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Seasonally loaded waste stabilisation ponds: a novel application for intermittent discharge

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Abstract

The research examined the use of a single facultative pond for treatment of an intermittent discharge from a UK campsite. The system was monitored over an 11-month period to determine the optimum time for discharge in terms of quality standards. The results showed that based on organic strength, discharge was possible in winter between November-March but February was the optimum to meet nutrient and suspended solids requirements. The pond showed rapid acclimatisation to the influent wastewater, with Biochemical Oxygen Demand removal rates during the filling period of around 50 kg ha⁻¹ day⁻¹ and removal efficiencies of ~95% after maturation. The system proved simple to operate. A major design factor is the requirement for storage of net incoming precipitation, which may provide dilution of residual pollutants but requires additional system capacity.

Keywords

Waste stabilisation ponds, intermittent discharge, seasonal loading, oxygen production

INTRODUCTION

Lockerley Water Farm is a campsite in Hampshire, UK (51.04 °N, 1.57 °W) accommodating around 100 campers in the summer period from July to early September. Wastewater from the site is collected in two underground storage tanks each of ~30 m³ capacity. These are regularly emptied and the contents removed from site for treatment and disposal, representing a significant cost to the charity that operates the campsite. As the site is not open all year round, one option is to adopt an intermittent discharge waste stabilisation pond (WSP) system, a typical approach in colder climates (Prince *et al.*, 1995). In this case wastewater is treated during the summer then stored until it reaches high quality in autumn or spring, when it is discharged over a relatively short period. Alternatively the system could discharge continuously during the summer once the effluent quality is adequate. The receiving watercourse for the site is the River Dun, which is a trout fishery and would be subject to stringent discharge conditions; another option would be land application. While this site has its own special features, however, the situation of an intermittent or increased seasonal load is not unusual, for example in small rural or seaside places which have an influx of summer visitors. A low-cost system and operating protocol to deal with this type of intermittent loading may therefore have widespread applications.

In the current research, wastewater was allowed to accumulate during the summer in a single pilot-scale facultative pond and stored through autumn and winter into spring, to ascertain the most suitable times for discharge. This single-pond batch-sequencing mode allows facultative treatment

during the filling period, followed by maturation to improve quality in terms of nutrient removal and pathogen reduction; and may offer a lower-cost solution compared to the traditional approach of separate ponds for these functions. One potential disadvantage is the need for extra capacity to store any net seasonal precipitation during the non-discharge period, but this may also provide further dilution of residual pollutants if the system is to discharge to a receiving water, or retention of additional water to meet irrigation requirements in spring.

MATERIALS AND METHODS

A pilot-scale pond was constructed with a maximum surface area of $\sim 41 \text{ m}^2$, base area 9 m^2 , maximum water depth of 1.3 m and capacity $\sim 30 \text{ m}^3$. During the camping season from 26 July to 10 September 2006, 160-litre batches of wastewater were pumped twice daily from the main storage tank into a 250-litre feed tank. Each batch was held for 15 minutes to allow time for sampling, and then gravity-fed into the pond 0.5 m below the surface to minimise sediment disturbance. This feed of 320 l day^{-1} represented approximately 5% of the campsite's wastewater production. The pond was initially inoculated with 33 litres of mixed micro-algal culture at a chlorophyll-*a* concentration of $\sim 27 \mu\text{g l}^{-1}$ to ensure rapid start-up. No effluent was discharged during the camping season, at the end of which the pond was left with freeboard for storage of net incoming precipitation. Exceptionally high rainfall in autumn and winter 2006, however, meant that between 5 December - 9 March approximately 10,470 litres had to be pumped out of the pond and back into the storage tanks to prevent overflow.

Sampling, monitoring and analysis. The pond was instrumented with two dissolved oxygen (DO) probes (Dryden Aqua Ltd, UK), one $\sim 30 \text{ cm}$ above the bottom and one near the surface. Air temperature was measured by a K-type thermocouple and irradiance by a RC/0308 photovoltaic cell (PV Systems, UK). Pond depth was recorded using a piezometer tube and a digital pressure sensor, with additional manual readings from a calibrated measuring rod. A D500 datalogger (DataTaker, UK) recorded these data, averaged over 10-minute intervals. Other climatic data were obtained from the University of Southampton's weather station, with additional air temperature and rainfall measurements recorded on site using Thermochron iButtons (Maxim, California) and a manually-read rain gauge. On-site water consumption was recorded from the consumer meter provided by Southern Water Plc. Influent wastewater samples were taken using a time-interval controlled peristaltic pump, with the composite stored in a refrigerated container.

Influent wastewater and pond water were analysed twice weekly for Chemical Oxygen Demand (COD), 5-day Biochemical Oxygen Demand (BOD_5), suspended solids (SS), ammonia, nitrate and orthophosphate using Standard Methods 5220 C, 5210 B, 2540 D, 4500-N B and 4500-P E respectively (APHA, 2005). pH was determined using a Jenway 3010 meter (Bibby Scientific Ltd, UK) with a combination glass electrode calibrated in buffers at pH 4, 7 and 9.2 (Fisher Scientific, UK). Chlorophyll-*a* was measured by method 10200 H (APHA, 2005), modified in that MgCO_3 was added to the filter rather than the acetone and the extract was not re-filtered after centrifuging. Gross and net photosynthesis were determined for samples of pond water by calculating the rates of change of dissolved oxygen concentrations when incubated under light and dark conditions (Whalley *et al.*, 2011). Testing for Faecal Coliforms was carried out using a Most Probable Number technique according to SCA (2003).

Loading and removal rates during the filling period were calculated on a mass balance basis taking into account the volume of the pond contents and the surface area at the time. Apparent removal efficiencies after wastewater inflow had ended were calculated from average influent concentration in the filling period (using unfiltered values for BOD_5 and COD) and the monthly average pond

water concentrations (using filtered BOD₅ and COD), without taking into account the effects of dilution by rainwater. To compensate for this and for the removal of pond water in December - March, actual efficiencies based on biologically-mediated removal only were calculated by mass balance. The volume of the pond based on its measured depth during the filling period showed good agreement with volumes calculated from inflow, precipitation and estimated evaporation ($R^2 = 0.99$, slope = 1.03), supporting the validity of this approach.

RESULTS AND DISCUSSION

Influent. The average characteristics of the influent wastewater were (mg l⁻¹): BOD₅ 180, COD 350, SS 162, NH₃-N 55, NO₃-N 0.3, P 24, with a pH of 7.66. The influent was thus typical of a low-to-medium strength domestic wastewater apart from ammonia and phosphorus concentrations which were in the high category (Metcalf and Eddy, 2003).

Pond water characteristics

BOD₅ and COD. Filtered and unfiltered BOD₅ and COD concentrations for the pond water throughout the monitoring period are shown in Figure 1. During the filling period unfiltered COD increased steadily from 300 to 600 mg l⁻¹, while filtered COD remained stable at around 120 mg l⁻¹ apart from a small peak in early September. BOD₅ values were more erratic, but filtered BOD₅ showed a reduction to around 40 mg l⁻¹. These filtered values are similar to the ranges of 50-80 mg BOD₅ l⁻¹ and 120-200 mg COD l⁻¹ given by von Sperling and Oliveira (2009) for facultative pond effluents. Values for all these parameters did not decrease immediately after filling ended, but reduced to low levels during the winter before increasing again in spring. The lowest monthly average unfiltered COD concentration of 82 mg l⁻¹ occurred in February, while unfiltered BOD₅ averaged 17 and 14 mg l⁻¹ in February and March respectively, making this the best period for discharge with respect to organic strength.

Chlorophyll-*a*. Chlorophyll-*a* concentrations rose steadily from the start of filling until late August then increased sharply, reaching 10 mg l⁻¹ in October. This continuing algal bloom was also evident in the difference between filtered and unfiltered COD and BOD₅ (Figure 1). Concentrations were generally higher than typical values of 1-1.5 mg Chl-*a* l⁻¹ seen in effluent from UK facultative ponds during summer (Mara *et al.*, 1992). This may have been due to a number of factors, including high light intensities, optimum water temperatures (Figure 2), and an influent high in ammonia and phosphate. High concentrations could also indicate uneven spatial distribution of algae within the water column and a sampling technique biased to surface water. Similar concentrations have been reported elsewhere, however, with values above 10 mg Chl-*a* l⁻¹ found in facultative pond effluents in Brazil (Mara, 2006). A sharp dip in chlorophyll concentrations at Lockerley in early September may have been due to a period of low light intensity, with the subsequent recovery a result of sustained nutrient concentrations in the water column. It was only with the onset of cold weather in November and the decrease in pond temperature that Chlorophyll-*a* reduced. Concentrations remained low until late February when a small bloom occurred which peaked at 1.3 mg Chl-*a* l⁻¹ on 20 March (Figure 1), then gradually declined to 0.15 mg Chl-*a* l⁻¹ by the end of June.

Filtered BOD₅ concentrations only reached low values after chlorophyll-*a* concentrations had decreased. The reasons for this are unknown, but it is possible that sedimentation of the algae caused a reduction in soluble or colloidal material.

Suspended solids. SS concentrations are shown in Figure 1. Linear regression analysis showed fairly strong relationships between SS and unfiltered COD (unfiltered COD = 1.32 SS + 96.2, mg l⁻¹; $R^2 = 0.91$, $p < 0.001$) and SS and unfiltered BOD₅ (unfiltered BOD₅ = 0.28 SS + 10.38, mg l⁻¹; R^2

= 0.72, $p < 0.001$). These agreed well with values quoted by von Sperling and Chernicharo (2005), who suggested that 1 mg l^{-1} of SS in facultative pond effluent corresponds to a BOD_5 of $0.3\text{--}0.4 \text{ mg l}^{-1}$ and COD of $1.0\text{--}1.5 \text{ mg l}^{-1}$. Outside the filling period there was a strong relationship between SS and Chlorophyll-*a* ($\text{SS} = 38.23 \text{ Chl-}a + 25.08 \text{ in mg l}^{-1}$; $R^2 = 0.94$; $p < 0.001$), indicating that SS concentrations above 25 mg l^{-1} during this time probably consist of algal solids.

Figure 1. Time-series values for key monitoring parameters during the study period.

Ammonia and phosphorus. Ammonia fell from 33 mg N l^{-1} to $\sim 16 \text{ mg N l}^{-1}$ in four days at the start of the filling period, and showed a continued downward trend to $< 1 \text{ mg N l}^{-1}$ at the end of October (Figure 1). Over the winter period concentrations began to rise, probably as a result of ammonia release into the water column from decaying sedimented biomass; then fell again from 7.8 mg N l^{-1} to $< 0.2 \text{ mg N l}^{-1}$ by the beginning of April as chlorophyll-*a* concentrations recovered. Nitrate concentrations remained low at $0.1\text{--}0.7 \text{ mg N l}^{-1}$.

Orthophosphate declined less quickly than ammonia, perhaps because there are fewer removal mechanisms: the main routes in WSP are via algal and bacterial uptake, and through precipitation under high pH conditions. pH values rose above 9.0 from early September until late October, but there was no significant relationship between orthophosphate concentration and pH or surface water temperature. There was a strong relationship between chlorophyll-*a* and orthophosphate

concentrations ($R^2 = 0.75$; $p < 0.001$), with phosphate generally 'leading' in time, suggesting this was a controlling factor for algal growth and that algal uptake was the primary removal mechanism. High phosphate concentrations immediately before the algal bloom in August also supported the high chlorophyll-*a* values observed. The decline in chlorophyll-*a* between May and June correlated with a decrease in orthophosphate ($R^2 = 0.96$; $p < 0.001$) and also, though less strongly, in ammonia ($R^2 = 0.66$; $p = 0.05$), suggesting the pond had become nutrient-limited following the bloom. Both ammonia and orthophosphate concentrations were $< 2 \text{ mg l}^{-1}$ for a period of 20 days in November, making this the best time to discharge with respect to nutrients. Average concentrations of 4 mg N l^{-1} and 2 mg P l^{-1} observed before the bloom in late February suggest this could also be a good time for discharge.

Dissolved oxygen and oxygen production potential. During the filling period average DO concentrations in the pond were low (Figure 2), reflecting the oxygen demand of the influent wastewater. Maximum DO concentrations started to rise as soon as filling ceased, indicating peak algal oxygen production in excess of community respiration. Average values remained low in this period due to the high night-time respiration demand of the whole community, but rose sharply in early November when the COD and BOD₅ concentrations were reduced to low levels. Average DO values then declined until late February, closely paralleling the changing chlorophyll-*a* concentration (Figure 1).

Figure 2. Dissolved oxygen concentrations and oxygen production potential throughout the study period and dissolved oxygen concentrations with light intensity in February and May

From late February into March average DO rose sharply, reaching over 200% saturation. The algal bloom that caused this did not appear to be produced by an increase in nutrient availability (Figure 1) or temperature (Figure 2), and was therefore probably due to increasing light levels. Welch (1992) noted that spring blooms usually occur when light intensity reaches a level where gross production exceeds respiration. From February until April average DO concentrations remained above saturation (Figure 2), presumably due to low levels of microbiological activity associated with the temperature, which remained around 10 °C, and to low BOD₅ and COD. Average DO concentrations started to decrease from mid-March, however, and maximum values fell from late April in parallel with decreasing chlorophyll-*a* concentrations. By May, although peak light intensities were much higher than in February, this was not sufficient to maintain DO saturation, with overnight concentrations falling to 2 mg l⁻¹ (Figure 2). From the viewpoint of DO the preferred discharge time would be in February, as community dark respiration was minimal in this period and would not exert an excessive oxygen demand on the receiving water.

In vitro testing showed that gross and net oxygen production reflected chlorophyll-*a* concentrations, with an increase during the filling period which stabilised at a high level until algal sedimentation in early November (Figure 2). The exception was a brief period in September where chlorophyll-*a* concentration, DO and gross and net oxygen production fell, suggesting that the algal population was adversely affected by a period of low light intensities. The highest recorded values for gross and net production were 16.6 and 13 mg O₂ l⁻¹ h⁻¹ respectively, both in mid-October. Average values for August-October were 10.52 mg O₂ l⁻¹ h⁻¹ for gross and 8.22 mg O₂ l⁻¹ h⁻¹ for net production, falling to zero in mid-winter. As expected, the average respiration rate was also highest during August-October at 2.30 mg O₂ l⁻¹ h⁻¹, reflecting the high level of bacterial activity associated with biodegradation of the wastewater. Respiration rates were very low from December to February due to low temperatures and the absence of organic loading; in spring both oxygen production and respiration increased with the rise in chlorophyll-*a* concentrations.

Microbiological quality. Samples were tested for faecal coliforms (FC) from February to April. No FC were found, suggesting that the long detention time and the absence of influent over the winter period were effective in reducing numbers, and making this a suitable period for discharge.

Overall performance

As the pond filled during the summer period the surface area and depth increased, leading to changes in surface and volumetric loading rates. The maximum loading should occur at the start of filling, and was calculated as 125 kg ha⁻¹ day⁻¹ and 44 g m⁻³ day⁻¹ for COD, and 64 kg ha⁻¹ day⁻¹ and 22.5 g m⁻³ day⁻¹ for BOD₅, based on average influent concentrations. These values should then decrease as the pond fills. In practice, however, fluctuations in the influent concentration meant that the actual maximum surface loading rates were 118 kg COD ha⁻¹ day⁻¹ and 82 kg BOD₅ ha⁻¹ day⁻¹ during mid-August. These loading rates were higher than the 80 kg BOD₅ ha⁻¹ day⁻¹ recommended for year-round use in the UK (Abis and Mara, 2005), but considerably below typical design values of 199-292 kg BOD₅ ha⁻¹ day⁻¹ based on ambient temperatures of 17-22 °C for the filling period only (Mara, 2005).

The pond responded rapidly to the relatively high influent wastewater load and was performing well within the first week of operation, indicating that a long acclimatisation period is not needed. This may have been due in part to the shallow depth during this period, which gave a high surface-area-to-volume ratio and good conditions for light exposure throughout the culture. This is supported by the BOD₅ removal rates, which increased in the first week of feeding then fluctuated around ~60 kg ha⁻¹ day⁻¹. COD removal rates were more stable and averaged ~85 kg ha⁻¹ day⁻¹ in the first four

weeks. Both BOD₅ and COD removal rates dipped sharply in early September, corresponding to the brief decrease in chlorophyll-*a* concentration and other parameters noted above. Despite the very high initial volumetric loading and the constantly changing conditions, BOD₅ and COD removal rates during the filling period were comparable to those achieved in typical facultative ponds and no problems were detected with odours or flies. The apparent BOD₅ removal efficiency during the filling period was 76%, but during the maturation period in October this increased to 96% and remained at this level until the end of the study. After taking into account both rainfall dilution and removal of pond water between December-March, the biologically-mediated removal for BOD₅ in the period from November to the end of the study was around 88%; removal rates for ammonia and phosphorus were above 95% from February onwards.

Table 1 summarises the most suitable times for discharge based on pond water quality in terms of BOD, SS and nutrient concentrations, according to the standards of the Urban Wastewater Treatment directive (UWWTD, 91/271/EEC), the Royal Commission on Sewage Disposal (1912) and the World Health Organisation guidelines (WHO, 2006).

Table 1. Possible periods for discharge to land or river according to different standards

Discharge	Parameter	Source	Standard (mg l ⁻¹)	Period
To river	COD (f)	UWWTD	125	Nov-May
	BOD (f)	UWWTD	25	Nov-May
	BOD (f)	Royal Commission	20	Feb-Mar
	Chlorophyll-a	n/a	n/a	Jan-May
	SS	UWWTD	150	Nov-May
	SS	Royal Commission	30	Jan-Feb
	Ammonia	UWWTD	2	Nov; April-May
	Phosphorus	UWWTD	1	Nov; April-May
To land	BOD (uf)	WHO	< 400	Aug-May
	Chlorophyll-a	n/a	n/a	Jan-Feb; April-May
	SS	WHO	< 100	Dec-May
	Ammonia	WHO	< 30	Aug-May
	Phosphorus	WHO	< 20	Aug-May

During the filling period filtered BOD₅ concentrations were higher than the UWWTD standard, making the treated wastewater unsuitable for discharge even if the SS concentration could be reduced from its average value of 300 mg l⁻¹. By late November filtered BOD₅ concentrations were ≤ 25 mg l⁻¹, filtered COD ≤ 125 mg l⁻¹ and SS ≤ 125 mg l⁻¹. With respect to these parameters, the pond water would meet UWWTD standards at any time from this point through to spring. To meet UWWTD nutrient standards, however, discharge would have to occur in either November or February. It was only in February and March that BOD₅ concentrations fell below 20 mg l⁻¹, as required by the more stringent Royal Commission standard which is commonly applied to discharges to river at a 1:8 dilution. In this case a February discharge would be preferable in order to avoid the risk of raised SS concentrations due to an early spring bloom. Based on all these results, a February discharge appears to be the best option as values for all parameters apart from ammonia were at their lowest at this time, while ammonia concentrations were only slightly higher than their minimum in November.

According to the WHO guidelines, BOD₅ and nutrient levels were low enough to allow discharge to land throughout the monitoring period. Faecal coliform concentrations were not monitored until spring, but none were found from January onwards, making this a possible discharge period, although with limited uses for irrigation water.

The system trialled provides treatment of the wastewater and offers a simple design with minimum requirements for maintenance and operator involvement, with discharge being required only once a

year over a relatively short time period. The mode of operation affects system design and performance, however, as the need to avoid extremely high volumetric loading rates at the start of the filling period leads to low surface loading rates later on and thus increases the required pond area. In the current design there is also a requirement for spare capacity to accommodate net precipitation entering the system between the start of filling and the time of discharge. In an average year net precipitation between October-March at the Lockerley site is ~350 mm, representing an increase of 25% in the required depth. Discharging the pond in November would reduce the need for storage capacity, but the quality would be too low for river discharge without further treatment, and in the case of land application the water is of little benefit locally at this time.

CONCLUSIONS

Pond performance during the filling period was good despite the high initial volumetric loading and the rapidly changing conditions. BOD₅ removal efficiencies reached 95% after a short maturation period following the end of wastewater addition. The main improvements in water quality occurred rapidly and corresponded to an autumn decrease in pond water temperature and a reduction in chlorophyll-*a* concentrations, indicating algal sedimentation. The optimum discharge period was in February when concentrations of all parameters (BOD, COD, SS, ammonia and phosphorus and faecal coliforms) reached very low levels. The requirement to store incoming net precipitation is a major factor in both design and performance.

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