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Unit Energy Consumption, Production, and Cost of Innovative Treatment Systems of Different Wastewater Streams

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UNIT ENERGY CONSUMPTION, PRODUCTION, AND COST OF INNOVATIVE TREATMENT SYSTEMS FOR DIFFERENT WASTEWATER STREAMS

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Jinze Li

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To: Dean John L. Volakis
College of Engineering and Computing

This dissertation, written by Jinze Li, and entitled Unit Energy Consumption, Production, and Cost of Innovative Treatment Systems for Different Wastewater Streams, having been approved in respect to style and intellectual content, is referred to you for judgment.

We have read this dissertation and recommend that it be approved.

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ABSTRACT OF THE DISSERTATION

UNIT ENERGY CONSUMPTION, PRODUCTION, AND COST OF INNOVATIVE TREATMENT SYSTEMS FOR DIFFERENT WASTEWATER STREAMS

by

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Innovative technologies such as micro-sieving, Anammox, and up-flow anaerobic sludge blanket (UASB) hold the key in the sustainable design of Water Resource Recovery Facility (WRRF). In the past, assessment metrics on the effectiveness and economic feasibility of these technologies have not been systematically investigated. According to the twelve design principles of Sustainable Environmental Engineering, Unit energy and cost metrics could provide universal benchmarks in the design of WRRF. Therefore, the objectives of this study are to design innovative WRRF systems to achieve energy positive. These WRRFs were modeled by developing an Excel model to estimate the unit energy metrics. Database of different wastewater quality was developed according to literature data. An Excel model was also developed to estimate the cost due to the energy saving of innovative systems.

In treating young, medium, and old leachate, systems with the innovative technologies could save the unit energy consumption by 2.24-4.07 kWh/kg $N_{removed}$ and the unit cost by \$0.86-2.09/kg $COD_{removed}$ than conventional technologies. Treatment of young leachate costs less than other leachate in terms of per kg COD removed. Although micro-sieving

decreases CH_4 production, it reduces the size of the UASB. As a result, micro-sieving could reduce the unit cost by 27% compared with systems without primary treatment. The major saving was contributed by UASB which converts BOD to CH_4. In addition, partial nitrification/anammox (PN/A) consumes less oxygen in removing nitrogen, which helps food processing treatment system achieve energy positive. In treatment of meat processing wastewater, tannery wastewater, and textile wastewater, the mean unit energy consumptions in innovative systems were 1.49, 1.37, and 1.39 kWh/kg $N_{removed}$. Mean unit energy consumption is close to the unit energy consumption of PN/A. The average unit costs for three types of industrial wastewater are 0.54, 0.57, and 1.12 \$/kg $COD_{removed}$, respectively. Therefore, meat processing wastewater can be the most efficiently treated by using innovative technologies due to its high biodegradability.

For WWTPs in China, anaerobic-oxic plus anaerobic-anoxic-oxic, oxidation ditch, and sequencing batch reactor were the main technologies. Due to lower energy consumption, SBR is the best technology in small and medium WWTPs in China.

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LIST OF ABBREVIATIONS

AAO	Anaerobic- Anoxic–Oxic
AD	Anaerobic Digester
AGMBR	Aerobic Granulation Membrane Bioreactor
AO	Anaerobic-Oxic
AO + AAO	Anaerobic-Oxic Plus Anaerobic-Anoxic-Oxic
AOBs	Ammonia Oxidizing Bacteria
AS	Activated Sludge
BOD	Biochemical Oxygen Demand
bpBOD	Biodegradable Particulate BOD
CAS	Conventional Activated Sludge
CH_4	Methane
CHP	Combined Heat and Power
CO_2	Carbon Dioxide
COD	Chemical Oxygen Demand
E&I	Electrical and instrumentation equipment
EPA	Environmental Protection Agency
GWh	Gigawatt hours
HDPE	High Density Polyethylene
HRT	Hydraulic Retention Time
MBBR	Moving Bed Biofilm Reactor
MBR	Membrane Bioreactor

MGD	Million Gallons Per Day
MLSS	Mixed Liquor Suspended Solids
N	Nitrogen
N_2	Nitrogen Gas
nbCOD	Nonbiodegradable COD
nbpCOD	Nonbiodegradable Particulate COD
nbsCOD	Nonbiodegradable Soluble COD
NH_4^+_N	Ammonium-Nitrogen
NO_2^-	Nitrite
NO_2_N	Nitrite-Nitrogen
NO_3_N	Nitrate-Nitrogen
NOBs	Nitrite Oxidizing Bacteria
O & M cost	Operation and Maintenance Cost
O_2	Oxygen
OD	Oxidation Ditch
P.E.	Population Equivalents
pbCOD	Particulate Biodegradable Organic Nitrogen
PC	Primary Clarifier
PN/A	Partial Nitrification/Anammox
PN/A	Nitrification/Heterotrophic Denitrification
RBF	Rotating Belt Filter
sBOD	Soluble BOD

sbON	Soluble Biodegradable Organic Nitrogen
SBR	Sequencing Batch Reactor
sCOD	Soluble Chemical Oxygen Demand
SPSS	Statistical Package for The Social Sciences
SRT	Sludge Retention Times
TN	Total Nitrogen
TOC	Total organic carbon
TSS	Total Suspended Solids
kWh	Kilowatt hours
UASB	Up-Flow Anaerobic Sludge Blanket
WRRF	Water Resources Recovery Facilities
WWTPs	Wastewater Treatment Plants

1. INTRODUCTION

1.1. Statement of problem

Most wastewater treatment plants applied energy-intensive aerobic activated sludge processes [1, 2]. According to statistics, WWTPs in China consumed electricity of 10,000 GWh in 2011, which made up for 0.2 % of the total industrial electricity consumption [3]. From 2011 to2017, the water supply and wastewater treatment accounted for approximately 0.67-0.75% of the total industrial energy consumption in China [4-10]. Energy consumption of electricity in the water production and supply industry increased by the rate of 5.7-10.7% per year in China. Increasing number and capacity of WWTPs to meet fast urbanization results in increased greenhouse gas production and resource consumption. Therefore, energy consumption has become a critical issue in the operation of WWTPs [11-13].

After vigorous research and development in the last two decades, innovative technologies such as micro-sieving, mainstream up-flow anaerobic sludge blanket, and partial nitrification/Anammox became effective, efficient, and practicable technologies in reducing energy consumption in WWTPs. These innovative technologies could be used either in retrofitting WWTPs to recover energy and nutrients from the wastewater or designing a new Water Resources Recovery Facility (WRRF). Properly adopted, they could turn conventional WWTPs from energy negative to energy-positive facilities. In literature, research focused on comparing the energy consumption and the demand of wastewater treatment systems based on specific wastewater quality. However, wastewater could be classified as leachate, food processing wastewater, industrial wastewater, and municipal wastewater. To my knowledge, there has never been a study that systematically

assesses innovative technologies as key processes in designing energy-positive WWTPs or WRRF. The effectiveness, efficiency, and economic feasibility of these technologies have never been evaluated in terms of unit energy consumption, production as well as operation cost per unit mass of pollutants removed for the different types of wastewater. This research attempts to apply the design metrics of Sustainable Environmental Engineering and to quantitatively evaluate innovative systems to achieve an energy positive design [14].

1.2. Literature review

1.2.1. Micro-sieving

Micro-sieving, or a rotating belt filter (RBF), offers more sustainable alternative than a traditional primary clarifier with a smaller footprint and lower capital costs [15, 16]. Since the land requirement of RBF is typically 10% of that of a primary clarifier [17], the capital cost of primary treatment could be saved by approximately 50% with RBFs compared with the conventional primary clarifier (PC) [18]. Typically, RBF can divert at least 50% of total suspended solids and 20% of biochemical oxygen demand from the main treatment stream to a mesophilic anaerobic digester [15, 19]. As a result, micro-sieving decreases the amount of oxygen needed for aeration by reducing the organic loading rate to aerobic biological treatment processes such as activated sludge. In addition, it increases biogas production for energy recovery because major chemical oxygen demand is diverted directly to anaerobic digesters in the form of primary sludge.

1.2.2. Up-Flow Anaerobic Sludge Blanket (UASB)

Mainstream UASB is the key technology in converting a wastewater treatment plant from energy consumption to energy production. In the treatment scheme, biodegradable COD in wastewater is converted into methane at an operating temperature between 15 and

35℃ [20]. Mainstream UASB eliminates the aeration process required during the removal of dissolved biodegradable COD, produces energy in the form of CH_4, and generates a much smaller amount of sludge to be disposed of [21]. These advantages reduce the operational cost when compared with conventional-wastewater treatment systems [22, 23].

1.2.3. Partial Nitrification/Ammonia (PN/A)

Developed in the beginning of the 1990s, the PN/A process is a cost-effective way to remove total nitrogen in wastewater [24]. PN/A process is a two-step biological transformation of ammonia to nitrogen gas. The first step is termed as partial nitrification. Ammonia oxidizing bacteria convert approximately 50% of the ammonia to nitrite under conditions that restrict nitrite-oxidizing bacteria growth. The second step is referred to as anaerobic ammonium oxidation or Anammox. Ammonium-Nitrogen is oxidized under anaerobic conditions by anaerobic ammonium oxidation bacteria that can use nitrite as the electron acceptor to produce N_2 [25]. Compared to the nitrification-denitrification process, PN/A process could reduce 60% of oxygen demand in the aeration process and eliminate organic carbon source requirements.

2. OBJECTIVES AND APPROACH

2.1. Objectives

The objectives of this dissertation are to assess the efficiency and economic feasibility of innovative technologies in treating different streams of wastewater. This is accomplished by characterizing the influence of industrial wastewater and leachate on the wastewater treatment system. Industrial wastewater and leachate have different concentrations of pollutants and biodegradability, which significantly affect the performance of micro-sieving, UASB, and PN/A. Besides wastewater quality, parameters such as treatment technologies, design capacity, operation loading rate, and COD removal efficiency also have a significant influence on the energy consumption or production of WWTPs. Therefore, the objectives are as follows:

1) To evaluate the effects of primary treatment of micro-sieving vs. traditional primary clarifier and secondary treatment (biological process) in leachate treatment and analyze the unit energy and cost of different leachate treatment systems with innovative technologies such as Anammox and UFAB.

2) To evaluate the influence of micro-sieving on the treatment performance of the system with UASB and PN/A and compare sustainable assessment metrics of treatment systems using micro-sieving in terms of unit energy and cost.

3) To compare the unit energy consumption and production of different industrial wastewater treatments and conduct a unit cost analysis of the system based on various industrial wastewater qualities.

4) To compare the unit energy consumption of different treatment technologies in WWTPs in China and study relationships between the unit energy consumption and design

capacity, operation loading rate, COD removal efficiency, and influent concentration of contaminants.

2.2. Materials and methods

Data concerning industrial wastewater, leachate at various landfill ages, and municipal wastewater are from peer-reviewed papers. Municipal wastewater flow rate, influent pollutant concentration, and treatment technologies were compiled from the China Urban Drainage Statistical Yearbook 2016. Innovative technologies can save energy and reduce carbon sources for removing COD and TN in WWTPs. However, it requires investment in innovative technologies and increases capital cost. On the other hand, the O&M costs could be significantly reduced due to the increased heat and electricity harvested by the new innovative technologies as well as sustainable engineering design. For quantifying the impacts of innovative technologies and design, it is necessary to determine if the cumulative benefit of energy-saving would be greater than increased capital cost and the reduced cumulative operation and maintenance (O&M) cost by the innovative technologies in the life cycle of the equipment. Therefore, unit energy and cost analysis are used to compare the economic feasibility of conventional methods and innovative technologies for treating different kinds of wastewater. For designing an energy positive WRRF, data was compiled from the peer-reviewed papers [26-28]. Excel 2010 and IBM Statistical Package for the Social Sciences (SPSS) Statistics 21.0 software were used to estimate energy and cost for different wastewater treatment systems. The effect of wastewater quality, flow rate, operation loading rate, treatment technologies on energy consumption was quantified by using these models. The current study collected different wastewater quality data and then an Excel-based model was developed. To reduce the space of the section, the performance

of treatment systems with different technologies is shown in Appendix. Figure A.18, A.19, A.20, A.21, A.22, A.23, A.24 illustrate these results.

The unit energy consumption (kWh/kg $COD_{removed}$) of different treatment processes in China were calculated based on design capacity, operation loading rate, COD removal, influent and effluent concentration of contaminants, and energy consumption in Excel as shown in Figure A.25. The unit energy consumption, production, and cost of leachate, food processing wastewater, and industrial wastewater system were estimated by the following equations [17, 26, 29-39]:

$$\text{Energy production(kWh/d)} = \text{(Energy recovery) (Enthalpy of combustion)}$$
$$(CH_4 \text{ production from UASB and anaerobic digester (kg/d)})$$

where
Energy recovery = 38%
Enthalpy of combustion = 13.9 kWh/kg CH_4

$$\text{Unit energy production(kWh/kg } BOD_{removed}) = \frac{\text{Energy production(kWh/d)}}{\text{BOD removal (kg/d)}} \quad (2.1)$$

$$\text{Energy consumption} = \frac{\text{Aeration demand of aeration process (kg}O_2\text{/d)}}{OT_a}$$

where

OT_a =Oxygen transfer efficiency (actual)=1.2 kg O_2 /kWh

$$\text{Unit energy consumption (kWh/ kg } N_{removed}) = \frac{\text{Energy consumption (kWh/d)}}{\text{N removal (kg/d)}} \quad (2.2)$$

It was assumed that UASB combining the heat and power technologies could convert 38% of CH_4 (formed in the mainstream and the side stream anaerobic digestion) to electricity with an energy density of approximately 13.9 kWh/kg CH_4 [40]. The energy demand of a WWTP mainly includes the power consumption of the wastewater lift pump, aeration equipment, and sludge treatment. The energy consumption of biological treatment accounts for 50–70% of the overall energy consumption [41]. Therefore, the aeration energy of the biological treatment process was the dominant energy demand. Since biodegradable COD in wastewater is converted into CH_4 at 15 and 35 ℃ [20], the operating temperature of the treatment system was assumed to be 25 ℃. The unit energy consumption is estimated based on O_2 mass demand and oxygen transfer efficiency at 25 ℃ [26].

Capital cost of preliminary treatment($)= EXP [3.25972 +0.61915 ln(flow to the treatment plant(MGD))]1000

Operation labor hour of preliminary treatment (hr/yr) = EXP [6.39872 + 0.23096 x + 0.16496 x^2 - 0.0146 x^3]

Maintenance labor hour of preliminary treatment (hr/yr) = EXP [5.8461+ 0.20651 x + 0.06884 x^2 + 0.02382 x^3 - 0.00441 x^4]

Total material and supply cost of preliminary treatment ($/yr) =EXP [7.23566 + 0.39994 x - 0.22498 x^2 + 0.1101 x^3 - 0.01103 x^4]

Electrical energy demand of grit removal (kwh/yr) = EXP [6.30864 + 0.23453 x - 0.35844 x^2 + 0.00871x^3]

Electrical energy demand of flow measurements and screening (kwh/yr) = EXP [7.1497 + 0.28856 x - 0.07886 x^2 + 0.014662 x^3]

The total electrical energy demand of preliminary treatment (kWh/yr) = Electrical energy demand of grit removal (kWh/yr) + Electrical energy demand of flow measurements and screening (kWh/yr)

where

x=In(Q)

Q=flow to the treatment plant, MGD

(2.3)

Capital cost of rotating belt filter($)= 0.55Capital cost of priamry clarifier with 50% TSS removal($)

Capital cost of priamry clarifier=$-0.00002(Q)^2+19.29(Q)+220,389$

Q=32.6/surface overflow rate (influent flow rate)

Operating Power Consumption of rotating belt filter =129.6 kWh/d

Maintenance Materials cost of rotating belt filter ($/yr) = 2% of capital costs of rotating belt filter

(2.4)

TSS removal of primary clarifier (%) =-0.004006[Surface overflow rate(m/d)] +0.73782

TSS removal of rotating belt filter (%) =16.45ln [Influent TSS concentration(mg/L)]-29.1

(2.5)

Capital and O&M costs of preliminary and primary treatment are related to the flow rate. To quantify the influence of different wastewater quality on treatment systems, 1 MGD was assumed as the flow rate for different wastewater treatment systems. A rotating belt filter has a typical operating power consumption of 1.8-3.6 kW for a flow rate of 0.6 MGD [17]; two rotating belt filters with total energy consumption of 5.4 kW were used for

treating wastewater. Since micro-sieving could reduce the capital cost by 30%-60% when it is compared with conventional primary clarifier, it is assumed that the capital cost of micro-sieving is 55% of that of primary clarifier with 50% TSS removal according to the literature data. Maintenance cost is equal to 2% of capital cost [1]. Since the cost of the primary clarifier is calculated based on the same flow rate, the costs of rotating belt filters are the same for different wastewater treatment systems. However, TSS removal efficiency is positively correlated to the TSS concentration before the micro-sieving process [30]. Different wastewater quality leads to different treatment performance of micro-sieving.

The capital cost of UASB($)=(cost factors)(Volume of UASB (m^3))

(unit capital cost of UASB($/$m^3$))

Volume of UASB (m^3)=(sludge production from UASB(m^3/d))(sludge age(d))

where

sludge age=60 d for the leachate treatment system

sludge age=40 d for food processing and industrial wastewater treatment system

Cost factors=(f_i)(f_{ac})

f_{ac}= factor for costs for construction of additional units=1.43

f_i= factor for additional capital costs (non-construction related) =1.72

unit capital cost of UASB=721.31 $/$m^3$ volume

(2.6)

Cavalcanti et al. [42] studied the performance of UASB reactors for treating raw wastewater. The results show that UASB could produce stable sludge at an anaerobic sludge age of 40 to 50 days at 25 ℃. In the municipal wastewater treatment system, 40 to 60% of the influent COD could be converted into CH_4 by UASB reactors at an anaerobic

sludge age of 30 to 60 days at 15 to 30 ℃. In practice, the sludge age of UASB is greater than 20 to 40 days [26]. Old leachate has very low biodegradability, sludge age is assumed to be 60 days and 40 days for keeping stable performance of leachate and industrial wastewater treatment systems. When the flow rate of different wastewater treatment systems is 1 MGD or 3785.4 m^3/d, f_{ac}, f_i, and unit capital cost of UASB are 1.43, 1.72, and 721.31 $/$m^3$ volume, which were calculated based on equations (A.6, A.7, and A.1).

Capital cost of PN/A($) =Equipment Cost of PN/A($)(Capital cost/Equipment Cost)

where

Capital cost/Equipment Cost=2.405

Equipment Cost of PN/A($) =(Volume of PN/A(MG))166438+155108

(2.7)

In the current study, the PN/A process was operated in a sequencing batch reactor (SBR). SBR with different volumes and corresponding equipment costs were summarized by the U.S. Environmental Protection Agency report [32]. Therefore, the equipment cost of PN/A SBR was estimated based on reactor volume, as shown in figure A.19 and equation (A.9). Capital cost includes equipment and construction costs. Cost of piping, installation, electrical, instrumentation, engineering, construction management (structural and civil) accounts for 85% of equipment cost [29]. Due to contingency, the capital cost is 30% higher than the designed capital cost [29]. Therefore, the ratio between capital cost and equipment cost is 2.405.

Capital cost of activated sludge process with nitrification and denitrification($)

=(costing factors)(Volume of activated sludge process with nitrification and denitrification (m^3)) (unit capital cost of activated sludge tank ($/$m^3$))

$= f_{ac}\ f_i\ V_{\text{activated sludge process with nitrification and denitrification}}$(Unit capital cost of activated sludge tank)

where

Cost factors=$(f_i)(f_{ac})$

f_{ac}= factor for costs for construction of additional units=1.43

f_i= factor for additional capital costs (non-construction related) =1.72

unit capital cost of activated sludge tank =244.64 \$/m^3 volume

(2.8)

If the flow rate of different wastewater treatment systems is 1 MGD or 3785.4 m^3/d, f_{ac}, f_i, the unit capital cost of the activated sludge tanks are 1.43, 1.72, and 244.64 \$/m^3 volume, which were calculated based on equations (A.6, A.7, and A.2).

Capital cost of final clarifier(\$)=(cost factors)(flow rate to final clarifier(m^3/d))(S_{fd} (H_d/v_0)EXP (kX_t)) (unit capital cost of final clarifier(\$/m^3))

where

S_{fd}=settler safety factor=2

H_d=settler height (m)=4m

v_0= vesilind constant=216 m/d

k= vesilind constant=0.35 L/gTSS

X_t=activated sludge concentration=4.006 g/L

Cost factors=$(f_i)(f_{ac})$

f_{ac}= factor for costs for construction of additional units=1.43

f_i= factor for additional capital costs (non-construction related) =1.72

unit capital cost of final clarifier=412.40 \$/m^3 volume

(2.9)

van Haandel and van der Lubbe [26] used 2 and 4 m as the default values for the safety factor and height of the final clarifier. Thus, these default values were selected for estimating the cost of the final clarifier. Sludge settleability affects the value of the Vesilind constants [26]. Without UASB, sludge to the final clarifier shows poor settleability. The Vesilind constants v_0 and k are assumed to be 144 m/d and 0.46 L/ g TSS. Since the settleability of sludge following mainstream UASB is good, Vesilind constants v_0 and k are 216 m/d and 0.35 L/ g TSS. As shown in table A.4, solid content and the specific gravity of the bulk sludge for activated sludge are 0.2-0.6% and 1-1.003. 0.4% and 1.0015 were assumed to be the corresponding solid content and the specific gravity of the bulk sludge. Therefore, the wet bulk sludge concentration of activated sludge equals 4.006 g/L. When the flow rate of different wastewater treatment systems is 1 MGD or 3785.4 m^3/d, f_{ac}, f_i, the unit capital cost of final clarifier are 1.43, 1.72, and 412.40 \$/m^3 volume, which were calculated based on equations (A.6, A.7, and A.3).

Capital cost of anaerobic digestion (\$)

=(costing factors)(Volume of anaerobic digestion(m^3))(unit capital cost of anaerobic digestion (\$/m^3))

$= f_{ac}\, f_i V_{\text{sludge thickener}}$(Unit capital cost of anaerobic digestion)

where

Cost factors=(f_i)(f_{ac})

f_{ac}= factor for costs for construction of additional units=1.43

f_i= factor for additional capital costs (non-construction related) =1.72

unit capital cost of anaerobic digester=700.46 \$/m^3 volume

(2.10)

Since combined heat and power could convert the CH_4 formed in UASB and anaerobic digestion to heat for operating anaerobic digester at high temperature, operation temperature is assumed to be 35 ℃ in the anaerobic digester. Whene the flow rate of different wastewater treatment systems is 1 MGD or 3785.4 m^3/d, f_{ac}, f_i, and unit capital cost of anaerobic digester are 1.43, 1.72, and 700.46 $\$/m^3$ volume, respectively, which are calculated based on equations A.6, A.7, and A.4.

Capital cost of Package Granular Activated Carbon Columns($)=(1+(Miscellaneous costs)/(construction cost))(0.0084 Q^3–5.2233Q^2+1318.4Q+27144)

Capital cost of Granular Carbon Regeneration ($)=(1+(Miscellaneous costs)/(construction cost))(2E–8x^3 – 0.0018 x^2 + 93.965 x +2000000)

O&M cost of Package Granular Activated Carbon Columns($/yr)=(0.0005 Q^3–0.3763 Q^2+140.13Q+4959)

O&M cost of Granular Carbon Regeneration ($/yr)=15.503 x + 128481

O&M cost of virgin Carbon ($/yr)=(Activated carbon loss)(365d/y)(Cost of virgin carbon) x

where

Q = plant flow (gpm)

x = regeneration capacity of activated Carbon (lb/day)

(Miscellaneous costs)/(construction cost)=0.28

Activated carbon loss=6%

Cost of virgin carbon=0.95$/lb

(2.11)

Sharma [36] summarized the capital and the O&M cost of granular activated carbon columns, while the cost of granular carbon regeneration was calculated based on the flow rate and regeneration capacity of activated carbon. Capital cost includes construction cost and miscellaneous costs such as those for site works, overhead, profit, administration, and interest, which are equal to 28% of construction cost [36]. On average, the price of virgin carbon was \$0.70/lb-\$1.20/lb, while 5% to 7% of Granular Activated Carbon is lost during normal operation [43]. Thus, the cost of virgin carbon and activated carbon loss were assumed to be 0.95\$/lb and 6%, respectively.

Capital cost of aeration equipment(\$)=(cost factors)(Energy consumption of aeration process (kWh/d)) (unit capital cost of aeration equipment(\$/kW))/(24h/d)

where

Cost factors=$(f_i)(f_{ac})$

f_{ac}= factor for costs for construction of additional units=1.43

f_i= factor for additional capital costs (non-construction related) =1.72

unit capital cost of aeration equipment=5276.43 \$/kw

(2.12)

Since the flow rate of different wastewater treatment systems is 1 MGD or 3785.4 m^3/d, f_{ac}, f_i, the unit capital cost of aeration equipment are 1.43, 1.72, and 5276.43 \$/kW, which were calculated based on equations A.6, A.7, and A.5.

O&M cost (\$/yr)=$(p+o+n+ 0.3m_{civ}+0.7\ m_{me\&i})$Capital cost

where

p= personnel cost =3.5 % of capital cost/yr

o= operation cost =1 % of capital cost/yr

m_{civ}=maintenance cost for civil works=0.75 % of capital cost/yr

$m_{me\&i}$ =maintenance cost for mechanical, electrical and instrumentation equipmen=1.75 % of capital cost/yr

n=insurance cost=0.3% of capital cost/yr

(2.13)

The operation and maintenance costs (O&M costs) include items such as cost for personnel, operation, insurance, and maintenance, which are taken as percentages of the capital costs for cost estimation. Personnel, operation, insurance, and maintenance costs for civil works and for mechanical, electrical and instrumentation equipment (E&I) are equal to 2-5%, 0.5-1.5%, 0.2-0.4%, 0.5-1%, and 1-2.5% of the capital cost, respectively. 30% and 70% were used as the percentages of maintenance cost for civil works and for mechanical and E&I [26]. Therefore, it was assumed that the personnel, operation, insurance, and maintenance costs for civil works and for mechanical and E&I are 3.5%, 1%, 0.3%, 0.75%, and 1.75% of the capital cost, respectively.

O&M costs of sludge(US$/yr)=(sludge production from AD(ton TSS/yr))(sludge disposal cost($/ton TSS))

where

sludge disposal cost($/ton TSS)=280 $/ton TSS

(2.14)

On average, the price of sludge disposal is 60 $/ton TSS-500 $/ton TSS [26]. Thus, the cost of sludge disposal is assumed to be 280$/ ton TSS.

Capital cost($/yr)=Capital cost($)/a_(i, n)

$$\text{Capital cost(\$/yr)} = \frac{\text{Capital cost(\$)}}{a_{i,n}}$$

$$a_{i,n} = \frac{(1+i)^n - 1}{i(1+i)^n}$$

where

$a_{i,n}$=annualization factor

i =interest rate (annual)=6%

n=economic lifetime of the treatment plant in years=20 (yr)

$$\text{Unit capital cost (\$/kg COD}_{\text{removed}}) = \frac{\text{Capital cost(\$/yr)}(365\,\text{d/yr})}{\text{COD removal (kg/d)}} \tag{2.15}$$

$$\text{Unit O \& M cost (\$/kg COD}_{\text{removed}}) = \frac{\text{O \& M cost(\$/yr)}(365\,\text{d/yr})}{\text{COD removal (kg/d)}} \tag{2.16}$$

$$\text{Unit cost(\$/kg COD}_{\text{removed}}\text{)=Unit capital cost(\$/kg COD}_{\text{removed}}\text{)+Unit O\&M cost(\$/kg COD}_{\text{removed}}\text{)} \tag{2.17}$$

For comparing the costs of different treatment systems, the total capital cost ($) was annualized over the expected lifetime of the WWTP. Economic lifetime and interest rates were assumed to be 20 years and 6% [26]. Capital costs ($) were transformed into annualized capital costs ($/yr) based on the interest rate and lifetime. The total unit cost was calculated as the combination of the unit capital costs and the unit operational costs.

3. UNIT ENERGY AND COST COMPARISON OF TWO INNOVATIVE TREATMENT SYSTEMS OF YOUNG, MEDIUM AND OLD LEACHATES

3.1. Abstract

An Up-flow anaerobic sludge blanket reactor (UFAB) followed by a partial nitrification/anammox (PN/A) reactor was reported as one of the energy-positive systems in removing N from municipal wastewater. However, there are few cases of research assessing the efficiency and economic feasibility of this energy-positive process in treating young, medium, and old leachates with different treatment trains. The current study collected leachate quality data at different ages and developed an energy and cost estimation Excel model to compare systems using these innovative technologies with conventional leachate treatment. From energy and cost perspectives, two innovative biological processes could save conventional leachate treatment unit energy by 2.24-4.07 kWh/kg $N_{removed}$ and cost by \$0.86-2.09/kg $COD_{removed}$. Since the secondary treatment could only remove 10.2-37.3% of COD from old leachate, COD removal of old leachate mainly depends on the physical-chemical process rather than the biological process. Therefore, innovative biological technologies are more suitable for treating young leachate than old leachate. This study quantifies the cost benefits of the innovative technologies for treating leachate at different ages, which provides a valuable reference for future sustainable design of leachate treatment systems.

3.2. Introduction

Solid waste production increases with population and economic growth [44]. Landfilling is one of the common and important disposal methods of urban solid wastes worldwide [45]. In the United States and China, more than 53% and 80% of total solid

waste were disposed of in landfills, respectively [46-48]. Leachate, which is an unavoidable by-product in solid waste landfilling, contains a high concentration of organic matter and ammonia nitrogen [49, 50]. It would have a long negative effect on the aquatic environment if it were discharged directly into surface water and groundwater [51-53]. Therefore, landfill leachate as a unique wastewater pollution source must be properly treated before being discharged [45, 50, 54]. Leachate exhibits different compositions depending upon a variety of factors such as the quality and quantity of solid waste received, mode of operation, climatic conditions, depth of fill, and age of the landfill [55-58].

Landfilling age, which represents the degree of stabilization of the solids, is one of the most important parameters used in selecting a suitable leachate treatment method. Properties of leachate such as chemical oxygen demand (COD), biological oxygen demand (BOD), biodegradability, the content of humic substances, and ammonium nitrogen (NH_4^+_N) dramatically change during solid waste decomposition in landfills [59-61]. Based on age, landfill leachates are commonly divided into young, medium, and old groups [59, 62-64]. Young leachate is commonly characterized by a high level of COD, high ratio of BOD/COD (ranging from 0.5 to 1), and a relatively low concentration of NH_4^+_N [64-67]. The features of old leachate are relatively low COD concentrations, little biodegradability (BOD/COD≈0.1), high content of non-biodegradable soluble organic matter in the form of humic substances, high ammonia-nitrogen concentration, and low carbon-to-nitrogen ratio [64, 67-71]. The characteristics of medium leachate are somewhere between those values of the young and the old [72, 73].

Since leachate at various ages have a high concentration of NH_4^+_N, COD, and different BOD/COD ratios, conventional treatment systems include primary physical

treatment, biological processes, and physical-chemical treatmentunits to meet the discharge requirements. Among these processes, biological processes were mostly applied for removing nitrogen and BOD from leachate [74-79]. For example, the activated sludge system with an autotrophic nitrification/heterotrophic denitrification process was commonly applied [50, 80]. Nitrification in a conventional activated sludge system converts most of the organic matters and ammonia nitrogen in leachate to CO_2 and NO_3^--N under aerobic condition, while denitrification converts NO_3^- to N_2 with carbon source under anaerobic or anoxic conditions. However, nitrification and denitrification are relatively expensive in removing nitrogen, because it commonly requires external carbon sources, consumes a lot of energy for aeration, and produces large amounts of sludge [67].

Compared to the conventional methods, innovative technologies such as mainstream up-flow anaerobic sludge blanket (UASB) and partial nitrification/Anammox (PN/A) are considered as sustainable technologies. Mainstream UASB is the key technology in converting a wastewater treatment plant from energy consumption to energy production. In the treatment scheme, biodegradable COD in wastewater is converted into CH_4 at the operating temperature between 15 and 35℃ [20]. Mainstream UASB eliminates the aeration process required during the removal of dissolved biodegradable COD, produces energy in the form of CH_4, and generates a much smaller amount of sludge to be disposed [21]. These advantages reduce operational costs when compared with conventional wastewater treatment [22, 23].

PN/A discovered in the 1990s is a two-step biological transformation of ammonia to nitrogen gas (N_2) [24, 81, 82]. During the partial nitrification step, 50-60% of ammonia in the influent is oxidized to nitrite [83, 84]. The remaining ammonia is subsequently

denitrified with nitrite to form N_2 and small amounts of nitrate in the Anammox step [85, 86]. In general, the PN/A process could save oxygen demand in nitrification by approximately 60%, eliminate external organic carbon source requirement for denitrification, and reduce sludge by 80-90% as compared to conventional nitrogen removal process [87-89]. As a result, mainstream UASB and partial nitrification/Anammox are considered as the most sustainable technologies to remove nitrogen. Garrido et al. [90] made mass and energy balances analysis of three kinds of N-removal systems and demonstrated that the use of the mainstream anammox process offered an opportunity to recover the electricity. McCarty [91] compared O_2 consumption, CH_4 production, and biosolid production of different N-removal systems and mainstream anaerobic treatment followed by mainstream anammox treatment. It has been shown conceptually to be the best energy-positive system for removing N. This research focused on comparing energy consumption and demand of wastewater treatment systems based on assumed wastewater quality using average values of reported wastewater quality. However, there are few cases of research assessing the efficiency and economic feasibility of this kind of energy positive process in treating young, medium, and old leachates with different compositions. Therefore, the current study aims to assess the leachate treatment system using UASB and PN/A from the energy and cost dimensions. The objectives of this study are to (1) evaluate the efficiency of secondary treatment (biological process) and tertiary treatment (physical-chemical process) for treating leachate at various landfill ages; (2) compare a system using innovative technologies with a conventional leachate treatment system in terms of unit energy consumption and production; (3) analyze and compare the unit cost of different leachate treatment systems. The ultimate goal of this study is to

quantify the cost-benefit of the innovative technologies for treating leachate at different ages and to offer an innovative design approach for future sustainable treatment of leachate.

3.3. Material and methods

In the current study, unit energy and cost were calculated based on different types of leachate quality. Leachate quality data was collected from published peer-reviewed papers, as shown in Table 3.1. The noticeable characteristics of young leachate samples were slightly high concentrations of suspended solids, BOD, COD, and BOD/COD ratio above 0.5 as shown in Table 3.1. The features of old leachate samples were considerably low BOD concentrations and biodegradability. An Excel-based model was developed in designing treatment systems of 1 MGD leachate using different innovative technologies [26-28]. To compare treatment performance and cost of the systems using UASB and PN/A technologies against a conventional leachate treatment system in treating leachate at different ages, the contaminant removal efficiency was modeled according to the influent leachate quality and effluent discharge requirements. Process flow diagrams of leachate treatment systems A and B are illustrated in Figure 3.1 to 3.2. To meet discharge requirements and compare the performance of system A and B in terms of unit energy and cost metrics, systems A and B were designed to achieve the same discharge standard (COD≤50mg/L, BOD≤20mg/L, and NH_4^+_N≤4.9mg/L) [61, 92, 93], so that systems A and B have similar COD and N removal efficiency. In both systems, a screen, grit chamber, rotating belt filter, and granular activated carbon system were used for preliminary treatment, primary treatment, and tertiary treatment for removing particulate matter and non-biodegradable soluble organic matter from leachate. The difference of the two systems is the secondary treatment processes. Specifically, system A uses the nitrification-

denitrification process as the main process for treating leachate to meet the discharge requirement by converting biodegradable COD and N to CO_2 and N_2. System B uses mainstream UASB and PN/A as the main systems for removing soluble BOD and N, which reduces aeration consumption through decreasing BOD and N loading to the nitrification-denitrification system.

The unit energy consumption, production, and cost are estimated by the following equations [20, 26, 40]:

$$\text{Energy production(kWh/d)} = (\text{Energy recovery})\,(\text{Enthalpy of combustion})\,(\text{CH}_4 \text{ production from UASB and anaerobic digester (kg/d)})$$

where

Energy recovery = 38%

Enthalpy of combustion = 13.9 kWh/kg CH_4

$$\text{Unit energy production(kWh/kg BOD}_{\text{removed}}) = \frac{\text{Energy production(kWh/d)}}{\text{BOD removal (kg/d)}} \tag{3.1}$$

$$\text{Energy consumption} = \frac{\text{Aeration demand of aeration process (kgO}_2\text{/d)}}{\text{OT}_a}$$

where

OT_a =Oxygen transfer efficiency (actual)=1.2 kg O_2 /kWh

$$\text{Unit energy consumption (kWh/ kg N}_{\text{removed}}) = \frac{\text{Energy consumption (kWh/d)}}{\text{N removal (kg/d)}} \tag{3.2}$$

It was assumed that UASB combining heat and power technology could convert 38% of CH_4 formed in the mainstream and the side stream anaerobic digestion to electricity with

an energy density of approximately 13.9 kWh/kg CH_4 [40]. The energy demand of a WWTP mainly includes power consumption of the wastewater lift pump, aeration equipment, and sludge treatment. The energy consumption of biological treatment accounts for 50–70% of the overall energy consumption [41]. Since UASB removes COD without consuming oxygen, the energy consumption of PN/A and nitrification–denitrification processes was the dominant energy demand. Biodegradable COD in wastewater were converted into CH_4 at 15 and 35℃ [20], the operating temperature of the treatment system was assumed to be 25 ℃. Therefore, energy consumption is estimated based on O_2 mass demand and oxygen transfer efficiency at 25 ℃ [26].

$$\text{Capital cost(\$/yr)} = \frac{\text{Capital cost(\$)}}{a_{i,n}}$$

$$a_{i,n} = \frac{(1+i)^n - 1}{i(1+i)^n}$$

where

$a_{i,n}$=annualization factor

i =interest rate (annual)=6%

n=economic lifetime of the treatment plant in years=20 (yr)

$$\text{Unit capital cost (\$/kg COD}_{\text{removed}}) = \frac{\text{Capital cost(\$/yr)}(365\,\text{d/yr})}{\text{COD removal (kg/d)}} \quad (3.3)$$

$$\text{Unit O \& M cost (\$/kg COD}_{\text{removed}}) = \frac{\text{O \& M cost(\$/yr)}(365\,\text{d/yr})}{\text{COD removal (kg/d)}} \quad (3.4)$$

$$\text{Unit cost}(\$/\text{kg COD}_{\text{removed}}) = \text{Unit capital cost}(\$/\text{kg COD}_{\text{removed}}) + \text{Unit O\&M cost}(\$/\text{kg COD}_{\text{removed}}) \quad (3.5)$$

For comparing the cost of treatment systems, the total capital costs ($) were annualized over the expected lifetime of the WWTP. Economic lifetime and interest rates were assumed to be 20 years and 6% [26]. Capital costs ($) were transformed into annualized capital costs ($/yr) based on the interest rate and lifetime. The total unit cost was calculated as the combination of the unit capital costs and the unit operational costs. Except for preliminary and primary treatment, the cost of main treatment units and aeration equipment is related to reactor volume and aeration energy consumption. These are calculated based on characteristics of leachate and have a significant impact on the treatment system design.

Table 3.1 Leachate quality at different ages of landfill sites

Ages	TSS (mg/L)	BOD (mg/L)	COD (mg/L)	TN (mg/L)	Reference
Young	1,894	4,979	9,310	1,172	[94]
	1,255	10,550	18,565	2,500	[95]
	2,400	10,800	24,400	1,766	[96]
	1,013	6,380	10,750	2,159	[97]
	1,540	9,660	18,420	2,517	[97]
	907	5,300	10,500	2,320	[98]
	724	4,680	9,890	2,225	[98]
Medium	154	2,307	8,938	2,550	[99]
	257	2,450	13,646	2,928	[99]
	143	2,684	5,348	2,192	[100]
	168	965	4,975	1,365	[101]
	393	1,957	6,638	2,072	[102]
	232	1,263	5,565	2,042	[103]
	595	5,550	12,554	2,102	[99]
Old	218	410	3,360	2,325	[104]
	30	372	3,105	1,937	[105]
	216	683	4,592	2,919	[106]
	542	420	3,580	1,889	[104]
	223	190	3,520	3,070	[98]
	430	1,375	5,200	3,147	[107]
	633	568	4,314	2,484	[104]

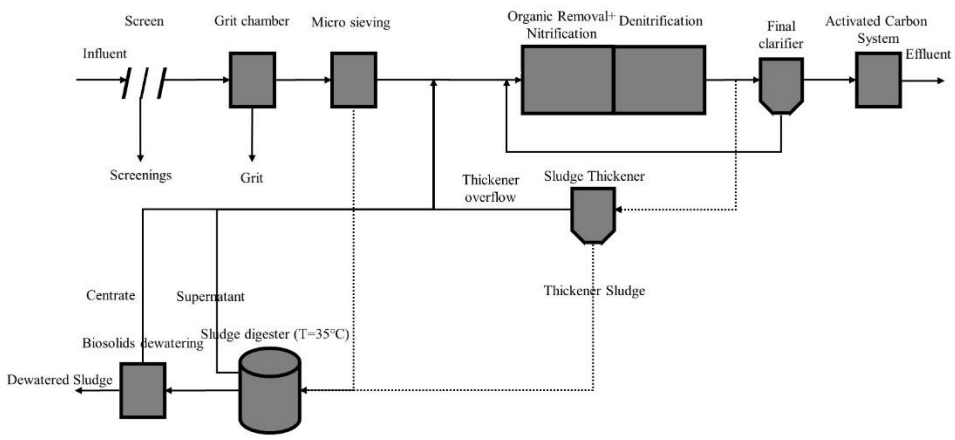

Figure 3.1 Schematic graphs of leachate treatment system A- direct line nitrification–denitrification process

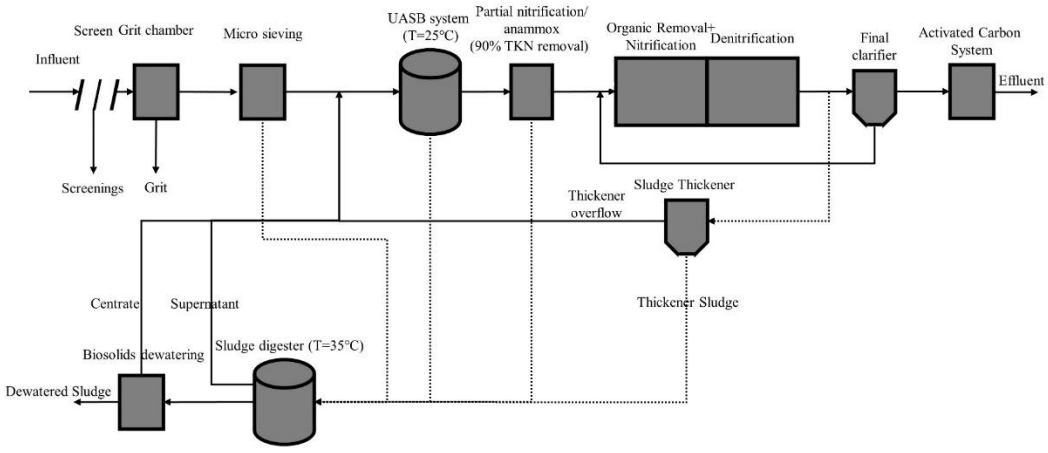

Figure 3.2 Schematic graphs of leachate treatment system B- mainstream anaerobic treatment and anammox treatment

3.4. Results and discussions

3.4.1. Treatment efficiency of systems A and B

Since systems A and B were designed to achieve the same effluent concentration of contaminant and similar COD removal efficiency, the points that represent COD removal of systems A and B are superimposed over each other in Figure 3.3. Figures 3.4 and 3.5 shows that COD removal rises with increasing biodegradability (the BOD/COD mass concentration ratio). COD is reduced by anaerobic and aerobic biological processes; the BOD/COD ratio represents the ability of a substance to be removed by microorganisms [108]. Amin et al. [109] reported that conventional biological technologies could achieve COD removal of 63% and 80% or higher when the BOD/COD ratio was 0.2-0.3 and above 0.4, respectively. COD removal predicted by the Excel model is similar to the results reported by Amin et al [109]. The ratio of BOD/COD varies greatly depending on the

landfill age [45]. There are abundant biodegradable organic compounds in the young landfill leachate. With the increase of the landfill age, the biodegradable substrates were depleted, and non-biodegradable organic matters such as humic substances were dominant in leachate composition [110-112]. Figures 3.3 and 3.4 confirms that the leachate biodegradability decreased with the age of the landfill, and the COD removal by biological processes is negatively correlated to the age of the landfills. Figure 3.5 shows that COD removal of young, medium and old leachate by biological processes are 57.1-77.9, 25.5-69.9, and 10.2-37.3%, respectively. After secondary treatment, humic substances, which are dominant in leachate composition, are adsorbed by the activated carbon system. Tertiary treatment needs to remove 13.5-32.6, 28.2-73.2, and 56.0-84.6 % of COD from the young, medium, and old leachate to achieve the same effluent COD concentration. Since the BOD/COD ratio decreases with rising age, COD removal of young leachate mainly depends on biological processes whereas old leachate mainly depends on physical-chemical methods. Therefore, UASB and PN/A are more suitable for treating young leachate than old leachate.

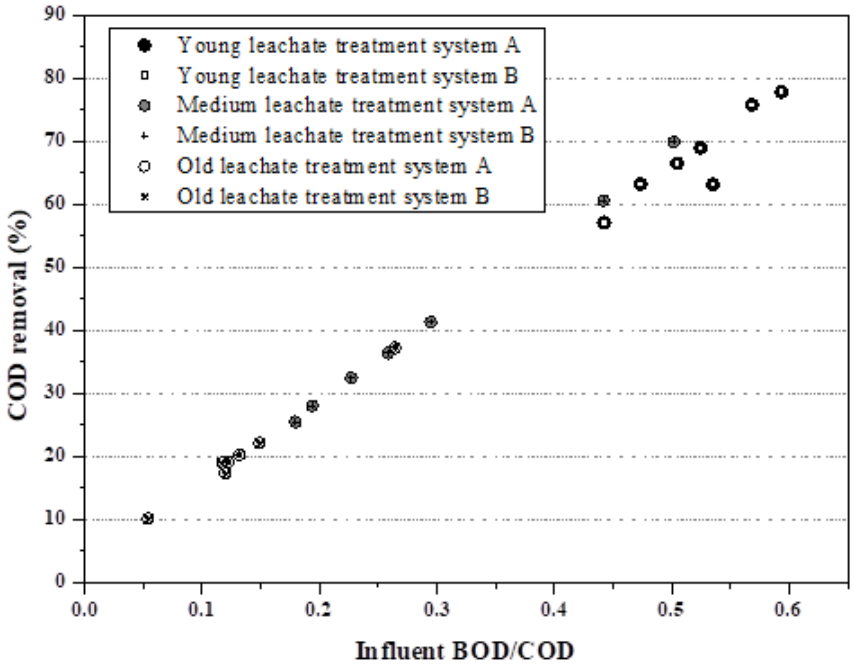

Figure 3.3 COD removal by biological processes in system A and B

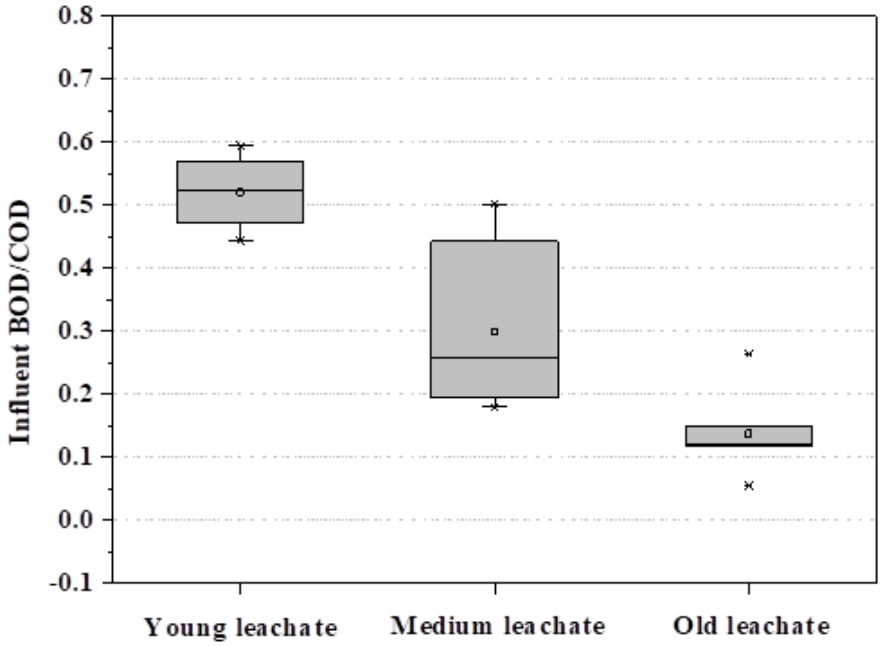

Figure 3.4 Biodegradability of young, medium and old leachate

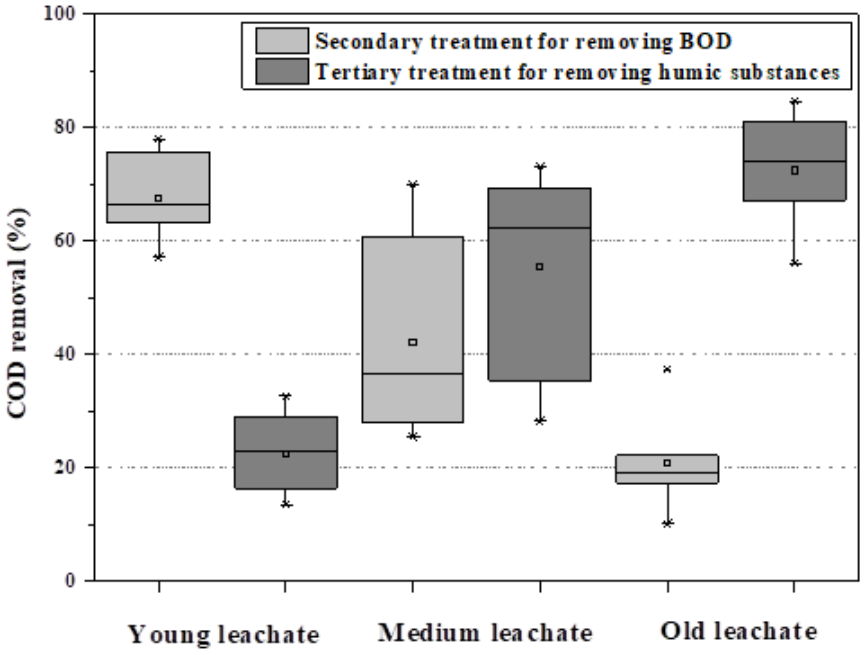

Figure 3.5 COD removal by secondary and tertiary treatment

3.4.2. Unit energy consumption and production

In energy benchmarking, unit energy metrics, which are key performance indicators of the wastewater treatment system, include the ratio between the daily energy consumption and treated wastewater flow (kWh/m^3), COD mass removal (kWh/kg $COD_{removed}$), BOD mass removal (kWh/kg $BOD_{removed}$), or TN mass removal (kWh/kg $N_{removed}$) [33, 38]. Since leachate treatment systems were designed based on the same flow rate and different leachate quality, the unit energy consumption as a function of treated wastewater volume is not an appropriate performance metric. This is because the unit energy consumption and production as well as the unit operation cost will be significantly different if the sustainable metrics such as COD mass removal (kWh/kg $COD_{removed}$), BOD mass removal (kWh/kg $BOD_{removed}$), or TN mass removal (kWh/kg $N_{removed}$) in this study.

Aeration demand for the aerobic biological process is the dominant energy requirement for treating wastewater [113]. Since leachate found from published peer-reviewed papers contains a high concentration of N, most of the O_2 would be required to convert ammonia to nitrate, especially in system B. Mainstream UASB and side-stream anaerobic digesters convert soluble BOD in leachate and biodegradable particulate BOD sludge to CH_4 for energy recovery. Therefore, the unit energy consumption per unit of N removal (kWh/kg $N_{removed}$) and the unit energy production per unit of BOD removal (kWh/kg $BOD_{removed}$) were used in this study.

Unit energy consumption of the two systems, as shown in Figure 3.6, rises with increasing N removal. The unit energy consumption of system B has a better correlation with the removed N load ($R^2 = 0.9516$) and requires less O_2 than system A. Activated sludge process with nitrification in system A consumes O_2 to remove BOD and N. However, most of the O_2 is used for removing N in the PN/A reactor due to soluble BOD removal by mainstream UASB in system B.

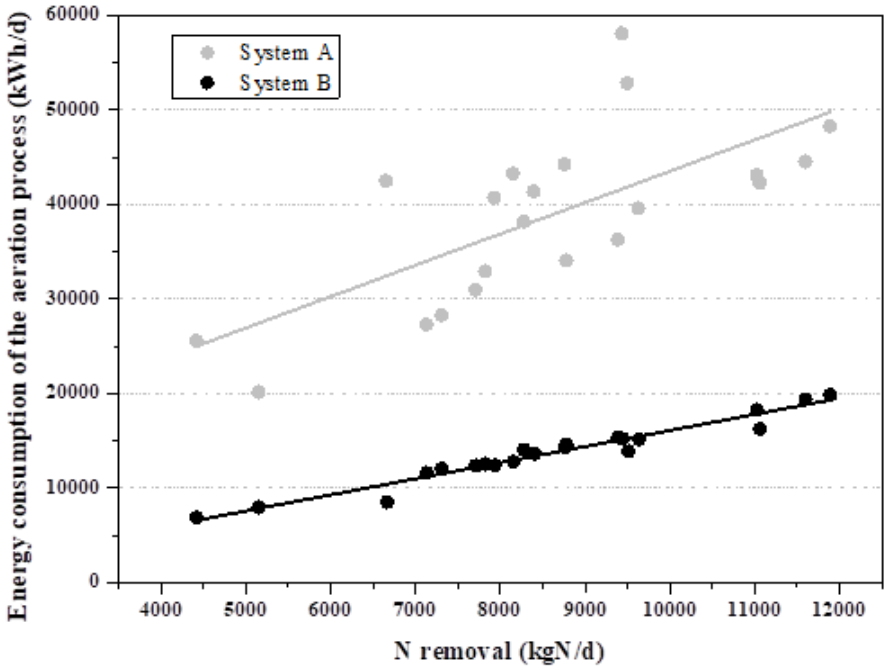

Figure 3.6 Unit energy consumption of the aeration process as a function of daily N removal

Figures 3.7 to 3.9 compare unit energy consumption (kWh/kg $N_{removed}$) and production (kWh/kg $BOD_{removed}$) of the two systems. The statistical parameters of the unit energy consumption of systems A and B are presented in Figure 3.7. The average value, as well as the values at the 0th, 25th, 50th, 75th, and 100th percentiles, are shown in the box plot. Since the average value is affected by extreme values, the median (50th percentile) is considered a more reliable indicator in assessing the performance of the leachate treatment system. Median unit energy consumption of systems A and B are 4.1 and 1.6 kWh/kg $N_{removed}$, which are close to the specific energy demand for conventional nitrification/denitrification and de-ammonification [114]. Figure 3.7 demonstrates that the unit energy consumption of system A decreases as landfills' age increases, while system B

stays constant with age. Young leachate has a higher concentration of COD and BOD/COD ratio than intermediate and mature leachate, which increases the oxygen requirement of the nitrification reactor in system A. Mainstream UASB in system B reduces BOD to lower concentrations before the N removal process. Therefore, the O_2 demand of system B is not significantly affected by the biodegradability of the leachate. Compared with system A, unit energy saving of system B is 4.07, 2.68, and 2.24 kWh/kg $N_{removed}$ for young, medium, and old leachate, respectively. Therefore, system B could save energy by 56%-80% compared to system A.

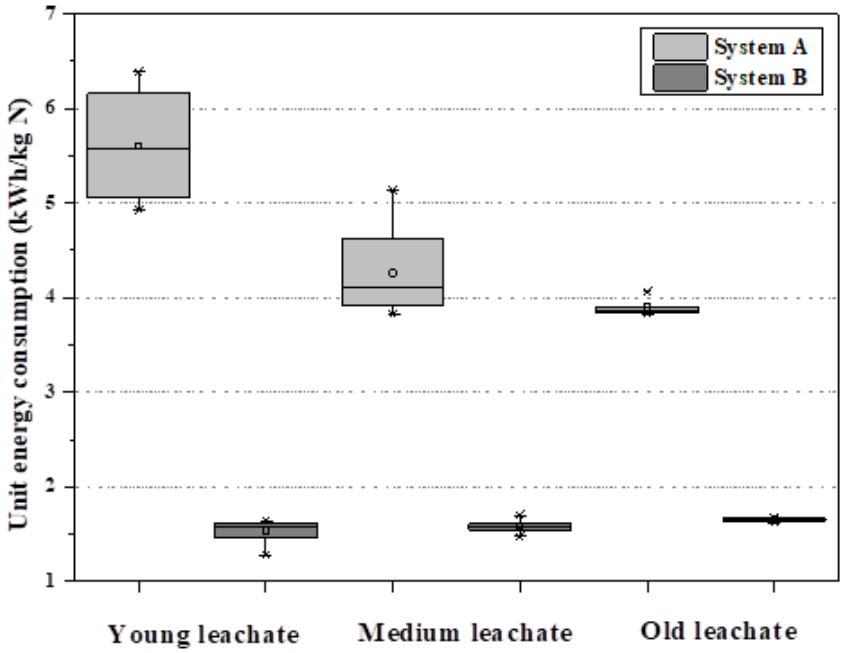

Figure 3.7 Unit energy consumption of leachate at different age by system A and B

The unit energy production of two systems, which are shown in Figure 3.8, increases as BOD removal increases. BOD in leachate contributes mainly to CH_4 production for generating electricity production except in the COD conversion to CO_2. Figure 3.9 compares the unit energy production of two treatment systems. Since UASB and the anaerobic digester in system B convert most of BOD to CH_4, system B could produce more electricity than system A at the same amount of BOD removed when treating young and medium leachate. However, the unit energy production of system A is higher than that of system B for treating old leachate. Although old leachate contains a lower concentration of COD and BOD/COD ratio than the young and medium leachate, the external carbon source in the denitrification process increases electricity production for system A. Due to the high BOD removal in the nitrification reactor, the heterotrophic denitrification process needs exogenous organic carbon sources such as methanol for the conversion of nitrate or nitrite to nitrogen. Since PN/A reactor in system B could convert 90% of NH_4^+-N to N_2 without adding organic carbon, system B needs less carbon sources than system A. On the other hand, old leachate has a higher concentration of NH_4^+_N and soluble, biodegradable, organic nitrogen than young and medium leachate, so system A needs a greater amount of carbon source to achieve high N removal. For treating young and medium leachate, the differences of unit electricity production between system B and A are 0.99 and 0.55 kWh/kg $BOD_{removed}$, respectively. This translates to unit energy productions of system B which are 112% and 36% higher than that of system A. Although system B produces less electricity than system A when treating old leachate, it needs a smaller amount of carbon source for the denitrification process.

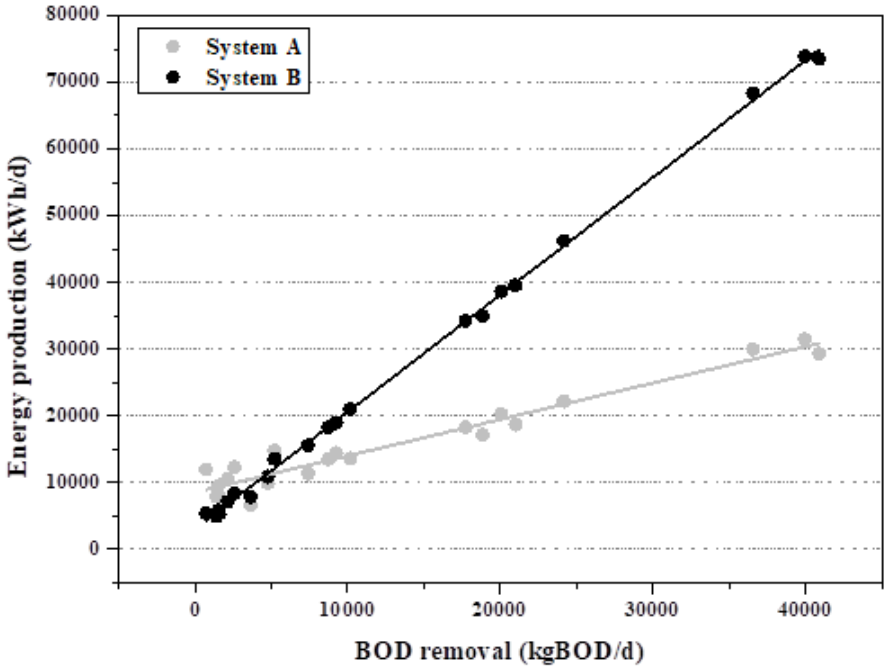

Figure 3.8 Unit energy production as a function of daily BOD removal

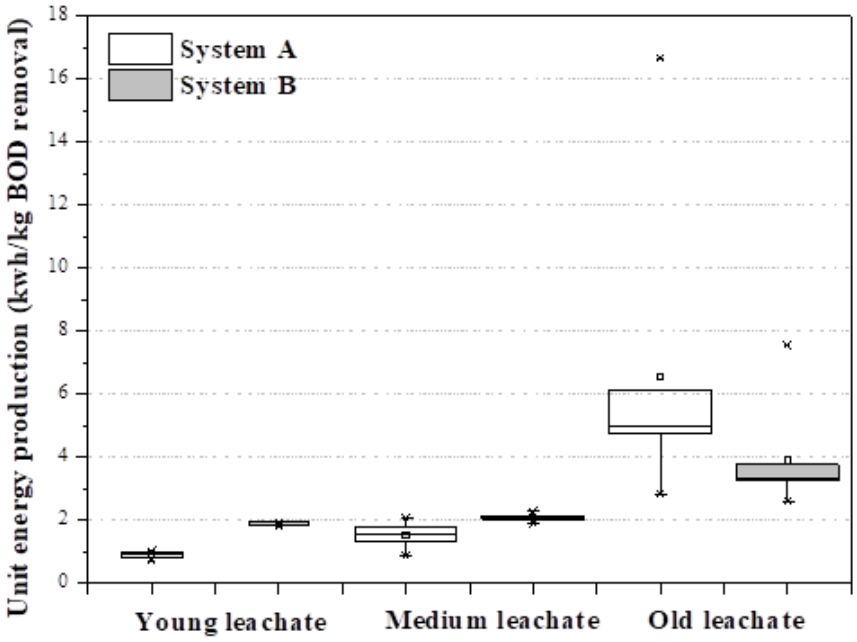

Figure 3.9 Unit energy production of leachate at different age by system A and B

In system B, mainstream UASB is the key technology for producing energy in the form of CH_4. Leachate quality at different ages has a significant effect on the performance of the UASB reactor. Figure 3.10 suggests that the CH_4 production/COD removal (%) rises with an increasing BOD/COD mass concentration ratio. Because of the higher biodegradability (BOD/COD>0.5), the COD of young leachate can be effectively converted to CH_4 during the mainstream anaerobic digestion process. On the other hand, UASB is ineffective for treating leachate from old landfills with a low BOD/COD ratio and a high fraction of refractory soluble organic matter such as humic and fulvic acids [115]. Due to the low biogas production rate, COD removal of old leachate in the UASB reactor depends on the process of sedimentation. However, humic substances are soluble

and discharged with the effluent of the UASB reactor. Therefore, UASB technology is applicable to young leachate treatment.

In tertiary treatment, adsorption is used for removing soluble substances by the accumulation of those substances on activated carbon. Based on the mass balance and Freundlich adsorption isotherm [27], adsorption performance and amount of activated carbon are mostly related to soluble COD concentration. For meeting the discharge requirements, secondary treatments in two systems are designed to achieve the same effluent BOD concentration. Humic substances are not easily removed by biological processes due to their solubility and nonbiodegradability [60]. Therefore, influent COD concentrations of tertiary treatment are similar in the two systems, and different biological methods have less effect on the treatment performance of the activated carbon system.

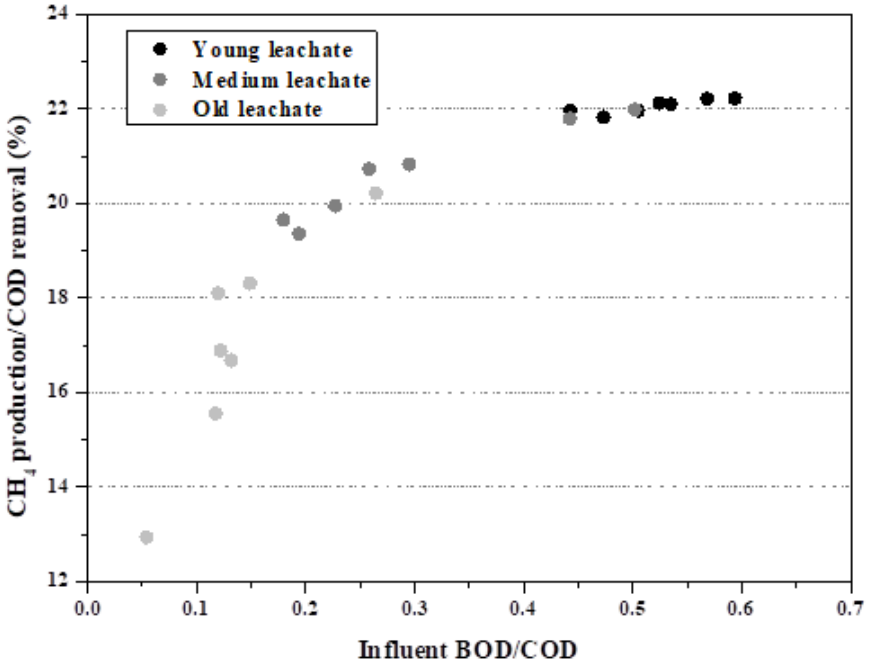

Figure 3.10 CH_4 production/COD removal by UASB reactor for treating leachate at different age in system B

3.4.3. Cost analysis

Young landfill leachate is characterized by high organic content, whereas old leachate is characterized by low biodegradability. COD removal by secondary treatment is positively related to the BOD/COD ratio. Therefore, the unit cost metric (\$/kg $COD_{removed}$) was selected as the indicator for this study. Cost was estimated by using the Excel-based model developed for this study to compare the unit cost of system B with that of system A as shown in Figure 3.11. It clearly shows that the unit cost of two systems increase with landfill age, while the COD concentration and BOD/COD ratio decrease. Young leachate has a higher biodegradability and lower concentration of NH_4^+_N and soluble, biodegradable organic nitrogen than medium and old leachate. The young landfill leachate treatment system needs less carbon source and saves unit capital and O&M costs through smaller N removal reactors when compared to other systems. These aspects reduce the unit cost of the young landfill leachate treatment system. In contrast to system A, the unit cost savings of system B are 0.86, 1.08, and 2.09 \$/kg $COD_{removed}$ for young, medium, and old leachate, respectively. System B reduced the unit cost by 53%-72% compared to system A. The nitrification–denitrification process in system A consumes a great amount of O_2 and external carbon source for the conversion of BOD and ammonia nitrogen to CO_2 and N_2 resulting in large amounts of sludge production. UASB and the PN/A reactor in system B removed most of the soluble BOD and 90% of NH_4^+-N before the activated sludge system, which decreased external carbon source requirement, aeration demand, sludge production, and size of the nitrification-denitrification system. Due to a high sludge

concentration, sludge generation from UASB and PN/A is not required to be concentrated by a sludge thickener, which reduces the size of the thickener and anaerobic digester. Therefore, system B is more economical than system A.

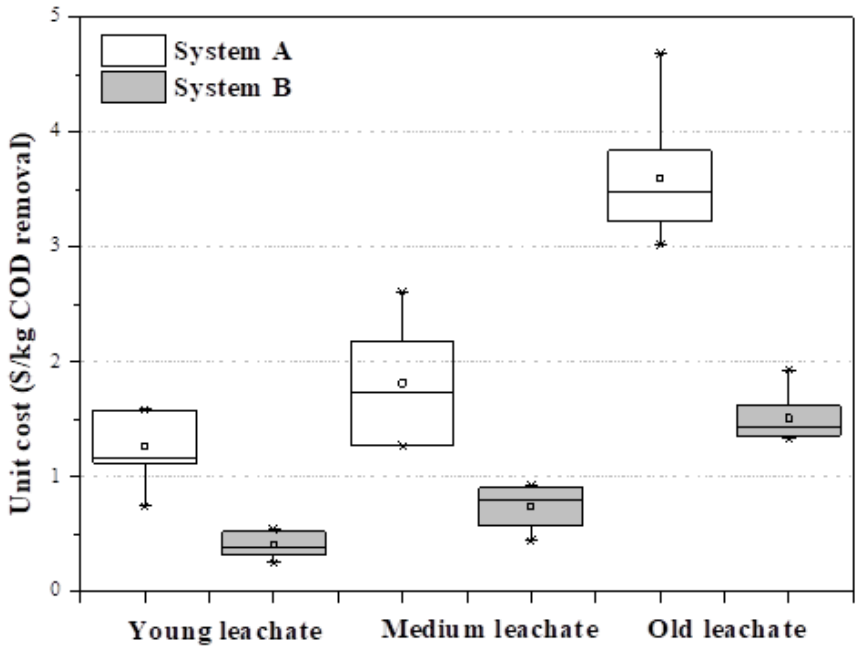

Figure 3.11 Unit daily cost of leachate at different landfill age by system A and B

3.5. Conclusions

The current study collected leachate quality data at different landfill ages and developed an Excel-based model for comparing the performance of a leachate treatment system with different technologies, as shown in Figure A.18 and A.19. COD removal of leachate by the biological treatment process was affected by the BOD/COD ratio, which is negatively correlated with the age of the landfills. The energy and economic analysis of the system using mainstream UASB and PN/A and a conventional leachate treatment

system showed that energy consumption, production, and cost are related to BOD, COD, and N removal of leachate. For young, medium, and old leachate, system B could save unit energy by 4.07, 2.68, and 2.24 kWh/kg $N_{removed}$ than system A. System B improves unit energy production by 0.99 and 0.55 kWh/kg $BOD_{removed}$ compared to system A for treating young and medium leachate, respectively. Furthermore, the unit cost saving of system B is 0.86, 1.08, and 2.09 $/kg $COD_{removed}$ for young, medium, and old leachate. COD removal from young leachate mainly depends on biological processes while old leachate mainly depends on physical-chemical methods. UASB is ineffective for converting COD of old leachate to biogas. These results suggest that the system using mainstream UASB and partial nitrification/Anammox is the best young leachate treatment system in unit energy and cost.

4. EFFECT OF MICRO-SIEVING ON THE UNIT ENERGY AND COST IN FOOD WASTEWATER TREATMENT SYSTEM

4.1. Abstract

According to the twelve design principles of Sustainable Environmental Engineering, micro-sieving is an effective way to prevent organic pollutants from entering the main biological treatment unit and has been identified as one of the major critical technologies in designing water resource recovery facilities. It has been successfully applied to improve treatment capacity and WWTP treatment performance. However, few researchers studied the effect of the micro-sieving on food processing wastewater treatment systems which use mainstream anaerobic digestion and the anammox process. The current study collected food processing wastewater quality data and targeted at different suspended solids removal. The hypothesis is that micro-sieving significantly reduce energy consumption and reduce unit operation cost. Excel-based models were developed to compare the performance of historical, common, and innovative nitrogen removal methods in terms of energy metrics with different primary suspended solids removal. Results showed that micro-sieving is more effective in reducing oxygen requirement and increasing biogas production and CH_4/O_2 energy ratio of historical and common systems. However, micro-sieving alone could not achieve energy positive because of high energy consumption in activated sludge process and nitrification. When micro-sieving was used together with UASB and PN/A, energy positive could be achieved.Results show that CH_4/O_2 energy ratio in an innovative system decreases at the average rate of 0.1 % per 5% TSS removal. The impact of micro-sieving on energy recovery of an innovative system is negative. On the other hand, micro-sieving also reduces the required size and cost of the UASB reactor through decreasing BOD mass load to UASB, systems with different primary

treatments have significant impact of micro-sieving on innovative systems through reduction of unit operation cost. In addition, micro-sieving could increase the biodegradability of wastewater and total COD removal by 5-40% and 1-4%, respectively. Unit energy analysis showed that a primary clarifier and micro-sieving could save unit energy by 1-4% and 1-8%, respectively, compared with a treatment system without primary treatment Micro-sieving decreases total CH_4 production through reducing BOD loading to UASB, which leads to low unit energy production. On the other hand, micro-sieving also reduces the size and cost of UASB and PN/A reactors. The economic benefit due to reduced size of mainstream reactors compensates for the disadvantage of low biogas production. In cost analysis, systems with micro-sieving could reduce unit cost compared to systems without primary treatment by 27% and those with primary clarifiers by 16%. This study quantifies the positive impact of micro sieving on energy-positive food processing wastewater treatment systems. The innovative treatment train is intended to provide a valuable reference for future sustainable food processing wastewater treatment designs.

4.2. Introduction

Food processing wastewater is an unavoidable by-product of industrial activity and increases with population and industrial growth [116]. Effluent from food processing contains large amounts of suspended organic matter, protein, and fat (i.e., bones, meat, and viscera). High concentration of COD, BOD, TN, and TSS is typical for food processing wastewater [117-119]. For example, the average COD, BOD, TN, and TSS concentration of food wastewater could be as high as 4,221 mg/L, 1,209 mg/L, 427 mg/L, and 1,164 mg/L, respectively [117]. Therefore, food processing wastewater could be a major pollution source if it is inadequately treated. River deoxygenation, algal blooms, and eutrophication are all potential environmental damages [120, 121].

Since food processing wastewater has a high concentration of COD, BOD, and TN, biological processes are the preferred treatment method due to its effectiveness and energy saving in treating high-strength wastewater. Biological processes include conventional treatment such as an activated sludge (AS) process with autotrophic nitrification/heterotrophic denitrification process and innovative treatment such as up-flow anaerobic sludge blanket and partial nitrification/anammox (PN/A). UASB converts biodegradable COD to CH_4 without using oxygen. PN/A saves approximately 60% of oxygen demand in nitrification, eliminates external organic carbon source requirement of denitrification, and reduces 80-90% of sludge when compared to the conventional nitrogen removal process [20, 21, 87, 89, 122]. For the first time, this study explores the possibility of applying UASB or PN/A as the main processes for reducing soluble BOD and N to achieveenergy positive food processing wastewater treatment.

Due to the high concentration of TSS in food processing wastewater, primary treatment could be used to reduce the particulate matter in the wastewater. The primary clarifier (PC), which uses gravity separation to remove total suspended solids and reduce chemical oxygen demand that passes through preliminary treatment, is the most widely used primary treatment in a traditional wastewater treatment plant. However, PCs have a large footprint and are costly to install [123]. Micro-sieving, such as a rotating belt filter, offers a more sustainable alternative with a smaller footprint, and lower capital costs when compared to large setting tanks [15, 16]. Since the land requirement of a RBF is typically 10% of that of a primary clarifier [17], the capital cost of primary treatment could be saved approximately 50% with RBFs compared with the conventional PC [18]. RBF could divert at least 50% of TSS and 20% of biochemical oxygen demand from the main treatment stream to mesophilic anaerobic

digester [15, 19]. As a result, micro-sieving decreases the amount of oxygen needed for aeration through reducing the organic loading rate to biological reactors and increases biogas production for energy recovery because of diverting COD in the form of primary sludge from mainstream to anaerobic digesters. Therefore, micro-sieving is considered as one of the most sustainable technologies to separate TSS in the early stage of the treatment train to prevent intensive energy required in traditional aeration of activated sludge system.

Micro-sieving has been identified as a major critical technology in designing water resource recovery facilities according to the twelve design principles of Sustainable Environmental Engineering [14]. Although some studies on suspended solids removal and micro-sieving were done over 40 years ago [124, 125], recent studies about the application of micro-sieving for treating municipal wastewater are relatively sparse. Noticeably, Rusten and Ødegaard [15] quantified the TSS removal efficiency of RBF in the primary treatment of municipal wastewater. Their results showed that most primary clarifiers were unable to meet the primary treatment requirements of the European Union (EU). Micro-sieving achieved the TSS removal of primary treatment as required by EU regulations. Sarathy et al. [126] compared the characteristics of primary solids separated from municipal wastewater by PC and RBF. RBF sludge contained higher organic content than PC sludge and increased methane production without increasing the size of the anaerobic digester. This research focused on comparing TSS removal efficiency and sludge production of PC and RBF in the wastewater treatment system. Currently, there are few cases of research studying the impact of micro-sieving on food wastewater treatment systems using mainstream anaerobic digestion and anammox treatment from energy and cost dimensions. Therefore, the objectives of this study are to (1) compare the performance of primary

clarifier and micro-sieving according to the twelve design principles of Sustainable Environmental Engineering [14]; (2) evaluate the influence of micro-sieving on treatment performance of energy positive system; (3) compare the effect of micro-sieving and primary clarifier on food wastewater treatment system in terms of unit energy consumption and production; (4) carry out unit cost analysis to quantify the effect of micro sieving on the innovative food wastewater treatment trains. The ultimate goal of this paper is to quantify the positive impact of the micro-sieving on treatment systems with innovative technologies and to offer an innovative design approach for future sustainable wastewater treatment of food wastewater.

4.3. Materials and methods

4.3.1. Data and process flow

The current study collected data of food processing wastewater from peer-reviewed papers, as shown in Table 4.1. The databases of untreated wastewater quality included conventional wastewater quality such as the concentration of TSS, COD, BOD, TN, and NH_4^+_N. The noticeable characteristics of food processing wastewater samples were slightly high concentrations of suspended solids, BOD, COD, and BOD/COD ratio of between 0.29-0.57 in Table 4.1. Unit energy and cost were calculated based on various kinds of food wastewater quality. An Excel-based model was developed in designing a one million gallon a day (MGD) food processing wastewater treatment system with different primary and secondary treatments, based on literature [26-28]. The treatment flow diagrams of food processing wastewater treatment systems A-C are illustrated in table 4.2 and Figures 4.1 to 4.3. The differences of system A, B and C were the secondary treatment technologies. System A represents the historical nitrogen removal approach. It consumes O_2 to convert BOD and NH_4^+_N to CO_2, and NO_3_N first, followed by denitrification using an external organic carbon source as the electron

donor for converting NO_3_N to N_2. System B represents the common nitrogen removal processes. It uses denitrification as the first stage followed by activated sludge process and nitrification. NO_3_N produced in the nitrification zone is recycled to the denitrification zone. Denitrifying bacteria uses influent BOD as a carbon source to reduce NO_3_N. System C represents the innovative method using UASB and PN/A for removing pollutants without consuming O_2 or organic carbon sources. Schmidt and Dhulashia [127] reported TSS removal through a conventional primary clarifier and micro-sieving are 30-50% and 50-90%, respectively. Hence, the hypothesis is that micro-sieving could remove 60%, 65%, 70%, 75%, 80%, 85%, and 90% of TSS. TSS removal through primary clarifiers is assumed to be 30% and 50%. According to the hypothesis, we will study the effect of the removal of different primary suspended solids in terms of energy consumption and the production. In the innovative system C, micro-sieving could divert more particulate BOD to side stream high-rate anaerobic digester than primary clarifier. However, this process decreases CH_4 production of mainstream anaerobic digesters through reducing the BOD load to UASB. Therefore, micro-sieving with higher TSS removal could reduce energy production. On the other hand, lower BOD mass load to a UASB also reduces the required size and cost of the UASB reactor. Since the impact of different primary treatments on the cost of an innovative system is unknown, for comparing the cost of systems with different primary treatments, system D, E and F are developed, as shown in table 4.2 and Figures 4.3 and 4.4. System D eliminates primary treatment to maximize the BOD load to UASB. System E and F use primary clarifiers and micro-sieving to divert particulate BOD to side stream anaerobic digester. Since micro-sieving could save 30-60 % of capital cost as compared to the primary clarifier with 50% TSS removal [17], 50% is assumedas TSS removal of clarifier in system E, according to the literature values. For example,

Behera et al. [30] reported TSS removal changes through micro sieving (360 micron screen size) for treating wastewater with different TSS concentrations. Then, they developed an empirical model for estimating TSS removal. For reflecting actual work conditions of micro-sieving, estimated values based on the empirical model are used for TSS removal of micro sieving in system F.

Table 4.1 Food Wastewater Quality

TSS (mg/L)	BOD (mg/L)	COD (mg/L)	TN (mg/L)	Reference
950	1200	2100	220	[128]
1400	1070	2350	317	[129]
1164	1209	4221	427	[117]
662	891	1697	246	[130]
625	1320	3900	217	[131]
3438	1602	5422	361	[132]

Table 4.2 Systems with different primary and secondary treatment

System	Figure	Primary treatment		Secondary treatments	
		Unit process	Pollutant removal	Unit process	Pollutant removal
A	4.1	Primary clarifier	TSS removal: 30% and 50%	Activated sludge process	BOD removal
		Micro sieving	TSS removal: 60%, 65%, 70%, 75%, 80%, 85%, and 90%	Nitrification	NH_4^+_N removal
				Denitrification	NO_3_N removal with organic carbon source
B	4.2	Primary clarifier	TSS removal: 30% and 50%	Denitrification	NO_3_N removal with influent BOD
		Micro sieving	TSS removal: 60%, 65%, 70%, 75%, 80%, 85%, and 90%	Activated sludge process	BOD removal
				Nitrification	NH_4^+_N removal
C	4.3	Primary clarifier	TSS removal: 30% and 50%	UASB	BOD removal
		Micro sieving	TSS removal: 60%, 65%, 70%, 75%, 80%, 85%, and 90%	Partial nitrification	NH_4^+_N removal
				Anammox	NH_4^+_N and NO_2_N removal with CO_2
D	4.4			UASB	BOD removal
				Partial nitrification	NH_4^+_N removal
				Anammox	NH_4^+_N and NO_2_N removal with CO_2
E	4.3	Primary clarifier	TSS removal: 50%	UASB	BOD removal
				Partial nitrification	NH_4^+_N removal
				Anammox	NH_4^+_N and NO_2_N removal with CO_2
F	4.3	Micro sieving		UASB	BOD removal

			Estimated TSS removal based the empirical model	Partial nitrification	NH_4^+_N removal
				Anammox	NH_4^+_N and NO_2_N removal with CO_2

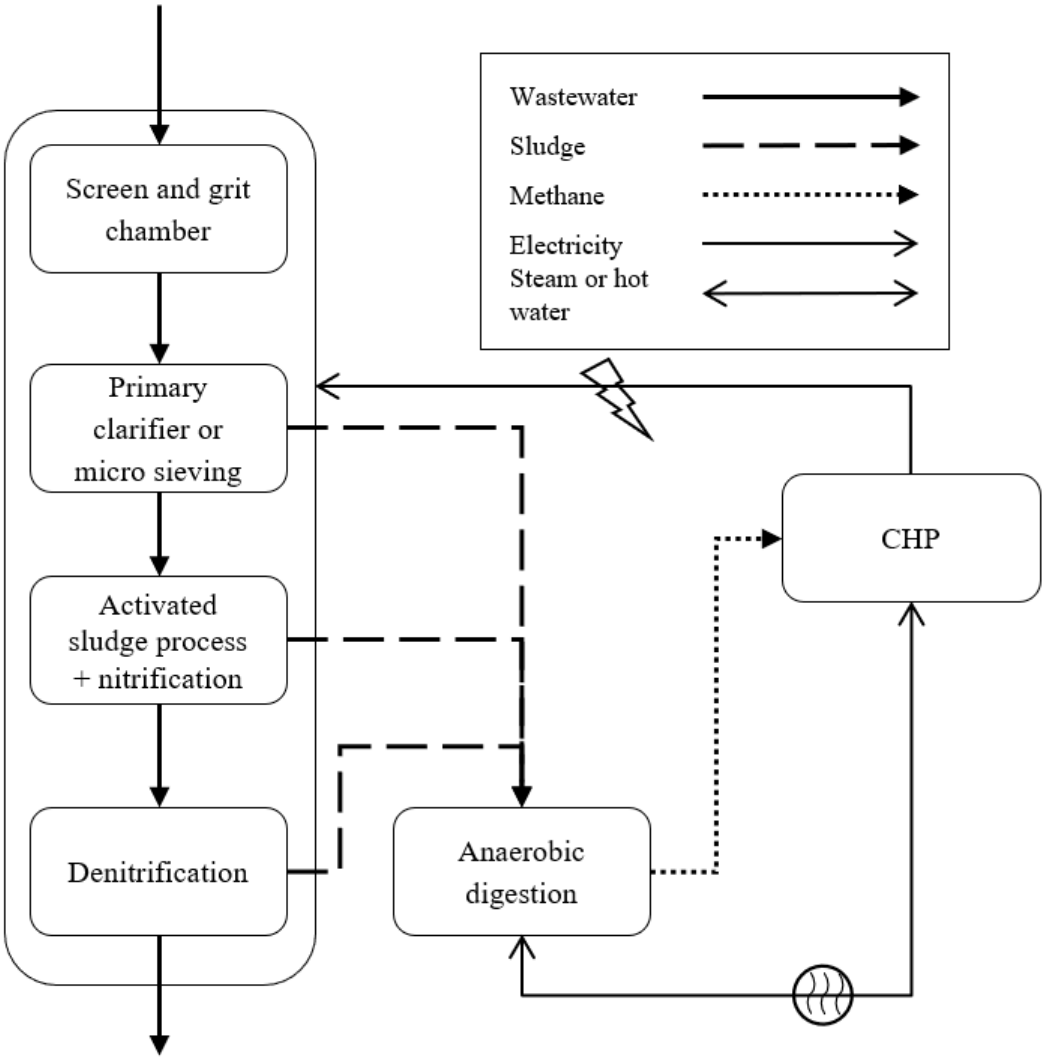

Figure 4.1 Scheme of food wastewater treatment system A- direct line nitrification/denitrification with primary clarifier or micro-sieving

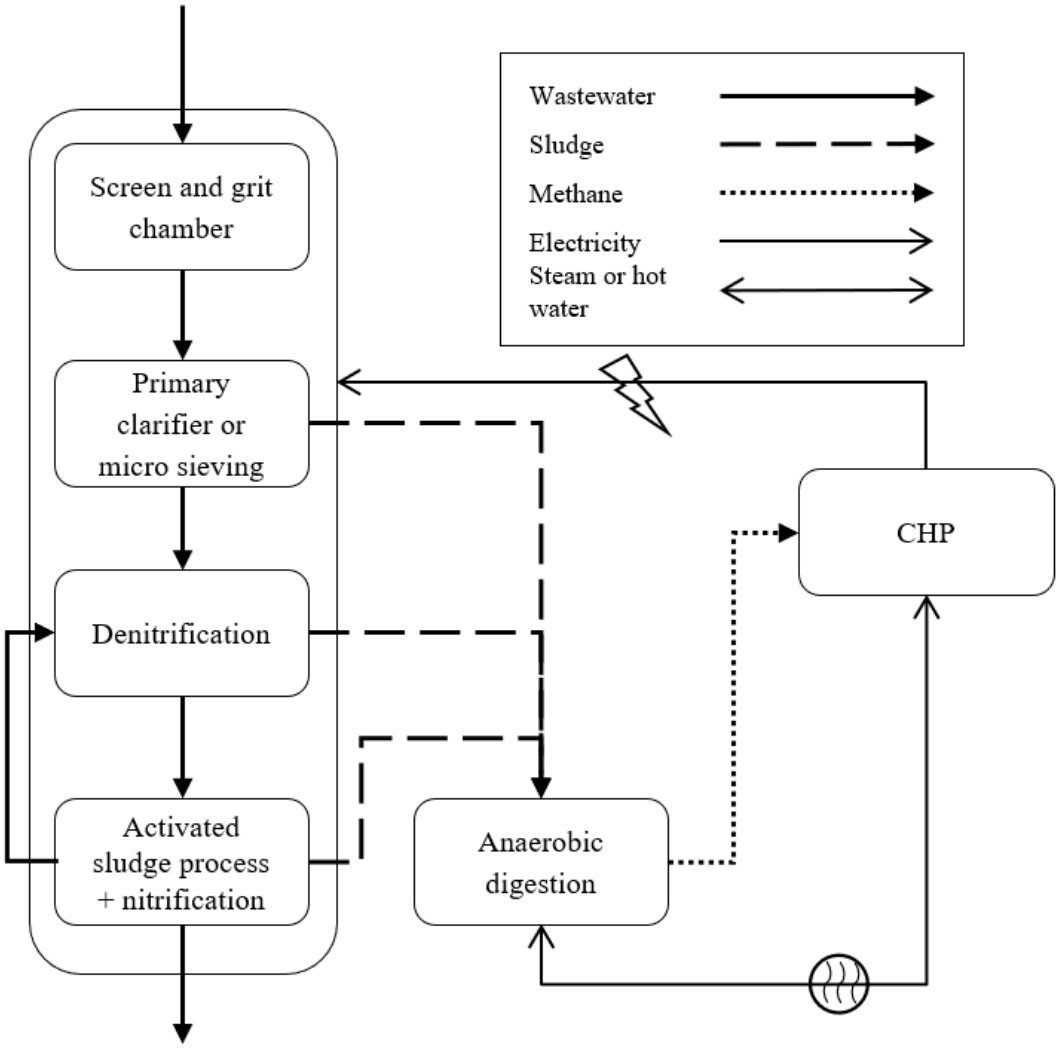

Figure 4.2 Scheme of food wastewater treatment system B- recycle line nitrification/denitrification with primary clarifier or micro-sieving

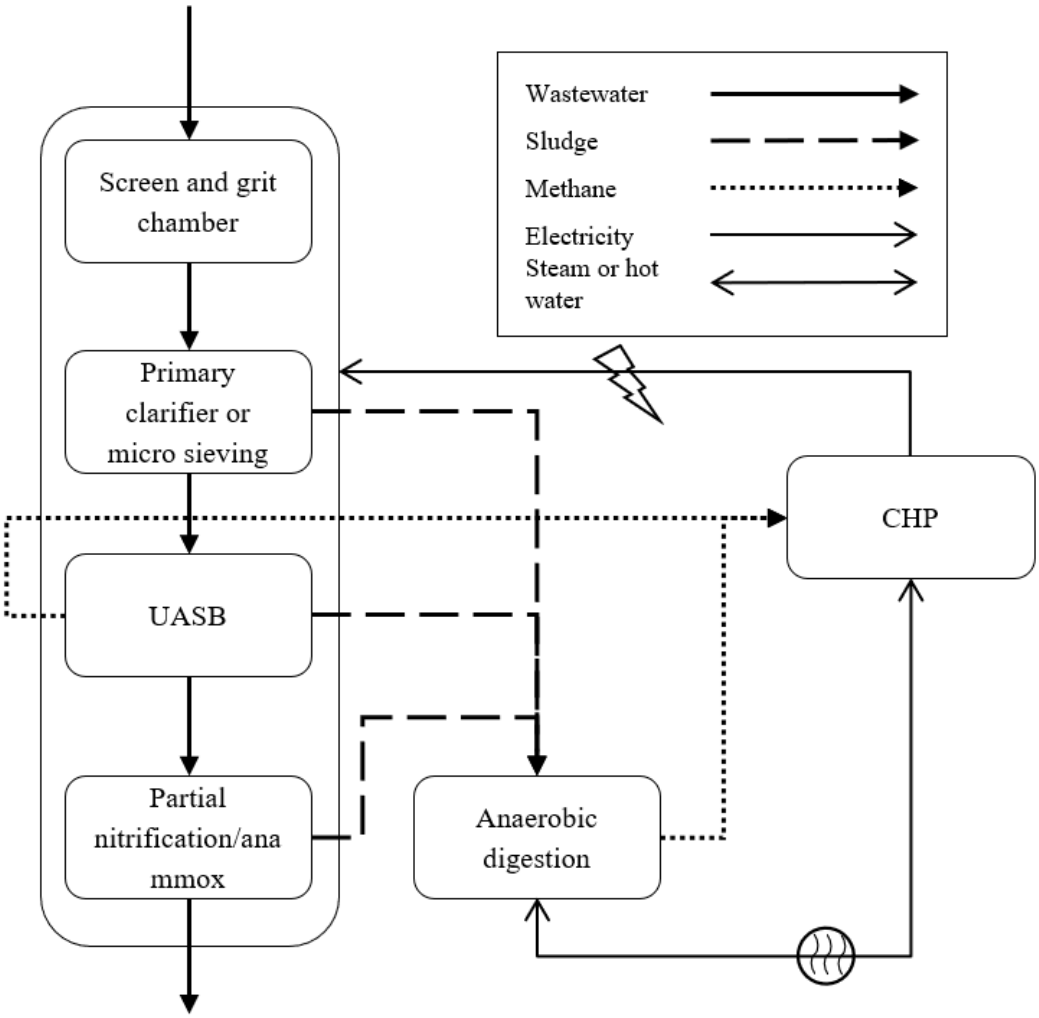

Figure 4.3 Scheme of food wastewater treatment system C, E and F- mainstream anaerobic treatment and anammox treatment with primary clarifier or micro-sieving

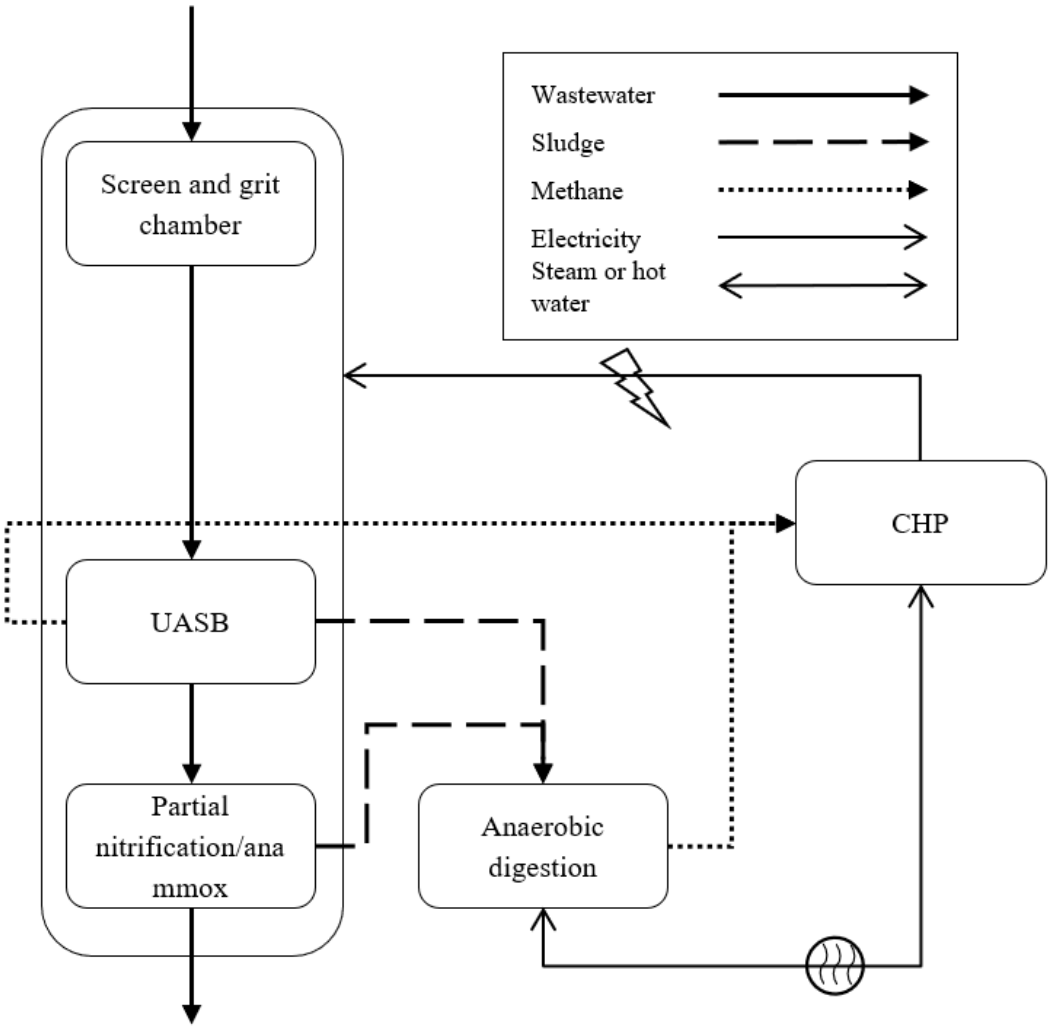

Figure 4.4 Schematic graphs of food wastewater treatment system D- mainstream anaerobic treatment and anammox treatment without primary

4.3.2. Equations for estimating energy and cost

4.3.2.1. Energy and cost of primary treatment

TSS removal, power consumption, and cost of primary treatment are calculated by the following equations [17, 30, 35, 37, 133]:

TSS removal of primary clarifier (%) = -0.004006[Surface overflow rate(m/d)]

+0.73782

(4.1)

TSS removal of rotating belt filter(%) =16.45ln [Influent TSS concentration(mg/L)]-29.1

(4.2)

Power Consumption of rotating belt filter=129.6 kWh/d

(4.3)

Power Consumption of primary clarifier=30 kWh/d

(4.4)

Capital cost of rotating belt filter($)= 0.55Capital cost of priamry clarifier with 50% TSS removal($)

(4.5)

Capital cost of priamry clarifier=-0.00002$(Q)^2$+19.29(Q)+220,389

Q=32.6/surface overflow rate (influent flow rate)

(4.6)

Capital and O&M costs of primary treatment are related to the flow rate. This study examines the influence of different food processing wastewater quality on treatment systems, 1 MGD was used as the flow rate of different treatment systems. A rotating belt filter has a typical operating power consumption of 1.8-3.6 kW for a flow rate of 0.6 MGD [17]; two rotating belt filters with total energy consumption of 5.4 kW were used for treating wastewater. The TSS concentration of food wastewater varies from 625 to 3,438.22 mg/L, micro-sieving could achieve a high TSS removal. However, the power consumption of micro-sieving is higher than that of the primary clarifier, which is 1.25 kW for the treated flow of 1 MGD [133]. When the TSS removal was 50%, the capital cost of micro- sieving is 50%-60% of the conventional primary treatment. It is assumed that the capital cost of micro sieving is 55% of that of primary clarifier with 50% TSS removal. Micro-sieving only had a 10% footprint of the clarifier [17].

Therefore, low capital cost and high TSS removal compensate for the disadvantage of high-power consumption.

4.3.2.2. Energy production of anaerobic digestion

Wastewater quality, retention time, temperature affects COD removal and methane production of UASB. Cavalcanti et al. [42] reported detailed empiric equations that was used to predict the division of the influent COD over the effluent. These account for wastewater quality, sludge age, and temperature.

$$mSeu = fns + \frac{0.27\exp[-0.04\cdot(Rsu-4)]}{1.067^{T-25}} \tag{4.7}$$

$$mSxvu = fnp + fcv\cdot Yan\cdot(1-fns-fnp) + \frac{0.25\exp[-0.04\cdot(Rsu-4)]}{1.067^{T-25}} \tag{4.8}$$

$$mSdu = 1 - mSeu - mSxvu \tag{4.9}$$

where:

mSeu = fraction of influent COD ending up as non-settleable COD in the UASB effluent (g COD/g COD)

mSxvu = influent COD fraction converted into anaerobic excess sludge (g COD/g COD)

mSdu = fraction of influent COD digested in UASB (g COD/g COD)

T=temperature=25°C

fns = non-biodegradable, soluble influent COD fraction (g COD/g COD)

fnp = non-biodegradable particulate influent COD fraction (g COD/g COD)

Rsu = anaerobic sludge age in the UASB reactor

Yan = yield coefficient in an anaerobic environment (0.05 g VSS/g COD) as determined by Cavalcanti et al. [42]

Since the sludge age of UASB is greater than 20 days [26], sludge age is assumed to be 40 days. Since biodegradable COD in wastewater could be converted into CH_4 at 15 and 35°C [20], the operating temperature of the treatment system was assumed to be 25 °C.

$$\text{Energy production of UASB (kWh/d)} = (\text{mSdu})\,(Q_{UASB})(S_{UASB})\,(0.25\ \text{g } CH_4/\text{g COD})\,(10^{-3}\text{kg/g})(13.9\ \text{kWh/kg } CH_4)\,(38\%) \tag{4.10}$$

$$\text{Theoretical unit energy production of UASB (kWh/kg COD}_{\text{removed}}) = \frac{\text{Energy production of UASB (kWh/d)}}{\text{COD removal (kg/d)}} \tag{4.11}$$

where:

mSdu = fraction of influent COD digested in UASB (g COD/g COD)

S_{UASB}= Influent COD concentration to UASB (g/m^3)

Q_{UASB}=Flow rate to UASB (m^3/d)

COD removal in UASB (kg/d) =COD digested in UASB+COD discharged as sludge

Qasim and Zhu [28] reported that 1 kg COD could be converted to 0.25 kg CH_4 by UASB. It was assumed that UASB, which combines heat and power technologies, could convert 38% of CH_4 formed in the anaerobic digestion to electricity with an energy density of approximately 13.9 kWh/kg CH_4 [40].

$$RT_{AD}=15d \tag{4.12}$$

$$T_{AD}=35°C$$

(4.13)

biodegradable COD destruction value =0.7

(4.14)

A side stream anaerobic digester is used to produce CH_4 from sludge in the absence of air at a specific solid retention time and a specific temperature. Mean values of solids retention time and temperature shall be between 15 days at 35℃ to 55 ℃ and 60 days at 20℃ [134], respectively. Kabouris et al. [135] reported that the biodegradable volatile solids destruction value of anaerobic digester was 69% at a retention time of 12 days and 35℃. It was reported 70% of biodegradable COD could be converted to CH_4 at 35℃ [40]. As a result, retention time, temperature and biodegradable COD destruction value were assumed to be 15 days, 35℃ and 70%, respectively.

4.3.2.3. Energy consumption of PN/A

The reaction of the PN/A process can be depicted as the following equation [28]:

$NH_4^+ + 0.799\ O_2 + 1.109\ HCO_3^- \rightarrow 0.436\ N_2 + 0.111\ NO_3^- + 1.034\ CO_2 + 2.496\ H_2O$+New biomass

(4.15)

The overall oxygen requirement for conversion of ammonia to nitrogen gas and nitrate is 1.83 g O_2/g NH_4-N converted.

$$\text{Theoretical unit energy consumption of PN/A} = \frac{1.83\,\text{kg}O_2/\text{kg}\ NH_4\_N}{OT_a}$$
$$= 1.32\,\text{kWh/ kg}\ N_{\text{removed}}$$

(4.16)

where

OT_a=Oxygen transfer efficiency (actual)=1.2 kg O_2/kWh [26]

Since biodegradable COD in wastewater could be converted into CH_4 at 15 and 35℃ [20], the operating temperature of the treatment system was assumed to be 25 ℃. Therefore, energy consumption is estimated based on O_2 mass demand and oxygen transfer efficiency at 25 ℃ [26].

4.3.2.4. Energy and cost of WWTP

The following equations are used for estimating unit energy consumption, production, and cost of WWTPs [26, 39, 40].

$$\text{Unit energy production(kWh/kg BOD}_{\text{removed}}\text{)} = \frac{(13.9\ \text{kWh/kg CH}_4)\,(38\%)\,(\text{CH}_4\ \text{production (kg/d)})}{\text{BOD removal (kg/d)}} \tag{4.17}$$

$$\text{Unit energy consumption (kWh/m}^3\text{)} = \frac{\text{Aeration demand of PN/A (kgO}_2\text{/d)}}{\text{Oxygen transfer efficiency (actual)Q}}$$

$$\text{Unit energy consumption (kWh/ kg N}_{\text{removed}}\text{)} = \frac{\text{Aeration demand of PN/A (kgO}_2\text{/d)}}{\text{Oxygen transfer efficiency (actual)(N removal (kg/d))}} \tag{4.18}$$

where

Q=flow rate (m^3/d)

Oxygen transfer efficiency (actual)= 1.2 kg O_2/kWh

Electricity consumption of a WWTP is mainly comprised by power consumption of the wastewater lift pump, aeration equipment, and sludge treatment. The energy consumption of biological treatment accounts for 50–70% of the overall energy

consumption [41]. Since UASB removes COD without consuming oxygen, aeration energy of the PN/A process was the dominant energy demand.

$$\text{Capital cost(\$/yr)} = \frac{\text{Capital cost(\$)}}{a_{i,n}}$$

$$a_{i,n} = \frac{(1+i)^n - 1}{i(1+i)^n}$$

where

$a_{i,n}$=annualization factor

i =interest rate (annual)=6%

n=economic lifetime of the treatment plant in years=20 (yr)

$$\text{Total cost(\$/d)=Capital cost(\$/d)+O\&M cost(\$/d)} \tag{4.19}$$

$$\text{Unit capital cost (\$/kg COD}_{\text{removed}}\text{)} = \frac{\text{Capital cost(\$/yr)(365 d/yr)}}{\text{COD removal (kg/d)}} \tag{4.20}$$

$$\text{Unit O \& M cost (\$/kg COD}_{\text{removed}}\text{)} = \frac{\text{O \& M cost(\$/yr)(365 d/yr)}}{\text{COD removal (kg/d)}} \tag{4.21}$$

$$\text{Unit cost(\$/kg COD}_{\text{removed}}\text{)=Unit capital cost(\$/kg COD}_{\text{removed}}\text{)+Unit O\&M cost(\$/kg COD}_{\text{removed}}\text{)} \tag{4.22}$$

For comparing the cost of systems with different primary treatment, the total capital costs ($) were annualized over the expected lifetime of the WWTP. Economic lifetime and interest rate were assumed to be 20 years and 6%[26]. Capital costs ($) were transformed into annualized capital costs ($/yr) based on the interest rate and lifetime.

The total unit cost was calculated as the combination of the unit capital costs and the unit operational costs.

4.4. Results and discussion

4.4.1. Effect of different primary suspended solids removal for treating the specific food wastewater quality

4.4.1.1. Pollutant mass load

The first row of data in table 4.1 was selected as the specific food wastewater quality for studying the impact of different primary suspended solids removal on the performance of systems A, B, and C. According to the coefficients of the wastewater obtained from GPS-X 7.0 in table 4.3, the total COD, BOD and N of the specific food processing wastewater were divided into fractions as shown in table 4.4. The analytical result shows that the influent particulate COD, BOD and N fractions are 48, 42 and 15%. BOD, COD and N removal rise with increasing TSS removal in Figure 4.5. Due to a lower particulate N fraction, the effect of micro-sieving on BOD and COD removal is greater than N removal.

Table 4.3 Coefficients of the wastewater obtained from GPS-X 7.0

Parameter	Value	Reference
Volatile Suspended Solids/Total Suspended Solids, VSS/TSS	0.75	[135]
Particulate COD/Volatile Suspended Solids, pCOD/VSS	1.42	[91]
Colloidal COD/slowly biodegradable COD, cCOD/sbCOD	0.15	[135]
BOD to COD ratio of soluble and colloidal biodegradable substrates, fBOD/(bsCOD+cCOD)	0.717	[135]
BOD to COD ratio of particulate biodegradable substrate, pBOD/pCOD	0.58	[135]
Biodegradable particulate organic nitrogen/biodegradable particulate COD, bpON/bpCOD	0.0268	[123]
Non-biodegradable particulate organic nitrogen/non-biodegradable particulate COD, nbpON/nbpCOD	0.068	[123]
Non-biodegradable soluble organic nitrogen/non-biodegradable soluble COD, nbsON/nbsCOD	0.05	[135]

Table 4.4 Specific food processing wastewater characteristics

Characteristics	Concentration (mg/L)
Inert suspended solids (ISS)	237.50
Volatile Suspended Solids (VSS)	712.50
Non-biodegradable particulate COD (nbpCOD)	144.58
Biodegradable particulate COD (bpCOD)	867.17
Colloidal COD (cCOD)	153.03
Biodegradable soluble COD (bsCOD)	819.13
Non-biodegradable soluble COD (nbsCOD)	116.09
Biodegradable particulate COD (bCOD)	1839.33
Particulate BOD (pBOD)	502.96
Filtered BOD (fBOD)	697.04
Non-biodegradable particulate organic nitrogen (nbpON)	9.83
Biodegradable particulate organic nitrogen (bpON)	23.24
Non-biodegradable soluble organic nitrogen (nbsON)	5.80
Ammonium-Nitrogen (NH4+-N)	181.12
Total nitrogen (TN)	220.00

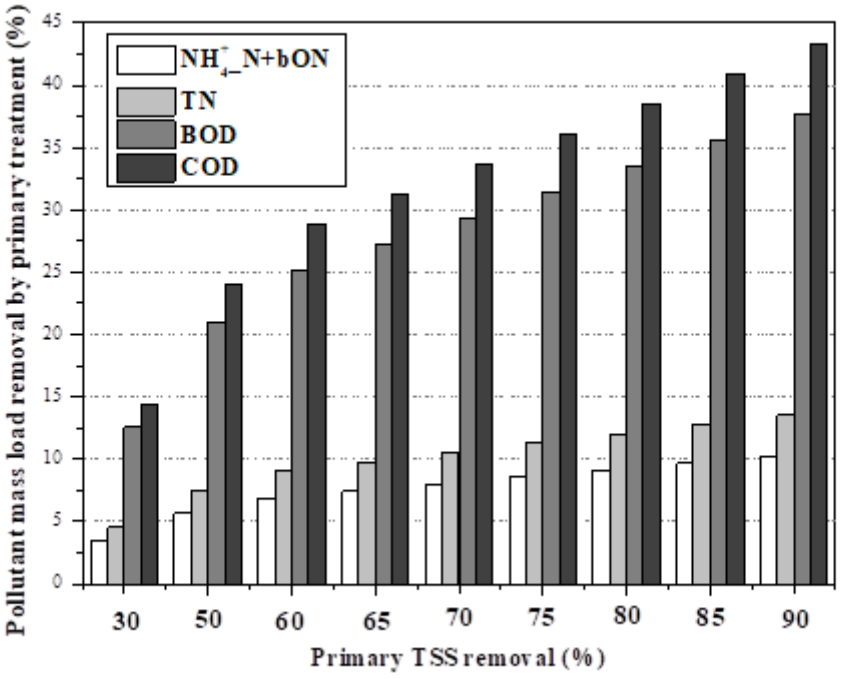

Figure 4.5 Pollutant mass load removal by primary treatment

4.4.1.2. Oxygen demand

Primary treatment removes particulate BOD and N to reduce influent BOD and bON mass load to the secondary treatment. Oxygen demand decreases with increasing primary suspended solids removal in Figure 4.6. Compared to the conventional primary clarifier with 30-50% TSS removal, micro-sieving is more effective in reducing oxygen requirement of historical, common, and innovative systems because of higher solids capture. In an innovative system, O_2 is only used for the partial nitrification process due to BOD removal by micro sieving and UASB without consuming O_2. System C with different suspend solids removal needs less O_2 than systems A and B. Oxygen requirement in systems A, B, and C decrease at the average rate of 1.7%, 2.3%, and 0.5% per 5% TSS removal. The change of oxygen demand reduction in system C is smaller than those in system A and B. Micro-sieving has very low efficiency in removing soluble biodegradable N and NH_4^+_N from wastewater. All soluble N enters into PN/A reactor. Due to low N removal, the impact of micro-sieving on oxygen demand of system C is smaller than those of system A and B.

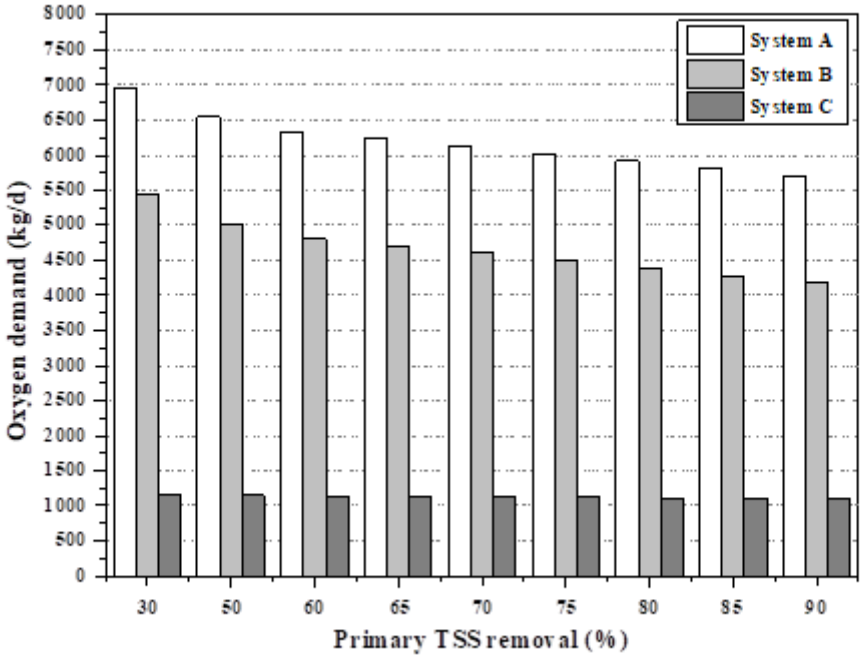

Figure 4.6 Oxygen demand for different primary suspended solids removal

4.4.1.3. CH_4 production

Primary treatment captures particulate BOD for direct anaerobic digestion prior to biological conversion to sludge. In the activated sludge process, BOD in wastewater is first converted to CO_2 and biomass. Energy of biomass is subsequently recovered through anaerobic digestion at low energy efficiency. In systems A and B, total CH_4 production at standard temperature and pressure condition rise with increasing primary suspended solids removal, as shown in Figure 4.7. Compared to the conventional primary clarifier with 30-50% TSS removal, micro-sieving is more effective in increasing CH_4 production of conventional systems because of higher particulate BOD division to side stream anaerobic digester. The external organic carbon source, which is added to system A for denitrification, is converted to biomass for indirect anaerobic digestion. Adding an external organic carbon source increases cost of chemicals but also increases CH_4 production. Therefore, system A could produce more CH_4 than

system B, as shown in Figure 4.7. In an innovative system, UASB and anaerobic digestion directly converts most of BOD in wastewater to CH_4. Hence, methane production of system C is 1.6-2.5 times that of system A and B. However, methane production in system C decreases at the average rate of 0.35% per 5% TSS removal. Primary treatment with higher TSS removal could separate more particulate BOD from wastewater to high-rate anaerobic digester. On the other hand, this process also decreases CH_4 production of mainstream anaerobic digester by reducing the BOD mass load to UASB. Mainstream UASB with high sludge age of 40 d could convert BOD more efficiently to CH_4 than the high-rate anaerobic digester. In general, the impact of micro-sieving on CH_4 production of system C is negative.

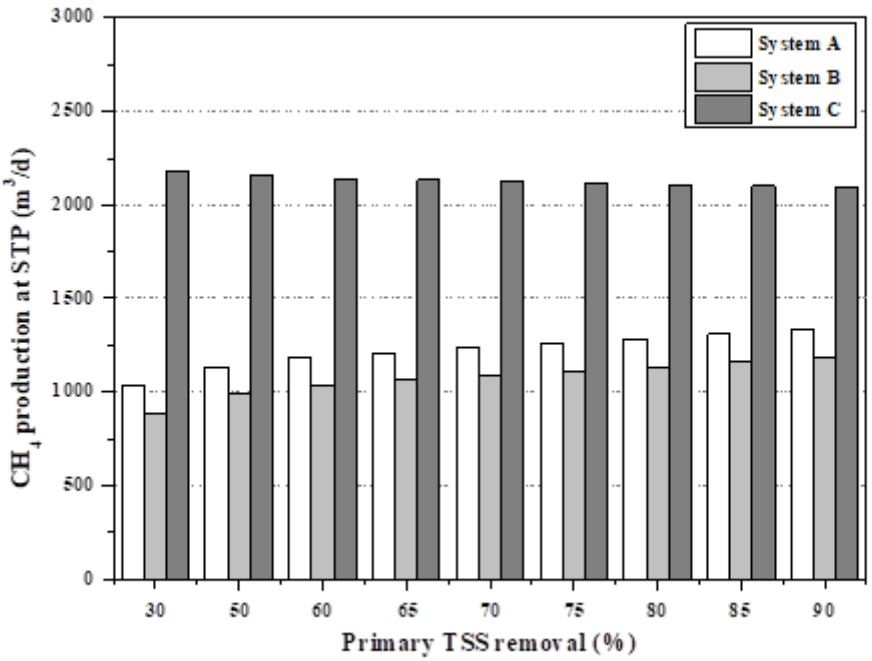

Figure 4.7 CH_4 production for different primary suspended solids removal

4.4.1.4. CH_4/O_2 energy ratio

Figure 4.8 presents CH_4/O_2 energy ratio for all treatment systems. Since the energy ratio is above 1, system A with greater than 85% TSS removal and system B with greater than 65 % TSS removal could produce enough electricity to satisfy the energy demand for oxygen production. Due to higher BOD to division to an anaerobic digester, micro sieving is more effective in recovering energy of historical and common systems when compared to the conventional primary clarifier. The energy consumption of biological treatment accounts for 50–70% of the overall energy consumption [41]. Oxygen demand is assumed to be 60% of total energy consumption, in which case only system C could achieve electrical self-sufficiency because its energy ratio is above 8. Figure 4.8 shows that CH_4/O_2 energy ratio in system C decreases at the average rate of 0.1 % per 5% TSS removal. Micro-sieving with higher TSS removal reduces CH_4 production of system C, which leads to a negative influence of micro- sieving on energy recovery. However, micro-sieving also reduces the required size and cost of UASB reactor by decreasing BOD mass load to UASB. Therefore, the impact of micro-sieving on the unit operational cost of the innovative systems is explored in the next section.

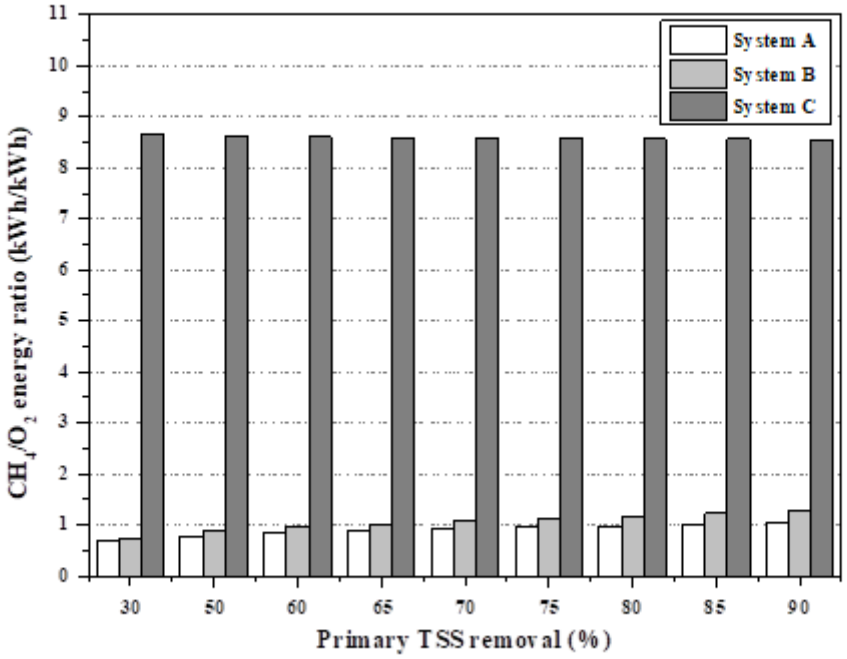

Figure 4.8 CH_4/O_2 energy ratio for different primary suspended solids removal

4.4.2. Assessment of the primary treatment

The performance of primary clarifier and micro-sieving could be compared based on the twelve design principles of Sustainable Environmental Engineering, as shown in Table 4.5 [14]. On the spatial scale, the land requirement of RBF is typically 10% of that of the primary clarifier [17]. Since primary clarifier is often construted open to the weather, treatment efficiency is impacted by temperature under weather change. Micro-sieving is a modular and flexible design that could be installed indoors or outdoor to serve WWTPs with any capacity requirement. In terms of materials, primary sedimentations are circular concrete tanks, and the major construction material of micro sieving is stainless steel, while concrete and steel are recyclable materials. However, the quality of recycled concrete is 15-25% lower than that of the original concrete. Steel could be recycled repeatedly without loss of strength [136]. In terms of separation, the solids content of micro-sieving sludge is 3-8%, which is significantly higher than that

of primary clarifier sludge [17]. Gupta [123] compared the removal efficiency of cellulose by different primary treatments. The cellulose contents in micro-sieved sludge and primary clarifier sludge are 37±1 % and 18±0.2 % cellulose, respectively. Since cellulose represents a major component of the particulate matter in raw municipal wastewater, micro-sieving has the ability to separate more sludge with higher percent dry solids than clarifiers. From energy and cost perspectives, the typical operating power consumption of micro sieving is 1.8-3.6 kW for a flow rate of 0.6 million gallons per day (MGD), and the energy demand of primary is 1.25 kW for the treated flow of 1 MGD [17, 133]. The capital cost of primary treatment could be reduced by approximately 50% with RBFs as compared to the conventional PC [18]. On the other hand, micro-sieving consumes more electricity than a primary clarifier for treating the same volume of wastewater, which leads to its higher O&M cost. On the other hand, the separation process by micro sieving reduces BOD and COD mass to biological reactors to decrease oxygen demand, required size and cost of mainstream equipment. Micro-sieving could be proved to be an economical and efficient alternative to the conventional primary clarifier when the benefit due to cost-saving is higher than its increased cost.

Table 4.5 Comparison between primary clarifier and micro sieving

	Primary clarifier	**Micro sieving**
Land requirement	Large	10% of the clarifier
Installation	Outdoor	Indoor or outdoor
Major construction material	Concrete	Stainless steel
TSS Removal Efficiency	30-50	50-90
Solids Content (%)	1-6	3-8
Capital cost	High	40%-70% of the clarifier
Operating power consumption (kwh/m^3)	0.008	0.019-0.038

4.4.3. Influence of primary treatment

4.4.3.1. Treatment performance of primary and secondary treatment

Figure 4.9 shows that the COD diversion to side stream anaerobic digester by micro-sieving is higher than that by the primary clarifier. The primary clarifier depends on the settling velocity of particles, flow rate, and surface overflow rate. In micro-sieving, particulate removal is achieved by filter-mesh under pressure. As wastewater flows through the filter-mesh, the suspended solids are retained on the mesh, and the filtered wastewater is transported by gravity to the outlet pipe [16]. Mesh pore size, the particle size distribution of influent wastewater, and flow rate dictate the performance of micro-sieving [137]. Since the retained suspended solids form an extra layer on the mesh, which contributes to retaining a greater number of particulates on the filter, TSS removal efficiency is positively related to TSS concentration to micro-sieving [30]. Most of the food wastewater has a high concentration of TSS, as shown in table 4.1, micro-sieving could retain more particulates on the filter and divert more COD to anaerobic digester than clarifier.

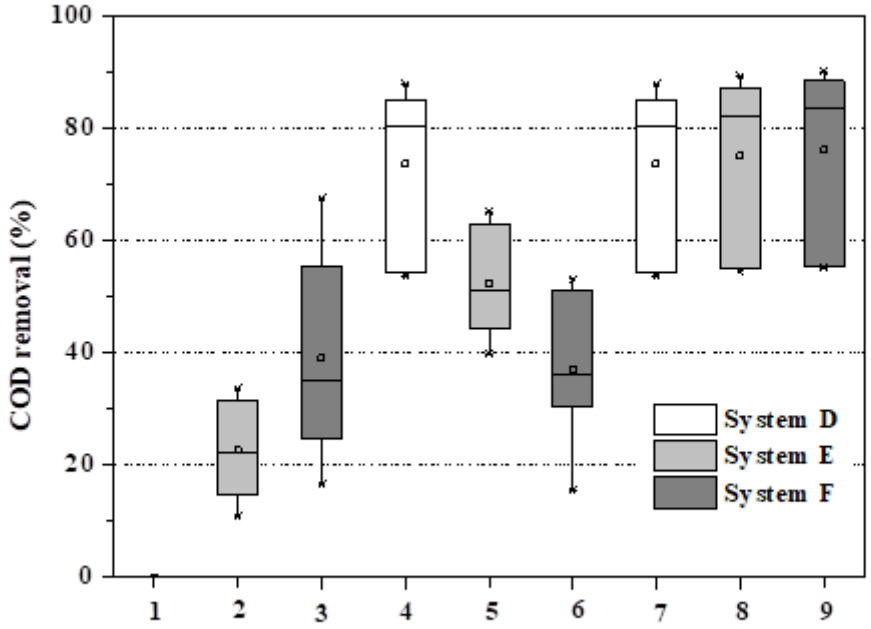

Figure 4.9 COD removal of primary and secondary treatment
Note: 1: primary treatment in system D, 2: primary treatment in system E, 3: primary treatment in system F, 4: secondary treatment in system D, 5: secondary treatment in system E, 6: secondary treatment in system F, 7: primary and secondary treatment in system D, 8: primary and secondary treatment in system E, 9: primary and secondary treatment in system F

Since the PN/A process does not require organic carbon sources for denitrification, mainstream UASB in the secondary treatment removes most of the remaining COD after primary treatment, as shown in Figure 4.9. The COD removal by the UASB in system C is less than those of other systems due to high COD removal of micro sieving. However, total COD removal of systems A, B, and C are 53.7%-87.9%, 54.6%-89.4%, and 55.3%-90.3%, respectively. COD removal of food processing wastewater increased by 1-2% with primary clarifier and by 1-4% with micro-sieving. Therefore, the COD removal of wastewater treatment could be improved by micro-sieving.

4.4.3.2. Treatment performance of primary and secondary treatment

Biodegradability (BOD/COD ratio), which represents the ability of a substance to be removed by microorganisms, is commonly used as an indicator of wastewater

biodegradability [108]. High biodegradability is a sufficient condition of biodegradation. Figure 4.10 shows the BOD/COD ratio of untreated wastewater and treated wastewater by primary clarifier and micro-sieving. Primary treatment reduces COD mass to secondary treatment, as shown in Figure 4.9. This process results in increased biodegradability of wastewater due to the reduction of particulate non-biodegradable COD in wastewater. Figure 4.10 demonstrates that the BOD/COD ratio of food processing wastewater increased by 3-11% with the primary clarifier and by 5-40% with micro-sieving. Gupta [123] reported the effect of different primary treatments on biodegradable (biodegradable COD/COD). The results showed that primary clarifier and micro-sieving increased the biodegradable fraction from 71% to 80% and from 71% to 74%, respectively. The differences between Gupta's study and the current study are TSS removal efficiency of primary treatment. In Gupta's investigation [123], primary clarifier could achieve 67%-73% of TSS removal, and micro sieving could remove 26%-28% of TSS from wastewater due to different wastewater quality and design of primary clarifier. In addition, biodegradability increases with increasing TSS removal. Due to high TSS concentration and particulate matter fraction, micro-sieving could improve the biodegradability of food processing wastewater more significantly as compared with the primary clarifier.

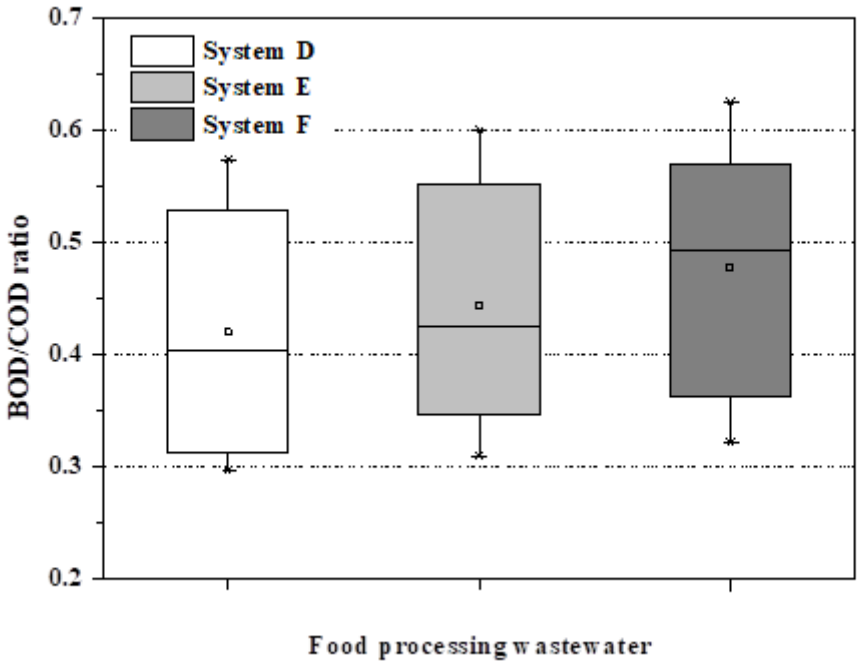

Figure 4.10 Biodegradability of food processing waster after different primary treatment.

Micro-sieving could change food processing wastewater quality. However, it has very little impact on biodegradable COD (bCOD) digestion rate, which represents the ratio between bCOD digested mass in UASB and influent bCOD mass to UASB. Biodegradable COD removal in UASB includes bCOD that is converted to CH_4 and bCOD discharged as sludge. High biodegradability increases the fraction of COD digested and COD discharged as sludge according to equation (8-9), which leads to similar bCOD digestion rates of three systems in Figure 4.11. Therefore, micro-sieving could increase biodegradable COD fraction to UASB, but it could not significantly change the conversion rate of biodegradable COD to CH_4.

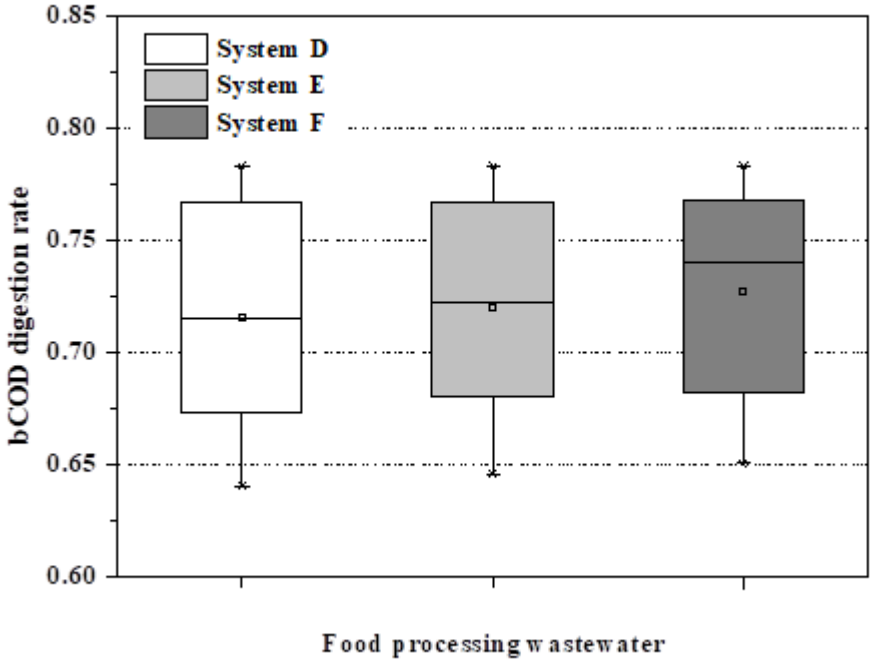

Figure 4.11 biodegradable COD digestion rate in UASB

4.4.3.3. Effluent wastewater quality

Micro-sieving with high TSS removal could remove more particulate organic matter than clarifier, as shown in Figures 4.5 and 4.9. Therefore, the effluent COD, BOD and TN concentration in system F are the lowest among all treatment systems in Figure 4.12. As shown in table 4.4, the influent particulate N fractions of the specific food wastewater quality is only 15% and other food wastewater quality has similar characteristics. TN in food processing wastewater is mostly composed of NH_4^+_N and soluble ON. Hence, the influence of micro-sieving on COD or BOD removal is more significant than that on TN removal.

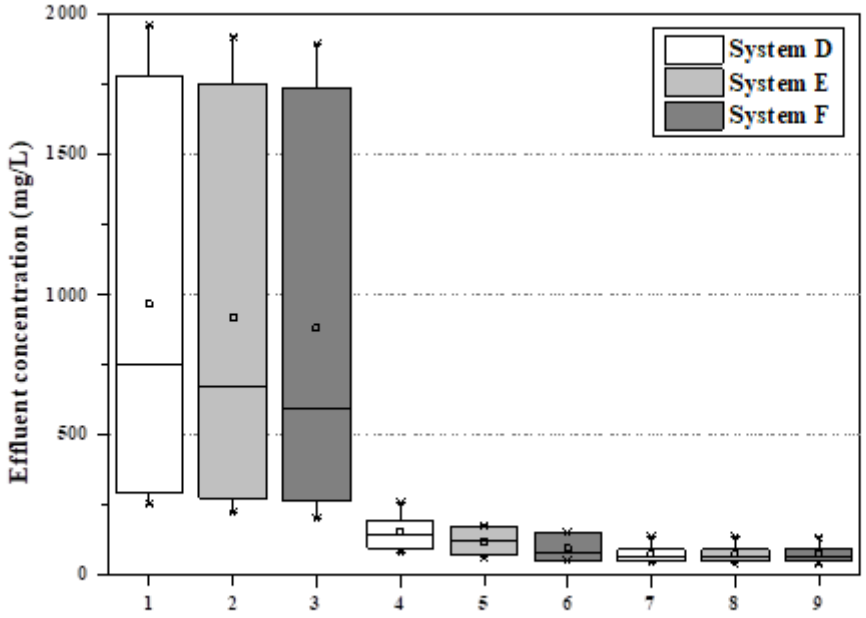

Figure 4.12 Wastewater quality in the effluent

Table 4.6 shows current discharge standards of organics and nutrients in food wastewater in different countries, including European [138], the USA [120], China [139], and India [140]. Figure 4.12 indicates that the effluent wastewater quality of system F is 53.88-153.45 mg/L BOD and 41.65-137.24 mg/L TN, respectively. This energy-positive system with micro-sieving could achieve COD removal of 55.3%-90.3% and NH_4^+_N removal of 90%. However, effluent wastewater quality of energy positive system might still do not meet the current discharge standards due to the high concentration of COD, BOD, and TN in food processing wastewater. Therefore, mainstream anaerobic treatment followed by mainstream anammox treatment was proposed as the best energy-positive system for treating municipal wastewater [91]. In addition, this energy-positive system should combine with aerobic treatment and physical-chemical treatment to increase pollutants removal efficiency to meet the discharge requirement of food processing wastewater.

Table 4.6 Discharge standards of food wastewater in different countries

Parameter	EU		USA		China		India	
	Min	Max	Min	Max	Min	Max	Min	Max
BOD (mg/L)	25		16	26	20	100	30	100
TN (mg/L)	10	15	4	8	15	20	10	50

4.4.4. Unit energy consumption and production

4.4.4.1. Theoretical and actual energy production of UASB

In this study, empiric equations (4.7-4.11) were used for estimating theoretical COD removal and unit energy consumption of UASB, as shown in table 4.7. Theoretical COD removal, unit methane production, and unit energy production at 25 ℃ are 54%-89%, 0.15-0.30 m^3 CH_4/kg $COD_{removed}$, and 0.52-1.03 kWh/kg $COD_{removed}$.

Table 4.7 Theoretical COD removal, unit methane production, and unit energy production of UASB in system D without primary treatment

BOD/COD ratio	0.57	0.46	0.29	0.53	0.34	0.30
COD concentration (mg/L)	2100.00	2350.00	4221.00	1697.00	3900.00	5422.25
COD removal (%)	88.08	85.45	53.85	82.66	54.56	79.01
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.27	0.22	0.17	0.26	0.21	0.14
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.30	0.24	0.19	0.28	0.23	0.15
Unit energy consumption (kWh/kg $COD_{removed}$)	1.03	0.83	0.65	0.97	0.80	0.52

* The following reaction is used for converting COD digested in UASB to methane:

$CH_4+2O_2 \rightarrow CO_2+2H_2O$

1 kg COD digested could be converted to 0.25 kg CH_4 or 0.35m^3 CH_4 theoretically at 0℃ and 1 atm [28]

In table 4.8, data about COD removal and methane production of UASB for removing untreated food processing wastewater was collected from published peer-reviewed papers [141-143]. Actual COD removal, unit methane production, and unit

energy production at 25 ℃ are 43%-95%, 0.01-0.38 m^3 CH_4/kg $COD_{removed}$, and 0.03-1.30 kWh/kg $COD_{removed}$.

As shown in Table 4.7, 4.8, and Figure 4.13, theoretical energy productions are close to the actual values, which indicates the validity of empirical equations used in my study. Theoretical energy production in system F is higher than those in systems D and F. Micro-sieving could remove more particulate non-biodegradable COD, which leads to a higher BOD/COD ratio of wastewater to UASB in system F. Non-biodegradable particulate COD, which could be not converted to biogas, increases COD removal of UASB in systems E and F. This results in lower unit energy production in systems E and F. The unit energy production per BOD removal shows an opposite trend in Figure 4.14. BOD removal does not include non-biodegradable particulate organic matter removal. This study considers the physical leakage of methane. 2.5 mg/L is assumed to be the dissolved CH_4 concentration in the anaerobic effluent based on The Handbook of Biological Wastewater Treatment [6]. In system F, micro sieving reduces BOD mass load to UASB. However, dissolved CH_4 concentration in the anaerobic effluent does not change in all systems. This leads to lower biogas production and unit energy production per BOD removal in system F.

Table 4.8 Actual COD removal and unit methane production and unit energy production of UASB

HRT (h)	22	22	22	18	18	14	14	7.1	6.8	6.7	4.1	2.3
SRT (d)								60.3	23.4	14	14.4	3.3
Temperature (℃)	35	35	35	35	35	35	35	33	33	33	33	33
COD (mg/L)	2800	3200	4200	3000	6500	3900	6330	8201	5719	5256	5495	5514
COD removal (%)	83.2	84.4	89.2	82.5	90.6	78.5	85.7	78	73	77	83	68
L biogas/g COD removed	0.555	0.47	0.402	0.46	0.192	0.3	0.157					
L CH_4/kg SCOD removed (at 25℃ and 1 atm)								213	254	283	201	199

Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0°C and 1 atm) *	0.34	0.29	0.25	0.29	0.12	0.19	0.1	0.07	0.15	0.23	0.15	0.11
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25°C and 1 atm)	0.38	0.32	0.27	0.31	0.13	0.2	0.11	0.08	0.17	0.25	0.16	0.12
Unit energy consumption (kWh/kg $COD_{removed}$)	1.30	1.10	0.94	1.08	0.45	0.70	0.37	0.27	0.58	0.85	0.56	0.41
Reference	[141]							[142]				
HRT (h)	24	24	24	24	24	24	24	24	24	24	24	24
SRT (d)												
Temperature (°C)	36	36	36	36	36	36	36	36	36	36	36	36
COD (mg/L)	3500	6000	10000	20000	28000	32000	3500	6000	10000	20000	28000	32000
COD removal (%)	>90	>90	>90	48	45	43	>90	>90	>90	95	72	68
Specific methane production (LCH_4/g COD added at 36 °C)	0.21	0.15	0.12	0.02	0.01	0.004	0.28	0.19	0.18	0.21	0.08	0.04
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0°C and 1 atm) *	0.21	0.15	0.12	0.04	0.02	0.01	0.27	0.19	0.18	0.2	0.1	0.05
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25°C and 1 atm)	0.23	0.16	0.13	0.04	0.02	0.01	0.3	0.2	0.19	0.21	0.11	0.06
Unit energy consumption (kWh/kg $COD_{removed}$)	0.78	0.56	0.44	0.14	0.07	0.03	1.04	0.70	0.67	0.74	0.37	0.20
Reference	[143]											

* Unit CH_4 production at the standard temperature and pressure is estimated based on the assuming methane/biogas ratio of 70%, [144], 0°C, and 1 atm.

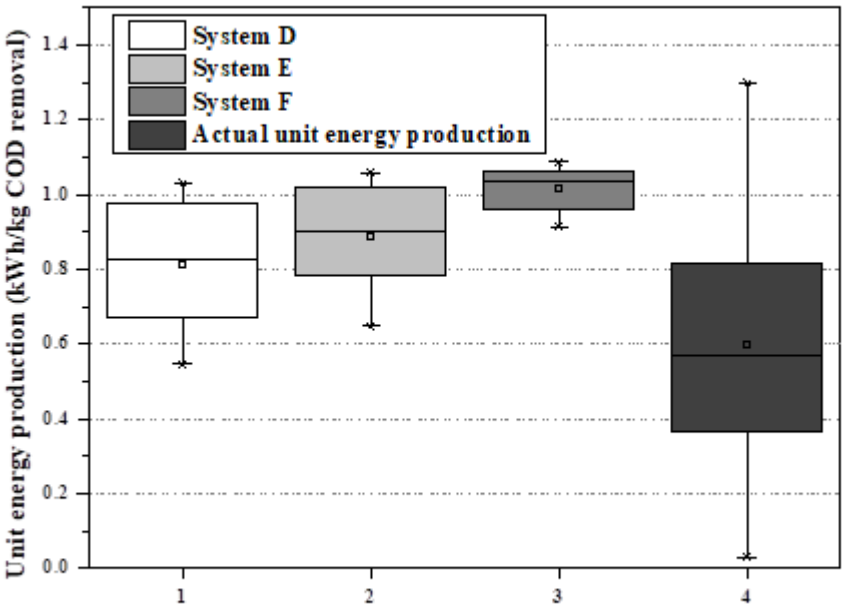

Figure 4.13 Actual and Theoretical unit energy production of UASB
Note: 1: theoretical unit energy production of UASB in system D, 2: theoretical unit energy production of UASB in system E, 3: theoretical unit energy production of UASB in system F, 4: actual unit energy production of UASB.

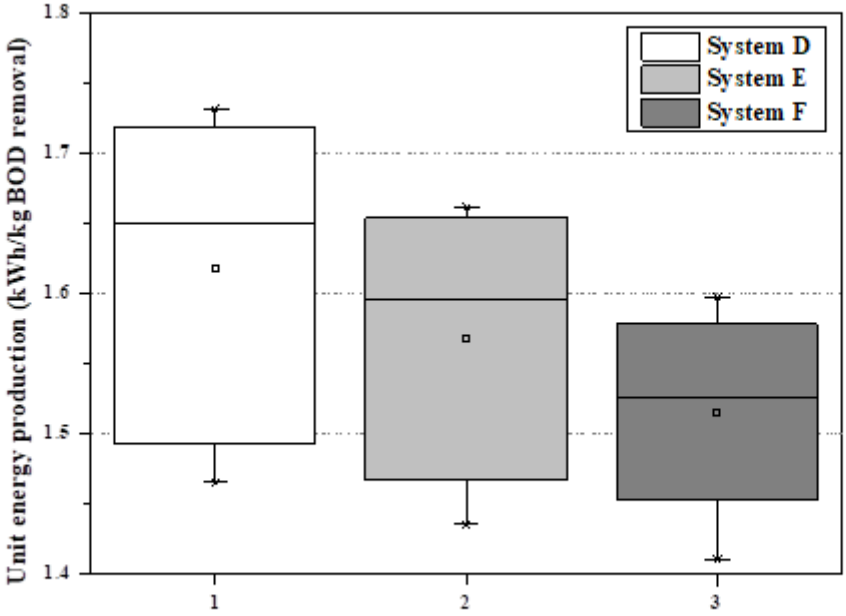

Figure 4.14 Theoretical unit energy production of UASB per BOD removal
Note: 1: theoretical unit energy production of UASB in system D, 2: theoretical unit energy production of UASB in system E, 3: theoretical unit energy production of UASB in system F

4.4.4.2. Theoretical and actual energy consumption of PN/A

In Table 4.9, the actual unit energy consumption of PN/A is 0.8-1.92 kWh/kg $N_{removed}$. Based on equations (4.15-4.16), 1.32kWh/kg $N_{removed}$ was used as the theoretical unit energy consumption of PN/A, which is close to the actual value, as shown in Figure 4.15. This proves the validity of the theoretical unit energy consumption used in my study. According to the benchmarking method reported by Yang et al. [145], the mean actual unit energy consumptions of 1.4 kWh/kg $N_{removed}$ could be selected as the benchmark data for comparing with actual unit energy consumption to check the energy efficiency of PN/A reactor.

Table 4.9 Energy consumption of PN/A reactors

Plant	Reactor type	Unit energy demand (kWh/kg $N_{removed}$)	Reference
Apeldoorn	SBR	1.1	[146]
Balingen	SBR	0.92	
Heidelberg	SBR	1.67	
Ingolstadt	SBR	1.92	
Nieuwegein	SBR	0.8	
Zurich	SBR	1.11	
Olburgen		1.86	
Malmö	MBBR	1.45-1.75	[88]

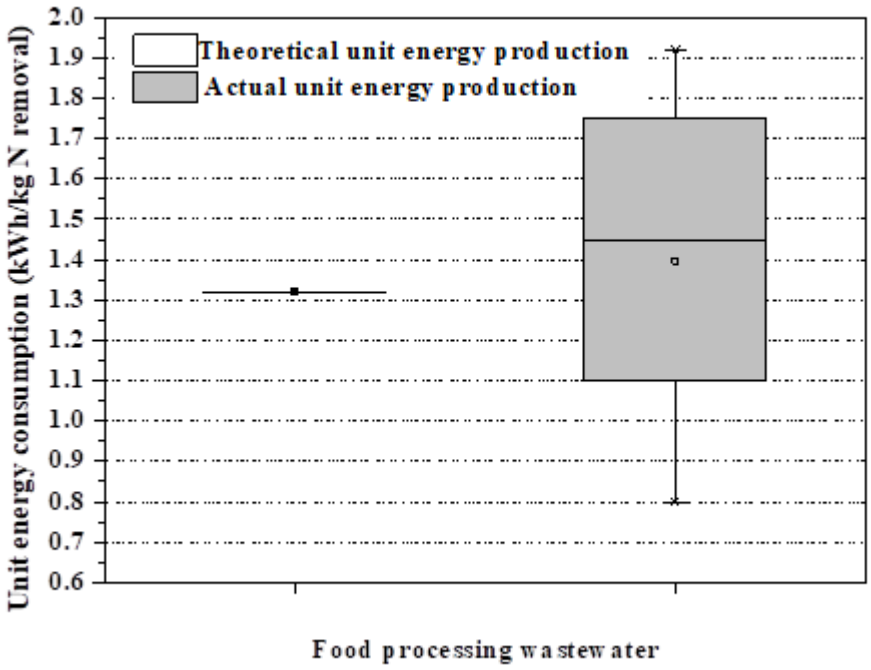

Figure 4.15 Actual and Theoretical unit energy consumption of PN/A

4.4.4.3. Energy consumption and production of WWTP

According to Sustainable Environmental Engineering [14], the unit energy indicator is critical when evaluating the energy performance of various WWTPs with different technologies. Unit energy metrics could be defined as the ratio between the daily energy consumption and treated COD mass removal (kWh/kg $COD_{removed}$) or TN mass removal (kWh/kg $N_{removed}$) [33, 147]. Since aeration demand for the aerobic biological process is the dominant energy demand in treating wastewater [113], aeration energy consumption was considered as electrical energy consumption. Since UASB reduces BOD to low concentration without oxygen consumption, O_2 is used for reducing N in the PN/A reactor. Mainstream UASB and side stream anaerobic digesters convert biodegradable soluble COD in wastewater and biodegradable particulate COD in sludge to CH_4 for energy recovery. Therefore, the unit energy consumption per unit of

N removal (kWh/kg $N_{removed}$) and unit of the unit energy production per unit of BOD removal (kWh/kg $BOD_{removed}$) were used in this study.

In figure 4.16, the unit energy consumptions of three systems are 0.72-1.46, 0.69-1.44, and 0.67-1.42 kWh/kg total N removal, respectively. Since oxygen is only consumed in the PN/A process, the unit energy consumption of the treatment system is close to the actual unit energy consumption of the PN/A reactor, as shown in table 4.9. Systems show low unit energy consumption because a part of NH_4^+-N and bON contributes to biomass production in UASB without consuming oxygen. Figures 4.16 suggests that the unit energy consumption of system F is lower than those of systems D and E because micro-sieving could remove more particulate biodegradable organic nitrogen than clarifier. Since pbON from micro-sieving to mainstream UASB is lower than that from the clarifier to UASB, more NH_4^+-N was consumed to form biomass during mainstream anaerobic digestion, and lower NH_4^+-N was transferred to PN/A reactors in system F. In contrast to system D, unit energy saving of systems E and F are 0.01-0.04 and 0.01-0.07 kWh/kg $N_{removed}$, which illustrates that primary treatment could help system D save unit energy by 1-4% and 1-8%.

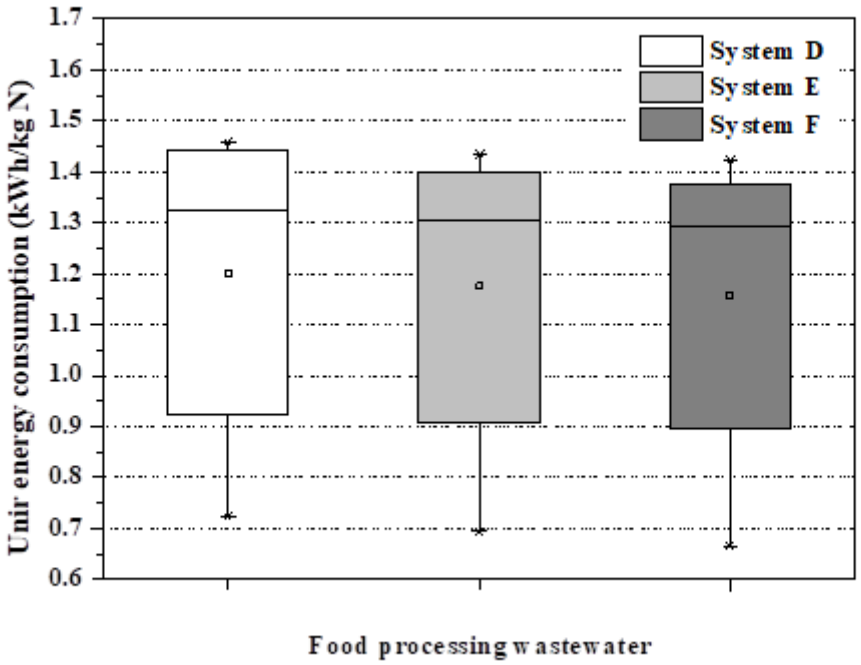

Figure 4.16 Unit energy consumption of system D, E, and F

Biodegradable COD contributes to CH_4 production to generate electricity. Since micro-sieving could separate more particulate BOD from wastewater to high-rate anaerobic digester than primary clarifier, anaerobic digester of system F could produce more electricity than that system D and E. However, this process decreases CH_4 production of mainstream anaerobic digester by reducing the BOD loading rate to UASB. In Figure 4.11, the bCOD digestion rate of UASB is 64-78 %, which is higher than the bCOD digestion rate of a high-rate anaerobic digester in equation (6.13). UASB with high sludge age of 40 d could convert BOD more efficiently to CH_4 than the high-rate anaerobic digester. As a result, system D could produce more electricity for removing the same amount of BOD than systems E and F, as shown in Figure 4.17. The mean value of unit energy production for systems D, E, and F are 2.04, 1.93, and 1.86, respectively. In contrast to system D and E, system F reduces unit energy

production by 4-15% and 1-8% because micro-sieving decreases particulate COD to UASB.

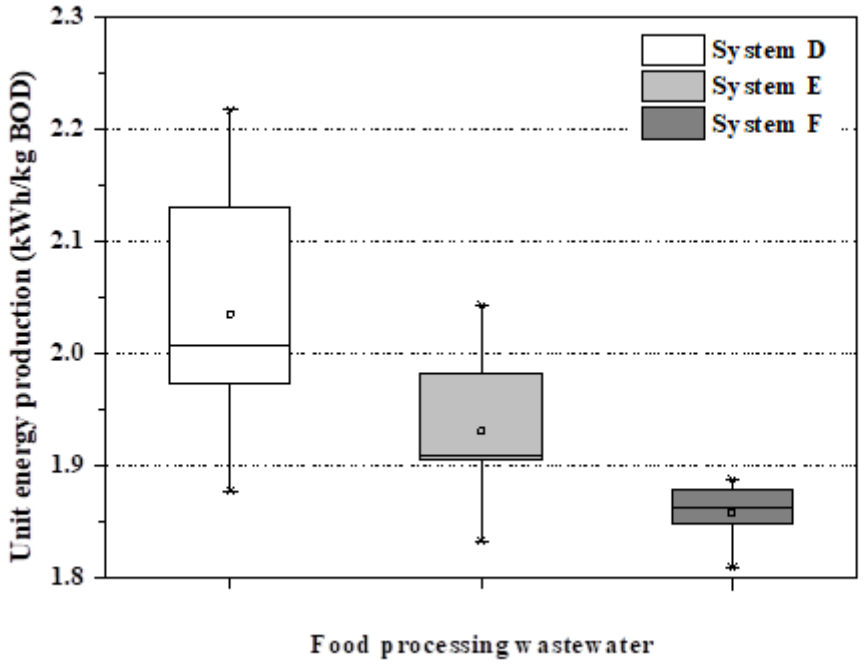

Figure 4.17 Unit energy production by system D, E, and F

4.4.5. Cost analysis

Treatment systems used different primary treatments for separating particulate COD, which leads to the different effluent COD concentrations of three systems in Figure 4.12. To compare the cost of three systems, unit cost metric ($/kg $COD_{removed}$) is selected as an economic indicator in this study. Cost analysis was performed according to the calculation of the Excel-based model to compare the unit cost of systems D, E and F as shown in Figure 4.18. The average unit cost of systems D, E and F are 0.60, 0.52, and 0.44 $/kg $COD_{removed}$, respectively. Compared with systems D and E, the average unit cost saving of system F was 0.16 and 0.08 $/kg $COD_{removed}$, respectively. System F could reduce unit cost by 27 % and 16 % than system D and E. As shown in figures 4.19 and 20, the percentage of capital cost or O&M cost for UASB

reactor for system F is lower than those for systems D and E. The cost of mainstream UASB is higher than the cost of other reactors in table 4.10, reducing the size of mainstream UASB could significantly decrease the total capital and O&M cost of a WWTP. Figure 4.21 shows that primary clarifier and micro sieving could reduce 32%-47% and 50%-93% of the total cost for UASB reactor. Low cost of mainstream UASB make up for disadvantage of low energy recovery efficiency in system F. Therefore, system with micro- sieving is more economic than other systems.

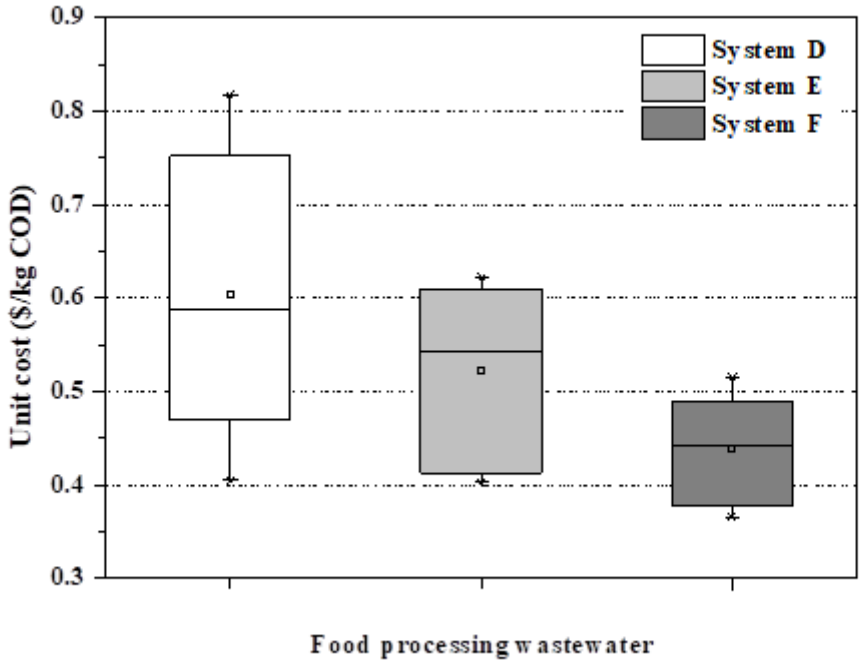

Figure 4.18 Unit daily cost of system D, E, and F

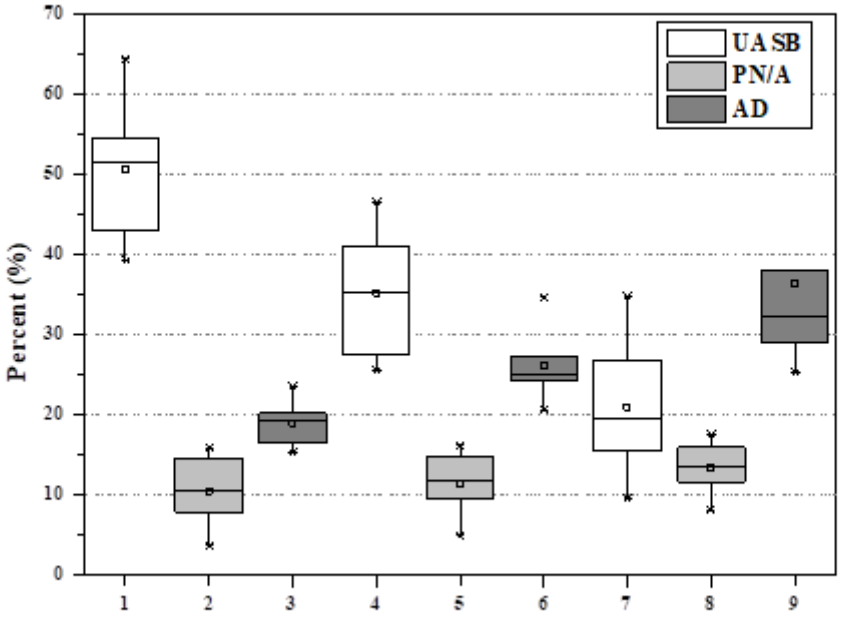

Figure 4.19 Percent distribution of capital costs

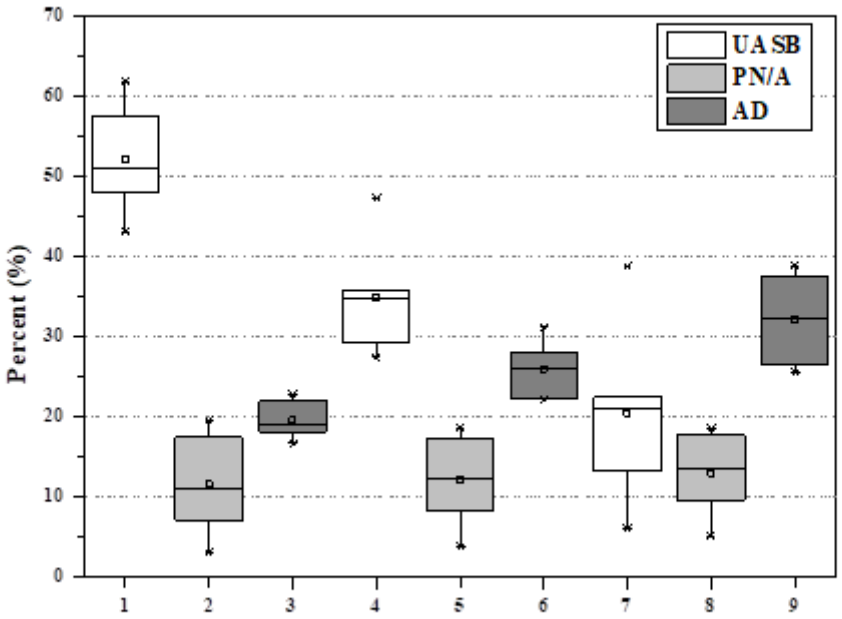

Figure 4.20 Percent distribution of O&M costs

Table 4.10 Costing parameters for different WWTP sizes

Capacity (P.E)	25000			50000			100000			200000		
Flow rate[a] (m^3/d)	3000			6000			12000			24000		
Capital cost ($/$m^3$ volume)	min	max	mean	min	max	Mean	Min	Max	mean	min	max	mean
UASB	600	1000	800	500	700	600	350	500	425	250	400	325
Aeration tank	220	300	260	180	250	215	150	200	175	120	170	145
final settler	350	550	450	300	400	350	250	330	290	200	260	230
Anaerobic digester	600	1000	800	450	700	575	300	400	350	250	350	300

[a]Flow rates were estimated based on capacity (P.E.) and wastewater production of 120 L/P.E./d (Schaum et al., 2015; Fatta and Anayiotou, 2007)

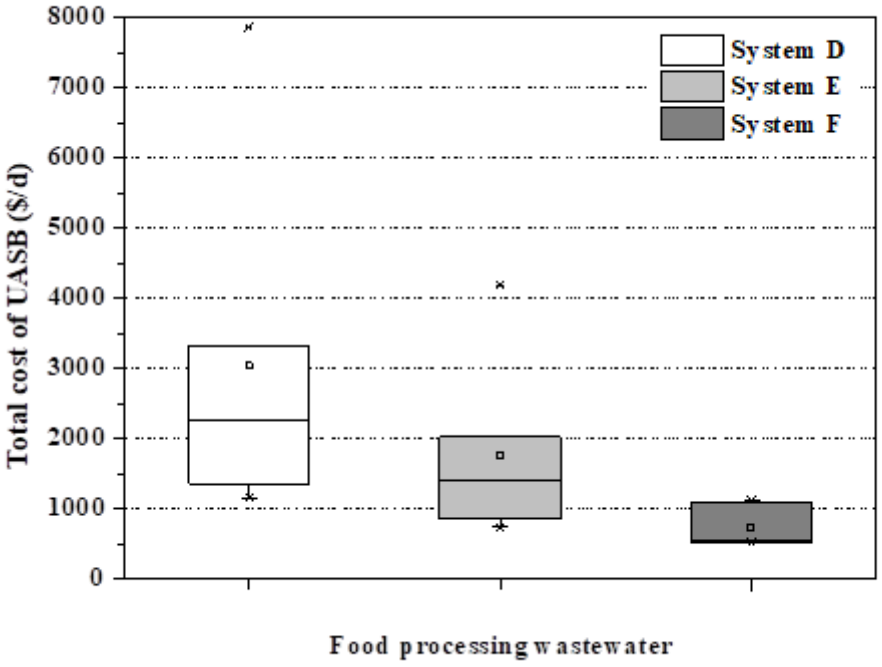

Figure 4.21 Total costs of UASB in system D, E and F

4.5. Conclusions

The energy and cost of three different treatment designs were quantified according to the design principles of Sustainable Environmental Engineering. Food wastewater quality data was compiled from peer reviewed papers and Excel-based models were developed to compare the performance of food wastewater treatment system with different treatment technologies. 30%, 50%, 60%, 65%, 70%, 75%, 80%, 85%, and 90% were assumed as TSS removal of primary clarifier and micro sieving. Based on the assumptions, Excel-based models were developed to compare the performance of conventional and the innovative nitrogen removal method with different primary suspended solids removal in terms of energy metrics. Oxygen requirements in systems A, B and C decrease at the average rate of 1.7%, 2.3% and 0.5% per 5% TSS removal. Since UASB converts soluble BOD in wastewater to biogas, methane production of system C is 1.6-2.5 times that of system A and B. Innovative system C with any primary suspended solids removal could achieve electrical self-sufficiency due to its CH_4/O_2 energy ratio of higher than 8. However, methane production and CH_4/O_2 energy ratio of system C decrease at the average rate of 0.35% and 0.1 % per 5% TSS removal. The impact of micro-sieving on energy recovery of an innovative system is negative. On the other hand, micro sieving also reduces the required size and cost of UASB reactor through decreasing BOD mass load to UASB, systems with different primary treatment were explored to quantify the impact of micro-sieving on innovative systems from cost dimensions. Compared to the primary clarifier, micro-sieving has various advantages such as small footprint, modular and flexible design, recyclable materials, and high solid removal with high percent dry solids. For treating wastewater with high TSS concentration, micro-sieving could retain more particulates on the filter and divert more COD to anaerobic digester than clarifier. Compared to the system without primary

treatment, micro-sieving could increase the biodegradability of wastewater by 5-40% and increased COD removal of 1-4% in food processing wastewater treatment system. However, it could not significantly change biodegradable COD digestion rate because high biodegradability increases the fraction of COD digested, and COD discharged as sludge. Energy-positive systems with micro-sieving could remove 55.3%-90.3% COD and 90% NH_4^+_N, respectively. Effluent wastewater quality of an energy positive system might still do not meet the discharge standards due to the high concentration of COD, BOD, and TN in food processing wastewater. This energy-positive system should combine with physical-chemical treatment to increase pollutants removal efficiency to meet the discharge requirement. Since primary treatment could reduce O_2 demand of PN/A reactor by separating particulate N and COD to side stream anaerobic digester, the energy analysis showed that primary clarifier and micro-sieving could help the treatment system without primary treatment save unit energy by 1-4% and 1-8%, respectively. Micro-sieving decreases CH_4 production of mainstream anaerobic digester through reducing BOD loading rate to UASB, which leads to a low unit energy production of system F. Since micro-sieving also reduces the size and cost of UASB, the economic benefit due to the reduced size of mainstream reactor compensates for the disadvantage of low biogas production. In operation cost analysis, the mean unit cost saving of system F is 0.16 and 0.08 \$/kg $COD_{removed}$, respectively. System F could reduce unit cost by 27% and 16% than system D and E. These aspects demonstrate that micro-sieving could help energy-positive food wastewater treatment system save cost.

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5. TREATMENT PERFORMANCE, ENERGY, AND COST ANALYSIS OF ENERGY-POSITIVE WASTEWATER TREATMENT SYSTEMS FOR DIFFERENT INDUSTRIAL WASTEWATERS

5.1. Abstract

Different industrial wastewaters have vastly different wastewater qualities. When biodegradability is high, an energy-positive treatment train could be designed according to the twelve design principles of Sustainable Environmental Engineering. Two of the major indicators are unit energy and cost per chemical oxygen demand removed. In the past, the activated sludge process has been used as the conventional method in the treatment of industrial wastewaters. However, intensive energy requirements for aeration made it impossible for the WWTP to be energy positive. In addition, there is little research assessing the efficiency and economic feasibility of innovative technologies in treating different kinds of industrial wastewater. This study tries to assess the effect of different kinds of industrial wastewater on treatment efficiency, energy, and cost of micro-sieving, UASB, and PN/A from the perspectives of unit energy and cost. An Excel model was developed for comparing the unit energy and cost of the treatment system for treating different kinds of industrial wastewater. Results showed that micro-sieving could remove 32.50-39.72% of COD for industrial wastewater with different TSS concentrations. Due to the relatively high TSS/COD ratio, primary treatment could achieve high COD removal efficiency of textile wastewater with low TSS concentration. A UASB reactor could remove 15.8%-53.5%, 14.0%-49.0%, and 22.9%-51.0% of COD for three kinds of wastewater. UASB converts more COD to CH_4 in meat process wastewater with high BOD concentration because of high biodegradability. For meat processing wastewater, tannery wastewater, and textile wastewater, mean unit energy production in innovative systems are 1.80,

1.77, and 1.73 kWh/kg $BOD_{removed}$, respectively. The energy-positive system for treating meat processing wastewater could produce more energy with the same BOD removal than that for treating tannery wastewater and textile wastewater. The average unit cost for three kinds of wastewater is 0.54, 0.57, and 1.12 \$/kg $COD_{removed}$, respectively. The meat processing wastewater system consumes more oxygen for removing N in PN/A. However, this process increases the CH_4 production of AD through increasing biomass. These aspects reduce the unit cost of the meat processing wastewater treatment system. Therefore, it is more economical to treat meat processing wastewater with innovative technologies.

5.2. Introduction

Industrial wastewater has vastly different water quality. For example, the water quality of food processing, pulp and paper, textile, chemical, pharmaceutical, petroleum, tannery, and manufacturing industries vary significantly [116]. The major wastewater quality parameters include chemical oxygen demand, biochemical oxygen demand, suspended solids, ammonium nitrogen, heavy metals, pH, color, turbidity, and biological parameters. Compared with municipal wastewater, industrial wastewaters usually have a high organic matter concentration, extreme physicochemical nature (e.g., pH, temperature, salinity), and humic substances that may inhibit biological treatment processes. Municipal wastewater has a low strength concentration of COD (250–800 mg/L), whereas strong (>1,000 mg COD/L) to extremely strong wastewaters are often produced by industries [148]. Olive mills and beverage production industries can generate extremely strong industrial wastewaters (COD>200,000 mg/L) [149, 150]. Characteristics of industrial wastewaters strongly depend on the type of industrial wastewaters and industrial processes. Water used by meat processing industries accounts for 29% of agricultural freshwater worldwide [151, 152]. Food processing

wastewater is typically generated from slaughtering houses. Accordingly, there is a large quantity of suspended organic matter, protein, and fat (i.e., bones, meat, and viscera) in meat processing wastewater. A high concentration of COD, BOD, TN, and TSS is typical for food processing wastewater [118, 119, 153]. The average COD, BOD, TN and TSS concentration of food wastewater can reach 4,221 mg/L, 1,209 mg/L, 427 mg/L, and 1,164 mg/L, respectively [117]. Meat processing wastewater is considered one of the most detrimental food industrial wastewaters because its inadequate disposal can lead to river deoxygenation, algal blooms, and eutrophication [120, 121]. The tannery industry, which can be defined as the production process of leather by treating sheep and goat skins and bovine hides, is one of the oldest industries worldwide [154]. Over the centuries, leather was used for producing high durability clothes and footwear. Nowadays, leather is still one of the few available materials to produce clothing and footwear because of its unique properties [155]. The tannery industry consumes large amounts of water and chemicals. Wastewaters are mainly produced during the wet processes of the tannery industry. For producing 250 kg of leather, 15–80 m^3 of tannery wastewater with 230–250 kg COD, 100 kg BOD, 150 kg TSS, 5–6 kg chrome, and 10 kg sulphide is produced [156, 157]. Tannery effluents are characterized by high pollution loads of conventional pollutants and suspended solids with low BOD/COD ratio, which cause deleterious effects on the natural water system [158-165]. The textile industry, representing one of the largest and most widespread production industries in the world, produces a large number of dyestuffs ($>7\times10^5$ tons) per year worldwide [166]. The textile industry consumes not only a large amount of process water but also a variety of chemicals. Complex wastewater with dyestuff, surface-active materials, and textile additives is produced from the spent dye bath and rinsing waters [167-169]. Textile effluent is typically a complex mixture of organic

and inorganic matters with relatively strong color, high COD, high salinity, high temperature, variable pH, and low BOD [170, 171]. The direct discharge of textile wastewater into rivers and streams can cause severe environmental problems [172]. Many dyes give the textile effluent color and are carcinogenic to humans and toxic to aquatic organisms [168, 173-175]. The discharge of untreated textile wastewater deteriorates water quality, adversely impacting aquatic life, crops, and humans [176-178].

Biological and chemical processes are traditionally used in treating meat processing, tannery, and textile wastewater [179-199]. Physical and chemical methods are very expensive because they consume a large amount of energy and chemicals and produce a large amount of excessive sludge [200-202]. Compared with oxidation processes and chemical processes, biological treatment methods have many advantages: (1) low capital and operating costs; (2) oxidation of a wide variety of organic matters, versus mere phase separation, such as air stripping or carbon adsorption; (3) removal of reduced inorganic compounds, such as ammonia, and total nitrogen removal by nitrification and denitrification; and (4) operational flexibility to treat different kinds of wastewater with a wide range of flows. The unit energy consumption of meat processing plants using biological process is reported as 0.81 kWh/kg $COD_{removed}$ [203]. Yang et al. [204] compared the performance and energy of textile wastewater treatment by conventional activated sludge, MBR, and moving bed biofilm reactor. Results showed that the unit energy consumption of CAS is 0.58 kW/kg $COD_{removed}$.

Activated sludge systems with autotrophic nitrification/heterotrophic denitrification process with primary clarifier are conventional processes for treating industrial and municipal wastewater. However, they show high energy consumption, a

large footprint, and a large amount of sludge production as compared with innovative biological technologies such as micro-sieving, UASB, and PN/A. Micro-sieving is one of the most sustainable technologies to separate TSS, with significant capital costs and footprint savings. Mainstream UASB can achieve energy recovery by converting COD to biogas without consuming O_2 at operating temperatures between 15 and 35°C [20, 21]. PN/A, which is an innovative N removal technology discovered in the 1990s [24, 81, 82], can remove 90% of NH_4 _N through converting ammonia and nitrite N_2 in the anammox process [83-86]. As compared to the conventional nitrification/denitrification process, the PN/A process could save O_2 demand of nitrification by approximately 60%, eliminate the external organic carbon source requirement for denitrification, and decrease sludge by 80-90% [87, 89, 122]. Currently, there are few cases of research assessing the efficiency and economic feasibility of innovative technology in treating different kinds of industrial wastewater. Therefore, the effects of different kinds of industrial wastewater on the performance, energy, and cost of using micro-sieving, UASB, and PN/A were quantified from energy and cost dimensions.

The objectives of this study are to (1) evaluate the influence of different industrial wastewater on the performance of primary treatment and secondary treatment; (2) compare the unit energy consumption and production of different industrial wastewater treatment; and (3) analyze unit cost of treatment systems based on different industrial wastewater quality. In addition, the economic advantages and limitations of innovative technologies for treating different industrial wastewater are addressed to offer insight for sustainable treatment design of different industrial wastewater.

5.3. Material and methods

In the current study, energy and cost were calculated based on different kinds of industrial wastewater quality. Wastewater quality data were collected from published peer-reviewed papers, as shown in Table 5.1. The database of untreated wastewater quality includes basic information such as the concentration of TSS, COD, BOD, and NH_4^+_N. The concentrations of TSS, COD, BOD, and NH_4^+_N of meat processing wastewater is higher than those of tannery wastewater and textile wastewater, which result in a higher amount of primary sludge production, oxygen demand, and CH_4 production. An Excel-based model was developed. One million gallons per day (MGD) wastewater treatment systems with different primary and secondary treatment processes were designed [26, 33, 34, 39, 40, 147]. Figures 5.1 and 5.2 show the flow diagram of industrial wastewater treatment systems. In all systems, the screen and grit chamber were selected as the preliminary treatment in removing large particulate matter from wastewater. The difference between systems is the primary and secondary treatment processes. In Systems A and B, primary clarifier and micro-sieving removed particulate COD and N and produced a certain amount of primary sludge, which resulted in the difference of aerobic treatment, UASB, PN/A reactor, and side stream AD design. The nitrification-denitrification process was used as the main system of System A for treating wastewater to meet the discharge requirement by converting biodegradable COD and N to CO_2 and N_2. System B used mainstream UASB and PN/A as main systems for removing soluble BOD and N, which reduces aeration consumption through decreasing BOD and N loading to the nitrification-denitrification system. To meet discharge requirements and compare the performance of the wastewater treatment system for different industrial wastewater, System B used UASB, PN/A, and nitrification–denitrification to achieve the same effluent BOD and N concentration as

System A. The following equations are used in calculating the unit energy consumption, production and capital cost, and operation and maintenance (O&M) cost [26, 40]:

$$\text{Unit energy consumption (kWh/m}^3\text{)} = \frac{\text{Aeration demand of PN/A and NDN (kgO}_2\text{/d)}}{\text{Oxygen transfer efficiency (actual)Q}} \tag{5.1}$$

$$\text{Unit energy consumption (kWh/kg N}_{\text{removed}}\text{)} = \frac{\text{Aeration demand of PN/A and NDN (kgO}_2\text{/d)}}{\text{Oxygen transfer efficiency (actual)(N removal (kg/d))}}$$

where

Q=flow rate (m^3/d)

Oxygen transfer efficiency (actual)= 1.2 kg O_2/kWh

$$\text{Energy production(kWh/d)} = \text{(Energy recovery) (Enthalpy of combustion) (CH}_4\text{ production from UASB and anaerobic digester (kg/d))} \tag{5.2}$$

where

Energy recovery = 38%

Enthalpy of combustion = 13.9 kWh/kg CH_4

$$\text{Unit energy production(kWh/kg BOD}_{\text{removed}}\text{)} = \frac{\text{Energy production(kWh/d)}}{\text{BOD removal (kg/d)}} \tag{5.3}$$

(5.4)

It was assumed that UASB combining the heat and power technologies could convert 38% of CH_4 formed in the mainstream and the side stream anaerobic digestion to electricity with an energy density of approximately 13.9 kWh/kg CH_4 [40]. The energy demand of a WWTP mainly includes power consumption of the wastewater lift

pump, aeration equipment, and sludge treatment. The energy consumption of biological treatment accounts for 50–70% of the overall energy consumption [41]. UASB removes COD without consuming oxygen, the aeration energy of PN/A and the nitrification–denitrification process was the dominant energy demand. biodegradable COD in wastewater could be converted into CH_4 at 15 and 35℃ [20], the operating temperature of the treatment system was assumed to be 25 ℃. Therefore, energy consumption is estimated based on O_2 mass demand and oxygen transfer efficiency at 25 ℃ [26].

$$\text{Capital cost(\$/yr)} = \frac{\text{Capital cost(\$)}}{a_{i,n}}$$

$$a_{i,n} = \frac{(1+i)^n - 1}{i(1+i)^n}$$

where

$a_{i,n}$=annualization factor

i =interest rate (annual)=6%

n=economic lifetime of the treatment plant in years=20 (yr)

(5.5)

$$\text{Unit capital cost (\$/m}^3\text{)} = \frac{\text{Capital cost(\$/yr)}}{\text{(Q)(365d/yr)}}$$

(5.6)

$$\text{Unit O \& M cost (\$/m}^3\text{)} = \frac{\text{O \& M cost(\$/yr)}}{\text{(Q)(365d/yr)}}$$

(5.7)

$$\text{Unit cost(\$/m}^3\text{)=Unit capital cost(\$/m}^3\text{)+Unit O\&M cost(\$/m}^3\text{)}$$

$$\text{Unit cost (\$/kg COD}_{\text{removed}}\text{)} = \frac{\text{Capital cost(\$/yr)}}{\text{(COD removal (kg/d))(365d/yr)}} + \frac{\text{O \& M cost(\$/yr)}}{\text{(COD removal (kg/d))(365d/yr)}}$$

(5.8)

For comparing the cost of treatment systems, the total capital costs ($) were annualized over the expected lifetime of the WWTP. Economic lifetime and interest rate were assumed to be 20 years and 6% [26]. Capital costs ($) were transformed into annualized capital costs ($/yr) based on the interest rate and lifetime. The total unit cost was calculated as the combination of the unit capital costs and the unit operational costs.

Table 5.1 Industrial Wastewater quality

Type of wastewater	TSS	BOD	COD	TN	Reference
Meat processing wastewater	950.00	1200.00	2100.00	220.00	[128]
	1400.00	1070.00	2350.00	317.22	[129]
	1164.00	1209.00	4221.00	427.00	[117]
	662.00	891.00	1697.00	246.00	[130]
	625.00	1320.00	3900.00	217.00	[131]
	3438.22	1602.00	5422.25	361.25	[132]
Tannery wastewater	890.00	665.83	2,290.00	282.00	[205]
	2,690.00	1,470.00	3,700.00	293.93	[154]
	915.00	1,024.89	2,155.00	228.00	[154]
	1,147.00	1,126.00	3,114.00	131.70	[154]
	2,229.00	1,760.00	5,094.00	358.00	[206]
	1,150.00	1,746.00	6,240.00	327.00	[207]
	1,550.00	463.25	3,280.00	260.00	[205]
Textile wastewater	137.00	455.00	1411.00	49.20	[208]
	324.00	283.00	513.00	28.70	[209]
	150.00	150.00	910.00	40.00	[210]
	460.00	198.00	714.00	18.70	[211]
	520.00	225.00	770.00	23.00	[211]
	438.00	220.00	798.00	24.70	[211]

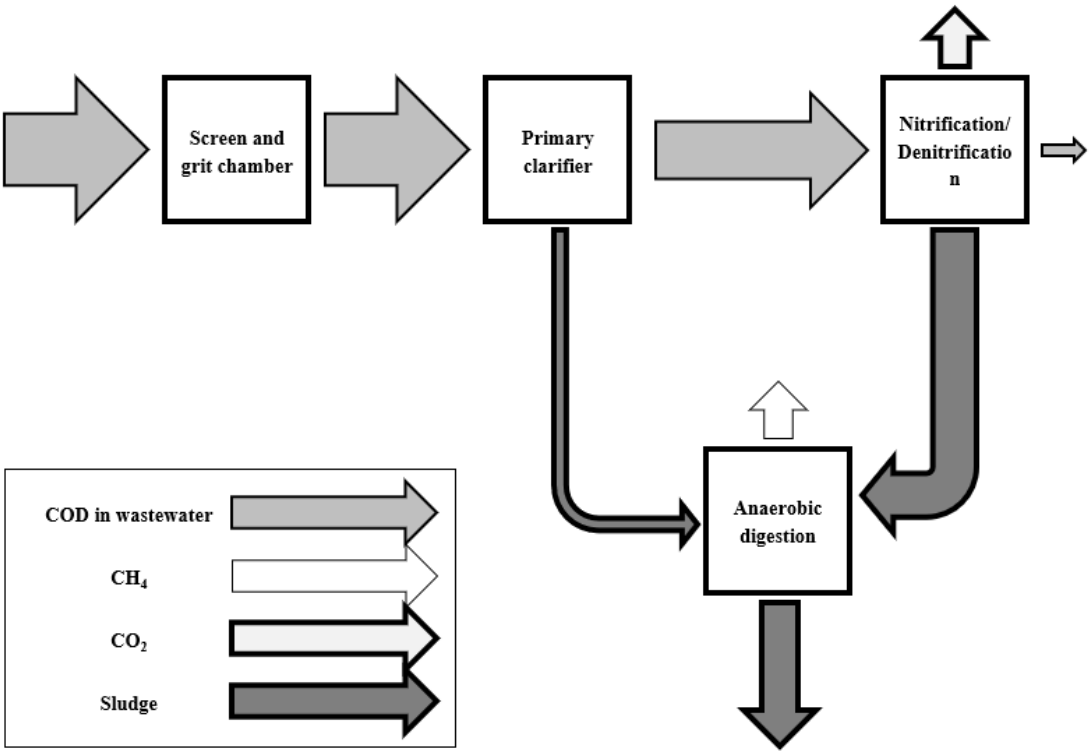


Figure 5.1 Schematic graphs of wastewater treatment system A- direct line nitrification–denitrification process

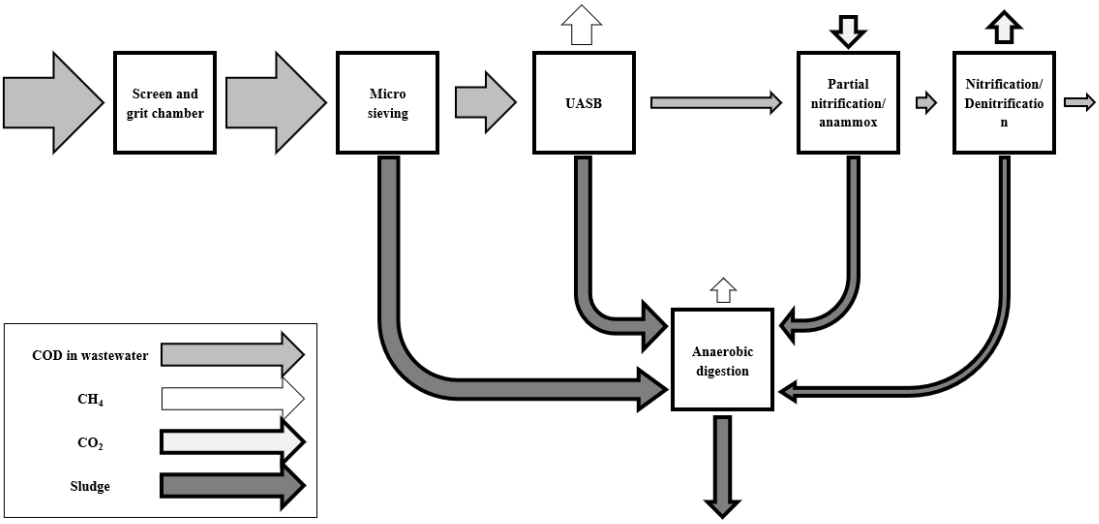


Figure 5.2 Scheme of wastewater treatment system B: mainstream anaerobic treatment and anammox treatment

5.4. Results and Discussions

5.4.1. Impact of different industrial wastewater quality

5.4.1.1. Treatment performance of primary treatment

Pollutants concentration and flow rate determine the mass of the contaminants need to be removed, whereas performance of wastewater treatment equipment depends on characteristics of wastewater such as particulate matter fraction and biodegradability. TSS/COD ratio and BOD/COD ratio are widely used as indicators of these two kinds of wastewater characteristics. Figure 5.3 compares these indicators of different industrial wastewater. It shows that the TSS/COD ratio of textile wastewater is higher than those of other industrial wastewaters. Due to relatively high TSS concentration, micro-sieving shows TSS removal of greater than 50% and significantly higher COD removal than the primary clarifier as shown in Figure 5.4. For tannery wastewater, meat processing wastewater, and textile wastewater, the average COD removal by micro-sieving are 35.36%, 39.08%, and 32.50%, respectively, which are higher than that by primary clarifier. Textile wastewater has a relatively lower TSS concentration than other industrial wastewaters in Table 5.1. However, primary clarifier in textile wastewater treatment shows higher COD removal than tannery wastewater, due to the high TSS/COD ratio of textile wastewater as shown in Figure 5.4. To study the influence of wastewater quality on COD removal of primary clarifier and micro-sieving, Pearson correlation analysis was carried out to evaluate the correlation between COD removal and related parameters at a level of significance of $p < 0.01$.

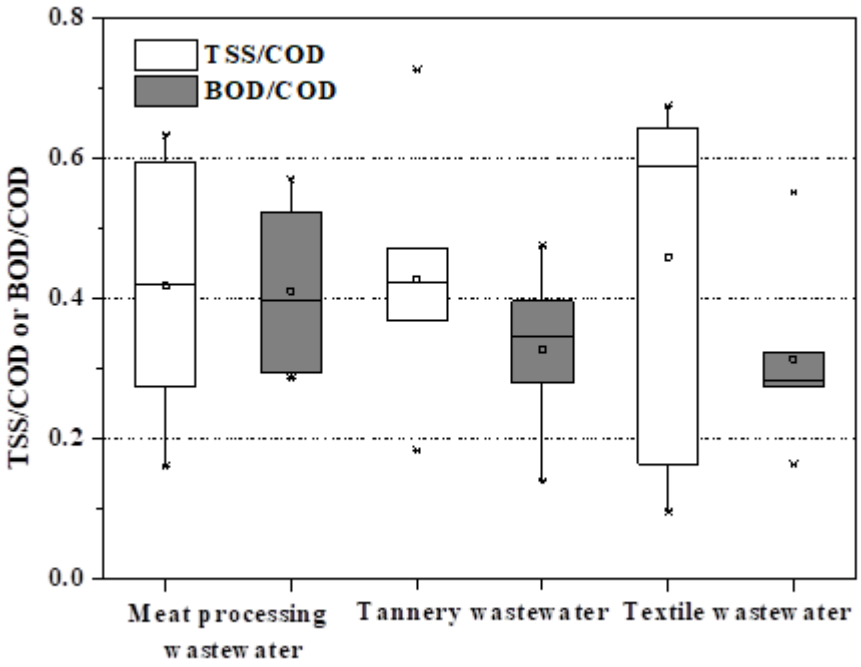


Figure 5.3 TSS/COD ratio and BOD/COD ratio for different industrial wastewater

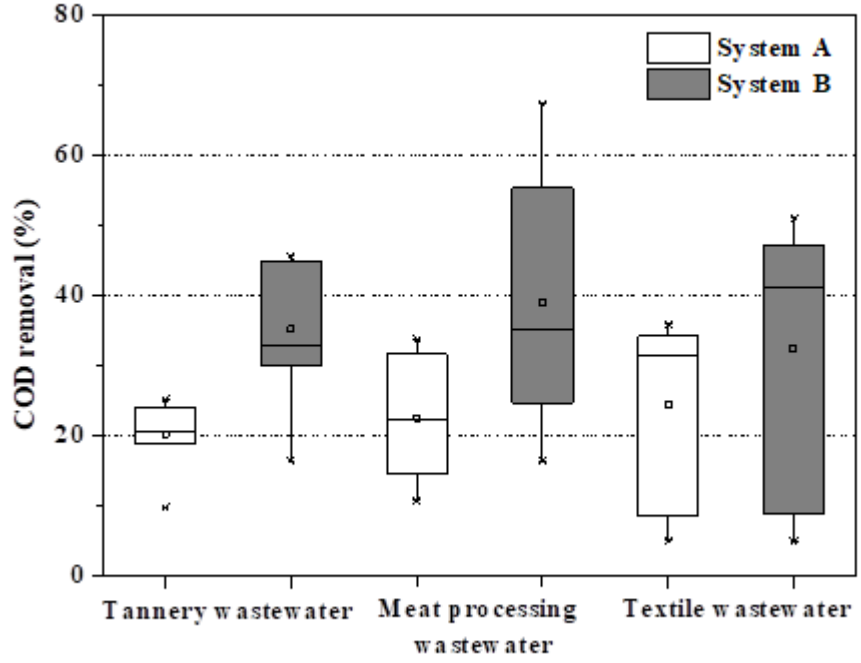


Figure 5.4 COD removal of primary treatment for different industrial wastewater

Table 5.2 shows the Pearson's correlations between COD removal of the two systems, concentration of TSS, and TSS/COD. Compared with TSS concentration, the ratio between TSS and COD showed stronger positive correlations with COD removal at the 0.01 level. Therefore, regression models of COD removal were developed based on TSS/COD ratio. The trend lines for predicting COD removal of primary treatment from TSS/COD is depicted in Figure 5.5. TSS concentration of textile wastewater is lower than those of other industrial wastewaters. However, textile wastewater has relatively low concentration of COD, as shown in Table 5.1. Particulate COD fraction of textile wastewater is higher than those of other industrial wastewaters. TSS/COD ratio is associated with particulate COD fraction, COD removal of the two systems rises with the increasing TSS/COD ratio of influent wastewater, as shown in Figure 5.5. Therefore, TSS/COD ratio is the key parameter for determining COD removal of primary treatment. Due to the high particulate matter fraction, primary treatment shows excellent separation performance for treating textile wastewater.

Table 5.2 Correlations between COD removal, TSS concentration and TSS/COD ratio of influent wastewater

Pearson Correlation	COD removal of system A (%)	COD removal of system B (%)
TSS (mg/L)	0.278	0.595**
TSS/COD (%)	0.941**	0.910**

** Correlation is significant at the 0.01 level (2-tailed).

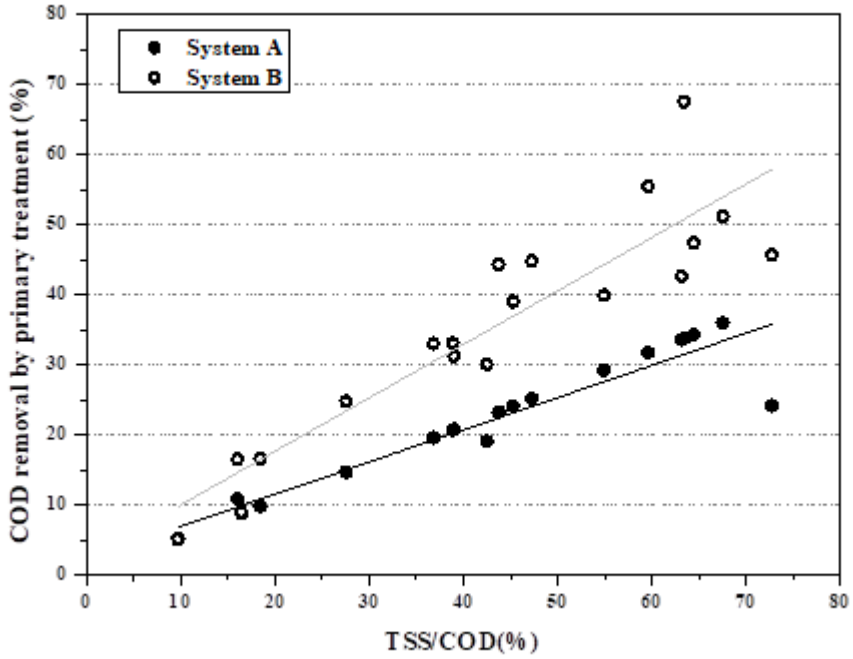


Figure 5.5 Variation of COD removal in primary treatment with TSS/COD ratio

Biodegradability, expressed as the BOD/COD ratio, represents the ability of a substance to be removed by microorganisms. It is commonly used as an index of the extent of biodegradation of wastewater [108]. High biodegradability is a sufficient condition of biodegradation. the biodegradability of meat processing wastewater, tannery wastewater, and textile wastewater are in Figure 5.3, which presents average BOD/COD ratios of 0.412, 0.327, and 0.314, respectively. Meat processing wastewater has higher biodegradability than other industrial wastewaters. Figure 5.6 shows the BOD/COD ratio of untreated wastewater and treated wastewater by primary clarifier and micro-sieving. Biodegradability rises with the increasing BOD/COD ratio of untreated wastewater. Figure 5.6 demonstrates that the slope of the linear fitting equation for System B is greater than that of System A, which illustrates that micro-sieving could improve the biodegradability of wastewater more significantly as

compared with the primary clarifier. Primary treatment increases the biodegradability of wastewater by reducing the particulate non-biodegradable COD fraction of industrial wastewater. Compared to System A, the BOD/COD ratio of wastewater increased by 0.59%-7.50%, 0.60%-3.57%, and 0.01%-2.19% for meat processing wastewater, tannery wastewater, and textile wastewater, respectively. Due to the relatively low TSS concentration of textile wastewater, TSS removal by primary clarifier and by micro-sieving and the variation of the BOD/COD ratio is smaller. Because meat processing wastewater has higher biodegradability and TSS concentration, micro-sieving increases mean and maximum BOD/COD ratio to 0.48 and 0.63. Hence, the biodegradability of industrial wastewater with a high concentration of TSS is easier to be improved by micro-sieving.

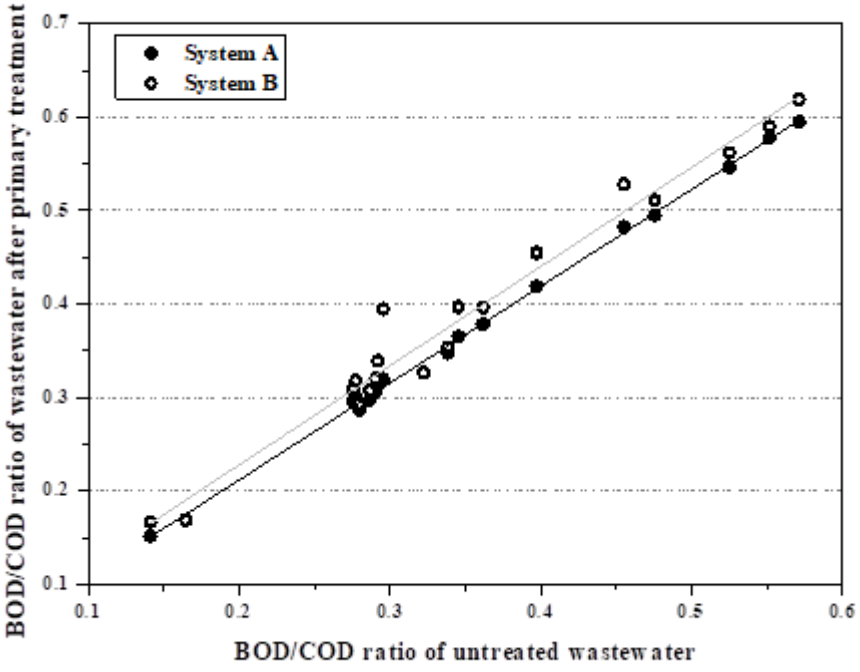


Figure 5.6 Variation of biodegradability after primary treatment

5.4.1.2. Treatment performance of secondary treatment

Due to the high COD removal of micro-sieving and the UASB reactor in System B, COD removal of aerobic treatment in System B decreases by 40.7%-66.5%, 33.6-60.0%, and 23.0%-60.0% for meat processing wastewater, tannery wastewater, and textile wastewater as compared to aerobic treatment in System A, as shown in Figure 5.7. Micro-sieving and UASB could help treatment systems save oxygen demand for all kinds of industrial wastewater. In secondary treatment, the UASB reactor removes 15.8%-53.5%, 14.0%-49.0%, and 22.9%-51.1% of COD for three kinds of industrial wastewater. Because of the higher BOD/COD ratio of treated meat processing wastewater by micro-sieving, UASB could convert more COD to CH_4 than other industrial wastewater, which resulted in less oxygen demand to remove the remaining COD.

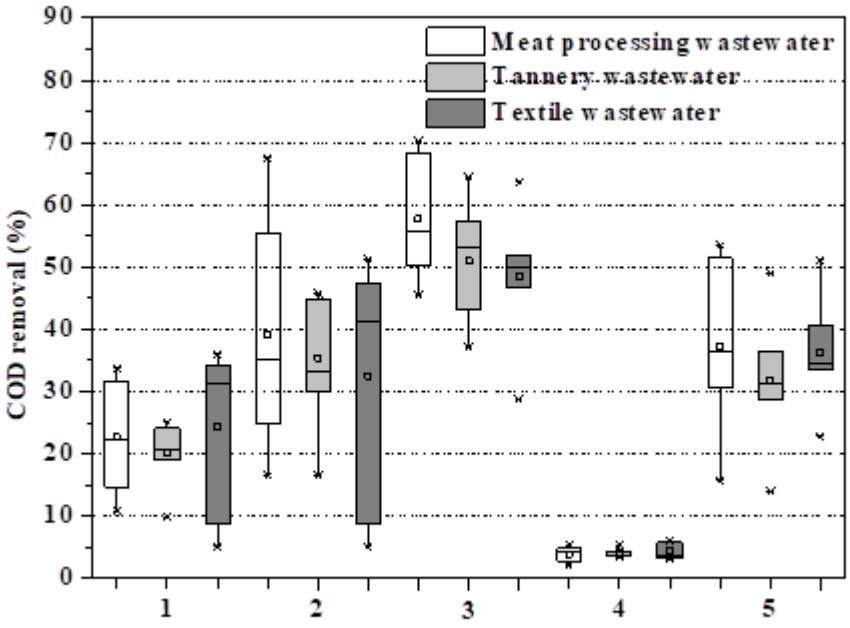


Figure 5.7 COD removal of primary and secondary treatment (1: Primary clarifier, 2: Micro sieving, 3: Aerobic treatment in system A, 4: Aerobic treatment in system B, 5: UASB reactor.)

Mainstream UASB converts biodegradable COD in wastewater to CH_4 for energy recovery. Since the BOD/COD ratio of treated wastewater by primary treatment is positively related to the influent wastewater quality as shown in Figure 5.6, CH_4 production/COD influent to secondary treatment (%) rises with increasing biodegradability in Figure 5.8. UASB removes COD through conversion of organic matter to biogas, COD removal by UASB reactor also shows strong positive correlations with the BOD/COD ratio of raw wastewater, as shown in Figure 5.8. This suggests that the biodegradability of wastewater could be used for predicting the methane conversion rate and COD removal of UASB reactor in industrial wastewater treatment systems with micro-sieving. Therefore, UASB shows excellent biogas production efficiency for treating industrial wastewater with high biodegradability.

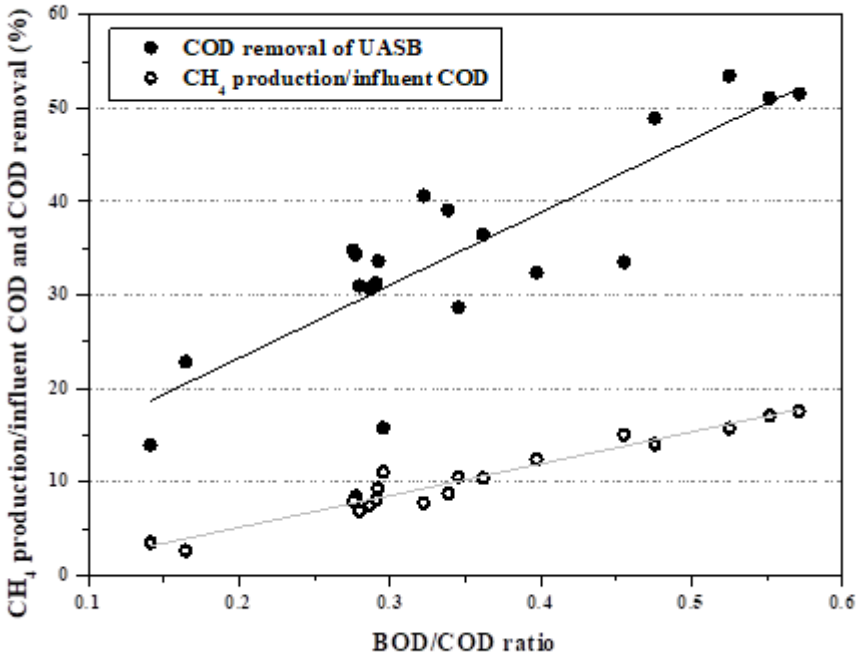


Figure 5.8 CH_4 production/influent COD to UASB and COD removal of UASB reactor

5.4.2. Effluent wastewater quality

Since aerobic treatment in Systems A and B is designed based on low effluent BOD and COD concentration, the two treatment systems could achieve similar COD removal and concentration in the effluent, as shown in Figures 5.7 and 5.9. Table 5.3 shows current discharge standards of organics in industrial wastewater in different countries, including European [138], the USA [120], China [139, 212], and India [140, 213, 214].

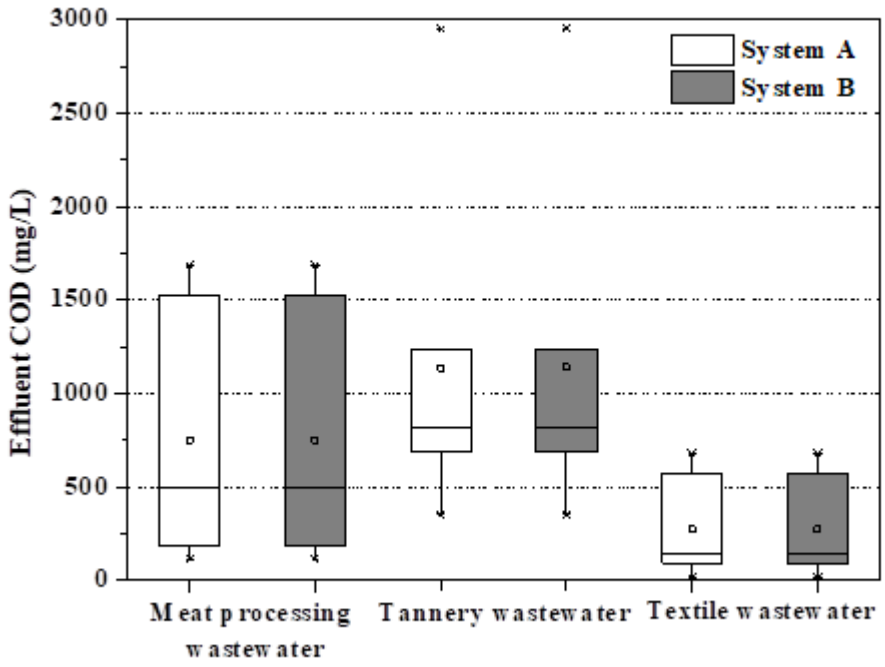


Figure 5.9 Effluent COD concentration of 3 industrial wastewater in two treatment systems

Table 5.3 Discharge standards of industrial wastewater in different countries

Type of wastewater Meat processing wastewater	Parameter	EU		USA		Canada		China		India	
		Min	Max	Min	Max	Min	Max	Min	Max	Min	Max
Meat processing wastewater	BOD (mg/L)	25		16	26	5	30	20	100	30	100
	COD (mg/L)	125						100	300	250	
Tannery wastewater	BOD (mg/L)									30	

	COD (mg/L)								250	
Textile wastewater	BOD (mg/L)							25	80	250
	COD (mg/L)							100	156	400

In Figure 5.9, effluent COD concentrations are 116.61-1683.80, 356.13-2952.27 and 14.19-680.48 mg/L, respectively. Most of the effluent wastewater quality in the two system does not meet the current discharge standards. This is due to a much higher concentration of COD, relatively low biodegradability, and the relatively high content of refractory soluble organic matter. Most of the remaining COD, which is soluble nonbiodegradable matter, is not easily removed by physical primary treatment and biological processes. Physical-chemical treatment such as adsorption might be required to remove soluble substances by the accumulation of those substances on activated carbon to increase COD removal efficiency and meet the discharge requirement. Figure 5.9 illustrates that the effluents of some meat processing wastewater and textile wastewater achieve the discharge requirement of several countries due to their low concentration of COD and relatively high BOD/COD ratio. Therefore, it is necessary to study the relationship between the total COD removal of primary treatment and biological processes and wastewater quality.

Table 5.4 shows the Pearson's correlations between COD removal, BOD/COD ratio, and TSS/COD ratio. The ratio between BOD/COD and the ratio between TSS and COD show strong positive correlations with COD removal at the 0.01 level. Since the BOD/COD ratio and TSS/COD ratio affect COD removal by UASB and primary treatment, regression models of COD removal were developed based on BOD/COD ratio and TSS/COD ratio.

Table 5.4 Correlations between COD removal, BOD/COD ratio and TSS/COD ratio of influent wastewater

Pearson Correlation	COD removal (%)
BOD/COD (%)	.712**
TSS/COD (%)	.804**

** Correlation is significant at the 0.01 level (2-tailed).

The equation for predicting COD removal by primary treatment and biological processes from BOD/COD and TSS/COD can be depicted as the following equation:

COD removal (%) =75.69(BOD/COD ratio) +57.53(TSS/COD ratio) R^2=0.929 +23.15 (5.9)

The squared correlation coefficient R^2 shows that the BOD/COD ratio and TSS/COD ratio of influent wastewater can explain 92.9% of COD removal. The BOD/COD ratio and TSS/COD ratio are positivly related to COD removal efficiency. Micro-sieving and UASB can achieve higher COD removal when treating industrial wastewater with higher particulate matter fraction and biodegradability. Effluent COD concentration can be calculated based on COD removal and influent COD concentration. Therefore, characteristics of wastewater can be used for predicting effluent COD concentration of industrial wastewater in systems with primary treatment and biological processes to determine if physical-chemical treatment is required.

5.4.3. Energy demand and production

5.4.3.1. Theoretical and actual energy production of UASB

In this study, empiric equations were used (7-11) for estimating theoretical COD removal and unit energy consumption of UASB, as shown in Table 5.5. Theoretical

COD removal, unit methane production, and unit energy production at 25 ℃ are 54%-88%, 0.15-0.30 m^3 CH_4/kg $COD_{removed}$, and 0.52-1.03 kWh/kg $COD_{removed}$.

Table 5.5 Theoretical COD removal, unit methane production, and unit energy production of UASB without primary treatment

BOD/COD ratio	0.57	0.46	0.29	0.53	0.34	0.30
COD concentration (mg/L)	2100.00	2350.00	4221.00	1697.00	3900.00	5422.25
COD removal (%)	88.08	85.45	53.85	82.66	54.56	79.01
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.27	0.22	0.17	0.26	0.21	0.14
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.30	0.24	0.19	0.28	0.23	0.15
Unit energy consumption (kWh/kg $COD_{removed}$l)	1.03	0.83	0.65	0.97	0.80	0.52

* The following reaction is used for converting COD digested in UASB to methane:

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$$

One kg COD digested can be converted to 0.25 kg CH_4 or 0.35m^3 CH_4 theoretically at 0℃ and 1 atm [28]

In Table 5.6, data about COD removal and methane production of UASB for removing untreated food processing wastewater was collected from published peer-reviewed papers [141-143]. Actual COD removal, unit methane production, and unit energy production at 25 ℃ are 43%-95%, 0.01-0.38 m^3 CH_4/kg $COD_{removed}$, and 0.03-1.30 kWh/kg $COD_{removed}$. As shown in Tables 5.5 and 5.6, and Figure 5.10, theoretical energy production is close to the actual values, which indicates the validity of empiric equations used in this study.

Table 5.6 Actual COD removal and unit methane production and unit energy production of UASB

HRT (h)	22	22	22	18	18	14
SRT (d)						
Temperature (℃)	35	35	35	35	35	35
COD (mg/L)	2800	3200	4200	3000	6500	3900

COD removal (%)	83.2	84.4	89.2	82.5	90.6	78.5
L biogas/g COD removed	0.555	0.47	0.402	0.46	0.192	0.3
L CH_4/kg SCOD removed (at 25℃ and 1 atm)						
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.34	0.29	0.25	0.29	0.12	0.19
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.38	0.32	0.27	0.31	0.13	0.2
Unit energy consumption (kWh/kg $COD_{removed}$)	1.30	1.10	0.94	1.08	0.45	0.70
Reference	[141]					
HRT (h)	14	7.1	6.8	6.7	4.1	2.3
SRT (d)		60.3	23.4	14	14.4	3.3
Temperature (℃)	35	33	33	33	33	33
COD (mg/L)	6330	8201	5719	5256	5495	5514
COD removal (%)	85.7	78	73	77	83	68
L biogas/g COD removed	0.157					
L CH_4/kg SCOD removed (at 25℃ and 1 atm)		213	254	283	201	199
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.1	0.07	0.15	0.23	0.15	0.11
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.11	0.08	0.17	0.25	0.16	0.12
Unit energy consumption (kWh/kg $COD_{removed}$)	0.37	0.27	0.58	0.85	0.56	0.41
Reference	[141]			[142]		
HRT (h)	24	24	24	24	24	24
SRT (d)						
Temperature (℃)	36	36	36	36	36	36
COD (mg/L)	3500	6000	10000	20000	28000	32000
COD removal (%)	>90	>90	>90	48	45	43
Specific methane production (LCH_4/g COD added at 36 ℃)	0.21	0.15	0.12	0.02	0.01	0.004
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.21	0.15	0.12	0.04	0.02	0.01
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.23	0.16	0.13	0.04	0.02	0.01
Unit energy consumption (kWh/kg $COD_{removed}$)	0.78	0.56	0.44	0.14	0.07	0.03
Reference	[143]					
HRT (h)	24	24	24	24	24	24
SRT (d)						
Temperature (℃)	36	36	36	36	36	36
COD (mg/L)	3500	6000	10000	20000	28000	32000
COD removal (%)	>90	>90	>90	95	72	68
Specific methane production (LCH_4/g COD added at 36 ℃)	0.28	0.19	0.18	0.21	0.08	0.04
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 0℃ and 1 atm) *	0.27	0.19	0.18	0.2	0.1	0.05
Unit CH_4 production (m^3 CH_4/kg $COD_{removed}$ at 25℃ and 1 atm)	0.3	0.2	0.19	0.21	0.11	0.06
Unit energy consumption (kWh/kg $COD_{removed}$)	1.04	0.70	0.67	0.74	0.37	0.20
Reference	[143]					

* Unit CH_4 production at the standard temperature and pressure (STP) is estimated based on the assuming methane/biogas ratio of 70% [144], 0℃, and 1 atm.

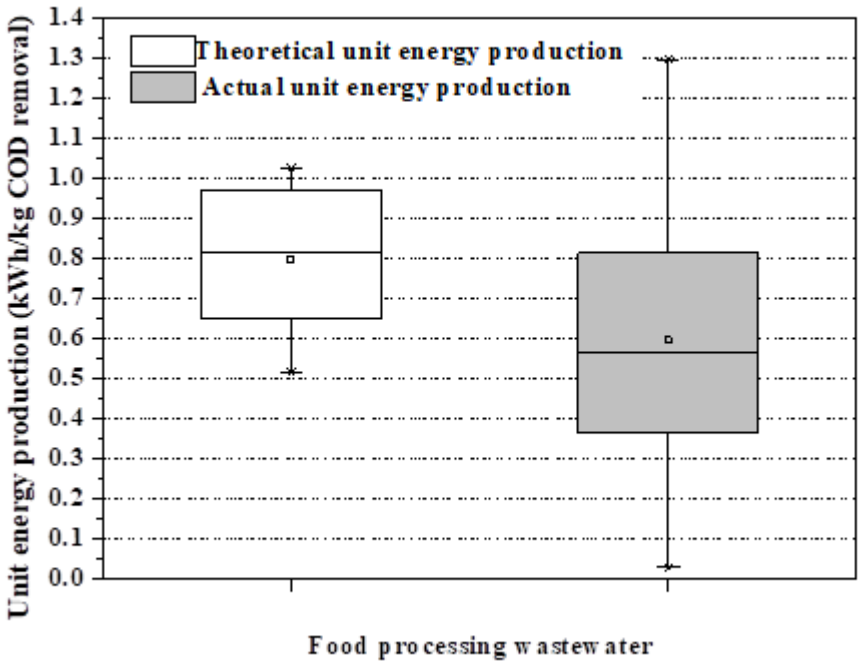


Figure 5.10 Actual and Theoretical unit energy production of UASB

5.4.3.2. Theoretical and actual energy consumption of PN/A

In Table 5.7, the actual unit energy consumption of PN/A is 0.8-1.92 kWh/kg $N_{removed}$. Based on equations (4.15-4.16), 1.32kWh/kg $N_{removed}$ was used as the theoretical unit energy consumption of PN/A, which is close to the actual value, as shown in Figure 5.11. This proves the validity of the theoretical unit energy consumption used in my study. According to the benchmarking method reported by Yang et al. [145], the mean actual unit energy consumption of 1.4 kWh/kg $N_{removed}$ can be selected as the benchmark data for comparing with actual unit energy consumption to check the energy efficiency of PN/A reactor.

Table 5.7 Energy consumption of PN/A reactors

Plant	Reactor type	Unit energy demand (kWh/kg $N_{removed}$)	Reference
Apeldoorn	SBR	1.1	[146]
Balingen	SBR	0.92	
Heidelberg	SBR	1.67	

Ingolstadt	SBR	1.92	
Nieuwegein	SBR	0.8	
Zurich	SBR	1.11	
Olburgen		1.86	
Malmö	MBBR	1.45-1.75	[88]

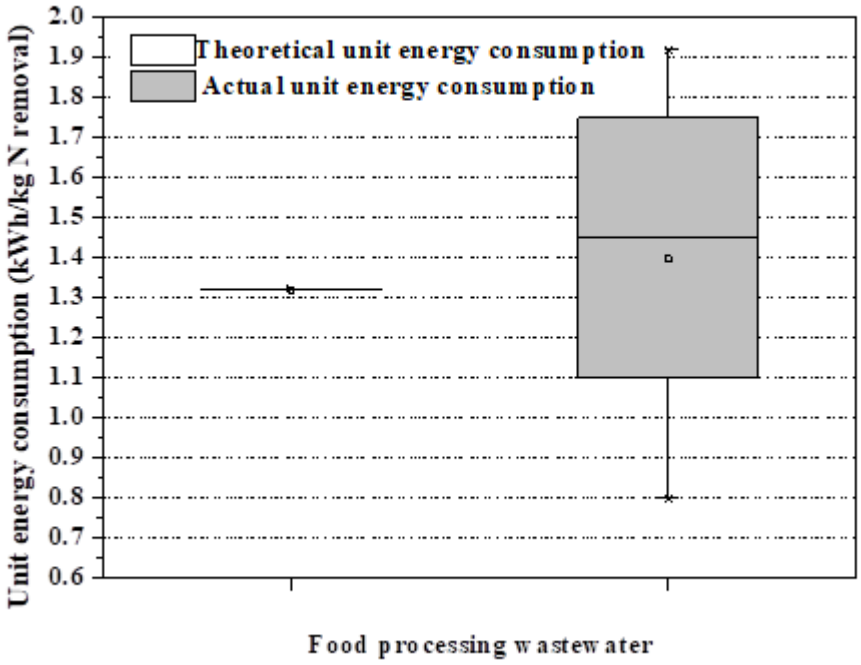


Figure 5.11 Actual and Theoretical unit energy consumption of PN/A

5.4.4. Energy demand

Electricity consumption of a WWTP is mainly from the wastewater pump station, aeration equipment, and sludge treatment. Secondary biological treatment consumes 50–70% of the overall energy consumption of a WWTP [41]. Therefore, energy required by aeration was the dominant energy consumption. Figure 5.12 shows that electricity consumption rises as the influent concentration of NH_4^+_N and soluble biodegradable organic nitrogen (sbON) increases. Therefore, energy required by aeration was the dominant energy consumption. Figure 5.12 shows that electricity consumption rises as the influent concentration of NH_4^+_N and soluble biodegradable

organic nitrogen increases. The unit energy consumption (kWh/kg $N_{removed}$) is used for comparing energy efficiency of different industrial wastewater systems, as shown in Figure 5.13. The oxygen requirement of the PN/A process and the nitrification process are 1.83 and 4.57 g O_2/g NH_4_N converted. The PN/A process in System B effectively reduces aeration consumption for removing NH_4_N. Therefore, System B shows lower unit energy consumption than System A. In Figure 5.13, mean unit energy consumption of System B for meat processing wastewater, tannery wastewater, and textile wastewater are 1.49, 1.37, and 1.39 kWh/kg $N_{removed}$, respectively. Since primary treatment and UASB removes COD without using oxygen, most of the O_2 is used for the PN/A process, especially in System B. Therefore, mean unit energy consumption of System B is close to than actual unit energy consumption of PN/A, as shown in Figure 5.11.

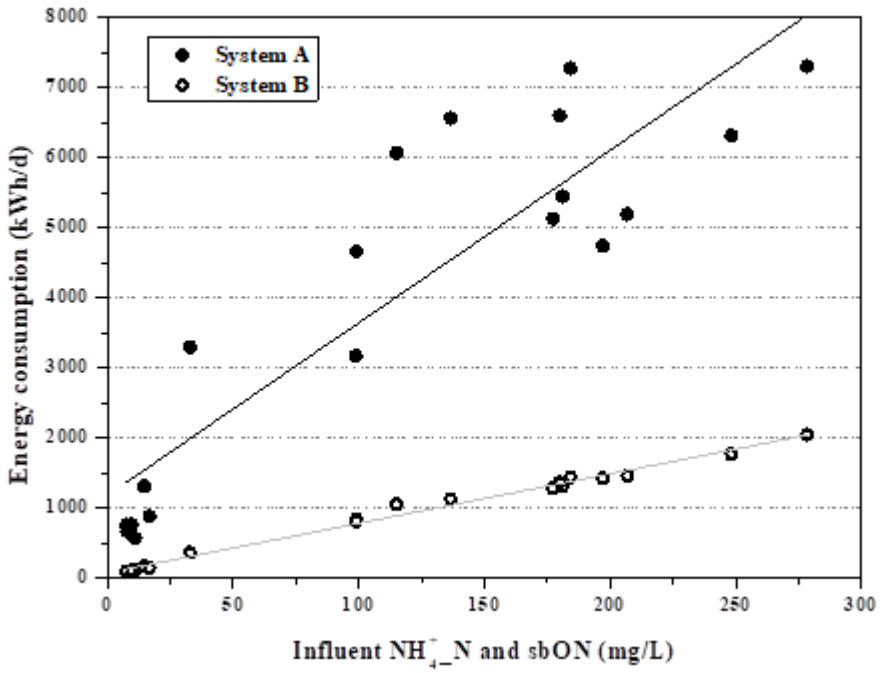


Figure 5.12 Electricity consumption and influent NH_4^+_N and sbON

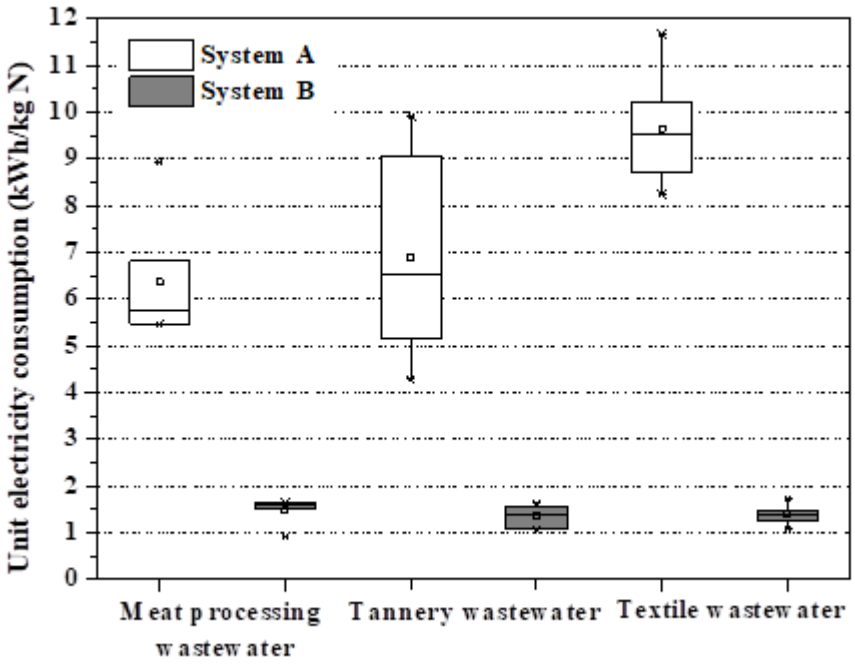


Figure 5.13 Unit electricity consumption

5.4.5. Energy production

Except for BOD that is converted to CO_2 in aerobic treatment, most of the soluble BOD in industrial wastewater and biodegradable particulate BOD in primary and secondary sludge are converted to CH_4 to produce electricity and heat by UASB and the side-stream anaerobic digester. Figure 5.14 illustrates energy production rises with increasing influent BOD concentration. The unit energy production (kWh/kg $BOD_{removed}$) is used as an indicator in assessing the influence of different industrial wastewaters on two treatment systems, as shown in Figure 5.15. The average unit energy production of System B for meat processing wastewater, tannery wastewater, and textile wastewater are 1.80, 1.77, and 1.73 kWh/kg $BOD_{removed}$, respectively. The unit energy production for food processing wastewater is higher than that of other industrial wastewaters. On the one hand, meat processing wastewater has higher BOD

concentration and biodegradability (BOD/COD ratio), as shown in Table 5.1 and Figure 5.3. This increases biogas production efficiency and unit energy production of UASB reactors, as shown in Figure 5.16. In addition, sludge production from the PN/A process and denitrification process increase the CH_4 production of the side-stream anaerobic digester. Therefore, the energy-positive system for treating meat processing wastewater can produce more energy with the same BOD removal than that for treating tannery wastewater and textile wastewater.

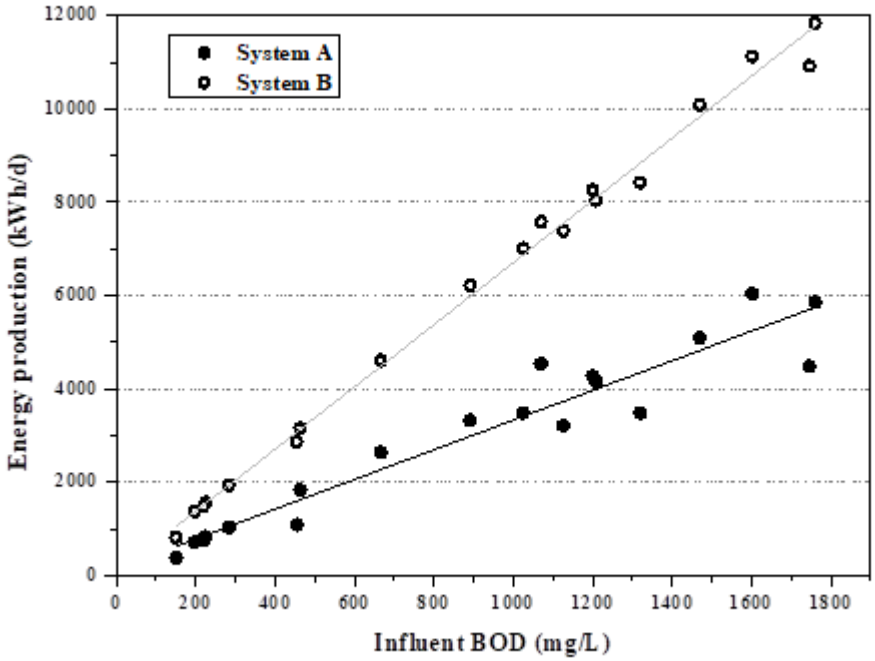


Figure 5.14 Electricity production and influent BOD

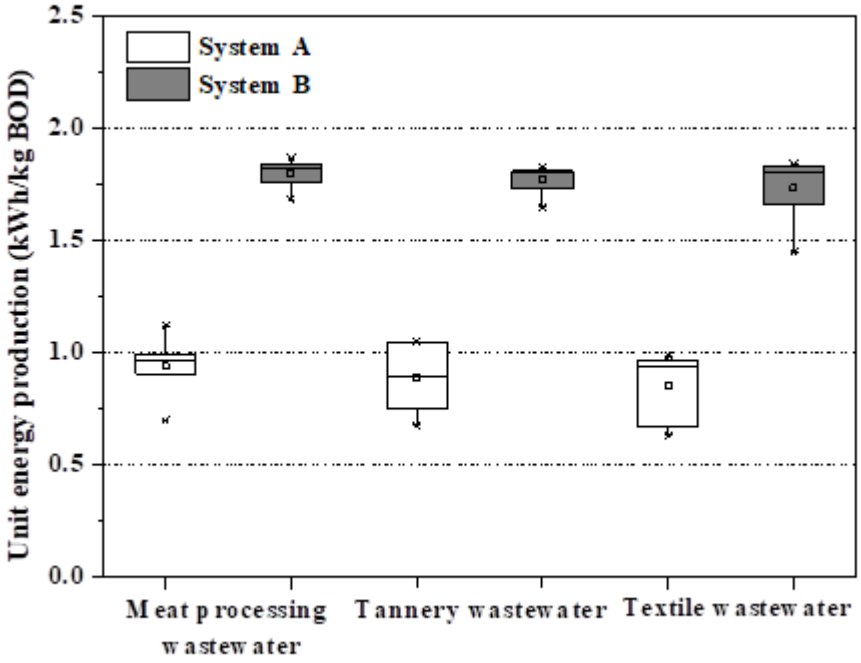


Figure 5.15 Unit electricity production

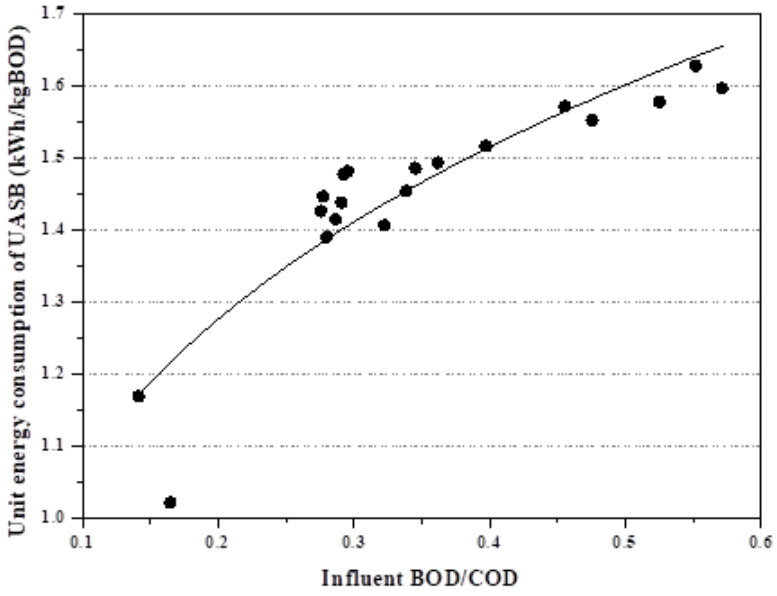


Figure 5.16 CH_4 production/influent COD to UASB and COD removal of UASB reactor

5.4.6. Cost analysis

Innovative technologies such as UASB can convert COD to CH_4 to increase electricity production. However, the unit capital cost ($/reactor volume m^3) of UASB is much higher than that of an aeration tank for small-sized WWTPs, as shown in Table 5.8. Innovative biological technologies could replace old processes when economic income due to energy saving and production is higher than the increased cost of innovative technologies. The unit cost analysis of treatment systems based on different industrial wastewater quality would provide operation cost of these systems. Since COD removal by primary treatment and secondary treatment are positively related to TSS/COD ratio and BOD/COD ratio, the unit cost metric ($/kg $COD_{removed}$) is used as the indicator in this study. Figure 5.17 compares the unit cost ($/kg $COD_{removed}$) of two systems for treating different industrial wastewaters. In System B, the average unit cost for meat processing wastewater, tannery wastewater, and textile wastewater are 0.54, 0.57, and 1.12 $/kg $COD_{removed}$, respectively. System B uses micro-sieving, UASB, and PN/A to remove COD and N with lower energy consumption. However, the unit cost of System B is higher than that of System A for textile wastewater due to low BOD/COD ratio and BOD concentration. UASB is inefficient in converting the BOD of textile wastewater to biogas. Meat processing wastewater has higher biodegradability and COD removal and a relativly lower particulate COD ratio than other wastewaters, which increases the CH_4 production of the mainstream anaerobic digester through increasing the COD loading rate to the UASB. Although the meat processing wastewater consumes more oxygen for removing N in PN/A and the aerobic process, this process increases the CH_4 production of AD through increasing biomass. These factors decrease the unit cost of the meat processing wastewater treatment

system. Therefore, it is easier to achieve energy-positive treatment with meat processing wastewater than with other wastewaters.

Table 5.8 Unit capital cost for different treatment capacity of WWTPs

Unit capital cost (US$ / reactor volume m^3)	UASB		Aeration tank	
Capacity (P.E.)	Min	Max	Min	Max
25,000	600	1000	220	300
50,000	500	700	180	250
100,000	350	500	150	200
200,000	250	400	120	170

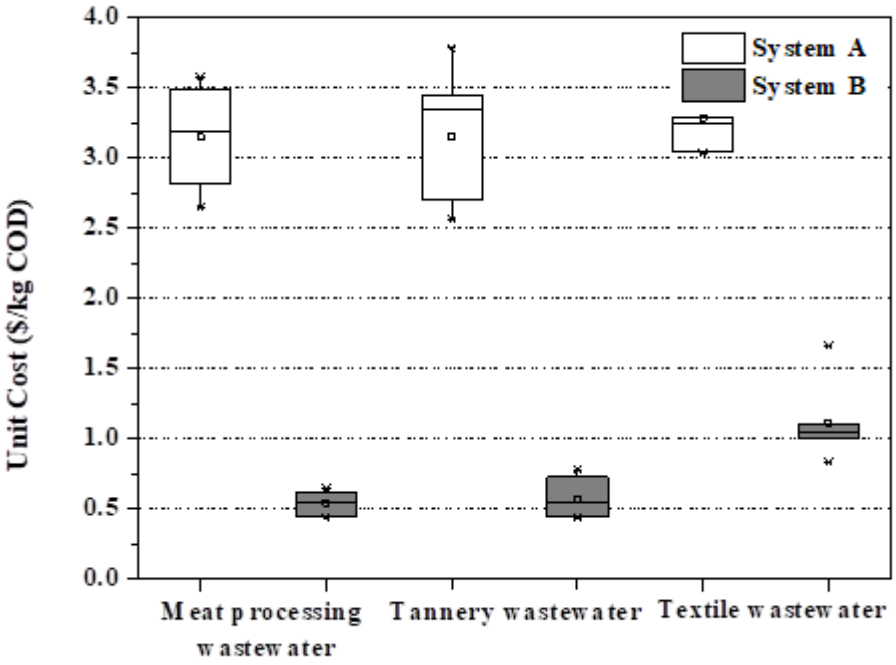


Figure 5.17 Unit cost of different industrial wastewater treatment

5.5. Conclusions

In the current study, an Excel-based model was developed to compare the unit energy and cost of energy-positive systems for treating different kinds of industrial wastewaters. Results showed that micro-sieving can remove 35.36%, 39.08%, and 32.50% of COD for tannery, meat processing, and textile wastewater, respectively. Due to the relatively high TSS/COD ratio, primary treatment can achieve higher COD removal efficiency of textile wastewater with lower TSS concentration as compared to tannery wastewater. A UASB reactor can remove 15.8%-53.5%, 14.0%-49.0%, and 22.9%-51.0% of COD for three kinds of wastewater, respectively. UASB converts more COD to CH_4 in meat processing wastewater with high BOD concentration because of the high BOD/COD ratio. Pearson correlation analysis shows that the TSS/COD ratio and the BOD/COD ratio are the key parameters in determining the COD removal and methane conversion rates of primary treatment and secondary treatment. These regression equations can be used to predict COD removal by the primary and biological processes as follows:

COD removal (%) =78.126(BOD/COD ratio) +67.022(TSS/COD ratio) R^2=0.975 +18.349

As shown in the above equation, the TSS/COD ratio and BOD/COD ratio can be effectively used to predict the total COD removal and effluent COD concentration of industrial wastewater in the treatment system with primary treatment and biological processes. For meat processing wastewater, tannery wastewater, and textile wastewater, the mean unit energy consumptions were 1.49, 1.37, and 1.39 kWh/kg $N_{removed}$, which are close to actual unit energy consumption of the PN/A process. Mean unit energy production for three kinds of wastewater in innovative systems are 1.80, 1.77, and 1.73 kWh/kg $BOD_{removed}$, respectively. The energy-positive system for treating meat

processing wastewater can produce more energy for removing the same amount of BOD than other industrial wastewater treatment systems. The average unit costs for meat processing wastewater, tannery wastewater, and textile wastewater are 0.54, 0.57, and 1.12 \$/kg $COD_{removed}$, respectively. Furthermore, meat processing wastewater has higher biodegradability, COD removal, and lower particulate COD ratio than other wastewaters, which increases electricity production through increasing COD loading rate to UASB and AD. A high CH_4 production rate improves the unit cost of meat processing wastewater treatment systems. Therefore, it is more economical and easier to treat meat processing wastewater with micro-sieving, UASB, and PN/A technologies than other industrial wastewaters.

6. ENERGY CONSUMPTION OF WASTEWATER TREATMENT PLANTS WITH ANAEROBIC-OXIC PLUS ANAEROBIC-ANOXIC-OXIC, OXIDATION DITCH AND SEQUENCING BATCH REACTOR IN CHINA

6.1. Abstract

Unit energy consumption in wastewater treatment plants in China is proposed as one of the critical sustainable metrics in assessing the sustainability of environmental engineering design according to Sustainable Environmental Engineering. The national data of 1,215 WWTPs in China were collected to evaluate the unit energy efficiency of different main biological technologies. 80.3% of the WWTPs in China used anaerobic-oxic plus anaerobic-anoxic-oxic, oxidation ditch, and sequencing batch reactor as main wastewater treatment technologies. The number of small and medium-sized plants accounted for 97% of the WWTPs. Pearson correlation analysis and comparison between theoretical and actual unit energy consumption show that the unit energy consumption (kWh/m^3) is not a suitable unit energy indicator. Therefore, the unit energy consumption ($kWh/kg\ COD_{removed}$) was used to rank the three main technologies. The energy efficiency of WWTPs increases with increasing key parameters such as design flow rate, operation loading rates, COD removal efficiency, and influent COD concentration. The average unit energy consumption of SBR decreased from 2.76 kWh/kg to 0.83 kWh/kg when the design flow rate increased from less than 10,000 m^3/d to 100,000-20,0000 m^3/d. The mean unit energy consumption of SBR decreases from 1.71 $kWh/kgCOD_{removed}$ to 1.32 $kWh/kgCOD_{removed}$ and 2.27 $kWh/kgCOD_{removed}$ to 1.30 $kWh/kgCOD_{removed}$ as the operation loading rates and COD removal increase from 40% to 100% and from 70% to over 90%, respectively. SBR technology shows lower energy consumption than other technologies under different conditions of key parameters. Therefore, SBR is the best treatment technology

for treating wastewater in small and medium-scale WWTPs in China. The equations between the benchmark data of unit energy consumption, the design flow rate, actual flow rate, and influent COD mass were developed in this study to provide the benchmark value for future sustainable treatment design.

6.2. Introduction

With population and economic growth, the number of municipal wastewater treatment plants in China and the corresponding capacity has reached 3,543 and 1.70×10^8 m^3/d in 2015 [215]. WWTPs mainly used energy-intensive aerobic technologies for treating wastewater [1, 2]. The electricity used for water supply and wastewater treatment accounts for more than 2% of the world's energy consumption [216, 217]. In the United States, wastewater treatment consumes approximately 2%-4% of electric energy [133, 218-220], whereas WWTPs in other countries use up to 5% of electricity consumption [11, 221]. According to statistics, WWTPs in China consumed 10^{10} kWh in 2011, which made up for 0.2 % of the total industrial electricity consumption [3]. From 2011-2017, the water supply and wastewater treatment account for approximately 0.67-0.75% of the total industrial energy consumption in China [4-10]. The water production and supply industry are not the largest electricity consumers in China. However, their energy consumption has increased very rapidly at a rate of 5.7-10.7% per year. Electricity costs account for 25 to 60% of the operating costs in conventional WWTPs, while the aeration process typically makes up approximately 50–75% of energy demand [222-228]. With the increase in the number and capacity of WWTPs to meet fast urbanization, energy consumption, which results in increased greenhouse gas production and resource consumption, has become a critical issue in the operation of WWTPs [11-13]. Currently, there is no sustainbale metrics in assessing

plant energy efficiency. Therefore, this study attempts to introduce new sustainable environmental engineering metrics to assess WWTPs.

According to Sustainable Environmental Engineering [14], the unit energy consumption indicator is critical when evaluating the energy performance of various WWTPs with different technologies. The energy gaps and energy-saving potential of WWTPs could be identified by comparing with benchmark data of unit energy consumption. Since the flow rate of wastewater is easy to measure, energy consumption per unit volume of wastewater was used as sustainable metrics for evaluating the operation energy efficiency in most plants in the past [229-233]. However, unit energy consumption (kWh/m^3) is significantly affected by the dilution of pollutant loads [38]. For example, WWTPs with combined sewer systems show higher energy efficiency than plants with separate sewer systems because the larger amount of water reduces the influent concentration of pollutants such as BOD or COD [231]. Some papers showed that treatment technology has a strong impact on the unit energy consumption (kWh/m^3), while others showed an opposite trend [234-236]. Therefore, energy consumption expressed as kWh/m^3 might not be a useful indicator in comparing the energy efficiency of WWTPs [225, 237].

The electricity power of WWTPs is mainly consumed by aeration equipment, pumping, and sludge treatment [238-242]. Typically, aeration is the largest electricity consumer of the plant. The energy consumption of pumping and sludge treatment is mainly determined by the actual flow rate and dewatered sludge mass. The oxygen demand of aeration equipment is determined not only by flow rate but also by the concentration of oxygen-consuming pollutants. The energy consumption of the aeration process is often evaluated according to either BOD or COD reduction (kWh/kg· BOD or kWh/kg· COD) [225, 242]. Unit energy metric such as kWh/kg $COD_{removed}$, reflects

the differences in the influent concentration (or mass loading) of pollutants in the wastewater, could be used to define benchmark values for evaluating the energy-saving potential of WWTPs. Therefore, unit energy consumption (kWh/kg $COD_{removed}$) is selected for assessing the energy efficiency of WWTPs in this study.

Several studies benchmarked energy consumption to provide useful information for optimizing the operation of the plants to achieve energy saving [34, 145, 243]. For example, Hanna et al. [244] and Vaccari et al. [38] benchmarked unit energy consumption of 266 small US and 241 Italian wastewater treatment facilities. They also explored the relationship between the unit energy consumption of all WWTPs and key parameters such as design flow rate, COD concentration, and operation loading rate. However, there were a few studies composing the energy efficiency of different treatment technologies under different conditions of design flow rate, operation loading rate, COD removal, and influent COD concentration. Therefore, this study tries to quantify the unit energy consumption of different treatment technologies in terms of unit energy metrics per kg $COD_{removed}$. The objectives of the current study are to (1) assess the suitability of unit energy metricsuch as kWh/m^3 and kWh/kg $COD_{removed}$ for the energy audits of main wastewater treatment technologies at different sizes of WWTPs in China; (2) explore the difference between the actual and theoretical unit energy consumption of main treatment technologies in WWTPs; (3) compare the unit energy consumption of different technologies under different conditions of design flow rate, operation loading rate, COD removal and influent concentration of contaminants. The unit energy consumption of main biological technologies is to offer valuable references for future sustainable treatment design.

6.3. Materials and methods

6.3.1. Data and processing

In the current study, unit energy consumption was calculated based on national data obtained from the China Urban Drainage Statistical Yearbook 2016 [215]. The national database of 1,399 WWTPs contains necessary information of wastewater treatment plants (WWTPs) such as design flow rate, treatment technology, operation loading rate, annual energy consumption, influent and effluent concentration of chemical oxygen demand (COD). However, the energy consumption of some WWTPs was not recorded in the Yearbook. Therefore, 1,215 WWTPs in China were selected after data screening. The main wastewater treatment technologies in China could be found based on the number of WWTPs with specific secondary processes. Pearson correlation analysis identified the feasibility and sustainability of unit energy indicators by using IBM SPSS Statistics 21.0 software. Secondly, Excel-based models were developed for composing the theoretical and actual unit energy consumption of WWTPs with main treatment technologies based on literature [26-28]. Thirdly, the actual unit energy consumption (kWh/kg $COD_{removed}$) was used as an indicator for ranking the energy efficiency of treatment technology in different design flow rates, operation loading rates, COD removal, and influent concentration of contaminants. Finally, regression models of unit energy consumption of main technologies were developed based on the corresponding parameters.

6.3.2. Unit energy consumption

6.3.2.1. Actual unit energy consumption

Based on the actual flow rate, the actual unit energy consumption (kWh/m^3) was calculated by the following equation:

Actual unit energy consumption (kWh/m^3)=(actual energy consumption (kWh/d)/Q

(6.1)

where

Q=Actual flow rate(m^3/d)=(Design flow rate (m^3/d)(Operation loading rate(%))

The daily COD removal is calculated by COD concentration in wastewater and actual flow rate:

$$\text{Daily COD removal (kgCOD/d)} = (S_0 - S)(Q)/(10^3 \text{ g/kg}) \quad (6.2)$$

where

S_0= Influent biodegradable COD (g/m^3)

S=Effluent biodegradable COD (g/m^3)

Based on COD concentration in influent and effluent, the COD removal efficiency (%) was estimated by the following equation:

$$\text{COD removal rate (\%)} = (S_0 - S)/S_0 \quad (6.3)$$

Actual unit energy consumption (kWh/ kg $COD_{removed}$) is calculated as the ratio between daily energy consumption and daily COD removal:

$$\text{Actual unit energy consumption (kWh/(kg } COD_{removed} \text{))} = (\text{actual energy consumption (kWh/d)})/(\text{COD removal (kgCOD/d)}) \quad (6.4)$$

6.3.2.2. Theoretical unit energy consumption

Theoretical aeration demand was estimated based on municipal wastewater quality from the China Urban Drainage Statistical Yearbook 2016 [215]. The Yearbook recorded the annual mean value of wastewater quality, data of some WWTPs is not suitable for estimating aeration consumption in the theoretical model. Therefore, the

wastewater quality of 881 WWTPs was selected after data screening. Theoretical oxygen consumption (g/d) is calculated by the following equation [27]:

$$R=Q(S_0-S)-1.42P+4.57Q(NO_x)-2.86Q(NO_x- NO_3\_N\ e) \quad (6.5)$$

where

R=oxygen demand (g/d)

P=sludge production (g/d)

NOx=Amount of NO_3_N production during nitrification (g/m^3) = Influent NH_4_N concentration -Effluent NH_4_N concentration -0.12P/Q

NO_3_N e= Effluent NO_3_N concentration (g/m^3)

$$P = \frac{QY_H\,(S_0 - S)}{1+b_H(SRT)} + \frac{f_d QY_H\,(S_0 - S)SRT}{1+b_H(SRT)} + \frac{QY_n\,(NO_x)}{1+b_n(SRT)} \quad (6.6)$$

where

b_H=0.088 g VSS/g VSS·d

b_n=0.135 g VSS/g VSS·d for AO+AAO and OD

b_n=0.082 g VSS/g VSS·d for SBR

Y_H=0.45 g VSS /g biodegradable COD

f_d=0.15

Y_n=0.15 g VSS/g NH_4_N for AO+AAO and SBR

Y_n=0.2 g VSS/g NH_4_N for OD

SRT= Solids retention time (d)

Based on design parameters from table 6.1, SRT of AAO, OD, and SBR are assumed to be 7.5, 27.5, and 22.5 d.

Table 6.1 Design parameters of various technologies [28]

Technology	SRT (d)	HRT (h)		MLSS (mg/L)
CAS	5-15	4-8		1000-3000
AO or AAO	5-10	Anaerobic zone	1-2	2000-4000
		Anoxic zone	1-2	
		Aerobic zone	4-6	
MBR	10-25	Anoxic zone.	1-2	5000-15000
		Aerobic zone.	2-4	
OD	15-40	15-30		2000-5000
SBR	15-30	Total cycle time	4-9	2000-5000
Biolak	30-70	15-40		1500-4000

According to the Handbook of biological wastewater treatment [26], 1.2 kg O^2/kWh is assumed to be the actual oxygen transfer rate (AOTR).

$$\text{Theoretical aeration demand (kWh/d)} = \frac{R}{AOTR} \tag{6.7}$$

where

AOTR=1.2 kg O^2/kWh

Based on the actual flow rate, the theoretical unit aeration demand (kWh/m^3) was calculated by the following equation:

$$\text{Theoretical unit aeration demand (kWh/m}^3\text{)=(theoretical aeration demand (kWh/d))/Q} \tag{6.8}$$

Theoretical unit aeration demand (kWh/ kg $COD_{removed}$) is calculated as the ratio between the theoretical aeration demand and the daily COD removal:

$$\text{Theoretical unit aeration demand (kWh/(kg COD}_{removed}\text{))} = \text{(theoretical aeration demand (kWh/d))/(COD removal (kgCOD/d))} \tag{6.9}$$

Since secondary biological treatment consumes 50–70% of the overall energy consumption of a WWTP [41], 0.6 is assumed to be the ratio between energy consumption and aeration demand.

$$\text{Theoretical unit energy consumption } (\text{kWh/m}^3) = \frac{\text{Theoretical unit aeration demand } (\text{kWh/m}^3)}{0.6}$$

$$\text{Theoretical unit energy consumption } (\text{kWh/kg COD}) = \frac{\text{Theoretical unit aeration demand } (\text{kWh/kg COD})}{0.6} \quad (6.10)$$

6.4. Results and discussion

6.4.1. Main technologies of WWTPs in China

In the Yearbook of 2016, WWTPs in China used different kinds of wastewater treatment technologies, which include anaerobic- anoxic–oxic, anaerobic-oxic, oxidation ditch, sequencing batch reactor, membrane bioreactor, BIOLAK, biofilm, wetland, activated sludge, and a combination of these technologies. In this study, the selected WWTPs were classified as AO+AAO, OD, SBR, Biofilm, Biolak Technology, constructed wetland, activated sludge process, MBR, and others, as shown in Figure 6.1.

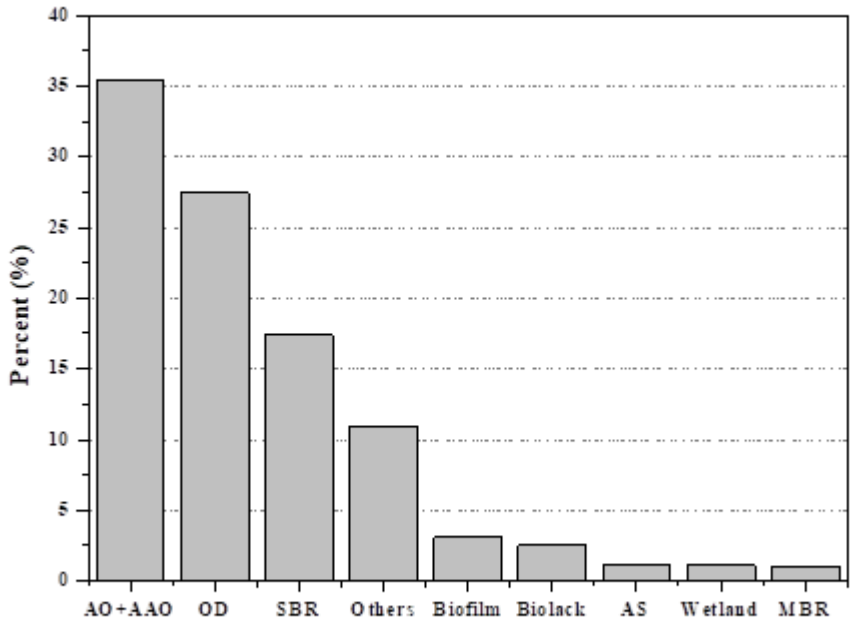


Figure 6.1 Distribution of treatment technologies in wastewater treatment plants (WWTPs)

Since the main pollutant of wastewater is organic matter, WWTPs used biological methods as the main wastewater treatment technologies in China. Biological methods can be classed into activated sludge process and biofilm. However, biofilm only accounted for 3.07 % of WWTPs in China, as shown in Figure 6.1, due to its low treatment efficiency. The activated sludge process mainly included the conventional activated sludge process, AO + AAO, OD, and SBR. Figure 6.1 shows that AO + AAO, OD, and SBR were the most widely used technologies in China, accounting for 36.5%, 27.4%, and 17.4 % of WWTPs, respectively. Jin et al. [13] reported similar findings on mainstream treatment technologies of WWTPs.

6.4.2. Size distribution

According to the design flow rate, WWTPs could be classified into small (<50,000m^3/d), medium (50,000-200,000 m^3/d) and large (>200,00m^3/d) sized plants.

Figure 6.2 demonstrates that the number of small and middle-sized WWTPs accounted for 94% of all WWTPs. Small WWTPs are generally built-in small coastal cities and towns. Because of economic and population growth, the number of small-sized plants has exceeded that of medium and large-sized plants in 2015, as shown in Figure 6.2, which accounted for 66% of WWTPs. Figure 6.3 shows that the actual flow rate and energy demand of the small and medium-sized WWTPs accounted for 61 % and 62% of all plants, which led to considerable total energy consumption. Since more dispersed wastewater sources need longer sewers for collecting wastewater, large wastewater treatment is not appropriate for the development of cities and improving the energy efficiency of small-medium-sized WWTPs.

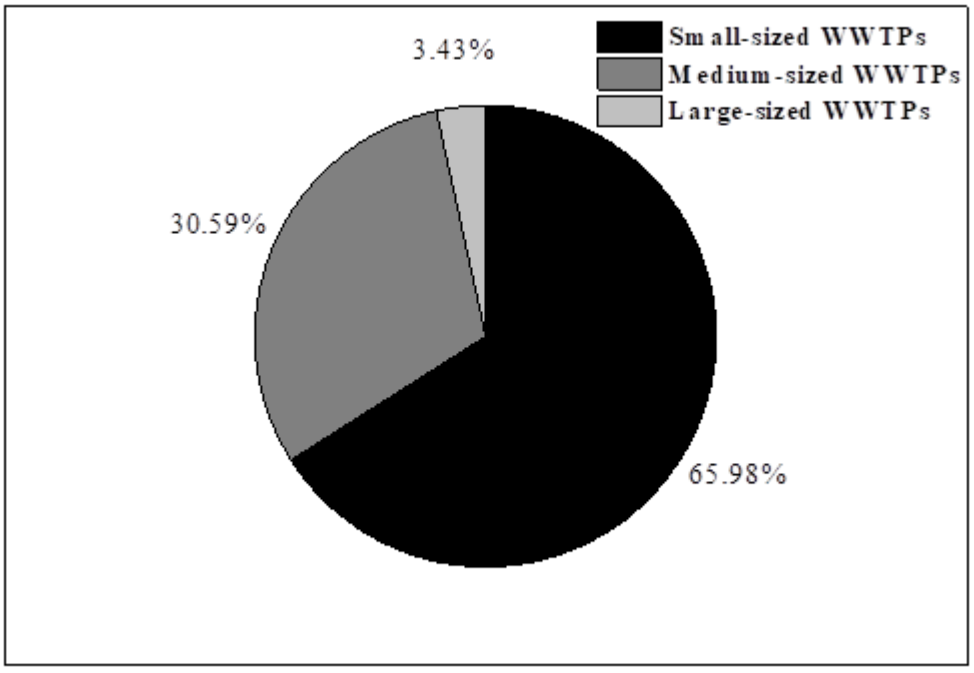


Figure 6.2 Size distribution of WWTPs in China

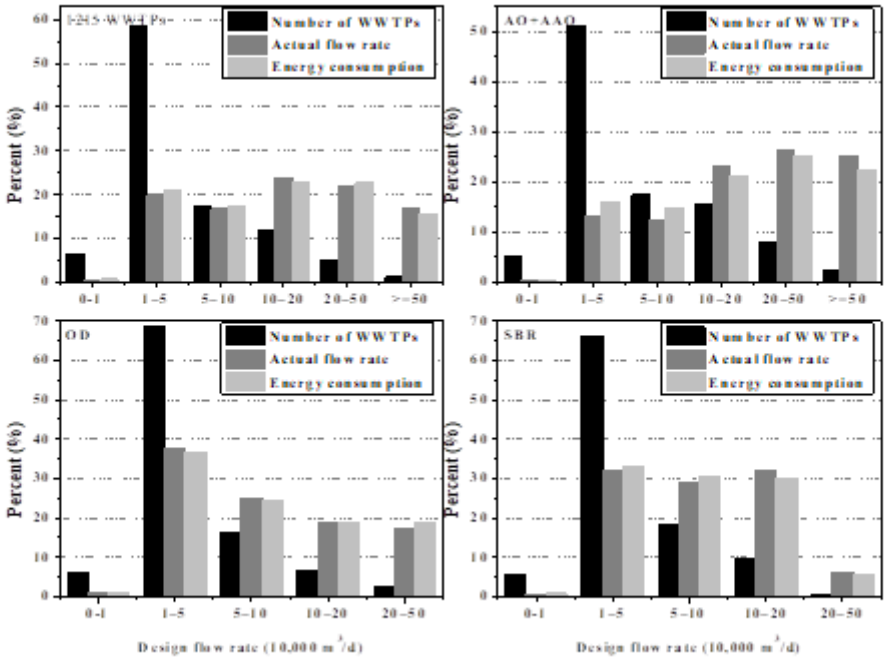


Figure 6.3 Distribution of number, actual flow rate, and energy consumption of WWTPs in different design capacities

6.4.3. Unit energy consumption of the treatment process

Different unit energy indicators lead to varying results of energy audits. Table 6.2 shows variations of wastewater characteristics and the operation loading rate of Gaobeidian WWTP. In 2015, the operation loading rate, influent COD, and BOD concentration of the WWTP increased to 93.26%, 409 mg/L, and 202 mg/L, which result in a 7% and 15% reduction in unit energy consumption (kWh/m^3 and kWh/kg $COD_{removed}$). Two different energy audit methods could produce diametrically opposite results. In table 6.2, Mulan Town WWTP and Hutou Town WWTP have the same flow rate and operation loading rate in 2015. Mulan Town WWTP shows a lower unit energy consumption (0.070 kWh/m^3) than Hutou Town WWTP (0.075 kWh/m^3). However, unit energy consumption per kg COD removal of Mulan Town WWTP is more than twice that of Hutou Town WWTP because Hutou Town WWTP received wastewater

with higher COD concentration. Unit energy consumption per m^3 flow rate ignores the change in the concentration of pollutants in the wastewater and lead to incorrect results of energy audits. Unit energy metrics expressed as kWh/kg $COD_{removed}$ could better reflect the energy efficiency of WWTPs.

Table 6.2 Characteristics and unit energy consumption of Gaobeidian WWTP, Mulan Town WWTP, and Hutou Town WWTP in China

Parameter	Gaobeidian WWTP [245, 246]	Gaobeidian WWTP [215]	Mulan Town WWTP [215]	Hutou Town WWTP [215]
Design flow rate (10,000 m^3/d)	100	100	2	2
Operation loading rate (%)	66.64	93.26	39.9	39.9
Influent COD concentration (mg/L)	173	202	110	159
Influent BOD concentration (mg/L)	346	409	51	21.8
Unit energy consumption (kWh/m^3)	0.258	0.240	0.070	0.075
Unit energy consumption (kWh/kg $COD_{removed}$)	0.746	0.631	1.189	0.545

For assessing feasibility and sustainability of unit energy indicators, the correlations between energy consumption, COD reduction, and actual flow rate of WWTPs with the main biological process in China were carried out, and results are presented in Table 6.3. The Pearson correlation coefficients are close to 1, which shows significant positive correlations between different parameters. Table 6.3shows that the COD reduction has stronger correlations with the energy consumption of 1,215 plants as compared with the actual flow rate. For mainstream treatment technologies, all the correlations were significant at a level of significance of $p < 0.01$ with Pearson coefficients of 0.906-0.933. As a result, the unit energy consumption (kWh/kg $COD_{removed}$) is a suitable indicator to evaluate energy efficiency in WWTPs.

Table 6.3 Correlations between energy consumption, daily COD removal, and actual flow rate of WWTPs using different technologies.

Pearson Correlation	Actual flow rate (10,000 m^3/d)	Daily COD removal (kg/d)
Energy consumption of all plants (kWh/d)	0.869[a]	0.900 [a]
Energy consumption of AO+AAO (kWh/d)	0.901 [a]	0.915 [a]
Energy consumption of OD (kWh/d)	0.916 [a]	0.906 [a]
Energy consumption of SBR (kWh/d)	0.905 [a]	0.933 [a]

[a] Correlation is significant at the 0.01 level (2-tailed).

Figure 6.4 presents the average unit energy consumption values of 1.811 kWh/kg $COD_{removed}$ and 0.318 kWh/m^3 for 1215 plants in China. Due to the differences in wastewater quality, the standard deviation of unit energy consumption (kWh/kg $COD_{removed}$) is higher than that of unit energy consumption (kWh/m^3). The mean unit energy consumptions of AO+AOO, OD, and SBR are 1.903, 1.877, and 1.705 kWh/kg $COD_{removed}$. SBR technology shows higher energy efficiency than other mean treatment technologies in China.

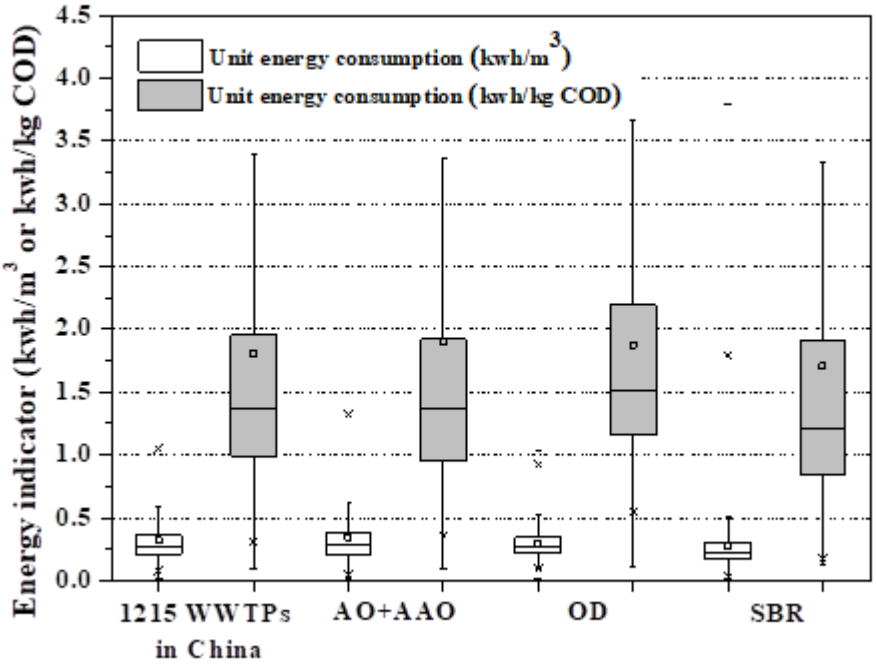


Figure 6.4 Unit energy consumption of WWTPs with main technologies

6.4.4. Actual and theoretical unit energy consumption

Figures 6.5, 6.6, and 6.7 show that average theoretical unit energy consumptions of AO+AAO, OD, and SBR are 0.71-0.86, 0.79-0.94, and 0.89-0.94 kWh/kg $COD_{removed}$, which is close to the energy indicators (kWh/kg $COD_{removed}$) reported in the literature [2, 247, 248]. AO+AAO, OD, and SBR are aerobic treatment methods that consume oxygen to convert COD and NH_4_N to CO_2 and N_2. Theoretical oxygen demand is related to wastewater quality and sludge production, as shown in equation (6.5). As shown in figures 6.5, 6.6, and 6.7, the difference between actual and theoretical unit energy consumption decreased with the increased design flow rate, while the theoretical unit energy consumption is not affected by design capacities. In practice, the smaller wastewater treatment systems have higher energy demand due to the lower operation loading rates. Larger WWTPs used more efficient pumps and compressors for treating wastewater [228]. Many cases of research and books do not consider the effect of design flow rate, operation loading rates, and equipment efficiency on theoretical unit energy consumption [26, 27, 40, 232], which leads to similar theoretical unit energy consumption in different design capacities. Figure 6.8 presents the ratio between theoretical and actual unit energy consumption. Average values of the ratio in table 6.4 could be used for making up the difference between actual and theoretical unit energy consumption.

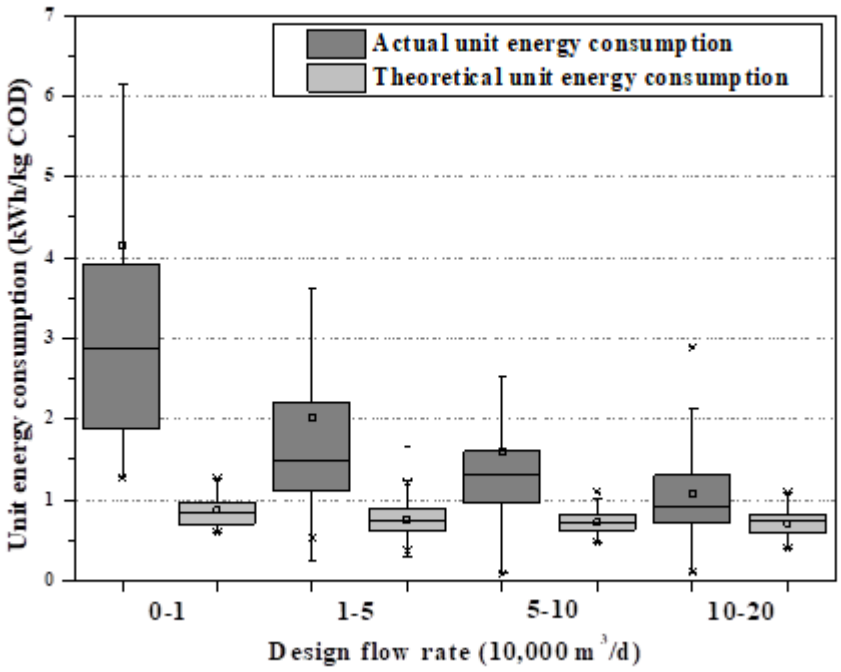


Figure 6.5 Actual and theoretical unit energy consumption per kg COD removal of WWTPs with AO+AAO technology

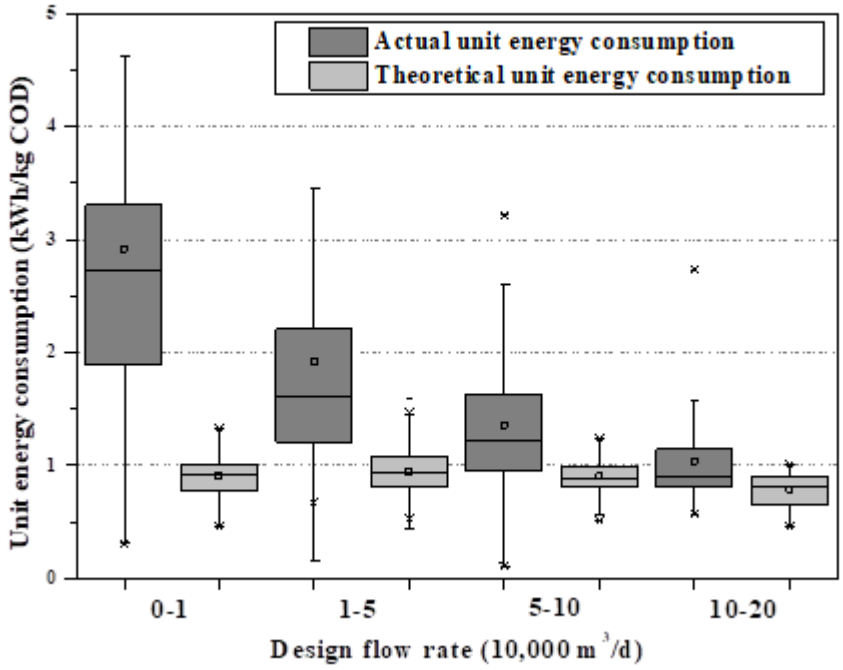


Figure 6.6 Actual and theoretical unit energy consumption per kg COD removal of WWTPs with OD technology

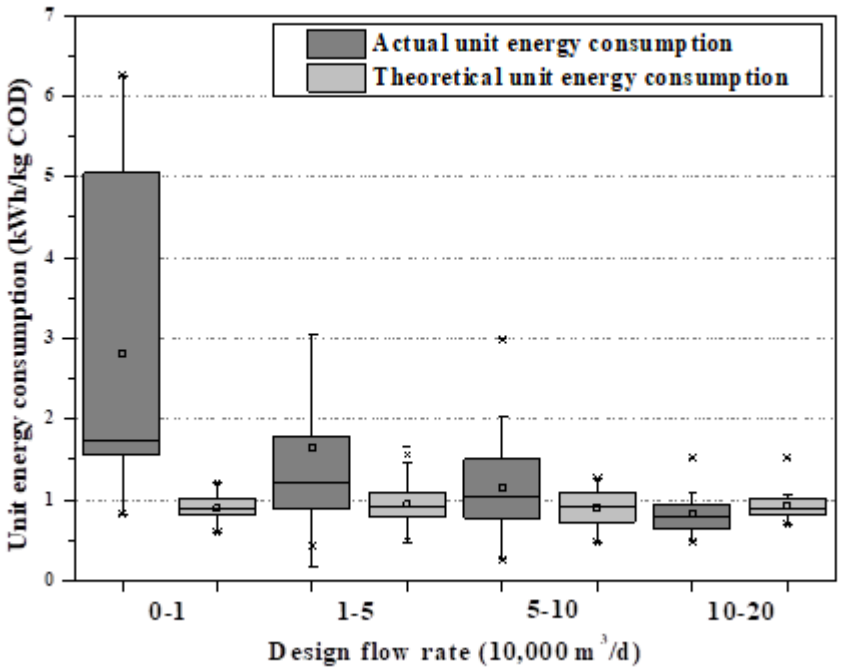


Figure 6.7 Actual and theoretical unit energy consumption per kg COD removal of WWTPs with SBR technology

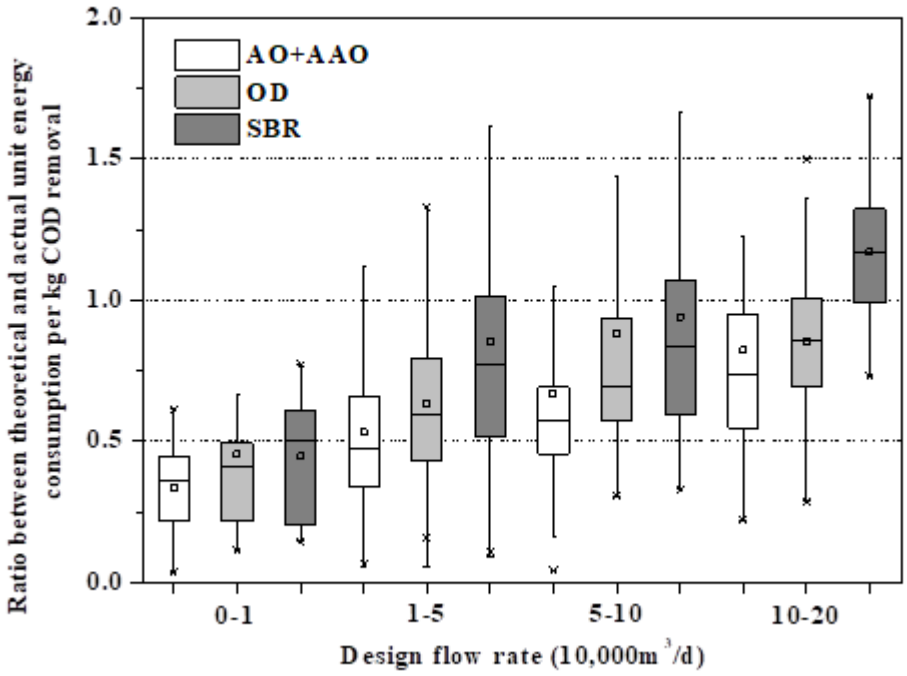


Figure 6.8 Ratio between theoretical and actual unit energy consumption per kg $COD_{removed}$

Table 6.4 Average ratio between theoretical and actual unit energy consumption

Design flow rate (10,000 m^3/d)	AO+AAO	OD	SBR
0-1	0.333	0.453	0.446
1-5	0.532	0.635	0.854
5-10	0.669	0.881	0.938
10-20	0.821	0.852	1.169

Figure A.10, A.11, and A.12 compared actual and theoretical unit energy consumption (kWh/m^3) in different design capacities. Actual unit energy consumption per unit volume of wastewater decreased as the design flow rate increased. However, theoretical unit energy consumption shows an opposite trend. In Figure 6.9, COD influent concentration is positively related to the design flow rate, which results in the abnormal change of theoretical unit energy consumption per m^3 flow rate. Figure 6.9 illustrates that larger WWTPs received wastewater with a higher concentration of COD.

Due to the change of the COD concentration in different design capacities, the effect of the COD concentration should not be ignored when evaluating the energy efficiency of WWTP. The unit energy consumption (kWh/m^3) is not a suitable unit energy indicator.

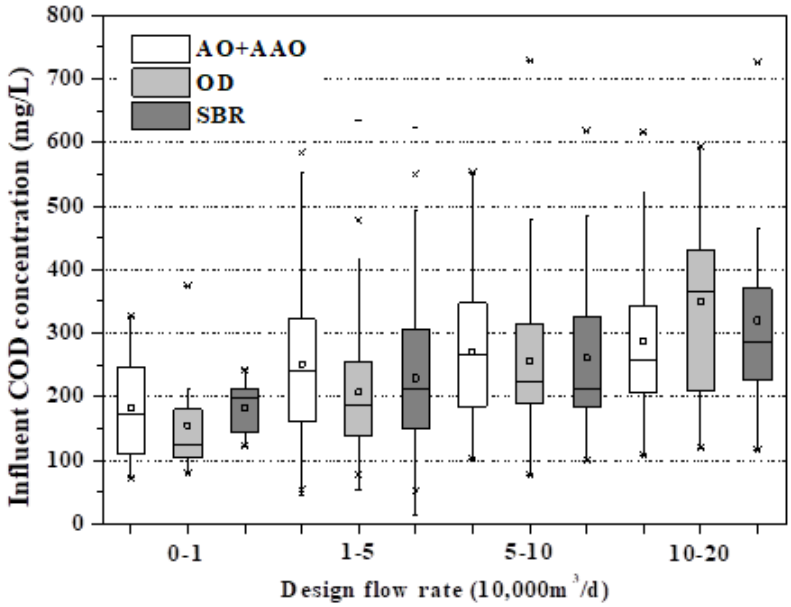


Figure 6.9 Variation of COD influent concentration with the design flow rate

6.4.5. Effect of flow rate, operation loading rate, COD removal efficiency, and COD concentration on unit energy consumption

6.4.5.1. Flow rate

Figure 6.10 indicates that Biolak has the lowest unit mean energy consumption with values of 0.896 and 1.751 kWh/kg $COD_{removed}$ among treatment technologies in small and medium-size plants due to its highly efficient submerged aeration methods and long sludge retention times (SRT). This indicates that it is high energy performance treatment system. Biolak, which is an activated sludge process in an earthen basin with high-density polyethylene (HDPE) basin sealing, was developed in Germany and introduced to China in 1999. Biolak systems use the floating aeration chain system to

supply fine air bubbles for microbial growth with long SRT ranging from 30 to 70 d, which leads to a relatively stable process with very efficient oxygen transfer, lesser energy consumption, and lesser sludge production than an oxidation ditch. Crites et al. [249] have reported that Biolak systems have achieved discharge standards in Colorado. The 2-year mean effluent TSS, BOD, and total nitrogen concentrations of Nevada, Ohio, Biolak system (100,000 gpd) are less than 10 mg/L, 10mg/L, and 8 mg/L, respectively [249]. The Biolak system is simpler to operate and requires less knowledgeable staff than the SBR system. The advantages of Biolak system include low capital cost, simple operation, and energy efficiency. Therefore, the Biolak process has great potential to be the most energy-efficient and economically activated sludge process. However, Biolak could cause problems in the settleability of the effluent particle because of its long sludge age, over oxidation, and the formation of pin-point floc, which results in a lower quality effluent. Mean and monthly performance data of 13 Biolak systems were summarized by the US Environmental Protection Agency report [250]. In Arkansas, the average effluent concentration of NH_3_N in Blytheville South WWTP and BOD and TSS in Piggot WWTP, are 30.9, 20.8, and 34.8 mg/L, which do not satisfy the effluent discharge requirements. Mirbagheri et al. [251] assessed BOD and COD removal efficiency of MBR, SBR, Biolak, and CAS at Ekbatan wastewater treatment plant, in the west of Tehran. The results show that effluent wastewater quality is not higher than MBR and SBR. The other disadvantage is its carbon footprint is larger than SBR and OD due to its considerably long retention time. The second energy-saving technology is the activated sludge process with unit energy consumption of 1.028 and 1.849 kWh/kg $COD_{removed}$. The low energy consumption of the activated sludge process is due to its primary clarifier, and poor treatment performance of N and P.

The old wastewater treatment plants in China were built based on conventional activated sludge technologies. In the past few decades, many existing CAS systems have been upgraded to efficient systems to remove more N, and P. Biofilm is the third lowest energy consumption technology with a unit energy demand of 1.115 kWh/COD kg in small plants due to its oxygen supply methods. The Biofilm reactor uses natural draft as the primary mean of providing airflow, which results in low oxygen supply and reduced removal rate of contaminants in the wastewater. Biofilm submerged reactor, which has been applied for treating industrial wastewater since the 1970s and is applied to treat municipal wastewater in China. However, only 9 WWTPs used biofilm submerged reactors in China in 2002 [252]. The reason is mainly related to the biofilm carriers and packing media. The ideal carriers and packing media could provide a specific large surface area, low cost, excellent mechanical strength, low density, stability, high bio-affinity, resistance to biofilm, which results in good system performance and low-maintenance requirements [253-256]. Biofilm system generally used inorganic materials-based carriers, this can result in several problems in mass transfer caused by low porosity and easy blockage [256]. Organic materials have large surface areas [257, 258], but have low bio-affinity due to the smooth surface [254]. MBR overcomes the disadvantages of the CAS, including large space requirements for secondary clarifiers, excess sludge generation, and a high removal rate of contaminants [259]. Upgrading the existing treatment plant to an MBR system can increase the treatment capacity by up to threefold without adding space [26]. In reports, MBR could reach a high COD removal of 97.1% [251]. However, MBR technology has disadvantages such as high energy costs, membrane fouling problems, and high costs of periodic membrane replacement, which results in high maintenance and operating costs [27]. The capital costs of MBRs are approximately 50% higher than those of CAS

for treating low strength wastewater [26]. The unit energy consumption is the highest, as shown in Figure 6.10. Due to low treatment performance, large footprint, and high energy consumption, biolak, biofilm, activated sludge process, and MBR accounted for 2.50%, 3.07%, 1.14%, and 1.00% of WWTPs in China, as shown in Figure 6.1. Because of high pollutant removal performance and small place requirement, AO + AAO, OD, and SBR were the main treatment technologies in China, accounted for 80.3 % of WWTPs.

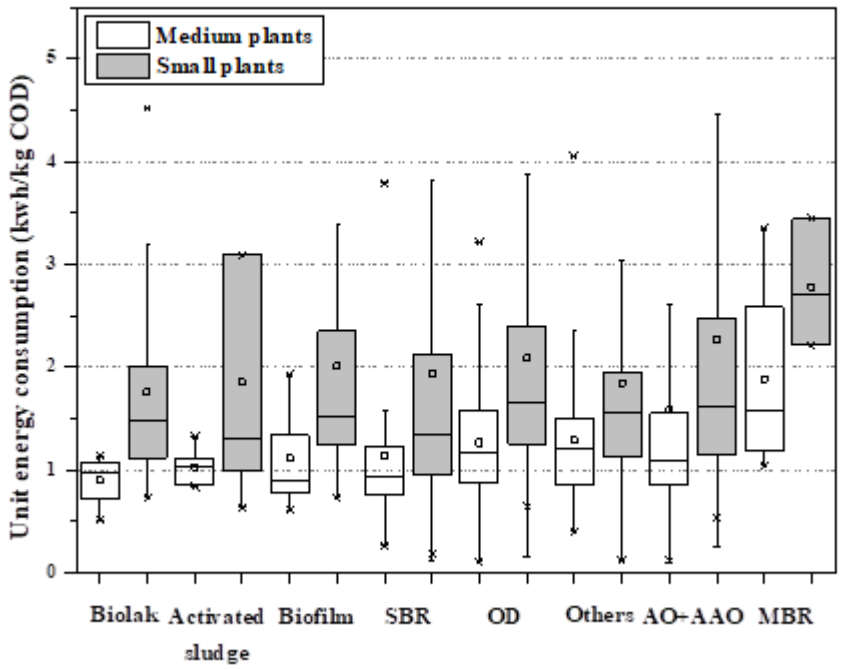


Figure 6.10 Unit energy consumption of small and medium-sized WWTPs with different technologies

Qasim and Zhu [28] summarized retention time and sludge concentration for different kinds of suspended growth biological treatment processes in table 6.1. Solids retention times of SBR and OD are higher than those of CAS and AAO, which reduce sludge production due to high SRT. AAO, OD and SBR have similar mixed liquor suspended solids values, which indicates they could produce sludge with similar

concentrations. Hydraulic retention time represents the average time that wastewater stays in a reactor, which determines the reactor volume based on the flow rate. Table 6.1 shows that SBR has a relatively lower HRT than AAO and OD. Therefore, building an SBR reactor requires smaller land and should be the preferred secondary treatment process due to relatively low sludge production with a high concentration of solids and a small footprint.

Since large-sized (>200,000 m^3/d) plants often used AO+AAO technology as mainstream treatment, there are few large WWTPs with OD and SBR technology, as shown in Figure 6.3. Figure 6.11 presents the unit energy consumption of three main technologies in small and medium-sized WWTPs in China. Variations of energy demand in small plants are higher than those in medium plants, which is affected by several factors such as wastewater quality and climate. For AO+AAO, OD, and SBR technology, the unit energy demand of WWTPs decrease with increasing design flow rate. Gu et al. [260] and Trapote et al. [243] have also reported similar energy trends. Large WWTPs are more energy efficient than small ones due to economies of scale [261-263]. Sludge treatment technologies of large and small facilities are different. Aerobic digestion was used for stabilizing the sludge of small or older plants [264]. Anaerobic digestion was applied for producing CH_4 in plants with an influent flow rate of greater than 5 million gallons per day [265]. In China, large plants such as Gaobeidian WWTP and Hangzhou sibao WWTP have used anaerobic digestion for treating sludge [266]. For example, Gaobeidian WWTP could achieve 31% of energy recovery [245]. Large size plants used energy-efficient equipment and anaerobic digestion to reduce electricity consumption.

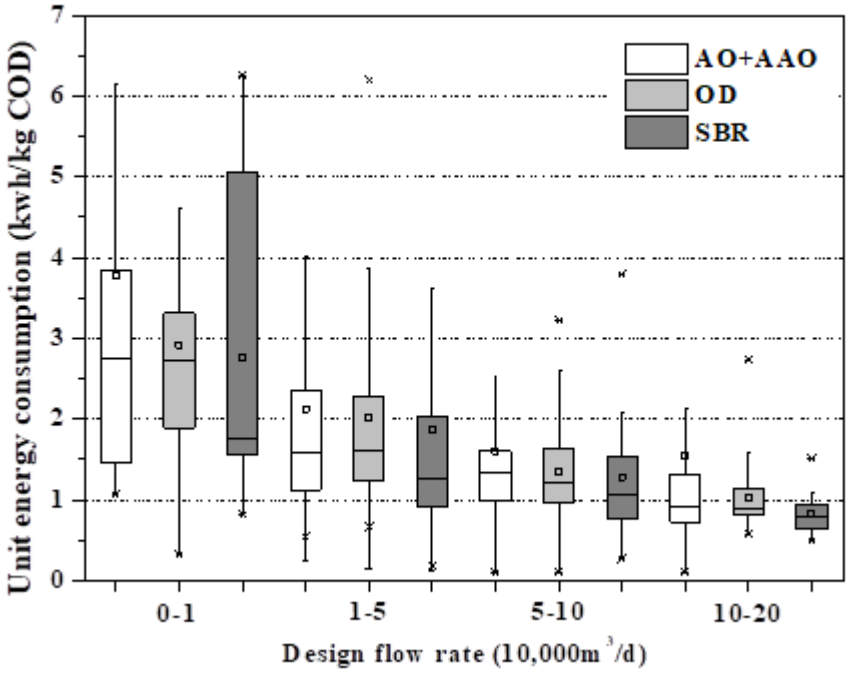


Figure 6.11 Unit energy consumption of small and medium-sized WWTPs

Since the number, actual flow rate, and energy consumption of plants with main technologies were the largest in design flow rate range between 1×10^4 m^3/d and 5×10^4 m^3/d in Figure 6.3, the mean unit energy consumption of mainstream technologies is close to the national average value of 1.811 kWh/kg $COD_{removed}$. At a design flow rate of less than 1×10^4 m^3/d, the unit energy consumption of three technologies is much higher than the national average value, which indicated that there is room for improving the energy efficiency of small-sized WWTPs using AO+AOO, OD, and SBR. In the design flow rate range between 1×10^4 m^3/d and 20×10^4 m^3/d, the unit energy consumption of SBR is lower than the national average value. Since SBR eliminates the need for secondary sedimentation tanks and return sludge flow, it consumes lower energy and requires a smaller footprint than AO, AAO, and OD

systems. For aeration methods, SBR processes generally use blast aeration, while OD typically uses mechanical aeration. For the same oxygen demand, blast aeration could save more electricity than mechanical aeration. As shown in Figure 6.11, the average unit energy consumption of SBR decreased from 2.76 kWh/kg to 0.83 kWh/kg when the design flow rate increased from less than 10,000 m^3/d to 100,000-20,0000 m^3/d, which shows that SBR is more energy-efficient than other activated sludge process processes in small and medium WWTPs in China. Similar results could be found in Figure 6.12. Therefore, SBR technology is more appropriate for small and medium-sized plants than other biological methods.

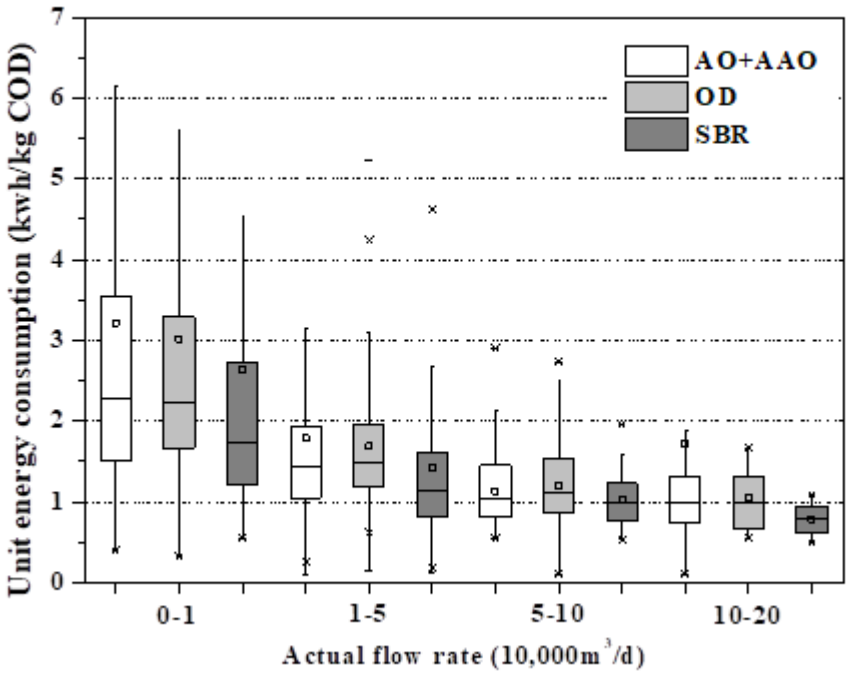


Figure 6.12 Variation of unit energy consumption with the actual flow rate

6.4.5.2. Operation loading rate

The operation loading rate of a WWTP represents the ratio between the actual flow rate and the design flow rate. Since the flow rate varies widely throughout the day, the

design treatment flow rate of WWTPs is based on the peak flow rate. The operation loading rate is lower than 100% under normal conditions. However, the flow rate to different wastewater treatment units may be higher than their design flow rate under certain conditions and weather. Figure A.13 ranks unit energy consumption of treatment technology at different operation loading rates. Biolak, biofilm, and activated sludge have the lowest energy consumption with an operation loading rate of 20%-40%, 40%-60%, 80-100%, and >=100%. Since Biolak and biofilm use moving aeration chains and natural draft as the primary aeration methods, respectively, they show high energy efficiency at a relatively low operation loading rate. The activated sludge process could save more energy than other technology at an operation loading rate of greater than 80%. This is because it is mainly applied for removing BOD and uses primary clarifiers for separate particulate COD without consuming oxygen. Due to the disadvantages, such as low treatment performance and large space requirements, they are not the main treatment technology in China. The unit energy consumption deteriorates as the operation loading rate increases, as shown in Table A.34. The low operational rate affects the flow rate and concentration of organic matter, which results in worse energy performance. The influence of the operation loading rate on MBR is relatively lower than the effect on other biological treatment technologies. Park et al. [267] investigated the impact of influent wastewater flowrate on the COD removal efficiency of MBR, SBR, and AAO through changing the influent flowrate from 100% to 70%, 40%, and 10% of the design treatment flow rate. At 10% of the design flow rate, A2O and SBR could only remove 74% and 70% of soluble chemical oxygen demand, whereas the MBR process has the ability to remove 89% of soluble sCOD from wastewater. Due to the high MLSS concentration of 5000-15000 mg/L, as shown in Table 6.1, the MBR reactor has higher BOD and COD removal than the conventional

activated sludge process and good adaptability to various changes of flow rate and wastewater quality [268]. However, suspended solids and active microorganisms in MLSS, colloids, solutes, and sludge flocs accumulate on the membrane surface and clog the pore, which causes membrane fouling [269, 270]. The aerobic granulation membrane bioreactor combines aerobic granulation biotechnology and membrane separation to control membrane fouling. This leads to better treatment performance due to excellent settling properties, smaller land requirements, higher biomass concentration, lower sludge generation, and strong resistance to shock loading rates [271-276]. High levels of MLSS in MBRs leads to high total suspended solids content and viscosity values [277]. The high viscosity in MBRs can limit aeration rates, which increases the energy consumption and results in a higher cost for aeration [278, 279]. The unit energy consumption value of 1.043-2.698 kWh/kg $COD_{removed}$ is higher than the other treatment technologies, as shown in Table A.34.

Figure 6.13 illustrates that the unit energy consumption of the WWTPs decreased as the operation loading rate increased. The unit energy consumption is significantly higher when the actual flow rate is lower than the design flow rate. However, energy consumption decreased when the annual flow rate approached the design value. Long term operation of WWTPs at a low operation loading rate leads to increased energy consumption. At an operation loading rate of less than 20%, the unit energy consumption of OD is significantly higher than those at other operation loading rates, as shown in Figure 6.13. The energy demand of OD systems is higher at a low operation loading rate due to longer hydraulic retention time. Therefore, the operation loading rate strongly affects the unit energy consumption of OD plants. The mean values for AO+AAO and OD are higher than the national average value at the operation loading rate of 0-80%. In contrast, at an operation loading rate of 40%-100%, the mean energy

consumption of SBR is lower than the national average value and decreased gradually from 1.71 kWh/kg$COD_{removed}$ to 1.32 kWh/kg$COD_{removed}$, which confirms that effect of operation loading rate on energy demand of SBR system is relatively small as reported by a previous study [145]. The unit energy consumption of AO+AAO and OD systems decreased at an operation loading rate of greater than 100%. However, hydraulic and organic overloadings lead to reduced aeration time and insufficient nutrient availability, which reduces the pollutants removal performance of treatment plants [280]. Therefore, an operation loading rate of 80–100% is recommended for mainstream technology in China.

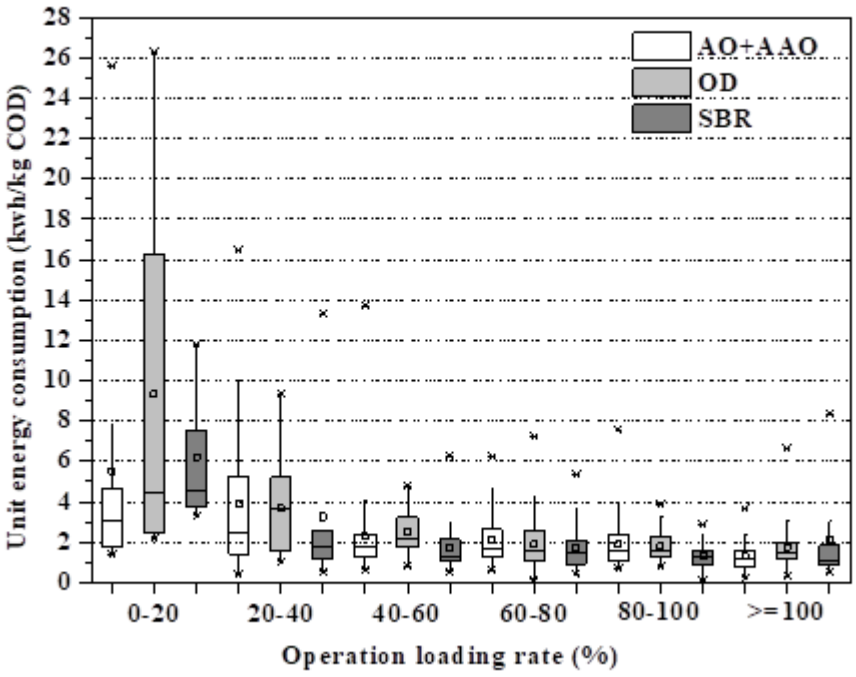


Figure 6.13 Variation of unit energy consumption with the WWTP operation loading rate in China

6.4.5.3. Chemical oxygen demand (COD) removal

Typically, pollutant removal efficiency is closely related to energy consumption [281]. However, such a correlation has not been quantitatively reported. The COD removal efficiency of a WWTP represents the treatment performance of removing organic matter. Figure A.14 ranks unit energy consumption of treatment technology in different COD removal scale. Wetland and activated sludge have the lowest energy consumption with operation loading rates of 70%-80% and 90%-100%. The number of WWTPs with COD removal of 80%-100% is much larger than that with COD removal of 60%-80%. Figure A.14 and Table A.35 show that all the treatment technologies have the ability to achieve COD removal of greater than 80%. COD removal efficiency is impacted by influent and effluent wastewater quality. Biological treatment technologies could achieve high COD removal. However, effluent wastewater quality may not meet discharge requirements due to the high concentration of influent COD. Based on the discharge standard, 97.45% of 1215 WWTPS in this study with effluent concentration COD of less than 50 mg/L meets the effluent standard of grade I-A. This demonstrates that most WWTPs in China could achieve COD removal of greater than 80% and achieve the effluent COD requirement.

In Yearbook 2016, 91.3% of WWTPs could treat 96.8 % of wastewater with COD removal efficiency of 80%-100%, and power consumption accounted for 97.4 % of the total energy demand, which indicates that the COD removal efficiency of WWTPs in China was satisfactory. According to COD removal, the WWTPs with different mainstream technologies were classified into four categories as follows: 60%-70%, 70%-80%, 80%-90%, and 90%-100%, as shown in Figure 6.14. The unit energy consumption of the WWTPs is negatively related to COD removal. OD system with COD removal of 60%-70% shows significantly low energy efficiency than other

technologies, which may be attributed to low operation loading rates of the OD plants. Moreover, the mean unit energy consumption of WWTPs with COD removal of 80%-100% was lower than the national average value because of the high operation loading rate. Due to the small influence of the operation loading rate on the SBR system, the mean energy consumption of SBR decreased from 2.27 kWh/kg$COD_{removed}$ to 1.30 kWh/kg$COD_{removed}$ as COD removal increase from 70% to over 90%, which confirms that the change of unit energy consumption in SBR plants is lower than those of other technologies in Figure 6.14. Since effective treatment and operation of plants could improve energy performance, high COD removal led to more efficient energy consumption.

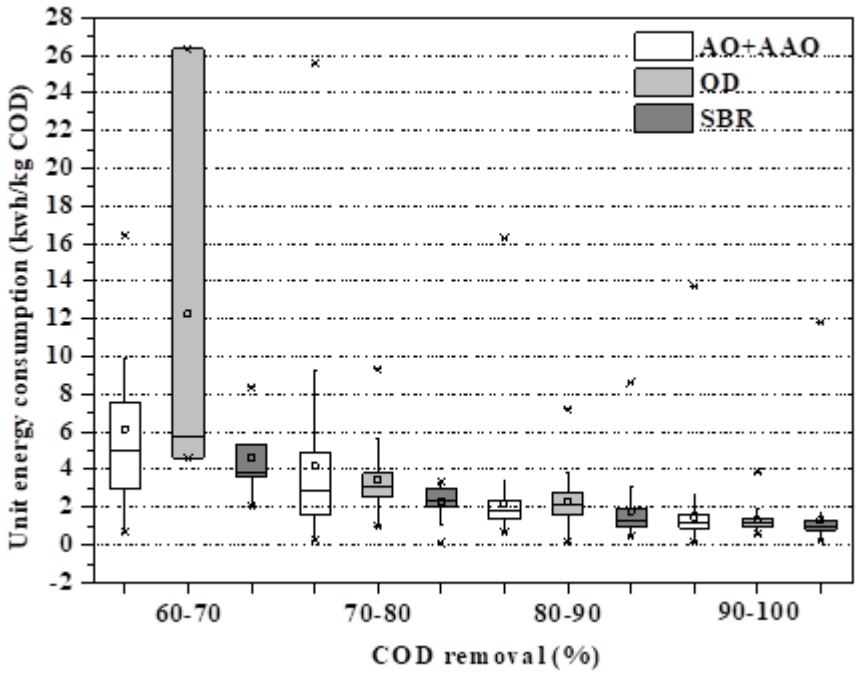


Figure 6.14 Variation of energy consumption per unit mass of COD removal with COD removal in selected WWTPs in China.

6.4.5.4. Influent chemical oxygen demand (COD) concentration

The influent COD of a WWTP is an important parameter that affects COD removal efficiency and energy consumption. Figure A.15 ranks unit energy consumption of treatment technology in different influent concentrations of COD. Wetland, AS and SBR have the lowest mean unit energy consumption with COD concentration of 0 mg/L-150 mg/L, 150 mg/L-250 mg/L, 250 mg/L-450 mg/L and 450 mg/L-1000 mg/L. In figure A.15 and table A.36, all treatment technologies have the ability to treat low and medium strength wastewater with COD concentration of 150 mg/L-450 mg/L. Wetland and AS show relative higher energy efficiency for treating low and medium strength wastewater than other treatment technologies. However, they are not applied for treating wastewater with a COD concentration of greater than 450 mg/L due to the limited treatment performance of high strength wastewater. WWTPs have higher unit energy consumption (kWh/m^3) when treating wastewater with a higher concentration of COD [282]. In addition, unit energy consumption(kWh/m^3) is positively related to influent COD concentration. However, this might mislead our understanding of the impact of influent COD concentration on energy consumption [38, 225, 237, 283]. Silva and Rosa [233], Vaccari et al. [38], and Niu et al. [282] confirmed that unit energy consumption (kWh/kg $COD_{removed}$) of WWTPs decreases as influent concentrations of COD increases. As shown in Table A.36, the average unit energy consumption of biolak, SBR, and MBR decreases up to approximately 50% when the COD concentration is higher than 450 mg/L. In most areas of China, rainwater and groundwater often dilute the wastewater, which results in a low COD influent concentration [284]. WWTPs waste energy when receiving large amounts of stormwater runoff [38, 231]. This explains why the energy consumption of WWTPS

with combined pipelines is lower than that of plants with separate systems for wastewater and stormwater.

Based on COD concentration, untreated wastewater could be classified into ultra-low strength (0-150 mg/L), low strength (150-250 mg/L), medium strength (250-450 mg/L), and high strength (450-1000 mg/L). The unit energy consumption decreases with increasing COD concentration in Figure 6.15. WWTPs, which receive diluted wastewater with a lower concentration of COD, have higher unit energy consumption. Because WWTPs use fixed energy for removing lower strength wastewater and consume more energy in pumping the same flow rate of water. The average unit energy consumption of SBR decreased from 3.89 kWh/kgCOD to 0.63 kWh/kgCOD when treating wastewater with low, medium, and high concentrations of COD, which show lower unit energy consumption than other technology. Thus, SBR is energy-saving technology for treating all types of wastewater.

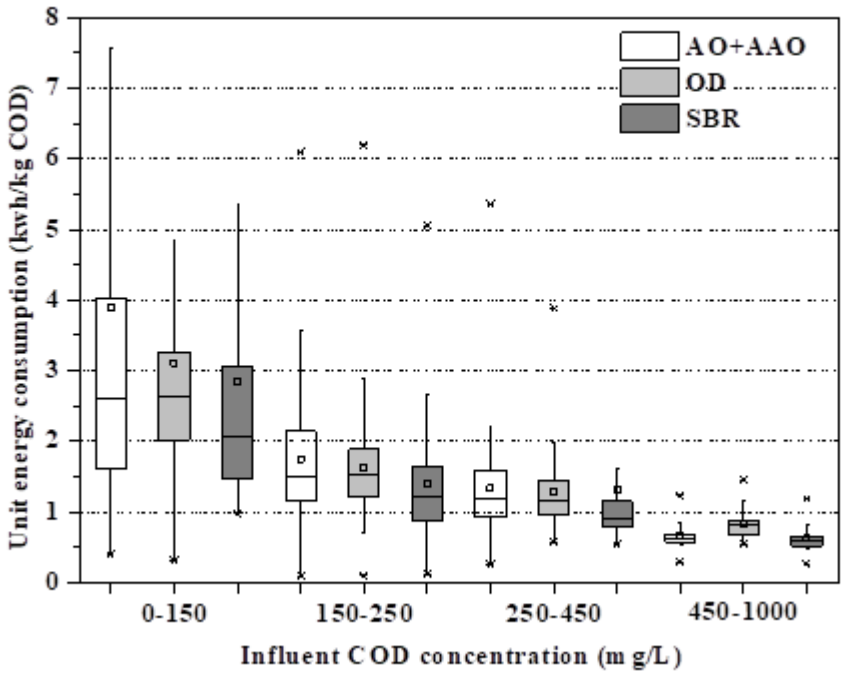


Figure 6.15 Variation of unit energy consumption with influent COD concentration in China.

6.4.6. Energy benchmarking based on the regression model

Table 6.5 shows the Pearson's correlations between the energy consumption of different technologies, design flow rate, operation loading rate, influent COD concentration, and influent COD mass. All the correlations were significant at the 0.01 level. Compared with operation loading rate and influent COD concentration, the design flow rate, actual flow rate, and influent COD mass showed stronger positive correlations with energy consumption. According to the benchmarking method reported by Yang et al. [145], the mean unit energy consumptions of three main technologies were selected as the benchmark data of unit energy consumption under different conditions of design flow rate, actual flow rate, and influent COD mass. The equations for predicting benchmark data of unit energy consumption from the design flow rate, actual flow rate, and influent COD mass are listed in Table 6.6.

Table 6.5 Correlations between the energy consumption of three main technologies, design flow rate, operation loading rate, influent COD concentration, and influent COD mass.

Pearson Correlation	Design flow rate (10,000 m^3/d)	Operation loading rate (%)	Influent COD concentration (mg/L)	Actual flow rate (10,000 m^3/d)	Influent COD mass (kg/d)
Energy consumption of AO+AAO (kwh/d)	.889**	.233**	.247**	.901**	.915**
Energy consumption of OD (kwh/d)	.902**	.181**	.407**	.916**	.910**
Energy consumption of SBR (kwh/d)	.894**	.329**	.368**	.905**	.935**

** At the 0.01 level (2-tailed), correlation is significant.

Table 6.6 Statistical results of mean unit energy consumption

Treatment technology	Regression equation	Label	R Square
AO+AAO	$UEC=3.0089(Q_{design})^{-0.276}$	6.11	0.9672
OD	$UEC=2.4924(Q_{design})^{-0.320}$	6.12	0.9608
SBR	$UEC=2.3607(Q_{design})^{-0.339}$	6.13	0.9408
AO+AAO	$UEC=2.4867(Q_{actual})^{-0.237}$	6.14	0.6684
OD	$UEC=2.3971(Q_{actual})^{-0.319}$	6.15	0.9953
SBR	$UEC=2.0744(Q_{actual})^{-0.354}$	6.16	0.9994
AO+AAO	$UEC=18.541(W_{COD})^{-0.254}$	6.17	0.8471
OD	$UEC=13.529(W_{COD})^{-0.23}$	6.18	0.8697
SBR	$UEC=12.758(W_{COD})^{-0.254}$	6.19	0.8226

Note: UEC=Unit energy consumption (kWh/kg $COD_{removed}$); Q_{design}= Design flow rate (10,000 m^3/d), Q_{actual}= Actaul flow rate (10,000 m^3/d) and W_{COD}=Influent COD mass (kg/d)

The R^2 value in equation (6.14) is relatively lower because the mean unit energy consumption is affected by extreme values, as shown in Figure 6.12. Due to the high R^2 value in most equations, regression equations are suitable for estimating benchmark data of unit energy consumption. Equations (6.11, 6.12, 6.13), which are based on the design flow rate, are applied for energy consumption predictions during the WWTPs design stage. When plants are put into operation, wastewater quality and operation loading rate and could be measured. Then unit energy could be calculated based on an equation (6.14, 6.15, 6.16, 6.17, 6.18, 6.19). The estimated benchmark value could be compared with actual unit energy consumption to check the energy efficiency of WWTPs.

6.5. Conclusions

The current study used the China national data of WWTPs and compared the performance of wastewater treatment systems with different main biological technologies. In China, the WWTPs used AO + AAO, OD, and SBR as main wastewater treatment technologies accounted for 80.3% of WWTPs. The number of small and medium-sized plants accounted for 97% of WWTPs. Larger WWTPs

received wastewater with a higher concentration of COD. The effect of the COD concentration should not be ignored when evaluating the energy efficiency of WWTP. The unit energy consumption (kWh/m^3) leads to incorrect results of energy audits. Pearson correlation analysis shows that COD removal is strongly related to energy consumption for all plants with main treatment technologies. The unit energy consumption (kWh/kg $COD_{removed}$) was used for evaluating the energy efficiency of mainstream technologies. Statistical and classification analyses were carried out to study the effect of design flow rate, operation loading rate, COD removal, and influent concentration of contaminants on the energy consumption of different treatment processes. Larger centralized WWTPs have high energy performance due to their high operation loading rates and high COD removal. However, centralized wastewater treatment might not be appropriate for the development of cities because more dispersed wastewater sources need longer sewers for collecting wastewater. Since SBR eliminates the need for secondary sedimentation tanks and return sludge, and uses blast aeration, SBR technology shows higher energy efficiency for treating all kinds of wastewater in small and medium-sized plants than other biological methods. Power regression equations between the benchmark value, design flow rate, and influent COD were developed in this study, which will provide a valuable reference for predicting the benchmark data of unit energy consumption for the future sustainable environmental design of WWTPs.

7. COMPARISON OF UNIT ENERGY AND COST OF SYSTEMS WITH INNOVATIVE TECHNOLOGIES IN TREATING LEACHATE AND INDUSTRIAL WASTEWATER

To compare the unit energy and cost of leachate and industrial wastewater treatment systems with innovative technologies, Excel models were developed in designing treatment systems of 1 MGD leachate using different innovative technologies based on leachate and industrial wastewater in table 3.1 and 5.1. Process flow diagrams of the treatment system are illustrated in Figure 3.2. To meet discharge requirements and assess the influence of different kinds of wastewater on treatment systems in terms of unit energy and cost metrics, systems were designed to achieve the same discharge standard (COD≤50mg/L, BOD≤20mg/L, and N≤5mg/L) so that systems have similar COD and N removal efficiency. In the systems, screen, grit chamber, micro sieving, UASB, PN/A, nitrification-denitrification process, and granular activated carbon system were used as a preliminary treatment, primary treatment, secondary treatment, and tertiary treatment for removing particulate matter, biodegradable COD and N and non-biodegradable soluble organic matter from wastewater, respectively. Due to low biodegradability of old leachate, sludge age is assumed to be 60 days for keeping stable performance of leachate and industrial wastewater treatment systems.

Since secondary biological treatment consumes 50–70% of the overall energy consumption of a WWTP [41], energy required by aeration was used as the dominant energy consumption in this study. Figure 7.1 shows that the electricity consumption rises as influent concentration of NH_4^+_N and soluble biodegradable organic nitrogen increases. Because most of the O_2 is used for PN/A process and nitrification process. Most of the sbON was converted to NH_4^+-N by ammonification prior to or during the aeration process. Since primary treatment and UASB removes COD without using

oxygen, main energy consumption is strong positive related to NH_4^+_N plus soluble biodegradable organic nitrogen. Except for BOD that is converted to CO_2 in aerobic treatment, most of soluble BOD and biodegradable particulate BOD in sludge are converted to CH_4 to produce electricity and heat by UASB and the side stream anaerobic digester. Figure 7.2 illustrates energy production rises with increasing influent BOD concentration. Since main energy consumption and energy production are related to influent concentration of NH_4^+_N and soluble biodegradable organic nitrogen (sbON) and BOD, unit energy consumption (kwh/kg $N_{removed}$) and unit energy consumption (kWh/kg $BOD_{removed}$) are used as indicators in assessing the influence of different wastewater on energy and consumption of treatment systems as shown in Figure 7.3 and 7.4.

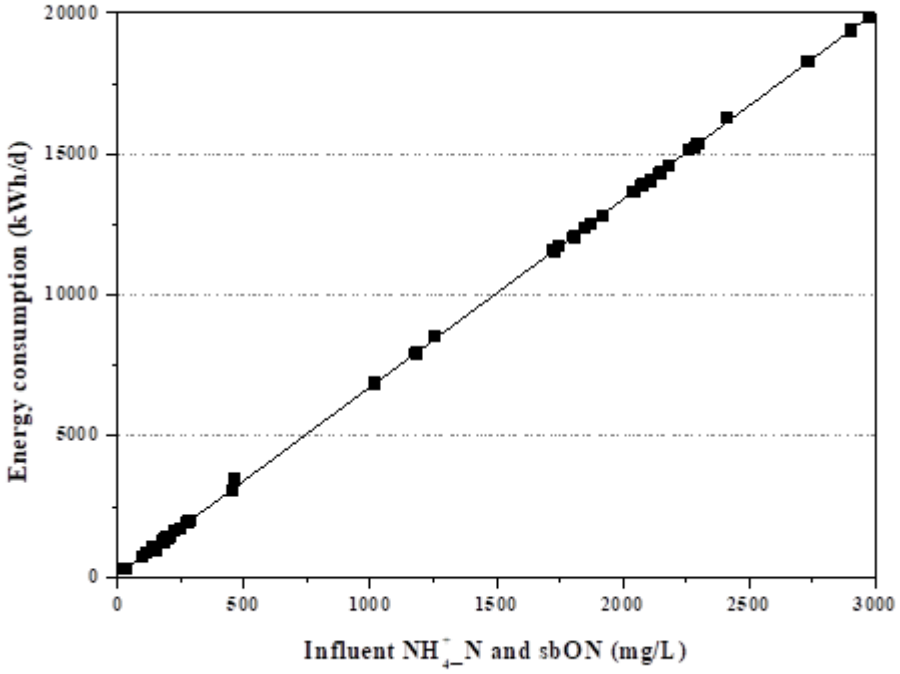


Figure 7.1 Electricity consumption and influent NH_4^+_N and sbON concentration

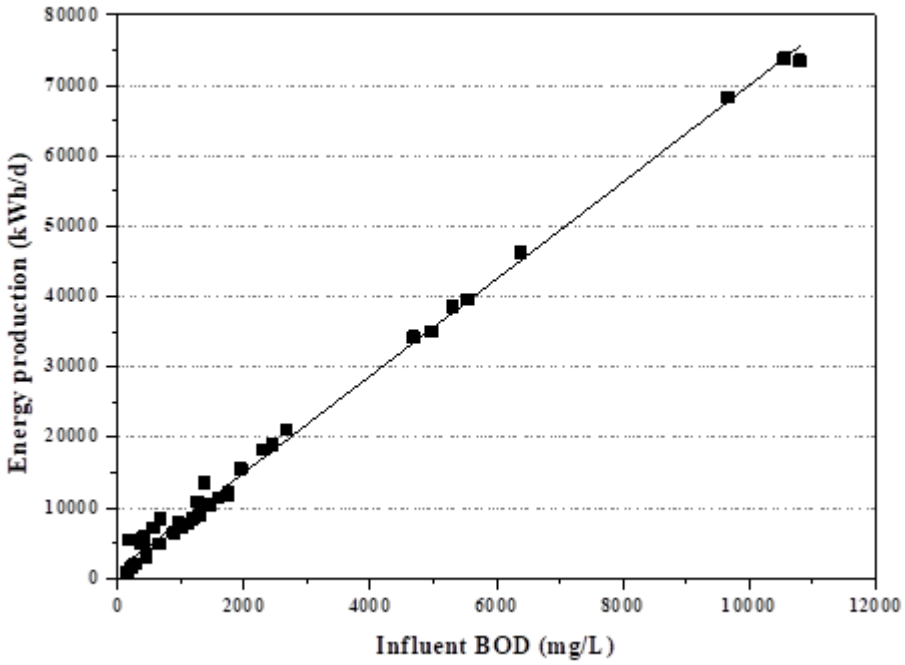


Figure 7.2 Electricity production and influent BOD concentration

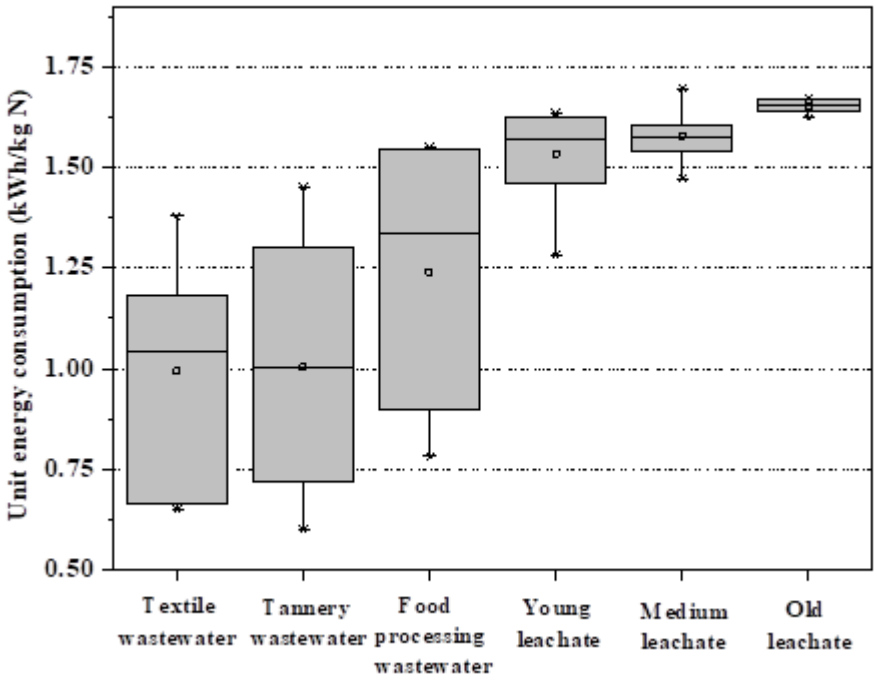


Figure 7.3 Unit energy consumption

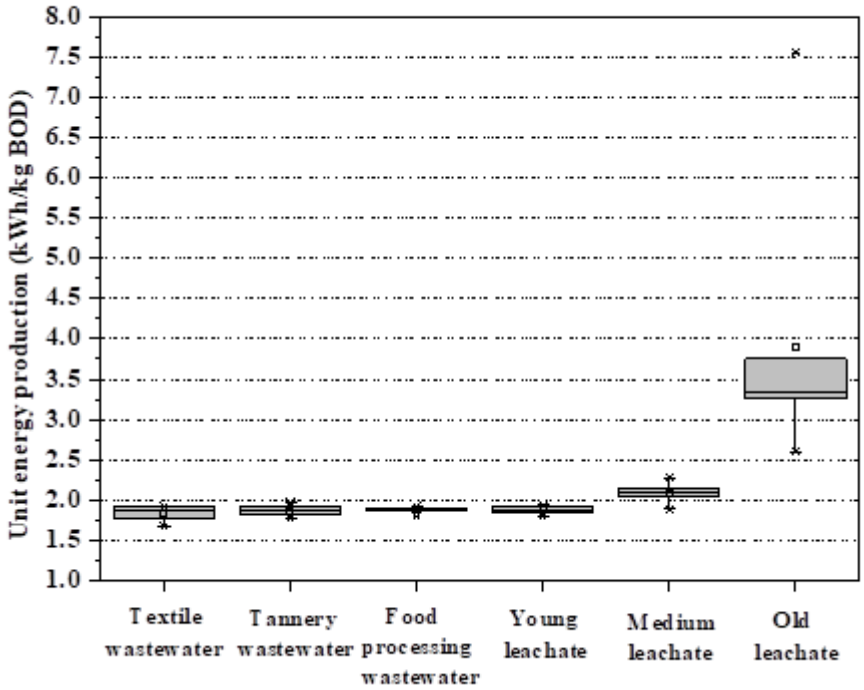


Figure 7.4 Unit energy production

In figure 7.3, average unit energy consumptions are 0.99, 1.00, 1.24, 1.53, 1.58 and 1.65 kWh/kg $N_{removed}$ for textile, tannery, food processing wastewater, young, medium, and old leachate, respectively. The unit aeration energy consumption for textile wastewater is lower than that of the other industrial wastewaters and leachate. TN removal depends on the separation of particulate N by primary treatment, oxidation of NH_4^+_N to nitrite-nitrogen or nitrate-nitrogen (NO_3_N), and conversion of NO_2_N or NO_3_N to N_2 by secondary treatment and adsorption of soluble nonbiodegradable N by tertiary treatment. N removal of textile wastewater depends more on sedimentation and adsorption than other wastewaters. As shown in Figure 7.5, the average ratio between N removal by secondary, tertiary treatment and total N removal are 45.42%, 34.18%, 31.45%, 11.40%, 10.74%, and 6.25%, respectively. Since particulate and soluble nonbiodegradable N removal do not consume oxygen (O_2) in the treatment system, textile wastewater needs a smaller amount of O_2 to remove N as compared to other wastewaters.

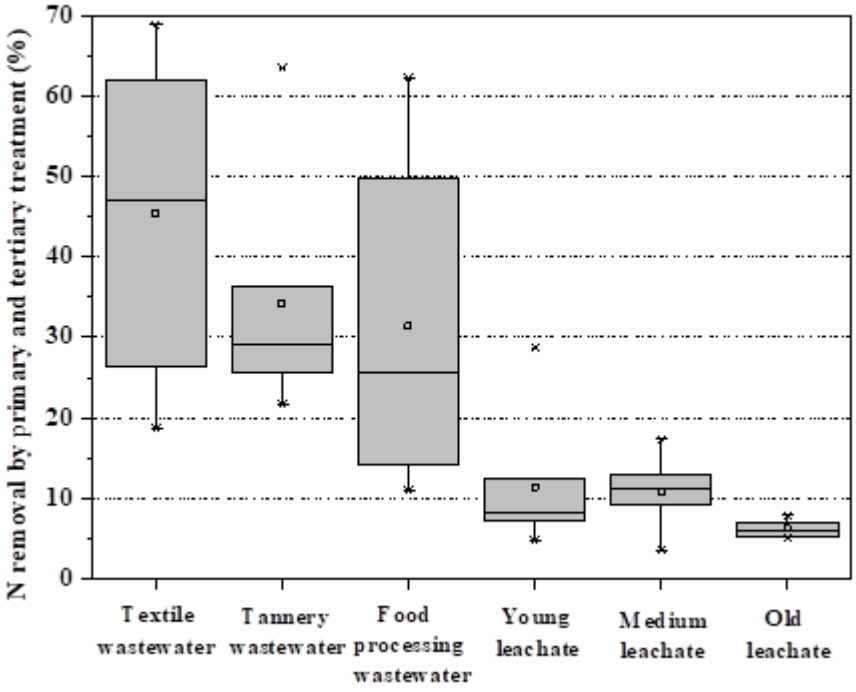


Figure 7.5 N removal by primary and tertiary treatment (%)

Figure 7.4 compares the unit energy production of treatment systems for different wastewater. Average unit energy productions are 1.84, 1.87, 1.87, 1.88, 2.09, and 3.90 kWh/kg $BOD_{removed}$ for textile, tannery, food processing wastewater, young, medium and old leachate, respectively. The unit energy production for old leachate is higher than that of the other wastewaters. Although old leachate contains has a low concentration of COD and BOD/COD ratio, external carbon source in PN/A, nitrification and the denitrification process increase electricity production through increasing sludge side stream AD. The reaction of nitrification, denitrification and PN/A process can be depicted as the following equations:

$$NH_4^+ + 1.86O_2 + 1.98HCO_3^- \rightarrow 0.021C_5H_7NO_2 + 0.98NO_3^- + 1.88CO_2 + 2.92H_2O \quad (7.1)$$

$$NO_3^- + 1.08CH_3OH + 0.24CO_2 \rightarrow 0.47N_2 + 0.065\ C_5H_7NO_2 + 1.43\ H_2O + HCO_3^-$$

(7.2)

$NH_4^+ + 0.799\ O_2 + 1.109\ HCO_3^- \rightarrow 0.436\ N_2 + 0.111\ NO_3^- + 1.034\ CO_2 + 2.496\ H_2O + \text{New biomass}$

(7.3)

The nitrification and PN/A process used inorganic matters as a source for carbon. Due to the high BOD removal of UASB, the heterotrophic denitrification process needs exogenous organic carbon source such as methanol for the conversion of nitrate to nitrogen, or nitrite to nitrogen. Nitrification, denitrification and the PN/A process converted these carbon sources to organic matter in sludge as shown in equation (7.1, 7.2 and 7.3). Since these carbon sources are not from wastewater, BOD removal from wastewater does not include an external carbon source. Since external carbon sources could be converted to biomass to increase CH_4 production of AD, external carbon sources could be considered as external BOD. Total BOD is the combination of total BOD removal from wastewater and external BOD as shown in equation (7.4):

Total BOD=BOD removal from wastewater + external BOD

(7.4)

The ratio between external BOD and total BOD represents the dependency of electricity production on the external carbon source. In figure 7.6, the mean external BOD/total BOD ratio are as follows: 6.42%, 7.77%, 9.83%, 16.21%, 38.48%, and 75.73% for textile, tannery, food processing wastewater, young, medium, and old leachate, respectively. Old leachate has a high concentration of NH_4^+_N and soluble biodegradable organic nitrogen, the system needs a great amount of carbon source to achieve high N removal. Energy production of old leachate treatment systems depends more on the external carbon source as compared to other wastewater treatment systems.

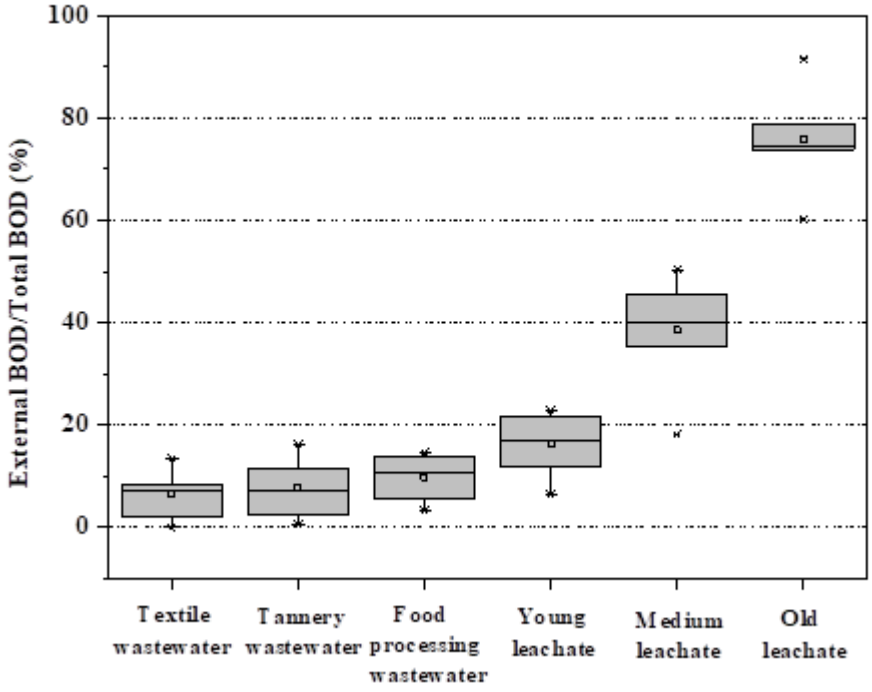


Figure 7.6 Ratio between external BOD and total BOD

Since preliminary treatment, primary treatment, secondary treatment, and tertiary treatment are used for removing particulate COD, biodegradable COD, and non-biodegradable soluble COD from wastewater, respectively, unit cost metric ($/kg $COD_{removed}$) is selected as the indicator for this study. As shown in equation (2.17), the total unit cost is the combination of the unit capital costs and the unit operational costs. Figure 7.7 compares the unit cost of treatment systems for different wastewater. Average unit costs are 0.41, 0.61, 0.64, 0.74, 1.32, and 1.51 for young leachate, tannery wastewater, food processing wastewater, medium leachate, textile wastewater, old leachate, respectively. The unit cost for young leachate is lower than that for the other wastewaters. As compared to other wastewater, young leachate has a higher biodegradability and concentration of BOD as shown in figure A.16 and A.17, which could produce a large amount of CH_4 for energy recovery to reduce the unit cost of a

treatment system. On the other hand, COD removal strongly affects unit costs of the unit process as shown in figures 7.8 and 7.9. The unit cost of micro sieving and UASB systems decrease with increasing COD removal. TSS/COD and BOD/COD ratio determine performance and COD removal efficiency (%) of micro sieving and UASB as shown in figures 5.5 and 5.8. COD concentration, flow rate, and COD removal efficiency determine the mass removal of COD. Young leachate has relatively higher COD and BOD concentration than other wastewaters as shown in table 3.1 and figure A.17, this leads to lower unit costs of the unit process. Therefore, systems with innovative technologies is more suitable for treating young leachate than other wastewaters.

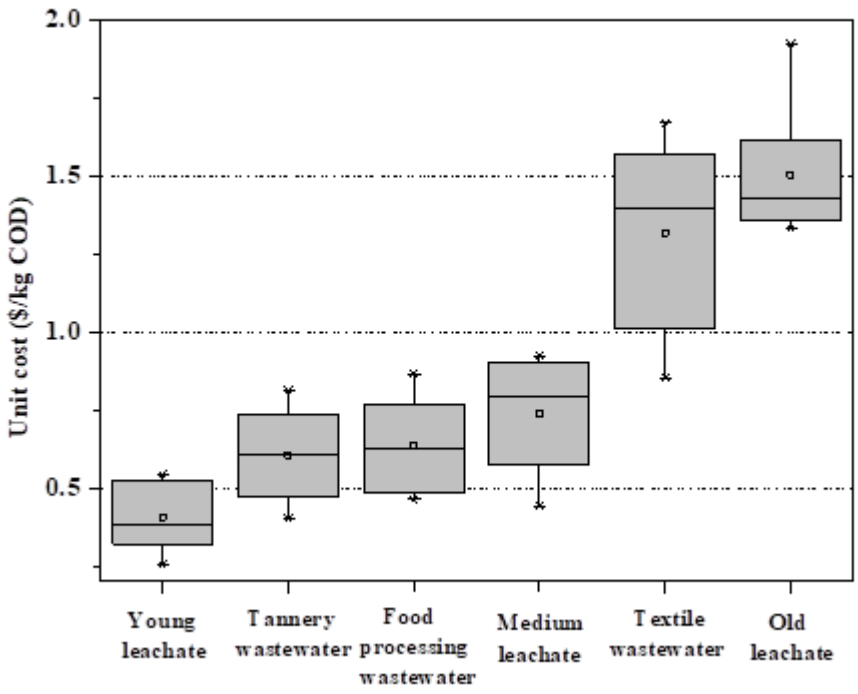


Figure 7.7 Unit cost

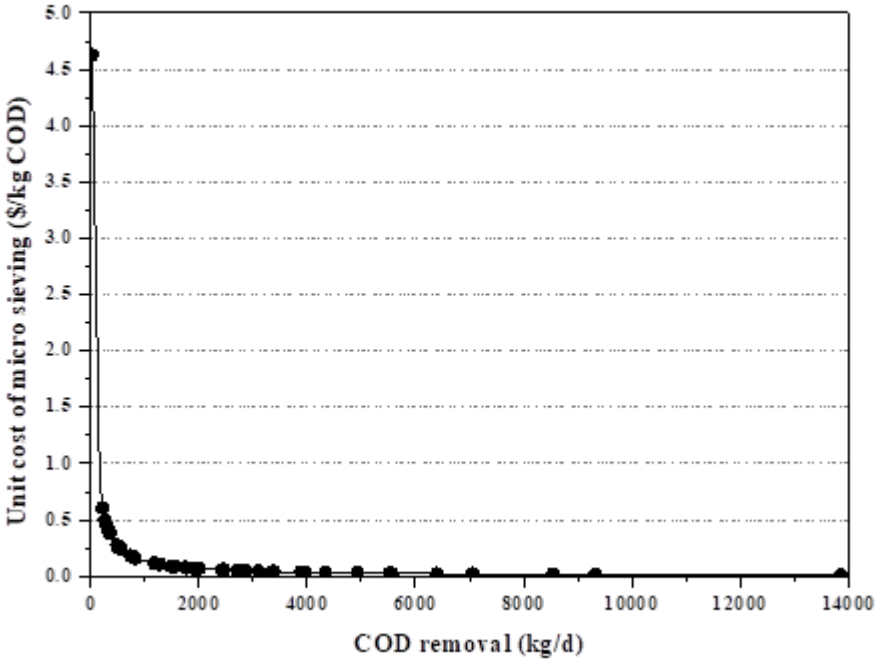


Figure 7.8 Unit cost of micro sieving

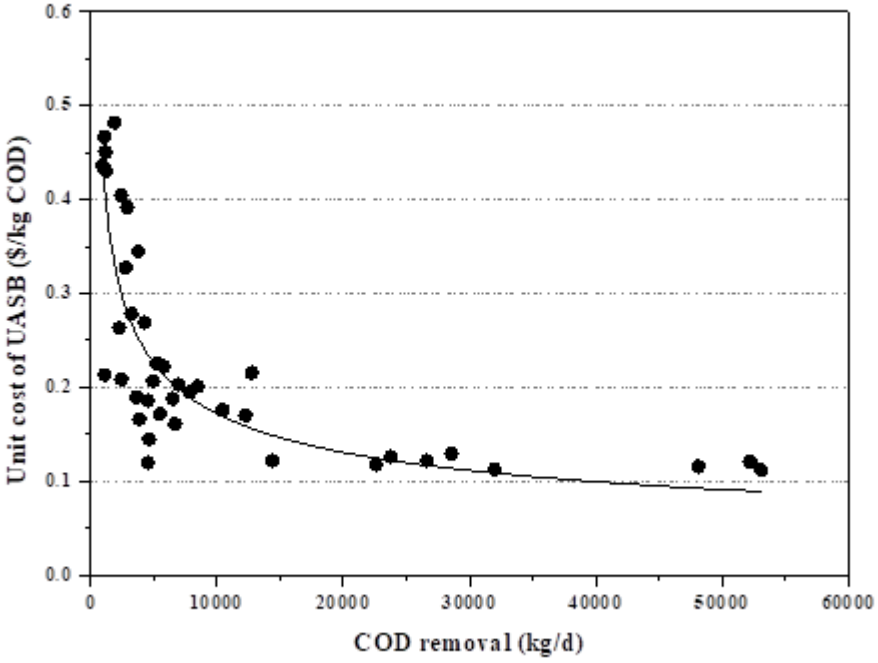


Figure 7.9 Unit cost of UASB

8. CONCLUSION

The current study collected different leachate and industrial wastewater quality data and developed Excel-based models for comparing the performance of treatment systems with innovative and conventional technologies for different wastewater quality. Systems with innovative technologies could remove most of COD and NH_4^+_N of industrial wastewater and leachate. However, effluent wastewater quality of energy positive system might still do not meet the discharge standards due to the high concentration of COD, BOD, and TN in industrial wastewater and leachate. This system should combine with physical-chemical treatment to increase pollutants removal efficiency to meet the discharge requirement. The energy and economic analysis showed that energy consumption, production, and cost are related to BOD, N, and COD removal, especially for system innovative technologies. The results show that systems using innovative technologies is the best treatment system for treating wastewater with high COD concentration and BOD/COD ratio in unit energy and cost.

Micro-sieving consumes more electricity than primary clarifiers for treating the same volume of wastewater. However, micro sieving shows a lot of advantages such as small footprint, modular and flexible design, recyclable materials, and high solid removal with high percent dry solids according to the design principles of Sustainable Environmental Engineering. In treating wastewater with high TSS concentration, micro-sieving could retain more particulates on the filter and divert more COD and N in sludge with higher percent dry solids to anaerobic digester than clarifier, which decreases CH_4 production of mainstream UASB, O_2 demand of PN/A reactor, size and cost of UASB and PN/A reactor. These aspects demonstrate that micro-sieving could help energy-positive wastewater treatment systems save costs.

Pearson correlation analysis in a study about industrial wastewater shows that TSS/COD and BOD/COD ratio are the key parameters in determining COD removal and methane conversion rate of primary treatment and secondary treatment. UASB is ineffective for converting COD of wastewater with a low BOD/COD ratio to biogas. The TSS/COD ratio and BOD/COD ratio could be effectively used to predict the total COD removal and effluent COD concentration of wastewater in the treatment system with primary treatment and biological processes.

In the Excel model, wastewater quality affects treatment performance, energy, and costs of the innovative treatment systems. In practice, aeration method, design capacity, operation loading rate, COD removal, and influent concentration of contaminants have a significant influence on the energy consumption of WWTPs with different conventional biological technologies which include anaerobic- anoxic–oxic, anaerobic-oxic, oxidation ditch, sequencing batch reactor, membrane bioreactor, BIOLAK, biofilm, activated sludge, and combination of these technologies. Biolak and Biofilm use the floating aeration chain system and natural draft as an aeration method, which reduces oxygen demand and improves energy efficiency. The activated sludge process shows low energy consumption because it is mainly applied for removing BOD and uses primary clarifiers for separate particulate COD without consuming oxygen. However, due to disadvantages such as low treatment performance and a large carbon footprint, they are not the main treatment technology in China. MBR overcomes the disadvantages such as large space requirement for secondary clarifiers, excess sludge generation, and high removal rate of contaminants. However, MBR technology has disadvantages such as high energy costs, membrane fouling problems, and high costs of periodic membrane replacement. All of these result in high maintenance and operating costs. Due to these advantages, Biolak and Biofilm, activated sludge process

and, MBR accounted for 2.50%, 3.07%, 1.14% and 1.00% of WWTPs in China. The WWTPs in China used AO + AAO, OD, and SBR as main wastewater treatment technologies which accounted for 80.3% of WWTPs. The number of small and medium-sized plants accounted for 97% of WWTPs. The unit energy consumption (kWh/kg $COD_{removed}$) was used to rank the three main technologies. The energy efficiency of WWTPs increases with increasing design capacity, operation loading rates, COD removal efficiency, and influent COD concentration. The average unit energy consumption of SBR decreased from 2.76 kWh/kg to 0.83 kWh/kg when design capacity increased from less than 10,000 m^3/d to 100,000-20,0000 m^3/d. SBR technology is more energy efficient in treating wastewater than other biological methods in small and medium scale WWTPs in China. The unit energy consumption of SBR decreases from 1.71 kWh/kg $COD_{removed}$ to 1.32 kWh/kg $COD_{removed}$ and 2.27 kWh/kg $COD_{removed}$ to 1.30 kWh/kg $COD_{removed}$ as the operation loading rates increase from 40% to 100% and COD removal from 70% to over 90%, respectively. Therefore, the energy demand of the SBR system is the most significant by design capacity while operation loading rate and COD removal contributes to the decrease to a less degree.

The last section compared unit energy and cost of systems with innovative technologies for treating leachate and industrial wastewater. Average unit energy consumptions are 0.99, 1.00, 1.24, 1.53, 1.58 and 1.65 kWh/kg $N_{removed}$ for textile, tannery, food processing wastewater, young, medium and old leachate, respectively. Since N removal of textile wastewater depends more on sedimentation and adsorption than other wastewater, textile wastewater needs a smaller amount of O_2 to remove N as compared to other wastewaters. Average unit energy productions are 1.84, 1.87, 1.87, 1.88, 2.09, and 3.90 kWh/kg$BOD_{removed}$ for textile, tannery, food processing

wastewater, young, medium and old leachate, respectively. Although old leachate contains has a high concentration of NH_4^+_N and soluble biodegradable organic nitrogen and a low concentration of COD and BOD/COD ratio, a great amount of external carbon source in PN/A, nitrification and denitrification process increases electricity production through increasing sludge side stream AD. Energy production of old leachate treatment system depends more on the external carbon source rather than BOD in wastewater as compared to other wastewater treatment systems. Average unit costs are 0.41, 0.61, 0.64, 0.74, 1.32, and 1.51 for young leachate, tannery wastewater, food processing wastewater, medium leachate, textile wastewater, old leachate, respectively. Compared to other wastewater, young leachate has a higher biodegradability and concentration of BOD, which could produce a large amount of CH_4 for energy recovery to reduce the unit cost of treatment system. On the other hand, unit costs of the mainstream unit process decrease with increasing COD removal. Young leachate has relatively higher COD and BOD concentration than other wastewaters, this leads to lower unit costs of the unit process. Therefore, systems with innovative technologies are more suitable for treating wastewater with a high concentration of COD and BOD/COD ratio.

REFERENCES

[1] Tarallo, S., 2014. Utilities of the Future Energy Findings. IWA Publishing. https://doi.org/10.2166/9781780406800

[2] Tarallo, S., Shaw, A., Kohl, P.,Eschborn, R., 2015. A Guide to Net-Zero Energy Solutions for Water Resource Recovery Facilities. IWA Publishing. https://doi.org/10.2166/9781780407685

[3] Wang, C., 2013. Analysis of Energy Consumption Distribution and Opportunity of Saving Energy of Sewage Treatment Plant. Civil Engineering Technology. 031, 148-151.

[4] NBS, 2013. China statistical yearbook 2013. http://www.stats.gov.cn/tjsj/ndsj/2013/indexeh.htm

[5] NBS, 2014. China statistical yearbook 2014. http://www.stats.gov.cn/tjsj/ndsj/2014/indexeh.htm

[6] NBS, 2015. China statistical yearbook 2015. http://www.stats.gov.cn/tjsj/ndsj/2015/indexeh.htm

[7] NBS, 2016. China statistical yearbook 2016. http://www.stats.gov.cn/tjsj/ndsj/2016/indexeh.htm

[8] NBS, 2017. China statistical yearbook 2017. http://www.stats.gov.cn/tjsj/ndsj/2017/indexeh.htm

[9] NBS, 2018. China statistical yearbook 2018. http://www.stats.gov.cn/tjsj/ndsj/2018/indexeh.htm

[10] NBS, 2019. China statistical yearbook 2019. http://www.stats.gov.cn/tjsj/ndsj/2019/indexeh.htm

[11] Curtis, T.P., 2010. Low-energy wastewater treatment: strategies and technologies. Environmental microbiology. 2. https://doi.org/10.1002/9780470495117.ch13

[12] Li, W.W., Yu, H.Q.,Rittmann, B.E., 2015. Chemistry: reuse water pollutants. Nature. 528, 29-31. https://doi.org/10.1038/528029a

[13] Jin, L., Zhang, G.,Tian, H., 2014. Current state of sewage treatment in China. Water Res. 66, 85-98. https://doi.org/10.1016/j.watres.2014.08.014

[14] Tang, W.Z.,Sillanpää, M., 2018. Sustainable Environmental Engineering. Wiley. https://books.google.com/books?id=nyNnDwAAQBAJ

[15] Rusten, B.,Ødegaard, H., 2006. Evaluation and testing of fine mesh sieve technologies for primary treatment of municipal wastewater. Water Science and Technology. 54, 31-38. https://doi.org/10.2166/wst.2006.710

[16] Franchi, A.,Santoro, D., 2015. Current status of the rotating belt filtration (RBF) technology for municipal wastewater treatment. Water Practice and Technology. 10, 319-327. https://doi.org/10.2166/wpt.2015.038

[17] Salsnes, 2017. Eco-Efficient Solids Separation Benchmarking water solutions. https://www.salsnes-filter.com/wp-content/uploads/sites/3/2013/11/Salsnes-detailer-13_LR.pdf

[18] Rusten, B.,Lundar, A., 2006. How a simple bench-scale test greatly improved the primary treatment performance of fine mesh sieves. Proceedings of the Water Environment Federation. 2006, 1919-1935. https://doi.org/10.2175/193864706783750114

[19] Bixio, D., Van Hauwermeiren, P.,Thoeye, C. Year. Impact of primary treatment technologies on BNR: the case study of the STP of Ghent. in The 5th IWA international symposium on systems analysis and computing in water quality management (Watermatex 2000), conference proceedings, Ghent, Belgium. 2000. Citeseer. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.500.2271&rep=rep1&type=pdf

[20] Gujer, W.,Zehnder, A.J.B., 1983. Conversion Processes in Anaerobic Digestion. Water Science and Technology. 15, 127-167. https://doi.org/10.2166/wst.1983.0164

[21] Chong, S., Sen, T.K., Kayaalp, A.,Ang, H.M., 2012. The performance enhancements of upflow anaerobic sludge blanket (UASB) reactors for domestic sludge treatment--a state-of-the-art review. Water Research. 46, 3434-3470. https://doi.org/10.1016/j.watres.2012.03.066

[22] Speece, R.E., 2008. Anaerobic Biotechnology and Odor/corrosion Control for Municipalities and Industries. R.E. Speece. https://books.google.com/books?id=onjLPgAACAAJ

[23] Gong, H., Wang, Z., Zhang, X., Jin, Z., Wang, C., Zhang, L.,Wang, K., 2017. Organics and nitrogen recovery from sewage via membrane-based pre-concentration combined with ion exchange process. Chemical Engineering Journal. 311, 13-19. https://doi.org/10.1016/j.cej.2016.11.068

[24] Abeling, U.,Seyfried, C.F., 1992. Anaerobic-aerobic treatment of high-strength ammonium wastewater-nitrogen removal via nitrite. Water Science and Technology. 26, 1007-1015. https://doi.org/10.2166/wst.1992.0542

[25] Jetten, M.S., Niftrik, L.v., Strous, M., Kartal, B., Keltjens, J.T.,Op den Camp, H.J., 2009. Biochemistry and molecular biology of anammox bacteria. Critical

reviews in biochemistry and molecular biology. 44, 65-84. https://doi.org/10.1080/10409230902722783

[26] van Haandel, A.C.,van der Lubbe, J.G.M., 2012. Handbook of Biological Wastewater Treatment. IWA Publishing. https://books.google.com/books?id=rGX9sgEACAAJ

[27] Metcalf,Eddy, 2014. Wastewater Engineering: Treatment and Resource Recovery. McGraw-Hill Education. https://books.google.com/books?id=6KVKMAEACAAJ

[28] Qasim, S.R.,Zhu, G., 2018. Wastewater Treatment and Reuse: Theory and Design Examples. Taylor & Francis, CRC Press. https://books.google.com/books?id=-kTnAQAACAAJ

[29] Alanezi, A., 2018. Cost and Environmental Impacts of Leachate Nitrogen/Phosphorus Management Approaches. http://purl.fcla.edu/fcla/etd/CFE0007147

[30] Behera, C.R., Daynouri-Pancino, F., Santoro, D., Gernaey, K.V.,Sin, G. Year. An Empirical Model for Carbon Recovery in a Rotating Belt Filter and Its Application in the Frame of Plantwide Evaluation. in Frontiers International Conference on Wastewater Treatment and Modelling. 2017. Springer. https://doi.org/10.1007/978-3-319-58421-8_5

[31] Bhunia, P.K.,Stenstrom, M.K., Optimal design and operation of wastewater treatment plants. 1986, Citeseer.

[32] EPA, 1999. Wastewater Technology Fact Sheet: Sequencing Batch Reactors. UEP Agency, Editor. https://www3.epa.gov/npdes/pubs/sbr_new.pdf

[33] Fitzsimons, L., Clifford, E., McNamara, G., Doherty, E., Phelan, T., Horrigan, M., Delauré, Y.,Corcoran, B., 2016. Increasing Resource Efficiency in Wastewater Treatment Plants. Environmental Protection Agency, Johnstown Castle, Ireland. https://www.researchgate.net/publication/303974985_Increasing_Resource_Efficiency_in_Wastewater_Treatment_Plants

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

[35] Libhaber, M.,Jaramillo, A.O., 2012. Sustainable Treatment and Reuse of Municipal Wastewater. IWA Publishing. https://books.google.com/books?id=k5LcxcGvbHcC

[36] Sharma, J.R., 2010. Development of a preliminary cost estimation method for water treatment plants. https://rc.library.uta.edu/uta-ir/handle/10106/4924

[37] United Nations, 2003. Waste-water Treatment Technologies: A General Review. UN. https://books.google.com/books?id=EJ3eAAAAMAAJ

[38] 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. https://doi.org/10.2166/wst.2018.035

[39] Verstraete, W.,Vlaeminck, S.E., 2011. ZeroWasteWater: short-cycling of wastewater resources for sustainable cities of the future. International Journal of Sustainable Development & World Ecology. 18, 253-264. https://doi.org/10.1080/13504509.2011.570804

[40] Khiewwijit, R., Temmink, H., Rijnaarts, H.,Keesman, K.J., 2015. Energy and nutrient recovery for municipal wastewater treatment: How to design a feasible plant layout? Environmental Modelling and Software. 68, 156-165. https://doi.org/10.1016/j.envsoft.2015.02.011

[41] Qiao, J.,Zhou, H., 2018. Modeling of energy consumption and effluent quality using density peaks-based adaptive fuzzy neural network. IEEE/CAA Journal of Automatica Sinica. 5, 968-976. https://doi.org/10.1109/JAS.2018.7511168

[42] Cavalcanti, P.F.F., 2003. Integrated application of the UASB reactor and ponds for domestic sewage treatment in tropical regions. https://library.wur.nl/WebQuery/wurpubs/320369

[43] Brooks, D., Roll, R.R.,Naylor, W., 2000. Wastewater technology fact sheet granular activated carbon adsorption and regeneration. Environmental Protection Agency, USA. 832.

[44] Dang, Y., Lei, Y., Liu, Z., Xue, Y., Sun, D., Wang, L.Y.,Holmes, D.E., 2016. Impact of fulvic acids on bio-methanogenic treatment of municipal solid waste incineration leachate. Water Research. 106, 71-78. https://doi.org/10.1016/j.watres.2016.09.044

[45] Renou, S., Givaudan, J.G., Poulain, S., Dirassouyan, F.,Moulin, P., 2008. Landfill leachate treatment: Review and opportunity. Journal of Hazardous Materials. 150, 468-493. https://doi.org/10.1016/j.jhazmat.2007.09.077

[46] Amor, C., Torres-Socias, E.D., Peres, J.A., Maldonado, M.I., Oller, I., Malato, S.,Lucas, M.S., 2015. Mature landfill leachate treatment by coagulation/flocculation combined with Fenton and solar photo-Fenton processes. Journal of Hazardous Materials. 286, 261-268. https://doi.org/10.1016/j.jhazmat.2014.12.036

[47] Behrend, P.,Krishnamoorthy, B., 2017. Considerations for waste gasification as an alternative to landfilling in Washington state using decision analysis and optimization. Sustainable Production and Consumption. 12, 170-179. https://doi.org/10.1016/j.spc.2017.07.004

[48] Xiaoli, C., Yongxia, H., Guixiang, L., Xin, Z.,Youcai, Z., 2013. Spectroscopic studies of the effect of aerobic conditions on the chemical characteristics of humic acid in landfill leachate and its implication for the environment. Chemosphere. 91, 1058-1063. https://doi.org/10.1016/j.chemosphere.2013.01.052

[49] Bernard, C., Colin, J.R.,Anne, L.D., 1997. Estimation of the hazard of landfills through toxicity testing of leachates: 2. Comparison of physico-chemical characteristics of landfill leachates with their toxicity determined with a battery of tests. Chemosphere. 35, 2783-2796. https://doi.org/10.1016/S0045-6535(97)00332-9

[50] Horan, N.J., Gohar, H.,Hill, B., 1997. Application of a granular activated carbon-biological fluidised bed for the treatment of landfill leachates containing high concentrations of ammonia. Water Science and Technology. 36, 369-375. https://doi.org/10.1016/S0273-1223(97)00410-1

[51] Aziz, S.Q., Aziz, H.A., Yusoff, M.S., Bashir, M.J.,Umar, M., 2010. Leachate characterization in semi-aerobic and anaerobic sanitary landfills: a comparative study. Journal of Environmental Management. 91, 2608-2614. https://doi.org/10.1016/j.jenvman.2010.07.042

[52] Ganigue, R., Lopez, H., Balaguer, M.D.,Colprim, J., 2007. Partial ammonium oxidation to nitrite of high ammonium content urban landfill leachates. Water Research. 41, 3317-3326. https://doi.org/10.1016/j.watres.2007.04.027

[53] Zolfaghari, M., Jardak, K., Drogui, P., Brar, S.K., Buelna, G.,Dube, R., 2016. Landfill leachate treatment by sequential membrane bioreactor and electro-oxidation processes. Journal of Environmental Management. 184, 318-326. https://doi.org/10.1016/j.jenvman.2016.10.010

[54] Sri Shalini, S.,Joseph, K., 2012. Nitrogen management in landfill leachate: application of SHARON, ANAMMOX and combined SHARON-ANAMMOX process. Waste Management. 32, 2385-2400. https://doi.org/10.1016/j.wasman.2012.06.006

[55] Pasalari, H., Farzadkia, M., Gholami, M.,Emamjomeh, M.M., 2019. Management of landfill leachate in Iran: valorization, characteristics, and environmental approaches. Environmental Chemistry Letters. 17, 335-348. https://doi.org/10.1007/s10311-018-0804-x

[56] Hasar, H., Unsal, S.A., Ipek, U., Karatas, S., Cinar, O., Yaman, C.,Kinaci, C., 2009. Stripping/flocculation/membrane bioreactor/reverse osmosis treatment of municipal landfill leachate. Journal of Hazardous Materials. 171, 309-317. https://doi.org/10.1016/j.jhazmat.2009.06.003

[57] Wiszniowski, J., Robert, D., Surmacz-Gorska, J., Miksch, K.,Weber, J.V., 2006. Landfill leachate treatment methods: A review. Environmental Chemistry Letters. 4, 51-61. https://doi.org/10.1007/s10311-005-0016-z

[58] Nanny, M.A.,Ratasuk, N., 2002. Characterization and comparison of hydrophobic neutral and hydrophobic acid dissolved organic carbon isolated from three municipal landfill leachates. Water Research. 36, 1572-1584. https://doi.org/10.1016/S0043-1354(01)00359-1

[59] Ahmed, F.N.,Lan, C.Q., 2012. Treatment of landfill leachate using membrane bioreactors: A review. Desalination. 287, 41-54. https://doi.org/10.1016/j.desal.2011.12.012

[60] Kjeldsen, P., Barlaz, M.A., Rooker, A.P., Baun, A., Ledin, A.,Christensen, T.H., 2002. Present and Long-Term Composition of MSW Landfill Leachate: A Review. Critical Reviews in Environmental Science and Technology. 32, 297-336. https://doi.org/10.1080/10643380290813462

[61] EPA, 2000. Development document for final effluent limitations guidelines and standards for the landfills point source category. https://www.epa.gov/sites/production/files/2015-11/documents/landfills-eg_dd_2000.pdf

[62] Kurniawan, T.A., Lo, W.H.,Chan, G.Y., 2006. Physico-chemical treatments for removal of recalcitrant contaminants from landfill leachate. Journal of Hazardous Materials. 129, 80-100. https://doi.org/10.1016/j.jhazmat.2005.08.010

[63] Alvarez-Vazquez, H., Jefferson, B.,Judd, S.J., 2004. Membrane bioreactors vs conventional biological treatment of landfill leachate: a brief review. Journal of Chemical Technology and Biotechnology. 79, 1043-1049. https://doi.org/10.1002/jctb.1072

[64] Foo, K.Y.,Hameed, B.H., 2009. An overview of landfill leachate treatment via activated carbon adsorption process. Journal of Hazardous Materials. 171, 54-60. https://doi.org/10.1016/j.jhazmat.2009.06.038

[65] Calace, N., Liberatori, A., Petronio, B.M.,Pietroletti, M., 2001. Characteristics of different molecular weight fractions of organic matter in landfill leachate and their role in soil sorption of heavy metals. Environmental Pollution. 113, 331-339. https://doi.org/10.1016/S0269-7491(00)00186-X

[66] Harmsen, J., 1983. Identification of organic compounds in leachate from a waste tip. Water Research. 17, 699-705. https://doi.org/10.1016/0043-1354(83)90239-7

[67] Wang, K., Li, L., Tan, F.,Wu, D., 2018. Treatment of Landfill Leachate Using Activated Sludge Technology: A Review. Archaea. 2018, 1039453. https://doi.org/10.1155/2018/1039453

[68] Rivas, F.J., Beltran, F., Carvalho, F., Acedo, B.,Gimeno, O., 2004. Stabilized leachates: sequential coagulation-flocculation + chemical oxidation process. Journal of Hazardous Materials. 116, 95-102. https://doi.org/10.1016/j.jhazmat.2004.07.022

[69] Fernandes, A., Pacheco, M.J., Ciríaco, L.,Lopes, A., 2015. Review on the electrochemical processes for the treatment of sanitary landfill leachates: Present and future. Applied Catalysis B: Environmental. 176-177, 183-200. https://doi.org/10.1016/j.apcatb.2015.03.052

[70] Robinson, H., 2007. The composition of leachates from very large landfills: an international review. Communications in waste and resource management. 8, 9-32. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.458.4471&rep=rep1&type=pdf

[71] Cataldo, F.,Angelini, G., 2012. Radiolysis and ozonolysis of a landfill leachate. Journal of Radioanalytical and Nuclear Chemistry. 293, 141-148. https://doi.org/10.1007/s10967-012-1729-7

[72] Gabarro, J., Ganigue, R., Gich, F., Ruscalleda, M., Balaguer, M.D.,Colprim, J., 2012. Effect of temperature on AOB activity of a partial nitritation SBR treating landfill leachate with extremely high nitrogen concentration. Bioresource Technology. 126, 283-289. https://doi.org/10.1016/j.biortech.2012.09.011

[73] Chian, E.S.K.,Dewalle, F.B., 1976. Sanitary landfill leachates and their treatment. Journal of the Environmental Engineering Division. 102, 411-431. https://cedb.asce.org/CEDBsearch/record.jsp?dockey=0006590

[74] Bohdziewicz, J., Neczaj, E.,Kwarciak, A., 2008. Landfill leachate treatment by means of anaerobic membrane bioreactor. Desalination. 221, 559-565. https://doi.org/10.1016/j.desal.2007.01.117

[75] Peng, Y., Zhang, S., Zeng, W., Zheng, S., Mino, T.,Satoh, H., 2008. Organic removal by denitritation and methanogenesis and nitrogen removal by nitritation from landfill leachate. Water Research. 42, 883-892. https://doi.org/10.1016/j.watres.2007.08.041

[76] Hoilijoki, T.H., Kettunen, R.H.,Rintala, J.A., 2000. Nitrification of anaerobically pretreated municipal landfill leachate at low temperature. Water Research. 34, 1435-1446. https://doi.org/10.1016/S0043-1354(99)00278-X

[77] Neczaj, E., Okoniewska, E.,Kacprzak, M., 2005. Treatment of landfill leachate by sequencing batch reactor. Desalination. 185, 357-362. https://doi.org/10.1016/j.desal.2005.04.044

[78] Yusof, N., Hassan, M.A., Phang, L.Y., Tabatabaei, M., Othman, M.R., Mori, M., Wakisaka, M., Sakai, K.,Shirai, Y., 2010. Nitrification of ammonium-rich sanitary landfill leachate. Waste Management. 30, 100-109. https://doi.org/10.1016/j.wasman.2009.08.018

[79] Visvanathan, C., Choudhary, M.K., Montalbo, M.T.,Jegatheesan, V., 2007. Landfill leachate treatment using thermophilic membrane bioreactor. Desalination. 204, 8-16. https://doi.org/10.1016/j.desal.2006.02.028

[80] Laitinen, N., Luonsi, A.,Vilen, J., 2006. Landfill leachate treatment with sequencing batch reactor and membrane bioreactor. Desalination. 191, 86-91. https://doi.org/10.1016/j.desal.2005.08.012

[81] van de Graaf, A.A., de Bruijn, P., Robertson, L.A., Jetten, M.S.M.,Kuenen, J.G., 1996. Autotrophic growth of anaerobic ammonium-oxidizing micro-organisms in a fluidized bed reactor. Microbiology. 142, 2187-2196. https://doi.org/10.1099/13500872-142-8-2187

[82] Strous, M., Kuenen, J.G.,Jetten, M.S.M., 1999. Key Physiology of Anaerobic Ammonium Oxidation. Applied and Environmental Microbiology. 65, 3248-3250. https://doi.org/10.1128/AEM.65.7.3248-3250.1999

[83] Giusti, E., Marsili-Libelli, S.,Spagni, A., 2011. Modelling microbial population dynamics in nitritation processes. Environmental Modelling and Software. 26, 938-949. https://doi.org/10.1016/j.envsoft.2011.02.001

[84] O'Shaughnessy, M., 2016. Mainstream Deammonification. IWA Publishing. https://doi.org/10.2166/9781780407852

[85] Rosenwinkel, K.H.,Cornelius, A., 2005. Deammonification in the Moving-Bed Process for the Treatment of Wastewater with High Ammonia Content. Chemical Engineering and Technology. 28, 49-52. https://doi.org/10.1002/ceat.200407070

[86] Cui, F., 2012. Cold CANON: Anammox at low temperatures. The Netherlands: Delft University of Technology. http://resolver.tudelft.nl/uuid:2bc15934-7e46-4d5e-98a6-d8f7cfdf8d9e

[87] Bi, Z., Qiao, S., Zhou, J., Tang, X.,Zhang, J., 2014. Fast start-up of Anammox process with appropriate ferrous iron concentration. Bioresource Technology. 170, 506-512. https://doi.org/10.1016/j.biortech.2014.07.106

[88] Christensson, M., Ekström, S., Chan, A.A., Le Vaillant, E.,Lemaire, R., 2013. Experience from start-ups of the first ANITA Mox plants. Water Science and Technology. 67, 2677-2684. https://doi.org/10.2166/wst.2013.156

[89] Jenkins, D.,Wanner, J., 2014. Activated Sludge – 100 Years and Counting. IWA Publishing. https://doi.org/10.2166/9781780404943

[90] Garrido, J.M., Fdz-Polanco, M.,Fdz-Polanco, F., 2013. Working with energy and mass balances: a conceptual framework to understand the limits of municipal wastewater treatment. Water Science and Technology. 67, 2294-2301. https://doi.org/10.2166/wst.2013.124

[91] McCarty, P.L., 2018. What is the Best Biological Process for Nitrogen Removal: When and Why? Environmental Science and Technology. 52, 3835-3841. https://doi.org/10.1021/acs.est.7b05832

[92] Norashiddin, F.A., Kamaruddin, M.A., Emmanuel, I.M.,Pakir, M.F., 2019. Activated carbon for landfill leachate treatment: A review. International Journal of Development and Sustainability. 8, 19-29. https://isdsnet.com/ijds-v8n1-02.pdf

[93] Department of Environment, 2009. Environmental Quality (Control of Pollution from Solid Waste Transfer Station and Landfill. http://www.fao.org/faolex/results/details/en/c/LEX-FAOC099122/

[94] Ye, Z.L., Xie, X., Dai, L., Wang, Z., Wu, W., Zhao, F., Xie, X., Huang, S., Liu, M.,Chen, S., 2014. Full-scale blending treatment of fresh MSWI leachate with municipal wastewater in a wastewater treatment plant. Waste Management. 34, 2305-2311. https://doi.org/10.1016/j.wasman.2014.06.019

[95] Kabdasli, I., Safak, A.,Tunay, O., 2008. Bench-scale evaluation of treatment schemes incorporating struvite precipitation for young landfill leachate. Waste Management. 28, 2386-2392. https://doi.org/10.1016/j.wasman.2007.10.020

[96] Im, J.H., Woo, H.J., Choi, M.W., Han, K.B.,Kim, C.W., 2001. Simultaneous organic and nitrogen removal from municipal landfill leachate using an anaerobic-aerobic system. Water Research. 35, 2403-2410. https://doi.org/10.1016/S0043-1354(00)00519-4

[97] Çeçen, F.,O., A., 2004. Aerobic Co-Treatment of Landfill Leachate with Domestic Wastewater. Environmental Engineering Science. 21, 303-312. https://doi.org/10.1089/109287504323066941

[98] Calli, B., Durmaz, S.,Mertoglu, B., 2006. Identification of prevalent microbial communities in a municipal solid waste landfill. Water Science and Technology. 53, 139-147. https://doi.org/10.2166/wst.2006.244

[99] Gómez, M., Corona, F.,Hidalgo, M.D., 2019. Variations in the properties of leachate according to landfill age. Desalination and Water Treatment. 159, 24-31. https://doi.org/10.5004/dwt.2019.24106

[100] Kang, K.H., Shin, H.S.,Park, H., 2002. Characterization of humic substances present in landfill leachates with different landfill ages and its implications. Water Research. 36, 4023-4032. https://doi.org/10.1016/S0043-1354(02)00114-8

[101] Yong, Z.J., Bashir, M.J.K., Ng, C.A., Sethupathi, S.,Lim, J.W., 2018. A sequential treatment of intermediate tropical landfill leachate using a sequencing batch reactor (SBR) and coagulation. Journal of Environmental Management. 205, 244-252. https://doi.org/10.1016/j.jenvman.2017.09.068

[102] Carabalí-Rivera, Y.S., Barba-Ho, L.E.,Torres-Lozada, P., 2017. Determination of leachate toxicity through acute toxicity using Daphnia pulex and anaerobic toxicity assays. Ingeniería e Investigación. 37, 16-24. http://www.scielo.org.co/scielo.php?script=sci_arttext&pid=S0120-56092017000100003&nrm=iso

[103] Melidis, P., 2014. Landfill Leachate Nutrient Removal Using Intermittent Aeration. Environmental Processes. 1, 221-230. https://doi.org/10.1007/s40710-014-0022-x
[104] Rada, E., Istrate, I., Ragazzi, M., Andreottola, G.,Torretta, V., 2013. Analysis of Electro-Oxidation Suitability for Landfill Leachate Treatment through an Experimental Study. Sustainability. 5, 3960-3975. https://doi.org/10.3390/su5093960

[105] Oulego, P., Collado, S., Laca, A.,Diaz, M., 2016. Impact of leachate composition on the advanced oxidation treatment. Water Research. 88, 389-402. https://doi.org/10.1016/j.watres.2015.09.048

[106] Wukasch, R.F., 1993. Proceedings of the 48th Industrial Waste Conference Purdue University, May 1993. Taylor & Francis. https://books.google.com/books?id=lyM6SgHXimEC

[107] El Fadel, H., Merzouki, M., Faouzi, M., Laamayem, A., Najem, M.,Benlemlih, M., 2013. PURIFICATION PERFORMANCE OF FILTRATION PROCESS FOR LEACHATE IN MOROCCO BY MARINE SANDS, CLAYS AND FLY ASH. Journal of Biotechnology Letters. 4, 51. http://bioinfopublication.org/viewhtml.php?artid=BIA0001693

[108] Tusseau-Vuillemin, M.H., Dispan, J., Mouchel, J.M.,Servais, P., 2003. Biodegradable fraction of organic carbon estimated under oxic and anoxic conditions. Water Research. 37, 2242-2247. https://doi.org/10.1016/S0043-1354(02)00582-1

[109] Amin, M.M., Hashemi, H., Bina, B., Ebrahimi, A., Pourzamani, H.R.,Ebrahimi, A., 2014. Environmental pollutants removal from composting leachate using anaerobic biological treatment process. International Journal of Health System and Disaster Management. 2, 216-219. https://doi.org/10.4103/2347-9019.142191

[110] de Morais, J.L.,Zamora, P.P., 2005. Use of advanced oxidation processes to improve the biodegradability of mature landfill leachates. Journal of Hazardous Materials. 123, 181-186. https://doi.org/10.1016/j.jhazmat.2005.03.041

[111] Batarseh, E.S., Reinhart, D.R.,Daly, L., 2007. Liquid Sodium Ferrate and Fenton's Reagent for Treatment of Mature Landfill Leachate. Journal of Environmental Engineering. 133, 1042-1050. https://doi.org/10.1061/(ASCE)0733-9372(2007)133:11(1042)

[112] Naumczyk, J., Prokurat, I.,Marcinowski, P., 2012. Landfill Leachates Treatment by H2O2/UV, O3/H2O2, Modified Fenton, and Modified Photo-Fenton Methods. International Journal of Photoenergy. 2012, 1-9. https://doi.org/10.1155/2012/909157

[113] Pakenas, L.J., 1995. Energy efficiency in municipal wastewater treatment plants: Technology assessment. New York State Energy Research and Development Authority. https://www.nyserda.ny.gov/-/media/Files/Publications/Research/Environmental/Energy-Efficiency-Municipal-WWTP.pdf

[114] Wett, B., 2007. Development and implementation of a robust deammonification process. Water Science and Technology. 56, 81-88. https://doi.org/10.2166/wst.2007.611

[115] Rodriguez, J., Castrillon, L., Maranon, E., Sastre, H.,Fernandez, E., 2004. Removal of non-biodegradable organic matter from landfill leachates by adsorption. Water Research. 38, 3297-3303. https://doi.org/10.1016/j.watres.2004.04.032

[116] Burian, S.J., Nix, S.J., Pitt, R.E.,Durrans, S.R., 2000. Urban wastewater management in the United States: Past, present, and future. Journal of Urban Technology. 7, 33-62. https://doi.org/10.1080/713684134

[117] Bustillo-Lecompte, C., Mehrvar, M.,Quiñones-Bolaños, E., 2016. Slaughterhouse wastewater characterization and treatment: an economic and public health necessity of the meat processing industry in Ontario, Canada. Journal of Geoscience and Environment Protection. 4, 175-186. https://doi.org/10.4236/gep.2016.44021

[118] Cao, W.,Mehrvar, M., 2011. Slaughterhouse wastewater treatment by combined anaerobic baffled reactor and UV/H2O2 processes. Chemical Engineering Research and Design. 89, 1136-1143. https://doi.org/10.1016/j.cherd.2010.12.001

[119] Khamtib, S.,Reungsang, A., 2014. Co-digestion of oil palm trunk hydrolysate with slaughterhouse wastewater for thermophilic bio-hydrogen production by Thermoanaerobacterium thermosaccharolyticm KKU19. International Journal of Hydrogen Energy. 39, 6872-6880. https://doi.org/10.1016/j.ijhydene.2014.02.073

[120] EPA, 2004. Effluent limitations guidelines and new source performance standards for the meat and poultry products, point source category. Environmental Protection Agency (EPA): Federal Register. 69. https://www.federalregister.gov/documents/2004/09/08/04-12017/effluent-limitations-guidelines-and-new-source-performance-standards-for-the-meat-and-poultry

[121] Kundu, P., Debsarkar, A.,Mukherjee, S., 2013. Treatment of slaughter house wastewater in a sequencing batch reactor: performance evaluation and biodegradation kinetics. BioMed research international. 2013. https://doi.org/10.1155/2013/134872

[122] Christensson, M., Ekstrom, S., Andersson Chan, A., Le Vaillant, E.,Lemaire, R., 2013. Experience from start-ups of the first ANITA Mox plants. Water Science and Technology. 67, 2677-2684. https://doi.org/10.2166/wst.2013.156

[123] Gupta, M., 2018. Microsieving as a primary treatment for biological nitrogen removal from municipal wastewater. https://ir.lib.uwo.ca/etd/5288/

[124] EPA, 1975. Process design manual for suspended solids removal. US Environmental Protection Agency. http://nepis.epa.gov/Exe/ZyPDF.cgi/9101AVYN.PDF?Dockey=9101AVYN.PDF

[125] Ljunggren, M., 2006. Micro screening in wastewater treatment-an overview. Vatten. 62, 171. https://www.tidskriftenvatten.se/wp-content/uploads/2017/04/48_article_2139.pdf

[126] Sarathy, S., Ho, D., Murray, A., Batstone, D.,Santoro, D., 2015. Engineered fractionation of primary solids–A comparison of primary treatments using rotating belt filters and primary clarifiers. Proceedings of the Water Environment Federation. 2015, 4950-4959.

[127] Schmidt Jr, H.E.,Dhulashia, S., Carbon Diversion and its Role in Energy Efficiency. https://fwrj.com/techarticles/0320%20t3.pdf

[128] Del Pozo, R., Diez, V.,Beltran, S., 2000. Anaerobic pre-treatment of slaughterhouse wastewater using fixed-film reactors. Bioresource Technology. 71, 143-149. https://doi.org/10.1016/S0960-8524(99)90065-2

[129] Latiffi, N.A.A., Mohamed, R.M.S.R., Apandi, N.M.,Tajuddin, R.M. Year. Preliminary assessment of growth rates on different concentration of microalgae scenedesmus sp. in industrial meat food processing wastewater. in MATEC Web of Conferences. 2017. EDP Sciences. https://doi.org/10.1051/matecconf/201710306010

[130] Vu, T., Bach, Q.D., Pham, H.B., Do, T.A.,Do, Q., 2017. The Performance of a Gaslift MBR for Slaughterhouse Wastewater Treatment in 1 m3/day Scale. Modern Environmental Science and Engineering. 03, 349-354. https://pdfs.semanticscholar.org/2263/f924843e6487cbcafc6c8e9b4508ca5f2e96.pdf?_ga=2.158854392.352607616.1602798023-98483532.1585087819

[131] Rajakumar, R.,Meenambal, T., 2008. Comparative Study on Start–Up Performance of HUASB and AF Reactors Treating Poultry Slaughterhouse Wastewater. International Journal of Environmental Research. 2, 401-410. https://ijer.ut.ac.ir/article_221_026486e1ef33561c63ec2f49bd6981b4.pdf

[132] Yaakob, M.A., Mohamed, R.M.S.R., Al-Gheethi, A.A.S.,Kassim, A.H.M., 2018. Characteristics of chicken slaughterhouse wastewater. Chemical Engineering Transactions. 63, 637-642. https://doi.org/10.3303/CET1863107

[133] Pabi, S., Amarnath, A., Goldstein, R.,Reekie, L., 2013. Electricity use and management in the municipal water supply and wastewater industries. Electric Power Research Institute, Palo Alto. 194. https://www.sciencetheearth.com/uploads/2/4/6/5/24658156/electricity_use_and_management_in_the_municipal_water_supply_and_wastewater_industries.pdf

[134] Davis, M.L., 2010. Water and wastewater engineering: design principles and practice. McGraw-Hill Education.

[135] Hydromantis, E., Inc. GPS-X Technical Reference. 2017, Hydromantis ESS, Inc.: Hamilton, ON, Canada.

[136] Cheng, Z., Wang, X.,Yang, J. Year. Experimental study on recycled steel fiber concrete. in 2011 International Conference on Electric Technology and Civil Engineering (ICETCE). 2011. IEEE. https://doi.org/10.1109/ICETCE.2011.5774777

[137] Lema, J.M.,Martinez, S.S., 2017. Innovative Wastewater Treatment & Resource Recovery Technologies: Impacts on Energy, Economy and Environment. IWA Publishing. https://books.google.com/books?id=aNYoDwAAQBAJ

[138] CEC, 1991. Council Directive of 21. May 1991 concerning urban waste water treatment (91/271/EEC). J. Eur. Commun. 34, 40. http://translation-centre.am/pdf/Translat/EU_Direct/environment/Dir_91_271_en.pdf

[139] CMEP, 1992. Discharge standard of water pollutants for meat packing industry GB 13457-92.

[140] CPCB, 1986. General standards for discharge of environmental pollutants part-a: Effluents. https://cpcb.nic.in/GeneralStandards.pdf

[141] Caixeta, C.E.T., Cammarota, M.C.,Xavier, A.M.F., 2002. Slaughterhouse wastewater treatment: evaluation of a new three-phase separation system in a UASB reactor. Bioresource Technology. 81, 61-69. https://doi.org/10.1016/S0960-8524(01)00070-0

[142] Torkian, A., Eqbali, A.,Hashemian, S.J., 2003. The effect of organic loading rate on the performance of UASB reactor treating slaughterhouse effluent. Resources, Conservation and Recycling. 40, 1-11. https://doi.org/10.1016/S0921-3449(03)00021-1

[143] Musa, M.A., Idrus, S., Harun, M.R., Tuan Mohd Marzuki, T.F.,Abdul Wahab, A.M., 2020. A comparative study of biogas production from cattle slaughterhouse wastewater using conventional and modified Upflow Anaerobic Sludge Blanket (UASB) reactors. International journal of environmental research and public health. 17, 283. https://doi.org/10.3390/ijerph17010283

[144] Federation, W.E., 2017. Design of Water Resource Recovery Facilities, Manual of Practice No.8, Sixth Edition. McGraw-Hill Education. https://books.google.com/books?id=8Mw2DwAAQBAJ

[145] Yang, L.B., Zeng, S.Y., Chen, J.N., He, M.,Yang, W., 2010. Operational energy performance assessment system of municipal wastewater treatment plants. Water Sci Technol. 62, 1361-1370. https://doi.org/10.2166/wst.2010.394

[146] Lackner, S., Gilbert, E.M., Vlaeminck, S.E., Joss, A., Horn, H.,van Loosdrecht, M.C., 2014. Full-scale partial nitritation/anammox experiences–an application survey. Water Research. 55, 292-303. https://doi.org/10.1016/j.watres.2014.02.032

[147] 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 Science and Technology. 77, 2242-2252. https://doi.org/10.2166/wst.2018.035

[148] Tchobanoglus, G., Burton, F.,Stensel, H.D., 2003. Wastewater engineering: Treatment and reuse. American Water Works Association. Journal. 95, 201.

[149] Paraskeva, P.,Diamadopoulos, E., 2006. Technologies for olive mill wastewater (OMW) treatment: a review. Journal of Chemical Technology & Biotechnology: International Research in Process, Environmental & Clean Technology. 81, 1475-1485. https://doi.org/10.1002/jctb.1553

[150] Boopathy, R., Larsen, V.F.,Senior, E., 1988. Performance of anaerobic baffled reactor (ABR) in treating distillery waste water from a scotch whisky factory. Biomass. 16, 133-143. https://doi.org/10.1016/0144-4565(88)90022-4

[151] Gerbens-Leenes, P.W., Mekonnen, M.M.,Hoekstra, A.Y., 2013. The water footprint of poultry, pork and beef: A comparative study in different countries and production systems. Water Resources and Industry. 1, 25-36. https://doi.org/10.1016/j.wri.2013.03.001

[152] Mekonnen, M.M.,Hoekstra, A.Y., 2012. A global assessment of the water footprint of farm animal products. Ecosystems. 15, 401-415. https://doi.org/10.1007/s10021-011-9517-8

[153] Bustillo-Lecompte, C.F.,Mehrvar, M., 2015. Slaughterhouse wastewater characteristics, treatment, and management in the meat processing industry: A review on trends and advances. Journal of Environmental Management. 161, 287-302. https://doi.org/10.1016/j.jenvman.2015.07.008

[154] Durai, G.,Rajasimman, M., 2011. Biological Treatment of Tannery Wastewater-A Review. Journal of Environmental science and Technology. 4, 1-17. https://doi.org/10.3923/jest.2011.1.17

[155] Famielec, S.,Wieczorek-Ciurowa, K., 2011. Waste from leather industry. Threats to the environment. Czasopismo Techniczne. Chemia. 108, 43-48. https://suw.biblos.pk.edu.pl/resources/i4/i5/i4/i6/r4546/FamielecS_WasteLeather.pdf

[156] United Nations Industrial Development Organization, 1991. Tanneries and the Environment: A Technical Guide to Reducing the Environmental Impact of Tannery Operations. United Nations Environment Programme, Industry and Environment Office. https://books.google.com/books?id=FHDbAAAAMAAJ

[157] European Commission, 2001. Integrated Pollution Prevention and Control (IPPC) Reference Document on Best Available Techniques for the Tanning of Hides and Skins May 2001. https://p2infohouse.org/ref/21/20575.pdf

[158] Lofrano, G., Meriç, S., Zengin, G.E.,Orhon, D., 2013. Chemical and biological treatment technologies for leather tannery chemicals and wastewaters: a review.

Science of the Total Environment. 461, 265-281. https://doi.org/10.1016/j.scitotenv.2013.05.004

[159] Mwinyihija, M., 2010. Ecotoxicological Diagnosis in the Tanning Industry. Springer New York. https://books.google.com/books?id=a53Z9sUKAdUC

[160] Abdulla, H.M., Kamal, E.M., Mohamed, A.H.,El-Bassuony, A.D. Year. Chromium removal from tannery wastewater using chemical and biological techniques aiming zero discharge of pollution. in Proceeding Of Fifth Scientific Environmental Conference. 2010.

[161] de Aquim, P.M., Hansen, É.,Gutterres, M., 2019. Water reuse: An alternative to minimize the environmental impact on the leather industry. Journal of Environmental Management. 230, 456-463. https://doi.org/10.1016/j.jenvman.2018.09.077

[162] Elabbas, S., Ouazzani, N., Mandi, L., Berrekhis, F., Perdicakis, M., Pontvianne, S., Pons, M.-N., Lapicque, F.,Leclerc, J.-P., 2016. Treatment of highly concentrated tannery wastewater using electrocoagulation: influence of the quality of aluminium used for the electrode. Journal of Hazardous Materials. 319, 69-77. https://doi.org/10.1016/j.jhazmat.2015.12.067

[163] Hassen, A.S.,Woldeamanuale, T.B., 2017. Evaluation and Characterization of Tannery Wastewater in each process at batu and modjo tannery, Ethiopia. Int J Rural Dev Environ Health Res. 1, 17-26.

[164] Islam, B.I., Musa, A.E., Ibrahim, E.H., Sharafa, S.A.A.,Elfaki, B.M., 2014. Evaluation and characterization of tannery wastewater. Journal of Forest Products & Industries. 3, 141-150.

[165] Nandy, T., Kaul, S.N., Shastry, S., Manivel, U.,Deshpande, C.V., 1999. Wastewater Management in Cluster of Tanneries in Tamil Nadu through Implementation of Common Effluent Treatment Plants+. http://nopr.niscair.res.in/bitstream/123456789/17844/1/JSIR%2058%287%29%20475-516.pdf

[166] Robinson, T., McMullan, G., Marchant, R.,Nigam, P., 2001. Remediation of dyes in textile effluent: a critical review on current treatment technologies with a proposed alternative. Bioresource Technology. 77, 247-255. https://doi.org/10.1016/S0960-8524(00)00080-8

[167] Villegas-Navarro, A., Ramírez-M, Y., Salvador-SB, M.S.,Gallardo, J.M., 2001. Determination of wastewater LC50 of the different process stages of the textile industry. Ecotoxicology and Environmental Safety. 48, 56-61. https://doi.org/10.1006/eesa.2000.1986

[168] Wang, C., Yediler, A., Lienert, D., Wang, Z.,Kettrup, A., 2002. Toxicity evaluation of reactive dyestuffs, auxiliaries and selected effluents in textile finishing industry to luminescent bacteria Vibrio fischeri. Chemosphere. 46, 339-344. https://doi.org/10.1016/S0045-6535(01)00086-8

[169] EPA, 1997. EPA Office of Compliance Sector Notebook Project : profile of the textile industry. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=50000HE9.PDF

[170] Volmajer Valh, J., Majcen Le Marechal, A., Vajnhandl, S., Jeric, T.,Šimon, E., 2011. Water in the textile industry. Treatise on Water Science; Wilderer, P., Ed.; Elsevier: Oxford, UK. 685-706.

[171] Vijayaraghavan, G.,Shanthakumar, S., 2015. Removal of sulphur black dye from its aqueous solution using alginate from Sargassum sp.(Brown algae) as a coagulant. Environmental Progress & Sustainable Energy. 34, 1427-1434. https://doi.org/10.1002/ep.12144

[172] Noreen, M., Shahid, M., Iqbal, M.,Nisar, J., 2017. Measurement of cytotoxicity and heavy metal load in drains water receiving textile effluents and drinking water in vicinity of drains. Measurement. 109, 88-99. https://doi.org/10.1016/j.measurement.2017.05.030

[173] Ganesh, R., Boardman, G.D.,Michelsen, D., 1994. Fate of azo dyes in sludges. Water Research. 28, 1367-1376. https://doi.org/10.1016/0043-1354(94)90303-4

[174] Gottlieb, A., Shaw, C., Smith, A., Wheatley, A.,Forsythe, S., 2003. The toxicity of textile reactive azo dyes after hydrolysis and decolourisation. Journal of Biotechnology. 101, 49-56. https://doi.org/10.1016/S0168-1656(02)00302-4

[175] Meriç, S., Kaptan, D.,Ölmez, T., 2004. Color and COD removal from wastewater containing Reactive Black 5 using Fenton's oxidation process. Chemosphere. 54, 435-441. https://doi.org/10.1016/j.chemosphere.2003.08.010

[176] Ullah, T., Sultana, T., Khan, L., Feroz, K., Ahmad, Q.A.,Hussain, S., 2017. Histopathological alteration in Gill, Kidney and Liver of Cirrhinus mrigala, Catla catla, Hypophthalmichthys molitrix and Labeo rohita due to sub-lethal exposure of textile industries effluents in Faisalabad, Pakistan. Pakistan. J. Entomol. Zool. Stud. 5, e62.

[177] Najam-us-Sahar, Hussain, A., Mustafa, A., Waqas, R., Ashraf, I.,Zaman Akhtar, M.F.U., 2017. Effect of textile wastewater on growth and yield of wheat (Triticum aestivum L.). Soil & Environment. 36, 28-34. https://www.researchgate.net/publication/316169367_Effect_of_textile_wastewater_on_growth_and_yield_of_wheat_Triticum_aestivum_L

[178] Daud, M.K., Nafees, M., Ali, S., Rizwan, M., Bajwa, R.A., Shakoor, M.B., Arshad, M.U., Chatha, S.A.S., Deeba, F., Murad, W., Malook, I.,Zhu, S.J., 2017. Drinking water quality status and contamination in Pakistan. BioMed research international. 2017. https://doi.org/10.1155/2017/7908183

[179] Al-Mutairi, N.Z., Hamoda, M.F.,Al-Ghusain, I.A., 2007. Slaughterhouse wastewater treatment using date seeds as adsorbent. Journal of Environment Science and Health. 55, 678-710.

[180] Boopathy, R., Bonvillain, C., Fontenot, Q.,Kilgen, M., 2007. Biological treatment of low-salinity shrimp aquaculture wastewater using sequencing batch reactor. International Biodeterioration & Biodegradation. 59, 16-19. https://doi.org/10.1016/j.ibiod.2006.05.003

[181] Kim, H.S., Choung, Y.K., Ahn, S.,Oh, H.S., 2008. Enhancing nitrogen removal of piggery wastewater by membrane bioreactor combined with nitrification reactor. Desalination. 223, 194-204. https://doi.org/10.1016/j.desal.2006.12.021

[182] Al-Mutairi, N.Z., Al-Sharifi, F.A.,Al-Shammari, S.B., 2008. Evaluation study of a slaughterhouse wastewater treatment plant including contact-assisted activated sludge and DAF. Desalination. 225, 167-175. https://doi.org/10.1016/j.desal.2007.04.094

[183] Roy, D., Hassan, K.,Boopathy, R., 2010. Effect of carbon to nitrogen (C: N) ratio on nitrogen removal from shrimp production waste water using sequencing batch reactor. Journal of industrial microbiology & biotechnology. 37, 1105-1110. https://doi.org/10.1007/s10295-010-0869-4

[184] Rajagopal, R., Rousseau, P., Bernet, N.,Béline, F., 2011. Combined anaerobic and activated sludge anoxic/oxic treatment for piggery wastewater. Bioresource Technology. 102, 2185-2192. https://doi.org/10.1016/j.biortech.2010.09.112

[185] Palatsi, J., Viñas, M., Guivernau, M., Fernandez, B.,Flotats, X., 2011. Anaerobic digestion of slaughterhouse waste: main process limitations and microbial community interactions. Bioresource Technology. 102, 2219-2227. https://doi.org/10.1016/j.biortech.2010.09.121

[186] Kern, C.,Boopathy, R., 2012. Use of sequencing batch reactor in the treatment of shrimp aquaculture wastewater. Journal of Water Sustainability. 2, 221-232. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.453.7245&rep=rep1&type=pdf

[187] Wang, F., Liu, Y., Wang, J., Zhang, Y.,Yang, H., 2012. Influence of growth manner on nitrifying bacterial communities and nitrification kinetics in three lab-scale bioreactors. Journal of industrial microbiology & biotechnology. 39, 595-604. https://doi.org/10.1007/s10295-011-1065-x

[188] Jawahar, A.J., Chinnadurai, M., Ponselvan, J.K.,Annadurai, G., 1998. Pollution from tanneries and options for treatment of effluent. Indian Journal of Environmental Protection. 18, 672-678.

[189] Eckenfelder, W.W., 2000. Industrial Water Pollution Control. McGraw-Hill. https://books.google.com/books?id=hRtSAAAAMAAJ

[190] Tare, V., Gupta, S.,Bose, P., 2003. Case studies on biological treatment of tannery effluents in India. Journal of the air & waste management association. 53, 976-982. https://doi.org/10.1080/10473289.2003.10466250

[191] Ahn, D.H., Chung, Y.C., Yoo, Y.J., Pak, D.W.,Chang, W.S., 1996. Improved treatment of tannery wastewater using Zoogloea ramigera and its extracellular polymer in an activated sludge process. Biotechnology letters. 18, 917-922. https://doi.org/10.1007/BF00154621

[192] Vijayaraghavan, K.,Murthy, D.V.S., 1997. Effect of toxic substances in anaerobic treatment of tannery wastewaters. Bioprocess Engineering. 16, 151-155. https://doi.org/10.1007/s004490050302

[193] Wiemann, M., Schenk, H.,Hegemann, W., 1998. Anaerobic treatment of tannery wastewater with simultaneous sulphide elimination. Water Research. 32, 774-780. https://doi.org/10.1016/S0043-1354(97)00309-6

[194] Di Iaconi, C., Lopez, A., Ramadori, R.,Passino, R., 2003. Tannery wastewater treatment by sequencing batch biofilm reactor. Environmental Science & Technology. 37, 3199-3205. https://doi.org/10.1021/es030002u

[195] Farabegoli, G., Carucci, A., Majone, M.,Rolle, E., 2004. Biological treatment of tannery wastewater in the presence of chromium. Journal of Environmental Management. 71, 345-349. https://doi.org/10.1016/j.jenvman.2004.03.011

[196] Schrank, S.G., Jose, H.J., Moreira, R.F.P.M.,Schröder, H.F., 2004. Elucidation of the behavior of tannery wastewater under advanced oxidation conditions. Chemosphere. 56, 411-423. https://doi.org/10.1016/j.chemosphere.2004.04.012

[197] Sekaran, G., Chitra, K., Mariappan, M.,Raghavan, K.V., 1996. Removal of sulphide in anaerobically treated tannery wastewater by wet air oxidation. Journal of Environmental Science & Health Part A. 31, 579-598. https://doi.org/10.1080/10934529609376375

[198] Dogruel, S., Genceli, E.A., Babuna, F.G.,Orhon, D., 2004. Ozonation of nonbiodegradable organics in tannery wastewater. Journal of Environmental Science and Health, Part A. 39, 1705-1715. https://doi.org/10.1081/ESE-120037871

[199] Song, Z., Williams, C.J.,Edyvean, R.G.J., 2004. Treatment of tannery wastewater by chemical coagulation. Desalination. 164, 249-259. https://doi.org/10.1016/S0011-9164(04)00193-6

[200] Churchley, J.H., 1994. REMOVAL OF DYEW ASTE COLOUR FROM SEW AGE EFFLUENT-THE USE OF A FULL SCALE OZONE PLANT. Water Science and Technology. 30, 275. https://search.proquest.com/openview/7db123fa0f6d45f15604e6f854e6d5b6/1?pq-origsite=gscholar&cbl=2044520

[201] Stern, S.R., Szpyrkowicz, L.,Rodighiero, I., 2003. Anaerobic treatment of textile dyeing wastewater. Water Science and Technology. 47, 55-59. https://doi.org/10.2166/wst.2003.0537

[202] Chu, W., 2001. Dye removal from textile dye wastewater using recycled alum sludge. Water Research. 35, 3147-3152. https://doi.org/10.1016/S0043-1354(01)00015-X

[203] Dąbrowski, W., Żyłka, R.,Rynkiewicz, M., 2016. Evaluation of energy consumption in agro-industrial wastewater treatment plant. Journal of Ecological Engineering. 17, 73--78. https://doi.org/10.12911/22998993/63306

[204] Yang, X., López-Grimau, V., Vilaseca, M.,Crespi, M., 2020. Treatment of Textile Wastewater by CAS, MBR, and MBBR: A Comparative Study from Technical, Economic, and Environmental Perspectives. Water. 12, 1306. https://doi.org/10.3390/w12051306

[205] Ayoub, G.M., Hamzeh, A.,Semerjian, L., 2011. Post treatment of tannery wastewater using lime/bittern coagulation and activated carbon adsorption. Desalination. 273, 359-365. https://doi.org/10.1016/j.desal.2011.01.045

[206] Doble, M.,Kumar, A., 2005. Biotreatment of Industrial Effluents. Elsevier Science. https://books.google.com/books?id=4qsNn-DZPScC

[207] Durai, G., Rajamohan, N., Karthikeyan, C.,Rajasimman, M., 2010. Kinetics studies on biological treatment of tannery wastewater using mixed culture. International Journal of Chemical and Biological Engineering. 3, 105-109. http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.309.2483&rep=rep1&type=pdf

[208] Yigit, N.O., Uzal, N., Koseoglu, H., Harman, I., Yukseler, H., Yetis, U., Civelekoglu, G.,Kitis, M., 2009. Treatment of a denim producing textile industry wastewater using pilot-scale membrane bioreactor. Desalination. 240, 143-150. https://doi.org/10.1016/j.desal.2007.11.071

[209] Tara, N., Arslan, M., Hussain, Z., Iqbal, M., Khan, Q.M.,Afzal, M., 2019. On-site performance of floating treatment wetland macrocosms augmented with dye-degrading bacteria for the remediation of textile industry wastewater. Journal of Cleaner Production. 217, 541-548. https://doi.org/10.1016/j.jclepro.2019.01.258

[210] Meriç, S., Selçuk, H.,Belgiorno, V., 2005. Acute toxicity removal in textile finishing wastewater by Fenton's oxidation, ozone and coagulation–flocculation processes. Water Research. 39, 1147-1153. https://doi.org/10.1016/j.watres.2004.12.021

[211] Patel, H.,Vashi, R.T., 2015. Characterization and Treatment of Textile Wastewater. Elsevier Science. https://books.google.com/books?id=TvicBAAAQBAJ

[212] Wang, Z., Xue, M., Huang, K.,Liu, Z., 2011. Textile dyeing wastewater treatment. Advances in treating textile effluent. 5, 91-116. https://www.intechopen.com/books/advances-in-treating-textile-effluent/textile-dyeing-wastewater-treatment

[213] Buljan, J., Kral, I.,Clonfero, G., 2011. Introduction to treatment of tannery effluents. UNIDO, Vienna. https://www.unido.org/sites/default/files/2011-11/Introduction_to_treatment_of_tannery_effluents_0.pdf

[214] Rajaganesh, K., Sumedha, N.,Ameer, B., 2014. Characterization of textile dye effluent from Komarapalayam, Namakkal district, Tamilnadu, India. Ind. Streams Res. J. 4, 2230-7850.

[215] Urban CUWA, 2016. Drainage statistical yearbook. Beijing: China Urban Water Association.

[216] Olsson, G., 2012. Water and Energy: Threats and Opportunities. IWA Publishing. https://doi.org/10.2166/9781780400693

[217] Plappally, A.K.,Lienhard V, J.H., 2012. Energy requirements for water production, treatment, end use, reclamation, and disposal. Renewable and Sustainable Energy Reviews. 16, 4818-4848. https://doi.org/10.1016/j.rser.2012.05.022

[218] Rothausen, S.,Conway, D., 2011. Greenhouse-gas emissions from energy use in the water sector. Nature Climate Change. 1, 210-219. https://doi.org/10.1038/nclimate1147

[219] McCarty, P.L., Bae, J.,Kim, J., 2011. Domestic Wastewater Treatment as a Net Energy Producer–Can This be Achieved? Environmental Science & Technology. 45, 7100-7106. https://doi.org/10.1021/es2014264

[220] EPA, 2006. Wastewater Management Fact Sheet: Energy Conservation. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100IL6T.PDF

[221] Scherson, Y.D.,Criddle, C.S., 2014. Recovery of Freshwater from Wastewater: Upgrading Process Configurations To Maximize Energy Recovery and Minimize Residuals. Environmental Science & Technology. 48, 8420-8432. https://doi.org/10.1021/es501701s

[222] EPA, 2008. Ensuring a Sustainable Future An Energy Management Guidebook for Wastewater and Water Utilities. https://nepis.epa.gov/Exe/ZyPDF.cgi/P1003Y1G.PDF?Dockey=P1003Y1G.PDF

[223] EPA, 2013. Local Government Climate and Energy Strategy Guides: Energy Efficiency in Water and Wastewater Facilities a Guide to Developing and Implementing Greenhouse Gas Reduction Programs. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100GIF5.PDF

[224] Young, D.F.,Koopman, B., 1991. Electricity Use in Small Wastewater Treatment Plants. Journal of Environmental Engineering. 117, 300-307. https://doi.org/10.1061/(ASCE)0733-9372(1991)117:3(300)

[225] 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. https://doi.org/10.2166/wst.2015.306

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

[227] Gude, V.G., 2015. Energy and water autarky of wastewater treatment and power generation systems. Renewable and Sustainable Energy Reviews. 45, 52-68. https://doi.org/10.1016/j.rser.2015.01.055

[228] 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. Applied Energy. 179, 1251-1268. https://doi.org/10.1016/j.apenergy.2016.07.043

[229] Kneppers, B., Birchfield, D.,Lawton, M., 2009. Energy-water relationships in reticulated water infrastructure systems. Water Supply (kWh/Capita). 76, 75.15. https://www.beaconpathway.co.nz/images/uploads/Final_Report_WA7090(2)_Energy_and_water_relationships.pdf

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

[231] Bodik, I.,Kubaska, M., 2013. Energy and sustainability of operation of a wastewater treatment plant. Environment Protection Engineering. 39, 15–24. https://www.researchgate.net/publication/278147489_Energy_and_sustainability_of_operation_of_a_wastewater_treatment_plant

[232] Garrido, J.M., Fdz-Polanco, M.,Fdz-Polanco, F., 2013. Working with energy and mass balances: a conceptual framework to understand the limits of municipal wastewater treatment. Water Sci Technol. 67, 2294-2301. https://doi.org/10.2166/wst.2013.124

[233] 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. https://doi.org/10.2166/wst.2015.189

[234] 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. https://doi.org/10.1016/j.envint.2016.03.024

[235] Gu, Y., Dong, Y., Wang, H., Keller, A., Xu, J., Chiramba, T.,Li, F., 2016. Quantification of the water, energy and carbon footprints of wastewater treatment

plants in China considering a water–energy nexus perspective. Ecological Indicators. 60, 402-409. https://doi.org/10.1016/j.ecolind.2015.07.012

[236] Li, W., Li, L.,Qiu, G., 2017. Energy consumption and economic cost of typical wastewater treatment systems in Shenzhen, China. Journal of Cleaner Production. 163, S374-S378. https://doi.org/10.1016/j.jclepro.2015.12.109

[237] Balmer, P.,Hellstrom, D., 2012. Performance indicators for wastewater treatment plants. Water Sci Technol. 65, 1304-1310. https://doi.org/10.2166/wst.2012.014

[238] Henderson, M., 2002. Energy Reduction Methods in the Aeration Process at Perth Waste Water Treatment Plant. http://www.esru.strath.ac.uk/Documents/MSc_2002/henderson.pdf

[239] Thompson, L., Song, K., Lekov, A.,McKane, A., 2008. Automated demand response opportunities in wastewater treatment facilities. https://www.osti.gov/servlets/purl/944125

[240] Awe, O.W., Liu, R.,Zhao, Y., 2016. Analysis of Energy Consumption and Saving in Wastewater Treatment Plant: Case Study from Ireland. Journal of Water Sustainability. 6, 63-76. https://doi.org/10.11912/jws.2016.6.2.63-76

[241] Boujelben, I., Samet, Y., Messaoud, M., Ben Makhlouf, M.,Maalej, S., 2017. Descriptive and multivariate analyses of four Tunisian wastewater treatment plants: A comparison between different treatment processes and their efficiency improvement. J Environ Manage. 187, 63-70. https://doi.org/10.1016/j.jenvman.2016.11.023

[242] Wang, H.T., Yang, Y., Keller, A.A., Li, X., Feng, S.J., Dong, Y.N.,Li, F.T., 2016. Comparative analysis of energy intensity and carbon emissions in wastewater treatment in USA, Germany, China and South Africa. Applied Energy. 184, 873-881. https://doi.org/10.1016/j.apenergy.2016.07.061

[243] Trapote, A., Albaladejo, A.,Simón, P., 2014. Energy consumption in an urban wastewater treatment plant: the case of Murcia Region (Spain). Civil Engineering and Environmental Systems. 31, 304-310. https://doi.org/10.1080/10286608.2013.866106

[244] Hanna, S.M., Thompson, M.J., Dahab, M.F., Williams, R.E.,Dvorak, B.I., 2018. Benchmarking the Energy Intensity of Small Water Resource Recovery Facilities. Water Environ Res. 90, 738-747. https://doi.org/10.2175/106143017X15131012153176

[245] Gans, N., Mobini, S.,Zhang, X., Mass and energy balances at the Gaobeidian wastewater treatment plant in Beijing, China. 2007, Masters Thesis.

[246] Zhou, Y., Zhang, D.Q., Le, M.T., Puah, A.N.,Ng, W.J., 2013. Energy utilization in sewage treatment–a review with comparisons. Journal of water and climate change. 4, 1-10. https://doi.org/10.2166/wcc.2013.117

[247] Lorenzo-Toja, Y., Alfonsín, C., Amores, M.J., Aldea, X., Marin, D., Moreira, M.T.,Feijoo, G., 2016. Beyond the conventional life cycle inventory in wastewater treatment plants. Science of the Total Environment. 553, 71-82. https://doi.org/10.1016/j.scitotenv.2016.02.073

[248] Taboada-Santos, A., Lema, J.M.,Carballa, M., 2019. Opportunities for rotating belt filters in novel wastewater treatment plant configurations. Environmental Science: Water Research & Technology. 5, 704-712. https://doi.org/10.1039/C8EW00899J

[249] Crites, R.W., Middlebrooks, E.J.,Reed, S.C., 2010. Natural Wastewater Treatment Systems. CRC Press. https://books.google.com/books?id=X5uuikcVyfEC

[250] EPA, 1990. Assessment of the biolac technology. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=2000GHJ2.PDF

[251] Mirbagheri, S.A., Saberi Yahyaei, S.M., Rafeidehkordi, N.,Vakilian, R. Year. Comparison of CAS, MBR, SBR, and Biolak Treatment Systems in Removal of BOD and COD from Municipal Wastewater—Case Study: Ekbatan Wastewater Treatment Plant. in 15th International Conference on Environmental Science and Technology, Rhodes, Greece. 2017. https://cest.gnest.org/sites/default/files/presentation_file_list/cest2017_00559_poster_paper.pdf

[252] Zhao, L.J., Wang, H.J.,Liu, J.L., 2002. Evaluation and Analysis on Two-stage Biological Contact Oxidation Process for Treatment of Urban Wastewater. CHINA WATER AND WASTEWATER. 18, 28-30.

[253] Wang, S., Zhou, M., Zeng, L.,Dai, W., 2016. Investigation on the Preparation and properties of reticulate porous ceramic for organism carrier in sewage disposal. J. Ceram. Process. Res. . 17, 1095-1099.

[254] Liu, Y., Zhu, Y., Jia, H., Yong, X., Zhang, L., Zhou, J., Cao, Z., Kruse, A.,Wei, P., 2017. Effects of different biofilm carriers on biogas production during anaerobic digestion of corn straw. Bioresour Technol. 244, 445-451. https://doi.org/10.1016/j.biortech.2017.07.171

[255] Xu, S.,Jiang, Q., 2018. Surface modification of carbon fiber support by ferrous oxalate for biofilm wastewater treatment system. Journal of Cleaner Production. 194, 416-424. https://doi.org/10.1016/j.jclepro.2018.05.159

[256] Zhang, X., Zhou, X., Ni, H., Rong, X., Zhang, Q., Xiao, X., Huan, H., Liu, J.F.,Wu, Z., 2018. Surface Modification of Basalt Fiber with Organic/Inorganic Composites for Biofilm Carrier Used in Wastewater Treatment. ACS Sustainable Chemistry & Engineering. 6, 2596-2602. https://doi.org/10.1021/acssuschemeng.7b04089

[257] Feng, Q., Wang, Y., Wang, T., Zheng, H., Chu, L., Zhang, C., Chen, H., Kong, X.,Xing, X.H., 2012. Effects of packing rates of cubic-shaped polyurethane foam

carriers on the microbial community and the removal of organics and nitrogen in moving bed biofilm reactors. Bioresour Technol. 117, 201-207. https://doi.org/10.1016/j.biortech.2012.04.076

[258] Chu, L., Wang, J., Quan, F., Xing, X.-H., Tang, L.,Zhang, C., 2014. Modification of polyurethane foam carriers and application in a moving bed biofilm reactor. Process Biochemistry. 49, 1979-1982. https://doi.org/10.1016/j.procbio.2014.07.018

[259] Mutamim, N.S.A., Noor, Z.Z., Hassan, M.A.A., Yuniarto, A.,Olsson, G., 2013. Membrane bioreactor: Applications and limitations in treating high strength industrial wastewater. Chemical Engineering Journal. 225, 109-119. https://doi.org/10.1016/j.cej.2013.02.131

[260] 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. Applied Energy. 204, 1463-1475. https://doi.org/10.1016/j.apenergy.2017.02.069

[261] Cornejo, P.K., 2015. Environmental Sustainability of Wastewater Treatment Plants Integrated with Resource Recovery: The Impact of Context and Scale. https://scholarcommons.usf.edu/etd/5669/

[262] Tillman, A.M., Svingby, M.,Lundström, H., 1998. Life cycle assessment of municipal waste water systems. The international journal of life cycle assessment. 3, 145-157. https://doi.org/10.1007/BF02978823

[263] Lundin, M., Bengtsson, M.,Molander, S., 2000. Life cycle assessment of wastewater systems: influence of system boundaries and scale on calculated environmental loads. Environmental Science & Technology. 34, 180-186. https://doi.org/10.1021/es990003f

[264] Capodaglio, A.G.,Olsson, G., 2020. Energy issues in sustainable urban wastewater management: Use, demand reduction and recovery in the urban water cycle. Sustainability. 12, 266. https://doi.org/10.3390/su12010266

[265] EPA, 2011. Opportunities for Combined Heat and Power at Wastewater Treatment Facilities Market Analysis and Lessons from the Field. https://nepis.epa.gov/Exe/ZyPDF.cgi?Dockey=P100N1G3.PDF

[266] Wu, J., Jiang, J., Zhou, H.M.,Bi, L., 2008. Current operation status of sludge anaerobic digestion system in municipal wastewater treatment plants in China. China Water & Wastewater. 22.

[267] Park, N., Oh, J., Lee, M., Jang, H.,Ahn, Y., 2018. Effect of influent flowrate on the performance of biological wastewater treatment processes. Desalination and Water Treatment. 112, 278-281. https://doi.org/10.5004/dwt.2018.22149

[268] Kraume, M., Bracklow, U., Vocks, M.,Drews, A., 2005. Nutrients removal in MBRs for municipal wastewater treatment. Water Science and Technology. 51, 391-402. https://doi.org/10.2166/wst.2005.0661

[269] Meng, F., Chae, S.R., Drews, A., Kraume, M., Shin, H.S.,Yang, F., 2009. Recent advances in membrane bioreactors (MBRs): membrane fouling and membrane material. Water Res. 43, 1489-1512. https://doi.org/10.1016/j.watres.2008.12.044

[270] Tay, J., Yang, P., Zhuang, W.Q., Tay, S.T.L.,Pan, Z.H., 2007. Reactor performance and membrane filtration in aerobic granular sludge membrane bioreactor. Journal of Membrane Science. 304, 24-32. https://doi.org/10.1016/j.memsci.2007.05.028

[271] Tay, J., Liu, Y., Tay, S.T.,Hung, Y., 2009. Aerobic granulation technology, in Advanced Biological Treatment Processes, Springer. 109-128

[272] Thanh, B.X., Visvanathan, C.,Ben Aim, R., 2013. Fouling characterization and nitrogen removal in a batch granulation membrane bioreactor. International Biodeterioration & Biodegradation. 85, 491-498. https://doi.org/10.1016/j.ibiod.2013.02.005

[273] Liu, Y.,Tay, J.H., 2004. State of the art of biogranulation technology for wastewater treatment. Biotechnol Adv. 22, 533-563. https://doi.org/10.1016/j.biotechadv.2004.05.001

[274] Thanh, B.X., Visvanathan, C., Spérandio, M.,Aim, R.B., 2008. Fouling characterization in aerobic granulation coupled baffled membrane separation unit. Journal of Membrane Science. 318, 334-339. https://doi.org/10.1016/j.memsci.2008.02.058

[275] Show, K.Y., Lee, D.J.,Tay, J.H., 2012. Aerobic granulation: advances and challenges. Appl Biochem Biotechnol. 167, 1622-1640. https://doi.org/10.1007/s12010-012-9609-8

[276] Kong, Y., Liu, Y.Q., Tay, J.H., Wong, F.S.,Zhu, J., 2009. Aerobic granulation in sequencing batch reactors with different reactor height/diameter ratios. Enzyme and Microbial Technology. 45, 379-383. https://doi.org/10.1016/j.enzmictec.2009.06.014

[277] Moreau, A.A., Ratkovich, N., Nopens, I.,van der Graaf, J.H.J.M., 2009. The (in)significance of apparent viscosity in full-scale municipal membrane bioreactors. Journal of Membrane Science. 340, 249-256. https://doi.org/10.1016/j.memsci.2009.05.049

[278] Verrecht, B., Judd, S., Guglielmi, G., Brepols, C.,Mulder, J.W., 2008. An aeration energy model for an immersed membrane bioreactor. Water Res. 42, 4761-4770. https://doi.org/10.1016/j.watres.2008.09.013

[279] Nywening, J.P.,Zhou, H., 2009. Influence of filtration conditions on membrane fouling and scouring aeration effectiveness in submerged membrane bioreactors to

treat municipal wastewater. Water Res. 43, 3548-3558. https://doi.org/10.1016/j.watres.2009.04.050

[280] Spellman, F.R., 2008. Handbook of Water and Wastewater Treatment Plant Operations, Second Edition. Taylor & Francis. https://books.google.com/books?id=Ja28hvTxVpwC

[281] Yang, L.B., Zeng, S.Y., Ju, Y.P., He, M.,Chen, J.N., 2008. Statistical analysis of energy consumption of municipal wastewater treatment plants in China. Water Suppl. Drain. 034, 42-45. https://doi.org/10.3969/j.issn.1002-8471.2008.10.009

[282] 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. https://doi.org/10.1016/j.scitotenv.2019.03.159

[283] Spruston, S., Kolesov, A.,Main, D., 2012. Leveraging the Energy of the Group to Manage the Energy of the Utility: The NWWBI Adopts Industry Tools to Improve Energy Performance. Proceedings of the Water Environment Federation. 2012, 2383-2402. https://doi.org/10.2175/193864712811726365

[284] Fan, L.F., Hu, J.Z.,Ou, L., 2016. Research on Low Inlet Concentration of Urban Sewage Treatment Plant and Countermeasures. Environmental Science and Management. 41, 132-135.

[285] Midkiff, R.C., 2016. A Cultural and Technical Study of Wastewater Treatment Plant Maintenance in a Small Community in Peru. https://digitalcommons.mtu.edu/cgi/viewcontent.cgi?referer=https://scholar.google.com/&httpsredir=1&article=1268&context=etdr

[286] Klotz, M.G., 2011. Research on Nitrification and Related Processes, Part A. Elsevier Science. https://books.google.com/books?id=o0lcIwXpvS4C

[287] Kabouris, J.C., Tezel, U., Pavlostathis, S.G., Engelmann, M., Dulaney, J.A., Todd, A.C.,Gillette, R.A., 2009. Mesophilic and thermophilic anaerobic digestion of municipal sludge and fat, oil, and grease. Water Environment Research. 81, 476-485. https://doi.org/10.2175/106143008X357192

[288] ENR, 2020. Construction economics. Engineering News-Record. http://enr.construction.com/economics

[289] EIA, 2020. Average Price of Retail Electricity to Ultimate Customers by End-Use Sector, by State. US Energy Information Administration. https://www.eia.gov/electricity/monthly/current_month/january2021.pdf

[290] Schaum, C., Lensch, D.,Cornel, P., 2015. Water reuse and reclamation: a contribution to energy efficiency in the water cycle. Journal of Water Reuse and Desalination. 5, 83-94.

[291] Fatta, D.,Anayiotou, S., 2007. MEDAWARE project for wastewater reuse in the Mediterranean countries: An innovative compact biological wastewater treatment system for promoting wastewater reclamation in Cyprus. Desalination. 211, 34-47.

[292] Methanex, 2020. Methanex posts regional contract methanol prices for North America, Europe and Asia. . https://www.methanex.com/our-business/pricing

APPENDIX

Leachate characteristics

The current study collected the data of leachate from peer-reviewed papers, as shown in Table 3.1. The databases of untreated leachate quality included basic information such as the concentration of TSS, COD, BOD and TN. The first row of data in table A.1 was selected as the specific young leachate quality for presenting the specific processes for estimating energy and cost of innovative treatment system. According to coefficients of the wastewater obtained from GPS-X 7.0 in table A.2, we could divide TSS, COD, BOD and TN of the specific young leachate into fractions, as shown in table A.3 and figures A.1-A.4.

Table A.1 Leachate characteristics

TSS (mg/L)	BOD (mg/L)	COD (mg/L)	TN (mg/L)	Reference
1894.11	4979.41	9309.80	1171.83	[94]

Table A.2 Coefficients of the wastewater obtained from GPS-X 7.0 and corresponding equations

Parameter	Value	Reference
Volatile Suspended Solids/Total Suspended Solids, VSS/TSS	0.75	[135]
Particulate COD/Volatile Suspended Solids, pCOD/VSS	1.42	[91]
Colloidal COD/slowly biodegradable COD, cCOD/sbCOD	0.15	[135]
BOD to COD ratio of soluble and colloidal biodegradable substrates, fBOD/(bsCOD+cCOD)	0.717	[135]
BOD to COD ratio of particulate biodegradable substrate, pBOD/bpCOD	0.58	[135]
Biodegradable particulate organic nitrogen/biodegradable particulate COD, bpON/bpCOD	0.0268	[123]
Non-biodegradable particulate organic nitrogen/non-biodegradable particulate COD, nbpON/nbpCOD	0.068	[123]
Non-biodegradable soluble organic nitrogen/non-biodegradable soluble COD, nbsON/nbsCOD	0.05	[135]
Biodegradable soluble COD, bsCOD	(bCOD)(sCOD)/COD	[27]

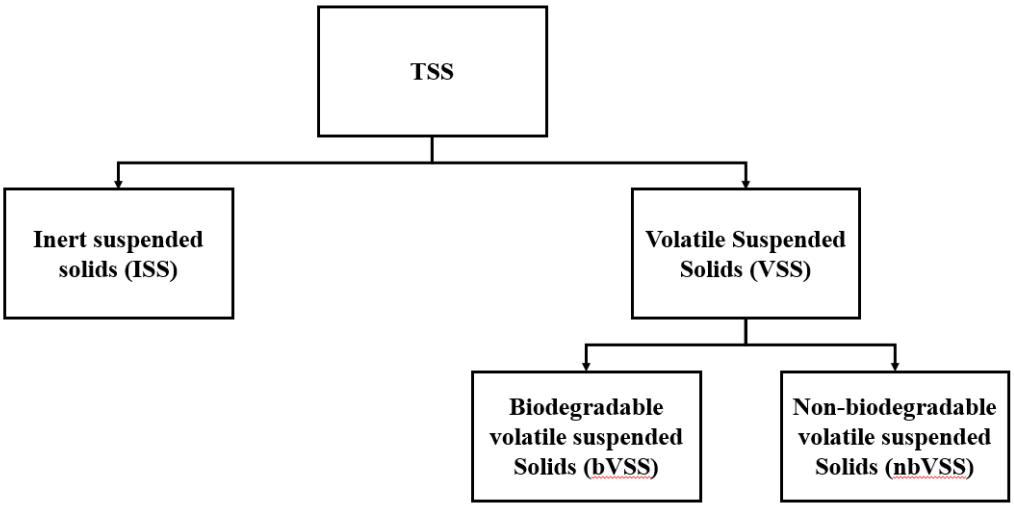


Figure A.1 Fractionation of SS

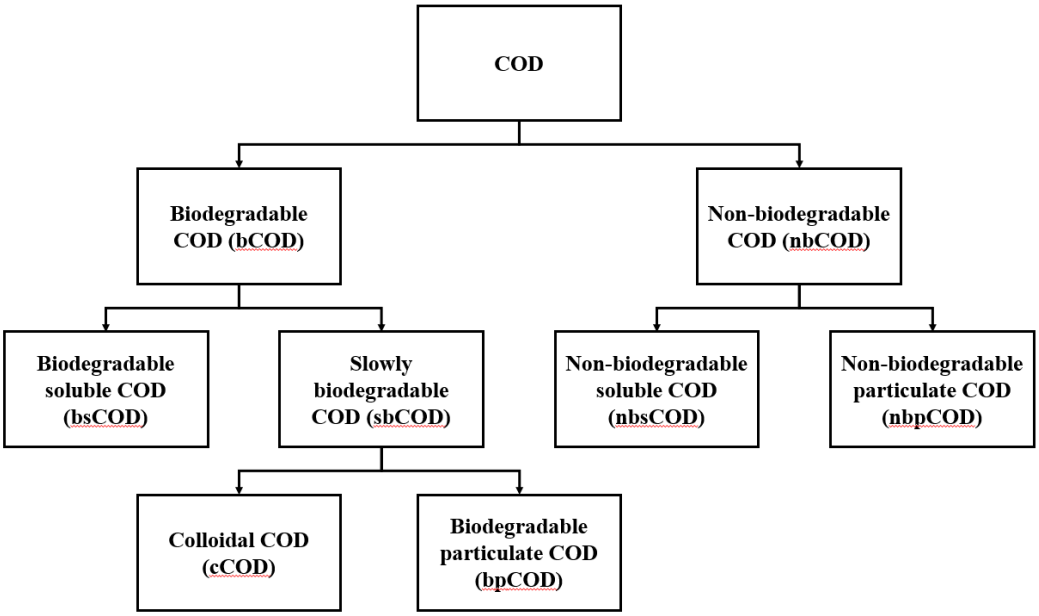


Figure A.2 Fractionation of COD

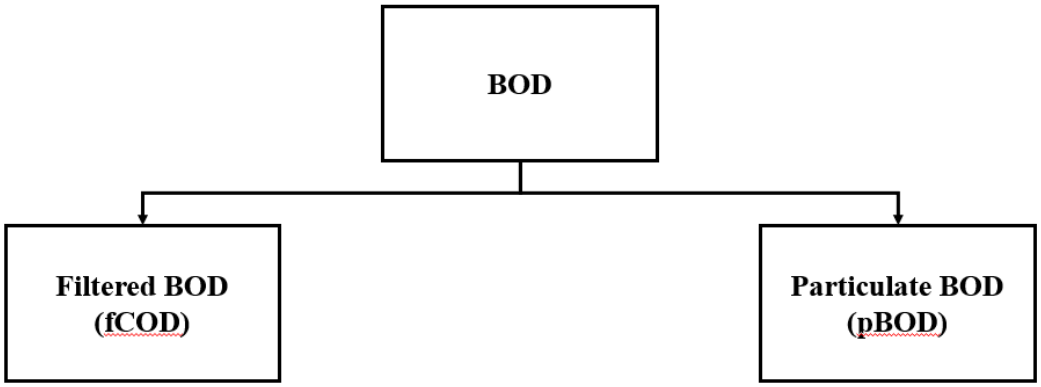


Figure A.3 Fractionation of BOD

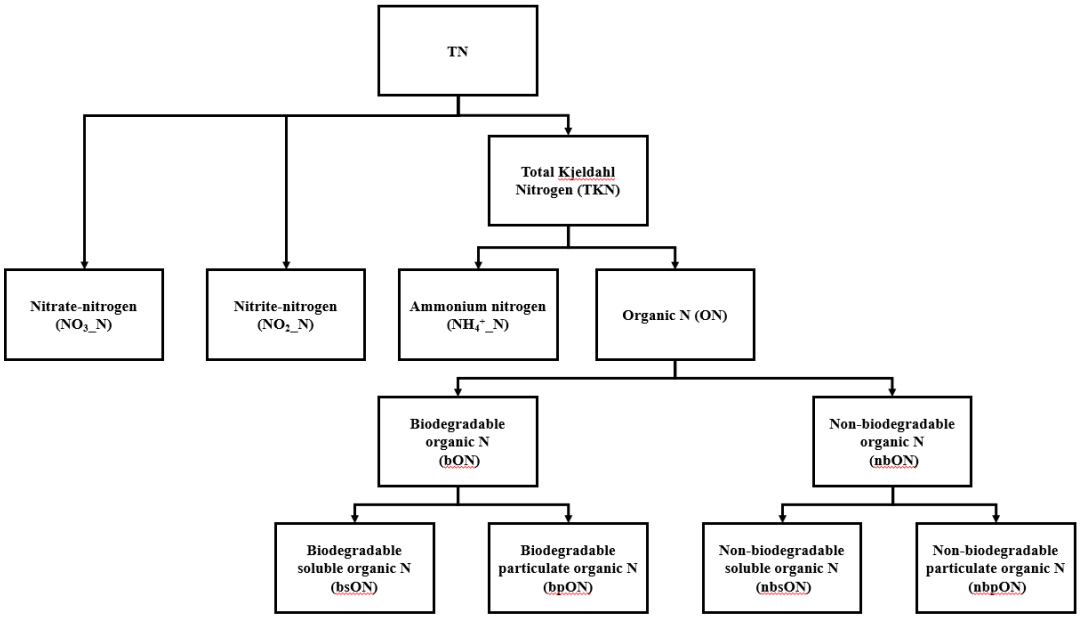


Figure A.4 Fractionation of N

Table A.3 Specific young leachate characteristics

Characteristics	Concentration (mg/L)
Inert suspended solids (ISS)	473.53
Volatile suspended solids (VSS)	1420.58
Biodegradable volatile suspended solids (bVSS)	1061.86
Non-biodegradable volatile suspended solids (nbVSS)	358.72
Non-biodegradable particulate COD (nbpCOD)	509.38
Biodegradable particulate COD (bpCOD)	1507.85
Colloidal COD (cCOD)	266.09
Biodegradable soluble COD (bsCOD)	5458.96
Non-biodegradable soluble COD (nbsCOD)	1567.53
Biodegradable COD (bCOD)	7232.89
Particulate BOD (pBOD)	874.55
Filtered BOD (fBOD)	4104.86
Non-biodegradable particulate organic nitrogen (nbpON)	34.64
Biodegradable particulate organic nitrogen (bpON)	40.41

Non-biodegradable soluble organic nitrogen (nbsON)	78.38
Biodegradable organic nitrogen plus ammonium-Nitrogen (bON+NH_4^+_N)	1058.82

Determine the SS parameters

VSS=(0.75)TSS=(0.75)(1894.11)=1420.58mg/L

ISS=TSS-VSS=1894.11-1420.58=473.53 mg/L

Determine the COD parameters

Since fBOD/(bsCOD+cCOD)= 0.717

pBOD/bpCOD=0.58

BOD=fBOD+pBOD

bCOD=bsCOD+cCOD+bpCOD

0.58<BOD/bCOD<0.717

1.39<bCOD/BOD<1.72

Hence, assume bCOD/BOD=1.5

bCOD=(1.4)BOD=(1.5)(4979.41)=7469.12 mg/L

pCOD=(1.42)VSS=(1.42)(1420.58)=2017.23 mg/L

fCOD=cCOD+sCOD=COD-pCOD=9309.8-2017.23=7292.57 mg/L

nbCOD=COD- bCOD=9309.8-7469.12=1840.69 mg/L

Since bpCOD<pCOD=2017.23 mg/L

Hence, assume bpCOD=1500 mg/L

bVSS=bpCOD/1.42=1500/1.42=1056.34 mg/L

bpCOD=sbCOD-cCOD=sbCOD-0.15sbCOD=0.85sbCOD

sbCOD=bpCOD/0.85

cCOD=0.15sbCOD=0.15bpCOD/0.85=264.71 mg/L

sCOD=fCOD-cCOD=7292.57-264.71=7027.87 mg/L

nbpCOD=pCOD-bpCOD=2017.23-1500=517.23 mg/L

nbVSS=nbpCOD/1.42=517.23/1.42=364.24 mg/L

bsCOD=((bCOD)(sCOD))/COD=((7469.12)(7027.87))/9309.8=5638.35 mg/L

nbsCOD=sCOD-bsCOD=7027.87-5638.35=1389.51 mg/L

pBOD=(0.58)(bpCOD)=(0.58)(1500)=870 mg/L

fBOD=(0.717)(bsCOD+cCOD)=(0.717)(5638.35+264.71)=4232.49 mg/L

BOD= pBOD+ fBOD=5102.49 mg/L

bCOD= bpCOD+ bsCOD+cCOD=7403.06 mg/L

bCOD/BOD=7403.06/5102.49=1.45

The calculated BOD and bCOD/BOD are close to the given value and the assumed ratio. We change the assumptions of bCOD/BOD ratio and bpCOD concentration until the calculated BOD value and bCOD/BOD ratio are equal to the given BOD concentration and assumed bCOD/BOD ratio. The final result of SS and COD parameters is shown in table A.3.

Determine the BOD parameters

pBOD=(0.58)(bpCOD)=(0.58)(1507.85)= 874.55 mg/L

fBOD=(0.717)(bsCOD+cCOD)=(0.717)(5458.96+266.09)= 4104.86 mg/L

Determine the N parameters

nbpON=(0.068) nbpCOD =(0.068)(509.38)=34.64 mg/L

bpON=(0.0268) bpCOD =(0.0268)(1507.85)=40.41 mg/L

nbsON=(0.05) nbsCOD =(0.05)(1567.53)=78.38 mg/L

bON+NH_4^+_N=TN-NO_3^+_N-NO_2^+_N-nbpON-nbsON=1171.83-0-0-34.64-78.38=1058.82 mg/L

Innovative treatment system for SS, COD, BOD and N removal.

Process flow diagrams of innovative treatment system is illustrated in Figure A.5.

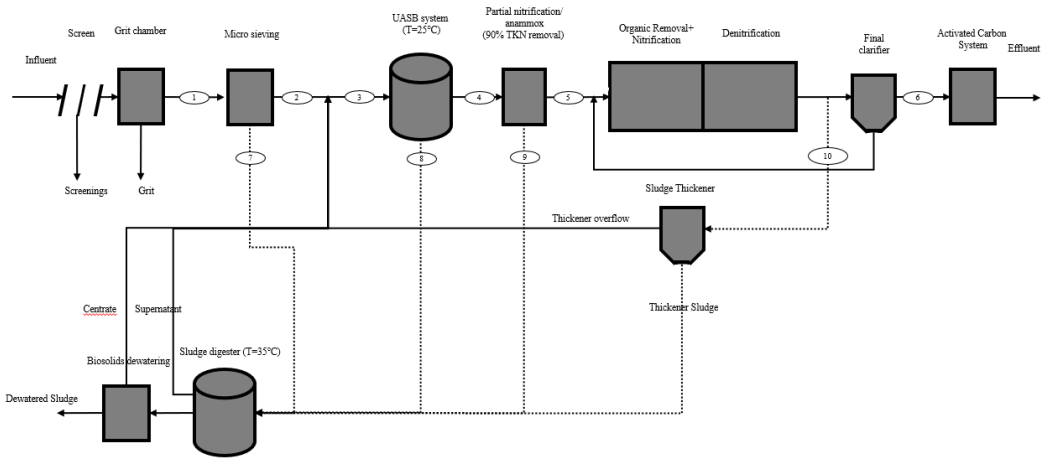


Figure A.5 Schematic graphs of innovative treatment system - mainstream anaerobic treatment and anammox treatment

Preliminary treatment

Preliminary treatment could remove screening and grit form wastewater. It was reported that volumes of screening and grit production are 4-100 L/1,000 m^3 and 4-37 L/1,000 m^3 for wastewater without stormwater inputs [27]. Due to the relatively small volumes of screening and grit production, the changes of flow rate through preliminary treatment could be ignored in this study. In BioWin software, ISS portion of the TSS was removed by preliminary treatment. 60% was assumed as ISS removal efficiency of preliminary treatment [285].

Calculate the mass loadings to the primary treatment (Stream 1).

$$Q_1=Q_i=1MGD=3{,}785.4\ m^3/d$$

Mass of TSS, $W_{TSS,\,1}=(C_{VSS,\,1}+C_{ISS,\,1})\,Q_1=(C_{VSS,\,i}+(1\text{-}60\%)\,C_{ISS,\,i}\,Q_i$

$=(1{,}420.58\ g/m^3+(40\%)(473.53\ g/m^3))\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 6{,}094.47\ kg/d$

Mass of ISS, $W_{ISS,\,1}==(C_{ISS,\,1})\,Q_1=((1\text{-}60\%)\,C_{ISS,\,i})Q_i$

$=((40\%)(473.53\ g/\ m^3))\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 717.00\ kg/d$

Mass of COD, $W_{COD,\,1}=(C_{COD,\,1})Q_1=(C_{COD,\,i})Q_i$

$= 9{,}309.80\ g/\ m^3\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 35{,}241.32\ kg/d$

Mass of pCOD, $W_{pCOD,\,1}=(C_{pCOD,\,1})\,Q_1=(C_{pCOD,\,i})\,Q_i=(C_{nbpCOD,\,i}+C_{bpCOD,\,i})\,Q_i$

$= (509.38\ g/\ m^3+1{,}507.85\ g/\ m^3)\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 7{,}636.01\ kg/d$

Mass of pCOD, $W_{bpCOD,\,1}=(C_{bpCOD,\,1})\,Q_1=(C_{bpCOD,\,i})\,Q_i$

$= (1{,}507.85\ g/\ m^3)\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 5{,}707.80\ kg/d$

Mass of nbpCOD, $W_{nbpCOD,\,1}=(C_{nbpCOD,\,1})\,Q_1=(C_{nbpCOD,\,i})\,Q_i$

$= (509.38\ g/\ m^3)\times 3{,}785.4\ m^3/d\times 10^{-3}\ kg/g = 1{,}928.21\ kg/d$

Mass of nbsCOD, $W_{nbsCOD,\,1}=(C_{nbsCOD,\,1})\,Q_1=(C_{nbsCOD,\,i})\,Q_i$

$= (1{,}567.53\ g/\ m^3) \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 5{,}933.71\ kg/d$

Mass of BOD, $W_{BOD,\,1}=(C_{BOD,\,1})\,Q_1=(C_{BOD,\,i})\,Q_i$

$= 4{,}979.41\ g/\ m^3 \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 18{,}849.06\ kg/d$

Mass of pBOD, $W_{pBOD,\,1}=(C_{pBOD,\,1})Q\_1=(C_{pBOD,\,i})Q_i$

$= 874.55\ g/\ m^3 \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 3{,}310.52\ kg/d$

Mass of N, $W_{N,1}=(C_{N,\,1})Q_1=(C_{N,\,i})Q_i$

$= 1{,}171.83\ g/\ m^3 \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 4{,}435.85\ kg/d$

Mass of pON, $W_{pON,1}=C_{\,pON,\,1}\,Q_1= C_{\,pON,\,i}\,Q_i=(C_{\,nbpON,\,i}+C_{bpON,\,i})Q_i$

$= (34.64\ g/m^3 + 40.41\ g/m^3) \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 284.09\ kg/d$

Mass of nbsON, $W_{nbsON,1}=(C_{nbsON,\,1})Q_1=(C_{nbsON,\,i})Q_i$

$= (\ 78.38\ g/\ m^3\ ) \times 3{,}785.4\ m^3/d \times 10^{-3}\ kg/g = 296.69\ kg/d$

Primary treatment

Micro-sieving, or a rotating belt filter (RBF), offers a more sustainable alternative than a traditional primary clarifier with a smaller footprint and lower capital costs [15, 16]. In the current study, we use micro sieving as primary treatment. Behera et al. [30] reported actual TSS removal of a rotating belt filter with the screen size of 350 microns for treating wastewater with different TSS concentration. The result showed that micro sieving could achieve TSS removal of greater than 90% when TSS influent concentration is higher than 1200 mg/L. Behera et al. [30] also developed the following empirical model for estimating TSS removal.

$$\text{TSS removal of rotating belt filter} = \frac{16.45\ln[C_{TSS,1}\ ] - 29.1}{100}$$

$$\text{Concentration of TSS, } C_{TSS,1} = \frac{W_{TSS,1}}{Q_1} = \frac{6{,}094.47\,\text{kg/d}}{(3{,}785.4\,\text{m}^3/\text{d})(10^{-3}\ \text{kg/g})}$$

$$= 1{,}609.99\,\text{g/m}^3 = 1{,}609.99\,\text{mg/L}$$

$$\text{TSS removal of rotating belt filter} = \frac{16.45\ln(1{,}609.99\,\text{mg/L}) - 29.1}{100} = 92.37\%$$

For reflecting actual work conditions of micro sieving, estimated values based on the empirical model is used as TSS removal of micro sieving.

Calculate the primary sludge characteristics (Stream 7).

Mass of TSS, $W_{TSS,7}=(92.37\%)W_{TSS,1}=92.37\%\times 6{,}094.47$ kg/d= 5629.25 kg/d

Mass of ISS, $W_{ISS,7}=(92.37\%)\ W_{ISS,1}=92.37\%\times 717.00$ kg/d= 662.26 kg/d

Mass of VSS, $W_{VSS,7}= W_{TSS,7}- W_{ISS,7}$

=5629.25 kg/d- 662.26 kg/d= 4,966.99 kg/d

Mass of COD, $W_{COD,7}=W_{pCOD,7}=(92.37\%)W_{pCOD,1}$

= 92.37%×7,636.01 kg/d= 7,053.12 kg/d

Mass of bpCOD, $W_{bpCOD,7}=(92.37\%)W_{bpCOD,1}$

= 92.37%× 5,707.80 kg/d= 5,272.10 kg/d

Mass of nbpCOD, $W_{nbpCOD,7}=(92.37\%)W_{nbpCOD,1}$

= 92.37%× 1,928.21 kg/d= 1,781.02 kg/d

Mass of BOD, $W_{BOD,7}=W_{pBOD,7}=(92.37\%)W_{pBOD,1}$

= 92.37%×3,310.52 kg/d= 3,057.82 kg/d

Mass of N, $W_{N,7}=W_{pON,7}=(92.37\%)W_{pON,1}$ = 92.37%×284.09 kg/d=262.40 kg/d

As shown in table A.4, the specific gravity of the bulk sludge for primary sludge are 1.001-1.02. The solids content of micro-sieving sludge is 3-8% [17]. 0.055 and 1.008 are assumed to be the corresponding solid content and the specific gravity of the micro-

sieving sludge. Therefore, the wet bulk sludge concentration of micro-sieving sludge is equal to 55.44 kg/m^3.

Table A.4 Physical characteristics properties of sludge

Source of Sludge	Primary clarifier		Micro sieving		Anaerobic digestion		PN/A		Activated sludge		Reference
	Min	Max	Min	Max	Min	Max	Min	Max	Min	Max	
Solids Content (%)	1	6	3	8	3	6			0.2	0.6	[17, 28]
The specific gravity of the bulk sludge	1.001	1.02	1.001	1.02	1.01	1.03			1	1.003	
Wet bulk sludge concentration (kg/m^3)	10.01	61.2	30.03	81.6	30.3	61.8	50	75	2	6.018	

$$Q_7 = \frac{W_{TSS,7}}{C_{TSS,7}} = \frac{5629.25\,kg/d}{55.44\,kg/m^3} = 101.54\,m^3/d$$

Calculate the mass loadings to the secondary treatment (Stream 2).

Mass of TSS, $W_{TSS,\ 2}=W_{\ TSS,\ 1}-W_{\ TSS,\ 7}$

= 6,094.47 kg/d-5629.25 kg/d= 465.22 kg/d

Mass of ISS, $W_{ISS,\ 2}= W_{\ ISS,\ 1}-W_{\ ISS,\ 7}$

= 717.00 kg/d- 662.26 kg/d= 54.73 kg/d

Mass of COD, $W_{\ COD,\ 2}=W_{\ COD,\ 1}-W_{COD,\ 7}$

= 35,241.32 kg/d-7,053.12 kg/d= 28,188.20 kg/d

Mass of nbpCOD, $W_{\ nbpCOD,\ 2}=W_{\ nbpCOD,\ 1}-W_{\ nbpCOD,\ 7}$

= 1,928.21 kg/d-1,781.02 kg/d= 147.19 kg/d

Mass of nbsCOD, $W_{\ nbsCOD,\ 2}=W_{\ nbsCOD,\ 1}$=5,933.71 kg/d

Mass of BOD, $W_{\ BOD,\ 2}=W_{\ BOD,\ 1}-W_{\ BOD,\ 7}$

= 18,849.06 kg/d-3,057.82 kg/d= 15,791.24 kg/d

Mass of N, $W_{N,\ 2}=W_{N,\ 1}-W_{N,\ 7}$

$=$ 4,435.85 kg/d-262.40 kg/d=4,173.44 kg/d

$Q_2=Q_1-Q_7=3{,}785.4\ m^3/d-\ 101.54\ m^3/d=\ 3{,}683.86\ m^3/d$

In the first iteration, the combined sidestream is ignored.

$Q_{\text{combined sidestream}}=0\ m^3/d$

$Q_3=Q_2+Q_{\text{combined sidestream}}=(3{,}683.86+0)\ m^3/d=3{,}683.86\ m^3/d$

Mass of TSS, $W_{TSS,\ 3}=W_{TSS,\ 2}+W_{TSS,\ \text{combined sidestream}}=$ (465.22+0)kg/d=465.22kg/d

Mass of ISS, $W_{ISS,\ 3}=W_{ISS,\ 2}+W_{ISS,\ \text{combined sidestream}}=$ (54.73+0) kg/d= 54.73 kg/d

Mass of COD, $W_{COD,\ 3}=W_{COD,\ 2}+W_{COD,\ \text{combined sidestream}}$

$=$ (28,188.20+0) kg/d=28,188.20 kg/d

Mass of nbpCOD, $W_{nbpCOD,\ 3}=W_{nbpCOD,\ 2}+W_{nbpCOD,\ \text{combined sidestream}}$

=(147.19+0) kg/d=147.19 kg/d

Mass of nbsCOD, $W_{nbsCOD,\ 3}=W_{nbsCOD,\ 2}+W_{nbsCOD,\ \text{combined sidestream}}$

=(5,933.71+0) kg/d=5,933.71 kg/d

Mass of N, $W_{N,\ 3}=W_{N,\ 2}+W_{N,\ \text{combined sidestream}}$

=(4,173.44+0) kg/d=4,173.44 kg/d

UASB

The COD fraction

Mass of COD, $W_{COD,\ 3}=$ 28,188.20 kg/d

$$\text{nbpCOD fraction}, f_{nbpCOD,3} = \frac{147.19\,kg/d}{28{,}188.20\,kg/d} = 0.01$$

$$\text{nbsCOD fraction}, f_{nbsCOD,3} = \frac{5{,}933.71\,kg/d}{28{,}188.20\,kg/d} = 0.21$$

Cavalcanti et al. [42] studied the performance of UASB reactors for treating raw wastewater. The results show that UASB could produce stable sludge at an anaerobic sludge age of 40 to 50 days at 25 ℃. In the municipal wastewater treatment system, 40–60% of the influent COD could be converted into CH_4 by UASB reactors at an anaerobic sludge age of 30 to 60 days at 15 to 30 ℃. In practice, the sludge age of UASB is greater than 20 to 40 days [26]. Since old leachate has bad biodegradability (BOD/COD less than 0.1), sludge age, R_{su} is assumed to be 60 d for keeping stable performance of leachate treatment systems. Since biodegradable COD in wastewater could be converted into CH_4 at 15 and 35℃ [20], the operating temperature of the treatment system was assumed to be 25 ℃.

$$R_{su}=60d$$

$$T_{UASB}=25℃$$

$$\text{fraction of influent COD ending up as non-settleable COD in the UASB effluent, } f_{COD,3}$$

$$= f_{nbsCOD,3} + \frac{0.27\exp[-0.04(R_{su}-4)]}{1.067^{T_{UASB}-25}}$$

$$= 0.21 + + \frac{0.27\exp[-0.04(60-4)]}{1.067^{25-25}} = 0.24$$

$$\text{influent COD fraction converted into anaerobic excess sludge, } f_{COD,8}$$

$$= f_{nbpCOD,3} + f_{cv}\ Y_{an}\ (1 - f_{nbpCOD,2} - f_{nbsCOD,2}) + \frac{0.25\exp[-0.04(R_{su}-4)]}{1.067^{T_{UASB}-25}}$$

$$= 0.01 + (1.42)(0.05)(1-0.01-0.21) + \frac{0.25\exp[-0.04(60-4)]}{1.067^{25-25}}$$

$$= 0.09$$

where

Y_{an} = yield coefficient in an anaerobic environment =0.05 mg VSS / mg COD [42]

f_{CV}=ratio between mass of COD and bacterial mass =1.42 mg COD / mg VSS [91]

$$\text{fraction of influent COD digested in UASB, } f_{COD,CH_4}$$
$$= 1 - f_{COD,3} - f_{COD,8}$$
$$= 1 - 0.24 - 0.09$$
$$= 0.67$$

Calculate the UASB sludge characteristics (Stream 8).

Mass of COD, $W_{COD,\ 8} = f_{COD,\ 8}\ W_{COD,\ 3} = (0.09)(28{,}188.20\ kg/d) = 2{,}467.03\ kg/d$

Mass of nbpCOD, $W_{nbpCOD,\ 8} = W_{nbpCOD,\ 2} = 147.19\ kg/d$

Mass of bpCOD, $W_{bpCOD,\ 8} = W_{COD,\ 8} - W_{nbpCOD,\ 3}$

$= 2{,}467.03\ kg/d - 147.19\ kg/d = 2{,}319.84\ kg/d$

Mass of ISS, $W_{ISS,\ 8} = W_{ISS,\ 3} = 54.73\ kg/d$

Mass of VSS, $W_{VSS,\ 8} = W_{COD,\ 8}/f_{cv} = (2{,}467.03\ kg/d)/1.42 = 1{,}737.34\ kg/d$

Mass of bVSS, $W_{bVSS,\ 8} = W_{bCOD,\ 8}/f_{cv} = W_{bpCOD,\ 8}/f_{cv}$

$= (2{,}319.84\ kg/d)/1.42 = 1{,}633.69\ kg/d$

Mass of TSS, $W_{TSS,\ 8} = W_{VSS,\ 8} + W_{ISS,\ 8} = 1{,}737.34\ kg/d + 54.73\ kg/d = 1{,}792.07\ kg/d$

Since ratio between mass of nbpON and mass of nbpCOD is equal to 0.068 [123]

Mass of nbpON, $W_{nbpON,\ 8} = 0.068 W_{nbpCOD,\ 8} = (0.068)(147.19\ kg/d) = 10.01\ kg/d$

Since Biomass, which can be represented by the formula ($C_5H_7NO_2$), contains 0.12 g N/g biomass, Nitrogen content of biomass is equal to 0.12 gN/gVSS [27].

Mass of bpON, $W_{bpON,\ 8} = 0.12 W_{bVSS,\ 8} = (0.12)(1{,}633.69\ kg/d) = 196.04\ kg/d$

Mass of N, $W_{N,\ 8} = W_{nbpON,\ 8} + W_{bpON,\ 8} = 10.01\ kg/d + 196.04\ kg/d = 206.05\ kg/d$

As shown in table A.4, the specific gravity of the bulk sludge and solids content for anaerobic digestion sludge are 1.01-1.03 and 3-6%. 0.045 and 1.02 are assumed to be the

corresponding solid content and the specific gravity of the UASB sludge. Therefore, the wet bulk sludge concentration of UASB sludge is equal to 45.90 kg/m^3.

$$Q_8 = \frac{W_{TSS,8}}{C_{TSS,8}} = \frac{1{,}792.07\,\text{kg/d}}{45.90\,\text{kg/m}^3} = 39.04\,\text{m}^3/\text{d}$$

$$V_{UASB} = R_{su}\,Q_8 = (39.04\ \text{m}^3/\text{d})(60\text{d}) = 2{,}342.58\ \text{m}^3$$

Calculate the mass loadings to the PN/A process (Stream 3).

$$\text{Mass of COD, } W_{COD,\,4} = f_{COD,\,3}\,W_{COD,\,3}$$

$$= (0.24)(28{,}188.20\ \text{kg/d}) = 6{,}743.95\ \text{kg/d}$$

$$\text{Mass of nbsCOD, } W_{nbsCOD,\,4} = W_{nbsCOD,\,3} = 5{,}933.71\ \text{kg/d}$$

$$\text{Mass of bsCOD, } W_{bsCOD,\,4} = W_{COD,\,4} - W_{nbsCOD,\,4}$$

$$= 6{,}743.95\ \text{kg/d} - 5{,}933.71\ \text{kg/d} = 810.24\ \text{kg/d}$$

$$\text{Mass of N, } W_{N,\,4} = W_{N,\,3} - W_{N,\,8} = 4{,}173.44\ \text{kg/d} - 206.05\ \text{kg/d} = 3{,}967.39\ \text{kg/d}$$

$$\text{Mass of nbsN, } W_{nbsON,\,4} = W_{nbsON,\,3} = W_{nbsON,\,2} = W_{nbsON,\,1} = 296.69\ \text{kg/d}$$

$$\text{Mass of bON and } NH_4^+\text{\_N, } W_{bON+NH_4^+\_N,\,4} = W_{N,\,4} - W_{nbsON,\,4}$$

$$= 3{,}967.39\,\text{kg/d} - 296.69\ \text{kg/d} = 3{,}670.71\,\text{kg/d}$$

$$Q_4 = Q_3 - Q_8 = 3{,}683.86\ \text{m}^3/\text{d} - 39.04\ \text{m}^3/\text{d} = 3{,}644.82\ \text{m}^3/\text{d}$$

$$\text{Mass of COD, } W_{COD,\,CH_4} = f_{COD,\,CH_4}\ W_{COD,\,2} = (0.67)(28{,}188.20\,\text{kg/d}) = 18{,}977.22\,\text{kg/d}$$

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O$$

1 kg COD digested could be converted to 0.25 kg CH_4 or 0.35m^3 CH_4 theoretically at 0°C and 1 atm [28]

$$\text{Mass of } CH_4\text{, } W_{CH_4,UASB} = 0.25 W_{COD,\,CH_4} = (0.25)(18{,}977.22\,\text{kg/d}) = 4{,}744.31\,\text{kg/d}$$

Not all of the methane produced can be collected, a part of the methane will remain dissolved in the anaerobic effluent. Most of the dissolved methane in the anaerobic effluent is lost to the atmosphere at some point in the effluent collection system. Thus, it was assumed that the methane concentration eventually lost to the atmosphere is equal to the dissolved CH_4 concentration in the anaerobic effluent. 2.5 mg/L is assumed to be the dissolved CH_4 concentration in the anaerobic effluent based on Handbook of Biological Wastewater Treatment [26].

$$\text{Physical leakage of methane (kg/d)} = \frac{(2.5\text{g/ m}^3) \times Q_4\ (\text{m}^3/\text{d})}{1000\text{g/kg}}$$

$$\frac{(2.5\text{g/ m}^3) \times 3{,}644.82\,\text{m}^3/\text{d}}{1000\text{g/kg}} = 9.11\text{kg/d}$$

$$\text{Mass of } CH_4,\ W_{CH_4,UASB} = 0.25 W_{COD,CH_4} = 4{,}744.31\text{kg/d} - 9.11\text{kg/d} = 4{,}735.19\,\text{kg/d}$$

PN/A

Nitrification could convert nitrogen (N) in the form of ammonia (NH_3) or ammonium (NH_4^+) or organic N to oxidized N in the form of nitrite (NO_2^-) or nitrate (NO_3^-) [286]. Biodegradable organic nitrogen was converted to NH_4^+-N by ammonification prior to or during aeration process. PN/A, which is an innovative N removal technology discovered in the 1990s [24, 81, 82], could remove 90% of NH_4 _N through converting ammonia and nitrite to N_2 in the anammox process [83-86]. In this study, it was assumed that Biodegradable organic nitrogen was converted to NH_4^+-N by ammonification prior to PN/A process and NH_4^+-N removal of PN/A process is 90%

$$\text{Mass of } NH_4^+\_N,\ W_{bON+NH_4^+\_N,5} = (1-0.9) W_{bON+NH_4^+\_N,4}$$
$$= (0.1)(3{,}670.71\,\text{kg/d}) = 367.07\,\text{kg/d}$$

An overall reaction of PN/A process is given by the following equation:

$$NH_4^+ + 0.799\ O_2 + 1.109\ HCO_3^- \rightarrow 0.436\ N_2 + 0.111\ NO_3^- + 1.034\ CO_2 + 2.496\ H_2O + \text{New cell biomass}$$

Based on the reaction, it could be found that PN/A process does not consume organic matter. For greater ease in calculation, COD consumption during PN/A process is ignored.

$$\text{Mass of COD, } W_{COD,\ 5} = W_{COD,\ 4} = 6{,}743.95 \text{ kg/d}$$

$$\text{Mass of nbsCOD, } W_{nbsCOD,\ 5} = W_{nbsCOD,\ 4} = 5{,}933.71 \text{ kg/d}$$

$$\text{Mass of bsCOD, } W_{bsCOD,\ 5} = W_{bsCOD,\ 4} = 810.24 \text{ kg/d}$$

Based on the reaction, it could be found that the oxygen requirement is 1.83 g O_2/g NH_4_N converted, NO_3_N produced is 0.111 g NO_3_N/g NH_4_N converted and the new cell mass synthesis is 0.12 g VSS/g NH_4_N converted.

$$\text{Mass of } O_2 \text{ requirement, } W_{O_2,PN/A} = (1.83)(0.9)W_{bON+NH_4^+\_N,\ 4}$$
$$= (1.83)(0.9)(3{,}670.71 \text{kg/d}) = 6033.38 \text{kg/d}$$

Table A.5 Design parameter values of the PN/A SBR

Parameter	Unit	Value	Reference
Ratio between carbonaceous oxygen demand and nitrogenous oxygen demand	unitless	0.06	**[27]**
Number of cycles	Cycles/d	3	
Reaction time	h react/cycle	6	
Intermittent aeration during react period	h aerobic/h react	0.66	
Design oxygen uptake rate (OUR)	$g/m^3 \cdot h$	150	

$$\text{Maximum nitrogenous oxygen demand} = W_{O_2,PN/A}\ 6033.38 \text{kg/d}$$

Maximum carbonaceous oxygen demand=0.06(Maximum nitrogenous oxygen demand)

=0.06(6033.38kg/d)=362.00kg/d

Total maximum oxygen demand=6033.38kg/d+362.00kg/d=6395.39kg/d

Total aerobic time=(3 cycles/d) (6h react/cycle) (0.66h aerobic/h react)=12h aerobic/d

$$\text{Reactor volume at minimum liquid level} = \frac{(6395.39\text{kg/d})(10^3\ \text{g/1 kg})}{(12\text{h aerobic/d})(150\,\text{O}_2\ \text{g/m}^3\cdot\text{h}))}$$
$$= 3552.99\text{m}^3$$

$$\text{Maximum hydraulic load/cycle} = \frac{Q_4}{\text{Number of cycles}} = \frac{3{,}644.82\,\text{m}^3/\text{d}}{3\,\text{cycles/d}} = 1214.94\,\text{m}^3$$

$$V_{PN/A}=3552.99\text{m}^3+1214.94\ \text{m}^3=4767.93\ \text{m}^3$$

$$\text{Mass of } NO_3\_N, W_{NO_3\_N,4} = (0.111)(0.9)W_{NO_3\_N,3}$$
$$= (0.111)(0.9)(3{,}670.71\,\text{kg/d}) = 366.70\,\text{kg/d}$$

$$\text{Mass of VSS, } W_{VSS,9} = (0.12)(0.9)W_{bON+NH_4^+\_N,4} = (0.12)(0.9)(3{,}670.71\text{kg/d})$$
$$= 401.86\ \text{kg/d}$$

$$\text{Mass of TSS, } W_{TSS,9}=W_{VSS,9}= 401.86\ \text{kg/d}$$

It was reported that sludge concentration is equal to 50-75 kg/m^3 in anammox reactor [26]. Therefore, 50 kg/m^3 is assumed to be concentration of PN/A sludge.

$$Q_9 = \frac{W_{TSS,9}}{C_{TSS,9}} = \frac{401.86\,\text{kg/d}}{50\,\text{kg/m}^3} = 8.04\,\text{m}^3/\text{d}$$

$$Q_5=Q_4-Q_9= 3{,}644.82\ \text{m}^3/\text{d}-8.04\ \text{m}^3/\text{d}= 3{,}636.78\ \text{m}^3/\text{d}$$

Activated sludge process, nitrification, and denitrification.

Activated sludge process design for BOD removal with nitrification

Table A.6 Activated sludge design kinetic coefficients for nitrification at 20°C

Coefficient	Unit	COD oxidation	NH_4 oxidation	Reference
$\mu_{max,20℃}, \mu_{max,AOB,20℃}$	gVSS/gVSS·d	6	0.9	[27]
$K_s, K_{NH_4\_N}$	mg/L	8	0.5	
$Y_H, Y_{NH_4\_N}$	g VSS/g substrate oxidized	0.45	0.15	

$b_{H,20℃}, b_{AOB,20℃}$	gVSS/gVSS·d	0.12	0.17	
f_d	unitless	0.15	0.15	
$K_{o,AOB}$	mg/L	0.2	0.5	
Θ Value				
$\theta_{\mu_{max}}, \theta_{\mu_{max,AOB}}$	unitless	1.07	1.072	
$\theta_{b_H}, \theta_{b_{AOB}}$	unitless	1.04	1.029	

Since the nitrifying organisms grow more slowly than the heterotrophic organisms that remove organic matter, the nitrification rate will determine solids retention time (SRT) of activated sludge process with nitrification.

Determine the nitrification rate μ_{AOB}

$$\mu_{AOB} = \mu_{max,AOB}\left[\frac{C_{NH_4\_N,6}}{C_{NH_4\_N,6} + K_{NH_4\_N}}\right]\left[\frac{C_{DO}}{C_{DO} + K_{o,AOB}}\right] - b_{AOB}$$

$$T_{nitrification}=T_{UASB}=25℃$$

$$\mu_{max,AOB} = \mu_{max,AOB,20℃}\,\theta_{max,AOB}^{T_{nitrification}-20} = (0.9g/g\cdot d)(1.072)^{(25-20)} = 1.27gVSS/gVSS\cdot d$$

$$b_{AOB} = b_{AOB,20℃}\,\theta_{b,AOB}^{T_{nitrification}-20} = (0.17g/g\cdot d)(1.029)^{(25-20)} = 0.20gVSS/gVSS\cdot d$$

The minimum dissolved oxygen (DO) concentration of 2.0 mg/L is recommended for nitrification [27].

$$C_{DO}=2\ g/m^3$$

0.50 g/m^3 is selected as effluent NH_4_N concentration

$$C_{NH_4\_N,6} = 0.5g/m^3$$

$$\mu_{AOB} = \mu_{max,\,AOB}\left[\frac{C_{NH_4\_N,6}}{C_{NH_4\_N,6} + K_{NH_4\_N}}\right]\left[\frac{C_{DO}}{C_{DO} + K_{o,AOB}}\right] - b_{AOB}$$

$$= (1.27\text{gVSS/gVSS}\cdot\text{d})\left[\frac{0.5\text{g/m}^3}{0.5\text{g/m}^3 + 0.5\text{g/m}^3}\right]\left[\frac{2\text{g/m}^3}{2\text{g/m}^3 + 0.5\text{g/m}^3}\right] - 0.20\,\text{gVSS/gVSS}\cdot\text{d}$$

$$= 0.31\text{gVSS/gVSS}\cdot\text{d}$$

$$\text{Theoretical SRT} = \frac{1}{\mu_{AOB}} = \frac{1}{0.31\text{gVSS/gVSS}\cdot\text{d}} = 3.19\text{d}$$

$$= 0.31\text{gVSS/gVSS}\cdot\text{d}$$

Safety factors is in the range of 1.3 to 2.0 [27].

Safety factors (SF) is assumed to be 1.5.

$$\text{Design SRT=(SF)(Theoretical SRT)=(1.5)(3.19d)=4.78d}$$

$$\mu_{max} = \mu_{max,\,20℃}\,\theta_{\mu_{max}}^{T_{nitrification}-20} = (6\text{g/g}\cdot\text{d})(1.07)^{(25-20)} = 8.42\text{gVSS/gVSS}\cdot\text{d}$$

$$b_H = b_{H,\,20℃}\,\theta_{b_H}^{T_{nitrification}-20} = (0.12\text{g/g}\cdot\text{d})(1.04)^{(25-20)} = 0.15\text{gVSS/gVSS}\cdot\text{d}$$

$$\text{Effluent bCOD concentration, } C_{bCOD,6} = \frac{K_s\,[1 + b_H\,(\text{Design SRT})]}{\text{Design SRT}(\mu_{max} - b_H) - 1}$$

$$= \frac{(8\text{g/m}^3)[1 + (0.15\text{gVSS/gVSS}\cdot\text{d})\,(4.78\text{d})]}{(4.78\text{d})(8.42\text{gVSS/gVSS}\cdot\text{d} - 0.15\text{gVSS/gVSS}\cdot\text{d}) - 1} = 0.35\text{g/m}^3$$

$$C_{bCOD,5} = \frac{W_{bsCOD,5}}{Q_5} = \frac{810.24\,\text{kg/d}}{3{,}636.78\,\text{m}^3/\text{d}} = 222.79\text{g/m}^3$$

$$\text{Mass of VSS, } W_{VSS,\text{ activated sludge process with nitrification},10}$$

$$= \frac{Q_5\,Y_H\,(C_{bCOD,5} - C_{bCOD,6})}{1 + b_H\,(\text{Design SRT})} + \frac{f_d b_H Q_5\,Y_H\,(C_{bCOD,5} - C_{bCOD,6})(\text{Design SRT})}{1 + b_H\,(\text{Design SRT})}$$

$$+ \frac{Q_5\,Y_{NH_4\_N}\,(NO_x)}{1 + b_{AOB}\,(\text{Design SRT})}$$

$$C_{NH_4\_N,5} = \frac{W_{NH_4\_N,5}}{Q_5} = \frac{367.07\,\text{kg/d}}{3{,}636.78\,\text{m}^3/\text{d}} = 100.93\text{g/m}^3$$

For obtaining $W_{VSS,\ activated\ sludge\ process\ with\ nitrification,\ 10}$, we assume that the NO_x concentration is equal to 80% of the influent NH_4_N.

$$\text{Assumed } NO_x = 0.8(100.93 g/m^3) = 80.75 g/m^3$$

$$\text{Mass of VSS, } W_{VSS,activated\ sludge\ process\ with\ nitrification,10} = 259.51 kg/d$$

$$NO_x = C_{NH_4\_N,5} - C_{NH_4\_N,6} - 0.12\frac{W_{VSS,\ activated\ sludge\ process\ with\ nitrification,10}}{Q_4}$$

$$= 100.93 g/m^3 = 100.93 g/m^3 - 0.5 g/m^3 - (0.12)\frac{259{,}507.30 g/d}{3{,}636.78 m^3/d} = 91.87 g/m^3$$

The computed value is not equal to the 80.75 g/m^3 assumed. We change the assumptions of NO_x concentration until the calculated NO_x concentration is equal to the assumed NO_x concentration.

When assumed NO_x concentration is equal to 91.77 g/m^3, the calculated value is equal to the assumed value.

$$\text{Mass of VSS, } W_{VSS,\ activated\ sludge\ process\ with\ nitrification,\ 10} = 262.61 \text{ kg/d}$$

As shown in table A.4, the specific gravity of the bulk sludge and solids content for activated sludge are 1.000-1.003 and 0.2-0.6%. 0.004 and 1.0015 are assumed to be the corresponding solid content and the specific gravity of activated sludge with nitrification and denitrification. Therefore, the wet bulk sludge concentration of activated sludge is equal to 4.006 kg/m^3.

$$C_{TSS,\ 10} = 4.006 \text{ kg/m}^3$$

$$V_{activated\ sludge\ process\ with\ nitrification} = \frac{(W_{VSS,\ activated\ sludge\ process\ with\ nitrification,\ 10})(\text{Design SRT})}{C_{TSS,\ 10}}$$

$$= \frac{(262.61 kg/d)(4.78 d)}{4.006 kg/m^3} = 313.62 m^3$$

$$\text{Mass of } O_2 \text{ requirement}, W_{O_2,\text{activated sludge process with nitrification}}$$
$$= Q_5\ (C_{bCOD,5} - C_{bCOD,6}) - 1.42\frac{Q_5\ Y_H\ (C_{bCOD,5} - C_{bCOD,6})}{1 + b_H\ (\text{Design SRT})}$$
$$+ \frac{f_d b_H Q_5\ Y_H\ (C_{bCOD,5} - C_{bCOD,6})(\text{Design SRT})}{1 + b_H\ (\text{Design SRT})} + 4.57 Q_5\ NO_x$$
$$= 1{,}997.91 \text{kg/d}$$

Denitrification

Table A.7 Coefficients for denitrification [28]

Coefficient	Unit	Value	Reference
$\mu_{max,H,20℃}$	gVSS/gVSS·d	1.2	[28]
$K_{NO,H}$	mg/L	0.1	
$K_{s,H}$	mg/L	1.5	
$Y_{H,denitrification}$	g VSS/g bCOD	0.3	
$k_{d,H,20℃}$	gVSS/gVSS·d	0.05	
f_d	unitless	0.15	
$K_{DO,H}$	mg/L	0.2	
Θ Value			
$\theta_{\mu_{max,H}}$	unitless	1.1	
$\theta_{k_{d,H}}$	unitless	1.04	

The denitrification rate is sensitive to DO level in the anoxic zone. Denitrification is inhibited at a DO concentration above 0.1 mg/L [28]. DO level of 0.05 mg/L in the anoxic zone is assumed for design of denitrification process.

$$\mu_H = \mu_{max,H}\left[\frac{C_{NO_{34}\_N,6}}{C_{NO_3\_N,6} + K_{NO,H}}\right]\left[\frac{K_{DO,H}}{C_{DO,\text{denitrification}} + K_{DO,H}}\right]$$

$$T_{\text{denitrification}} = T_{\text{nitrification}} = 25℃$$

$$\mu_{max,H} = \mu_{max,H,20℃}\ \theta_{\mu_{max,H}}^{T_{\text{denitrification}}-20} = (1.2 \text{g/g} \cdot \text{d})(1.1)^{(25-20)} = 1.93 \text{gVSS/gVSS} \cdot \text{d}$$

$$k_{d,H} = k_{d,H,20℃}\,\theta_{k_{d,H}}^{T_{denitrification}-20} = (0.05g/g\cdot d)(1.04)^{(25-20)} = 0.06gVSS/gVSS\cdot d$$

$$C_{DO,\ denitrification} = 0.05\ g/m^3$$

2 g/m^3 is selected as effluent NO_3_N concentration

$$C_{NO_3\_N,6} = 2\ g/m^3$$

$$\mu_{max,\ H,\ F} = \mu_{max,\ H}\left[\frac{C_{NO_{34}\_N,6}}{C_{NO_3\_N,6} + K_{NO,\ H}}\right]\left[\frac{K_{DO,\ H}}{C_{DO,\ denitrification} + K_{DO,\ H}}\right] = 1.47gVSS/gVSS\cdot d$$

It was reported that the theoretical oxygen consumption ratio for NO_3^- ($OE_{NO_3\_N}$) is 2.86 mg O_2/mg NO_3_N [28].

$$\text{The net oxygen consumption ratio for } NO_3\_N, CR_{NO_3\_N} = \frac{OE_{NO_3\_N}}{1-1.42Y_{H,denitrification}} = 4.98gO_2/gNO_3\_N$$

$$NO_3\_N \text{ reduction rate by heterotrophs, } r_{NO_3\_N,F} = \frac{\mu_{max,H,F}}{CR_{NO_3\_N}}$$

It was reported that fraction of heterotrophic bacteria active for denitrification $f_{AX,}$ varies from 0.2 to 1 [28]. f_{AX} is assumed to be 0.9.

$$\text{Specific denitrification rate, } SDNR = \frac{\mu_{max,\ H,F} C_{bCOD,6}}{CR_{NO_3\_N} Y_{H,denitrification}\,(K_{s,H} + C_{bCOD,6})} f_{AX}$$
$$= 0.17gNO_3\_N/gVSS\cdot d$$

$$NO_x = 91.77\ g/m^3$$

$$C_{NO_3\_N,5} = \frac{W_{NO_3\_N,5}}{Q_5} = \frac{366.77\,kg/d}{3{,}636.78\,m^3/d} = 100.83g/m^3$$

$$C_{NO_3\_N,6} = 2\ g/m^3$$

$$W_{NO_3\_N,denitrfication} = (C_{NO_3\_N,5} + NO_x - C_{NO_3\_N,6})Q_5 = 693.17\,kg/d$$

$$V_{denitrification} = \frac{W_{NO_3\_N,denitrification}}{(C_{TSS,10})(SDNR)} = 1{,}026.01 m^3$$

$V_{\text{activated sludge process with nitrification and denitrification}} = 313.62\ m^3 + 1{,}026.01\ m^3 = 1{,}339.63\ m^3$

$$\text{The amount of organic consumption due to denitrification, } \Delta S_{r,NO_3\_N} = CR_{NO_3\_N} W_{NO_3\_N,denitrification}$$
$$= 3{,}453.77 kgCOD/d$$

$$C_{r,NO_3\_N} = \frac{\Delta S_{r,NO_3\_N}}{Q_5} = 949.68 gO_2/m^3$$

It was reported that return sludge ratio, R varies from 0.5 to 1 [28]. R is assumed to be 0.6.

The concentration of organic consumption due to deoxygenation, $C_{r,\ DO}$

$=C_{DO}(1+R)=(2gO^2/m^3)(1+0.6)= 3.2\ g\ O^2/m^3$

The concentration of organic consumption due to denitrification and deoxygenation,

$C_{r,\ AX}=949.68\ g\ O^2/m^3 + 3.2\ g\ O^2/m^3 = 952.88\ g\ O^2/m^3$

$CH_3OH + 1.5O_2 \rightarrow CO_2 + 2H_2O$

Based on the above equation, the equivalent oxygen for methanol,

$$O_{2,CH_3OH} \text{ is equal to } 1.5 mg\ O_2/mg\ CH_3OH$$

$$\text{The concentration of methanol required by the denitrification, } C_{CH_3OH}$$
$$= \frac{C_{r,NO_3\_N}}{O_{2,CH_3OH}} = 635.25 g/m$$

Methanol requirement $W_{CH_3OH} = C_{CH_3OH} Q_5 = (635.25\ gO_2/m^3)(3{,}636.78\ m^3/d)$

$= 2{,}310.27\ kg/d$

$NO_3^- + 1.08CH_3OH + 0.24CO_2 \rightarrow 0.47N_2 + 0.065\ C_5H_7NO_2 + 1.43\ H_2O + HCO_3^-$

Based on the above equation, 1kg CH_3OH could porduce 0.21 kg VSS

Mass of VSS, $W_{VSS,\ denitrification,\ 10}$=(0.21kgVSS/kg$CH_3$ OH) W_{CH_3OH} = 491.00 kg/d

Mass of TSS, $W_{TSS,\ 10}$=$W_{VSS,\ 10}$

=$W_{VSS,\ activated\ sludge\ process\ with\ nitrification,\ 10}$+$W_{VSS,\ denitrification,\ 10}$

= 262.61 kg/d+491.00 kg/d= 753.61 kg/d

$$Q_{10} = \frac{W_{TSS,10}}{C_{TSS,10}} = \frac{753.61\,kg/d}{4.006\,kg/m^3} = 188.12\,m^3/d$$

Q_6=Q_5-Q_{10}=3,636.78 m^3/d-188.12 m^3/d= 3,448.66 m^3/d

Mass of nbsCOD, $W_{nbsCOD,\ 6}$=$W_{nbsCOD,\ 5}$=5,933.71 kg/d

$$C_{nbsCOD,6} = \frac{W_{nbsCOD}}{Q_6} = \frac{5,933.71\,kg/d}{3,448.66\,m^3/d} = 1,720.58\,g/m^3$$

$C_{bCOD,\ 6}$=0.35g/m^3

$C_{COD,\ 6}$= $C_{nbsCOD,\ 6}$+ $C_{bCOD,\ 6}$=1,720.96 mg/L

Final clarifier

Table A.8 Design parameter values of the final clarifier

Parameter	Unit	Value	Reference
Vesilind parameter, k	L/g	0.35	[26]
Vesilind parameter, v_0	m/h	9	
Settler height H_d	m	4	
Settler safety factor s_{fd}	unitless	2	

$V_{final\ clarifier}$=s_{fd} (H_d/v_0)exp(k$C_{TSS,\ 10}$)Q_i

=(2)(4m/(9m/h))exp((0.35L/g)(4.006 g/L))(3,785.4 m^3/d)/(24h/d)=569.73 m^3

Activated Carbon Adsorption system

Table A.9 Design parameter values of the activated carbon adsorption system

Parameter	Value	Reference
COD/TOC	1.75	**[27]**
Freundlich capacity factor, K_f	150 (mg/g)(L/mg)$^{1/n}$	

Freundlich intensity parameter, 1/n	0.5	

$$C_{TOC,6} = \frac{C_{COD,6}}{1.75} = 983.40\,\text{mg/L}$$

Since system was designed to achieve the discharge standard (COD≤50 mg/L, BOD≤20 mg/L, and N≤4.9 mg/L) [92, 93].

40 mg/L is assumed to be effluent COD concentration.

$$C_{COD,\,e}=40\ \text{mg/L}=40\ \text{g/m}^3$$

$$C_{TOC,e} = \frac{C_{COD,e}}{1.75} = 22.86\,\text{mg/L}$$

$$\text{Activated Carbon dose concentration} = \frac{C_{TOC,6} - C_{TOC,e}}{K_f C_{TOC,e}^{\,1/n}}$$

$$= 1.34\,\text{g/L} = 1.34\,\text{kg/m}^3$$

Activated Carbon dose requirement=(Activated Carbon dose concentration) Q_6

$$=(1.34\ \text{kg/m}^3)(3{,}448.66\ \text{m}^3/\text{d})(2.205\text{lb/kg})=10{,}185.33\ \text{lb/d}$$

Sludge thickener

Table A.10 Design parameter values of the thickener

Parameter	Unit	Value	Reference
Vesilind parameter, k	**L/g**	**0.35**	**[26]**
Vesilind parameter, v_0	**m/h**	**9**	
Settler height H_{th}	**m**	**3**	
Settler safety factor s_{th}	**unitless**	**1.5**	
Thickened sludge concentration, $C_{TSS,\ \text{thickener sludge}}$	**kg/m^3**	**25**	
Solids recovery	**unitless**	**0.9**	**[28]**

$$\text{limiting sludge concentration } C_l$$

$$= \left(\frac{C_{TSS,\text{ thickener sludge}}}{2}\right)\left[1 + \left(1 - \frac{4}{kC_{TSS,\text{ thickener sludge}}}\right)^{0.5}\right]$$

$$= \left(\frac{25}{2}\right)\left[1 + \left(1 - \frac{4}{(0.35)(25)}\right)^{0.5}\right]$$

$$= 21.71 \text{kg/m}^3$$

limiting solids flux $F_l = C_{TSS,\text{ thickener sludge}}\, v_0\, (k \cdot C_1 - 1)\exp(-k \cdot C_1)$

$= 0.74 \text{kg/m}^2 \cdot \text{h} = 17.86 \text{kg/ m}^2 \cdot \text{d}$

$V_{\text{sludge thickener}} = H_{th}\, s_{th}\, W_{TSS,\ 10}/F_l = (3)(1.5)(753.61)/17.86 = 189.88 \text{m}^3$

Mass of TSS, $W_{TSS,\text{ thickener sludge}} = W_{VSS,\text{ thickener sludge}} = 0.9 W_{TSS,\ 10}$

$= 0.9(753.61)\text{ kg/d} = 678.25\text{ kg/d}$

Mass of VSS, $W_{VSS,\text{ thickener overflow}} = W_{VSS,\ 10} - W_{VSS,\text{ thickener sludge}}$

$= 753.61\text{kg/d} - 678.25\text{kg/d} = 75.36\text{ kg/d}$

Mass of COD, $W_{COD,\text{ thickener overflow}} = (1.42) W_{VSS,\text{ thickener overflow}}$

$= (1.42)(75.36\text{ kg/d}) = 107.01\text{ kg/d}$

Mass of N, $W_{N,\text{ thickener overflow}} = (0.12)\, W_{VSS,\text{ thickener overflow}}$

$= (0.12)(75.36\text{ kg/d}) = 9.04\text{ kg/d}$

$$\text{Thickener sludge flow } Q_{\text{thickenersludge}} = \frac{W_{\text{thickenersludge}}}{C_{\text{thickenersludge}}} = \frac{678.25\,\text{kg/d}}{25\,\text{kg/m}^3} = 27.13\,\text{m}^3/\text{d}$$

Thickener overflow $Q_{\text{thickener overflow}} = Q_{10} - Q_{\text{thickener sludge}}$

$= 188.12\text{ m}^3/\text{d} - 27.13\text{ m}^3/\text{d} = 160.99\text{ m}^3/\text{d}$

Anaerobic digestion

Side stream anaerobic digester converted sludge to CH_4 in the absence of air at a specific solids retention time and a specific temperature. Mean values of solids retention time and temperature shall be between 15 days at 35℃ to 55 ℃ and 60 days at 20℃ [134].

Kabouris et al. reported that the biodegradable volatile solids destruction value of anaerobic digester was 69% at a retention time of 12 days and 35℃ [287]. It was reported 70% of biodegradable COD could be converted to CH_4 at 35℃ [40]. As a result, retention time, temperature and biodegradable COD destruction value are assumed to be 15d, 35℃ and 70%.

$$RT_{AD}=15d$$

$$T_{AD}=35℃$$

$$Qt_{\text{total sludge to AD}}=Q_7+Q_8+Q_9+Q_{\text{thickener sludge}}$$

$$= 101.54\ m^3/d+39.04\ m^3/d+8.04\ m^3/d+27.13\ m^3/d = 175.75\ m^3/d$$

$$V_{AD}=Q_{\text{total sludge to AD}}\times RT_{AD} = 175.75\ m^3/d \times 15d = 2{,}636.22\ m^3/d$$

$$\text{Mass of VSS, } W_{\text{VSS, total sludge to AD}}=W_{\text{VSS, 7}}+W_{\text{VSS, 8}}+W_{\text{VSS, 9}}+W_{\text{VSS, thickener sludge}}$$

$$= 4{,}966.99\ kg/d+1{,}737.34\ kg/d+401.86\ kg/d+678.25\ kg/d = 7{,}784.44\ kg/d$$

$$\text{Mass of ISS, } W_{\text{ISS, total sludge to AD}}=W_{\text{ISS, 1}}=717.00\ kg/d$$

$$\text{Mass of TSS, } W_{\text{TSS, total sludge to AD}}=W_{\text{ISS, total sludge to AD}}+W_{\text{VSS, total sludge to AD}}$$

$$=717.00\ kg/d+7{,}859.80\ kg/d = 8{,}501.44\ kg/d$$

$$\text{Mass of pCOD, } W_{\text{pCOD, total sludge to AD}}=(1.42\ kg\ COD / kg\ VSS)W_{\text{VSS, total sludge to AD}}$$

$$=(1.42\ kg\ COD/kg\ VSS)(7{,}784.44\ kg/d) = 11{,}053.90\ kg/d$$

$$\text{Mass of nbpCOD, } W_{\text{nbpCOD, total sludge to AD}}=W_{\text{nbpCOD, 1}}=1{,}928.21\ kg/d$$

$$\text{Mass of bpCOD, } W_{\text{bpCOD, total sludge to AD}}=W_{\text{pCOD, total sludge to AD}}-W_{\text{nbpCOD, total sludge to AD}}$$

$$= 11{,}053.90\ kg/d-1{,}928.21\ kg/d = 9{,}125.69\ kg/d$$

$$\text{Mass of bVSS, } W_{\text{bVSS, total sludge to AD}}=W_{\text{bpCOD, total sludge to AD}}/1.42=6{,}426.54\ kg/d$$

$$CH_4+2O_2\rightarrow CO_2+2H_2O$$

1 kg COD digested could be converted to 0.25 kg CH_4 or 0.35m^3 CH_4 theoretically at 0℃ and 1 atm [28]

Mass of CH_4, $W_{CH_4,\ AD}$=(0.25)(0.7)$W_{\text{bpCOD, total sludge to AD}}$

=(0.25)(0.7)(9,125.69 kg/d)= 1,597.00 kg/d

Mass of VSS destroyed, $W_{\text{VSS, DG}}$=(0.7)$W_{\text{bVSS, total sludge to AD}}$= 4,498.58 kg/d

Mass of TSS remaining, $W_{\text{TSS, REM}}$=$W_{\text{TSS, total sludge to AD}}$- $W_{\text{VSS, DG}}$

= 8,501.44kg/d- 4,498.58 kg/d= 4,002.86 kg/d

As shown in table A.4, the specific gravity of the bulk sludge and solids content for anaerobic digestion sludge are 1.01-1.03 and 3-6%. 0.045 and 1.02 are assumed to be the corresponding solid content and the specific gravity of the anaerobic sludge. Therefore, the wet bulk sludge concentration of anaerobic sludge is equal to 45.90 kg/m^3. It was reported that the solid concentration in supernatant is 2000-15000 mg/L [28]. 5000 mg/L is assumed as the solid concentration in supernatant in this study.

$$Q_{\text{total sludge to AD}} = \frac{W_{\text{TSS, REM}} - W_{\text{TSS, Supernatant}}}{45.9\ \text{kg/m}^3} + \frac{W_{\text{TSS, Supernatant}}}{5\ \text{kg/m}^3}$$

$$175.75\ \text{m}^3/\text{d} = \frac{4{,}002.86\ \text{kg/d} - W_{\text{TSS, Supernatant}}}{45.9\ \text{kg/m}^3} + \frac{W_{\text{TSS, Supernatant}}}{5\ \text{kg/m}^3}$$

$W_{\text{VSS, Supernatant}}$= 496.82 kg/d

$W_{\text{COD, Supernatant}}$=(1.42)496.82 kg/d= 705.48 kg/d

$W_{\text{N, Supernatant}}$=(0.12) 496.82 kg/d= 59.62kg/d

$Q_{\text{Supernatant}}$= (496.82 kg/d)/(5 kg/ m^3)=99.36 m^3/d

$Q_{\text{sludge to dewatering}}$= 175.75 m^3/d-99.36 m^3/d= 76.38 m^3/d

$W_{\text{TSS, sludge to dewatering}}$= 4,002.86 kg/d-496.82 kg/d=3,506.04 kg/d

Sludge dewatering

Table A.11 Design parameter values of the Sludge dewatering

Parameter	Unit	Value	Reference
Solids Content	%	22	**[27]**
The specific gravity of the bulk sludge	unitless	1.06	
Solids capture	%	93	

$$W_{TSS,\ sludge\ cake}=(0.93)\ \ 3{,}506.04\ \ kg/d=(0.93)\ \ 3{,}506.04\ \ kg/d=\ 3{,}260.61 kg/d$$

$$Q_{sludge\ cake}\ =\frac{3{,}260.61 kg/d}{1.06(0.22)(1000 kg/m^3)}=\ 13.98\, m^3/d$$

$$Q_{centrate}=Q_{sludge\ to\ dewatering}-Q_{sludge\ cake}=76.38\ m^3/d-13.98\ m^3/d\ =\ 62.40\ m^3/d$$

$$W_{VSS,\ \ centrate}=W_{TSS,\ sludge\ to\ dewatering}-W_{TSS,\ sludge\ cake}$$

$$=3{,}506.04\ \ kg/d-3{,}260.61 kg/d=\ 245.42\ kg/d$$

$$W_{COD,centrate}=(1.42)\ 245.42\ \ kg/d=\ \ 348.50\ \ \ kg/d$$

$$W_{N,centrate}=(0.12)\ \ 245.42\ \ kg/d=\ 29.45\ kg/d$$

$$Q_{combined\ sidestream}=Q_{thickener\ overflow}+Q_{Supernatant}+Q_{centrate}$$

$$=160.99\ m^3/d+99.36\ m^3/d\ +\ 62.40\ m^3/d\ =\ 322.76\ m^3/d$$

$$W_{VSS,\ \ combined\ sidestream}=W_{VSS,thickener\ overflow}+W_{VSS,Supernatant}+W_{VSS,centrate}=\ 817.60 kg/d$$

$$W_{COD,\ \ combined\ sidestream}=W_{COD,thickener\ overflow}+W_{COD,Supernatant}+W_{COD,centrate}=\ 1{,}160.99 kg/d$$

$$W_{N,\ \ combined\ sidestream}=W_{N,thickener\ overflow}+W_{N,Supernatant}+W_{N,centrate}=98.11 kg/d$$

Since the combined sidestream is returned to the UASB, the calculations in the first iteration will remain the same for preliminary and primary treatment. Other calculations need to be revised. The above computational procedure was repeated two more iterations by a spreadsheet program. After the fourth iteration, stable flow and mass loadings were

obtained. The final results are summarized in Table A.12. We could determine energy and cost based on final iteration.

Table A.12 Final parameter values for estimating energy and cost

Iteration	1.00	2.00	final
Oxygen requirement (kg/d)	8,031.30	8,230.30	8,234.00
Mass of CH_4 (kg/d)	6,332.19	6,612.02	6,617.19
Excess sludge production (kg/d)	3,260.61	3,280.58	3,280.94
V_{UASB} (m^3)	2,342.58	2,446.91	2,448.84
$V_{PN/A}$ (m^3)	4,767.93	4,960.63	4,964.66
$V_{\text{activated sludge process with nitrification and denitrification}}$($m^3$)	1,339.63	1,375.17	1,375.77
Methanol requirement W_{CH_3OH}(kg/d)	2,310.27	2,362.95	2,363.94
Q_6 (m^3/d)	3,448.66	3,764.10	3,771.33
Activated carbon dose requirement (lb/d)	10,185.33	10,163.36	10,162.86
$V_{\text{sludge thickener}}$ (m^3)	189.88	195.31	195.40
V_{AD} (m^3)	2,636.22	2,676.85	2,677.57
$Q_{\text{sludge to dewatering}}$ (m^3/d)	76.38	76.85	76.86

Energy

Energy consumption

Table A.13 Equations for calculating the energy consumption

Unit energy metrics	Related equations	Reference
Energy consumption	Energy consumption of areation process (kWh/d) = (Aeration demand (kgO_2/d))/OT_a OT_a = Oxygen transfer efficiency (actual) = 1.2kgO_2/kWh	[26]

$$\text{Oxygen requirement} = \text{aeration demand} = W_{O_2,\text{PN/A}} + W_{O_2,\text{activated sludge process with nitrification}}$$
$$= 8{,}234.00\ \text{kg}\ O_2/\text{d}$$

$$\text{Energy consumption of areation process}$$
$$= \frac{\text{Aeration demand}\ (\text{kg}O_2/\text{d})}{OT_a} = \frac{8234.00\ \text{kg}O_2/\text{d}}{1.2\ \text{kg}\ O_2/\text{kWh}}$$
$$= 6{,}861.67\ \text{kWh/d} = 2{,}504{,}509.32\ \text{kWh/yr}$$

Energy production

Table A.14 Equations for calculating the energy production

Unit energy metrics	Related equations	Reference
Energy production	Energy production(kWh/d) = (Electricity recovery) (Enthalpy of combustion) (CH_4 production from UASB and anaerobic digester (kg/d)) Electricity recovery=38% Enthalpy of combustion=13.9 kWh/kg CH_4	[34, 38-40]

Mass of CH_4=CH_4 production from UASB and anaerobic digester= 6,617.19 kg/d

$$\text{Energy production(kWh/d)} = (\text{Energy recovery})\,(\text{Enthalpy of combustion})\,(\text{CH}_4 \text{ production from UASB and anaerobic digester (kg/d)})$$
$$= (38\%)(13.9\text{kWh/kg})(6,617.19\,\text{kg/d}) = 34,951.98\,\text{kWh/d}$$
$$= 12,757,470.91\ \text{kWh/yr}$$

$$\text{Ratio between electricity prodcution and electricity consumption} = \frac{\text{Electricity prodcution}}{\text{Electricity consumption}}$$
$$= \frac{12,757,470.91\ \text{kWh/yr}}{2,504,509.32\,\text{kWh/yr}} = 5.09$$

Cost

Preliminary treatment.

Table A.15 Equations for calculating cost of preliminary treatment.

Unit cost metrics	Related equations	Reference
Preliminary treatment	Capital cost of preliminary treatment($)= EXP [3.25972 +0.61915 x]1000 OHRS = EXP [6.39872 + 0.23096 x + 0.16496 x^2 - 0.0146 x^3] XMHRS = EXP [5.8461+ 0.20651 x + 0.06884 x^2 + 0.02382 x^3 - 0.00441 x^4] TMSU =EXP [7.23566 + 0.39994 x - 0.22498 x^2 + 0.1101 x^3 - 0.01103 x^4] EERG = EXP [6.30864 + 0.23453 x - 0.35844 x^2 + 0.00871x^3] EERMS = EXP [7.1497 + 0.28856 x - 0.07886 x^2 + 0.014662 x^3] EER = EERG + EERMS where x=In(Q_i) Q_i =flow to the treatment plant, MGD, OHRS=operation man-hour requirements, hr/yr, XMHRS= maintenance man-hour requirements, hr/yr, TMSU=total material and supply cost, $/yr, EERG=electrical energy required for grit removal, kWh/yr EERMS= electrical energy required for flow measurements and screening, kWh/yr, EER=total electrical energy required, kWh/yr .	[31]

Q_i=3785.4 m^3/d=1 MGD

Capital cost of preliminary treatment($)= EXP [3.25972 +0.61915 ln(1)]1000=26042.24$

Costs are based on 1986 cost data. The cost of preliminary treatment is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

1986 ENR CCI=4295.00

$$\text{Capital cost of preliminary treatment}(\$) = (26042.24\$)\ \frac{\text{November 2010 ENR CCI}}{\text{1986 ENR CCI}}$$

$$= (26042.24\$)\ \frac{11579.02}{4295.00} = 70208.07\$$$

$$OHRS = EXP\ [6.39872 + 0.23096\ \ln(1)\ + 0.16496\ \ln(1)^2 - 0.0146\ \ln(1)^3] = 601.08\ \text{hr/yr}$$

$$XMHRS = EXP\ [5.8461 + 0.20651\ \ln(1) + 0.06884\ \ln(1)^2 + 0.02382\ \ln(1)^3 - 0.00441\ \ln(1)^4] = 345.88\ \text{hr/yr}$$

$$TMSU = EXP\ [7.23566 + 0.39994\ \ln(1) - 0.22498\ \ln(1)^2 + 0.1101\ \ln(1)^3 - 0.01103\ \ln(1)^4] = 1388.06\ \$/\text{yr}$$

$$EERG = EXP\ [6.30864 + 0.23453\ \ln(1) - 0.35844\ \ln(1)^2 + 0.00871\ln(1)^3] = 549.30\ \text{kWh/yr}$$

$$EERMS = EXP\ [7.1497 + 0.28856\ \ln(1) - 0.07886\ \ln(1)^2 + 0.014662\ \ln(1)^3] = 1273.72\ \text{kWh/yr}$$

The updated November, 2020 unit price of electricity, and labor are 0.1045 $/kWh and 59.31$/hr, respectively [288, 289].

$$\text{O \& M cost of preliminary treatment}$$

$$= (601.08\,\text{hr/yr} + 345.88\,\text{hr/yr})(59.31\$/\text{hr}) + (1388.06\$/\text{yr})\frac{11579.02}{4295.00}$$
$$+ (549.30\,\text{kWh/yr} + 1273.72\,\text{kWh/yr})(0.1045\$/\text{kWh})$$
$$= 60{,}096.69\$/\text{yr}$$

Primary treatment

Table A.16 Equations for calculating cost of primary treatment.

Unit cost metrics	Related equations	Reference
Micro sieving	Capital cost of rotating belt filter ($)= 0.55 Capital cost of primary clarifier with 50% TSS removal($) Capital cost of priamry clarifier=-0.00002(Q)^2+19.29(Q)+220,389 Q=32.6/surface overflow rate (influent flow rate) TSS removal of primary clarifier = -0.004006[Surface overflow rate(m/d)] +0.73782 Operating Power Consumption =129.6 kWh/d Labor hour =216 hour/yr Maintenance Materials cost ($/yr) = 2% of capital costs	[1, 17, 30, 35, 37]

Capital cost of rotating belt filter ($)= 0.55 Capital cost of priamry clarifier with 50%

TSS removal($)

TSS removal of primary clarifier = -0.004006[Surface overflow rate(m/d)] +0.73782

$$\text{Surface overflow rate} = \frac{0.73782 - \text{TSS removal of primary clarifier}}{0.004006}$$
$$= \frac{0.73782 - 0.5}{0.004006} = 59.37\text{m/d}$$

$$Q = \frac{(32.6)(\text{influent flow rate})}{\text{surface overflow rate}} = \frac{(32.6)(Q_i)}{\text{surface overflow rate}}$$
$$= \frac{(32.6)3785.4}{59.37} = 2{,}078.70\,\text{m}^3/\text{d}$$

Capital cost of primary clarifier=-0.00002(Q)^2+19.29(Q)+220,389

=-0.00002(2,078.70)^2+19.29(2,078.70)+220,389= 260,400.72 $

Costs are based on 1996 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

1996 ENR CCI=5620.00

$$\text{Capital cost of primary clarifier}(\$) = (260{,}400.72\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{1996 ENR CCI}}$$

$$= (260{,}400.72\,\$)\ \frac{11579.02}{5620.00} = 536{,}509.80\$$$

Capital cost of rotating belt filter = 0.55 Capital cost of primary clarifier with 50% TSS removal($)= 0.55 ×536,509.80 $= 295,080.39$

Operating Power Consumption =129.6 kWh/d

Labor hour =216 hour/yr

Maintenance Materials cost ($/yr) = 2% of capital costs=(0.02)(295,080.39$)= 5,901.61 $/yr

The updated November, 2020 unit price of electricity, and labor are 0.1045 $/kWh and 59.31$/hr, respectively [288, 289].

O&M cost of rotating belt filter =(216 hour/yr)(59.31$/hr)+(5,901.61 $/yr)+(129.6 kWh/d)(365d/yr)(0.1045 $/kWh) = 23,655.84 $/yr

UASB

van Haandel and van der Lubbe [26] summarized the unit capital cost and cost factors in Table A.17. Flow rates were estimated based on capacity (P.E.) and wastewater production of 120 L/P.E./d [290, 291].

Based on the average values in table A.17, regression models of capital cost and cost factors are developed as shown in Figure A.6, A.7, and A.8.

Table A.17 Costing parameters for different WWTP sizes

Capacity (P.E)	**25000**			**50000**			**100000**			**200000**		
Flow rate (m^3/d)	3000			6000			12000			24000		
Unit capital cost ($/$m^3$ volume)	min	max	mean	min	max	mean	min	max	mean	min	max	mean
UASB	600	1000	800	500	700	600	350	500	425	250	400	325
Activated sludge tank	220	300	260	180	250	215	150	200	175	120	170	145
Final clarifier	350	550	450	300	400	350	250	330	290	200	260	230
Sludge thickener	700	1000	850	500	800	650	300	500	400	250	400	325
Anaerobic digester	600	1000	800	450	700	575	300	400	350	250	350	300
Capital cost (US$/kW):	min	max	mean	min	max	Mean	Min	Max	mean	min	max	mean
Aeration equipment	4500	7000	5750	4000	5200	4600	3200	4000	3600	2800	3500	3150
Cost factors	min	max	mean	min	max	mean	min	max	mean	min	max	mean
f_{ac}	1.4	1.5	1.45	1.35	1.45	1.4	1.3	1.4	1.35	1.25	1.35	1.3
f_i	1.6	1.9	1.75	1.5	1.8	1.65	1.5	1.7	1.6	1.4	1.6	1.5

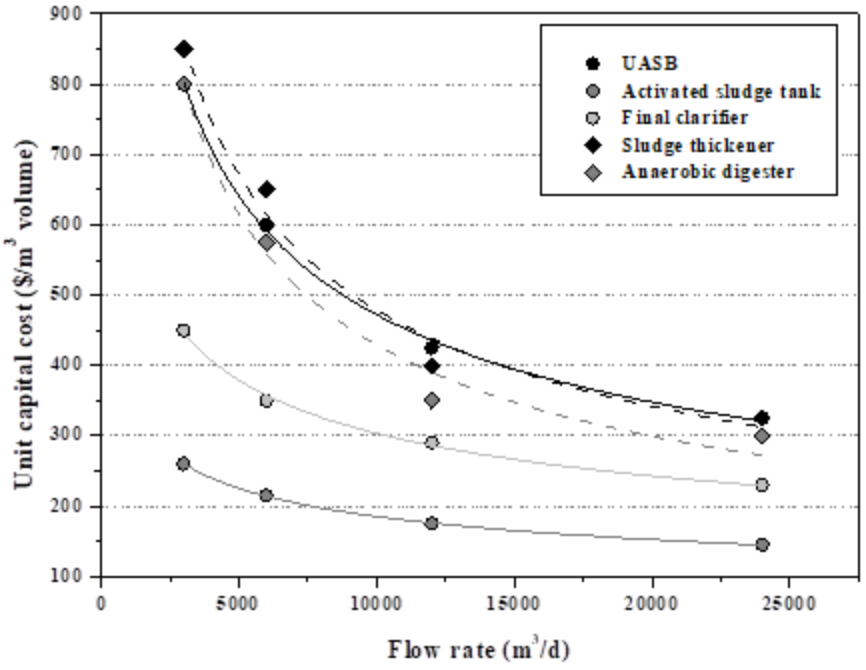


Figure A.6 Capital cost of UASB, aeration tank, final clarifier, sludge thickener and anaerobic digester

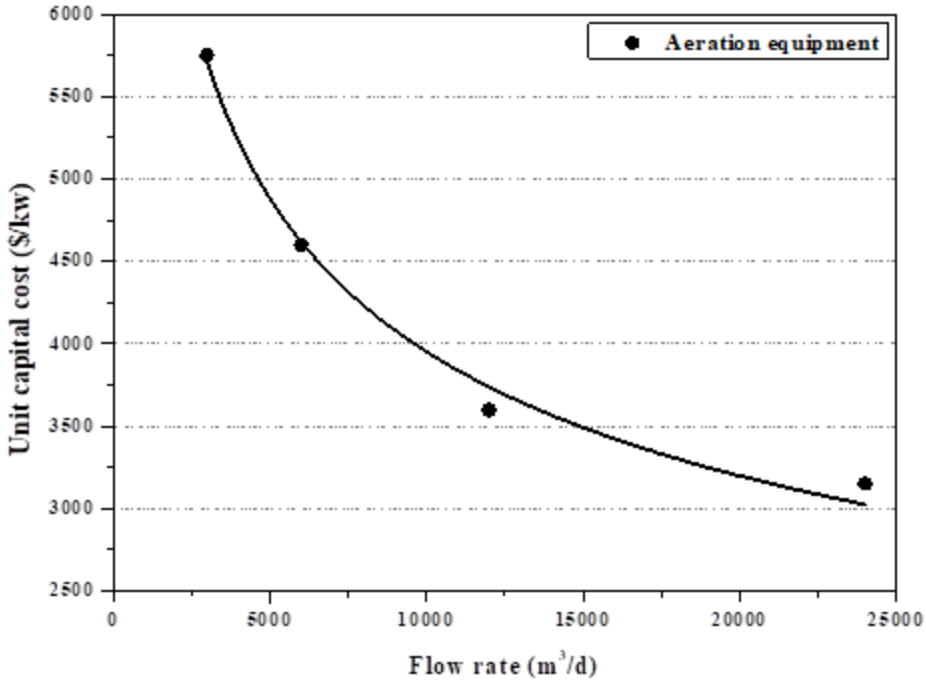


Figure A.7 Capital cost of aeration equipment

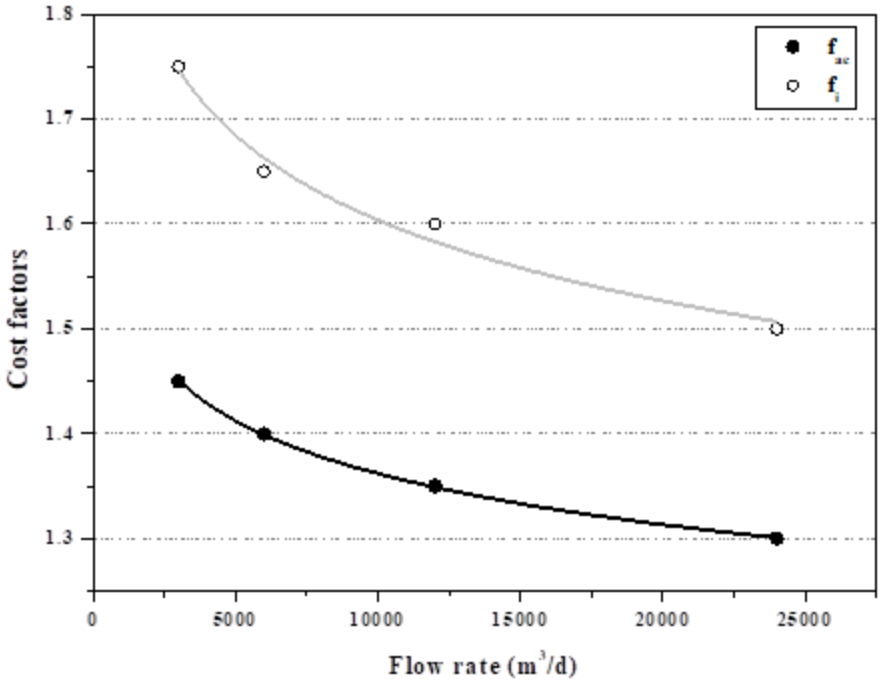


Figure A.8 Cost factors

The following equations could be used for predicting capital cost and cost factors:

Unit capital cost of UASB ($/m^3 volume) =27070(Flow rate (m^3/d))$^{-0.44}$ R^2=0.9979

(A.1)

Unit capital cost of activated sludge tank ($/m^3 volume) =2497.8(Flow rate (m^3/d))$^{-0.282}$ R^2=0.9997

(A.2)

Unit capital cost of final clarifier ($/m^3 volume) =5664.5(Flow rate (m^3/d))$^{-0.318}$ R^2=0.9973

(A.3)

Unit capital cost of sludge thickener ($/$m^3$ volume) =42072(Flow rate (m^3/d))$^{-0.486}$ R^2=0.9773

(A.4)

Unit capital cost of anaerobic digester ($/$m^3$ volume) =41699(Flow rate (m^3/d))$^{-0.496}$ R^2=0.9662

(A.5)

Unit capital cost of aeration equipment ($/kW) =60459(Flow rate ($m^3$/d))$^{-0.296}$ R^2=0.9864

(A.6)

Cost factors, f_{ac} =2.2092(Flow rate (m^3/d))$^{-0.053}$ R^2=0.9997

(A.7)

Cost factors, f_i=3.0884(Flow rate (m^3/d))$^{-0.071}$ R^2=0.9839

(A.8)

Flow rate =Q_i=1MGD=3,785.4 m^3/d

Unit capital cost of UASB =27070(Flow rate (m^3/d))$^{-0.44}$=27070(3,785.4 m^3/d)$^{-0.44}$=721.31$/$m^3$ volume

Unit capital cost of activated sludge tank =2497.8(Flow rate (m^3/d))$^{-0.282}$=2497.8(3,785.4 m^3/d)$^{-0.282}$=244.64 $/$m^3$ volume

Unit capital cost of final clarifier ($/$m^3$ volume) =5664.5(Flow rate (m^3/d))$^{-0.318}$=5664.5(3,785.4 m^3/d)$^{-0.318}$=412.40 $/$m^3$ volume

Unit capital cost of sludge thickener ($/$m^3$ volume) =42072(Flow rate (m^3/d))$^{-0.486}$=42072(3,785.4 m^3/d)$^{-0.486}$=767.42 $/$m^3$ volume

Unit capital cost of anaerobic digester ($/$m^3$ volume) =41699(Flow rate (m^3/d))$^{-0.496}$=41699(3,785.4 m^3/d)$^{-0.496}$=700.46 $/$m^3$ volume

Unit capital cost of aeration equipment ($/kW) =60459(Flow rate ($m^3$/d))$^{-0.296}$=60459(3,785.4 m^3/d)$^{-0.296}$=5276.43 kW

Cost factors, f_{ac} =2.2092(Flow rate (m^3/d))$^{-0.053}$=2.2092(3,785.4 m^3/d)$^{-0.053}$=1.43

Cost factors, f_i=3.0884(Flow rate (m^3/d))$^{-0.071}$=3.0884(3,785.4 m^3/d)$^{-0.071}$=1.72

Table A.18 Equations for calculating cost of UASB.

Unit cost metrics	Related equations	Reference
Mainstream UASB	Capital cost of UASB($)=(costing factors)(Volume of UASB (m^3)) (unit capital cost of UASB($/$m^3$)) Volume of UASB ($m^3$) =(sludge production from UASB(m^3/d))(sludge age(d)) O&M cost (US$/year)=(p+o+n+ 0.3$m_{civ}$+0.7 $m_{me\&i}$)Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26]

The capital cost of UASB($)=(cost factors)(Volume of UASB (m^3))

(unit capital cost of UASB($/$m^3$))=$f_{ac}$ f_i V_{UASB} (Unit capital cost of UASB

)=(1.43)(1.72)(2,448.84 m^3)(721.31$/ m^3 volume)= 4,338,325.70 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

$$\text{Capital cost of UASB}(\$) = (4{,}338{,}325.70\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2006 ENR CCI}}$$

$$= (4{,}338{,}325.70\,\$)\ \frac{11579.02}{7749.37} = 6{,}482{,}280.11\$$$

$$\text{O\&M cost of UASB} = (p+o+n+\ 0.3m_{civ}+0.7\ m_{me\&i}\ )\text{Capital cost of UASB}$$

$$=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(\ 6{,}482{,}280.11\ )$$

$$=\ 405{,}142.51\ \$/yr$$

PN/A

Based on the data in table A.19, regression models of equipment cost and are developed as shown in Figure A.9.

Table A.19 Equipment cost of SBR reactor

Flow rate (MGD)	Reactors			Equipment cost ($)	Reference
	Volume of a reactor (MG)	Number of reactors	Total volume (MG)		
0.012	0.021	1	0.021	94000	[32]
1	0.479	2	0.958	339000	
1.4	0.678	2	1.356	405000	
1.46	0.91	2	1.82	405000	
2	0.958	2	1.916	564000	
4.25	1.556	4	6.224	1170000	

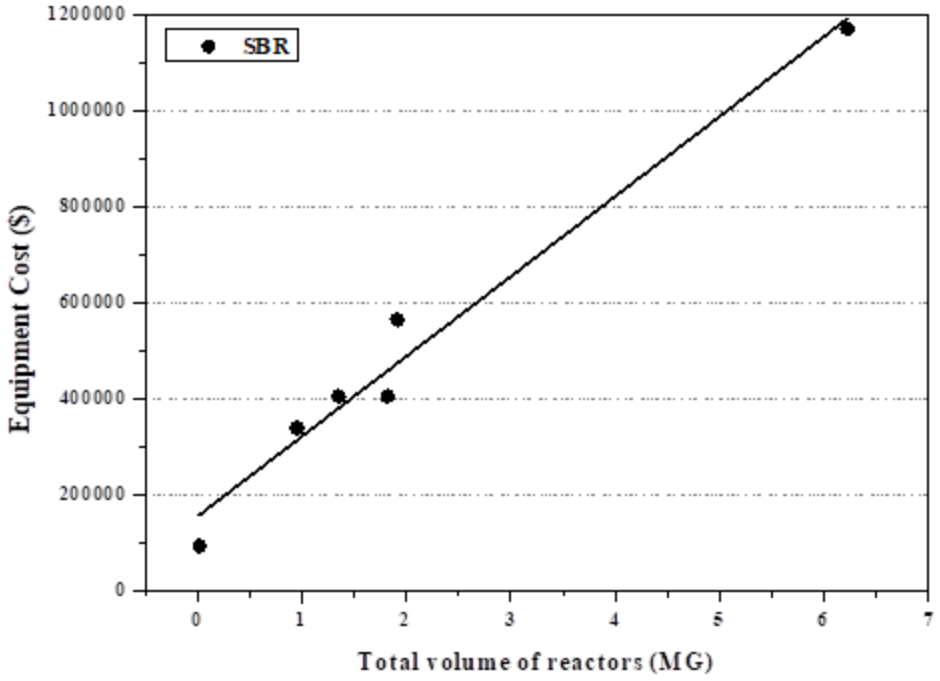


Figure A.9 Equipment cost of SBR

The following equation could be used for predicting equipment cost of SBR:

Equipment Cost of SBR ($) =(Volume of SBR(MG))166438+155108 R^2=0.9839 (A.9)

Table A.20 Equations for calculating cost of partial nitrification/anammox.

Unit cost metrics	Related equations	Reference
Partial nitrification /Anammox	Capital cost of PN/A($) =Equipment Cost of PN/A SBR($) (Capital cost⁄Equipment Cost) Capital cost/Equipment Cost=2.405 Equipment Cost of PN/A($) =(Volume of PN/A(MG))166438+155108 O&M cost (US$/year)=(p+o+n+ 0.3$m_{civ}$+0.7 $m_{me\&i}$)Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26, 29, 32]

$V_{PN/A}$= 4,964.66 m^3/d=1.31 MG

Equipment Cost of PN/A SBR($) =(Volume of PN/A(MG))166438+155108

=(1.31)166438+155108= 373,219.15$

Capital cost of PN/A($) =Equipment Cost of PN/A($)(Capital cost/Equipment Cost)

=(2.405)(373,219.15$)= 898,017.68 $

Costs are based on 1998 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

1998 ENR CCI=5920

$$\text{Capital cost of PN/A(\$)} = (898{,}017.68\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{1998 ENR CCI}}$$

$$= (898{,}017.68\,\$)\ \frac{11579.02}{5920} = 1{,}756{,}446.73\,\$$$

O&M cost of PN/A=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost of PN/A

=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(1,756,446.73)= 109,777.92 $/yr

Activated sludge system with nitrification, and denitrification

Table A.21 Equations for calculating cost of activated sludge system with nitrification, and denitrification

Unit cost metrics	Related equations	Reference
Activated sludge process with nitrification, and denitrification	Capital cost of activated sludge process with nitrification, and denitrification($)= (costing factors)(Volume of activated sludge process with nitrification, and denitrification (m^3)) (unit capital cost of activated sludge tank tank ($/$m^3$)) O&M cost (US$/year)=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost O&M cost of CH_3OH(US$/yr)=(Price of CH_3OH($/1000kg))(Methanol requirement (kg/d))(365d/yr) where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075	[26, 292]

	$m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175 Price of CH_3OH=379$/1000kg	

$V_{\text{activated sludge process with nitrification and denitrification}} = 1{,}375.77\ m^3$

Capital cost of activated sludge process with nitrification and denitrification($)

=(costing factors)(Volume of activated sludge process with nitrification and denitrification (m^3))(unit capital cost of activated sludge tank ($/$m^3$))

$=f_{ac}\ f_i\ V_{\text{activated sludge process with nitrification and denitrification}}$ (Unit capital cost of activated sludge tank)=(1.43)(1.72)(1,375.77 m^3)(244.64 $/ m^3 volume)= 826,645.82 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

Capital cost of activated sludge process with nitrification and denitrification ($)

$$= (826{,}645.82\ \$)\ \frac{\text{November 2010 ENR CCI}}{\text{2006 ENR CCI}}$$

$$= (826{,}645.82\ \$)\ \frac{11579.02}{7749.37} = 1{,}235{,}165.39\ \$$$

O&M cost of activated sludge process with nitrification and denitrification

$=(p+o+n+0.3m_{civ}+0.7\ m_{me\&i})$Capital cost of activated sludge process with nitrification and denitrification=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(1,235,165.39)

= 77,197.84 $/yr

$$\text{O \& M cost of } CH_3OH(\text{US\$/yr}) = (\text{Price of } CH_3OH(\$/1000\text{kg}))(\text{Methanol requirement } W_{CH_3OH})\ (\text{kg/d}))(365\text{d/yr})$$

$$= (379\$/1000\text{kg})(2{,}363.94\ \text{kg/d})(365\text{d/yr})$$

$$= 327{,}015.17\ \$/\text{yr}$$

Final clarifier

Table A.22 Equations for calculating cost of final clarifier

Unit cost metrics	Related equations	Reference
Final clarifier	Capital cost of final clarifier($)=(costing factors)(volune of final clarifier(m^3)) (unit capital cost of final clarifier($/$m^3$)) O&M cost (US$/year)=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26]

$$V_{\text{final clarifier}}=569.73 m^3$$

Capital cost of final clarifier($)=(costing factors)(Volume of final clarifier (m^3))(unit capital cost of final clarifier ($/$m^3$))=$f_{ac}f_i$ V $_{\text{final clarifier}}$ (Unit capital cost of final clarifier)=(1.43)(1.72)(569.73 m^3)(412.40$/ m^3 volume)= 577,082.28 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

$$\text{Capital cost of final clarifier (\$)} = (577{,}082.28\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2006 ENR CCI}}$$

$$= (577{,}082.28\,\$)\ \frac{11579.02}{7749.37} = 862{,}270.21\$$$

O&M cost of final clarifier=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost of final clarifier=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(862,270.21)= 53,891.89 $/yr

Activated Carbon Adsorption system

Table A.23 Equations for calculating cost of activated carbon adsorption system

Unit cost metrics	Related equations	Reference
Activated Carbon System	Capital cost of Package Granular Activated Carbon Columns($) =(1+(Miscellaneous costs)/(construction cost))(0.0084 x^3 – 5.2233 x^2 + 1318.4 x +27144) (Miscellaneous costs)/(construction cost)=0.28 x = plant flow to activated Carbon System(gpm) Capital cost of Granular Carbon Regeneration ($)=(1+(Miscellaneous costs)/(construction cost))(2E–8x^3 – 0.0018 x^2 + 93.965 x+2000000) x = regeneration capacity (lb/day) O&M cost of Package Granular Activated Carbon Columns($/yr)=(0.0005 x^3 – 0.3763 x^2 + 140.13 x +4959) x = plant flow (gpm) O&M cost of Granular Carbon Regeneration ($/yr)=15.503 x + 128481 O&M cost of virgin Carbon ($/yr)=(Activated carbon loss)(365d/y)(Cost of virgin carbon) x Activated carbon loss=6% Cost of virgin carbon=1.2$/lb x = regeneration capacity (lb/day)	[36, 43]

Activated Carbon dose requirement=regeneration capacity= 10,162.86 lb/d

Q_6=3771.33 m^3/d=691.86gpm

Since the maximum limit of plant flow to activated carbon system is 350 gpm, number of package granular activated carbon columns is 2.

$$x = \frac{691.86\text{gpm}}{2} = 345.93 \text{ gpm}$$

Capital cost of package Granular Activated Carbon Columns($)=(number of package Granular Activated Carbon Columns)(1+(Miscellaneous costs)/(construction cost))(0.0084 x^3 – 5.2233 x^2 + 1318.4 x +27144)

=(2)(1+0.28)(0.0084 (345.93)^3 – 5.2233 (345.93)^2 + 1318.4 (345.93) +27144)

= 527,081.60 $

Capital cost of Granular Carbon Regeneration ($)=(1+(Miscellaneous costs)/(construction cost))(2E–8x^3 – 0.0018 x^2 + 93.965 x +2000000)

=(1+0.28)(2E–8(10,162.86)^3 – 0.0018(10,162.86)^2 + 93.965(10,162.86) +2000000)

$$= 3{,}571{,}245.24\ \$$$

Costs are based on 2009 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2009 ENR CCI=8585.71

$$\text{Capital cost of package Granular Activated Carbon Columns (\$)} = (527{,}081.60\ \$)\ \frac{\text{November 2010 ENR CCI}}{\text{2009 ENR CCI}}$$

$$= (527{,}081.60\ \$)\ \frac{11579.02}{8585.71} = 710{,}842.60\ \$$$

$$\text{Capital cost of Granular Carbon Regeneration (\$)} = (3{,}571{,}245.24\ \$)\ \frac{\text{November 2010 ENR CCI}}{\text{2009 ENR CCI}}$$

$$= (3{,}571{,}245.24\ \$)\ \frac{11579.02}{8585.71} = 4{,}816{,}319.22\ \$$$

Capital cost of activated carbon system ($)=710,842.60 $+4,816,319.22 $=5,527,161.82$

O&M cost of Package Granular Activated Carbon Columns($/yr)=(number of package Granular Activated Carbon Columns)$(0.0005\ x^3 - 0.3763\ x^2 + 140.13\ x + 4959)$

$=(2)(0.0005\ (345.93)^3 - 0.3763\ (345.93)^2 + 140.13\ (345.93) + 4959) = 58{,}203.34\ \$/yr$

O&M cost of Granular Carbon Regeneration ($/yr)=15.503 x + 128481

=15.503(10,162.86) + 128481=286,035.76 $/yr

The 2009 unit price of electricity, natural gas and labor are 0.0981 $/kWh, 0.00898 and 45.82 $/hr, respectively. The updated November, 2020 unit price of electricity, natural gas and labor are 0.1045 $/kWh ,0.00766 $/scf and 59.31$/hr, respectively [288, 289]. Component percent of electricity, labor and maintenance materials are 23%, 49% and 28% for Package Granular Activated Carbon Columns. Costs are based on 2009 cost

data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

$$\text{O \& M cost of Package Granular Activated Carbon Columns(\$/yr)}$$
$$= (58{,}203.34\$/\text{yr})(0.23)\ \frac{0.1045}{0.0981} + (58{,}203.34\$/\text{yr})(0.49)\ \frac{59.13}{45.82}$$
$$+ (58{,}203.34\$/\text{yr})(0.28)\frac{11579.02}{8585.71} = 73{,}154.98\$/\text{yr}$$

Component percent of electricity, natural gas, labor and maintenance materials are 9%, 42%, 36% and 13% for Granular Carbon Regeneration.

$$\text{O \& M cost of Granular Carbon Regeneration(\$/yr)}$$
$$= (286{,}035.76\$/\text{yr})(0.09)\ \frac{0.1045}{0.0981} + (286{,}035.76\$/\text{yr})(0.42)\ \frac{0.0766}{0.0898}$$
$$+ (286{,}035.76\$/\text{yr})(0.36)\ \frac{59.13}{45.82} + (286{,}035.76\$/\text{yr})(0.13)\frac{11579.02}{8585.71}$$
$$= 313{,}336.74\ \$/\text{yr}$$

O&M cost of virgin Carbon ($/yr)=(Activated carbon loss)(365d/y)(Cost of virgin carbon) x=(0.06)(365d/y)(1.2$/lb) (10,162.86 lb/d)= 267,079.86 $/yr

O&M cost of activated carbon system ($/yr)

=73,056.35$/yr+313,345.28 $/yr+267,093.08 $/yr=653,571.58 $/yr

Sludge thickener

Table A.24 Equations for calculating cost of sludge thickener

Unit cost metrics	Related equations	Reference
Sludge thickener	Capital cost of sludge thickener($)=(costing factors)(volune of sludge thickener(m^3)) (unit capital cost of sludge thickener($/$m^3$)) O&M cost (US$/year)=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26]

$V_{\text{sludge thickener}} = 195.40 m^3$

Capital cost of sludge thickener ($)=(costing factors)(Volume of sludge thickener(m^3)) (unit capital cost of sludge thickener ($/$m^3$))

$= f_{ac}\ f_i\ V_{\text{sludge thickener}}$ (Unit capital cost of sludge thickener)=(1.43)(1.72)(195.40 m^3)(767.42$/ m^3 volume)= 368,304.04 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

$$\text{Capital cost of sludge thickener } (\$) = (368{,}304.04\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2009 ENR CCI}}$$

$$= (368{,}304.04\,\$)\ \frac{11579.02}{7749.37} = 550{,}315.98\ \$$$

O&M cost of sludge thickener=$(p+o+n+ 0.3m_{civ}+0.7\ m_{me\&i})$Capital cost of sludge thickener=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(550,065.07)= 34,394.75 $/yr

Anaerobic digestion

Table A.25 Equations for calculating cost of anaerobic digestion

Unit cost metrics	Related equations	Reference
Anaerobic digestion	Capital cost of anaerobic digestion($)=(costing factors)(volune of anaerobic digestion(m^3)) (unit capital cost of anaerobic digestion($/$m^3$)) O&M cost (US$/year)=$(p+o+n+ 0.3m_{civ}+0.7\ m_{me\&i})$Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26]

$V_{AD} = 2{,}677.57 m^3$

Capital cost of anaerobic digestion ($)=(costing factors)(Volume of anaerobic digestion(m^3)) (unit capital cost of anaerobic digestion ($/$m^3$))=$f_{ac}$ f_i V_{AD} (Unit capital cost of anaerobic digestion)=(1.43)(1.72)(2,677.57 m^3)(700.46$/ m^3volume)

= 4,606,458.17 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

Capital cost of anaerobic digestion ($)

$$= (4{,}606{,}458.17\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2009 ENR CCI}}$$

$$= (4{,}606{,}458.17\,\$)\ \frac{11579.02}{7749.37} = 6{,}882{,}920.79\$$$

O&M cost of anaerobic digestion=(p+o+n+ 0.3m_{civ}+0.7 $m_{me\&i}$)Capital cost ofanaerobic digestion=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(6,882,920.79)

= 430,182.55 $/yr

Sludge dewatering

Table A.26 Equations for calculating cost of sludge dewatering

Unit cost metrics	Related equations	Reference
Sludge dewatering	Capital cost of sludge dewatering($)=(1+(Miscellaneous costs)/(construction cost))(-0.0727 x^3+48.326 x^2 + 13071x+389081) (Miscellaneous costs)/(construction cost)=0.28 x = flow to sludge dewatering(gpm) O&M cost of sludge dewatering($/yr)=0.5981 x^2+ 1598.4 x+48127 x = flow to sludge dewatering(gpm)	[36]

$Q_{\text{sludge to dewatering}}$=76.86 m^3/d=14.10gpm

Capital cost of sludge dewatering ($)=(1+(Miscellaneous costs)/(construction cost))(-0.0727 x^3+48.326 x^2 +13071 x +389081)=(1+0.28)(-0.0727 $(14.10)^3$+48.326$(14.10)^2$ + 13071(14.10) +389081)=745,971.85 $

Costs are based on 2009 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2009 ENR CCI=8585.71

$$\text{Capital cost of sludge dewatering (\$)} = (745{,}971.85\,\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2009 ENR CCI}}$$

$$= (745{,}971.85\,\$)\ \frac{11579.02}{8585.71} = 1{,}006{,}046.43\ \$$$

O&M cost of sludge dewatering($/yr)=0.5981 x^2+ 1598.4 x+48127

=0.5981$(14.10)^2$+ 1598.4(14.10)+48127=70,783.85 $/yr

The 2009 unit price of electricity, natural gas and labor are 0.0981 $/kWh, 0.00898 and 45.82 $/hr, respectively. The updated November, 2020 unit price of electricity, natural gas and labor are 0.1045 $/kWh ,0.00766 $/scf and 59.31$/hr, respectively [288, 289]. Component percent of electricity, labor and maintenance materials are 30%, 62% and 8% for

sludge dewatering. Costs are based on 2009 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

$$\text{O \& M cost of sludge dewatering (\$/yr)} = (70{,}783.85\ \$/\text{yr})(0.30)\ \frac{0.1045}{0.0981} + (70{,}783.85\ \$/\text{yr})(0.62)\ \frac{59.13}{45.82}$$

$$+ (70{,}783.85\ \$/\text{yr})(0.08)\ \frac{11579.02}{8585.71} = \ 87{,}064.07\ \$/\text{yr}$$

Sludge disposal

Table A.27 Equations for calculating cost of sludge disposal

Unit cost metrics	Related equations	Reference
Sludge disposal	O&M costs of sludge(US$/year)=(Excess sludge production (ton TSS/yr))(sludge disposal cost($/ton TSS)) Sludge disposal cost=280$/ton	[26]

Excess sludge production= 3,280.94 kg/d= 1,197.54 ton/yr

O&M costs of sludge disposal ($/year)=(Excess sludge production

(ton TSS/yr))(sludge disposal cost($/ton TSS))

=(1,197.54 ton/yr)(280$/ton)= 335,312.36 $/yr

Aeration equipment

Table A.28 Equations for calculating cost of aeration equipment

Unit cost metrics	Related equations	Reference
Aeration equipment	Capital cost of aeration equipment($)=(costing factors)(Installed capacityof aeration equipment (kW))(unit capital cost of aeration equipment($/kW)) Installed capacity of aeration equipment=1.5 (Energy consumption of areation process)/24 O&M cost (US$/year)=(p+o+n+ 0.3$m_{civ}$+0.7 $m_{me\&i}$)Capital cost where p=personnel cost=0.035, o=operation cost=0.01, n=insurance cost=0.003, m_{civ}=maintenance costs for civil works=0.075 $m_{me\&i}$=maintenance costs for mechanical, electrical and instrumentation equipment (E&I)=0.00175	[26]

Energy consumption of areation process=6,861.67 kWh/d

Installed capacity of aeration equipment=1.5(Energy consumption of areation process)/24

=1.5 (6,861.67)/24=428.85 kW

Capital cost of aeration equipment($)=(costing factors)(Installed capacityof aeration

equipment (kW))(unit capital cost of aeration equipment($/kW))

$=f_{ac}\ f_i$ (Installed capacityof aeration equipment (kW))(unit capital cost of aeration equipment($/kW))=(1.43)(1.72)(428.85 kW)(5,276.43 $/kW)=5,557,674.11 $

Costs are based on 2006 cost data. The cost is updated to November, 2020 by using Engineering News Record (ENR) Cost Index [288].

November 2010 ENR CCI=11579.02

2006 ENR CCI=7749.37

$$\text{Capital cost of aeration equipment (\$)} = (5{,}557{,}674.11\$)\ \frac{\text{November 2010 ENR CCI}}{\text{2006 ENR CCI}}$$

$$= (5{,}557{,}674.11\ \$)\ \frac{11579.02}{7749.37} = 8{,}304{,}217.54\$$$

O&M cost of aeration equipment=(p+o+n+ $0.3m_{civ}$+$0.7\ m_{me\&i}$)Capital cost of aeration equipment=(0.035+0.01+0.003+(0.3)(0.075)+(0.7)(0.0175))(8,304,217.54)

= 519,013.60 $/yr

Electrical energy

Table A.29 Equations for calculating cost of electrical energy

Unit cost metrics	Related equations	Reference
Electrical energy	O&M costs of electrictrical energy(US$⁄year)=(electricity consumption-electricity prodcution)(electricity cost($/kWh)	[26]

Energy consumption of areation process= 2,504,509.32 kWh/yr

Energy production= 12,757,470.91 kWh/yr

The updated November, 2020 unit price of electricity is 0.1045 $/kWh [289].

O&M costs of electrictrical energy(US$⁄year)=(electricity consumption-electricity prodcution)(electricity cost($/kWh)=(2,504,509.32 kWh/yr- 12,757,470.91 kWh/yr)(0.1045$/kWh)= - 1,071,434.49 $/yr

Table A.30 Wastewater quality for 1215 WWTPs in China

No.	Treatment technology	Design capacity (10,000 m^3/d)	Operation loading rate (%)	Influent BOD concentration (mg/L)	Influent COD concentration (mg/L)	Influent NH_4 concentration (mg/L)
1	MBR	10	84.26	364	663	42.7
2	AO+AAO	4	110.55	264	553	53.4
3	CAS	100	93.26	202	409	39.2
4	AO+AAO	60	104.99	224	468	52.1
5	AO+AAO	40	102.89	227	485	48.3
6	MBR	15	113.55	352	646	35.2
7	SBR	8	95.27	188	395	50
8	OD	20	100.74	214	444	48.3
9	AO+AAO	55	90.7	157	387	40.3
10	AO+AAO	45	82.39	183	487	48.6
11	AO+AAO	40	94.01	317	639	47.3
12	AO+AAO	10	91.75	259	616	34.9
13	AO+AAO	20	97.79	198	358	40.6
14	OD	4	75.7	66.7	305	18.6
15	AO+AAO	3	93.6	131	373	22
16	AO+AAO	3	94	98.9	385	39.8
17	OD	6	93.9	79.7	262	24.7
18	OD	5	87.8	58	224	23.8
19	AO+AAO	3	82.8	94	331	37.2
20	AO+AAO	7	87.3	131	310	36.1
21	AO+AAO	3.7	89.6	92	231	20.9
22	SBR	1	66.6	64	172	13.9
23	Others	3	96.2	114	193	22
24	OD	3	99	101	187	42.2
25	AO+AAO	2	68.4	68.4	187	32.2
26	AO+AAO	1	79.9	32.8	107	12
27	AO+AAO	3	90	51.7	120	20.6
28	AO+AAO	1	110	54.4	219	23.7
29	CAS	1	101	105	333	36.6
30	Biolak	3	74.3	35.3	125	17.5
31	SBR	10	95.4	147	371	13.9
32	AO+AAO	5	58	131	401	16.9
33	SBR	1	59.8	43.6	141	7.04
34	AO+AAO	0.5	68	108	173	15.21
35	AO+AAO	1	49.5	52.1	364	23.8

36	AO+AAO	16	98.5	114	243	36.9
37	AO+AAO	20	98.5	116	305	32
38	AO+AAO	60	82.2	182	394	60.2
39	Biolak	2	98	102	123	19.4
40	OD	10	85.7	134	534	116
41	AO+AAO	6	98.7	260	448	25.4
42	Biofilm	4	96	111	305	36.5
43	SBR	2	51.8	98.6	218	23.9
44	Biofilm	6	92.8	52	189	30.5
45	Biolak	3	56.4	113	271	33.7
46	OD	2	103	72	212	20.3
47	SBR	1	69.4	210	226	28.8
48	SBR	2	91.5	82.6	219	14.6
49	SBR	2	41.9	109	322	42.4
50	AO+AAO	5.3	100	160	431	26.4
51	SBR	4	75	150	200	40
52	Biolak	5	40.2	181	357	26.3
53	SBR	5	69.7	216	432	26.5
54	OD	15	66.4	172	490	45.5
55	AO+AAO	16	63.5	236	504	55.7
56	OD	15	69.4	105	393	20.5
57	CAS	4	41.5	68.1	192	29.1
58	AO+AAO	8	121	260	552	43
59	OD	8	51	137	314	21.4
60	Others	8	75.7	55.7	167	14.2
61	SBR	4	69.7	68.7	219	55.9
62	OD	4	52.5	85	192	27.4
63	CAS	4	41.5	68.1	192	29.1
64	SBR	2	74.3	123	211	22.7
65	OD	4	116	130	449	26
66	AO+AAO	4	74.9	255	480	48.4
67	AO+AAO	12	97.2	220	624	46.9
68	OD	7	59.9	86.7	230	25.3
69	OD	7	82.8	140	316	11.9
70	AO+AAO	4	71.6	193	322	41.6
71	OD	2	70.8	117	334	25.3
72	OD	5	65	170	479	16.6
73	Others	3	60.4	159	193	22.6
74	OD	6	76.1	175	286	30.1
75	OD	10	80.5	216	487	45

76	Biofilm	10	80	165	450	43
77	Biolak	6.6	80.1	105	331	29.2
78	Biolak	3	100	89.2	225	28.5
79	SBR	2	82.9	98	323	35
80	Biolak	2.5	79.5	180	256	26.8
81	Others	3	81.1	112	374	31.2
82	OD	2.5	64.3	120	350	35.5
83	Biolak	3	61	145	235	24.4
84	Biolak	3	63.3	150	260	20
85	SBR	3	79.4	185	278	47
86	Others	16	16.6	173	616	33.9
87	Others	8	70.1	136	454	56.3
88	Others	3	91.2	136	454	56.3
89	SBR	3	88.3	135	242	9.3
90	SBR	2	63.6	32.4	93	32.6
91	OD	3	97	20.3	173	16.9
92	AO+AAO	4	110	180	214	24
93	SBR	4	85.4	107	319	32.2
94	AO+AAO	3	98.9	121	309	26.1
95	SBR	1.2	86.6	103	187	42.5
96	Biolak	1	81.6	88.4	195	23.1
97	OD	3	118	97.3	221	21.1
98	Biolak	3	66.7	100	220	22.3
99	Others	20	82	143	475	6.79
100	AO+AAO	2	110	112	313	23.2
101	OD	1.25	98.2	132	239	25.2
102	OD	1.5	91.9	113	220	21.9
103	Biolak	4	82.9	151	276	14.4
104	SBR	2	97.3	165	276	34.3
105	AO+AAO	2	99.5	167	276	30
106	SBR	3	88.3	135	242	9.3
107	OD	1	83.9	192	495	25
108	AO+AAO	1	98.3	96.7	144	21.1
109	SBR	2	88.1	76.6	141	29.5
110	AO+AAO	10	96.8	211	477	51.3
111	AO+AAO	12	91.2	130	501	22
112	Others	1.1	99.4	28.7	255	28.1
113	OD	2	37.3	137	353	34.9
114	SBR	1.8	5.96	23.4	271	37.4
115	OD	3	51	171	434	56.9

116	OD	3	44.6	153	352	41
117	OD	8	78	90.4	189	17.5
118	Biofilm	5	59.1	77.4	216	32.4
119	Biolak	2	26.3	178	201	27.3
120	Biolak	3	78.4	91.9	230	23.7
121	OD	2	103	189	267	18.3
122	Biolak	3	58.9	56.4	386	42.4
123	Biolak	2	87.4	144	428	33.5
124	Biolak	2	96.6	121	287	33
125	AO+AAO	3.5	32.6	155	317	35.3
126	AO+AAO	2	107	84.7	232	19.1
127	AO+AAO	2.5	76.2	131	259	35.4
128	AO+AAO	7	50.9	90.4	186	19.7
129	Wetland	1.5	54.1	99.3	167	37.6
130	Others	1.5	90.2	114	236	34.8
131	SBR	5	28.7	100	325	11.2
132	OD	1.5	86.8	104	218	23.3
133	OD	2	89	108	328	33
134	OD	1.5	95.9	110	270	23
135	OD	3	60	124	292	35.6
136	Others	4	52.8	54.6	164	14.7
137	Others	2	80	142	289	19
138	AO+AAO	16	99.5	210	418	49.3
139	OD	4	99.4	183	398	24.7
140	CAS	6	77.1	174	441	37.7
141	Biofilm	2	106	132	266	25.1
142	SBR	5	91.7	87.7	192	36.2
143	AO+AAO	16	103	210	522	26.9
144	OD	1.5	89.8	132	277	29.7
145	OD	1	49.2	160	336	44.9
146	OD	0.8	78	178	374	29
147	AO+AAO	4	86.7	89.6	242	17.9
148	SBR	0.5	61.2	89.7	209	14.4
149	AO+AAO	8	81.9	100	402	34.6
150	Wetland	2	47.9	89.7	370	44
151	Others	3	69.2	99	342	33
152	AO+AAO	0.8	18.8	41.8	91.5	11.3
153	OD	2	80.8	145	324	26.4
154	AO+AAO	7.5	85.41	163	276	27
155	OD	1.5	46.8	159	298	33.6

156	AO+AAO	1	51	156	346	42
157	SBR	2	36.1	108	214	25.4
158	AO+AAO	1.5	45.2	146	335	39.8
159	OD	1.5	50.5	167	325	28
160	MBR	0.8	89.7	181	323	25.7
161	AO+AAO	0.8	79.8	114	183	22.2
162	AO+AAO	1	69.6	112	234	35.29
163	Biofilm	2	58	67.5	222	32.6
164	OD	12	84	133	364	42.8
165	AO+AAO	1	64.6	86.8	212	27.6
166	OD	1	88.5	121	234	24.5
167	OD	1.8	123	175	300	37.3
168	Others	0.2857	100	168	388	44
169	OD	1.5	76.9	100	312	48.3
170	OD	2	43.4	173	361	30.3
171	Others	3.5	62.2	209	360	38
172	AO+AAO	1	58.1	150	324	32.5
173	OD	1	84.1	187	364	29.2
174	AO+AAO	2	68.8	177	328	31.4
175	OD	3	71.3	216	478	47.1
176	SBR	2.5	55	155	398	63.6
177	Others	0.8	70.6	103	332	34.5
178	AO+AAO	1	64.1	123	227	27.6
179	AO+AAO	1	54.3	101	250	31.2
180	AO+AAO	1.5	50.1	109	315	30.7
181	Others	1.2	53	125	293	31.6
182	AO+AAO	1.8	56.8	212	439	43.7
183	AO+AAO	2	41.2	142	400	50.5
184	SBR	1.5	55.3	181	462	62
185	AO+AAO	1	74.3	181	330	47
186	Others	3	57.4	125	306	45.2
187	OD	1.5	57.4	126	200	30
188	AO+AAO	1.2	45.2	177	310	45
189	AO+AAO	8	68.7	154	347	46
190	AO+AAO	4	24.1	163	408	66.8
191	Others	2	29	149	309	41
192	OD	1.6	33.6	140	324	33
193	AO+AAO	2	55.9	107	367	48.9
194	AO+AAO	3	66.4	157	254	29.6
195	SBR	0.5	80	94.8	216	30.3

196	AO+AAO	0.7	33.7	94	287	28.8
197	AO+AAO	1	45.5	160	294	40.7
198	AO+AAO	0.25	77.6	116	313	28
199	AO+AAO	0.5	86.1	142	327	37.9
200	AO+AAO	0.5	35.1	134	368	78.3
201	Biofilm	1	36.9	126	287	29.5
202	AO+AAO	1	45.3	130	255	37.5
203	Others	4	48.2	112	350	46
204	Biofilm	3.5	83.2	160	317	30.5
205	Biofilm	0.35	97.7	118	287	28.1
206	OD	2	25.9	100	256	30.5
207	OD	1	79.2	131	295	31.9
208	AO+AAO	0.75	76.6	123	251	32.3
209	AO+AAO	1.5	11.1	190	336	30.7
210	AO+AAO	2	31.3	220	320	46
211	AO+AAO	0.65	40.9	181	262	36.2
212	AO+AAO	0.6	42.9	131	261	29.7
213	AO+AAO	0.6	52.3	288	390	39.7
214	OD	1.5	59.2	165	331	51.1
215	Others	4	61.6	540	1200	57.9
216	OD	10	59	134	414	23.2
217	AO+AAO	15	56	128	400	27.2
218	AO+AAO	15	59.6	111	332	23.8
219	AO+AAO	39	72.8	129	392	19.6
220	AO+AAO	10	82.4	103	302	26
221	Biolak	3	71.3	200	277	29.5
222	MBR	15	88.5	127	239	21.9
223	Biolak	2	87.9	178	386	16.3
224	AO+AAO	5	102	114	267	25.8
225	SBR	10	85.9	195	302	36.1
226	SBR	5	80.4	199	299	37.8
227	SBR	1.5	82.9	81.6	176	28.2
228	Biolak	2	86.4	200	235	25.8
229	SBR	1	72.3	120	200	30
230	SBR	1.5	87.9	150	231	33
231	AO+AAO	10	89.8	132	245	17.8
232	SBR	3	92.2	75.3	186	22.5
233	SBR	1.5	73.7	50.5	107	14.5
234	SBR	5	100	163	235	26
235	SBR	15	24.2	112	210	43.1

236	AO+AAO	10	82.4	152	472	40.5
237	AO+AAO	32.5	84.7	159	291	32
238	Others	32.5	93	172	317	34.6
239	SBR	15	103	179	462	29.1
240	Others	6	62.2	136	283	28.2
241	AO+AAO	5	90.4	206	427	18
242	SBR	10	37	156	318	35
243	SBR	2	25.1	100	240	19
244	SBR	2	25	100	240	19
245	SBR	1	49.3	180	350	25
246	SBR	1	77.5	132	466	36.9
247	SBR	2	39.9	52	110	17.7
248	SBR	2	89.6	200	400	30
249	SBR	10	37	156	318	35
250	AO+AAO	2	100	100	260	20
251	SBR	1	65.8	120	280	20
252	AO+AAO	10	71.2	137	270	53.8
253	Biofilm	2	27.9	184	393	30.8
254	Others	20	60.7	115	289	44.6
255	SBR	5	103	99.5	295	18.8
256	SBR	1.5	91.3	180	230	22
257	SBR	1.5	110	4.3	42.1	268
258	SBR	5	89	345	618	19.8
259	SBR	3	35.4	174	362	26.4
260	Others	1	96.2	117	204	38.6
261	AO+AAO	5	96.8	83.7	219	24
262	SBR	2	106	136	263	28.7
263	SBR	1	11.1	61	156	17.5
264	SBR	2	59.6	136	327	60
265	Biofilm	6	39.4	195	387	53.9
266	Biofilm	13	87.8	111	330	31.7
267	SBR	0.3	57.5	300	192	30.5
268	SBR	6	99.2	89	196	26
269	SBR	10	36	152	397	25.6
270	AO+AAO	2	60.3	55.4	228	43.5
271	SBR	1.5	100	114	245	34.8
272	SBR	1	74.8	160	315	22.4
273	SBR	1	77.5	200	157	19.8
274	AO+AAO	1.5	47.5	286	337	38
275	AO+AAO	2	61.9	200	200	30

276	CAS	5	76.8	160	348	25
277	AO+AAO	4	8.36	60.5	121	18.3
278	AO+AAO	10	100	86.2	247	18.5
279	AO+AAO	2	100	103	345	20.6
280	Wetland	2	95.9	100	240	19.4
281	SBR	2	90	110	160	28
282	SBR	2.5	91	134	406	49.7
283	SBR	3	74	111	170	16.9
284	AO+AAO	1.5	81.8	192	522	46.8
285	SBR	4	52.5	97.6	278	32
286	SBR	5	93.8	301	484	28
287	SBR	5	99.5	86.9	392	32.3
288	SBR	2	54.9	114	400	29
289	SBR	2	3.01	200	350	30
290	AO+AAO	4.5	67.1	102	257	35.7
291	SBR	2	36.9	141	294	32.8
292	AO+AAO	280	83.2	113	254	24.8
293	AO+AAO	170	93	142	251	19.8
294	CAS	7.5	95.6	151	337	31.7
295	Others	6	74	169	298	30.6
296	SBR	6	105	106	302	18.2
297	AO+AAO	2	113	129	290	37.5
298	AO+AAO	4	98.8	192	370	36.8
299	AO+AAO	10.5	90.8	122	259	28
300	AO+AAO	2.2	98.3	124	356	27.1
301	AO+AAO	5	92.4	140	236	25
302	SBR	40	97.2	176	364	28.9
303	AO+AAO	50	102	127	228	23.3
304	Others	4	82.7	127	279	29.5
305	Others	10	97.4	166	333	24
306	AO+AAO	10	116	162	316	21.1
307	AO+AAO	6	76.5	141	406	36.5
308	AO+AAO	5.5	62.6	111	273	17.6
309	AO+AAO	3.5	84.5	104	222	20.9
310	OD	12	97.6	154	344	23.4
311	OD	3	72.5	59.1	121	11.7
312	AO+AAO	1.2	99.3	71.6	184	11
313	OD	3.4	91.1	115	416	17.3
314	OD	1	123	105	315	13.7
315	SBR	0.25	103	57.8	122	15.9

316	OD	1	102	93.1	349	15.3
317	SBR	0.5	54.3	50.5	123	9.78
318	AO+AAO	2.8	76.3	192	368	18
319	OD	6	72.9	182	399	16.2
320	Others	10	90.4	162	371	23.6
321	SBR	1	124	271	624	18.9
322	AO+AAO	1	105	201	365	39.4
323	AO+AAO	15	86.9	144	327	28.4
324	AO+AAO	20	74.4	120	445	24.7
325	AO+AAO	1.25	111	226	621	34.6
326	AO+AAO	5	103	74.8	184	27.5
327	AO+AAO	14	91.4	159	343	30.8
328	AO+AAO	13.8	88.1	196	358	24.3
329	OD	1.7	93.6	164	370	13.5
330	OD	0.4	84.3	158	362	9.1
331	OD	1.2	81.7	143	321	10
332	AO+AAO	2.5	60.4	122	216	34
333	AO+AAO	2.5	122	80.8	134	20.2
334	AO+AAO	1.75	33.5	62.3	109	22.5
335	AO+AAO	1.25	80.2	51.3	75.2	14.7
336	AO+AAO	0.5	94.8	69	115	10.8
337	AO+AAO	12	71.3	176	517	25.2
338	SBR	7	102	108	333	26.9
339	AO+AAO	5	110	116	360	25.1
340	OD	8	78.5	102	267	22.8
341	OD	8	73.8	85.3	223	24.1
342	OD	4	99	64	163	22.8
343	AO+AAO	2	41.4	78.6	157	13.3
344	OD	3.5	28	10.9	46.8	7.04
345	Others	15	92.3	84.9	242	27.5
346	Others	11	92.9	150	213	26.3
347	Others	4	90.3	105	291	30.4
348	Others	5	94	102	341	24.7
349	OD	4	106	80.6	225	31
350	AO+AAO	2.5	89.6	20.7	68.9	17.2
351	AO+AAO	2	61.8	18.2	59.2	13.5
352	AO+AAO	4.5	106	54	156	30.2
353	AO+AAO	2	19.2	26.1	117	24.1
354	AO+AAO	16.5	100	91	182	29
355	AO+AAO	15	98	120	239	20.7

356	AO+AAO	3	107	79.3	183	34
357	AO+AAO	1.5	97.5	89.7	212	22.3
358	AO+AAO	20	93	51.8	223	20.4
359	AO+AAO	15	104	104	247	27.9
360	AO+AAO	10	70.5	93.1	232	24.3
361	AO+AAO	5	91.7	180	168	22.2
362	AO+AAO	3	91.2	55.5	159	23.6
363	AO+AAO	4	91.2	127	399	14.6
364	AO+AAO	3	33.6	34.4	207	6.79
365	Others	4	27.1	222	576	14.6
366	AO+AAO	9.8	67.1	211	464	26.5
367	OD	3.5	77.6	100	217	25.9
368	OD	7	57	84	179	29.1
369	OD	2	61.5	204	384	35.8
370	AO+AAO	1	63.1	94	186	26.4
371	AO+AAO	3	48.8	146	308	30.3
372	OD	1.1	72.6	88	172	25.2
373	AO+AAO	2	68.4	106	226	17.9
374	AO+AAO	2.5	41.6	92	186	32.5
375	AO+AAO	0.4	63.9	124	228	47.5
376	OD	9	75.6	121	359	20.8
377	AO+AAO	3	57.2	82.1	249	21.8
378	OD	3	105	102	305	22
379	OD	8	73.5	187	428	25.9
380	OD	8	65.3	198	420	28.8
381	SBR	4	63.9	160	390	33.8
382	SBR	4	38.5	134	306	28.4
383	SBR	4	46.4	137	316	24.8
384	CAS	18	79.9	129	395	31.7
385	CAS	14	79.6	130	286	22
386	AO+AAO	4	105	148	335	18.8
387	Others	6	96.4	131	284	16.4
388	Others	10.3	97	98.2	420	16.3
389	OD	25	76.4	134	291	31.2
390	AO+AAO	4	87.9	156	315	27.6
391	OD	4	88.8	165	327	36.1
392	SBR	2.5	79	129	550	24.6
393	Others	9.8	82.7	200	388	25.9
394	AO+AAO	2	84.8	45.9	125	15.9
395	AO+AAO	2	94.8	83.5	264	29.5

396	AO+AAO	2	4.58	30	100	20
397	AO+AAO	1.5	67	155	583	27
398	SBR	18	99.3	140	285	25.2
399	Others	15	105	124	289	24.3
400	AO+AAO	5	91.8	93	224	22.4
401	SBR	14.5	98.5	90.5	240	28.7
402	AO+AAO	4	97.8	90.2	227	28.1
403	AO+AAO	1	91.9	94.3	225	26.7
404	SBR	2	86.7	104	230	25.7
405	AO+AAO	2	87.3	86.2	306	20.4
406	Biolak	2	61.4	128	252	27.8
407	AO+AAO	1.5	23.1	70.5	176	13.6
408	AO+AAO	0.5	65.7	74	173	20
409	SBR	1.4	15.7	35	100	14
410	AO+AAO	5	48	65.1	182	22.8
411	AO+AAO	60	97.8	85.7	256	24.6
412	AO+AAO	10	102	114	309	29.1
413	AO+AAO	30	94.6	225	744	20.3
414	Others	4	84.8	106	241	17.5
415	Others	4.5	88.2	71	178	19.1
416	OD	0.8	77.2	33.8	185	13.3
417	Others	2	30.9	356	810	22.9
418	OD	6	96.8	124	341	32.5
419	AO+AAO	8	99.7	169	385	17.3
420	SBR	1	108	120	320	10.8
421	Others	10	105	73	187	20
422	AO+AAO	16	101	85	196	20
423	AO+AAO	8	89.2	73	186	26
424	AO+AAO	16	86.9	99	229	13
425	AO+AAO	20	62	96	253	20
426	Others	8	71.1	143	219	28.9
427	AO+AAO	10	78.3	47	207	14.7
428	SBR	6	101	95	237	21.7
429	OD	22	103	88.8	180	18.1
430	AO+AAO	3	94.4	109	222	21.9
431	AO+AAO	5	97.4	68.6	215	18.8
432	SBR	6	111	68.2	200	16.1
433	AO+AAO	0.25	64.3	35.8	106	11
434	AO+AAO	0.25	64.5	40	115	10.4
435	OD	0.3	69.1	34.8	104	11

436	AO+AAO	50	7.52	134	353	25.3
437	SBR	6	111	68.2	200	16.1
438	OD	20	109	63	124	19.8
439	OD	8	87.6	162	362	28.3
440	Biofilm	1	94.8	98	135	16.5
441	AO+AAO	30	108	124	399	18.4
442	Others	2	77.7	99.1	361	27.6
443	SBR	15	87.4	121	369	23.4
444	Others	16	58.1	120	290	19.1
445	AO+AAO	6	74.6	123	346	13
446	AO+AAO	5	88.7	62	167	24.7
447	AO+AAO	7.5	70.6	108	300	14.8
448	OD	3	101	49.9	259	17.9
449	SBR	2	81.6	122	215	26.7
450	AO+AAO	3	65.3	76.9	299	10.5
451	SBR	2.64	99.3	43.4	170	8.4
452	SBR	1.5	65	23.8	90.5	15.8
453	AO+AAO	5	87.5	82.6	227	23.2
454	Others	90	84.2	221	620	61.6
455	OD	15	93.2	222	527	17.3
456	OD	7	94.9	70	217	27.3
457	AO+AAO	3.5	103	83.6	294	17.8
458	OD	2	122	33.1	104	10.8
459	OD	6	106	50.1	202	20.5
460	OD	4	102	40.8	151	17.4
461	AO+AAO	15	98.6	62	236	22.7
462	AO+AAO	4	95.2	59	268	21.8
463	AO+AAO	2	110	31.9	161	9.38
464	Others	5	66.5	53	190	17.3
465	OD	4	97.7	89	235	26.3
466	SBR	4	62.7	71.5	312	14.9
467	OD	10	82.4	96.4	210	13.7
468	Others	15	95.5	98.7	401	22.2
469	OD	1.95	87.4	33	118	21.2
470	Others	0.5	106	59.1	98.7	8.3
471	SBR	1.8	95.3	63.1	174	24.7
472	Others	0.5	105	38.1	227	12.5
473	Others	0.5	112	47	93	13
474	Others	0.5	101	102	224	24.9
475	SBR	5	70.2	100	387	32.1

476	OD	5	91.3	167	287	16.4
477	SBR	5.5	116	98.8	204	25.7
478	OD	30	114	71	181	26.2
479	OD	18	112	109	272	22.5
480	Others	10	103	104	197	31.5
481	OD	20	112	98	206	22.7
482	OD	5	107	85	208	23
483	OD	10	107	66.7	169	21.1
484	AO+AAO	3	35.9	88.6	216	39
485	OD	2	53.8	51.6	107	24.9
486	AO+AAO	5	2.53	43	122	20.2
487	OD	2	120	58.3	136	22.5
488	AO+AAO	20	100.5	122	392	19.5
489	AO+AAO	15	95.1	103	260	29.4
490	AO+AAO	2.5	57	62.9	219	26
491	OD	2.5	83.5	125	229	24
492	OD	3	93.1	95.6	260	23.2
493	Biofilm	2.5	77.2	180	199	25.3
494	AO+AAO	2.5	80.7	16.8	44.2	5.4
495	Others	2	102	61.9	160	19.4
496	OD	2.5	66.2	69	131	20
497	OD	6	86	115	182	25.2
498	OD	10	91.4	103	212	17.9
499	AO+AAO	2	27.4	22.5	48.9	2.97
500	AO+AAO	2.75	73.3	131	262	19.8
501	AO+AAO	4.5	100	59.4	173	13.2
502	AO+AAO	2	75	78.1	223	9.8
503	SBR	3	79.5	48.6	149	16.9
504	Others	3	75.6	52	130	14
505	AO+AAO	10	105	86	177	22.3
506	OD	5	101	70	199	26.1
507	AO+AAO	0.5	98.6	80	150	20
508	Others	4.5	105	105	273	30
509	AO+AAO	1	84.7	65	137	15.5
510	OD	3	96.4	83.8	179	28.4
511	OD	6	86.4	110	219	25.8
512	AO+AAO	1.1	69.2	171	399	49.6
513	OD	5	90.7	79.3	155	19.6
514	SBR	10	109	97.3	215	19
515	AO+AAO	3	106	80.9	196	20.9

516	OD	4	96.9	80.5	234	35.6
517	OD	4	79.4	60	253	31.6
518	CAS	2	99.5	160	233	23.5
519	OD	2.58	100	118	234	32.6
520	OD	5	75	99	195	25
521	OD	8	96.4	82.3	203	20
522	OD	2	97.4	81.3	191	17.2
523	OD	4	75.6	104	219	25
524	OD	2.5	105	91	203	24
525	OD	1.5	94.6	87.2	191	19.9
526	OD	4	46.4	88.1	189	20
527	AO+AAO	2	53.4	94	161	12.9
528	OD	1	16.4	53.3	128	18.6
529	OD	8	67.5	56	134	14.6
530	Others	2.5	67.8	38	265	20
531	OD	1	53.7	17.2	106	12.7
532	OD	2	67.6	59.6	128	12.6
533	Others	2	90.4	60	158	16
534	AO+AAO	6.65	85.3	189	149	17.9
535	OD	3	88.6	97.6	203	18.3
536	OD	1	73.9	79.5	199	15.1
537	OD	4	70.2	85	153	14.9
538	OD	2	72.9	85.3	160	18.2
539	OD	0.75	103	26.5	105	12.5
540	OD	4	87.7	84.3	171	15
541	AO+AAO	8	110	48.1	106	16
542	Others	60	71.2	81.4	167	15.2
543	SBR	5	53	60.9	176	24.9
544	SBR	5	84.8	149	322	25.7
545	AO+AAO	10	65.5	59	149	19.7
546	Others	2.5	90.5	77.4	219	17.8
547	OD	2.5	91.5	92	226	21.7
548	SBR	1.25	61.3	135	353	16.9
549	SBR	5	59.8	41.8	112	22.3
550	OD	3	68.8	61.2	171	18.8
551	OD	1	2.46	21.8	53.1	11.7
552	OD	1	31.2	32	117	13.8
553	SBR	4	81.2	63	193	28.3
554	OD	1	115	58	165	28
555	OD	1	71	45	143	18.8

556	AO+AAO	12	92.8	78.1	159	7.88
557	SBR	5	76	95	184	23.3
558	OD	2	97	104	294	33.2
559	Others	30	93.1	141	332	22.5
560	Others	20	90.2	120	279	30.3
561	AO+AAO	6	91.9	146	340	20.4
562	AO+AAO	10	66.1	111	258	27.1
563	Others	9	65	111	249	27.5
564	OD	10	73.2	181	367	17.7
565	OD	2.5	114	81	172	17.6
566	OD	3	100	54	177	23.8
567	AO+AAO	16	106	79.6	199	18.5
568	SBR	1.75	99.4	62.5	197	20.7
569	OD	1	61.7	72.9	187	28
570	OD	1	46	52.9	170	15.9
571	OD	1.5	87.9	42.5	137	19.9
572	SBR	4	69.1	42.7	134	12.6
573	OD	1	72	58	183	17.6
574	Others	1	44.2	55.3	136	26.9
575	SBR	2	55	58.6	153	14.7
576	OD	2	63.1	58.9	156	22.8
577	OD	2	129	59.9	204	21.8
578	Others	0.4	18.5	180	109	8.1
579	OD	3	92.5	45.3	136	13.4
580	AO+AAO	1	85.5	49.4	160	17.6
581	OD	2	66.5	51.1	167	17.2
582	OD	1.25	102	46.1	146	20.9
583	OD	4	95	60.4	198	17
584	AO+AAO	15	92.5	78.8	165	17.9
585	SBR	4.5	100	74.7	160	12.5
586	SBR	4.5	62.1	42	131	25.9
587	OD	2.5	25.7	39	103	21.8
588	OD	2.5	48.7	49.8	172	20
589	OD	2.5	13.5	47.9	151	28.8
590	OD	7	77.8	83	200	22.2
591	SBR	1	25.8	34	111	21
592	AO+AAO	4.5	84.7	119	242	25.8
593	Biofilm	0.25	85.7	93	162	23.1
594	OD	3	102	80.9	206	17.9
595	OD	4	83.2	90.8	217	26.1

596	Others	10	101	37	109	18.2
597	AO+AAO	10	91.7	55.4	149	19.8
598	OD	5	72.2	74.5	196	19.1
599	OD	3	44.5	54.3	115	18.8
600	OD	1.25	75.2	84.9	181	18.2
601	OD	2	65.4	41	167	14.3
602	SBR	2	39.9	103	159	16.9
603	OD	1	40.6	48.3	117	24.6
604	Others	10	78.6	63.4	232	24.5
605	OD	2.5	93.9	63.7	159	21.6
606	OD	4	65.2	75.2	222	22
607	AO+AAO	2	73	60	131	23.7
608	OD	3	67.3	97	202	17.2
609	Biofilm	0.8	92.2	111	197	43.8
610	OD	3	83.2	203	373	36.8
611	OD	1	105	65	170	21.2
612	OD	2	100	65	186	20.4
613	OD	0.5	82.6	74	186	24.6
614	SBR	1	8.79	84	211	25.8
615	OD	2.5	68.8	79.6	189	18
616	OD	2	114	89	188	14.8
617	AO+AAO	5	80.3	52.1	237	20.2
618	AO+AAO	3	25.9	33.8	129	18.8
619	OD	1	85.5	76	148	25.7
620	OD	1.5	105	51.3	154	13.1
621	OD	1	104	35.6	133	11.6
622	OD	1	84.8	44.1	107	17.3
623	OD	2	90.4	66.2	186	19.3
624	Wetland	1.5	96.8	52	172	10.2
625	OD	1	37.8	47.1	137	15.5
626	OD	1.5	83.9	68.1	164	16
627	OD	1.5	95.4	81.9	128	13.8
628	AO+AAO	15	93.1	72.7	220	13.9
629	AO+AAO	2	39	103	302	14.9
630	AO+AAO	4	75	44.1	172	15.4
631	AO+AAO	2.5	108	55	154	23
632	OD	2	83.3	40.6	126	16.6
633	OD	2	73.8	62	173	15.3
634	OD	1	77	48.7	139	17.3
635	AO+AAO	1.5	75.8	30.6	89.9	6.73

636	OD	2	93.5	37.8	141	21.4
637	OD	4	94.5	71.5	176	23.6
638	SBR	4	59.1	66.4	194	25.3
639	OD	2	109	78.3	176	18.9
640	OD	1	77.3	69.5	168	26.3
641	SBR	1	66.2	51.2	178	21.9
642	Others	1	98	56.6	160	26.6
643	OD	5	68.5	89.1	247	24.9
644	OD	5	102	109	235	26.9
645	OD	1	44.4	40.1	98.8	26.5
646	OD	2	23.3	56.9	139	20.6
647	Others	50	98.3	60.6	175	17.6
648	CAS	8	102	63.5	188	23.9
649	OD	10	72.3	65.6	184	15.8
650	AO+AAO	20	84.3	79	216	24.4
651	OD	4	79.5	30.1	131	12.7
652	AO+AAO	8	95.2	65.9	178	18.1
653	OD	8	89.3	36.3	85.7	10.7
654	OD	1	37.8	46	105	15.2
655	OD	1.5	54.4	54.6	128	10.1
656	OD	0.75	68.7	46.9	113	9.26
657	SBR	10	103	45.9	116	11.4
658	OD	3	67.9	65.3	165	15.5
659	OD	0.75	120	25	80	6.5
660	OD	2	71.2	25.4	77.9	7.63
661	CAS	2.5	98.3	78	179	20.1
662	OD	1.5	84.4	87	164	20.6
663	OD	1	99.2	49.7	133	13.4
664	OD	6	100	51	114	11.9
665	OD	6	27	36.1	110	10.4
666	Others	5	90.1	52	110	22.9
667	OD	2	98.9	33.1	101	7.2
668	OD	0.5	100	42.3	82.4	11.1
669	SBR	1	94.1	67.3	119	10.5
670	SBR	6	102	64.8	144	23.1
671	OD	4	69.8	34.9	93.2	16.2
672	OD	1.5	106	62.2	146	22.1
673	OD	0.5	111	39.2	106	17.4
674	OD	1	91.8	47.8	111	16.8
675	OD	2	92.3	55.2	131	20.2

676	OD	1	113	34.7	77.4	13.2
677	OD	1	106	48.1	112	14.8
678	OD	2	98.5	32.5	95.3	11.6
679	OD	4	64	59	107	17.1
680	OD	0.75	110	46	103	19.2
681	OD	8	104	86.5	219	21.4
682	OD	1	77.7	37.3	90.3	14.7
683	OD	0.3	68	55.8	142	19.3
684	OD	1	126	66.4	154	16.1
685	OD	1	100	91.7	202	16.4
686	OD	0.6	82.2	70.9	175	24.6
687	OD	1.5	80.2	81	165	17.1
688	OD	1	61.4	46.8	102	14.3
689	OD	0.75	102	54.4	166	19.8
690	OD	1	157	40.6	140	16.9
691	AO+AAO	2	85	85	276	26
692	OD	1	105	67.5	132	22.7
693	AO+AAO	4	63.7	40.5	97.9	13.7
694	OD	2.25	77.3	54.6	133	11.2
695	OD	2	61	68.3	157	19.6
696	OD	12	89.7	77.6	172	23.3
697	OD	0.75	114	59.2	138	14.1
698	OD	2	96.3	75.7	242	10.2
699	OD	1.5	87.3	55.2	149	21
700	OD	3	98.6	49	100	9.67
701	OD	1.5	89.1	41.9	106	9.8
702	OD	2	87.6	45	125	10.6
703	AO+AAO	2	60.6	55.8	141	12.5
704	AO+AAO	2	75.4	48	112	8.9
705	OD	0.75	99	45.7	121	12
706	OD	1	80	52.4	114	10.4
707	OD	0.5	99.6	34.2	102	7.25
708	AO+AAO	30	106	169	363	38.1
709	AO+AAO	20	110	145	437	46.4
710	AO+AAO	20	99.7	142	452	40
711	AO+AAO	3	158	104	303	29.8
712	AO+AAO	3	110	169	397	49.5
713	Biofilm	0.5	76.4	137	263	38.7
714	AO+AAO	5	50.4	122	362	54.7
715	OD	4	39.8	56.8	168	28.9

716	AO+AAO	4	83.6	81.5	341	20.9
717	AO+AAO	2	65.8	106	322	34.8
718	AO+AAO	1	36.2	13.6	70.2	11.4
719	AO+AAO	5	90.9	125	266	36.8
720	AO+AAO	4	38.2	89.6	194	24.6
721	AO+AAO	3	46.1	61.5	241	25.8
722	Others	10	66.8	331	762	55.3
723	Others	17	109	367	827	41.3
724	SBR	16	86.3	303	725	61
725	Biofilm	14	85	219	482	47
726	Others	15	61.9	109	408	21.5
727	Others	15	77.3	297	781	24.2
728	AO+AAO	1	69.8	161	354	31.1
729	SBR	2	24.4	154	428	35
730	AO+AAO	11	76.1	167	553	41
731	Others	8	92.7	155	354	36.3
732	AO+AAO	8	76.8	206	411	42.4
733	AO+AAO	3	100	157	482	43.8
734	AO+AAO	3	43.7	125	345	25.8
735	Others	20	77.1	105	342	52.8
736	Biofilm	5	69.8	99	354	26.6
737	AO+AAO	10	133	111	373	31.8
738	AO+AAO	4	101	105	333	34.6
739	AO+AAO	7.5	78.4	88.2	258	33.6
740	Others	8	72.7	109	325	19.3
741	AO+AAO	5	49.1	127	365	38
742	Others	6	66.3	214	376	33.5
743	AO+AAO	2	35.9	108	222	25.6
744	OD	7	94.1	98.7	225	25.8
745	AO+AAO	4	83.7	70.9	226	26.9
746	AO+AAO	2	46.4	86.7	272	39.7
747	AO+AAO	4	82.8	142	316	18.3
748	AO+AAO	4	43.2	191	229	21.7
749	Others	6	95.6	88.9	286	27.5
750	OD	3	88	53.6	199	36.3
751	AO+AAO	4	52.2	163	413	16.4
752	Others	6	89.5	42.8	291	16.8
753	AO+AAO	7.5	70.9	104	338	35.5
754	AO+AAO	2	39.7	110	420	45.1
755	AO+AAO	4	12.7	40.7	279	28.9

756	AO+AAO	6	31.5	83.6	243	34.1
757	Others	5	61.4	50.49	220	22.1
758	AO+AAO	20	101	522	1137	45.5
759	Others	12	88.2	289	760	44.9
760	AO+AAO	5	97.2	94	390	35
761	Biolak	2	89.2	149	302	27.5
762	AO+AAO	3	77.9	234	405	36
763	OD	2	77.4	127	314	32.5
764	AO+AAO	4	80.8	153	318	37.8
765	Biolak	5	93.2	183	379	26.7
766	MBR	3	16.6	50	260	35
767	OD	2	74.1	180	342	18
768	AO+AAO	5	92.2	139	339	34.4
769	AO+AAO	4	64	90	220	20.3
770	AO+AAO	4	26.3	35	414	25.8
771	Biolak	10.6	76.1	261	740	52.2
772	AO+AAO	8	88.1	280	520	30
773	AO+AAO	12	70	130	400	26.7
774	OD	3	51.5	50.8	263	26.2
775	Others	4	76.3	66.7	365	14.7
776	AO+AAO	8	71.2	187	401	25.6
777	AO+AAO	3.5	86.3	107	237	52.8
778	OD	5	74.6	88.6	411	24.9
779	OD	10	93.5	76.6	431	27.2
780	AO+AAO	5	71.2	115	338	26
781	AO+AAO	2	97.1	142	356	20.9
782	AO+AAO	5	94	108	336	44.6
783	OD	12	71.5	132	331	28.9
784	AO+AAO	3	100	192	424	39.3
785	Others	20	98.9	115	383	39.7
786	Others	6	81.6	53	126	22.9
787	Biolak	4	53.8	88	236	18.1
788	Others	4	108	157	312	13.9
789	AO+AAO	3	74.3	102	287	17
790	AO+AAO	8	96.2	52	184	24
791	AO+AAO	3	41.1	58	190	42
792	Others	5	80.7	90	179	22.7
793	Biolak	3	83.2	80	100	30
794	SBR	4	73.3	71	131	13.3
795	OD	3	66.8	96.6	224	30.4

796	AO+AAO	4	79.6	53.5	270	16.9
797	AO+AAO	3	16.6	71.3	250	17
798	AO+AAO	3	89.4	47.2	96.4	7.89
799	Biofilm	2.5	26.7	14.1	99.6	7.1
800	AO+AAO	4	52.1	53	121	19.9
801	AO+AAO	6	55.2	36	120	12.4
802	AO+AAO	2	76.6	29.6	70	12.7
803	AO+AAO	2	44.7	43	104	16
804	OD	4	83.3	84.5	233	22.2
805	OD	2	111	128	347	23.3
806	Others	6	83.6	138	350	32.2
807	AO+AAO	3	41.5	75.6	204	22.3
808	OD	5	81.4	125	281	28.6
809	AO+AAO	4	95.4	115	308	25.2
810	AO+AAO	4	102	154	361	31.6
811	AO+AAO	5	84.3	122	274	24.8
812	OD	4	95.6	97.2	285	28.5
813	AO+AAO	12	75.8	108	326	30
814	OD	3	95	126	285	30.4
815	Others	2	53.5	121	265	24
816	Biofilm	4	110	178	443	53.9
817	MBR	1	61.7	165	258	19.8
818	OD	8	95.3	102	299	34
819	AO+AAO	4	8.9	493	1011	43.2
820	AO+AAO	4	46.9	118	304	22
821	Biofilm	2	78.2	116	456	32
822	AO+AAO	2	49.4	142	422	29.7
823	SBR	5	77	137	233	26.2
824	OD	5	86.5	172	337	25.6
825	AO+AAO	2.5	90.2	159	319	31.7
826	Wetland	1	89.9	155	289	26.3
827	Biofilm	1	93.7	142	301	32.2
828	AO+AAO	2	100	151	374	29
829	Others	0.8	88.7	155	334	37.9
830	OD	6	90.1	124	300	23.4
831	AO+AAO	4	72.7	101	289	24.7
832	OD	2.5	92.2	136	261	17
833	AO+AAO	1	20.8	39.2	105	10.2
834	AO+AAO	2	91.8	169	287	28
835	Biolak	1.5	50.3	134	266	38.6

836	AO+AAO	2	67.2	173	360	31.3
837	AO+AAO	10	90.8	133	257	25
838	Others	3	104	104	282	14.5
839	OD	2	104	120	235	22.9
840	AO+AAO	4	89	115	282	34.8
841	Others	2	62.8	143	379	28.8
842	Biolak	3	102	121	272	28
843	AO+AAO	4	106	168	264	28.4
844	AO+AAO	3	90.5	90.5	174	28.4
845	AO+AAO	6	52.9	95.7	269	26.1
846	AO+AAO	8	90	149	398	20.7
847	AO+AAO	8	88.6	137	298	21.9
848	Others	2	80.3	144	268	30.6
849	AO+AAO	3	100	133	355	19.6
850	AO+AAO	3	100	101	345	25.8
851	AO+AAO	2	20	67	111	16.7
852	AO+AAO	15	104	150	270	22.4
853	AO+AAO	1.5	94.6	82	240	24
854	Others	6	73.7	114	313	27.2
855	AO+AAO	4	106	41	154	18
856	AO+AAO	4	61.8	215	267	26
857	AO+AAO	5	93.6	122	259	22
858	AO+AAO	3	55.8	92.2	306	30.6
859	AO+AAO	3	55.2	142	372	33.8
860	Others	5	64.8	96	366	24
861	AO+AAO	4	74.3	80.5	273	33
862	OD	2	81.2	82.5	331	25
863	OD	2	99.5	124	255	31.5
864	AO+AAO	3	60.6	116	275	22.1
865	AO+AAO	2.5	38.3	94.5	270	23.6
866	AO+AAO	7.5	88.5	67	216	20.6
867	AO+AAO	4	84.5	53.9	210	24.1
868	AO+AAO	4	52.4	80	329	34.4
869	Others	8	88.6	92.2	272	35.8
870	AO+AAO	1.5	36.2	180	420	55
871	AO+AAO	1.9	66.8	86.7	210	34
872	AO+AAO	4	53.5	57.6	247	26.4
873	AO+AAO	2	14.3	25.5	120	8.38
874	AO+AAO	2	18.5	120	230	30
875	AO+AAO	2	41	85	183	18.2

876	AO+AAO	6	72.1	123	294	20.5
877	AO+AAO	8	88.3	117	378	24.8
878	AO+AAO	16	79.8	101	319	31.3
879	OD	1.5	84.3	86.5	221	31.2
880	AO+AAO	1.5	81.6	243	411	29.2
881	AO+AAO	3	87.9	103	232	26.7
882	SBR	3	79.3	180	170	30
883	AO+AAO	8	103	113	280	29.2
884	AO+AAO	4	110	102	282	26.5
885	AO+AAO	40	124	93.7	264	32.8
886	OD	20	110	213	391	34.2
887	AO+AAO	60	78.6	232	413	37.7
888	OD	3	78.1	141	352	26
889	OD	2	99.8	85.4	267	27.5
890	SBR	3	71.6	278	493	65.8
891	AO+AAO	10	131	174	255	25.3
892	OD	3	93.7	17.3	235	18.5
893	OD	3	24.5	89.7	239	23.8
894	OD	2	101	106	205	24
895	OD	3	60.3	72.5	338	7.64
896	OD	3	85.4	60.2	100	15.3
897	OD	3	100	107	193	28.8
898	AO+AAO	30	113	92	200	20.3
899	OD	7	95.5	58	154	13.5
900	AO+AAO	7	80.3	58.3	163	8.47
901	Others	18	67.4	124	263	13.2
902	AO+AAO	4.5	87.4	25.1	74.3	7.1
903	AO+AAO	10	65.9	103	249	22.3
904	AO+AAO	10	73.4	102	248	21.2
905	AO+AAO	10	75.6	98.6	234	24.7
906	SBR	1.2	76	109	263	23
907	AO+AAO	8	76.4	64.2	194	20.3
908	OD	4	61.2	51.7	132	11
909	Biofilm	1	61.8	51	122	11.4
910	Others	2	99.9	120	182	19.8
911	OD	4	101	70.6	253	13.6
912	OD	2	107	51.6	140	14.8
913	AO+AAO	5	72.3	123	320	11.7
914	Biofilm	1	92.1	75	150	14
915	OD	2.5	74.5	62	158	16.6

916	OD	2	102	63.6	212	15.6
917	OD	2	112	45	165	16
918	OD	2	98.2	57.6	166	11.3
919	AO+AAO	17	77.2	87.2	246	13.2
920	SBR	2	95.8	225	365	29
921	OD	3	50.2	71.3	139	17.3
922	OD	10	94.8	39.5	120	12.1
923	OD	3	99.9	63.4	149	19.8
924	OD	2	137	68.7	145	19
925	OD	3	83.5	65.5	166	19.4
926	Others	2	103	41.3	90.4	10.5
927	OD	10	97	45	182	16.6
928	OD	3	105	33	100	15
929	AO+AAO	10	89.3	65.3	208	8.3
930	SBR	1	93.8	30.1	92.5	23.4
931	Biofilm	10	92.2	85.7	239	17.5
932	AO+AAO	1	92.5	45.4	129	21.5
933	SBR	1	89.3	51.8	134	20.2
934	SBR	1	92.3	60.2	140	21.9
935	SBR	2	93.3	52.4	134	17.5
936	OD	10	86	135	410	17.1
937	OD	2	94.5	48.3	135	17.6
938	Others	22	106	131	203	20.6
939	AO+AAO	33	68.5	90	257	18.5
940	AO+AAO	20	97	90.5	186	24
941	AO+AAO	20	116	80.1	144	14.4
942	AO+AAO	20	78.3	70.4	141	19.1
943	AO+AAO	5	92.3	116	188	20.7
944	AO+AAO	3	97.6	74	165	16.5
945	AO+AAO	15	63.6	105	217	24.8
946	MBR	10	85.3	81.2	158	23.8
947	OD	5	62.8	50.6	130	16.6
948	OD	2	44.3	76.8	190	12.2
949	SBR	2	32.3	5.1	13.4	3.99
950	OD	1.1	41	31	81.7	9.47
951	AO+AAO	1.6	99.7	65.2	147	28.1
952	OD	1	39.5	289	708	11.3
953	AO+AAO	2	41.5	49.1	153	12.8
954	AO+AAO	4.8	100	31.2	91	10.8
955	AO+AAO	4.9	125	68.1	127	19.8

956	AO+AAO	2.5	27.9	23.1	83.3	14.3
957	SBR	0.13	40	67.4	213	11.1
958	SBR	4	88.2	41.6	97	18.2
959	SBR	2	67.9	59.7	147	13.8
960	SBR	3	92.9	37.7	101	10.6
961	SBR	20	113	71.4	136	14.2
962	SBR	4	81.8	32	91	22.3
963	OD	4	98.2	88	180	21.5
964	SBR	2	106	24.4	53	4.83
965	Others	2	90.5	51	104	13
966	SBR	4	87.1	35.3	95	9.29
967	SBR	10	99.9	91.6	226	18.1
968	AO+AAO	2.5	93.7	60.5	147	8.59
969	AO+AAO	10	100	52	156	16.8
970	AO+AAO	15	86.7	100	261	15.6
971	AO+AAO	2	47.6	50	137	21
972	AO+AAO	0.3	85.3	25	72.5	4.62
973	AO+AAO	22	126	61.7	172	20.8
974	AO+AAO	30	94.6	90.5	186	24
975	Others	30	92	202	415	38
976	Others	35	93.3	229	582	20
977	SBR	6	92.6	47.3	153	23.8
978	AO+AAO	73.6	98.2	147	316	28.8
979	Others	8	47.4	129	280	29.8
980	AO+AAO	8	72.2	112	293	19.2
981	SBR	10	101	92.2	254	21.9
982	AO+AAO	20	101	59.2	193	21
983	AO+AAO	4	128	189	334	29.6
984	Others	20	7.84	107	335	52.6
985	OD	20	73.2	74.9	159	15.8
986	OD	3	63	231	634	29.1
987	SBR	12	61.8	134	267	19.8
988	AO+AAO	20	115	75.3	221	22.2
989	AO+AAO	15	77	84	213	14.5
990	Biofilm	40	97.5	48.7	156	21
991	AO+AAO	15	105	32.7	117	28.6
992	AO+AAO	15	70.5	143	368	27.6
993	AO+AAO	24	87.3	132	375	23.9
994	Biofilm	5	65	111	255	23.7
995	AO+AAO	12.5	92.9	77.5	186	25.9

996	AO+AAO	15	106	61.5	178	22
997	SBR	3	67.4	52.2	116	17.9
998	AO+AAO	5	79.9	86.4	204	20
999	AO+AAO	5	121	47	142	13.8
1000	AO+AAO	10	118	3.46	5.55	0.43
1001	SBR	4	103	33.8	87.9	7.54
1002	SBR	4	52.2	41.6	115	8.83
1003	AO+AAO	20	105	56.3	187	25
1004	AO+AAO	25	96.4	180	289	28.8
1005	AO+AAO	10	93.9	145	291	21.1
1006	OD	3	112	94.2	230	23.2
1007	OD	5	111	94.2	230	23.2
1008	OD	5	83.3	78.7	193	19.7
1009	SBR	1.8	83.3	115	254	19.1
1010	SBR	5	119	53	161	16.9
1011	AO+AAO	8	99.3	115	206	21.8
1012	AO+AAO	5.5	102	138	252	17.4
1013	Others	3	110	49.4	160	14.7
1014	AO+AAO	2.646	99.4	47.4	156	16.6
1015	OD	2	110	51.4	170	17.5
1016	Others	5	72.8	41.5	196	10.1
1017	SBR	3.5	99.4	53.5	184	21
1018	SBR	3.5	91.7	84	182	15.2
1019	Others	3	105	62.2	172	13.8
1020	AO+AAO	6	78.3	64.1	162	14.9
1021	AO+AAO	7.5	58	53.9	148	18.8
1022	AO+AAO	26	98.7	80.4	155	24.9
1023	Others	12	82.4	95.7	241	7.74
1024	AO+AAO	10	20.2	73.3	151	16.2
1025	AO+AAO	3.2	69.3	85	230	13

1026	Others	3	79.8	81.7	191	12.1
1027	Others	3	87.8	99.6	259	12.9
1028	Others	20	95.6	57.3	191	26.9
1029	Biofilm	4	80.5	59.6	189	20.8
1030	Biofilm	0.5	94.2	69.2	186	16.5
1031	Others	4	81	58.6	187	19.4
1032	Others	8	88.2	74.8	200	18
1033	OD	0.5	45.2	69.2	127	16.5
1034	Others	10	101	77.1	182	21.8
1035	OD	4	102	67.7	168	18.4
1036	Others	0.5	42.3	49.7	112	16.4
1037	AO+AAO	8	123	65.3	149	11.3
1038	Wetland	0.4	82.2	35.2	135	7.97
1039	Others	0.4	57.7	37.3	183	13.7
1040	Others	0.2	82.6	96.3	168	24.3
1041	OD	5	95.4	62	140	16.2
1042	SBR	0.5	78.2	58	202	16
1043	Wetland	0.5	72.6	65	148	36
1044	SBR	8	62.9	72.7	184	23.7
1045	AO+AAO	0.8	94.3	65.2	131	30.2
1046	Wetland	0.15	67.4	86.9	140	13.6
1047	Wetland	0.3	75.6	95.7	131	13.3
1048	AO+AAO	1	30.1	30.4	95.6	15.5
1049	AO+AAO	0.3	60.7	45	105	11
1050	Others	2	120	84.1	169	14.3
1051	Wetland	0.3	90.4	60	150	10
1052	Others	5	23.6	47	128	20.7
1053	AO+AAO	5	98.6	39.8	109	19
1054	AO+AAO	6	94	34.2	117	14.9

1055	AO+AAO	10	39.2	27.5	107	10.7
1056	SBR	8	46.8	38.4	100	10.4
1057	SBR	2	102	25.8	95.2	15.4
1058	OD	4	8.1	40	140	18
1059	OD	5	99.7	40	140	18
1060	AO+AAO	2	77	38.4	99.4	19.1
1061	AO+AAO	2	70.2	60	186	17.7
1062	AO+AAO	2	32.7	19.8	94.4	11.2
1063	AO+AAO	0.5	104	30.7	85.4	13
1064	Others	3	99.4	28.3	104	16
1065	SBR	5	88	24.7	87.3	21.1
1066	SBR	5	86.8	24.7	98.8	15.7
1067	Wetland	2	88.1	41	118	18.2
1068	AO+AAO	2	87.2	42	117	18
1069	AO+AAO	4	93	52.1	127	17.3
1070	AO+AAO	5	55.8	70.4	177	19.9
1071	AO+AAO	8	80.9	68.9	175	18
1072	AO+AAO	1.5	82.7	55.5	125	15.2
1073	AO+AAO	5	26.3	39.5	101	8.93
1074	OD	4	100	35.6	111	19
1075	OD	4	102	39.6	122	21
1076	OD	2	95.7	13.4	73.4	7.75
1077	Others	4	71.5	485	1413	135
1078	Others	1	47.2	5.5	70.2	3.01
1079	Others	3	36	17.3	96.2	9.24
1080	OD	2	102	27.5	118	13.4
1081	Others	1	68.7	11.9	66.4	7.39
1082	Wetland	3	90.4	35	118	18.2
1083	AO+AAO	6	67.5	63.3	126	16.3

1084	Wetland	1	49.7	16.8	51.4	4.18
1085	AO+AAO	1	53.4	18	52.7	5.02
1086	AO+AAO	1	76.6	22.4	66.2	5.54
1087	Others	1.5	86.2	15.9	43.2	4.55
1088	Others	2	87.3	42.5	136	10.2
1089	OD	2	95.5	60	124	12.4
1090	SBR	4	122	75	200	18.1
1091	OD	5	108	64.1	159	18.2
1092	AO+AAO	10	93.9	54.1	170	17.2
1093	AO+AAO	5	112	65	145	11.8
1094	OD	1.5	75.8	74	153	15.2
1095	AO+AAO	4	90.6	70	168	18.3
1096	SBR	12	120	105	200	20.6
1097	AO+AAO	12	76.7	65.3	145	15.4
1098	OD	4	130	57.3	200	16
1099	SBR	6	101	64.7	127	22
1100	SBR	3	84.9	23.7	99.5	11.5
1101	Others	10	90.7	56.5	140	16.5
1102	Others	10	90.7	56.5	140	16.5
1103	SBR	2	87.6	68.8	173	19.8
1104	Others	1	100	81.5	179	19.9
1105	OD	1	102	57	126	19.7
1106	Wetland	1	71.6	94.6	167	19.2
1107	SBR	2	100	71.1	169	19.8
1108	OD	3	91.6	106	191	23.5
1109	AO+AAO	4	79.5	67.7	165	18.7
1110	Others	9	35.4	84.1	160	20.5
1111	OD	1	86.9	95.7	254	21.7
1112	SBR	4	81.1	67.8	162	20.2

1113	OD	4	79.9	52	122	11.1
1114	AO+AAO	2	60	69	169	20
1115	Others	10	94	79.2	177	12.9
1116	OD	6	86.1	54.7	164	14.5
1117	Wetland	1.2	68	13.9	50.3	14.2
1118	SBR	1	119	92.8	243	19.6
1119	SBR	1	37.1	35	60.1	13
1120	Biofilm	0.3	88.7	68.9	166	27.7
1121	AO+AAO	10	77.2	144	260	20.4
1122	AO+AAO	4.5	65.8	143	260	20.3
1123	AO+AAO	10	64.4	155	275	26.8
1124	AO+AAO	3.35	95.6	99	179	22.2
1125	SBR	2	55.9	117	217	24.9
1126	AO+AAO	3	105	82	152	8.06
1127	SBR	1	73.1	51	82	10.3
1128	SBR	3	59.5	51.8	90	9.9
1129	SBR	7	103	81	157	15.3
1130	Others	3	85.5	60	161	23.2
1131	AO+AAO	20	69.5	62	160	14.8
1132	AO+AAO	1	92.8	49.8	111	20
1133	OD	1.5	91.3	52	122	18.6
1134	SBR	1	82.6	44.4	132	22.3
1135	AO+AAO	1	96.7	80.2	143	18.7
1136	OD	1.5	84.5	40.5	135	16.7
1137	OD	2	41.9	47.2	112	19.6
1138	Others	30	77.1	149	332	19.7
1139	AO+AAO	1.5	51.9	45	139	18.3
1140	AO+AAO	20	76.4	81.5	176	18
1141	OD	5	41	28.9	75.9	12.9

1142	Others	8	132	72.7	222	29.1
1143	CAS	1	71.4	48.6	133	11.9
1144	AO+AAO	1.5	55	52.4	160	29.2
1145	Biofilm	1.5	63.9	46.5	155	20.5
1146	AO+AAO	3	34.7	35.5	103	12.1
1147	CAS	1.5	90.4	80.5	274	24.8
1148	AO+AAO	1.5	70.9	15	86.8	11.5
1149	AO+AAO	1.5	29.9	39	110	14
1150	OD	0.5	93.7	40.7	105	19.6
1151	SBR	1.5	65.3	13.8	86.7	12.4
1152	OD	2.5	63.4	20	87.2	13.5
1153	OD	3	83.5	35.9	126	14.4
1154	Biofilm	0.7	78.9	18.6	107	7.31
1155	SBR	2.5	45.1	28.6	131	12.8
1156	SBR	1.4	57.5	20.8	73.2	11.7
1157	SBR	3	77.8	19.8	93.6	12.1
1158	AO+AAO	1.5	65.1	29.5	121	15.6
1159	OD	1	82.5	38.1	116	18.9
1160	OD	1.5	106	21.6	90.4	13.8
1161	AO+AAO	1	54.5	11.8	82.7	10.9
1162	AO+AAO	3	62.2	20.7	85.3	10.8
1163	SBR	6	87.7	65	270	35.2
1164	SBR	6	90.9	56.4	281	15.4
1165	Biofilm	4	63.9	60	167	24.3
1166	OD	2	90.4	94.8	219	30.7
1167	SBR	6	65.3	85.8	210	21.1
1168	SBR	8	100	97.1	217	23.7
1169	AO+AAO	0.8	79.4	89.6	227	29.9
1170	OD	0.6	86.1	73	213	31

1171	AO+AAO	0.6	105	100	243	33.6
1172	AO+AAO	2	100	41.4	165	5.56
1173	SBR	1.5	70.7	70	221	26.3
1174	OD	1	77	126	270	30
1175	AO+AAO	1	72.9	58.5	189	29.7
1176	SBR	1	50.6	83.8	214	30.9
1177	SBR	1.5	81.6	80	195	17.9
1178	SBR	1	42.5	126	245	41.4
1179	OD	2.5	95.1	72	215	25.7
1180	SBR	0.75	53.3	56	145	17.7
1181	SBR	1	75.1	93	192	34.2
1182	SBR	1	70.5	59.2	188	21.4
1183	SBR	1	70.6	72	196	29.9
1184	SBR	1	62.1	73.4	174	23
1185	SBR	1	59.6	63.6	145	22.6
1186	SBR	1	52.9	98	187	24.5
1187	SBR	0.6	62.4	51	165	22.9
1188	SBR	0.75	42.5	105	242	40.3
1189	SBR	1	72.8	135	234	36.9
1190	SBR	1.5	33.9	126	240	30
1191	Others	1	57	73.2	164	23.3
1192	SBR	1	81.8	81	288	18.8
1193	SBR	0.8	49.7	74.7	194	19
1194	OD	20	84.2	270	700	34.8
1195	AO+AAO	50	75.8	220	420	23
1196	AO+AAO	20	87.2	249	483	35.1
1197	AO+AAO	10	86.1	117	418	28.6
1198	OD	4	97.5	107	236	34.2
1199	SBR	1.5	72.7	151	402	28.2

1200	OD	8	84	168	462	38.3
1201	OD	3	58.1	175	328	37.9
1202	OD	5	126	180	359	23.4
1203	OD	10	60.9	175	593	17.6
1204	OD	10	115	186	365	27.9
1205	OD	5	92.1	208	728	42
1206	AO+AAO	2.5	89	57.1	255	19.1
1207	OD	1.5	76.5	130	304	42.7
1208	SBR	10	101	246	465	29.8
1209	OD	6	62.3	145	318	46
1210	Biolak	2	38.4	128	280	41
1211	Biolak	2	53.4	221	487	40
1212	Others	1.5	35.6	249	618	67.3
1213	Others	1	70.7	171	453	64
1214	Others	3	46.2	11.2	70.9	105
1215	SBR	3	77.8	120	329	26.3

Table A.31 Mean, maximum and minimum value of wastewater quality for 1215 WWTPs in China

Parameter	Influent BOD concentration (mg/L)	Influent COD concentration (mg/L)	Influent NH4 concentration (mg/L)
Mean value	103.00	248.54	24.65
Maximum value	540.00	1413.00	268.00
Minimum value	3.46	5.55	0.43

Table A.32 Mean value and ranges of leachates, and the industrial wastewater quality

Parameter		TSS (mg/L)	BOD (mg/L)	COD (mg/L)	TN (mg/L)
Mean value	Young leachate	1,390	7,478	14,548	2,094
	Medium leachate	277	2,454	8,238	2,179
	Old leachate	327	574	3,953	2,539
	Meat processing wastewater	1,373	1,215	3,282	298
	Tannery wastewater	1,510	1,179	3,696	269
	Textile wastewater	338	255	853	31
Maximum value	Young leachate	2,400	10,800	24,400	2,517

	Medium leachate	595	5,550	13,646	2,928
	Old leachate	633	1,375	5,200	3,147
	Meat processing wastewater	3,438	1,602	5,422	427
	Tannery wastewater	2,690	1,760	6,240	358
	Textile wastewater	520	455	1,411	49
Minimum value	Young leachate	724	4,680	9,310	1,172
	Medium leachate	143	965	4,975	1,365
	Old leachate	30	190	3,105	1,889
	Meat processing wastewater	625	891	1,697	217
	Tannery wastewater	890	463	2,155	132
	Textile wastewater	137	150	513	19

Table A.33 Treatment systems for different papers

Paper	System	Figure	Wastewater	Primary treatment	Secondary treatments	Tertiary treatment
Unit Energy and Cost Comparison of Two Treatment Systems of Young, Medium and Old Leachates	A	3.1	Young, medium and old leachate	Micro sieving	Activated sludge process with direct line nitrification and denitrification process	Activated carbon
	B	3.2	Young, medium and old leachate	Micro sieving	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A), and activated sludge process with direct line nitrification and denitrification process	Activated carbon
Effect of Micro-Sieving on the Unit Energy and Cost in Food Wastewater Treatment System	A	4.1	Meat processing wastewater	Primary clarifier or micro sieving	Activated sludge process with direct line nitrification and denitrification process	
	B	4.2	Meat processing wastewater	Primary clarifier or micro sieving	Activated sludge process with recycle line nitrification/denitrification	
	C	4.3	Meat processing wastewater	Primary clarifier or micro sieving	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A)	
	D	4.4	Meat processing wastewater		Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A)	
	E	4.3	Meat processing wastewater	Primary clarifier	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A)	
	F	4.3	Meat processing wastewater	Micro sieving	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A)	
Treatment performance, energy, and cost analysis of	A	5.1	Meat processing wastewater, tannery	Primary clarifier	Activated sludge process with direct line nitrification and denitrification process	

energy-positive wastewater treatment systems for different industrial wastewaters			wastewater and textile wastewater			
	B	5.2	Meat processing wastewater, tannery wastewater and textile wastewater	Micro sieving	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A), and activated sludge process with direct line nitrification and denitrification process	
Last section: Unit energy and cost of systems with innovative technologies for treating leachate and industrial wastewater	A	3.2	Young, medium, old leachate, meat processing wastewater, tannery wastewater and textile wastewater	Micro sieving	Up-flow anaerobic sludge blanket (UASB), partial nitrification/Anammox (PN/A), and activated sludge process with direct line nitrification and denitrification process	Activated carbon

Table A.34 Unit energy consumption with different operation loading rate

Operation loading rate (%)	0-20	20-40	40-60	60-80	80-100	>=100
SBR	6.189	2.631	1.771	1.638	1.225	1.464
OD	9.349	3.596	2.332	1.713	1.625	1.625
AO+AAO	5.119	3.653	2.170	1.697	1.531	1.575
Biolak		2.820	1.447	1.755	1.556	0.933
AS			3.095	1.383	1.011	0.792
Biofilm		1.745	1.590	2.524	1.418	1.294
Wetland			2.496	1.988	1.388	
Others	2.257	1.945	2.417	1.208	1.485	1.717
MBR	2.698			2.210	2.483	1.043

Table A.35 Unit energy consumption with different COD removal efficiency

COD removal efficiency (%)	60-70	70-80	80-90	90-100
SBR	4.634	2.275	1.618	1.296
OD	12.241	3.422	1.850	1.284
AO+AAO	6.157	4.178	1.769	1.466
Biolak		3.478	1.748	1.348
AS			2.252	0.943
Biofilm		1.782	2.043	1.303
Wetland	5.151	1.277	1.746	1.086
Others	4.054	3.450	1.486	1.276
MBR			2.210	2.279

Table A.36 Unit energy consumption with different influent COD concentration

Influent COD concentration (mg/L)	0-150	150-250	250-450	450-1000
SBR	2.846	1.408	1.310	0.629
OD	3.010	1.525	1.292	0.828
AO+AAO	3.894	1.780	1.291	0.770
Biolak	3.823	1.573	1.395	0.667
AS	2.810	1.887	0.937	
Biofilm	3.830	1.691	1.342	1.044
Wetland	2.364	1.120	1.315	
Others	2.952	1.631	1.189	
MBR		2.584	2.783	1.182

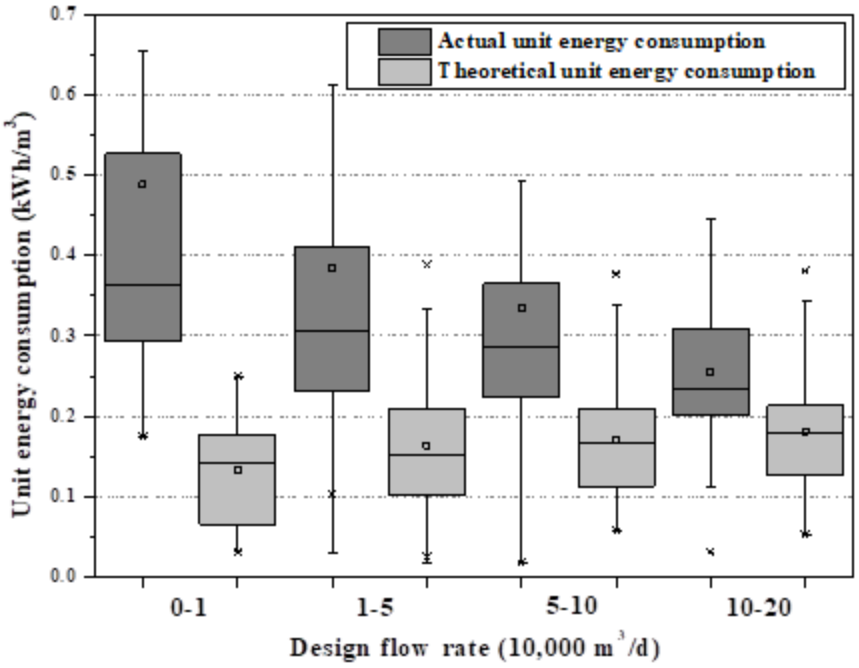


Figure A.10 Actual and theoretical unit energy consumption per m^3 flow rate of WWTPs with AO+AAO technology

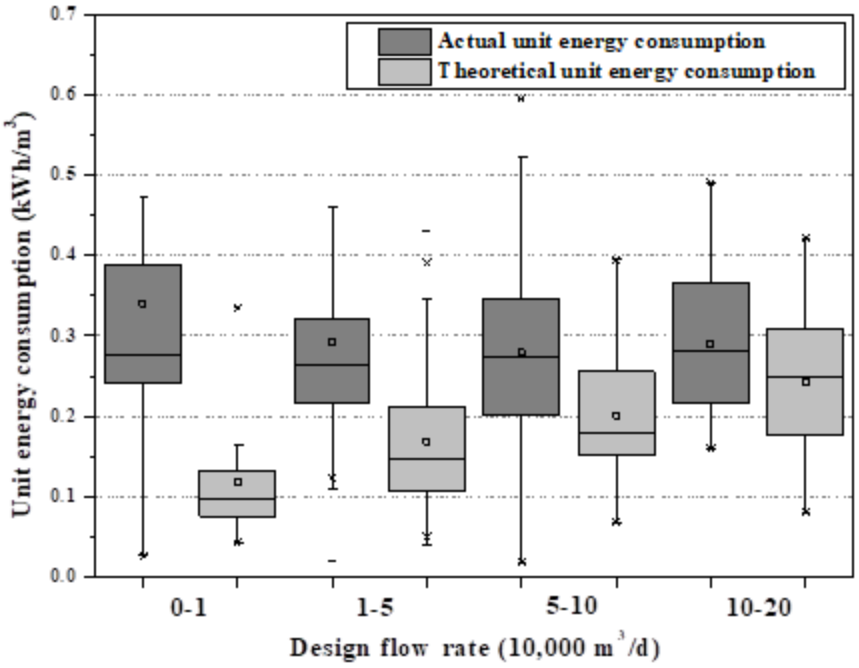


Figure A.11 Actual and theoretical unit energy consumption per m^3 flow rate of WWTPs with OD technology

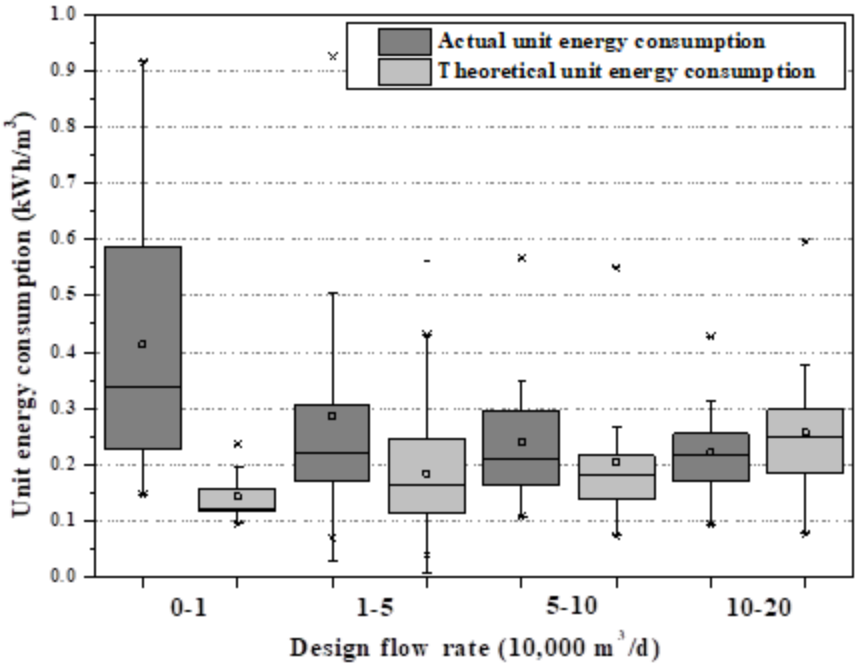
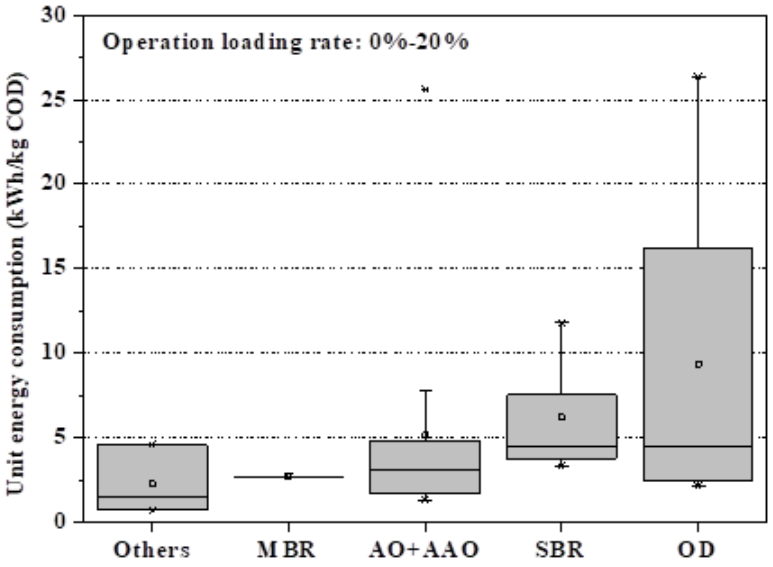
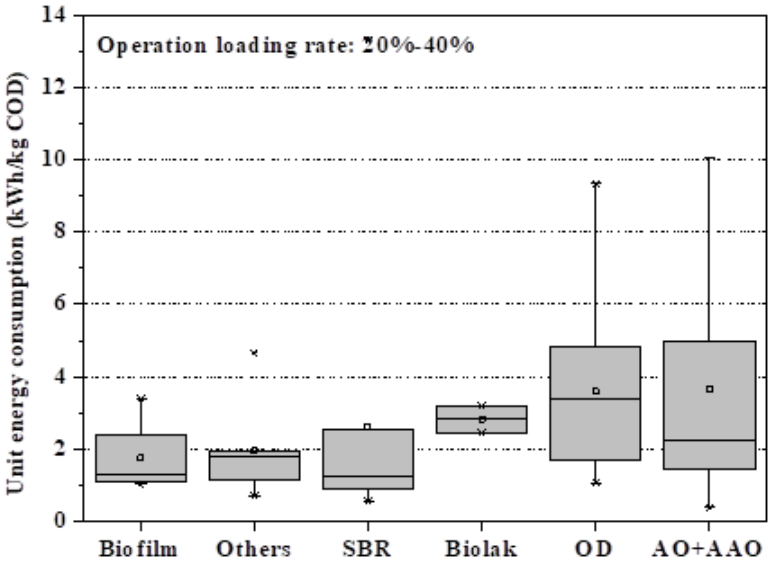
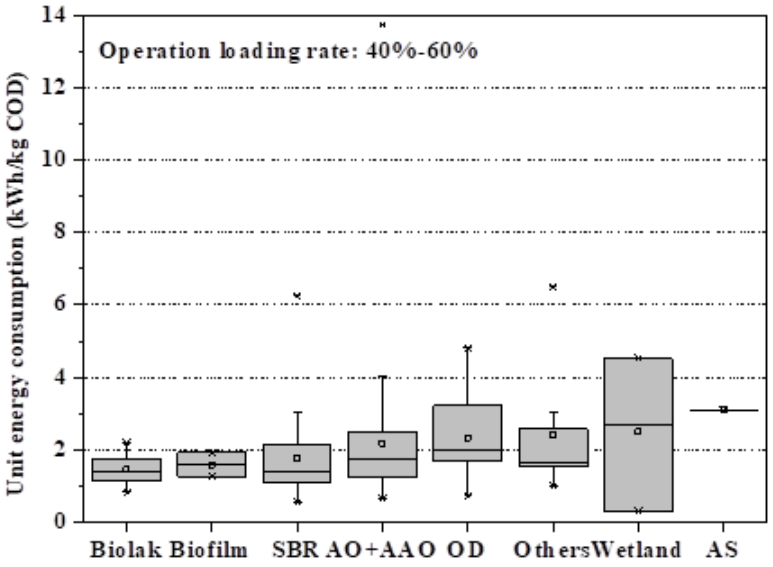
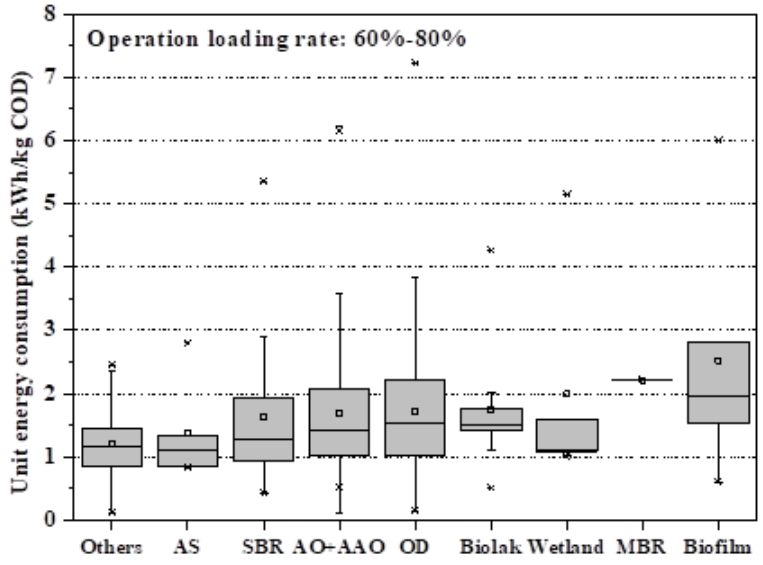


Figure A.12 Actual and theoretical unit energy consumption per m^3 flow rate of WWTPs with SBR technology


Operation loading rate: 0%-20%
Unit energy consumption (kWh/kg COD)
30
25
20
15
10
5
0
Others
MBR
AO+AAO
SBR
OD


Operation loading rate: 20%-40%
Unit energy consumption (kWh/kg COD)
14
12
10
8
6
4
2
0
Biofilm
Others
SBR
Biolak
OD
AO+AAO


Operation loading rate: 40%-60%
Unit energy consumption (kWh/kg COD)
0
2
4
6
8
10
12
14
Biolak
Biofilm
SBR
AO+AAO
OD
Others
Wetland
AS


Operation loading rate: 60%-80%
Unit energy consumption (kWh/kg COD)
0
1
2
3
4
5
6
7
8
Others
AS
SBR
AO+AAO
OD
Biolak
Wetland
MBR
Biofilm

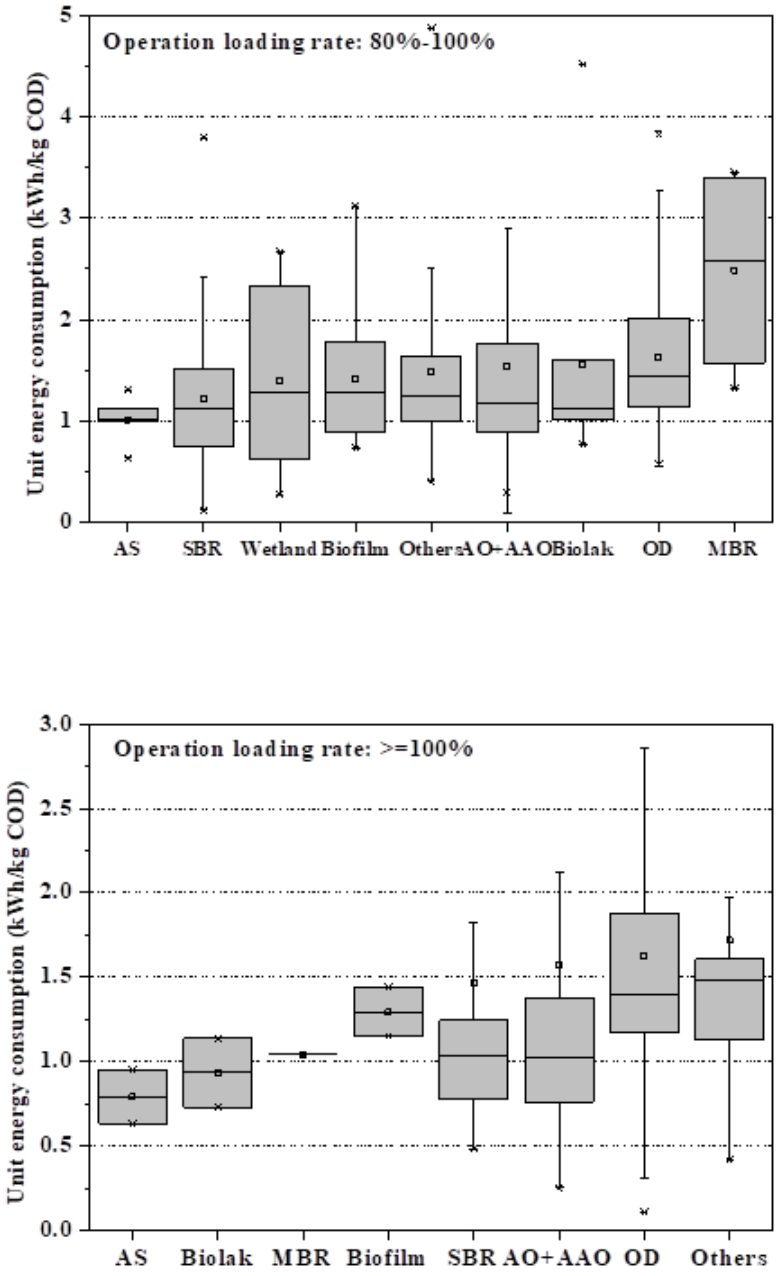
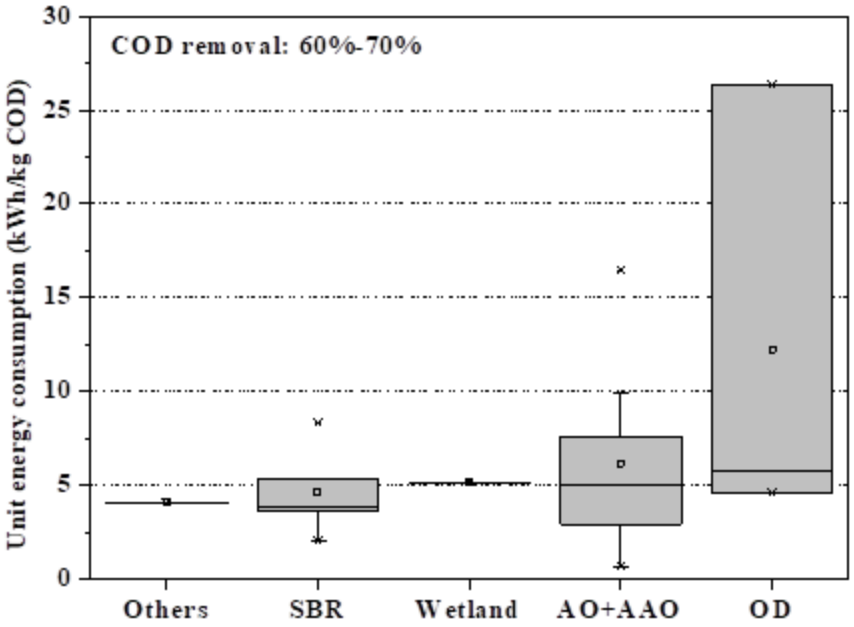
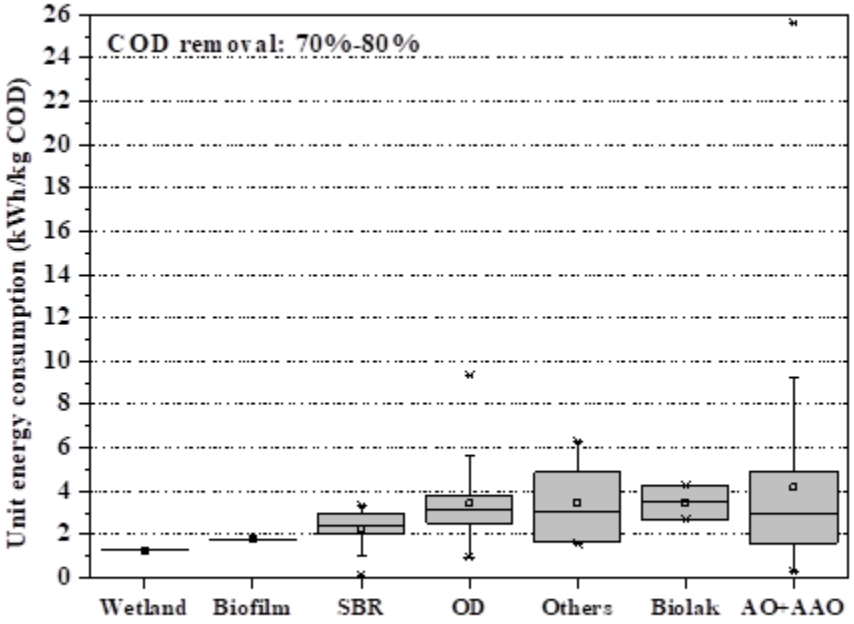


Figure A.13 Unit energy consumption with different operation loading rate


COD removal: 60%-70%
Unit energy consumption (kWh/kg COD)
30
25
20
15
10
5
0
Others
SBR
Wetland
AO+AAO
OD


COD removal: 70%-80%
Unit energy consumption (kWh/kg COD)
26
24
22
20
18
16
14
12
10
8
6
4
2
0
Wetland
Biofilm
SBR
OD
Others
Biolak
AO-AAO

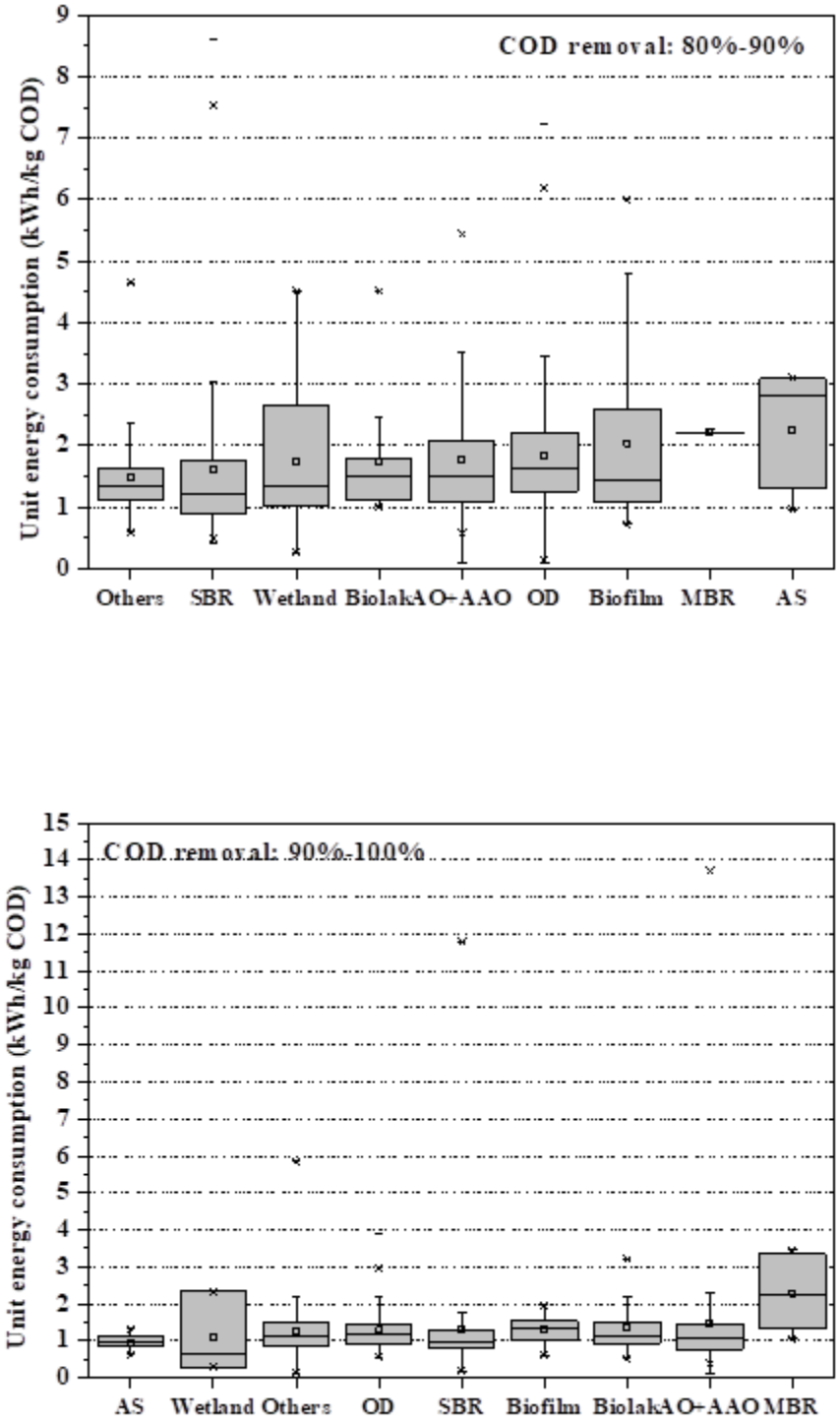
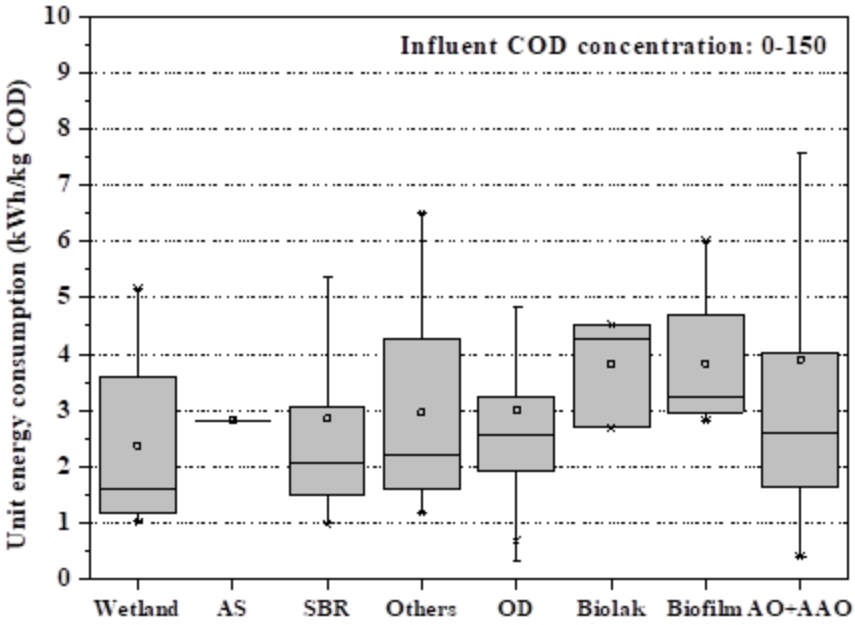
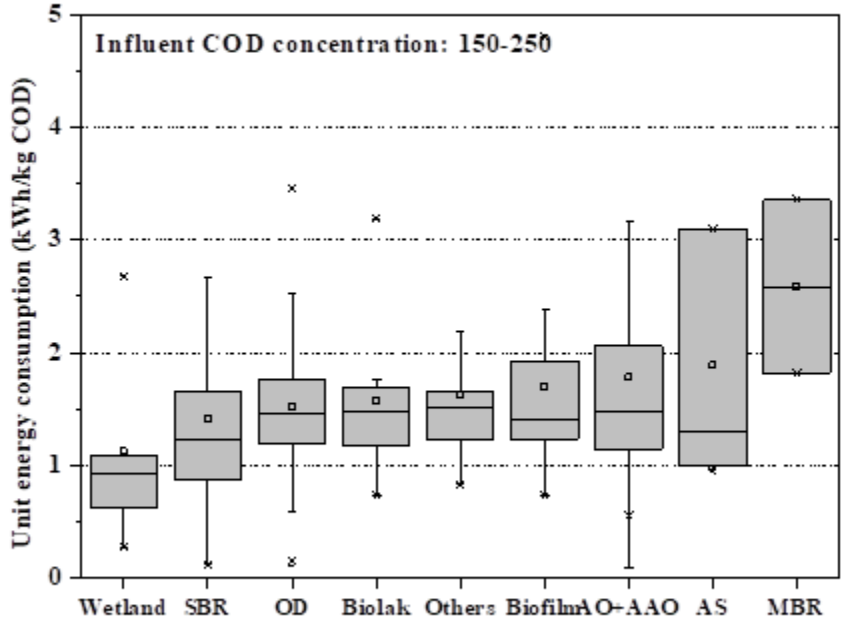
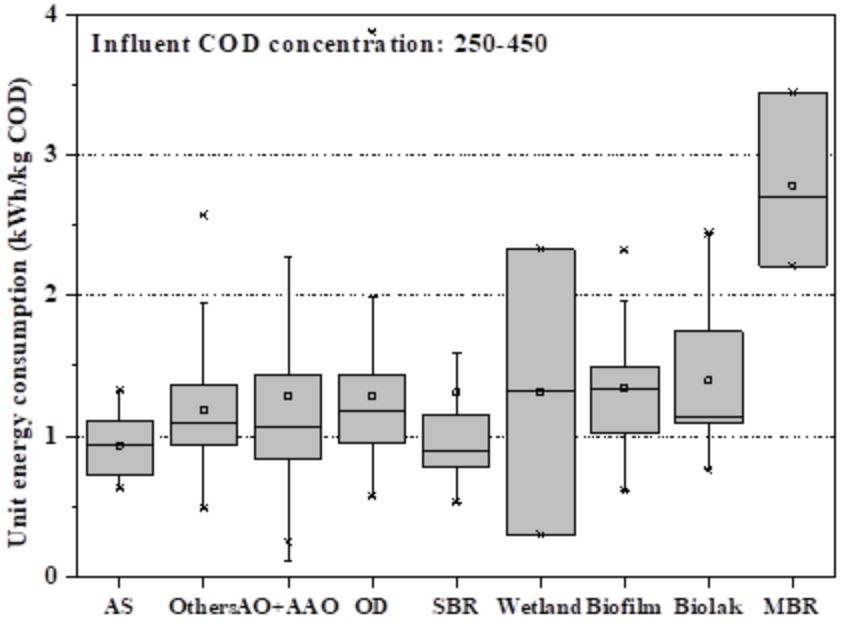


Figure A.14 Unit energy consumption with different COD removal efficiency


Influent COD concentration: 0-150
Unit energy consumption (kWh/kg COD)
0
1
2
3
4
5
6
7
8
9
10
Wetland
AS
SBR
Others
OD
Biolak
Biofilm
AO-AAO


Influent COD concentration: 150-250
Unit energy consumption (kWh/kg COD)
0
1
2
3
4
5
Wetland
SBR
OD
Biolak
Others
Biofilm
AO-AAO
AS
MBR



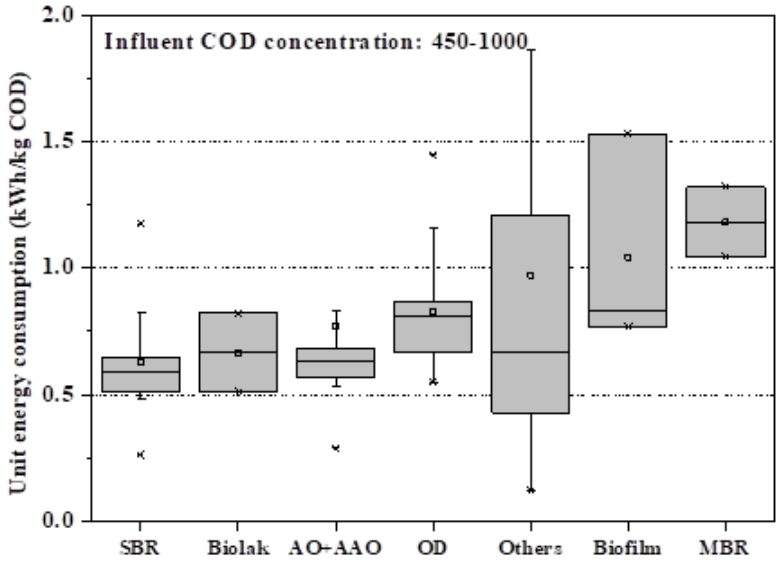


Figure A.15 Unit energy consumption with different influent COD concentration

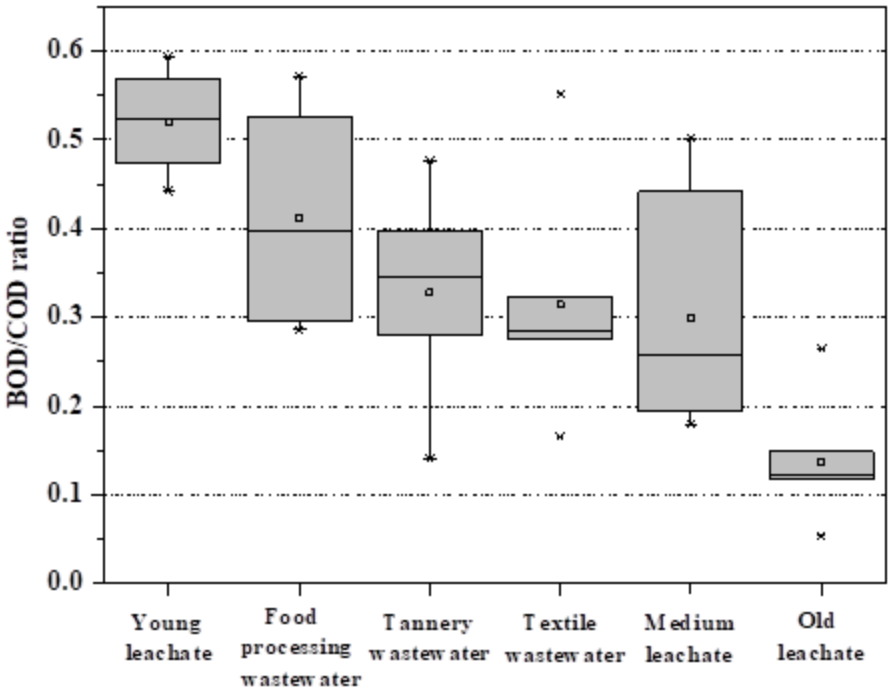


Figure A.16 BOD/COD ratio of leachate and industrial wastewater

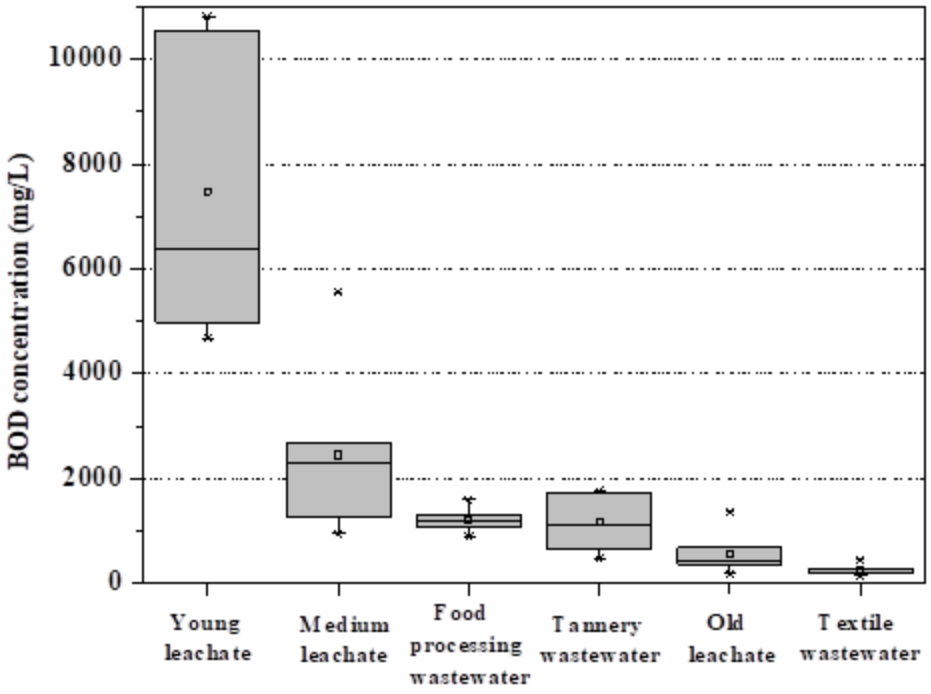
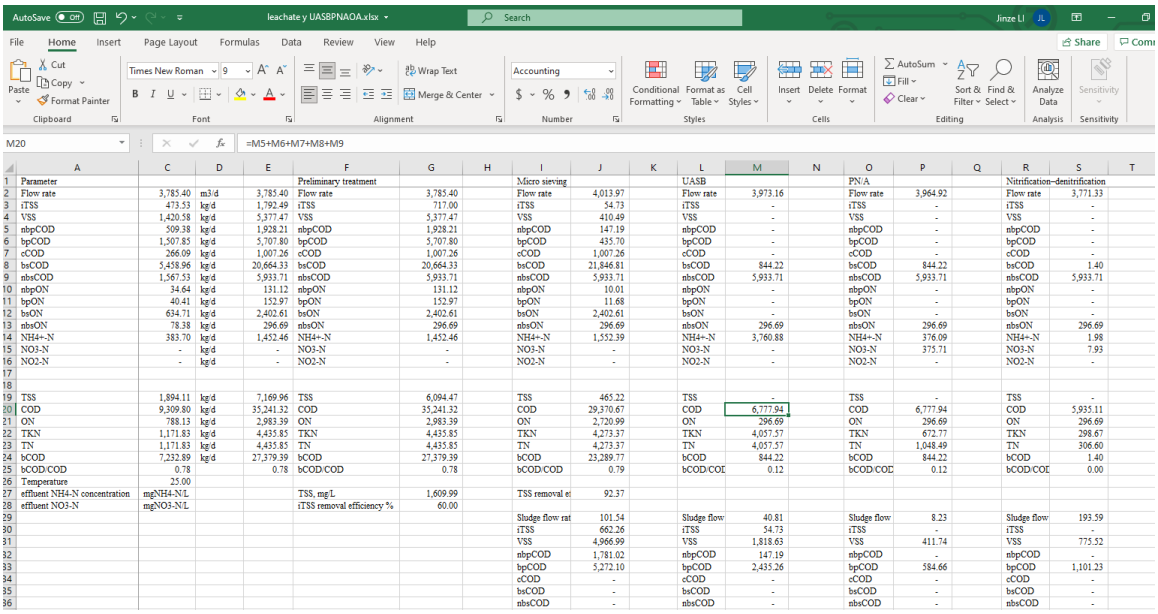
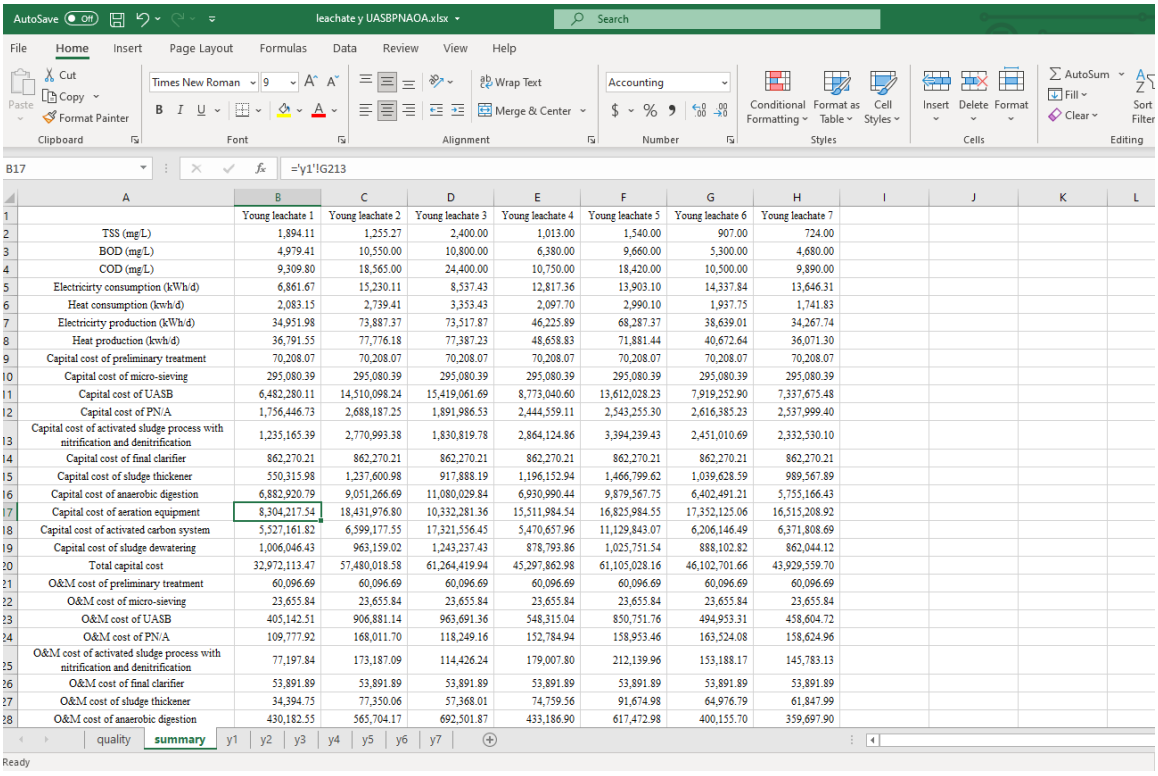


Figure A.17 BOD concentration of leachate and industrial wastewater



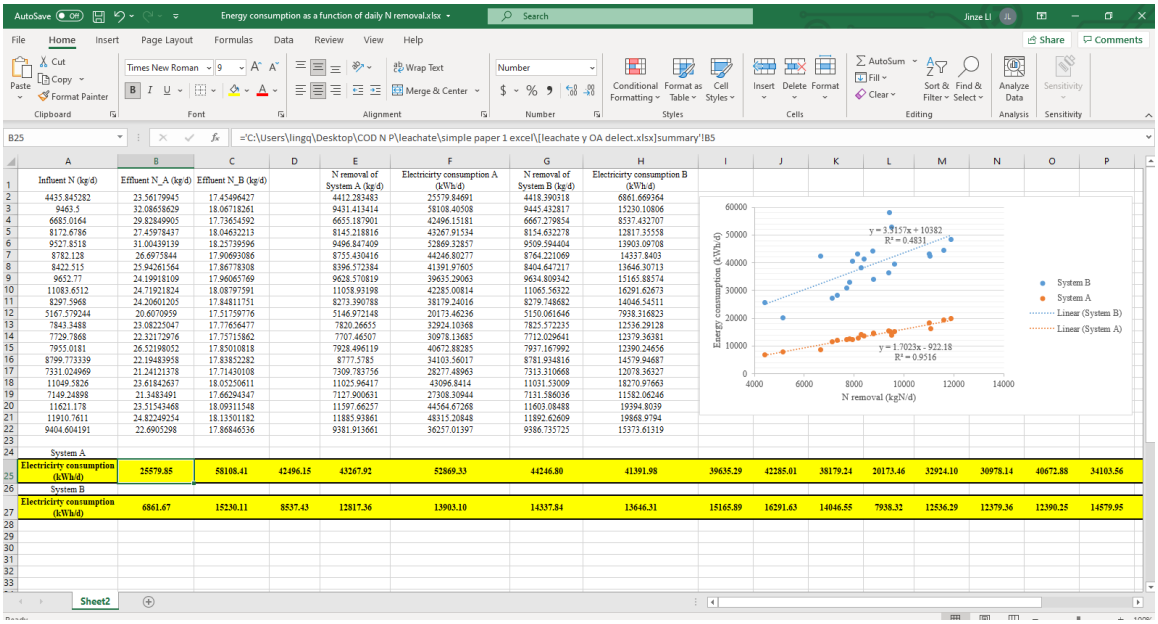
	A	C	D	E	F	G	H	I	J	K	L	M	N	O	P	Q	R	S	T
1	Parameter				Preliminary treatment			Micro sieving			UASB			PN/A			Nitrification–denitrification		
2	Flow rate	3,785.40	m3/d	3,785.40	Flow rate	3,785.40		Flow rate	4,013.97		Flow rate	3,973.16		Flow rate	3,964.92		Flow rate	3,771.33	
3	iTSS	473.53	kg/d	1,792.49	iTSS	717.00		iTSS	54.73		iTSS	-		iTSS	-		iTSS	-	
4	VSS	1,420.58	kg/d	5,377.47	VSS	5,377.47		VSS	410.49		VSS	-		VSS	-		VSS	-	
5	nbpCOD	509.38	kg/d	1,928.21	nbpCOD	1,928.21		nbpCOD	147.19		nbpCOD	-		nbpCOD	-		nbpCOD	-	
6	bpCOD	1,507.85	kg/d	5,707.80	bpCOD	5,707.80		bpCOD	435.70		bpCOD	-		bpCOD	-		bpCOD	-	
7	cCOD	266.09	kg/d	1,007.26	cCOD	1,007.26		cCOD	1,007.26		cCOD	-		cCOD	-		cCOD	-	
8	bsCOD	5,458.96	kg/d	20,664.33	bsCOD	20,664.33		bsCOD	21,846.81		bsCOD	844.22		bsCOD	844.22		bsCOD	1.40	
9	nbsCOD	1,567.53	kg/d	5,933.71	nbsCOD	5,933.71		nbsCOD	5,933.71		nbsCOD	5,933.71		nbsCOD	5,933.71		nbsCOD	5,933.71	
10	nbpON	34.64	kg/d	131.12	nbpON	131.12		nbpON	10.01		nbpON	-		nbpON	-		nbpON	-	
11	bpON	40.41	kg/d	152.97	bpON	152.97		bpON	11.68		bpON	-		bpON	-		bpON	-	
12	bsON	634.71	kg/d	2,402.61	bsON	2,402.61		bsON	2,402.61		bsON	-		bsON	-		bsON	-	
13	nbsON	78.38	kg/d	296.69	nbsON	296.69		nbsON	296.69		nbsON	296.69		nbsON	296.69		nbsON	296.69	
14	NH4+-N	383.70	kg/d	1,452.46	NH4+-N	1,452.46		NH4+-N	1,552.39		NH4+-N	3,760.88		NH4+-N	376.09		NH4+-N	1.98	
15	NO3-N	-	kg/d	-	NO3-N	-		NO3-N	-		NO3-N	-		NO3-N	375.71		NO3-N	7.93	
16	NO2-N	-	kg/d	-	NO2-N	-		NO2-N	-		NO2-N	-		NO2-N	-		NO2-N	-	
17																			
18																			
19	TSS	1,894.11	kg/d	7,169.96	TSS	6,094.47		TSS	465.22		TSS	-		TSS	-		TSS	-	
20	COD	9,309.80	kg/d	35,241.32	COD	35,241.32		COD	29,370.67		COD	6,777.94		COD	6,777.94		COD	5,935.11	
21	ON	788.13	kg/d	2,983.39	ON	2,983.39		ON	2,720.99		ON	296.69		ON	296.69		ON	296.69	
22	TKN	1,171.83	kg/d	4,435.85	TKN	4,435.85		TKN	4,273.37		TKN	4,057.57		TKN	672.77		TKN	298.67	
23	TN	1,171.83	kg/d	4,435.85	TN	4,435.85		TN	4,273.37		TN	4,057.57		TN	1,048.49		TN	306.60	
24	bCOD	7,232.89	kg/d	27,379.39	bCOD	27,379.39		bCOD	23,289.77		bCOD	844.22		bCOD	844.22		bCOD	1.40	
25	bCOD/COD	0.78		0.78	bCOD/COD	0.78		bCOD/COD	0.79		bCOD/COD	0.12		bCOD/COD	0.12		bCOD/COD	0.00	
26	Temperature	25.00																	
27	effluent NH4-N concentration	mgNH4-N/L			TSS, mg/L	1,609.99		TSS removal ef	92.37										
28	effluent NO3-N	mgNO3-N/L			iTSS removal efficiency %	60.00													
29								Sludge flow rat	101.54		Sludge flow	40.81		Sludge flow	8.23		Sludge flow	193.59	
30								iTSS	662.26		iTSS	54.73		iTSS	-		iTSS	-	
31								VSS	4,966.99		VSS	1,818.63		VSS	411.74		VSS	775.52	
32								nbpCOD	1,781.02		nbpCOD	147.19		nbpCOD	-		nbpCOD	-	
33								bpCOD	5,272.10		bpCOD	2,435.26		bpCOD	584.66		bpCOD	1,101.23	
34								cCOD	-		cCOD	-		cCOD	-		cCOD	-	
35								bsCOD	-		bsCOD	-		bsCOD	-		bsCOD	-	
36								nbsCOD	-		nbsCOD	-		nbsCOD	-		nbsCOD	-	

Figure A.18 Excel model of leachate treatment system for treating young leachate



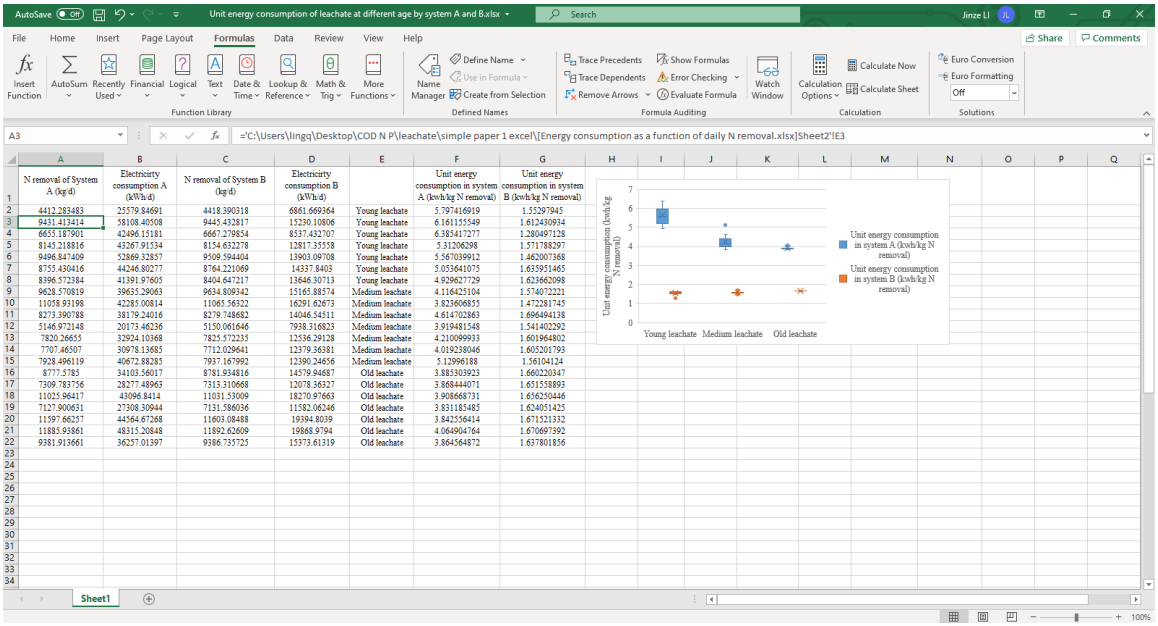
	A	B	C	D	E	F	G	H	I	J	K	L
1		Young leachate 1	Young leachate 2	Young leachate 3	Young leachate 4	Young leachate 5	Young leachate 6	Young leachate 7				
2	TSS (mg/L)	1,894.11	1,255.27	2,400.00	1,013.00	1,540.00	907.00	724.00				
3	BOD (mg/L)	4,979.41	10,550.00	10,800.00	6,380.00	9,660.00	5,300.00	4,680.00				
4	COD (mg/L)	9,309.80	18,565.00	24,400.00	10,750.00	18,420.00	10,500.00	9,890.00				
5	Electricirty consumption (kWh/d)	6,861.67	15,230.11	8,537.43	12,817.36	13,903.10	14,337.84	13,646.31				
6	Heat consumption (kwh/d)	2,083.15	2,739.41	3,353.43	2,097.70	2,990.10	1,937.75	1,741.83				
7	Electricirty production (kWh/d)	34,951.98	73,887.37	73,517.87	46,225.89	68,287.37	38,639.01	34,267.74				
8	Heat production (kwh/d)	36,791.55	77,776.18	77,387.23	48,658.83	71,881.44	40,672.64	36,071.30				
9	Capital cost of preliminary treatment	70,208.07	70,208.07	70,208.07	70,208.07	70,208.07	70,208.07	70,208.07				
10	Capital cost of micro-sieving	295,080.39	295,080.39	295,080.39	295,080.39	295,080.39	295,080.39	295,080.39				
11	Capital cost of UASB	6,482,280.11	14,510,098.24	15,419,061.69	8,773,040.60	13,612,028.23	7,919,252.90	7,337,675.48				
12	Capital cost of PN/A	1,756,446.73	2,688,187.25	1,891,986.53	2,444,559.11	2,543,255.30	2,616,385.23	2,537,999.40				
13	Capital cost of activated sludge process with nitrification and denitrification	1,235,165.39	2,770,993.38	1,830,819.78	2,864,124.86	3,394,239.43	2,451,010.69	2,332,530.10				
14	Capital cost of final clarifier	862,270.21	862,270.21	862,270.21	862,270.21	862,270.21	862,270.21	862,270.21				
15	Capital cost of sludge thickener	550,315.98	1,237,600.98	917,888.19	1,196,152.94	1,466,799.62	1,039,628.59	989,567.89				
16	Capital cost of anaerobic digestion	6,882,920.79	9,051,266.69	11,080,029.84	6,930,990.44	9,879,567.75	6,402,491.21	5,755,166.43				
17	Capital cost of aeration equipment	8,304,217.54	18,431,976.80	10,332,281.36	15,511,984.54	16,825,984.55	17,352,125.06	16,515,208.92				
18	Capital cost of activated carbon system	5,527,161.82	6,599,177.55	17,321,556.45	5,470,657.96	11,129,843.07	6,206,146.49	6,371,808.69				
19	Capital cost of sludge dewatering	1,006,046.43	963,159.02	1,243,237.43	878,793.86	1,025,751.54	888,102.82	862,044.12				
20	Total capital cost	32,972,113.47	57,480,018.58	61,264,419.94	45,297,862.98	61,105,028.16	46,102,701.66	43,929,559.70				
21	O&M cost of preliminary treatment	60,096.69	60,096.69	60,096.69	60,096.69	60,096.69	60,096.69	60,096.69				
22	O&M cost of micro-sieving	23,655.84	23,655.84	23,655.84	23,655.84	23,655.84	23,655.84	23,655.84				
23	O&M cost of UASB	405,142.51	906,881.14	963,691.36	548,315.04	850,751.76	494,953.31	458,604.72				
24	O&M cost of PN/A	109,777.92	168,011.70	118,249.16	152,784.94	158,953.46	163,524.08	158,624.96				
25	O&M cost of activated sludge process with nitrification and denitrification	77,197.84	173,187.09	114,426.24	179,007.80	212,139.96	153,188.17	145,783.13				
26	O&M cost of final clarifier	53,891.89	53,891.89	53,891.89	53,891.89	53,891.89	53,891.89	53,891.89				
27	O&M cost of sludge thickener	34,394.75	77,350.06	57,368.01	74,759.56	91,674.98	64,976.79	61,847.99				
28	O&M cost of anaerobic digestion	430,182.55	565,704.17	692,501.87	433,186.90	617,472.98	400,155.70	359,697.90				

Figure A.19 Summary of capital and O&M cost of unit process in treatment system



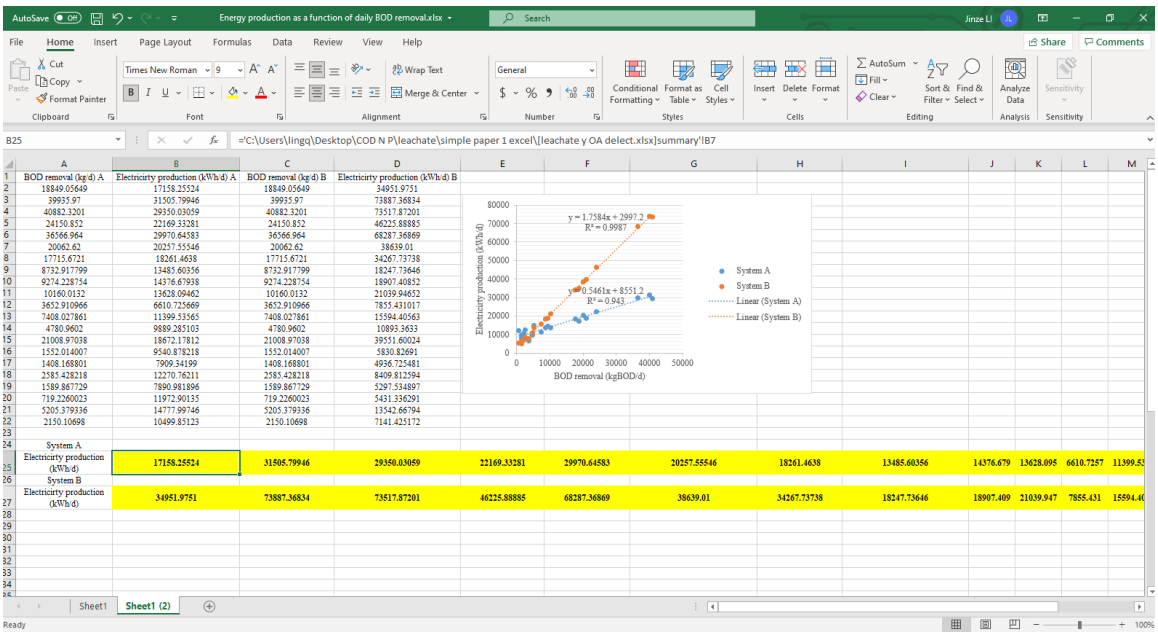
Influent N (kg/d)	Effluent N_A (kg/d)	Effluent N_B (kg/d)	N removal of System A (kg/d)	Electricity consumption A (kWh/d)	N removal of System B (kg/d)	Electricity consumption B (kWh/d)
4435.845282	23.56179945	17.45496427	4412.283483	25579.84691	4418.390318	6861.669364
9463.5	32.08658629	18.06718261	9431.413414	58108.40508	9445.432817	15230.10806
6685.0164	29.82849905	17.73654592	6655.187901	42496.15181	6667.279854	8537.432707
8172.6786	27.45978437	18.04632213	8145.218816	43267.91534	8154.632278	12817.35558
9527.8518	31.00439139	18.25739596	9496.847409	52869.32857	9509.594404	13903.09708
8782.128	26.69975844	17.90693086	8755.430416	44246.80277	8764.221069	14337.8403
8422.515	25.94261564	17.86778308	8396.572384	41391.97605	8404.647217	13646.30713
9652.77	24.19918109	17.96065769	9628.570819	39635.29063	9634.809342	15165.88574
11083.6512	24.71921824	18.08797591	11058.93198	42285.00814	11065.56322	16291.62673
8297.5968	24.20601205	17.84811751	8273.390788	38179.24016	8279.748682	14046.54511
5167.579244	20.6070959	17.51759776	5146.972148	20173.46236	5150.061646	7938.316823
7843.3488	23.08225947	17.77656477	7820.26655	32924.10368	7825.572235	12536.29128
7729.7868	22.32172976	17.75715862	7707.46507	30978.13685	7712.029641	12379.36381
7955.0181	26.52198052	17.85010818	7928.496119	40672.88285	7937.167992	12390.24656
8799.773339	22.19483958	17.83852282	8777.5785	34103.56017	8781.934816	14579.94687
7331.024969	21.24121378	17.71430108	7309.783756	28277.48963	7313.310668	12078.36327
11049.5826	23.61842637	18.05250611	11025.96417	43096.8414	11031.53009	18270.97663
7149.24898	21.3483491	17.66294347	7127.900631	27308.30944	7131.586036	11582.06246
11621.178	23.51543468	18.09311548	11597.66257	44564.67268	11603.08488	19394.8039
11910.7611	24.82249254	18.13501182	11885.93861	48315.20848	11892.62609	19868.9794
9404.604191	22.6905298	17.86846536	9381.913661	36257.01397	9386.735725	15373.61319
System A						
Electricity consumption (kWh/d)	**25579.85**	**58108.41**	**42496.15**	**43267.92**	**52869.33**	**44246.80**	**41391.98**	**39635.29**	**42285.01**	**38179.24**	**20173.46**	**32924.10**	**30978.14**	**40672.88**	**34103.56**
System B															
Electricity consumption (kWh/d)	**6861.67**	**15230.11**	**8537.43**	**12817.36**	**13903.10**	**14337.84**	**13646.31**	**15165.89**	**16291.63**	**14046.55**	**7938.32**	**12536.29**	**12379.36**	**12390.25**	**14579.95**

Figure A.20 Energy consumption as a function of daily N removal



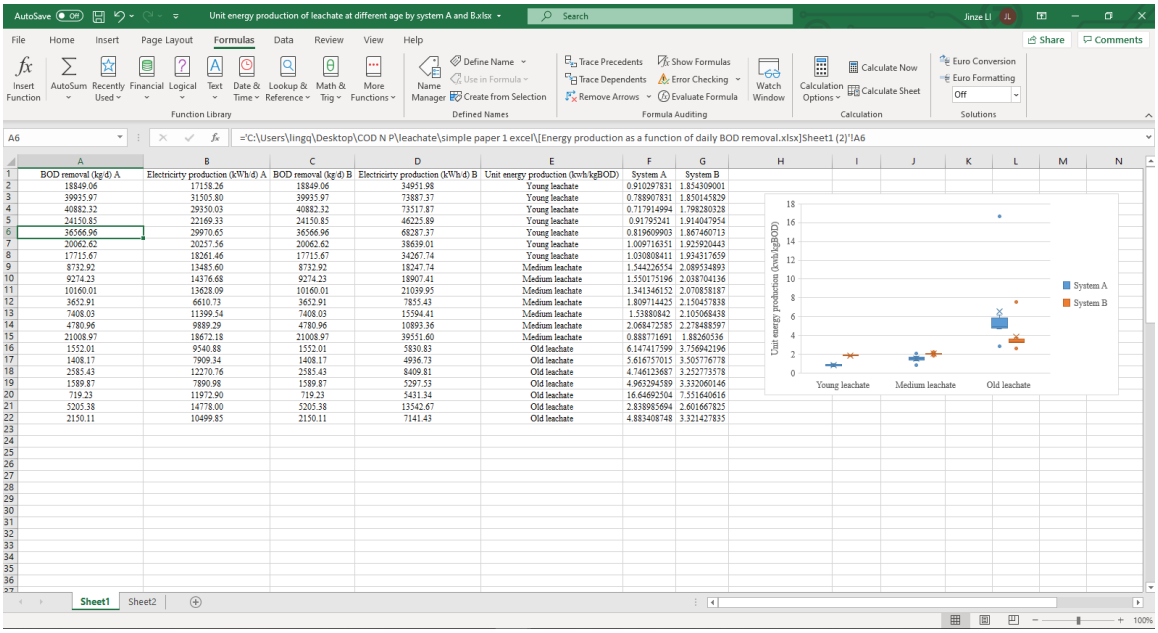
N removal of System A (kg/d)	Electricity consumption A (kWh/d)	N removal of System B (kg/d)	Electricity consumption B (kWh/d)		Unit energy consumption in system A (kwh/kg N removal)	Unit energy consumption in system B (kwh/kg N removal)
4412.283483	25579.84691	4418.390318	6861.669364	Young leachate	5.797416919	1.55297945
9431.413414	58108.40508	9445.432817	15230.10806	Young leachate	6.161155549	1.612430934
6655.187901	42496.15181	6667.279854	8537.432707	Young leachate	6.385417277	1.280497128
8145.218816	43267.91534	8154.632278	12817.35558	Young leachate	5.31206298	1.571788297
9496.847409	52869.32857	9509.594404	13903.09708	Young leachate	5.567039912	1.462007368
8755.430416	44246.80277	8764.221069	14337.8403	Young leachate	5.053641075	1.635951465
8396.572384	41391.97605	8404.647217	13646.30713	Young leachate	4.929627729	1.623662098
9628.570819	39635.29063	9634.809342	15165.88574	Medium leachate	4.116425104	1.574072221
11058.93198	42285.00814	11065.56322	16291.62673	Medium leachate	3.823606855	1.472281745
8273.390788	38179.24016	8279.748682	14046.54511	Medium leachate	4.614702863	1.696494138
5146.972148	20173.46236	5150.061646	7938.316823	Medium leachate	3.919481548	1.541402292
7820.26655	32924.10368	7825.572235	12536.29128	Medium leachate	4.210099933	1.601964802
7707.46507	30978.13685	7712.029641	12379.36381	Medium leachate	4.019238046	1.605201793
7928.496119	40672.88285	7937.167992	12390.24656	Medium leachate	5.12996188	1.56104124
8777.5785	34103.56017	8781.934816	14579.94687	Old leachate	3.885303923	1.660220347
7309.783756	28277.48963	7313.310668	12078.36327	Old leachate	3.868444071	1.651558893
11025.96417	43096.8414	11031.53009	18270.97663	Old leachate	3.908668731	1.656250446
7127.900631	27308.30944	7131.586036	11582.06246	Old leachate	3.831185485	1.624051425
11597.66257	44564.67268	11603.08488	19394.8039	Old leachate	3.842556414	1.671521332
11885.93861	48315.20848	11892.62609	19868.9794	Old leachate	4.064904764	1.670697392
9381.913661	36257.01397	9386.735725	15373.61319	Old leachate	3.864564872	1.637801856

Figure A.21 Unit energy consumption of leachate treatment system



BOD removal (kg/d) A	Electricity production (kWh/d) A	BOD removal (kg/d) B	Electricity production (kWh/d) B
18849.05649	17158.25524	18849.05649	34951.9751
39935.97	31505.79946	39935.97	73887.36834
40882.3201	29350.03059	40882.3201	73517.87201
24150.852	22169.33281	24150.852	46225.88885
36566.964	29970.64583	36566.964	68287.36869
20062.62	20257.55546	20062.62	38639.01
17715.6721	18261.4638	17715.6721	34267.73738
8732.917799	13485.60356	8732.917799	18247.73646
9274.228754	14376.67938	9274.228754	18907.40852
10160.0132	13628.09462	10160.0132	21039.94652
3652.910966	6610.725669	3652.910966	7855.431017
7408.027861	11399.53565	7408.027861	15594.40563
4780.9602	9889.285103	4780.9602	10893.3633
21008.97038	18672.17812	21008.97038	39551.60024
1552.014007	9540.878218	1552.014007	5830.82691
1408.168801	7909.34199	1408.168801	4936.725481
2585.428218	12270.76211	2585.428218	8409.812594
1589.867729	7890.981896	1589.867729	5297.534897
719.2260023	11972.90135	719.2260023	5431.336291
5205.379336	14777.99746	5205.379336	13542.66794
2150.10698	10499.85123	2150.10698	7141.425172

Figure A.22 Energy production as a function of daily BOD removal



BOD removal (kg/d) A	Electricity production (kWh/d) A	BOD removal (kg/d) B	Electricity production (kWh/d) B	Unit energy production (kwh/kgBOD)	System A	System B
18849.06	17158.26	18849.06	34951.98	Young leachate	0.910297831	1.854309001
39935.97	31505.80	39935.97	73887.37	Young leachate	0.788907831	1.850145829
40882.32	29350.03	40882.32	73517.87	Young leachate	0.717914994	1.798280328
24150.85	22169.33	24150.85	46225.89	Young leachate	0.91795241	1.914047954
36566.96	29970.65	36566.96	68287.37	Young leachate	0.819609903	1.867460713
20062.62	20257.56	20062.62	38639.01	Young leachate	1.009716351	1.925920443
17715.67	18261.46	17715.67	34267.74	Young leachate	1.030808411	1.934317659
8732.92	13485.60	8732.92	18247.74	Medium leachate	1.544226554	2.089534893
9274.23	14376.68	9274.23	18907.41	Medium leachate	1.550175196	2.038704136
10160.01	13628.09	10160.01	21039.95	Medium leachate	1.341346152	2.070858187
3652.91	6610.73	3652.91	7855.43	Medium leachate	1.809714425	2.150457838
7408.03	11399.54	7408.03	15594.41	Medium leachate	1.53880842	2.105068438
4780.96	9889.29	4780.96	10893.36	Medium leachate	2.068472585	2.278488597
21008.97	18672.18	21008.97	39551.60	Medium leachate	0.888771691	1.88260536
1552.01	9540.88	1552.01	5830.83	Old leachate	6.147417599	3.756942196
1408.17	7909.34	1408.17	4936.73	Old leachate	5.616757015	3.505776778
2585.43	12270.76	2585.43	8409.81	Old leachate	4.746123687	3.252773578
1589.87	7890.98	1589.87	5297.53	Old leachate	4.963294589	3.332060146
719.23	11972.90	719.23	5431.34	Old leachate	16.64692504	7.551640616
5205.38	14778.00	5205.38	13542.67	Old leachate	2.838985694	2.601667825
2150.11	10499.85	2150.11	7141.43	Old leachate	4.883408748	3.321427835

Figure A.23 Unit energy production of leachate treatment system

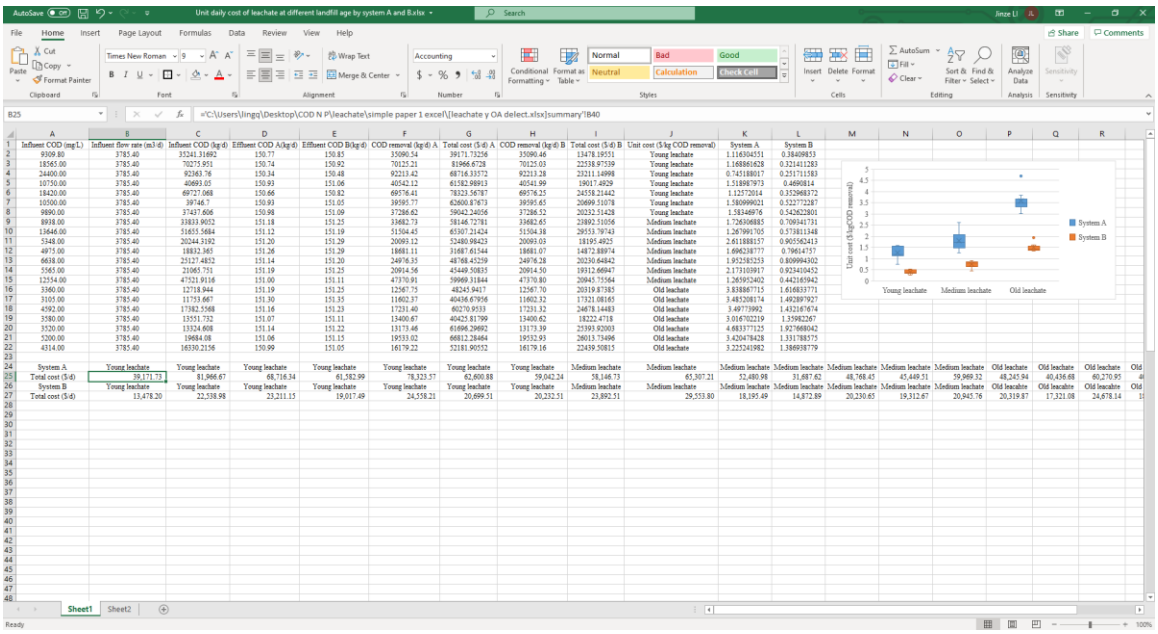
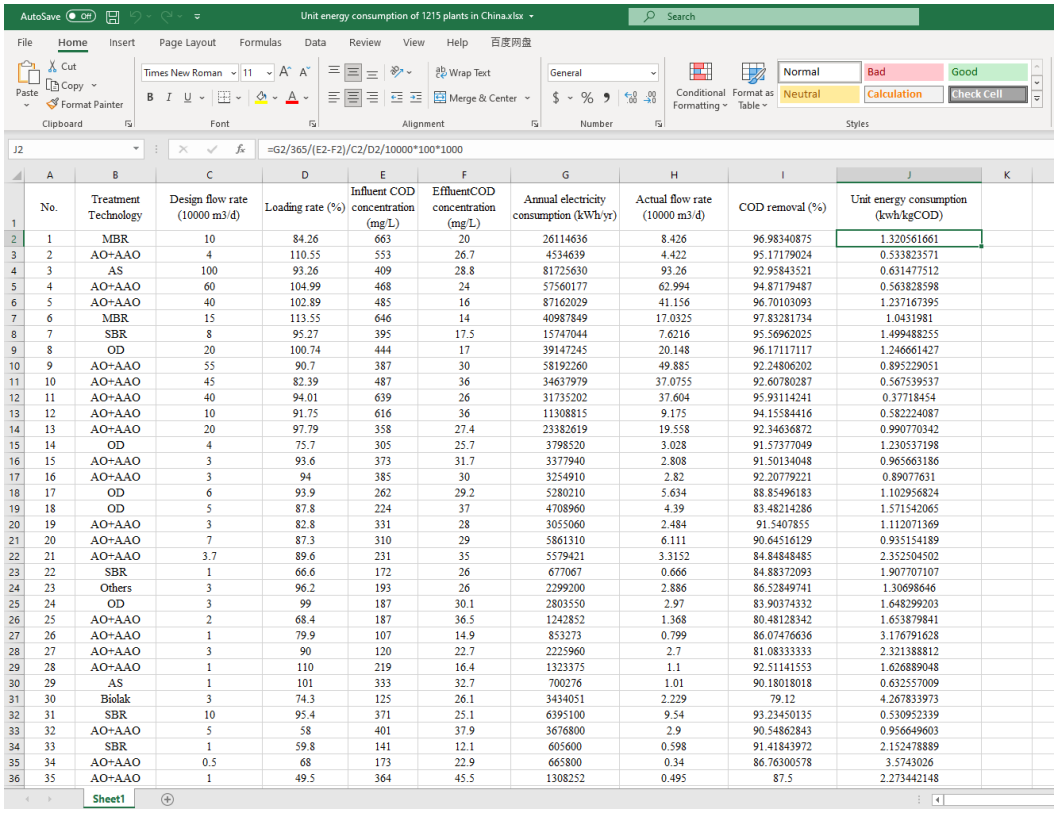


Figure A.24 Unit cost of leachate treatment system



No.	Treatment Technology	Design flow rate (10000 m3/d)	Loading rate (%)	Influent COD concentration (mg/L)	EffluentCOD concentration (mg/L)	Annual electricity consumption (kWh/yr)	Actual flow rate (10000 m3/d)	COD removal (%)	Unit energy consumption (kwh/kgCOD)
1	MBR	10	84.26	663	20	26114636	8.426	96.98340875	1.320561661
2	AO+AAO	4	110.55	553	26.7	4534639	4.422	95.17179024	0.533823571
3	AS	100	93.26	409	28.8	81725630	93.26	92.95843521	0.631477512
4	AO+AAO	60	104.99	468	24	57560177	62.994	94.87179487	0.563828598
5	AO+AAO	40	102.89	485	16	87162029	41.156	96.70103093	1.237167395
6	MBR	15	113.55	646	14	40987849	17.0325	97.83281734	1.0431981
7	SBR	8	95.27	395	17.5	15747044	7.6216	95.56962025	1.499488255
8	OD	20	100.74	444	17	39147245	20.148	96.17117117	1.246661427
9	AO+AAO	55	90.7	387	30	58192260	49.885	92.24806202	0.895229051
10	AO+AAO	45	82.39	487	36	34637979	37.0755	92.60780287	0.567539537
11	AO+AAO	40	94.01	639	26	31735202	37.604	95.93114241	0.37718454
12	AO+AAO	10	91.75	616	36	11308815	9.175	94.15584416	0.582224087
13	AO+AAO	20	97.79	358	27.4	23382619	19.558	92.34636872	0.990770342
14	OD	4	75.7	305	25.7	3798520	3.028	91.57377049	1.230537198
15	AO+AAO	3	93.6	373	31.7	3377940	2.808	91.50134048	0.965663186
16	AO+AAO	3	94	385	30	3254910	2.82	92.20779221	0.89077631
17	OD	6	93.9	262	29.2	5280210	5.634	88.85496183	1.102956824
18	OD	5	87.8	224	37	4708960	4.39	83.48214286	1.571542065
19	AO+AAO	3	82.8	331	28	3055060	2.484	91.5407855	1.112071369
20	AO+AAO	7	87.3	310	29	5861310	6.111	90.64516129	0.935154189
21	AO+AAO	3.7	89.6	231	35	5579421	3.3152	84.84848485	2.352504502
22	SBR	1	66.6	172	26	677067	0.666	84.88372093	1.907707107
23	Others	3	96.2	193	26	2299200	2.886	86.52849741	1.30698646
24	OD	3	99	187	30.1	2803550	2.97	83.90374332	1.648299203
25	AO+AAO	2	68.4	187	36.5	1242852	1.368	80.48128342	1.653879841
26	AO+AAO	1	79.9	107	14.9	853273	0.799	86.07476636	3.176791628
27	AO+AAO	3	90	120	22.7	2225960	2.7	81.08333333	2.321388812
28	AO+AAO	1	110	219	16.4	1323375	1.1	92.51141553	1.626889048
29	AS	1	101	333	32.7	700276	1.01	90.18018018	0.632557009
30	Biolak	3	74.3	125	26.1	3434051	2.229	79.12	4.267833973
31	SBR	10	95.4	371	25.1	6395100	9.54	93.23450135	0.530952339
32	AO+AAO	5	58	401	37.9	3676800	2.9	90.54862843	0.956649603
33	SBR	1	59.8	141	12.1	605600	0.598	91.41843972	2.152478889
34	AO+AAO	0.5	68	173	22.9	665800	0.34	86.76300578	3.5743026
35	AO+AAO	1	49.5	364	45.5	1308252	0.495	87.5	2.273442148

Figure A.25 Unit energy consumption of WWTPs in China

VITA

JINZE LI

EDUCATION, FIELDS OF RESEARCH AND ACTIVITIES

Florida International University, Miami, USA

Jan 2015-Dec 2016	Master of Science in Environmental Engineering
Jan 2017-May 2021	Doctor of Philosophy in Civil Engineering
2017	Rain harvest system and wastewater reuse system in Indian Creek - The roof area of rainwater harvesting system was designed based on the water demand and precipitation -MatLab design tool was developed for automatically estimating size and cost of rainwater harvesting system
2018	Life cycle cost and benefit of Water Resource Recovery Facilities (WRRF) -Energy analysis of facility configurations A1~P1E from WERF ENER1C12a is done. -COD and energy flow of B1, B1E, G1E, side-stream N-removal model and CEPT model were simulated using MatLab and Simulink. - Life cycle cost and benefit analysis were carried out. Time required to recover capital cost and cumulative O & M cost for technologies such as N-removal, CEPT, CHP at different flow rates ranging from 10 to 1,000 MGD of WWTPs was identified.
2019	Energy positive treatment system design for leachate and industrial wastewater -COD fraction analysis of leachate, municipal wastewater and different industrial wastewater was quantified. -Energy positive wastewater treatment plants for all the different wastewater were designed.

	-Capital cost and O & M cost of technologies such as CEPT, UASB, PN-anammox and CHP for different wastewater quality was analyzed.
2020	Unit energy and cost analysis of wastewater treatment systems with different technologies -Effect of different leachate and industrial wastewater quality on unit energy consumption and cost in plants with micro sieving, Up-flow anaerobic sludge blanket (UASB), partial nitrification-anammox technologies was quantified. -Micro sieving effects on unit energy consumption and production in energy positive wastewater treatment plant design were quantified. -Life cycle cost and benefit analysis of industrial wastewater treatment plants were carried out. -Data of municipal wastewater plants in China, which included the treatment process, design flow rate, average operation loading rate, energy consumption, influent and effluent COD, BOD5, and $NH4^{+}$-N concentrations were collected. -A novel index of total oxygen demand was developed for evaluating the performance of municipal wastewater plants.