The ability of an artificially established wetland system to upgrade oxidation pond effluent to meet water quality criteria

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Abstract

A pilot-scale artificially established wetland planted with *Phragmites australis* and referred to as a reed bed (or AEWL) received effluent from an experimental oxidation pond system (OPS). Over a 12-month period, concentrations of $PO_4 - P$, $NH_3 - N$, $NO_3 - N$ and total suspended solids (TSS) from the OPS were reduced by up to 70%; 50% of COD and all coliform bacteria were removed during passage through the AEWL. The water quality of the OPS effluent was upgraded to meet effluent standards of $NH_3 - N$, TSS, *E. coli* and $PO_4 - P$ (for the second half of the study) after passage through the AEWL.

The AEWL, a combination of gravel and soil (20% clay) removed $PO_4 - P$ from the OPS effluent, probably by the following mechanisms: P adsorption onto clay molecules; precipitation with calcium (Ca^{++}) and iron (Fe^{+++}); and uptake by microorganisms and *Phragmites*. $NH_3 - N$ and $NO_3 - N$ were probably removed by nitrification - denitrification in aerobic and anaerobic zones; adsorption onto soil particles; and uptake by microorganisms and *Phragmites*. Organic matter was probably primarily removed by filtration and sedimentation. An extrapolated harvest of 71 t ha^{-1} a^{-1} could be expected from such systems and this crop could probably be used for fodder or building material.

Introduction

Artificially established wetland systems (AEWL) have been utilised in West Germany (Seidel, 1976), the Netherlands (De Jong, 1976), the USA (Spangler *et al.*, 1976; Werblan *et al.*, 1978) and Australia (Finlayson and Chick, 1983) to treat rural, industrial and municipal waste waters. The benefits of such systems are:

- ease of operation;
- low installation and building costs;
- low operation costs;
- insensitivity to fluctuating loads; and
- good purification results (De Jong, 1976).

Phragmites, *Typha* and *Scirpus* are the main genera of plants used in AEWL. Swamp plants of these and other genera have lacunae which allow the movement of oxygen to their roots when growing in anaerobic sediments. Oxygen (O_2) can diffuse from the roots (e.g. Finlayson and Chick, 1983) and create aerobic zones within the rhizosphere, and this may have several effects (Althaus, 1976). Firstly, it stimulates the development of a diverse microflora and microfauna which assist in the breakdown of organic matter. Secondly, aerobic zones are created where nitrification can occur. The effluent therefore passes intermittently through aerobic and anaerobic zones where nitrification and denitrification lead to excellent nitrogen removal properties. Thirdly, the aerobic conditions also contribute to phosphate adsorption and precipitation (Toerien and Wrigley, 1984).

The filter media used in AEWL can consist of gravel (Seidel, 1976; Spangler *et al.*, 1976; Werblan *et al.*, 1978; and Finlayson and Chick, 1983), soil (De Jong, 1976) or sand and gravel (Pope, 1981). Gravel acts as a porous support and anaerobic filter (Young and McCarty, 1969) and is primarily responsible for reductions in chemical oxygen demand (COD) and suspended solids (SS). Reductions of total nitrogen from 42 to 90% and total phosphorus from 55 to 79% have been achieved in AEWL with gravel (Finlayson and Chick, 1983; Wolverton, 1982; Wolverton *et al.*, 1983). However, experiments with a sandy clay (De Jong, 1976) resulted in 80% removal of total N and over 99% removal of the total P from a sewage effluent. A sand layer above gravel led to clogging problems, but *Phragmites* rootlets aided in the degradation of the surface sludge layer (Pope, 1981).

The purpose of this study was to evaluate the ability of a pilot-scale AEWL to upgrade the quality of effluent received from an experimental oxidation pond system (OPS). Several parameters e.g. TSS, $NH_3 - N$ and COD included in the standards for the discharge of industrial effluents, were used as a basis for evaluation.

Materials and methods

A fibreglass box 2,4 m long, 0,7 m deep and 0,8 m wide was filled with gravel (about 2 cm size) and soil (Fig. 1). Gravel was placed in the first 0,34 m^3 of the container, the remaining volume was made up by equal proportions of gravel and soil. The box was connected to an experimental OPS (Wrigley *et al.*, 1988) by a 50 mm polyvinylchloride (PVC) pipe containing a sampling port. Effluent from the OPS passed through the pipe and a right-angle joint inserted 0,3 m into the gravel matrix. Holes in this joint allowed effluent to disperse at different levels within the matrix. Effluent was collected from a hole (4 cm diameter) at the bottom

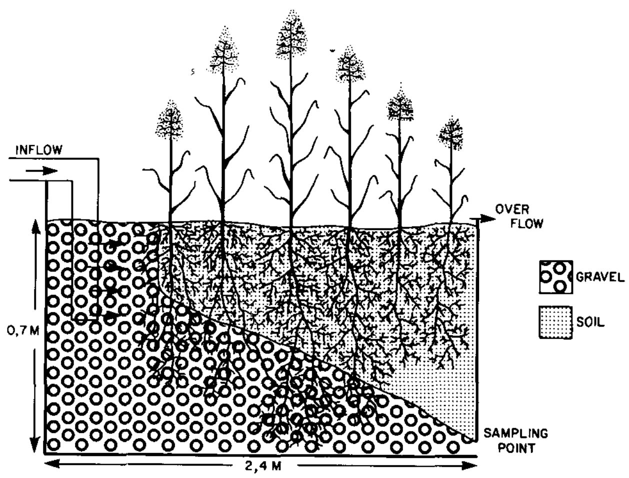

Figure 1
Cross-section of an artificially established reed bed system.

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of the end of the box (Fig. 2). Samples from both points were taken three times a week from March 1982 to April 1983.

TSS concentrations were determined by filtration on 0,45 μm Sartorius filters and oven drying (105 °C). The filtrate was collected and weekly composite samples analysed for NH_3 – N, NO_3 – N and PO_4 – P (APHA, 1975). COD and oxygen absorbed (OA) were determined once per week on unfiltered samples for each sampling point (APHA, 1975).

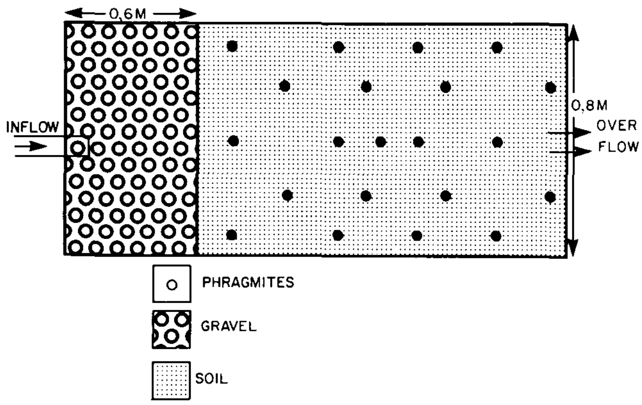

Figure 2
Overhead view of an artificially established reed bed system.

Bacterial counts were determined using the pour-plate method (APHA, 1967). Total bacterial numbers were determined on nutrient agar (Merck) and coliform numbers determined on endo fuchsin lactose agar Type C (Merck).

Phragmites australis rhizomes were collected from Wuras Dam, 70 km from Bloemfontein. Twenty-one rhizomes were planted into the gravel and soil of the container and all became well established (Fig. 2). All above-ground material in the container was harvested in March 1983 and again 38 d later, along with rhizomes and roots. Above-ground material was separated into leaves and stems and weighed after drying to constant weight at 105 °C. Below-ground material was washed before weighing. Nitrogen (N) and phosphorus (P) concentrations in the above-ground plant material were measured by Kjeldahl and molybdate ascorbate after digestion in sulphuric and nitric acid respectively (Wiltshire, 1980) Potassium (K), sodium (Na), calcium (Ca) and magnesium (Mg) concentrations were measured by atomic absorption spectrophotometry after digestion with perchloric acid (Wiltshire, 1980).

Soil samples were taken randomly in March 1983, using a soil auger at 0,7 m intervals along the bed sampling the depth of the soil column. The inorganic N concentration in the soil was measured by Kjeldahl digestion in an extract made with 2 mol dm^{-3} KCL (Bremner, 1965). Extractable (or plant available) P was extracted from the soil with 0,5 mol dm^{-3} $NaHCO_3$ and determined by the Olsen method (Olsen and Dean, 1965). Total P was determined after HNO_3/H_2SO_4 digestion, Zn was determined after extraction with 0,1 mol dm^{-3} HCl, whereas Ca, Mg, K and Na were determined by atomic absorption spectrophotometry after extraction with a solution of $NH_4C_2H_2O_2$ (pH7). Texture was determined by the hydrometer and sieve method (Bremner, 1965) and pH was measured in solution (soil: water, 1:2,5).

Results and discussion

Four factors were examined to assess the performance of the AEWL over a period of a year. Firstly, the ability to upgrade effluent to meet standard water quality criteria was determined. Secondly, the productivity and chemical characteristics of waste-grown reeds were compared with reeds from natural systems to determine the effect of nutrient-rich waste water on these properties. Thirdly, productivity measurements of reeds were made to indicate the magnitude of the potential harvest and its uses. Fourthly, chemical studies were carried out on the soil of the systems in order to provide an insight into the mechanisms of phosphate uptake and denitrification — nitrification processes essential for the improvement of effluents.

Flow regime

The approximate flow to the AEWL was 1 m^3 d^{-1}. To determine the retention period, fluorescent dye (pyranine) was added to the inflow and recorded in the outflow every day for a week. Flow took place in part across the surface of the AEWL before percolation to the bottom and outflow point or in part by entering through the gravel-soil matrix and horizontal percolation to the outflow. The soil remained saturated at all times. Overflow from the surface occurred regularly; measurements were not taken of this flow, so mass nutrient balance determinations were not made.

This constant overflow indicated that hydraulic conductivity of AEWL was exceeded. However, the following nutrient removal data suggest likely performance criteria in properly designed systems. Permeability problems of this nature appear to be overcome in small greenhouse upwelling bucket systems where nutrient removal of settled sewage is of the order of 90% for 3-year-old systems (Breen and Rogers, 1988). However, caution is needed when translating these results to large-scale trench systems (Breen, 1988).

Chemical characteristics

Chemical oxygen demand (COD)

The mean COD concentration for the year after passage through the system was 55% that of the inflow (Table 1 and Fig. 3). There was a large variation in the COD concentration of the inflow which was not reflected in the effluent from the AEWL. This reduction is similar to that reported for similar systems elsewhere Kickuth (1975), (1976); Seidel (1976) and Pope (1981). The mean average COD concentration of the final effluent was 121 $mg\ell^{-1}$, the gazetted Government standard is 75 $mg\ell^{-1}$ (Table 2).

The major components of the COD concentrations entering the AEWL were algal cells, some detritus and zooplankton. This particulate matter was trapped and probably used as respiratory substrate within the gravel matrix by a large microbial population.

The development of a more advanced design should allow further reductions in COD concentrations to meet the effluent standard.

Total suspended solids (TSS)

TSS concentrations were reduced from 75 to 23 $mg\ell^{-1}$ after passage through the AEWL (Table 1, Fig. 4). This reduction of 69% brought the effluent below the effluent standard of 25 $mg\ell^{-1}$ (Table 2).

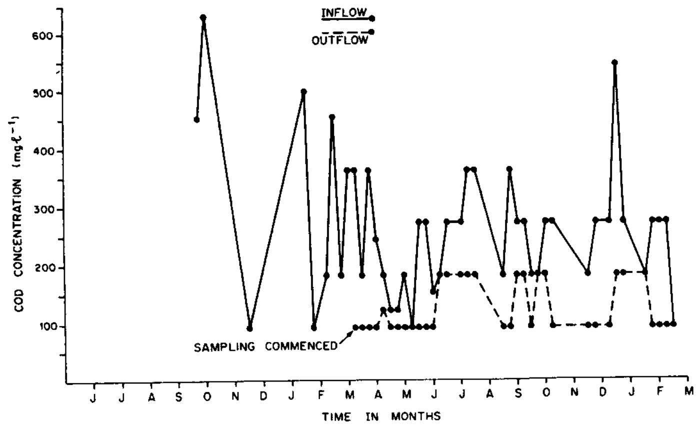

Figure 3
Chemical oxygen demand (COD) of the inflow and outflow from an artificially established reed bed system fed by oxidation pond effluent from March 1982 to April 1983.

TABLE 1
ANNUAL MEAN AND STANDARD DEVIATION OF TSS, PO_4-P, NH_3-N, NO_3-N, OA AND COD CONCENTRATIONS AT THE INFLOW AND OUTFLOW OF THE ARTIFICIALLY ESTABLISHED REED BED (ALL VALUES IN $mg\ell^{-1}$)

Parameter	Inflow	Outflow	% Reduction
TSS	75 ± 42	23 ± 15	69,3
PO_4-P	6,3 ± 3,7	2,0 ± 1,9	68,3
NH_3-N	3,7 ± 3,4	0,8 ± 0,8	78,4
NO_3-N	1,03 ± 1,61	0,27 ± 0,42	73,8
COD	267 ± 128	121 ± 43	54,7
OA	23,0 ± 8,1	19,2 ± 7,9	16,4

Similar reductions for TSS have been achieved by other AEWL e.g. (Finlayson and Chick, 1983; Pope, 1981).

Wolverton (1982) compared two systems; both had gravel filter media, but only one was planted with reeds (*Phragmites*). In the system planted with *Phragmites* the TSS concentration of settled sewage was reduced in 48h from 68 to 6 $mg\ell^{-1}$. By contrast, in the gravel only, the TSS concentration was reduced from 51 $mg\ell^{-1}$ to 15 $mg\ell^{-1}$ over the same time. Wolverton (1982) proved for other chemical constituents that a planted gravel filter medium was superior in reducing the concentrations of these constituents in comparison to a gravel only filter medium.

The physical filtering ability of the gravel and the developed microbial populations enhanced by the presence of plants were probably primarily responsible for the large reductions in TSS and COD concentrations.

Ammonia (NH_3-N) and nitrate (NO_3-N)

Both NH_3-N and NO_3-N concentrations were reduced by 70 to 80% in passage through the AEWL (Table 1 and Figs. 5 and 6), and to concentrations less than the Government standards (Table 2). Wolverton *et al.* (1983) and Althaus (1976) suggested that such reductions occur in *Phragmites* reed beds because the plant roots are contributing sufficient oxygen to the substrate around the roots to enhance nitrification, but not enough to sustain the entire substrate in a completely aerobic state. Therefore, anoxic conditions conducive to denitrification also occur in the system as a whole.

Similar reductions in total nitrogen have been recorded by Kickuth (1976), Finlayson and Chick (1983) and Gesberg *et al.* (1983). These large reductions in nitrogen are a factor of the available carbon source for denitrifying bacteria. Gesberg *et al.* (1983) using mulched plant biomass as an added carbon additive in an artificial wetland system increased the loss of total nitrogen from 25% to 86%.

It appears that in this extremely well vegetated system, decaying plant material was an adequate source of carbon necessary for the high rate of denitrification which occurred.

Inorganic phosphorus (PO_4-P)

PO_4-P concentrations were reduced by 68% from 6,3 to 2,0 $mg\ell^{-1}$ (Table 1 and Fig. 7). This reduction appeared to be dependent on the rate of establishment of *Phragmites* within the substrate. During the first six months, the concentration of PO_4-P in the effluent varied considerably. Subsequently the PO_4-P concentration averaged 0,7 $mg\ell^{-1}$ which is below the recently introduced phosphate standard of 1 $mg\ell^{-1}$ of PO_4-P for sensitive catchments. This increased removal of PO_4-P with time is probably a factor of increased aeration of the soil mass and the exposure of a greater surface area of soil by expansion of the root systems, thus making more sites available for P adsorption.

TABLE 2
A COMPARISON OF REED BED EFFLUENT CONCENTRATIONS WITH MATURATION POND DESIGN CRITERIA AND THE GENERAL EFFLUENT STANDARD FOR INDUSTRIAL EFFLUENT.*

Parameter	Reed bed effluent	Maturation pond effluent according to design criteria (Meiring *et al.*, 1968)	General standard for industrial effluents (Government Gazette, 1962)
COD	121	130	75
TSS	23	no value	25
NH_3-N	0,8	10	10
NO_3-N	0,27	no value	no value
PO_4-P	2,0	no value	1**
OA	19	15	10
E. coli/mℓ	0	1 000/100 mℓ (97,5% probability)	nil/100 mℓ

* All values in mg ℓ^{-1} except *E. coli*
** A 1 mg ℓ^{-1} PO_4-P effluent standard has been promulgated and came into effect on 1 August 1985.

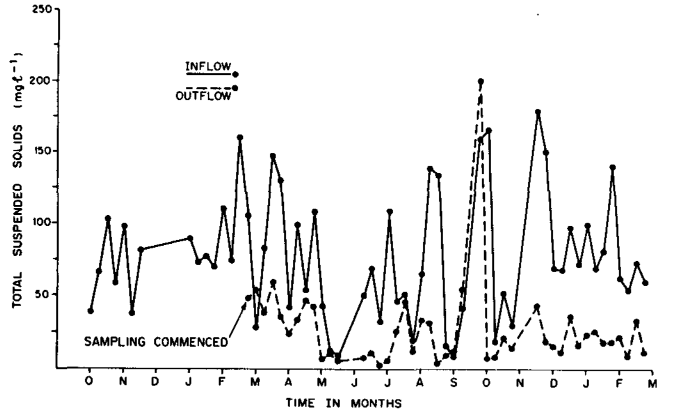

Figure 4
Total suspended solids (TSS) concentrations of the inflow and outflow from an artificially established reed bed system fed by oxidation pond effluent from March 1982 to April 1983.

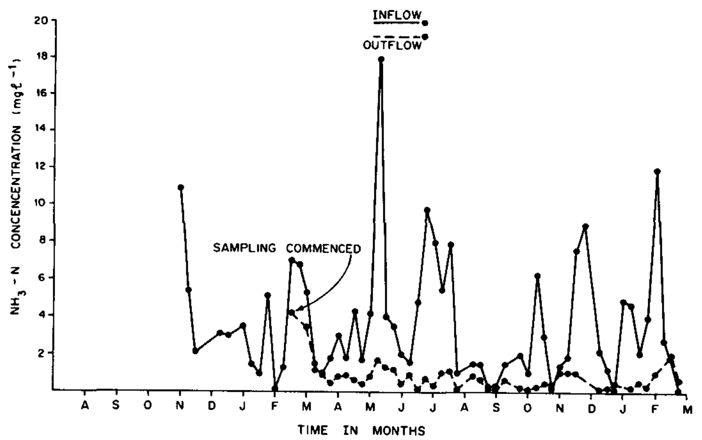

Figure 5
Ammonia ($NH_3 - N$) concentrations of the inflow and outflow from an artificially established reed bed system fed by oxidation pond effluent from March 1982 to April 1983.

The increased concentration of O_2 in the system promotes the adsorption of P from the liquid phase onto soil or sediments (Wetzel, 1975). High O_2 levels also keep iron (Fe) in the ferric form which is necessary for P immobilisation in soil (Bouwer, 1976).

The soil component of the AEWL consisted of 20% clay and 70% fine sand. It has been shown that phosphates will bind onto the positively charged Al^{3+} edges of clay plates and substitute for silica in clay structures (Stumm and Morgan, 1970). Van Riensdijck *et al.* (1979) found that more phosphate was sorbed onto soil from waste water than from pure solutions, probably because phosphate, Al^{3+} and a cation of waste water were involved and formed a stable compound. In laboratory studies, much of the adsorption of phosphate onto soil occurs within the first few minutes, but slower reactions continue to remove phosphate for days to several months (Nichols, 1983). This slower phosphate fixation has been attributed to the shift of physically adsorbed phosphate to chemically complexed forms, the diffusion of adsorbed phosphate on the surface of structurally porous oxides of Fe and Al to positions inside the matrix and the precipitation of crystalline Fe, and Al phosphates (Nichols, 1983). Toerien and Wrigley (1984) also suggest that the clay may have effectively adsorbed organic compounds and the soil may thus contribute to the removal of organics from the effluent.

Similar removal percentages have been reported by Wolverton (1982) and Finlayson and Chick (1983).

The removal of P from waste water is an important consideration since P is a major contributor to eutrophication (e.g. Toerien, 1977). The inclusion of soil in AEWL must be further evaluated, as the success of these systems may well depend in the long term on the ability of the soil to adsorb P. The incorporation of soil reduces the hydraulic conductivity of an AEWL but the extra land needed to compensate for the extra flow may well compensate for the reduction in eutrophication.

Oxygen absorbed (OA)

Concentrations were reduced by 16% (Table 1). The final effluent exceeded the effluent standard (Table 2); however, this may be as a result of the collection of numerous flying insects breeding at the sampling sites.

Total and coliform bacteria

Bacteriological analyses of the effluent from the AEWL indicated an excellent removal of coliform bacteria, though total bacterial numbers sometimes increased (Table 2). It is generally observed that bacteria are removed from infiltrating effluents mainly by filtration, sedimentation and adsorption (Gerba *et al.*, 1975), whereas viruses are mainly removed by adsorption (Bitton 1975; Gerba *et al.*, 1975) onto soil particles. Clay particles in soil may be important in providing adsorption sites for pathogen removal.

Productivity of *Phragmites* in the AEWL

The above-ground biomass of *Phragmites* at the end of the experiment in the AEWL was 6 334 g m^{-2} (Table 3). *Phragmites* growing at Wuras Dam, a mesotrophic impoundment 70 km from Bloemfontein, yielded 1 846 g m^{-2} a^{-1} of above-ground biomass (Wiltshire, 1981). World-wide biomass production reported for *Phragmites* ranged from 654 to 3 990 g m^{-2} a^{-1} (Westlake, 1963; Dykyjova, 1968 and Ho,1981). In this experiment the regrowth of reed for a further 38d after the initial harvest resulted in a biomass of 793 g m^{-2} (Table 3). The total above-ground biomass of *Phragmites* over the entire growing period was therefore 7 127 g m^{-2} a^{-1}, or 71,3 t ha^{-1} a^{-1}.

The below-ground biomass of *Phragmites* was 5 348 g m^{-2} (Table 3), substantially higher than the 3 144 g m^{-2} a^{-1} recorded for Wuras Dam (Wiltshire, 1981). Kvet and Husak (1978) reported that the annual below-ground biomass of *Phragmites* ranged from 1 600 to 8 000 g m^{-2} world-wide.

The effluent and the soil-gravel filter media used in the study promoted satisfactory growth of *Phragmites*. The very high production of *Phragmites* was probably related to the uptake of nutrients from the effluent. This probably increased the surface area of the filter media, allowed for greater P adsorption, the development of microorganisms and increased the void area of the filter media, which improved filtration, hydraulic conductivity and increased the aeration of the filter media.

Chemical composition of *Phragmites*

The concentrations of the major ions and nutrients of the waste-grown *Phragmites* are compared with *Phragmites* harvested from Wuras Dam and a hypertrophic Scottish loch in Table 4. N, P,

TABLE 3
HARVESTABLE BIOMAS FROM THE ARTIFICIALLY ESTABLISHED *PHRAGMITES* REED BED AFTER 12 MONTHS

	Productivity in reed bed (1,92 m^2)	
	(kg)	(kg m^{-2})
Below-ground	10,268	5,348
Above-ground	12,162	6,334
Leaves	5,727	2,983
Stems	6,435	3,356
Regrowth (38 d)	1,523	0,793

TABLE 4
THE CHEMICAL COMPOSITION OF STEMS AND LEAVES OF *PHRAGMITES* IN AN ARTIFICIALLY ESTABLISHED REED BED (THIS STUDY), IN AN IMPOUNDMENT IN THE SOUTHERN ORANGE FREE STATE, SOUTH AFRICA (WILTSHIRE, 1981) AND FROM A STAND GROWN IN HYPERTROPHIC WATER (SCOTTISH LOCH) (HO, 1981). THE CHEMICAL COMPOSITION OF THE NATURAL STAND FROM WURAS DAM IS EXPRESSED AS A TOTAL FOR STEM AND LEAF.

	% of dry mass					
Stem	N	P	K	Na	Ca	Mg
AE	0,728	0,101	0,832	0,067	0,027	0,068
SL	1,601	0,237	2,230	0,096	0,524	0,162
Leaf						
AE	2,095	0,159	1,724	0,055	0,135	0,169
SL	3,473	0,208	1,720	0,049	0,973	0,428
Total						
AE*	1,184	0,120	1,129	0,063	0,063	0,102
SL**	2,069	0,248	2,103	0,084	0,629	0,229
WD	0,780	0,088	1,209	0,074	0,056	0,043

AE - artificially established (this study)
SL - Scottish loch
WD - Wuras Dam
* - Shoot to leaf ratio = 2:1
** - Shoot to leaf ratio = 3:1

Ca, Mg and K concentrations in the stems were higher than in the leaves while the reverse occurred for Na. Ulehlova *et al.* (1973), Kovaces *et al.* (1978) and Ho (1981) have reported similar results.

The concentrations of $PO_4 - P$, $NH_3 - N$ plus $NO_3 - N$ of water from the AEWL, Wuras Dam and the hypereutrophic Scottish loch are listed in Table 5. The waters associated with the different *Phragmites* stands were comparable in terms of N but differed by 500% in $PO_4 - P$ concentrations. The lower nutrient concentration of the water at Wuras Dam probably accounted for the lower production of *Phragmites* at this site. Helophytes (e.g. *Phragmites*) growing in nutrient-rich habitats, whether littoral or limnosel, accumulate more mineral nutrients than populations growing in nutrient-poor habitats. Greater nutrient availability not only enhances the accumulation of mineral elements but also the net production (Dykyjova, 1978).

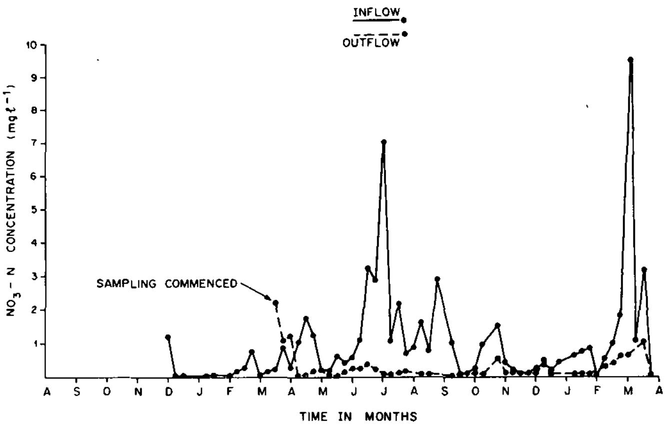

Figure 6
Nitrate ($NO_3 - N$) concentrations of the inflow and outflow from an artificially established reed bed system fed by oxidation pond effluent from March 1982 to April 1983.

The finely branched aquatic roots of *Phragmites* acquire a significant portion of the nutrients for the plant from the water body (Ho, 1981). The reed not only absorbs nutrients through these fine roots but also through the immersed part of the stem and the foliar system (Roman *et al.*, 1971).

The nutrient poor gravel and high productivity of *Phragmites* at Ho's (1981) study site, re-emphasises the importance of the surrounding water as a major source of nutrients for growth of *Phragmites*. A similar phenomenon is probably responsible for the excellent productivity of *Phragmites* in the AEWL. Though the $PO_4 - P$ concentrations of the effluent used in this experiment exceeded those found by Ho (1981) in the Scottish loch, P concentrations in the above-ground tissue of *Phragmites* were significantly lower. This may have been caused by the increased production of *Phragmites* in the AEWL which brought about a dilution in the percentage contents of the nutrient absorbed. Dykyjova (1978) presented a similar explanation for such a paradox for *Phragmites* grown in different ponds in Czechoslovakia.

Removal of N and P from the AEWL by harvesting the above-ground biomass material, amounted to 750 and 76 kg $ha^{-1}a^{-1}$ respectively. Similar figures for the Scottish loch and Wuras Dam were 822 kg ha^{-1} of N and 99 kg ha^{-1} of P; 222 kg ha^{-1} of N and 25 kg $ha^{-1}a^{-1}$ of P respectively. Other figures listed by Gallagher and Plumley (1979), Dykyjova (1978), De Jong (1976), Dykyjova and Hradecka (1976), Mason and Bryant (1975) and Kvet (1973) for the N and P removal potential of *Phragmites* ranged from 330 to 880 and 38 to 74 kg $ha^{-1}a^{-1}$ for above-ground biomass respectively.

Chemical composition of soil

Concentrations of Ca, Mg, Zn and Na in the soil in the AEWL were higher with distance from the input (Table 6). Total inorganic N, total P and pH also increased, while electrical resistance and plant available P and K decreased. The cation pattern was probably a result of the solubility and accumulation of

TABLE 5

PO_4 – P AND NH_3 – N + NO_3 – N CONCENTRATIONS OF THE WATER IN AN ARTIFICIALLY ESTABLISHED REED BED (THIS STUDY), A NATURAL STAND IN HYPERTROPHIC WATER (SCOTTISH LOCH) AND IN MESOTROPHIC WATER (WURAS DAM)

Reed stand	Nutrient concentration of the water	
	PO_4 – P ($mg\ \ell^{-1}$)	NH_3 – N + NO_3 – N ($mg\ \ell^{-1}$)
Artificially established	6,300	4,730
Scottish loch (from Ho, 1981)	1,000	4,610
Wuras Dam (from Stegmann, 1982)	0,006	0,080

TABLE 6

THE CHANGE IN CHEMICAL COMPOSITION OF THE SOIL USED IN THE PILOT-SCALE *PHRAGMITES* REED BED FROM INFLOW TO OUTFLOW (ALL VALUES IN mg kg^{-1} EXCEPT pH AND ELECTRICAL RESISTANCE)

Parameter	Inflow	Middle	Outflow
pH	7,20	7,87	8,23
Electrical resistance (ohm)	756	560	279
Plant available phosphorus	53	36	30
Total phosphorus	200	270	290
Total inorganic nitrogen	28	26	16
Ca	1 260	2 200	4 460
Mg	220	232	740
K	304	302	294
Na	124	196	622
Zn	1,1	2,4	4,4

the cations towards the end of the AEWL. The cations may well have combined with PO_4 – P to form insoluble compounds.

Nichols (1983) reported that soluble inorganic P is readily immobilised in soils by adsorption and precipitation with Ca, which was more likely at the end of the AEWL because reactions with PO_4 – P occur mainly under alkaline conditions while reactions with Al and Fe predominate in acid to neutral soil (Nichols, 1983). Such reactions would have occurred in the AEWL closer to the inflow. The increase in total P and decrease in plant available P suggested that the above reactions might have occurred in conjunction with the substitution of P ions within the lattice hydroxyl ions of the clay, thereby becoming an immobile part of its structure (Golterman, 1973). The very energetic sorption of P which ensures near saturation uptake by clay minerals with minimal release of P during storage (Barrer, 1978), may have been another factor in the change of P concentrations within the AEWL. The decrease of total inorganic N suggested that most of the N in the waste water had been lost by denitrification, though organic sediments which have a high affinity for ammonium ions may have retained a proportion of inorganic N.

A decrease in electrical resistance occurred as concentrations of cations increased along the AEWL, suggesting many cations were still unattached in the soil mass.

The potential uses of *Phragmites*

Results from this study indicated yields of up to 70 t $ha^{-1}a^{-1}$ could be achieved with *Phragmites* grown in AEWL systems. The plant has many established uses both in South Africa and elsewhere.

A comprehensive review of propagation, cultivation and exploitation of *Phragmites* was compiled by Verber (1978). The plant is easy to propagate and harvest. The crude-protein content of the leaves in this study was 13,1% and that of the shoot 7,4%. In a stand of natural *Phragmites* of 1 and 2 m high grown in the Karoo, Viljoen (1976) found that the protein content of the shoot was 10,3% (Table 7). In comparison, the protein content of good summer grass in the Free State (red grass, *Themeda triandra*) and hay have a protein content of approximately 12% and 5,1% respectively. The high yield of *Phragmites* in conjunction with the moderate protein content of the shoot, suggests that this crop (grown without artificial fertilizers) could be a major fodder crop

TABLE 7

THE CHEMICAL COMPOSITION *PHRAGMITES AUSTRALIS* AND HAY GROWN IN THE KAROO (VILJOEN, 1976)

P. australis	% of dry mass								
	NFI	protein	fibre	fat	ash	sugar	Ca	P	Mg
1 to 2 m high	40,6	10,30	35,7	2,1	11,4	40,1	0,37	0,11	—
Before flowering 1,5 to 1,8 m high	42,0	7,3	38,4	1,2	11,1	42,0	0,37	—	—
Hay	44,7	5,1	41,4	0,9	7,9	—	0,18	0,05	0,11

NFI – nitrogen free index
P – phosphorus
Mg – magnesium

if harvested before senescence. A comprehensive breakdown of the components of *Phragmites* was compiled by Viljoen (1976) (Table 7).

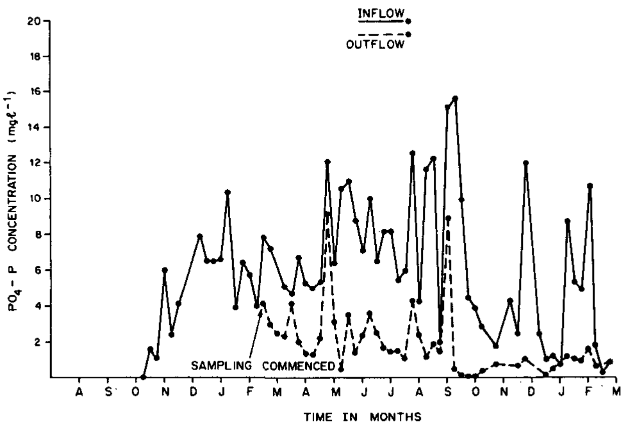

Figure 7
Orthophosphate ($PO_4 - P$) concentrations of the inflow and outflow from an artificially established reed bed system fed by oxidation pond effluent from March 1982 to April 1983.

In South Africa, *Phragmites* are used in Zululand for thatching and hut building. Viljoen (1976) recommended the use of the plant in paper pulp, insulating and shelter material and in medical ointments. In Czechoslovakia, the demand for *Phragmites* outstrips supply (Verber, 1978). Products made from the plant are mainly used in the building industry (insulation, mats, etc.) followed by horticulture (shades and other screens) and by various domestic crafts (Verber, 1978). In Germany, the plant is used in the cellulose and paper industry (Weise and Jorga, 1981). Harvesting of *Phragmites* in Romania has been developed to the extent that thousands of tons of the plant are used annually; 5 000 km^2 of wetland at the mouth of the Danube River are managed to ensure maximum productivity (Sainty and Jacobs, 1981).

Conclusions

The removal of the major nutrients through the AEWL was in the order of 70%. The concentrations of most of these parameters in the effluent were below effluent standards. Although COD concentrations were reduced by 50%, the final effluent concentration remained above the effluent standard. Coliform bacteria were completely removed.

An extrapolated harvest of 70 t $ha^{-1}a^{-1}$ represents a harvest of some 750 kg ha^{-1} of N and 76 kg ha^{-1} of P. The harvestable material had a low shoot to leaf ratio of 2:1, and a protein content of 7%.

The increased concentrations of cations and total available P in the soil mass indicated that precipitation was a major removal pathway of $PO_4 - P$ from the effluent. $PO_4 - P$ appeared to be adsorbed by the clay present in the soil while N appeared to have been lost primarily by denitrification. The relatively robust operations, ease of maintenance and the wastewater removal potential of this pilot-scale AEWL suggest that further studies on these systems should be undertaken in the RSA, amongst others to develop design criteria. The results from a study of this nature may have significant dividends for water quality improvement of effluents in South Africa and for application in areas where technological expertise is lacking.

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