Plants as ecosystem engineers in subsurface-flow treatment wetlands

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Abstract Mass balance performance data from side by side studies of planted and unplanted gravel-bed treatment wetlands with horizontal subsurface-flow are compared. Planted systems showed enhanced nitrogen and initial phosphorus removal, but only small improvements in disinfection, BOD, COD and suspended solids removal. Direct nutrient uptake by plants was insufficient to account for more than a fraction of the improved removal shown by planted systems. Roles of plants as ecosystem engineers are summarised, with organic matter production and root-zone oxygen release identified as key factors influencing nutrient transformation and sequestration.

Keywords Aeration; constructed wetlands; natural systems; nutrient removal; phytoremediation; wetland plants

Introduction

The success of human civilisation is largely due to our skills as physical ecosystem engineers. Although these engineering activities are primarily directed towards achieving some specific purpose (e.g. agricultural production), many have major indirect and unintended effects on ecosystems (e.g. water pollution, global warming). We are not, however, the only organisms that directly or indirectly control or modulate the availability of resources to other organisms by modifying the physical state of biotic or abiotic materials (Jones et al., 1997). Beaver dams, termite mounds, coral reefs and forests are obvious examples of non-human engineering that come to mind, but all manner of organisms including microbes, play a role in the creation, modification and maintenance of habitats at a range of spatial scales.

The ecosystem engineering role of plants in treatment wetlands is probably most obvious in surface-flow (SF) systems. Here emergent plant shoots and litter form the main physical structure in the water column, moderating water flow, stabilising sediments, shading and sheltering the water column, providing surfaces for biofilm growth, and providing refuge and habitat for other biota. Here, however, I wish to focus primarily on the role of plants in horizontal subsurface-flow (SSF) wetlands, where wastewaters pass laterally through gravel media in which emergent plants are rooted. In this case, the media provides the main physical structure and the plants’ role, apart from nutrient uptake, is more indirect.

Do plants influence treatment performance?

Dense beds of emergent wetland plants are the most obvious visual feature of SSF wetlands. They undoubtedly play a major role in enhancing their wildlife habitat values, aesthetics and perceived naturalness, but do they actually make much difference to treatment performance?

Because evapotranspiration by plants can significantly affect the hydrological balance of SSF treatment wetlands, comparative assessments ideally need to be made on the basis of mass balances (Howard-Williams, 1985). Many early studies (e.g. Wolverton et al.,
1983) reported greater reductions in the concentration of particular contaminants in planted than unplanted gravel-beds, but most either did not measure flows or did not present data in a way which enabled mass balances to be readily calculated. A range of studies have also reported enhanced transformation of other contaminants in the rhizosphere of wetland plants, such as metals, pesticides and organic compounds.

Published data from side-by-side studies of planted and unplanted SSF treatment wetlands where mass balances could be calculated for biochemical or chemical oxygen demand (BOD, COD), nitrogen or faecal coliform bacteria removal was compiled and compared (see Figures 1 and 2). The studies included hydraulic loading rates of 25–182 mm d$^{-1}$, domestic and agricultural wastewaters with varying levels of preceding treatment, and eight different emergent plant species. Sizes of pilot-scale systems ranged from 18–400 m$^2$ and experimental mesocosm studies from 0.08–6 m$^2$. Plant effects may have been exaggerated in the smaller-scale experimental studies, where high edge to volume ratios result in ramification of rhizomes and roots at the edges of the container and elevated shoot densities and plant biomass (e.g. Tanner, 1994b). Although all of these studies monitored inflow rates, not all reported outflows (Bavor et al., 1987; Gersberg et al., 1983; 1986; 1987) and here mass removals have been calculated assuming overall hydrological balance.

**Oxygen demand reduction**
Comparison of oxygen demand removal (relative to mass loading) for planted and unplanted SSF wetlands shows obvious monotonic relationships between BOD (and COD) mass loading and removal rates (Figure 1), with little difference between planted and unplanted beds. Comparison of paired systems does, however, commonly show reduced (by 2–5 g m$^{-3}$) effluent BOD concentrations for planted beds. Removal of suspended solids, which is primarily a physical process of settling and retention, is also very similar for planted and unplanted beds.

![Figure 1](image-url)

**Figure 1** Comparison of mass loading and removal rates of BOD and COD for planted and unplanted wetlands. Log scales are used to improve visibility of clumped data; note different vertical scales. Lines shown are power fits ($R^2 \geq 0.99$); unplanted $n = 16$ and 7, planted $n = 30$ and 7, respectively. Data sources: pilot-scale studies at Santee, California, Gersberg et al. (1983, 1986); Sydney; Bavor et al. (1987) and Melbourne, Australia, Cottingham et al. (1999); Hamilton, NZ, Van Oostrom and Cooper (1990) and Tanner et al. (1995a); and experimental mesocosm studies reported by DeBusk et al. (1990), Burgoon et al. (1991a), Tanner et al. (1999) and Soto et al. (1999)
**Disinfection**

Although comparative data for faecal bacterial indicator removal in planted and unplanted beds is more limited, mean areal removal rates show a very consistent linear relationship to areal loading over a broad range of application rates ($10^7$–$10^{13}$ m$^{-2}$ d$^{-1}$; data from comparative studies listed for Figures 1 and Gersberg et al., 1987; $n=6$, includes Santee CA data for total, not faecal, coliforms). When comparative data for planted and unplanted systems are plotted together the data points and linear regression lines ($R^2 >0.999$) almost completely obscure each other. However, in side by side comparisons the planted beds commonly show small but consistent improvements in inactivation rates of faecal coliform and a range of other bacterial indicators (e.g. Soto et al., 1999). Planted SSF systems have also shown lower effluent concentrations of viruses than unplanted systems (e.g. Gersberg et al., 1987).

The water column in SSF systems is not exposed to sunlight and does not undergo significant diurnal variations in pH and dissolved oxygen, which together are predominant means of disinfection in natural treatment systems such as waste stabilisation ponds. Other mechanisms including settling, adsorption, protozoan grazing, and possibly release of anti-microbial compounds, are believed to account for the pathogen attenuation observed in SSF treatment wetlands. Decamp et al. (1999) found different ciliate community composition in planted and unplanted SSF wetlands, with greater abundance and grazing rates of fluorescently labelled *E. coli* in planted systems. They considered that the more oxidised conditions in the plant rhizosphere provided a more favourable habitat for ciliate protozoa.

**Nutrient reduction**

Comparison of total nitrogen (TN) removal performance for planted and unplanted systems is shown in Figure 2. Here, despite considerably more data scatter, the planted wetlands show a clear trend of improved TN removal. Some of the systems showed markedly poorer overall performance than others. These included systems receiving highly nitrified, low BOD wastewaters (presumably organic C limited, Gersberg et al., 1983) and others receiving ammonia-rich meat-processing wastewaters containing high levels of COD and sulphur (presumably oxygen limited, Van Oostrom and Cooper, 1990).

Planted wetlands have also shown enhanced P and metal mass removal compared to unplanted controls (e.g. DeBusk et al., 1990; Dunbabin et al., 1988; Soto et al., 1999; Tanner et al., 1995b). Unless specialised P sorbing media are employed, the primary long-term mechanism for wetland P removal is accumulation in accreting sediments (Kadlec and Knight, 1996). The studies noted above involved relatively immature systems where plant uptake and sediment adsorption pools were still actively filling, and it is unlikely that reported P removal rates would be sustainable (Richardson and Craft, 1993; Tanner et al., 1998c).

**Can plant uptake explain differences in nutrient removal?**

The quantity of nutrients able to be taken up and accumulated in live plant biomass per unit of wetland surface area is finite for a given plant species, nutrient regime and set of environmental conditions. Once live plant storage pools approach this limit, little further net annual uptake is possible (Howard-Williams, 1985). In pilot-scale trials where plant storage pools were still actively filling, Gersberg et al. (1986) estimated potential plant uptake could only account for 12–16% of the N removal recorded in systems planted with bulrushes. This was 5–7 times less than the additional removal recorded for the planted systems (over that of an unplanted system). In higher loaded systems achieving relatively low N removal (see above), Van Oostrom and Cooper (1990) estimated net N uptake by
bulrush over an annual period accounted for 25% of wetland TN removal, representing 66% of the additional removal recorded for the planted systems.

Detailed measurements of seasonal uptake by bulrush during the second growth season in four equivalent SSF systems operated over a range of loading rates (Tanner, 2001) showed mean rates of 0.2–0.3 g N m\(^{-2}\) d\(^{-1}\) and 0.05–0.1 g P m\(^{-2}\) d\(^{-1}\) during late spring. Maximum autumn accumulations (including below-ground and standing dead tissues) of 26–47 g N m\(^{-2}\) and 5.8–12.2 g P m\(^{-2}\) were recorded in late summer of the second growth season, rising to 48–69 g N m\(^{-2}\) and 8.8–13.4 g P m\(^{-2}\) in the third growth season. Seasonal senescence after the second growth season resulted in the net release of 0.1–0.25 g N m\(^{-2}\) d\(^{-1}\) and 0.02–0.06 g P m\(^{-2}\) d\(^{-1}\) from live plant tissues. Over an annual period net storage in live plant tissues thus only accounted for 2–8% of TN removal and 1.9–5.3% of TP removal (Tanner et al., 1995b). Comparative measurements in planted and unplanted systems showed net annual plant uptake was responsible for only 3–19% of the additional TN removal and 3–60% of the additional TP removal recorded for the planted systems. This shows that, even in immature systems where plant nutrient pools are actively building, uptake and storage of N and P in live plant biomass can usually only account for a fraction of the improved performance of planted systems. This suggests plants primarily facilitate improved nutrient removal indirectly through their effects on other removal processes.

**Ecosystem engineering by plants**

**Nutrient spiralling**

Although, on an annual basis, little net accumulation of nutrients in plant tissues may occur in mature wetlands, this does not mean nothing has happened. Depending on the phenology of the species and the climate, senescence of aboveground tissues may occur on a seasonal basis or be spread more gradually over the year. Nutrients taken up by plants in dissolved inorganic forms are thus returned sometime later in complex organic forms. This means uptake and return of nutrients is often separated in time and occurs on different temporal scales. In temperate climates, this generally provides a period of rapid plant uptake in early spring when rising temperatures stimulate mineralisation of organic matter accumulated in the wetland over the previous winter. Although some of the nutrients assimilated in active

![Figure 2](image_url)  
**Figure 2** Comparison of mass loading and removal rates of TN for planted and unplanted wetlands. Lines shown are power fits ($R^2 = 0.34$ and 0.39). Data sources listed in Table 1; unplanted $n = 24$, planted $n = 35$. Data sources as for Figure 1, plus Burgoon et al. (1991b), Rogers et al. (1991), Tanner et al. (1995b)
growth phases are internally recycled during senescence by translocation to below-ground storage tissues (rhizomes, corms and bulbs), the magnitude of this is likely to be reduced in the eutrophic conditions of wastewater treatment wetlands (Shaver and Melillo, 1984).

Senescing aerial shoots of tall emergent species such as *Phragmites*, *Schoenoplectus* and *Typha* are not generally abscised. Dead shoots thus remain standing for a period of months during which considerable leaching of mobile nutrients (e.g. K) and in situ decomposition occurs. By the time the dead shoots fall much of the readily available carbon compounds will have already been consumed, so that mainly slowly degradable and recalcitrant litter fractions are returned to the wetland surface. Interactions with invertebrate shredders, grazers and burrowing worms are likely to be important in the decomposition of this fallen litter and its incorporation in the substratum.

After 5 years operation, Tanner *et al.* (1998a) found substantial accumulation of organic matter (6.8–15 kg m\(^{-2}\)) in SSF wetlands treating farm dairy wastewaters. Organic matter accumulated particularly on the surface and in the top 100 mm of the gravel media. Comparison with unplanted controls after ~2 y operation showed 1.6 to 6-fold higher organic matter accumulation in the presence of plants (Tanner and Sukias, 1995). Molar C:N ratios in the accumulated organic matter were also higher in the planted (mean 15.6) than unplanted (mean 11.0) wetlands. Organic matter accumulating in the sediments and as fallen litter in these systems represented a significant pool of both N (~ twice that present in live and standing dead plant tissues at maximum seasonal biomass, Tanner, 2001) and P (~ 6-fold higher than present in live and standing dead plant tissues at maximum seasonal biomass, Tanner *et al.*, 1998c). Organic matter accumulated from wastewaters and plant turnover provides additional sorption sites, sources of complexing and biochemically active substances (humic and fulvic acids), and substrates for microbial processes (e.g. denitrification). This intensifies nutrient spiralling (repeated cycling) along treatment wetlands, markedly elevating the residence time of nutrients relative to that of the wastewaters passing through them (Howard-Williams, 1985; Tanner *et al.*, 1998b).

**Root-zone aeration**

Diffusion, and in some cases convective flows, of oxygen down through the internal spaces (aerenchyma) of wetland plants enables root growth and survival in flooded sediments (Armstrong *et al.*, 1990). However, there has been controversy about how much of this oxygen is actually released into the root-zone (Sorrell and Armstrong, 1994). Increased rates of BOD removal and ammonia oxidation from wastewaters and elevated dissolved oxygen concentrations have been recorded in the root-zone of wetland plants (Dunbabin *et al.*, 1988; Reddy *et al.*, 1989a). Higher interstitial redox potentials (indicating more oxidised conditions) have also been reported for planted than unplanted SSF systems in comparative studies (Figure 3, Burgoon *et al.*, 1995; Dunbabin *et al.*, 1988; Tanner *et al.*, 1999). Root oxygen release has been postulated to account for improved rates of ammoniacal N removal in SSF wetland by stimulating nitrification, the rate limiting step (Gersberg *et al.*, 1986; Reddy *et al.*, 1989b), and higher densities and activity of nitrifiers have been recorded in biofilms associated with wetland plant roots and rhizomes than in the gravel media (Williams *et al.*, 1994).

The predicted depth of plant root penetration, and thus potential for oxygen release, has been proposed as a rational basis for determining the appropriate depth of SSF treatment wetlands (e.g. Reed *et al.*, 1995). In common with many other studies, Tanner (1996; 2001) found considerably shallower root penetration (mostly <300 mm) in gravel-bed systems than those reported by Gersberg *et al.* (1986; 300–760 mm depending on species) and commonly cited in guidelines. In mesocosm studies comparing bulrush root growth over a range of wastewater dilutions in the presence of excess nutrients, Tanner (1994a) found
that increasing concentration of BOD rather than nutrients was the primary environmental factor influencing the depth of root penetration. This is consistent with current theories and models of plant aeration (Armstrong et al., 1990; Sorrell et al., 2000) which predict that for a given root morphology, length will be controlled by cumulative radial and longitudinal consumption due to root respiration and leakage from the root. The extent of leakage is dependant both on anatomical properties of the root (e.g. diameter, degree of endodermal and lateral development, Sorrell et al., 2000) and the intensity of rhizospheric oxygen consumption (Sorrell and Armstrong, 1994). Obviously, it will normally be in the plants’ interest to restrict radial oxygen losses from the root as much as possible; generally to areas of active root growth and nutrient assimilation. Further studies are needed to quantify and compare the oxygen release characteristics of different emergent species and their response to environmental conditions in the root-zone of SSF treatment wetlands.

As well as acting as a conduit for gas transport into (O2) and out (e.g. CH4, CO2, N2O, H2S) of the substratum (Sebacher et al., 1985), the internal ventilation systems of emergent wetland plants may regulate the balance between gas production and consumption processes in the sediments. Tanner et al. (1997) found lower methane emissions for planted than unplanted SSF treatment wetlands, postulating that plant oxygen release was suppressing methanogenesis (a strictly anaerobic process) in the gravel media and/or enhancing root-zone methane oxidation.

Synthesis
Table 1 uses the ecosystem engineering paradigm of Jones et al. (1997), to synthesise the concepts discussed above with a range of other roles identified for plants in treatment systems (e.g. Brix, 1994). Some of the ecosystem engineering roles identified probably only have tenuous links to treatment performance. However, this approach does emphasise the importance of the indirect effects of plants on carbon and nutrient cycling, and ultimately the structure and functioning of treatment wetland ecosystems. The fundamental importance of factors such as the decomposition dynamics of plant litter and the efficacy of root-zone aeration, which ultimately determine the rate and sustainability of carbon and nutrient transformation and sequestration, are highlighted.

Conclusions
Data from comparative studies of planted and unplanted SSF systems suggests that, in
In general, wetland plants provide only small improvements in BOD and COD removal, and disinfection performance. However, wetland plants provide measurable enhancement of nutrient removal, mainly by promoting transformations to gaseous forms and sequestration in accumulating organic matter. Unless nutrient loadings are very low, net removal by direct plant uptake is generally a relatively small proportion of total removal. Plants primarily affect treatment performance through ecosystem engineering, enhancing key nutrient transformation processes (e.g. nitrification and denitrification) by root-zone oxygen release and supply of organic matter. Cycling and accumulation of plant-derived organic matter provides a sustained supply of organic C for microbes, sequesters organically bound nutrients, and buffers nutrient release.

<table>
<thead>
<tr>
<th>Action</th>
<th>Examples of ecosystem engineering by wetland plants in SSF treatment wetlands</th>
<th>Creation of resource</th>
<th>Control over resources</th>
<th>Modulation of forces</th>
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</thead>
<tbody>
<tr>
<td>Σ Growth of plant canopy</td>
<td></td>
<td>Σ Shaded, moist &amp; concealed habitat</td>
<td>Σ Nutrient uptake &amp; accumulation</td>
<td>Σ Temperature buffering</td>
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<td></td>
<td>Σ Aerial shoot habitat</td>
<td>Σ Light attenuation</td>
<td>Σ Shelter from wind &amp; rain impact</td>
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<td></td>
<td>Σ Shoot habitat for burrowing invertebrates</td>
<td>Σ Evapotranspirative water loss</td>
<td>Σ Increased humidity &amp; evapotranspirative cooling</td>
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<td>Σ Condut for oxygen &amp; respiratory gas transport</td>
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<td>Σ Senescence of above ground growth</td>
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<td>Σ Standing dead shoot habitat for microbes &amp; burrowing invertebrates</td>
<td>Σ Nutrient retention, immobilisation &amp; release</td>
<td>Σ Shelter from wind &amp; rain impact</td>
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<td></td>
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<td>Σ Rainfall interception &amp; evaporation</td>
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<td>Σ Surface insulation</td>
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<td>Σ Clogging of media – reduced hydraulic conductivity</td>
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<td>Σ Litter fall</td>
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<td>Σ Aerobic/anoxic/ an aerobic surface litter habitats</td>
<td>Σ Nutrient retention, immobilisation &amp; release</td>
<td>Σ Temperature buffering</td>
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<td></td>
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<td>Σ Supply of humus-modifying sediment texture, cation exchange capacity etc.</td>
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<td>Σ Supply of humic substances – buffering of pH, binding/chelation of nutrients, heavy metals &amp; toxins</td>
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<td>Σ Substrate for production of toxic fermentation products e.g. fatty acids</td>
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<td>Σ Occupation of interstitial spaces causing reduced hydraulic conductivity &amp; increased short-circuiting</td>
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<td>Σ Filtering of suspended solids</td>
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<td>Σ Root &amp; rhizome growth</td>
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<td>Σ Additional surface area &amp; attachment sites</td>
<td>Σ Nutrient uptake &amp; accumulation</td>
<td>Σ Binding/armouring of media surface</td>
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<td>Σ Nutrient &amp; metal retention in iron plaques</td>
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<td>Σ Oxygenation controls microbial pathways, decomposition &amp; mineralisation rates, nutrient and toxin transformations (including ammonia), &amp; methane oxidation</td>
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<td>Σ Transpirational draw of water &amp; dissolved constituents to root-zone</td>
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**Table 1** Examples of ecosystem engineering by wetland plants in SSF treatment wetlands
Acknowledgements
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References


