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Floating Treatment Wetlands (FTWs) in Water Treatment: Treatment efficiency and potential benefits of activated carbon.

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Literature Review:

Floating Treatment Wetlands (FTWs) in Water Treatment: Treatment efficiency and potential benefits of activated carbon.

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Executive Summary

Floating Treatment Wetlands (FTWs) have many benefits over Free Water Surface (FWS) wetlands:

1. Plant roots assisting in filtering and settling processes for sediment bound P and metals
2. Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
3. Mild acidification of water due to release of humic acids; and a C input from senescent vegetation, assisting denitrification.
4. They can adjust to varying water levels
5. A higher retention time is possible as they can be made deeper without submerging the vegetation

Percentage removal of nutrients and metals from effluent is around 20-40% higher in FTWs than in conventional FWS ponds. Removal efficiency, particularly of nitrogen, can be further increased with tighter control on the water chemistry (aeration; adding $CaCO_3$; adding a carbon source). 20% coverage of islands is optimal for aerobic basins. 100% cover is optimal for anaerobic basins or aerobic basins where there is artificial aeration. The design the FTW and the control of basin water chemistry is essential for optimising treatment efficiencies. The passive use of activated carbon within layers of floating islands is unlikely to be cost effective.

Introduction

Definition

Floating Treatment Wetlands (FTWs) comprise of wetland basins or cells, on which there are artificial mats containing emergent plants (Figure 1). This is not to be confused with treatment using floating leaved plants such as *Eichhornia crassipes* (Water Hyacinth), *Pistia stratiotes* (water lettuce), *Lemna* spp. (duck weed) or *Azolla* spp. (water fern) e.g. Reddy & Smith (1987); Kivaisi (2001), or where natural floating islands have established. Floating Treatment Wetlands are also referred to as Constructed Floating Wetlands (CFWs) or Floating Mat Constructed Wetlands, but we will use FTW throughout the review. Floating Islands (FIs) will be used to refer only to the islands within the treatment system. 'Effluent' refers to the water being treated at any stage within the wetland and 'inflow' refers to effluent entering the wetland, and 'outflow' as effluent leaving the wetland. Comparison will regularly be made between FTWs and other wetlands. Where 'conventional wetlands' is referred to, this means other treatment wetlands in general. Basins where there is open water but no islands, are known as Free Water Surface (FWS) wetlands.

The core of this review assesses process, performance and design of FTWs and includes a section on the potential for incorporating activated carbon into FTWs.

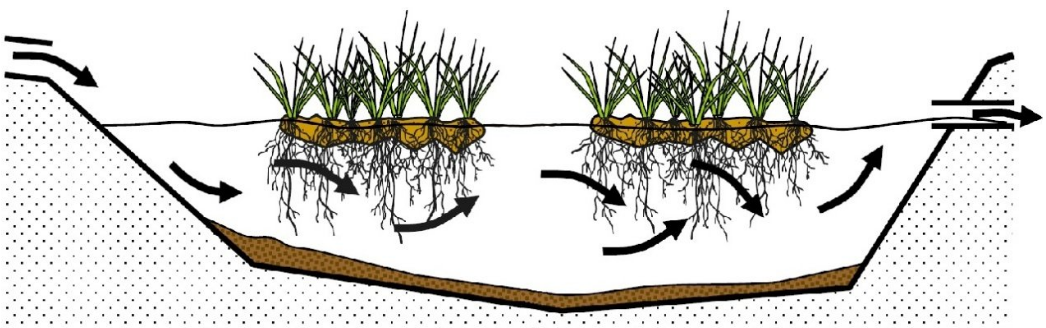

Figure 1. A Floating Treatment Wetland (FTW). Emergent plants are grow within a floating artificially constructed material. The roots are directly in contact with the effluent and can intercept suspended particles. The roots also provide a high surface area for microbiological activity. Image: Headley and Tanner (2006).

1.1 History

Floating islands are a natural occurrence, and can be found where emergent aquatic plants have broken from the land, sometimes developing in highly nutrient rich or sulphurous pools (Duzer, 2004). Floating leaved plants for treatment date back to the 11th Century, when the floating *Azolla* fern was used by Chinese and Vietnamese farmers to extract dissolved nutrients from wetlands and rice paddies, after which it was dried and applied as soil fertiliser (Whitton & Potts, 2002). The use of Water Hyacinth (*Eichhornia crassipes*) to remove nutrients also developed in South East Asia, and both have been used for centuries for water treatment within this region (Whitton & Potts, 2002). *E. crassipes* was suggested for use in the early 20th Century in both Auckland (Australia) and Yorkshire (UK) (Dymond, no date), and then in 1975 NASA used it to treat a sewage lagoon in the USA (Wolverton & Mcdonald, 1978).

Constructed floating islands were first developed in Japan in the 1990s, with *Cana generalis* being grown in floating beds to absorb nutrients from fish ponds and treatment basins (Wu et al., 2000). Twenty percent coverage of soilless artificial floating islands, again using *C. generalis*, was later recommended to improve water quality in China (Bing & Chen, 2001).

Floating mats have also developed unintentionally in many open water treatment systems. Sometimes detrimental effects were observed, such as in Florida, where mats which had grown to 50% coverage moved with the wind across a shallow basin; scraping the bottom and disturbing sediments, resulting in increased outflow turbidity and phosphorus release (Kadlec & Wallace, 2009).

1.2 Range of applications

FTWs can be specifically designed or they can be installed in currently operating open water wetlands i.e. retrofitting. Potentially FTWs can be used with the same waste streams as conventional wetland systems. Some examples from the literature are:

Domestic wastewater treatment

Conventional vegetated wetlands have often been advocated for wastewater polishing, rather than heavy nutrient loads, since they can become clogged and plants are very good at removing low concentrations of nutrients. However, given that there is primary sedimentation, FTWs can potentially deal with larger nutrient loadings since they have higher P and N removal capacity compared to conventional wetland treatment systems. The exposed roots aid sediment deposition, thus reducing turbidity, and there is greater surface area available for microbiologically mediated nitrification/denitrification reactions. Treatment with floating islands has been done on domestic waste in a highly controlled environment i.e. as a hydroponic system (Vaillant et al., 2003).

Metals treatment

Wet detention ponds are used as a Best Management Practice for stormwater run-off in the USA (Chang et al., 2013). FTWs have thus become a popular choice as a retrofit for stormwater run-off treatment in these ponds (Chimney et al., 2006; Headley & Tanner, 2006; Tanner & Headley, 2008, 2011; Hwang & Lepage, 2011; Chang et al., 2012; Borne et al., 2013; White & Cousins, 2013; Winston et al., 2013). FIs are beneficial since they can treat water effectively even with the large fluctuations in water depth that occur during storms.

Strosnider & Nairn (2010) stated that FTWs are ideal for acid mine drainage, particularly if anaerobic conditions are maintained using high island cover. The resulting anaerobic conditions and the decomposing plant material aids denitrification, making the water more alkaline.

Agricultural waste

The enhanced nitrate removal rate of FTWs makes them appealing in reducing pollution from agricultural run-off (Stewart et al., 2008; Yang et al., 2008) as well as for more concentrated wastewaters, such as from swine effluent (Hubbard et al., 2004).

Habitats

FIs have sometimes been constructed specifically to create habitats e.g. to protect birds from land-based predators (Hancock, 2000), including a huge floating island of 3 700 m^2 in Sheepy lake (California) as a habitat for nesting Caspian Terns (Patterson, 2012). Only islands designed for effluent treatment will be covered in this review, however FIs do provide habitats as a secondary function. Emergent grasses can attract waterfowl and terrestrial birds because of the seeds, nesting material, nesting cover and available water. Fish have been introduced into some open water wetlands, however those that feed or nest on the bottom have been found to disturb sediments, increasing suspended solids (Kadlec & Wallace, 2009, p.779).

Although usually not problematic, there have been incidences where large bird communities have contaminated open water treatment wetlands with faeces (Orosz-Coghlan et al., 2006), or have disturbed sediments, increasing turbidity (Knowlton et al., 2002). Geese herbivory can devastate the establishment of wetland plants, especially if planted during the spring or autumn migratory period (Kadlec & Wallace, 2009). However, a benefit of floating islands is that other herbivores e.g. rabbits, cannot usually access the islands. Mosquitoes may also be a problem with an open water system, particularly where monotypic vegetation such as cattail, bulrush and common reed restrict predator access (Knight et al., 2004). However, removing leaf litter, and ensuring that water depth is greater than 40cm (Sinclair et al., 2000) can reduce the problem.

Tourism

Treatment wetlands have been effectively marketed for tourism, especially those which provide good natural habitats for birds (Kadlec & Wallace, 2009). If the FTW is operated for tourism the design and operation is likely to have to include walkways, bird viewing areas and education centres. There may also be conflicting aims for depth regulation between habitat provision and treatment.

2. Processes

FTWs, as with other wetland treatment systems, remove pollutants by four main processes (in order of importance): physical; biogeochemical; microbial and plants. These processes are similar in conventional wetlands, so much of the details provided here comes from that research. However, the larger surface area created by plant roots in FTWs tends to increase sedimentation (by filtering), microbiological decomposition, nitrification and denitrification, and also alter the water chemistry i.e. pH and dissolved oxygen (DO) concentrations. Processes will be discussed relative to the effluent constituents being removed.

2.1 Phosphorous removal

Phosphorous within wetland effluents is usually as dissolved orthophosphate (PO_4^{3-}), or organic phosphorus (Masters, 2012). The scarcity of P in natural environments results in efficient nutrient cycling within ecological systems (Kadlec & Wallace, 2009), thus there are few permanent routes for removal of P within treatment wetlands (Figure 1). The major mechanisms for P removal are accretion in peat/soil and soil adsorption.

Settling and peat accretion

Settling is the main process by which phosphorous bound sediments and BOD are removed from the water column (Kadlec & Wallace, 2009). Settling is a physical process whereby phosphate bound in particles sink to the bottom. Settling is increased in FTWs both by the roots (Masters, 2012) which filter the particles from the water column to later slough off to settle on the bottom, and by reducing currents and circulation caused by surface wind disturbance or water movements (e.g. from pumps) (Headley & Tanner, 2006; Chang et al., 2013). The reduction in movement is essential for preventing resuspension of sediment bound phosphorous into the water column, however, this reduction in currents also contributes to the risk that the basin will become anoxic (Van de Moortel et al., 2010). P retention within different conventional wetlands ranges from 40-60%, around 45 to 75 g/m^2/yr (Vymazal, 2007), most of this being due to settling (and associated processes such as accretion and soil adsorption). P removal from FTWs is usually higher due to the additional filtering properties of the roots, reaching 81% (White & Cousins, 2013).

Soil adsorption

Phosphorus is retained in the soils by binding to the soil surface. Soils with high clay content have high P adsorption capacity, which increases with lower pHs. Organic soils also adsorb P, with the adsorption capacity dependent on mineral components (Rhue & Harris, 1999). Al and Fe fix phosphorus in acidic soils, whilst Ca and Mg fix it in alkaline soils (Kadlec & Knight, 1996). This adsorption process is reversible, with an equilibrium between the bound P and the dissolved P in the soil porewater. The soil minerals and binding sites result in a 'phosphate buffering capacity' which determines where this equilibrium exists (Barrow, 1983). This has important implications for P removal, since reducing inflow P can cause P desorption from the sediments, actually producing a higher P outflow than inflow (Belmont et al., 2009).

Precipitation of P

P adsorption occurs in aerobic waters, but as conditions become anoxic (reducing conditions) metals within the soil change valency, becoming soluble. This causes the release of phosphorus as a co-precipitate (precipitating due to the action of a true precipitate) from the soil (Kadlec & Wallace, 2009). In very low oxygen conditions, where the soils are anaerobic (Eh < -200 mV) sulphate reduction occurs (Figure 4). This creates free sulphide which preferentially binds with Fe (as iron sulphide) preventing iron mineralisation of P. Thus, anaerobic conditions promote the release of P back into the water column (Kadlec & Wallace, 2009).

Plant uptake

Plant uptake of P reaches only around 6% (Masters, 2012). If a FTW has a P removal up to 81% (White & Cousins, 2013), this means around 75% is removed predominantly by settling or storage in other sinks. Much of the P in plant uptake is also difficult to remove permanently from the system by harvesting because it is stored in the roots, or it re-enters the system as litter (see Section 2.8 Harvesting). Vymazal (2007) considers that harvesting of conventional wetlands is only useful in low P effluents (e.g. polishing) with around 10-20 g P/m^2/yr, where uptake is not limited by growth rate. FTWs may be able to absorb more P, due to their roots being suspended directly in the effluent, and plant roots are more accessible for harvesting, but dredging is still likely to be the most effective method of permanent removal.

Microbial and Algal uptake

Bacteria and algae are important in P cycling within the soils, rhizosphere and water column (Vymazal, 2007). P uptake by microbes in conventional wetlands is very fast, but they store very little (Vymazal, 2007). Thus, having higher surface area and consequently higher microbial mass, microbes in a FTW are likely to be a larger sink of P than in conventional treatment wetlands, however nutrient cycling is likely to result in little net removal, except through sedimentation of dead organic microbial matter.

Fish Uptake

In South East Asia it is common to use fish for nutrient recovery in ponds receiving human effluent (Cairncross & Feachem, 1993). Fish eat periphyton (such as algae, cyanobacteria, heterotrophic microbes, and detritus) (Azim et al., 2005) as well as fungi, protozoa, phytoplankton, zooplankton, invertebrates and invertebrate larvae, and some species are piscivorous. In treatment wetlands fish are usually chosen for their adaptation to low oxygen levels, for example *Gambusia affinis* (mosquito fish) in warm temperate to tropical conditions, and *Notrophus fundulus* (black-stripped top minnow) or *Umbra limi* (central mudminnow) in temperate climates with over 77 different fish species being used in North American treatment wetlands (Kadlec & Wallace, 2009). Sometimes *Oreochromis* spp.

(Tilapia) and Bass have colonised previously unpopulated treatment wetlands (Kadlec & Wallace, 2009).

Li & Li (2009) examined nutrient removal from aquaculture effluent using floating islands (17% cover) planted with the aquatic vegetable *Ipomoea aquatic.* There was artificial aeration and it was populated with *Aristichthys nobilis* (silver carp), *Siniperca chuatsi* (mandarin fish; carnivorous) and *Carassius auratus gibelio (*crucian carp). Around 34% of TN and 18% of TP was removed from the system, and of this around a third (34%) of removed TP and TN was removed by fish. This was around the same that was removed by sedimentation.

Kania (2014, unpublished) suggests that FTW facilitate the sustainable growth of fish and demonstrates that FTW significantly increase fish biomass that can be harvested from the waterway. Fish harvesting enables P removal from the effluent with fish being made into meal which can be used for pork or poultry farming or in pet food. There must be no toxins or toxic metal contaminants in the effluent, especially contaminants that may bioaccumulate. Also, if it is to be sold for human consumption the fish need to be cooked well since there is the potential for contamination by pathogens, particularly the tapeworm *Clonorchis sinensis* (Cairncross & Feachem, 1993).

Fish can disturb bottom sediments, releasing P, particularly those that feed or nest on the bottom e.g. *Cyprinus carpio* (Carp) (Kadlec & Wallace, 2009, p.696).

Problems with phosphorous removal

Generally, wetland treatment only produces temporary storage of P, in contrast to N and C which can be released as gases through microbiological degradation (N_2 and CO_2). Indeed, Yousefi and Mohseni-Bandpei (2010) stated that P can be considered as a conserved entity. Most P is stored in sinks such as sediments (95%; Masters, 2010), plants, microbes and algae, but this P is recycled. These sinks give an initial period of apparent P removal. However, once the wetland is established, nutrient cycling results in similar outflow P levels to inflow. Even regular harvesting of plants only removes around 6% (Masters, 2012) of P inflow, if both the roots and shoots are harvested. Thus, Kavanagh & Keller (2007) concluded that at least 90% of P eventually passes through a wetland system and is released in the effluent.

Some wetland treatment systems can even export more P than they receive, such as a stormwater wetland in North Carolina which had median removal efficiencies of – 95% to 70%; at times exporting twice as much P as it was receiving (Line et al., 2008). This can occur both due to physical disturbance of the sediments releasing P, the re-release of P from biodegradation of organics (Sundaravadivel & Vigneswaran, 2001), or anoxia which can also result in the sudden release of P as a co-precipitate (Maine et al., 2005).

Sudden P releases into the water column can potentially have other detrimental effects. Since P is usually the limiting factor for biological activity in freshwaters (Schindler et al., 2008), a large P release can result in nitrogen becoming limiting. This promotes the growth of Cyanobacteria blooms which as well as producing harmful toxins, also extract N from the atmosphere (Conley et al., 2009).

Masters (2012) is thus emphatic that dredging is important for long term removal of phosphorus from a FTW. Kadlec and Wallace (2009) detail projected working life of different types of wetlands with different soils, ranging from around 10 to 170 years, but dredging around every 10 years (Masters, 2010) would be ideal for sustained P removal with most effluents.

A minor route of P removal is phosphine (PH_3). It is usually found in very low amounts (e.g. 47 ng/m^3 of water in marshes), mostly bound to sediments but with around 10% of this dissolved in the water (Hana et al., 2010). However, it can be released from highly anaerobic wetlands (Eh < - 200mV) (Gassmann & Glindemann, 1993) as phosphine gas. Devai and Delaune (1995) calculated a gaseous release rate of 1.7 g P/m^2/yr from a bulrush wetland treatment system.

Thus, treatment wetlands have various sinks (algae, plants, microbes, soils) which vary in their capacity to absorb P from the effluent based on conditions such as available surface area, soil type, pH and redox potential. FTWs limit the resuspension of particulates since the islands reduce water movement within the wetland and the roots filter out particulates (Borne et al., 2013), thus increasing P sedimentation. However, dredging is essential to long term functioning of a FTW for P removal, and regular harvesting can be useful at low P loadings (Figure 1).

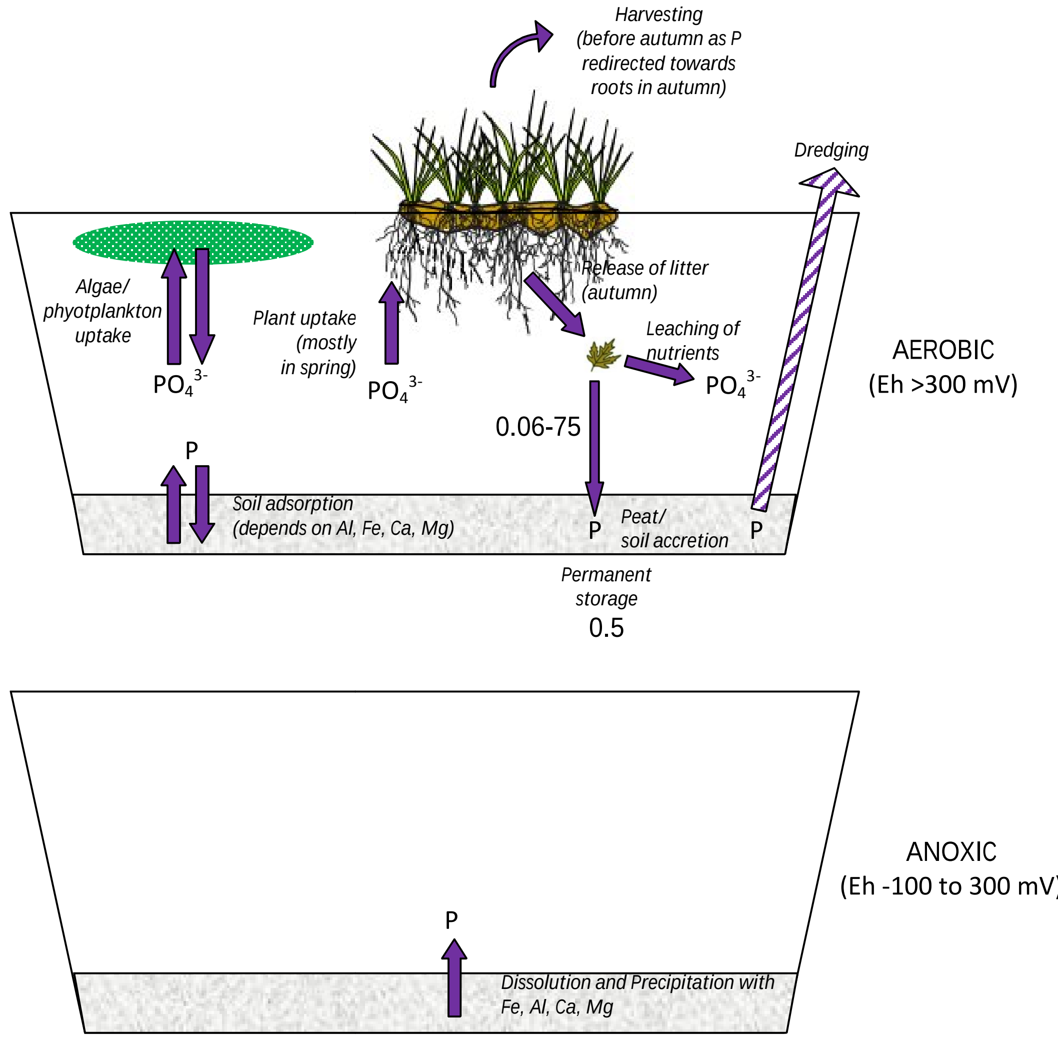

Figure 2. Summary of phosphorus processes in aerobic and anoxic wetlands.
Soil/peat accretion and soil adsorption is the major process and major (95%) sink. However, sorption of P into the soil is reversible. Without harvesting or dredging P removal eventually stops. Numbers in **bold** are g P/m^2/yr that may be removed or added during the processes; *italics* indicate the name of the process, with specific conditions required in brackets.

2.2 Nitrogen removal

Nitrogen is the principal target for treatment in many wetlands. Effluents contain organic nitrogen compounds, which break down principally to ammonia, from which nitrite and nitrate can form through a microbiological nitrification process. Different micro-organisms within anoxic zones can denitrify this nitrate to permanently release N_2 gas from the basin. Agricultural wastes may already have high concentrations of nitrate nitrogen as they enter the wetland. Conversion between different forms of N depends on many factors, including DO, available carbon and pH.

2.2.1 Nitrogen removal in aerobic water

Ammonification (mineralisation)

Dead and decaying organic material is broken down in to ammonia by microbes, either utilising the energy released or absorbing the ammonia for use as microbial biomass. Ammonification increases with temperature, being optimal at 40-60 °C, and with organic compound availability (especially when they have low C/N ratios) (Reddy & Patrick, 1984). Optimum pH is between 6.5 and 8.5 (Vymazal, 2007). Ammonification usually takes place under aerobic conditions (oxidative deamination).

Equation 1: Break down of organic N (example with amino acid) to ammonia

$$RCH(NH_2)COOH + H_20 \rightarrow NH_3 + CO_2$$

Ammonification rates can vary greatly e.g. between 0.004 and 0.53 g $N/m^2/d$ (Reddy & D'Angelo, 1997; Tanner et al., 2002). The root zone of a FTW is likely to be a good location for ammonification.

Ammonia volatilisation

Ammonia exists in an equilibrium between its dissolved ammonium form (NH_4^+) and its gaseous form (NH_3); Equation 2. Below pH 8.0 ammonia loss as gas is negligible (Reddy & Patrick, 1984). At a pH around 9.3 losses due to volatilisation can become significant (Vymazal, 2007). N removal rates due to ammonia volatilisation have been measured at 2.2 g $N/m^2/d$ in wetlands (Stowell et al., 1981).

Equation 2: Conversion of dissolved ammonium to ammonia gas

$$NH_4^+ + OH^- \Leftrightarrow NH_3 + H_2O$$

Algal photosynthesis often elevates pH values during the day (Vymazal, 2007), thus increasing ammonia volatilisation. However, FTWs may inhibit this due to (i) islands shading algae and reducing the area of the air-water interface, and (ii) plants releasing humic acid, which reduces the pH (Van de Moortel et al., 2010).

Nitrification

Within aerobic water micro-organisms convert ammonium to nitrate in a process called nitrification. Directly adjacent to plant roots there is an aerobic zone (Reddy et al., 1989), which means that FTW are likely to have elevated denitrification rates due to the availability of root surface area.

Kadlec & Wallace (2009; p.280) note that nitrification in wetlands is quite different from nitrification in conventional Waste Water Treatment Works (WWTWs). Whilst nitrification is commonly considered a two step process in conventional WWTWs, in natural wetlands it is now believed to have three stages (Bothe et al., 2000); Equation 3.

Equation 3: The three stage nitrification process, converting ammonium to nitrite, then nitrate.

Nitritation (2 stages)

$$NH_3 + O_2 + 2H + 2e^- \xrightarrow{Nitrosomonas} NH_2OH + H_20$$

$$NH_2OH + H_20 \xrightarrow{Nitrosomonas} NO_2^- + 5H + 4e^-$$

Nitrification (1 stage)

$$2NO_2^- + O_2 \xrightarrow{Nitrospira\ or\ Nitrobacter} 2NO_3^-$$

Due to the different processes less oxygen and alkalinity is consumed in wetlands during nitrification than in conventional WWTWs (Kadlec & Wallace, 2009). *Nitrospira* is also much more prominent as a nitrifier than *Nitrobacter* in wetlands (Austin et al., 2003).

Nitrification is influenced by temperature (optimum 25-35 °C), pH (optimum 6.6-8), alkalinity, microbial populations present, DO and ammonium concentrations (Vymazal, 1995). Below 4 °C nitrifying bacteria *Nitrosomonas* and *Nitrobacter* do not grow (Paul & Clark, 1996). Kadlec & Wallace (2009; p.280) note, unlike WWTWs, there is little evidence that a low C/N ratio in wetland effluents improves nitrification rates.

In wetlands, for every g of ammonium oxidised to nitrate 2.28 g of oxygen and 7.1 g of alkalinity as calcium carbonate are consumed (Kadlec & Wallace, 2009; p.279) i.e. nitrification requires aerobic conditions and will consume alkalinity and oxygen, becoming increasingly acidic and anaerobic. Wetlands have nitrification rates of 0.01 to 2.15 g $N/m^2/d$ (mean of 0.048) (Reddy & D'Angelo, 1997; Tanner et al., 2002), though this may be much higher for FTWs due to the large root surface area within the aerobic zone.

Low oxygen conditions can result in nitrite (NO_2^-) being produced instead of completing the process toward nitrate (Bernet et al., 2001). The consequence of this is that in a later denitrification stage, some of the nitrite is converted into nitrous oxide (N_2O), a potent greenhouse gas. Sufficient oxygenation in nitrification basins is therefore recommended.

2.2.1 Nitrogen removal in anoxic water

Denitrification

Denitrification is the microbiologically mediated conversion of nitrate into nitrogen gas, which is then released from the wetland into the atmosphere. A carbon source is required for denitrification. The equation can be written in many ways, depending on the source assumed (Equation 4).

Equation 4: Denitrification of methanol, producing nitrogen gas and alkalinity

$$6NO_3^- + 5CH_3OH \longrightarrow 3N_2 + 5CO_2 + 7H_20 + 6OH^-$$

In many ways denitrification is the converse of nitrification, making the water more alkaline and requiring anoxic or anaerobic conditions. Microorganisms denitrify because in the absence of dissolved oxygen for reduction, they reduce nitrate. Although methanol is used for illustration here as a source of carbon, usually it is large organic molecules. It is calculated that per g of NO_3^- around 3.02 g of organic matter (or 2.3g of BOD) is consumed, and around 3g of alkalinity as $CaCO_3$ is produced (Kadlec & Wallace, 2009).

The optimum pH is 6 to 8 (Paul & Clark, 1996) being negligible below pH4 (Vymazal, 2007). Denitrification is very slow below 5 °C, but increases with temperature up to 60 or 75 °C, then decrease rapidly (Paul & Clark, 1996). More nitrate can speed up the process, but the limiting factor in denitrification is often the carbon supply (Kadlec & Wallace, 2009), especially if BOD has settled out in previous treatment basins. A C/N ratio of 5:1 is suggested to ensure carbon does not become limiting (Baker, 1998) although this may be an overestimate if much of the C is labile (Kadlec & Wallace, 2009). Lower pHs can assist with breaking down lignin in cell walls, increasing the litter quality for denitrification processes (Ding et al., 2012).

Often an anaerobic denitrification basin is placed after an aerobic nitrification basin. This enables all the ammonium to be converted to nitrate prior to denitrification, thus maximising total N removal. However, even in a well oxygenated basin there are areas of low mixing, and deeper waters and sediments, where oxygen levels are low enough to produce denitrification (Figure 3, Figure 4), and in anoxic basins nitrification can occur on the surface of roots where the plants have transported oxygen (Kadlec & Wallace, 2009, p.281). Thus both nitrification and denitrification processes can be achieved within a single basin, though controlling the treatment efficiency may be more difficult.

Floating islands can aid denitrification by producing anoxic conditions through the restriction of oxygen diffusion into the water column. Also, roots and plant litter, as well as coconut coir on islands (Baquerizo et al., 2002),can act as sorption sites, with biofilms developing which increase denitrification rates and thus NO_3 removal rates (Vymazal, 2007). Denitrification releases are about 0.003 to 1.02 g $N/m^2/d$ in wetlands (Vymazal, 2007), though this could be higher in FTWs due to more biofilm area and more sorption sites.

Anaerobic Ammonia Oxidation (ANAMMOX)

The bacteria involved in this process were only discovered in 1999. *Planctomycetes Nitrosomonas eutropha* utilises ammonium ions and nitrite (from nitrification of ammonium) to produce nitrogen gas. This can be represented as in Equation 5.

Equation 5. The ANAMMOX process

Formation of nitrite

$$2NH_4^+ + 3O_2 \longrightarrow 2NO_2^- + 4H+ + 2H_2O$$

ANAMMOX

$$NH_4^+ + NO_2^- \longrightarrow N_2 + 2H_2O$$

This denitrification process uses less than half the oxygen (1.94g O per gram of NH_4^+) of the standard denitrification process, and requires no carbon substrate (Kadlec & Wallace, 2009). ANAMMOX processes occur in many types of wetlands when there is severely restricted oxygen. Bishay & Kadlec (2005) found that in a Free Water Surface wetlands there were more ammonia losses than could be accounted for by the oxygen consumed under normal dentification. There was also a lot of nitrite present in this wetland, and very little carbon, suggesting that these conditions were conducive to the ANAMMOX reaction.

Plant uptake

Nitrogen uptake by plants in conventional wetland treatment is low (up to 6-8%) compared to microbial denitrification (up to 61-63%) (Metheson et al., 2002). Vymazal (2007) estimates that for conventional wetland systems plant harvesting is useful for N removal if loading is only around 100-200 g N/m^2/yr. If N removal is a priority, designing and operating the basins to maximise nitrification/denitrification by microorganisms is probably more cost effective.

N is predominantly taken up by plants in the form of ammonia, but also as nitrate. Much of this is returned to the system when tissues senesce (Kadlec & Wallace, 2009).

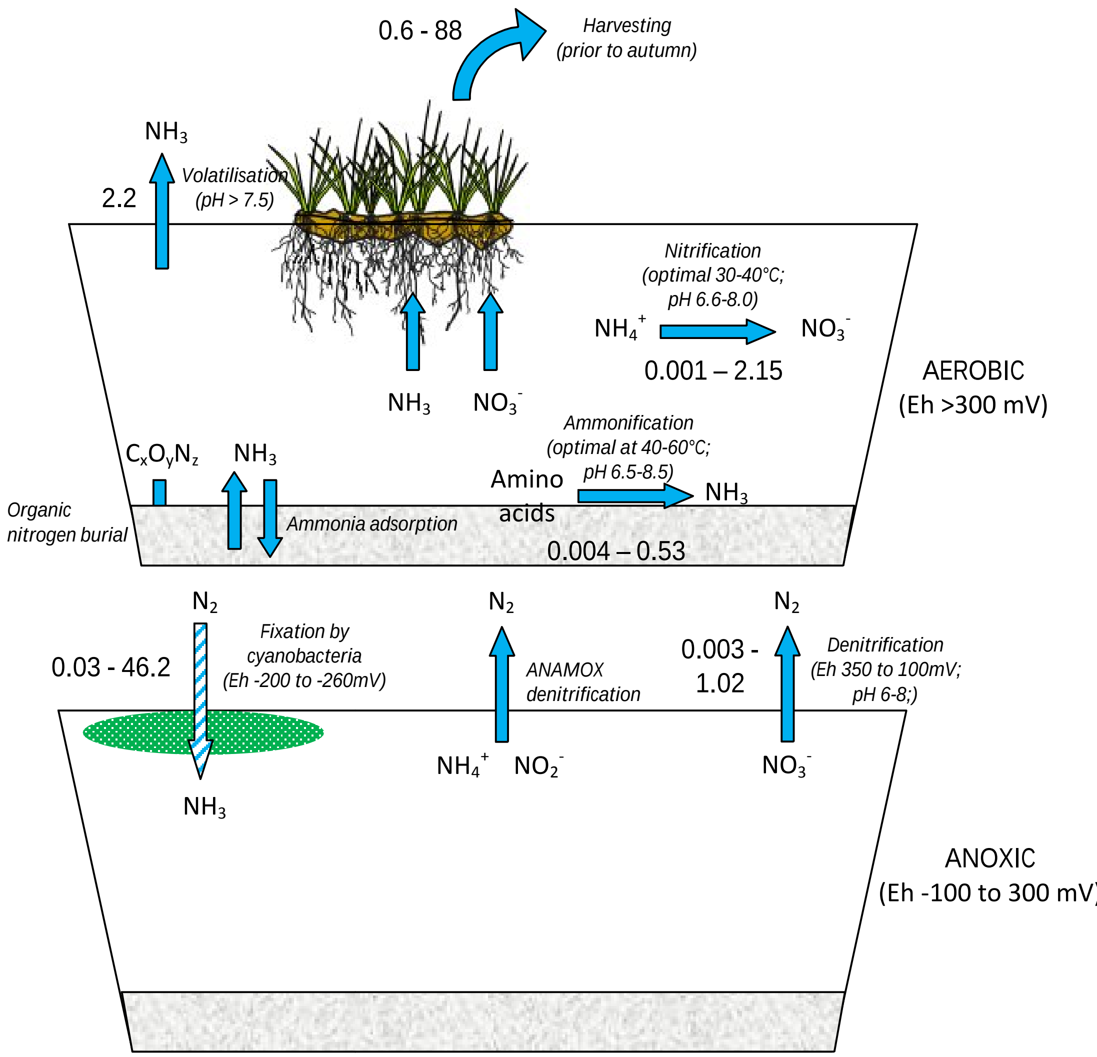

Figure 2. Summary of nitrogen processes in aerobic and anoxic wetlands. Primary settlement of effluent is assumed prior to entering the wetland. Numbers in bold are g N/m^2/yr that may be removed or added during the process; *italics* indicates the process, with specific conditions required in brackets. The striped blue arrow indicates nitrogen fixation that would not normally occur unless the anoxic pond becomes anaerobic (Eh < -200 mV).Permanent removal of N is only through ammonia volatilisation (minor), denitrification (including ANAMOX) and harvesting. Organic nitrogen burial (associated with litter) and ammonia adsorption (associated with clay soils) are relatively minor processes.

Problems with nitrogen removal

NH_4 removal rates in conventional wetlands vary between 35 and 50% in Europe (Verhoeven & Meuleman, 1999; Vymazal, 2002). FTWs have shown removal rates from -45% to 75% for NH_4 and between 36% and 40% for total nitrogen (Boutwell, 2002; DeBusk & Hunt, 2005; Gonzalez et al., 2005).

Problems with nitrogen removal are associated with producing the correct microbiological conditions; aerobic for nitrification and anoxic for denitrification, as well as ensuring sufficient carbon supply for the later. These are discussed in the design section.

2.3 Oxygen

Factors influencing oxygen concentrations

Wetlands typically have slow flow, incomplete mixing, and rapidly decreasing oxygen profiles with depth (Figure 3). Anoxic zones develop just below the substrate in shallower basins and also in the lower regions of the water column in deeper basins (Kadlec & Wallace, 2009). Oxygen can be rapidly depleted in wetlands due to microbiological activity, particularly with nitrification and decomposition (Kadlec & Knight, 1996). FTWs exacerbate oxygen depletion both due to high rates of microbiological activity (nitrification) and due to the islands restricting diffusion of oxygen back in to the water i.e. reducing air-water contact area and reducing wind disturbance (Van de Moortel et al., 2010). This makes FTWs particularly susceptible to unwanted drops in DO, especially at high percentage cover of islands.

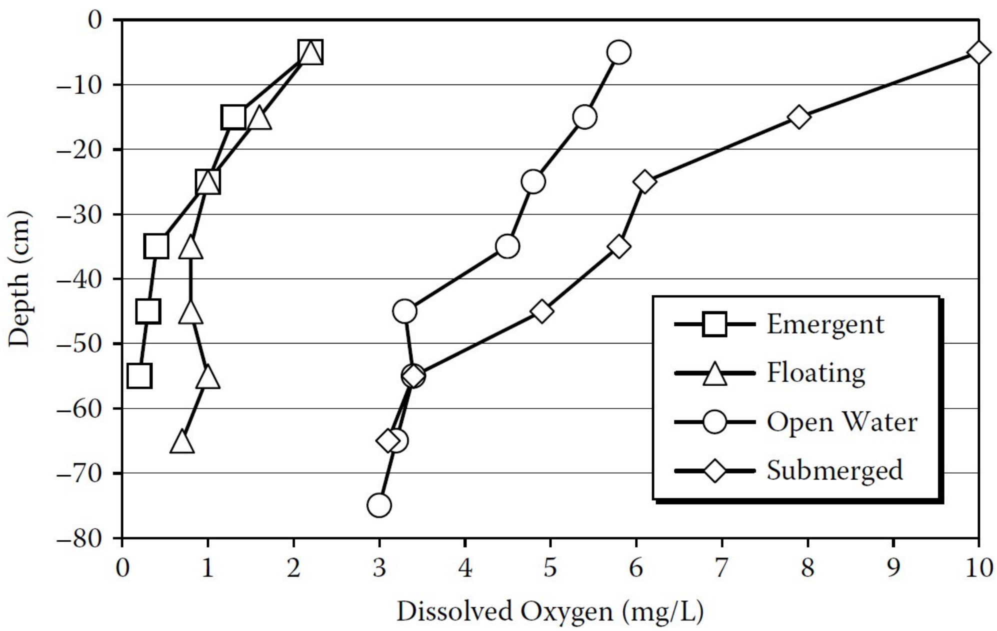

Figure 3. Vertical profiles of dissolved oxygen in various types of FWS (Free Water Surface) wetlands, Florida. Data from 141 profiles collected over a 2½ year period. Data from Chimney et al. (2006), Figure from Kadlec & Wallace (2009). FTWs are most readily compared to floating plant systems.

Temperature affects

Oxygen saturation of water varies with temperature: at 25 °C dissolved oxygen is 8.2 mg/l, and at 5 °C it is 12.8 mg/l. However Kadlec and Wallace (2009) note that the poor mixing of waters limits the dissolution of oxygen such that reaeration is very slow, even in open wetlands. They estimate that it takes 2 to 4 days to reaerate an open wetland basin from 0 to 90% DO, with typical winds. This is likely to be even slower in FTWs.

Plants

Submerged photosynthesising plants and algae release in the range of 0.26 and 0.96 g/m^2/d of O_2 during photosynthesis (Kadlec & Wallace, 2009, p138), oxygenating the water. Emergent plants bring O_2 to the roots, but O_2 delivery usually matches respiration requirements, so there is little net input into the water column (Brix & Schierup, 1990). Studies by Tanner and Headley (2011) and White and Cousins (2013) both found a high level of oxygen depletion in basins due to floating islands.

Tanner and Headley (2011) not only illustrated how oxygen depletion is higher in FTWs, but also that oxygen depletion is higher when there are plants rather than mats with artificial roots (sisal) (Table 1). This oxygen depletion is likely due to the higher rate of microbiological activity associated with plant roots. Although the relationship between oxygen depletion and root biomass was weak, there was little oxygen depletion due to the floating mat alone and even the mat with artificial roots.

Table 1. Oxygen depletion (%DO) at subsurface and bottom of mesocosms after 7 days due to the effect of Floating Islands, ordered from highest to lowest. Influent DO was 95%, floating island coverage was 50%. Root biomass (dry weight) also shown. Adapted from Tanner and Headley (2011).

	subsurface DO (%)	bottom DO (%)	Root biomass (g/m^2)
Control (no floating mat, but equivalent shading)	87	85	
Floating mat only	85	84	
Mat + soil + artificial roots	85	84	
Mat + soil media	80	79	
Mat + soil + Juncus edgariae	68	66	299
Mat + soil + Schoenoplectus tabernaemontani	68	67	184
Mat + soil + Carex virgata	58	57	533
Mat + soil + Cyperus ustulatus	50	48	329

Van de Moortel et al. (2010) found redox potentials to be decreased due to floating islands: at both 5cm and 60cm depths the FTW has much lower O_2 than an open water basin: at 5cm redox is 68 (open water) cf. -25 (FTW); at 60cm redox is: -93 (open water) cf. -122 (FTW). They did claim that roots can aerate island matting. However, there was little difference between the mat redox potential (72 mV ± 478) and the redox potential 5cm below the surface of an open water basin at (68 mV ±225).

A liability with FTWs is that during summer periods, due to high rates of microbiological activity and insufficient O_2 exchange with the atmosphere, the basin can become anaerobic, causing sulphide toxicity which then kills the plant roots (Lamers et al., 2002) and consequently reducing the effectiveness of treatment. Reduction in treatment efficiency due to anoxia was found in several studies, but usually when the floating islands occupied 50% or more of the surface water area (Van de Moortel et al., 2010; Borne et al., 2013).

2.4 Redox potential

Oxidation is the loss of electrons during a reaction. This is usually through a substance combining with oxygen, as it is energetically the most favourable oxidant. Redox potential is the tendency of a system to oxidise substances i.e. in high redox potential water, incoming organic substances will be rapidly oxidised (an oxidising environment) whereas in low redox potential waters substances will be reduced (a reducing environment). An example of reduction would be where hydrogen combines with carbon to produce methane.

Redox potential is strongly associated with the oxygenation of the water, but it is not identical, since substances other than O_2 can oxidise. Zonation usually occurs in a wetland with oxygen being the oxidiser near the surface, then as DO decreases other substances become oxidisers, with reactions releasing less energy with successively weaker oxidisers. This is in the order O_2, NO_3^-, MnO_2, FeOOH, SO_4^{2-} then CO_2.

The decline in free oxygen reflects the redox potential (Eh), also known as the oxidation-reduction potential (ORP), of the water i.e. the tendency of a chemical to acquire electrons, measured as electric potential (mV). At Eh > 300mV (measured with a platinum electrode) conditions are considered aerobic, at < -100 mV conditions are anaerobic, and between these (near-zero Dissolved Oxygen) conditions are anoxic (Figure 4).

Redox Potential	Reactions	Zone	
> +300 mV	Oxygen reduction	I	Aerobic

+ 100 to +300 mV	NO_3^- and Mn_4^+ reduction	II	Anoxic
+100 to – 100 mV	Fe_3^+ and Mn_3^+ reduction	III	Anoxic
-100 to -200 mV	SO_4^{2-} reduction	IV	Anaerobic
< -200 mV	CH_4 formation	V	Anaerobic

Figure 4. Redox zonation in wetlands, based on Kadlec & Wallace (2009). This vertical zonation can be found in deep lentic environments, particularly where there is high oxygen consumption e.g. my microorganisms.

At high redox potential phosphorus can form insoluble complexes with oxidised iron, calcium and aluminium. Organic compounds which comprise most of the BOD are oxidised using oxygen by bacteria, releasing carbon dioxide. At lower redox potentials organic material does not decay quickly. The water is anoxic, with reducing conditions predominating. Manganese and iron are both reduced (Equation 6)

Equation 6. Reduction of manganese and iron in anaerobic conditions

$$Mn^{4+} + 2e^{-} \rightarrow Mn^{2+} \qquad\qquad Fe^{3+} + e^{-} \rightarrow Fe^{2+}$$

This reduction causes metals to precipitate out of the sediments back into the water column, bringing P with them, as a co-precipitate (Van de Moortel et al., 2010).

Further decreases in oxygen (below -100mV) result in anaerobic conditions, whereby sulphate is reduced to hydrogen sulphide, which although soluble, can be released as gas at low pH (Kadlec & Wallace, 2009). Usually this reduction is undesirable in wetlands, except in acid mine treatment.

Equation 7. Reduction of sulphate to hydrogen sulphide

$$SO_2^{-4} + 2CH_2O \rightarrow H_2S + SHCO_3^{-}$$

Eventually, at very low redox potential (below -200mV) CO_2 , formate, or acetate, is reduced to methane (CH_4) by bacteria.

Equation 8. Reduction of carbon dioxide to methane.

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_20$$

2.5 BOD, Suspended Solids and Carbon

Biological Oxygen Demand

Biological Oxygen Demand (BOD) is a measure of oxygen consumption by microorganisms due to the oxidation of organic matter; usually measured in the lab over 5 days (BOD_5). BOD of inflows are typically high, unless the treatment basin is being used just for polishing previously treated wastes. BOD decreases rapidly (around 50% decrease within 6 hours) as it passes through a wetland due to decomposition and settling of organic carbon, finally reaching a non-zero plateau (Kadlec & Wallace, 2009). Even if the waters are not aerobic, fermentation and sulphate reduction can remove carbon from the system.

Carbon

Most carbon entering a wetland is organic. Microbiological processes are the main method for removing carbon, through the oxidation of organic compounds, releasing energy. In aerobic waters, respiration takes place (Equation 9), releasing CO_2 to the atmosphere. In anaerobic zones there are four main processes which can take place: (i) fermentation producing either lactic acid or ethanol (ii) methanogenesis producing gaseous methane (iii) sulphate (SO_4^{2-}) reduction producing carbon dioxide and hydrogen sulphide, and (iv) denitrification, producing carbon dioxide and gaseous nitrogen.

Settling is also an important removal method (although the carbon is retained in the sediments). In FTWs plants have been shown to remove around 5.9 g BOD/m^2/day. The large surface area provided by roots can produce a higher rate of microbial decomposition (Brisson & Chazarenc, 2009), but roots also physically entrap particulates onto the biofilm which then fall in clumps and settle out, providing a significant removal pathway for suspended solids (Smith & Kalin, 2000; Headley & Tanner, 2006; Van de Moortel et al., 2010; Borne et al., 2013). Settling is further encouraged by flow resistance through the roots and flow reduction caused by wind shielding of the surface. Particulate carbon, and carbon bound in litter, if it is not decomposed, accumulates in the sediments, particularly where conditions are anaerobic (Kadlec & Wallace, 2009).

Equation 9. Microbiological decomposition of organic compounds.

Respiration

$$C_6H_{12}O_6 \rightarrow CO_2 + H_20$$

Fermentation

$$C_6H_{12}O_6 \rightarrow 2CHCHOHCOOH \quad \text{(lactic acid)}$$
$$C_6H_{12}O_6 \rightarrow 2CH_3CH_2OH + CO_2 \quad \text{(ethanol)}$$

Methanogenesis

$$\text{(acetate)} \quad CH_3COO^- + 4H_2 \rightarrow 2CH_4 + H_20 + OH^-$$

Sulphate reduction

$$\text{(lactate)} \quad 2CH_3CHOHCOO^- + SO_4^{2-} + H^+ \rightarrow 2CO_2 + 2H_2O + HS^- + 2CH_3COO^- \quad \text{(acetate)}$$

Denitrification (see Equation 4)

Unlike submerged plants, which obtain carbon from the water, carbon uptake by emergent plants is from atmospheric CO_2. Plants thus bring carbon into the system through photosynthesis and the deposition of organic matter. However, the net effect of plants in wetlands is to reduce BOD due to plant respiration, increased settling, and increased decomposition processes (Masters, 2012). Also, where there is carbon limitation in anoxic or anaerobic basins, the C provided by the deposition of litter can be important in increasing denitrification rates (see Section 2.2.1).

Settling of BOD is also affected by basin depth, residence time and water movement (Kadlec & Wallace, 2009). Theoretically higher temperatures should increase microbial decomposition rates. Bacteria have limited activity below 5°C, but in conventional wetlands there is no significant temperature dependence above this (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009). This may be due to limitations in oxygen transfer rates or restricting factors in one or more of the many C processes (Kadlec & Wallace, 2009).

In anoxic (reducing) conditions, the presence of sulphate contributes to the removal of organic matter (BOD/COD) by acting as a coagulant and thus increasing settling rates (Huang 2005).

2.6 Metal removal

Metal removal from wetlands is predominantly through forming complexes with organic matter, and through being coated in iron or manganese oxyhydroxides (Kadlec & Wallace, 2009). This either occurs in the sediments, or they settle out into the sediments. Under anoxic conditions Cu, Zn, Pb, Ni and Ca form insoluble metal sulphides which will settle out. Even in aerobic basins, decomposition of organic matter usually means there is an anoxic layer just below the surface oxic layer (≈1cm) in which these metal sulphides can form.

Predicting metal removal from wetlands can be very difficult, depending on the structure of the sediments and many factors of the water chemistry, with models regularly being wrong by orders of magnitude (Kadlec & Wallace, 2009). Factors that affect metal removal include the Cation Exchange Capacity (CEC) of the sediments, pH (circumneutral usually being optimum), redox potential, the availability of sulphur for the formation of metal sulphides, and the formation of iron and manganese oxyhydroxides (which allow co-precipitation) (Kadlec & Wallace, 2009). Organic soils with humic acids and phenolics increase the CEC and thus adsorption of metals. Sedimentation of metals can result in long term storage, depending on the availability of organics with which metals can complex, although metal accumulation can eventually saturate the soil sink and result in biological toxicity (Kadlec & Wallace, 2009). Thus (careful) dredging is required in the long term to permanently remove metals and ensure the wetland continues to operate effectively.

Uptake by plants is much less important than that by sedimentation, and where metals are taken up, they are mostly stored in the roots. Table 2 shows percentage removal of metals by plants in a conventional wetland and how this is allocated in the roots and shoots.

Table 2. Percentage removal of metals by plants in a conventional treatment wetland and how this is allocated to the roots and shoots (adapted from Nolte and Associates, 1998).

Metal	Roots (%)	Shoots (%)	Total (%)
Ag	2.0	0.0	2.0
As	10.1	0.6	10.7
Cd	13.3	0.0	13.3
Cr	16.8	2.2	19.0
Cu	5.5	0.6	6.1
Hg	6.7	0.0	6.7
Ni	4.7	0.3	5.0
Pb	11.8	2.0	13.8
Zn	6.1	0.4	6.5

Despite plant uptake being low, FTWs have been shown to greatly increase metal removal compared to unvegetated retention ponds. For example Borne et al. (2013) compared treatment in a normal stormwater retention pond with one retrofitted with a floating island. With concentrations of 0.0092 mg Cu/l and 0.035 mg Zn/l in the inflow, particulate Cu and Zn removal was 19% and 40% (respectively) in the normal pond, and 50% and 65% with a floating island. Tanner & Headley (2011) found the removal of dissolved Cu and Zn to be 5% and 1% without a floating island, and 50% and 47% with an island. These authors believe that the benefit of the floating island wasn't principally due to plant uptake. Indeed, Tanner & Headley (2011) found mean plant uptake rates were 0.059-0.114 mg Cu/m^2/d and 1.2-3.3 mg Zn/m^2/d, accounting for less than 4% of Cu removal and less than 10% of Zn removal. This was a mesocosm experiment without bottom sediments and with predominantly dissolved metals, so values of plant uptake were probably higher than they would be in a normal FTW.

Tanner & Headley (2011) and Borne et al. (2013) considered that the improved performance with floating islands was due mainly to: (i) interception by the plant roots, (ii) humic acid release from the plants, which reduced alkaline waters to circumneutral pH (Van de Moortel et al., 2010), improving metal complexation and therefore flocculation and settling (Mucha et al., 2008) and (iii) The islands reducing the redox potential to the extent that insoluble metal sulphides formed.

The exact mechanisms of metal removal depend on the specific metal. Most zinc within effluents is in particulate form and is removed predominantly through settling, sorption to organic sediments and chemical precipitation/co-precipitation (Kadlec & Wallace, 2009). It can form precipitates with sulphur (ZnS) and carbonate from the water ($ZnCO_3$) and it co-precipitates with Fe, Mn, Al oxyhydroxides. However, ZnS does not readily precipitate in neutral waters (Younger, 2000), only in more alkaline waters (>7.5). Also, for co-precipiration, the other metals must be present in the effluent, and even then, Fe and Mn oxides are not stable in anoxic waters (Knox et al., 2004). Warmer water temperatures are also correlated with Zn removal, probably due to increased sorption rates (Borne et al., 2013). Aerobic wetlands are expected to absorb about 0.04g $Zn/m^2/d$ (PIRAMID consortium, 2003). Similar to Zn, Cu removal rates increase with temperature, however adsorption is better at more neutral pH (Borne et al., 2013). They also concluded that reduced oxygen resulted in a high production of Cu sulphide precipitates in basins with floating islands.

High loadings of effluent and insufficient adsorption capacity or saturation of the potential sinks (organic carbon, metal hydroxides, high CEC soils) can result in decreasing treatment capacity as well as increasing toxicity. Toxicity can be a biological problem, particularly in open water treatment systems where birds, amphibians and freshwater invertebrates have direct access to the basin (as opposed to subsurface flow systems) (Kadlec & Wallace, 2009). Sorption capacity in studies listed by Kadlec and Wallace estimate between 20 and 780 years operation of a wetland with metal loading. Careful dredging (avoiding resuspension) can be applied to remove contaminated sludges/soils. In mixed wastewater effluents from WWTWs it is likely that the necessity for P removal through regular dredging is higher than that from metal accumulation.

2.7 pH

pH has a profound effect on the functioning of wetlands, as mentioned in previous sections. Several studies have confirmed the effect of floating vegetated islands in reducing pH. In a two year study by White and Cousins (2013) pH decreased from 8.6 to 6.2. After only 11 days Van de Moortel et al. (2010) found a significant pH decrease from 7.5 to 7.0 whilst the control (without an island) stayed constant at around 7.5. Borne et al. (2013) found a difference between the control (8.3) and the FTW (7.3), which aided Cu adsorption. Interestingly Tanner and Headley (2011) didn't notice a drop in pH in mesocosm tanks, although they still found that treatment was enhanced with floating islands, attributing the difference in to the release of bioactive compounds. The researchers who found differences in pH generally agreed that humic compounds were released by the plants, reducing pH. White and Cousins also acknowledged that alkalinity consumed during microbial nitrification on the plant roots could also be a driving force behind dropping pH within aerobic basins.

2.8 Harvesting of Floating Island Plants

FTWs are a relatively new technology with few long term studies, and few details on plant harvesting. The prime functions of plants in FTWs is (i) for their roots to intercept and filter particulates, aiding sedimentation, (ii) to increase the rates of microbiological processes by providing a high surface area on which microorganisms respire, nitrify or denitrify, and (iii) to alter the physic-chemical and chemical environment i.e. increase microbiological processing through the release of humic acids and through reducing DO exchange (acidity and lower oxygen increasing denitrification) and carbon deposition (increasing denitrification). Harvesting is therefore not essential to long term management of FTWs, and although it can help with permanent removal of nutrients and metals, removal rates are typically low. For example, in subsurface flow wetlands plants only removed 2-8% of total nitrogen (Tanner, 2001; Yousefi & Mohseni-Bandpei, 2010) and 3-12% of total phosphorous (Yousefi &

Mohseni-Bandpei, 2010), with microbes believed to be removing the rest of the N, and settling removing the rest of the P. Even with total uptake for N and P estimated at around 6%, all of this is unlikely to be harvested as it is stored in both the roots and shoots, and nutrients are returned back to the wetland through deposition of senescent material.

Practicalities of harvesting

Floating islands facilitate easy harvesting. Often larger islands are able to support the weight of humans, and so cutting could be done directly on the island. Smaller islands can be pulled towards the shore and even lifted out. In contrast with other wetlands where the vegetation is rooted in the sediments, in FTWs both roots and shoots can be removed, and with little disturbance to the sediments. Theoretically a replacement island could be installed immediately, although this may not be cost effective. Also, removal of root mass is likely to be more detrimental to treatment than the gains from permanent removal of the nutrients. For example, FTWs typically increase N and P removal rates by around 20-40% (Table 14), whereas P and N removal by harvesting the whole plant is at the most 6%.

Storage of nutrients in plants

The start of the growing season, in early spring and prior to maximum growth rate, is the time of highest P uptake. However, prior to autumn senescence, much of the P is relocated to the root stock for the following year (Vymazal, 2007). Thus, if removal of P is a priority, harvest timing and frequency is extremely important, with a recommendation that it is done not only prior to senescence, but also during the peak growth period. The P lost in the senescent material re-enters the basin system very rapidly; up to 30% lost through leaching within the first few days of decomposition (Vymazal, 2007).

Although shoot biomass tends to be larger than root biomass (see plants in design section), there is generally more N, P and K stored in the roots than in the shoots, especially when autumn approaches (White & Cousins, 2013; Winston et al., 2013) e.g. Figure 4.

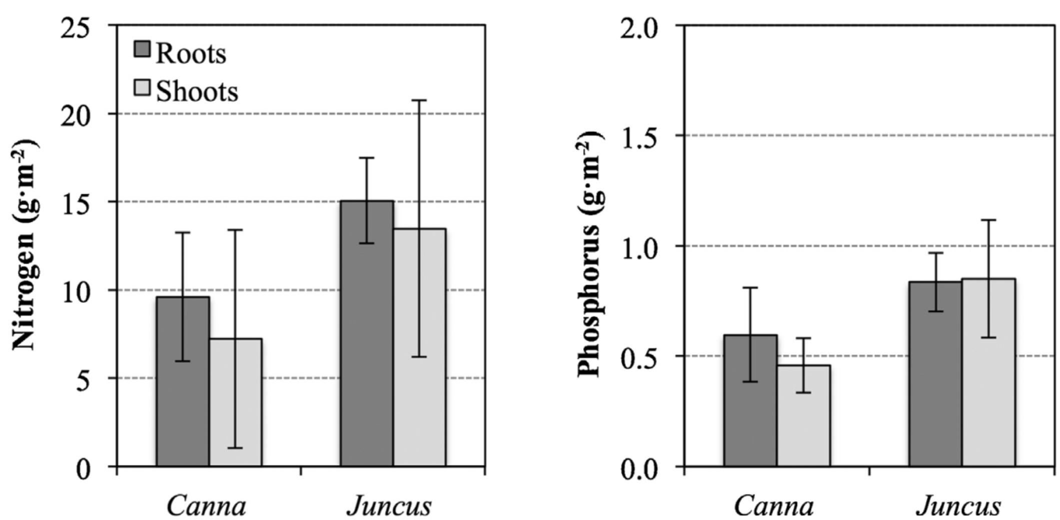

Figure 4. Nitrogen and phosphorus in roots and shoots of *Canna flaccida* and *Juncus effusus* after one summer of growth (harvested 18 September 2008). Nutrients are per m^2 of floating island. Three replicates per bar, with standard error indicated. From White & Cousins (2013).

Storage of metals in plants

Storage of metals tends to show either an even distribution between roots and shoots (e.g. Cu) or predominant storage in the roots (e.g. Zn) (Tanner & Headley, 2011). Table 3 shows uptake of copper and zinc in roots and shoots over a 7 day trial.

Table 3. Uptake of copper and zinc in roots and shoots of four different plant species over 7 days in a FTW, measured as μg/m^2/d. Adapted from: Tanner and Headley (2011).

	Cu		Zn	
Plant species	roots	shoots	roots	shoots
Cyprus ustulatus	54	61	3027	282
Carex virgata	54	89	1228	934
Juncus edgariae	38	41	1703	760
Schoenoplectus tabernaemontani	36	24	881	320

Relative importance of different processes

Restricting flow and intercepting particulates on roots is one of the prime benefits of FTWs, consistently removing more BOD and P than open water treatment ponds. However, using synthetic root structure (sisal) Borne et al. (2013) showed that the physical structure alone does not account for most of the benefits of FTWs; water chemistry changes, and to a much lesser extent plant uptake, assist with improving treatment.

3. Treatment Efficiency

Treatment efficiency obtained within a FTW is highly dependent on appropriate design and proper operation, as well as the characteristics of the inflow and the objectives of the treatment. At one end of the scale are FTWs designed for aerobic treatment with mixing or air bubbled into the system, often with low % island coverage and addition of calcium carbonate to aid nitrification reactions. These basins are predominantly to remove ammonium. At the other end of the scale are anaerobic basins with up to 100% island coverage, with addition of carbon in the form of e.g. molasses, to supply the denitrification process. Thus, in aerobic basins, ammonium removal may be high whereas nitrate is produced and may exceed inflow nitrate concentrations. In the latter, denitrification reactions remove nitrate, but ammonium may not be nitrified, resulting in NH_4^+ increasing (due to organic carbon decomposition) such that outflow exceeds inflow. Sometimes floating islands achieve very high rates of removal because of a tightly controlled DO, pH and carbon supply in a hydroponic system. The concentrations of pollutants also affects the removal rate, with higher inflow concentrations often resulting in higher removal rates.

There can be different flow regimes, such as plug flow, where a quantity of effluent is kept in the basin for around 3-7 days, continuous flow, or sporadic flow (such as storm events). Some mesocosm and lab based studies use synthetic effluent with dissolved nutrients, which may exaggerate treatment efficiencies, especially for P and metals which are usually bound to particulates.

Thus, the main considerations when examining performance of a FTW are:

1. Dissolved oxygen: aerobic/anoxic/anaerobic. Natural aeration or artificial aeration through bubblers. With aerobic basins tending to towards nitrification and anaerobic basins tending towards denitrification.
2. Carbon sources: either naturally, through organic carbon, or added artificially, to enhance denitrification rates. Decomposition of organic carbon also results in increased ammonia production within the basin.
3. pH: with alkaline pH increasing nitrification and acidic pH increasing denitrification.
4. Root mass: aiding removal of particulates due to physical filtering and settling processes
5. Mixing: circulation of water to aid the nutrient supply to microbiological processes.
6. Plug flow or continuous flow: affecting residence times and nutrient gradients.
7. Concentrations of inflow pollutants: with higher nutrient supply increasing rates of decomposition/nitrification/denitrification unless limited by another factor.
8. Changes in the FTW chemistry with time. Often pH and redox potential drops due to microbiological processes and restriction of oxygen diffusion from the surface.

Thus, direct comparison between different FTWs has little meaning, and the best comparison is with a relevant control basin. This is often a basin without an island which is receiving the same effluent, however sometimes it is before and after the retrofitting of an island, which doesn't guarantee exactly the same effluent inputs.

New treatment systems can take over a year to stabilise, and even then they can have high variation in treatment efficiency, especially if environmental conditions vary or sinks (such as sediment adsorption) become saturated. However, significantly higher performance of FTWs can be noticed in as little as two days (Van de Moortel et al., 2010), particularly in relation to nitrification/denitrification and other processes which are predominantly dependent on microorganisms, due to their fast response time (Kadlec & Wallace, 2009).

In this section, treatment efficiency from peer-reviewed FTW studies will be summarised separately, with relevant details supplied, and compared to a control where possible. Where complete columns are blank, there was no information.

Abbreviations follow this system: NO_x represents nitrate in the form or either NO_2 or NO_3; TN is Total Nitrogen; N_{org} is organic nitrogen; Cu_{tot} is total copper; Cu_{part} is particulate copper; Cu_{diss} is dissolved copper; DRP is dissolved reactive phosphorus; BOD is biological oxygen demand; COD is chemical oxygen demand; PBP is Particle Bound Phosphorus; TKN is Total Kjeldahl Nitrogen.

Table 4. Removal rates in the study by Van de Moortel *et al.* (2010). Close to 100% coverage of the island resulted in reduced redox potential and anoxic conditions. This resulted in high NO_3 removal rates, but poor NH_4 (thus TN) and P removal rates.

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l	2.16	1.77	18		1.9	12
TN	mg/l	21.8	13.1	40	*identical*	19.5	11
NH_4	mg/l	16.1	10.8	33	*to*	16.5	-2
NO_3	mg/l	0.37	0.2	46	*control*	0.08	78
N_{org}	mg/l	4.31	1.6	63	*inflow*	2.87	33
TOC	mg/l	27.7	16.4	41		23	17
COD	mg/l	81.3	46.6	43		51.4	37
Cu	mg/l	10.0	5.5	45		8.4	16
Fe	mg/l	454	325	28		259	43
Mn	mg/l	164	153	7		176	-7
Ni	mg/l	10.0	6.1	39		5.75	43
Pb	mg/l	6.10	3.4	44		4.58	25
Zn	mg/l	57.5	29.7	48		47.6	17
SO_4^{2-}	mg/l	64.2	49.8	22		53.7	16
pH	mg/l	7.35	7.08	4		7.48	-2
cond	µS/cm	1035	1017	2		1015	2

Table 5. Removal rates in the study by White & Cousins (2013). Troughs of 1.15 m^2 and 3.03 m^2 were used with 100% island coverage and soluble fertiliser added to pond water as the inflow.

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	$mg/m^2/day$				37.2	15.4	59
TN	$mg/m^2/day$				320	106	67

Table 6. Removal rates in the study by Yang *et al.* (2008). This was a lab based hydroponic study with synthetic effluent (dissolved P and N, but also organic matter used), although the objective was to represent a nursery run-off treatment system, which naturally has few suspended solids. 100% island coverage was used with purposely anaerobic conditions, 3 day batch process, and glucose added to aid denitrification. Thus, high NO_x removal rates were obtained, but NH_3 removal was negative as decomposition of organics was still taking place but with limited or no nitrification.

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l				1.25	1.17	6
TN	mg/l				3.76	2.59	31
NH3	mg/l				0.93	1.19	-28
NO_x	mg/l				1.39	0.12	91
COD	mg/l				41.8	34.8	17
DO	mg/l				0.01	0	100

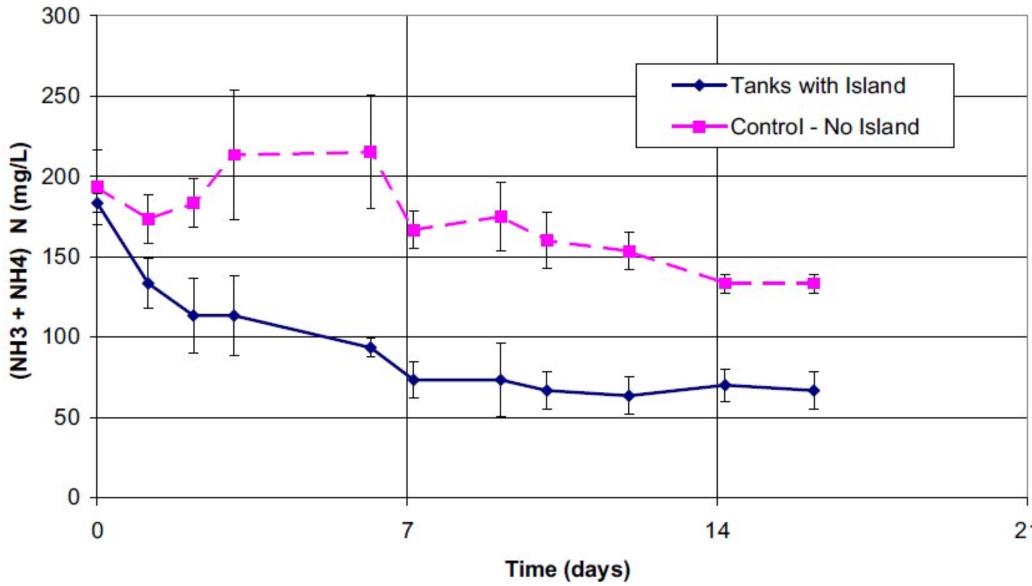

Figure 5. Ammonium removal rates; graph from the study by Stewart *et al.* (2008). An aerobic lab experiment with 100% island coverage, calcium carbonate added, and aerated with a bubbler. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for ammonium removal.

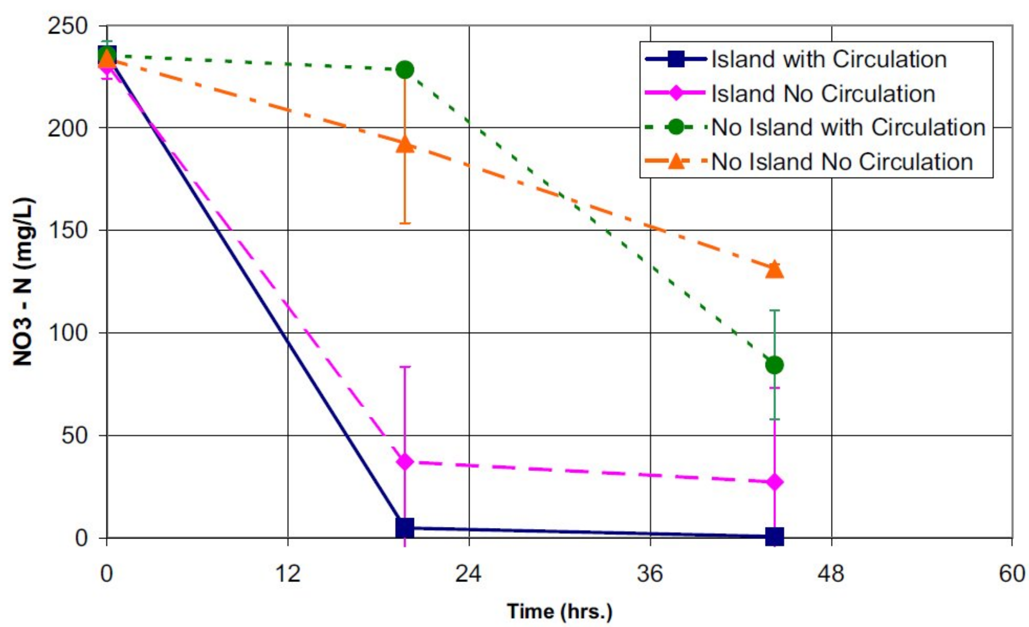

Figure 6. Nitrate removal rates; graph from the study by Stewart *et al.* (2008). An anaerobic lab experiment with 100% island coverage and carbon (molasses) added. In some of the replicates water was circulated by a pump. Synthetic effluent was created using liquid fertiliser (soluble). Conditions were optimised for nitrate removal. Redox potential in the control decreased from +200mV to +48mV, but in tanks with islands it decreased to -200mV (much better for denitrification).

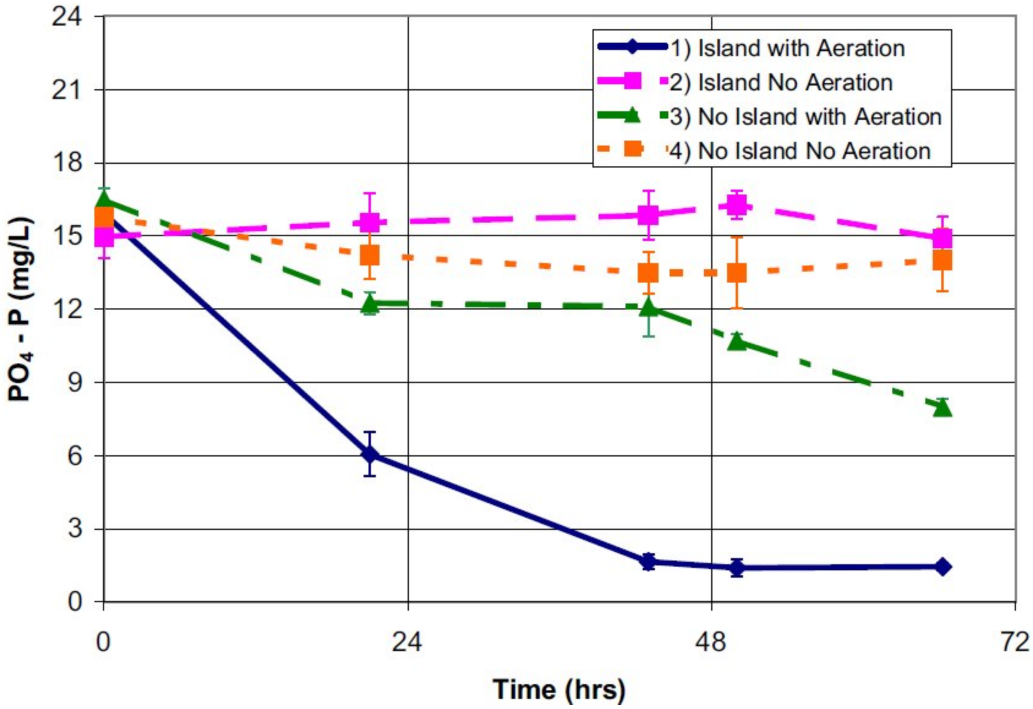

Figure 7. Phosphate removal rates; graph from the study by Stewart *et al.* (2008). Both anaerobic and aerobic basins were tested (conditions as in Figures 5 and 6) without islands and with 100% cover of islands. Phosphate removal was best achieved when there was both aeration and floating islands.

Table 7. Removal rates in the study by Borne et al. (2013). A control stormwater retention pond was compared with a retention pond with 50% cover of floating island, receiving the same effluent. Data was retrieved from a graphical presentation of inflow and outflow effluent concentrations.

	control			FTW		
	inflow	outflow	% removal	inflow	outflow	% removal
TSS	30	24	20	*identical*	12	60
Cu_{tot}	0.0090	0.0075	17	*to*	0.0057	37
Cu_{part}	0.0035	0.0030	14	*control*	0.0019	46
Cu_{diss}	0.0049	0.0044	10	*inflow*	0.0038	22
Zn_{tot}	0.035	0.022	37		0.013	63
Zn_{part}	0.027	0.017	37		0.010	63
Zn_{diss}	0.006	0.005	17		0.005	17

Table 8. Removal rates in the study by Tanner & Headley (2011). After 7 days batch experiment with 1m x 1m mesocosms and 36% island cover. Artificial stormwater used. FTW results are from the plant species which gave best results (*Cyperus ustulatus*).

	control			FTW		
	inflow	outflow	% removal	inflow	outflow	% removal
TP			3			58
DRP			-5			60
Cu_{tot}			7			57
Cu_{diss}			5			50
Zn_{tot}			-1			19
Zn_{diss}			1			37
Turbidity - subsurface			24			67
Turbidity - bottom			24			67
DO (subsurface)			8			39
DO (bottom)			11			40

Table 9. Removal rates in the study by Stefani *et al.* (2011) based on median values. Effluent was from aquaculture, following conventional activated sludge treatment. There was a 19% cover of islands and a continuous flow (0.09 m/s).

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l				0.55	0.19	65
SS	mg/l				350	320	9
COD	mg/l				15	5	67
BOD	mg/l				4.2	2	52
pH					7.3	7.2	1
cond	µS/cm				645	645	0

Table 10. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an 18% coverage of floating island. Data is a mean over different storm events.

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l	0.26	0.11	58	0.41	0.05	88
PBP	mg/l	0.13	0.04	69	0.17	0.03	82
OP	mg/l	0.13	0.07	46	0.24	0.02	92
TN	mg/l	1.01	0.41	59	3.49	0.43	88
NH_3	mg/l	0.10	0.05	50	1.6	0.04	98
TKN	mg/l	0.88	0.35	60	3.32	0.37	89
NO_x	mg/l	0.12	0.06	50	0.17	0.06	65
N_{org}	mg/l	0.89	0.34	62	1.72	0.33	81
TSS	mg/l	216	24	89	252	13	95

Table 11. Removal rates in the study by Winston *et al.* (2013). The study examined a stormwater retention pond before (control) and after (FTW) retrofitting an 9% coverage of floating island. Data is a mean over different storm events.

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l	0.26	0.17	35	0.19	0.12	37
PBP	mg/l	0.13	0.05	62	0.07	0.05	29
OP	mg/l	0.14	0.12	14	0.12	0.07	42
TN	mg/l	1.64	1.05	36	1.17	0.61	48
NH_3	mg/l	0.12	0.11	8	0.11	0.05	55
TKN	mg/l	1.43	0.97	32	0.84	0.55	35
NO_x	mg/l	0.20	0.08	60	0.34	0.06	82
N_{org}	mg/l	1.50	0.93	38	0.72	0.5	31
TSS		354	30	92	101	22	78

Table 12. Removal rates in the study by Chang *et al.* (2013) during storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over several different storm events. The pond contained a fountain.

		Control			FTW		
		inflow	Outflow	% removal	inflow	outflow	% removal
TP	mg/l	0.028	0.027	4	0.058	0.050	14
OP	mg/l	0.006	0.006	0	0.021	0.010	52
TN	mg/l	0.300	0.377	-26	0.626	0.526	16
NH_3	mg/l	0.048	0.052	-8	0.102	0.104	-2
NO_x	mg/l	0.006	0.017	-183	0.062	0.029	53

Table 13. Removal rates in the study by Chang *et al.* (2013) outside of storm events in a functioning stormwater retention pond; assessed before (control) and after (FTW) fitting floating islands with 8.7% cover. Nutrient concentrations are given as the means over different sampling times. Notice that the treatment rates are much higher than during storm events (probably due to lower flows and thus higher retention times).

		control			FTW		
		inflow	outflow	% removal	inflow	outflow	% removal
TP	mg/l	0.037	0.034	8	0.055	0.029	47
OP	mg/l	0.003	0.002	33	0.020	0.004	80
TN	mg/l	0.303	0.349	-15	0.655	0.552	16
NH_3	mg/l	0.121	0.103	15	0.208	0.102	51
NO_x	mg/l	0.025	0.022	12	0.032	0.025	22

Overview

With more cover of floating islands there is a tendency for redox potential to drop due to reduced O_2 diffusion from atmosphere. This leads to denitrification processes dominating in which there is high removal of NO_3 but low removal of NH_4 (Yang et al., 2008; Van de Moortel et al., 2010). Indeed NH_4 removal may be negative due to decomposition of organics to NH_4 without subsequent removal by nitrification (Table 4 and 6). Aeration can prevent this NH_4 accumulation by encouraging nitrification, and it also prevents P release from sediments that can occur at low redox potentials (Figure 7).

Low % island cover had detrimental effects on treatment efficiency, as did lower residence times. For example, TN removal was 48% with 9% island cover in the Winston et al. study (2013), but this increased to 88% TN removal with 18% cover (Tables 10 and 11). Similarly Chang *et al.* (2013) found only 14% TP removal with 9% cover during storms, but outside of storm flows this removal increased to 47% (Tables 12 and 13).

Around 20% cover seems optimal if the basin is to be maintained as an aerobic system without artificial aeration, and still achieve good removal efficiency. Beyond this point it is probably worth using 100% cover, with a choice between a high nitrate removal anaerobic basin, or artificial aeration (bubbling) to produce a high treatment rate aerobic basin. Stewart et al. (2008) illustrates how tightly controlled conditions and addition of calcium carbonate (nitrification) or carbon (denitrification) can be used to optimise treatment rates. Stewart et al. (2008) also showed that nitrification and denitrification processes can be achieved in a single aerobic tank if tightly controlled. Treatment efficiencies noted by Stewart and White & Cousins (2013) are likely to be around the maximum achievable in FTWs due to the use of soluble fertilisers in their experiments and their tightly controlled hydroponic systems. Therefore when assessing potential performance of a new FTW we must decide whether it will be a tightly controlled situation or more of a field based FTW.

Table 14 summarises these studies. The improvements through using Floating Islands, as discussed, vary due to conditions, however we can expect between around 2 and 55% increase in P removal compared to a Free Water Surface wetland, and a 12 to 42% increase in N removal. Metal removal is also considerably higher in FTWs (20-50% higher). Most importantly, if conditions are tailored for denitrification (anaerobic and sufficient carbon supply) NO_3 removal can be up to 100% in FTWs.

Table 14. Summary of % removal rates in different studies for main nutrients and metals. ‘Improvement’ examines the increase of treatment efficiency of FTWs beyond the control wetlands (FST wetlands). The Van de Moortel study is excluded from the fianl comparison since anaerobic conditions produced P release and is not an example of good FTW management.

	Study	Moortel	White	Yang	Stewart*	Stewart*	Borne*	Tanner	Stefani	Winston	Winston	Chang	Chang	RANGE
	% cover	100	100	100	100	100	50	50	19	18	9	8.7	8.7	8.7-100
	notes	anaerobic	anaerobic	anaerobic	aerobic	anaerobic						storm	non-storm	
FTW	TP	12	59	6	91			58	65	88	37	14	47	6-91
FTW	TN	11	67	31						88	48	16	16	11-88
FTW	NH_4	-2		-28	66					98	55	-2	51	-28-66
FTW	NO_3/NO_x	78		91		100				65	82	53	22	22-100
FTW	Cu_{tot}	16					37	57						16-57
FTW	Zn_{tot}	17					63	19						17-63
controls	TP	18			53			3		58	35	4	8	3-58
controls	TN	40								59	36	-26	-15	-26-59
controls	NH_4	33			26					50	8	-8	15	-8-33
controls	NO_3/NO_x	46				42				50	60	-183	12	-183-60
controls	Cu_{tot}	45					17	7						7-45
controls	Zn_{tot}	48					37	-1						-1-48
improvement	TP				38			55		30	2	10	39	2-55
improvement	TN									29	12	42	31	12-42
improvement	NH_4	*study*			40					48	47	6	36	6-48
improvement	NO_3/NO_x	*excluded*				58				15	22	236	10	10-236
improvement	Cu_{tot}						20	50						20-50
improvement	Zn_{tot}						26	20						20-26

Notes:
(*) indicates data extracted from graphs.
In Stewart study, PO_4^{3-} was assessed instead of P

3.1 Seasonal Variation

Seasonal variation in FTW is due to (i) temperature variations, which affect plant and especially microbial productivity, (ii) consequent DO variations due to increased oxygen demand when there is increased microbiological activity, and to some extent the solubility of oxygen in water at different temperatures, and (iii) seasonal growth patterns in plants.

The effect of season on treatment efficiency depends on the main processes involved in their removal, particularly how temperature and oxygen variations affect these processes. For example, spring and autumn are peak P uptake periods for vegetation in wetlands (Kadlec & Wallace, 2009) however, the main process of P removal is settling and adsorption, so seasonal P removal was found to vary less than that of nitrogen (Wittgren & Maehlum, 1997).

Studies have shown conflicting results over how variable treatment efficiency is over different seasons, particularly with N removal, but this is likely to be due to differences in limiting factors. As previously mentioned, plant uptake as NO_3 or NH_4^+ tends to be relatively small compared to microbiological processes (Riley et al., 2005). Thus, studies have found N removal to be affected by seasonal temperature variation (Spieles & Mitsch, 2000; Picard et al., 2005). However, Maehlum and Stalnacke (1999) and Mander et al. (2000) found little difference in N removal between warm and cold climates and Van de Moortel et al. (2010) found more variation due to temperature in P than in N. It is likely that these differences are due to other factors that may be limiting, particularly anoxia. For example, in the study by Van de Moortel et al. (2010) there was low NH_4 removal as 100% island coverage produced low DO and reducing conditions, nullifying any further potential N removal increases due to increased temperature. Also, as mentioned previously, in practice decomposition is not found to be highly temperature dependent in wetlands (Akratos & Tsihrintzis, 2007; Kadlec & Wallace, 2009) and therefore ammonia production rates are not likely to change much with temperature. Thus, interactions between season, light, temperature and DO mean that an individual variable is not a good predictor of activity, and the net effect can be counter-intuitive (Stein & Hook, 2005; Kadlec & Wallace, 2009). However, despite these interacting effects very low temperatures (5 °C) certainly restrict microbiological activity and plant growth (Mitsch and Gosselink 1993).

Rainfall

Rainfall can have a large and varied effect on pollutants entering a basin. If the inflow is from a combined sewer system rainfall events can massively increase dilution and flow rates into the wetland. Van de Moortel et al. (2010) found that heavy rainfall caused a significant reduction in inflow conductivity from 1102 µS to 733 µS, and total nitrogen from 23.1 mg TN/l to 16.9 mg TN/l. Then, after rain events although other constitutions remained diluted in the pond, ammonium and nitrate concentrations actually increased (probably due to microbiological activity). With stormwater treatment ponds, the inflow comes from road run-off which has often had an accumulation of metals during the dry period, so initial concentrations during a storm are usually high, as the metals and particulates get washed off the road, but then rapidly decrease as the storm continues and the concentrations become diluted (Barbosa & Dodkins, 2010).

Rainfall and evaporation also have an effect on dilution within the basins (Kadlec & Wallace, 2009). The addition of rainwater can alter the water chemistry (oxygen, pH), rates of microbiological activity, and affect physical processes e.g. increased depth increasing settling. It is also important to consider that when measuring inflow and outflow concentrations, differences may be due to changes in dilution, rather than any removal within the basin, and loading capacities must take rainfall input and evaporation (and drainage) into consideration (Kadlec & Wallace, 2009).

Shading and Temperature

Floating islands can significantly reduce water temperature (by shading) in the warmer months, and also reduce temperature variation if there is sufficient cover (Van de Moortel et al., 2010). However, Winston et al. (2013) with only 18% island cover, found there was little water temperature reduction (preventing them producing conditions for trout to live in the FTW). Van de Moortel et al (2010) also found that although summer tempertures were lower in FTW compared to open water wetlands, winter temperatures were not lower. However, ice still persisted longer in FTWs during the winter due to reduced wind disturbance at the surface in FTWs.

4. Design Considerations

To achieve treatment objectives careful consideration must be taken in design and operation of the wetland. These must be specific to the flow volume, flow variation, the concentrations of pollutant and the required characteristics of outflow. Wetlands can easily be overloaded with sludge so pre-treatment (removal of large material by bar screen or settling of grit and stones) and primary treatment (sedimentation) are essential for domestic effluents prior to entering the wetland. Good design of these initial stages is also extremely important in maximising the treatment efficiency and the cost of running a FTW and to prevent them becoming unnecessarily clogged by high sludge loadings.

4.1 Island Cover

Since floating islands can restrict oxygen diffusion from the air into the water (Smith & Kalin, 2000), island coverage is an extremely important design factor. For example, an almost complete coverage by islands resulted in poor P retention in sediments due to anoxia (Van de Moortel et al., 2010).

The percentage cover of a pond by the island is one of the most important considerations in FTW design. High cover (>50%) can cause anoxia but low cover (9 to 18%) may produce little additional treatment effect (e.g. Winston et al., 2013). The anoxia is not only caused by islands reducing air-water contact, but also because of the high rate of microbiological processes such as nitrification and decomposition. Thus, the optimum size of the island to prevent anoxia is likely to be dependent on the quality of the influent, particularly ammonia/nitrate and organic carbon concentrations. Flow design, mixing and aeration will also be major factors (See section 3. Treatment Efficiency).

Using more island coverage should increase microbiological activity due to the larger root area, however if high rates of aerobic microbiological activity is to be maintained (e.g. nitrification) oxygen consumption will necessarily be high. To maintain high island coverage without depleting oxygen, bubblers can be installed (Stewart et al., 2008). This requires investment and energy costs, and therefore their use depends on a cost-benefit analysis, although energy can be provided by e.g. solar power. Baffles have also been introduced in some FTWs to increase circulation around the roots. Mixing waters to promote aeration should be done with care as disturbance of sediments can liberate trapped P.

4.2 Optimising for N removal

Since P is effectively conservative, but N can be released as gas through correctly managing the microbial environment, strategies for permanently removing N are very different from those for removing P.

Aerobic and Anaerobic basins

Floating islands may increase denitrification by increasing anoxia, although it is preferable to have an oxygenated basin with a high residence time as a first stage to convert most of the ammonia to nitrate in the nitrification process.

Thus, with N the main objective is to convert as much ammonium as possible to nitrate, usually through an aerobic 1st stage, and then to convert as much of this nitrate to N_2 gas, through an anoxic 2nd stage. Oxygen in the aerobic stage can be rapidly depleted with high coverage of FIs and high rates of microbiological activity, so FI cover has to be carefully managed, or artificial aeration has to be included. Sufficient alkalinity must also be available for nitrification, which can be achieved through the addition of $CaCO_3$ (Stewart et al., 2008).

Nitrification reduces DO and pH, though these conditions are ideal for the next (anoxic) denitrification stage. FI cover can be much higher at this stage. Yang et al. (2008) achieved 97% N removal rates with 0% DO in a hydroponic system, though in more natural systems anoxia can cause sulphide toxicity (Lamers et al., 2002) that kill or restrict root growth.

Recycling

Denitrification is predominantly limited by C supply, with a recommended C:N loading ratio of 5:1 (Bishay & Kadlec, 2005). In a two stage system Carbon limitation often occurs because much of the organic C is removed by settling in the earlier aerobic stage (Kadlec & Wallace, 2009), thus releasing nitrate. Additional C can be supplied artificially, e.g. as glucose syrup (Yang et al., 2008) but for most effluent treatment systems it is cheaper and more practical to seed the anoxic basin with raw effluent that has not gone through the anaerobic stage (Kadlec & Wallace, 2009).

Recycling is also used to return anaerobic outflow back to the aerobic stage; denitrification makes the effluent more alkaline, ideal for further nitrification of ammonia (Kadlec & Wallace, 2009). Recycling is now in Danish treatment wetland guidelines (Brix & Schierup, 1990). Figure 5 gives an example of how FTW wetlands could be designed for treatment of domestic effluent, including recycling.

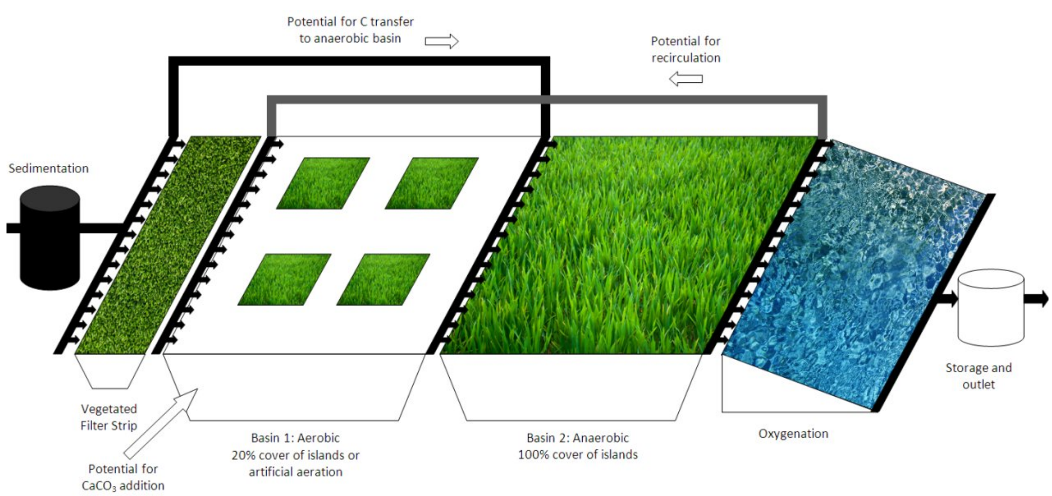

Figure 5. A theoretical design using FTWs for treating low volume domestic effluent (mixture of P, BOD, NH_4^+ and NO_3^- inputs) showing basic design features of combined basins. The vegetated filter strip would have to be adapted/increased/removed depending on solids input from sedimentation.

4.3 Plants

The treatment potential within a FTW depends mostly on the filtering capacity of the roots (root depth and density) and their surface area as a microbiological habitat. Choice of plant species will also affect the rates of nutrient and metal uptake, root/shoot biomass division, growth rates and the way in which the basin water chemistry is altered due to the release of humic acids and protons by plant roots.

Plant dimensions

Tanner & Headley (2011) examined four species growing on floating islands in mesocosms, providing detailed measurements. 90th percentile of root depth averaged between 24 and 48cm, depending on species. The root surface area was between 4.6 and 9.3 m^2/m^2 of floating mat. Above mat biomass was between 834 and 2350 g/m^2 and root biomass of 184-

533 g/m^2 (see Table 3 for more details) with shoot to root ratios of between 3.7 and 4.5. Winston *et al.* (2013) found that *Hibiscus* had shoot:root ratio of 6.3. Indeed, most species have an above mat biomass greater than the below mat biomass, except for Carex spp. such as Carex stricta (Winston et al., 2013), Carex virgata and Cyperus ustulatus (Tanner & Headley, 2011) (Table 3).

Table 5. Mean biomass and shoot:root ratio for FTW plants.

Plant species	Shoots: Mean above mat biomass (g/m^2)	Roots: Mean below mat biomass (g/m^2)	Biomass ratio
[1]*Juncus spp.*	86.3	43.4	2.0
[1]*Carex stricta*	131.4	207.6	0.6
[1]*Spartina pectinata*	121.7	48.1	2.5
[1]*Hibiscus moscheutos*	269	58.9	4.6
[1]*Pontederia cordata*	72	57.7	1.2
[2]*Cyperus ustilatus*	1528	329	4.6
[2]*Carex virgate*	2350	533	4.4
[2]*Juncus edgariae*	1113	299	3.7
[2]*Schoenoplectus tabernaemontani*	834	184	4.5

[1] Winston et al. (2013) in stormwater retention pond
[2] Tanner & Headley (2011) in mesocosm with much more intensive planting

Plant uptake appears to be more associated with total plant biomass rather than root density (Tanner & Headley, 2011), although White & Cousins (2013) found that uptake of N and P by *Juncus effusus* (60.6 N and 3.71 g P/m^2/growing season) was higher than that of *Canna flaccida* (3.71 N and 2.27 g P/m^2/growing season) despite having a similar shoot length, and this was attributed to the much longer roots of *J. effusus*. Nutrient uptake by *J. effusus* was also found to be much higher than that of *Pontederia cordata* in a study by (Chang et al., 2013).

Floating islands are usually allowed 6 months of plant growth to establish before assessing efficiency e.g. (Borne et al., 2013). Once plant growth has reached a maximum (maximum density and shoot biomass) there is no additional net uptake of nutrients by the plants i.e. litter deposition is equal to growth (Kadlec & Wallace, 2009). However, although some of this litter will accumulate on the island, some will sink into the basin with some nutrient release but also with some C and P storage in the sediments.

White & Cousins (2013) found that an increase in nutrient loading increased shoot growth (but not root growth) and suggested that this may be due to a shift in allocation strategy towards shoots when nutrients are plentiful, following Muller, Shmid & Weiner (2000). However, harvesting of the shoots does not appear to affect the root biomass (Borne et al., 2013).

Plant establishment

Vogel (2011) noted that floating island plants have more establishment success and establish quicker, with more cover, when the starting biomass is higher. She recommends planting of as much biomass stock as possible at the start, to aid establishment. The growth rate for some plants may be higher in the first year of establishment, whilst other plants may have higher growth rate in the second year (Svengsouk & Mitsch, 2001).

Buoyancy of islands

Buoyancy of vegetated islands changes seasonally, with mats sinking several centimetres during the spring and summer as the biomass increases (Hogg and Wein 1988 a). Seasonal effects become less pronounced with age, as dead biomass accumulates and decomposition increases (matching biomass accumulation).

4.4 Activated Carbon

Activated carbon is being considered by Frog Environmental Ltd. as a possibility for improving floating island performance prior to the complete establishment of plants by incorporating the carbon within the floating island material. Performance of floating islands usually relates to their ability to increase removal rates for P and N as well as to remove metals, commonly Cu and Zn.

Activated carbon is used in water filters and chemical purification processes. It is highly porous carbon with a high surface area which has been treated by oxygen or sulphuric acid to increase adsorption. It has a surface area of 300-2,000 m^2/g and can adsorb a wide range of pollutants including large organic molecules. Because adsorption works by chemically binding the impurities to the carbon, the active sites in the carbon eventually become filled and adsorption stops. The effectiveness of activated carbon depends on pore size, the carbon source and the manufacturing process.

Typically activated carbon is used to remove metals or organic pollutants rather than nutrients. This is because the surface of activated carbon is negatively charged, attracting positive ions (e.g. Cu^{2+}, Zn^{2+}) rather than negative ions (NO_2^-, NO_3^-). Bhatnagar & Sillanpää (2011) reviewed the adsorption of nitrate on to various carbon substances. Results vary with 1mg/g (Mizuta et al., 2004), 1.7 mg/g (Bhatnagara* et al., 2008) and 4 mg/g (Oztürk & Bektaş, 2004) adsorption of NO_3^-, although these studies are all done in lab conditions and are better than can be expected in the field. Biochar (a form of charcoal) has been tested in field for nitrate removal, though it has tended to have low effectiveness except where nut shells have been the carbon source of biochar (Knowles et al., 2011; Yao et al., 2012).

Nitrate adsorption depends on contact time. Oztürk and Bektaş (2004) achieved complete adsorption within 1 hour at pH<5.0 and 25 °C. Optimal pH for activated carbon adsorption of nitrate occurs at pH2. This is because H+ ions bind to surface and reduce –ve charge, increasing uptake of –ve ions (NO_3^-). Problems with extrapolating these results to the field include (i) the nitrate could be bound to other substances within the water column or sediments, (ii) there would be a diffusion gradient between the site of adsorption (the island) and the bottom of the basin, (iii) the effluent is not being passed through the carbon, so adsorption is passive (iv) optimal pHs for adsorption would not be suitable for a treatment basin, which should be kept around neutral pH.

Ammonia adsorption is around 5.08 mg NH_3/g of carbon at 20 °C increasing to 5.80 mg/g at 60 °C (Long et al., 2008). The temperature of activation of the carbon also affects the adsorption capacity, with higher activation temperatures increasing ammonia adsorption (Ghauri et al., 2012).

P removal in wetlands tends to be predominantly through physical sedimentation processes, which are aided by particle interception by plant roots. When P removal was tested with activated carbon adsorption capacity was 1.11 mg /g at high P concentrations, decreasing with lower P concentrations (Liang et al., 2011). Optimum pH for adsorption ranges between 6 and 10 (Kumarab et al., 2010), which is ideal for FTWs, although again, these studies use data for filtration of nutrients rather than passive adsorption at the surface of the basin.

Adsorption rates of different nutrients are dependent on the nutrient concentration in the effluent, and at low concentrations close to 100% removal is theoretically possible. However, with low circulation and a diffusion gradient within a treatment basin it is unlikely that high percentage removal rates are possible. In addition, we would expect to use around 100 times more carbon (by weight) than the nutrient we are reclaiming, which is likely to be prohibitively expensive.

Activated carbon can provide a carbon source for improving denitrification when C is limiting (Isaacs & Henze 1995; Yang *et al.* 2008). This may be particularly important prior to the establishment of vegetation, which would then provide a source of carbon through decaying organic matter. However, addition between the layers of the floating island may be less useful than simply mixing the powdered activated carbon into the effluent as it enters the basin. Also, a soluble carbon source such as glucose (Yang et al., 2008), acetate or hydrolysate (Isaacs & Henze, 1995) may be better for encouraging denitrification than powdered activated carbon.

5. Conclusions and Recommendations

The main function of floating islands in removing pollutants from effluents is:

- Plant roots assisting in filtering and settling processes for P
- Plant roots acting as a large surface area for micro-organism activity in: decomposition, nitrification, and denitrification (removal of BOD and N).
- Mild acidification of water due to release of humic acids, and a C input from senescent vegetation; assisting denitrification.

P removal is predominantly a physical process. It binds to particulates and removal is assisted by the reduced water movement and the filtering effect of roots on these particulates. This sloughs off to the bottom sediments. Metals are also removed predominantly through binding to particles and sedimentation. Reduced DO in the basin and disturbance of the sediments can result in release of P and metals from the sediments. P is effectively conservative, and if dredging of the sediments is not done (around every 10 years is suggested) the sediment bound P and dissolved P will reach an equilibrium whereby there is no net P removal (and potential for pulses of P in the outflow which are higher than that in the inflow).

N removal is predominantly a microbiological process with NH_4^+ being nitrified to NO_3^- in aerobic basins by nitrifying bacteria, then NO_3^- being denitrified to N_2 gas (and thus released) in anaerobic basins by denitrifying bacteria. FTWs have excellent potential for removing N from effluents. An initial aerobic basin (up to 20% island cover or 100% with aeration) can be used for nitrification and then a second anaerobic basin (100% island cover) can be used for denitrification. Up to 100% N removal is possible, with more tightly controlled conditions increasing the ability to remove N. At the aerobic stage the addition of $CaCO_3^-$ can assist with nitrification (as alkalinity is used up during this process). At the anaerobic stage the addition of C can assist with denitrification (as carbon compounds are used during this process). This C may be added as e.g. glucose or molasses, or as BOD fed from the FTW inlet.

With good management but without hydroponic conditions (i.e. aeration, $CaCO_3$ or artificial C addition) we could expect a FTW wetland to achieve around 60% removal of TP, 75% removal of TN, 50% removal of NH_4^+, 80% removal of NO_3 and 40% removal of metals. All these are expected to be significant improvements (around 20-40% higher) than with basins without islands, depending on specific conditions. More controlled conditions could considerably increase the treatment rates.

Plant uptake only accounts for up to 6% of nutrient (N and P) removal in FTWs. This is also recycled into the system through decomposition unless harvesting is undertaken. Although concentrations of nutrients and some metals (e.g. Zn) are higher in the roots, shoot biomass of plants tends to be higher. Thus shoot harvesting often removes a little over half of the nutrients taken up by the plants. Floating islands also provide access for root harvesting, but harvesting of roots is unlikely to be beneficial as it is more time consuming and also reduces the filtering capacity and microbiological activity associated with the root network: the principal mechanisms of nutrient removal in FTWs. Evidence suggests that removal of shoots does not negatively affect the roots.

FTWs have other advantages over conventional Free Water Surface Wetlands:

- They can adjust to varying water levels

- A higher retention time is possible as they can be made deeper without submerging the vegetation
- Habitat value for birds/amphibians

Recommendations for domestic effluent treatment:

Domestic effluent usually has high BOD, NH_4^+, NO_3^- and P although specific operation and design of FTWs should be tailored to the specific characteristics of the inflow.

1. N removal is the principal benefit of FTWs:
   - An aerobic basin for nitrification is required to convert ammonia to nitrate
   - An anaerobic basin for denitrification is required to convert nitrate to N_2 gas.
   - Although nitrification and denitrification can be achieved in the same basin, separate aerobic/anaerobic basins can be used to more easily control the processes.
2. Depending on cost considerations and inflow water alkalinity, $CaCO_3$ can be added in the aerobic basin to aid nitrification.
3. C can be added in the anaerobic basin to aid denitrification. In smaller treatment systems requiring high water quality outflow a hydroponic system with glucose or molasses addition can be used. For larger treatment systems with greater costs considerations, input of C can come from a controlled input of BOD directly from the FTW inflow.
4. 100% cover of islands, with mixing, is optimal for N reduction in the anaerobic basin.
5. 100% cover of islands with aeration (bubbling) is optimal for aerobic nitrification. If cost considerations prevent aeration, 20% island cover is recommended in the aerobic basin to prevent anoxia occurring.
6. A recycling system from the anaerobic to the aerobic basin, although not always necessary, may be useful when there is excessive NH_4^+ in the outflow i.e. to increase nitrification rates.
7. Aeration is required after the denitrification basin to prevent the release of anoxic waters to the environment.
8. Circum-neutral pH should be maintained in anaerobic and aerobic basins. If pH drops considerably there is a danger of P release.
9. Dredging, particularly of the first (aerobic) basin is recommended every 10 years to remove P trapped in sediments, as well as accumulating metals. Alternative (dormant) treatment basins may be required to be made operational treat effluent as dredging operations are undertaken in the main basin.
10. Plants with high root surface area and high plant biomass are recommended for the floating islands e.g. *Juncus effusus*. Ecological considerations may result in other species being chosen or plant mixtures being used.
11. Harvesting should be done, but only of the shoots.

Use of Activated Carbon

The use of activated carbon between layers of floating island material to assist in pollutant removal will probably have limited effectiveness. This is due to a diffusion gradient between the surface of the basin and the bottom of the basin, and the passive nature of adsorption i.e. the effluent is not being filtered through the medium. At the very most (with high retention times and full adsorption) N and P removal is likely to be about 1g for every 100g of activated carbon used. The proper establishment of plants, a focus on correct basin design, and water chemistry control, is likely to be a much more effective use of resources.

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