

# UC Santa Barbara

## UC Santa Barbara Previously Published Works

### Title

Quantification of the water, energy and carbon footprints of wastewater treatment plants in China considering a water-energy nexus perspective

### Permalink

<https://escholarship.org/uc/item/92b076hm>

### Authors

Gu, Yifan  
Dong, Ya-nan  
Wang, Hongtao  
[et al.](#)

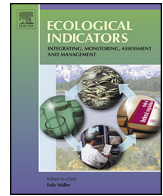
### Publication Date

2016

### DOI

10.1016/j.ecolind.2015.07.012

Peer reviewed



# Quantification of the water, energy and carbon footprints of wastewater treatment plants in China considering a water–energy nexus perspective

Yifan Gu<sup>a,1</sup>, Ya-nan Dong<sup>b,1</sup>, Hongtao Wang<sup>a</sup>, Arturo Keller<sup>c</sup>, Jin Xu<sup>b</sup>, Thomas Chiramba<sup>d</sup>, Fengting Li<sup>a,\*</sup>

<sup>a</sup> State Key Laboratory of Pollution Control and Resource Reuse, Tongji University, Shanghai 200092, China

<sup>b</sup> College of Environmental Science and Engineering, Tongji University, Shanghai 200092, China

<sup>c</sup> Bren School of Environmental Science and Management, University of California, Santa Barbara, CA 93106, United States

<sup>d</sup> United Nations Environment Programme (UNEP), Nairobi, Kenya

## ARTICLE INFO

### Article history:

Received 12 January 2015

Received in revised form 9 July 2015

Accepted 11 July 2015

### Keywords:

Wastewater treatment plants  
Water footprint compensation  
Energy intensity  
Carbon emission  
Environment impacts

## ABSTRACT

Water and energy are closely connected and both are very important for human development. Wastewater treatment plants (WWTPs) are central to water–energy interactions as they consume energy to remove pollutants and thus reduce the human gray water footprint on the natural water environment. In this work, we quantified energy consumption in 9 different WWTPs in south China, with different treatment processes, objects, and capacities. The energy intensity in most of these WWTPs is in the range of 0.4–0.5 kWh/m<sup>3</sup> in 2014. Footprint methodologies were used in this paper to provide insight into the environmental changes that result from WWTPs. A new indicator “gray water footprint reduction” is proposed based on the notion of gray water footprint to better assess the role of WWTPs in reducing human impacts on water resources. We find that higher capacity and appropriate technology of the WWTPs will result in higher gray water footprint reduction. On average, 6.78 m<sup>3</sup> gray water footprint is reduced when 1 m<sup>3</sup> domestic sewage is treated in WWTPs in China. 13.38 L freshwater are required to produce the 0.4 kWh electrical input needed for treating 1 m<sup>3</sup> domestic wastewater, and 0.23 kg CO<sub>2</sub> is emitted during this process. The wastewater characteristics, treatment technologies as well as management systems have a major impact on the efficiency of energy utilization in reducing gray water footprint via these WWTPs. The additional climate impact associated with wastewater treatment should be considered in China due to the enormous annual wastewater discharge. Policy suggestions are provided based on results in this work and the features of China’s energy and water distribution.

© 2015 Elsevier Ltd. All rights reserved.

## 1. Introduction

As driving forces and limiting factors for sustainable development, water and energy are key resources for global production and life (Dincer, 2002; Gleick, 1994; Walker et al., 2013). The nexus between water and energy pervades modern economies. These two inextricably intertwined fundamental resources have become a fascinating topic (Hellegers et al., 2008; Jägerskog et al., 2014; Kenway et al., 2011; Perrone et al., 2011; Scott et al., 2011; U.S. Department of Energy, 2014; Water in the west, 2013). Water

supply, consumption, transportation and wastewater treatment require various forms of energy (Lazarova et al., 2012; Stokes and Horvath, 2006, 2009), while almost every stage in the energy supply chain needs water (Blackhurst et al., 2010; Dominguez-Faus et al., 2009; Gerbens-Leenes et al., 2009; International Energy Agency, 2012; Rio Carrillo and Frei, 2009). The energy sector is the second largest water user in the world in terms of withdrawals, following irrigation (Hightower and Pierce, 2008). For example, water used in thermoelectric power generation accounted for nearly 49% of total fresh water withdrawals in the United States (Scown et al., 2011). In addition, some large water transfer projects, such as China’s South-to-North water diversion project, need enormous energy supply. The water–energy nexus has increasingly become prominent in domestic and international policy discourse and prompted a number of studies to explore managing the

\* Corresponding author. Tel.: +86 21 65980567.

E-mail address: [fengting@tongji.edu.cn](mailto:fengting@tongji.edu.cn) (F. Li).

<sup>1</sup> These authors contributed equally to this work.

links between energy and water for a sustainable future (Hardy et al., 2012; Kahrl and Roland-Holst, 2008; Kenney and Wilkinson, 2011; Lofman et al., 2002; Malik, 2002; Siddiqi and Anadon, 2011).

Wastewater treatment plants (WWTP) are a typical case of water–energy interactions. In the WWTP, water quality is improved at the cost of energy consumption. The emission of greenhouse gases from respiration and power consumption in WWTP has caused wide concern (Gori et al., 2011; Stokes and Horvath, 2009). Schnoor pointed out that probably the greatest water story of the 21st century is to treat wastewater through membranes and reverse osmosis for drinking water supplies with significant energy consumption (Schnoor, 2011). In developed countries, wastewater treatment accounts for about 3% of the electrical energy load (Curtis, 2010). It was reported that the high energy costs for wastewater treatment due to aeration requirement in the U.S. cannot be borne by developing countries (Liu et al., 2004). Therefore, the water–energy nexus in wastewater treatment needs further study.

Assessing humanity's “environmental footprint” is one way of reflecting the total human pressure on the planet (Hoekstra and Wiedmann, 2014). Water footprint (WF) (Hoekstra et al., 2011) and energy footprint (EnF) (Wiedmann, 2009) refer to the total freshwater and energy directly and indirectly required to produce a commodity or service. The Water Footprint Network uses energy water footprint to link energy with water, which makes it possible to assess the virtual water consumption through the usage of energy (Hoekstra et al., 2011; Gu et al., 2014b). As a reflection of growing concerns about the increasing pressures of energy and water consumption, there have been an increasing number of studies aimed to systematically quantify the energy–water nexus by using water footprint tool, such as water footprint of biofuels (Dominguez-Faus et al., 2009); bioenergy (Gerbens-Leenes et al., 2009); bio-ethanol (Gerbens-Leenes and Hoekstra, 2012); nonfood biomass fuel (Zhang et al., 2014); and electricity from hydropower (Mekonnen and Hoekstra, 2012). These studies are meaningful and helpful to understand the water–energy nexus and guide policy making. However, there are few studies on the water–energy nexus in WWTPs from the water footprint point of view. The energy used in wastewater treatment also consumes some direct and indirect water withdrawals and results in wastewater discharge. Research by Shao and Chen, 2013 shows that the water footprint of electricity accounts for 57% of the total water footprint for a medium scale WWTP in Beijing, China. These links between water embodied in energy use are considerable but usually not included to assess the efficiency of WWTPs.

In this study, we evaluate the water–energy nexus in WWTPs in south China considering their water and energy footprints to reduce their environmental impacts. We investigate 9 different WWTPs in south China with different treatment techniques, sources (domestic/industrial wastewater) and treatment capacities in 2014. We quantify energy consumption and the virtual water embodied in energy consumed by these WWTPs. A new indicator “gray water footprint reduction” is proposed based on the notion of gray water footprint (GWF) (Hoekstra et al., 2011) to better assess the role of WWTPs in reducing human impacts on water resources. Thus, this study also contributes to the development of footprint methodologies. Our aims are (1) to quantify the water–energy nexus in WWTPs by accounting for the freshwater, energy and carbon footprints as they seek to reduce the GWF; (2) to assess the efficiency of the energy utilization of WWTPs in reducing the GWF; (3) to understand how WWTPs interact with the hydrologic cycle, energy resources and climate; and (4) to make policy suggestions for future WWTP construction in consideration of the energy–water implications.

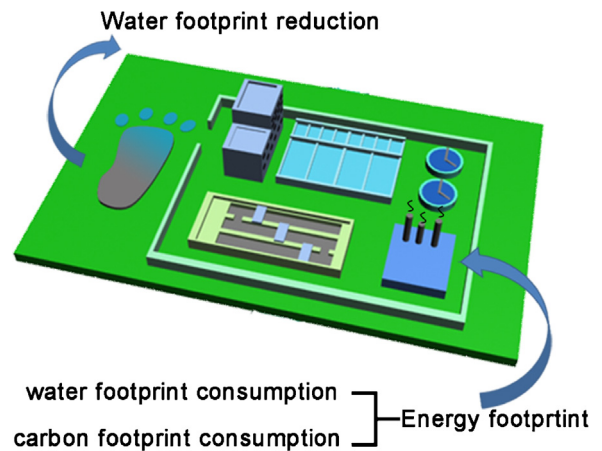


Fig. 1. Input and output footprints of a WWTP.

## 2. Materials and methods

### 2.1. Water footprint compensation and energy efficiency assessment model

Fig. 1 shows the connections between various footprints of a WWTP based on the water–energy nexus. Every stage in the WWTP needs energy input, such as wastewater collection, physical treatment, chemical treatment, sludge treatment, and discharge. In most WWTPs, electricity is used as the only energy source for pumping and carrying wastewater through pipes, as well as operating most of the equipment. Electricity production needs water withdrawals and causes CO<sub>2</sub> emissions. Thus, there are both WF and carbon footprint (CF) (Wiedmann and Minx, 2008) behind the EnF input, based on a life cycle analysis. GWF refers to pollution and is defined as the volume of freshwater that would be required to dilute the pollutants to meet given natural background concentrations and existing water quality standards (Hoekstra et al., 2011). It assumes that dilution is the only treatment, although in almost all cases this is not the case. The GWF of the region can be reduced via the WWTP, reducing the impact on the environment. Treated WWTP effluent can also be reused for irrigation, industrial purposes, drinking and many other activities, reducing blue water footprint (i.e. groundwater and surface water consumption) to realize water footprint compensation. However, there are trade-offs in water footprint reduction since it can increase EnF and CF.

GWF is a useful metric for comparing effluent water quality when there are multiple pollutants at different concentrations and perhaps different water quality standards due to the sensitive nature of some water bodies in water footprint assessment. Although it refers to a hypothetical dilution volume, GWF is important in the assessment of environmental effects on a water resource. In the existing water footprint methodologies, wastewater treatment can reduce the GWF down to zero when the concentrations of pollutants in the treated effluent are equal to or lower than the water quality standards or the concentrations from the water source (Hoekstra et al., 2011). However, to better reflect the role of WWTPs in reducing the impact on human activities on water resources, a new indicator “water footprint reduction” ( $\Delta GWF$ ) is proposed here. The  $\Delta GWF$  (in m<sup>3</sup> of freshwater) of a WWTP for a specific period of time is defined as follows:

$$\Delta GWF = \text{MIN} \left[ \frac{Q_i - B_i}{B_i} \right] \times V$$

where  $Q_i$  are the concentrations of main pollutants in the WWTP influent (in mass/volume);  $B_i$  are the concentrations of main pollutants  $i$  after treatment (in mass/volume) and  $V$  is the wastewater



Fig. 2. Geographical locations of the WWTPs investigated.

volume treated by the WWTP in the specific time period. In this study, we mainly consider Chemical Oxygen Demand (COD), Biochemical Oxygen Demand (BOD), and Total Nitrogen (TN), which are common water quality criteria for a WWTP. The  $B_i$  value of these indicators should be greater than zero.  $\Delta GWF$  reflects the reduction in freshwater volume required to assimilate the pollutants due to the WWTP. Thus, a greater  $\Delta GWF$  value corresponds to more contribution to water footprint compensation by the WWTP.

The gray water footprint reduction efficiency ( $GWFRE$ ) of a WWTP is determined as follows:

$$GWFRE = \text{MIN} \left[ \frac{Q_i - B_i}{B_i} \right]$$

$GWFRE$  is dimensionless. It quantifies the efficiency with which the WWTP reduces the gray water footprint of the surrounding community (i.e. the sources of wastewater). A higher  $GWFRE$  corresponds to higher GWF reduction efficiency for a WWTP.

The energy utilized to reduce the GWF ( $eGWFR$ , in  $\text{m}^3/\text{kWh}$ ) via the WWTP is:

$$eGWFR = \frac{\Delta GWF}{E_l} = \frac{\text{MIN} \left[ \frac{Q_i - B_i}{B_i} \right] \times V}{E_l}$$

where  $E_l$  is the total energy input during the wastewater treatment process ( $E_l$ , in kWh). A greater  $eGWFR$  corresponds to increased gray water footprint reduction for a unit energy input into the wastewater treatment process.

With the progression of water footprint methodology, carbon footprint methodology and energy footprint methodology research, the footprint methods are widely accepted to be implemented for the analysis of human activities processes and services (Hoekstra and Wiedmann, 2014). The methods proposed in this research are a simple improvement on existing methodologies of footprints assessment, which are easy to understand and apply. In addition, the original dates needed for this research are not complex. Therefore, our research could be repeated and applied to other cases.

## 2.2. Description of case study WWTPs and data sources

Herein, we investigate 9 different WWTPs in the south China. The basic information and geographical location of these WWTPs are shown in Table 1 and Fig. 2. These are WWTPs treating industrial or domestic wastewater with different treatment techniques like Cyclic Activated Sludge System (CASS), Anaerobic–Anoxic–Oxic technique ( $A^2/O$ ), humic packing filter technology, and others. For this study, the primary data on wastewater intake (flow and quality), wastewater discharge (flow and quality), and energy consumption of the selected WWTPs were accurately obtained from the engineers of the facilities with an estimated error of 5% or less in 2014. Detailed calculation information of the carbon footprint and the virtual water embodied in energy input in the WWTPs investigated can be found in Appendix A.

## 3. Results and discussion

### 3.1. Water–energy nexus in case study WWTPs

Energy use in the water sector is associated with abstraction, conveyance and treatment of fresh water and wastewater, together with end-use processes (particularly heating of water). The direct energy input per unit wastewater treated for the case study WWTPs is shown in Fig. 3, considering only operational electricity. The wastewater treatment technologies chosen have a significant impact on the energy requirements, more than the overall plant throughput. The WWTP #3, WWTP #6 and WWTP #7 use Anaerobic–Anoxic–Oxic ( $A^2/O$ ) treatment technology to treat domestic wastewater.  $A^2/O$  is one of the most mature and popular wastewater treatment technologies in the world. The electrical energy intensities in these three plants are similar and around  $0.45 \text{ kWh}/\text{m}^3$ . The slight distinction between them may due to differences in wastewater inflow, quality and management in these plants. Electric energy intensity in the WWTP #2 with a humic filter treatment technology is lowest at  $0.25 \text{ kWh}/\text{m}^3$ . At  $0.6 \text{ kWh}/\text{m}^3$ , the electric energy intensity of the MBR process in the WWTP #8 is the highest. Other treatment technologies like oxidation ditches, Anoxic/Oxic ( $A/O$ ), and Constructed Rapid Infiltration

**Table 1**  
Basic information of the WWTPs investigated.

Number	Wastewater type	Treatment technology	Capacity (m <sup>3</sup> per day)	Location
#1	Domestic sewage and industrial wastewater	Oxidation Ditch	$3 \times 10^5$	Zhejiang Province
#2	Domestic sewage	Humus Filter	40	Jiangsu Province
#3	Domestic sewage	Anaerobic–Anoxic–Oxic	$4 \times 10^4$	Jiangsu Province
#4	Domestic sewage	Constructed Rapid Infiltration Technology (CRI)	$2.2 \times 10^4$	Guangxi Province
#5	Domestic sewage	4S-MBR	200	Jiangxi Province
#6	Domestic sewage	Anaerobic–Anoxic–Oxic	5000	Jiangsu Province
#7	Domestic sewage and industrial wastewater	Anaerobic–Anoxic–Oxic	$2.25 \times 10^4$	Jiangsu Province
#8	Electroplating wastewater	Membrane Bio-Reactor (MBR)	300	Guangdong Province
#9	Dyeing wastewater	Anoxic/Oxic	2000	Guangdong Province

Technology (CRI) have a medium energy intensity, in the range of 0.4–0.5 kWh/m<sup>3</sup>.

The virtual water embodied in the electricity in different regions in China is presented in Table A.1 (in Appendix A). Regional differences of water withdrawal intensity for energy production are significant (Zhang and Anadon, 2013), with Shanghai the highest at >60 m<sup>3</sup>/MWh, and Guangxi the lowest at just over 20 m<sup>3</sup>/MWh. WWTP #8 and WWTP #9 have a higher amount virtual water embodied, because they are located in Guangdong province which has higher water intensity for electricity than the WWTPs in the other regions. Thus, virtual water needed for wastewater treatment needs to be considered in the overall water footprint of these facilities.

### 3.2. Water footprint compensation by WWTPs

$\Delta$ GWF per day for the case study WWTPs (Fig. A.1), is a function of the WWTP throughput (m<sup>3</sup>/day) and the effectiveness of the treatment technologies, converted to the reduction in the theoretical amount (m<sup>3</sup>) of water needed to dilute the effluent to the water quality objective (based on the common definition of GWF). The WWTP #1 has by far the largest  $\Delta$ GWF ( $-1116 \times 10^3$  m<sup>3</sup>/day) with the largest throughput ( $300 \times 10^3$  m<sup>3</sup>/day). The WWTP #3 and #4 have similar  $\Delta$ GWF ( $-119 \times 10^3$  and  $-125 \times 10^3$  m<sup>3</sup>/day) although they have significantly different throughputs ( $40 \times 10^3$  and  $22 \times 10^3$  m<sup>3</sup>/day, respectively). The reason for this discrepancy is that WWTP #4 uses CRI, which results in higher treatment efficiency (85%) than WWTP #3's A<sup>2</sup>/O process (75% removal). Thus, wastewater treatment can significantly reduce the regional gray water footprint, but treatment technology matters to achieve this objective.

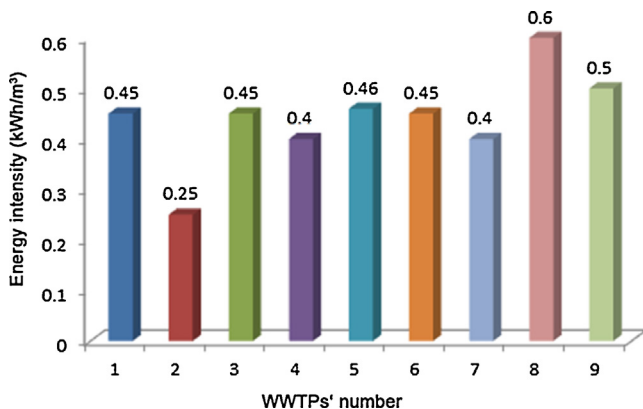
The GWFRE for the case study WWTPs are showing in Fig. 4. The WWTP #9 has the largest GWFRE (39). Note that this is different from the traditional treatment efficiency, and reflects the reduction in theoretical freshwater needed to dilute the original wastewater to the water quality objectives in the absence of the

WWTP. This high GWFRE reflects the high inlet concentration of the specific industrial wastewater for this WWTP, with a very high COD<sub>Cr</sub> ( $\approx 2000$  mg/L). Thus, a relatively high gray water footprint is reduced by treating dyeing wastewater. The average GWFRE of domestic sewage treatment plants (WWTPs #2, #3, #4, #5, and #6) is 6.8, but there is significant variability. The WWTP #5 has a relatively high GWFRE (15), since it uses 4S-MBR technology to treat sewage with a high COD removal rate (90%). GWFRE is thus a strong function of both inlet concentrations and treatment technology.

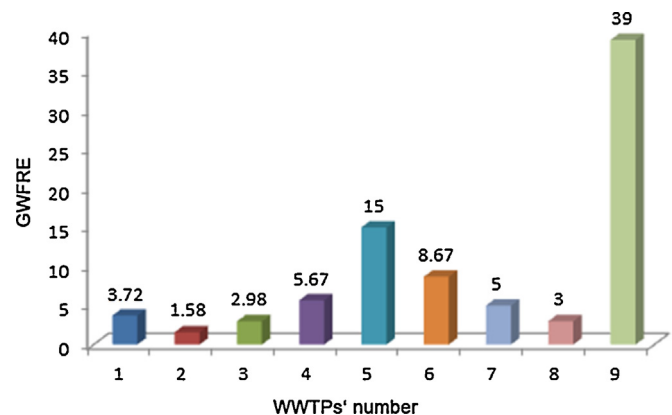
The common definition of GWF is based on the volume of freshwater that is required to assimilate pollutants on the basis of natural background concentrations and existing water quality standards. Therefore, the amount of GWF depends on the water quality standards, which reflect national policies and even regional differences (Gu et al., 2014a,b). The proposed indicators ( $\Delta$ GWF and GWFRE) are independent on the water quality standards, yet provide a useful metric for comparing WWTPs across large regions. Thus, this is an important contribution to water footprint methodologies.

### 3.3. Energy utilized to reduce gray water footprint for the case study WWTPs

Though the energy intensity is similar in the case study WWTPs (generally 0.4–0.6 kWh/t), there are large differences in eGWFR (Fig. 5). The WWTP #9 has the largest eGWFR (78 m<sup>3</sup>/kWh), indicating very high energy utilization efficiency for wastewater treatment, driven by the high input concentration and high removal efficiency without requiring a much higher energy input. This is followed by the WWTP #5 with an eGWFR of 33 m<sup>3</sup>/kWh, reflecting the high energy efficiency of the 4S-MBR process as well as the higher waste load removal. The CRI and A<sup>2</sup>/O wastewater treatment processes for domestic wastewater are markedly lower in eGWFR, although in general they outperform the humic filter used in the small WWTP #2. The result shows that wastewater characteristics, treatment technology as well as WWTP operation can significantly affect the efficiency of energy utilization in reducing gray water



**Fig. 3.** Energy input per unit wastewater treated for the case study WWTPs.



**Fig. 4.** GWFRE for the case study WWTPs.

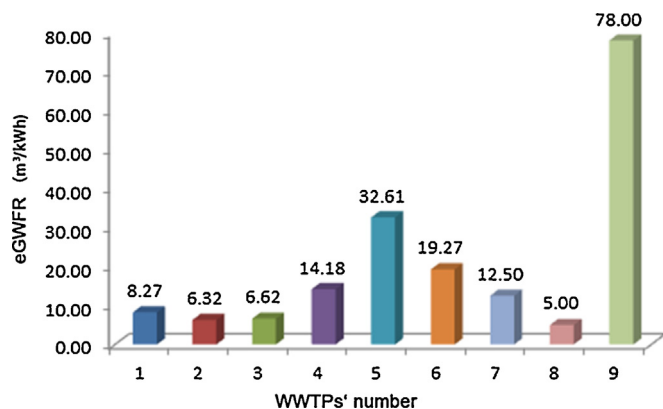


Fig. 5. eGWF of cases WWTPs.

footprint. It is also notable that a higher input pollutant load, as is the case for some industrial wastewaters, can be removed with relatively lower energy input per unit of GWF reduction.

### 3.4. Water, energy and carbon footprints and impact assessment

Fig. 6 shows the average water, energy and carbon footprints for 1 m<sup>3</sup> of domestic sewage treated, considering WWTPs #2, #3, #4, #5, and 6. On average, 6.78 m<sup>3</sup> (with a standard deviation of 5.3) GWF is reduced when 1 m<sup>3</sup> of domestic wastewater is treated in these WWTPs. 13.4L (0.0134 m<sup>3</sup>) (with a standard deviation of 4.2) of freshwater are required to produce the 0.4 kWh (with a standard deviation of 0.1) input for treating 1 m<sup>3</sup> wastewater. Since the treatment technologies and capacities in these WWTPs are typical of China, the average result can be used to roughly estimate the footprints attributable to domestic wastewater treatment in China. It is estimated that  $4.3 \times 10^{10}$  m<sup>3</sup> domestic wastewater was treated in 2011 (Ministry of Environmental Protection, 2011). This would require an energy input of  $1.3 \times 10^7$ – $2.2 \times 10^7$  MWh, with  $4.0 \times 10^8$ – $7.6 \times 10^8$  m<sup>3</sup> freshwater embodied. The estimated energy input accounts for 0.21–0.49% of the total  $4.7 \times 10^9$  MWh electricity consumption in 2011 (National Bureau of Statistics of China, 2013). Although it is less than 1% of total electricity use, it is a very significant amount, growing with increasing wastewater treatment and population. Therefore, energy efficient technologies such as 4S-MBR should be considered for new facilities as well as upgrades and capacity increases.

The estimated gray water footprint reduction would be  $0.6$ – $5.2 \times 10^{11}$  m<sup>3</sup>. Compared with the amount of gray water footprint reduced, the freshwater consumed to generate the electricity is negligible. However, it is important to understand how WWTPs impact the hydrologic cycle. According to official statistics, in 2010 about 768 million people in China had rural status, about 57% of the population. However, the domestic wastewater treated was only about 11% in provincial towns and less than 1% for rural villages (Wu et al., 2011). Until 2013, there were 4316 WWTPs installed, with a total capacity of 0.126 billion m<sup>3</sup> wastewater treated per day (Ministry of Environmental Protection, 2013). Thus there is a major need to consider energy efficient WWTPs at all levels in China and other developing countries.

The energy footprint of WWTPs connects the water and carbon footprints. One impact of wastewater treatment is an increase in greenhouse-gas emissions that results from energy use for wastewater treatment. However, few studies have consider the trade-offs between water and energy in assessing the CF of the water sector (Rothausen and Conway, 2011). Fig. 6 shows that on average, 0.28 kg CO<sub>2</sub> are produced when electricity is used to treat 1 m<sup>3</sup> domestic wastewater. At a national level this would

Table 2

Comparison of carbon footprint in the wastewater treatment system.

Name	Reference	Estimated carbon footprint (kg/m <sup>3</sup> )
WWTP #2	This work	0.20
WWTP #3	This work	0.36
WWTP #4	This work	0.37
WWTP #5	This work	0.45
WWTP #6	This work	0.36
Southern wastewater treatment works in eThekweni Municipality, South Africa	Friedrich et al. (2009)	0.11
City of Toronto municipal water treatment system	Racoviceanu et al. (2007)	0.12

represent about  $1.2 \times 10^{10}$  kg CO<sub>2</sub> produced if all domestic wastewater discharged in 2011 was treated. The additional climate impact associated with wastewater treatment should be seriously considered in China due to enormous wastewater discharged annually. Our calculations indicate that the CF of the domestic sewage WWTPs in this study are two to three times greater than WWTPs in other countries (Table 2). This is maybe because thermal power accounts for 83.2% of electricity power generation in China (Electricity Regulatory Commission of China, 2007). The extensive use of coal will result in major carbon emissions. Thus, there is room for improving the CFs of WWTPs in China, which can have major global climate implications.

### 3.5. Data limitation and uncertainty analysis

The primary data on the wastewater intake, wastewater discharge, and energy consumption of the selected WWTPs were accurately obtained from the plant engineers, resulting in a low uncertainty ( $\pm 5\%$ ).

Uncertainties also result from the assumptions to establish the research scope. Energy use in the wastewater sector includes facility construction and operation. In this study, we focus on the wastewater treatment operation processes and do not consider the construction of the WWTPs. There are no studies that have conducted a systematic accounting of the energy and water needed to build WWTPs, and it was beyond the scope of the current study to do so. There is also no information on the decommissioning of WWTPs, with regards to energy or water requirements. However, given the long life of most WWTPs (30–50 years), it is likely that most of the energy and water requirements occur in the operational phase. There are some uncertainties in the calculation of the energy water footprint and carbon footprint. In the calculation of the water footprint of electricity in China, the conversion coefficients are from Zhang and Anadon (2013). The authors do not provide an estimate of their uncertainties, but it is likely that there is variability even within each region, and there may be some under or over prediction.

### 3.6. Policy suggestions

Historically, wastewater treatment has been regulated through the pollutants removal rate. The new indicator “gray water footprint reduction” proposed here can assess WWTPs’ effectiveness from water footprint sustainable perspective in a specific region. In addition, it is clear that water/wastewater treatment encompasses highly energy-intensive processes. The water sector faces great challenges in the coming decades. Greater focus on its energy requirements will be a crucial part of the policy response to these challenges (Rothausen and Conway, 2011). The water footprint compensation and energy efficiency assessment model proposed in this paper provide tools for understanding how WWTPs interact with the hydrologic cycle, energy resources and climate based on

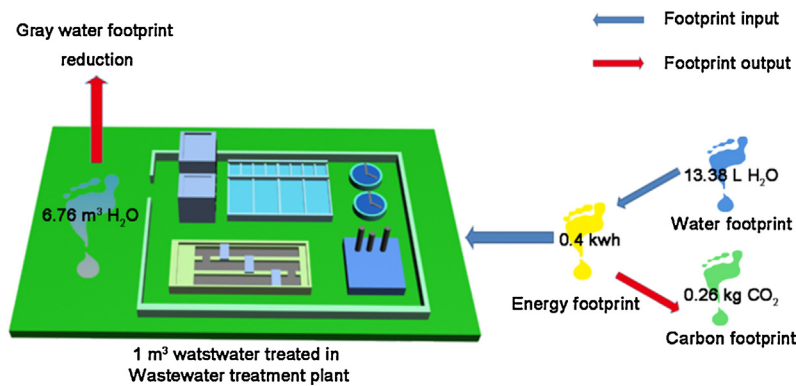


Fig. 6. Average footprints accounting of 1 m<sup>3</sup> wastewater treated in cases WWTPs.

the water–energy nexus, as such, more comprehensive information is obtained. Thus, it can improve strategies for future WWTPs construction and assessment in consideration of energy–water implications.

Water–energy portfolios ultimately must be studied at the local scale (Pandya Swargo and Abe, 2014). China faces greater natural resources (both water and energy) challenges than other major countries. China is the second largest energy consumer following the United States. However, China's per capita use of energy in 2001 was only a ninth of that in the United States, and half of the world average (Liu and Diamond, 2005). In addition, both water and energy resources are distributed unevenly. Southern China has more water resources than northern China, and the east more than the west. Three provinces (Shanxi, Shaanxi, and Inner Mongolia, all located in northwest of China) contain the largest coal (the dominant primary energy source in China) deposits and production in China, contributing more than half of the total national coal output and 16% thermal power generation (Zhang and Anadon, 2013). To meet the higher effluent water quality standard and larger covering range, further improvements of planning and management are needed for wastewater treatment in rural and urban China. Under these circumstances and based on the findings in this work, it is suggested that low energy footprint technologies (e.g. humus filter) be considered in southern China where the water resources are abundant but the energy resources are limited. For wastewater in which the concentration of pollutants is not high, low energy consumption treatment technology is also suitable. Artificial wetland is specially suggested as it can reduce gray water footprint and carbon footprint with very low energy consumption. In northern China where the water resources are scarce but the energy resources are relatively abundant, high *eGWFR* treatment technologies are recommended to achieve more gray water footprint reduction and more gray water reused. These suggestions also can spread to other countries. The decision-makers can adopt different treatment technologies with low energy footprint or high *eGWFR* from a top-down perspective according to overview of the scarcity/abundance degree of water and energy resources in different regions. Virtual water embodied in energy consumption in WWTPs should be given more consideration particularly in water-limited regions. From the bottom-up approach, the decision-makers can obtain information of the WWTPs already built in specific regions through the water footprint compensation and energy efficiency assessment model. Thus, the sustainability of these WWTPs can be evaluated from water and energy perspective, as such, adjustment strategies could be founded.

#### 4. Conclusion

While the full extent of the water–energy nexus in wastewater treatment system is difficult to assess, in this paper, we

considers a footprints methodology to provide a path for understanding how WWTPs interact with the hydrologic cycle, energy resources and climate based on the water–energy nexus as such, more comprehensive information is obtained. A new indicator “gray water footprint reduction” was proposed based on the notion of gray water footprint to better assess the role of WWTPs in reducing human impacts on water resources. Nine real WWTPs in the south of China with different treatment processes, objectives, and capacities were modeled with this procedure.

Our results show that higher capacity and appropriate technology of the WWTPs will result in higher gray water footprint reduction. On average, 6.78 m<sup>3</sup> gray water footprint is reduced when 1 m<sup>3</sup> domestic sewage is treated in WWTPs in China in 2014. It requires 13.38 L freshwater to produce the 0.4 kWh electricity needed for treating 1 m<sup>3</sup> domestic wastewater, and 0.23 kg CO<sub>2</sub> is emitted during this process. Therefore, the water–energy nexus in wastewater could be quantified based on our methods to how WWTPs interact with the hydrologic cycle, energy resources and climate. A set of indices are devised to reveal the purification efficiency and renewability of a wastewater treatment system at the expense of energy consumption. Based on that, we assessed the efficiency of the energy utilization of the nine real cases WWTPs in China. We found that the wastewater characteristics, treatment technologies as well as management systems have a major impact on the efficiency of energy utilization in reducing gray water footprint via these WWTPs. Policy suggestions are provided based on results in this work and the features of China's energy and water distribution, which also can be applied other countries. This work is expected to contribute to better planning and operation of WWTPs by considering the water–energy nexus.

#### Acknowledgements

The research was partially supported by the Fundamental Research Funds for the Central Universities (0400219276 and 0400219184).

#### Appendix A.

The virtual water embodied in energy production in China in 2007 (Table A.1) is based on Zhang et al.'s work (Zhang and Anadon, 2013). They studied the life cycle water withdrawals, consumptive water use, and wastewater discharge of China's energy sectors by using a mixed-unit multiregional input-output (MRIO) model at the provincial level.

Carbon emission model developed by Intergovernmental Panel on Climate Change (IPCC) is chosen to assess the carbon emission situations of every wastewater treatment plant. According to the 1996 Guidelines and IPCC Good Practice Guidance

**Table A.1**  
Water withdrawal coefficients of electricity in China.

Region	Water withdrawal coefficients of electricity (m <sup>3</sup> /MWh)
Shanghai	60.1
Jiangsu	37.4
Zhejiang	32.8
Jiangxi	37.8
Guangdong	41.8
Guangxi	21.4

**Table A.2**  
National EF of electricity for carbon emission calculation.

Region	EF <sub>grid,OM,y</sub> (tCO <sub>2</sub> /MWh)
North China	1.0303
Northeast China	1.1120
East China	0.8100
Central China	0.9779
Northwest China	0.9720
South China	0.9223

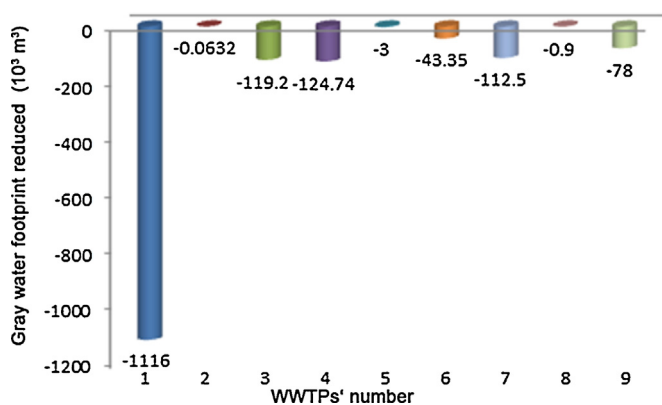


Fig. A.1. ΔGWF per day for the case study WWTPs.

(Intergovernmental Panel on Climate Change, 2006), the most common simple methodological approach is to combine information on the extent to which a human activity takes place (called activity data or AD) with coefficients which quantify the emissions or removals per unit activity. These are called emission factors (EF). Here the basic equation is as follow:

$$\text{Emissions} = \text{AD} \cdot \text{EF}$$

In the equation, AD can be get in the enterprise daily production process when calculating the carbon of enterprise, and the EF value mainly comes from literatures (i.e. IPCC Guidelines, IPCC Emission Factor Database (EFDB), International Emission Factor Databases: USEPA, etc.). Here, we aim to the EF of electricity, thus we get the EFs come 2013 Baseline Emission Factors for Regional Power Grids in China in the website of National Development and Reform Commission (National Development and Reform Commission, 2013,). The EFs are shown in Table A.2.

## References

Blackhurst, B.M., Hendrickson, C., Vidal, J.S.I., 2010. Direct and indirect water withdrawals for US industrial sectors. *Environ. Sci. Technol.* 44, 2126–2130.

Curtis, T.P., 2010. Low-energy wastewater treatment: strategies and technologies. *Environ. Microbiol.* 2.

Dincer, I., 2002. The role of exergy in energy policy making. *Energy Policy* 30, 137–149.

Dominguez-Faus, R., Powers, S.E., Burken, J.G., Alvarez, P.J., 2009. The water footprint of biofuels: a drink or drive issue? *Environ. Sci. Technol.* 43, 3005–3010.

Electricity Regulatory Commission of China, 2007. Electricity Supply and Demand Situation in 2006 and Situation Analysis of 2007.

Friedrich, E., Pillay, S., Buckley, C.A., 2009. Carbon footprint analysis for increasing water supply and sanitation in South Africa: a case study. *J. Clean. Prod.* 17, 1–12.

Gerbens-Leenes, W., Hoekstra, A.Y., 2012. The water footprint of sweeteners and bio-ethanol. *Environ. Int.* 40, 202–211.

Gerbens-Leenes, W., Hoekstra, A.Y., van der Meer, T.H., 2009. The water footprint of bioenergy. *Proc. Natl. Acad. Sci. U. S. A.* 106, 10219–10223.

Gleick, P.H., 1994. Water and energy. *Annu. Rev. Energy Environ.* 19, 267–299.

Gori, R., Jiang, L.-M., Sobhani, R., Rosso, D., 2011. Effects of soluble and particulate substrate on the carbon and energy footprint of wastewater treatment processes. *Water Res.* 45, 5858–5872.

Gu, Y., Li, Y., Wang, H., Li, F., 2014a. Gray water footprint: taking quality, quantity, and time effect into consideration. *Water Resour. Manage.* 28, 3871–3874.

Gu, Y., Xu, J., Wang, H., Li, F., 2014b. Industrial water footprint assessment: methodologies in need of improvement. *Environ. Sci. Technol.* 48, 6531–6532.

Hardy, L., Garrido, A., Juana, L., 2012. Evaluation of Spain's water-energy nexus. *Int. J. Water Resour. Dev.* 28, 151–170.

Hellegers, P., Zilberman, D., Steduto, P., McCornick, P., 2008. Interactions between water, energy, food and environment: evolving perspectives and policy issues. *Water Policy* 10, 1.

Hightower, M., Pierce, S.A., 2008. The energy challenge. *Nature* 452, 285–286.

Hoekstra, A.Y., Chapagain, A.K., Aldaya, M.M., Mekonnen, M.M., 2011. The Water Footprint Assessment Manual: Setting the Global Standard. Earthscan, London, UK.

Hoekstra, A.Y., Wiedmann, T.O., 2014. Humanity's unsustainable environmental footprint. *Science* 344, 1114–1117.

Intergovernmental Panel on Climate Change, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories. Institute for Global Environmental Strategies, Hayama, Kanagawa, Japan.

International Energy Agency, 2012. Methodology for estimating water requirements for energy production, [http://www.worldenergyoutlook.org/media/weowebsite/energymodel/documentation/Methodology\\_water\\_energy\\_nexus.pdf](http://www.worldenergyoutlook.org/media/weowebsite/energymodel/documentation/Methodology_water_energy_nexus.pdf)

Jägerskog, A., Clausen, T.J., Holmgren, T., Lexén, K. (Eds.), 2014. Energy and Water: The Vital Link for a Sustainable Future. Report Nr. 33. SIWI, Stockholm.

Kahrl, F., Roland-Holst, D., 2008. China's water-energy nexus. *Water Policy* 10, 51.

Kenny, D.S., Wilkinson, R., 2011. The Water-Energy Nexus in the American West. Edward Elgar Publishing.

Kenway, S., Lant, P., Priestley, A., Daniels, P., 2011. The connection between water and energy in cities: a review. *Water Sci. Technol.* 63, 1983–1990.

Lazarova, V., Choo, K.H., Cornel, P. (Eds.), 2012. Water-Energy Interactions in Water Reuse. IWA Publishing.

Liu, J., Diamond, J., 2005. China's environment in a globalizing world. *Nature* 435, 1179–1186.

Liu, H., Ramnarayanan, R., Logan, B.E., 2004. Production of electricity during wastewater treatment using a single chamber microbial fuel cell. *Environ. Sci. Technol.* 38, 2281–2285.

Lofman, D., Petersen, M., Bower, A., 2002. Water, energy and environment nexus: the California experience. *Int. J. Water Resour. Dev.* 18, 73–85.

Malik, R., 2002. Water-energy nexus in resource-poor economies: the Indian experience. *Int. J. Water Resour. Dev.* 18, 47–58.

Mekonnen, M.M., Hoekstra, A.Y., 2012. The blue water footprint of electricity from hydropower. *Hydrol. Earth Syst. Sci.* 16, 179–187.

Ministry of Environmental Protection, 2011. Environment Statistical Yearbook in 2011 (in Chinese).

Ministry of Environmental Protection, 2013. Announcement on list of the national urban sewage treatment facilities, [http://www.mep.gov.cn/gkml/hbb/bgg/201404/t20140415\\_270550.htm](http://www.mep.gov.cn/gkml/hbb/bgg/201404/t20140415_270550.htm) (in Chinese).

National Bureau of Statistics of China, 2013. <http://www.stats.gov.cn/> (in Chinese).

National Development and Reform Commission, 2013. Chinese baseline emission factor for regional power grid 2013, <http://cdm.ccchina.gov.cn/Detail.aspx?newsId=41387&Tid=19> (in Chinese).

Pandyaaswargo, A.H., Abe, N., 2014. The Water-Energy Nexus and Development of Basic Needs of Urban-Life (BNU) Index (POST 2015 Working Paper No.1). Tokyo, <http://t2r2.star.titech.ac.jp/rrows/file/CTT100675168/ATD100000413/>

Perrone, D., Murphy, J., Hornberger, G.M., 2011. Gaining perspective on the water-energy nexus at the community scale. *Environ. Sci. Technol.* 45, 4228–4234.

Racovitanu, A., Karney, B., Kennedy, C., Colombo, A., 2007. Life-cycle energy use and greenhouse gas emissions inventory for water treatment systems. *J. Infrastruct. Syst.* 13, 261–270.

Rio Carrillo, A.M., Frei, C., 2009. Water: a key resource in energy production. *Energy Policy* 37, 4303–4312.

Rothausen, S.G., Conway, D., 2011. Greenhouse-gas emissions from energy use in the water sector. *Nat. Clim. Change* 1, 210–219.

Schnoor, J.L., 2011. Water-energy nexus. *Environ. Sci. Technol.* 45, 5065.

Scott, C.A., Pierce, S.A., Pasqualetti, M.J., Jones, A.L., Montz, B.E., Hoover, J.H., 2011. Policy and institutional dimensions of the water-energy nexus. *Energy Policy* 39, 6622–6630.

Scown, C.D., Horvath, A., McKone, T.E., 2011. Water footprint of U.S. transportation fuels. *Environ. Sci. Technol.* 45, 2541–2553.

Shao, L., Chen, G., 2013. Water footprint assessment for wastewater treatment: method, indicator, and application. *Environ. Sci. Technol.* 47, 7787–7794.

Siddiqi, A., Anadon, L.D., 2011. The water-energy nexus in Middle East and North Africa. *Energy Policy* 39, 4529–4540.

Stokes, J., Horvath, A., 2006. Life cycle energy assessment of alternative water supply systems. *Int J Life Cycle Assessment* 11, 335–343.



- Stokes, J.R., Horvath, A., 2009. Energy and air emission effects of water supply. *Environ. Sci. Technol.* 43, 2680–2687.
- U.S. Department of Energy, 2014. The Water-Energy Nexus: Challenges and Opportunities, <http://energy.gov/sites/prod/files/2014/07/f17/Water%20Energy%20Nexus%20Full%20Report%20July%202014.pdf>
- Walker, M.E., Lv, Z., Masanet, E., 2013. Industrial steam systems and the energy-water nexus. *Environ. Sci. Technol.* 47, 13060–13067.
- Water in the west, 2013. Water and Energy Nexus: A Literature Review, <http://waterinthewest.stanford.edu/sites/default/files/Water-Energy.Lit.Review.pdf>
- Wiedmann, T., 2009. A first empirical comparison of energy Footprints embodied in trade—MRIO versus PLUM. *Ecol. Econ.* 68, 1975–1990.
- Wiedmann, T., Minx, J., 2008. A definition of 'carbon footprint'. *Ecol. Econ. Res. Trends* 1, 1–11.
- Wu, S., Austin, D., Liu, L., Dong, R., 2011. Performance of integrated household constructed wetland for domestic wastewater treatment in rural areas. *Ecol. Eng.* 37, 948–954.
- Zhang, C., Anadon, L.D., 2013. Life cycle water use of energy production and its environmental impacts in China. *Environ. Sci. Technol.* 47, 14459–14467.
- Zhang, T., Xie, X., Huang, Z., 2014. Life cycle water footprints of nonfood biomass fuels in China. *Environ. Sci. Technol.* 48, 4137–4144.