Malt house wastewater treatment with settleable algal-bacterial flocs

Luboš Stříteský, Radka Pešoutová and Petr Hlavínek

ABSTRACT

This paper deals with biological treatment of malt house wastewater using algal-bacterial flocs. During three months of testing, optimisation of growth conditions and biomass separation leads to maximisation of biomass production, improved flocs settleability and increased pollutant removal efficiency while maintaining low energy demand. At a high food to microorganism ratio (0.16 to 0.29 kg BOD_5 kg^{-1} TSS d^{-1}), the biological oxygen demand (BOD_5), chemical oxygen demand (COD_{Cr}), total phosphorus (P_{tot}) and total suspended solids (TSS) removal efficiencies were all higher than 90%. At a food to microorganism ratio of 0.06 kg BOD_5 kg^{-1} TSS d^{-1}, BOD_5, COD_{Cr}, total nitrogen (N_{tot}), P_{tot} and TSS removal efficiencies of 99.5%, 97.6%, 91.5%, 97.8% and 98.4%, respectively, were achieved. The study also proved a strong dependence of removal efficiencies on solar radiation. The results suggest the algae-bacteria system is suitable for treatment of similar wastewater in locations with available land and sufficient solar radiation and temperature during the whole year.

Key words | algae-bacteria floc, biomass production, phytoremediation

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INTRODUCTION

Conventional activated sludge wastewater treatment technologies have a large carbon footprint due to high energy consumption for aeration and high CO_2 emissions produced by heterotrophic bacteria during the conversion of organic carbon into CO_2 (approximately half of CO_2 produced corresponds to the production from heterotrophic bacteria, the rest of the carbon footprint is related to electricity generation) (Suwanteep *et al.* 2011). In conventional technologies, nitrogen is mainly wasted to the atmosphere during nitrification and denitrification. These reasons led scientists to develop new wastewater treatment technologies producing fewer emissions to the atmosphere based on microalgal treatment. The advantage is that microalgae use solar radiation as an energy source and CO_2 as a carbon source during photosynthesis. Compared to higher plants, their photosynthetic potential and growth is higher (e.g. more efficient use of solar radiation) (Fraunhofer 2010).

Symbiotic colonies of microalgae and bacteria for wastewater treatment were introduced in the scientific literature more than four decades ago (Humenik & Hanna 1971). In these systems, algae and bacteria exist in a classic symbiotic relationship. Bacteria metabolise organic waste for growth and energy, producing new bacterial biomass, releasing CO_2 and inorganic nutrients (Muñoz & Guieysse 2006). Algae then utilise CO_2 through photosynthesis, assimilating nutrients into algal biomass and release O_2 supporting the aerobic bacterial activity (Muñoz & Guieysse 2006).

Prior research on this topic is limited and no research on malt house or brewery wastewater treatment with gravity settling microalgae-bacteria flocs was found to date. Research has been conducted in the field of microalgae immobilised with bacteria. Some research focused on microalgae-bacteria flocs treatment but none of them on treating similar wastewater in real-life conditions. For example, Van Den Hende *et al.* (2011, 2014) performed research on the treatment of different wastewaters with algal-bacterial culture. In these studies, the potential of flue gas treatment and municipal and industrial wastewater treatment was investigated on a laboratory scale and in scaled-up raceway pond reactors with results showing a potential for use of a microalgae-bacteria consortium for a low cost treatment of wastewater and for flue gas treatment. Su *et al.* (2011, 2012) achieved similar results when treating municipal wastewater.

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This study is part of the international project ALBAPRO, whose goal is to develop a technology using symbiotic colonies of microalgae and bacteria for treatment of wastewater from paper industry, food industry and municipal sources on pilot scale.

The novelty of this study lies in the cultivation of a settleable algal-bacterial culture for the treatment of malt house wastewater and investigation of pollution removal efficiencies and biomass settleability.

METHODS

Microalgae cultivation

An autochthonic mix of microalgae and cyanobacteria was used for cultivation of biomass in the reserve tank. The aforesaid mix originated from a lagoon located in the close proximity of the industrial wastewater treatment plant of the malt house Bernard – Rajhrad, the Czech Republic. Throughout the experiment, the most abundant species were: *Pediastrum boryanum*, *Coelastrum astroideum*, *Monoraphidium contortum*, *Ulothrix* sp., *Gomphonema truncatum*, *Gomphonema parvulum*, *Navicula* sp. and *Navicula gregaria*. The mixed algae culture in the reserve tank was used as an inoculum for the reaction tank start-ups.

Material

Solution of 40% $FeCl_3$ of a technical quality was supplied by FIMA Brno, the Czech Republic. Aerobic activated sludge was sampled in the aeration tank at the wastewater treatment plant (WWTP) of the malt house Bernard – Rajhrad. The total suspended solids (TSS) of the mixed liquor was 6 g L^{-1}, sludge volume index (SVI) was 90 mL g^{-1} and sludge age was > 20 days.

Experimental set up

The pilot plant (see Figure 1 for its schematic) installed at the wastewater treatment plant of the malt house Bernard – Rajhrad consisted of two tanks, the reaction tank and the reserve tank, with a total volume of 1.0 m^3 (diameter of 1.6 m and height of 0.5 m) for the reaction tank and a total volume of 0.4 m^3 (diameter of 1.0 m and height of 0.5 m) for the reserve tank. Two low speed mixers (diameter of 1.0 m for the reaction tank and 0.6 m for the reserve tank) continuously agitated the cultures in both tanks. A fixed speed mixer (15 rpm) agitated the reserve tank and a variable speed mixer (0 to 23 rpm) agitated the reaction tank. Separation of biomass proceeded in the sedimentation tank with a total volume of 0.48 m^3 (diameter of 0.8 m and total height of 1.1 m). Two identical peristaltic pumps (WMC 401i-IND-F-100-230-100) controlled by a time relay were used. One pumped raw wastewater to the reaction tank and the second one pumped settled biomass from the bottom of the sedimentation tank back to the reaction tank. Raw wastewater was pumped continuously through the whole experiment from the equalisation tank of the WWTP malt house Bernard – Rajhrad. Dissolved oxygen (DO) concentration, pH, turbidity, electrical conductivity, redox-potential and water temperature in the reaction tank were measured *in situ* by a multi-parametric probe and logged every 30

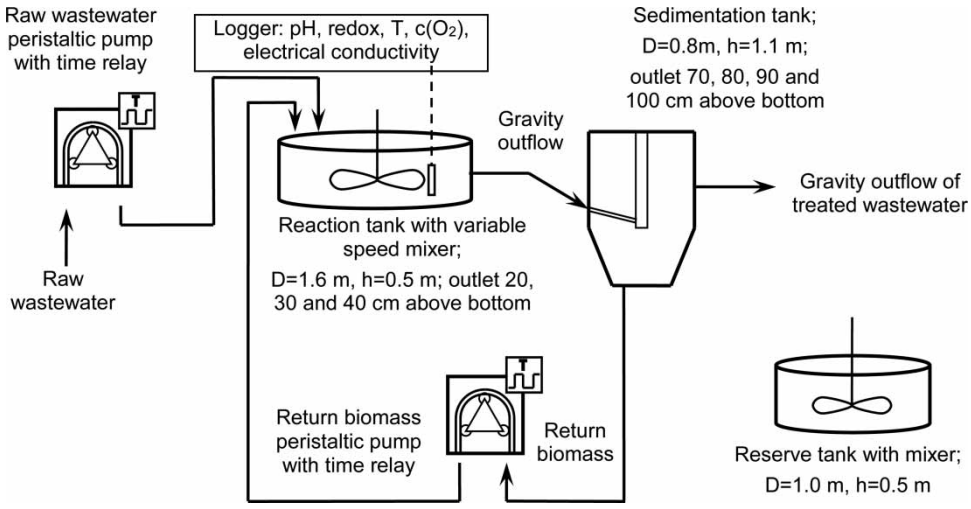

Figure 1 | Schematic of the pilot plant.

minutes (Aquaread AP-2000 probe with a dedicated logger). Each tank was illuminated only by natural sunlight; pH and water temperature were not controlled.

Sampling and analytical methods

Values of daily solar radiation were provided by the Institute of Geodesy of the Faculty of Civil Engineering, Brno University of Technology, the Czech Republic. Chemical analyses were performed by Brněnské vodárny a kanalizace, a.s. (Laboratory Brno – Modřice). Standardised methods used for the analyses are: SOP/M-06 (ČSN EN872) for TSS; SOP/M-08 (ČSN EN 1899-1; ČSN EN 1899 2) for biological oxygen demand (BOD_5); SOP/M-10 (ČSN EN ISO 15705), SOP/M-09 (TNV 75 7520; ČSN ISO 6060, Z1) for chemical oxygen demand (COD_{Cr}); SOP/M-19 (ČSN EN ISO 15681-2) for total phosphorus (P_{tot}) and SOP/M-21 (ČSN EN 12260) for total nitrogen (N_{tot}). The identification of microalgae and cyanobacteria was performed according to Komárek & Anagnostidis (2005) and Ettl & Gärtner (1995) using an Olympus CX 31 microscope.

The reactor liquor was sampled by collecting 1.0 L of mixed liquor and analysed for the total suspended solids. In the first and the second cycle, the treated wastewater was sampled by collecting 1.0 L of water from the surface of the sedimentation tank. In the remaining cycles, the treated water was sampled by collecting 1.0 L of supernatant from the reaction tank after 30 minutes without mixing (45 minutes at the end of the fourth cycle). Raw wastewater was sampled by pumping 1.0 L of wastewater from the equalisation tank. Sampling of treated wastewater was discontinued when excessive *Daphnia sp.* were present and/or biomass concentrations were lower than 400 mg L^{-1} TSS.

Settled volume (V_{30}) was determined in the reaction tank mixed liquor after 30 minutes of sedimentation in a 1000 mL graduated cylinder. The corresponding sludge volume index was calculated as $SVI = V_{30}/TSS$. Pollution removal efficiencies were calculated as $R = (c_{raw} - c_{treated})/c_{raw}$. The results are shown as $A \pm B$, where A is the mean value and B is the standard deviation.

Experimental operation

The experiment lasted for 116 days, the first 21 days served for cultivation of the inoculum only. The reserve tank was initially filled with water from the lagoon with the initial biomass concentration of 15 mg L^{-1} TSS. During the whole experiment, raw wastewater served as a source of nutrients for biomass in the reserve tank. The food to microorganism ratio (F:M) in the reserve tank was kept near 0.04 kg BOD_5 kg^{-1} TSS d^{-1}.

For the remaining 95 days, the experiment was conducted in four different cycles. Each cycle started with inoculation of the reaction tank with the inoculum from the reserve tank. The first cycle started 21 days after the beginning of the experiment by inoculating the reaction tank with biomass from the reserve tank and ended after 34 days due to a long-term power failure (with no mixing, settled biomass at the bottom of the reaction tank started to decompose anaerobically). The second cycle lasted 23 days and was terminated due to complex problems. The third cycle was successful in terms of biomass growth and pollution removal rates and was terminated on day 14 after harvesting the biomass for anaerobic digestion trials. The last, the fourth cycle, was terminated after 20 days due to the interruption of the malt house production.

The pilot plant was operated in a batch mode at the beginning of the first and the second cycle (16 days and 7 days, respectively) and then in a continuous mode. In the subsequent cycles, the pilot plant was operated in the sequencing batch reactor (SBR) mode due to the problems with the sedimentation tank encountered in the first and the second cycles. Water level and volume in all cycles were 0.27 ± 0.07 m and 0.55 ± 0.13 m^3, respectively.

Activated sludge was added to the reaction tank only in the first and the second cycles (on day 16 and day 21 of the first cycle and on day 20 of the second cycle). The weight ratio of added activated sludge to the biomass in the reaction tank was 1:5. To improve settleability, $FeCl_3$ was used as a substitution for activated sludge at a dose of 1.0 mL L^{-1} on day 25 of the first cycle; 0.2 mL L^{-1} on day 2 and 0.8 mL L^{-1} on day 10 of the second cycle; 0.6 mL L^{-1} on day 3 of the third cycle and 0.7 mL L^{-1} on day 4 of the fourth cycle.

RESULTS AND DISCUSSION

Operating conditions

Operating conditions during the whole experiment varied significantly over time and are shown in Figure 2 (biomass concentration as TSS and SVI) and in Figure 3 (F:M and daily solar radiation).

Concentrations of DO in the reaction tank were near saturation at the beginning of each cycle (from 6 to 15 mg O_2 L^{-1}). After coagulation with $FeCl_3$ the dissolved oxygen

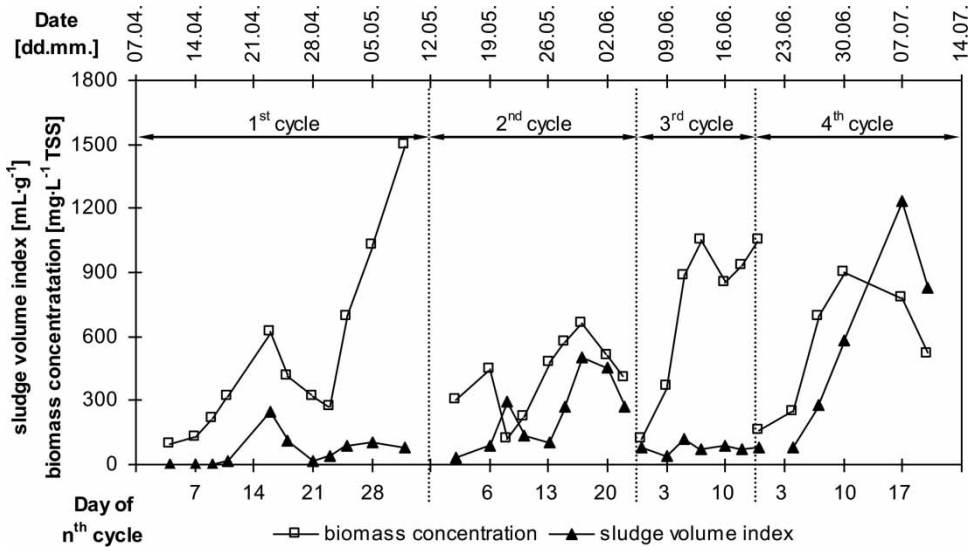

Figure 2 | Sludge volume index and biomass concentrations.

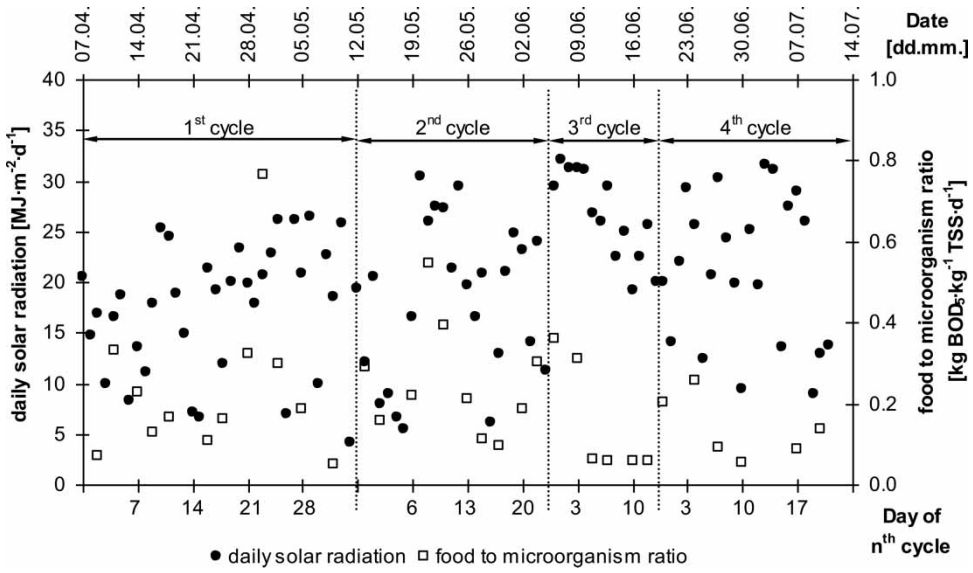

Figure 3 | Daily solar radiation and food to microorganism ratio.

concentrations decreased considerably as a consequence of newly formed flocs and higher biomass concentrations associated with the addition of iron, both resulting in higher mutual shading. Concentrations of dissolved oxygen after coagulation ranged from 0 to 2 mg O_2 L^{-1} at night and up to 8 mg O_2 L^{-1} during the day.

The impact of mixing and air diffusion across the water surface on aeration of the reaction tank was not examined. The reason was a very low speed of the mixer (the maximum speed of the mixer used for mixing of the reaction tank was 14 rpm) and the relation of the dissolved oxygen concentrations to solar radiation (see Figure 4). Solar radiation and the DO concentrations over a selected period of the third cycle are shown in Figure 4 and indicate that the role of solar radiation is crucial to the process. At the solar radiation intensity close to zero, the dissolved oxygen concentrations are below 0.2 mg O_2 l^{-1} and the role of mixing and air diffusion across the water surface and the impact on the reaction rank aeration is thus negligible.

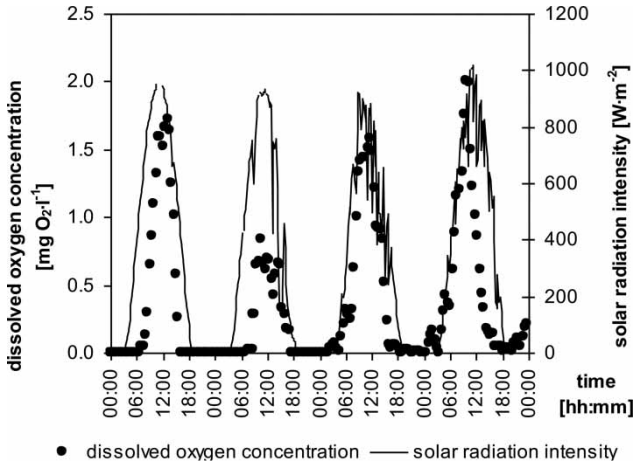

Figure 4 | Solar radiation and oxygen concentrations on days 3 to 5 of the third cycle.

The pH was not controlled and its values varied throughout the day as a result of the consumption of CO_2 by microalgae. Values of pH before coagulation were 8.9 ± 0.6. Coagulation with $FeCl_3$ resulted in a drop of pH to 7.9 ± 0.6. High pH at the beginning of each cycle (i.e. before coagulation) was caused mainly by alkaline microalgae inoculum (pH 9.0 ± 0.6).

Water temperature in the reaction tank was 14.3 ± 3.9 °C, 17.2 ± 4.3 °C, 22.4 ± 3.5 °C and 20.8 ± 2.7 °C in the first, second, third and fourth cycles, respectively.

Standard reduction–oxidation potential (E_H) development corresponded to the dissolved oxygen concentrations: non-zero concentrations of dissolved oxygen resulted in a positive E_H (up to 380 mV) and zero dissolved oxygen concentrations resulted in a low/negative E_H (down to −90 mV).

The F:M was controlled according to the dissolved oxygen concentrations and reduction–oxidation potential in a way that the dissolved oxygen concentrations were at least 1.0 mg O_2 L^{-1} or the standard reduction–oxidation potential was positive.

Hydraulic retention time in the reaction tank was 16.2 ± 10.9 days, 13.2 ± 4.2 days, 17.0 ± 3.9 days and 19.2 ± 5.3 days in the first, second, third and fourth cycles, respectively. The mean solids retention time during the experiment was 21 days.

Mixing power in the reaction tank was 19.4 ± 12.5 W m^{-3} during the experiment although 2 W m^{-3} is sufficient to prevent biomass from settling as found at the end of the experiment.

Raw wastewater in the equalisation tank contained: BOD_5 935 ± 170 mg O_2 L^{-1}, COD_{Cr} 1840 ± 320 mg O_2 L^{-1}, N_{tot} 43.9 ± 9.3 mg N L^{-1}, P_{tot} 15.2 ± 6.4 mg P L^{-1} and TSS 416 ± 94 mg L^{-1}.

Pollution removal with algal-bacterial culture

The achieved effluent concentrations of BOD_5, COD_{Cr}, N_{tot}, P_{tot} and TSS in all samples (see Figure 5(a)) were lower than the requirements given by the national legislation. Treated malt house wastewater discharged into surface water must meet these parameters: pH 6–8.5, COD_{Cr} 130 mg L^{-1} O_2, BOD_5 40 mg L^{-1} O_2, TSS 40 mg L^{-1}, N_{tot} 20 mg N L^{-1}, P_{tot} 5 mg P L^{-1} (Governmental Decree of the Czech Republic No. 23/2011 Coll).

The BOD_5 removal efficiencies were stable during the entire testing (99.3 ± 0.2%, see Figure 5(b)) and BOD_5 concentrations of treated wastewater were low (7.1 ± 2.0 mg L^{-1} O_2, see Figure 5(a)). BOD_5 removal efficiencies were independent of the water temperature, daily solar radiation and the TSS in the reaction tank.

COD_{Cr} removal efficiencies (95.9 ± 1.0%) showed a lower stability compared to BOD_5 removal efficiencies and

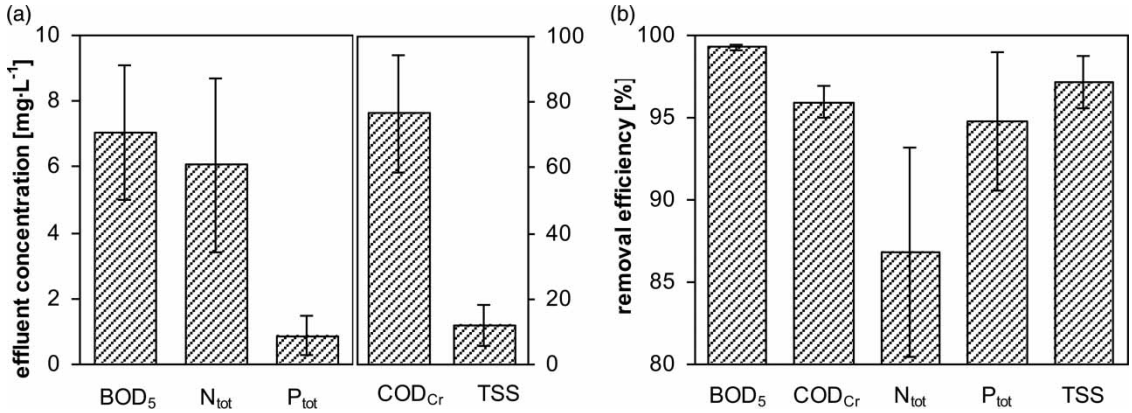

Figure 5 | Performance of microalgae-bacteria biomass: (a) treated wastewater concentrations; (b) removal efficiencies. Error bars represent standard deviation.

were moderately dependent on daily solar radiation and effluent TSS in the third cycle. The linear relationship is $R_{COD} = 0.32 \cdot E_5 + 87$ ($R^2 = 0.52$) and $R_{COD} = 1.41 \cdot TSS - 41$ ($R^2 = 0.56$), where R_{COD} stands for COD_{Cr} removal efficiency, TSS is effluent TSS and E_5 is the average daily solar radiation of 5 days prior to the sampling day.

N_{tot} removal efficiencies (86.8 ± 6.3%) were the least stable of all the monitored parameters (see Figure 5(b)). A strong dependence of N_{tot} removal efficiencies on solar radiation caused this instability. The linear relationship for the third cycle is: $R_N = 2.49 \cdot E_5 + 17$ ($R^2 = 0.98$), where R_N is the removal efficiency of N_{tot} and E_5 is the average daily solar radiation of 5 days prior to the sampling day.

P_{tot} removal efficiencies achieved in the third cycle were of 94.8 ± 4.2% and strongly depended on solar radiation: $R_P = 0.29 \cdot E_5 + 89$ ($R^2 = 0.75$), where R_P is the removal efficiency of P_{tot} and E_5 is the average daily solar radiation of 5 days prior to the sampling day.

The above correlations only concern the third cycle when all other operational parameters were most stable. Linear correlations between all measured removal efficiencies and each operational parameter were mostly weak.

The low effluent concentrations of TSS (12 ± 6.2 mg L^{-1}, see Figure 5(a)) were the result of biomass coagulation. Even though coagulation is considered as an expensive method for biomass harvesting (Hung *et al.* 2010; Milledge & Heaven 2013), this is not the case since the biomass was coagulated only at the beginning of the cycle (twice in the second cycle). The effluent TSS concentrations remained low during all cycles without the need to coagulate the biomass repeatedly.

Compared to other studies, the overall removal efficiencies achieved in this study were generally higher. For example, the removal efficiencies in Van Den Hende *et al.* (2014) were 28 ± 48%, 53 ± 56%, 31 ± 17%, and 64 ± 22% for the removal of COD_{Cr}, BOD_5, N_{tot} and P_{tot}, respectively, for the treatment of pikeperch culture wastewater. Municipal wastewater treatment with microalgae-bacteria flocs also resulted in lower removal efficiencies as shown by Van Den Hende *et al.* (2011): 56% for N_{tot} and 57% for P_{PO4} without flue gas sparging and up to 80% for N_{tot} and 29% for P_{PO4} with flue gas sparging.

Similarly to this study, Su *et al.* (2011) achieved COD_{Cr} removal around 98% and N_{tot} removal from 73% to 83% after 8 days of municipal wastewater treatment in batch reactors. Results achieved by Su *et al.* (2011) were accomplished with wastewater as a source of bacterial inoculum instead of activated sludge used by Van Den Hende *et al.* (2011, 2014).

For the above reasons, $FeCl_3$ was used as a substitute for activated sludge to improve settleability. Substitution of activated sludge by $FeCl_3$ had no or insignificant effect on the pollution removal efficiencies. Microscopic observations of biomass proved that the flocs were composed of microalgae and bacteria even without the activated sludge addition. In conclusion, the combination of activated sludge with microalgae appears to be ineffective not only in this study but also when considering lower efficiencies achieved in other studies (for example Van Den Hende *et al.* 2011, 2014).

With respect to the results achieved in the third cycle at the given conditions (F:M 0.06 kg BOD_5 kg^{-1} TSS d^{-1}; TSS_{RT} 950 mg L^{-1}; water level 0.3 m, specific mixing power 2 W m^{-3}) the required reactor volume (V), area (A) and electric energy consumption for mixing (E) for one population equivalent (one PE = 60 g BOD_5 PE^{-1} day^{-1}) are: $V = F{:}M \cdot TSS_{RT} \cdot BOD_{1PE}^{-1} = 0.95$ m^3 PE^{-1}; $A = V\,h^{-1} = 3.2$ m^2 PE^{-1} and $E = P_{mix} \cdot V \cdot 31.54 \cdot 10^6 = 60.0$ MJ PE^{-1} $year^{-1}$ (e.g. 16.7 kWh PE^{-1} $year^{-1}$). The maximal total electric energy consumption for treatment is estimated to be 18 kWh PE^{-1} $year^{-1}$ including energy required for pumping, control and measurement. Annual electric energy consumption of the conventional activated sludge wastewater treatment plant of the malt house Bernard – Rajhrad is 29 kWh PE^{-1} $year^{-1}$. The 40-year LCC cost is 61% compared to the conventional activated sludge process (the cost assessment is not covered by this paper).

Settleability of the algal-bacterial culture

The project proposed that an addition of activated sludge will enhance the aggregation of microalgae with flocs of activated sludge and will lead to the improvement of pollution removal and biomass settleability. However, the addition of activated sludge resulted in serious problems with biomass settleability. Except for the first cycle, $FeCl_3$ was added at the beginning of each cycle to improve the settleability of microalgae culture.

In the third cycle, only $FeCl_3$ was dosed and no activated sludge was added. This resulted in a considerable improvement of settleability. Treated wastewater TSS was 6.5 to 14 mg L^{-1} and sludge volume index 41 to 146 mL g^{-1} (see Figure 2).

In the fourth cycle, settleability was poorer compared to the third cycle; sludge volume index started at 80 mL g^{-1} on day 4 and increased up to 1230 mL g^{-1} on day 17. Even though the sludge volume index was high, the treated wastewater had no more than 30 mg L^{-1} of TSS (settled volume

was 800 mL L^{-1}). High sludge volume index was a result of domination of filamentous bacteria in the flocs.

Compared to other studies, the settleability of the microalgae-activated sludge bacteria biomass was considerably worse in this study. For example, Su *et al.* (2012) found that the biomass with a wide range of weight ratio of microalgae to sludge settled well (supernatant TSS dropped to less than 60 mg L^{-1} within an hour of settling). Similar results were obtained with a mixed culture of autochthonic microalgae with wastewater used as bacterial inoculum (Su *et al.* 2011) or with a culture composed of autochthonic microalgae and activated sludge (Van Den Hende *et al.* 2011). Settleability of coagulated biomass in this study was similar to the results of the studies described above. Coagulation performed in this study was used only to trigger flocculation and no additional coagulation was needed within one cycle.

CONCLUSION

The potential of mixed algae-bacteria biomass to treat food industry wastewater on a pilot scale was investigated in this study. The following conclusions were drawn from the data obtained:

- Microalgae-bacteria wastewater treatment technology performed well on malt house wastewater.
- Treatment efficiencies achieved were up to 97.6%, 99.6%, 92.7% and 98.9% for COD_{Cr}, BOD_5, N_{tot} and P_{tot}, respectively.
- The land requirement for the treatment described in this study is 3.2 m^2 PE^{-1} while the energy demand is approximately 18 kWh PE^{-1} $year^{-1}$.
- Further research aimed at more effective use of solar radiation and land is needed along with the research focused on increasing the culture stability in real-life conditions and on validating the long-term sustainability of coagulation.

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REFERENCES

Ettl, H. & Gärtner, G. 1995 *Syllabus der Boden-, Luft- und Flechtenalgen*. Gustav Fischer Verlag, Stuttgart, Germany, 721 pp.

Fraunhofer, I. G. B. 2010 Microalgae – a sustainable resource for valuable compounds and energy. http://www.igb.fraunhofer.de/content/dam/igb/en/documents/brochures/Microalgae__a_sustainable_resource_for_valuable_compounds_and_energy.pdf (accessed 27 February 2015).

Governmental Decree of the Czech Republic No. 23/2011 Coll.

Humenik, F. J. & Hanna, G. P. 1971 Algal-bacterial symbiosis for removal and conservation of wastewater nutrients. *Water Pollution Control Federation* **43** (4), 580–594.

Hung, Y., Amuda, O., Alade, A., Amoo, I., Tay, S. & Li, K. 2010 *Handbook of Environmental Engineering*, Vol. 11: Environmental Bioengineering. Humana Press, New York City.

Komárek, J. & Anagnostidis, K. 2005 Cyanoprokaryota, 2. Teil: Oscillatoriales. In: *Süsswasserflora von Mitteleuropa*, vol. 18/2 (B. Büdel, G. Gärtner, L. Krienitz & M. Schagerl, eds). Elsevier, München, 759 pp.

Milledge, J. J. & Heaven, S. 2013 A review of the harvesting of micro-algae for biofuel production. *Rev. in Environ. Sci. and Biotechnol.* **12** (2), 165–178.

Muñoz, R. & Guieysse, B. 2006 Algal–bacterial processes for the treatment of hazardous contaminants: A review. *Water Res.* **40** (15), 2799–2815.

Su, Y., Mennerich, A. & Urban, B. 2011 Municipal wastewater treatment and biomass accumulation with a wastewater-born and settleable algal-bacterial culture. *Water Res.* **45** (11), 3351–3358.

Su, Y., Mennerich, A. & Urban, B. 2012 Synergistic cooperation between wastewater-born algae and activated sludge for wastewater treatment: Influence of algae and sludge inoculation ratios. *Bioresour. Technol.* **105**, 67–73.

Suwanteep, K., Chavalparit, O. & Ongwandee, M. 2011 Evaluation of Carbon footprint of activated sludge wastewater treatment system. In: *Fostering Economic Growth through Low Carbon Initiatives in Thailand*, C. Tingsabadh (ed.), 1st edn, Chula Global Network, Chulalongkorn University, Bangkok, Thailand, pp. 219–232.

Van Den Hende, S., Vervaeren, H., Desmet, S. & Boon, N. 2011 Bioflocculation of microalgae and bacteria combined with flue gas to improve sewage treatment. *New Biotechnol.* **29** (1), 23–31.

Van Den Hende, S., Carré, E., Cocaud, E., Beelen, V., Boon, N. & Vervaeren, H. 2014 Treatment of industrial wastewaters by microalgal bacterial flocs in sequencing batch reactors. *Bioresour. Technol.* **161**, 245–254.

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