THE POTENTIAL OF NATURAL AND ARTIFICIAL WETLANDS FOR PHOSPHORUS REMOVAL IN THE HARVEY CATCHMENT

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ABSTRACT

An assessment was made of the potential for using various management options, involving wetland filters, to reduce phosphorus loadings into Harvey Estuary. The study involved a literature survey, study of existing wetlands in the Harvey catchment, and examination of phosphorus uptake by experimental wetlands. Up to 90% reduction in phosphorus concentration was achieved at high input concentration and low flow rates. However, the use of artificial wetlands along the borders or outlets of drains was not considered feasible, as high rates of water movement into the drainage system would preclude efficient phosphorus retention by the filter. The use of artificial wetlands at the outlet of certain point sources, e.g. piggeries, is likely to be effective, although further work on design and efficiency of the wetland filter would be required before implementation.

Diversion of agricultural runoff into existing natural wetlands was not recommended as the majority of wetlands in the catchment have higher phosphorus concentrations than the runoff, due to removal of fringing vegetation from swamps in farmland. Overflow from these swamps into drainage canals may provide a significant source of phosphorus to the estuary, although the actual quantity, on a catchment basis, is not known. There appears to be no practicable way of reducing the amount of phosphorus contributed in this overflowing water, which would be effective in the short term. In the long term, reintroduction of fringing vegetation (including sedges and trees) sited around wetlands and at areas of overflow, may be profitable.

1. INTRODUCTION

Eutrophication in the Peel-Harvey estuarine system has been attributed to the high loading of phosphorus from the rivers draining into the system, with the major source (60%) being the catchment of the Harvey River and associated drains (Hodgkin et al. 1980). This catchment, situated predominantly on the Swan Coastal Plain, has soils naturally deficient in phosphorus which are heavily supplemented with superphosphate for agricultural purposes. This supplementation has resulted in large losses of phosphorus into drainage canals (Birch 1980).

A number of methods have been suggested for reducing phosphorus input to the estuary (Hodgkin et al. 1980). These include:-

(i) modifying present techniques of superphosphate application to coastal plain farmland;

(ii) diverting phosphorus-rich water away from the estuary;

(iii) introducing a "biological filter" into the coastal plain drainage system, that is, to use wetland plants to remove phosphorus from drainage water leaving the catchment.

This report is concerned with the third of these options. The use of wetlands or aquatic plants to filter nutrients from water has been successful in many parts of the world, such systems having achieved up to 98% removal of phosphorus and 99% removal of nitrogen from wastewater (Appendix 1). Two factors favour the use of biological filters in the coastal plain catchments:-

(i) the low-lying plain of the Harvey Catchment already contains numerous swamps fringed by wetland plants.

(ii) the area is drained, and the drains may provide focal points for nutrient removal systems.

To study the feasibility of introducing a biological filter system, two major areas were investigated. Firstly the ecology, hydrology and nutrient partitioning of selected wetlands in the Harvey Catchment were studied, to see where and when nutrients might be trapped in the system. Much of the detailed information on this work is provided in a separate report on the Harvey wetlands (Chambers 1983) and only conclusions relevant to the biological filter concept are summarized here. Secondly, the ability of an aquatic macrophyte, common in the area, to take up nutrients was studied in two experiments. These investigations were to provide information about various management options to reduce the output of phosphorus from the catchment. These include:-

(i) fringing agricultural drains with wetland plants;

(ii) the diversion of runoff through existing wetlands;

(iii) the use of artificial wetlands at the outlet of swamps and major drains;

(iv) the use of artificial wetlands at point sources.

Successful biological filter systems reported in the literature have a number of important characteristics including long residence times (low flow rates), shallow water allowing good water/sediment interaction, optimum emergent plant growth, and high nutrient uptake by plants and sediments. The majority of existing biological filter systems are involved in reducing nutrient concentrations from a controlled point source (e.g. sewage effluent). The Coastal Plain catchment, however, involves a wide area of fertilized agricultural land where runoff is much more dispersed and usually of a lower phosphorus concentration than that of sewage effluent.

The catchment is characterized by highly seasonal runoff. Peak runoff is during winter, soon after the fertilization of pastures, which results in high phosphorus losses during a time of rapid water flow in the drainage canals. In addition, the Bassendean sands of the catchment are readily leached and not effective at retaining phosphorus. Each of these characteristics differs from those mentioned above for optimum use of wetland filters, but it nevertheless appeared important to assess the magnitude of phosphorus reduction which might be achieved using artificial and natural wetlands.

2. ARTIFICIAL WETLANDS

2.1 Introduction

Investigations were made into the possible use of wetland plants to remove nutrients from agricultural runoff into drains, from point sources of effluent, and from swamp outlets and major drains. Two experiments were carried out, the first to examine the effect of residence time on phosphorus removal from water in a small section of wetland, and the second to study the effectiveness of *Lepidosperma* and two substrates, peat and sand, to remove phosphorus from water flowing at various rates through the experimental system. The rates of drainage and concentrations of nutrients in the drainage water were monitored by others in the Catchment Studies Group and this information is brought together with the experimental work in the discussion.

2.2 Experiments

Lepidosperma longitudinale (or Common Sword Sedge) was chosen for the experiments. This plant is the most common emergent macrophyte in the wetlands studied (Chambers 1983); emergent sedges were preferred over floating plants for the reasons outlined in Appendix 1. *Lepidosperma* was found to have a high productivity and nutrient content in its natural environment (Chambers 1983), although these properties appeared to be highly dependent on water level during winter.

2.2.1 Methods

Plant material required for the experiments was removed from a swamp located in a reserve, 15 km north-west of Harvey. This wetland (Swamp 3) had the lowest phosphorus loading of the three swamps studied in the Meredith Drain subcatchment (Chambers 1983). Segments of rhizomes with leaves and leaf-like stems (referred to collectively as culms) were excavated and trimmed to 20 cm to reduce transpiration losses. The plants were brought back from the field in plastic bags with much of the original substrate around the roots. Culms brought back in this way grew well under a wide range of phosphorus loadings, although the older leaves tended to die after a few days. No seeds were found in the field, rhizome extension being the major form of propagation.

In the first experiment, a container was made from half of a "44 gallon drum" which had a surface area of 0.25 m^2 (Figure 1). In this container was placed a section of wetland, containing peat and *Lepidosperma*, brought intact from Swamp 3. After being left to establish for three months, 20 l of water were drawn from the bottom of the tub and replaced by 20 l of water from Swamp 1, which had a phosphorus concentration at the time of 9.9 mg l^{-1}. After 3 hr, 5 l was drawn from the bottom of the tub and recycled to the top, a small sample being withdrawn for assay. The procedure was carried out each day. Water lost to evapotranspiration was replaced with distilled water at weekly intervals.

In the second experiment, four treatments were carried out, consisting of *Lepidosperma* planted into peat from Swamp 3, *Lepidosperma* and grey sand of the Bassendean association (collected adjacent to a drainage canal), and peat and sand without plant material. There were two replicates of each treatment. Plants and substrate were initially left to establish for three months, during which the tubs (48 l plastic rubbish bins with a surface area of 0.12 m) were kept waterlogged. The apparatus is shown in Figure 1.

Prior to the addition of experimental solutions, each tub was flushed with distilled water at a rate of 5 l day^{-1} for 6 days. Total phosphorus

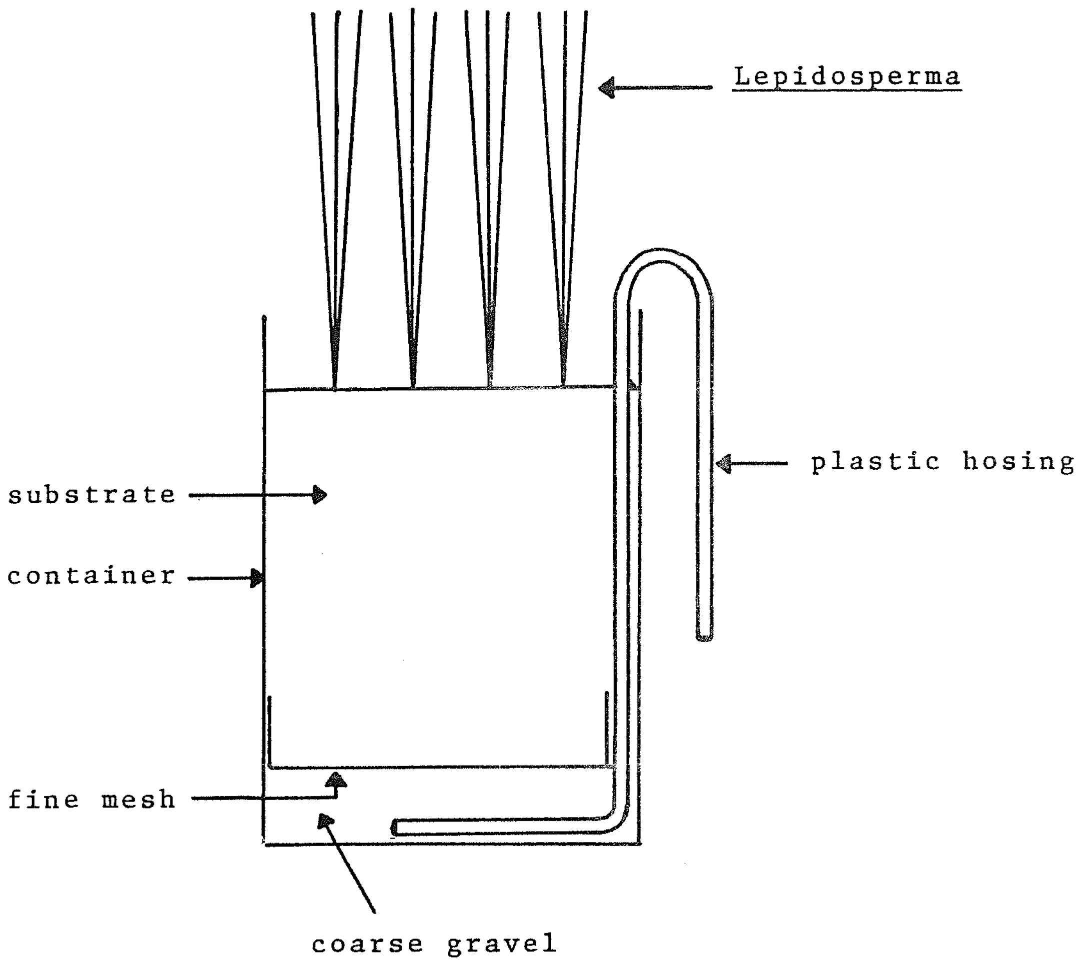


Figure 1 : Design of experimental apparatus.

concentrations were monitored during this time. After this, 2 l of water from Swamp 1 were added to each tub, while 2 l were drawn from the bottom and sampled for total phosphorus. Loss of water by evapotranspiration was replaced with distilled water on each sampling occasion.

This procedure was carried out five times each week for 60 days, giving a flow rate of 11.96 $l\ m^{2}$ day. The mean phosphorus concentration of applied water during this time was 12.78 $mg\ l^{-1}$. Between days 60-161 the sampling was reduced to once per week, giving a flow rate of 2.39 $l\ m^{-2}$ day; the mean phosphorus concentration of applied water was 19.71 $mg\ l^{-1}$. Water was brought from Swamp 1 every two weeks and stored in 20 l plastic containers in a cool dark place. Each solution was sampled for total phosphorus immediately before use.

2.2.2 Results

Figure 2 shows the time course of change in phosphorus concentration in water recycled through the section of undisturbed wetlands contained in the half drum. A 67% reduction in phosphorus concentration occurred within the first week, concentrations being reduced to those present initially in the soil water within 40 days. The total amount of phosphorus taken up in the trial was 792 $mg\ P\ m^{-2}$, and the minimum con-centration reached was 0.12 $mg\ l^{-1}$.

Figure 3 a-d show the time course of change in phosphorus concentration in water passed through tubs containing Lepidosperma and peat (3a), peat (3b), Lepidosperma and sand (3c), and sand (3d). In the initial flushing period (-12 - 0 days) phosphorus levels remained constant in both peat treatments at approximately 0.1 $mg\ l^{1}$, while levels increased slightly in the sand treatments to 0.25 $mg\ l^{-1}$ (Lepidosperma/sand) and 0.45 $mg\ l^{-1}$ (sand).

After application of the phosphorus-rich water there was a "lag-phase"

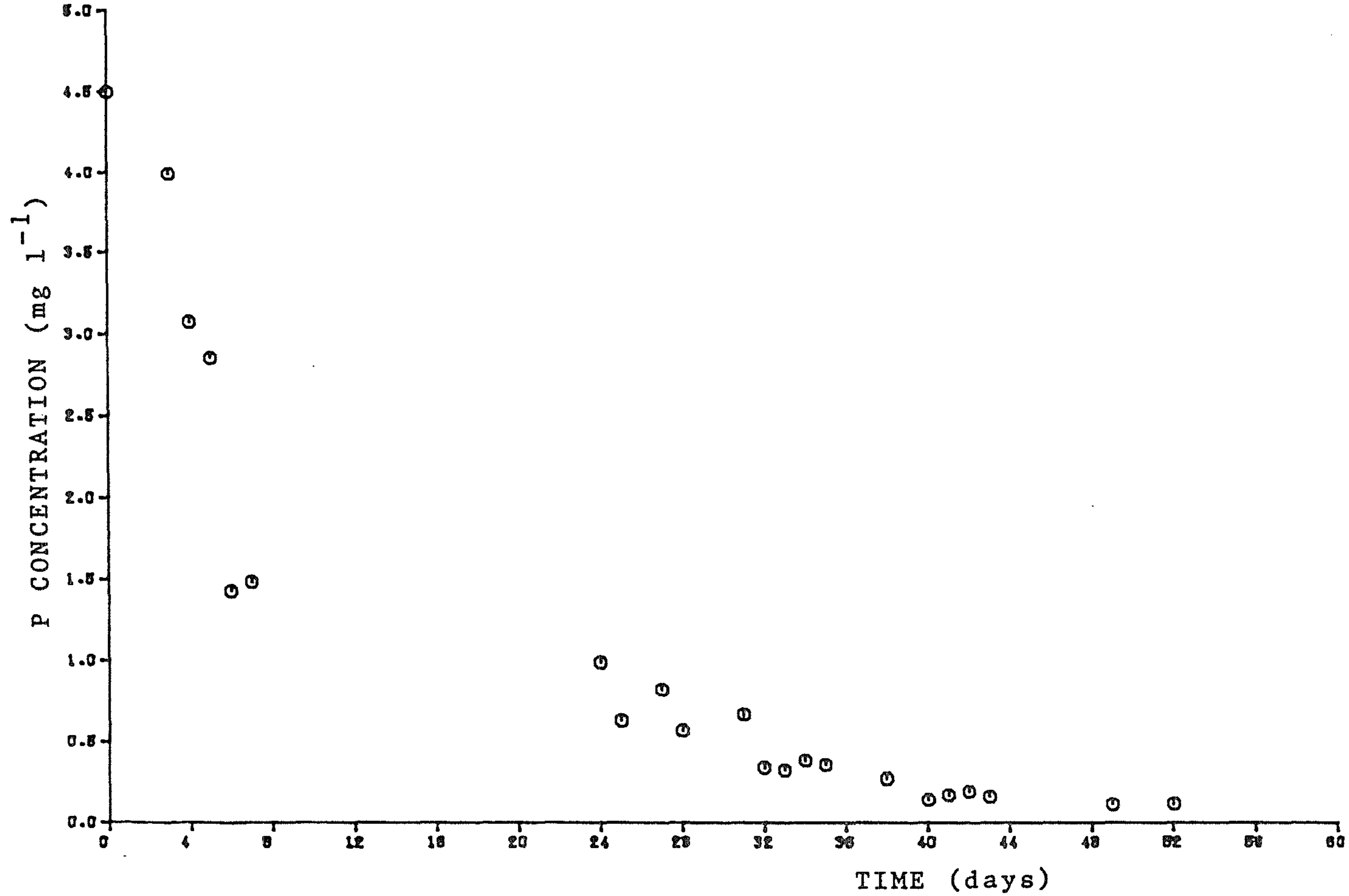
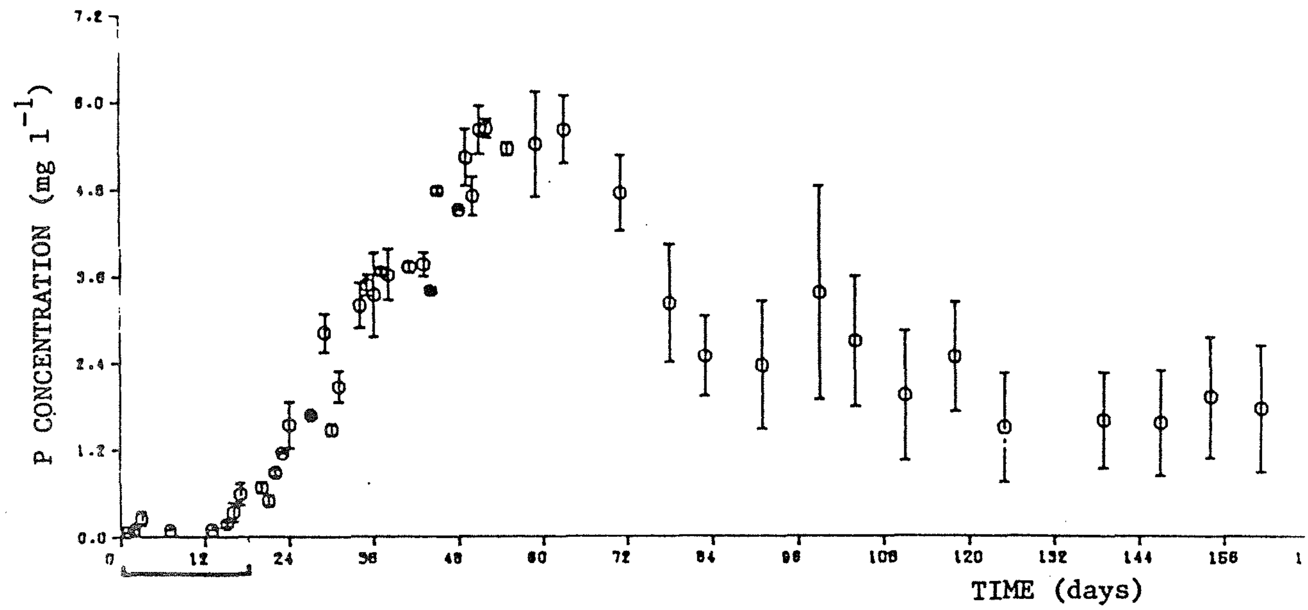
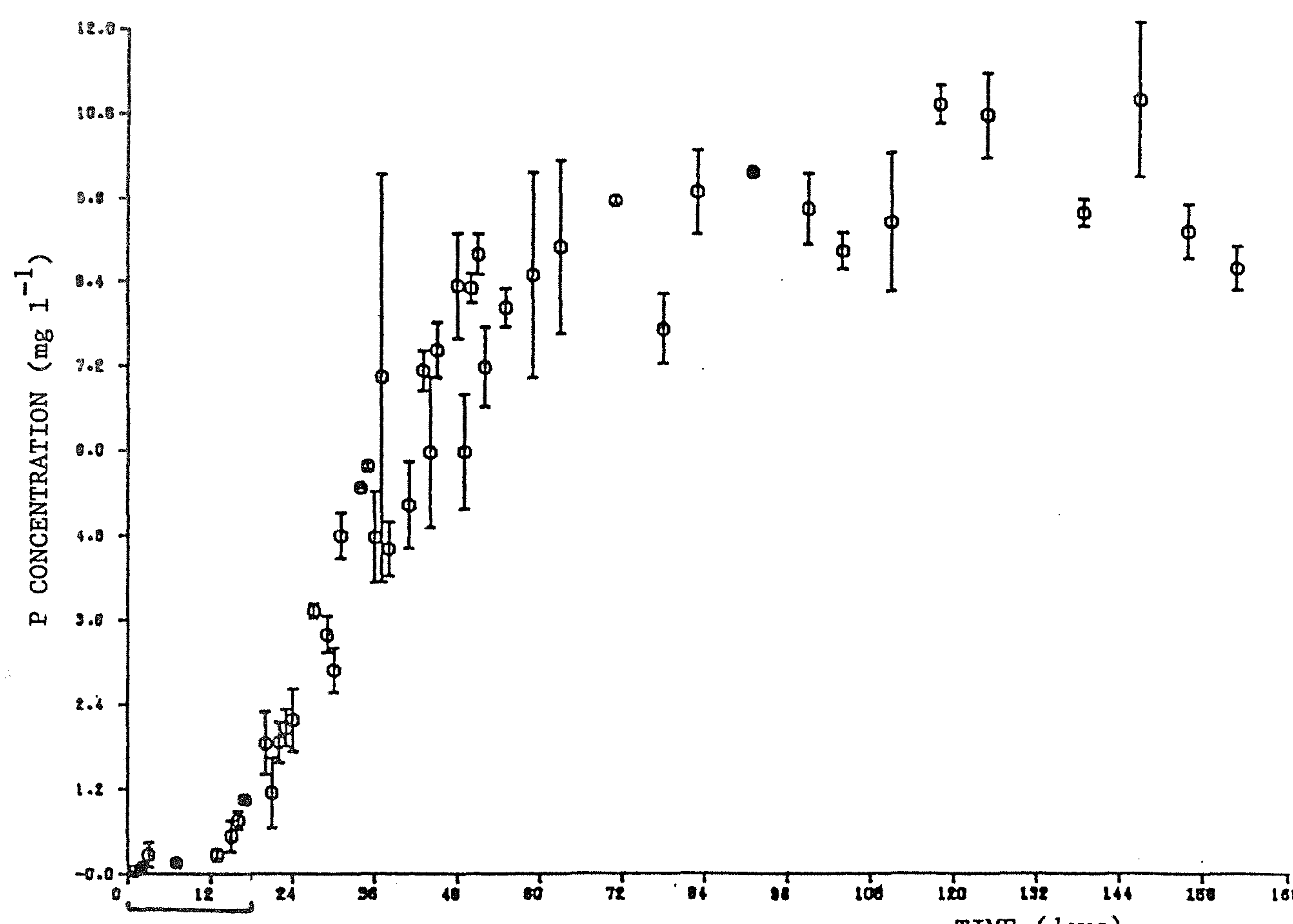


Figure 2: Experiment 1 - Phosphorus concentration of soil water over time.

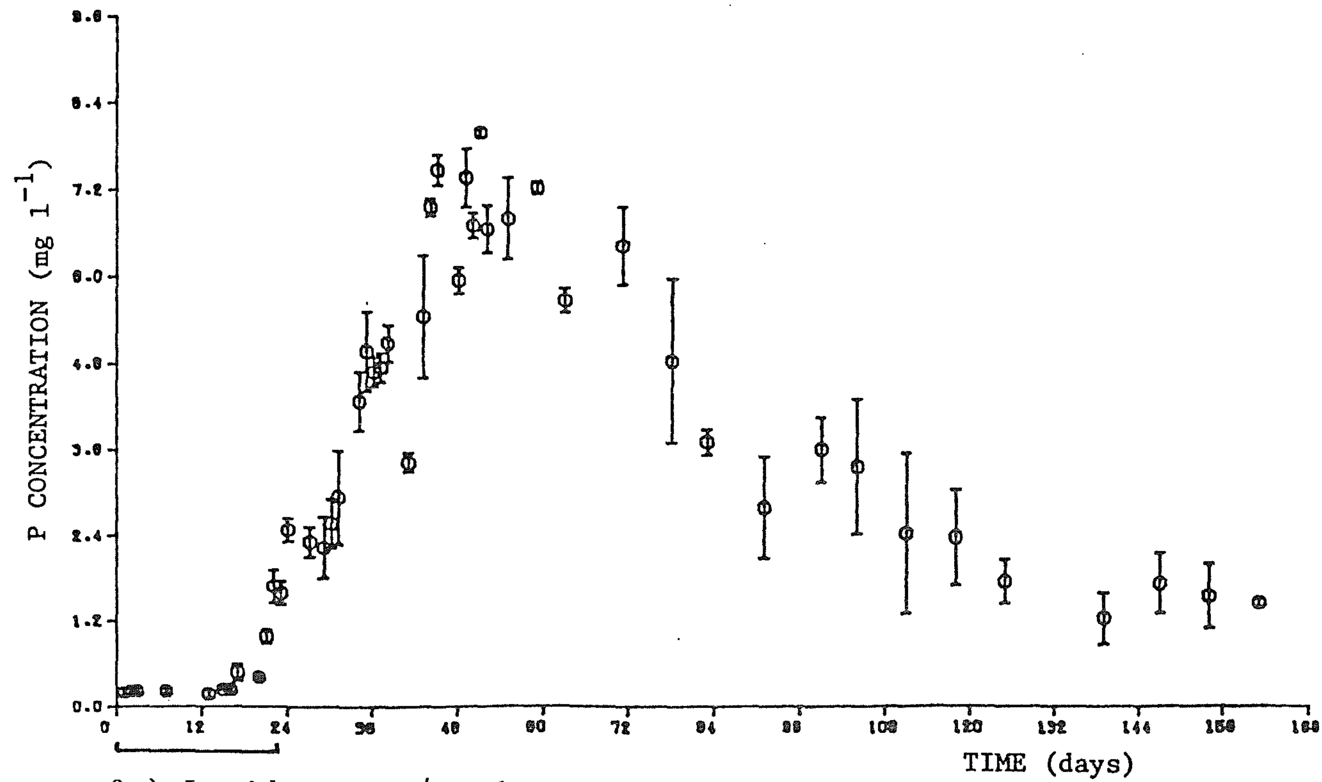


3a) Lepidosperma/peat

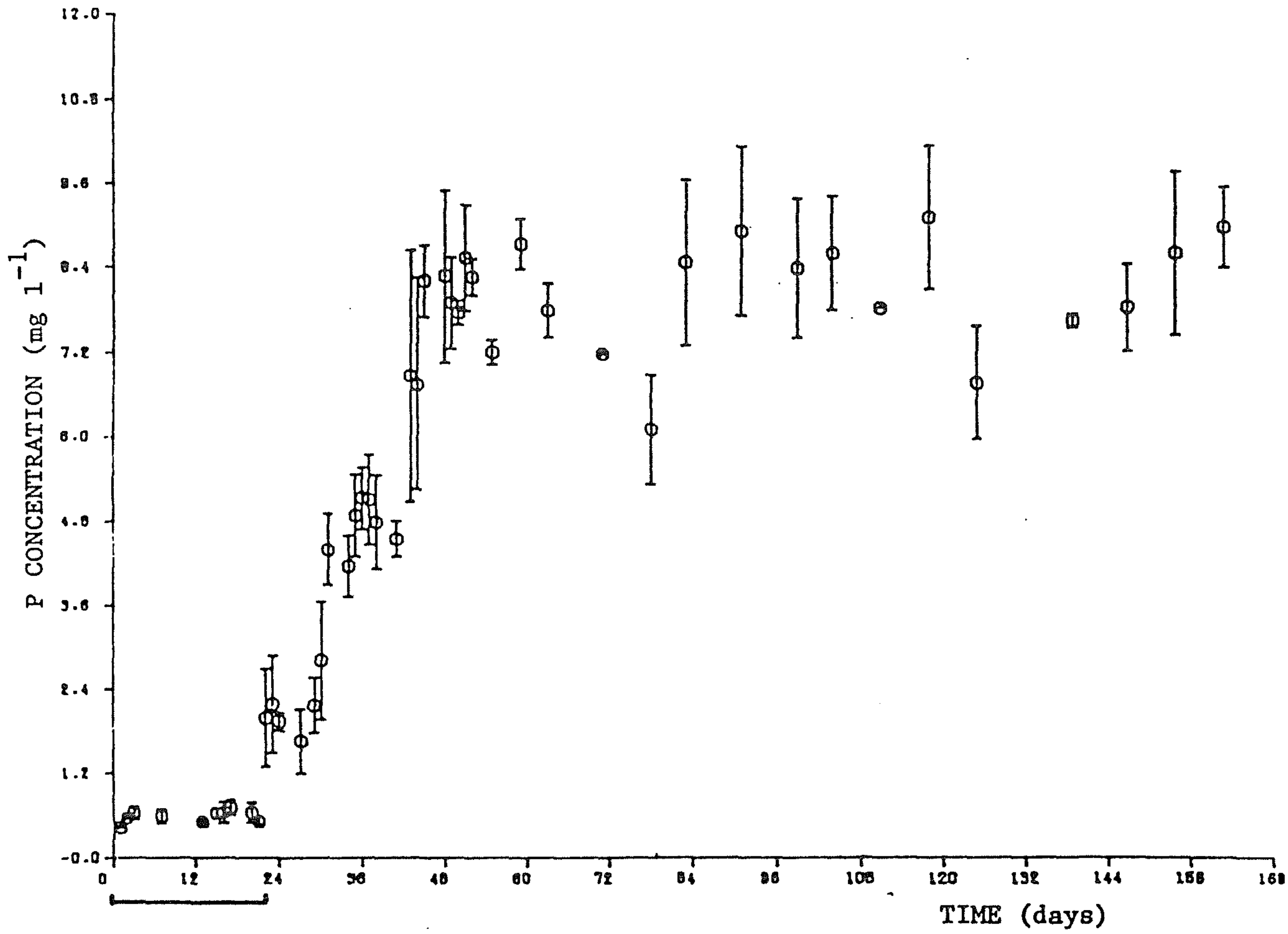


3b) Peat

Figure 3: Experiment 2 - Phosphorus concentration of effluent over time.(Bracket under x-axis marks "lag-phase").

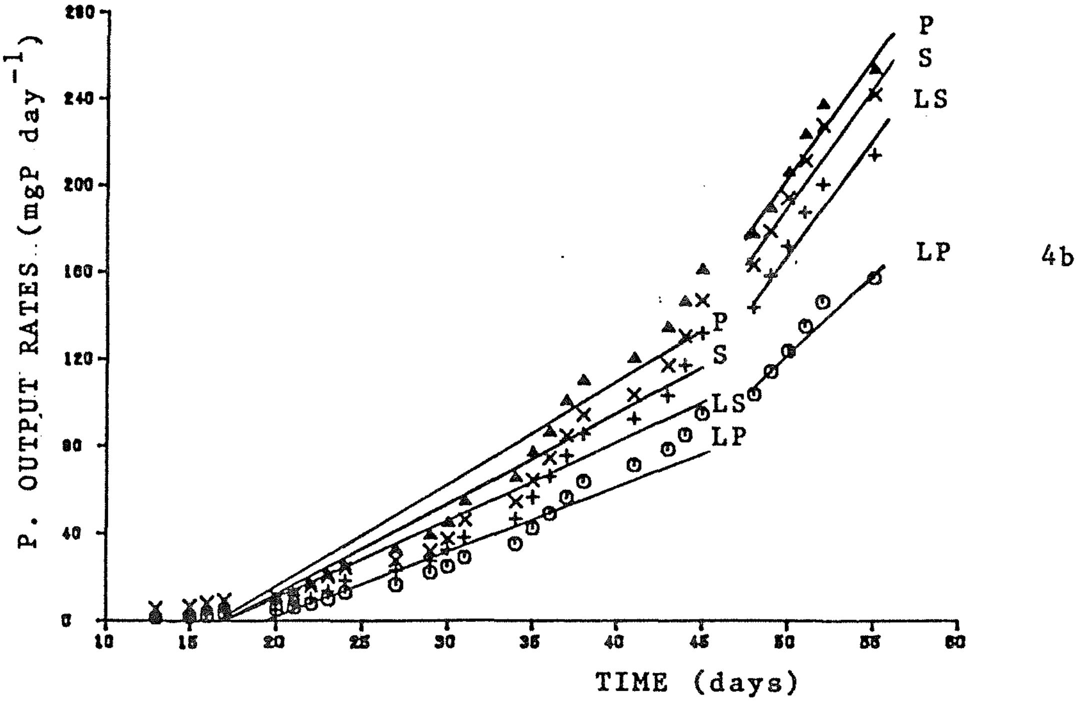


3c) Lepidosperma/sand



3d) Sand

Figure 3: Experiment 2 - Phosphorus concentration of effluent over time. (Bracket under x-axis marks "lag-phase").



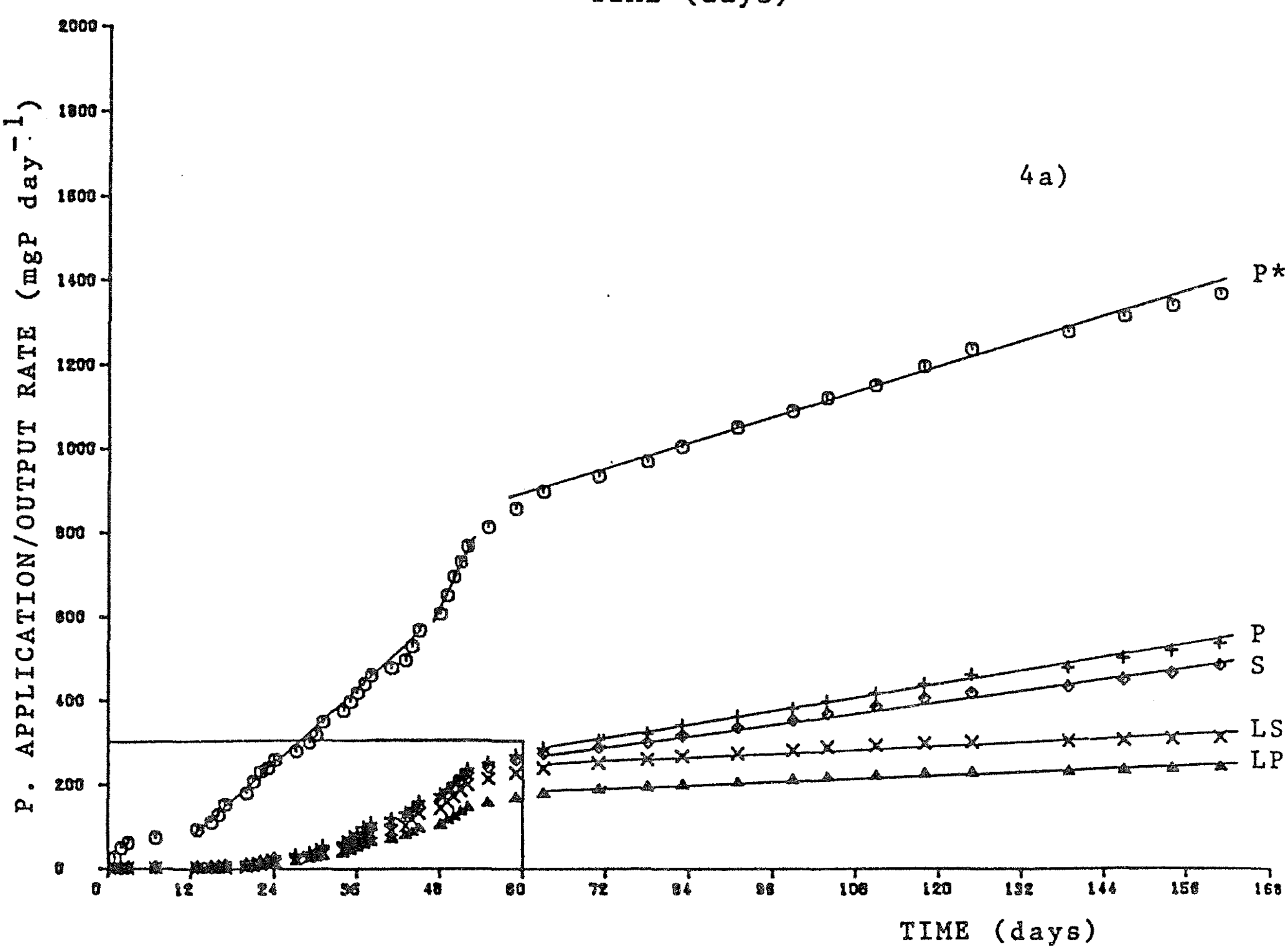


Figure 4: Experiment 2 - Phosphorus application/output rates.

P* = phosphorus applied.

Output rates:
- LP = <u>Lepidosperma</u>/peat.
- LS = <u>Lepidosperma</u>/sand.
- P = Peat.
- S = Sand.

Note: 4b) represents larger scale diagram of area shown in box in 4a).

(All lines r $\geqslant$ 0.95, p = 0.005) of about 18 days in the peat treatments and 24 days in the sand treatments during which phosphorus concentrations did not increase discernably. After this period all treatments showed a linear increase in phosphorus concentration, reaching the concentrations shown in Table 1 in 60 days.

Table 1. Phosphorus concentrations of effluent after 60 days

Treatment	P concentration $mg\ l^{-1}$
<u>Lepidosperma</u>/peat	5.1
<u>Lepidosperma</u>/sand	7.0
Peat	7.7
Sand	8.0

As phosphorus concentrations continued to rise the flow rate was reduced to once-weekly sampling. This resulted in decreasing phosphorus concentrations in both <u>Lepidosperma</u> treatments (r = -0.8356, p 0.001 <u>Lepidosperma</u>/peat; r = -0.9069, p 0.001 <u>Lepidosperma</u>/sand), while the peat and sand treatments showed no significant trends (r = +0.3315, p 0.5-0.2 (peat); r = +0.2012, p 0.5-0.2 (sand). After 65 days at this slower rate, soil water phosphorus concentrations were reduced to 1.66 $mg\ l^{-1}$ for <u>Lepidosperma</u>/peat and 1.53 $mg\ l^{-1}$ for <u>Lepidosperma</u>/sand, while peat and sand treatments fluctuated around 9.79 $mg\ l^{-1}$ and 7.98 $mg\ l^{-1}$ respectively.

Figure 4 shows the cumulative amounts of phosphorus applied to the experimental tubs, together with the cumulative outputs of each of the four treatments. Linear regressions were used to determine the slope of each line; the slope defining the phosphorus application and output rates. Each of the regressions were highly significant (p = 0.005). As changes in phosphorus application rates resulted in immediate changes in output rates

(Figure 4), a direct comparison between inputs and outputs could be made and the percent reduction in the phosphorus concentration of water passing through the tubs could be calculated (Table 2). At each flow rate those treatments containing Lepidosperma showed greater reductions in phosphorus than substrate only treatments, reductions ranging from 74-87% for Lepidosperma and peat to 48-66% for peat alone.

2.2.3 Discussion

The section of wetland used in the recycling experiment showed an excellent uptake capacity, with a residence time of one week removing two-thirds of the applied phosphorus, while longer residence times produced reductions of up to 99%. However, such residence times must be longer than the periods during which water would pass through artificial wetlands in a field situation (see below) unless water were ponded artificially. In experiments of this kind, little information is gained on uptake capacity in the longer term. For example, it may be argued that the experimental wetland was taken from a comparatively oligotrophic lake, and may have had numerous sites available for phosphorus uptake by plants and peat.

The efficiency of a biological filter is determined in part by the nutrient concentration of the treated water. In the second experiment a flow rate of 12.0 $l\ m^{-2}\ day^{-1}$ caused effluent concentrations to increase in all treatments. At the slower flow rate of 2.4 $l\ m^{-2}\ day^{-1}$, however, the effluent concentrations decreased from those shown in Table 1 by 69-79% in the Lepidosperma treatments while effluent from "substrate-only" treatments remained constant. The reduction between input and output concentrations at the slow flow rate was 91-92% for Lepidosperma treatments, and 50-59% for substrate treatments.

If the rate of phosphorus output is expressed as the percentage of the rate of input, then quite high rates of phosphorus retention were achieved

Table 2. A summary of application/output rates and concentrations for Experiment 2.

Applied phosphorus					Output rates* (mg m^{-2} day^{-1})				% Reduction				P concentration of output. Day = 48, 59 & 161 (mg l^{-1})				% Reduction			
Order in which rate was applied	*Phosphorus application rate mg m^{-2} d^{-1}	Flow rate l m^{-2} d^{-1}	Mean P concentration mg l^{-1}	Days of application	Lepidosperma/peat	Lepidosperma/sand	Peat	Sand	Lepidosperma/peat	Lepidosperma/sand	Peat	Sand	Lepidosperma/peat	Lepidosperma/sand	Peat	Sand	Lepidosperma/peat	Lepidosperma/sand	Peat	Sand
1.	120.7	12.0	11.3	13–45	24.3	32.8	41.5	35.3	80	73	66	71	4.5	5.9	8.3	8.3	60	48	27	27
2.	248.3	12.0	19.6	48–55	65.2	86.8	95.5	97.6	74	65	62	61	5.4	7.2	8.5	8.7	72	63	57	56
3.	42.0	2.4	19.7	59–161	5.5	6.4	21.7	18.1	87	85	48	57	1.7	1.5	9.8	8.0	91	92	50	59

*Phosphorus application/output rates determined from linear regressions of data shown in Figure 4.

for nearly all treatments and flow rates (Table 2). The percentage of phosphorus retained in the peat and sand fell with time however, despite decreased rates of phosphorus application, suggesting that these treatments were becoming saturated with phosphorus. The Lepidosperma treatments showed reduction rates which depended on the application rates, the percentage of phosphorus retained increasing when less phosphorus was applied.

Another trend apparent in the results is the difference in the "lag-phase" between sand and peat treatments, which may be explained by different flow rates through the substrates. The sand formed a cohesive mass in the tubs and flow rates were slow (it took 9 ± 0.7 minutes for 2 l of water to be siphoned from the "sand only" treatment). However, flow rates through peat were much higher (0.7 ± 0.1 min siphon rate), suggesting preferential flow through cracks and decreased contact between water and substrate. Lepidosperma had a modifying influence on these flow rates by binding the peat (1.5 ± 0.5 min siphon rate) and by providing root channels through the sand (6 ± 0.9 min siphon rate). The longer "lag times" in sand substrate are therefore attributed to greater contact between water and sand allowing greater initial nutrient uptake when compared with peat.

The difference between Lepidosperma and "substrate only" treatments shows that Lepidosperma is an important component in the biological filter system. Although artificial flow patterns were created in the peat treatments of this experiment and possibly better reduction rates could be achieved in the field, the difference between sand and peat substrates appears marginal.

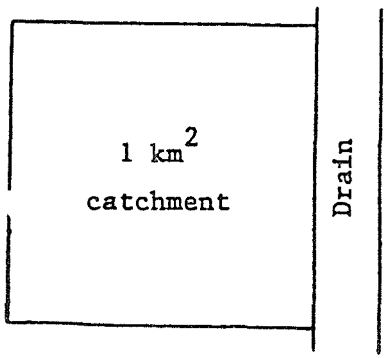
2.3 The possible use of artificial wetlands in the Harvey Catchment

After analysis of experimental and field data, the feasibility of using artificial wetlands to remove phosphorus from the outlets of major drains and point sources, and from runoff into agricultural drains, can be assessed.

Experimental data showed that at a flow rate of 2.4 l m^{-2} day^{-1} and an input concentration of 20 mg l^{-1} a wetland filter consisting of Lepidosperma and sand could remove 92% of applied phosphorus. Flow, rainfall and phosphorus concentration data for the Meredith Drain sub-catchment for the 1982 winter period were obtained from Mr G. Forbes of the Department of Conservation and Environment (cited Humphries and Croft 1983). From these data Humphries and Croft (1983) calculated the volume of flow from various catchment areas under different rainfall events. Using these data an example is given in Figure 5 of the runoff from a 1 km^2 catchment after 10 mm of rain.

It can be seen from this figure that after 10 mm of rainfall, 3100 l of water would cross each metre of drain edge in a day. If the drain were lined by a 2 m strip of wetland plants, the loading rate would be 1550 l m^{-2} day^{-1}, as compared with the flow of 2.4 l m^{-2} day^{-1} found to be satisfactory in the experiments. To achieve a loading comparable to that used in the experiment, it can be estimated that a wetland would have to be 1.3 km wide to treat the runoff from a 1 km^2 catchment after 10 mm of rain!

In addition, concentrations entering the Meredith Drain ranged from 0.25-4.5 mg l^{-1}, as compared with 20.0 mg l^{-1} in the experiment. The phosphorus loading figures would be almost 3100 mg d^{-1} in the field, as compared with 42 mg d^{-1} in the experiment. As noted in Appendix 1, wetland filters are most successful at low flow rates and high concentrations, and this is consistent with the experimental results; in contrast, flow rates in the field are very high and concentrations generally low, suggesting that the possibility of using wetland filters along drainage canals would



* 1 km edge between catchment and drain.
* 10 mm (rain) X 1 km^2 (catchment area) = 10 m^3 (runoff) per metre of drain.
* catchment yield = 31% of which: 26% is surface flow and
  74% is longer term sub-surface flow.
* 31% X 10 m^3 = 3100 l per metre of drain per day,
  of which 806 l is surface flow and
  2294 l is sub-surface flow.

Figure 5: Flow rate of runoff from a 1 km^2 after 10mm rain.

be unsuccessful.

Two further points should be made. The first is that, despite this conclusion, there is very little quantitative information in the literature about the relationship between flow rate nutrient concentrations and uptake rates. It is possible that at high flow rates and low concentrations a wetland filter may be somewhat more effective than the generalization suggests, and this should be investigated. The second point to note is that high flow rates are unlikely to physically damage the wetland filters, as at present the bare sand lining the agriculural drains is coping with flows of 2000-3000 $l\ m^{-1}\ day^{-1}$.

The use of artificial wetlands at the outlet of major drains is discussed by Humphries and Croft (1983). Using the experimental wetland filter flow rate of 2.4 $l\ m^{-2}$ day, treatment of the Harvey Main Drain with an assumed flow of 200,000 ML over a 15-week period, would require a 794 km^2 wetland filter, almost the area of the whole Pinjarra-Waroona-Harvey coastal plain drainage area. Treatment of drains with particularly high phosphorus loadings, e.g. the Meredith Drain, would require an 18 km^2 wetland filter at a flow rate of 43 ML day^{-1}. Once again the phosphorus concentrations of these drainage waters are much lower than the experimental solution and at these very high flow rates neither of these suggestions appears feasible.

Information on point sources collected by Humphries amd Croft (1983) suggests that there are few areas which might be treated by wetland filters. Piggery effluent was the only source recommended for treatment, pigs producing an effluent volume of 3 m^3 pig^{-1} yr^{-1} or 2.9 kg P pig^{-1} yr^{-1}. Of this, about 1% is liquid material, and 99% solid. Effluent from a settling and oxidation tank could be passed by gravity feed through a wetland filter. Croft estimates from the data provided in this report that a piggery containing 2000 pigs would require only a 6724 m^2 (82 x 82 m) wetland filter. Further work on design detail and efficiency of phosphorus retention would be needed before a wetland filter could be implemented.

In summary, artificial wetlands sited at the outlets of major drains and fringing agricultural drains do not appear to provide a feasible method for reducing the amount of phosphorus entering the Peel-Harvey estuarine system. It is likely, however, that small-scale wetlands would be useful in reducing the phosphorus concentration of certain "point source" effluents such as piggeries.

3. NATURAL WETLANDS

The wetlands studied in the Harvey Catchment (Chambers 1983) are characterized by relatively deep lake basins surrounded by a band of wetland vegetation. The lake basins, up to 1.8 m deep, are not suitable for nutrient removal as little sediment/water interaction is possible at such depths. The sediments at the bottom of these lakes are quite anaerobic, which is not conducive to phosphate removal from the water column (Appendix 1).

Any nutrient removal would therefore take place predominantly in the peripheral band of vegetation, and hence the width of this band is important in determining the water quality of the lake. This was exemplified at the three swamps in the Meredith Catchment. Swamp 3, located in a reserve, had relatively low phosphorus concentrations in its water (0.04-0.25 mg l^{-1}) and had a wide band of vegetation surrounding the lake. In winter, the swamp flooded over a large area of vegetation creating excellent conditions for nutrient removal. Swamp 1, however, had very little wetland vegetation and acted as a collection site for phosphorus-rich water (3.5-24 mg l^{-1}).

The exact area of vegetation required to keep nutrients at low levels within wetlands is difficult to determine, being a function of catchment area flow rates and the nutrient status of runoff entering the swamp.

However, from general observation of swamps in the Harvey Catchment a minimum area equal to that of mean lake area is suggested to maintain phosphorus concentrations at an acceptable level (< 0.2 mg l^{-1}). For example a lake of 2 ha mean area would require at least 2 ha of fringing vegetation. This figure is arbitary, however, and more or less vegetation may be required depending on lake geomorphology, and nutrient and flow characteristics of the runoff.

The vegetation fringing a wetland has two roles in maintaining low nutrient status in the lake. Firstly, it intercepts runoff, removing nutrients and reducing the rate of flow via evapotranspiration; and secondly, it lessens the area of swamp catchment subject to fertilization and hence lowers the actual amount of nutrients available to enter the swamp.

In areas where the swamps overflow in winter (e.g Swamps 1 and 2, Chambers 1983) the concentration of phosphorus in the lake water is important. Table 3 summarizes the loss of phosphorus from Swamps 1 and 2. Despite the considerably lower volume of water lost over a shorter time period, Swamp 1 released over twice the amount of phosphorus released from Swamp 2. Although this export of phosphorus may be small compared to that leaving the whole catchment in the drainage system, there is a large number of swamps in the catchment. The nutrient loading and the number of swamps which overflow into the drainage system is not known and would be difficult to measure.

However, a large number of swamps are located on farmland, and the vegetation fringing them has been cleared, resulting in higher phosphorus concentrations in the lake water (Table 4). To reduce these nutrient loadings a number of options have been suggested:-

(i) reintroduce the natural fringing vegetation;

Table 3. Phosphorus export from Swamps 1 and 2

Swamp number	Dates of overflow	Number of days	Estimated volume of output (m^3)	Mean P concentration ($mg\ l^{-1}$)	Total amount P lost (kg)
1.	07/09-05/10/83	28	1,871	8.20	15.34
2.	27/07-05/10/83	39	18,074	0.39	7.05

Table 4. Phosphorus concentrations of wetlands in farmland and reserves.*

Farmland		Reserves	
Swamp number	P concentration $mg\ l^{-1}$	Swamp number	P concentration $mg\ l^{-1}$
1.	10.15	3.	0.22
2.	0.46	7.	0.08
4.	8.80	8.	0.04
5.	1.04	9.	0.15
6.	5.34		
	$\bar{x}$ = 5.16		$\bar{x}$ = 0.12

* This trend was found to be consistent over 1982 sampling period. Example given 21/02/82.

Table 5. Wetland filter area required to treat effluent from Swamps 1 and 2

Swamp number	Mean lake area (ha)	Flow rate during overflow $l\ day^{-1}$	Experimental filter flow rate $l\ m^{-2}\ day^{-1}$	Area of wetland required (ha)
1.	1.5	66,820	2.4	2.8
2.	10	463,440	2.4	19.3

(ii) introduce an artificial wetland at the site of outflow;

(iii) reduce outflow.

Replacement of the natural fringing vegetation would reduce flow and the amount of phosphorus entering the swamp, but the nutrients already present in the lake water and sediments are unlikely to be substantially reduced within the next few years. In the long term, however, fringing vegetation would achieve this aim and improve the ecology of the wetlands.

The area of artificial wetland required to treat the outflow of Swamps 1 and 2 can be estimated using the data outlined in the previous chapter (Table 5). Using the flow rates of the successful biological filter, an area approximately twice that of mean lake area would be required to treat the overflow. However, the water lost from the swamps had a much lower concentration at the time, than that used in the experiment (8.2 and 0.39 mg l^{-1} for Swamps 1 and 2 respectively compared to 20 mg l^{-1} used in the experiment). It is possible that higher flow rates may be used if concentrations are low, so lessening the area required. However, the relationship between concentration and flow rate of solution, and the ability of a wetland to remove phosphorus efficiently, is not well understood. Further work is planned to clarify this relationship.

The final option, to reduce outflow from the swamps, shall only be dealt with briefly. It may be possible, in some cases, to block the site of overflow by filling in nearby drains; however, this is unlikely to be popular with farmers who would lose land to winter flooding. Another option is to increase evapotranspiration by planting trees around and in shallow swamps, thus reducing the volume of water held in the lakes and preventing overflow. In a property in the Meredith Drain catchment, Eucalyptus globulus and E. robusta have been planted reducing the water table by approximately 30 cm after 10 years of growth. Both of these trees provide commercially-viable timber (Croft, pers. comm. 1983). Drying of the swamps brings about changes in the species composition and ecology of the area, although this is of less importance in farmed areas where natural vegetation has already been removed.

The initial concept of directing runoff through wetlands does not appear feasible. Many wetlands are located in farmland and have insufficient fringing vegetation to maintain lake phosphorus concentrations at acceptable levels at the present rate of runoff, without further loading. Monitoring of one swamp which lies on the C-subdrain of the Meredith Catchment has suggested a 50% reduction in concentration between inlet and outlet channels but it is unknown at this stage whether this is due to dilution or swamp storage effects.

In summary, the wetlands of the Harvey Catchment would not be suitable for reducing the nutrient concentration of runoff directed into them. At present few wetlands in the catchment have acceptable levels of phosphorus, as many of them lie in farmland and have had their fringing vegetation removed. They act as collection sites for phosphorus-rich water, which may overflow into the drainage system. No short term solution to this problem has been found, apart from structural alteration of the outflow channels. In the long term, reintroduction of fringing vegetation and further work on the possibility of siting artificial wetlands at points of outflow may be beneficial.

4. CONCLUSION

The aim of the report was to assess the potential for using natural and artificial wetlands to reduce the phosphorus content of water leaving the Harvey catchment. The study involved a literature survey, study of existing wetlands in the Harvey catchment, and examination of phosphorus uptake by experimental wetlands. Up to 90% reduction in phosphorus concentration was achieved at high input concentration and low flow rates. From experimental and field data it was found that the flow rates of agricultural runoff into and within drains were too high to allow efficient nutrient removal by wetland filters sited along the borders or at the outlet of major drains. The use of artificial wetlands at certain point sources, however, does appear feasible, but further work on the design and efficiency of a wetland filter would be required before implementation.

Studies of natural wetlands in the catchment revealed that the removal of vegetation fringing swamps located in farmland, created higher phosphorus loadings than those wetlands located in reserves. As many of the wetlands overflow into the drainage system during winter, reduction of this phosphorus loading was considered desirable. No short term solution to this problem was found, apart from physically blocking the swamp overflow, but long term solutions such as reinstating the fringing vegetation and planting trees to reduce the water level were considered feasible.

The use of artificial wetlands to intercept swamp overflow was considered. Using the flow rate of the successful experiment biological filter an area twice that of mean lake area would be required. However, the water lost from the swamps is generally of a much lower phosphorus concentration to that in the experiment and possibly higher flow rates than that of the experiment may be used in the field. This would lessen the area of wetland filter required. As the majority of wetlands in the Harvey catchment already have a high phosphorus loading (> 0.2 mg l^{-1}) the possibility of directing runoff through these wetlands to remove phosphorus would not be effective.

The effectiveness of wetland filters at high flow rates and low nutrient concentrations is generally considered to be low. Despite this there is very little quantitative information in the literature about the relationship between flow rate, phosphorus concentration and the efficiency of a wetland filter. Further work on this point would be profitable, together with detailed work on wetland filter design for certain point sources.

5. ACKNOWLEDGEMENTS

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APPENDIX 1

THE USE OF WETLANDS AND VASCULAR AQUATIC PLANTS TO REMOVE NUTRIENTS FROM WATER.

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INTRODUCTION

Growing concern over the eutrophication of lakes and estuaries around the world has stimulated considerable research into ways to alleviate eutrophication. The major cause of the problem is the large input of nutrients by agricultural, industrial, and urban effluents, which causes algal blooms, bacterial growth, and a general deterioration of the lake environment.

This essay discusses the possibility of using vascular aquatic plants and wetland ecosystems to remove nutrients from urban, industrial, and agricultural effluent before it enters the water body. Attention is focused on using plants and wetlands to remove nutrients from sewage effluent, as this is a major point source of pollution, but similar systems might be used for lining agricultural drains and lakeside margins, and in treatment plants for industrial and cattle feedlot effluent.

The nutrients of particular concern are those which may limit plant growth in lakes and estuaries, since reduction in the level of 'limiting' nutrient will reduce the biomass of plants in the receiving water body. Nitrogen and phosphorus are of particular importance and, of the two, phosphorus is of critical importance because a variety aquatic of organisms can "fix" nitrogen from the atmosphere. There is no comparable source of phosphorus, from which plants might make up a relative deficiency.

The following account examines the efficiency with which natural wetlands, or artificial systems containing one or two species of plants, remove nitrogen and phosphorus from water.

AQUATIC PLANTS AS NUTRIENT FILTERS.

Most of the literature dealing with the use of individual species or genera of plants as nutrient filters results from work on tertiary treatment of sewage effluent. Although other areas of high nutrient output exist, such as cattle feedlot effluent and fertilized agricultural runoff, sewage effluent is probably the easiest to manage experimentally.

Conventional wastewater treatment is not primarily involved in the removal of plant nutrients, but rather the reduction of biochemical oxygen, demand (BOD), chemical oxygen demand (COD), suspended solids and coliform bacteria. However, conventional treatment processes are biological, and plant nutrients are essential for various metabolic functions which take place throughout the system. Both the trickling filter and activated sludge secondary treatment processes use micro-organisms to remove oxygen-demanding materials and waste constituents, which are then adsorbed onto filter beds. In general, primary and secondary treatment can be expected to remove 20-50% of nitrogen and phosphorus from raw wastewater. The remaining 50-80% is more difficult to remove, and much effort has been expended to find economical, reliable means of reducing this nutrient content before effluent is discharged into natural waters (Harvey and Fox 1973).

As aquatic plants are known to absorb large amounts of nutrients, considerable research has been done to see if such plants, when introduced to artificial systems, can be used to reduce the concentration of nutrients in conventional wastewater effluent.

Boyd (1970) has outlined a number of criteria which would suggest a plant species useful in the economic removal of nutrients from wastewater effluent. They are:-

1) Large standing crops per unit area - this allows efficient use of space i.e. maximum nutrinet uptake per unit area.

2) Rapid growth rate - rapid growth requires a large amount of nutrients over a short period of time.

3) Accumulation of large quantities of nitrogen and phosphorus, as these nutrients are the most important in eutrophication.

4) Readily harvestable - so that the nutrients in the biomass of the plant can be removed easily and without decrease in the rate of nutrient removal from the system; ie, the species must be able to withstand harvest.

5) Some economic value - this is useful in offsetting the cost of harvest and upkeep of the plants. Many aquatic plants can be used as feed for animals, for example.

Seasonality of plant growth, is also important. If plants have very seasonal growth patterns, then nutrient uptake may be confined to certain periods during the year. This may be especially true of exotic species such as tropical Eichornia crassipes transferred to cold or temperate climates.

Another important factor is the level to which the species can reduce the nutrient concentration of the water. Clearly, the more efficient a plant is at removing nutrients, the lower will be the residual nutrient content of the effluent. Other factors which determine the choice of a plant is whether it is toxic and whether it has many pests. (Culley and Epps 1973).

A number of studies have been made to discover plants which fulfill these criteria. Those studied to date can be divided into four groups:

1/ Floating plants - Lemnaceae

2/ Floating plants - water hyacinth (Eichornia crassipes)

3/ Submerged plants and,

4/ Emergent plants

Each of these groups will be discussed in turn and then compared.

1) Floating plants - Lemnaceae.

The prolific growth of the Lemnaceae family in eutrophic lakes pointed to the potential of the plant for nutrient removal from sewage effluent. The major species studied are Lemna minor (Harvey and Fox 1973), Lemna gibba (Sutton and Ornes 1975), Spirodela polyrhiza (Sutton and Ornes 1977), and Spirodela oligorrhiza (Culley and Epps 1973). Some work has also been done on Wolffia sp. (eg. Harvey and Fox 1973 studied W. columbiana), but what little information is available does not suggest that this species is particularly useful for wastewater treatment, and it is not considered further here.

The standing crop of Lemna and Spirodela spp. is quite high, although mixing of different species tends to increase the standing crop per unit area as compared to single species stands. Culley and Epps (1973) found that Spirodela oligorrhiza never occurred as a layer more than one centimetre thick on the water surface (except when windblown), but mixed stands involving some of the submerged Lemnaceae occupied two centimetres of the top layer, doubling the biomass available for nutrinet removal. Sutton and Ornes (1975) used a mixed stand of Lemna minor and L. gibba in their experiments.

The growth rate of the Lemnaceae is rapid. Harvey and Fox (1973) calculated that L. minor doubled its frond number, and hence area, every four days but, as Sutton and Ornes (1975) point out, this required a continual supply of sewage effluent. In their experiments in static sewage effluent, they found a maximum growth rate of 4 m^{-2} day^{-1} during the second to third week of growth; the lowest was 0.66 g m^{-2} day^{-1} found during the seventh to eighth weeks, when the nutrient concentration was diminishing due to weekly harvests. A continuation of growth under conditions of this study would result in an annual productivity of 240-1460 g m^{-2} $year^{-1}$. Growth of the duckweeds was closely related to the phosphorus content of the sewage effluent. Spirodela polyrhiza was found to increase its initial biomass of 1.7 g DW m^{-2} to 13.5 g DW m^{-2} over a period of seven weeks followed by a general decline. This gives a maximum growth rate of 1.9 g m^{-2} day^{-1} and a minimum of 0.5 g m^{-2} day^{-1} (mean of 1.2 g m^{-2} day^{-1} over the twelve week study period).

Abdulayef (1969 - cited Harvey and Fox 1973) found a summer productivity of 27-36 metric tons per hectare for Lemna minor. He also found L. minor to grow vegetatively at 1-3 ^{o}C, and so it 'over winters' well. As it is a floating plant, ice forming on the water body would stop its production and hence nutrient removal capability, but it does survive until the following summer.

The nitrogen and phosphorus absorption properties of the Lemnaceae are excellent. Lemna minor grown in sewage effluent for more than on month contained 4.6% nitrogen and 0.8% phosphorus on a 70% dry weight basis. All other nutrinets were either average or above average when compared to terrestial plants. Dry weight of Lemna minor per acre (0.4 ha) of wastewater effluent was approximately 101 kg of nitrogen and 0.81 kg of phosphorus. Sutton and Ornes (1977) found that the concentration of phosphorus in Lemna minor and L. gibba tissues was directly related to the concentration of phosphate in the sewage effluent. Culley and Epps (1973) found up to 2.84% of phosphorus in Spirodela oligorrhiza and hence this species may be even more effective at removing phosphorus from enriched water, († 2.1 g P ml^{-1}). In experiments done by Harvey and Fox (1973) the effect of Lemna minor on the wastewater concentrations of Kjeldahl nitrogen, nitrate, nitrite, and phosphate was studied. Ammonia was not included. The results of these experiments can be seen in Figure 1 (a - d).

Total nitrogen concentratins in the control tank containing no Lemna minor were found to increase between 27 to 55% over a ten day period. This increase was thought to be due to algae and bacteria dying in the tank and releasing nutrients. The tank containing Lemna minor, however, showed 75 - 89% removal over the ten day. (Figure 1). The reduction of Kjeldahl nitrogen over the ten day period can be seen in Figure 1d.

Nitrate showed a 15 - 89% increase in the control run as compared to a 21 - 60% decrease for the Lemna minor tank. The percent reduction of nitrate can be seen to be less efficient than that of Kjeldahl nitrogen. Nitrite was found to increase in the control, probably as a result of the oxidation of ammonia present in the tank. The test run also increased by 340-460%, but nitrite concentrations were always lower than 1.0 g l^{-1}.

Phosphate showed a decrease of 18-21.5% in the control tank, probably due to adsorption on containers. The test run showed a further decrease of 56-81%, (Figure 1). By subtracting the control data from the test data a real Lemna minor removal rate could be calculated, although Harvey and Fox (1973) think that in the natural environment phosphorus would be adsorbed onto substrate surfaces so the control tank loss of phosphate is not significant. The rate of phosphate reduction over the ten days can be seen in Figure 1d.

Sutton and Ornes (19875) found a 90% decrease in the 2.67 g/ml of phosphorus sewage in four weeks using Lemna minor and L. gibba. A final concentration of 0.08 g/l was achieved after eight weeks, although over the last four weeks the reduction rate was very slow removing only another 7%, to make an overall reduction of 97%. The rate of removal of phosphorus was dependant on the amount of duckweed, the concentration of the effluent, and the length of contact time.

Sutton and Ornes (1977) have also done experiments on *Spirodela polyrhiza* in static sewage effluent. They found that phosphorus, which averaged 3.53 g ml^{-1} at the beginning of the experiment, was reduced to 0.09 g ml^{-1} at the end of twelve weeks, representing a 97% decrease. Statistical analysis found that the amount of phosphorus abosorbed in plant tissue correlated with that in the effluent at the 1% level. Such analysis found the uptake of phosphorus by *Spirodela polyrhiza* to be directly related to the phosphorus content of the sewage up to an estimated 2.0 g ml^{-1} (ie. *S. polyrhiza* would take up phosphorus until the effluent reached this phosphorus concentration). An overall removal rate of 8.8 mg P m^{-2} day^{-1} was calculated for the study, with a maximum of 14.9 mg P m^{-2} day^{-1} for the twelfth week.

Duckweeds could be harvested by a modification of the skimmer systems used by petrochemical industries (Culley and Epps 1973).

The rate of harvest could be 50% of the total biomass every four days, when grown under optimum conditions, allowing the removal of 50 kg of nitrogen and 0.4 kg of phosphorus every four days (Harvey and Fox 1973) Harvest encourages nutrient uptake by keeping the population in a constant state of growth. More nutrients are absorbed at this stage than when the population has become established (Boyd 1970).

After harvesting, *Lemna* becomes as excellent source of stock food, the main problem being a very high water content (95%). So far no economically-viable means of drying *Lemna* has been described.

Of the dry weight, 87% is organic matter, of which 64% is digestable. (Pasture grass has 50-60% digestable organic matter). Lemna minor contains 30-35% albumins, 4-5% raw fat, and 30-35% starch and is rich in vitamins E, B_1, B_6 and carotene. Muzafanou et al. (1968 cited Harvey and Fox 1973) found duckweed to have 12-14% more protein than wheat and 18-19% more than corn, so it can be considered very good green vitamin feed for stock, which offsets the cost of harvesting it.

Possible problems concerning the growth of duckweeds to remove nutrients from wastewater include providing a breeding site for mosquitoes, and harbouring toxic micro-organisms.

Culley and Epps (1973) found that a dense growth of Spirodela oligorrhiza prevented mosquito reproduction by preventing the larvae from reaching the surface to obtain oxygen. Regrowth after each harvest was rapid enough to cover the pond in three to four days.

Viruses and bacteria are associated with wastewater and studies are needed to determine if pathogenic forms might be present in harvested and processed duckweed. Toxic blue-green algae may be assoicated with lagoon wastes and become established when harvest allows light penetration. Such algae could pass into animal feed if processing does not remove or destroy the toxicant.

Culley and Epps (1973) report that few pests attack duckweed. For effective growth and nutrient removal the ponds would have to be sheltered, as the plants are very easily windblown into heaps around the edge of the ponds.

2) Floating plants - Water Hyacinth (Eichornia crassipes)

The water hyacinth (Eichornia crassipes) is probably the most researched aquatic plant to date, with respect to nutrient removal from wastewater. As early as 1948, Dymond (cited Cornwell et al. 1977) suggested using hyacinths to remove nutrients from wastewater, and since then a variety of nutrient removal schemes have been studied and developed.

Despite the amount of literature, there is little information on size of standing crop. Boyd (1970) gave a theoretical maximum standing crop of 12.8 metric tons ha^{-1} (dry weight), but considered this to be a conservative estimate. Westlake (1963 - cited Wooten and dodd 1976) calculated a seasonal maximum biomass of 1473 g m^{-2} (or 14.73 metric tons DW ha^{-1}) for hyacinth growing under natural conditions in Louisiana. Wooten and Dodd (1976) estimated a total wet weight of 645 metric tons ha^{-1} or an oven dry weight of 29.7 metric tons ha^{-1} for hyacinths growing in treated wastewater effluent. This is considerably higher than Boyd's (1970) and Westlake (1963) estimates and significantly higher than that of crop plants (Boyd 1970). Cornwell et al. (1977) noted that hyacinths grown in effluent grow much larger after they have covered the surface water of the pond, as compared with their counterparts in natural waters, and this may contribute to the higher standing crop.

Eichornia crassipes grows rapidly. Cornwell et al. (1977) noted an area doubling time of 6.2 days, while Wolverton and McDonald (1976) found a doubling time of 8-10 days. Production figures for hyacinth growth vary, depending on climate and nutrient availability (Table 1). Note that figures for hyacinth grown in effluent are far higher than those in natural environments. Scarsbrook and Davis (1971) compared directly the growth of *Eichornia crassipes* grown on sewage effluent and natural waters, and found that hyacinth grown in ordinary water produced 57 g $pool^{-1}$ in a 23 day period (pools were approximately $6m^2$) whereas plants grown in effluent produced 235 g $pool^{-1}$ in the same period.

Eichornia crassipes is a tropical plant, and winter productivities in temperate climates are very low. Frost kills the leaves, but unless the water freezes, viable rhizomes will respond to warm water ($10^{o}C$ or more) by producing new growth. The rhizome is protected from frosts by shields of folding leaves. Water hyacinths grow best at water temperatures above $21^{o}C$ but die within a few hours as water approaches freezing temperatures (Dinges 1978). The death of such plants would cause a release of their assimilated nutrients to the water body when temperatures were sufficiently warm for active decay.

Nutrient assimilation by water hyacinth has been studied on a number of occasions, with diverse results. Dymond (1948, cited Cornwell et al. 1977) found the plant to contain 2% nitrogen and up to 8% phosphorus. Penfold and Earle (1948) and Westlake (1963, both cited Steward 1970) found hyacinths in natural waters to contain 4% nitrogen and 0.4% phosphorus, Boyd (1970) cited values of 3% nitrogen and 0.4% phosphorus, while Wolverton and McDonald (1979) found water hyacinth grown in sewage to contain 3-4% nitrogen and 0.4-1.0% phosphorus. These variable results may be due to different ambient concentration of these nutrients in the water. Rogers and David (1972), Scarsbrook and David (1971), and Rogers and David (1972) found that a five-fold increase in the nutrient concentration of the water caused a three-fold increase of both nitrogen and phosphorus in the tissue of water hyacinth.

The reduction of the nutrient concentration of water by *Eichornia crassipes* depends on a variety of factors including the plant biomass of (Sutton and Ornes 1975), the length of contact time, the depth of the tank, and the initial concentration of nutrients in the effluent. Water hyacinth seems more efficient at removing nitrogen than phosphorus, giving an average of 70% reduction for nitrogen as compared to 40% for phosphorus (Table 2). (Cornwell et al. (1977) points out that nitrogen is often the limiting nutrient in sewage, and hence nitrogen is taken up more readily.

The influence of these variables on the nutrient reduction potential of Eichornia crassipes can be seen in Table 2 and 4. Cornwell et al. (1977) show the importance of contact time in Table 2 and 3. Very short contact (or retention) times result in actual export of nutrients (Table 2; 6 & 11 hours), whereas longer times, up to four days, improved the quality of the tank effluent (Table 3).

The depth of the tank defines the amount of water in actual contact with the plant, deeper water allowing a passage below the root systems of the hyacinth mat. The water can then pass out of the tank without being processed. The depth of the ponds should not be much deepter than the length of the root systems. Cornwell et al. (1977) noted that water hyacinths grown in fertile systems had very short roots extending only 50 mm below the water surface. Normally hyacinth root systems extend 0.3m below the surface and this would be an excellent depth for nutrient removal ponds. Table 3 shows that if ponds are kept shallower, shorter effective retention times are possible, and hence the system is more efficient.

The initial concentration of the wastewater determines the level of nutrients in the tank effluent. Rogers and Davis (1972) found that at normal nitrogen and phosphorus levels for sewage, 22 mg l^{-1} and 3.7 mg l^{-1} respectively, E. crassipes was quite efficient at removing the nutrients, producing a 46% reduction in nitrogen and a 93% reduction in phosphorus (Table 4). At very high levels (96 mg N l^{-1} and 16 mg P l^{-1}), *E. crassipes* actually exported nitrogen and took up phosphorus at a very slow rate (2.6% after four days; Table 4).

Sinclair and Forbes (1980) point out the necessity of harvesting hyacinths for optimum nutrient reduction, a practice not carried out by the researchers listed in Table 2. Harvest keeps the population at a continual growth state which requires a larger amount of nutrients than when the population is established (Boyd 1970) and also prevents dead and decaying hyacinths from releasing nutrients into the water column (Sinclair and Forbes 1980).

Harvesting of *E. crassipes* requires some sort of heavy-duty skimmer, or possibly the plants could be grown in removable mesh traps. Uses of the plant, once harvested, include stock food, production of methane gas and fertilizer and compost (Wolverton et al. 1976), the major problem being the high water content (approximately 95%; Boyd 1970).

Problems assoicated with the use of this species as a nutrient filter include possible mosquito problems, odour, and the fact that *E. crassipes* is a noxious weed. Cornwell et al. (1977) did not find mosquitoes a problem during their studies, although they could see no reason why mosquitoes did not breed in the ponds. Wolverton and McDonald (1979) found that the anaerobic nature of hyacinth ponds could result in odour problems due to the activities of anaerobic micro-organisms. They suggest mechanical aeration of the ponds during photosynthetically inactive periods to combat this problem. The fact that *E. crassipes* is a noxious weed has led Mitchell (1978) to recommend against its use in Australia. The properties which make *E. crassipes* a good plant for nutrient removal schemes also allow its rapid spread through natural ecosystems, causing serious damage to them.

3) Emergent Plants

The use of emergent plants in nutrient removal systems was pioneered by Seidel (1976, 1978) in West Germany, and the majority of work in this field is still carried out in Europe.

In a subsequent section of this essay the use of natural marshes for nutrient removal will be discussed in detail, and such systems include emergent plants. This section will deal only with artificial systems containing emergent plants. In artificial systems the plants are grown in a suitable substrate (eg. gravel), and effluent is passed through this substrate, interactings with the substratum, roots and rhizomes of the plants and associated micro-organisms. The processes of nutrient removal and the importance of micro-organisms in these systems will be covered in the subsequent section.

The main genera of emergent plants used in nutrient removal systems are *Typha*, *Scirpus**, and *Phragmites*, although *Iris* and *Phalaris* have also been investigated. Table 5 shows that the standing crops and productivities of these plants are high and compare favourably with *Eichornia crassipes*. It must be noted that most of these figures are based on natural systems in cool, temperate climates and represent only above ground data. Fertilized systems in warmer climates may produce better growth rates and standing crops.

The amount of nitrogen and phosphorus contained in these plants is somewhat lower than those for the Lemnaceae and water hyacinth (Table 5), but this lower nutrient concentration (and uptake potential) is though to be balanced by the nutrient removal activities of the microbes associated with emergent plants. The combined nutrient removal potential of

* Footnote - now *Schoenoplectus*

emergent plants and their microbes is generally thought to be higher than those of floating plants (Seidel 1976, de Jong 1976).

Existing wastewater treatment systems containing emergent plants vary in their efficiency to remove nitrogen and phosphorus. De Jong (1976, 1977) found a 98% reduction of phosphorus in effluent from Scirpus lacustris and Phragmites australis holding ponds. These high reduction figures were also found for a pond containing no vegetation, during the summer. the role of the plants in nutrient removal systems is not clearly understood, but possibly they are important in long term nutrient removal as a growing sink for nutrients. Both living plants and litter are capable for assimilating nutrients over a long period to time, whereas a purely sedimental system would become saturated relatively quickly.

Woodwell (1977) found up to a 91% reduction in nitrogen and a 98% reduction in phosphorus when effluent was passed through a Phalaris arundinacea meadow. Spangler et al. (1976) found Scirpus validus to remove 81% of phosphorus, S. acutus to remove 84% phosphorus, and Iris versicolor to remove 80% of phosphorus from effluent after a five day retention time. Finlayson & Chick (1982) found a Scirpus validus system to reduce nitrogen by 74% and phosphorus by 79%, Phragmites australis to reduce nitrogen by 62% and phosphorus by 68%, and Typha (domingensis and orientalis mixture) to reduce nitrogen by 42% and phosphorus by 68%. They found that Scirpus and Phragmites oxygenated the effluent, making conditions suitable for nitrification. This not only removes ammonia, but produces nitrate which can later be denitrified and removed from the system. Pope (1981 - cited Finlayson & Chick 1982) found this process to remove a large quantity of nitrogen from systems containing Phragmites australis and Scripus lacustris, and Kickuth (1976 - cited Finlayson & Chick 1982) attributed 70% of the 97% reduction in nitrogen in a system containing *Phragmites vulgaris*, due to denitrification. Finlayson & Chick (1982) also showed that systems containing emergent plants are capable of purifying effluents of very high nutrient concentrations. Their study used effluent from a abbatoir which contained up to 100 mg l^{-1} of nitrogen and up to 15 mg l^{-1} of phosphorus.

A mature *Scirpus* system (3-5 years old) is capable of removing 300-500 kg N ha^{-1} and 50-75 kg P ha^{-1} annually. From this amount 150-300 kg N ha^{-1} and 20-40 kg P ha^{-1} can be harvested and removed from the system (de Jong 1976, 1977) suggests harvesting *Scirpus* only every second or third year. *Phragmites* can withstand harvest every year. The need for harvest when using emergent plants is not as important as when floating plants are used. It has been suggested that systems containing emergent plants do not need to be harvested but left to form an artificial marsh ecosystem. It is not known for how long artificial emergent systems would continue to remove nutrients, but work in West Germany suggests a potential life of hundreds to 5,000 years (Kickuth 1976 - cited Finlayson & Chick 1982).

4) *Submerged and other plants*

Few plants, apart from those previously discussed, have been studied in any detail as regards their potential for removing nutrients from water. The only major habit not so far described is that of the submerged group and some of these plants will be discussed in this section.

Howard-Williams (1981) studied the ability of a *Potamageton pectinatus* community to remove dissolved nitrogen and phosphorus compounds from lake water by nutrient additions at four different rates, ranging from 5-100 mg P m^{-3} $week^{-1}$ and 50-1000 mg N m^{-3} $week^{-1}$. The most obvious effect of enrichment was the increase in the filamentous algae which was thought to be responsible for the initial removal of phosphorus from the lakewater. Only at maximum enrichment rates (100 mg P and 1000 mg N) was there any increase in the concentration of nutrients in the algae, plants or sediments. Plant tissues under high enrichment conditions contained 1.8% nitrogen and 0.13% phosphorus on a dry weight basis. After nine weeks at all levels of enrichment, the community was able to remove all of the added nitrogen and phosphorus within one day. The nutrient removal capabilities of Potamogeton pectinatus were found to be lower than those of Eichornia crassipes and other floating plants. P. pectinatus also requires aerobic conditions and hence it would not be suitable for tertiary sewage treatment. It could be very useful however, in removing nutrients from a eutrophic lake and thus possibly preventing the later stages of phytoplankton blooms.

Sinclair and Forbes (1980) tested the ability of Najas guadalupensis community to remove nutrients in runoff from an organic farm. Reductions of 64% ammonia, 33% nitrate and 93% phosphorus were achieved. The high reduction in phosphorus was attributed to the aerobic nature of the 0.4 ha reservoir, allowing the removal of phosphorus to the sediment. Sinclair and Forbes (1980) found the Najas system to be more efficient than the E. crassipes and swamp systems that they studied and suggested running the E. crassipes and N. guadalupensis systems in series as the hyacinth system was more efficient at removing nitrate, the only failing of the naiad system.

Work on submerged vascular plants in wastewater ponds has been carried out in Michigan by McNabb (1976). He found aerobic ponds containing Potamogeton foliosus, Elodea canadensis and Ceratophyllum demersum could remove 20-25% P and 50-70% N from the influent water, when plants were harvested. The major problems incurred were keeping the light penetration of the water at a high standard, which was achieved by introducing the zooplankton Daphnia to remove phytoplankton, and keeping a viable population over winter (only 1-2% of the summer biomass was present in winter).

Alligator weed (<u>Alternanthera philoxeroides</u>) may be an emergent, or submerged plant, or it may form floating mats on the surface of the water. Its potential for nutrient removal from water was first recognized by Boyd (1970) who cited a standing crop of 8 tonnes DW ha^{-1} and an annual productivity of 62 tonnes DW ha^{-1}. The plant can contain up to 3% nitrogen and 0.3% phosphorus, and Boyd (1970) suggested 1779 kg N ha^{-1} and 198 kg ha^{-1} could be removed per year by these plants. These figures are below those found for <u>Eichornia crassipes</u> and <u>Typha latifolia</u> but still represent sizeable removals of these nutrients from the system. Wolverton et al. (1976) researched the possible use of <u>A. philoxeroides</u> to remove nutrients from raw sewage and secondary effluent. He found a 97% reduction in nitrogen concentration and a 50% reduction in phosphorus concentrations after a seven day retention time for raw sewage. An increase retention time of fourteen days improved the phosphorus reduction to 78% although no further reductions in nitrogen were observed. Using seconday effluent <u>A. philoxeroides</u> reduced nitrogen by 61% and 76%, for a seven and fourteen day retention times respectively, and phosphorus by 44% and 62%. Once again these nutrient reductions are less than those for <u>E. crassipes</u>.

Other plants which researchers believe have potential for nutrient removal from water include <u>Salvinia rotundifolia</u> and <u>S. molesta</u> (Yount and Crossman 1970, Jackson and Gould 1981) <u>Myriophyllum aquaticum</u> (Jackson and Gould 1981). Egeria (<u>Egeria densa</u>), and slender majas (<u>Najas flexilis</u> - Scarsbrook and Davis 1971) and water willow (<u>Justicia americana</u> - Boyd 1970), although very little information is available for any of these plants.

Submerged plants do not reach the same potential for nutrient removal as the floating and emergent plants discussed previously, but they may be especially useful in shallow reservoirs and lakes.

Conclusion

Vascular aquatic plants have been shown to provide an efficient and economical way of removing nutrients from water. Floating and emergent plants show greater nutrient uptake than submerged plants but all can be used in nutrient filter systems.

The Lemnaceae and water hyacinth (*Eichornia crassipes*) would be suitable for tertiary sewage treatment as they are effective at removing nutrients from anaerobic water. No Lemnaceae treatment plants exist at present and so nutrient reduction figures were based on laboratory and small pond experiments. Slightly less efficient nutrient removal may occur in larger scale treatment ponds. The Lemnaceae are easily maintained in controlled conditions but in a natural environment they are readily windblown and show erratic growth patterns decreasing their effectiveness as a nutrient filter. *Eichornia crassipes* is a noxious weed in many parts of the world and stringent controls on its propagation would be required to protect natural wetlands.

Submerged plants are restricted by the transmission of light through water. Water of high nutrient concentration tends to produce algal blooms which reduces the light available for submerged plants. However, at early stages of eutrophication submerged plants do remove nutrients and reduce the rate of deterioration of the water body.

Emergent plants are useful both in artificial systems to remove nutrients from sewage and other effluent and in the natural environment to provide a "buffer zone" around lakes and estuaries reducing the nutrient concentration of water entering and residing in these water bodies. Emergent plants do not require harvest and provide an artificial wetland similar to the natural wetlands described in the next section.

NATURAL WETLANDS

Nutrient Cycling

The use of natural wetlands for the removal of nutrients from water if far more complicated than the use of artificial systems containing only one of two species of plants, as natural wetlands are delicately balanced systems of inter-related flora, fauna, water, sediments and the atmosphere. In order to use a natural wetland effectively for nutrient removal from water it is necessary to understand the nutrient dynamics and stores within this complex ecosystem.

Nitrogen Cycling in Wetlands

A simplified model of the chemical tranformations, major fluxes and stores of nitrogen in a hypothetical wetland can be seen in Figure 2. Algal and bacterial populations in the water column have not been included in this model, but their importance in chemical transformations, and as a store in some cases, should not be overlooked.

The main sources of nitrogen entering a wetland are through the input of water containing nitrates, nitrites, ammonia and organic nitrogen and through the fixation of nitrogen gas from the atmosphere.

A variety of microbial genera are able to carry out nitrogen fixation including free-living blue-green algae, (eg. *Aphanizomenon, Anabaena, Nostoc.*) photosynthetic bacteria, (*Chlorobium* sp.) or the heterotrophic aerobic bacterium *Azotobacter*. Certain anaerobic bacteria may also fix nitrogen (eg. *Clostridium pasteurianum.*) Nitrogen fixation is not confined to the water column. Bristow (1974) found nitrogen fixation rates of 60 kg ha^{-1} yr^{-1} in the anaerobic rhizosphere of *Glyceria borealis* and found that nitrogen fixation in the rhizospere of *Typha* sp. could account for 10-20% of the plants requirements. Granhall and Selander (1973, cited van der Valk 1978) found that nitrogen fixation by blue-green algae growing as epiphytes in wetlands contribute nitrogen, as can legume nodules (eg. _Viminaria_ spp.). The exact contribution of nitrogen to wetlands via nitrogen fixation has evidently never been studied (van der Valk et al. 1978).

Losses of nitrogen from wetland ecosystems occur mainly through denitrification, the transformation of nitrate (NO_3) to nitrogen gas (N_2), which is carried out by such micro-organisms as _Pseudomonas denitrificans_, _Thiobaccilus denitrificans_, and _Spirillium_ sp. These require an organic energy source, a circum neutral pH, and anaerobic conditions. Such an environment is found in most wetland sediments, or at the sediment/water interface. Tilton (1977) reports a nitrate removal rate of 2.5 mg l^{-1} day^{-1} from water overlying sediment, compared to 0.5 mg^{-1} day^{-1} from wetland surface water not in contact with sediment. The nitrogen gas formed by denitrification will only escape to the atmosphere if it is not refixed in the ecosystem. Nitrogen gas is most likely to escape if the water in the wetlands is very shallow and anaerobic. The greater the volume of aerobic water above the sediment, the lower the removal rate of nitrogen from the wetland. Losses of nitrogen from different types of wetlands vary. Patrick (1974) found sediments and associated microbes to remove 9.15 ppm N^{-1} in a continually flooded saline marsh and 4.38 ppm^{-1} in a freshwater swamp in Louisiana, U.S.A.. Tilton and Kadlec (1979) suggest such rates would be sufficient to account for all the nitrate removal from their study of a Michingan marsh. However, although no other avenues are available for total removal of nitrogen from the wetland, there are other mechanisms for nitrogen loss. Nitrogen, ammonia and organic nitrogen introduced into the wetland via runoff and other water inputs may be transformed or removed by a number of processes:-

Dissolved organic nitrogen (eg. amino acids) are rapidly assimilated by bacteria. Particulate organic nitrogen may also be present as protein, nucleic acids, or other amino compounds, which may be decomposed by various bacteria (Bott 1976). The organic nitrogen in thus converted to ammonia by microbial activity.

Ammonia may be removed by nitrification. This involves the oxidation of ammonia to nitrates under highly aerobic conditions with a neutral to slightly basic pH by the autotrophic bacteria Nitrosomonas which converts ammonia to nitrite and Nitrobacter which converts nitrite to nitrate. (Kadlec and Tilton 1979). there is evidence that some heterotrophic organisms are also capable of nitrification. Ammonium ions are readily exchangeable and tightly bound to organic matter, hence the organic sediments of wetlands, which generally have a high cation exchange, provides an excellent sink for ammonia. This is not true for the negatively charged nitrate, which is readily leached through organic soils. A final sink for ammonia is via the uptake of plants and their epiphytes. Bresonik (1968 - cited Klopatek 1978) has shown that ammonia may be the prime inorganic nitrogen source for freshwater plants that obtain their nutrients directly from the water, and that it may be assimilated very rapidly.

Nitrites, as described earlier, are predominantly removed by denitrification. Plants also assimilate nitrate, (Tilton and Kadlec 1979) although the literature suggests that the ammonium ion is the main source of nitrogen for plants growing in waterlogged soils. It must be noted that chemical and biological tranformations described above are in equilibrium within the wetland. If some factor increases the concentration of nitrate in the water, nitrate would likely be converted, via denitrification and nitrogen fixation, to ammonia thus making the nitrogen available for plant assimilation.

The removal of nitrogen by weland plants has been intensively studied and the uptake rates for plants have been described in the previous section. The role of epiphytes on these plants, however, has been somewhat neglected. For practical reasons researchers tend to lump the epiphyte and host plants together. In marshes where emergent plants are dominant this may be a mistake, as it is possible that the epiphyte population is most active in nutrient removal from the water, while the emergent plants which derive their nutrients predominately from the soil, take longer to react to water nutrient concentrations. Allen (1971, cited van der Valk et al. 1978) has shown that epiphytes account for more than 31% of the annual production of Lawrence Lake, U.S.A., and Toth (1972) attributes most of the 98% and 95% reduction in phosphorus and nitrogen respectively in Lake Balton, Hungary, to precipitation and uptake by epiphytic algae and bacteria, and fauna on the reed stems.

Although aquatic plants and their epiphytes are excellent sinks for nitrogen, especially during their growing season where uptake rates reach their maximum, there is a great deal of evidence to show a return of this nitrogen to the water column during autumn, winter, and spring (Sloey et al. 1978), Spangler et al. 1976, Lee et al. 1975). Nitrogen is returned or released to the water by wetland macrophytes and epiphytes through leaching, litter fall, and root excretion. The timing, speed, and amount of nitrogen released during these processes determines the water quality of the wetland effluent (van der Valk et al. 1978). Studies on nitrogen and phosphorus release during decomposition have been confined to emergent macrophytes in cool temperate climates. Literature on floating (Lemnaceae; Laube and Wohler 1973) and submerged plants (_Potamogeton diversifolius_ - Wohler et al. 1975) does not include information about nitrogen or phosphorus. From studies on _Typha_, _Scirpus_, _Phragmites_, and _Carex_ sp., a general pattern of nitrogen cycling can be seen (Davis and van der Valk 1978 A, B, Bernard and Solsky 1977, Klopatek 1975, 1978, Mason and Bryant 1975). An increase of nitrogen is found in the above-ground parts of these plants at the beginning of the growing season, where there is a maximum nutrient concentration in the leaves. Towards the end of the growing season bacteria and fungi may invade submerged leaf material, releasing ammonia to the water. After a time, sufficient nutrients are released to the water for the micro-organisms to survive there and the nitrogen content of the plant once more increases. (Boyd (1970), Klopatek (1975-cited Klopatek 1978). Mason and Bryant (1975), in their study of a reed swamp in Norfolk, England, did not detect any increase in nutrients in the rhizomes in autumn in _Typha_ and _Phragmites_, whereas a large increase in the interstitial waters was noticed. During the winter, rising waters of the lake inundated the swamp and the large nutrient content was passed to the lake. David and van der Valk (1978), Bernard and Solsky (1977), and Prentki et al. (1978) found _Typha_ sp., _Scirpus_ sp., and _Carex_ sp. to translocate nutrients to their rhizomes and roots before the death of the shoots in December. Despirte this, at the end of a 525 day study Davis and van der Valk (1978a) found _Typha_ litter to have released 71 kg N/ha, (60% of which was lost in the early weeks of submergence due to leaching, the remainder due to fragmentation) and _Scirpus_ 10 kg N ha^{-1}, (39% of which was lost in the first few weeks).

These studies include only a few species of aquatic vascular plants and were carried out in cool to cold temperate climates where winters are severe. In contrast Dolan et al. (1981), in their study on a Florida marsh, detected no such release of nutrients. Nevertheless, some species are seasonal irrespective of the climate in which they grow eg. _Typha_ (personal observation , Mitchell 1978), and wetlands containing plants which show seasonality may be likely to release nutrients. Further, soft-tissued species (eg. _Sagittaria latifolia_) decay more quicly than plants with more structural tissue (eg. _Typha scirpus_) and hence release nutrients more quickly (van der Valk et al. 1978).

Although leaching of fresh litter releases large quantities of nitrogen, older litter often acts as a sink for nitrogen (Brinson 1977). Chamie and Richardson (1978) found that sedge leaves of an initial nitrogen concentration of 1.4% decreased to 0.9% in seven months, followed by a steady rise to 1.3% during the remaining five months. The leaves of all woody species (willow, bog birch) showed an increase during the year, willow showing an increase of almost 40%. Micro-organism growth was thought to be responsible for these increses. Davis and van der Valk et al. (1978b) determined in their studies at Goose Lake, Iowa, that litter of _Typha glauca_ contained 131% more nitrogen in the standing and fallen litter after a period of ten months, while nitrogen values in _Scirpus fluviatilis_ decreased. It would seem that the ability of the litter to trap nitrogen is species dependent. Wetlands which contain a large proportion of soft-tissued species and annuals may lose up to 80% of the nitrogen in the first month, and significant nutrient retrieval by older, fallen litter is unlikely to occur after such a large initial release.

Whether nitrogen is removed from the wetland by water outflow depend on hydrology and seasonality of litter fall. Wetlands that have significant surface water flow at times of high litter fall will export more nutrients than those who have little of no outflow during litter fall, or whose litter fall is not highly seasonal (as in the studies of Dolan et al. 1981). Release of nutrients is further aided by artificial drainage, or by natural drying out of the wetlands. This allows the oxidation of normally anaerobic sediments and litter. Lee et al. (1975) found that a drained portion of Shakey marsh, Wisconsin, contained 36% less nitrate, 57% less ammonia, and 60% less organic nitrogen than the natural marsh. Ammonia would be rapidly oxidised to nitrate under aerobic conditions and then leached. Amundson (1970-cited Lee et al. 1975) found the drainage of this swamp to release fifty times the annual agricultural output of 0.5 kg/ha/yr over a period of several years. Klopatek (1978) found that Teresa marsh, Wisconsin, showed a three-fold increase in export during drainage, although innundation following drainage caused a large increase in nitrogen concentration in the soil.

The nitrogen cycle within a marsh, then, is regulated by the type of vegetation, soil, climate, and hydrology of each particular marsh. The capabilities of a wetland to remove nitrogen from incoming water depends on the intrinsic qqalities of each marsh, and no general pattern can be outlined.

Phosphorus Cycling in Wetlands

Phosphorus cycling in wetlands generally parallels nitrogen cycling, although there are fewer chemical transformations (Figure 3). A major reasonf for this is because there is no atmospheric source or sink for phosphorus.

Phosphorus enters a wetland either as inorganic or organic material. Organic phosphorus may be converted to phosphate by heterotrophic bacteria and *vice versa*, in both the soil and water column (eg. van der Valk et al. 1978). Bott (1976) suggests that inorganic phosphate is rapidly utilized by algae, bacteria, and macrophytes and tends to be assimilated and removed rather than transformed to another state in the water column.

A few abiotic reactions are capable of removing phosphate from the water column. Under high oxygen tension, phosphate combines with iron to precipitate ferric phosphate. This reaction is in equilibrium with ferric sulphide which forms during anaerobic conditions, releasing the phosphorus (Bott 1976). Mortimer (1971, cited Sloey et al. 1978) comments that phosphorus will not be released unless the oxygen concentration drops below 2 mg O_2/l. Dissolved phosphorus is adsorbed onto peat particles, clay and silt, and is known to form stable organo-metallic phosphates with fulvic and humic acids. Phosphorus is more likely to remain in the sediments at pH 5-7 than at more extreme pH. Spangler et al. (1976) further suggest co-precipitation and precipitation as removal mechanisms for phosphorus.

The second area of phosphorus removal is by plant assimilation, which may be closely linked to absorption of phosphorus by soils. For example Wetzel (1969, cited Spangler et al. 1977) demonstrated that aquatic macrophytes play a significant role in the precipitation of phosphorus in waters such as Spring Creek, U.S.A., which have enough calcium to precipitate calcium carbonate as a result of photosynthesis. Phosphorus is readily adsorbed by calcium carbonate (Sloey et al. 1978).

The uptake of phosphorus by aquatic macrophytes has been covered in the previous section. During the growing season uptake of phosphorus is at a maximum, resulting in phosphorus removal rates from water of up to 97% (Dolan et al. 1981). During the autumn Prentki et al. (1978) found _Typha latifolia_ to translocate 23% of this phosphorus to underground parts, the roots continuing to take up phosphorus after shoot death, accumulating 9% of the next seasons phosphorus requirements. Of the original 3.2 g P m^{-2} in the plant at the end of the growing season, 2.5 g P m^{-2} was released to the marsh surface water as a result of litter fragmentation and leaching. Mason and Bryant (1975) found no such translocation in _Typhus angustifolia_ or _Phragmites communis_, and a large proportion of the phosphorus in these plants was found to be leached from litter within the first month. Such a loss of phosphorus during the autumn and winter seasons caused an 83% detention of phosphorus in the summer to drop to only a 10% annual average in studies of Lake Winga, Wisconsin (Loucks et al. 1977 - cited Sloey et al. 1978). It would seem that phosphorus parallels nitrogen discharge patterns.

Although phosphorus is leached rapidly from fresh litter, older fallen litter accumulates phosphorus by microbial activity. At Goose Lake the total amount of phosphorus in fallen and standing litter of Scirpus fluviatilis and Typha glauca after ten months was 192% and 230% respectively, of that of fresh standing litter. (Davis and van der Valk 1978b). These authors found Scirpus fluviatilis to have incorporated 8 kg of phosphorus during a 525 day period. Typha litter however, released 10 kg of phosphorus (Davis and van der Valk 1978a). Davis and Harris (1978 - cited van der Valk et al. 1978) demonstrated that Typha angustifolia and Cladium jamaicense accumulate more phosphorus in areas where water has a high phosphorus concentration, than in areas of lower phosphorus concentration. For Typha litter phosphorus accumulation in areas of high phosphorus concentrations amounted to 372% of phosphorus content of fresh standing litter. David and Harris (1978) concluded from their study that "the growth response of the detritus biota to phosphate enrichment appeared to be much greater than the response of living vegetation".

Sediments appear to be the major permanent sink for phosphorus (Sloey et al. 1978, Dolan et al. 1981), as plants tend to release nutrients to the water column on decay. However, the importance of plants cannot be overlooked in terms of epiphyte growth, microbial action, their contribution of phosphorus to the sediment on decay, and then provision of peat substrate.

Lee et al. (1975) and Spangler et al. (1977) suggest that wetlands are only temporary phosphorus sinks, plants assimilating nutrients in the growing season and returning them to the water in autumn, winter and early spring, causing the annual input and output of a wetland to be more or less equal as regards phosphorus. Van der Valk (1978) and this author believe that this does not apply universally. Both Lee et al. (1975) and Spangler et al. (1977) conducted studies in areas where winters were severe, resulting in the death of the wetland plants studied over winter. In warmer climatic areas, and regions where plant growth is less seasonal, less phosphorus is likely to be released in the wetland effluent. Dolan et al. (1981) observed no spring flushing of phosphorus in their study on a Florida marsh, and Odum et al. 81975, cited Sloey at al. 1978) found a 96% removal of phosphorus from the water column in a study on cypress domes in Florida.

The number of studies done in warmer climates is limited, and it is likely that wetlands with more efficient phosphorus removal will be found in these areas. Even wetlands in cooler climates should provide phosphorus traps, if not particularly efficient ones, as the data on phosphorus accumulation by old litter shows. Permanent burial in sediments by peat and organic sediments should provide a further avenue for phosphorus removal from water in wetlands.

Management of Wetlands for Nutrient Assimilation

With the knowledge of nutrient dynamics in wetlands, possible avenues for the removal of nutrinets from wetland influent can be recognized, and the wetlands managed to provide optimum conditions for nutrient uptake from waste water. Figure 4 shows the nutrient dynamics and stores within a hypothetical wetland receiving waste water. The reate at which wastewater can be applied to a wetland (R_1 Figure 4) depends on three factors:-

1/ The rates of input of nutrients from rain and surface water, groundwater and nitrification (R_1, R_3, R_N)

2/ The rates of tranfer to and from storage compartments. (R_5 - R_{13})

3/ The rate at which nutrients may be permitted to enter receiving waters (R_{14} ie. water quality standards)

Presuming that the water quality standards of the wetland effluent will be to achieve as alow a concentration of nutrients as possible, and that contol of the reates of input of nturients by rain, surface water, and groundwater is not feasible, it remains to manage the wetlands in terms of minimizing nutrient contributions to the water column by the stores, and maximizing the removal rates from the water to the stores and tne atmosphere.

Denitrification has been shown to require anaerobic conditions, a circumneutral pH, an organic carbon source, and a small depth of water between the active site of denitrification (the sediment/water interface) and the atmosphere in order to aid nitrogen escape to the atmosphere. Sediment/water interaction has also been seen to be important for both abiotic and biotic removal of phosphate. Hence management of the wastewater input should cause its flow to extend over the whole region of the wetland in a shallow film rather than channelling water across the wetland. The studies of Mudroch and Capobianco (1979) at Cootes Paradise, Ontario, and of Lee et al. (1975) at Horicon, Shakee and Waunakee Marshes, Wisconsin, and Fetter et al. (1978) at Brillion Marsh, Wisconsin, all show poor nutrient removal rates as compared with other wetland studies (Table 6). In those three cases there was channelling of water, and/or a large proportion of deep water in the marsh. This leads to short retention times and little interaction between the water inflow and the organic sediments. Kadlec and Tilton (1979) suggests the use of trickle flood irrigation to prevent this effect.

Impacts on a sedge willow community from small single-point discharges of the order of 6-60 gal/min in their Houghton Lake study showed that damage was localized. Kadlec and Tilton's Bellaire study (cited Kadlec and Tilton 1979) found that ditching of the wetland to distribute the water can have deleterious effects on wetland flow patterns. Ditches tend to channel the water rather than spread it, forming small rivers across the wetland. Spraying of the wastewater resulted in canopy damage.

Factors involved in the effectiveness of nutrient assimilation of a wetland include the concentration of nutrients in the wastewater, sediment/water interaction, and retention times; these factors being related. More concentrated wastewater requires a longer retention within the wetland and, as has been noted, little sediment/water interaction occurs when relatively fast-flowing channels of water are flowing across the wetland.

The retention rate and rate of input into a wetland depends on the concentration of the wastewater, the acceptable standards of water quality of the effluent, and the type of wetland. Every wetland has a saturation point however, and hydraulic overloading can cause serious damage including complete washout of loosely aggregated suspended solids and algal debris, removal of sediment, vegetation damage, and little nutrient removal. Too concentrated a discharge can cause root exposure and canopy damage (Kadlec and Tilton 1979). Odum et al. (1975-9 - cited Kadlec and Tilton 1979) found applications of 18 cm of wastewater over the wetland area, each week were too large for adequate treatment whereas 2-5cm week^{-1} allowed adequate nutrient removal. The rate of wastewater input to the wetland needs to be determined by studying the nature of each individual wetland. In any case, inputs should cause minimal increases in water depth over the whole wetland if nutrient assimilation is to be effective.

Adopting the preceding management option allows for good nutrient interaction between the water and the stores outlined in Figure 4, and for nitrogen removal by denitrification. Further management may allow the rates of uptake by these stores to exceed their contributions to the water.

The removal of nutrients by vegetation has been shown to be at a maximum during the growing season. In order to lower the nutrient contribution of these plants to the water upon their death, it seems that harvesting plants towards the end of the growing season would be advantageous. So as not to disturb the sediment, and cause nutrient release from this store, only above ground shoots would be harvested. This is also the easiest form of harvest. In the case of floating plants this course of action may be feasible (see data on the Lemnaceae and *Eichornia crassipes*), but for emergent plants in their natural environment, harvest was found only to remove 6% of the phosphorus input (Spangler et al. 1977). Spangler et al. (1976) found that after each successive harvest the quantity of material was smaller, such that in biweekly harvests of *Typha angustifolia* the weight per square metre dropped from 99 g DW/m^2 on June 6, 1971 to only 7.8 g DW/m^2 on August 7, 1971. Harvesting was also found to slow growth rates. The cost of harvesting such plants would not be worth the 6% decrease in phosphorus. Harvesting within a natural wetland would not be an easy task and could upset and balance and perhaps nutrient uptake properties of the ecosystem.

The only remaining stores are the micro-organisms, the activities of which can be manipulated only by making conditions suitable for denitrification or phosphate precipitation, and the temporary and permanent stores within the sediment. As noted above, phosphate can be held in the sediments if the water is not too anaerobic while denitrification requires anaerobic conditions. Although the exact oxygen concentration for optimum denitrification is not available, it would seem that different conditions are needed for these two important processes of nutrient removal. Possibly other methods of phosphorus precipitation could be utilized, which would lock phosphorus away under anaerobic conditions, a field of study which might provide useful results.

The final management problem is if plants which have seasonal growth patterns cannot be prevented from releasing nutrients in between growing seasons, then a method must be devised to prevent this nutrient-enriched water from entering lakes and estuaries. Spangler et al. (1977) suggest three methods by which this may be done.

1/ The water from the marsh could be irrigated on land;

2/ The water could be lagooned for recycling through the marsh at a later time;

3/ The water could be treated in a conventional treatment plant.

Any of these three options should reduce the seasonal nutrient flush into a receiving body.

Under proper management most wetlands should be able to filter a large percentage of nutrients from influent wastewater, even if only seasonally. The effectiveness of existing systems varies due to variations in the geographical location, topography, hydrology, and vegetation of the wetland.

The efficiency of Existing Wetlands used for Nutrient Assimilation

Existing wetlands that have been used for nutrient uptake from water can be broadly subdivided into two groups, those that are efficient (greater than 80% removal) and those that are inefficient (50% or less nutrient removal) (Table 6). The inefficient wetlands shown in Table 6 will be discussed first. Table 7 shows studies for which no quantitive data is available.

Theresa Marsh, Wisconsin (Table 6) is the least efficient at nutrient uptake, with an actual export of organic and total nitrogen during the course of the study. This may be explained by the draining of the lake during this period for a fish removal program. As described earlier, this would result in the oxidation of nitrogen in the wetland to a soluble form, allowing its removal during the following inundation, and physical transport of organic nitrogen after draining has occurred.

Brillion Marsh, Wisconsin, was shown to achieve a very low nutrient removal from influent water (Fetter et al. 1978). The nutrient loading on this marsh, however, was extremely high, 1.6km^2 of Typha marsh receiving 148 kg ha^{-1} yr^{-1}. The percent reduction figures quoted represent only one major input of phosphorus, and together with other inputs a mass balance of phosphorus into and out of the marsh yielded a reduction of 32%. This is still not particularly efficient, but the marsh has a fairly high hydraulic loading, especially in winter and early spring, and the vegetation (mainly Typha) shows very seasonal behaviour as regards nutrient uptake and release. Spangler et al. (1976) have observed somewhat better nutrient removal rates for this same marsh (Table 6).

Prentki et al. (1978) observed very poor phosphorus removal (10%) over a year in Lake Wingra, Wisconsin. This was the result of a highly seasonal uptake of phosphorus. The peatlands absorbed 83% of phosphorus in summer but only 1% and 8% in winter and early spring respectively. Although there was no net discharge period, these long seasons of little uptake reduced the annual phosphorus uptake figure drastically.

The final inefficient system for nutrient uptake was a deep water Typha marsh in California (Nute 1977 - cited Kadlec and Tilton 1979). Although California has a mild climate, it may be the seasonality of the plant Typha reduced the nutrient uptake rates. Rates of nitrite and nitrate uptake were reasonable, while ammonia uptake was particularly poor.

The efficient systems in Table 6 are far more numerous than the inefficient ones. The most efficient wetlands studied (Dolan et al. 1981, Kadlec and Tilton 1977, 1979, Boyd et al. 1977, Hartland-Rowe and Wright 1975, and Toth 1972) are ones of fairly shallow water allowing good sediment/water interaction and good retention times. Deeper water swamps (Yonika and Lowry 1978, Nute 1977, Fetter et al. 1978, Prentki et al. 1978) tend to be less efficient, the exception perhaps being Cootes Paradis (Semkin et al. 1976, Murdroch and Capobianca 1979).

Table 7 gives qualitative information that most wetlands are capable of removing nutrients even if only seasonally. The exceptions are the lack of removals of nitrogen in Chandler Slough, Florida and Olifantsvlei, South Africa. At Chandler Slough this lack of nitrogen removal is likely to be due to periodic drying of the wetland, causing oxidation and removal of nitrogen (van der Valk et al. 1978).

The efficiency of existing wetlands used as biological filters for removing nutrients from wastewater effluent and other nutrient-rich water can be seen to be based on the patters of nutrient cycling, the more efficient systems being ones in which conditions are more suitable for nutrient uptake than export for long periods of time.

The Benefits and Adverse Effects of Wetland Utilization for Nutrient Removal

The input of nutrient-rich water or wastewater to a natural wetland will cause a number of floral, faunal, and sediment changes as the wetland adjusts to a new equilibrium. The most obvious changes can be expected in the flora. Whigham and Simpson (1976) found that after one seaons of sewage application to a tidal marsh *Impatiens capensis* was eliminated, *Bidens laevis* was sensitive to continuous aerial spraying, perrenials, *Zizania aquatica* var. *aquatica* and *Acrida cannabina* were not affected and *Polygonum arifolium* increased in numbers. Overall a general decrease in vegetation heights was noticed. Ewell (1976 - cited Sloey et al. 1978) found a considerable increase in Lemnaceae and Azolla where cypress domes were irrigated with sewage and dominant Lyonia sp. were replaced by dog fennel (Eupatorium compostifolium) and fireweed. (Erechtites hieracifolia). Water lilies (Nymphaea odorata) and bladderwort (Utricularia sp.) also decreased. Other authors who have described floral changes due to effluent input include Richardson et al. (1976) and Valiela et al. (1975). The importance of these changes in composition to nutrient uptake are not known, but probably species more vigorous at high nutrient levels would replace less efficient ones. However, on an ecological basis this change in the flora of natural wetlands must be given serious thought before effluent is introduced.

If harvesting is to be used as a method to reduce nutrients in the wetland then studies on the plant intended to be harvested should be carried out. Hanster (1975, cited Sloey et al. 1978) found Scirpus acutus and S. validus recovered well and sustained high yields even when harvested every two weeks. Sparganium eurycarpum, decreased in size and shoot numbers after harvest. Plant biomass is a critical factor in nutrient removal by wetland and if it is lowered significantly the efficiency of nutrient removal by the wetland will suffer. Timing of the havest can also be of critical importance (Sloey et al. 1978).

The impact of nutrient loading on insects within the wetland, and in particular mosquitoes, has not been studied. As the most efficient water level for nutrient removal is also the one most beneficial to mosquito reproduction, this may be a problem. However, as proper management of effluent inflows to a natural wetland should not cause much change to the hydrology of the wetland, the increase in mosquito numbers may be minimal. There is urgent need to study this facet of wetland ecology in wetlands receiving a greater nutrient load.

The effect of nutrient loading on wildlife is also not known. Certainly an increase in water level caused by management practices will affect wildlife and heavy metals, phenolic compounds, and other pollutants present in wastewater are likely to be harmful to wildlife in the wetland. Once again study in needed in this area.

Another problem is the nutrient capacity of a wetland. Although wetlands may absorb high nutrient inputs at the beginning of a program it may be possible for such wetlands to become super-saturated and their efficiency as nutrient filters to decrease over time. Too few studies have been conducted for a long enough time to make a sound judgement on this. Brillion Marsh, Wisconsin, has been used for tertiary treatment of effluent since 1923 (Sloey et al. 1978), and is still operational. Studies in Europe predict that marshes should be able to absorb nutrients for long periods of time, perhaps up to 5,000 years (Kickuth 1976, cited Finalyson 1982). The longevity of a marsh as a nutrient filter is probably a function of its sediment properties, peat production rates, vegetation types, hydrology, and possibly no generalization can be made as to the lenght of time a wetland may be used for nutrient asismilation.

Despite these drawbacks the use of wetlands as nutrient filters for effluent or nutrient-rich waters has been convincingly shown to be a cheap, effective method of water purification. Even in areas where nutrient removal is only seasonal, fairly inexpensive steps may be taken to ensure that nutrient inputs into lakes and estuaries may be minimized.

CONCLUSION

Artifical systems, containing one of two species of plants, and natural wetland ecosystems have both been shown to be very efficient at removing nutrients from water. This author, however, recommends the use of artifical systems over natural wetlands for the following reasons:-

1) Full control of all aspects of nutrient removal is possible in artificial wetlands. In a complex natural ecosystem, control is very difficult.

2) Artificial systems containing plants of known high nutrient removal capacity can be put into operation quickly and efficiently. Use of a natural system would require a full ecological survey and extensive pilot tests, before a decision could be made about its possible operation efficienty as a nutrient removal system.

3) In the past, lakes and estuaries were used as disposal sites for urban, industrial, and agricultural effluent. For a while they were capable of self-purification by similar methods to those found in wetlands, but eventually they became overloaded and eutrophic. It is important that the same mistake is not made with our valuable and diminishing wetlands. Long-term, carefully monitored experiments at a limited number of sites around the world, are required before natural wetlands are commonly used for nutrient removal from effluent. It has already been shown that input of effluent causes changes in the species composition of wetlands, and further problems may arise if we base our decisions on short-term observations.

Under proper management, the nutrient removal capabilities of artificial systems, containing vascular aquatic plants, and natural wetland ecosystems, may provide an economical and effecitive means of purifying industrial, urban, and agricultural effluent, and help reduce eutrophication of our lakes and esturies.

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APPENDIX 1: ABBREVIATIONS

C = Control.

T = Test run.

TN = Total nitrogen.

Org. N = Organic nitrogen.

NO_3-N = nitrate.

NO_2-N = nitrite.

NH_4-N = ammonia.

PO_4-P = phosphate.

TP = Total phosphorus.

TDP = Total dissolved phosphorus.

N = nitrogen.

P = phosphorus.

DW = dry weight.

WW = wet weight.

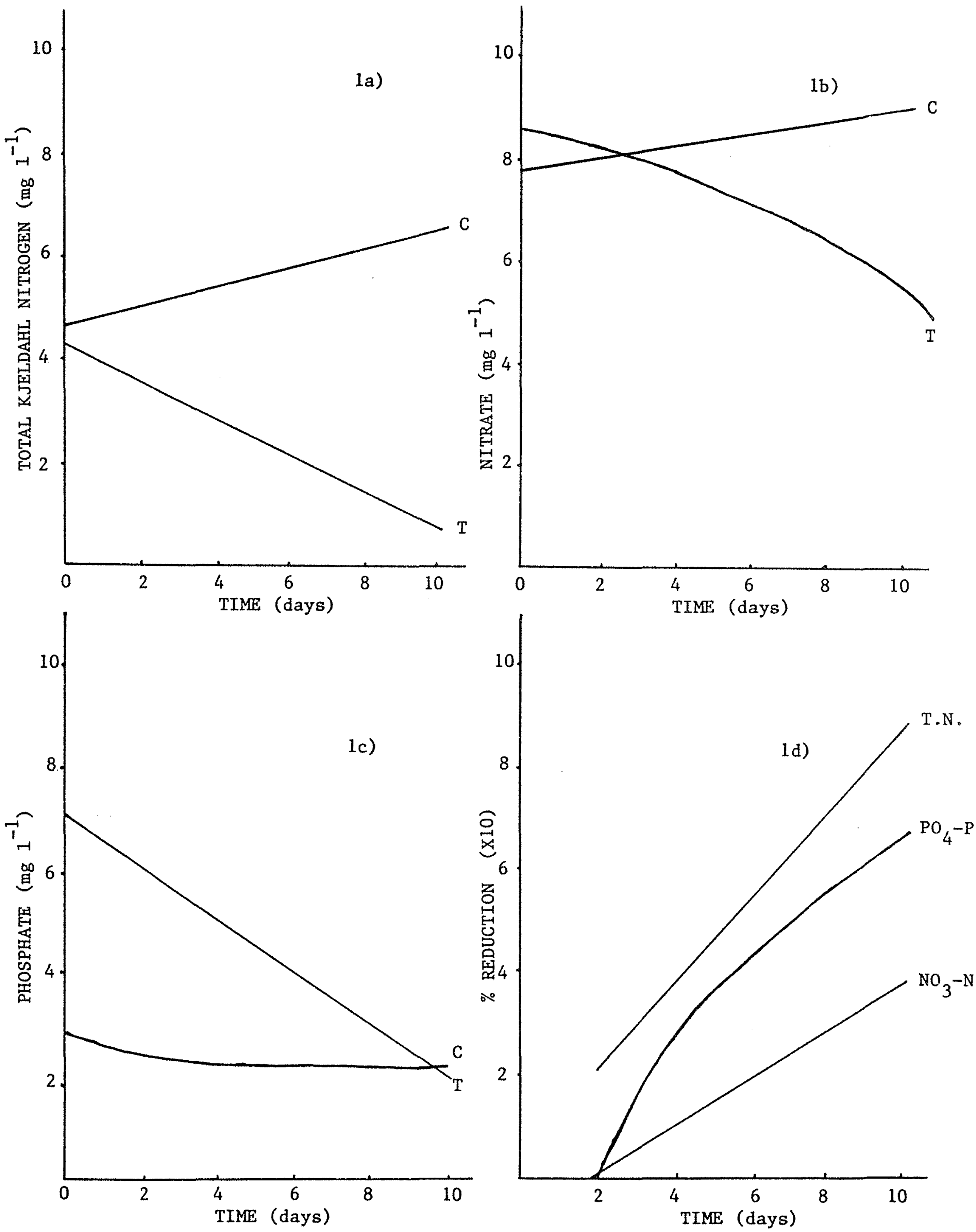
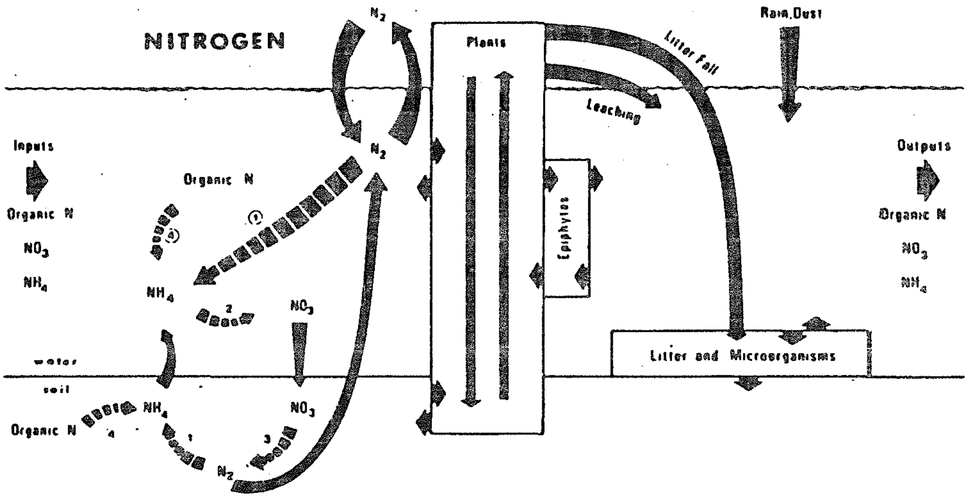


FIGURE 1: THE RATE OF TOTAL NITROGEN, NITRATE, AND PHOSPHATE REMOVAL FROM EFFLUENT. - Graphs derived from data given in Harvey and Fox (1973).

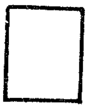
FIGURE 2: NITROGEN CHEMICAL TRANSFORMATIONS, FLUXES, AND STORAGES IN A HYPOTHETICAL WETLAND RECEIVING LAND RUNOFF.



<u>KEY</u>:

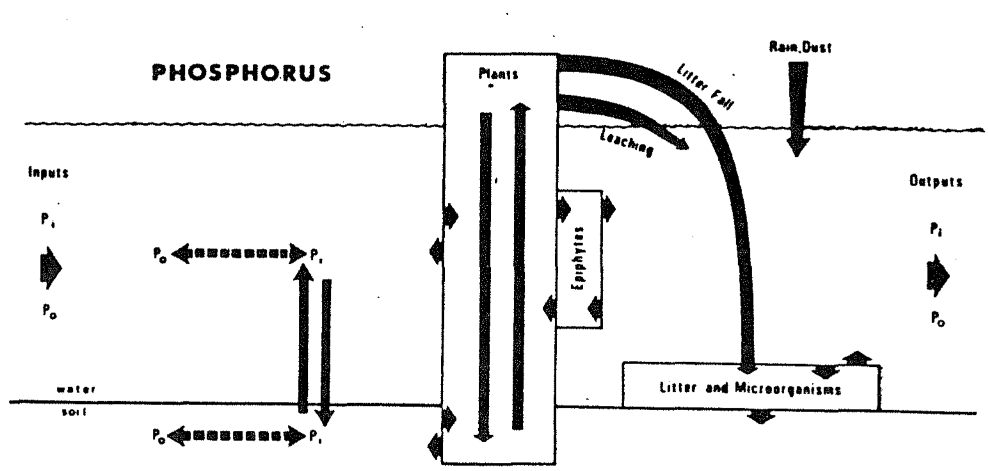
 Nitrogen flux

 Chemical transformation

 Store

1 Nitrogen fixation
2 Nitrification
3 Denitrification
4 Mineralization

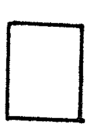
FIGURE 3: PHOSPHORUS CHEMICAL TRANSFORMATIONS, FLUXES, AND STORAGE IN A HYPOTHETICAL WETLAND RECEIVING LAND RUNOFF - cited van der Valk et al (1978).



KEY:

 Phosphorus flux

 Chemical transformation

 Store

P_i Inorganic phosphorus

P_o Organic phosphorus

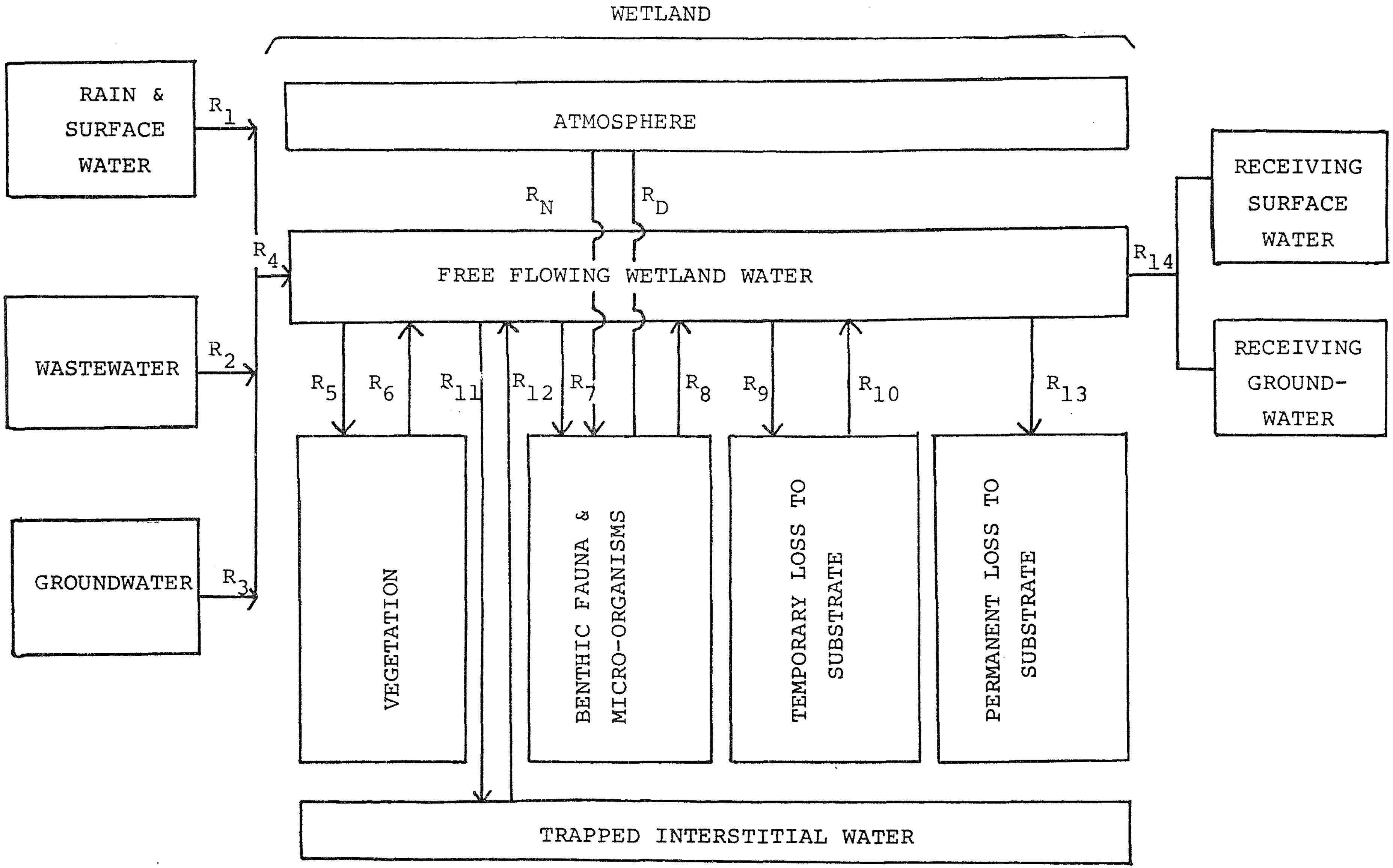


FIGURE 4: DIAGRAMMATIC REPRESENTATION OF SOURCES, RATES OF TRANSFER, AND STORAGE COMPARTMENTS OF NUTRIENTS IN A WETLAND ECOSYSTEM RECEIVING WASTEWATER. (cited Sloey et al 1978)

TABLE 1: PRODUCTIVITY OF *Eichornia crassipes*

LOCATION & REFERENCE	DW/DAY	WW/DAY	DW/YEAR
Mississippi[1] Wolverton & McDonald (1976)	-	17.8 tonnes/ha 1780 g/m^2	212 tonnes/ha 21200 g/m^2
Iowa[1] Wooten & Dodd (1976)	0.29 tonnes/ha 29 g/m^2	6.14 tonnes/ha 614 g/m^2	-
Louisiana[2] Penfold & Earle (1948 - cited Steward 1970)	-	-	5.9 tonnes/ha 590 g/m^2
Sub Tropics[2] Westlake (1963- cited Steward 1970)	-	-	24 tonnes/ha 240 g/m^2
Theoretical Boyd (1970)	-	-	54.7 tonnes/ha 5470 g/m^2

NB: 1 = sewage effluent. 2 = natural environment.

TABLE 2: NUTRIENT REMOVAL BY *Eichornia crassipes* SYSTEMS.

DESCRIPTION OF SYSTEM.	FLOW RATE	DETENTION TIME	NUTRIENT	INPUT	OUTPUT	% REDUCTION	REFERENCE
Single 2 ha pond; depth = 1.22 m	473 m^3/day	54 days	TN	12.01	3.35	72	Wolverton & McDonald (1979).
580 m^2 pond divided into 4 sections	1) 1.26 l/sec.	5.3 days	TN	8.16	2.47	70	Dinges (1978).
1) 1 m deep; mixed vege^n & fauna	2) 1.26 l/sec.	4.5 days	TN	9.94	3.59	64	
2) 0.85 m deep; pure *E. crassipes*							
16.2 ha reservoir	-	-	NO_3-N	0.76	0.36	53	Sinclair & Forbes (1980).
			PO_4-P	0.74	0.37	53	
5 ponds in series each 465m^2; 0.8 m deep	481 l/min.	two weeks	NH_4-N	13.1	0.6	94	Wooten & Dodd (1976).
			NO_3-N	1.0	-	100	
			PO_4-P	23.1	21.0	9	
7600 m^2 pond; 1.4 m deep 1) winter	1) 1900 m^3/day	13.5	TN	13.87	11.26	19	Cornwell & Zoltek (1977)
			NO_3-N	3.10	2.75	11	
			PO_4-P	4.75	4.18	12	
2) April-January		15 hours	TN	5.79	4.69	19	
			NH_3-N	4.45	3.53	21	
			NO_3-N	5.46	5.00	8.4	
			PO_4-P	3.2	3.0	6.6	
			TP	5.46	4.51	17	
3) February - April		5.96 hours	NH_3-N	2.07	3.07	-32.5	
			NO_3-N	8.10	7.91	2.3	
			PO_4-P	3.28	3.18	3	
			TP	3.42	3.44	-0.5	
		11.4 hours	NH_3-N	1.42	1.57	-9.5	
			NO_3-N	9.28	7.69	17	
			PO_4-P	2.47	2.47	0	
			TP	3.24	3.15	2.8	
	23.5		NH_3-N	3.47	3.21	7.4	
			NO_3-N	8.38	8.22	2.0	
			PO_4-P	3.40	3.21	5.3	
			TP	3.64	3.57	2.0	
No data	-	5 days	PO_4-P	-	-	61	Clock et al (1968) - cited Cornwell et al (1977)
			NO_3-N	-	-	75	
4, 0.2 ha ponds 1.07 m deep	-	5 days	NO_3-N	-	-	100	Furman & Gilcreas (1965 - cited Cornwell et al 1977).
			NH_3-N +org. N	-	-	75	
No data		10 days	PO_4-P	-	-	52	Sheffield (1966-cited Cornwell et al 1977).
		30 days	PO_4-P	-	-	8	
		10 days	NO_3-N +NH_3-N	-	-	94	
Lab tests raw sewage	-	7 days	TN	-	-	92	Wolverton et al (1976)
			TP	-	-	60	
secondary effluent		7 days	TN	-	-	75	
			TP	-	-	87	
		14 days	TN	-	-	89	
			TP	-	-	99	
Lab Tests	static	4 days	P	3.7	0.8	78	Rogers & Davis (1972).
			N	21	12.8	39	

TABLE 3: THE EFFECT OF RETENTION TIMES AND POND DEPTH ON THE NUTRIENT REMOVAL EFFICIENCY OF *Eichornia crassipes*.

NUTRIENT	AVERAGE INFLUENT VALUE (mg/l)	DETENTION TIME (HOURS)	AVERAGE EFFLUENT CONCENTRATIONS. (mg/l) 0.34m POND	% REDUCTION	0.64m POND	% REDUCTION	0.70m POND	% REDUCTION
PO_4-P	3.37	12	3.31	18	3.32	1.5	-	-
		24	2.69	20	2.87	15	3.17	6
		48	1.86	45	2.38	29	2.84	16
		96	-	-	-	-	1.90	44
TP	3.44	12	3.42	1	3.55	-3	-	-
		24	2.86	17	3.08	11	3.33	3
		48	1.82	47	2.30	33	2.90	16
		96	-	-	-	-	1.95	43
NO_3-N	8.08	12	8.56	-6	8.35	-3	-	-
		24	4.25	48	5.35	34	8.35	-3
		48	1.30	84	3.13	61	5.36	33.7
		96	-	-	-	-	0.76	91
Org. N	3.93	12	2.57	35	2.88	27	-	-
		24	1.49	62	1.95	50	2.18	45
		48	1.42	64	1.78	55	1.37	65
		96	-	-	-	-	1.16	71
NH_3-N	1.67	12	0.92	45	1.25	25	-	-
		24	0.15	91	0.65	61	0.96	43
		48	0.00	100	0.07	96	1.12	33
		96	-	-	-	-	1.17	30
TN	13.68	12	12.05	12	12.48	9	-	-
		24	5.89	57	7.95	42	12.49	16
		48	2.72	80	4.98	64	6.85	50
		96	-	-	-	-	3.09	78

cited Cornwell et al (1977).

TABLE 4: THE EFFECT OF EFFLUENT NUTRIENT CONCENTRATION ON THE NUTRIENT REMOVAL EFFICIENCY OF *Eichornia crassipes*

NITROGEN				PHOSPHORUS			
INITIAL CONC^N (mg/l)	RETENTION TIME (DAYS)	FINAL CONC^N (mg/l)	% REDUCTION	INITIAL CONC^N (mg/l)	RETENTION TIME (DAYS)	FINAL CONC^N (mg/l)	% REDUCTION
22	1	14.5	34.1	3.9	1	2.4	35.2
sewage	2	13.5	38.7	sewage	2	2.0	45.9
	3	13.0	40.9		3	1.6	56.8
	4	12.0	45.5		4	0.1	93.3
22	1	21.0	4.6	3.1	1	3.3	-6.5
10% h.s.*	2	18.5	15.1	10% h.s.	2	3.0	3.3
	3	17.0	22.7		3	1.8	41.9
	4	17.0	22.7		4	1.1	64.5
54	1	54.0	0	7.7	1	8.1	-5.2
25% h.s.	2	50.0	7.5	25% h.s.	2	7.7	0
	3	48.5	10.2		3	6.5	15.6
	4	48.0	11.2		4	5.5	28.6
96	1	96.25	0	15.5	1	15.4	0.7
50% h.s.	2	96.5	-0.5	50% h.s.	2	15.3	1.3
	3	96.75	-0.78		3	15.2	1.9
	4	97	-1.0		4	15.1	2.6

* h.s. = Hoaglands solution.

TABLE 5: STANDING CROP, ANNUAL PRODUCTIVITY AND NUTRIENT CONTENT OF SOME EMERGENT PLANTS.

GENERA	STANDING CROP g/m^2	ANNUAL PRODUCTIVITY g/m^2	% N CONTENT	% P CONTENT	REFERENCE
Typha	1300	820-19200	1.4	0.2	Bray (1959 - cited Steward 1970), Boyd (1970), Mason & Bryant (1975)
Phragmites	500-1000	520-1080	1.5	0.15	Rudescu (1969 - cited Steward 1970), Mason and Bryant (1975)
Scirpus	606	1000-2000	1.3	0.2	Auclair (1979), Whigham et al (1978), deJong (1977)

TABLE 6 : THE REMOVAL OF NUTRIENTS FROM WATER BY NATURAL WETLANDS.

LOCATION	DESCRIPTION	NUTRIENT	INPUT mg/l	OUTPUT mg/l	% REDUCTION	REFERENCE
Northern Lower Michigan, USA.	Natural *Carex* Peatland	NO_3-N	16.05kg	0.08kg	99	Tilton (1977) cited Kadlec and Tilton (1979)
		NH_4-N	2.57kg	0.74kg	71	
		TDP	16.80kg	0.87kg	95	
Northern Lower Michigan, USA. (Bellaire)	Natural white Cedar swamp (*Thuja*)	NH_4-N +NO_3-N	4.95	0.46	91	Kadlec & Tilton (1977) cited Kadlec & Tilton (1979)
		TDP	3.48	0.11	97	
Wisconsin USA. (Brillion Marsh)	Natural Cattail marsh (*Typha*) (deep water)	NO_3-N	1.17	0.57	52	Fetter et al (1978)
		PO_4-P	3.13	2.93	6.4	
		TP	3.43	2.97	13	
Central Florida USA.	natural forested swamps	TP	6.40	0.12	98	Boyt et al (1977)
		TN	15.30	1.60	90	
Louisiana USA.	*Phragmites* marsh	NH_4-N	547	37	93	Price (1975) cited Kadlec & Tilton (1979)
California USA.	Cattail marsh (*Typha*) (deep water)	NH_4-N	8.3	7.5	9.6	Nute (1977) cited Kadlec & Tilton (1979)
		NO_3-N +NO_2-N	5.8	1.3	78	
Massachusetts USA.	Deep water marsh	PO_4-P	2.2	0.7	68	Yonika & Lowry (1978) cited Kadlec & Tilton (1979)
		NH_4-N	8.8	0.3	97	
		NO_3-N	1.4	0.6	57	
Northern Lower Michigan USA.	natural *Carex* wetland	NO_3-N	1.50	0.10	93	Tilton & Kadlec (1979)
		NH_4-N	0.11	0.03	73	
		TDP	1.57	0.07	96	
Florida USA.	Natural marsh	TP	8.88	0.20	98	Dolan et al (1981)
Wisconsin USA. (Theresa marsh)	Natural riverine marsh	NO_2-N	0.28	0.25	11	Klopatek (1978)
		NO_3-N	1.00	0.89	11	
		NH_4-N	1.72	0.86	50	
		Org. N	1.19	2.73	+229	
		TN	3.63	4.15	+114	
		TDP	0.61	0.30	51	
		TP	0.63	0.40	37	
Michigan USA. (Houghton Lake)	forested peatland	NH_4-N +NO_3-N	331kg/ha	64kg/ha	81	Richardson et al (1978)
		PO_4-P	262kg/ha	85kg/ha	68	
Wisconsin USA. (Brillion Marsh)	Natural *Typha* marsh (deep water)	PO_4-P	3.75	1.21	68	Spangler et al (1976)
		TP	4.28	1.41	67	
Wisconsin USA. (Lake Wingra)	*Typha* marsh (deep water)	TP	5.6 $g/m^2/yr$	5.04 $g/m^2/yr$	10	Prentki et al (1978)
Wisconsin USA. (Waunakee/ Horicon marsh)	Natural marsh	TP	2.0	0.21	89	Lee et al (1975)
		PO_4-P	2.0	0.11	94	
		NH_4-N	10.0	0.39	96	
		NO_3-N	1.5	0.28	81	
		Org. N	0.5-20	3.6	(65)	
Ontario Canada (Cootes Paradise)	Deep water marsh and open water	NH_4-N	11.2	0.5	96	Semkin et al (1976) cited Kadlec & Tilton (1979)
		NO_3-N +NO_2-N	4.4	0.8	82	
		TN	22.7	2.5	89	Mudroch and Capobianco (1979)
		TP	4.73	0.4	93	
North-Western Territory Canada	Sedge meadow	NH_4-N	10.3	0.39	96	Hartland-Rowe & Wright (1975) cited Kadlec & Tilton (1979)
		TP	11.0	0.26	97	
		PO_4-P	9.95	0.25	98	
Hungary	*Phragmites* swamp	TP	4.5	0.08	98	Toth (1972)
		TN	19.97	0.81	96	

TABLE 7: QUALITATIVE DATA ON WETLANDS AS NUTRIENT REMOVAL SYSTEMS (cited van der Valk et al 1978)

LOCATION	DESCRIPTION	PERIOD OF SAMPLING	NUTRIENT	NUTRIENT TRAP?	REFERENCE
South Carolina USA (Santee Swamp)	Riverine	Winter & Spring	N P	Yes Yes	Kitchens et al (1975)
Uganda (Chambura Payrus swamp)	Riverine	Spring	N P	- Yes	Beadle (1932)
Sudd, Sudan	Riverine	All year	N P	Yes -	Harrison et al (1960)
South Africa Olifantsvlei	Riverine	Spring & Winter	N P	No S*	Talling (1968)
Sweden (Lillan)	Riverine	All Year	N P	S S	Stake (1967,1968)
Pennsylvania USA (Tinicum Marsh)	Tidal	Summer	N P	Yes Yes	Grant & Patrick (1970)
New Jersey USA Hamilton marsh	Tidal	All Year	N P	S S	Simpson et al (1978)
Poland (Masurian Lakes)	Lacustrine	Summer	N P	Yes Yes	Pieczynka et al (1975)
Florida USA (Waldo Cypress Stand)	Palustrine	All Year	N P	- Yes	Nessel (1978)
Florida USA (Chandler Slough)	Palustrine	All Year	N P	No Yes	Shih et al (1978)
Iowa USA (Eagle Lake)	Palustrine	All Year	N P	Yes Yes	van der Valk et al (unpubl.)

S* = seasonably (ie., wetland acts as a nitrogen or phosphorus trap for only part of the year.)