THE HUMBLE WETLAND – A CARBON SUPERHERO?

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ABSTRACT

The effects of changing climate are significant and vary by location. New Zealand has significant coastline and the risk of climate-related disasters is increasing, threatening our way of life. Strategies to mitigate climate change are needed. Wetlands can play a role in this strategy and could be called a "carbon superhero".

In New Zealand, in a municipal wastewater context, wetlands are a common treatment process. Wetlands store significant volumes of carbon in plant biomass and soil. Peat wetlands are considered "super" carbon sinks, holding twice as much carbon as the world's forests.

The majority of drained peatland in Aotearoa is used for intensive farming. Dried peatland emits carbon and is responsible for up to 6% of agricultural emissions in New Zealand. A number of studies have recommended that natural and coastal wetlands be restored globally to mitigate this.

This paper outlines how wetlands can accumulate peat, which is needed for carbon sequestration, to help achieve decarbonisation. A case study demonstrating the production of nitrous oxide and methane from wetlands compared to a conventional wastewater treatment process has been summarised. Overall, this paper shows that the wetland could be reinvigorated as a critical piece of wastewater infrastructure with multi-faceted benefits.

KEYWORDS

wetland, sustainability, decarbonisation, greenhouse gas

PRESENTER PROFILE

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Kate Simmonds is an environmental engineer with over 18 years' experience in the water industry. Kate is passionate about strategic planning and innovation and is always looking for ways to improve the performance and benefits of our existing infrastructure, to achieve positive environmental, social and economic benefits for our communities.

Lucy Ferris

Lucy Ferris is a civil engineer with over five years' experience in the water industry. Lucy is passionate about the environment and incorporating innovative, sustainable and comprehensive engineering solutions in order to leave a positive impact on the world today and for future generations.

INTRODUCTION

It is well understood that the climate is changing at an extraordinary rate. The effects of this changing climate are significant and vary by location. It is predicted that these changes will result in intensifying storm activity, rising sea levels and more frequent floods and droughts (Intergovernmental Panel on Climate Change, 2014). Globally, the risk of climate-related disasters is increasing, particularly in coastal areas. New Zealand has significant coastline, the 9th longest in the world, and therefore climate related disasters are a real threat to our way of life.

Strategies to mitigate climate change and adapt to its changing conditions are needed now, more than ever. Wetlands can play a vital role in this strategy and the humble wetland might even be called a "carbon superhero".

In New Zealand, in a municipal wastewater context, wetlands are a common process in the overall wastewater treatment train. Wetlands as part of the treatment process are particularly evident in smaller and more rural wastewater treatment plants (WWTPs).

Wetlands, particularly peatlands and coastal systems (i.e., mangroves), store significant volumes of carbon in plant biomass and especially in the soil. Peat wetlands are considered "super" carbon sinks, holding twice as much carbon as all of the world's forests combined, estimated at between 180 and 450 Gt globally (Joosten, et al., (2016)), yet covering less than 3% of the earth's surface.

This poses the question – is the wetland a carbon superhero – and could strategic use of wetlands in WWTP treatment trains result in improved overall environmental outcomes for wastewater management and treatment?

CURRENT NEW ZEALAND SITUATION

The Resource Management Act 1991 defines wetlands as "permanently or intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions". It is estimated that New Zealand has lost approximately 90% of our wetlands following European Settlement (Clarkson, et al., 2013).

Wetlands are important for biodiversity, birds, fish, plants and people, for example providing spawning grounds for native plants, birds and fish and helping produce weaving materials such as raupo and harakeke. The numerous values and uses of wetlands have been well documented. Healthy wetlands are part of a healthy environment; yet wetlands continue to be lost, degraded, undervalued, ignored, and destroyed both deliberately and through lack of understanding of their importance (Johnson & Gerbeaux, 2004).

Photograph 1 Example of a wetland (Crown Copyright: Department of Conservation: Te Papa Atawhai, n.d.)

Constructed wetlands (CWs) are an established technology for secondary or tertiary treatment of wastewater, with over 80 systems in operation in New Zealand (Sukias & Tanner, 2004). Wetlands as part of the treatment process are particularly evident in smaller and more rural wastewater treatment plants (WWTPs). They appeal to communities due to being a "natural" treatment process, and potentially providing social benefit (bird watching and recreation) and in some instances can be considered as providing contact with Papatuanuku to help restore the waters' mauri.

WETLANDS IN WASTEWATER TREATMENT

Wetlands can be used for wastewater polishing as an alternative for small communities that are either situated near an existing wetland or in a position to construct one (Kadlec & Bevis, 1990). In early 1950s Germany, Käthe Seidel carried out the first experiments using wetland macrophytes for wastewater treatment by designing horizontal sub-surface flow constructed wetlands (HF CWs) (Vymazal, 2005). Since then, the HF CWs technology has spread worldwide and is now used to treat different types of wastewaters other than the usual domestic and municipal, such as industrial and agricultural, landfill leachate and runoff waters (Vymazal, 2010).

CWs are engineered systems that have been designed and constructed to utilise the natural processes involving wetland vegetation, soils, and their associated microbial assemblages to assist in treating wastewater. CWs are designed to take advantage of many of the processes that occur in natural wetlands, but do so within a more controlled environment (Vymazal, 2007). Vymazal (2007) found that single-stage CWs could not achieve high removal of total nitrogen (N) due to their inability to provide both aerobic and anaerobic conditions at the same time and therefore various types of CWs, i.e. HF, vertical flow, or free water surface, may be combined in order to achieve a higher treatment effect, especially for N removal.

To evaluate the efficiency of CWs, Environmental Waikato requested NIWA to undertake an assessment of the performance of seven CWs that treat domestic wastewater in the Waikato Region (Sukias & Tanner, 2004). Sukias and Tanner (2004) found that the wetlands had good rates of removal of BOD and suspended solids, moderate rates (17-33%) of total kjedahl nitrogen removal and negligible phosphorus removal. However, low removal rates of phosphorus are typical in all types of well-established CWs (Vymazal, 2007). A study by Kadlec and Wallace (2009) spanning a number of sites around the world, shows median removal rates of 36% for total phosphorus, and examples in New Zealand of -76% to +80% phosphorus removal. These removal rates demonstrate that CWs can be an efficient step in a wastewater treatment train.

WETLAND RESTORATION

It is noted that the majority of the drained wetlands (peatland) in New Zealand have been reclaimed and are used for intensive farming. Dried peatland emits carbon and is responsible for up to 6% of agricultural emissions in New Zealand (Forest and Bird, 2021). In recognition of this, a number of studies globally have recommended that natural and coastal wetlands be restored all around the world.

Restoration of wetlands can lead to a reversal of the carbon oxidation that came about as a result of drainage of peatlands, which would then make restored wetlands a sink of atmospheric carbon dioxide (CO_2) again (Hemes, et al., 2019). However, there is a time lag after the restoration until processes in restored wetlands become similar to those of natural wetlands (Lal, 2008).

With restored wetlands, a minimum of 55% vegetation cover is needed to become a net carbon sink, which most wetlands can achieve once vegetation is established, sometimes as early as two years after restoration. Bathymetry design and water depth are the key factors of vegetation establishment after restoration. A reduction in bathymetry variations can lead to higher vegetation to water ratios, which is key when designing wetlands for carbon sequestration (Valach, et al., 2021).

A global net carbon sink by 2100 could be achieved through peatland protection and restoration policies, if around 60% of present-day degraded peatlands could be restored in the coming decades, along with the protection of existing peatlands (Humpenöder, et al., 2020). Peatland restoration may become more attractive to policy makers in the near future as it provides a new opportunity for investing in ecosystem-based mitigation through the development of carbon markets due to peatland restoration’s cost effective climate mitigation and abatement potential, comparable to other measures (Bonn, et al., 2014). Bonn, et al. (2016) elaborate that there is encouraging progress in this space, including the renegotiation of the Kyoto Protocol and other instruments under the United Nations Framework Convention on Climate Change (UNFCCC) , the Wetland (Ramsar) Convention and the Convention on Biological Diversity (CBD), the European Union (EU) Common Agricultural Policy (CAP) reform, and the implementation of the EU Water Framework Directive (WFD). All of which recognise the need to conserve peatlands for their essential ecosystem services and for underpinning biodiversity.

Photograph 2: Waiatarua Reserve – New Zealand's biggest urban wetland restoration project, Meadowbank, Auckland, North Island, New Zealand. (Photo: Jay Farnworth. (Stuff Limited, 2019))

An example of a wetland restoration project undertaken by Jacobs, is the 4G Wetlands in southwest Florida, USA. The 4G Wetlands consisted of a 176-acre groundwater recharge wetland system built on wet pastures with the aim of reversing groundwater drawdowns in areas affected by public water supply wellfields in the region. Flow to the wetland is 18.9 MLD of nitrified, secondary effluent. The natural design of the wetland cells blended in with the natural environment, creating biological diversity and providing significant additional acreage of wildlife habitat. Benefits achieved from the wetland restoration included water quality improvement, such as the removal of nitrate-nitrogen inherent in the reclaimed water to protect groundwater and adjacent surface water quality. Water quality improvements occur through biological processes of wetland surface and soil treatment as water infiltrates through the wetland subsoils protecting aquifer water quality. Water percolating through wetland sediment undergoes complete denitrification.

For restoration of peatlands, the wider scale hydrological effects of the 4G wetlands are of special interest. The zone of rehydration, restoring the status quo ante of wet meadows, is well over ten times the area of the wetland itself. Thus, design of similar wetlands in areas of former peat wetlands have the potential to rewet peatlands at similar scales in which 1 ha of treatment wetland may restore over 10 ha of peatlands.

Photograph 3: 4G Wetlands –Addressing Florida's groundwater supply with the largest groundwater recharge wetland in the world. (Photo: Jacobs.com)

WETLANDS FOR DECARBONISATION

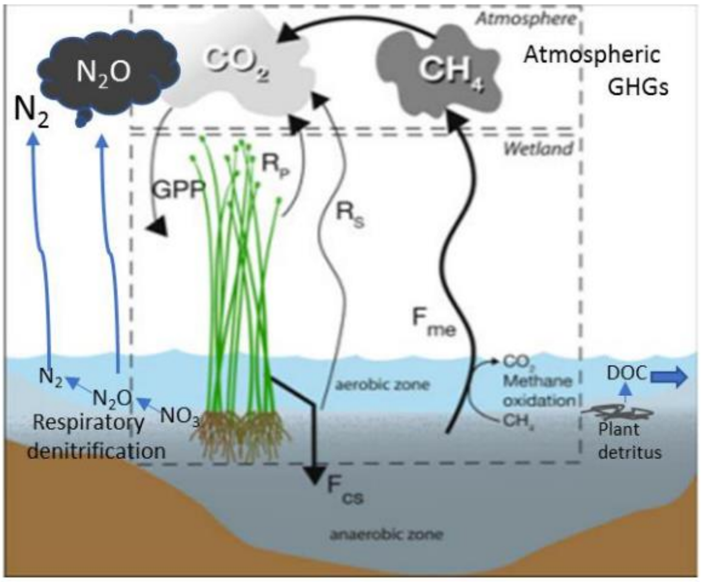
Wetlands, particularly peatlands and coastal systems (i.e., mangroves), store significant volumes of carbon in plant biomass and especially in the soil. Peat wetlands are considered "super" carbon sinks, holding twice as much carbon as all of the world's forests combined, estimated at between 180 and 450 Gt globally (Joosten, et al., (2016)), yet covering less than 3% of the earth's surface.

Wetlands are dynamic and natural ecosystems, characterised by waterlogged conditions or standing water conditions during at least part of the year (Adhikari, et al., 2009). Inundated wetlands can potentially sequester substantial amounts of soil carbon long-term due to slow decomposition and high primary productivity, particularly in climates with long growing seasons (Valach, et al., 2021). Figure 1 shows the rate of carbon sequestration in a wetland is the change in carbon dioxide equivalent of all GHGs (CO_{2e}) storage, including emissions of the greenhouse gases methane (CH_4) and nitrous oxide (N_2O) as CO_{2e} (Burrows, et al., 2018).

Peatlands are wetlands with an organic soil layer of at least 30 cm, which may extend up to 15-20 m depth (Limpens, et al., 2008). The habitat requirements for peat initiation and accumulation are similar in every geographical location, those being waterlogging, low pH, low nutrient availability, low oxygen supply and reduced decomposition rate. However, the physical and chemical characteristics differ according to specific site characteristics of landscape area and topography, climate, water depth and flow, nutrient availability and biogeographical availability of plant species (International Peatland Society, 2021).

Peat formation is the result of incomplete decomposition of the remains of plants growing in waterlogged conditions. (International Peatland Society, 2021). Peat accumulation occurs when plant production exceeds organic losses from a site. This usually occurs in wetlands where very cold or anaerobic sediments inhibit soil respiration, resulting in mean long-term rates of peat accumulation being higher in boreal and temperate peat deposits (Ovenden, 1990). Ovenden (1990) found that the mean long-term rate of carbon accumulation in a peat deposit (g $C/m^2/yr$) is the product of the peat accumulation rate (cm/yr) and its carbon concentration (g $C/cm^3 \times 10^4$). This rate depends on the productivity of the aquatic vegetation that in turn is determined by the nutrient content of the water and, initially, the mineral substrate (International Peatland Society, 2021).

Figure 1: Simplified conceptual model of carbon and nitrogen flows in a wetland and exchanges with the atmosphere (Burrows, et al., 2018)



Note: F_{cs} = carbon sequestration; F_{me} = methane emissions; GPP = gross primary productivity; R_p = plant respiration; R_s = soil respiration; DOC = dissolved organic carbon.
Adapted from Mitsch et al. 2012. The gas clouds indicate relative strength of CO_2, CH_4 and N_2O. Conversion factors were 3.7 for C to CO_2 and 21 and 298 for CH_4 and N_2O respectively (IPCC 2007). N_2 has no GHG effect.

Long-term carbon sequestration is a function explicitly restricted to actively peat accumulating systems. Peat accumulation is only possible when the water level in the peatland is - on average in the long-term - near the surface. The exact level depends on the peatland type. Both too low and too high-water levels are detrimental to peat accumulation and the associated functions (Schumann & Joosten, 2008). Yin, et al. (2019) found that the topsoils (0-10 cm) in a wetland contained the highest soil organic carbon contents.

LITERATURE REVIEW SHOWING WETLAND DECARBONISATION

Generally, the distribution of peatlands globally follows that of wetlands. According to Xu, et al., (2018), the majority of the worlds' peatlands are situated in Asia (38.4%) and North America (31.6%, mostly Canada & Alaska). European peatlands make up 12.5%, followed by South America (11.5%), Africa (4.4%), and Australasia and Oceania (1.6%). This correlates to a distribution of 83.3, 4.0, and 12.7% from the boreal (and polar), temperate, and tropical zones respectively. The global applicability and vast carbon sequestering potential of wetlands has led to studies being conducted around the world.

Mitsch, et al. (2013) modelled the carbon flux results from their own studies of seven temperature and tropical wetlands combined with that from 14 other wetland studies by others and showed that methane emissions became negligible within 300 years compared to carbon sequestration in wetlands. As a result of their research, Mitsch, et al. (2013) estimated that the world's wetlands were net carbon sinks of about 830 Tg/year of carbon with an average of 118 g-C/m^{2}year of net carbon sequestration.

Whiting and Chanton (2001) found that over a large time scale, well established wetlands may be a greenhouse gas sink when the amount of CO_2 that is removed from the atmosphere and stored in the carbon pool is greater than the release of greenhouse gas equivalents associated with methane (CH_4) emission. A wetland is a greenhouse source when the CH_4/CO_2 ratio is elevated, and the Global Warming Potential of the methane is considered over a short time scale. It is around the 100-year time horizon that wetlands switch from being carbon sources to carbon sinks (Whiting & Chanton, 2001).

It is important to note that treatment wetlands do not necessarily need to be methane producers. Nitrified secondary effluent contains concentrations of nitrate well in excess of 40 mg/L (as the nitrate ion) which is an energy terminal electron acceptor for bacterial oxidation of methane. Research is needed to understand the differences in methane balance between denitrifying treatment wetlands and natural wetlands.

Adhikari, et al. (2009) provides a review that summarises carbon storage along with the mechanisms and factors affecting carbon dynamics in wetland ecosystems. They found that wetlands may affect the atmospheric carbon cycle in four ways. Firstly, many wetlands especially boreal and tropical peatlands have highly changeable carbon and these wetlands may release carbon if water level is lowered or if management practices results in oxidation of soils. Secondly, carbon dioxide enters a wetland system via photosynthesis by wetland plants giving it the ability to alter its concentration in the atmosphere by sequestrating this carbon in the soil. Thirdly, wetlands are prone to trap carbon rich sediments from watershed sources and may also release dissolved carbon into adjacent ecosystems. This in turn affects both sequestration and emission rates of carbon. Lastly, wetlands are also known to contribute in the release of methane to the atmosphere even in the absence of climate change (Adhikari, et al., 2009).

NEW ZEALAND LITERATURE

Ausseil et al. (2015) conducted a study on the estimation of carbon stocks of 126 freshwater wetland sites across New Zealand. The wetlands were classed by their soil types, either as mineral or organic (peat) and then further classed as either, a bog, fen, swamp, marsh, ephemeral or pakihi. These are six out of the nine total wetland classes that are recognised; those aforementioned as well as seepage, shallow water and saltmarsh. Differentiation between the types of wetlands is governed by combinations of substrate factors, water regime, and the consequent factors of nutrient status and pH (Johnson & Gerbeaux, 2004). The carbon density for mineral and organic wetlands were found to have means of 121 and 1348 t C ha^{-1}, respectively; which is comparable to the range of values reported in literature for wetlands in other parts of the world (Ausseil, et al., 2015).

Extensive areas of peat bogs in the Waikato region of New Zealand have been converted to dairy farming, which has resulted in subsidence of the natural peat stores (Schipper & McLeod, 2002). Schipper and McLeod (2002) studied the Moanatuatua peat bog, south of Hamilton, North Island, New Zealand, in order to estimate peat subsidence rates and total carbon losses, due to 40 years of dairy farming in the region. They measured the thickness of peat and total carbon of the Moanatuatua peat bog and surrounding farmland above a marker tephra layer that was deposited about 200 AD. Subsidence rates averaged 3.4 cm yr^{-1} (95% confidence interval of 3.2 to 3.5 cm yr^{-1}) and carbon loss averaged 3.7 t ha^{-1} yr^{-1} (95% confidence interval of 2.5 to 5.0 t ha^{-1} yr^{-1}).

Two notable studies on the rate of carbon sequestration in natural wetlands in New Zealand have been carried out (Campbell, et al., 2014, Goodrich, et al., 2017). The results of the studies showed that New Zealand has favourable conditions for carbon sequestration due to low altitudes and a moderate climate. However, the main differentiation for New Zealand wetlands having larger CO_2 sink capabilities in contrast to the values reported for peatland ecosystems in the Northern Hemisphere, was the year-round productivity of the evergreen vegetation cover in New Zealand (Campbell, et al., 2014).

ABILITY OF WASTEWATER WETLANDS TO SEQUESTER CARBON

Considerable information is available on treatment wetland (TW) design and GHG issues associated with natural wetlands, but much less information is available on GHG emissions from TWs, where nutrient and carbon loading tend to be considerably greater than in nature (Jordahl, et al., 2008).

Photograph 4: Aerial view of Maungarei Springs (Stonefields) Wetland, Auckland, North Island, New Zealand. (Photo: Stu Preece)

The release of GHG, especially methane, is the inevitable result of inundating land rich in organic matter with water. These conditions support microbial carbon processing reactions, and are characteristic of all constructed wetlands, including TWs. Releases of GHG from TWs have been found to be comparable to natural wetlands. It is clear that, in general, the creation of wetlands will sequester large amounts of carbon in living vegetation and detritus, but there will be a release of CO_2, N_2O, and CH_4 that will vary with climate, season of the year, wetland type, and loading rate.

There is a rapidly growing demand for developing detailed carbon footprints for all human activities. CO_2 release from water treatment facilities may be considered entirely biogenic, in that CO_2 from degradation of plant tissue represents a cyclic return to the atmosphere. The degradation of the organic load in wastewater to CO_2 also represents no net increase in GHG emission unless there is an increase in CH_4/N_2O as compared to other treatment processes. The CO_2 captured by wetland plants through photosynthesis that is subsequently degraded and released as CO_2 provides no net contribution of GHG. Degradation of carbonaceous compounds from wastewater in a TW eventually leads to CO_2 release to the environment, and mineralisation within the wetland to CO_2 again does yield a net contribution of GHG. On the other hand, conversion of organic compounds in wastewater or wetland biomass to CH_4, and conversion of organic and mineralised N to N_2O, represent the major potential negative impacts of TW on GHG that need to be understood (Jordahl, et al., 2008).

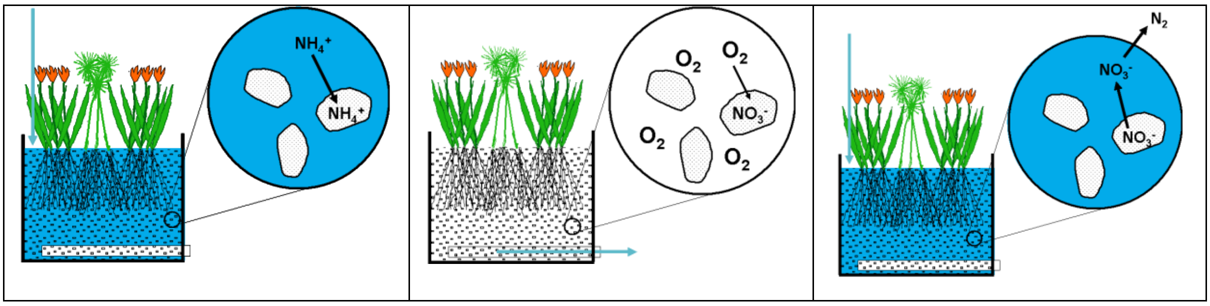
Some studies have shown that vegetated zones in free water surface and horizontal sub-surface flow (HSSF) TWs have reduced emission of CH_4 compared to unvegetated zones. As a percentage of carbon loading, approximately 2 to 4 percent of the carbon in wastewater applied to vegetated wetlands is released as CH_4, as compared to 7 to 8 percent for unvegetated areas (Tanner, et al., 1997). Plant roots can introduce oxygen into the root zone through arenchyma (specialised tissues in wetland plants that facilitate the exchange of gases between the root zone and the atmosphere), shifting the balance toward methanotrophy.

The data on GHG emissions from TWs under various conditions suggests a number of design options that could be explored to reduce emissions. However, some of the measures to mitigate GHG emissions come at the cost of a reduction in other types of environmental benefits, the efficiency treatment, or cost-effectiveness.

FLOOD AND DRAIN WETLANDS

Operating the wetlands with variable water levels, including lowering the water table below the soil surface during the growing season, can help reduce CH_4 emissions, but will significantly reduce treatment capacity for wastewater constituents such as nitrate, and may increase releases of phosphorus and metals accumulated in wetland sediments. However, for seasonally discharging wetlands, this technique could be considered. Flood and drain (tidal) designs (Figure 2), with regular and frequent fluctuations in water level, can still provide high levels of treatment for a number of constituents and would likely have very low emission of CH_4 relative to more common constant-flow designs. They also have the benefit of a significantly reduced footprint.

Figure 2: Flood and Drain (Tidal) Flow Wetland Schematic



Tidal flow wetlands use cation exchange for oxygen transfer. Positively charged ammonium ions (NH_4^+) adsorb to negatively charged aggregate surfaces when wetland is flooded. When drained, ammonium ions oxidise to nitrate (NO_3^-) in the presence of atmospheric oxygen (O_2). When flooded again, the negatively charge nitrate ions desorb from aggregate surfaces and denitrify if organic carbon is present.

AMENDMENTS

The addition of amendments could be considered to help regulate microbial processes that impact GHG emissions. Acidic conditions tend to reduce the ratio of N_2O/N_2 produced, so pH adjustments could be considered as a means to decrease emissions of N_2O. Molybdenum is an essential cofactor for the enzymes that perform nitrate and sulfate reduction, and micromolar additions to some wastewaters could potentially help favour growth of denitrifiers and sulfate reducers over methanogens. Substantial quantities of amendments would reduce the operations and maintenance cost advantages of TW over conventional treatment however, which may limit this approach in comparatively large systems. Gypsum or other sources of sulfate also have the potential to limit methane generation by favouring utilisation of organic substrates by sulfate reducers.

AERATION OF INLET ZONES

Especially for HSSF designs, the CH_4 emissions could likely be reduced by increasing the redox potential of the inlet zone. This may be accomplished through recirculation of nitrified effluent, aeration, or intermittent loading to favour aerobic degradative processes.

COMPARISON OF EMISSIONS – TREATMENT WETLANDS VERSUS CONVENTIONAL TECHNOLOGIES

The most common use of wetlands in wastewater treatment in NZ is polishing to supplement secondary treated wastewater rather than raw wastewater treatment, particularly for surface flow wetlands. However, globally there are over 4,000 treatment wetlands in France that receive coarsely screened, raw influent, and more than 10,000 wetlands in Europe that treat septic tank effluent. In terms of total area of wetland, by far the largest area is devoted to stormwater treatment (e.g. > 16,000 ha in Florida).

To allow comparison of emissions from wetlands versus conventional advanced wastewater treatment in treating typical municipal wastewater, calculations of infrastructure and carbon footprints are required. Treatment wetlands typically use less energy comparative to conventional treatment technologies. A typical wetland system at a WWTP may require pumping of wastewater to its inlet works, but under normal design circumstances water inside the wetlands will flow entirely by gravity. Typical energy requirements are therefore limited to an inflow pump station and powering of automated control systems. The nature of plant growth that drives microbial and physical pollutant removal processes in the wetland means no further energy or chemical inputs are generally required.

A study by Jacobs (Jordahl, et al., 2008) assessed the emissions from typical surface flow wetlands treating varying qualities of secondary effluent to a tertiary level. The estimated footprint to achieve the target water quality from each of the three influent waters was found to require significant footprint (assuming treatment of 38 megalitres per day of flow). However, it should be noted that emerging wetland technology can significantly reduce the required area (>90%) needed for nitrification (Austin, 2019).

The estimated carbon dioxide equivalents required for conventional tertiary treatment for the same flows and loads were also calculated, to allow comparison. The results factor GHG emissions due to power demand (excluding decarbonized power sources) and chemical usage. The numbers also include direct N_2O emissions in the tertiary treatment process. For the analysis, direct CH_4 and CO_2 emissions were considered biogenic and omitted.

The assessment concluded that while there are emissions of greenhouse gases from treatment wetland facilities, they are less than conventional treatment facilities, and these releases must be weighed against the ecosystem function and environmental services provided by treatment wetlands that conventional advanced water treatment facilities lack. Therefore it is important to assess the net environmental benefit.

NET ENVIRONMENTAL BENEFIT ANALYSIS

Natural resources can be managed to produce direct services for humans (e.g. outdoor recreation, potable water), indirect services for humans (e.g. wastewater treatment, flood moderation), and ecosystem support (e.g. primary productivity, wildlife habitat). These services all have economic value to humans and are called “environmental benefits”. Net environmental benefit analyses (NEBAs) are applied strategies for assessing the environmental consequences of any action. These analyses compare and rank the net environmental benefits associated with multiple management alternatives and other actions impacting natural resources. Environmental benefits may be quantified in monetary units or non-monetary units or they can be described in qualitative terms. As growing attention is paid to carbon footprints and global climate change implications of infrastructure development, NEBAs are likely to be more commonly used to evaluate the ecological and environmental consequences among various comparable wastewater treatment technologies (Jordahl, et al., 2008).

Recent applications of NEBA in regional wastewater planning efforts (Madison, et al., 2008) illustrate that wetlands provide a number of benefits over conventional treatment process alternatives including:

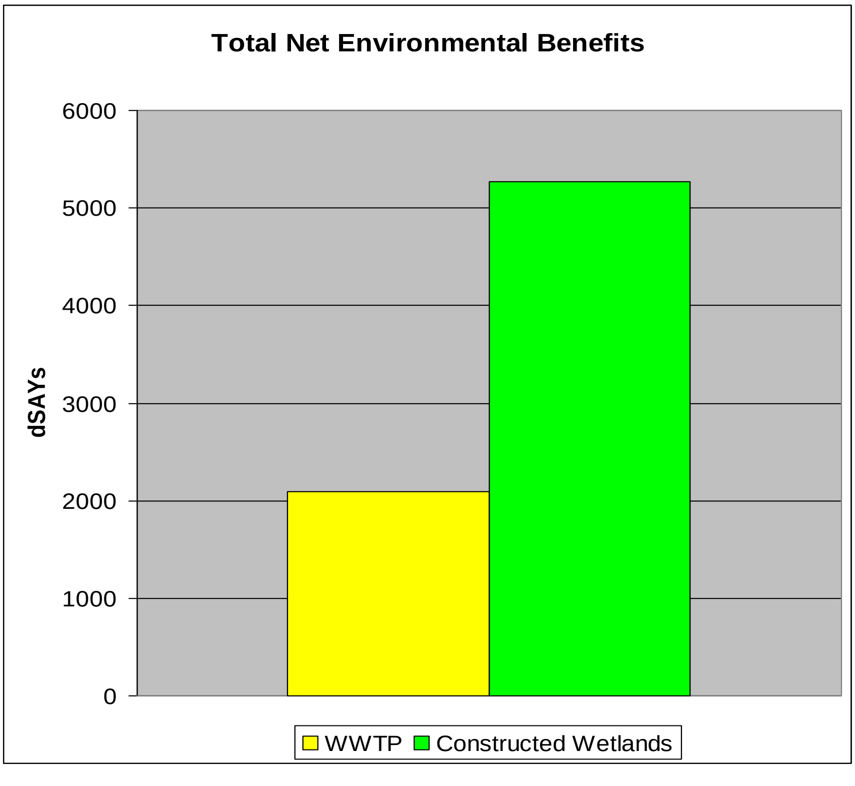
- Open space or habitat creation/restoration
- groundwater recharge
- watershed/hydrologic connectivity
- creation of recreational amenities and promotion of "green" technologies among the public

Ecosystem support services are best quantified in ecological units, and this approach was used to compare the ecological performance of constructed wetlands versus conventional wastewater treatment through NEBA (Madison, et al., 2008).

The NEBA was applied to 312.6 acres of land proposed for permanently constructed wetland cells, pipelines, pump stations, and outfalls. A range of habitat types were applied including pasture and emergent wetlands. The constructed wetlands alternative used mostly emergent wetlands (approximately three quarters of the total area). For the conventional wastewater treatment option, it was assumed no landscape changes would occur. The methodology and assumptions for valuing ecological services in this NEBA are available upon request from the authors.

The analysis showed that the constructed wetlands treatment alternative would provide about a 2.5 times greater amount of valued ecological services (in terms of discounted-service-acre-year – DSAY) than the conventional wastewater treatment alternative (Figure 3).

Figure 3: Comparison of Total Net Environmental Benefits (DSAYs) for Two Wastewater Treatment Options (Jordahl, et al., 2008).



The margin of overall improvement allows a degree of comfort that the constructed wetlands treatment option would provide net ecological benefits, even before water quality benefits are considered.

Other attributes in addition to GHG emissions and total carbon footprint for treatment wetlands options could include such things as the value of wetlands as environmental buffers in indirect potable reuse projects and the potential for mitigation credits, although regulatory approval remains a challenge.

CONCLUSIONS

It is clear that wetlands, in particular peat accumulating wetlands, can act as a carbon sink, sequestering significant volumes of carbon. Restoration of 60% of the degraded peat wetlands could achieve a global net carbon sink by 2100 provided existing peat wetlands are protected. The use of wetlands in the wastewater treatment process, designed to achieve peat accumulation, can further support this. The concern being whether the emissions from the wetlands outweigh the ability for carbon sequestration.

In the end, some form of wastewater treatment is required, so the more fundamental question is whether wetlands can achieve tertiary treatment of municipal wastewater, and how the emissions footprint compares to a more conventional treatment process. Recent advances in constructed wetlands has demonstrated the ability to achieve tertiary treatment, and to reduce nitrogen and phosphorous to levels comparable with well-performing conventional process, as outlined in this paper. In addition, engineered wetlands, such as flood and drain wetlands, can reduce the footprint requirements (when compared to a traditional wetland) by up to 90 times.

The emerging empirical literature on GHG emissions from wetlands is providing a means by which assessment can begin of potential design and management adjustments that could be considered to reduce emissions. This paper shows that amendments, such as pH adjustment, and process mechanisms can be implemented to improve performance and reduce the emissions from wetlands.

Any net release of GHG needs to be considered within the context of the small fraction of total GHG release that wetlands represent, and the net environmental benefits and reductions in energy use when compared to conventional treatment. The major GHG impact of conventional wastewater treatment is energy demand rather than direct emissions, and natural treatment systems such as wetlands substitute land area for this energy input, noting that the footprint requirements are reducing as treatment advances are made.

Other benefits of wetlands include replacement of lost habitat, aesthetics, recreational facilities, and the role wetlands can play as environmental buffers in indirect potable reuse, which will likely be increasingly needed as global climate change and increasing water demands put increasing pressure on fresh water supplies.

Use of wetlands in the treatment process should be encouraged due to the significant potential to sequester carbon, and the low emissions footprint when compared to more conventional approaches, as well as the technology advancements which are seeing improvements in the effluent quality and reduction in required footprint. An additional benefit of treatment wetlands for wastewater treatment is that they set the community on a path towards restoring lost wetlands and perseverance of large tracts of green space near urban areas. The value of the land will increase over time and the net environmental benefits will become a community asset that improves the value of adjacent neighbourhoods relative to building a conventional tertiary treatment plant.

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