LWRRDC Project QPI 26

Nutrient control in irrigation drainage systems using artificial wetlands

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Abstract

The potential for using constructed wetlands to improve the quality of irrigation drainage waters was assessed at an experimental site in the Burdekin River Irrigation Area in north Queensland. Two detailed performance trials were undertaken in 1999 to quantify changes in concentrations and loads of suspended solids, phosphorus and nitrogen between wetland inlets and outlets. In addition, concentrations were monitored along each wetland bay to assess the rate of change with distance from the inlet. A study was also made of wetland removal of the herbicides atrazine and diuron.

Intake water to the wetlands during the trials contained mean concentrations of suspended solids of ≤95 mg L^{-1}, total phosphorus ≤0.09 mg L^{-1}, and total nitrogen ≤0.63 mg L^{-1}. The wetlands removed 60-70% of the suspended solids load (compared with 16-49% from a control bay without vegetation) and concentrations at bay outlets were significantly lower than at inlets. However, there was a net increase (ranging from 0.4% to 67%) in total phosphorus loads, and concentrations at the outlets of vegetated bays were significantly higher than at inlets. Changes in total nitrogen loads were relatively small and variable (within the range ± 22%), and concentrations at outlets were generally not significantly different from those at inlets. The wetlands at the time of these trials had been established for five years. Results from monitoring in 1994/95 indicated a much greater ability of the wetlands to remove phosphorus, although results for suspended solids and nitrogen were comparable. Reasons for the diminished phosphorus removal in 1999 may have been due to the changed condition of the wetlands as well as differences in the phosphorus composition of water entering the wetlands.

The vegetated wetlands were effective in removing around 40% of atrazine and diuron applied at the inlets, comparable with the removal of 52% of a bromide tracer added at the same time. In complementary glasshouse studies, growth and photosynthetic activity of *Phragmites australis*, *Schoenoplectus validus*, *Veteveria zizanioides* and *Typha* sp. were found to be unaffected when exposed to atrazine and diuron (applied once-off to sediment) at concentrations up to 200 µg L^{-1}, although growth of phragmites was markedly reduced at 2000 µg L^{-1}. In addition, *Iris pseudocorus* and *Veteveria zizanioides* (but not *Schoenoplectus validus*), when grown in pots under simulated wetland conditions were found to enhance the degradation of atrazine. Over twenty-five days, iris plants increased the loss of atrazine by 58% and vetiver by 40%, compared with pots without plants.

These studies have raised several important issues concerning the potential for using constructed wetlands with irrigation drainage waters. Implications of the findings for long-term, effective management of wetlands need to be fully evaluated by key specialists and practitioners.

Project team

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Project objectives (as agreed in the amended Project Schedule of 13 July, 1999)

Project QPI 26 investigated the potential for using constructed wetlands to improve the quality of irrigation drainage waters in tropical and sub-tropical areas. It is envisaged that constructed wetlands could play a role in minimising adverse impacts of drainage waters on downstream water bodies, possibly in conjunction with other measures such as on-farm recycling of irrigation runoff. The project objectives were:

1. To obtain information on the rate of contaminant removal in drainage water with distance and time.
2. To undertake a desk audit of at least three key publications and prepare an attachment for these publications which aims to enhance existing knowledge and in particular:
   - include data on nutrient/sediment kinetics in drainage water
   - make comment on wetland issues as they relate to the tropics
3. To present key findings to a national workshop on nutrient stripping.

Objective 1: Research findings

Two aspects of the project are reported here: 1) field studies of wetland removal of contaminants from irrigation drainage waters; and 2) complementary glasshouse studies that addressed issues concerning the use of constructed wetlands to treat waters containing herbicide residues.

1. Field site investigations

Site description

The field site for the project was located in the Burdekin River Irrigation Area (BRIA) in north Queensland, adjacent to drain 1 at the Burdekin Agricultural College. The drain's catchment area at that point was approximately 19 km^2, with sugar cane farming the dominant (>90%) upstream land use. The experimental site was located in a 'borrow pit' area on a sodic duplex soil, from which the surface layers had been removed. The soil was strongly sodic at the surface of the wetlands and remained so at depth (appendix 1), indicating little likelihood of water and solutes leaching below the wetland bays, particularly if kept wet. Water quality and other monitoring commenced at the site in 1994. A major site upgrade was undertaken in 1995/96 so that water flows into and out of the bays could be controlled and measured. However, two major cyclonic events in 1997 caused serious flooding at the site and necessitated a revision of the project objectives. Details of the wetland design and information on the early stages of the project are provided in appendix 1.

Briefly, there were four wetland bays, each 60 m long, 6 m wide and of varying depth (*ca.* 1 m, see appendix 1) and each set up with calibrated tipping buckets and data loggers to record water flows at bay inlets and outlets. Water was pumped to the inlets from the adjacent drain. An automatic weather station at the site recorded rainfall, air temperature, solar radiation and wind run. Three bays (A, B and C) were planted with local wetland plant species in 1994; the fourth was a 'control' of similar dimensions (in 1999), but with no macrophytes. For the 1999 trials, all three planted bays contained dense stands of mature macrophytes, dominated by *Phragmites karka*.

Performance trials and tracer studies, 1999

To address the revised objective 1 (above), we undertook two detailed field performance trials in September and November 1999. Sugar cane crops in the district were at the harvest/early crop stage of development during the first performance trial (3-13/9/99), while the second trial (17-25/11/99) occurred during a peak period in the cropping cycle for fertiliser use and irrigation.

Each trial involved intensive water quality monitoring over approximately ten days, of water flowing into and out of each bay. Wetland removal of suspended solids (SS), nitrogen (N), phosphorus (P) and the herbicides atrazine and diuron was assessed (the herbicides in November only). For SS and nutrients, the monitoring period for loads at outlets was offset from that at inlets to take account of the hydraulic detention time. Monitoring was also carried out daily (for five days) at six transect points, to assess changes in concentrations along each bay. Tracer studies using bromide (Br) and Rhodamine-WT (Rwt) were carried out in conjunction with each trial to determine the hydraulic detention times for each bay. The first tracer study was carried out prior to the first performance trial and included both Br and Rwt; the second tracer study (Br only) was carried out at the same time as the second performance trial. Results of the tracer studies are presented in appendix 2 and details of experimental procedures for the performance trials (including sampling, analysis and data calculations) in appendix 3.

Suspended solids

Mean SS concentrations in drain water entering the wetlands were 72 mg L^{-1} in September and 95 mg L^{-1} in November (figure 1a), typical of concentrations measured at the inlet over the course of the project (1994/95 median 95 mg L^{-1}, n = 50).

During both performance trials mean SS concentrations at all bay outlets were significantly lower (P<0.05) than at the inlet (figure 1a) and mean concentrations at outlets of bays A, B and C were significantly lower than that at bay D (control, with no vegetation). Over the two trials, bays A-C removed 60-70% of the total SS load, while bay D removed 16% in September and 49% in November (figure 1b). The average rate of SS removal form vegetated bays was 1.1 kg day^{-1} in September and 1.4 kg day^{-1} in November. Flow rates in bay D were comparable for the two trials so it is not clear why there was a much greater reduction in the SS load in November. Although particle size was not measured, it is possible that the SS load in November contained a higher proportion of coarse-grained particles that settled out more readily during transit in bay D. The ionic strength of inlet waters was similar for both trials (electrical conductivity *ca.* 0.17 dS m^{-1}).

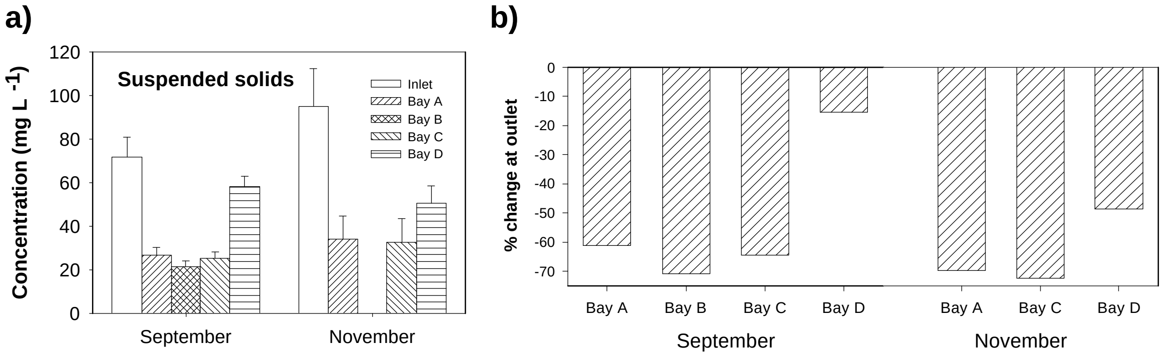

Figure 1 Suspended solids, a) mean concentrations at wetlands inlet and outlets in the September and November trials; and b) changes in loads between inlet and outlets. Vertical lines in a) show standard deviations. Note that the changes in loads were based on flow-weighted mean concentrations (appendix 3).

Phosphorus

Mean total P concentrations in inlet waters were 0.06 mg L^{-1} in September and 0.09 mg L^{-1} in November (figure 2a), compared with a median concentration of 0.09 mg L^{-1} for previous monitoring at the site (1994/95). These concentrations were higher than those typically found in the Burdekin Dam and irrigation supply channel (Congdon and Lukacs 1996) and exceeded recommended guideline levels for protection of aquatic ecosystems in lowland streams (ANZECC & ARMCANZ 1999). The P composition of inlet waters differed between trials; particulate P comprised a much higher proportion of the total P in November, with relatively less of the other forms of P (figure 3). During both performance trials mean total P concentrations at the outlets of bays A-C (but not bay D) were significantly higher (P<0.05) than at the inlet (figure 2a).

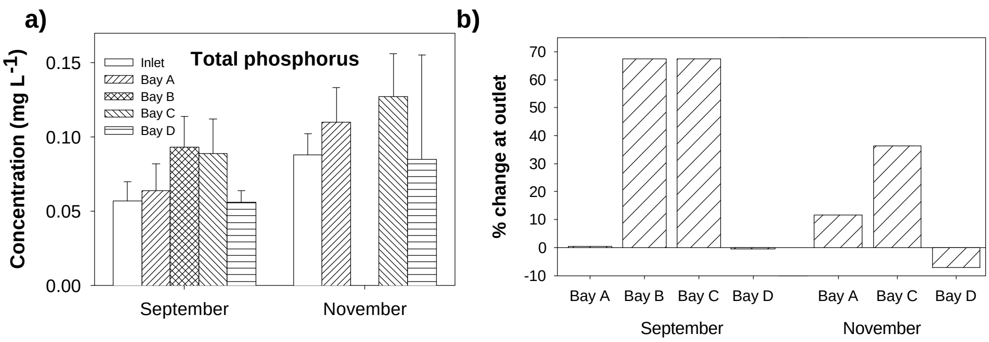

Figure 2 Total phosphorus, a) mean concentrations at wetlands inlet and outlets in the September and November trials; and b) changes in loads between inlet and outlets. Vertical lines in a) show standard deviations. Note that the changes in loads were based on flow-weighted mean concentrations (appendix 3).

In both trials, total P loads at outlets of bays A-C were higher than at the inlets, although the increase in bay A in September was negligible (figure 2b). In contrast, total P loads in bay D decreased on both occasions, although by a negligible amount in September. Thus, the marked decreases found in SS loads in all bays were not reflected in decreased total P loads in bays A-C, even though in November, 39% of P in the inlet water was particulate P (figure 3). The average rate of net increase in P loads at the outlets of vegetated bays was 0.4 g day^{-1} in September and 0.3 g day^{-1} in November. Although there were marked decreases in orthophosphate-P (PO_4-P) loads in both trials (excluding bay B in November), in bays A-C these were offset by larger increases (combined) in particulate P and dissolved organic P loads (appendix 4). Thus, in both trials the vegetated bays were net sources of P, rather than sinks.

These changes in the P composition of water passing through the wetlands are consistent with increased P uptake and cycling by the microflora (and possibly also by macrophytes), enrichment of the SS load through settling of coarser particles of lower P content and sorption of PO_4-P onto the SS fraction. The increased P content of the SS fraction at bay outlets (particularly bays A-C) compared with that at the inlet (figure 4a) further suggests that physical, chemical or biochemical enrichment processes occurred during transit through the bays. However, it is not known to what extent these processes actually occurred to account for the measured changes in P loads.

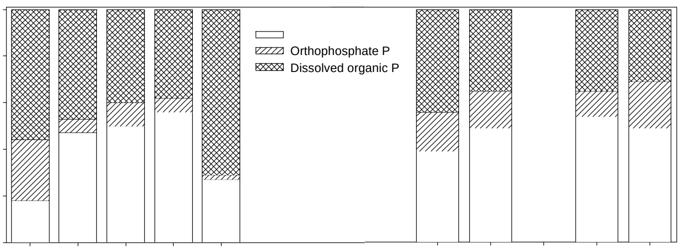

Figure 3 Relative contents (%) of particulate, dissolved organic and orthophosphate forms of phosphorus in inlet and outlet waters in the September and November trials.

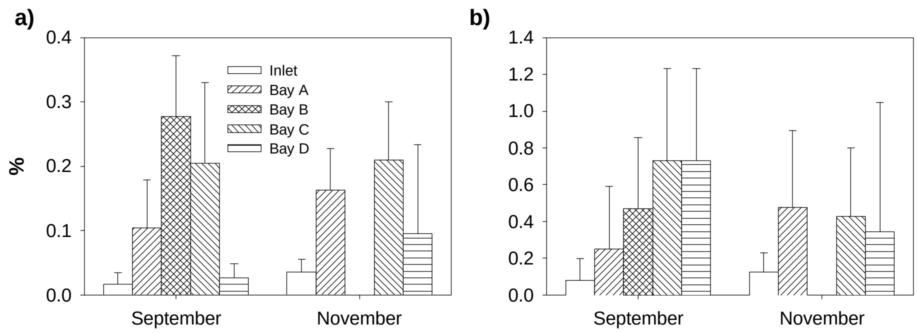

Figure 4 Percentage composition of a) phosphorus and b) nitrogen in suspended solids at inlet and outlets.

Nitrogen

Mean total N concentrations in inlet water from the drain were 0.42 mg L^{-1} in September and 0.63 mg L^{-1} in November (figure 5a). These concentrations were typical of those measured previously at the site (1994/95 median 0.64 mg L^{-1}, n = 41). Although these concentrations were relatively low and within suggested guideline values for the protection of aquatic ecosystems (ANZECC & ARMCANZ 1999), there was considerable variability (eg highest concentrations measured in the November trial was 1.4 mg L^{-1} and 1.5 mg L^{-1} in 1994/95). The N composition of inlet waters differed between trials, in November being lower in dissolved organic N but higher in other forms of N (figure 6).

Wetland performance showed variable results for total N. In September, N loads were reduced by 18% in bay A, but were increased by 11-21% in bays B, C and D; while in November, bays A, C and D all reduced total N loads by 8-22% (figure 5b). However, the reductions (where they occurred) were relatively small and (apart from bay A in September) were not reflected in significantly lower (P<0.05) mean total N concentrations at bay outlets compared with the inlet (figure 5a). The average rate of N removal from vegetated bays was 0.6 g day^{-1} in September and 3.7 g day^{-1} in November.

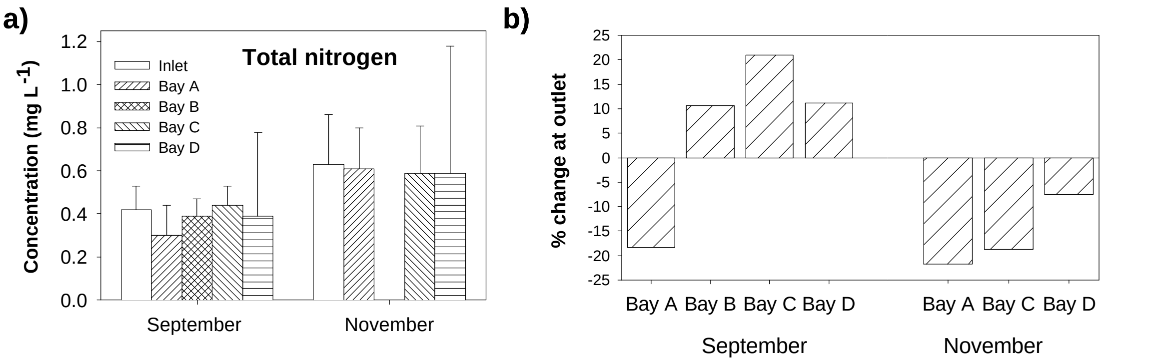

Figure 5 Total nitrogen, a) mean concentrations at wetlands inlet and outlets in the September and November trials; and b) changes in loads between inlet and outlets. Vertical lines in a) show standard deviations. Note that the changes in loads were based on flow-weighted mean concentrations (appendix 3).

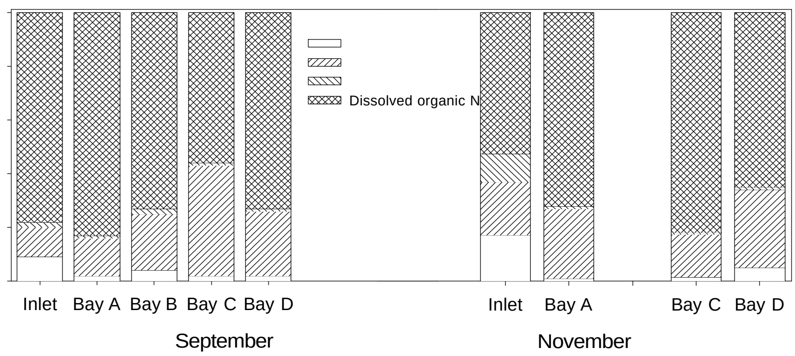

Figure 6 Relative contents (%) of particulate, dissolved organic and inorganic forms of nitrogen in inlet and outlet waters in the September and November trials.

The wetlands consistently reduced ammonium-N (NH_4-N) and oxidised N (NO_x-N) loads (appendix 4). This occurred in all bays but was generally less marked in bay D and is reflected in the lower dissolved inorganic N content of outlet waters, when compared with inlet water (figure 6). However, the lower outlet NO_x-N concentrations that resulted from the large percentage reductions in loads (appendix 4) were relatively less important, since inlet concentrations of NO_x-N were low in both trials (mean concentrations <0.1 mg L^{-1}).

The wetlands had variable effects on loads of dissolved organic N and particulate N (appendix 4). Even though outlet SS concentrations were much lower than at the inlet, the N content of the SS fraction at bay outlets was much higher, particularly bays A-C (figure 4b).

As noted for P, these changes in the N composition of water passing through the wetlands may reflect increased biological uptake and cycling and possible enrichment of the SS load through settling of coarser particles of lower N content. Nitrification of NH_4-N and denitrification of NO_x-N may have also occurred. Typically, water discharged from the wetlands (particularly bays A-C) was higher in dissolved organic N and particulate N but lower in dissolved inorganic N (figure 6) than water at the inlet.

Changes in concentrations between inlet and outlets

Daily monitoring at six transect points within each bay was carried out over five days to assess changes in concentrations with increasing distance from the inlet. Although there was considerable variability in concentrations, some consistency in results was evident (summarised in table 1).

Table 1 Summary of the trends observed in mean concentrations of nutrients and suspended solids with increasing distance from bay inlets

Parameter	Comments
Suspended solids	• concentrations decreased with distance along bays • more variable in September than November
Total N and P	• higher concentrations in bays A-C than in bay D (the 'control') • higher concentrations within each bay than at the inlet or outlet
Organic N and P	• variable results, no clear trends
Particulate N and P	• lowest concentrations in bay D • concentrations at inlet and outlets (A-C) lower than within bays (September) • variable changes in concentration along bays (November)
Ammonium	• concentrations decreased with distance from inlet (except D in November) • greatest reduction in concentration occurred in first 20 m
Oxidised N	• inlet concentrations very low in September, higher in November • concentrations in November decreased with increasing distance from inlet (all bays) but remained higher in bay D than in bays A and C
Orthophosphate	• concentrations decreased with distance from inlet in all bays in September, but this trend was less evident in November

Overall, the decreases in concentrations of SS, NH_4-N, NO_x-N and PO_4-P with distance along each bay were the most consistent feature of the results. In addition there was a trend for concentrations of total and particulate N and P to be higher in bays A-C than in bay D, and higher within the bays than at the inlet and outlets (figure 7a, b, c; for clarity only examples for SS, total P and N in bay A are shown). The SS results are consistent with increasing settling out of particulates with distance from the inlet, as would be expected with the passage of water through the bays, particularly those with dense stands of macrophytes (bays A-C). The trend for higher nutrient concentrations within bays A-C than at the inlet and outlets (eg figures 7b, c) may reflect higher organic matter loadings associated with the macrophytes and microflora within these bays. Decreases in inorganic forms of N and P along the bays are consistent with biological uptake and transformation.

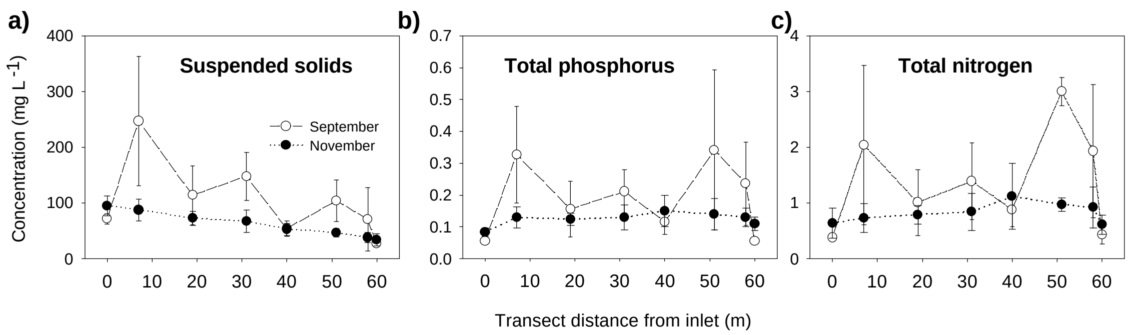

Figure 7 Changes in mean concentration with increasing distance from the inlet of bay A for a) suspended solids, b) total P and c) total N. Vertical bars show standard deviations; (the number of samples at the inlet, transects 1-6 and outlets were 34, 10, 16 respectively in September and 20, 5, 30 in November).

Comparisons with earlier wetland performance

As discussed above, under the conditions that prevailed for the two performance trials in 1999, the vegetated wetlands generally had relatively little net effect on N loadings (although some reductions ≤22% did occur) and they generally were a source of P (increases of up to 67%). Water draining out of the wetlands, although lower in concentrations of SS, was higher in particulate and dissolved organic forms of N and P. However, these findings need to be considered in the context of the site history and the condition of the wetlands at the time of the trials. Particular features that may have affected wetland performance included:

- vegetated bays contained dense stands of mature macrophytes (five years old) and associated microflora and litter, the macrophytes had never been harvested;
- flow through the wetlands was intermittent in the months prior to the trials (due to pump problems) and they may not have been sufficiently well flushed; for example, to remove accumulated litter, and P released from sediments under (possibly) lower redox conditions;
- there may have been considerable build-up of sediment in the bays, not only due to settling out of SS from inflowing water under 'normal' flows, but also from flooding and bank erosion; thus the bays may have been much shallower than originally designed.

While the importance of these issues in determining wetland performance in the trials is not known, a comparison with results from the site in 1994/95 suggests that under the conditions at that time, the wetlands were much more effective in removing P than in 1999 (figure 8b), although their performance in removing N was little better (figure 8c). Wetland effectiveness in removing SS (bay A) was comparable between 94/95 and 1999, although less so in September (figure 8a). Detention times in 1994/95 are not known. Although the factors noted above may have contributed to the less effective P removal in 1999, it is also worth noting that in 1994/95, particulate P comprised 55% (median value) of the total P load in inlet water, but in the 1999 trials it was only 18% and 39%. Thus, the differences in effectiveness between the two years may have reflected different processes of P removal, since there was less potential in 1999 for P to be removed by flocculation and settling out of the suspended solids fraction.

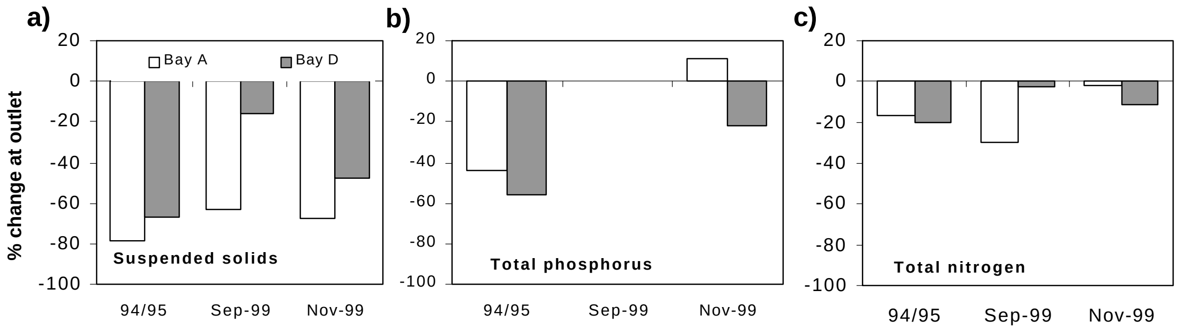

Figure 8 Comparison of wetland performance between 1994/95 and 1999, showing changes in loads of a) suspended solids, b) total phosphorus and c) total nitrogen between bay inlets and outlets. For clarity, only results for bays A and D are shown. Note that in b) the changes in September were within ± 0.5%; note also that the changes in loads were based on flow-weighted mean concentrations in 1999 and on median concentrations in 1994/95 (appendix 3).

Herbicide residues

The herbicides atrazine and diuron are commonly used in sugar cane growing areas in Australia, including the BRIA (Hamilton and Haydon 1996). In the second performance trial, small amounts of each herbicide were added at the inlets of each bay to simulate concentrations that potentially could be found in irrigation drainage waters following field applications. During the trial water at the intake to the wetlands was monitored at least daily for background concentrations of the two herbicides. Eighty-three percent of these samples contained detectable (but low) concentrations of atrazine (0.06-0.50 $\mu g\ L^{-1}$) and 92% contained diuron (0.37-1.2 $\mu g\ L^{-1}$). Much higher concentrations (up to 88 $\mu g\ L^{-1}$ atrazine and 69 $\mu g\ L^{-1}$ diuron, figure 9a) were measured at bay outlets following application at inlets; herbicide recoveries at outlets were calculated to assess the extent of wetland removal.

Recoveries of atrazine at bay outlets ranged from 41% to 75% of that applied, while recoveries of diuron ranged from 51% to 89%. Highest recoveries of both herbicides were recorded from the non-vegetated ‘control’ bay D (figure 9b). For both herbicides, there was little difference in recoveries between bays A and C, with an average of 42% of the atrazine and 53% of the diuron recovered from these bays. Concentrations on the last day of sampling (eight days after application at the inlets) were < 10 $\mu g\ L^{-1}$ for both herbicides (figure 9a), so although these levels were not as low as background concentrations, it is unlikely that recoveries would have been markedly higher had sampling been continued beyond day eight.

Recovery of a bromide tracer solution added at the same time as the herbicides (appendix 2) ranged from 50% to 79% for bays A, C and D, with the highest recovery again recorded from bay D. The average recovery from bays A and C was 52% (figure 9b). Thus, recoveries of bromide were comparable with those of the two herbicides. Moreover, the concentrations of bromide, atrazine and diuron at bay outlets showed similar changes with time over the monitoring period (figure 9a), suggesting that they experienced similar flow paths and fates (eg physical, chemical and/or biological processing) in being transported through the bays. However, there was some evidence of (a) a slight lag in the rate of increase of bromide concentrations at the outlet compared with the herbicides, and (b) relatively lower concentrations of bromide by day eight (figure 9a).

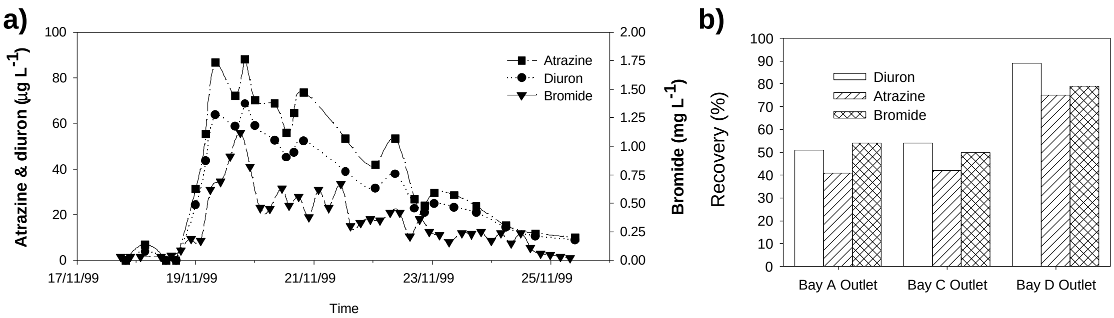

Figure 9 Diuron, atrazine and bromide (a) concentrations at the outlet of bay A, and (b) recoveries at outlets of bays A, C and D expressed as a percentage of amounts applied at inlets on 17 November 1999.

The similarity of the results for atrazine, diuron and bromide was surprising given their differences in chemical properties. The high recoveries of diuron were particularly surprising because of its tendency (more so than atrazine) to sorb onto suspended clay particles (Hamilton & Haydon 1996). Thus, both atrazine and diuron appear to have effectively performed as tracers of water and solute movement in this study. The bromide recoveries recorded were comparable with those reported for other wetland tracer studies (Netter 1994).

Conclusions from the 1999 field trials

In evaluating the wetland performance in 1999, it is important to recognise that the condition of the wetlands at that time may have influenced their performance (discussed above), but this may not necessarily have reflected inherent weaknesses in the concept of using constructed wetlands to improve the quality of irrigation drainage waters. Rather, the issues raised may be more to do with how constructed wetlands are managed so that they maintain their effectiveness. Aspects of wetland design may also need evaluation in assessing the results of the performance trials.

The wetlands showed a considerable capacity for removing SS from incoming waters, but generally showed only a limited ability for net N removal (insufficient to significantly lower concentrations in effluent waters) and they generally increased total P loads and concentrations in waters discharged. Previous results from 1994/95 showed a much greater capacity for P removal, but it is not known whether this reflected the changed condition of the wetlands in 1999, or the differing P compositions of the inflowing waters over the two study periods. Certainly, there was a greater potential for P removal by flocculation and settling in 1995/95 due to the higher proportion of P associated with the SS fraction.

Results of the herbicide trial indicated that the wetlands were effective in removing around 40% of the applied atrazine and diuron. This suggests there is some potential for using constructed wetlands to reduce concentrations of these two herbicides in irrigation drainage waters. However, the mechanisms by which these reductions in herbicide loadings occurred are not known. The very similar results for Br added at the same time suggests similar processes occurred, which is somewhat surprising given their very different chemical properties, and suggests that further work is required in this area. A better understanding of these processes is required before the real potential for using wetlands for herbicide removal can be evaluated and the concept applied.

Hence, the following recommendations are made:

- Consolidate existing knowledge in the design and management of constructed wetlands for irrigation tailwaters - this may be best achieved through a small workshop with key scientists to scope guidelines for industry use. LWRRDC/ANCID are likely to be best placed to facilitate this.
- Undertake further research to better define constructed wetland performance - both mass balance data and the quantification of relative removal processes is required (for nutrients, suspended solids and pesticides). Cost:benefit analyses are likely to be needed to justify the significant expenditure for the establishment of such a rigorously monitored and studied system.
- Improve technology transfer to government, community and industry professionals. This should initially focus on explaining limitations of the technology and providing appropriate guidelines for their use (see above).

2. Glasshouse studies

In addition to the field trials, two glasshouse studies were undertaken:

1. To assess the performance of several wetland species following exposure to atrazine and diuron solutions at environmentally relevant concentrations; and
2. To examine whether certain wetland plant species can enhance the degradation of atrazine.

Effects of herbicides on wetland plant species

For wetlands to be effective in enhancing the quality of drainage waters from agricultural lands, the plant species used need to be able to withstand exposure to commonly used herbicides in these areas, at least at the concentrations likely to be found in runoff and drainage waters. In this trial, wetland ecosystems were simulated in 200 mm diameter plastic-lined pots filled with 4 kg of a clay loam sediment, with a free water table of 15 mm maintained above the surface of the sediment. Plants of each of the four species evaluated (*Phragmites australis*, *Schoenoplectus validus*, *Vetiveria zizanioides* and *Typha*) were established in the pots and after approximately eight weeks they were exposed to atrazine or diuron at concentrations in the free water above the sediment of 20, 200 and 2000 $\mu g\ L^{-1}$, in a once-only addition. Application rates were chosen to represent concentrations that might be found in natural waterways, the highest rate simulating an accidental spillage event. Measurements were made over the following 28 days of cumulative plant water use, leaf area, chlorophyll fluorescence (physiological response, measured by PAM fluorometry), plant dry weights and tissue herbicide content. The experiment was a replicated, completely randomised block design; data for each species were analysed separately by ANOVA (Statistica 5.1®) and means compared using protected least squares differences at each time of measurement. Rachael Cull presented a paper on the vetiver component of this work at the 2nd International Conference on Vetiver, held in Thailand in January 2000 (Cull *et al.* 2000).

Key findings of the study included:

- The growth and activity of phragmites was seriously impaired by both herbicides at the highest rate of application (2000 $\mu g\ L^{-1}$), but not at lower rates.
- Growth of the other three species was not affected by exposure to either herbicide at rates up to 2000 $\mu g\ L^{-1}$; however, both schoenoplectus and typha showed a significant ($P<0.05$) decline in photosynthetic activity at 2000 $\mu g\ L^{-1}$ of diuron (shown for typha in figure 10).
- Vetiver was not affected by exposure to the herbicides, showing no impairment of growth or photosynthetic activity at the rates tested.

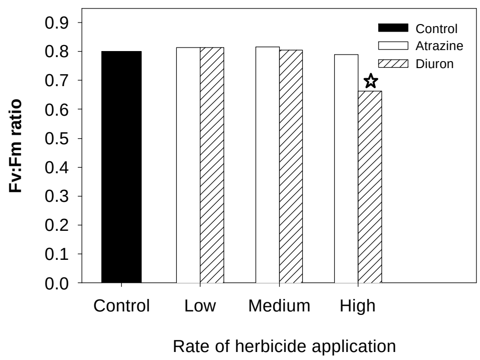

Figure 10 Effects of low (20 $\mu g\ L^{-1}$), medium (200 $\mu g\ L^{-1}$) and high (2000 $\mu g\ L^{-1}$) rates of atrazine and diuron application on photosynthetic activity of typha. The Fv:Fm ratio is a measure of chlorophyll fluorescence; the star indicates means were significantly different at P <0.05.

Conclusions from 1st glasshouse trial

Growth and photosynthetic activity of *Phragmites australis, Schoenoplectus validus, Veteveria zizanioides* and *Typha* sp. were unaffected when exposed to atrazine and diuron at concentrations up to 200 µg L^{-1}. However, the sensitivity of phragmites to both atrazine and diuron at higher concentrations raises some concern about its long-term viability for use in wetlands in agricultural areas where these herbicides are applied. Even though the concentration at which this occurred (2000 µg L^{-1}) is unlikely to occur often in natural waterways, concentrations of 200 µg L^{-1} or more are not uncommon when rainfall events occur soon after herbicide applications on farms (BW Simpson, pers. comm.). Although no adverse effects were found at 200 µg L^{-1} of either herbicide, further research is needed to determine the safe upper limit of exposure within the range, 200-2000 µg L^{-1}. Moreover, we do not know what effects these herbicides have (on phragmites and other species) when applied in combination, rather than individually as occurred in this study. The implications of the more subtle effects of diuron (at 2000 µg L^{-1}) on the photosynthetic activity of schoenoplectus are presently not clear but may be of concern for its long-term viability in constructed wetlands, particularly if exposure is chronic (rather than once-off), even at lower concentrations.

Enhanced degradation of atrazine by wetland plants

In a second glasshouse trial using the simulated wetland pot system outlined above, we examined whether wetland plants could enhance the removal or degradation of atrazine. The study was undertaken to follow up on preliminary reports that certain wetland species could reduce atrazine levels in water or sediment (Rice *et al.* 1996, Coglan 1997). The species examined were *Iris pseudocorus*, *Schoenoplectus validus* (club rush) and *Vetiveria zizanioides*, in a replicated, completely randomised block design. Eight weeks after planting, a once-only application of 1000 µg L^{-1} atrazine (in one litre of solution) was made to each pot (as described above) and the plants were harvested 25 days later. Measurements made during the trial included plant water use, redox potential and photosynthetic activity (by PAM fluorometry). Total amounts of atrazine recovered from each pot were determined by analysis of sediment, water and plant tissues (roots and shoots) for atrazine. Data were analysed by ANOVA, with means compared using protected least squares differences.

Steven Winters submitted a report on this work (Winters 1999) as part of his fourth-year course (B. Land Resource Science) at the University of Queensland and was subsequently awarded a medal and cash prize by the Australian Institute of Agricultural Science (SE Qld branch) for the best project in the category of *Sustainability in agricultural practice*.

The major findings included:

- Total recovery of atrazine from pots growing iris was significantly lower (P <0.05) than for other species, and lower with vetiver than with schoenoplectus and pots without plants (figure 11).
- Overall, 41-42% of the applied atrazine was recovered from control pots and those growing schoenoplectus, while recovery from vetiver pots was 26% and iris, 18%.
- Iris plants increased the loss of atrazine by 58% and vetiver by 40%, compared with pots without plants.
- Only very low levels of desethyl atrazine, a common degradation product of atrazine, were detected in any pots.

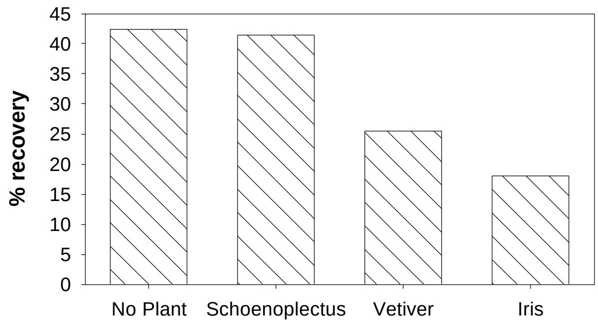

Figure 11 Effect of different plant species on the recovery of atrazine residues from simulated wetland systems, 25 days after application.

Conclusions form 2nd glasshouse trial

Although the mechanisms for the enhanced degradation of atrazine in the presence of iris and vetiver plants were not determined in this study, the results suggest that these species may offer the potential for enhancing water quality, when used in constructed wetlands in agricultural areas where atrazine is widely used. Further research is needed to identify these mechanisms, to determine whether other (particularly native) species show similar capabilities and to evaluate how such species can be successfully incorporated into constructed wetland design.

Objective 2: Desk audit

A review of studies that have examined the treatment of agricultural (non-point) source pollution by constructed wetland technology was undertaken as part of this research project. Few studies were found that had specifically targeted the role of constructed wetlands to mitigate the effects of irrigation drainage waters; however, sufficient examples exist to enable a worthwhile review.

In Australia, Bowmer *et al.* (1992), Chambers *et al.* (1993), Raisin and Mitchell (1995), Cottingham *et al.* (1997) and Roberts *et al.* (1999) have undertaken the key studies. Breen and Craigie (1997) have also outlined salient design elements to consider in their potential use as part of the Shepparton Irrigation Area Land and Water Salinity Management Plan. These few studies collectively represent the sum of published experimental knowledge in Australia on the incorporation of constructed wetlands as part of irrigation drainage systems. Only one study (Raisin and Mitchell 1995) attempted to sample the movement of the same parcel of water through the constructed wetland to derive accurate measurements of contaminant removal abilities. Additionally, none of these studies attempted to attribute (empirically) the difference between influent and effluent concentrations/loads to specific removal mechanisms (eg direct measurements of denitrification rates utilising acetylene reduction techniques – see Comin *et al.* 1997)

The studies have all been undertaken in the temperate zone, and except for the work by Chambers *et al.* (1993) have been undertaken in the Murray Darling Basin. Such climatic differences prevent direct comparisons to tropical systems, especially in relation to relative seasonal contaminant loads. Despite these serious limitations (acknowledged by many of the authors), results from most of these studies share several features in common:

- Highly variable influent nutrient concentrations, particularly daily and seasonal differences
- Abilities to reduce nutrient loads were variable – they act as both sinks and sources of nutrients
- Conclusions on their prospective use as part of irrigation drainage systems are equivocal

Given such results, Breen and Craigie (1997) proposed a design approach based on the incorporation of removal mechanisms specifically tailored to the nature of the contaminants.

1. Biological and chemical processes involving soluble materials (eg uptake of nutrient by epiphytes, ad/de-sorption of phosphorus on/from particles).
2. Coagulation and filtration of small (colloidal) particles – ie those particles in the size/density range that are too small to settle under all but the most quiescent of conditions (eg adhesion of colloids and particles on the surface of macrophytes and epiphytes).
3. Physical sedimentation of particles (eg sedimentation occurring in wetlands due to decreased water velocity – enhanced by presence of macrophytes that further reduce turbulence and water velocities).

The quality of tailwaters in this study (in the Burdekin River Irrigation Area) has previously been described by Congdon and Lukacs (1996). It is typical of many irrigation areas – a high proportion of fine colloids and therefore a high phosphorus load, with nitrogen loads being more "spiky". Breen and Craigie (1997) have listed desirable features of a constructed wetland for treating such types of irrigation tailwaters – based on the Shepparton Irrigation Area. Features should include:

- over-excavation of drain bed to create semi-permanent pools at zero flow – up to 1m deeper.

- alternate the deep zones with shallow strips (0.1m deep) - to allow emergent vegetation to colonise
- suppression of emergent vegetation in deep zones by riparian tree plantings (shading)
- deep zones at all drain inlets, and on upstream and downstream sides of road crossings
- installation of simple pervious rock walls to separate pool zones – increases water levels and retention times. Height of structures to not influence a 1:2 year event.

These design principles (except for riparian shading and rock walls) were incorporated into this research study. However, size and scale effects make successful incorporation into experimental approaches difficult. Other practical considerations need to be considered, such as the required retention time (suggested to be 10-20 days by Breen and Craigie 1997). Incorporation of such retention times, whilst not increasing the impact of a 1:2 year ARI event – seem unachievable. The modification of drains to include wetland habitats appears possible; however, the lack of quantifiable mass-balance studies continues to hamper its application.

Objective 3: Presentation at national workshop

Heather Hunter and George Lukacs presented a paper on the research being undertaken in Project QPI 26 at the Workshop on *Management of Nutrients and Sediment in Irrigation Return Water*, sponsored by ANCID and LWRRDC and held in Brisbane, 20-22 June 1999. An extended abstract of the paper, *Use of constructed wetlands to improve water quality in the Burdekin River Irrigation Area* and a copy of the accompanying slides are presented in Appendix 5.

As noted previously, other communication activities in 1999 included presentations on the glasshouse research findings. These were made by Rachael Cull at the 2nd International Conference on Vetiver in Thailand in January 2000 (Cull *et al.* 2000), and by Steven Winters at a meeting of the Australian Institute of Agricultural Science (SE Qld branch) meeting in December 1999 (Winters 1999).

During the early stages of the project, a field tour to the wetlands site was held in conjunction with the National Wetlands Conference held in Townsville (September 1995). Local government personnel, farmers and other interested groups were invited to attend, as well as conference delegates. Members of the project team gave presentations on various aspects of the project and considerable interest was expressed in the concept and potential of constructed wetlands for improving water quality. A series of posters was also displayed at the field day and were used several times later to inform interested groups about the project.

Publications and reports to date (from 1999 studies)

Cull R, Hunter H, Hunter M & Truong P 2000. Application of vetiver grass technology in off-site pollution control II. Tolerance to herbicides under selected wetland conditions. Preceedings, 2nd International Conference on Vetiver, Phetchaburi, Thailand, 17-21 January, 2000, pp. 407-410.

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Winters S 1999. *Plants reduce atrazine levels in wetlands*. Fourth year report AG421, School of Land and Food, University of Queensland, St Lucia, 37 pp.