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Impact of Inflow Loading and Algal Productivity on Water Quality in the Chikugo Barrage Reservoir, Japan

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Abstract

The Chikugo Barrage is located at 23 km upstream from the mouth of the Chikugo River, which is the largest river in the Northern Kyushu Region of Japan. Main purposes of the Chikugo Barrage are flood control, water supply, maintaining flow of the Chikugo River and preventing effect of seawater intrusion from the Ariake Sea. There are two rivers flowing into the reservoir, the Chikugo River (the main stream) and the Homan River (a tributary). According to water intake for urban area and irrigation and the operation of the Chikugo Barrage, previous researches reported that the hydraulic retention time (HRT) in the reservoir of the Chikugo Barrage became longer and resulted in high growth of phytoplankton in the reservoir during 1985-2008, while inflow loading was a cause of high phytoplankton when HRT was short. In this study, relationship between loading from upstream, HRT, algal productivity and water quality in the Chikugo Barrage Reservoir was analyzed by using water quality model. Calculated results confirm that chlorophyll-a (Chl-a) in the reservoir increased when inflow loading was high or when the HRT in the reservoir was long. Algal productivity significantly affected on the nutrient level and also led to the increase in COD and SS in the reservoir. Without loading from the Homan River, simulated result of COD in the reservoir became lower indicating that loading from the Homan River has subsidiary impact on concentration of algae in the reservoir. Since the result of loading analysis shows that ratio of Chl-a loading from the Homan River to the main stream tends to increase, loading control in the Homan River Basin is suggested as an effective measure for water quality management in the Chikugo Barrage Reservoir.

Keywords : Chikugo Barrage; Homan River; inflow loading; algal productivity; finite volume model

Introduction

When considering about water resources in one country especially an island country like Japan, precipitation is a valuable source of freshwater in that country. Based on the data of FAO [1] in June 2018, average annual precipitation in Japan is 1,668 mm which is around 1.6 times of the world average. On the other hand, total renewable water resources per capita of Japan is 3,373 m^3/capita-year which is less than half of the world average [2]. Despite of high precipitation, small amount of water resources is available for one person in Japan because of its large population.

After the World War II, economic development and population growth resulted in the rapid increase in water demand in Japan. In order to secure stable water supply, water resources development became necessary [3, 4].

According to the Water Resources Development Promotion Law, the Water Resources Development Basic Plans were issued for seven river systems namely Tone River, Arakawa River, Toyokawa River, Kiso River, Yodogawa River, Yoshino River and Chikugo River, so-called river systems for water resources development [4, 5]. Facilities including dams, barrages and water channels have been constructed to stabilize water flow at downstream and, at the same time, some of them also pay important roles in flood control, power generation and conservation of water environment [3, 4].

Designated as one of the river systems for water resources development, the Chikugo River is the largest river located in the Northern Kyushu Region. Length of the main stream is 143 km and the river basin covers 2,860 km^2 in Kumamoto Prefecture, Oita Prefecture, Fukuoka Prefecture and Saga Prefecture (Figure 1) [6].

Figure 1 Watershed of the Chikugo River [6]

According to the Water Resources Development Basic Plan of the Chikugo River System [7], the Chikugo Barrage was constructed at 23 km upstream from the river mouth and the operation has been conducted since April 1985. The purposes of the Chikugo Barrage are flood control, water supply, compensation for flow fluctuation in the Chikugo River and prevention of seawater intrusion [6]. At present, the Japan Water Agency (JWA) is responsible for operation and maintenance of the Chikugo Barrage. The JWA is an independent administrative agency established in 1962 under Water Resources Development Public Corporation Law, which has engaged in development and management of water resources in the seven river systems [4, 8].

The Chikugo Barrage is about 500 m long and consists of 5 main gates, a lock for ship passage and 2 fish ladders. Total capacity of the reservoir of this barrage is around 5.5 million m^3 [6]. Water is withdrawn from the reservoir by large pumping stations and distributed to water users in both sides of the Chikugo River. Water quality environmental standards of the Chikugo River [6] is shown in Figure 2. Water quality from Mamezu Bridge located at upstream of the Chikugo Barrage until the mouth of the Chikugo River is classified as B Class.

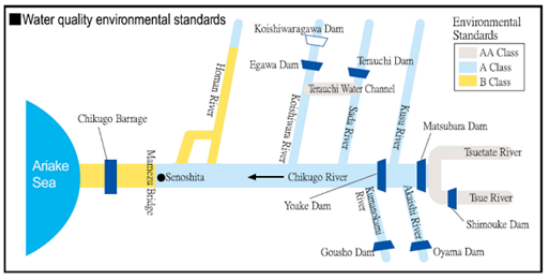

Figure 2 Water quality environmental standards of the Chikugo River [6]

Since 1977, the Chikugo Barrage Environmental Monitoring Liaison Council was established in order to monitor the environmental impact of the construction and to periodically assess the impact of the barrage operation on the environment of the main stream, tributaries and the Ariake Sea [6]. Figure 3 shows sites of environmental survey around the Chikugo Barrage.

Since water quality in the reservoir of the Chikugo Barrage is required to meet water quality environmental standards and demand of water users, several researches were conducted focusing on water quality in this reservoir and pollutant loading from upstream. It was found that loading of COD, T-N and T-P at upstream of the Chikugo Barrage had linear relationship with population and paddy field area [9]. Unit loading of COD from forest, urban area and paddy field area at upstream of the Chikugo Barrage were obtained from water quality simulation by the tank model [10] and the tank model sufficiently simulated pollutant loading of the Chikugo River at Senoshita Station [11]. According to the water intake for downstream and the operation of the Chikugo Barrage, the hydraulic retention time (HRT) in the reservoir became longer and resulted in the increase in phytoplankton in the reservoir during 1985-2008 [12], while the analysis result in [13] showed that loading from upstream was a cause of high concentration of phytoplankton in the reservoir during the period of short HRT.

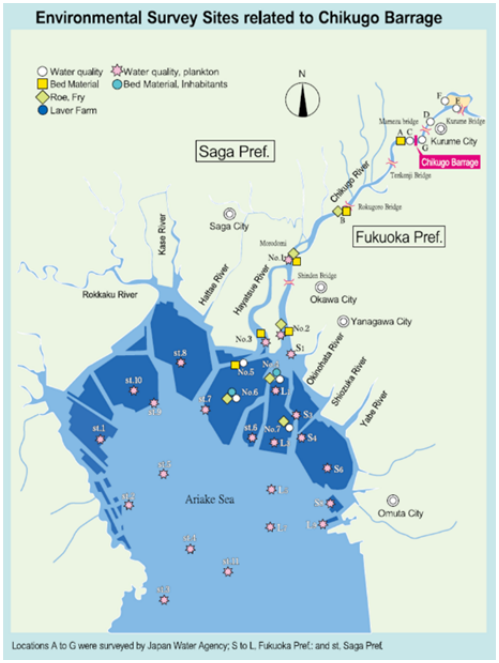

Figure 3 Environmental survey sites related to the Chikugo Barrage [6]

In order to provide useful information for water quality management in the Chikugo Barrage Reservoir, relationship between inflow loading, HRT, algal productivity and water quality in the Chikugo Barrage Reservoir from 2009 to 2014 was examined in this study by using water quality model.

Materials and Methods

Study Area

The study area consists of the reservoir of the Chikugo Barrage and two inflow rivers; the Chikugo River (the main stream) and the Homan River (a tributary). Map of the study area and monitoring stations are shown in Figure 4.

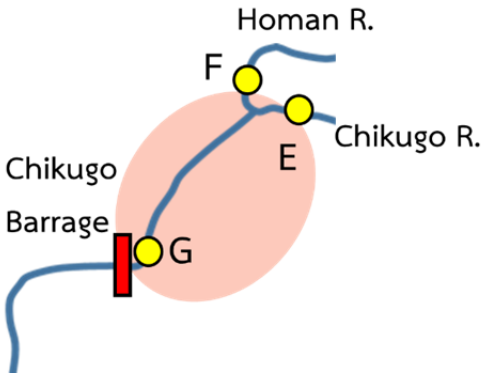

Figure 4 Study area and location of monitoring stations

Station E and Station F are the monitoring stations of the Chikugo River and the Homan River respectively while water quality in the reservoir is monitored at Station G. Flow rate and water quality in the study area are monitored by Chikugo Barrage Operation and Maintenance Office (Japan Water Agency) and Kyushu Regional Development Bureau (Ministry of Land, Infrastructure, Transport and Tourism).

In this study, characteristics of water quality in the reservoir and upstream rivers were examined based on the monitoring data. After that, impact of inflow loading, HRT and algal productivity on water quality in the reservoir was analyzed by using the finite volume model.

Data Analysis

Monitoring data related to flow and water quality in the reservoir and upstream rivers was analyzed to examine a trend of each water quality parameter and to find the relationship among the parameters. Water quality parameters considered in this study are chlorophyll-a (Chl-a), dissolved inorganic nitrogen (DIN), orthophosphate (PO4-P), Chemical Oxygen Demand (COD), suspended solids (SS) and dissolved oxygen (DO). Period of the study is 1985-2014.

Loading Analysis

Loading of the Chikugo River and the Homan River was estimated from monitoring data of Station E and Station F, respectively. Relationship between loading and flow rate (L-Q equation) of each river was examined. The L-Q equations were applied as boundary condition of the water quality model and impact of loading on water quality in the reservoir was analyzed.

Water Quality Model

Since water depth is shallow and spatial difference of concentration is small, the reservoir of the Chikugo Barrage can be considered as a completely-mixed water body. The water quality model was developed based on a finite volume model [16]. Inflow loading from upstream (main stream of the Chikugo River and the Homan River) was estimated from L-Q equations. Calculation period is 2009-2014. Time step of calculation is 1 day. Continuity equation and governing equations of water quality are listed below [13-15].

$$\frac{dV}{dt} = \sum Q_{in} - \sum Q_{out} \quad (1)$$

where V: Volume (m^3); Q_{in} : Inflow (m^3/s); Q_{out} : Outflow (m^3/s)

Governing equations of Chl-a are described in equations (2) – (4). Monod equation was adopted for growth rate of Chl-a [17]. Based on the monitoring data, three types of phytoplankton; namely diatom, green algae and blue-green algae were considered in this study.

$$\frac{d(V \cdot CH)}{dt} = \sum L_{in(CH)} - \sum L_{out(CH)} + G - D - w_{CH} \cdot CH \cdot A \quad (2)$$

$$G = \mu_{max} \cdot f_G \cdot \frac{N}{N+K_N} \cdot \frac{P}{P+K_P} \cdot \frac{I}{I+K_I} \cdot CH \cdot V \quad (3)$$

$$D = k_d \cdot \theta_{CH}^{(T-20)} \cdot CH \cdot V \quad (4)$$

where CH: Chl-a (μg/l); $L_{in(CH)}$: Chl-a inflow loading (g/s); $L_{out(CH)}$: Chl-a outflow loading (g/s); G: Growth; D: Decay; w_{CH} : settling velocity of Chl-a (m/d); A: surface area (m^2); μ_{max} : maximum specific growth rate (1/d); f_G : temperature coefficient of growth (-); N: dissolved inorganic nitrogen (mg/l); P: orthophosphate (mg/l); I: light intensity (cal/cm^2); K_N : half-saturation constant of nitrogen (mg/l); K_P : half-saturation constant of phosphorus (mg/l); K_I : half-saturation constant of light intensity (cal/cm^2); k_d: decay rate (1/d); θ_{CH} : temperature coefficient of decay; T: water temperature (°C)

Governing equations of DIN and PO4-P are shown in equation (5) and equation (6), respectively.

$$\frac{d(V \cdot N)}{dt} = \sum L_{in(N)} - \sum L_{out(N)} + Y_N(D - G) + r_N \cdot f_N \cdot B_N \cdot A_B - K_{DN} \cdot \theta_N^{(T-20)} \cdot N \cdot V \quad (5)$$

$$\frac{d(V \cdot P)}{dt} = \sum L_{in(P)} - \sum L_{out(P)} + Y_P(D - G) + r_P \cdot f_P \cdot B_P \cdot A_B \quad (6)$$

where $L_{in(N)}$: DIN inflow loading (g/s); $L_{out(N)}$: DIN outflow loading (g/s); Y_N : mass ratio of DIN to Chl-a; r_N : release rate of DIN (m/d); f_N : temperature coefficient of release; B_N : DIN in river bottom (mg/l); A_B : bottom area (m^2); K_{DN} : denitrification rate (1/d); θ_N : temperature coefficient of denitrification; $L_{in(P)}$: PO4-P inflow loading (g/s); $L_{out(P)}$: PO4-P outflow loading (g/s); Y_P : mass ratio of PO4-P to Chl-a; r_P : release rate of PO4-P (m/d); f_P : temperature coefficient of release; B_P : PO4-P in river bottom (mg/l)

COD is composed of dissolved COD, particulate COD and COD content of phytoplankton, while SS includes suspended solids and SS content of phytoplankton. Governing equations of COD and SS are as follows:

$$TCOD = DC + PC + Y_C \cdot CH \quad (7)$$

$$\frac{d(V \cdot DC)}{dt} = \sum L_{in(DC)} - \sum L_{out(DC)} - K_{DC} \cdot DC \cdot V \quad (8)$$

$$\frac{d(V \cdot PC)}{dt} = \sum L_{in(PC)} - \sum L_{out(PC)} - w_{PC} \cdot PC \cdot A \quad (9)$$

$$TSS = SS + Y_S \cdot CH \quad (10)$$

$$\frac{d(V \cdot SS)}{dt} = \sum L_{in(SS)} - \sum L_{out(SS)} - w_{SS} \cdot SS \cdot A \quad (11)$$

where TCOD: total COD (mg/l); DC : dissolved COD (mg/l); PC : particulate COD (mg/l); Y_C : mass ratio of COD to Chl-a; $L_{in(DC)}$: DC inflow loading (g/s); $L_{out(DC)}$: DC outflow loading (g/s); K_{DC} : degradation rate of DC (1/d); $L_{in(PC)}$: PC inflow loading (g/s); $L_{out(PC)}$: PC outflow loading (g/s); w_{PC} : settling velocity of PC (m/d); SS: suspended solids (mg/l); Y_S : mass ratio of SS to Chl-a; $L_{in(SS)}$: SS inflow loading (g/s); $L_{out(SS)}$: SS outflow loading (g/s); w_{SS} : settling velocity of SS (m/d)

Results and Discussion

Characteristics of Water Quality in the Chikugo Barrage Reservoir and Inflow Rivers

Ratio of flow rate of the Homan River at Station F (QF) to the flow rate of the Chikugo River at Station E (QE) is shown in Figure 5. The ratio QF/QE increased from 2009 and the average ratio was around 0.25. The increase in the ratio QF/QE indicates that impact of inflow loading from the Homan River on the water quality in the reservoir might increase when compared with the inflow loading from the main stream [15]. In loading analysis, inflow loading from each river

was evaluated and relationship between loading and flow rate was examined.

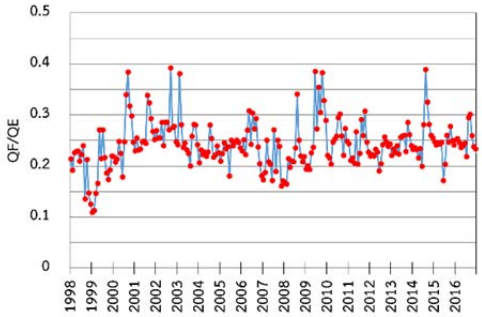

Figure 5 Ratio of flow rate of the Homan River (QF) to the Chikugo River (QE)

Chl-a observed at Station E, F and G are shown in Figure 6. Since Chl-a at all stations was high in the period of high rainfall, it can be said that high inflow loading resulted in high Chl-a in the reservoir. Chl-a in the Homan River was much higher than that in the main stream. Based on flow ratio of 0.25, Chl-a loading of the Homan River could be as high as half of the main stream. Moreover, Chl-a at Station G also increased when HRT in the reservoir was long. It indicates that both loading from upstream and HRT were the causes of the increase in Chl-a in the reservoir.

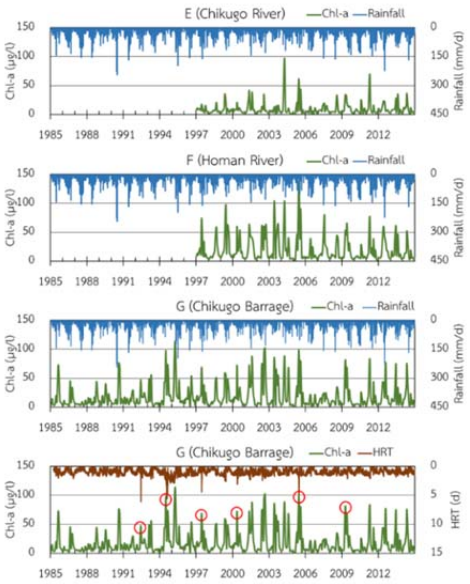

Figure 6 Chl-a in the Chikugo Barrage Reservoir and inflow rivers

In Figure 7, DO in the reservoir exceeded the saturated DO (Cs) when Chl-a was high which shows that large amount of oxygen was supplied by photosynthesis. Figure 8 shows COD at Station E, F and G. COD at Station G had the same trend with inflow rivers which represents the impact of inflow loading on COD in the reservoir. On the other hand, COD in the reservoir also increased in the period of high Chl-a. It indicates that growth of phytoplankton and inflow loading affected on COD in the reservoir.

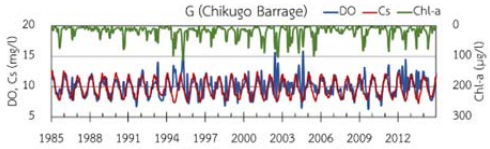

Figure 7 DO and Chl-a in the Chikugo Barrage Reservoir

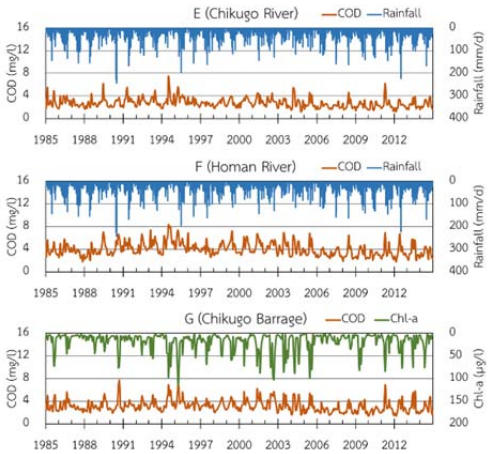

Figure 8 COD in the Chikugo Barrage Reservoir and inflow rivers

Figure 9 shows DIN at Station E, F and G. Trend and level of DIN in the reservoir were same with DIN of the main stream (Station E), while DIN of the Homan River was higher than other stations and changed seasonally. It can be said that DIN in the reservoir was affected by inflow loading from the main stream. Compared with Chl-a, DIN in the reservoir decreased when

Chl-a was high, which confirmed the consumption of DIN by algal growth.

PO4-P at each station is shown in Figure 10. Similar to DIN, POP4-P in the reservoir was raised by loading from upstream. The decline of PO4-P in the period of high Chl-a also confirms phosphorus consumption by phytoplankton in the reservoir.

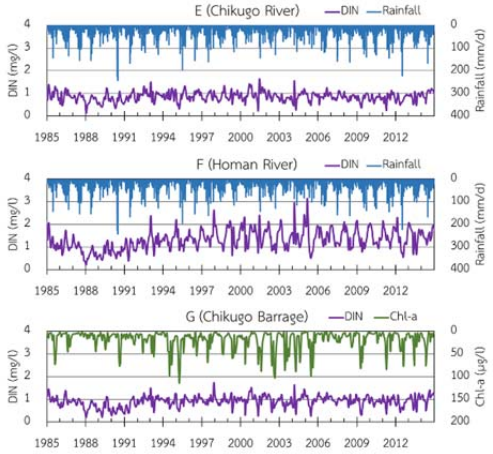

Figure 9 DIN in the Chikugo Barrage Reservoir and inflow rivers

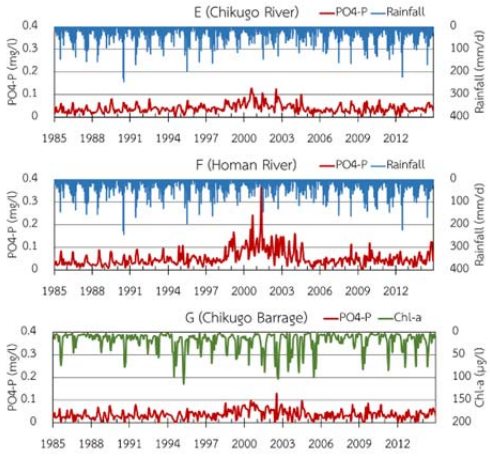

Figure 10 PO4-P in the Chikugo Barrage Reservoir and inflow rivers

From Figure 11, SS at each station increased when the rainfall was high. It indicates that high inflow loading resulted in high SS in the rivers and the reservoir. Since the period of high SS in the reservoir was different from the period of high Chl-a, it can be said that impact of algal growth on SS in the reservoir was less significant than the inflow loading. On the other hand, long HRT in the reservoir probably resulted in large settling amount of SS.

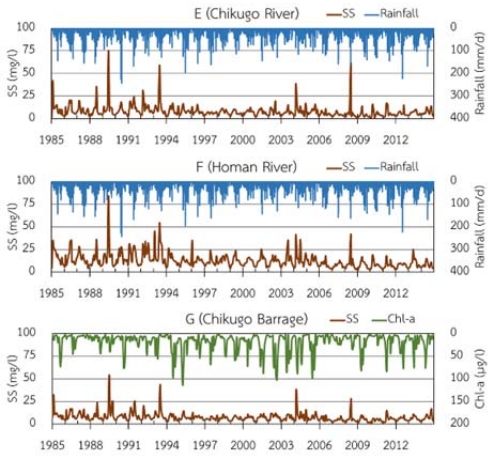

Figure 11 SS in the Chikugo Barrage Reservoir and inflow rivers

Loading Analysis

Loading of the main stream (LE) and the Homan River (LF) was estimated from the observed data. As shown in Figure 12, loading of COD in both rivers proportionally related to the flow rate.

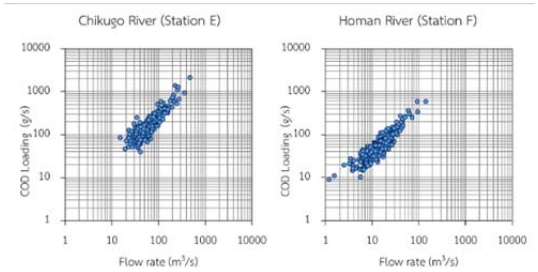

Figure 12 COD loading and flow rate of the Chikugo River and the Homan River

Other loading also had the similar relationship with flow rate. The L-Q equations of the main stream and the Homan River are listed in Table 1.

Table 1 L-Q equations of the Chikugo River and the Homan River

$L = aQ^b$	Chikugo River (Station E)		Homan River (Station F)	
	a	b	a	b
Chl-a	0.1	2.0	1.5	1.75
DIN	0.8	1.07	1.3058	0.9527
PO4-P	0.008	1.3648	0.033	1.1088
COD	1.8	1.0811	2.0	1.1913
SS	0.6	1.5454	4.5	1.2372

As shown in Figure 5, monthly average flow rate of the Homan River from 2009 to 2016 was around 25% of the flow rate in the main river. Loading of each river in 2009-2016 was estimated by using L-Q equations (the result of 2012 was excluded because of inadequate data). Ratio of loading of the Homan River to loading of the Chikugo River is summarized in Table 2.

Table 2 Ratio of loading of the Homan River (LF) to loading of the Chikugo River (LE)

Loading	LF / LE
Chl-a	0.39
DIN	0.25
PO4-P	0.25
COD	0.34

It was found that nutrient loading (DIN and PO4-P) of the Homan River was 25% of the main river. On the other hand, Chl-a loading and COD loading were 39% and 34%, respectively. Despite of its low flow rate, loading ratio of Chl-a and loading ratio of COD of the Homan River were larger than the ratio of flow rate. It is necessary to pay attention to the impact of the loading of the Homan River on the water quality in the reservoir especially concentration of phytoplankton and organic matter.

Contribution of Inflow Loading on Water Quality in the Chikugo Barrage Reservoir

Water quality in the reservoir was calculated by considering only the inflow loading from upstream and neglecting other mass transport and transformation terms. Calculated results are shown in Figure 13. Chl-a increased during the period of high rainfall (high flow rate). Except in the period of high rainfall, DIN in the reservoir was almost constant. At the same time, it also indicates that the low level of observed DIN was a result of nutrient consumption by algal growth in the reservoir.

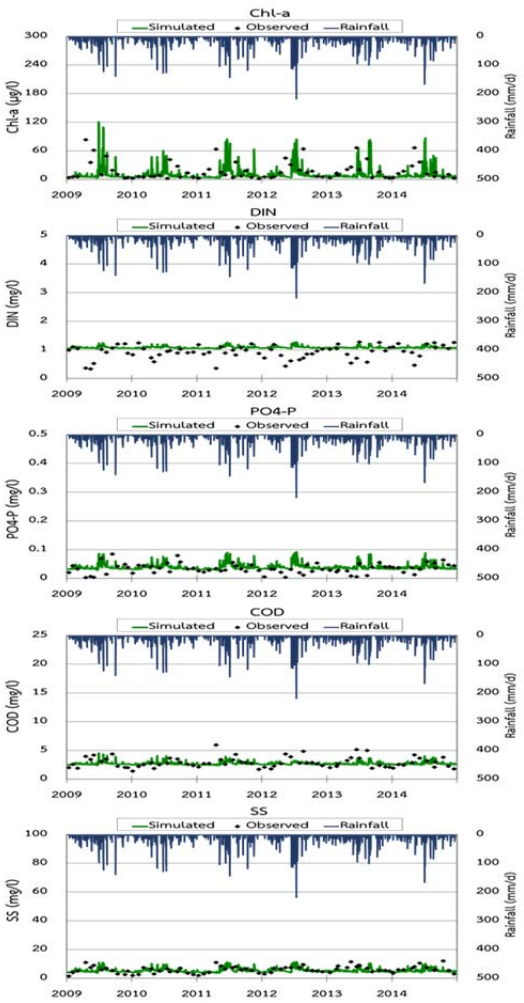

Figure 13 Contribution of inflow loading on water quality in the Chikugo Barrage Reservoir

Calculated result of PO4-P was high in the period of high flow rate which confirmed the impact of inflow loading on PO4-P in the reservoir. Similar to DIN, reduction of PO4-P according to productivity of algae in the reservoir was also confirmed.

Calculated results of COD and SS during the period of high loading were lower than observed data, which indicates that the effect of algal productivity on COD and SS could not be neglected.

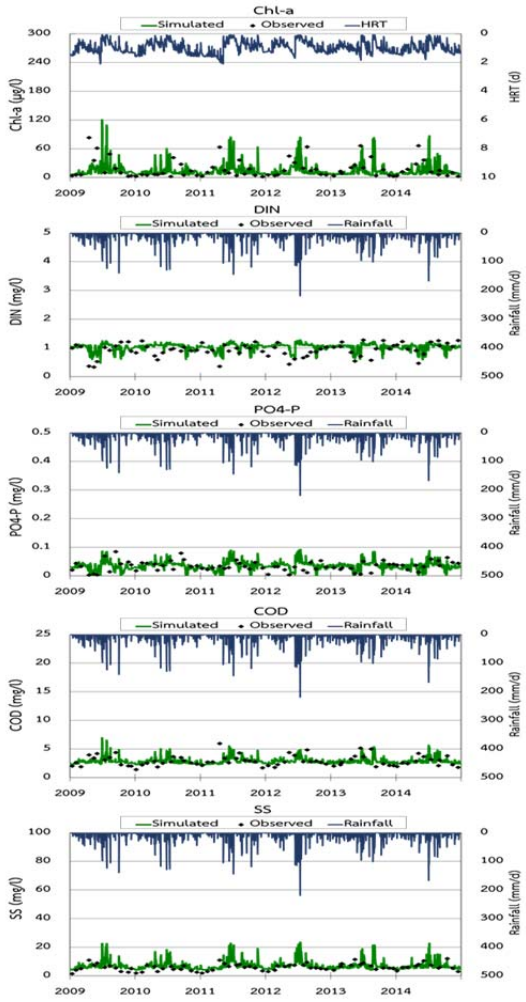

Figure 14 Simulated results considering inflow loading and algal productivity in the Chikugo Barrage Reservoir

Contribution of Algal Productivity on Water Quality in the Chikugo Barrage Reservoir

Productivity of algae was taken into account in the calculation together with the inflow loading from upstream. Figure 14 shows the calculated results. By including the impact of algal productivity, calculated results of all water quality parameters have good agreement with the observed data. Moreover, simulated result of the Chl-a also showed good agreement with observed data in the period of long HRT. The increase of Chl-a, COD and SS and the consumption of nitrogen and phosphorus by algal productivity were confirmed. Algal productivity and HRT placed significant impact on water quality in the reservoir. The parameters used in the water quality model are listed in Table 1.

Contribution of Loading of the Homan River on Water Quality in the Chikugo Barrage Reservoir

Calculated result without loading of the Homan River is shown in Figure 15. The calculated result in the reservoir is lower than the result in Figure 14. It is shown that loading of the Homan River also had subsidiary effect on concentration of phytoplankton in the reservoir. Reduction of loading of the Homan River can contribute to eutrophication control in the Chikugo Barrage Reservoir.

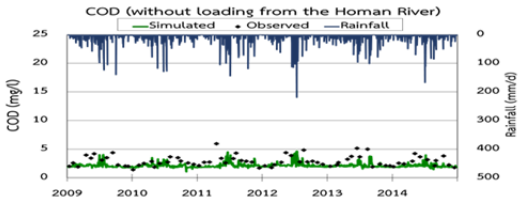

Figure 15 Simulated result by neglecting loading of the Homan River

Table 3 Parameters used in the water quality model of the Chikugo Barrage Reservoir

μ_{max}	maximum specific growth rate	1.3 – 1.8	1/d
K_N	half-saturation constant of nitrogen	0.02	g/m^3
K_P	half-saturation constant of phosphorus	0.002	g/m^3
K_I	half-saturation constant of light intensity	100	cal/cm^2
k_d	decay rate	0.05	1/d
θ_{CH}	temperature coefficient of decay	1.05	-
w_{CH}	settling velocity of chl-a	0.05	m/d
w_{PC}	settling velocity of PCOD	0.5	m/d
w_{SS}	settling velocity of SS	0.5	m/d
r_N	release rate of DIN	0.15	m/d
r_P	release rate of PO4-P	0.01	m/d
K_{DN}	denitrification rate	0.015	1/d
θ_N	temperature coefficient of denitrification	1.05	-
K_{DC}	degradation rate of DCOD	0.08	1/d
Y_N	DIN: Chl-a	0.02	mg DIN /µg Chl-a
Y_P	PO4-P: Chl-a	0.0015	mg PO_4-P /µg Chl-a
Y_C	COD: Chl-a	0.015 – 0.03	mg COD /µg Chl-a
Y_S	SS: Chl-a	0.1 – 0.18	mg SS /µg Chl-a

Conclusions

In this study, water quality analysis was conducted to define the impact of inflow loading from upstream and HRT regarding to the operation of the barrage on water quality in the Chikugo Barrage Reservoir. It was found that Chl-a in the reservoir increased when loading from upstream was high or when the HRT in the reservoir was long. Calculated results confirmed that algal productivity significantly affected on nutrient level (both DIN and PO4-P) in the reservoir while PO4-P also highly depended on the supply from upstream. Growth of phytoplankton also played an important role on increasing COD and SS in the reservoir. Since the effect of loading from the Homan River on concentration of algae in the reservoir was confirmed and ratio of Chl-a loading from the Homan River to the main stream tends to increase, loading control in the Homan River Basin is suggested as an effective measure for water quality management in the Chikugo Barrage Reservoir.

Acknowledgement

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Control of Algae in Swimming Pool by Ozonation

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Abstract

Algae proliferation is an important problem of swimming pool, making it unsightly and ultimately unacceptable to swimmers. Chlorination of the pool seems to inhibit growth of microorganism. However some types of algae can survive the condition and are frequently found (attached on tiles or in suspension). In this study algae samples were collected from a hotel swimming pool and cultured in Bold's Basal medium. Morphological identification by Direct Microscopic Examination of the culture showed that predominating algae were *Oscillatoria, Scenedesmus, Euglena* and *Phagus*. Algae in the culturing bottles were found in 3 zones namely attached on the wall (A), suspended (SS) and settled at the bottom (ST). Ozonation of algae culture with 22.1-141.5 mgO_3/L (5-30 min.ozonation time) was found to remove attached and suspended algae. These then settled down to the bottom resulting in an increase in the settled portion. It was noted that ozone consumption of the suspended portion (1.72 mgO3/mgAlgae at 30 min.) was higher than that of the attached portion (0.47 mgO_3/mgAlgae at 30 min.) indicating that the former was more resistant to ozone than the latter. In terms of algae type, of the 4 predominating spices, *Euglena sp.* was the most resistant.

Keywords : ozonation; algae control; swimming pool

Introduction

Swimming pool is a place where many people gather to swim, for exercise and recreation. Since cleaning before entering the pool is often perfunctorily observed, contamination from the bodies of swimmers, such as cosmetics, oil, and human excretes is common. Without precautionary measures, the swimming pool can be breeding ground for various diseases. Another source of objectionable condition in the pool is the presence of algae. Although most algae does not cause much harm to human health, but its presence makes the pool unsightly. Water quality is thus the point of concern and must be regulated so that it conforms to the standard set by the Ministry of Public Health. One method widely used to control the spread of the diseases is chlorination. It effectively reduces the growth of diseases and algae. However, some types of algae are resistant to this condition and can proliferate in the pool. Use of high chlorine dosage is not feasible due to the formation of carcinogenic trihalomethane (THM) [1]

Algae is a type of microorganism that can be found everywhere, especially in damp places. The distribution depends mainly on the type of

algae and environmental conditions including season, light, moisture, temperature and nutrient. The survey studies on algae proliferation in different area in Thailand reported the presence of Cyanophyta, Chlorophyta and Bacillariophyta. The most frequently found were *Scenedesmus sp.*, *Oscillatoria sp.* and *Euglena sp.* [2-5].

Algae can be controlled by oxidation process through various types of chemicals. In water treatment system, chlorine is a common chemical used. The use of chlorine coupled with oxidizing agent such as $KMnO_4$ and $CuSO_4$ could strengthen the control effect [6].

Ozone, because of its very high oxidizing power, is another candidate for oxidant for algae control. With a short half-life of 20-30 min., it emits O_2 as degradation product [7]. It has been used in food preservation [8] and disinfection purposes in various situations. It has performed better than chlorine and UV in destroying *Escherichia coli*. The destruction caused by oxidation of cell membrane leads to cell lysis [9]. It has been reported that the use of ozone for disinfection of domestic wastewater could reduce Heterotrophic Aerobic Bacteria, Total Coliform, Fecal Coliform and *E.coli* by up to 90% [10]. Algae was also controlled by ozonation, the study on algae from natural reservoir showed that 80% removal was achieved [11].

This study investigated the types of algae surviving in the swimming pool after it has been treated with chlorine and evaluated the efficiency of ozone in controlling algae. Algae morphological identification was achieved by Direct Microscopic Examination. Qzonation was conducted on the algae cultured in Bold's Basal medium. The results of the study should be beneficial as a guideline for algae control in swimming pools.

Materials and Methods

Algae used in the study was collected from a hotel swimming pool. The pool used Trichloro Isocyanuric Acid 90% as disinfectant and had algae proliferation problem. Grab sample was collected by scraping algae attached to the wall of the pool. It was examined under microscope (Direct Microscopic Examination) and identified by morphology according to Bold and Wynne (1978) [12]. To obtain enough population for further study, it was cultured in Bold's Basal medium under natural light with aquarium air pump aiding the mixing. The system setup for ozonation experiment, shown in Figure 1, consisted of a sample bottle (1.5 L) connected with 3 consecutive flasks of 2%KI solution serving as excessive ozone traps. An ozone generator supplied ozone to the sample bottle. The ozone production rate was determined using Wet Chemistry Iodide method [13] in the same ozonation system used for sample but omitting the sample bottle. The ozone production rate was calculated, according to equation 1, from the amount of standard $Na_2S_2O_3$ titrant used in titration with KI traps. The average production rate of 6.97 mgO_3/min was obtained from 10 tests.

*Ozone production rate (mg O_3/min) = (A+B+C) * N * 24 / T .....(1)*

where A, B, C = mL of $Na_2S_2O_3$ used for titrating trap A, B, C
N = normality of $Na_2S_2O_3$ titrant
T = ozonation time (min)

During experiment, ozone was continuously supplied to a batch of algae culture for 30 min. The treated algae culture was sampled at 10 min.

interval and determined for biomass and cell numbers (Suspended Solids and Natural Unit Count, according to APHA, AWWA and WEF, 2012) [14]. The amount of ozone in KI traps accounted for ozone amount remaining after the reaction. Actual amount used was calculated by subtracting the amount remaining from the amount supplied.

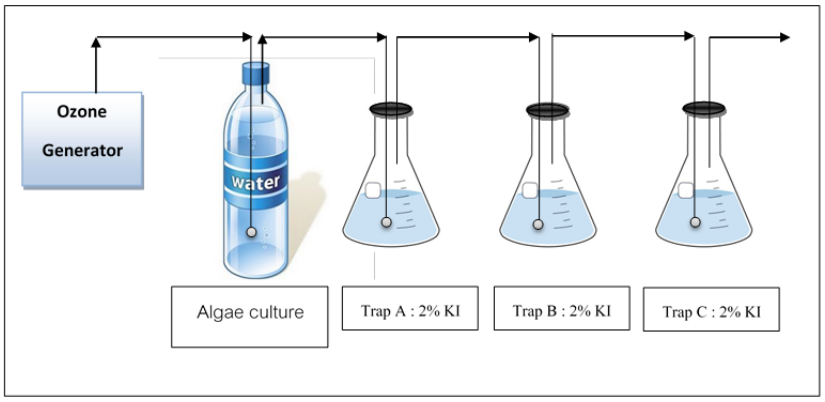

Figure 1 the setup of ozonation system for controlling algae population

Results and Discussion

Identification of algae in the swimming pool

The direct microscopic examination of algae in water sample collected from the swimming pool showed that algae surviving swimming pool condition were in the division Cyanophyta and Chlorophyta. Predominant species of Cyanophyta was *Oscillatoria* sp. (Figure 2). Those in Chlorophyta included *Scenedesmus* sp., *Euglena* sp. and *Phacus* sp. (Figure 3). These species were common species found in natural water bodies in Thailand [2-5].

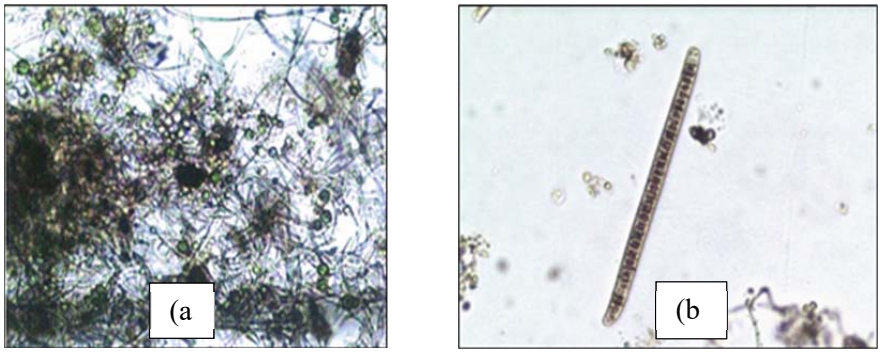

Figure 2 Algae (Division Cyanophyta) found in swimming pool
(a) *Oscillatoria* sp. (10x) (b) *Oscillatoria* sp. (100x)

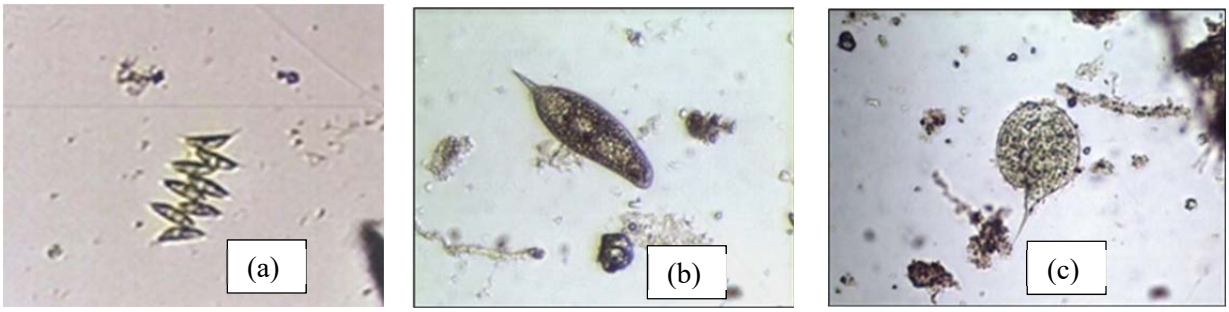

Figure 3 Algae (Division Chlorophyta) found in swimming pool
(a) *Scenedesmus* sp. (100x) (b) *Euglena* sp. (100x) (c) *Phacus* sp. (40x)

Ozonation of algae culture

The effect of ozone on algae population

The experiment on effect of ozone was conducted on swimming pool algae population cultured in Bold's Basal medium. In the culturing bottles, algae was found growing in 3 portions, namely, attached to the bottle surface (A), suspended (SS) and settled at the bottle bottom (ST). Of the 3 portions, biomass of the algae attached to the bottle was the highest. The biomass of the suspended and settled portions were the same. The change in population due to ozonation was investigated, the results are as shown in Figure 4. At the start, the attached biomass was 350 mg/L while the suspended and the settled portion were 85 mg/L. During early period (5-10 min.) of ozonation with 22.1-56.7 mgO_3/L, both the attached and suspended population decreased rapidly. The rate of decrease later slowed down. At the end of the experiment, with 141.5 mgO_3/L, 84% of attached algae was removed (56 mgDW/L remaining from 358 mgDW/L). Suspended population was more sensitive to ozone, 96% was removed (3.3 mgDW/L remaining from 85 mgDW/L) under the same condition. Simultaneous to the decrease in biomass of these 2 populations, the settled population increased. It should be noted that the total dry weight of the 3 portions after ozonation were not much different, in the range of 446-528 mg/L (Table 1). This showed that suspended and attached population settled down due to ozonation. This was confirmed by the appearance of the test bottles in Figure 5. Initially, the green color of attached and suspended algae was very strong and reduced with ozonation, while the amount of the settled portion increased.

Considering ozone consumption for algae removal – mgO_3/mgAlgae (Table 2), both attached and suspended portion showed the increasing trend with an increase in ozone. It was noted that ozone consumption of the suspended portion (1.72 mgO_3/mgAlgae at 30 min.) was higher than those of the attached portion (0.46 mgO_3/mgAlgae at 30 min.) all through the ozonation period, indicating that the former was more resistant to ozone than the latter.

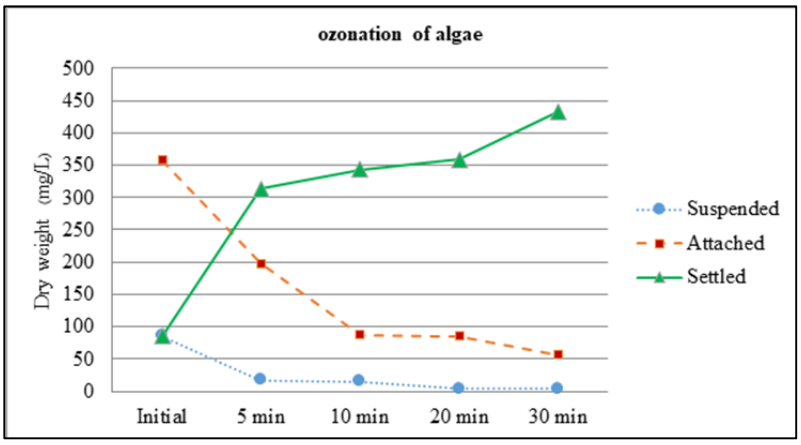

Figure 4 The changes in algae population (dry weight) during ozonation (initial = o min. ozonation time)

Table 1 Algae population (dry weight) during ozonation

Ozonation time	Dosage (mgO_3/L)	Algae population (dry weight, mg/L)			
		Suspended	Attached	Settled	Total
Initial	0	85.1	358	85.1	**528.2**
5 min	22.12	17	198.3	313	**528.3**
10 min	56.71	15.2	87.5	343.3	**446**
20 min	76.07	3.83	85.2	358.7	**447.7**
30 min	141.50	3.3	56	433	**492.3**

Figure 5 The appearance of algae culture at different ozonation times

Table 2 Ozone consumption for algae removal

Ozonation time	Dosage (mgO_3/L)	Ozone consumption (mgO_3/mgAlgae)	
		Suspended	Attached
5 min	22.12	0.32	0.14
10 min	56.71	0.81	0.21
20 min	76.07	0.93	0.27
30 min	141.50	1.72	0.47

The effect of ozone on different groups of algae

This part of the study aimed to investigate the effect of ozone on particular species. The predominating species were enumerated by direct counting before and after ozonation. The result is as shown in Table 3. *Oscillatoria* sp. population was found to be highest with 173,186 cell/L in the test culture. After ozonation, its population decreased to 22 cell/L (>99.99% decrease). *Scenedesmus* sp. and *Phacus* sp. were also very sensitive to ozone, 95 and 100% of the population was decreased, respectively. *Euglena* sp. was more resistant to ozone than the others. Its population was decreased by 68%.

Table 3 Effect of ozone on predominant algae in the swimming pool (141.5 mgO_3/L)

Division	Genus	Population number (Cell/L)		%decrease
		Initial	Ozonated	
Cyanophyta	*Oscillatoria* sp.	173,186	22	>99.99
Chlorophyta	*Scenedesmus* sp.	100	5	95
	Euglena sp.	50	16	68
	Phacus sp.	88	ND*	100

*ND = Not Detected

Conclusion

Algae found to survive swimming pool after disinfection by chlorination were in the genus Cyanophyta (*Oscillatoria sp.*) and genus Chlorophyta (*Scenedesmus sp., Euglena sp.* and *Phacus* sp. Ozone was an interesting choice for algae control in the swimming pool. By ozonation with 141.5 mgO_3/L, 84% of algae on the wall and 96% of algae suspended in the medium were removed and settled down. Among the 4 predominating species, *Euglena* sp. was the most resistant to ozone, only 68% of the population was removed while 95-100% removal could be achieved in other species.

Acknowledgement

This work was a part of the study on algae control by ozonation, The project funding was granted by the National Research Council of Thailand and the Faculty of Engineering, Kasetsart University.

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Water Removal from Crude Biodiesel using Microbubbles

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Abstract

Current production of biodiesel using transesterification .however, biodiesel is not yet available to the engine because there is contamination of the mixture. Therefore it is necessary to have a process that makes biodiesel purification. Washing biodiesel with water is the most common method of cleaning biodiesel. But the disadvantage of the process is high cost and longer manufacturing process.In this research, therefore, air bubbles at different sizes of diffusers (10-16 μm, 16-40 μm, and 40-100 μm) and air flow rates (100 cc/min, 200 cc/min, 300 cc/min) were used instead to remove water in biodiesel. The results show that the smallest size of bubbles with the highest air flow rate provided the highest removal rate of water. Using bubbling technique, can not only overcome those drawbacks suffering from the conventional processes. But also reduce the production cost

Keywords : Biodiesel; water removal; Air bubbles

Introduction

In recent years global warming and environmental pollution have become major global issues. The use of fuels coming from biomass such as biodiesel and bioethanol can help to mitigate such issues because of the renewable features of these energy sources. The above mentioned bioproducts are the most important biofuels employed up-to-date in the transport sector. Both can be utilized alone, in special motor engines or as additives in fossil diesel and gasoline blends. Actually, biodiesel exhibits many benefits as an alternative fuel: it is derived from a renewable source, its biodegradability and lower ecotoxicity contribute to its beneficial character compared to petroleum based diesel. Furthermore, biodiesel has a more favorable combustion emission profile, such as low emission of carbon monoxide, particulate matter and unburned hydrocarbons [1]. Current production of biodiesel using transesterification .which can be produced by mixing vegetable oil, animal fat, alcohol and catalyst with Sodium hydroxide. The main products consist of alkyl esters (biodiesel) and glycerol [2].

Finally, biodiesel is stored and distributed to be sold [3]. Once stored, biodiesel can absorb more humidity than petroleum diesel since fatty acid methyl esters (FAMEs) are hygroscopic compounds, making biodiesel much more hydrophilic than regular diesel [4]. Biodiesel clearly offers environmental, commercial and performance benefits, since biodiesel does not contain sulfur, aromatic hydrocarbons, metals or crude oil residues. However, it must be emphasized that biofuel's performance heavily depends on the purity of the final product and the complete absence of particulates or contaminants, e.g., water [5]. The free water content in biodiesel and diesel fuel promotes biological growth in storage tanks, which could lead to corrosion of metals (copper, iron, steel and others) and formation of sludge and slime, thereby causing blockage of fuel filters and fuel lines, which could damage vehicle fuel injection systems. Washing biodiesel with water is the most common method of cleaning biodiesel. But the disadvantages of the process are high cost and longer manufacturing process Therefore, this paper is interested in air bubbles to remove the water. Compared with conventional process, the advantages gained from this proposed technique are requirements of less space and less operating time. Also, it is a simple and cost effective method [6].

The objective of this research is to find relationship between pore size of diffusers, air flow rates and operating time for water removal in the bubble column.

Methodology

Biodiesel was prepared by transesterification process. The experimental setup is shown in Figure 1. A bubble column with a diameter of 50 mm was assembled with a diffuser connected to stainless tubes, a flow meter and an air pump. Bubbles were generated while air was flowed through the column. Bubble-sizes were varied depending on the diffuser used. Here, the diffusers are porous-glass discs with a diameter of 50 mm and have different ranges of pore sizes; P2 (40-100 μm), P3 (16-40 μm) and P4 (10-16 μm). The biodiesel of 50 mL was loaded into the bubble column. Air was then flowed from the air pump with different flowrates of 300, 500 and 700 cc/min through the diffuser and the biodiesel containing in the column. Samples were collected every 10 minutes for 1 hour. The concentration of water in biodiesel was analyzed using the coulometric Karl Fischer method. The reduction of water concentration in the bubble column with time was recorded in related to bubble-sizes and flowrates.

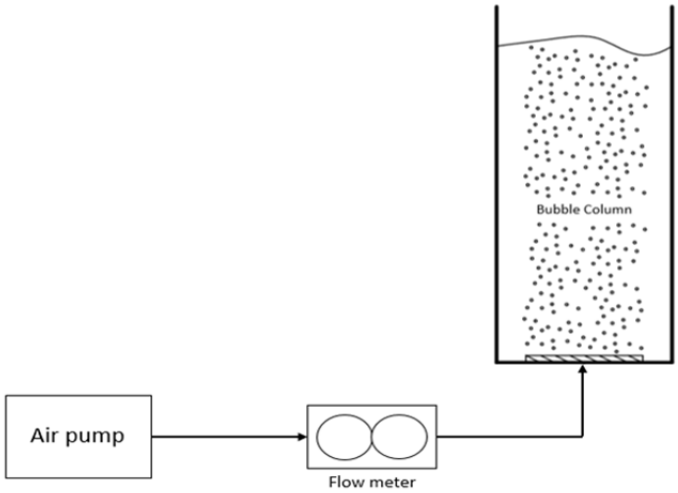
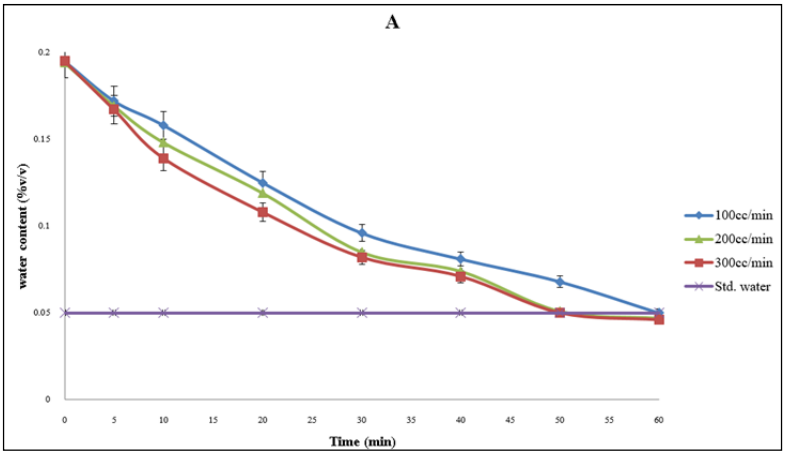
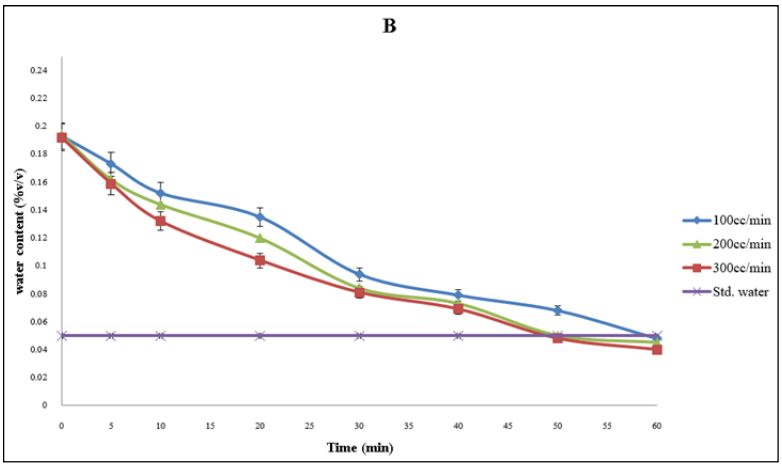


Figure 1 Experimental setup for water removal via bubbles. The inset is a photograph of the bubble column while flowing air with a flow rate 300 cc/min through a diffuser P2, P3, and P4.

Results and Discussions





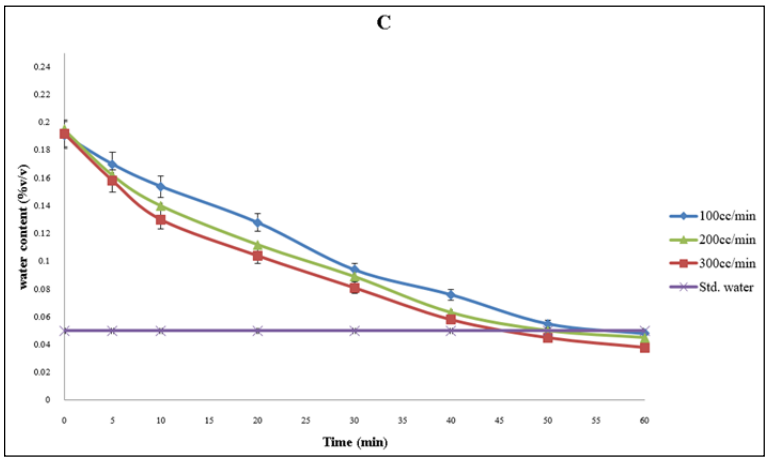


Figure 2 Remaining water content (%v/v) after air flowing for 60 minute through diffusers P2(A), P3(B), and P4(C) at air flow rates of 100, 200 and 300 cc/min.

A technique for water removal via air bubbles presented in Figure 1. The inlet temperature of air was set at room temperature. After bubbles formed, they float from the bottom to top of the bubble column. Water dissolved in biodiesel evaporates simultaneously into the bubbles leading to a reduction of water in the system.

The amount of water remaining in biodiesel at various bubbling times is shown in Figure 2. Overall, the concentration of water decreased rapidly within the first 5 minutes at all flow rates and diffusers. At air flow rate of 300 cc/min represented in Figure 2A, water content decreases from 0.195% to 0.05% (Standard ASTM D-2709) within 50 minutes, while longer bubbling time required at lower air flow rate. These characteristics were observed when the diffuser P3 and P4 were used as shown in Figure 2B and Figure 2C, respectively. At flow rate of 300 cc/min represented in Figure 2B, water content decreased from 0.194% to 0.05% within 52 minutes, whereas diffuser P4 represented in Figure 2C water content decreased from 0.195% to 0.05% within 45 minutes. This means that small pore size could reduce more amount of water than large pore size at the same air flow rate according to the work done by William study [7].

Moreover, at the same air flow rate, the highest removal rate of water was observed at the highest air flow rate as shown in Figure 2 A-C. This means that the amount of water transferred into air bubbles is proportional to air flow rate.

Conclusion

This research presented a novel technique used for water removal from biodiesel using air bubbles. Bubble size and air flow rate are the main parameters that affect the removal rate. Higher water removal rate was observed when both smaller bubbles were formed and higher flow rate was applied. In addition, the amount of water in biodiesel is less than 0.05% within 45 minutes for P4 diffuser and flow rate of 300 cc/min.

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Kinetics of Ethyl Benzene Degradation in Biofilter using Isolated Bacteria from Petrochemical Wastewater Treatment Plant

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Abstract

The objective of this study was to investigate the kinetics of ethyl benzene degradation in the biofilter packed with the isolated bacteria from a petrochemical wastewater treatment plant. The inlet synthetic contaminated air containing 50 ppm ethyl benzene gas was introduced up flow in a 21.2 lite reactor with the flow rate of 1.0, 1.5, 2.0, 2.5, 3.0, 3.5, 4.0, 4.5 and 5.0 L/min. The bacterias were fixed on plastic medias (polyethylene) 40 mm × 40 mm and packed in the reactor with about 30.0 cm height Both inlet and outlet ethyl benzene concentrations were measured using gas chromatography. The results were fit with the order of reaction equations. The relation between the concentrations and empty bed resident times (EBRT) show that the biofilter kinetics was Zero-Order Reaction Limited Model and The Kinetic constant (k_0) was 3.2192 ppm.min^{-1}. The kinetic of the biodegradation of the biofilter system may be described by an equation: C_0-C = (3.2192×t)-21.513.

Keywords : Biofiltration; Kinetic

Introduction

Treatment of volatile organic compounds (VOCs) using a biofilter is one of the technique most widely used. This technique is effectively applied to control air pollution containing volatile organic compounds and/or inorganic compounds. Generally biofilter is a kind of treatment system that is easy to control and operate.

Ethyl benzene is the VOCs. It is highly flammable, clear colorless with an aromatic odor [1]. It is important used to an intermediate in the production of styrene in the petrochemical industry. For health effect, ethyl benzene is in 2B group (Possibly carcinogenic to humans) of IARC Monographs on the identification of carcinogenic hazards to human [2].

The degradation of volatile organic compounds in biofiltration involves many physical, chemical and microbiological phenomena [3]. Biofilter system design should consider factors to get an appropriate system and conditions to operate. One of the most important factors is the kinetic of the bacteria degradation. This characteristic is specific for each bacteria and conditions. Kinetics of degradation of bacteria in

the biofilter are classified into three forms 1) Zero-Order Reaction-Limited Model 2) Zero-Order Diffusion-Limited Model and 3) First-Order Model [1]. A studied kinetics of ethyl acetate and xylene in biofilter using sugarcane bagasse base as packing material. The inlet mixed gas (ethyl acetate and xylene) with the concentration of 0.2-1.2 g/m^3 were introduced to the system. The result found that the kinetic of the biofiltration was Zero-Order Diffusion Limitation Model [4]. The studied of kinetics and modeling of H_2S in biofilter using cylindrical particle as medias and the empty bed resident time (EBRT) were controlled at 20, 30, 45 and 60 s. The result found that the kinetic of biofiltration was First-Order Model [5]. The studied of kinetics in biofilter using compost base as packing material. The H_2S were passing with flowrate 68 l/min to the reactor, EBRT was 16 s. The result found that the kinetic of biofiltration was First order model for low concentration (<200 ppm) and Zero order model for high concentration (>400 ppm) [6]. The Studied of Kinetic in biofilter using compost mix with granular activated carbon media. The 3 parallel biofilter using compost mix granular activated carbon for 0, 3.55 and 13% by weight for each reactor. The BTEX inlet gas were introduced to the biofilter. The result found the kinetic was first order for BTEX concentration 50 ppm and Zero order for BTEX concentration range 235-440 ppm [7].

Biofilter system designed to degradation ethyl benzene using bacteria isolated from a petrochemical industry wastewater treatment plant. The group, which as bacteria *P. aeruginosa* S19 and *B. Cereus* O5-1/1 [8]. Therefore, the study of kinetics of bacteria degradation is needed in full scale design biofilter system.

Materials and Methods

The study of kinetics ethyl benzene degradation in biofilter system used bacteria isolated from a petrochemical industry wastewater. Bacterias were fixed on plastic medias (polyethylene) 40 × 40 mm [9]. The surface area per unit volume of the media was 180 m^2/m^3. The medias were packed in a 21-liter reactor. Synthetic contaminant gas was generated by passing cleaned air over ethyl benzene surface in a closed glass impinger. The rich ethyl benzene gas was mixed with another cleaned air stream to control and vary the ethyl benzene concentrations. This synthetic ethyl benzene contaminated air with was introduce into the reactor at the flow rate of 1.0, 1.5, 2.0, 2.5, 3.0, 3.5, 4.0, 4.5 and 5.0 L/min. Essential nutrient was also introduced into the reactor. The input ethyl benzene concentration was maintained about 50 ppm. Both inlet and outlet ethyl benzene concentrations were measured using gas chromatography (8900 GC-TID, Baseline-Mocon, Inc.). The schematic of the biofilter experimental system is shown in Figure 1.

This study aims to verify the kinetic of the biodegradation reaction of the system. The characteristic of the kinetics may be represented by three equations, (1) - (3), as following [4].

Zero-Order Reaction-Limited Model

$$C_0 - C = k_0 t \qquad \text{(Eq. 1)}$$

Zero-Order Diffusion-Limited Model

$$C = C_0 \left[1 - t \left(\frac{a k_0 D_e}{2 m C_0 \delta} \right)^{1/2} \right]^2 \qquad \text{(Eq. 2)}$$

Where K_d is the rate coefficient of zero-order kinetic with diffusion limitation.

$$K_d = (ak_0D_e/2mC_0\delta)^{1/2}$$

First Order Model

$$\ln \frac{C}{C_0} = -k_1 t \qquad \text{(Eq. 3)}$$

Where C = Outlet concentration (ppm)

C_0 = Inlet concentration (ppm)

k_0 = Kinetic constant of Zero-Order Reaction Limited Model ($ppm.min^{-1}$)

K_d = The rate coefficient of Zero-Order Kinetic with Diffusion Limited Model (min^{-1})

k_1 = Kinetic constant of First-Order Model (min^{-1})

t = Empty Bed Resident Times (min)

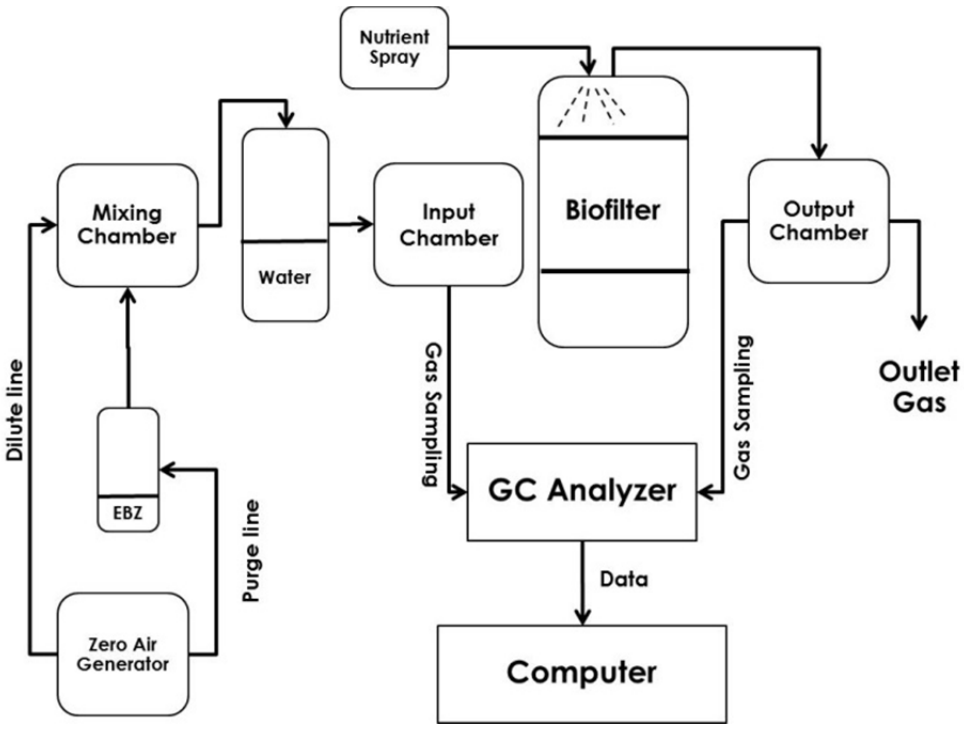

Figure 1 Schematic of the biofilter experimental system

Results and Discussions

The results of the study were exhibited as follows:

1. By plotting C_0-C with t as shown in Figure 2, the relation according to linear regression best fit was found 0.7965 for R^2. The Kinetic constant of the Zero-Order Reaction Limited Model was 3.2192 ppm.min^{-1}.

2. By plotting 1-$(C/C_0)^{(1/2)}$ with t as shown in Figure 3, the relation according to linear regression best fit was found 0.2744 for R^2. The rate coefficient of the Zero-Order Kinetic with Diffusion Limited Model was -0.0228 min^{-1}.

3. By plotting ln(C/C_0) with t as shown in Figure 4, the relation according to linear regression was found 0.5547 for R^2. The Kinetic constant of the First-Order Model was 0.5695 min^{-1}

The result concluded that the kinetics reaction of ethyl benzene degradation in the biofilter reactor was Zero-Order Reaction Limited Model (Highest R^2). This result indicated that the bacteria degradation ethyl benzene was the limited function of the reaction more than the diffusion rate of ethyl benzene from the gas phase into the biofilm layer. The efficiency of the system depends on the abilities to degrade the ethyl benzene. The rate of reaction wasn't depending on the inlet concentration. The inlet concentration of ethyl benzene increased the reaction rate was stable. The kinetic of the biodegradation of the biofilter system may be described by an equation: C_0-C = (3.2192xt) - 21.513. This equation can be applied for sizing the Empty Bed Resident Times (EBRT) of a biofilter reactor and also to predict the efficiency of the biofilter system.

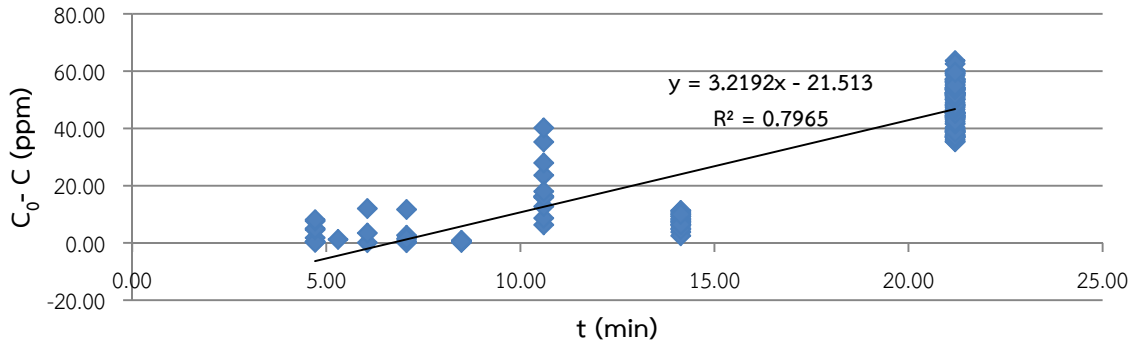

Figure 2 Zero-Order Kinetic with Reaction Limited Model by shown plotting C_0-C with t

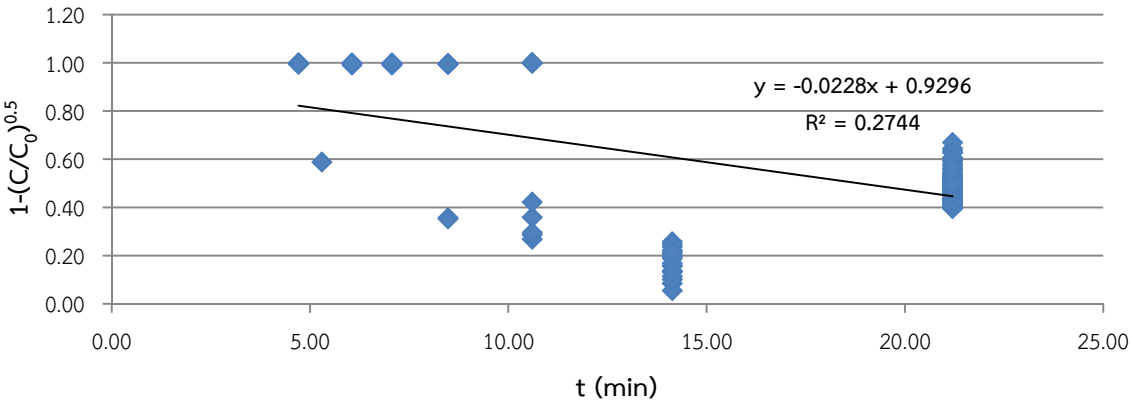

Figure 3 Zero-Order Kinetic with Diffusion Limited Model by plotting 1-$(C/C_0)^{(1/2)}$ with t

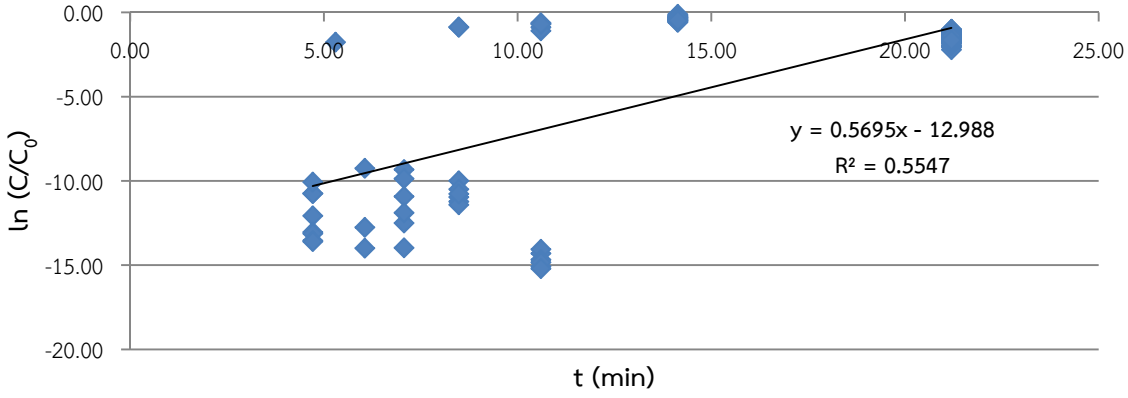

Figure 4 First-Order Model by plotting ln (C/C_0) with t

Conclusions

The kinetics of biofilter column packed with synthetics material using bacteria isolated from a petrochemical industry wastewater treatment plant. (*P. aeruginosa* S19 and *B. Cereus* O5-1/1). The Zero-Order Reaction Limited model was found for the kinetics degradation of ethyl benzene in biofilter system. The equation was C_0-C = (3.2192×t) - 21.513 and the constant of the kinetic reaction rate (k_0) was 1.8467 ppm.min^{-1}.

Acknowledgement

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Hybrid Aquatic System for Treatment of Domestic Wastewater Containing Pharmaceutical Residuals

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Abstract

This experimental research aimed to study the efficiencies of wastewater treatment and reducing pharmaceutical residuals by a hybrid aquatic system. Four laboratory-scale reactors, each filled duckweeds of 0.63 kg/m^2, were operated at the organic loading rate (OLR) of 12-50 kg COD/(ha-day) and illuminated with red LED at the light intensity of 150 µmol/(m^2-sec). Three reactors, each filled with plastic media at the specific area of 50 m^2/m^3, were fed with domestic wastewater at the hydraulic retention times (HRTs) of 5, 10 and 20 days; the fourth reactor was operated at the HRT of 10 days, but without plastic media. The experimental results showed the COD removal efficiency of 36.11, 40.74 and 76.85% at HRTs of 5, 10 and 20 days, respectively; while the reactor without plastic media had less COD removal efficiency. The hybrid aquatic system was also effective in removing sulfamethoxazole, a common anti-biotic drug, from the influent wastewater about 99%.

Keywords : hybrid aquatic system; domestic wastewater; pharmaceutical residuals; sulfamethoxazole; red LED

Introduction

Most Thai people usually consume antibiotic drugs for curing diseases and health protection. The residues antibiotic drugs will eventually be released together with domestic wastewater and can reach water sources, thus contaminating the ecosystem and food chain [1]. The hybrid aquatic system utilizes natural processes to treat wastewater and does not require complicated operation procedures. It involves aquatic plants to perform photosynthesis which produces oxygen for heterotrophic bacteria to decrease organic matters. These aquatic plants could also absorb nutrients (nitrogen (N) and phosphorus (P)) from the wastewater which help to purify the treated effluent. Duckweeds are a common aquatic plant in the tropics and have been employed for wastewater treatment [2, 3]. The objectives of

this study were: (1) to investigate the efficiencies the hybrid aquatic system incorporating duckweed and plastic media in treating a domestic wastewater, (2) to compare the efficiencies of the hybrid aquatic system equipped with red LED light and without red LED light, and (3) to investigate the efficiencies of the hybrid aquatic system in removing a common antibiotic drug (Sulfamethoxazole) [4].

Materials and Methods

Laboratory-scale reactors

Four laboratory-scale reactors, each with a size of 20×40×30 cm^3 (width × length × depth), were constructed at the Civil Engineering laboratory, Faculty of Engineering, Thammasat University, Rangsit Campus, Pathumthani province, Thailand (Figure 1). Three reactors were equipped with plastic media, each at the density of 50 m^2/m^3 (Figure 2) and duckweeds at the density of 0.63 kg/m^2. The last laboratory-scale reactor without plastic media was used as the control.

Experimental conditions

1) The three laboratory-scale reactors were operated at the hydraulic retention times (HRT) of 5, 10 and 20 days, corresponding to the organic loading rates (OLR) of 12-50 kgCOD/(ha-day). The control laboratory-scale reactor was operated at the HRT of 10 days. Experimental data obtained from this phase were used to determine the k_T value of COD removal by the hybrid aquatic system.

2) The effects of red LED light on the wastewater treatment efficiencies of the hybrid aquatic system were investigated by operating a laboratory-scale reactor at the HRT of 10 days and equipped with red LED light (Figure 3) at the light density of 150 μmole/m^2-sec which was previously found to be optimum for algal growth [5] (Figure 4). Another laboratory-scale reactor was operated at the HRT of 10 days, but without red LED light for comparing the treatment efficiencies.

3) The efficiency of the hybrid aquatic system in removing Sulfamethoxazole was investigated by feeding this antibiotic drug at the concentration of 35.60 μg/l to a laboratory-scale reactor equipped with red LED light and operated at the average HRT of 10 days. Three effluent samples were collected after 7 days of operation for determination of the Sulfamethoxazole concentration using LC-MS/MS technique at the department of Civil and Environmental Engineering, Mahidol University, Nakhon Pathom, Thailand in which the relative standard deviation was found to be 6.28% [6]. In addition, there effluent samples were examined for the presence of algal and protozoa species by a microscope at 10-40x.

Figure 1 Laboratory-scale reactors

Figure 2 Plastic media

Figure 3 Laboratory-scale reactors with red LED light

Figure 4 PAR Meter, Apogee, USA

Results and Discussion

Treatment efficiencies

The experimental results of the four laboratory-scale reactors are show in Table 1. The COD removal efficiencies of 36.11, 40.74 and 76.85% were found at the HRTs of 5, 10 and 20 days, corresponding to the effluent COD (filtered) concentrations of 70, 65 and 25 mg/l, respectively. The laboratory-scale reactors with the plastic media were very effective in removing SS and TKN with the effluent concentrations of below 10 mg/l and non-detectable, respectively. The control laboratory-scale reactor without plastic media apparently showed inferior treatment effciencies in which the effluent COD, SS, Coliform bacteria and TKN concentrations were higher than the laboratory-scale reactor operating at the same HRT of 10 days, suggesting the benefits of plastic media which had attached-growth microorganisms in wastewater treatment. The hybrid aquatic system operating at the HRT of 10 days was considered to be appropriate for domestic wastewater treatment because its effluent concentrations could meet the standards for discharge or reuse [7].

Table 1 Wastewater treatment efficiencies

Parameters	Influent concentrations	Effluent concentrations		
		HRT = 5 days	HRT = 10 days	HRT = 20 days
COD, filtered, mg/l	110±15	70±16	65±36 (75)*	25±6
SS, mg/l	22±18	8±1	6±3 (35)*	5±6
Coliform bacteria, CFU100/ml	3.51×10^{6}	3.19×10^{6}	9.84×10^{5} (1.30×10^{6})*	2.46×10^{5}
TKN, mgN/l	6	ND	ND (2)*	ND

( )* = Average water quality from reactor without plastic media

ND = non-detectable

Table 2 Wet weight of duckweeds

Parameters	Effluent concentrations		
	HRT = 5 days	HRT = 10 days	HRT = 20 days
Wet Weight of duckweeds before treatment, kg/m^2	0.63	0.63 (0.63)*	0.63
Wet Weight of duckweeds After treatment, kg/m^2	0.70	0.65 (0.73)*	0.60

( )* = Wet weight of duckweed in reactor without plastic media

Table 2 shows the wet weights of duckweeds at the begining and after 30 days of experiments which indicated some increases especially at the HRT of 5 days because of more nutrients inputs. The reactor without plastic media operating at the HRT of 10 days had higher duckweed growth than that without plastic media, probably because there were no attached-growth microorganisms to compete for their growth. These duckweeds could be harvested for reuses as animal feeds or as a raw marterial for composting [8].

By assuming plug flow conditions in the laboratory-scale reactors, the first-order kinetic value (k_T) for COD removal could be determined from equations (1) and (2) as follow :

$$\frac{C_e}{C_o} = e^{-k_T t} \quad (1)$$

$$k_T = k_{20}\,(1.06)^{T-20} \quad (2)$$

when

C_0 = Influent COD concentration, mg/l

C_e = Effluent COD concentration, mg/l

t = hydraulic retention time, days

k_T = kinetic coefficient at temperature T, day^{-1}

k_{20} = kinetic coefficient temperature at 20 °C

T = liquid temperature, °C

From the data of Table 1 in which the liquid temperature was 30 °C, the k_{30} was found to be 0.071 day^{-1} and the k_{20} was found to be 0.040 day^{-1}. These k values can be used for design and operation of a hybrid aquatic system to achieve the desired effluent COD concentration.

Effects of red LED light

Table 3 shows the wastewater treatment efficiencies of the 2 laboratory-scale reactors operating at the HRT of 10 days, one with red LED light and the other wiyhout. It can be seen that the effluent concentrations of the laboratory-scale reactor with red LED light were lower than those of the one without red LED light. These results suggest the benefits of red LED light in wastewater treatment by the hybrid aquatic system probably due to increased photosysthetic reactions which supported the algal-bacterial synthesis. In areas with limited space, the hybrid aquatic system equipped with red LED light could therefore be constructed indoor for wastewater treatment with high efficiencies.

Table 3 Wastewater treatment efficiencies with and without red LED light

Parameters	Influent concentrations	Effluent concentrations
COD, filtered, mg/l	164±41	48±6 (74±9)*
SS, mg/l	31±4	11±4 (19±5)*
Coliform bacteria, CFU100/ml	6.50×10^4	3.30×10^4 (4.30×10^4)*
TKN, mgN/l	5.6	ND (ND)*

( )* = Average water quality from reactor without red LED light

Efficiency of Sulfamethoxazole removal

A laboratory-scale reactor with plastic media and red LED lighting at 150 µmole/m^2-sec was operated at the HRT of 10 days and continuously fed with Sulfamethoxazole at the concentration of 35.60 µg/l. Three effluent samples were analyzed for the Sulfamethoxazole concentrations which showed the average value to be 0.02 µg/l (Table 4). This result suggested the effectiveness of the hybrid aquatic system in removing Sulfamethoxazole from domestic wastewater, probably through such mechanisms as biodegradation, plant uptake, absorbtion and sedimentation, further reseach in this area is suggested. Because antibiotic drug contemination of water resources is becoming more serious [4], the application of the hybrid aquatic system in removing these antibiotic drugs is highly strongly recommended (Figures 5-7) represent major algal species while (Figures 8-10) are the major protozoa species found in the hybrid aquatic system. These results suggest the diversity of mixed algal and protozoa species in the hybrid aquatic system which helped to support the algal-bacterial synthetic reactions respondsible for wastewater treatment including Sulfamethoxazole degradation.

Table 4 Sulfamethoxazole concentrations

Parameters	Influent concentrations	Effluent concentrations
Sulfamethoxazole concentrations,µ g/l	35.60±2.24	0.02±0.01

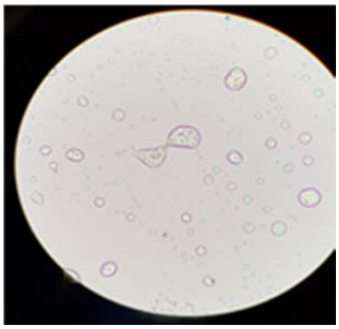

Figure 5 *Sphaerocystis sp.*

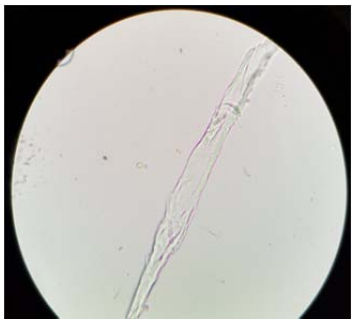

Figure 6 *Mougeotia sp.*

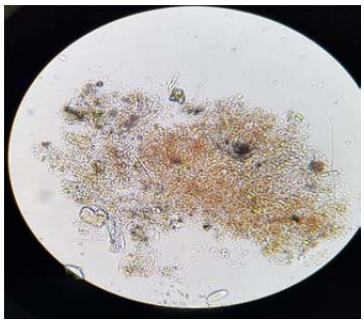

Figure 7 *Navicula sp.*

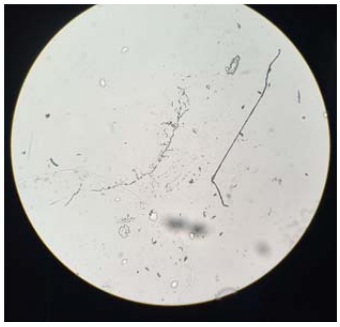

Figure 8 *Paramecium sp.*

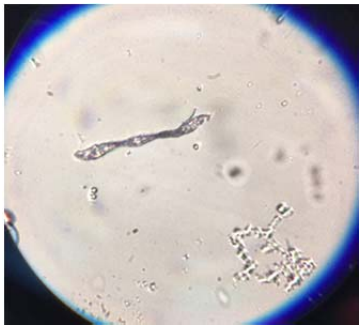

Figure 9 *Euglena tripteris*

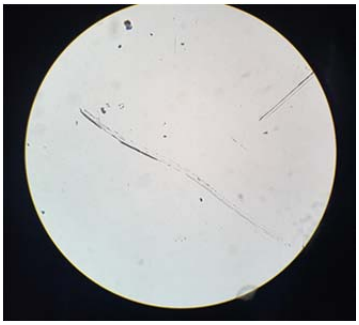

Figure 10 *Tintinnopsis radix*

Conclusions

Based on the experimental results of this study, the following conclusions are made:

1. The hybrid aquatic system was found to be effective in domestic wastewater treatment in which the treated effluent characteristics could meet the discharge standards when operated at the HRT of 10 days.

2. The k_{20} of the hybrid aquatic system was found to be 0.040 day^{-1}, applicable for system design and operation.

3. The hybrid aquatic system equipped with red LED light was found to be more effective in wastewater treatment than that without, suggesting its applicability for construction and operation indoor in areas with limited space.

4. The hybrid aquatic system with red LED light was found to be very effective in removing Sulfamethoxazole more than 99% from the influent wastewater.

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Hydrogen peroxide production in *Anubias barteri*, *Echinodorus ozelot* and *Cabomba caroliniana* by induction of 17α-Ethinylestradiol

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Abstract

The 17α-ethinylestradiol (EE2), a synthetic estrogen has become public concern on the contamination in the aquatic environment because of its endocrine-disrupting properties which can cause adverse effects on aquatic organisms. EE2 is reported to be effectively removed by advanced wastewater treatment technologies; however, due to its high operating cost other low-cost technologies such as constructed wetland might be more suitable. Since certain plant species were able to produce endogenous hydrogen peroxide (H_2O_2) under stress condition including EE2 and induce hydroxyl radicals (•OH) formation when exposing to external dissolved iron through the Fenton-like reaction. This Phyto-Fenton process might be possible to combine in constructed wetland for EE2 removal. In this study, aquatic commercial plants including Anubias *(Anubias barteri)*, Amazon *(Echinodorus ozelot)*, and Green Cabomba *(Cabomba caroliniana)* were introduced to the solution containing different EE2 concentrations as an environmental stressor to investigate H_2O_2 production. The results revealed the increasing of H_2O_2 production following the increase of EE2 concentration in Amazon, while Anubias and Green Cabomba showed no difference with the control. Based on morphological observation, Amazon branch appeared healthy compared to other plants after exposing to EE2 for 21 days. The results suggested that Amazon was the most suitable aquatic commercial plant for applying to remove EE2 by Phyto-Fenton in constructed wetlands.

Keywords : 17α-ethinylestradiol (EE2); Aquatic commercial plants; Fenton-like reaction; Hydrogen peroxide (H_2O_2); Phyto-Fenton process; Oxidative stress

Introduction

Nowadays, there is a public concern on the estrogen hormone contamination in the aquatic environment as they can interfere with the endocrine system in aquatic animals. The concerned estrogen hormones include natural estrogens such as estrone (E1), 17β-estradiol (E2), and estriol (E3) and the synthetic estrogen such as 17α-ethinylestradiol (EE2). The estrogen hormones have been detected in surface water at concentrations lower than 0.1-17 ng/L for E1, 0.05-15.5 ng/L for E2, lower than 0.1-3.4 ng/L for E3 and lower than 0.053-30.8 ng/L for EE2 [1]. While the predicted no-effect concentrations (PNEC) were 3 ng/L, 1 ng/L, and 0.1 ng/L for E1, E2, and EE2, respectively [2]. EE2 appeared to be the most recalcitrant compound in environment with no or small degradation rate [3]. Therefore, EE2 was the most concerned chemical among other estrogens. In addition, the occurrence of EE2 in surface water in developing countries was observed at higher levels than in developed countries [4].

Homogeneous fenton reaction is the mechanism involving with hydroxyl radical (•OH) that is produced by the interaction of H_2O_2 and dissolved ferrous salt in a bulk solution (Equation 1). The hydroxyl radical is a powerful oxidizing species (E^0 = 2.80 V vs. NHE) that can eliminate various organic pollutants [5]. Since the ferrous ion is unstable at a pH higher than 4.0, it can easily form unwanted solid iron-oxy hydroxide sludge FeO(OH) [6]. Its application under neutral pH in wastewater treatment plants is not practical. Moreover, the dissolved iron can leach to aqueous environment causing additional post-treatment cost. The heterogeneous Fenton reaction utilizing solid ferric iron materials as a catalyst is feasible over a wide pH range [7]. The reaction of hydrogen peroxide with ferric ion is referred to as a Fenton-like reaction (Equation 2) which produces superoxide radical (HO_2•) that has lower oxidation power (E^0 = 1.65 V vs. NHE) than hydroxyl radical. In addition, the ferric iron can be reduced further to ferrous iron (Equation 2-4), which then reacts with H_2O_2 to produce highly reactive •OH via Equation 1 [8].

$$Fe^{2+} + H_2O_2 + H^+ \longrightarrow Fe^{3+} + HO^- + \bullet OH \quad (1)$$

$$Fe^{3+} + H_2O_2 \longrightarrow Fe^{2+} + HO_2\bullet + H^+ \quad (2)$$

$$Fe^{3+} + HO_2\bullet \longrightarrow Fe^{2+} + O_2 + H^+ \quad (3)$$

$$Fe^{3+} + O_2^-\bullet \longrightarrow Fe^{2+} + O_2 \quad (4)$$

Plants can generate reactive oxygen species (ROS) responded to biotic or abiotic stresses. It is regulated by autonomous factors such as age, reproductive development, and phytohormone levels and by environmental signals including photoperiod, stresses, drought, ozone, nutrient deficiency, wounding, and shading [9]. Under normal condition, the production of ROS in cells is maintained at low level by antioxidant enzymes. This balance can be disrupted by a depletion of antioxidants or an accumulation of ROS, leading to an increasing of oxidative stress, and consequently to damage cellular macromolecules and membranes [10]. H_2O_2 is one of the major ROS in plant tissues. It is produced in chloroplasts and mitochondria via electron transport systems during normal metabolic processes [11]. H_2O_2 plays an important role in plants under stress conditions as a signaling molecule that mediates between different physiological processes in response to environmental stress [12].

Recent studies indicated that H_2O_2 is produced at a concentration range between 0.4-1.5 μmol/gFW in typha (*Typha spp.*), vetiver (*Vetiveria zizanioides*), reed (*Phragmites australis*) and bird of paradise (*Strelitzia reginae*) under E2

stress conditions [13]. Also, H_2O_2 produced in response to EE2 stress at concentration of 0.6-1.5 μmol/gFW in large-flowered waterweed (*Egeria densa*), green cabomba (*Cabomba caroliniana*), amazon frogbit (*Limnobium laevigatum*) and greater duckweed (*Spirodela polyrhiza*) [7]. As plants can produce endogenous H_2O_2, it is possible to react with iron catalysts and eliminate recalcitrant pollutants. This technology is defined as Phyto-Fenton process [7] and possible to incorporate into a constructed wetland with fenton like reaction at neutral pH.

Based on these background, this study investigated how the EE2 can affect to H_2O_2 production in different EE2 concentrations in three aquatic commercial plants including; Anubias, Amazon and Green Cabomba. The important criteria for aquatic plant selection were easily growing and economic value as they can be benefit to livestock farmer when apply to wastewater treatment. The plant with highest H_2O_2 production level might be possible to apply to the advanced oxidation Phyto-Fenton wetland for EE2 removal.

Methodology

Experimental setup and aquatic plant acclimatization

The study was conducted in glass containers with a dimension of 21 cm × 21 cm × 21 cm (Figure 1). There were three aquatic plants used in this study including Anubias (*Anubias barteri*), Amazon (*Echinodorus ozelot*), and Green Cabomba (*Cabomba caroliniana*). They were acclimatized in Hoagland's nutrient solution (59.04 mg/L $Ca(NO_3)_2{\cdot}4H_2O$, 25.28 mg/L KNO_3, 24.65 mg/L $MgSO_4{\cdot}7H_2O$, 6.80 mg/L KH_2PO_4, 0.18 mg/L $MnCl_2{\cdot}4H_2O$, 0.01 mg/L $ZnCl_2$, 0.005 mg/L $CuCl_2{\cdot}2H_2O$, 0.005 mg/L $Na_2MoO_4{\cdot}2H_2O$, and 0.286 mg/L H_3BO_3) [7] before starting the experiment. Each type of plant was cultured in an individual container with tree plants per tank. Initial fresh weights of Anubias, Amazon and Green Cabomba were 9 ± 0.7 g, 7 ± 0.9 g and 12 ± 0.5 g, respectively. Three concentrations of EE2 (Sigma-Aldrich, China) including 1 ng/L, 1 μg/L, and 1 mg/L were applied to triplicate sets. There was also a control set which was free of EE2.

Operation and Sampling

The artificial sunlight was provided with the 12 h of light followed by 12 h of dark cycle. The irradiation intensity ranged 2000-4000 lux in different locations. The 16 W LED daylight tubes (Ecofit LED tubes T8, PHILIPS) were used. The temperature in the growth chamber was about 26-30°C. The experiments were run in batch mode for 21 days and plants samples were collected at day 1, 3, 5, 7, 9, 11, 13, 17, 19 and 21.

H_2O_2 Analysis

The method of endogenous H_2O_2 analysis was followed Velikova *et al*. [14] with some modification. Leaf tissues (100-500 mg) were homogenized in liquid nitrogen and then introduced to 5 mL of 0.1% (w/v) Trichloroacetic acid (Merck, Germany). The homogenate was centrifuged at 12,000×g for 15 min and 0.5 mL of supernatant was subsequently transfered to a new tube containing 0.5 mL of 10 mM potassium phosphate buffer (pH 7.0) and 1 mL of 1 M Potassium iodide. The absorbance of the supernatant was measured at 390 nm by Spectorphotometer (Thermo Scientific GENESYS20, USA). The content of H_2O_2 (30%, Merck, Ausalia) in samples were determined relative to a standard curve.

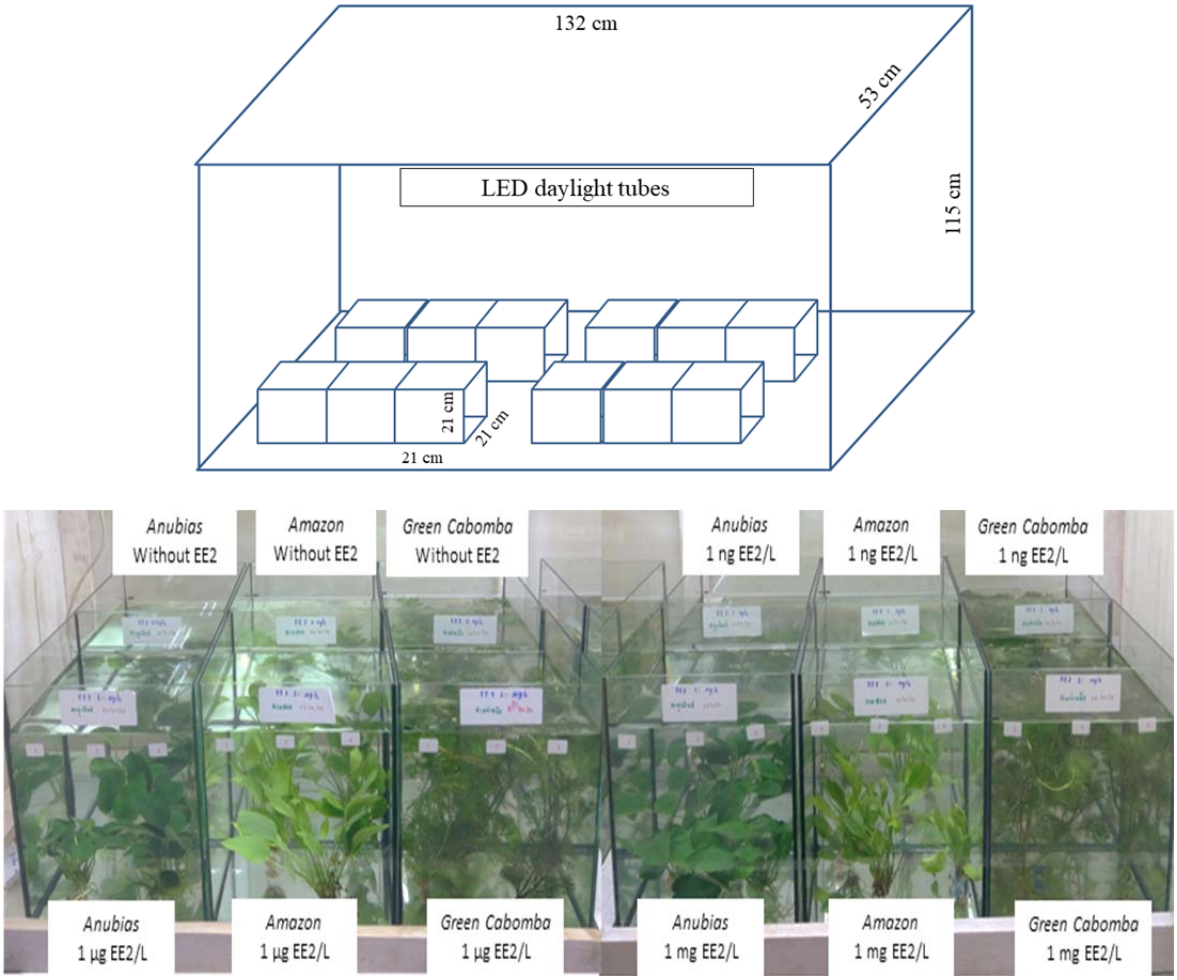
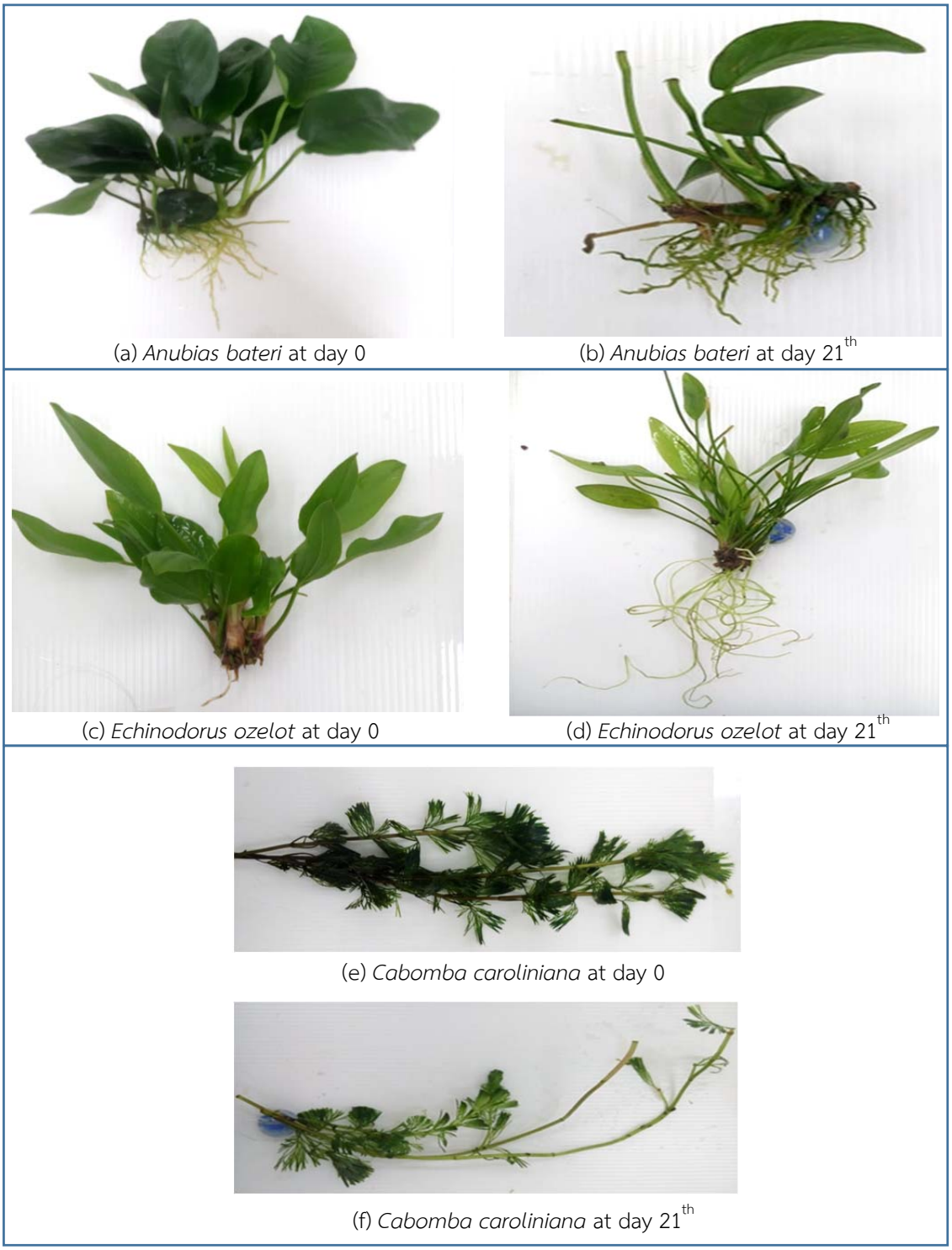


Figure 1 Experimental setup

Results and Discussions

Plant Morphology

In order to apply the aquatic commercial plants for EE2 removal with H_2O_2 induction, the resistance of the plants to the study conditions is important. We observed the physical appearance or morphology of the selected plants at 1 mg/L EE2 concentration during 21 days period. Figure 2 shows the morphology of three aquatic plants at day zero and the end of experiment at day 21. Judging from the appearance, Amazon was the most healthy compared to Anubias and Green Cabomba. Some leaf or stalk of Anubias and Green Cabomba were spoilage and roots were not regenerated as observed in Amazon. Therefore, amazon was more suitable to apply to the EE2 treatment among all selected plant species.



(a) *Anubias bateri* at day 0

(b) *Anubias bateri* at day 21th

(c) *Echinodorus ozelot* at day 0

(d) *Echinodorus ozelot* at day 21th

(e) *Cabomba caroliniana* at day 0

(f) *Cabomba caroliniana* at day 21th

Figure 2 The appearance of three aquatic plants at the start (day 0) and end (day 21th) of the operation

Concentration of endogenous H_2O_2

Figure 3 demonstrates the production of endogenous H_2O_2 under different EE2 concentrations. For the baseline condition without EE2 addition (Figure 3a), the H_2O_2 concentrations in Anubias were quite high during the beginning period and then reduced; however, the concentration in Amazon and Green Colomba returned to the high level at the end period.

The H_2O_2 concentrations after exposed to EE2 were shown in Figure 3b – d. The H_2O_2 concentrations in all plant species followed the same trend with the EE2 free condition. While increasing the EE2 concentrations, Anubias and Green Cabomba did not respond to the changes. Except for Amazon, the H_2O_2 concentration increased at higher EE2 concentrations. This indicated that only Amazon responded to the EE2 stress.

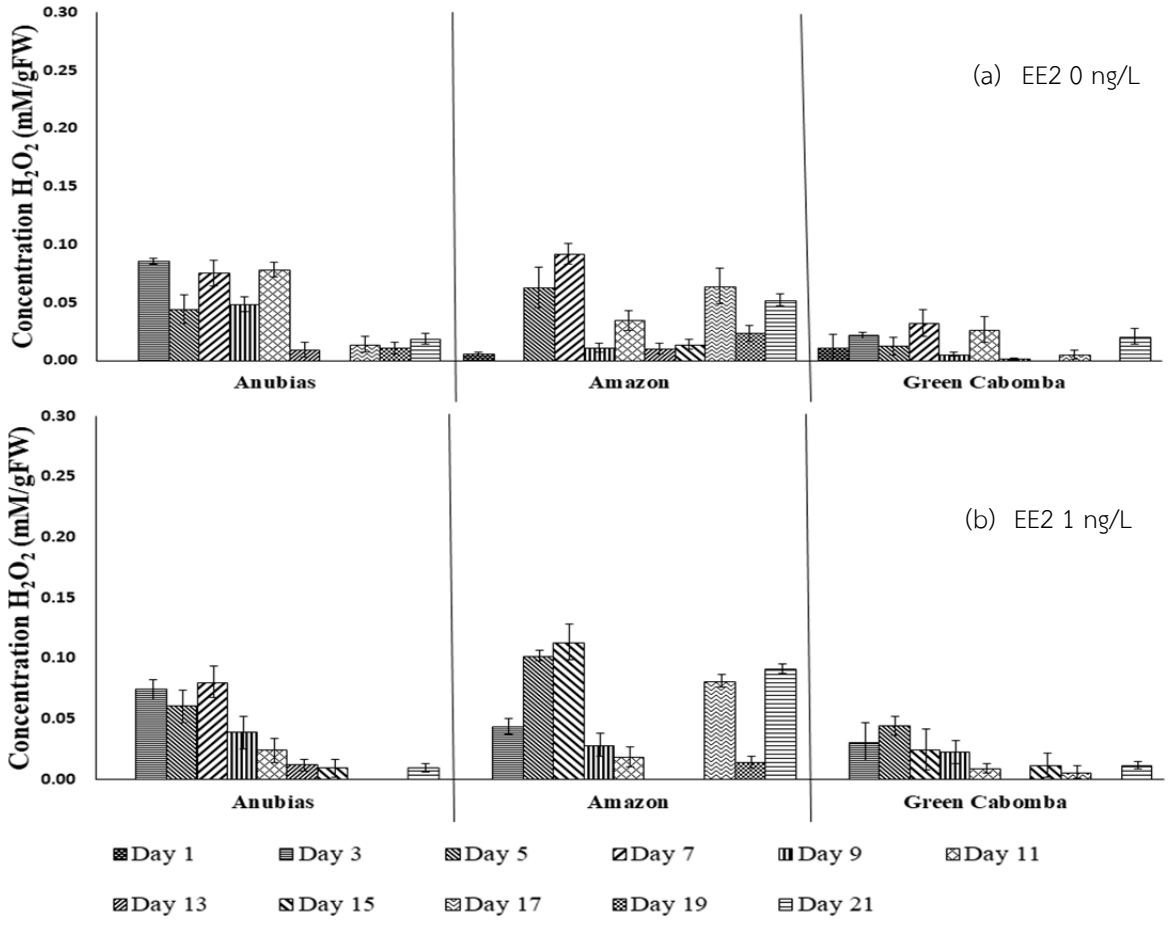


Figure 3 H_2O_2 concentration in tested aquatic plants induced by different EE2 concentrations. Error bars indicate standard error

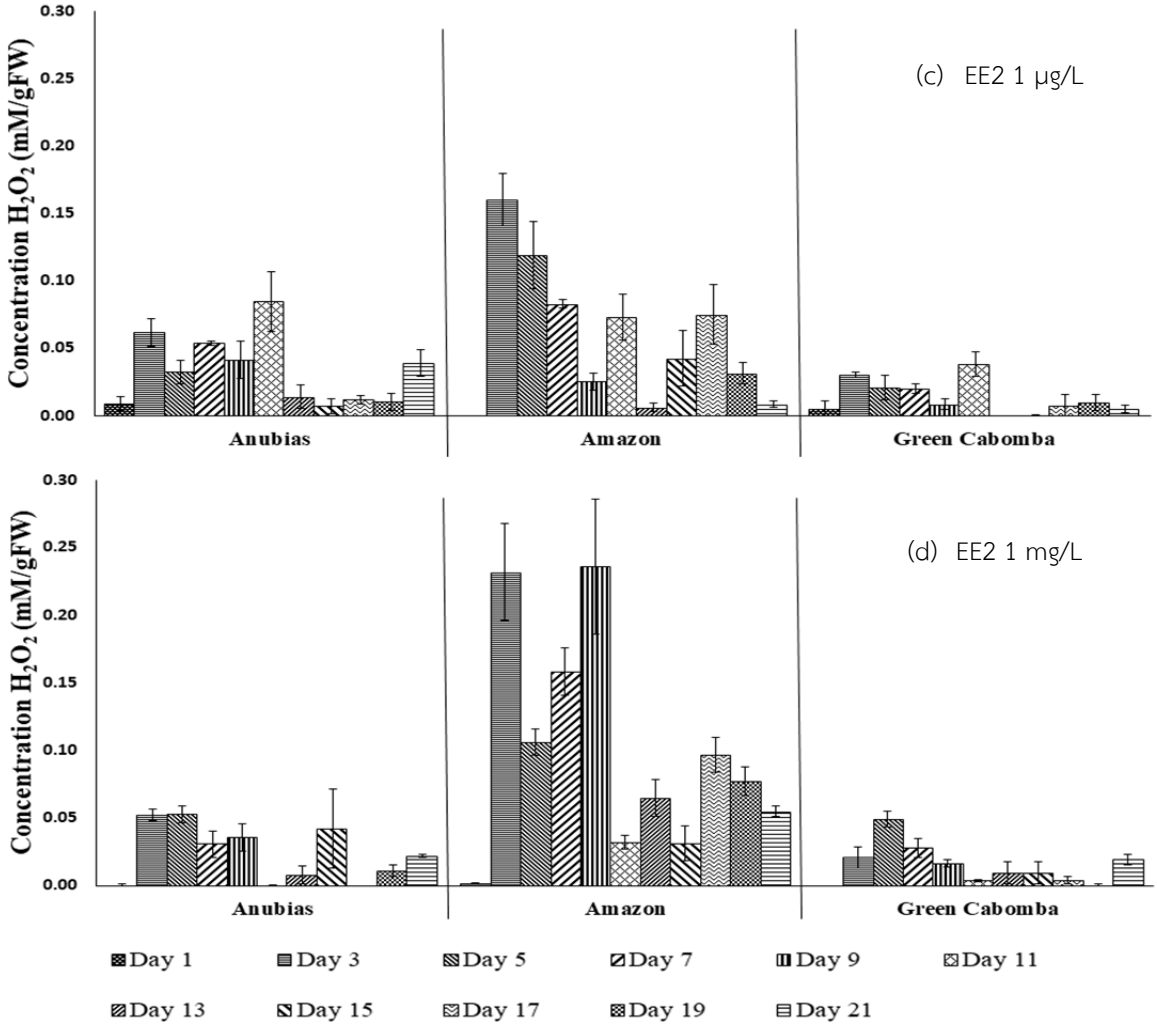


Figure 3 H_2O_2 concentration in tested aquatic plants induced by different EE2 concentrations. Error bars indicate standard error (cont')

Table 1 showed the averaged H_2O_2 concentrations in Anubias, Amazon, and Green Cabomba during operation were 0.035, 0.034, and 0.013 mM/gFW for without EE2, 0.028, 0.045, and 0.014 mM/gFW for 1 ng/L, 0.033, 0.057, and 0.013 mM/gFW for 1 µg/L, and 0.023, 0.099, and 0.015 mM/gFW for 1 mg EE2/L, respectively. Elevated levels of H_2O_2 at higher EE2 concentration were observed only in Amazon. The change of H_2O_2 with the EE2 addition did not observe in Anubias and Green Cabomba. These results suggested Amazon was the most suitable plant to apply to phyto-fenton advanced oxidation wetland among all selected plant species based on H_2O_2 production.

Table 1 H_2O_2 concentrations in three aquatic plants during operation pediod (21 day)

EE2 concentration	Aquatic plant	Concentration H_2O_2 (mM/gFW)		
		Average	Minimum	Maximum
Control	Anubias	0.035	<0.001	0.086
	Amazon	0.034	<0.001	0.092
	Green Cabomba	0.013	<0.001	0.032
1 ng/L	Anubias	0.028	<0.001	0.080
	Amazon	0.045	<0.001	0.113
	Green Cabomba	0.014	<0.001	0.044
1 µg/L	Anubias	0.033	0.007	0.085
	Amazon	0.057	<0.001	0.160
	Green Cabomba	0.013	<0.001	0.038
1 mg/L	Anubias	0.023	<0.001	0.053
	Amazon	0.099	0.002	0.236
	Green Cabomba	0.015	<0.001	0.049

Conclusion

Based on the experimental results obtained in this study, the following conclusions are made:

- The highest H_2O_2 concentration was found in Amazon (0.034-0.099 mM/gFW), followed by Anubias (0.023-0.035 mM/gFW) and Green Cabomba (0.013-0.015 mM/gFW).
- The H_2O_2 concentrations in Amazon increased with the higher EE2 concentration which was likely due to the environmental stress accumulation. The induction trend was not observed in the other selected plants.
- Amazon was more resistance to EE2 exposure than Anubias and Green Cabomba during 21 days.
- Among tested aquatic commercial plants, Amazon was the most suitable plant to apply for testing EE2 removal by Phyto-Fenton process.

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Duration of Elevated Starting Temperature Influencing Food Waste Composting

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Abstract

This study aims to improve composting efficiency by elevating the starting temperature at mesophilic phase in order to accelerate microbial activities. The controlled starting temperature at 40°C was applied for 12 and 24 hours to a pile of synthetic food waste in the designed composting reactor compared with the conventional composting process. Composting parameters such as temperature, mesophilic and thermophilic bacterial quantities were investigated. It was found that the controlled starting temperature at 40°C for different duration showed significant enhancement for composting temperature profile, maximum temperature and bacterial quantity compared to the conventional condition. Moreover, the condition of 40°C elevated temperature for duration of 24 hours exhibited the higher maximum temperature and bacteria quantities compared to that of 12 hours. It is concluded that the composting efficiency could be improved by using the suitable duration of controlled starting temperature.

Keywords : food waste; composting; temperature control; elevated temperature; starting temperature

Introduction

In Thailand, about 27.40 million tons of municipal solid waste (MSW) were generated in 2017. However, only 20.22 million tons were properly disposed [1]. The remaining 7.17 million tons thus have caused serious pollution to the environment. More than 60% of MSW were organic [2] that can be transformed into useful products. One of a well known methods to convert organic waste into valuable materials is composting. Generally, composting can reduce organic waste volume by 40 to 50% [3]. Composting is the biological decomposition and stabilization process of organic substrates, under suitable conditions to produce a final product that is stable and free of pathogens [4].

Composting temperature is found to be a significant issue affecting composting efficiency. Temperature is effective factor in the composting process related to the bacteria activities and the composting progress [3]. It was reported that bacteria play a major role in the decomposition of organic matter in composting pile with their type dependence [5]. However, general conventional composting processes usually

proceed under environmental temperature which occasionally cause low microbial activity and thus affect to low degradation efficiency. The aim of this study is to improve composting efficiency by elevating starting temperature at mesophilic phase in order to accelerate microbial activity. The controlled starting temperature at 40°C was applied for 12 and 24 hours to a pile of synthetic food waste in the composting reactor compared with conventional composting process. Composting efficiency including temperature, mesophilic and thermophilic bacteria quantities were investigated and discussed.

Materials and Methods

Materials for composting

The main substrate in the composting material is synthetic food waste consisting of 20% vegetable scraps, 30% fruit scraps, 25% fish scraps, and 25% rice scraps. Dry leaves were mixed with the synthetic food waste to make the starting carbon to nitrogen ratio 36, as this ratio should be in the range 25-50 [6]. Moisture content of the composting material mixture was 60%, while the recommended range for it is 50-70% [5,7]. The synthetic food waste and dry leaves were crushed to 1-2 cm pieces prior to composting.

Composting reactor

A composting material batch of 7.32 kg was placed in a simple composting reactor and the lid made of stainless steel in cylindrical shape with 36 cm diameter and 36 cm height. The total volume of the reactor is 0.036 m^3 and the composting material was filled in the composting reactor with maintaining of 15% free volume. The reactor is surrounded by 3,000 watts of heating system. The air supply inlet is a perforated plate at the bottom of the reactor, using an air flow rotameter (DWYER, model: RMA–22SSV) to control the aeration rate at 0.5 L/min/kg that was calculated based on oxygen requirement for decomposition of organic matter [6]. The perforated plate at the bottom of the reactor distributes the air supplied by a compressor and the plate also filters the leachate. The reactor has 3 thermocouple probes located at top, middle, and bottom along the centerline, to monitoring the temperature profile in the composting pile. The average composting temperature of the 3 positions was then calculated. The initial composting temperature was the ambient temperature. A schematic diagram of the reactor is shown in Fig. 1. The composting was carried out until the pile had returned to ambient temperature, carbon to nitrogen ratio was below 20, and oxygen concentration had returned to ambient level. The composting was completed when all these mentioned parameters remained constant. Each run was performed in triplicate.

Result and Discussion

The average temperature profiles of food waste composting when starting temperature at 40°C was controlled for 12 and 24 hours compared with conventional composting condition are shown in Fig. 2. In the conventional composting condition, the temperature started to increase immediately after composting begun. The compost temperature is in the range of 29.1-51.2°C from day 1 to 30 with reaching the maximum temperature of 51.2°C within 72 hours. In case of using elevated temperature at duration of 12 hours, when reaching the setting starting temperature of 40°C, the composting pile temperature stayed constant at 40°C for 11 hours before rising to the maximum

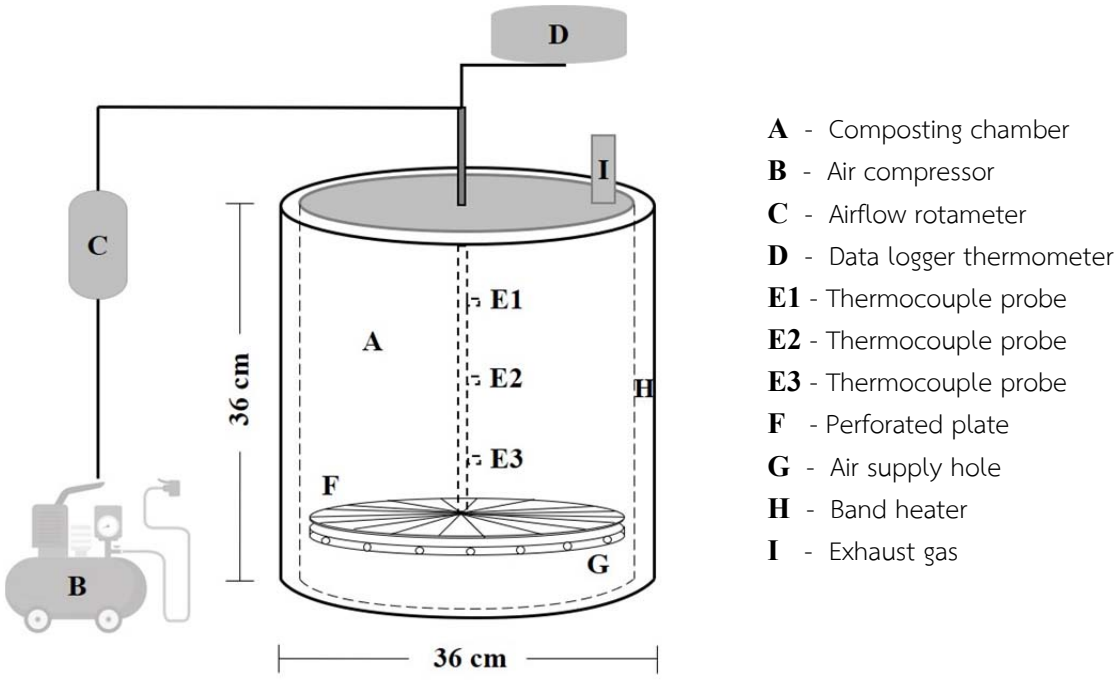

Fig. 1 Schematic diagram of the composting reactor

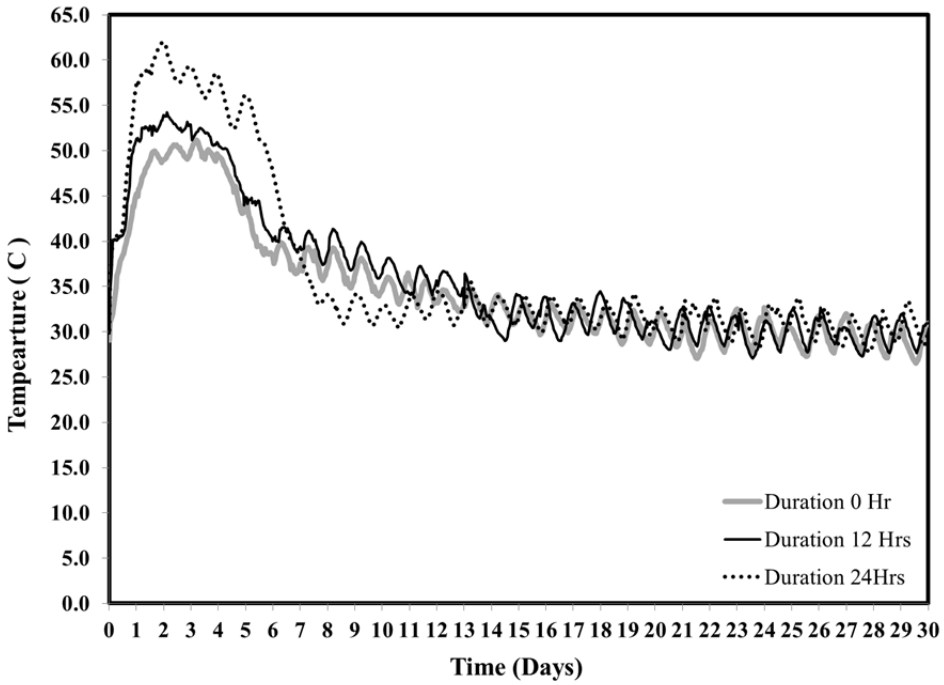

Fig. 2 Average temperature versus composting time

temperature of 54.2°C within 51 hours. The temperature was in the range of 30.8-54.2°C from day 1 to 30. In case of using elevated temperature at duration of 24 hours, after reaching the setting starting temperature of 40°C, the temperature of composting pile stayed unchanged for 9 hours and started to increase afterward**.** It is noticed that in this case the buffering time or lag time is decreased compared to a controlled starting temperature for 12 hours. This may be attributed to the duration of 24 hours is suitable condition without heat transfer to environment thus leads the microorganisms for a prompt switching from a low activity to a high activity [3]. The composting pile reached the maximum temperature of 62.0°C within 48 hours and the temperature was in the range of 30.2-62.0°C from day 1 to 30. It is seen that elevated temperature at duration of 24 hours exhibited the highest maximum temperature and the fastest decrease to the ambient temperature level.

The aerobic bacteria that involve in composting process have been measured and the quantities of mesophilic bacteria are shown in Fig. 3. It is found that there is significant difference of mesophilic bacteria quantity with varying elevated duration of starting temperature. In the conventional composting condition, the maximum quantity of mesophilic bacteria was 10.11 Log CFU/g. While, the maximum quantities of mesophilic bacteria were 9.98 and 9.96 Log CFU/g In case of using 12 and 24 hours, respectively. It is noticed that quantity of mesophilic bacteria is directly related to the composing pile temperature. That is when composting pile temperature is below than 45°C, the quantity of mesophilic bacteria is high which is due to a preference temperature of mesophilic bacteria is in the range of 20-45°C. However, the decreasing trend of mesophilic bacteria quantities is remarkably noticed when the composting pile temperature is higher than 50°C. This is because at the high temperature most of mesophilic bacteria is inactivated.

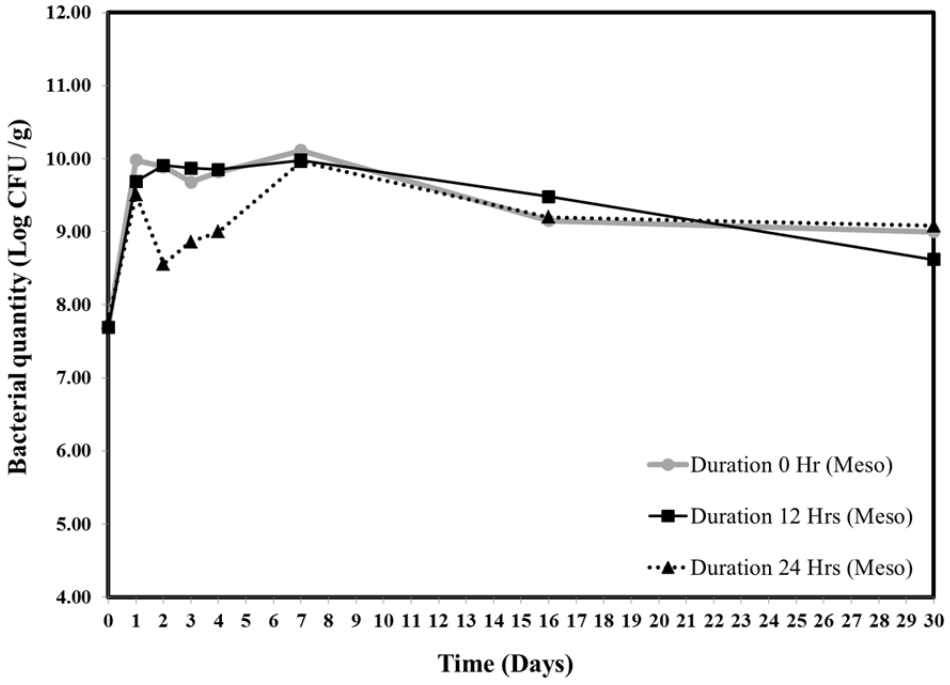

Fig. 3 Mesophilic bacteria quantity versus composting time

The quantity of thermophilic bacteria is also shown in Fig. 4. The increase of thermophilic bacteria quantity found to straightly correspond with the temperature raising in composting pile that was contribute from exothermic reaction via decomposition of organic matter. It was found that the condition in which exhibited a higher maximum temperature, a higher thermophilic bacteria quantity was found. Therefore, the highest maximum number of thermophilic bacteria (9.99Log CFU/g) is found in case of using elevated temperature at duration of 24 hours.

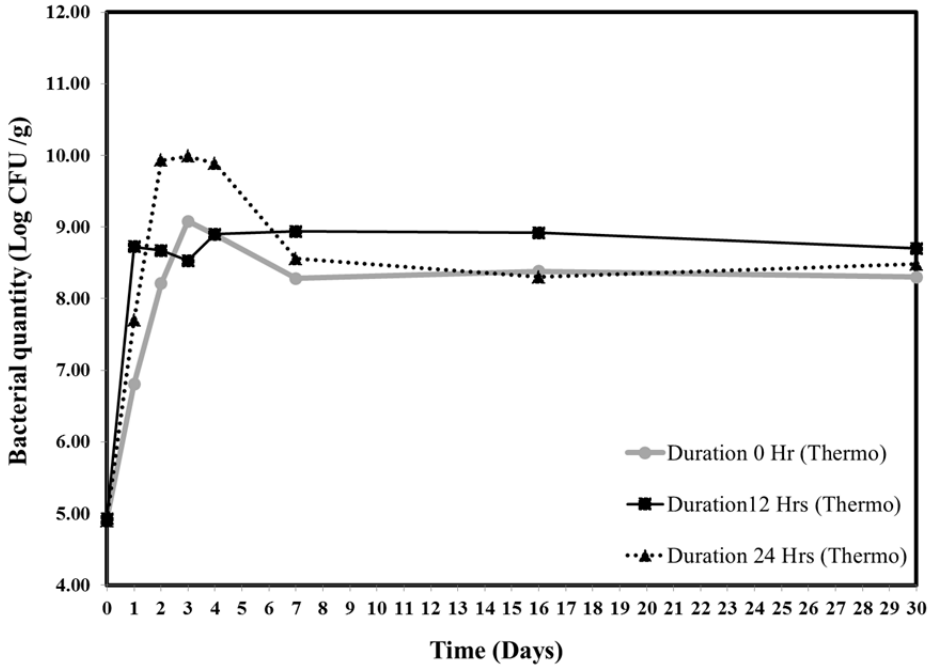

Fig. 4 Thermophilic bacteria quantity versus composting time

Conclusions

Controlled starting temperature at 40°C for different duration showed significant enhancement in composting temperature profile, maximum temperature and bacteria quantities compared to the conventional condition. Additionally, the condition of 40°C elevated temperature at duration of 24 hours exhibited the higher maximum temperature and bacterial quantities compared to that of 12 hours. Therefore, the composting efficiency can be improved by using suitable duration of controlled starting temperature.

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Font, font size & typeface

TH SarabunPSK font type is required for Thai text and English text. Font size [Pica] for various text function are tabulated as follow.

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Address of correspondence	14 [CT]	Normal
Abstract heading	16 [LRJ]	Bold
Abstract & Main Texts	14 [LJ]	Normal
Section Heading & Number*	16 [LJ]	Bold
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All titles of manuscript should be short and precise; long title should be condensed whenever possible (not more than 42 characters). Title should be printed with every first letter of every word capitalized, excluding prepositions and articles. Directly below the title, author should print their full names (first name then family name), address and institution. E-mail of corresponding author and between 3-5 key words should also be provided.

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Abstract in both Thai and English, should be provided on separate sheets and be not more than 300 words. International contributor who are unable to provide an abstract in Thai may submit an English abstract alone.

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Units of measurement should be indicated in SI units throughout.

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Tables and figures should be numbered with Arabic numerals, in order in which they are cited in the text. The table's titular heading should concisely detail the content of the table and include units of measure for all numerical data.

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[1] Inthorn, D., Sidtitoon, N., Silapanuntakul, S. and Incharoensakdi, A. 2002. Sorption of mercury, cadmium and lead in aqueous solution by the use of microalgae. Science Asia. 28 (3): 253-261.

Reference to article or abstract in a conference proceedings:

[1] Inthorn, D., Singhakarn, C. and Khan, E. Decolorization of reactive dyes by pre-treated Flute reed (phargmites karka (Retz)). At 34th Mid-Atlantic Industrial & Hazardous Conference, Annual Mid Atlantic Industrial and Hazardous Waste Conference at Rugers University, New Jersey, USA on September 20-21, 2002.

Reference to a book:

[1] Polprasert, C. 1996. Organic Waste Recycles. John Wiley & Sons Inc., New York.

Reference to article in a conference proceedings:

[1] Inthorn, D. Heavy metal removal. In: Kojima, H. and Lee, Y.K. Photosynthetic Microorganisms in Environmental Biotechnology, Springer-Verlag, 2001; 111-135.

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