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CHANGES IN THE WATER, SOIL, AND VEGETATION OF A WETLAND AFTER A DECADE OF RECEIVING A SEWAGE EFFLUENT

Summary: The impact of discharging an oxidation pond effluent into a wetland in the Waitangi Forest (Northland) was assessed by comparing the water, soil, and vegetation of this wetland (the sewage wetland) with that of an adjacent wetland not receiving effluent (the reference wetland). The hydroperiod of the two wetlands differs markedly with the sewage wetland now permanently flooded whereas the reference wetland is subject to summer drawdown. Marked differences were found in gross chemical indicators such as pH and redox potential between the soils of the 2 wetlands. Extreme differences were also found in the nutrient chemistry of the soils and this was related to differences in water chemistry. This was especially the case for the plant available forms of nitrogen and phosphorus. Typically the sewage wetland soil contained ammonium and inorganic phosphorus two orders of magnitude greater than that of the reference wetland. Major differences were noted also in the plant communities of the two wetlands. The sedge *Eleocharis sphacelata* and the bulrush *Typha orientalis* dominated communities covering a large part of the sewage wetland, whereas the reference wetland contained a much more diverse *Baumea-Isachne* grassy sedge community. Historical information indicated that the sewage wetland had similar hydrological, chemical, botanical characteristics to the reference wetland before discharge of the effluent commenced. It is concluded that most of the vegetation community changes that have occurred may be interpreted in terms of allogenic succession but that future community changes will have a strong autogenic component.

Keywords: Wetlands; sewage; water quality; succession; soil chemistry.

Introduction

As is the case internationally, wetlands in New Zealand have only recently been recognised widely as a valuable, but diminishing component of the natural landscape (Environmental Council, 1983). Because of the high intrinsic values of natural wetlands any management practice affecting their functions and uses should be examined critically. One such practice gaining popularity is the use of wetlands for treatment and disposal of sewage wastes (Venus, 1987). There is no published data on the ecological impacts of sewage discharges to wetlands in New Zealand, and little data about wetlands of similar type internationally, from which to decide whether such disposal practices will cause unacceptable changes to the ecosystem.

Theory would suggest (Howard-Williams, 1985; Kadlec, 1987) that sewage effluent inputs will affect the plant communities of a wetland, principally through changing the hydrological characteristics and/or the nutrient status of the system. Changes in plant diversity may be induced by increasing frequency and depth of flooding, changes in the velocity distribution, and masking of the natural hydroperiod (Richardson and Nichols, 1985; Whigham, 1985). Depending on the intial trophic status of the wetland, nutrient additions from wastewater should lead to greater nutrient availability which will result in greater cycling, decomposition, and ultimately, to a change in community composition to species more tolerant of high nutrient loads (Shaver and Melillo, 1984).

Such changes are likely to be subtle rather than dramatic. At a recent conference, Odum (1987) commented that more profound ecological changes in vegetation were difficult to detect on a short time scale and could take 20 years or more to become apparent. However, changes to microbial processes occur over much shorter time-scales and may, therefore, provide an early indicator of macro-ecological changes (Hodson *et al.,* 1985).

In this study, we use a variety of techniques, including microbial assays, to characterize two arms of a wetland, one of which has received a treated sewage effluent (oxidation pond) for a decade. We use these data, in combination with data on the botanical composition of the wetlands to deduce what the long term consequences of the discharge will be both in terms of the impact on the wetland, and the impact to downstream ecosystems.

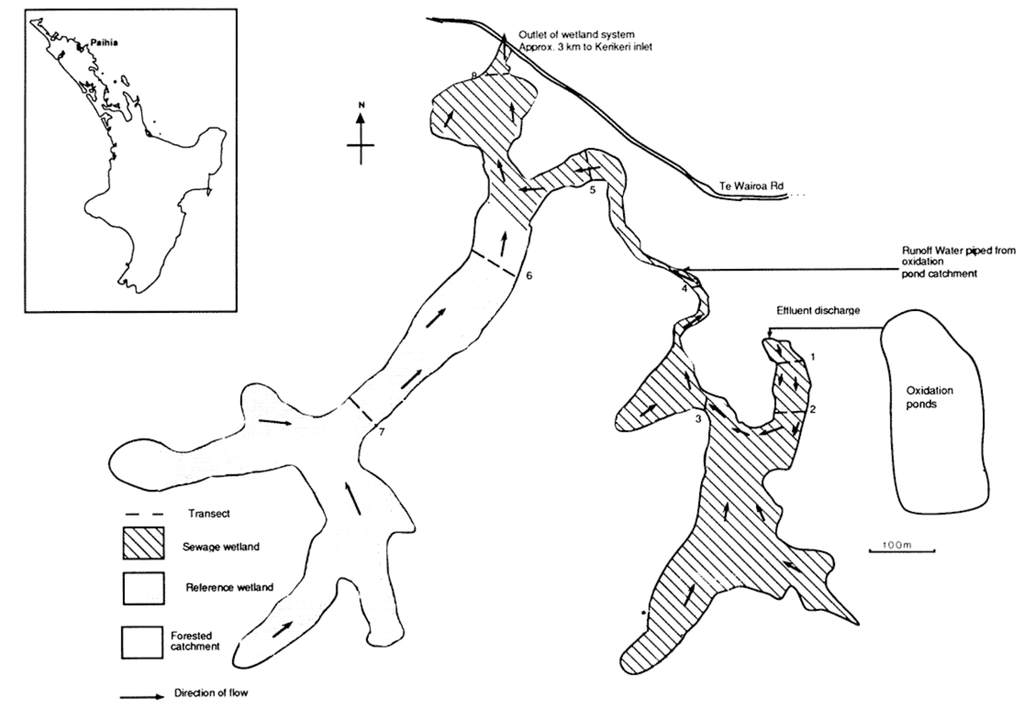


Figure 1: *Map of Waitangi Forest wetlands in relation to the discharge of oxidation pond effluent. The sampling sites for water and sediment are approximately at the mid-point of each transect.*

Site description

The study site forms part of an extensive wetland system occupying the valley. bottoms between sandstone spurs contiguous with a basalt flat. The hills surrounding the wetland system are afforested, mainly with *Pinus radiata*. Oxidation pond effluent from the township of Paihia has been discharged into an arm of the wetland (Fig. I) since 1978. No discharge records have been kept; however, the sewered population of Paihia has a marked 'seasonal variation from a stable base of 2000 in winter to a fluctuating summer population which currently peaks at approximately 8000 (Northland Regional Council, 1990). The designers of the scheme impounded the wetland at the point of discharge to force the effluent to flow around a loop of wetland (from discharge point to site 4, Fig. 1) which they envisaged would provide additional treatment. Treatment time in the 'loop swamp' was assisted further by piping runoff water from the catchment in which the oxidation ponds were sited, to a point just below a forestry road culvert (our site 4, Fig. 1). The 'loop swamp' and the effluent-impacted wetland between the culvert (site 4) and the outlet (site 8) we named the 'sewage wetland'. Approximately 200 m before the outlet, the sewage wetland merges with water from a large arm of wetland not affected by sewage effluent which we named the 'reference' wetland (Fig. 1). Shortly after this confluence the flow is markedly channelized and has characteristics more akin to a stream than a wetland. This stream then meanders a further 3 km before reaching the coast at Kerikeri inlet.

Methods

Transects

Six transects were laid out perpendicular to the flow path in the sewage wetland at approximately 200 m intervals from the discharge point (Fig. 1). A further two transects were laid out in the 'reference' wetland; one approximately 100 m upslope of the confluence with the sewage wetland (site 6) and the other (site 7) a further 300 m upslope. Near the centre of each transect, a permanent sampling station was established for water and soil sampling, and as a point to conduct experiments.

Water sampling and analysis

On 5 occasions at 3 monthly intervals over 1 year, dissolved oxygen (DO), electrical conductivity (EC), pH, and temperature, were measured *in situ* at each sampling station. Water samples were also taken for nutrients (nitrogen and phosphorus forms), and biochemical and chemical oxygen demand. Samples taken for nutrient analyses were immediately preserved (40 g m^{-3} $HgCl_2$) and a fraction filtered (< 0.45 _m cellulose acetate) within 3 hours of collection. Samples were refrigerated until analysis. References to the analytical methods used may be found in Cooke (1988) and Cooke and Cooper (1988). Significant differences between sites were calculated using one-way ANOVA and Tukey tests, or its non-parametric equivalent in cases where the data was non-normally distributed. Differences are reported significant with 95% confidence. The flow rates from the oxidation pond, the oxidation pond catchment, site 4, and site 8 were measured using standard gauging techniques. This also enabled calculation of the flow contribution from the loop swamp (minus oxidation pond effluent), and the reference wetland.

Soil sampling and analyses

Duplicate 20 cm long soil cores were taken at each site using a sharpened PVC pipe (10 cm diameter). The cores were transported to a field laboratory where the pipe was sliced open. Redox potential (Pt-calomel electrodes) and pH were measured immediately at 1,2,4,6,8,10 and 15 cm from the surface. Because of marked difference in pH, redox potentials (Eh) were corrected to pH 7.0 (Eh_7) using the formula given by Bohn (1971). The cores were then sliced at these same intervals and mixed before subsampling. Two g subsamples from each slice were shaken with 10 cm^3 M KCl containing 40 g m^{-3} mercuric chloride for 1 h on an end-over-end shaker. The extracts were centrifuged and decanted for later analysis of mineral nitrogen forms. Extraction was completed within 3 h of taking the cores. Further 1 g subsamples were taken for determination of readily mineralizable carbon (Cooper and Cooke, 1984), and water soluble phosphorus (Olsen and Sommers, 1982).

Additional analyses performed on the soil were total inorganic and organic phosphorus (Olsen and Sommers, 1982), Kjeldahl-nitrogen (Cooper and Cooke, 1984), total solids and volatile solids (gravimetrically after drying at 105 °C and ashing at 550°C, respectively), water soluble carbon (Burford and Bremner, 1975), and denitrifying enzyme activity (Tiedje, 1982). Denitrifying enzyme activity (DEA) was included because it was surmised that nitrate-nitrogen (NO_3-N) could be an important form of N exported from the oxidation pond, and because denitrification is the only process whereby N can be permanently lost (other than by export) from the wetland ecosystem (Brodrick, Cullen and Maher, 1988; Howard-Williams, 1985). The results for all analyses are reported as the mean of samples at each depth from sites 1,2,3,4,5 and 8, and 6 and 7, for the sewage and reference wetlands, respectively.

Vegetation survey'

Vegetation measures were made along each of the 8 marked transects (Fig. 1) and an additional transect (5a) 20 m downstream from 5, which was included to ensure adequate representation of a distinctive sedgeland type in the sample. At 5 m intervals along each transect (land to land), all vascular plant species within aim radius were identified and their percentage cover values estimated by eye within 7 class intervals. One hundred and eleven quadrats were examined in all during March 1989. Patterns in the distribution of the plants and their interrelationships were examined by subjecting the % cover measures of the 31 most abundant species (species recorded in >1 site) to multivariate cluster analysis using the polythetic divisive technique of two-way indicator species analysis (TWINSPAN: Hill, Bunce & Shaw, 1975; Hill, 1979).

Results

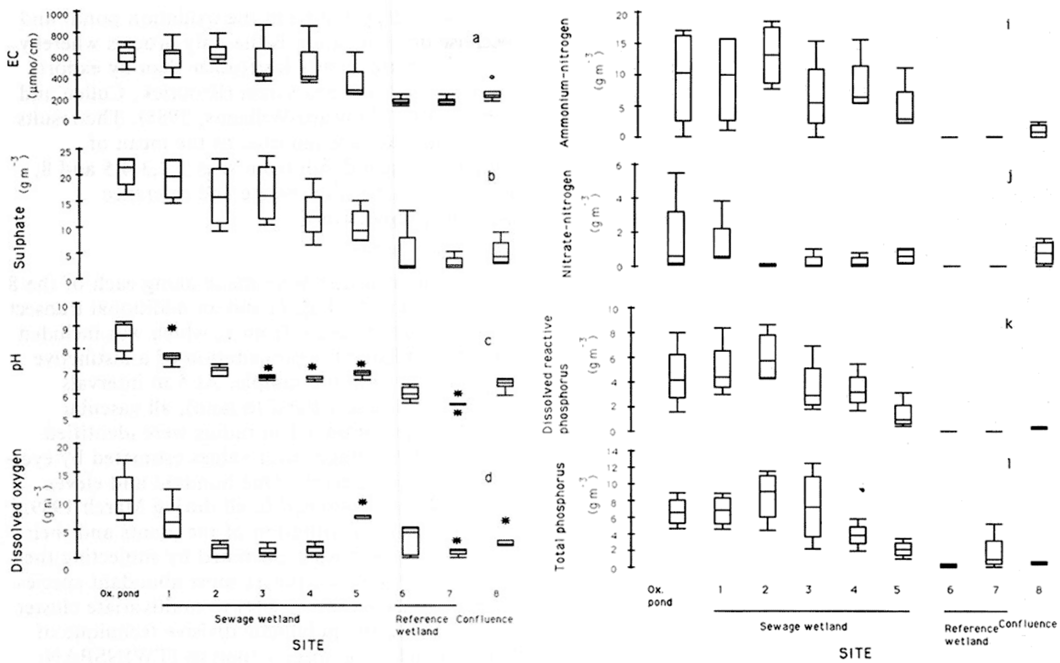
Hydrology

The addition of oxidation pond effluent has resulted in the sewage wetland being permanently flooded. Except under heavy rainfall conditions (Table 1), the specific discharge from the loop swamp was relatively constant throughout the year, with approximately 75% of the flow being accounted for by effluent input. In contrast, discharge from the reference wetland showed greater dependence on antecedent rainfall. For example, under near drought conditions which occurred in March 1990, there was no discharge at

Table 1: *Discharge estimates from sewage and reference wetlands.*

Date	Rainfall (mm/24h)	Sewage wetland (at site 4)	Reference wetland
		------------(1 S^{-1} ha^{-1} wetland)----------	
06/06/89	1	1.86 (0.73)	1.09
06/09/89	55	7.35 (0.33)	4.82
06112189	0	1.37 (0.76)	2.65
06/03/90	0	1.48 (0.76)	0.00

(Bracketed values are the proportion of discharge from the sewage wetland that can be accounted for by effluent inflow)



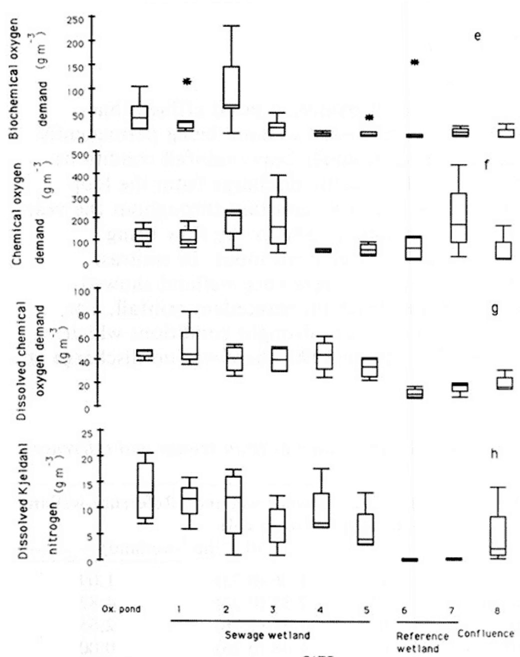


Figure 2: *Water chemistry of sites within the wetland system. The box plots approximately depict the* 25 *to* 75 *percentile and the horizontal line across the box marks the median value. The "whiskers" encompass the main body of the data ($\pm$.1.5 x interquartile range) and extreme values are denoted with a starburst. EC = electrical conductivity.*

all from the reference wetland indicating that evapotranspiration had caused a marked drop n water level.

Water chemistry

There were major differences in water chemistry (Fig. 2) between the sewage and reference wetlands. Furthermore, there were some significant differences in water chemistry of the sewage wetland with distance away from the point of discharge. Some of this difference can be ascribed to dilution of the effluent by 'natural' wetland waters during its passage through the wetland, and indeed one needs to know the mass flow (concentration x flow) before one can reasonably infer whether any removal processes are operative. A detailed analysis is beyond the scope of this paper and in any case it is the concentration in the water column which is likely to influence both the soil characteristics and the plant response. Nevertheless, since the electrical conductivity (EC) of the reference wetland was much lower than hat of the oxidation pond effluent (Fig. 2a), EC may be taken as a useful surrogate measure of dilution, from which to compare the concentration distributions of the other parameters (Fig. 2b-21). Marked departures from the EC distribution is indicative that processes other than dilution are operative. Sulphate (Fig. 2b) is an example of a concentration distribution which does not depart significantly from that of EC.

The alkaline discharge from the oxidation pond has clearly modified the pH of the wetland waters (Fig. 2c). The reference wetland waters were moderately acidic with a pH range of 5.2-6.5, whereas the sewage wetland waters were 1.5 pH units higher on average. The EC distribution, and also our flow gaugings and hydrodynamic studies (Williams, B.W.; Bleeker, F.; Cooke, J.G.; Cooper, A.B. Transport of wastewater through a natural wetland. Paper presented at New Zealand Water Supply and Disposal Association Conference, Tauranga, August 1990), show that water at site 2 is almost entirely derived from the oxidation pond. The significant drop in pH between sites 1 and 2 is therefore attributed to the buffering capacity of the wetland detritus and soil.

Similarly, there were significant differences in dissolved oxygen between sites (Fig. 2d). This was due to a number of factors. Firstly, the oxidation pond effluent was quite oxic. The effluent, and a ponded area immediately after the discharge (Site 1) were often supersaturated at the surface in summer due to a thick scum of the alga *Oscillatoria mougeotii*. By site 2, however, this alga was no longer evident. Decomposition of the algae probably contributed to the high biochemical oxygen demand (BOD, Fig. 2e) at this site and concomitantly its low DO. The significant rise in DO between sites 4 and 5 is probably due to: (i) all the algal-derived BOD having been exerted, (ii) dilution with catchment waters of low BOD (Fig. 2e), and, (iii) reaeration due to channelization and an approximately 0.5 m drop in bed level. The addition of lower DO waters from the reference wetland resulted in waters with a mean DO of 3.4 g m-3 at site 8.

There was a large range in the organic content of the waters (as reflected by oxygen demand). Much of the variation encountered in the reference wetland is due to sampling artifacts. During summer, it was often difficult to sample the water without disturbing bottom soil and much of the variation in BOD (Fig. 2e) and COD (Fig. 20 was due to the inclusion of this particulate material; when samples were filtered (Fig. 2g) the concentration of oxygen demanding organics in the reference wetland waters was significantly less than those in the sewage wetland.

Ammonium-nitrogen comprised more than 90% of the dissolved reduced-N (Kjeldahl-N) pool dissolved in the sewage wetland waters (only the dissolved component is reported for the same reasons applying to oxygen demand), whereas in the reference wetland most (>75%) of this pool was organic-N (Fig. 2h & i). There was approximately an order of magnitude difference in the size of the N pool between the sewage and reference wetlands reflecting the large extra N input from the oxidation ponds. Similarly there were big differences in the oxidized N ($N0_3$-N) pool between the two wetlands (Fig. 2j). However, because of the large variance in nitrate concentrations in the sewage wetland, no significant difference could be detected between the two wetlands. Most of the variation in NO_3-N concentration in the sewage wetland is due to seasonal variation in nitrate emanating from the oxidation pond, with high concentrations during the summer and low concentrations in winter. The reverse trend was observed at sites 4, 5 and 8. This is probably due to a combination of increased $N0_3$-N from catchment sources, and decreased nitrate reduction due to lower temperatures, more oxic conditions, and faster travel times.

Phosphorus showed a similar pattern to that of nitrogen in that P in the reference wetland was clearly significantly less than than in the sewage wetland (Fig. 2k & l). There were, however, differences in P between sites within the sewage wetland that did not occur with N. The major difference was that P at the outlet of the wetland system (site 8) was significantly less than the next site upstream in the sewage catchment (site 5) indicating significant P removal was occurring between the two sites.

Soil chemistry

Differences in water chemistry were reflected in the soil chemistry. The Eh_7 of the reference wetland indicated moderately aerobic condition in the top half centimetre of soil whereas soil in the sewage wetland was highly reducing even at this depth (Fig. 3a). The pH of the soil in the surface layer of the sewage wetland was approximately 1.0 pH unit higher than that of the reference wetland, and declined to the relatively constant pH of the reference wetland soil at 9 cm depth (Fig. 3b).

The organic-N (Kjeldahl-N - NH.-N) content of the sewage wetland soil was higher than that of the reference wetland (Fig. 3c), but not as unequivocally so as the ammonium-N fraction (Fig. 3d). The organic-N fraction clearly dominated the total N pool being 2 orders of magnitude greater than the ammonium fraction in the sewage wetland, and 3 orders of magnitude greater in the

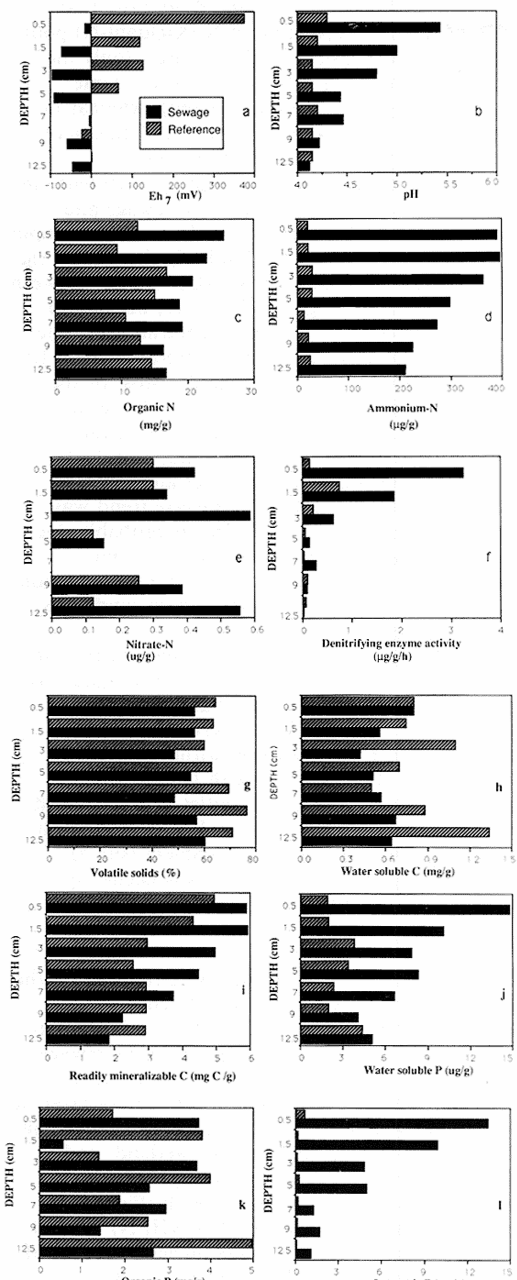


Figure 3: *Soil chemistry at sites within the wetland system. Bars represent the mean value calculated for each wetland at that depth. All concentrations are reported on a dry weight basis.*

reference wetland. In contrast, nitrate-N was found in only trace quantities in the soil of both wetlands (Fig. 3e). This is not unexpected in such a reduced environment; however, it is likely that nitrate from the water column is being reduced in the soil of the sewage wetland since denitrifying enzyme activity was elevated in its surface layers (Fig. 3f).

There was little difference in total carbon (as indicated by volatile solids) between the two wetland soils and in fact the reference wetland soil had a higher proportion of organics than the sewage wetland when expressed on a dry weight basis (Fig. 3g). This is probably due to a higher quantity of inorganics in the sewage wetland as indeed calcium and magnesium were present in much greater concentrations in the sewage wetland soil than that of the reference wetland (data not shown). Similarly there was little difference in water soluble carbon, with again the reference wetland soil generally having slightly higher concentrations (Fig. 3h). However, the sewage wetland soil had higher values of readily mineralizable carbon over the top 7 cm than the reference wetland (Fig. 3i) indicating a greater pool of potentially available carbon.

The difference in phosphorus form concentrations between the two soil types was consistent with that for nitrogen and carbon but even more striking. There were major differences in the levels of both water soluble P (Fig. 3j) and inorganic P (Fig. 31) between the sewage and reference wetland soils, and concentrations of both P forms in the sewage soil decreased with depth. In contrast there was no consistent difference in organic P levels between the two wetland soils (Fig. 3k). In direct contrast to nitrogen, most of the phosphorus in the sewage wetland soil was present in inorganic form.

Vegetation

A total of 50 species of vascular plants was recorded on the transects. Thirty-two species (64%) are native (28 angiosperms and 4 ferns), one of which *(Ranunculus urvilleanus)* is recognised as rare and endangered (Wilson and Given, 1989), and at least one other (the grass *Hierochloe redolens)* as uncommon.

Marked differences in the vegetation were noted between the sewage and reference wetlands. The vegetation of the reference wetland is predominantly a *Baumea-Isachne* grassy sedgeland with scattered shrubs (mostly of *Coprosma tenuicaulis).* This vegetation type, which has the greatest diversity of floristics and community structure, occurs in a modified form on part of the sewage wetland (transects 3 and 5a - Fig. 4). A much larger part of the sewage wetland is

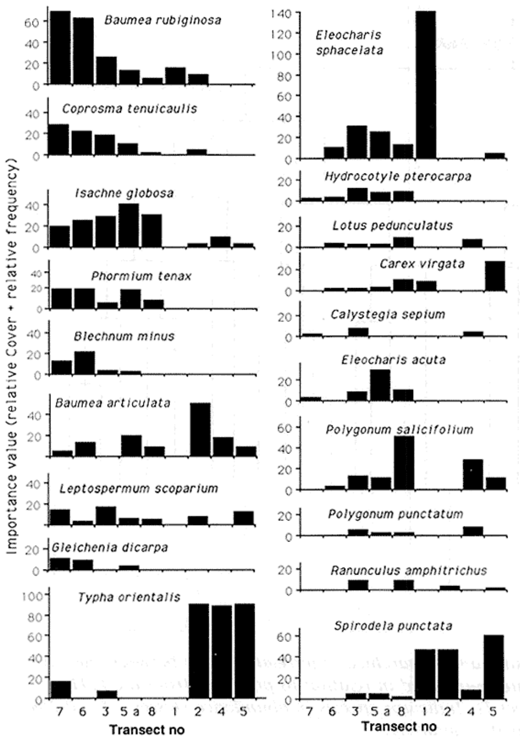


Figure 4: *Importance values (IV) calculated as an average for all sites in each transect for each of the 19 most plentiful species (species recorded in >2 transects and with IV >5 in one or more transects). The transects are ordered in sequence from the highest to lowest average IV of the* 3 *species most characteristic of vegetation in the reference wetland* (Baumea rubiginosa, Coprosma tenuicaulis, Isachne globosa). *Species are arranged from the highest to the lowest IV in the two reference transects (6 and 7).*

dominated by either the bamboo spike sedge *(Eleocharis sphacelata)* or the bulrush *Typha orientalis* (raupo). There was a low species diversity within stands dominated by either of these species (Fig. 4), with many of the native species growing in other communities absent. Compared with the reference wetland, there is a general reduction in abundance and/or presence of some native species. Most notable in this category are the sedge *Baumea rubiginosa,* the shrub *Coprosma tenuicaulis* (swamp coprosma), and the ferns *Blechnum minus* and *Gleichenia dicarpa* (tangle fern). Conversely some native herbs, notably *Ranunculus amphitrichus* and *Polygonum salicifolium* were found only in the sewage wetland (Fig. 4). On sewage wetland sites where the *Baumea-Isachne* grassy sedgeland prevails, some native species remain more or less equally abundant or increase; most notably the swamp grass *Isachne globosa,* flax *(Phormium tenax),* the sedges *Baumea articulata, Eleocharis acuta, E. sphacelata,* and species of *Carex,* especially C. *virgata,* and the herb *Hydrocotyle pterocarpa.* The sewage wetland is characterized also by the presence and particularly abundance of some exotic species, most notably the purple duckweed *Spirodela punctata* (which was not recorded in the reference wetland), and the herbs *Lotus pedunculatus* and *Polygonum punctatum.*

Multivariate cluster analysis (TWINSPAN) clearly distinguished patterns in plant distribution recorded at each site and their interrelationships (Fig. 5). The sites in which the 3 main vegetation types were found cluster consistently. *Eleocharis/Spirodela* stands were found in relatively deep water near the head of the sewage wetland. Raupo dominated stands (transects 2,4,5) were distinguished at a high level. It is notable that a small raupo dominated stand on the reference catchment (sites 7.1-7.4) clusters as a distinct entity with the other sites in the reference wetland. This is because these sites also contain native species (e.g. *Baumea rubiginosa, Coprosma tenuicaulis, Isachne globosa)* which were absent from raupo stands in the sewage wetland, and also lack the exotic duckweed.

Baumea-Isachne grassy sedgeland sites (reference transects 6 & 7, and sites on transects 3, Sa, and 9) clustered in a more complex pattern, but one which is interpretable ecologically. The sites in which *Eleocharis sphacelata* was characteristically in association with the native grass *Isachne globosa* in relatively shallow water (< 10 cm deep) cluster as a distinct entity with relatively close affinity to the *Eleocharis/Spirodela* stands of deeper water. Additionally, the reference catchment sites and many sites at the extremities of the sewage wetland transects separated out from sites which were obviously on the effluent pathway.

Discussion

The discharge of oxidation pond effluent to the wetland has clearly resulted in fundamental changes to the wetland hydrology, water and soil chemistry, and vegetation. The sewage wetland is now permanently flooded whereas the reference wetland is subject to drawdown under drought conditions. Water in the sewage wetland has become more alkaline and nutrient enriched than the adjacent reference wetland and this difference is also reflected in the soil environment. These differences in the physical and chemical environments of the wetland have apparently resulted in fundamental

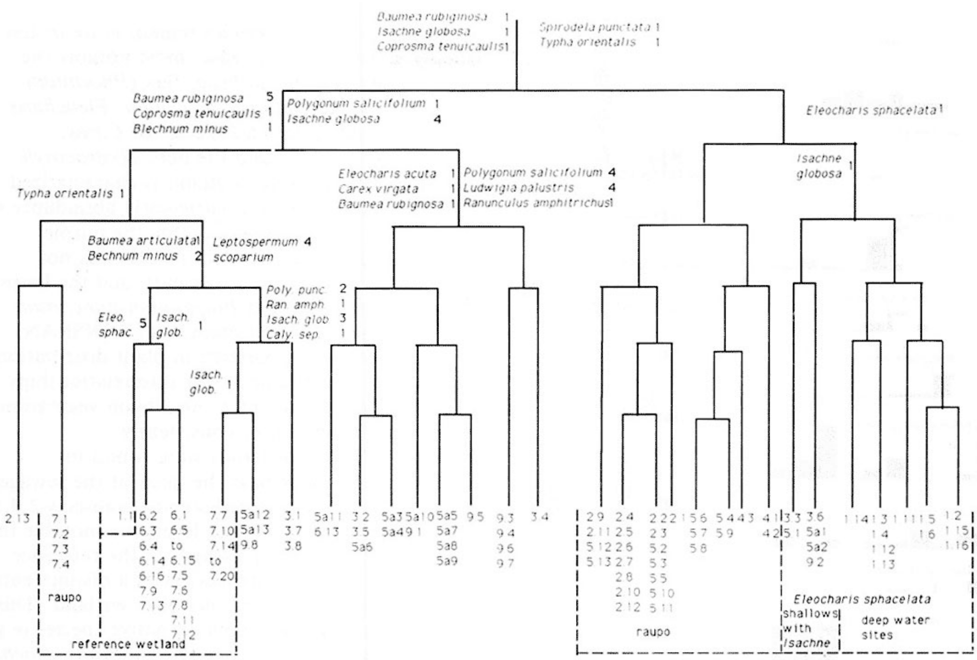


Figure 5: *TWINSPAN dendogram showing classification of sites and hierarchical interrelationships between the site groupings. Figures at the bottom represent each of the 107 sites numbered in relation to position (from the L.H. downstream bank) in each transect (e.g. 1.2 is site* 2 *in transect I). Indicator species in abundance classes 1* (= *0-2% cover) to* 5 (= *>20% cover) are shown for some divisions of the hierarchy.*

changes to the botanical character of the wetland. A diverse *Baumea/Isachne* grassy sedgeland has now been replaced by a relatively uniform community dominated by *Eleocharis sphacelata* and *Typha orientalis*.

Unfortunately, no detailed botanic survey was carried out before effluent disposal into the wetland commenced. Nevertheless, several sources of unpublished information support the notion that the differences we observed between the sewage and reference wetlands are indeed due to change ensuing since the start of the scheme. The Environmental Impact Report (Bay of Islands County Council - Proposed Paihia Sewerage Scheme, 1976) states that the dominant marsh vegetation in the water course below the then proposed discharge point (between our sites 1 and 4) was "dense beds of the rushes *Baumea articulata, Baumea teretifolia,* and *Baumea rubiginosa,* with *Juncus* sp. and *Eleocharis sphacelata* present in places." Patches of flax and raupo were also recorded in dryer marsh areas as well as "tea-tree and bracken". Thus it would appear that at this time, raupo was not dominant, and it would appear from the above qualitative description, and also from water chemistry data, that the 'sewage' wetland was similar in character to the reference wetland of today. After 5 years of effluent discharge to the wetland, a qualitative survey by one of us (NMUC) noted that much of the manuka was dead and that the remaining manuka showed a marked reduction in vigour (Clunie, N.M.U. and Esler, A.E. Unpublished Botany Division Vegetation Series Report No. 439). Clunie and Esler also suggested that in the long term the higher nutrient status may favour raupo ahead of the sedges. Photographic comparisons between the surveys done in 1983 and 1989 also show that there has been local expansion of both *E. sphacelata* and *T. orientalis* communities, probably mainly at the expense of *Baumea* spp. At one specific location (site 8), which was photographed in both the 1983 and 1989 surveys, it was noticeable that whereas in 1983 *Baumea rubiginosa* was dominant with some willow herb *Polygonum salicifolium* intermixed, by 1989 the *B. rubiginosa* had been almost totally displaced by the now dominant *P. salicifolium* with patches of *Isachne, Ludwigia, Eleocharis acuta,* and (less) *E. sphacelata* and *Carex* spp.

Although there is good evidence that changes in vegetation communities within the sewage wetland have occurred since discharge started in 1978, we can only speculate as to whether these changes are due to changes in either the hydrological or the chemical environment, or some combination of both. Evidence from the literature would suggest that water level is likely to be a major factor causing vegetation shifts in the short term. In addition to the obvious killing of vegetation (such as manuka) adapted to a drier environment, the continuous flooding brought about by the effluent discharge will eliminate the wetting and drying cycles that occur in unamended wetlands due to peaks in winter runoff, and summer evapotranspiration respectively. Thus species which require a period of drawdown in which to germinate would be eliminated (Whigham, 1985). This may account in some measure for the absence, e.g., of the fern *Blechnum minus* from the sewage wetland. However, it seems more likely that it is simply the depth of water that has caused the near elimination of these species and also *Baumea rubiginosa* and *Coprosma tenuicaulis*. For example it would appear that *E. sphacelata* is the only rooted macrophyte able to survive in deep (0.4-0.6 m) waters at the head of the wetland (site 1), and even this plant was absent when the water depth exceeded 0.6 m. Conversely, transport within the effluent stream may be responsible for the introduction of other species such as *Ranunculus amphitrichus* and *Polygonum punctatum* which were found only in the sewage wetland. It may be noted that each of these latter species (and also *Polygonum salicifolium* and *Lotus pedunculatus* which only occurred in trace proportions in the reference wetland) grow elsewhere only on sites of moderate to high trophic status, whereas species such as the fern *Gleichenia dicarpa* grow naturally on sites of very low trophic status.

While there are sound ecological reasons for believing that a large increase in nutrient availability in a wetland ecosystem will result in vegetation species shifts (Howard-Williams, 1985; Shaver and Melillo, 1984) much of the literature on this aspect is confusing. In general, it would appear that field surveys on wetlands which have received increased nutrients from sewage effluents have shown species changes (e.g. Finlayson *et al.,* 1986; Mudroch and Capobianco, 1979; Tilton and Kadlec, 1979) whereas specific field experiments designed to simulate sewage additions have often produced inconclusive results (Bayley *et al.,* 1985; Sanville, 1988; Valiela *et al.,* 1985). Notwithstanding the difficulty in conducting realistic field experiments of this type, we believe that a major reason for the inconclusive results from such work is their relatively short-term nature. The primary forcing function in determing ecosystem response of the nutrient additions is the <u>availability.</u> Similarly, for the micro-organisms involved in the recycling of extra nutrients within the wetland they must have a increased supply of <u>available</u> carbon. Our study has shown major differences in plant available forms of nutrients and microbially available forms of carbon, whereas differences in unavailable forms were very much less.

The increase in available nutrient forms in the sewage catchment soil has arisen for two reasons. Firstly the total pool of available nutrient in overlying water has increased, and secondly, the capacity of the wetland detritus and soil to immobilize these nutrient forms has been exceeded. The detrital components of wetland soil, which includes most of the microbial biomass, have a large capacity to trap and immobilize throughflowing nutrients (Richardson, 1985). In fact model results from Dixon and Kadlec (1975) suggest that incoming nutrients are accumulated for at least to years in the detrital components of a wetland. Thus in the case of fertilization experiments, even if the nutrient application rate may realistically simulate loading from wastewater, the initial immobilization potential of wetland detritus will almost certainly ensure that there is little nutrient available for plant uptake which could initiate change in nutrient cycling and species composition. Thus, we believe the definitive experiment on the response of wetland ecosystems to added nutrients remains to be done.

However, considering that hydrological conditions in the sewage wetland have remained relatively stable since at least 1983, it appears likely that it is the fertilization effect of the effluent that is responsible for the expansion of *T. orientalis* reedland. There are numerous examples cited in the literature where aggressive *Typha* spp. have replaced other species in similar conditions (Kadlec, 1981; Whigham, 1985) and at least one where *Typha* spp. have been replaced by an even more aggressive *Glyceria grandis* (Mudroch and Capobianco, 1979). Certainly the annual cycle of growth and die-back favours expansion of *T. orientalis* under non-limiting conditions. In a parallel study we (Cooke and Cooper, in prep.) have shown that the rate of decomposition of fallen *T. orientalis* leaves is much greater in the sewage wetland than the reference wetland. Observation would also suggest that production and litter fall of *T. orientalis* are also greater in the sewage catchment than in the reference catchment and further studies are underway to confirm this. Higher rates of production, litter fall, and decomposition in the sewage wetland result in a higher rate of wetland soil development and creation of highly reducing conditions at the soil-water interface. At the margins of *T. orientalis* communities, especially where this coincides with a channel of effluent-enriched water, this in turn appears to provide favourable conditions for further colonization of the species. The modification of the hydrologic environment caused by vegetation changes has important implications to the transport of wastewater through the wetland (Williams *et al.,* 1990, unpublished). Within the interior of *T. orientalis* communities in the sewage wetland, rapid nutrient cycling in combination with reduction in light climate at the surface of the wetland caused by the vigorous raupo growth, may well explain the almost total exclusion of other species.

Unlike the very gradual changes alluded to by Odum (1987) in the introduction of this paper, the vegetation changes which we have documented occurred very rapidly. As there also has been a drastic change in environmental conditions within the sewage wetland, the inescapable conclusion is that the vegetation changes have been allogenic. Undoubtedly the addition of effluent to the wetland has completely disrupted autogenic succession processes. However, it may be that the system has now reached hydrologic equilibrium beyond which allogenic influences will be spatially quite small. The aggradation of the wetland soil surface within the rapidly cycling raupo communities will eventually cause the wetland soil to dry out locally. Under these conditions it is conceivable that ombrotrophic conditions will eventually result and autogenic succession will again be dominant.

For logistical reasons, we restricted our study to a relatively small area close to the effluent discharge point. Downstream of our site 8 there are extensive wetland areas, albeit highly channelized. In this paper we have purposely avoided discussing nutrient removal processes. However, the changes in vegetation we have observed, and the processes leading to those changes, allow us to make some deductions as to downstream effects. The rapid allogenic succession that has accompanied the introduction of the effluent would, in the first instance result in rapid uptake of nutrients from floodwaters. However, as succession has proceeded, and with continuing input of nutrient from the effluent, the proportion of the total nutrient pool which is reutilized by wetland plants will decrease. The rise in swamp soil level will also decrease the effective area available for nutrient removal and result in increasing channelisation. Therefore the net result will be an increase in the pool of nutrients leaving the system. It is therefore likely that the vegetation changes that occurred within the sewage wetland, especially the expansion of *Typha* reedland, will also occur progressively downstream of site 8. However, the time taken for this to occur throughout the entire watercourse may be very large in terms of the life expectancy of a wastewater treatment system.

Are the environmental changes we have outlined unacceptable? The Environmental Impact Report for the Paihia sewage scheme stated that the wetland area below the discharge point was of 'low ecological value', so perhaps it could be argued that any changes which have occurred are not of concern. However, criteria determining this argument are largely anthropocentric. It is only by studies such as this one that the nature of the system, and the processes and changes in it can be determined, so that rational discussion and evaluation can ensue.

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