Wat. Res. Vol. 21, No. 5, pp. 591–599, 1987
Printed in Great Britain.

0043-1354/87 $3.00 + 0.00

NUTRIENT REMOVAL FROM EFFLUENTS BY AN ARTIFICIAL WETLAND: INFLUENCE OF RHIZOSPHERE AERATION AND PREFERENTIAL FLOW STUDIED USING BROMIDE AND DYE TRACERS

KATHLEEN H. BOWMER
CSIRO Institute of Biological Resources, Centre for Irrigation and Freshwater Research, Griffith, NSW 2680, Australia

(*Received March 1986*)

Abstract—Sewerage and effluents from rural industry can be treated by percolation through the root zones of emergent macrophytes growing in a gravel substratum. The hydrology of these systems is complex, being driven by both gravity and transpiration, and so measurements of nutrient transformations within the systems are complicated by incomplete mixing. Pulse addition of dye and bromide tracers concurrently with nutrients, has been used in one such experimental artificial wetland to investigate the rates and processes of nutrient removal. The tracer was used for comparison to compensate for incomplete mixing and concentration caused by evapotranspiration. Nitrogen removal efficiency is dependent on sequential mineralization of organic nitrogen to ammonium-nitrogen, followed by nitrification of the ammonium to nitrate or nitrite and denitrification of nitrate or nitrite to gaseous nitrogen products. The effluent from a rendering plant was dominated by organic and ammonium-nitrogen, and efficiency of nitrogen removal was probably impaired by inadequate rates of mineralization and nitrification. Aeration is required for the latter process. Apparently the macrophytes were not introducing sufficient oxygen into the effluent for nitrification to be complete. This may reflect an inadequate outward radial diffusion of oxygen into the rhizosphere, or the effects of channelling of the effluent in preferential flow paths around the aerating root masses, requiring changes in system design.

Key words—emergent aquatic macrophyte, filters (biological), waste waters, *Typa orientalis*, *Schoenoplectus validus*, *Eleocharis sphacelata*, Erochrome Acid Red, quantitative dye tracer, bromide tracer, rhizosphere aeration

INTRODUCTION

The use of low cost, low maintenance artificial wetland systems for the biological treatment of effluents was reviewed by Mitchell (1978). In the systems described by Kickuth (1975), Seidal (1976), Pope (1981), Finlayson and Mitchell (1982), Finlayson and Chick (1983) and Gersberg *et al.* (1983, 1984) nutrients are removed with varying degrees of success from effluents during percolation through the root zones of emergent macrophytes in a gravel substratum. The performance efficiency of these trenches can be determined by comparing inflow and outflow nutrient loads. However, information on processes of nutrient transformation within the wetlands is required to optimize the operating parameters, including the physical and biological design of the systems. This aspect has been neglected until recently, probably owing to difficulties in measurement of nutrient transformations within the system associated with incomplete mixing and complicated hydrology.

In the experiments reported here, information on the hydrology of the wetland system, and on the rates of nutrient removal from the effluent, was obtained by monitoring the behaviour of a pulse of stable tracers, together with nutrient ions, during transport along the trench and into the outflow.

The effect of chemical decay on the pulse form of the concentration-time curve for any pollutant addition in a flowing system was discussed by Hartley and Graham-Bryce (1980). For concentration-dependent decay specifically, the concentration of an unstable compound in any part of the system after time t is $\exp(-kt)$ times the concentration of a stable molecule, where k is the first order rate constant of dissipation or reaction. In the present study the pulse concentration-time curves of nitrate and phosphate were compared with those of two stable materials, a fluorescent dye tracer, and bromide ion. Background concentrations were obtained before arrival of the pulse at a sampling station within the wetlands. Concentrations of tracer and nutrient observed during the passage of the pulse were corrected for background concentrations then expressed as a ratio of the concentration added during the pulse at the inflow. The nutrient ratios would be coincident with the tracer ratios if both were stable in the system, while the extent of divergence with time reflects the rate of nutrient removal from the effluent. The use of a stable tracer for comparison thereby compensates

for incomplete mixing and concentration caused by evapotranspiration.

METHODS

Wetland system

The experimental wetlands at Griffith, New South Wales, Australia, were level trenches lined with an impermeable membrane of butyl polymer 1 mm thick and filled with gravel of dia 2–3 cm. Further details of general design are given by Finlayson and Chick (1983). The trenches were 50 m long, trapezoidal in section, about 2 m wide at the top and 0.5 m deep. The effluent from a rendering plant was stored in an open pond, then introduced at a constant rate of flow onto the surface of the gravel through four evenly-spaced tubes at one end of the system. The outflow was pumped from the opposite end of the system. The height of the effluent was adjusted to ensure no free water at the gravel surface. *Eleocharis sphacelata* R.Br.; *Typha orientalis* Presl; and *Schoenoplectus validus* Vahl A. Love and D. Love were planted in to the upper, middle and lower thirds of the wetland respectively in the winter of 1982, and by the summer of 1982–3 were well established. In previous work at a poultry abattoir (Finlayson and Chick, 1983), *S. validus* was able to oxygenate an anaerobic effluent more effectively than species of *Typha*, so *S. validus* was planted in the downstream portion with the aim of oxygenating the effluent before discharge. Later, in winter 1984, the planting pattern was revised to give a uniform random distribution of the three species over the whole wetland. A control trench with gravel but not plants was run in parallel with the planted system.

Initially, permeable pipes were inserted mid-stream as vertical sampling ports at sampling stations 17 and 34 m from the inflow. In June 1983 the pipes were replaced with six small flexible tubes located at sampling stations, 1, 17, 34 and 50 m downstream from the inflow. At each station three tubes were placed in 0.1 m from the surface, two at each side, and one mid-stream, and three more were located similarly but 0.4 m deep (0.1 m from the bottom of the gravel).

Pulse addition of dye and tracers

Three experiments were conducted;

Experiment 1. Stability of dye tracers. The stability of Eriochrome Acid Red dye tracer (EAR; Ciba-Geigy Ltd, Sydney, Australia) was investigated in a preliminary experiment conducted in mid-summer (December 1982). A concentrated solution of dye was prepared and added to the effluent from a constant head device over 4 h at a constant rate of $1.11 \times 10^{-6}\,m^3\,s^{-1}$ while the effluent continued to flow into the wetland at $130 \times 10^{-6}\,m^3\,s^{-1}$. Therefore the dilution of the effluent by the dye solution was $< 1\%$. The theoretical average retention time in the wetland, determined from the pore volume of the unplanted system, and assuming no losses from evapotranspiration, was about 48 h. (The pore volume was determined immediately after construction, and was the volume of effluent required to fill the trenches and replace the air spaces within the gravel substratum.) Dye concentrations in the trench outflow were monitored during the passage of the pulse to give a complete concentration-time curve.

Experiment 2. Comparison of tracers and nutrient removal efficiency. A pulse of nutrients and stable dye and bromide tracers was added concurrently for 4 h, under experimental conditions similar to those described in experiment 1 (Table 1). Concentrations of nutrient, dye and bromide observed during the passage of the pulse were corrected for background concentrations (b) and expressed as a ratio of the concentration added (p) during the pulse at the inflow. This corrected or proportion ratio (r) is given by equation (1). Background concentrations were obtained outside the time of passage of the tracer pulse, over about 12 h.

$$r = (c - b)/p. \quad (1)$$

This experiment was conducted in late summer (February 1983). Surface and depth samples were collected at the vertical sampling ports, 17 and 34 m downstream, and also at the outflow. Effluent composition at the inflow to the wetland and the concentration during pulse addition were also monitored (Table 1).

Experiment 3. Nutrient removal efficiency. Further experiments were conducted in late summer (March 1985), but using a shorter 1 h pulse and with a longer average theoretical retention time of 72 h (based on pore volume and inflow rate, and assuming no evapotranspiration). The actual retention time is much longer in severe conditions because, as will be discussed later, evapotranspiration can consume the inflow during the day. Details of dye and nutrient addition are given in Table 1. Two injections were made on separate occasions. Firstly, at the 34 m station, with collection at the outflow. Secondly, at the inflow with collection at the 17 m station. Background concentrations were obtained over about 5 days, and avoiding the elevated concentrations during the passage of the pulse.

Longer-term overall system performance

The water quality and nutrient load of inflowing and outflowing effluent were measured during winter 1984 and summer 1985. Effluent samples were collected from the inflow and outflow every 2 days over a period of about 3–4 weeks (Breen, 1987).

Analytical methods

Eriochrome Acid Red was determined in the field using a portable fluorimeter (Turner Designs) or in the laboratory using an autoanalyser system and an Aminco Flurocolorimeter fitted with a flow-through cell. Sulphate-sulphur, chloride, bromide, nitrate-nitrogen and phosphate-phosphorus were measured in filtered samples by suppressed ion chromatography using standard methods (Dionex Corp.,

Table 1. Water quality at the inflow of the wetland before (background) and after pulse addition (experiments 2 and 3)

Expt No.	Nutrient ion or EAR† dye	Concentration in inflow ($g\,m^{-3}$)	Pulse addition ($g\,m^{-3}$)	Total concn in pulse ($g\,m^{-3}$)
2	EAR	0	0.0475	0.0475
	Bromide	0	5.0	5.0
	Nitrate-N	<0.08	2.10	2.1
	Phosphate-P	10.9 ± 0.7 (4)*	17.23	28.1
	Chloride	93 ± 2 (5)*	0	93
	Sulphate-S	9.37 ± 0.2 (4)*	7.7	17.1
3	EAR	0.003	4.437	4.440
	Nitrate-N	0.12	46.4	46.5

*Mean ± SE (followed in brackets by numbers of samples).
†Eriochrome Acid Red.

Sunnyvale, Calif.; Tabatabai and Dick, 1983). Kjeldahl-nitrogen, and nitrate-nitrogen were determined by standard Technicon Autoanalyser methods as previously described (Finlayson and Chick, 1983). Ammonium-nitrogen was determined by the Berthelot colorimetric reaction (Technicon Industrial Method No. 98-70 W/A). Storage checks showed that no loss of nutrients occurred when unfiltered samples were stored frozen (−15°C) for 4 days, quickly thawed and filtered (0.45 μm Millipore type HA), or filtered in the field before storage at 4°C. Also, pH remained constant within the range 7.1–7.7.

In experiment 3 nitrate-nitrogen was measured by ion chromatography (x) and colorimetric Autoanalyser (y) methods, with good correlation; $r^2 = 0.942$ [equation (2)], but colorimetry gave nitrate-nitrogen results which were at least 24% higher than those obtained by chromatography. This may reflect the inclusion of nitrite-nitrogen in the colorimetric but not the chromatogaphy methods and the loss of precision at low concentrations (< 0.1 g m^{-3} nitrate-nitrogen in 26% of the samples). This error did not invalidate the general conclusion that nitrate-nitrogen was very rapidly lost from the effluent during pulse addition, as will be seen later.

$$y = 1.24\ x + 0.06. \quad (2)$$

RESULTS

Pulse addition of dye and tracers

Experiment 1. Stability of dye tracers. Results for experiment 1 are given in Fig. 1. In experiment 1 the theoretical pulse addition of EAR was 1.56×10^4 g m^{-3} s for an inflow rate of 130×10^{-6} m^3 s^{-1}, giving an inflowing load of EAR of 2.02 g. The concentration-time integral for the outflowing pulse (Fig. 1) was 1.28×10^4 g m^{-3} s and the discharge was 122×10^{-6} m^3 s^{-1} (94% of inflow rate) giving an outflowing load of 1.73 g. Therefore the load of dye in the outflow was 86% of the pulse addition, and EAR could be considered stable during transit through the wetland.

Experiment 2. Comparison of tracers and nutrient removal efficiency. Dye and bromide concentration-time curves for experiment 2 are given in Figs 2 and 3. Bromide and EAR dye concentration ratios were almost coincident [Fig. 2(a)], supporting the assumption that the tracers can be considered stable.

Selected results for nutrients in surface samples at the 17 m station are given in Fig. 2(b). Similar results (not shown) were obtained for depth samples.

The divergence between concentration ratios for stable tracers and nitrate-nitrogen was substantial, indicating that added nitrate was lost rapidly from the system. By contrast, tracer and phosphate-phosphorus concentration ratios were almost coincident, indicating that phosphate was much more stable.

Background concentrations (outside the region of pulse addition) showed similar trends (Table 2). Nitrate-nitrogen concentrations were not detectable (< 0.01 g m^{-3}) in the inflow, and increased only slightly towards the outflow. There was only a gradual decline in phosphate-phosphorus concentration along the system, although with 45% removal overall.

Sulphate-sulphur concentrations were extremely variable from sample to sample, and were positively correlated with nitrate-nitrogen concentration [Fig. 4 and equation (3)], where x and y are nitrate-nitrogen and sulphate-sulphur concentrations, respectively

$$y = 0.58 + 0.79y;\ P < 0.001;\ r^2 = 0.523. \quad (3)$$

Experiment 3. Nutrient removal efficiency. Concentration-time curves for dye and nitrate-nitrogen are given in Fig. 5; and background concentrations, obtained over about 5 days, are given in Table 3.

Results for the pulse addition experiment (Fig. 5) were consistent with those obtained in experiment 2, again indicating that nitrate-nitrogen was lost rapidly from the system.

Data on background concentrations showed there was negligible accumulation of nitrate-nitrogen in the wetland (Table 3), confirming previous results. Ammonium-nitrogen concentrations entering the wetland were only slightly reduced in the outflow, and organic-nitrogen represented 75–80% of the total nitrogen load in both inflow and outflow. The overall performance of the system, assessed by efficiency of nutrient removal and water recovery is shown in Table 4.

Longer-term system performance

Results for winter 1984 (June) and summer 1985 (January) are summarized in Table 5.

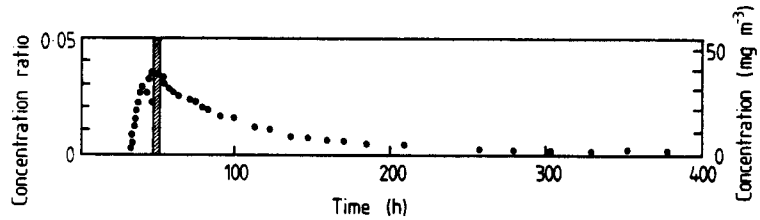

Fig. 1. Profiles of dye tracer concentration vs time in the outflow of the macrophyte treatment system after pulse addition for 4 h at the inflow. The rectangular hatched area gives the position of the theoretical pulse addition, assuming no dispersion or evapotranspiration.

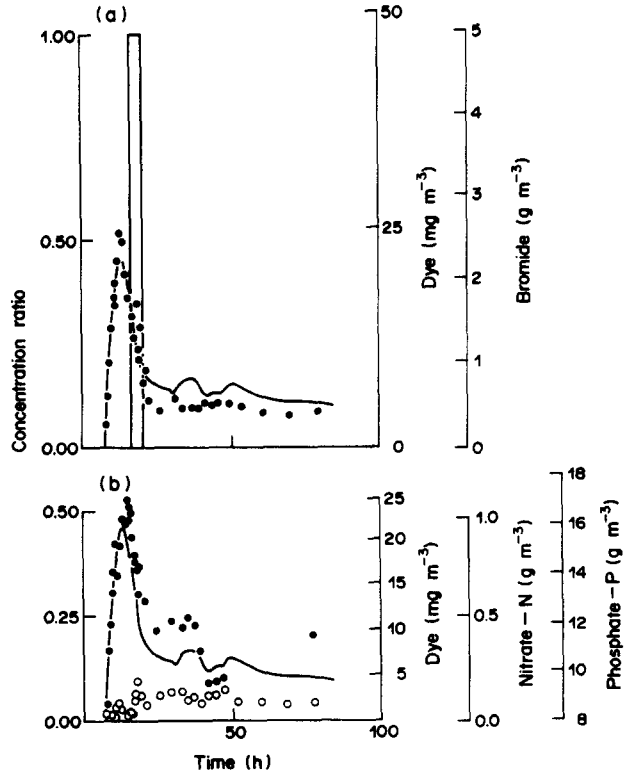

Fig. 2. Concentration-time profiles for addition of a pulse for 24 h at the inflow, observed at the surface sampling station 17 m downstream from the inflow (experiment 2). The rectangular area represents the theoretical position of the pulse addition assuming no dispersion or evapotranspiration. (a) Dye (——) and bromide (●); (b) Dye (——), nitrate-N (○), and phosphate-P (●).

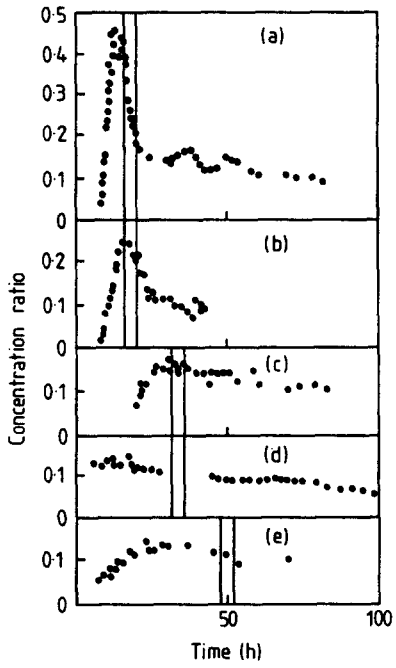

Fig. 3. Concentration-time profile for dye tracer, observed: (a) at the surface 17 m downstream; (b) at depth 17 m downstream; (c) at the surface 34 m downstream; (d) at depth 34 m downstream; and (e) at the outflow (experiment 2). The rectangular areas shows the position of the pulse assuming no dispersion.

DISCUSSION

Nitrogen removal and redox conditions

In the present study the efficient removal of nitrogen is dependent on a series of transformations. After mineralization of organic nitrogen to ammonium-nitrogen, nitrification must occur, followed by denitrification. Nitrification is a process of bacterial

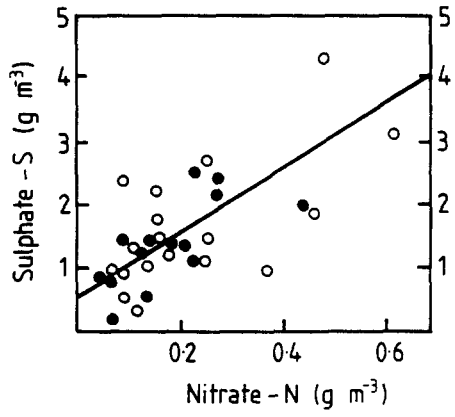

Fig. 4. Experiment 2. Relationship between sulphate-sulphur and nitrate-nitrogen concentrations at a sampling point 34 m from the inflow for surface (○) and depth (●) samples. Line fitted by regression given in equation (3).

Table 2. Experiment 2. Background concentrations in the macrophyte system on one day in February 1983 (summer)

Distance from inflow (m)	Nitrate-N* ($g\,m^{-3}$)	Phosphate-P* ($g\,m^{-3}$)	Sulphate-S* ($g\,m^{-3}$)	Chloride* ($g\,m^{-3}$)
0 Inflow	n.d.†	11.0 ± 0.7 (4)	9.4 ± 0.5 (4)	91.1 ± 3.45 (5)
17 Surface‡	0.03 ± 0.01 (5)	8.0 ± 0.6 (5)	—	—
Depth	0.08 ± 0.01 (14)	9.5 ± 0.4 (21)	—	—
34 Surface	0.03 ± 0.01 (5)	8.2 ± 0.5 (5)	—	—
Depth	0.12 ± 0.06 (5)	8.3 ± 0.2 (6)	—	—
50 Outflow	0.44 ± 0.07 (5)	6.1 ± 0.3 (6)	3.8 ± 1.8 (7)	90.0 ± 11.0 (5)

*Mean ± SE (followed in brackets by numbers of samples).
†Less than detectable ($< 0.01\ g\,m^{-3}$).
‡0.1 and 0.4 m, surface and depth, respectively.
— Not determined.

chemoautotrophic oxidation in which ammonium-nitrogen is converted to nitrite or nitrate-nitrogen. In denitrification, nitrate is reduced to nitrous oxides and di-nitrogen, with the nitrate acting as the terminal electron acceptor during the oxidation of organic compounds.

The efficiency of nitrogen removal in this system is probably limited by relatively slow rates of mineralisation and nitrification. Dentrification, in contrast, is occurring rapidly. Overall nitrogen removal efficiency for the current experiment (Table 5) was similar to that described by Finlayson and Chick (1983) for an abattoir effluent, where the forms of nitrogen were also dominantly organic and ammonium. Improved removal efficiencies were obtained by Gersberg *et al.* (1983) at Santee, Calif. but for a nitrified effluent with methanol added to aid denitrification (Table 5). The quality of the effluent clearly determines the efficiency of nutrient removal by the system.

The influence of physico-chemical conditions on the rate of nitrogen transformation has also been widely studied in natural and artificial wetlands, (e.g. Van Kessel, 1977; Nichols, 1983; Gersberg *et al.*, 1983, 1984) and also in microbial films (Strand *et al.*, 1985). Nitrification requires an oxygenated substratum (dissolved oxygen $> 2\ g\,m^{-3}$), whereas denitrification requires anaerobic conditions, together with an adequate supply of organic carbon compounds.

The rendering plant effluent was relatively high in chemical oxygen demand, typical values ranging from about 200 to 1700 $g\,m^{-3}$ (Breen and Chick; personal communications). Mean average water temperatures in summer and autumn ranged from about 23 to 31°C and pH was near-neutral. These conditions are suitable for rapid denitrification, providing oxidising and reducing sites are encountered sequentially. Theoretically, efficient nitrogen removal might be

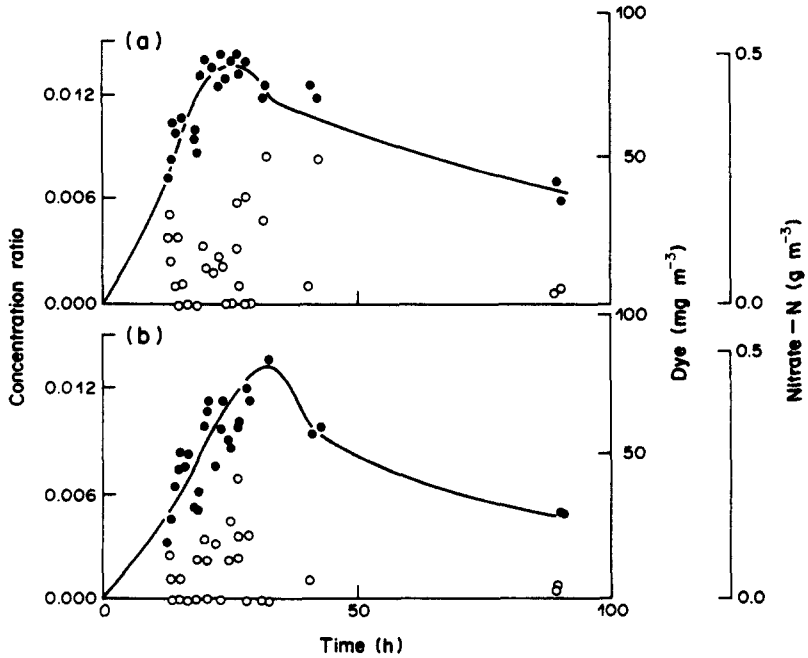

Fig. 5. Experiment 3. Concentration-time profiles with time after the start of addition of a pulse of dye (●) and nitrate-N (○) for 1 h at the inflow, observed 17 m downstream: (a) at the surface; (b) at depth. (Nitrate-nitrogen background concentrations assumed to be zero; see Table 3.)

Table 3. Experiment 3. Background concentrations in the macrophyte treatment system during 5 days in March 1985 (late summer)

Distance from inflow	Nitrate-N ($g\,m^{-3}$)	Ammonium-N† ($g\,m^{-3}$)	Kjeldahl-N ($g\,m^{-3}$)	Phosphate-P† ($g\,m^{-3}$)	Sulphate-S† ($g\,m^{-3}$)
0 Inflow	0.06–0.18*‡	0.05–0.07†‡	105.4 ± 105.6*‡	519	19.9–20.2‡	4.6–4.8‡
17 Surface¶	0.11 ± 0.02 (35)§	0.07 ± 0.02 (32)§	107.9 ± 0.4 (35)§	480 ± 3 (10)	16.0 ± 0.2 (33)§	0.73 ± 0.17 (32)§
Depth¶	0.08 ± 0.02 (35)	0.04 ± 0.01 (33)	107.3 ± 1.0 (35)	467 ± 6 (10)	11.6 ± 0.3 (33)	0.50 ± 0.07 (32)
34 Surface	0.46 ± 0.16 (9)	0.22 ± 0.05 (8)	105.7 ± 2.0 (6)	447 ± 9 (6)	15.7 ± 0.5 (6)	0.87 ± 0.17 (6)
Depth	1.82 ± 0.88 (9)	1.39 ± 0.69 (9)	106.8 ± 3.1 (6)	467 ± 9 (7)	16.2 ± 0.2 (6)	3.8 ± 1.1 (6)
50 Outflow	0.27 ± 0.05 (24)	0.10 ± 0.02 (24)	103.6 ± 0.4 (24)	415 ± 6 (4)	12.4 ± 0.7 (24)	0.57 ± 0.03 (24)

*Analysis by Auto Analyser methods.
†Analysis by ion chromatography.
‡Range of duplicate analyses.
§Mean ± SE (followed in brackets by numbers of samples).
¶0.2 and 0.4 m, surface and depth, respectively.

achieved by high and low redox microsites in juxtaposition as in deep microbial films with aerobic surfaces (Strand and McDonnell, 1985), or be effluent flow through aerobic and anaerobic microzones in sequence in the wetland.

During experiments 2 and 3 the overall status of the experimental system appeared to be mildly reducing in the upstream portion since nitrate-nitrogen was rapidly removed [Figs 2(b) and 5], and mildly oxidising downstream, since nitrate-nitrogen began to accumulate there (Tables 2 and 3). In experiments 2 and 3 sulphate-sulphur concentrations were highly variable (Tables 2 and 3), and in the mid-section of the wetland were positively correlated with nitrate-nitrogen concentration (Fig. 4). Sulphate reduction is indicative of very reducing conditions so this may reflect rapid and extreme fluctuations in the redox status of the effluent, compatible with the close juxtaposition of microsites of contrasting redox in the mid-part of the system. The occurrence of extremely reducing zones or microsites throughout the wetland is also suggested by the overall reduction in sulphate-sulphur concentration described in Tables 2 and 3. These highly reducing zones could have two fold implications in system operation. Firstly, they are essential for the anaerobic denitrification process. Secondly, they may reduce the rate of nitrification through the production of toxic concentrations of sulphide.

In the current experiments it seems that one of the limitations to efficient nitrogen removal is that oxygenation is inadequate for nitrification. It is well-known that emergent aquatic macrophytes are able to oxygenate their substratum (e.g. Armstrong, 1972; Oremland and Taylor, 1977; Sherr and Payne, 1979; Finlayson and Chick, 1983) but the magnitude of the process is difficult to assess. In the present study the oxygen demand of the influent is high so that reducing conditions are expected to prevail in the upstream portion of the system. The inability of the plants to supply sufficient oxygen to counter the oxygen demand of the effluent is also demonstrated by results obtained during April 1983 (cf. experiment 2 in February 1983). The chemical oxygen demand (COD) average ± SE for 15 samples) was: inflow, $212 \pm 9\,g\,m^{-3}$; outflow in an unplanted control system, $108 \pm 11\,g\,m^{-3}$; and outflow from the planted system, $81 \pm 9\,g\,m^{-3}$. Further details are given elsewhere (Bowmer *et al.*, 1985). Therefore the plants did provide oxygen, reducing the COD by 62%, compared with only 49% in the plant-free control, but there was still a substantial oxygen demand in the effluent after it had passed through the system.

Also, even if the macrophytes can introduce large quantities of oxygen into their substratum, the effects of channelling in preferential flow paths around the aerated root masses may be preventing sufficient diffusion of oxygen into the effluents. This could account for slow rates of mineralization and nitrification as discussed earlier.

Table 4. Experiment 3. Efficiency of nutrient removal in the macrophyte treatment system over 5 days in March

Kjeldahl-N		Ammonium-N		Water	
Load ($g\,m^{-2}\,d^{-1}$)	Removal (%)	Load ($g\,m^{-2}\,d^{-1}$)	Removal (%)	Load ($m\,d^{-1}$)	Recovery (%)
21.6	38	0.84	52	0.0416	78

Table 5. Comparison of total loads of nitrogen, nutrient removal efficiency, and recovery of water in artificial wetlands

			Total Kjeldahl-N		Effluent water	
Effluent	Macrophyte(s)	Operation	Load ($g\,m^{-2}\,d^{-1}$)	Removal (%)	Load ($m\,d^{-1}$)	Recovery (%)
Rendering* plant	*Eleocharis*, *Typha* and *Schoenoplectus*	Winter 1984‡	4.7	43	0.089	102
		Summer 1985‡	7.5	67	0.069	52
Abattoir	*Typha*		1.4	42	0.014	68
	Phragmites	Autumn 1981§	1.5	62	0.016	51
	Schoenoplectus		1.0	74	0.010	55
Nitrified sewage plus methanol	*Schoenoplectus* and *Typha*	Oct. '80–Sep. '81¶	48	95	0.17	n.a.†
		Oct. '81–May '82**	3.1	94	0.25	n.a.†

*Current experimental system.
†Not available.
References: ‡Breen (1987); §Finlayson and Chick (1983); ¶Gersberg *et al.* (1983); **Gersberg *et al.* (1984).

Hydrology

The presence of preferential flow paths is demonstrated by results for dye tracer experiments. As expected, dye concentration-time profiles (Figs 1 and 3) show large dispersion and asymmetry, as expected for flow through an irregular gravel substrate, with "dead zones" which are inaccessible to the main flow. Consequently, a substantial proportion of the effluent travels through the system faster than predicted by the theoretical retention time. The proportion not retained was 13% in experiment 1 (Fig. 1) and probably much higher in experiment 2 [Fig. 3(e)]. Major preferential pathways are also demonstrated in experiment 2 (late summer 1983) by the rapid arrival of tracer at the 34 m station at depth [Fig. 3(d)]. Yet by March 1985 this area became a large "dead zone", since dye injected there in experiment 3 remained in position for at least 5 days. Later examination suggested the accumulation of a sludge of detritus at the bottom of the ageing system, which, by blocking pore volume, is expected to increase channelling in the system, and reduce the overall retention time even further.

The hydrology of the experimental system is also complicated by the inland weather pattern in summer, with oscillations over 4–7 day intervals of hot dry weather and cooler conditions (Fig. 6). Occasionally, in the most severe weather, evapotranspiration consumes all the daily inflow, as indicated by cessation of outflow measured by tipping buckets. Superimposed on this pattern is a large diurnal change in water consumption (Table 6). The wetland must operate as a temporary sink and nutrient-concentrator during periods of high evapotranspiration, when the distribution of water-absorbing roots will be influential in determining the pattern of flow. Gravity induced flow will dominate at night, and in cooler or more humid weather.

Longer-term performance and implications for system design

The efficiency of nitrogen removal may be limited in the experimental system by insufficient oxygenation. Oxygen is required to accelerate mineralization and is essential for nitrification. Slow mineralization is demonstrated by the presence of organic-nitrogen in the outflow. (There were also substantial concentrations of cells of *Scenedesmus* spp which evidently travelled intact through the system.) Slow nitrification is suggested by the observations that nitrate-nitrogen did not accumulate in the system, and that ammonium-nitrogen was present in the outflow. However, in a substratum with high and low redox sites in close proximity, nitrate-nitrogen would probably be transitory, so that the rate of mineralization, rather than nitrification, could control the efficiency of nitrogen removal. Nitrate-nitrogen added to the inflow was rapidly removed, suggesting that denitrification would not limit system performance.

A further complication is that some of the organic-nitrogen removal may reflect physical entrapment in the substratum. Some of the nitrogen in algal and bacterial bodies may be recalcitrant and mineralization may only occur over very long periods (e.g. months). As a corollary, it is possible that the load of outflowing ammonium-nitrogen could reflect the mineralization of organic matter accumulated previously. Since the rate of mineralization is correlated with temperature, large fluctuations in outflow loads are expected in response to short-term changes in temperature, superimposed upon seasonal variation.

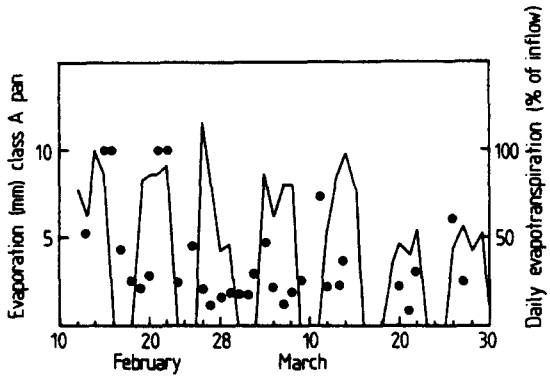

Fig. 6. Daily evapotranspiration (●) from a macrophyte treatment system during 50 days in late summer 1985, at a average inflow of 0.044 m day^{-1} (5.1×10^{-5} m^3 s^{-1}); together with class A pan evaporation at Griffith, New South Wales (——).

In the current study, observations were insufficient to quantify the relative importance of mineralization and nitrification as rate-limiting processes. However, circumstantial evidence is available from observations of the performance of other Australian wetlands, planted with *S. validus* or *Typha* species, at Richmond, New South Wales. Here, ammonium-nitrogen comprises about 80% of the inflowing nitrogen load in sewerage, so that mineralization should be less important than nitrification in limiting system performance.

However, the efficiency of ammonium-nitrogen removal seemed to be limited in the Richmond system (Table 7; H. J. Bavor, unpublished data) and nitrite-nitrogen and nitrate-nitrogen concentrations remained small (< 1 g m^{-3} on average). These results suggest that while slow mineralization of the organic fraction may contribute to poor performance, oxygen shortage, and its effect in limiting nitrification, is a critical factor in restricing the efficiency of both the Griffith and Richmond systems.

In summary, for efficient performance, it seems that more oxygen must be introduced into the effluent to stimulate nitrification. Alternatively, the loading of effluent might be reduced, either by increasing the retention time, or by diluting the effluent before introduction into the wetland. The potential for increasing retention time is not practicable at our experimental site in summer, because high evapotranspiration already reduces the efficiency of water recovery by about 50% (Table 5). An advantage of artificial wetlands over land disposal is the recovery of water for the maintenance of river flows and water supplies. If this advantage is to be maintaned the retention time cannot be prolonged, and nutrient loadings could only be reduced by dilution.

Gersberg *et al.* (1983, 1984) demonstrated efficient removal of nitrogen from nitrified effluents at flow rates up to about 4 times faster than those used in current experiments, although with similar nutrient loadings (Table 5). Clearly, the relationship between rate of flow, nutrient loading and system performance will need to be optimised if the objects of both nutrient removal and recovery of water are to be achieved.

Greater oxygenation and nitrification might be achieved outside the system by preliminary treatment in algal ponds or by mechanical aeration methods, but these are expensive. Possible changes in the artificial wetland, itself, include using plant species giving greater radial diffusion of oxygen from their roots into the substratum, and hydraulic improvements to minimize preferential channelling of flow. Perforated baffles might be useful to improve mixing, or sprinkler systems could be used to apply the effluent over a larger surface area. Periodic draining and filling is another possible strategy which could be explored to provide oxygenation alternately with reducing conditions, as required to impove the efficiency of nitrogen removal in this wetland system.

Table 6. Diurnal change in evapotransporation (% of inflow) during two hot days in March 1985, for an inflow of about 0.1 m day^{-1} (1.2×10^{-4} m^3 s^{-1})

		Evapotranspiration	
Date	Day*/night	+ Plants (%)	− Plants (%)
11/3	Day	56	4
	Night	1	0
12/3	Day	85	1
	Night	10	−3

*08.00–20.00 h.

Acknowledgements—Many colleagues at the CSIRO Centre for Irrigation and Freshwater Research made substantial contributions to this study, particularly Dr M. Finlayson,

Table 7. Ammonium-nitrogen and flow régime in artificial wetland sewerage treatment systems at Richmond, New South Wales; average values given for March 1984–March 1985*

Treatment	Site	Ammonium − N (g m^{-3})	Effluent load (m d^{-1})	Ammonium − N load (g m^{-2} d)	Removal (%)
Gravel control	Inlet	41.8	0.0985†	4.12	10
	Outlet	41.1	0.0903	3.71	
S. validus	Inlet	41.8	0.0773†	3.23	43
	Outlet	23.8	0.0768	1.83	
Typha spp	Inlet	41.8	0.0723†	3.02	44
	Outlet	23.8	0.0710	1.69	

*H. J. Bavor, Hawkesbury Agricultural College, unpublished report. "Joint Study on Nutrient Removal using Shallow Lagoon-Macrophyte Systems". Interim Report March 1984–May 1985.
†Excluding rain (2.3×10^{-3} m d^{-1}).

Mr A. Chick, Mr P. Breen, Mr P. Weerts, Mr L. Higgins and Ms M. Rowley, who established and maintained the system, or provided chemical analyses. Dr D. Mitchell enthusiastically initiated a project on effluent treatment by artificial wetlands, of which this study is a small part. I am grateful to Dr P. Boon and to two anonymous referees for many useful editorial additions and improvements.

REFERENCES

Armstrong W. (1972) A re-examination of the functional significance of aerenchyma. *Physiol. Pl.* **27**, 173–177.

Breen P. F. (1987) Wastewater treatment using artificial wetlands: the role of emergent aquatic macrophytes. *Wat. News* (Department of Resources and Energy, Canberra). In press.

Bowmer K. H., Weerts P. G. J., Chick A. J. and Rowley M. (1985) Kinetics of nutrient transformation in rendering plant effluent at Griffith, Research Report 1983–1984. CSIRO Centre for Irrigation Research, Griffith.

Finlayson C. M. and Chick A. J. (1983) Testing the potential of aquatic plants to treat abattoir effluents. *Wat. Res.* **17**, 415–422.

Finlayson C. M. and Mitchell D. S. (1982) Treatment of rural wastewaters in Australia with aquatic plants: a summary. *Tropenlandwirt* **83**, 155–165.

Gersberg R. M., Elkins B. V. and Goldman C. R. (1983) Nitrogen removal in artificial wetlands. *Wat. Res.* **17**, 1009–1014.

Gersberg R. M., Elkins B. V. and Goldman C. R. (1984) Use of artificial wetlands to remove nitrogen from wastewater. *J. Wat. Pollut. Control Fed.* **56**, 152–156.

Hartley G. S. and Graham-Bryce I. J. (1980) Principles of diffusion and flow. In *Physical Principles of Pesticide Behaviour* (Edited by Hartley G. S. and Graham-Bryce I. J.), pp. 177–179. Academic Press, London.

Kickuth R. (1975) Degradation and incorporation of nutrients from rural waste waters by plant rhizosphere under limnic conditions. In *Utilization of Manure by Land Spreading*, pp. 335–347. Co-ordination of Agricultural Research, Commission of The European Communities.

Mitchell D. S. (1978) The potential for wastewater treatment by aquatic plants in Australia. *Wat. Aust.* **5**, 15–17.

Nichols D. S. (1983) Capacity of natural wetlands to remove nutrients from wastewater. *J. Wat. Pollut. Control Fed.* **55**, 495–505.

Oremland R. S. and Taylor B. F. (1977) Diurnal fluctuations of O_2, N_2 and CH_4 in the rhizosphere in of *Thalassia testudinum*. *Limnol. Oceanogr.* **22**, 566–570.

Pope P. R. (1981) Wastewater treatment by rooted aquatic plants in sand and gravel trenches. (EPA-600/2/81-091), U.S. Environmental Protection Agency, Cincinnati, Ohio.

Seidel K. (1976) Macrophytes and water purification. In *Biological Control of Water Pollution* (Edited by Tourbier J. and Pierson R. W.), pp. 109–121. University of Pennsylvania Press, Philadelphia.

Sherr B. F. and Payne W. J. (1979) Role of the salt marsh grass *Spartina alterniflora* in the response of soil-denitrifying bacteria to glucose enrichment. *Appl. envir. Microbiol.* **38**, 747–748.

Strand S. E. and McDonnell A. J. (1985) Mathematical analysis of oxyen and nitrate consumption in deep microbial films. *Wat. Res.* **19**, 345–352.

Strand S. E., McDonnell A. J. and Unz R. F. (1985) Concurrent denitrification and oxygen uptake in microbial films. *Wat. Res.* **19**, 335–344.

Tabatabi M. A. and Dick W. A. (1983) Simultaneous determination of nitrate, chloride, sulfate and phosphate in natural waters by ion chromatography. *J. envir. Qual.* **12**, 209–213.

Van Kessel J. F. (1977) Factors affecting the denitrification rate in two water-sediment systems. *Wat. Res.* **11**, 259–267.