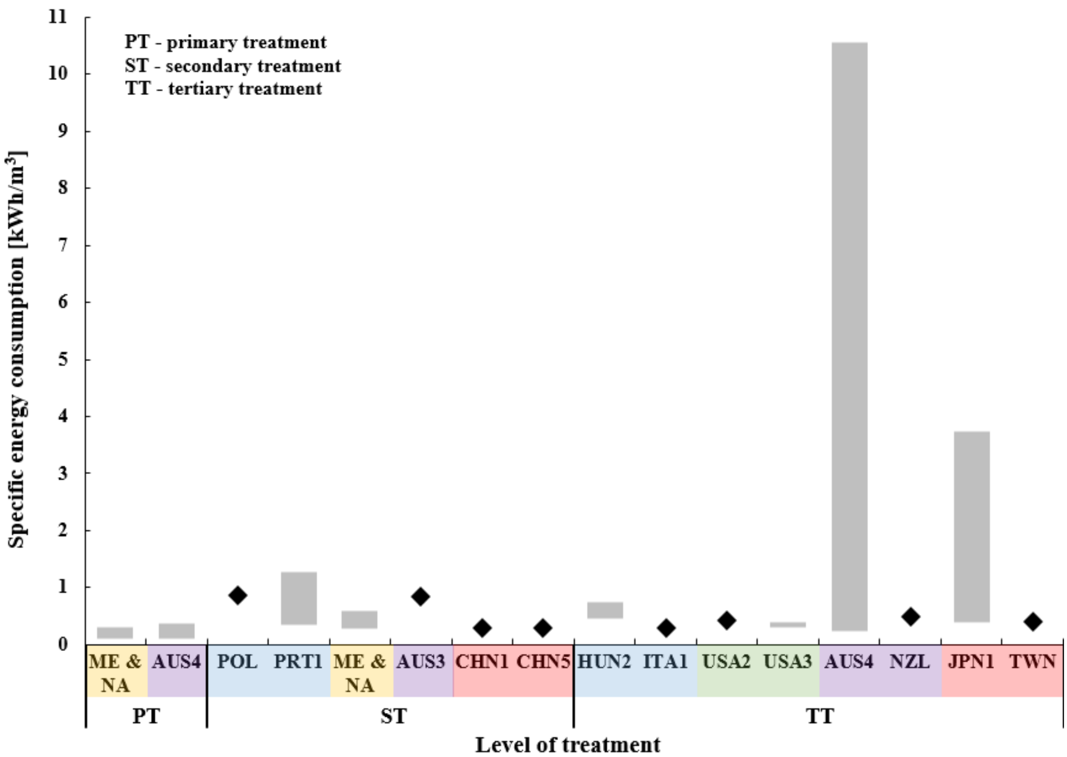


**Figure 4 -** Specific energy consumption for different treatment levels and locations.

Australia has a very demanding effluent quality requirement as its treated wastewater aims for human consumption or irrigation. This quality requirement justifies the wide range of values found in plants applying tertiary treatment, leading to the simultaneous application of several energy-intensive technologies (*e.g.*, microfiltration, nanofiltration, and reverse osmosis).

Secondary and tertiary treatments are more energy-intensive as these require more demanding treatment of wastewater and a higher quality discharge effluent than primary treatment. Therefore, one cannot compare the SEC of different treatment levels without considering the benefits of more demanding treatments, such as better effluent quality and the removal of nutrients and other pollutants.

Within the same treatment level, plants present diverse SEC values. This diversity is justified by different secondary treatment technologies, which have distinct EI values. By categorizing the types of treatment processes, it is easier to understand the differences. These processes include suspended-growth, attached-growth, oxidation pond, and hybrid or multi-stage.

Most of the SEC from the studies is related to suspended-growth processes, and this group has the broadest geographical coverage (Figure 5), indicating that they are well established worldwide. However, there is quite a significant difference between WWTPs with the highest and lowest SEC. Therefore, potential savings transversal to all secondary treatment types could be exploited by implementing EE measures. For instance, conventional activated sludge (CAS) treatment has an EI varying from 0.0845 kWh/m$^3$ to 3.18 kWh/m$^3$, whereas extended aeration (0.199 kWh/m$^3$ – 1.50 kWh/m$^3$) and oxidation ditch (0.057 kWh/m$^3$ – 2.12 kWh/m$^3$) have smaller variation ranges.

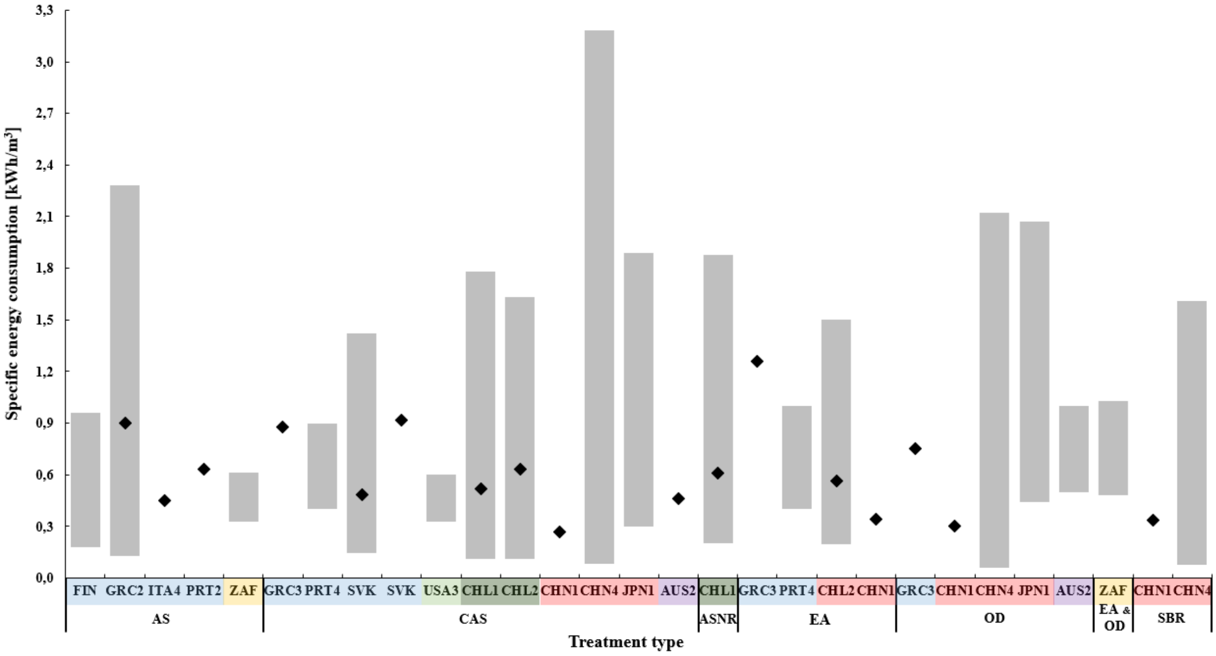


**Figure 5 -** Specific energy consumption for suspended-growth processes in several countries.

Similarly, for plants applying attached-growth processes, such as a trickling filter (TF) treatment, the values range from 0.19 kWh/m$^3$ to 1.82 kWh/m$^3$ (Figure 6). The different ranges of EI are due to several factors, such as different aeration requirements or hydraulic retention times (HRT). Attached-growth processes typically present lower EI than suspended-growth due to lower aeration requirements (Molinos-Senante, 2018). Similarly, the SEC of the oxidation ditch is higher than that of the CAS due to a longer HRT (Mizuta and Shimada, 2010). Nevertheless, even within the same type of treatment, there is significant variability of EI values. Plants are impacted by several factors, including size, flow rate, load factor, target effluent quality, or type of aeration system (Mizuta and Shimada, 2010; Zhang et al., 2016).

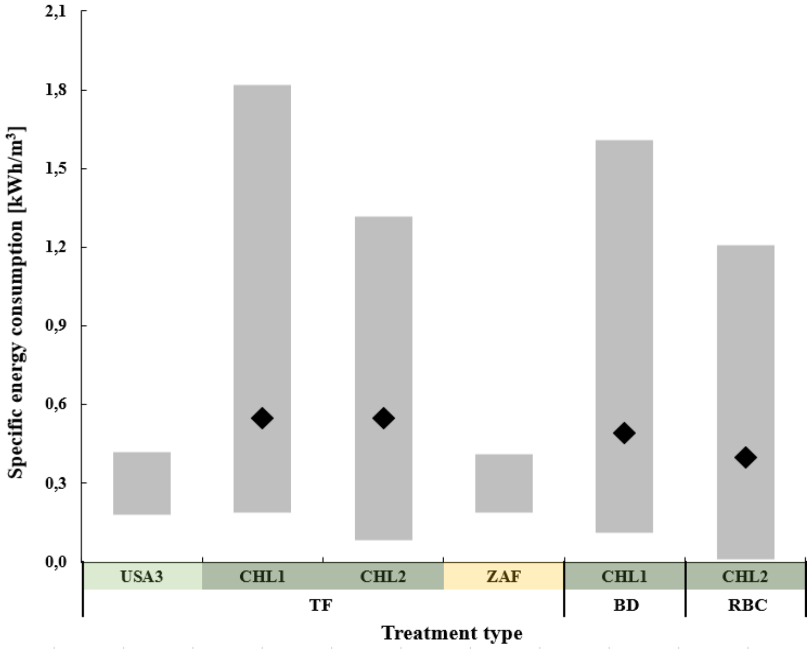


**Figure 6 -** Specific energy consumption for attached-growth processes in several countries.

Due to the reduced need for aeration, both lagoon systems and constructed wetlands display the lowest SEC, ranging between 0.079 kWh/m$^3$ and 0.29 kWh/m$^3$ (Figure 7). However, lagoon systems require a considerably larger land footprint than other treatment types, limiting their application in urban lands, coastal regions, and industrial areas (Wang et al., 2016). Still, when aeration is needed, as in aerated lagoons, the values are much higher, ranging between 0.01 kWh/m$^3$ and 2.1 kWh/m$^3$.

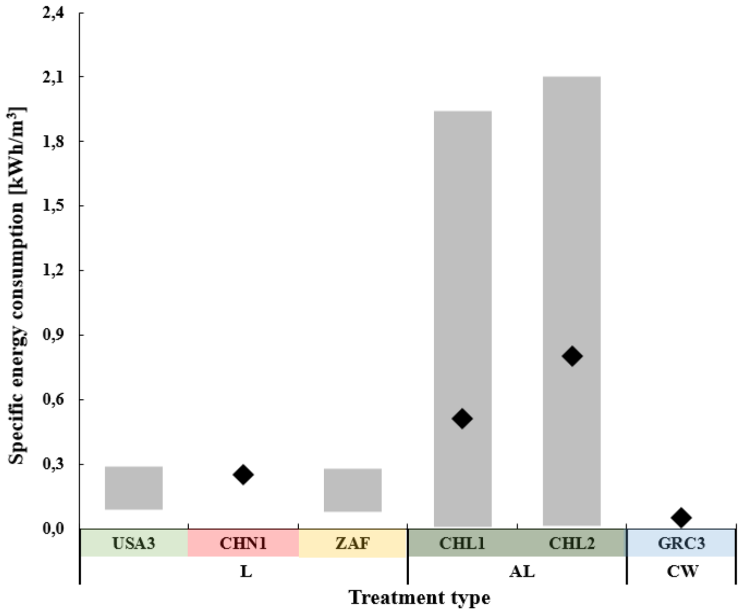


**Figure 7 -** Specific energy consumption for oxidation pond processes in several countries.

When analyzing hybrid and multi-stage processes (Figure 8), the SEC presented by membrane bioreactors (MBR) treatment ranges from 0.10 kWh/m$^3$ to 1.10 kWh/m$^3$, showing in the Chinese case an average value of 0.33 kWh/m$^3$. This technology is typically more energy-intensive than other systems because it applies a membrane separation step and requires intensive aeration rates to manage fouling and clogging (Krzeminski et al., 2012). The difference in values could be due to several factors, such as membrane type and configuration, level of membrane utilization, volume of treated flow, or even the adopted effluent discharge standard.

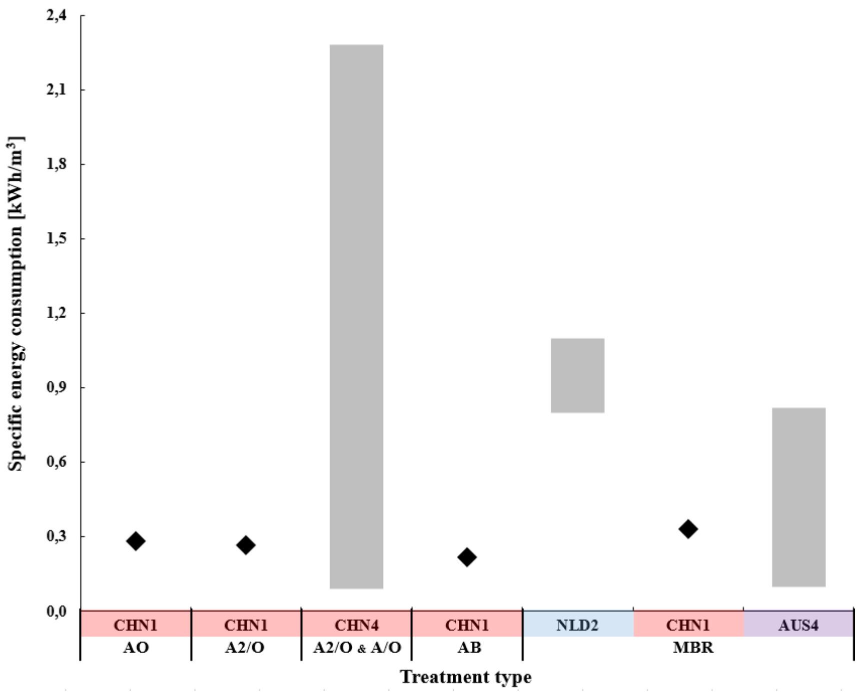


**Figure 8 -** Specific energy consumption for some hybrid and multi-stage processes in several countries.

SEC has significant variations within the same country and treatment type (Table S1 in the Supplementary Material), justified by other characteristics such as the facility's size or different effluent discharge standards. Therefore, comparing WWTPs based on SEC values alone should be done with caution, even for a similar treatment process or level of treatment, as conclusions may be misleading because several factors influence SEC. This highlights the need for implementing different methodologies which allow several inputs and outputs to be incorporated simultaneously in the evaluation (*e.g.*, the use of DEA or aggregated performance indicators, such as the Water Treatment Energy Index developed by the ENERWATER project (Longo et al., 2019, 2018)).

# 4 Energy performance factors

The WWTPs' energy consumption, EE, and performance differences on various levels (*e.g.*, economic, environmental) often result from distinct plant operations, characteristics, and context. Therefore, two main groups were created and analyzed individually, namely process-related and physical/context factors.

From factors analyzed in the following subsections, several recommendations are summarized in Figure 9. These recommendations may improve EE by ensuring an optimal plant design but also helping to adjust systems operation to the various dynamic conditions.

| Factor | Recommendations |
| --- | --- |
| *Load factor* | Adequately plan and design to prevent over-sizing or under-sizing. |
| | Implement flexibility and adaptability in the operational process to deal with seasonal variations of flow and pollutant load (e.g., automation, inverters). |
| | Increase the load factor by constructing an efficient sewage collection pipe network. |
| | Operate the WWTPs with a load factor between 80 % and 100 %. |
| *Dilution factor and polluted influent* | Improve and renovate sewage pipelines to reduce infiltrations. |
| | Accelerate the renovation of sewage systems and the transition to separate sewage systems to prevent mixing wastewater and rainwater. |
| | Implement, when viable, equalization tanks to deal with variations in load and flow rate. |
| *Secondary treatment technology* | Deploy anaerobic-anoxic-oxic systems in sensitive areas requiring high removal rates of pollutants and nutrients, while applying membrane bioreactors in situations with severe restrictions regarding effluent quality, such as wastewater reuse. |
| | Replace obsolete aeration systems with more efficient models, as well as apply variable frequency drivers and new automated controls to adjust aeration to oxygen demand. |
| *Tertiary treatment and effluent discharge standard* | Apply methodologies that incorporate multiple inputs and outputs in the analysis to carry out a holistic analysis and consider the advantages offered by each plant. |
| | Combine several techniques, e.g., chlorination and ultraviolet (UV) systems to improve disinfection efficiency and reduce by-products. |
| | Adopt dose-pacing control on UV systems to adjust the output according to flow rates and water quality. |
| *Sludge processing* | In medium and large WWTPs, apply co-digestion solutions with other wastes to improve the efficiency and stability of the anaerobic digestion process. |
| | In small WWTPs, dewatered sludge should be sent to centralized anaerobic digestion facilities located at large WWTPs. |
| *Size* | Centralization and construction of large WWTPs should be favored to exploit economies of scale provided that the overall system is economically and energetically viable. |
| | In situations requiring wastewater reuse, decentralized systems should be favored. |
| | Before aggregation of building new WWTPs a detailed analysis must be performed and extra costs and environmental impacts accounted. |
| | Ensure that automation, optimization, and cogeneration systems are implemented. |
| | Guarantee that the treatment process is stable as possible and the load factor is as close as possible to 100 %. |
| *Age* | Renovate old WWTPs when justified and replace obsolete equipment. |
| | Optimize process and resource management. |
| | Ensure adequate corrective and preventive maintenance of equipment. |
| | Periodically analyze the WWTP's performance to identify possible improvements and avoid efficiency decrease caused by growing age of the installation and equipment. |
| *Location* | Carry out regular (e.g., annual, biannual) energy benchmarking at national level to enhance the joint improvement of WWTPs. |
| | Replace existing pumps at WWTPs' inlet pumping stations with more energy-efficient models and install variable frequency drivers to adjust operation to dynamic conditions. |
| *Climate* | Maintain whenever possible the wastewater temperature in the biological process within the appropriate temperature range. |
| | Investigate in more detail the effect of the influent temperature in the biological process and determine the adequate temperature range to improve energy efficiency. |

**Figure 9 -** Summary of recommendations.

## 4.1 Process-related factors

The first group deals with process-related factors, such as load, dilution and polluted influent, secondary treatment technology, type of aeration, tertiary treatment and effluent discharge standard, and sludge processing.

### *4.1.1 Load factor*

WWTPs are typically designed to cope with extreme operating conditions to reduce the risk of failure (Torregrossa et al., 2019). Therefore, plants are oversized using safety coefficients to deal with variations in the volume and pollutant load of wastewater caused by new urban and industrial agglomerations (Guerrini et al., 2017) or seasonal variations of the population in tourist areas (Torregrossa et al., 2019).

Most European Union WWTPs are oversized. According to the European Environment Agency database, the average ratio between the served population equivalent and the design capacity —*i.e.*, average load factor (LF)— is near 80 % (Gandiglio et al., 2017). This oversizing is found in numerous facilities that operate nowhere near their design capacity for more than half of the year and low LF for extended periods (Castellet-Viciano et al., 2018b). Nonetheless, the largest WWTPs seem to operate closer to their design volume, with higher LF than small and medium-sized plants. Considering their long useful life and the unforeseen expansions of the served population, some WWTPs may become undersized and overloaded during their lifetime in rapidly developing countries (Lorenzo-Toja et al., 2015).

Various studies demonstrate a clear and intrinsic link between plant capacity utilization – the ratio between operational and design wastewater inflow – and energy consumption and efficiency. WWTPs with low LF have worse energy performance (Luo et al., 2019), and such low capacity utilization generates extra energy costs (Foladori et al., 2015). For example, a LF of 50 % has a SEC varying between 0.32 $kWh/m^3$ and 0.60 $kWh/m^3$, whereas this value ranged between 0.15 $kWh/m^3$ and 0.43 $kWh/m^3$for plants with a LF of 80 % (Silva and Rosa, 2015). Furthermore, when the LF approaches 100 %, the SEC decreases (Longo et al., 2018) and even continues to decrease in overloaded plants (Luo et al., 2019). Oppositely, low LF values are related to unsatisfactory SEC, such as the ones below 50 % (Silva et al., 2016), 40 % (Bodík and Kubaská, 2013), and 30 % (Tao and Chengwen, 2012). However, few studies found no relation between LF and energy consumption (Gómez et al., 2017; Lorenzo-Toja et al., 2015).

Fully loaded WWTPs were found to have a specific electricity consumption 12 % lower than overloaded and underloaded facilities (Niu et al., 2019), demonstrating that equipment and processes are more efficient when operating at design flow conditions. When load rates approach the optimum value, the operation of equipment and processes occurs more efficiently, as the treatment operation is more stable than with lower load rates. Moreover, the operation process has minimal changes in the amount of wastewater and concentration of pollutants, and conditions are more favorable for the growth of microorganisms and sludge (Luo et al., 2019).

In China, a rise in the load rate was followed by an increase in the average efficiency score (Figure 10), even for overloaded plants (Jiang et al., 2020). However, it is not specified whether these overloaded plants comply with the discharge standards. Thus, despite appearing to be more efficient, overloaded facilities deteriorate effluent quality and do not meet the discharge standards on certain occasions, thus impairing wastewater treatment performance. Therefore, the optimum LF should be around 100 % (Jiang et al., 2020) or at least 80 % (Luo et al., 2019).

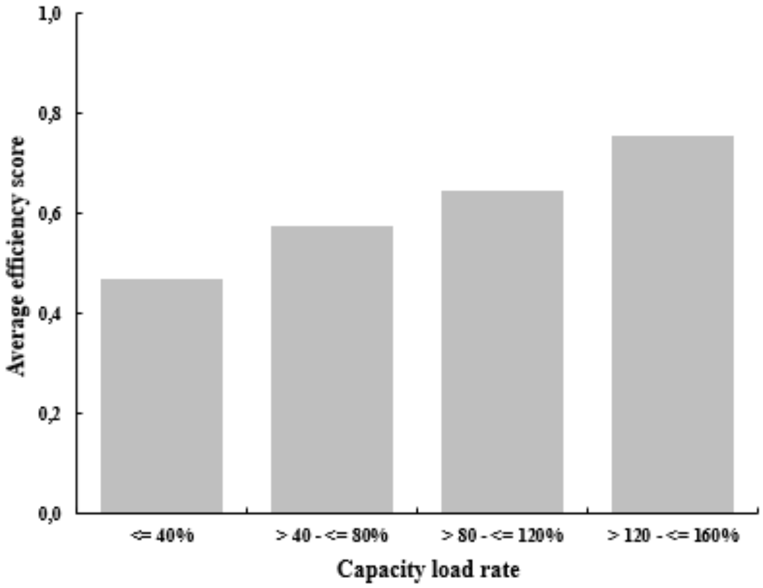


**Figure 10 -** Average efficiency score in China divided per capacity load rate. Data from Jiang et al. (2020).

Therefore, it is important to have adequate planning and design to prevent both over-sizing or under-sizing (Guerrini et al., 2017) and to implement flexibility and adaptability in the operational process to deal with seasonal variations of flows and pollutant load (*e.g.*, automation and inverters) (Foladori et al., 2015). Also, the construction of an efficient sewage collection pipe network is suggested to increase the LF, especially in undeveloped areas, rather than relying solely on centralizing plants to increase the amount of wastewater to treat (Niu et al., 2019). Ideally, an adequate LF should be between 80 % and 100 %.

#### *4.1.2 Dilution factor and polluted influent*

Gradual and drastic changes in the composition of the influent wastewater impact the WWTPs' energy consumption (Hernández-Sancho et al., 2011a), such as when the influent is highly polluted/loaded or conversely diluted. Typically, wastewater is diluted by groundwater infiltration in older networks and collected rainwater in combined or mixed sewage systems, leading to a low COD concentration in the influent and a higher volume treated (di Cicco et al., 2019). These diluted influents cause operational challenges and low organic removal rates (Lorenzo-Toja et al., 2015).

The reception of diluted wastewater with a low COD concentration makes the facility energy-intensive and equates to higher energy levels per population equivalent (kWh/PE). This increase in energy occurs due to the additional energy consumption required for hydraulically sized equipment, such as pumps and screens. A low value of energy consumption per volume of treated wastewater (kWh/$m^3$) originated from the dilution is misleading as this indicator is strongly influenced by dilution. It offers an apparent energy discount when, in fact, the installations require more energy per mass of pollutant removed, even for equal pollutant loads (Vaccari et al., 2018).

The negative impact of excessive dilution is also noticed by the increased energy required to remove a unit quantity of COD (di Cicco et al., 2019). Lower SEC values (kWh/kg COD rem) in Spain and Germany, when compared to France, result from the very low dilution factor levels (Figure 11) (Longo et al., 2016). When the influent COD concentration increases, this indicator decreases rapidly and maintains a stable decline for COD concentrations above 500 mg/L (Niu et al., 2019). Therefore, higher EE may be achieved by controlling the influent COD concentration by constructing new sewage pipelines and separating wastewater and rainwater in different pipe systems.

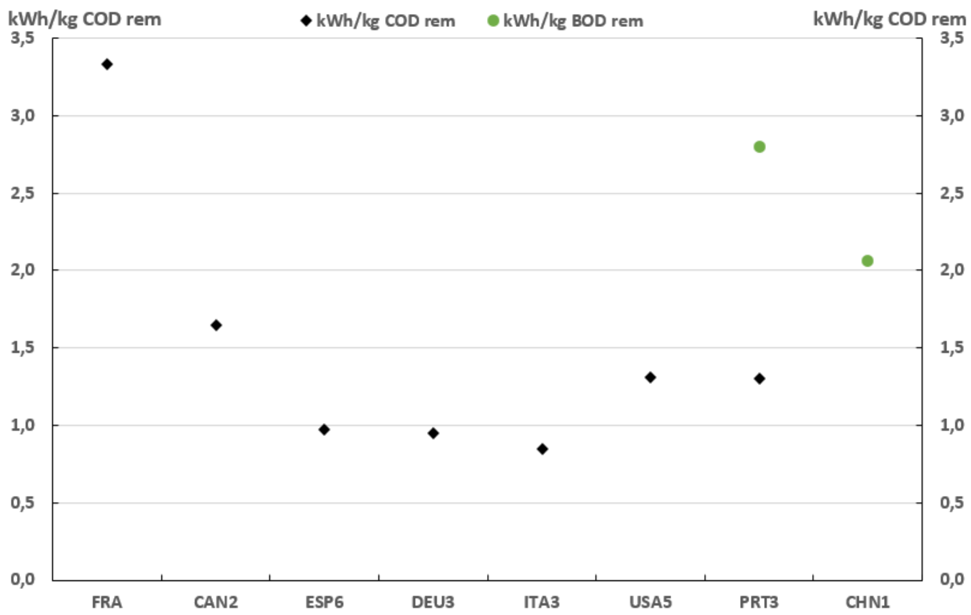


**Figure 11 -** Specific energy consumption of municipal WWTPs in several countries. Data from Longo et al. (2016b), Silva and Rosa (2015), and Yang et al. (2010).

In highly polluted and toxic influents, the SEC is affected negatively due to the larger concentrations of pollutants and toxins in the influent, such as nitrate and ammonia-nitrogen ($NH_3$-N), triggering an increase in aeration demand and sludge production rates, among others (Lorenzo-Toja et al., 2015). Also, wastewater rich in total nitrogen is perceived as one of the main reasons for higher energy consumption, which must be carefully controlled to meet the regulatory discharge levels (Yu et al., 2019).

As large plants require an equalization phase to buffer variations in pollutant load or inflow, equalization tanks are used to store and homogenize the wastewater, reduce operational problems, and increase the plant's efficiency. However, the construction of these tanks is not common practice in small plants, even though they may also suffer from this type of problem (Leu et al., 2009).

The sewage pipeline must be renovated and improved to reduce infiltrations to reduce wastewater dilution and increase influent pollutant concentration. Also, sewage systems must be separated to prevent mixing wastewater and rainwater, and equalization tanks must be applied to stabilize pollutant loads or flow rates.

#### 4.1.3 Secondary treatment technology

The type of secondary wastewater treatment is generally chosen to maximize treatment efficacy while considering EE, cost, and land availability (Luo et al., 2019). The impact of different secondary treatment types on energy performance has been widely studied. However, the conclusions are contradictory, despite general agreement on the influence of the type of treatment applied (Hernández-Sancho et al., 2011b; Molinos-Senante, 2018).

In Chile, plants using attached-growth processes – biodisk (BD) and TF – have higher EE values than those using suspend-growth processes – CAS, AL, and activated sludge with nutrient removal – (Molinos-Senante, 2018). Moreover, WWTPs using activated sludge with biological nutrient removal (BNR) were considered more energy-efficient despite having higher SEC than WWTPs with CAS processes. This EE is due to more efficient nutrient removal, which eventually compensates for increased energy consumption. In another study, BD technology proved to be, on average, the most efficient, partly due to the lack of artificial aeration (Hernández-Sancho et al., 2011b). Nonetheless, some studies did not find a significant performance impact from the treatment technology used (Molinos-Senante et al., 2014b; Zhou et al., 2013).

In the Chinese context, it is worth noting the lower SEC values exhibited by the CAS (0.269 kWh/m$^3$) and anaerobic-anoxic-oxic (A$^2$/O, 0.267 kWh/m$^3$) technologies, in addition to the already expected constructed wetlands (CW, 0.253 kWh/m$^3$) (Yang et al., 2010). Compared to CAS, A$^2$/O presents a lower value due to BNR systems achieving a better performance despite their higher treatment intensity. More efficient equipment can be implemented with better automation and regulation performance (Longo et al., 2016). A$^2$/O presenting moderate operating costs and electricity consumption with high levels of treatment efficiency are suggested by the literature to be adopted in economically disadvantageous regions (Jiang et al., 2020), in sensitive areas requiring simultaneous biological removal of phosphorous and nitrogen, or in cases when both the MBR fixed-asset investment cost is prohibitive and the extreme reduction of pollutant load is not required (Zeng et al., 2017).

Despite being more energy-intensive than most technologies, MBR presents a very high treatment efficiency and protects sensitive water bodies, making it the right choice in areas with high environmental requirements without constraints on investment cost and energy supply (Zeng et al., 2017), in small WWTPs receiving high-loaded influents and discharging to sensitive areas, or especially in situations requiring high effluent quality through advanced treatment, such as wastewater reuse (Molinos-Senante et al., 2012).

The variety of SEC values exhibited by many treatment technologies are largely due to different aeration requirements. As aeration is responsible for most of the plants' electricity consumption, obsolete systems should be replaced by more efficient models (turbo blowers) and variable frequency drivers (VFDs) (Liu et al., 2012). Furthermore, new automated control methods should also be implemented to adjust aeration to dynamic and real oxygen demand (Maktabifard et al., 2018).

Two suggestions can be made. First, $A^2/O$ systems should be implemented in sensitive areas requiring high removal rates of pollutants and nutrients (phosphorous and nitrogen), while MBR systems are recommended for situations with severe restrictions regarding effluent quality, such as wastewater reuse. Second, obsolete aeration systems should be replaced with more efficient models and implemented VFDs and new automated controls to adjust aeration to meet the oxygen demand.

#### *4.1.4 Aeration type*

WWTPs with diffusion systems were 28 % more energy-efficient than facilities with surface aeration systems (Hernández-Sancho et al., 2011a). It was found that although diffusers had a higher mean SEC (kWh/m$^3$) than turbines, they had a higher EE mean value and a higher percentage of efficient plants (14,3 % compared to 6,5 %) (Figure 12). Despite having higher energy consumption, diffusers provide better aeration and have a 15.6 % increase in EE over mechanical surface aerators (Guerrini et al., 2017). The differences between aeration systems were also found in other WWTPs with diffusion systems, which had a SEC of 26.5 kWh/PE, a much lower value than the 40 kWh/PE displayed by facilities with surface aeration systems (Mamais et al., 2015).

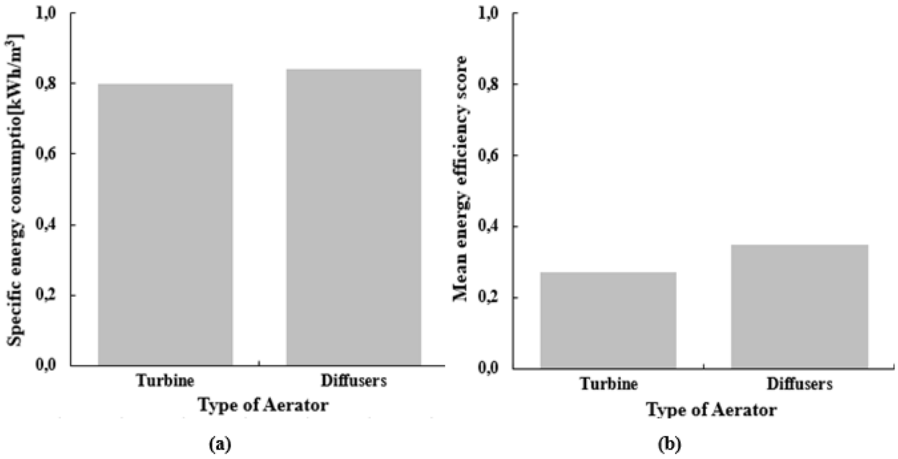


**Figure 12 –** Specific energy consumption (a) and mean energy efficiency score (b) of municipal WWTPs in Spain divided per type aeration system. Data from Hernández-Sancho et al. (2011a).

### *4.1.5 Tertiary treatment and effluent discharge standard*

Tertiary treatment or advanced treatments are typically applied to comply with more demanding regulations and discharge standards, which often imply removing common pollutants and other substances such as pathogens and pharmaceuticals (Zeng et al., 2017). Therefore, WWTPs with this treatment level are generally more energy-intensive than those which only have primary or secondary levels (Silva and Rosa, 2015). However, these plants consume energy more efficiently and have a lower unit energy consumption per volume treated or mass of pollutant removed (Silva and Rosa, 2015).

Still, in some cases, the improvement in effluent quality may not be sufficient to compensate for the higher energy requirements and GHG emissions (Zeng et al., 2017). Even though performance deteriorated, as the tertiary treatment capacity increased, the advantages over water reclamation, such as reducing pollution and resource conservation, were not fully accounted for in these analyses. Complementarily, applying this level of treatment can offer additional environmental benefits, such as reducing the use of chemicals (Liu et al., 2012).

The required effluent quality significantly influences energy consumption and efficiency (Mizuta and Shimada, 2010). Indeed, stricter discharge standards in different regions of China resulted in: (a) higher consumption of energy and chemicals and higher indirect emissions of gaseous pollutants, (b) higher release of gases – nitrous oxide ($N_2O$) and methane ($CH_4$) – at the biological treatment phase, and (c) higher emissions of GHG at the treatment stages and disposal of excess sludge (Li et al., 2020).

Nevertheless, stricter discharge standards reduce the potential for eutrophication. As such, their adoption should proliferate to protect natural water environments. Moreover, the implementation of measures to reduce the direct discharge of $N_2O$ and $CH_4$ gases is recommended, such as advanced treatment technologies (*e.g.*, simultaneous nitrification and denitrification (SND) and anaerobic ammonia oxidation (anammox)) to decrease $N_2O$ emissions during the biological treatment phase or combined cooling, heating, and power systems to reduce direct $CH_4$ emissions and generate energy and heat to be used in the process (Li et al., 2020).

Advanced treatment technologies have also been examined in the literature to replace conventional biological nitrogen removal processes (nitrification-denitrification) with high oxygen and organic carbon consumption levels. For example, SND, anammox, and nitrite shunt reduce energy consumption due to lower aeration requirements and reduce sludge production. In addition, SND has a smaller footprint, while anammox does not require an external carbon source and nitrite shunt is suitable for wastewater with low carbon to nitrogen ratio (Gurung et al., 2018; Lema and Suarez Martinez, 2017; Maktabifard et al., 2018).

Non-removed pathogens and other substances are eliminated by applying processes such as disinfection (chlorination, UV, ozonation) and filtration (microfiltration, nanofiltration) to attain a high-quality effluent in the tertiary treatment (Longo et al., 2016). Chlorination and UV are the most used disinfection techniques, but UV systems are becoming preferred due to health issues related to chlorination disinfection by-products (Plappally and Lienhard V, 2012).

A combination of technologies is recommended, such as chlorination/UV or Ozonation/UV, to improve the efficiency of the disinfection process (Pei et al., 2019). Moreover, WWTPs using UV systems may improve energy performance by adjusting the UV systems to the actual conditions. This improvement can be achieved by either reducing the number of lamps in operation or adopting a dose-pacing control, allowing the optimal adjustment of the UV lamp output based on flow rates and water quality (Daw and Hallett, 2012).

The suggestions are to (a) combine various disinfection techniques (*e.g.*, chlorination and UV) to improve the efficacy of the process and reduce by-products, and (b) adopt a dose-pacing control on UV lamps to adjust the output according to flow rates and water quality to increase EE.

#### *4.1.6 Sludge processing*

The impact of the technology used to treat sludge on the efficiency of a WWTP or the amount and destination of sludge generated has not been greatly investigated. Nonetheless, neither the method nor the destination of sludge deposition in agriculture has influenced the EE (Guerrini et al., 2017). There is also no evidence corroborating that efficiency was affected by the quantity of sludge generated as the cost of waste management only amounted to 7 % of the total costs (Molinos-Senante et al., 2014b).

The type of technology might be responsible for a difference in performance between facilities. For example, plants with aerobic sludge stabilization had a higher SEC (kWh/PE) than stations with mesophilic sludge digestion in municipal WWTPs in Austria (Figure 13) (Haslinger et al., 2016). The median SEC with aerobic sludge stabilization was 42 kWh/PE, while mesophilic sludge digestion, the value was 33 kWh/PE. Furthermore, these differences were also indirectly linked to size, as this affects the choice of the sludge stabilization process.

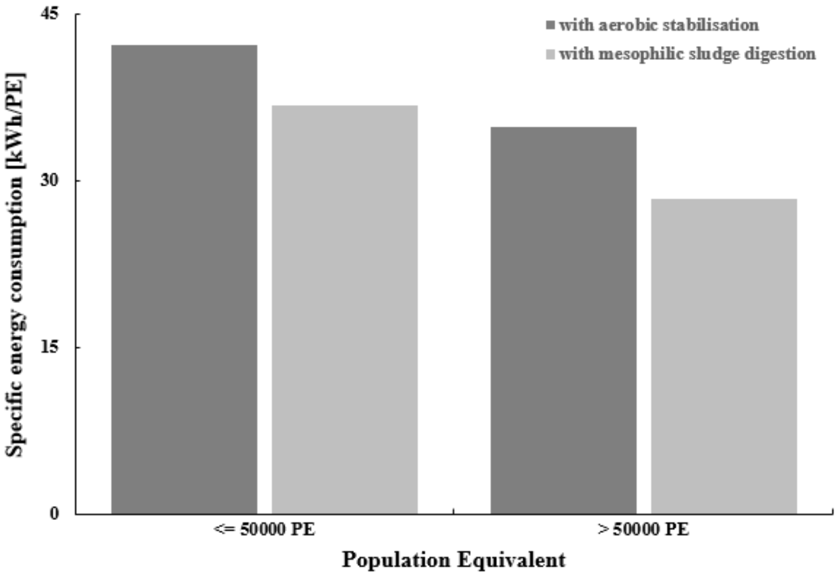


**Figure 13 -** Specific energy consumption of municipal WWTPs in Austria divided per population equivalent and sludge stabilization type. Data from Haslinger et al. (2016).

Comparing aerobic digestion and mechanical dewatering, anaerobic digestion (AD) was the most economically and environmentally efficient technology (Molinos-Senante et al., 2014a). Additionally, the AD enables biogas production, which can produce heat and electricity in the plant, increasing its energy self-sufficiency. Thus, combining energy self-production with the simultaneous application of energy consumption rationalization measures makes it possible to achieve high levels of self-sufficiency or even energy neutrality (Gu et al., 2017), as is the case in some WTTPs (Maktabifard et al., 2018).

The co-digestion of sludge with other wastes is currently considered the most promising solution to increase biogas production as it improves the efficiency and stability of the AD process by adding a suitable substrate. However, biogas production is not profitable for small WWTPs due to the costs of constructing and maintaining anaerobic digesters and is only advisable for plants with daily flows above 20,000 m3/day (Maslón et al., 2020).

Therefore, two recommendations can be drawn. First, enhance the AD of sludge and biogas production in medium and large WWTPs to increase self-sufficiency and apply co-digestion solutions with other wastes to increase the efficiency and stability of the processes. Second, to send dewatered sludge from small WWTPs to centralized AD facilities located in large WWTPs where biogas production is feasible.

## 4.2 Physical and context-related factors

The second group deals with the physical and context factors, namely facility size, age, location, and climate.

### *4.2.1 Size*

The dimension of the municipality affects the design of the WWTPs, requiring proportional operation processes which correlate with the size of the facilities (Hernández-Chover et al., 2018). Therefore, some correlation between the size of the WWTP and the energy consumed or its EE is expected. Although large WWTPs present high energy consumption levels, several studies show these to be more energy-efficient and present lower SEC values, expressed both per volume of wastewater treated or population equivalent served. For example, as the size of the facility increase, so did the COD removal rate, ammonia nitrogen removal rate, and reclaimed water yield, while average operating costs, energy consumption, and labor decreased (Jiang et al., 2020). To further exemplify, a 1 % size increase only had a 0.91 % rise in total energy consumption (Longo et al., 2017).

In Figure 14, Austrian WWTPs were grouped according to their size, expressed by the design capacity in PE (Haslinger et al., 2016), and Italian WWTPs using the PE served (Vaccari et al., 2018). Results indicate there is a relationship between size and energy consumption and that large WWTPs present, on average, lower SEC values.

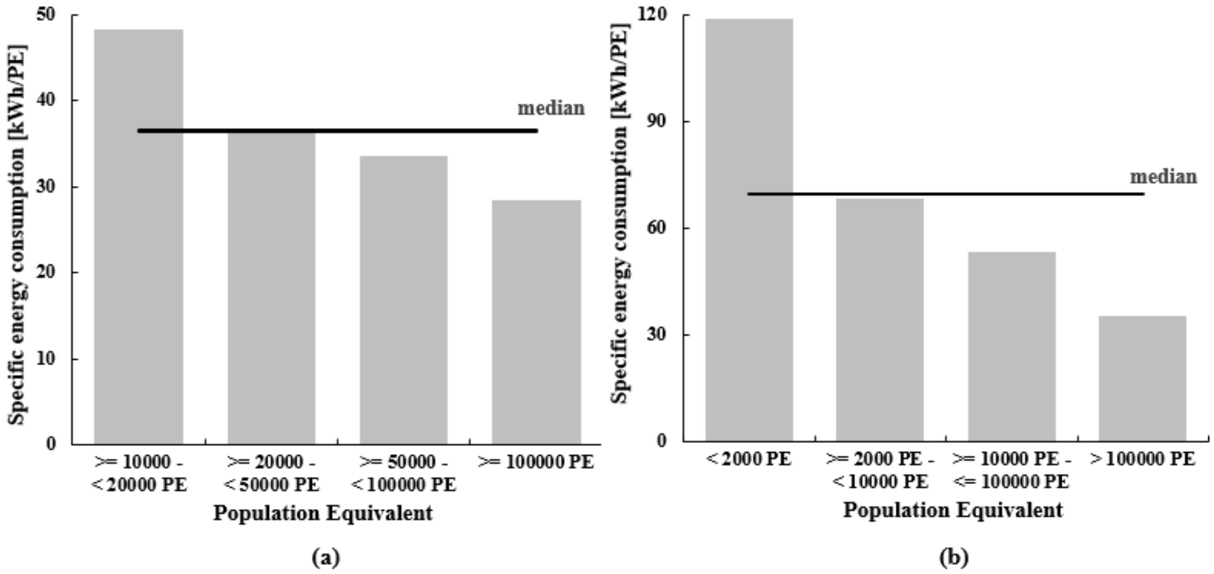


**Figure 14 -** Specific energy consumption of municipal WWTPs in Austria (a) and Italy (b). Data from Haslinger et al. (2016), and Vaccari et al. (2018).

Economies of scale affect efficiency, especially operating costs, such as personnel, waste management, maintenance, and others (Gómez et al., 2017). For example, when comparing the largest WWTPs (> 100,000 $m^3$/day) against the smallest (< 10,000 $m^3$/day) in China, large plants had a better performance with low values for SEC and specific fixed costs, thus demonstrating the effect of economies of scale (Zeng et al., 2017).

In Chile, four out of five types of secondary treatment (CAS, EA, TF, BD) revealed the annual flow rate as the largest contributor to EI (Molinos-Senante et al., 2018). However, no relation between capacity and EI for the remaining technology (AL) was found. In Spain, the mean SEC (kWh/$m^3$) decreased as the size increased, whereas the mean EE increased with size (Figure 15) (Hernández-Sancho et al., 2011a). Furthermore, only 3 % of the small plants were considered energy-efficient, as for large WWTPs, the value was substantially higher (32 %).

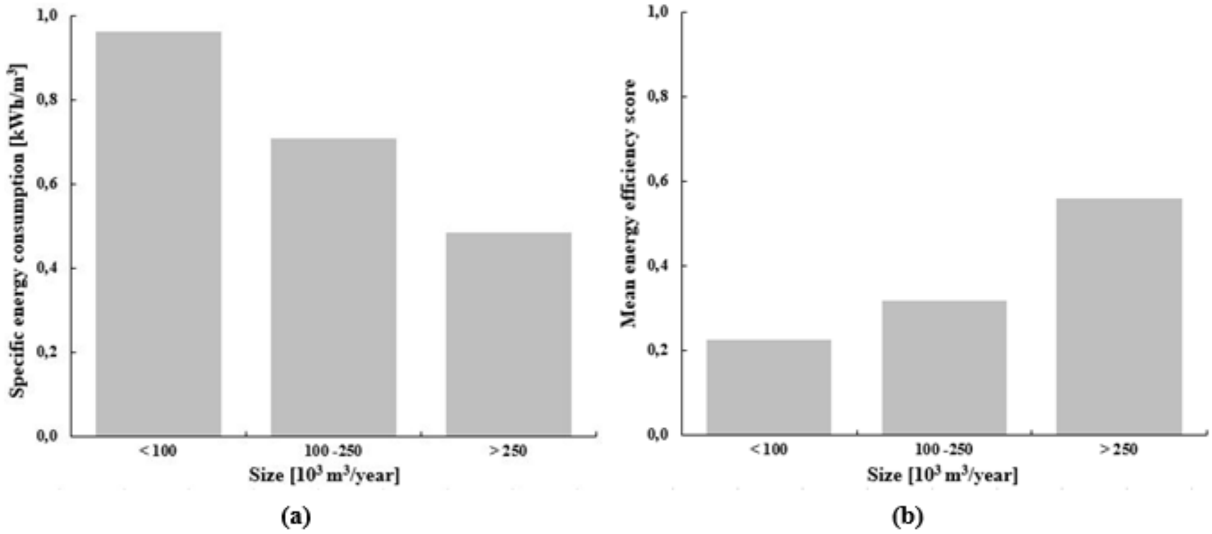


**Figure 15 -** Specific energy consumption *(a)* and mean energy efficiency score *(b)* in Spain divided according to their size ($10^3$ $m^3$/year). Data from Hernández-Sancho et al. (2011a).

Large WWTPs have, on average, higher efficiency levels and lower SEC levels due to several advantageous factors. These operate closer to the design volume and optimal operating point and have a higher LF (Castellet-Viciano et al., 2018b). Lower LF in small and medium WWTPs may be attributed to being typically located in small urban settlements with small populations and incomplete wastewater collection systems (Luo et al., 2019).

Low SEC is also attributed to the modulation of the treatment processes, cogeneration systems, and economies of scale (Trapote et al., 2014). Indeed, larger WWTPs typically employ automation and optimization tools, such as VFDs in aeration and pumping, and operate under more stable operating conditions than small plants, which typically undergo particularly energy-intensive transitional periods (Christoforidou et al., 2020).

Small WWTPs are generally projected with simplified configurations and are not usually equipped with systems to control the process or regulate volumetric and organic load fluctuations. Due to their size, an investment in online monitoring and process control systems is not justifiable, leading to reduced data availability for process trend monitoring in small plants (Sabia et al., 2020).

In Valencia, an analysis of extended aeration (EA) indicated that treating a large PE requires lower unit economic costs to treat wastewater, showing that a minimum of 3000 PE was needed to be efficient. In this sense, installing one plant per small agglomeration is not the best option for reducing operational costs. Instead, the aggregation of small urban agglomerations is suggested to gain dimension and reach an efficiency threshold. Therefore, the centralization of wastewater systems is generally advocated to obtain higher performances and EE levels and low operational costs (Hernández-Chover et al., 2018).

Literature does not provide a consensus on the optimal plant size. Moreover, there is an ongoing dispute about whether to build several small plants or just a larger one serving multiple urban settlements (Hernández-Chover et al., 2018). The reason is mostly associated with the sewage systems as additional costs for conveyance and pumping along the network should be considered (Vaccari et al., 2018).

The existence of statistically significant diseconomies of scale in larger water utilities has been reported in the literature, due to significant weight of the conveyance cost (Carvalho et al., 2012). The aggregation of two utilities may lead to conveyance and operation costs not being compensated by savings attained with a larger treatment facility. In addition to the economic costs associated with effluent transport, environmental costs (*e.g.*, environmental impact of large WWTPs and associated pipelines, possible local reuse) should not be neglected (Trapote et al., 2014).

The main recommendation is that a WWTP should be as large as possible to take advantage of economies of scale, provided that the overall system (plant and associated collection network) is economically, energetically, and environmentally viable. In order to achieve this, before aggregating or building new WWTPs, the following must be confirmed (a) a detailed analysis of all the advantages and disadvantages is performed, (b) the LF of the plant will be as close as possible to 100 %, (c) the treatment process will be as stable as possible, (d) automation, optimization, and cogeneration systems are implemented, and (e) the extra costs and environmental impacts of the network (*e.g.*, pumping, construction) are dully accounted for and are lower than the efficiency gains of the plant.

Given the climate change scenario of water scarcity and uneven water distribution, wastewater reuse is seen as a viable water source alternative in several regions of the planet, particularly those applying desalination. Therefore, in situations that require wastewater reuse, the adoption of decentralized systems appears to be the best solution because, in the case of large-scale WWTPs, the supply of reclaimed wastewater to consumers becomes quite expensive in materials and energy. Therefore, in these cases, the optimal plant's size should be determined by conducting energy and economic assessment of the wastewater treatment and reuse options incorporating the benefits such as reduced water losses and conveyance costs compared to centralized long-distance distribution systems, but also the costs of conveying the reclaimed wastewater to consumers and of building new distribution networks.

#### *4.2.2 Age*

The findings on the impacts of age (or time since renovation) are somewhat contradictory. Some studies found that age correlates both positively and negatively with efficiency (Castellet-Viciano et al., 2018a; Molinos-Senante et al., 2018), whereas several others dismiss the existence of any evidence (Gómez et al., 2017; Hernández-Sancho et al., 2011a). Nonetheless, as diffusers get blocked (the most common type of aeration systems for CAS and EA technologies), the EE deteriorates and EI increases (Molinos-Senante et al., 2018). For these two types of technologies, the wear and tear of equipment negatively affect energy costs (Castellet-Viciano et al., 2018a). In turn, the EI is not statistically affected by age when using aerated lagoons, TF, and BD (Molinos-Senante et al., 2018). For example, the lowest efficiency levels were attained by older plants when the age of the plants varied between 3 and 30 (Fuentes et al., 2015). However, the sample included plants that had already exceeded their service life and, in some cases, were technologically obsolete. In this case, older plants exhibited higher efficiency scores than newer ones (Gómez et al., 2017).

Proper maintenance, optimized process operation, and good resource management are critical factors for increasing efficiency and productivity (Hernández-Sancho et al., 2011a). When other factors are fixed, energy consumption falls for new WWTPs. For example, in plants with an age gap of 15 years, the EE difference is 20 %. When older plants are renovated (built before 2000), a 3.2 % reduction in energy consumption is obtained (Niu et al., 2019).

Therefore, some suggestions can be made: (a) renovate old WWTPs, (b) replace obsolete equipment, (c) optimize process and resource management, (d) ensure adequate corrective and preventive maintenance of equipment, and (e) periodically monitor the energy performance of the plants.

#### *4.2.3 Location*

Statistical differences between WWTPs in different countries were found. For example, on average, plants in Spain and Italy are less efficient than in Switzerland (Longo et al., 2018). In Austria, WWTPs consumed about 45 % less electricity than those in Sweden (Gu et al., 2017). These differences are justified with benchmarking processes and EE measures implemented in the previous years by those countries.

These examples highlight the importance of carrying out energy benchmarking on national levels. This benchmarking allows plants to identify potential energy performance inefficiencies and implement appropriate improvement measures. For instance, the inspection of half of the WWTPs in Austria led to improving EE, and one-third of the WWTPs increased more than 10 % (Haas et al., 2018). Nonetheless, only a few countries have successfully implemented energy benchmarking programs.

Electricity tariffs and existing treatment types resulting from economic and environmental constraints (Longo et al., 2016) and different target effluent quality standards and implemented strategies (Gu et al., 2017) have been highlighted as factors leading to differences among countries. Indeed, the lower effluent quality in China is typically highlighted as one of the main causes for low average energy consumption in wastewater treatment (Luo et al., 2019).

As mentioned earlier in Section 4.1.5., more stringent effluent quality requirements entail a higher pollutant removal rate and the application of advanced treatment and disinfection processes with higher energy and chemical demands than the conventional ones. Greater removal of oxygen-consuming pollutants (*e.g.*, $BOD_5$ and ammonia-nitrogen) requires more energy to provide enough oxygen for the same volume of treated wastewater. However, by evaluating the energy consumption required for a given amount of pollutant removed, it is noticeable that greater removal leads to lower energy consumption per amount of pollutant removed (Luo et al., 2019).

Another factor is the national industrial structure, as industrial wastewater contains higher concentrations of pollutants that are more difficult to degrade, requiring more energy consumption and affecting plant efficiency (Wang et al., 2016). This factor originates an energy efficiency bias between WWTPs in manufacturing-based countries and those in resource-based countries, where the influent is more biodegradable than the former.

Geographical and morphological characteristics of the area may also justify differences in energy consumption. For example, one study concluded that WWTPs built in hilly areas (200 m to 800 m above sea level) had lower energy consumption than those constructed in plains (< 200 m) or plateaus (> 800 m) (Niu et al., 2019). The location of the WWTP may, for example, lead to the existence of pumping stations at the beginning of the plant, which may account for 5 % to 18.9 % of the plant's overall energy consumption (Siatou et al., 2020). In these cases, either VFDs must be installed to adjust pump operation or current pumps replaced with new and more energy-efficient technologies (Liu et al., 2012).

Therefore, two suggestions are made. The first is to carry out regular (*e.g.*, annual, biannual) energy benchmarking studies on the national level to enhance the joint improvement of WWTPs. Second, replace existing pumps at the inlet pumping stations with more energy-efficient models and install VFDs to adjust pump operation to dynamic conditions.

#### *4.2.4 Climate*

Climate is another factor referenced in the literature. In Spain, for example, areas with milder average temperatures and greater rainfall values indicated better performance when combining DEA and life cycle assessment (Lorenzo-Toja et al., 2015). Contrarily, the worst performance is found in areas with the driest climate (lowest rainfall and highest average temperature).

Extreme weather events will become more recurrent, and temperature amplitude will be greater and more common due to climate change, thus, influencing the treatment process and energy consumption. For this reason, ambient temperature is considered a factor, as high and low temperatures of the wastewater will increase energy consumption (Yu et al., 2019). The temperature rise originates a growth in the biological activity, both in the rate of substrate absorption and in endogenous respiration, but this leads to a severe decrease in oxygen solubility and a consequent rise in the energy requirement for aeration (Longo et al., 2018).

The energy demand for aeration is more significant than the energy reduction driven by the growth of biological activity since there is a positive correlation between ambient temperature and energy consumption (Longo et al., 2018), thus demonstrating a negative relationship between temperature and EE. On the other hand, the high temperature of the biological process during summer (19.4 °C) led to a SEC approximately 5.6 % lower than in the winter (11.5 °C) and lower energy consumption from aeration. Thus, the EE may be increased by stimulating the microbial activity of activated sludge (Bartha et al., 2020).

In countries with high solar exposure, the implementation of photovoltaic systems may positively contribute to meeting the energy demand or even be used as a profitable resource. However, it should be emphasized that the supply side options (local generation) should not hamper the EE approach.

Some recommendations may be made, such as to carry out more detailed studies on the effects of influent temperature in the biological process, to determine an adequate temperature range that improves the EE of the process and maintain the wastewater temperature within that range, and lastly, to implement measures that help control the wastewater temperature.

# 5 Conclusions

This paper analyzes factors reported in the literature that influence the EE in wastewater treatment and provides recommendations to reduce energy consumption, environmental impacts, and use of other resources. The main conclusions of this study are:

- Despite existing a wide range of studies, most lack a proper in-depth analysis and improvement actions, thus limiting applicability and usefulness to facility managers, designers, and decision makers.
- Annual flow rate is considered an important factor in specific energy consumption because it influences plant performance through economies of scale and its relationship with the load factor and the quantity of pollutants and nutrients removed.

Besides, the influent's characteristics and pollutant concentration are also considered to have an important contribution to energy consumption as they influence the treatment process and aeration requirements.

- Since a large part of the energy consumption belongs to aeration, the implementation of automation, optimization, and advanced control systems that allow the adjustment of the plant’s load factor, and especially of aeration, to the real requirements of the treatment process becomes preponderant and can significantly reduce energy consumption.
- Large centralized WWTPs should be built to take advantage of economies of scale, providing that the overall system (including the network) is economically, energetically, and environmentally viable. On the contrary, decentralized systems are considered the best solution when wastewater reuse is required. Nonetheless, a detailed analysis of all the advantages and disadvantages should be carried out before the aggregation or construction of new plants.
- Anaerobic-anoxic-oxic systems are a good option in sensitive areas requiring high pollutant and nutrient removal rates. Membrane bioreactors are advised for situations with even more stringent effluent quality requirements, such as wastewater reuse.
- Proper corrective and preventive equipment maintenance should be ensured. The replacement of obsolete equipment and processes with more efficient solutions be guaranteed. Renovation of the sewage collection pipe network and implementation of equalization tanks are also recommended.
- Anaerobic digestion of sludge and biogas production should be improved by applying co-digestion solutions with other wastes. In addition, the energy self-sufficiency of the WWTPs should be improved through renewable energy sources (*e.g.*, photovoltaics).

As climate change and the deterioration of water resources continue to aggravate, it is essential to promote stringent wastewater treatment requirements and the transition to an era of energy and carbon neutrality. To this end, the implementation of energy-saving measures, energy production from biogas and renewable sources, and advanced automation and control strategies should be enhanced without neglecting treatment efficiency.

Thus, while making discharge standards more demanding, regulators should also leverage regular energy benchmarking and audit procedures. Moreover, since energy consumption and performance are influenced by several contextual factors, regulators should also complement the use of a single KPI with holist and multi-criteria approaches, such as data envelopment analysis or ENERWATER project's methodology and its aggregated indicator. Complementarily, utilities should promote technological innovation and turn to energy service companies to seek expertise and financial solutions.

# Acknowledgments

The presented work is framed under the *Energy for Sustainability Initiative* of the University of Coimbra.

The authors are grateful to Anabel Reis Alves for proofreading the manuscript.

Funding: Bruno J. Cardoso acknowledges the support provided by the Portuguese Foundation for Science and Technology (FCT) under Doctoral grant PD/BD/142812/2018.

# Credit Author Statement

**Bruno J. Cardoso:** Conceptualization; Methodology; Visualization, Investigation; Data curation, Writing – Original Draft Preparation

**Eugénio Rodrigues:** Methodology, Visualization; Writing – Review and Editing

**Adélio R. Gaspar:** Writing – Review and Editing; Supervision

**Álvaro Gomes:** Writing – Review and Editing; Supervision

# References


Ananda, J., 2018. Productivity implications of the water-energy-emissions nexus: An empirical analysis of the drinking water and wastewater sector. J. Clean. Prod. 196, 1097–1105. doi:10.1016/j.jclepro.2018.06.145

Aymerich, I., Rieger, L., Sobhani, R., Rosso, D., Corominas, L., 2015. The difference between energy consumption and energy cost: Modelling energy tariff structures for water resource recovery facilities. Water Res. 81, 113–123. doi:10.1016/j.watres.2015.04.033

Bartha, C., Jipa, M., Ignat, D.M., Ókos, A., Lingvay, I., 2020. The Energy Efficiency of Domestic Wastewater Treatment Processes: Case Analysis. Electroteh. Electron. Autom. 68, 30–36. doi:10.46904/eea.20.68.4.1108004

Belloir, C., Stanford, C., Soares, A., 2015. Energy benchmarking in wastewater treatment plants: The importance of site operation and layout. Environ. Technol. (United Kingdom) 36, 260–269. doi:10.1080/09593330.2014.951403

Bodík, I., Kubaská, M., 2013. Energy and sustainability of operation of a wastewater treatment plant. Environ. Prot. Eng. 39, 15–24. doi:10.5277/EPE130202

Cardoso, B.J., Gaspar, A.R., Gomes, Á., 2020. Energy Audits and Energy Efficiency in Small Wastewater Treatment Plants: A Case Study, in: Monteiro, J., João Silva, A., Mortal, A., Aníbal, J., da Silva, M., Oliveira, M., Sousa, N. (Eds.), INCREaSE 2019. Springer International Publishing, Cham, pp. 766–777. doi:10.1007/978-3-030-30938-1_59

Carvalho, P., Marques, R.C., Berg, S., 2012. A meta-regression analysis of benchmarking studies on water utilities market structure. Util. Policy 21, 40–49. doi:10.1016/j.jup.2011.12.005

Castellet-Viciano, L., Hernández-Chover, V., Hernández-Sancho, F., 2018a. Modelling the energy costs of the wastewater treatment process: The influence of the aging factor. Sci. Total Environ. 625, 363–372. doi:10.1016/j.scitotenv.2017.12.304

Castellet-Viciano, L., Torregrossa, D., Hernández-Sancho, F., 2018b. The relevance of the design characteristics to the optimal operation of wastewater treatment plants: Energy cost assessment. J. Environ. Manage. 222, 275–283. doi:10.1016/j.jenvman.2018.05.049

Castellet, L., Molinos-Senante, M., 2016. Efficiency assessment of wastewater treatment plants: A data envelopment analysis approach integrating technical, economic, and environmental issues. J. Environ. Manage. 167, 160–166. doi:10.1016/j.jenvman.2015.11.037

Chatzisymeon, E., 2015. Reducing the energy demands of wastewater treatment through energy recovery, in: Stamatelatou, K., Tsagarakis, K. (Eds.), Sewage Treatment Plants: Economic Evaluation of Innovative Technologies for Energy Efficiency. IWA Publishing, London, pp. 3–14, ISBN: 9781780405018.

Christoforidou, P., Bariamis, G., Iosifidou, M., Nikolaidou, E., Samaras, P., 2020. Energy Benchmarking and Optimization of Wastewater Treatment Plants in Greece. Environ. Sci. Proc. 2, 36. doi:10.3390/environsciproc2020002036

Copeland, C., Carter, N.T., 2017. Energy-Water Nexus : The Water Sector ' s Energy Use. Congressional Research Service, Washington, DC, USA.

Daw, J., Hallett, K., 2012. Energy Efficiency Strategies for Municipal Wastewater Treatment Facilities Energy Efficiency Strategies for Municipal Wastewater Treatment Facilities. doi:NREL/TP-7A30-53341

de Haas, D., Foley, J., Marshall, B., Dancey, M., Vierboom, S., Bartle-Smith, J., 2015. Benchmarking wastewater treatment plant energy use in Australia, in: Proceedings of the Australian Water Association Ozwater'15 Conference. Adelaide, Australia.

di Cicco, M.R., Spagnuolo, A., Masiello, A., Vetromile, C., Nappa, M., Corbo, G., Lubritto, C., 2019. Assessing energy performance and critical issues of a large wastewater treatment plant through full-scale data benchmarking. Water Sci. Technol. 80, 1421–1429. doi:10.2166/wst.2019.392

Di Fraia, S., Massarotti, N., Vanoli, L., 2018. A novel energy assessment of urban wastewater treatment plants. Energy Convers. Manag. 163, 304–313. doi:10.1016/j.enconman.2018.02.058

Foladori, P., Vaccari, M., Vitali, F., 2015. Energy audit in small wastewater treatment plants: Methodology, energy consumption indicators, and lessons learned. Water Sci. Technol. 72, 1007–1015. doi:10.2166/wst.2015.306

Fuentes, R., Torregrosa, T., Ballenilla, E., 2015. Conditional Order-m Efficiency of Wastewater Treatment Plants: The Role of Environmental Factors. Water 7, 5503–5524. doi:10.3390/w7105503

Gandiglio, M., Lanzini, A., Soto, A., Leone, P., Santarelli, M., 2017. Enhancing the energy efficiency of wastewater treatment plants through co-digestion and fuel cell systems. Front. Environ. Sci. 5, 1–21. doi:10.3389/fenvs.2017.00070

Gómez, T., Gémar, G., Molinos-Senante, M., Sala-Garrido, R., Caballero, R., 2017. Assessing the efficiency of wastewater treatment plants: A double-bootstrap approach. J. Clean. Prod. 164, 315–324. doi:10.1016/j.jclepro.2017.06.198

Gu, Y., Li, Y., Li, X., Luo, P., Wang, H., Robinson, Z.P., Wang, X., Wu, J., Li, F., 2017. The feasibility and challenges of energy self-sufficient wastewater treatment plants. Appl. Energy 204, 1463–1475. doi:10.1016/j.apenergy.2017.02.069

Guerrini, A., Romano, G., Indipendenza, A., 2017. Energy efficiency drivers in wastewater treatment plants: A double bootstrap DEA analysis. Sustain. 9, 1–13. doi:10.3390/su9071126

Gurung, K., Tang, W.Z., Sillanpää, M., 2018. Unit Energy Consumption as Benchmark to Select Energy Positive Retrofitting Strategies for Finnish Wastewater Treatment Plants (WWTPs): a Case Study of Mikkeli WWTP. Environ. Process. 5, 931. doi:10.1007/s40710-018-0340-5

Haas, D. de, Appleby, G., Charakos, G., Dinesh, N., 2018. Benchmarking Energy Use for Wastewater Treatment Plants - a summary of the 2015-16 benchmarking study. Water e-Journal 3, 1–26. doi:10.21139/wej.2018.023

Hao, X., Liu, R., Huang, X., 2015. Evaluation of the potential for operating carbon neutral WWTPs in China. Water Res. 87, 424–431. doi:10.1016/j.watres.2015.05.050

Haslinger, J., Krampe, J., Lindtner, S., 2016. Operating costs and energy demand of wastewater treatment plants in Austria: Benchmarking results of the last 10 years. Water Sci. Technol. 74, 2620–2626. doi:10.2166/wst.2016.390

Henriques, J., Catarino, J., 2017. Sustainable value – An energy efficiency indicator in wastewater treatment plants. J. Clean. Prod. 142, 323–330. doi:10.1016/j.jclepro.2016.03.173

Hernández-Chover, V., Bellver-Domingo, Á., Hernández-Sancho, F., 2018. Efficiency of wastewater treatment facilities: The influence of scale economies. J. Environ. Manage. 228, 77–84. doi:10.1016/j.jenvman.2018.09.014

Hernández-Sancho, F., Molinos-Senante, M., Sala-Garrido, R., 2011a. Energy efficiency in Spanish wastewater treatment plants: A non-radial DEA approach. Sci. Total Environ. 409, 2693–2699. doi:10.1016/j.scitotenv.2011.04.018

Hernández-Sancho, F., Molinos-Senante, M., Sala-Garrido, R., 2011b. Techno-economical efficiency and productivity change of wastewater treatment plants: The role of internal and external factors. J. Environ. Monit. 13, 3448–3459. doi:10.1039/c1em10388a

IEA, 2016. Water Energy Nexus- Excerpt from the World Energy Outlook 2016, International Energy Agency, ISBN 978-92-64-26494-6.

Jiang, H., Hua, M., Zhang, J., Cheng, P., Ye, Z., Huang, M., Jin, Q., 2020. Sustainability efficiency assessment of wastewater treatment plants in China: A data envelopment analysis based on cluster benchmarking. J. Clean. Prod. 244, 118729. doi:10.1016/j.jclepro.2019.118729

Krzeminski, P., Van Der Graaf, J.H.J.M., Van Lier, J.B., 2012. Specific energy consumption of membrane bioreactor (MBR) for sewage treatment. Water Sci. Technol. 65, 380–392. doi:10.2166/wst.2012.861

Lema, J., Suarez Martinez, S., 2017. Innovative Wastewater Treatment & Resource Recovery Technologies.

Leu, S.-Y., Rosso, D., Larson, L.E., Stenstrom, M.K., 2009. Real-Time Aeration Efficiency Monitoring in the Activated Sludge Process and Methods to Reduce Energy Consumption and Operating Costs. Water Environ. Res. 81, 2471–2481. doi:10.2175/106143009x425906

Li, Y., Xu, Y., Fu, Z., Li, W., Zheng, L., Li, M., 2020. Assessment of energy use and environmental impacts of wastewater treatment plants in the entire life cycle: A system meta-analysis. Environ. Res. doi:10.1016/j.envres.2020.110458

Lindtner, S., Schaar, H., Kroiss, H., 2008. Benchmarking of large municipal wastewater treatment plants treating over 100,000 PE in Austria. Water Sci. Technol. 57, 1487–1493. doi:10.2166/wst.2008.214

Liu, F., Ouedraogo, A., Manghee, S., Danilenko, A., 2012. A PRIMER ON ENERGY EFFICIENCY FOR MUNICIPAL WATER AND WASTEWATER UTILITIES, Energy Sector Management Assistance Program, Technical Report 001/12, World Bank. Washington, DC.

Longo, S., D'Antoni, B.M., Bongards, M., Chaparro, A., Cronrath, A., Fatone, F., Lema, J.M., Mauricio-Iglesias, M., Soares, A., Hospido, A., 2016. Monitoring and diagnosis of energy consumption in wastewater treatment plants. A state of the art and proposals for improvement. Appl. Energy 179, 1251–1268. doi:10.1016/j.apenergy.2016.07.043

Longo, S., Hospido, A., Lema, J.M., Mauricio-Iglesias, M., 2018. A systematic methodology for the robust quantification of energy efficiency at wastewater treatment plants featuring Data Envelopment Analysis. Water Res. 141, 317–328. doi:10.1016/j.watres.2018.04.067

Longo, S., Lema, J.M., Mauricio-Iglesias, M., Hospido, A., 2017. Measuring Energy Demand and Efficiency at WWTPs: An Econometric Approach, in: Frontiers in Wastewater Treatment and Modelling, Lecture Notes in Civil Engineering 4. Springer International Publishing. doi:10.1007/978-3-319-58421-8

Longo, S., Mauricio-Iglesias, M., Soares, A., Campo, P., Fatone, F., Eusebi, A.L., Akkersdijk, E., Stefani, L., Hospido, A., 2019. ENERWATER – A standard method for assessing and improving the energy efficiency of wastewater treatment plants. Appl. Energy 242, 897–910. doi:10.1016/j.apenergy.2019.03.130

Lorenzo-Toja, Y., Vázquez-Rowe, I., Chenel, S., Marín-Navarro, D., Moreira, M.T., Feijoo, G., 2015. Eco-efficiency analysis of Spanish WWTPs using the LCA+DEA method. Water Res. 68, 651–666. doi:10.1016/j.watres.2014.10.040

Luo, L., Dzakpasu, M., Yang, B., Zhang, W., Yang, Y., Wang, X.C., 2019. A novel index of total oxygen demand for the comprehensive evaluation of energy consumption for urban wastewater treatment. Appl. Energy 236, 253–261. doi:10.1016/j.apenergy.2018.11.101

Maktabifard, M., Zaborowska, E., Makinia, J., 2018. Achieving energy neutrality in wastewater treatment plants through energy savings and enhancing renewable energy production. Rev. Environ. Sci. Biotechnol. 17, 655–689. doi:10.1007/s11157-018-9478-x

Mamais, D., Noutsopoulos, C., Dimopoulou, A., Stasinakis, A., Lekkas, T.D., 2015. Wastewater treatment process impact on energy savings and greenhouse gas emissions. Water Sci. Technol. 71, 303–308. doi:10.2166/wst.2014.521

Marner, S.T., Schröter, D., Jardin, N., 2016. Towards energy neutrality by optimising the activated sludge process of the WWTP Bochum-Ölbachtal. Water Sci. Technol. 73, 3057–3063. doi:10.2166/wst.2016.142

Maslón, A., Czarnota, J., Szaja, A., Szulzyk-Cieplak, J., Lagód, G., 2020. The Enhancement of Energy Efficiency in a Wastewater Treatment Plant through Sustainable Biogas Use: Case Study from Poland. Energies 13. doi:10.3390/en13226056

Mizuta, K., Shimada, M., 2010. Benchmarking energy consumption in municipal wastewater treatment plants in Japan. Water Sci. Technol. 62, 2256–2262. doi:10.2166/wst.2010.510

Molinos-Senante, M., 2018. Comparative energy efficiency of wastewater treatment technologies: a synthetic index approach. Clean Technol. Environ. Policy 20, 1819–1834. doi:10.1007/s10098-018-1575-6

Molinos-Senante, M., Garrido-Baserba, M., Reif, R., Hernández-Sancho, F., Poch, M., 2012. Assessment of wastewater treatment plant design for small communities: Environmental and economic aspects. Sci. Total Environ. 427–428, 11–18. doi:10.1016/j.scitotenv.2012.04.023

Molinos-Senante, M., Hernández-Sancho, F., Mocholí-Arce, M., Sala-Garrido, R., 2014a. Economic and environmental performance of wastewater treatment plants: Potential reductions in greenhouse gases emissions. Resour. Energy Econ. 38, 125–140. doi:10.1016/j.reseneeco.2014.07.001

Molinos-Senante, M., Hernandez-Sancho, F., Sala-Garrido, R., 2014b. Benchmarking in wastewater treatment plants: A tool to save operational costs. Clean Technol. Environ. Policy 16, 149–161. doi:10.1007/s10098-013-0612-8

Molinos-Senante, M., Hernández-Sancho, F., Sala-Garrido, R., 2010. Economic feasibility study for wastewater treatment: A cost-benefit analysis. Sci. Total Environ. 408, 4396–4402. doi:10.1016/j.scitotenv.2010.07.014

Molinos-Senante, M., Sala-Garrido, R., Iftimi, A., 2018. Energy intensity modeling for wastewater treatment technologies. Sci. Total Environ. 630, 1565–1572. doi:10.1016/j.scitotenv.2018.02.327

Niu, K., Wu, J., Qi, L., Niu, Q., 2019. Energy intensity of wastewater treatment plants and influencing factors in China. Sci. Total Environ. 670, 961–970. doi:10.1016/j.scitotenv.2019.03.159

Panepinto, D., Fiore, S., Zappone, M., Genon, G., Meucci, L., 2016. Evaluation of the energy efficiency of a large wastewater treatment plant in Italy. Appl. Energy 161, 404–411. doi:10.1016/j.apenergy.2015.10.027

Pei, M., Zhang, B., He, Y., Su, J., Gin, K., Lev, O., Shen, G., Hu, S., 2019. State of the art of tertiary treatment technologies for controlling antibiotic resistance in wastewater treatment plants. Environ. Int. 131, 105026. doi:10.1016/j.envint.2019.105026

Plappally, A.K., Lienhard V, J.H., 2012. Energy requirements for water production, treatment, end use, reclamation, and disposal. Renew. Sustain. Energy Rev. 16, 4818–4848. doi:10.1016/j.rser.2012.05.022

Rodriguez-Garcia, G., Molinos-Senante, M., Hospido, A., Hernández-Sancho, F., Moreira, M.T., Feijoo, G., 2011. Environmental and economic profile of six typologies of wastewater treatment plants. Water Res. 45, 5997–6010. doi:10.1016/j.watres.2011.08.053

Rojas, J., Zhelev, T., 2012. Energy efficiency optimisation of wastewater treatment: Study of ATAD. Comput. Chem. Eng. 38, 52–63. doi:10.1016/j.compchemeng.2011.11.016

Sabia, G., Luigi, P., Avolio, F., Caporossi, E., 2020. Energy saving in wastewater treatment plants: A methodology based on common key performance indicators for the evaluation of plant energy performance, classification and benchmarking. Energy Convers. Manag. 220, 113067. doi:10.1016/j.enconman.2020.113067

Scholz, M., 2016. Sludge Treatment and Disposal. Wetl. Water Pollut. Control 157–168. doi:10.1016/B978-0-444-63607-2.00021-6

Siatou, A., Manali, A., Gikas, P., 2020. Energy consumption and internal distribution in activated sludge wastewater treatment plants of Greece. Water (Switzerland) 12, 1–15. doi:10.3390/W12041204

Silva, C., Alegre, H., Rosa, M.J., 2015. Introduction to energy management in wastewater treatment plants, in: Stamatelatou, K., Tsagarakis, K.P. (Eds.), Sewage Treatment Plants : Economic Evaluation of Innovative Technologies for Energy Efficiency. IWA Publishing, pp. 33–55, ISBN: 9781780405018.

Silva, C., Matos, J.S., Rosa, M.J., 2016. A comprehensive approach for diagnosing opportunities for improving the performance of a WWTP. Water Sci. Technol. 74, 2935–2945. doi:10.2166/wst.2016.432

Silva, C., Ramalho, P., Quadros, S., Alegre, H., Rosa, M.J., 2012. Results of “PASt21” - The Portuguese initiative for performance assessment of water and wastewater treatment plants. Water Sci. Technol. Water Supply 12, 372–386. doi:10.2166/ws.2012.004

Silva, C., Rosa, M.J., 2015. Energy performance indicators of wastewater treatment: A field study with 17 Portuguese plants. Water Sci. Technol. 72, 510–519. doi:10.2166/wst.2015.189

Tao, X., Chengwen, W., 2012. Energy Consumption in Wastewater Treatment Plants in China. World Congr. Water, Clim. Energy 1–6. doi:10.13140/2.1.1228.9285

Torregrossa, D., Castellet-Viciano, L., Hernández-Sancho, F., 2019. A data analysis approach to evaluate the impact of the capacity utilization on the energy consumption of wastewater treatment plants. Sustain. Cities Soc. 45, 307–313. doi:10.1016/j.scs.2018.11.036

Torregrossa, D., Hernández-Sancho, F., Hansen, J., Cornelissen, A., Popov, T., Schutz, G., 2017. Energy saving in wastewater treatment plants: A plant-generic cooperative decision support system. J. Clean. Prod. 167, 601–609. doi:10.1016/j.jclepro.2017.08.181

Trapote, A., Albaladejo, A., Simón, P., 2014. Energy consumption in an urban wastewater treatment plant: the case of Murcia Region (Spain). Civ. Eng. Environ. Syst. 31, 304–310. doi:10.1080/10286608.2013.866106

Tsagarakis, K.P., Mara, D.D., Horan, N.J., Angelakis, A.N., 2000. Small municipal wastewater treatment plants in Greece. Water Sci. Technol. 41, 41–48. doi:10.2166/wst.2000.0007

Vaccari, M., Foladori, P., Nembrini, S., Vitali, F., 2018. Benchmarking of energy consumption in municipal wastewater treatment plants - A survey of over 200 plants in Italy. Water Sci. Technol. 77, 2242–2252. doi:10.2166/wst.2018.035

Vera, I., Sáez, K., Vidal, G., 2013. Performance of 14 full-scale sewage treatment plants: Comparison between four aerobic technologies regarding effluent quality, sludge production and energy consumption. Environ. Technol. (United Kingdom) 34, 2267–2275. doi:10.1080/09593330.2013.765921

Wakeel, M., Chen, B., Hayat, T., Alsaedi, A., Ahmad, B., 2016. Energy consumption for water use cycles in different countries: A review. Appl. Energy 178, 868–885. doi:10.1016/j.apenergy.2016.06.114

Wang, H., Yang, Y., Keller, A.A., Li, X., Feng, S., Dong, Y. nan, Li, F., 2016. Comparative analysis of energy intensity and carbon emissions in wastewater treatment in USA, Germany, China and South Africa. Appl. Energy 184, 873–881. doi:10.1016/j.apenergy.2016.07.061

Yang, L., Zeng, S., Chen, J., He, M., Yang, W., 2010. Operational energy performance assessment system of municipal wastewater treatment plants. Water Sci. Technol. 62, 1361–1370. doi:10.2166/wst.2010.394

Yang, X., Wei, J., Ye, G., Zhao, Y., Li, Z., Qiu, G., Li, F., Wei, C., 2020. The correlations among wastewater internal energy, energy consumption and energy recovery/production potentials in wastewater treatment plant: An assessment of the energy balance. Sci. Total Environ. 714, 136655. doi:10.1016/j.scitotenv.2020.136655

Yu, Y., Zou, Z., Wang, S., 2019. Statistical regression modeling for energy consumption in wastewater treatment. J. Environ. Sci. (China) 75, 201–208. doi:10.1016/j.jes.2018.03.023

Zaborowska, E., Czerwionka, K., Makinia, J., 2017. Strategies for achieving energy neutrality in biological nutrient removal systems - A case study of the Slupsk WWTP (northern Poland). Water Sci. Technol. 75, 727–740. doi:10.2166/wst.2016.564

Zeng, S., Chen, X., Dong, X., Liu, Y., 2017. Efficiency assessment of urban wastewater treatment plants in China: Considering greenhouse gas emissions. Resour. Conserv. Recycl. 120, 157–165. doi:10.1016/j.resconrec.2016.12.005

Zhang, Q.H., Yang, W.N., Ngo, H.H., Guo, W.S., Jin, P.K., Dzakpasu, M., Yang, S.J., Wang, Q., Wang, X.C., Ao, D., 2016. Current status of urban wastewater treatment plants in China. Environ. Int. 92–93, 11–22. doi:10.1016/j.envint.2016.03.024

Zhou, Y., Zhang, B., Wang, H., Bi, J., 2013. Drops of energy: Conserving urban water to reduce greenhouse gas emissions. Environ. Sci. Technol. 47, 10753–10761. doi:10.1021/es304816h

# Highlights

- Long-term operation of low installed power equipment deserves more attention
- Energy intensity is mainly influenced by plant's size, load, and influent dilution
- Anaerobic-anoxic-oxic systems are a good option for impoverished areas
- Energy performance must be assessed in a holistic and multi-criteria approach