Constructed stormwater wetlands literature review, MUSIC uncertainty assessment and study of Melbourne Water's guidelines and procedures

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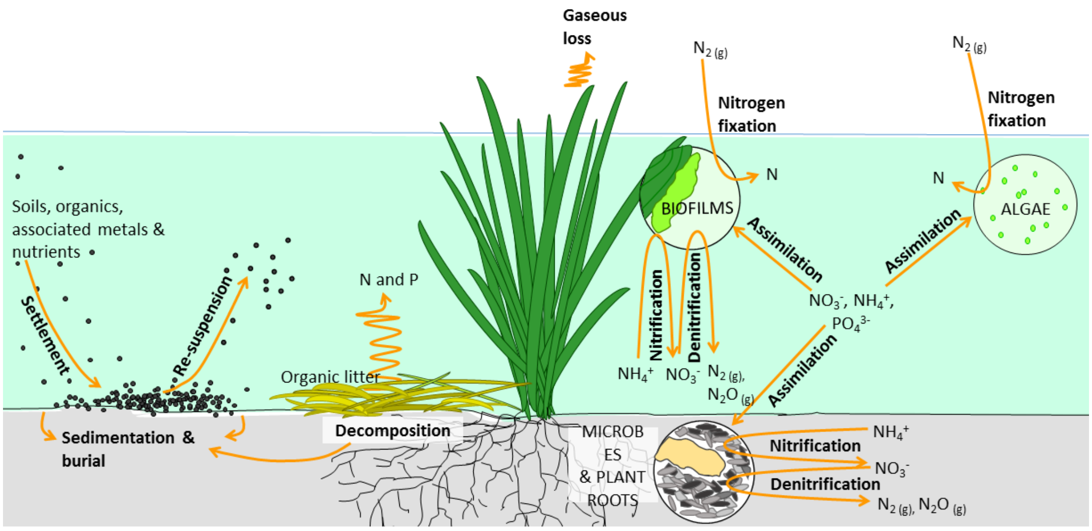

Waterway Ecosystem Research Group

The University of Melbourne

Melbourne Water

Technical Report

Melbourne Waterway Research-Practice Partnership

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Report to: Melbourne Water

Published: January 2015

Cite as: *Payne, E.G.I., Fletcher, T.D., Danger, A., Carew, D. (2015) Constructed stormwater wetlands literature review, MUSIC uncertainty assessment and study of Melbourne Water's guidelines and procedures. January 2015. Melbourne Waterway Research-Practice Partnership Technical Report.*

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Reviewed by: Melbourne Water

Cover figure: Processes acting on nutrients and sediment in a constructed treatment wetland (Figure 2 in the report)

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1 Executive Summary

Construction of wetlands is encouraged by regulatory bodies for the treatment of urban stormwater runoff, providing water authorities such as Melbourne Water with an expanding asset base to be monitored and maintained. While the potential benefits for water treatment are well established, practitioners are faced with the challenge of translating the theory into effective design, construction and long-term operation. The substantial degree of investment involved, both upfront and over the life of the wetland, demands a rigorous scientific foundation to ensure expected treatment performance is met (in the regulatory sense) and to prioritise the allocation of maintenance resources. The consequences of under performing wetlands resulting from poor design and/or maintenance will have long-term financial and environmental ramifications, and reduces confidence in the effectiveness of constructed wetlands for stormwater treatment.

To best inform design and management of constructed wetlands in the region, Melbourne Water need to identify; critical knowledge gaps relating to the drivers of wetland function, practical indications of performance, compatibility between modelled and actual wetland performance and develop a clear understanding of wetland life span. This includes understanding the performance implications and drivers of widespread vegetation loss, experienced across multiple Melbourne Water wetland assets.

This report includes the outcomes of a literature review, modelling study and direct comparison between Melbourne Water Guidelines and the literature. The literature review was commissioned to summarise recent advances in wetland function, design and operation to address uncertainty in Melbourne Water's current constructed wetlands program (Sections 3 - 6). This was further supported by a modelling study to assess uncertainty associated with parameter selection in MUSIC (Section 7). Knowledge from the literature review was applied to assess the justification behind Melbourne Water's constructed wetland design guidelines (2005, 2014) (Section 8.1). Finally, the report concludes with recommended changes to design and operation protocols, identification of critical knowledge gaps and potential future research investigations (Sections 8.2, 9 and 10).

Key outcomes of the literature review include a firm basis for the beneficial role of plants, support for inclusion of limited deep water zones in wetlands of sufficient size, the need for careful design of wetland hydraulics including consistent features across the width of the wetland, and the importance of a relatively shallow water regime, both during vegetation establishment and ongoing. The benefits of replicating natural wetland heterogeneity in terms of microtopography and a degree of water level variation are apparent, although optimal conditions in the specific context of constructed wetlands are poorly defined. Vegetation, hydraulics and bathymetry all exert strong influences on wetland performance, but multiple feedbacks between these parameters make constructed wetlands sensitive to poor design.

The modelling investigation concluded that, despite some uncertainty in the translation of modelled performance to actual performance, the (limited) evidence suggests that the performance predicted by MUSIC is likely to reasonably reflect the long-term performance of a well-functioning and well-monitored wetland. However, it is not known whether wetlands identified by Melbourne Water as being in 'poor condition' continue to effectively remove pollutants.

Many of the problematic issues raised in the literature review are addressed by Melbourne Water's guidelines, most particularly the recent revision (2014). When all requirements are met, constructed wetlands should comprise extensive shallow zones of emergent vegetation, limited deep water zones vegetated with submerged plants, suitable topsoil, healthy seedlings, a hydrological regime that does not excessively inundate plants and capacity to adjust water levels. However, further guidance is required from Melbourne Water to define measurable objectives, direct initial filling, manage substrate early in wetland life, support plant establishment, incorporate greater wetting and drying variability and conduct monitoring.

A suite of characteristics should be monitored to assess wetland performance, including water levels, plant cover, health, distribution and vegetation type and substrate characteristics. These can be measured using a variety of ground-based or aerial imagery techniques. However, little data is available in the literature to quantify relationships between these parameters and water quality, and wetland structure is dynamic and not necessarily indicative of function. Bioassays and other rapid analytical methods are also promising techniques for indicating the level of toxicity within systems, but their application in constructed treatment wetlands requires further research. Constructed wetlands may function effectively beyond a nominal 25 year lifespan, but data are scarce and net removal of sediments and nutrients ceases within some systems well before this time. There is a particular need for research to investigate practical issues relating to optimal maintenance, prolonging wetland lifespan and performance assessment using proxy indicators for water quality. These outcomes will inform a revised monitoring and maintenance program, and direct future wetland research by Melbourne Water.

In summary, recommended actions include:

- Definition of, and comparison against, measureable objectives specific to individual wetlands

- Provide further guidance/requirements surrounding water level manipulation during initial filling, plant establishment and across the operational life of the wetland
- Develop a monitoring program to assess wetland function, including aspects of the vegetation, substrate, flow paths and control structures. The program should include some long-term monitoring.
- Critically, development of a monitoring program requires direct anlysis of water quality in pilot studies to determine relationships between water treatment and wetland configuration, vegetation, substrate, age and other design or operational parameters
- Compare the hydroperiod from monitored water levels to the design, and against characteristics of vegetation surviving within each zone
- Investigate the benefit of (and potential to incorporate into design) greater heterogeneity, including microtopography and wetting and drying
- Investigate the influence of extended detention operation (i.e. the frequency, duration and extent of inundation) on vegetation cover, growth and survival
- Study key processes occurring within constructed wetlands to ascertain contaminant fate and optimise design and operation
- Conduct field trails in wetlands of varied ages to investigate how performance changes over time, wetland lifespan and indicators of end-of-life

2 Introduction and purpose

Constructed wetlands are widely utilised for treatment of wastewater streams globally. Melbourne Water manage a growing network to meet their regulatory obligations for stormwater treatment. However, to achieve continuous improvement, the theory and assumptions underpinning wetland design and operation require justification and revision using the current literature. In addition, Melbourne Water face multiple management challenges related to vegetation loss, contaminant accumulation and wetland lifespan, but there is a distinct lack of practical guidance readily available. Most critically, comparisons between systems are almost impossible due to an absence of methods to quantify wetland performance using cost-effective indicators. This review aims to investigate the crucial drivers of wetland function in the scientific literature and to identify the key knowledge gaps. To strengthen application to practical design and operation, and help identify recommended actions, the literature review is accompanied by a modelling study and a comparison with Melbourne Water guidelines.

2.1 Background

The use of constructed wetlands for the treatment of various effluent streams, including stormwater runoff, municipal and industrial wastewater, and agricultural effluent, has become widespread on a global basis across the past 30 years (Malaviya and Singh, 2011). In parallel, a wide body of scientific literature and practical design and operation principles have been developed. However, a disconnect between theory and practice can develop without regular review of research outcomes and constructed wetland guidelines. Given the growing investment in constructed wetlands, justified and sound principles underlying design and operation are essential for investment efficiency and performance outcomes.

Melbourne Water manages an extensive and expanding network of constructed wetlands to meet regulatory requirements for stormwater treatment. The design, operation and maintenance of these systems is based upon detailed guidelines, principles and modelling tools. However, the supporting theory dates back to literature published in the early 1990s and since this time there has been a lack of review by industry practitioners to incorporate more recent research findings into procedures. Equally, the available research stops short of many practical and long-term challenges faced by Melbourne Water. These include widespread vegetation loss and failure to establish across many constructed wetlands (Alluvium, 2010), and determining the best remediation options. Melbourne Water must also prioritise maintenance budgets, manage contaminant accumulation and the eventual need for expensive 're-setting' of wetland assets within limited resources. Further, the foundation of several key assumptions, such as a wetland lifespan of 20-25 years, has not been clearly demonstrated in practice. These issues loom larger as Melbourne Water's asset base expands and wetlands age.

Critically, assessing the efficiency of different wetland designs and management actions is restricted by an almost complete lack of performance quantification. Direct water quality monitoring is prohibitively expensive. Cost-effective means of measuring and comparing constructed wetland 'performance' are simply not available. As a result, Melbourne Water have limited knowledge of key performance drivers, including identifying the fundamental contaminant removal processes and their relationships with design parameters (such as the physical configuration, hydrology and vegetation). These knowledge gaps restrict Melbourne Water's ability to respond to management challenges across its constructed wetland network.

2.2 Objectives and structure

Melbourne Water commissioned a review of the primary and secondary scientific literature to understand i.) the factors driving wetland performance, ii.) actions to enhance or prolong wetland function, iii.) readily quantifiable indicators of wetland performance and iv.) identify key knowledge gaps for further research. The fundamental research questions are:

- What are the causes of vegetation loss in constructed stormwater wetlands and the ramifications on treatment efficiency?
- What are the key processes behind contaminant accumulation and release?
- What are the vital driving factors and influential conditions for these key processes?
- How does wetland performance and structure change over its lifespan?
- Are there opportunities to modify design or operational practices to enhance functionality?
- What are the indicators of wetland end-of-life and are there management actions or design features that may prolong system lifespan?

The literature review is accompanied by a MUSIC modelling study, which sought to determine uncertainty associated with the model parameters. An assessment of the theory supporting Melbourne Water's constructed wetlands guidelines (Melbourne Water, 2005, Melbourne Water, 2014) was also undertaken. Overall, the project aimed to document theoretical and practical knowledge and identify future management actions, research questions and potential future research studies.

The purpose of this study is to contribute to the process of continuous improvement in design and operation of constructed wetlands, not to question their overall effectiveness for the treatment of stormwater runoff, which has been repeatedly demonstrated (Birch et al., 2004, Greenway, 2010, Malaviya and Singh, 2011, Carleton et al., 2000, Collins et al., 2010). This study intends to provide background information and support for the revision of Melbourne Water's constructed wetland guidelines (Melbourne Water, 2014). It also accompanies recent studies investigating factors contributing to vegetation loss across the constructed wetland network (Webb et al., 2012, Dugdale and Ede, 2013). Together, these studies contribute towards Melbourne Water's legislated requirement; that is to discharge urban stormwater runoff to the receiving waterbody within the quality targets outlined in the relevant State Environmental Protection Policy (Victoria Government Gazette, 2003, Victoria Government Gazette, 2004, Victoria Government Gazette, 1997, Victoria Government Gazette, 1999).

The report is structured into the following broad sections:

- Introduction to wetland performance and objectives (Section 3)
- Factors driving wetland performance (Section 4 – Literature review)
- Indicators of performance (Section 5.3 – Literature review)
- Methods for monitoring performance (Section 5 – Literature review)
- Wetland lifespan and maintenance (Section 6 – Literature review)
- MUSIC modelling uncertainty assessment (Section 7 – MUSIC model study)
- Recommended management actions and assessment of Melbourne Water's guidelines (Section 8 – Comparison against literature review)
- Key questions and areas for future research (Section 9)
- Conclusions (Section 10)

Throughout the report key messages have been summarised in black text boxes at the start of sub-sections, and key recommendations are incorporated into green text boxes.

3 Wetland performance and its measurement

Current objectives for Melbourne Water's asset base meet legislative requirements but do not specify measurable objectives for individual wetland systems. Finding indicators of wetland performance is problematic given the expense of rigorous direct water quality monitoring. Proxy indicators, such as measures of wetland structure, are required but must be used with care and clear understanding of the relationship to wetland function.

For the purpose of this review, wetland performance is defined as the removal of suspended solids and nutrients from urban stormwater runoff.

3.1 Constructed systems and the challenges of the wetland environment

Constructed wetlands take advantage of the biogeochemical processing and water retention capacity of natural wetlands, but placed within a highly controlled (engineered) environment (Figure 1). This engineering of a natural system allows optimisation and control of wetland function (Malaviya and Singh, 2011, Wong, 1999).

Understanding the challenges associated with creation and operation of an engineered wetland system first requires an appreciation of the nature of natural wetlands. Natural wetlands are unique environments characterised by elements of both terrestrial and aquatic domains, high productivity and home to an enormous array of biological processes (McClain et al., 2003, Brix, 1997). They are also fragile, complex and typically develop over extremely long time frames (Brock and Casanova, 2000). The presence of water supports a wide diversity of processes including thriving floral and faunal communities and the accumulation of sediments, nutrients and carbon. However, water can also act as an environmental stressor. Water limits oxygen availability for respiration which cripples plant gas exchange and promotes anoxic conditions in the sediment, which can accumulate reduced compounds that are toxic to plants (e.g reactive oxygen species) (Wersal and Madsen, 2011, Colmer and Pedersen, 2008, Perata et al., 2011). Wetland organisms have evolved mechanisms to take advantage of the wetland environment while coping with, or escaping from, the stresses (Perata et al., 2011). However, the difference between flourishing vegetation and plant death can be finely balanced upon the hydrological regime as too much water over time will drown plants and severely restrict reproduction and establishment (Colmer and Pedersen, 2008, Raulings et al., 2010).

Natural wetlands and the biota therein, have evolved and adapted over long periods to suit local hydrological conditions. However, designers of constructed wetlands are challenged with engineering a suitable environment to support healthy vegetation and microbial communities yet also promote water retention (i.e. storing water for an extended period of hours or days following an inflow) to allow time for key processes to act (Wong, 1999). The constructed system must function almost immediately and is controlled by the hydrology of urban stormwater runoff, which differs substantially from pervious catchment runoff (Booth, 1991). Multiple engineered structures and design parameters impact the constructed wetland hydrology and function. These include inlet and outlet structures, bathymetry, shape treatment zone configuration, the relative volume of the permanent pool and wetland size relative to its catchment (Wong, 1999, Somes et al., 2000, Persson et al., 1999). Hence, ensuring an optimal water regime to support healthy stands of vegetation and balance water retention with treatment capacity, is inherently difficult.

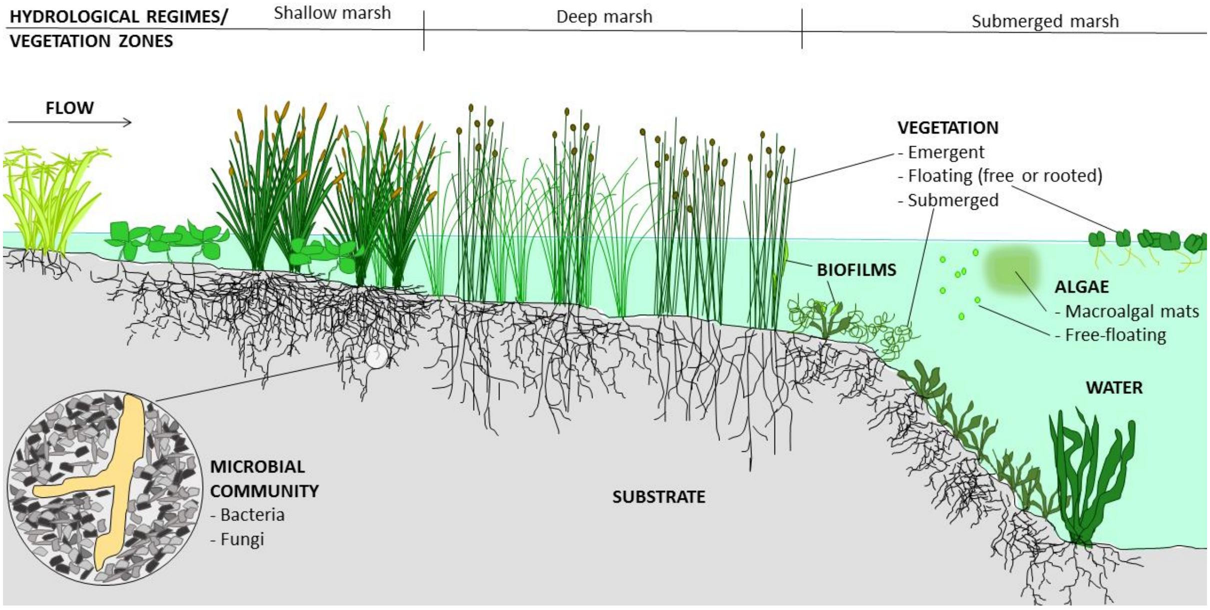


Figure 1 Key wetland functional components

3.2 Constructed wetland objectives

3.2.1 Primary objectives

The primary objectives for Melbourne Water's stormwater treatment wetlands are to satisfy their legislated requirements. Key requirements are outlined in the State Environment Protection Policies (SEPP) and the Urban Stormwater Best Practice Environmental Management Guidelines for Stormwater Treatment. The SEPPs outline the beneficial uses that must be maintained in waterways receiving runoff and water quality targets to protect these uses. The Best Practice Guidelines define water quality and hydrological targets for stormwater discharged from sites in terms of percentage pollutant load reductions and relative to the frequency and magnitude of pre-development flow peaks (Melbourne Water revised Guidelines, 2014). Specific targets are outlined in Table 1.

Table 1 Contaminant and flow reduction targets outlined in the SEPP

Contaminant	Reduction target
TSS	80%
TP	45%
TN	45%
Gross contaminants	70%
Flows	Flows < 1.5 year ARI similar to pre-development hydrology

These objectives respond to the recommendations that arose from the Port Phillip Bay Environmental Study (CSIRO, 1996). The study identified the critical need to reduce the nitrogen load input to Port Phillip Bay by 1,000 tonnes per year to protect its future health. It also recognised the importance of reducing nitrogen and sediment loads entering the bay as a result of urban stormwater runoff events.

3.2.2 Secondary objectives

While the primary objectives surround stormwater quality and discharge hydrology, the SEPPs also require Melbourne Water's constructed wetlands to provide additional values to the community and environment – including aesthetic, social, cultural, recreation and habitat values (Melbourne Water, 2014).

3.3 Are these objectives appropriate?

As a result of requirements outlined in the SEPPs, Melbourne Water's objectives focus on outcomes at the catchment outlet. This is beneficial to provide an overall target and protect the bay environment. However, it would also be valuable to define additional system-specific objectives which define outcomes for a given wetland. These could be targeted to protect an aspect of the local stream environment that may be particularly sensitive or valued, or for treatment of a contaminant of particular concern in that section of the waterway. Objectives could focus on specific aspects of the flow regime or water quality. Such localised objectives may already be defined during wetland inception or design, and may be detailed in conceptual or design reports, but they do not appear to be readily available or closely associated with ongoing system monitoring.

In addition to considering the target of the objectives, it is also important to also consider their ease of measurement. Measuring performance directly against Melbourne Water's primary objectives would require an extensive and ongoing monitoring program collecting water quality and flow data across the constructed wetland network. However, such a large-scale program is near impossible given to the prohibitive cost of water quality monitoring. Hence, secondary objectives are required that correlate with wetland performance for water treatment but are more readily quantified. This requires alternative indicators to water quality and robust relationships between these and wetland function. Such secondary objectives would provide benchmarks for monitoring programs, allow comparison between systems and assessment of performance. However, water quality monitoring on a pilot scale is first required to characterise the relationship to water treatment. The challenges associated with assessing wetland performance are discussed further in Section 5.

Hence, a process of clear definition and documentation of quantifiable objectives is required, including specific objectives for each system. Development of secondary objectives that can be readily monitored is critical but requires the development of robust relationships with water treatment, which itself will require water quality monitoring. Once objectives have been defined, a regular monitoring program should be undertaken to assess wetland performance against the objectives. The outcomes should feed into management of the wetland and design practices for new systems.

3.4 Wetland performance definition

For the purposes of this review wetland performance will be defined as the retention of contaminants in urban stormwater runoff, primarily suspended sediments (and associated contaminants), nitrogen and phosphorus. This definition is compatible with the scope of the literature review.

4 Factors affecting constructed wetland performance

4.1 Introduction

Constructed wetland performance is dictated by water retention (i.e. an extended storage period within the wetland) and the degree of contact between stormwater and the vegetation and sediments as it passes through the wetland. Hence, the flow of stormwater through the wetland and the environmental conditions encountered by each water parcel on the journey are crucial aspects.

Wetland performance is fundamentally dictated by the extent of contact between the stormwater and active biogeochemical processes occurring in the wetland. Multiple parameters in constructed wetland design influence flow through the system or the occurrence and magnitude of processes acting on contaminants. As a result, water retention is critical (Wong, 1999) and any factors that influence the vegetation, sediments, microbial population, algae or the flow hydraulics are also vital (Kadlec, 2010, Wetzel, 2001). Design can affect performance via the wetland layout, nature of the vegetation, presence of algae and biofilms, substrate characteristics, hydrological regime, hydraulics of flow through the system and an influence on water temperature (Kadlec, 2008). Each of these variables and the contributing design parameters are discussed in detail in the subsequent sections, but this is preceded by a brief overview of key wetland processes.

4.2 Dominant contaminant transformation/removal pathways

Understanding contaminant processing pathways is vital to optimise wetland design. An enormous range of chemical, physical and biological pathways occur in wetlands but certain processes can dominate contaminant retention (Figure 2). However, the dominant processes can differ between daily (inter-event and stormwater inflow events), seasonal and long-term (years to decades) temporal scales as the wetland ages (Johnston et al., 1990, Gottschall et al., 2007).

It is also vital to distinguish the timeframe of contaminant removal by distinguishing between permanent contaminant removal and processes that may only temporarily attenuate the contaminant or transform it into another form before export in the outflow (Malaviya and Singh, 2011) (Figure 2). Permanent sinks for contaminants include gases released to the atmosphere or transformation processes with entirely inert end products. Such processes include denitrification (conversion of nitrate to gaseous forms), volatilisation (conversion to a gaseous form, e.g. ammonia or volatile organic carbons) and some decomposition processes (such as phytodegradation or photodegradation of certain organic compounds) (Malaviya and Singh, 2011, Kadlec, 2011). Alternatively, contaminants may be stored within the wetland over time periods ranging from short to long-term associated with the sediments or incorporated into the biomass. If the timeframe of storage exceeds the constructed wetland lifespan, which may be the case for contaminants incorporated and buried in sediment layers or those stored in recalcitrant plant tissues, removal can be considered permanent (Kadlec, 2011), although appropriate disposal is required at some stage. In other cases, contaminants can be re-released back into the water column and exported downstream if sediment re-suspension occurs, or contaminants are released from the sediment or upon the death and decomposition of an organism (Kadlec et al., 2005, Fennessy et al., 2007). Drying out of the wetland can facilitate release of nutrients and dissolved organic compounds from organic matter (Kadlec, 2011). In addition, contaminants may only be partially broken down by transformation processes before export from the system, such as incomplete decomposition or nitrification without subsequent denitrification.

The principal contaminants in urban stormwater runoff and the key processes acting upon them have been summarised in the following sub-sections.

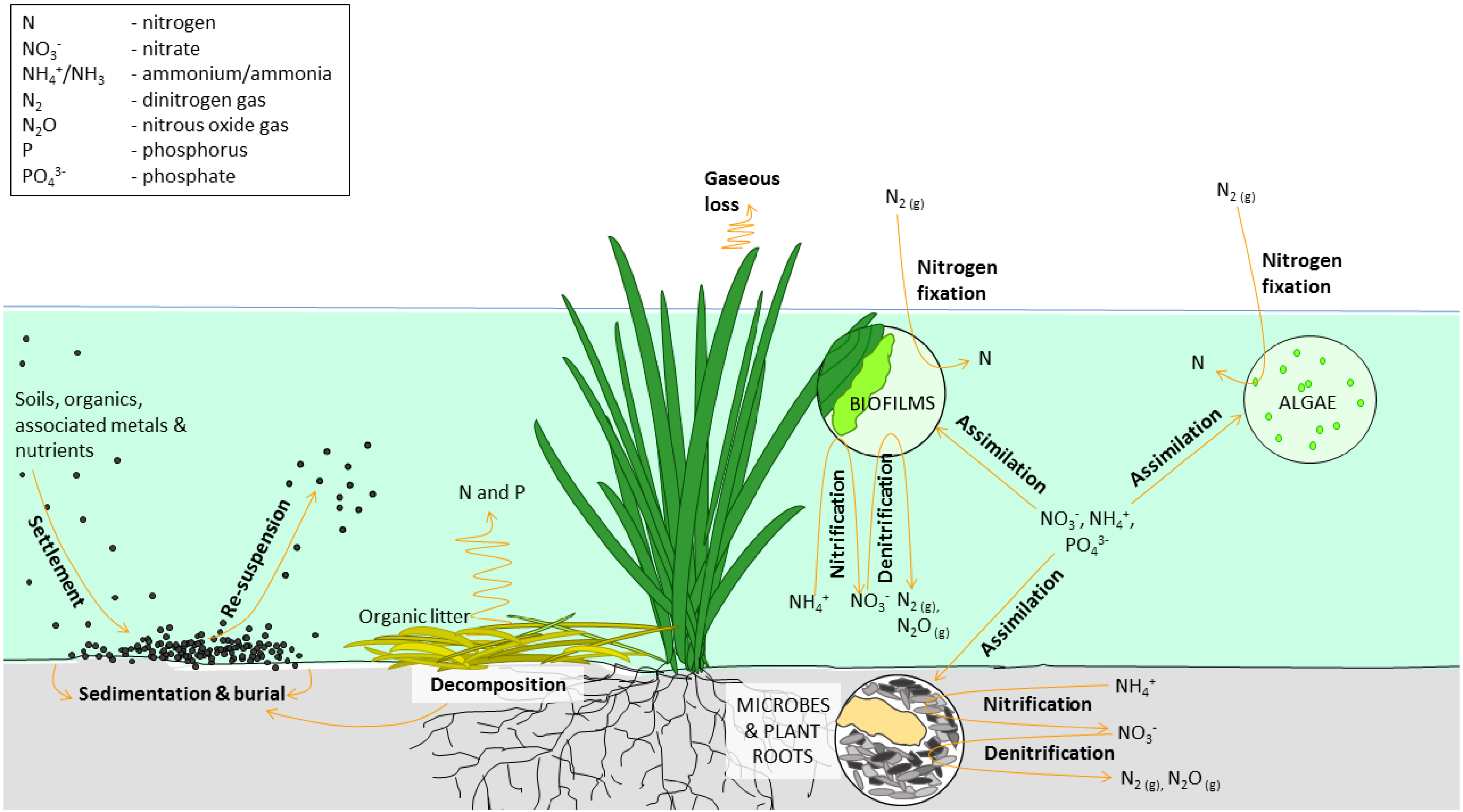


Figure 2 Processes acting on nutrients and sediment in a constructed treatment wetland

4.2.1 Key contaminants

Suspended particles

Sediment transported in stormwater runoff broadly incorporates all non-dissolved (i.e. particulate) matter, which includes soil minerals, a wide range of organic matter and multiple associated contaminants including heavy metals and phosphorus. Suspended particles reduce light penetration into the water column, accumulate in waterways and if organic, consume oxygen during decomposition (Malaviya and Singh, 2011).

Sediment and its associated contaminants may be removed via sedimentation, which is the process of settling and incorporation into the substrate (Figure 2). Processes such as decomposition of organic matter, may act within the sediment to transform different components. Conversely, sediment may be disturbed and resuspended by high velocity flow, wave action or bottom-dwelling organisms such as carp (Bodin, 2013).

Nitrogen

Nitrogen occurs in a wide range of chemical species in stormwater. On average, inorganic nitrogen (primarily ammonium NH_4^+ and nitrate NO_3^-) comprises almost 50% of total nitrogen in urban stormwater runoff in Melbourne, but only 30% on a global basis, and the remainder comprises diverse organic nitrogen compounds in both dissolved and particulate fractions (Taylor et al., 2005). It is important to note that nitrogen is an essential element in all organisms, it is only when available in excess that it causes environmental problems. When this report refers to nutrients as contaminants it is always in the context of excessive concentrations or loads. Anthropogenic sources generate an excess of the nutrient which leads to eutrophication, anoxia and loss of biodiversity in aquatic ecosystems (Vitousek et al., 1997, Galloway et al., 2008, CSIRO, 1996). Nitrogen is particularly critical in Melbourne’s creeks and waterways as algae in the receiving waterbody, Port Phillip Bay, is consistently limited by nitrogen (CSIRO, 1996).

Key nitrogen transformation and retention processes include (Figure 2):

- **Decomposition and mineralisation** incorporates a wide range of physiochemical processes acting to progressively breakdown organic compounds into simpler forms and eventually inorganic compounds. Decomposition is most effective under aerobic conditions (Kadlec and Wallace, 2008).
- **Biotic assimilation** - undertaken by plants or microbes and provides retention of inorganic nutrients in the tissues of living or dead organic matter. Decomposition processes will eventually re-release nutrients, but some proportion of recalcitrant compounds will be

incorporated into soil organic matter and stored over longer periods (Ingersoll and Baker, 1998, Kadlec et al., 2005). This pathway will be high during initial growth and its net contribution will decline once plant senescence occurs. If nutrient loading is high the plant storage can become saturated (Vymazal, 2007) or only form a small proportion of total removal (Gottschall et al., 2007). Nevertheless, the cycling of nutrients through the biomass attenuates the movement of nutrients, providing uptake and retention across the growing season (Gottschall et al., 2007, Kadlec et al., 2005), and provides the essential substrates for microbial processing (Tanner, 2001). Benefits to water quality can be increased if seasonal plant uptake matches seasonal changes in inflows and nutrient availability, patterns evident in natural wetlands (Tanner, 2001, Scholz and Lee, 2005).

- **Nitrification** is undertaken by select bacteria (nitrifiers), and converts NH_4^+ to NO_3^-, typically under aerobic conditions. It is an important step preceding permanent loss via denitrification, but if the two processes are not efficiently coupled, nitrogen losses can be exacerbated because NO_3^- is highly mobile within a system (whereas NH_4^+ can adsorb to soil and is typically the preferred nitrogen source for organisms).
- **Denitrification** is undertaken by denitrifying bacteria, primarily under anaerobic conditions, and transforms NO_3^- to N_2 gas or to a lesser extent N_2O. The process requires electrons from a carbon source. Coupled nitrification-denitrification requires close proximity of aerobic and anaerobic conditions, which may occur if pockets of oxygen or oxygen-depletion exist amongst contrasting conditions (e.g. if certain plant species release oxygen from their roots into an anaerobic substrate) (Faulwetter et al., 2009).
- **Nitrogen fixation**, undertaken by specialist microbes fixing atmospheric N_2 into NH_3, and can counteract the benefits of denitrification. However, the extent of this is poorly understood (Vymazal, 2007), but the likelihood of the process may increase moving downstream within a wetland as nitrogen is permanently removed via denitrification, which can produce a nitrogen deficiency relative to phosphorus availability (Scott et al., 2005).
- **Sorption/Adsorption** is a reversible process of attachment of a contaminant due to intermolecular attraction between charged clay surfaces or organic matter (e.g. ammonium) (Malaviya and Singh, 2011, Bodin, 2013). This process will depend upon various factors including pH and media characteristics, such as mineralogy and particle size (Akratos and Tsihrintzis, 2007). Sorption of nitrogen and phosphorus is increased by fine substrates, high in clay mineral content, or organic matter (e.g. peat) (Akratos and Tsihrintzis, 2007, Goonetilleke et al., 2005), although constructed wetland substrates typically have relatively low sorption capacity (Vymazal, 2007). In addition, the process will

decline over time as sorption sites become saturated, and will generally contribute less to water treatment in free water surface wetlands relative to vertical or horizontal-flow wetlands (Vymazal, 2007).

- **Ammonia volatilisation** is a physical process, highly dependent upon pH and favoured by alkaline conditions. NH_3 in an aqueous state is converted to gaseous NH_3 and lost to the atmosphere.
- **Sedimentation** may act on nitrogen in particulate form. In a similar manner to other components of suspended sediment, particles may settle and gradually become buried within the sediment. Conversely, re-suspension may occur to mobilise the contaminant or decomposition processes may release dissolved forms of nitrogen (either dissolved organic compounds, ammonia or nitrate) (Fennessy et al., 1994).

Many of these processes may act in a cycle to conserve nitrogen within the system, such as assimilation, eventually followed by decomposition, mineralisation and re-assimilation (Kadlec et al., 2005). Processes that recycle nitrogen may attenuate its movement downstream, but net removal from incoming stormwater requires long-term storage in the sediment, recalcitrant biomass or gaseous loss (primarily denitrification) (Vymazal, 2007).

Phosphorus

Similarly to nitrogen, phosphorus is a vital ingredient for all organisms. In excess it leads to excess quantities however, it can lead to excessive biological production of algal biomass that can rapidly deplete oxygen, subsequently killing aquatic fauna, and the change in nutrient availability can dramatically alter community structure (Kadlec, 2006). Phosphorus can undergo adsorption/desorption, precipitation/dissolution, biotic assimilation, decomposition, mineralisation and sedimentation (Vymazal, 2007), but unlike nitrogen, none of these lead to permanent removal via transformation into a gaseous phase (Kadlec, 2006) (Figure 2). Both adsorption and plant assimilation have limited capacity, but their contribution will depend in part upon loading (Vymazal, 2007, Bodin, 2013). When aerobic conditions prevail, phosphorus can form insoluble precipitates with iron, calcium or aluminium within the sediment (Malaviya and Singh, 2011, Scholz and Lee, 2005). Phosphorus can also adsorb to organic and clay materials, and sediments with iron or aluminium oxides (Scholz and Lee, 2005). Particulate-associated phosphorus may be incorporated into the sediment layers, acting as a long-term sink (Vymazal, 2007, Bodin, 2013). However, anaerobic conditions can cause the re-release of phosphorus (Malaviya and Singh, 2011). Reactions between phosphorus and components within the sediment are also influenced by pH, and is most available at neutral and slightly acidic pH (Scholz and Lee, 2005). Phosphorus can be exported downstream in the form of inorganic phosphate, attached to sediment or incorporated into organic materials (e.g. plant litter, fauna).

Organics

The decomposition of organic matter consumes oxygen, which can result in anoxic conditions toxic to aquatic fauna (Malaviya and Singh, 2011). Organic compounds may be in particulate or dissolved (defined as particles (< 0.45 μm diameter) forms. While the particulate fraction can be retained via sedimentation, dissolved organic compounds can be complex to remove. Processes include complexation, reactions with sunlight and transformation via a broad range of microbial respiration processes (Goonetilleke et al., 2005). These processes can decompose large organic compounds into simple organics (Faulwetter et al., 2009). While respiration is most effective when O_2 is available to accept electrons, processing continues under anaerobic conditions using alternate electron acceptors (Reddy and DeLaune, 2008).

Pathogens

Pathogens in stormwater can be diverse and cause a human health hazard when exposed via recreation activities or reuse of the water (Malaviya and Singh, 2011). Pathogens may be retained via adsorption, sedimentation of particulates, natural die-off, die-off from bactericidal compounds released by plants or microbes, burial, sunlight exposure, competition or consumption by other organisms (Stottmeister et al., 2003, Malaviya and Singh, 2011).

Heavy metals

Excessive concentrations of heavy metals have toxic effects on living organisms, and the impact is magnified for certain metals that can bioaccumulate within the food chain. Metals can undergo a range of reactions with minerals or organic matter in the soil, including adsorption, complexation, oxidation, precipitation and cation-exchange (Stottmeister et al., 2003, O'Sullivan et al., 2004). Long-term removal can be associated with the precipitation of insoluble substances (Malaviya and Singh, 2011). Metals can also be removed via plant uptake. Favourable conditions for retention include an alkaline pH, low redox potential, and available organic matter and sulphate (due to the formation of metal-sulphide compounds) (O'Sullivan et al., 2004, Faulwetter et al., 2009). Conversely, low pH typically increases metal availability in solution (O'Sullivan et al., 2004). Similarly to phosphorus, there are no permanent removal pathways and metals can accumulate near the surface of the substrate, eventually leading to saturation if the top layer is not removed (O'Sullivan et al., 2004). However, over time metals can become more securely bound into the substrate. For example, metals originally sorbed via cation exchange, a reversible process, over time become bound into more stable forms such as metal sulphides, facilitated by alkaline and reducing conditions and available carbon and sulphate (O'Sullivan et al., 2004).

4.2.2 Environmental conditions that influence processes

A range of redox conditions are required to support diverse processes in wetlands. In particular, nitrogen removal via nitrification and denitrification requires aerobic and anaerobic conditions respectively. The zone surrounding plant roots supports intense microbial activity, with carbon and oxygen released from the roots.

Some degree of wetting and drying also benefits diverse wetland processes, particularly nutrient processing, but beyond a point extreme drying leads to the release of nutrients and organic material upon re-wetting.

Wetland design is complicated by the diverse conditions required for a wide range of processes, which sometimes conflict. For example, aerobic conditions promote nitrification and decomposition, but anaerobic conditions facilitate denitrification and phosphorus retention (Kadlec and Wallace, 2008). Many biochemical processes are sensitive to redox potential (or oxygen availability) and the availability of carbon. Oxygen availability will shift with water availability, the extent of root oxygen release and in response to photosynthesis and respiration, which vary diurnally (Kadlec, 2010). Carbon availability is driven by the production of litter and root exudates from plants (Hinsinger et al., 2009). As a result, processing hot spots occur in the zone surrounding the plant root (the rhizosphere), where carbon availability is high and steep aerobic-anaerobic gradients can occur (McClain et al., 2003). These conditions are particularly favourable for coupled nitrification and denitrification (i.e. when the processes occur within close proximity) (Brune et al., 2000). The rhizosphere can support such intense and diverse microbial productivity that the bulk soil has been likened to a desert in comparison (Hinsinger et al., 2009).

Wetting and drying cycles can also benefit constructed wetland function (Wong, 1999). Reactants accumulate during unfavourable conditions, before shifting to favourable conditions (e.g. changed water availability, aerobic or anaerobic conditions establish) (McClain et al., 2003, Baldwin and Mitchell, 2000). Permanent flooding will slow decomposition and accumulate organic material, particularly if oxygen is scarce (Kadlec, 2011). However, anaerobic conditions can promote the release of phosphorus from sediments and dissolved organic matter from incomplete decomposition (Bai et al., 2005, Thullen et al., 2005, Baldwin and Mitchell, 2000). A cyclic wetting and drying regime will promote decomposition, and will benefit nutrient retention unless extreme drying occurs (Bai et al., 2005, Baldwin and Mitchell, 2000). Prolonged drying of sediments leads to the release of a nutrient pulse upon re-wetting (Baldwin and Mitchell, 2000, Birch, 1964), although the magnitude of this pulse may be reduced if sediments are frequently dried (Wilson and Baldwin, 2008).

The effects of sediment wetting and drying are complex and often contradictory, depending upon the frequency, severity and duration of wetting or drying and sediment characteristics, among other factors (Sommer, 2006, Baldwin and Mitchell, 2000). Hence, despite the potential to manipulate processes through wetting and drying (Wilson and Baldwin, 2008), identification of 'optimal' regimes is challenging. In general, 'partial' drying can be beneficial to nutrient retention whereas 'complete desiccation' is associated with nutrient release and greatly reduced capacity for nutrient processing (Baldwin and Mitchell, 2000), and permanent flooding is optimal for organic matter accumulation (Kadlec, 2011, Sommer, 2006). Sommer (Sommer, 2006) suggested understanding the sediment characteristics was key to predicting the implications of wetting and drying, and vital for wetland management.

4.3 The contribution of vegetation to performance

Vegetation is a vital component of constructed treatment wetlands, forming the basis of multiple functions. Wetland performance, particularly for nutrient removal, is substantially poorer in the absence, or only minimal presence, of plants.

Vegetation is essential in constructed wetlands. Plants form the foundation of key wetland functions and interact with processes via numerous pathways (Kadlec, 2008, Brix, 1997, Kadlec et al., 2005). As a result, vegetation is obligatory for effective water treatment (Brix, 1994a, Greenway, 2010). In the absence of plants, or following widespread vegetation loss, the capacity for constructed wetlands to remove pollutants from the water column is significantly reduced (Zhu and Sikora, 1995, Kadlec, 2008, Yu et al., 2012, Scholz and Lee, 2005). In particular, effective nutrient processing, especially nitrogen, is heavily dependent upon vegetation (Kadlec, 2008, Zhu and Sikora, 1995, Yu et al., 2012, Ruiz-Rueda et al., 2009).

The essential role of plants in wetlands is evident from their multiple interactions with wetland processes (Section 4.3.1) (Kadlec, 2008). Not only is a high quantity of vegetation vital, but its appropriate configuration within the wetland is also crucial to flow dynamics and treatment efficiency (Section 4.3.4) (Jenkins and Greenway, 2005, Persson et al., 1999). Further subtleties are introduced by considering the influence of vegetation type and diversity on water treatment (Section 4.3.2 and 4.3.3 respectively) (Mitsch et al., 2005a, Kadlec, 2011). Without suitable vegetation, wetlands look and function like ponds, and lose many of the water treatment benefits afforded by a vegetated wetland environment (Wong et al., 1999). This eventuality occurs in constructed wetlands suffering from vegetation loss or its poor establishment, a problem that is common to many regulated semi-aquatic environments (Section 4.3.5) (Gawne and Scholz, 2006, Vanderbosch and Galatowitsch, 2011).

4.3.1 Role of plants

Plants influence contaminant removal processes in multiple ways including altered hydrology, contaminant uptake, altered sediment transport, interactions with the microbial community, and shading and sheltering of the water column. Most plant functions have positive implications for contaminant removal but a few can be detrimental and are generally associated with dense vegetation (blocking out light and oxygen from the water).

The function of constructed wetlands is modified by plants through various pathways. On a broad level, plants interact with flow dynamics, microbial processing, substrate characteristics and directly influence concentrations of pollutants in the water column. Well-established mechanisms of plant contribution to water treatment include (Figure 3). :

- **Assimilation/uptake of nutrients (and limited heavy metals) –** assimilated compounds are typically incorporated into the synthesis of organic tissues, which are stored in the plant before re-release upon plant or tissue death. The contaminants may be incorporated into the sediments or re-released upon decomposition (Brix, 1994a, Kadlec et al., 2005).
- Providing **surface area for biofilm attachment** – in turn biofilms contribute significantly to wetland microbial processes (Section 4.5.1) (Weisner and Thiere, 2010).
- **Supporting a diverse microbial community in the rhizosphere** – this hot-spot of processing is fuelled by plant-derived carbon compounds (Hinsinger et al., 2009, Faulwetter et al., 2009).
- **Aeration of the sediments via root oxygen release** – which provides aerobic micro-pockets in the sediment, facilitating coupled nitrification and denitrification and more effective decomposition of organic matter (Brix, 1994a, Kadlec, 2008)
- **Alteration of flow dynamics** – Plants act to shift flow paths, filter particulates and reduce flow velocities. This has implications on particulate removal and transport (including settlement and re-suspension), retention time, development of short-circuit flow paths and hydraulic efficiency (Jenkins and Greenway, 2005, Wood, 1995) (Section 4.7).
- **Oxygenation of the water column via photosynthesis** – which is also associated positively with plant growth and demand for nutrients (Colmer and Pedersen, 2008)

- **Stabilisation of the substrate** and **reduction in erosion** – occurs from the action of roots binding the substrate (Guardo et al., 1995).
- **Volume reduction via evapotranspiration (ET)** – which reduces excess volume contributed from impervious surfaces and increases retention time for the remaining water, but can increase pollutant concentrations. ET can promote infiltration into the substrate for processing, and in particular can transport nitrate and ammonium towards the microbial-rich rhizosphere for transformation or assimilation (Shelef et al., 2013, Bodin, 2013, Heers, 2006).
- **Shading of the water column** – shading reduces water temperature and algal growth (Brock and Casanova, 2000).
- **Sheltering from the wind** – this can reduce re-suspension of sediments and minimise wind-driven short-circuit flow paths (Brix, 1994a).
- **Plant litter accumulation develops into a humus layer** - and this layer in the substrate facilitates multiple treatment processes (Hammer, 1992)
- **Contributing a seasonal pattern to wetland function** – with periods of growth, senescence and dormancy. This generates periods of high nutrient uptake and later, release (Kadlec et al., 2005).
- **Providing habitat, food or shelter for fauna** – including fish, water birds and invertebrates, although this role is typically considered from an ecological perspective and less frequently discussed I the context of water quality treatment (Brock and Casanova, 2000, Greenway, 2010, Woods et al., 2004).

Other roles of plants are less frequently mentioned in the literature, which may indicate they plant less critical roles in water treatment within constructed wetlands. These include the:

- **Release of allelochemicals by plant roots** - these can comprise a diverse array of compounds including antimicrobial agents but are typically very species-specific (Brix, 1997, Faulwetter et al., 2009).
- **Provision of odour biofiltering from plant litter accumulated at the base of wetlands** - particularly odours released by anaerobic decomposition processes (Wood, 1995, Environment Heritage and Local Government Ireland, 2010)
- **Provision of water resistant compounds in plant litter** – these help to seal the wetland base and retain water

Not all mechanisms of plant influence directly benefit water quality treatment objectives (Faulwetter et al., 2009). However, they are part of ecosystem functioning and overall do not outweigh the positive benefits of plants to water treatment (Kadlec, 2008, Brix, 1994a). These include:

- **An increase in short-circuiting and reduced wetland effectiveness** caused by poor planting configurations and high litter accumulation (e.g. dense fringing vegetation, broken bands across the wetland) (Wood, 1995, Heers, 2006, Keefe et al., 2010).
- **The release of nutrients, sediment and organic compounds from plant litter**; largely in a seasonal pulse during senescence (Kadlec et al., 2005).
- **Reduced direct re-oxygenation of the water column** due to lowered wind velocity from the effect of plant sheltering and reduced photosynthesis from plant shading (see below). However, it must be noted that, conversely, plant photosynthesis acts to oxygenate the water column during the day.
- **A reduction in biological process rates and reduced algal photosynthesis and biofilm growth** from plant shading, particularly dense vegetation stands (Brix, 1997, McCormick et al., 1997)
- Dense vegetation also **decreases the volume of the wetland**, reducing capacity and retention time at low flows (but once flow exceeds a critical point vegetation conversely acts to slow flows and increase retention time) (Jadhav and Buchberger, 1995, Bodin, 2013).
- Facilitating the **transport and release of greenhouse gases** produced in the sediment, including CO_2, CH_4 and N_2O via internal plant gas conduit pathways (Wetzel, 2001, Shelef et al., 2013)
- **Plants synthesise allelochemicals which can impede certain processes,** although this is highly species-and process-specific. Nitrification inhibition is one widely reported allelopathic effect (Faulwetter et al., 2009).

Distinguishing between the significance of these multiple roles is challenging, particularly under field conditions, hence it remains largely unknown and in some cases debated (Shelef et al., 2013). For example, Brix (Brix, 1997) suggested the key contribution by plants is via physical processes including substrate stabilisation, filtration, provision of surface area for biofilms and reducing wind and flow velocities. There is certainly a wide body of literature studying the interaction of vegetation with flow hydraulics and treatment efficiency (Jenkins and Greenway, 2005, Wong et al., 1999, Bodin et al., 2012, Leonard et al., 2006). Others focus on the inextricable interactions between plants and microbial functioning via carbon provision and oxygenation of the rhizosphere (Faulwetter et al., 2009, Brune et al., 2000).

Alongside these indirect influences, the direct contribution of plant uptake to water treatment has been questioned, as rooted macrophytes likely source most nutrients from the substrate (Brix, 1997), a significant proportion of nutrients are returned upon litter decomposition (Kadlec et al., 2005), and some authors suggest uptake is minimal relative to nutrient loading (apart from during the initial plant growth stage) (Gottschall et al., 2007, Vymazal, 2007, Stottmeister et al., 2003, Bodin, 2013). However, plants also synthesise recalcitrant compounds and some nutrients are translocated below ground before senescence, leading to some long-term nutrient storage in the sediment (Kadlec, 2006, Vymazal, 2007). Additionally, some authors suggest plant uptake is significant under low loading (such as stormwater), to the extent that plant harvesting could provide a feasible permanent removal pathway (Vymazal, 2007, Brix, 1997, Bodin, 2013).

Nonetheless, the functions of vegetation will change over time as the plant grows and ages (Bodin, 2013). Plants are clearly important to microbial processing in the substrate and in attached biofilms, the hydraulic efficiency of a wetland, cycling of nutrients, substrate stabilisation and incorporation of pollutants into sediment deposits. Irrespective of the mechanisms, a healthy vegetation community is obligatory if constructed wetlands are to achieve optimum water treatment (Kadlec, 2008, Zhu and Sikora, 1995, Yu et al., 2012).

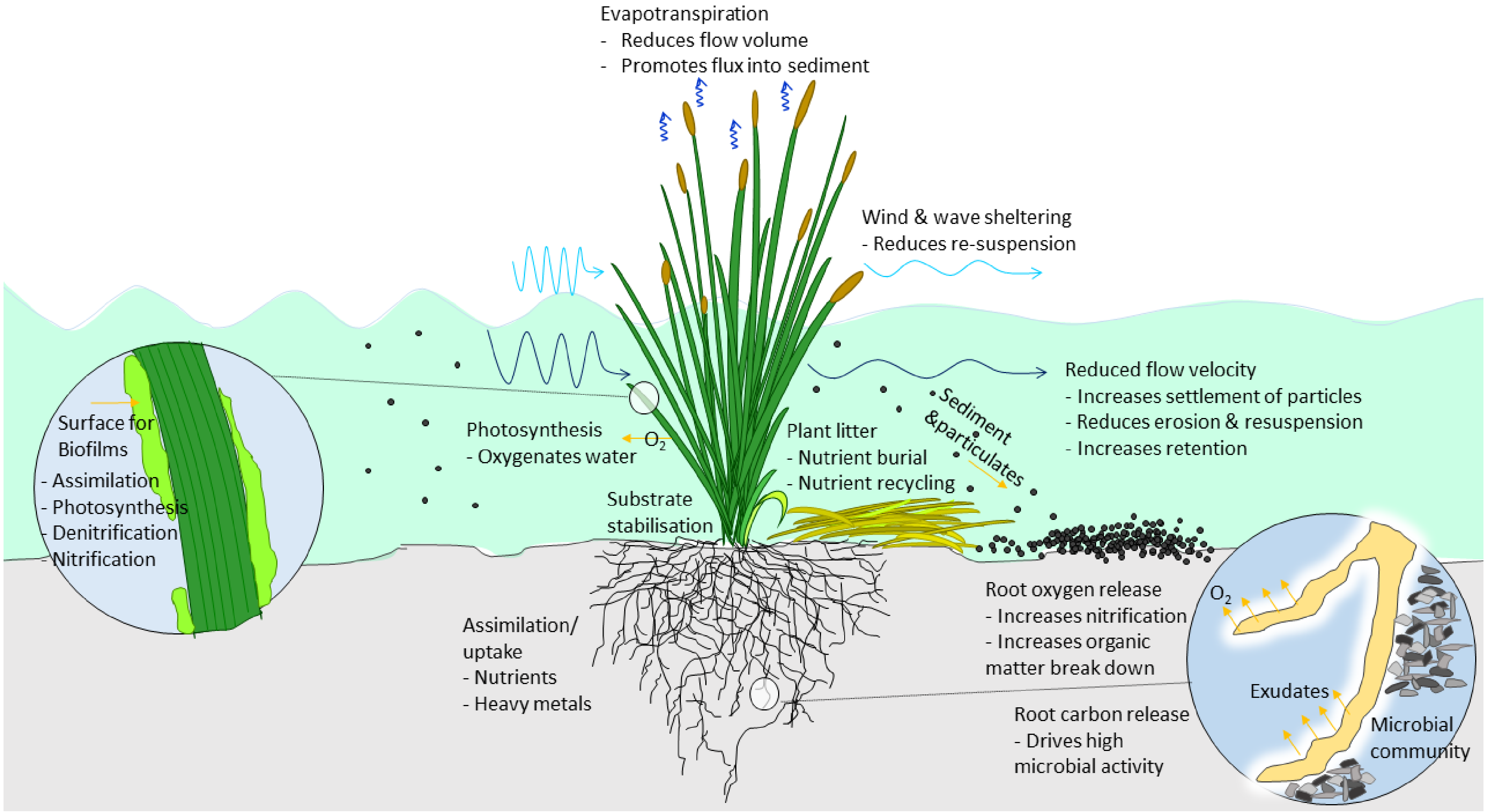


Figure 3 Key roles played by plants in water treatment processing

4.3.2 Vegetation type

Emergent vegetation is generally more productive and effective for nutrient removal than submerged vegetation, however, submerged vegetation can benefit denitrification due to the nature of their carbon flux. Floating vegetation uptakes nutrients directly from the water column, but interrupts other processes by restricting light and oxygen reaching the water below. A degree of mixed vegetation types is beneficial to multiple processes and hydraulic efficiency, but should incorporate a majority of emergent vegetation.

Differences in performance between plant species tend to be smaller than those between differing plant forms. High productivity, biomass and an extensive root system benefits nutrient removal. However, in general local species are used in constructed wetlands on the assumption that they are adapted to local climatic and soil conditions.

While dense stands of invasive or dominant species are not desirable, these species are commonly well suited to the disturbed environment of constructed wetlands and are not necessarily detrimental to performance *per se*.

Emergent, Submerged, Free-floating

Constructed wetlands typically include areas of both emergent and submerged vegetation, but a large proportion typically comprises shallow zones of emergent vegetation. It is clear from the proceeding discussion that the presence of vegetation is essential for optimal wetland performance. However, the contribution of submerged vegetation in deeper sections of the wetland to water treatment is less certain.

Emergent vegetation has been reported to remove nitrogen more effectively than submerged plants (Bastviken et al., 2009, Kadlec, 2011, Weisner and Thiere, 2010) and have greater tolerance for anaerobic conditions in the sediment (Carpenter and Lodge 1986). Given the challenges of plant growth in the aquatic environment (Colmer and Pedersen, 2008), it is not surprising that emergent species reportedly have higher productivity and biomass (Greenway and Woolley, 1999), as their protruding foliage can effectively exchange gases with the atmosphere. High plant productivity is associated with greater nutrient uptake (Brix, 1997, Greenway and Woolley, 1999), carbon provision to drive microbial processing (Eviner and Chapin, 2003) and provides a greater surface area to support biofilms and slow flow to facilitate settlement (Weisner and Thiere, 2010). However, overly dense emergent plants can be associated with poor hydraulic efficiency and a lack of oxygen (Mietto, 2010, Thullen et al., 2002).

In contrast to emergent vegetation, submerged plant species generally provide labile carbon at a more consistent turnover rate to fuel denitrification (Bastviken et al., 2005). However, it is debatable whether this translates to higher denitrification than emergent macrophytes, with some reports associating elevated denitrification potential with submerged vegetation (Bastviken et al., 2005), but others suggest a mixture of vegetation types is best for provision of a variety of labile and recalcitrant carbon (Bachand and Horne, 1999, Weisner et al., 1994, Kadlec, 2011). In terms of nutrient assimilation, free-floating vegetation provide the benefit of direct nutrient uptake from the water column, while rooted macrophytes primarily assimilate nutrients from the sediments (Vymazal, 2007). On the contrary, floating plants restrict light from reaching the water column below (Wetlands International, 2003).

Overall, many studies recommend incorporating a range of vegetation types as mixed vegetation benefits processing and hydraulic efficiency (Bachand and Horne, 1999, Mietto, 2010, Weisner et al., 1994). Despite the need for some variation, the majority of constructed wetlands should comprise of emergent macrophytes zones (Kadlec, 2011). The high biomass and productivity of emergent macrophytes is most likely to provide a high degree of interaction with wetland functions, from supporting microbes in the substrate, providing surfaces for biofilm attachment, direct nutrient uptake and interaction with flow dynamics.

Performance can also be sensitive to species, particularly for nitrogen removal (Bodin, 2013, Zhu and Sikora, 1995), although the influence of plant species on processing is generally less critical than plant type (Kadlec, 2011). Effective species for nutrient removal may be highly productive (Heers, 2006), with high above and below ground biomass (including a rhizome), extensive root systems with high surface area for nutrient uptake and capacity to oxygenate the rhizosphere (Shutes, 2001). However, it is challenging to obtain the necessary data about species physiology to inform plant species selection. As a result, local wetland species are commonly selected for constructed wetlands given their adaptation to local soils, climate and the surrounding flora and fauna (Melbourne Water, 2005, Kadlec, 2006). Such species will also enhance local biodiversity. Interestingly, several international studies report more effective water treatment from wetlands that were left unplanted to naturally develop a vegetation community (Kadlec, 2011, Mietto, 2010).

Weeds

Weeds are typically observed in constructed wetlands, particularly in the formative phase (Ahn and Dee, 2011, Kadlec, 2011). Zedler and Kercher (2004) concluded that wetland environments are particularly susceptible to invasion by weeds, likely because they are naturally a concentration point for sediment, nutrients and water from the catchment. Weedy species are not universally more productive, competitive or fast-growing than native species (Daehler, 2003, Zedler and Kercher, 2004). Instead, their characteristics tend to be advantageous under conditions of high resource availability and frequent disturbance (Zedler and Kercher, 2004, Green and Galatowitsch, 2001). As a result, stormwater inflows to constructed wetlands tend to exacerbate the advantage of invasive species (Zedler and Kercher, 2004, Miller and Zedler, 2003). Under different growth conditions, such as low nutrient or water availability, or natural disturbance regimes, native species tend to have superior growth and survival (Daehler, 2003).

Weedy species tend to be opportunistic and highly flexible in their growth strategy as nutrient availability changes or pockets of bare soil are created by disturbance (Zedler and Kercher, 2004, Green and Galatowitsch, 2001, Galatowitsch et al., 1999). Invasive species are typically efficient at shifting biomass allocation in response to changing conditions, often have low density tissues (efficient for growth if nutrients are abundant and herbivory low), and some species have the advantage of a prolonged growing season (Green and Galatowitsch, 2001, Daehler, 2003).

Weeds will alter the structure and function of the wetland (Zedler and Kercher, 2004). However the relative benefits and disadvantages in the context of water treatment in constructed wetlands are debated. Weedy species may be detrimental to wetland treatment function if present in dense stands by blocking light and oxygen, and if they out-compete desired species (Green and Galatowitsch, 2001, Wetlands International, 2003). This is a strong possibility given their tendency to form monotypes (Zedler and Kercher, 2004). However, weedy species are clearly effective at growth within the environmental conditions found in constructed stormwater wetlands (Zedler and Kercher, 2004, Kercher and Zedler, 2004, Miller and Zedler, 2003), and in some cases are tolerated by managers (Zedler and Kercher, 2004). There are reports that invasive species may lead to more effective nitrogen removal from water, but this is debateable and research is limited (Zedler and Kercher, 2004). Either way, invasive species do pose risks for maintaining heterogeneity within the system over time (Green and Galatowitsch, 2001).

Currently, there appears too little research to advocate definitively for allowing invasive species to dominate in constructed wetlands. However, their presence should be expected and if kept under control, they are likely to benefit, or at least not disadvantage, water treatment in constructed stormwater treatment wetlands.

Understanding the ecological niches of invasive species can help to manage their dominance. Given the dependence of invasive species on environmental conditions for their competitive advantage, changing the nutrient, soil disturbance or hydrological regimes can help control weeds (Daehler, 2003). While responses will be species-specific, native species may be promoted by slowly varying wetting and drying (as opposed to static water levels (Deegan et al., 2012) or rapid fluctuations (Miller and Zedler, 2003)), providing a range of ecological niches within the wetland (Daehler, 2003) and minimising areas of bare soil (Zedler and Kercher, 2004), such as preventing scour and ensuring rapid establishment of high plant cover.

4.3.3 Diversity

Despite being a commonly used indicator for 'healthy' systems, in constructed treatment wetlands high diversity does not necessarily produce better water treatment. For example, a system dominated by emergent macrophytes removed nitrogen more effectively than a wetland with higher vegetation diversity. In addition, diversity will invariably change over the wetland lifespan and while over the short-term it may correlate with function, the relationship may not exist over the long-term.

Biotic diversity is commonly used to infer the health of ecosystems. Nevertheless, is diversity beneficial in the context of constructed wetland function? Diversity is generally considered to provide greater resilience against stresses or disturbances, such as flooding, drought or herbivory (Mitsch et al., 2005a). Undoubtedly diversity is associated with a suite of environmental benefits, so managers ubiquitously seek to enhance it (Weisner and Thiere, 2010). Diversity may provide a broad range of substrates for microbial processing and heterogeneity in physical plant structure to provide flow resistance, biofilm surfaces and maximise light penetration of the canopy (Wetzel, 2001, Stern et al., 2001, Bachand and Horne, 1999). It can also provide flexibility for the system to 'self-design' (Mitsch and Wilson, 1996).

However, the relationship between diversity and system function can be ambiguous. Diversity may inevitably decline over time (Aronson and Galatowitsch, 2008, Weisner and Thiere, 2010). Some authors have reported higher performance from diverse systems relative to monocultures (Bachand and Horne, 1999). Nevertheless, Mitsch et al. (2005) observed a relationship between diversity and productivity in early wetland life, but over longer time periods (7 years in this case) there was no clear relationship. Similarly, a study by Weisner and Thiere (2010) found floral diversity between different wetlands (one planted and the other with self-established vegetation) changed over time as the systems developed. Interestingly, diversity declined when one vegetation type dominated (in this case tall emergent vegetation), but nitrogen treatment efficiency was highest in this system. In addition, floral diversity is not necessarily correlated with faunal diversity; Weisner and Thiere (2010) noted that low diversity macrophyte cover can harbour a diverse macroinvertebrate community. Hence, increasing wetland diversity does not necessarily produce increased function

(National Research Council, 2001, Drinkard et al., 2011), and may even lead to poorer water treatment (Mitsch et al., 2005a).

4.3.4 Cover, density and arrangement

The placement of vegetation within a wetland has strong implications on hydraulic efficiency and the extent of water treatment. Configuration is particularly important to good design – bands of vegetation are essential while patchy vegetation cover and fringing vegetation are detrimental to function, reducing the effective wetland area, retention time and contaminant removal. In addition, very dense vegetation can have negative implications on nutrient removal.

The influence of vegetation will change seasonally with plant senescence leading to reduced hydraulic efficiency and increasing export of nutrients and organic matter downstream. It will also vary across the flow spectrum.

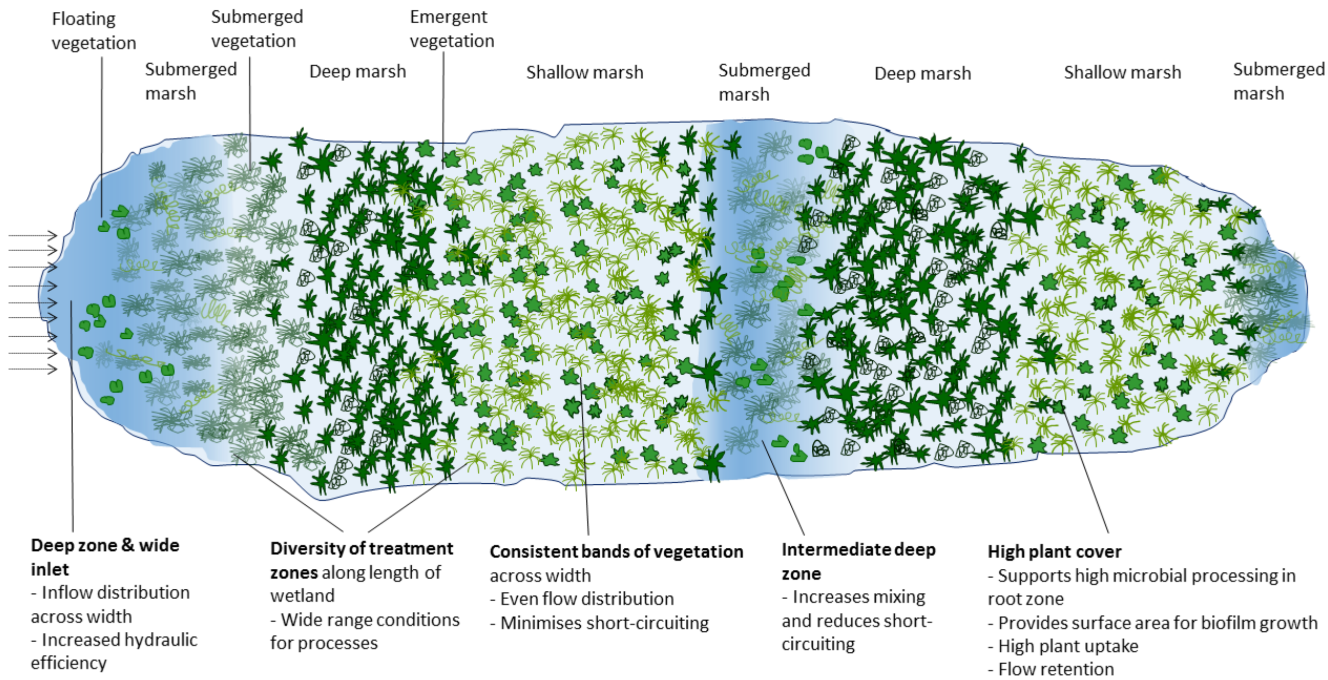
The benefits of vegetation to water treatment processing are well acknowledged and stem from the multiple modes of interaction with wetland processes acknowledged (Section 4.3.1) (Kadlec, 2008, Zhu and Sikora, 1995, Yu et al., 2012, Brix, 1994a). Wetland hydraulics are strongly influenced by plant cover, density and arrangement. In addition, the amount and placement of vegetation can indicate the degree of nutrient assimilation, microbial processing in the substrate (driven by plant carbon) and alteration of flow paths and velocity. Due to its ease of measurement, plant cover is commonly used to infer wetland efficiency (Kadlec, 2006, Cole, 2002) (further discussed in Section 5.6). However, the influence of plants on water treatment varies with flow conditions, seasons, vegetation configurations or density, among other factors.

Plants provide an effective hydraulic barrier (Greenway, 2010), and this can be either beneficial or detrimental depending upon plant configuration and density across the wetland. When planted in consistent bands perpendicular to the flow, all flow paths experience similar interaction with the vegetation (Figure 4). As a result, the entire flow is slowed, plug flow conditions are promoted, retention time increases and the physical and biochemical processes associated with plants (Section 4.3.1) can act more or less uniformly across the wetland width (Kadlec, 2008, Persson et al., 1999, Jenkins and Greenway, 2005). Conversely, fringing vegetation or a clumped or discontinuous arrangement reduces hydraulic efficiency and a majority of the flow will be channelled between the vegetation in short-circuit channels, experiencing a greatly reduced retention time and little, if any, interaction with vegetated zones and the associated processing (Lightbody et al., 2008, Jenkins and Greenway, 2005). In particular, dense fringing vegetation will reduce the effective wetland area and negate the benefit of a well-designed wetland shape (Jenkins and Greenway, 2005). In addition, very sparse vegetation can exacerbate erosion and reduce sedimentation to a greater extent than non-vegetated systems (Nepf, 1999) (Figure 4).

Even with consistent distribution, after a certain point, very dense vegetation may not provide a net benefit to water treatment due to an increased likelihood of short-circuit development (Fennessy et al., 1994, Mietto, 2010), the possibility of anaerobic conditions in the underlying water (Thullen et al., 2002) and reduced effective volume of the wetland (Jadhav and Buchberger, 1995, Keefe et al., 2010). Hydraulic efficiency can be improved by using a mixture of vegetation types in the place of dense emergent monocultures (Mietto, 2010, Stern et al., 2001, Kjellin et al., 2007). The influence of vegetation density on water treatment is further discussed in Section 5.6.2.

In addition, the positive influences of vegetation on water treatment are highest during the growing season, but are reversed to some extent following senescence. During the autumn and winter plant litter accumulates within the water column and can increase short-circuiting, although the extent of this will vary with species physiology (Wood, 1995, Heers, 2006, Keefe et al., 2010). Decomposition can release dissolved and particulate organic matter and nutrients downstream (Kadlec et al., 2005).

Finally, the interaction between vegetation and wetland functions will vary across the flow spectrum. For example, vegetation benefits sediment retention at low flows, but under high hydraulic loading its influence is reportedly not significant (Brueske and Barrett, 1994). Vegetation will also dictate hydraulics differentially across flows – at very low flows the topography controls flow paths and retention, but vegetation dominates at higher flows, an influence that will vary with species physical characteristics (Ahn and Dee, 2011, Choi and Harvey, 2014). Interestingly, at low flows vegetation biomass reduces the effective wetland area leading to a lower retention time, but at higher flows the vegetation impedes the flow and increases retention (Jadhav and Buchberger, 1995, Keefe et al., 2010).



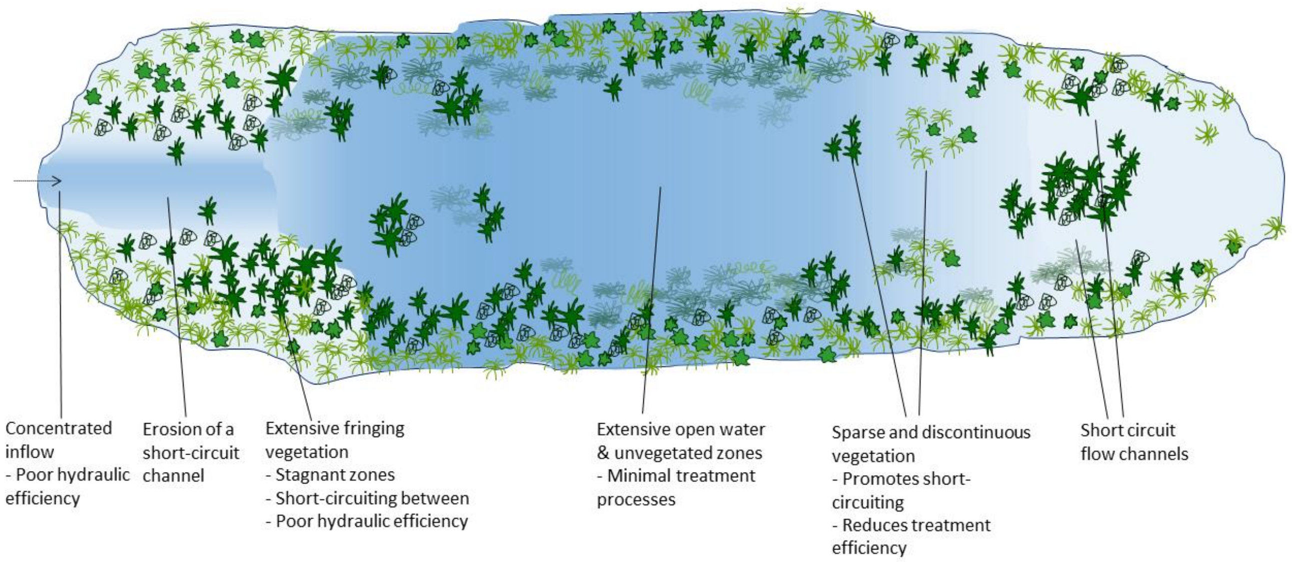


Figure 4 Example wetland configurations with high vegetation cover and flow distribution (above) and low vegetation cover and poor flow distribution (below)

4.3.5 *Loss of vegetation*

The loss of vegetation is a common problem across many regulated water bodies. The cause/s can be difficult to identify but excessive inundation depth and a loss of wetting and drying fluctuations are commonly key contributors. Other causes include erosion, grazing and animal damage, insufficient topsoil and inappropriate planting season. In addition, a separate hydrological regime is crucial during seedling establishment. While plant species responses to hydrological changes are complex and often contradictory, excessive inundation (in terms of both depth and time) and a lack of drawdown impede plant growth and reproduction.

A failure to establish and loss of vegetation in the initial years of wetland life is a widespread problem across Melbourne Water's constructed wetland network (Alluvium, 2010). A comprehensive investigation was recently undertaken on the relationship between vegetation decline and constructed wetland hydrological regime (Dugdale and Ede, 2013). Dugdale (Dugdale and Ede, 2013) noted that water levels in the ephemeral zone are deeper and sustained for longer periods relative to natural wetlands. The resultant flooding impedes plant growth below maintainable levels and prevents reproduction, typically producing death 2-5 years following construction. Despite variation in species tolerances, increasing water depth negatively influences emergent plant growth (Dugdale and Ede, 2013, Webb et al., 2012, Sorrell et al., 2012, Vretare et al., 2001, Perata et al., 2011, Hoban et al., 2006).

Similar problems with plant loss and altered community structure are being faced in other environments, where natural systems (or their components within an engineered system) are subjected to altered flow regime, largely driven by water management including dams, water extraction, flow releases, weirs, levees and diversions. Such systems include ephemeral deflation lakes (Gawne and Scholz, 2006), floodplains (Kingsford, 2000), riparian vegetation, floodplain wetlands (Greet et al., 2011) and lakeshores (Vanderbosch and Galatowitsch, 2011). While identifying the cause/s of vegetation loss can be elusive, even in research wetlands (Kadlec, 2008, Vanderbosch and Galatowitsch, 2011), in many of these environments the modification or complete loss of the natural wetting and drying regime is commonly identified as a critical factor. This may produce a wetter or drier system, or shift seasonal flow timing, but as a result wetland function, productivity and diversity are reduced (Gawne and Scholz, 2006). Constructed wetland design does incorporate some wetting and drying when extended detention volume (with slow drawdown following an inflow) is incorporated into design (Wong, 1999). Despite this, the dynamics of wetting and drying are not addressed specifically in design guidelines for Melbourne Water's constructed wetlands, and mentioned primarily in the context of mosquito control (Melbourne Water, 2005).

In addition to an altered water regime, other potential causes for vegetation decline include erosion, scouring or lack of topsoil, shallow underlying compacted clay layer, toxicity in the sediments, nutrient availability, herbivory/grazing (by water birds, musk rats, carp etc.), water bird roosting, wave action and inappropriate planting season (Brock and Casanova, 2000, Greenway et al., 2007, Kadlec, 2008, Mitsch et al., 2005b, Greet et al., 2011, Adcock et al., 1995, Vanderbosch and Galatowitsch, 2011). However, the investigation by Dugdale (2013) concluded inappropriate hydrology was the primary cause of poor plant cover, or at least that rectification of this would improve plant cover significantly. The relationship between hydrology and vegetation health is further discussed in Section 4.7.3.

A separate consideration is the hydrological regime during plant establishment, which is particularly crucial to achieving successful vegetation (Webb et al., 2012, Dugdale and Ede, 2013). Seedlings will be highly vulnerable to flooding until they attain certain height and biomass. The duration of the necessary establishment period is debated – some suggest a 3 year time period (Brock and Casanova, 2000), or at least an entire growing season (Hammer, 1992) while in practice only a 6-8 week period may be applied (Dugdale and Ede, 2013).

Thus, protocols must be developed and adhered to during plant establishment. This should include a significant reduction in water levels, as recommended by Dugdale (2013) and only gradual increase over time as seedling height increases. Careful management is required up to the point when plants reach their mature height and emerge sufficiently at the normal water level (this will require much longer than 6-8 weeks). Following this, ongoing monitoring of water levels, plant height and health, and appropriate adjustment to water levels, are required to ensure a suitable hydrological regime (including engagement and drawdown of extended detention) within each vegetated zone. Dugdale (2013) recommended drawdown events were required to allow vegetation recovery and expansion in Melbourne Water's wetlands. In addition, it is recommended that wetland design should include a table of species for each zone with their preferred hydroperiod and the hydroperiod provided by the wetland. This will ensure appropriate species selection and hydrological design, and provide a basis for comparison of monitoring data once the system is constructed.

4.4 Physical layout

Wetland size and location within the catchment will influence the capacity to treat incoming flows and contaminants, which in turn are governed by climate and catchment characteristics (Wong and

Somes, 1995, Johnston et al., 1990). The physical configuration of a wetland – including physical dimensions (e.g. length and width), bathymetry, microtopography and layout of treatment zones – dictates flow dynamics and the subsequent extent of contact between stormwater and treatment features in the wetland (Kadlec, 2008, Greenway and Polson, 2007). Parameters influenced by configuration include flow paths, velocity, retention time, vegetation presence and species/ vegetation type, and in turn these affect wetland performance. Therefore, layout strongly influences the degree of contaminant retention and/or processing, via the time and environmental conditions available for various processes to occur.

4.4.1 Size

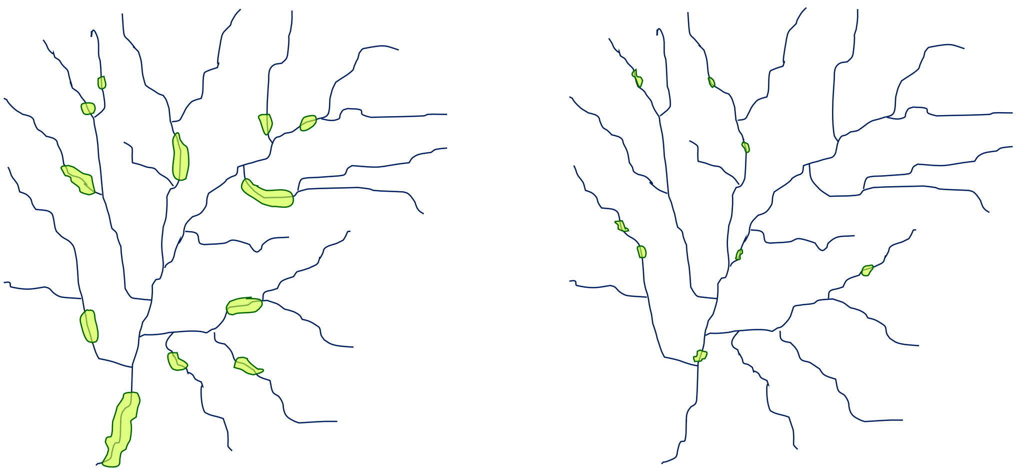
The size of a wetland relative to the inflows of stormwater runoff it receives is crucial to effective long-term water treatment. The necessary size varies but depends upon the target contaminant/s, sediment characteristics, the impervious surface area, rainfall characteristics (e.g. intensity and distribution) and wetland location within the broader catchment.

A well-established principle in constructed wetland design is the relationship between wetland size relative to the expected inflows - the balance is fundamental to stormwater treatment capacity (Wong and Somes, 1995). Wetlands that provide a relatively high volume can treat a high proportion of incoming flows, while a high surface area and prolonged retention are also essential for effective water treatment performance (Carleton et al., 2000, Kadlec, 2010, Wong, 1999, Somes et al., 2000). However, providing treatment capacity and retention in shallow zones are conflicting objectives, and designers must balance these objectives, all within the available land area (Wong and Somes, 1995). Critically, additional retention should not be gained by an increase in deep zones as these areas do not contribute significantly to water treatment (Kadlec, 2011) and do not support emergent plant growth (Vretare et al., 2001, Webb et al., 2012). Key characteristics of wetlands are a high surface area:volume ratio and variable water levels, which distinguish their water treatment performance from ponds (Wong et al., 1999). Adding to the conflict, retention time increases significantly as the volume of the permanent pool of water increases (relative to the airspace volume reserved for extended detention), but this does not outweigh the treatment benefits of shallow vegetated zones and variable wetting and drying provided by limiting the permanent pool volume (Wong et al., 1999). Regardless of these challenges, the consequences of undersizing include; inadequate retention time (Birch et al., 2004), high flow volume bypassing the wetland, frequent engagement of the extended detention depth leading to excessive inundation (which stresses plant health) (Hoban et al., 2006, Dugdale and Ede, 2013), and the risk of high flow damage (Storm Consulting, 2013, EDAW, 2008); all severely compromising wetland performance.

Catchment area, inflow hydrology (i.e. rainfall frequency and intensity, fraction of catchment area imperviousness, flow routing), particle size and pollutant chemistry are all key considerations for wetland sizing (Wong, 1999, Carleton et al., 2000, Wong and Somes, 1995). The characteristics of individual storm events and antecedent storm events, will dictate the proportion of runoff that can pass through the wetland, and the proportion that can be retained within the wetland between inflow events, allowing prolonged retention and the opportunity for inter-event processing (Carleton et al., 2000, Wong, 1999). Inflow characteristics, including the pollutant load, will vary significantly between storm events and as a result, stochastic data generation and analysis should be employed in design (Wong and Somes, 1995). The required retention period is influenced by the settling rate of the target particle size (CRC, 2012). Catchments that contribute a high proportion of fine sediments (e.g. clay colloids) and wetlands designed to remove nutrients and metals (if not in dissolved form, these contaminants are primarily attached to the finer sediment fraction) will require a longer retention period (Wong et al., 2006, Wong et al., 1999). In addition, biogeochemical processes such as nitrification and denitrification, will typically require prolonged retention (Kadlec, 2010, Carleton et al., 2000). Due to the sensitivity of process rates to temperature, the seasonality of flows will also influence wetland size requirements with longer retention required in the cold months (Kadlec, 2011).

The sufficiency of wetland size is often expressed as a ratio between wetland area and catchment area. There are conflicting reports of the wetland surface area necessary for water treatment. Some suggest 1-2% (Malaviya and Singh, 2011), up to 5% (Malaviya and Singh, 2011, Koskiaho, 2003), 3-7 % (Mitsch and Gosselink, 2000a) or even 10% (Cohen and Brown, 2007). This conflict may arise from different sizing requirements depending upon rainfall distribution, duration and intensity, target pollutants, sediment characteristics, location, impervious fraction or land use within the catchment (Tilley and Brown, 1998, Cohen and Brown, 2007, Wong et al., 1999). Even treatment objectives have a bearing, as concentration reductions are particularly sensitive to wetland area, but if pollutant mass (or load) reductions are the aim, these are primarily dictated by the hydraulic loading (Kadlec, 2011).

Hence, wetland size should be considered strategically in terms of the inflow hydrology, the target pollutants, treatment objectives, its location in the catchment, and other wetlands in the same catchment (Cohen and Brown, 2007, Kadlec, 2011) (see Section 4.4.2) (Figure 5). Importantly, Koskiaho (2003) noted that if wetlands are small relative to their catchment area, careful design for hydraulic efficiency is even more critical (further discussed in Sections 4.4.3, 4.3.4 and 4.7).



a.) b.)

Figure 5 Wetland sizing and location – example of a.) wetland sizing for upstream catchment and strategic catchment planning for water treatment, b.) undersized wetlands and no apparent catchment-wide strategy for wetland location

4.4.2 Location

Wetland positioning should be considered strategically as it will influence performance outcomes. Nutrient and sediment removal tends to be more effective when smaller wetlands are utilised in upstream reaches of the catchment, while hydrological benefits can be achieved by larger downstream systems. Wetland location should reflect the treatment objectives, in terms of the targeted contaminants, hydrological parameters and the stream or waterbody.

The location of constructed treatment wetlands is restricted to available land, however suitable. Yet their location within the landscape, including relative to other wetlands or alterative stormwater treatment measures within the catchment, are important considerations. Careful planning can yield substantially better outcomes for flow and pollutant retention (Cohen and Brown, 2007).

Location influences performance via the underlying soil characteristics and its development as a wetland substrate (Mitsch, 1992, Español et al., 2013). Constructed wetlands are frequently located on former terrestrial soils (Mitsch, 1992). However, hydric soils have been reported to develop from terrestrial soils within several years following wetland construction (Mitsch et al., 2005b). Inflow hydrology and contaminant loading is also governed by wetland position (Johnston et al., 1990), and inflows from the catchment should be sufficient to support a functioning wetland (Malaviya and Singh, 2011). In addition, wetland location relative to other stormwater treatment devices influences cumulative water treatment (Johnston et al., 1990, Koch et al., 2013).

Studies report differential contaminant removal depending upon location within the catchment. Cohen et al. (Cohen and Brown, 2007) suggest wetlands located high up in the catchment are efficient for sediment removal; intermediately placed wetlands target phosphorus and large wetlands near the catchment outlet best achieve flow reduction and attenuation goals. Mitsch et al. (1992) suggest small wetlands upstream remove a higher proportion of nutrients, but larger downstream systems have capacity to remove a higher load. Interestingly, a recent study showed that preserving healthy upland streams (which means treatment is necessary high in the catchment) was likely better for nitrogen removal than building downstream treatment systems (Kaushal and Belt, 2012). These findings all highlight the importance of considering wetland location in light of the specific treatment objectives – that is, the target contaminants, hydrological characteristics and reach or waterbody of interest.

4.4.3 Length:width ratio

A high l:w ratio optimises use of the wetland area leading to more effective water treatment. While l:w is an important parameter in design, its importance may be exceeded by vegetation placement. A high cover and density of fringing vegetation can lead to poor hydraulic efficiency even in wetlands with high l:w ratio.

An ideal wetland shape promotes flow dispersal and plug flow (i.e. hydraulic efficiency), high surface area to volume ratio between the stormwater, sediment and vegetation, and a diversity of treatment zones (Persson et al., 1999, Van Dam et al., 1998). Length:width ratio is a critical determinant of hydraulic efficiency (Thackston et al., 1987), influencing flow distribution (Persson et al., 1999) and the extent of stagnant zones (Jenkins and Greenway, 2005). It is worth noting the statement by Kadlec (2011) that hydraulic efficiency is most relevant when the treatment objectives seek concentration reductions, whereas load reductions are less sensitive. This is because mass removal is optimised by high hydraulic loading, whereas concentration reductions instead benefit from prolonged retention (i.e. low hydraulic loading and high hydraulic efficiency).

Hydraulic efficiency is maximised by a high length:width (l:w) ratio, as long as suitably low flow velocities are maintained. Conversely, square designs or those with l:w ratio < 4 are hydraulically poor from a water treatment perspective due to an inefficient use of the wetland area (Persson, 2000, Persson and Wittgren, 2003, Koskiaho, 2003). Low l:w ratio leads to a greater proportion of the wetland incorporated into zones of recirculation with low velocity, limited mixing and very minimal contribution to water treatment (Jenkins and Greenway, 2005). These recirculation zones invariably develop on either side of the inflow but stretch across an increasing length of the wetland as the l:w ratio decreases (Jenkins and Greenway, 2005). However, despite the benefits of a high l:w ratio, the ratio should not exceed 10 as beyond this point additional treatment benefits are minimal (Thackston et al., 1987).

Baffles greatly enhance hydraulic efficiency within a small land area and help to reduce the detrimental effect of wind on uniform flow conditions (Thackston et al., 1987, Persson, 2000, Koskiaho, 2003). Compartmentalisation, through the use of wetland cells, can also enhance hydraulic efficiency (Kadlec, 2011). However, this requires compromise as these structures reduce the wetland area, so beyond a point the use of too many baffles or compartments is detrimental to performance (Kadlec, 2011).

Further, poor vegetation positioning can completely undermine the designed l:w ratio. Dense fringing vegetation acts to significantly reduce hydraulic efficiency as its area and density increase, irrespective of the l:w ratio (Jenkins and Greenway, 2005). Critically, even a thin and long wetland will be associated with poor hydraulic efficiency if dense fringing vegetation is abundant, but consistent vegetation bands across the wetland have no detrimental effect (Jenkins and Greenway, 2005).

4.4.4 *Depth and bathymetry*

Water depth influences water treatment processes in wetlands, not least by dictating vegetation type, cover, density and species. Plant growth is extremely sensitive to water depth. Water depth also influences the occurrence of contaminant removal processes. Despite a majority of biogeochemical processing occurring within shallow vegetated zones, limited deep water zones can augment treatment via some denitrification in the sediments, sedimentation and algal uptake in the water column augment and improved hydraulic efficiency.

Water depth is a function of bathymetry, hydrology, sediment accumulation and design of the inlet and outlet structures (e.g. invert level, capacity, stage-discharge relationship). In turn, water depth has strong influence on vegetation cover and composition, flow velocity and contact between the water and wetland components (Webb et al., 2012, Greenway and Polson, 2007, Raulings et al., 2010, Thackston et al., 1987). Depth may change over time with sedimentation, causing a slow transition of submerged zones towards ephemeral and terrestrial zones (Zedler and Callaway, 1999).

Critically, water depth in vegetated zones must be carefully designed and managed. The influence of water depth and variation in water levels on plant survival is discussed in Section 4.7.3. The tolerance of macrophytes to inundation depth varies significantly across species, and growth parameters decline non-linearly as water depths increase (Sorrell et al., 2012, Webb et al., 2012). Species responses also differ with the duration and timing of flooding (Webb et al., 2012, Greet et al., 2011). Although the thresholds differ between species, deep water acts to severely reduce plant growth and rhizome development, which is critical to survival across the non-growing season (Vanderbosch and Galatowitsch, 2011, Webb et al., 2012, Perata et al., 2011). Hence, deep water may be sparsely vegetated (if at all). Further discussion of appropriate water depths for wetland zones is given in Section 4.7.3.

Relative to shallow vegetated zones, deep water provides inferior nutrient removal, with reduced nitrification and oxidation of organic matter, and lower adsorption of pesticides relative to shallow areas (Koch et al., 2013, Knight, 1992, Lange et al., 2011, Kadlec, 2011). However, the low velocity does assist sediment removal and helps to break short circuits (Lightbody et al., 2008), non-vegetated sediments do provide some microbial processes including denitrification (Bourgues and Hart, 2007) and algal uptake will also contribute to nutrient removal (Kadlec, 2011) (see Section 4.5.2).

Despite these benefits, deep zones should not be incorporated simply to improve retention capacity, as treatment primarily occurs from retention in shallow zones (Kadlec, 2011).

4.4.5 Microtopography

Heterogeneous microtopography is a characteristic of natural wetlands and can benefit nitrogen removal and plant survival. However, careful design is required to ensure hydraulic efficiency is not reduced by enhanced short circuiting or highly variable flow paths, and plant cover is not severely reduced. While the beneficial influence of microtopography is also restricted to lower flows, most studies recommend its incorporation into constructed wetland design. However, microtopography can develop naturally over time and until reliable design principles are developed, at present its inclusion may not be essential to good design.

Topographic variation across small vertical scales is known as microtopography and. This heterogeneity in the surface encompasses the roughness and relief of the surface at a scale relevant to individual plants (Moser et al., 2007) and includes features such as hummocks and depressions, ridges and sloughs (Choi and Harvey, 2014). While natural wetlands are characterised by heterogeneous topography, constructed wetlands tend to have little if any microtopographic variation as a result of exacting construction techniques and careful design of levels (Moser et al., 2007).

However, if present, microtopography can positively influence water treatment processes, particularly nitrogen removal. Firstly, it provides variation in water depth, and possibly also wetting and drying, across short distances (Figure 6) (Raulings et al., 2010). This fine-scale variation is highly significant to individual plants and can allow a diversity of species, with differing flood tolerances, to closely co-exist (Raulings et al., 2010, Ahn and Dee, 2011, Moser et al., 2007). Microtopography can also help to provide a buffer against the effects of prolonged flooding in a wetland, benefitting plant survival and regeneration (Raulings et al., 2011, Cramer and Hobbs, 2002). In addition, the water column may be oxygenated by a tortuous flow path, higher wind disturbance and algal photosynthesis (the latter two may occur if plant cover is not continuous across the microtopography) (Keefe et al., 2010, Thullen et al., 2002). Higher dissolved oxygen and a greater exposed soil surface area amongst microtopography has been shown to increase nitrification and ammonia removal, relative to dense and continuous vegetation cover (Thullen et al., 2002). Coupled nitrification-denitrification has also been demonstrated to increase as both aerobic and anaerobic conditions may exist across the height of the microtopographic feature, and plant litter can accumulate as a carbon source in the low elevation points (Wolf et al., 2011b).

However, the influence of microtopography on hydraulic efficiency is debated. While there are reports the meandering paths around complex topographic features can increase retention time (Choi and Harvey, 2014), others suggest it can instead increase short circuiting (Persson et al., 1999). As a result, some authors advocate for relatively flat bathymetry that instead facilitates wide and uniform flow distribution (Persson et al., 1999, Thackston et al., 1987). Velocity heterogeneity (i.e. non-uniform flow) can also increase due to microtopography, which leads to a wider range of retention times (Keefe et al., 2010). However, in a wetland reconfigured to include hummocks the velocity heterogeneity was accompanied by an overall longer retention due to a higher effective volume (Keefe et al., 2010). A further negative impact of microtopography can be reduced area available for emergent macrophyte cover and increased open water, if the depth gradient exceeds the shallow depths most favourable to plant growth. While some interspersion with open water is reported to enhance ammonia removal (Thullen et al., 2002), the reduction in plant cover can be significant - Keefe et al. (2010) reported more than a 90 % reduction in vegetation cover in some zones of a wetland reconfigured to include hummocks. However, depending upon the hydrology, microtopography can increase plant cover if it facilitates plant establishment and survival (Moser et al., 2007). The overall effect of microtopography will depend upon the method used to create it (e.g.

disking or excavation), including the resulting elevation difference, tortuosity and roughness (Moser et al., 2007), and the hydrological regime.

The influence of microtopography is restricted to the low flow end of the hydrological spectrum (Ahn and Dee, 2011). Ahn and Dee (2011) found under wet conditions (in this case rainfall at or above average), the invert levels controlling hydraulics and inflow hydrology exert greater influence on plant growth than microtopography. In terms of influence on flow hydraulics, vegetation will provide greater resistance to flow once water exceeds the height of the topography (Choi and Harvey, 2014). In addition, the benefits of microtopography may only be apparent during initial plant establishment (Ahn and Dee, 2011).

The likelihood of reduced hydraulic efficiency or poor plant cover resulting from microtopography could be controlled using good design principles. Most studies support the inclusion of some form of surface roughness or heterogeneity into wetland design. Microtopography can be created using an excavator or with agricultural disks or it will naturally establish and change over time, either increasing or decreasing depending upon hydrology, sedimentation and species (particularly those with clumped growth forms) (Ahn and Dee, 2011, Wolf et al., 2011b, Keefe et al., 2010). However, given its potential downsides and effectiveness only at low flows, incorporation of microtopography may not yet be essential to good constructed wetland design. This may change in the future as design guidelines further develop to define the optimal characteristics of microtopography.

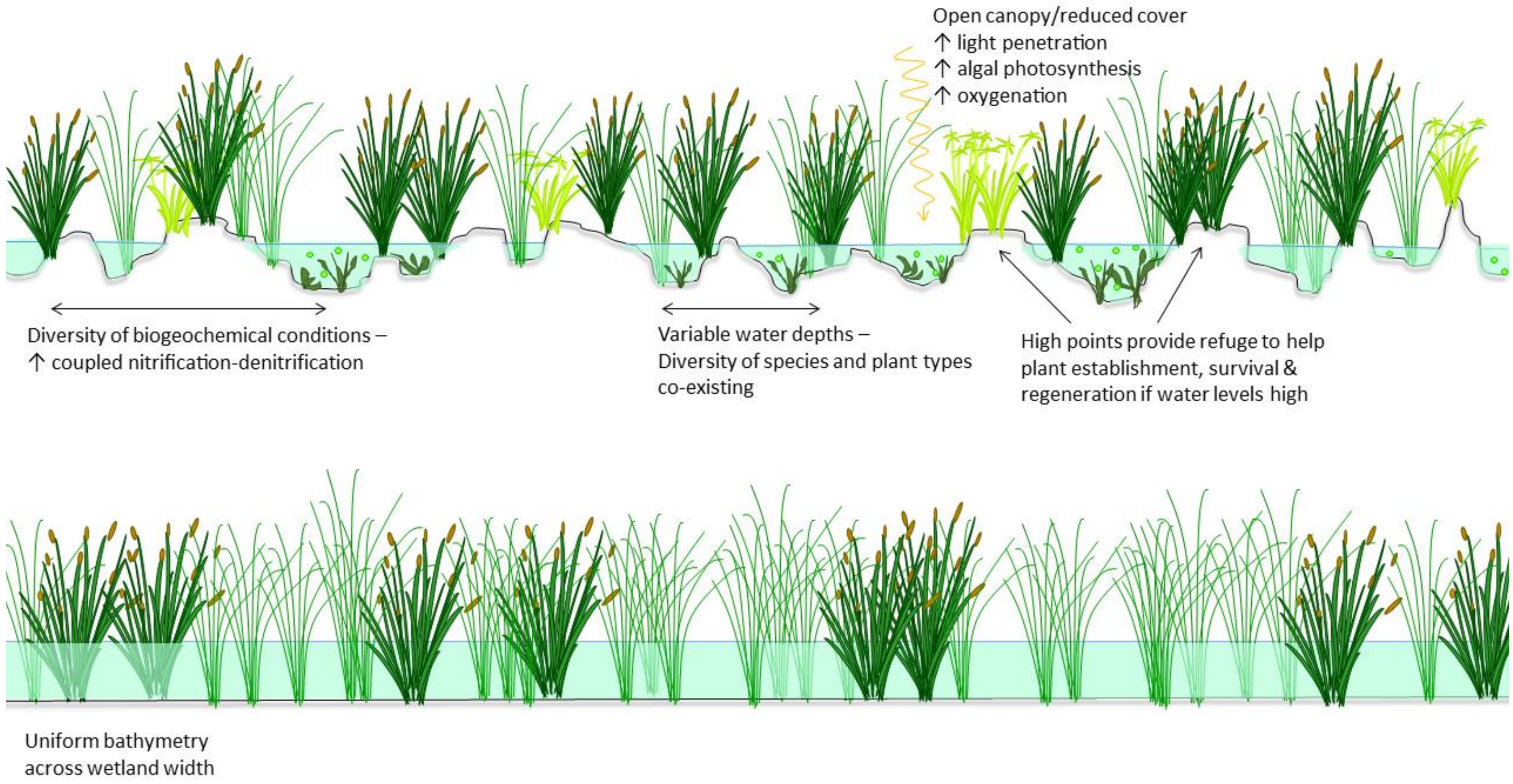


Figure 6 Microtopography and its influence on wetland function (above), with flat bathymetry shown for comparison (below)

4.5 Biofilms and algal communities

Biofilms and algae can contribute significantly to wetland function. They can provide some compensation for vegetation loss, providing functionality in the sediment or water column in non-vegetated zones. However, their contribution in isolation will not exceed the benefits of vegetation. Instead, biofilms and algae complement the role of vegetation in water treatment. This suggests the counteractive role of plants shading out algal photosynthesis in biofilms and the water column is not likely to be a primary concern.

Although not readily visible, bacteria, fungi and algae can be the powerhouses behind wetland functioning (Wetzel, 2001). The diversity of microorganisms and algae is vast, as is their combined substrate processing capacity (Hiraki et al., 2009, Faulwetter et al., 2009).

4.5.1 Biofilms

Biofilms are ubiquitous in the environment and facilitate highly efficient processing by bacteria and algae. They positively contribute to water treatment via nitrification, denitrification, photosynthesis and breakdown of organic compounds. Little information is available to guide direct management of biofilms or algae in constructed wetlands, but the best approach may be to promote high plant cover and hydraulics allowing high surface area contact between the water and wetland surfaces.

Biofilms represent an incredible synergy between single-celled organisms. They are a community of bacteria and algae within an extracellular polymeric matrix (or slime) formed on solid surfaces, and in the case of wetlands on the surface of plants, litter and within the soil (Brix, 1997). Biofilms are also known as epiphyton (on submerged plants) and periphyton (attached to plants, the sediment or floating) (Kadlec, 2006). Their benefits to wetland functioning include:

- Biofilms allow **highly efficient processing**. Individual processes occur within close proximity, allowing the outputs from one process to be readily utilised as reactants in another (Pollard, 2010) due to short diffusion distances and high concentration gradients (Wetzel, 2001). Reactants may also be readily sourced from the underlying substrate or passing flow of water.
- **Processes occurring in biofilms** can include nitrification, denitrification, photosynthesis, nitrogen fixation and decomposition of a range of organic

pollutants (Grismer and Shepherd, 2011, Eriksson and Weisner, 1997, Scott et al., 2005). A large proportion of wetland functionality and primary production may be attributable to biofilms, including biofilms in the substrate and surrounding plant roots (in the rhizosphere) (Brix, 1997, Wetzel, 2001).

- **Process coupling is particularly efficient** in biofilms, including coupled nitrification and denitrification. Nitrifiers occur near the surface utilising oxygen available from diffusion or algal photosynthesis, while anaerobic zones, particularly prevalent at night, support denitrification (Eriksson and Weisner, 1997).
- **Denitrification occurring in biofilms on plant surfaces** (the epiphyton) **can be of comparable magnitude to**, or may even exceed, **sediment denitrification** on an areal basis. This results from high plant surface area relative to that of the sediment, but dependent upon plant biomass and biofilm development (Eriksson and Weisner, 1997, Bourgues and Hart, 2007).
- **Biofilm assimilation is also significant**. The electrostatic charge of the surrounding matrix facilitates absorption of dissolved substances and ions may be concentrated by 100 to 1000 times that of the water column (Hiraki et al., 2009, Pollard, 2010). The magnitude of this contribution relative to that of the plant itself is debateable. For example, in a eutrophic system nitrogen uptake by epiphyton was half the magnitude of assimilation by submerged vegetation (Eriksson and Weisner, 1997), whereas in a seagrass bed the productivity of algae in epiphyton was of similar magnitude to submerged macrophytes on a surface area basis (Pollard and Kogure, 1993).
- Alongside the beneficial influences of plants, biofilms in the rhizosphere can also **assist in the decomposition of recalcitrant organic pollutants** (Grismer and Shepherd, 2011).
- Biofilms also contribute to the **water resistant base layer**, alongside phenolic compounds, helping to maintain the wetland environment (Environment Heritage and Local Government Ireland, 2010).

However, aspects of biofilm functioning that do not benefit water treatment include:

- **Highly efficient internal recycling** of nutrients (Mulholland et al., 1994)
- Extensive biofilm coverage of plant surfaces **may reduce plant photosynthesis** (Carpenter and Lodge, 1986, Pollard and Kogure, 1993), but there is little evidence for this.

It is clear that biofilms can contribute substantially to water treatment (Hiraki et al., 2009). In light of this, are there mechanisms to promote biofilms in constructed wetlands? Biofilms are effectively harnessed in wastewater treatment (Pollard, 2010), but little is known of their formation, functioning or contribution to performance in constructed wetlands (Grismer and Shepherd, 2011). Biofilm processing can correlate with macrophyte surface area and be promoted by dense vegetation (Eriksson and Weisner, 1997). However, the relationship is not necessarily straightforward. Shading from dense vegetation can reduce biofilm photosynthesis (McCormick et al., 1997, Kadlec, 2006) and biofilm growth is three-dimensional and spatially variable – so it may not be reliably predicted from the plant surface area (Pollard, 2010). Adding further complication, respiration and growth are not always coupled and biofilms are complex and heterogeneous structures. As a result, functioning may be variable and unexpected (Hall-Stoodley, 2004)(Pollard, 2010).

Despite this, biofilms are ubiquitous in aquatic environments (Hiraki et al., 2009) and conditions for their development include:

- Available plant surfaces near the water surface (e.g. within 1.5 cm) where photosynthetic epiphyton will develop most readily (Pollard, 2010).
- Biofilm structure and growth will be influenced by flow velocity, but they are adaptable to a range of flows and take a number of physical forms (Hall-Stoodley, 2004).
- Development can be rapidly (e.g. within 7 days), but also reach maturity quickly (within 21 days), with potentially slowed processes beyond this point (Hiraki et al., 2009).
- Rapid response and functionality may be promoted by maintaining a thin biofilm (as thick biofilms can be diffusion-limited) (Wetzel, 2001) with high respiration but low growth, achieved by nutrient pulses to physiologically stress the biofilm (Pollard, 2010).
- In the sediment biofilm development is facilitated by high litter and humus (Wetlands International, 2003).
- Biofilm functionality will vary between surfaces. For example, biofilms on pine and spruce twigs had higher nitrification relative to those in the sediment, on submerged macrophyte shoots and macro-algae, while denitrification was highest in the sediment (Bastviken et al., 2003).

However, the benefits of micro-engineering biofilm processes to the extent of adding pine twigs to wetlands are unknown and may have unforeseen ramifications on other processes. It may be best to simply increase the opportunity for biofilm development by facilitating high vegetation cover and promoting contact between water the biofilms (via hydraulic design) (Wetzel, 2001, Eriksson and Weisner, 1997).

4.5.2 Algae

Algae can occur in a number of forms within wetlands – in biofilms, single-celled or a wide variety of macroalgal structures. Their primary contribution to water treatment is via rapid nutrient assimilation and oxygenation of the water column. Macroalgae can additionally filter particulates and slow water velocity. Conversely, nitrogen fixation by cyanobacteria, the export of algal cells downstream as suspended organic matter, and the potential development of short-circuits around large macroalgal structures can all be detrimental to water quality treatment in wetlands. However, little information is available to quantify the contribution of these different mechanisms.

Algae should not be considered as a replacement for vegetation cover in wetlands. However, the contribution from algae alone may be greatest as the vegetation first establishes and by providing nutrient processing capacity in open water zones, albeit of a lesser magnitude than vegetated zones.

Algae occur within biofilms, but also in free-floating form (plankton) or in large assemblages suspended within the water column (macroalgae). Algae can contribute to wetland functioning in a variety of ways –

- Algae can provide an **effective sink for nutrients** (Thullen et al., 2002). Unlike rooted macrophytes, algae take up all their nutrients directly from the water column (McCormick et al., 2006, Greenway, 2010, Wu and Mitsch, 1998) and their high surface area to volume ratio drive higher nutrient uptake rates than macrophytes (Williams 1985).
- **Some nutrients assimilated by algae can be effectively stored in the long-term**. Although algal tissues are typically more labile than plant tissue and algae have a lower biomass, some algal taxa comprise a significant recalcitrant fraction. This, combined with high turnover rates, provides both short- and long-term nutrient storage (McCormick et al., 2006).

- **Algae respond rapidly** with high growth and uptake when a growth limiting nutrient becomes available. This is particularly beneficial for intermittent stormwater inflows (McCormick et al., 2006).
- **Algal photosynthesis can significantly oxygenate the water column** and remove carbon dioxide during the day (Kadlec, 2008, Wood, 1995). This increases ammonium removal via nitrification, exceeding the level of removal provided in vegetated but anaerobic waters (Bachand and Horne, 1999, Thullen et al., 2002).
- **Increased pH** as a result of photosynthesis – this effect can be significant if algae are abundant, and may increase ammonia volatilisation, nitrification and phosphate precipitation (Vymazal, 2007, Wood, 1995, Thullen et al., 2002).
- Macroalgal forms, particularly filamentous algae forming mats, can slow water velocity and **physically filter particulates from the water** (Mulholland et al., 1994, Wu and Mitsch, 1998).
- Due to the shading effect of plants, algal abundance is roughly inversely related to macrophyte cover, with algae typically most abundant in open water with maximum light penetration (Kadlec, 2008, Kadlec, 2006, McCormick et al., 1997). Hence, **algal nutrient uptake will help to somewhat compensate for vegetation loss**. However, Kadlec (2008) found this was not sufficient to offset a performance decline in the context of a wastewater treatment wetland suffering vegetation loss.

Conversely, potentially detrimental influences of algae include:

- **Excess algal biomass** (or algal blooms) **will deplete oxygen within the water column**, suffocating organisms and aerobic processes, and may release toxins (e.g. blue-green algae) (Kadlec, 2008)
- Algae are **highly mobile**, and the benefits of nutrient uptake may be negated if algae are exported downstream as suspended solids (Brix, 1994b, Kadlec, 2008, Thullen et al., 2002)
- Similarly to plants, **algal functioning is be seasonal** – positive uptake may occur during the growing season but die-back and nutrient release at the end of this season (Wu and Mitsch, 1998).
- **Cyanobacteria**, which under some definitions are classified as algae, **can fix nitrogen**, which may act to reverse the benefits of removal mechanisms (Kadlec, 2008). An imbalance in nitrogen and phosphorus removal down the length of a wetland (which may occur as a result of nitrogen denitrification and phosphorus

accumulation) can promote nitrogen fixation towards the outflow (Scott et al., 2005). However, little is known its significance and the process is often disregarded (Vymazal, 2007).

- **Macroalgal forms can alter flow dynamics** which can either have a positive effect on retention or negative implications by promoting stagnant zones and short-circuiting (Mulholland et al., 1994).

The contribution of algae to functioning will be constrained by the availability of light, nutrients and substrate for their growth (McCormick et al., 2006). Despite the general inverse relationship between algae and plant cover and the clear benefits of algae to water treatment (Kadlec, 2008, Kadlec, 2006, McCormick et al., 1997), this should not imply that low plant cover is desirable in constructed wetlands. The benefits of plants to water treatment are well founded. At any rate, some studies report algal productivity in biofilms can be high even amongst dense vegetation, indicating sufficient light can penetrate through (Pollard and Kogure, 1993). On their own algae can play a vital role in water treatment during early wetland life as the vegetation establishes (Wu and Mitsch, 1998) and contribute to the functionality of open water zones (Kadlec, 2008).

Hence, algae should be considered to complement, and not substitute, the positive contribution of plants to water treatment.

4.6 Substrate depth, composition and groundwater interactions

Substrate should not be ignored as an important contributor to wetland function, influencing the success of vegetation establishment, hosting a wide array of microbial processes and providing a sink for sediments and other contaminants. Depth and composition are key design parameters. Importantly, soil characteristics are critical for, and may help predict, denitrification capacity. Constructed wetland soils will evolve over time but do not necessarily develop the same characteristics as natural wetland soils.

While there could be some benefits from infiltrating stormwater into surrounding soils, the use of clay liners on wetlands is appropriate to protect the underlying groundwater from contamination and ensure the wetland is not flooded, dried or exposed to high salinity as a result of exchanges with the groundwater.

Wetland substrate is not simply an inert support material, but an active biogeochemical component of wetland function. While the substrate plays a more crucial role for treatment in subsurface flow wetlands (Akratos and Tsihrintzis, 2007, Vymazal, 2007) it still has multiple roles in free water surface wetlands and can have a strong influence on performance and wetland success (Wolf et al., 2011a, van der Valk, 2012). The substrate provides physical support and nutrients to macrophytes, a matrix to support biofilms and other microbial communities and a sink for pollutants retained by sedimentation, complexation or adsorption reactions (Hammer, 1992, Heers, 2006). Key parameters include carbon content, hydraulic conductivity, mineral composition, clay content and substrate depth (Dunbabin and Bowmer, 1992).

Constructed wetland substrates are typically terrestrial soils, varying with local conditions, ranging from mineral to organic (Kadlec, 2006, Campbell et al., 2002). Constructed wetlands soils typically have lower organic content, a higher bulk density, more rock and sand, but less silt than natural wetland soils (Campbell et al., 2002). Wetland substrate must have sufficient organic matter and nutrients to support plant growth, but not an excess which will act as a nutrient source (Martin and Reddy, 1997, Wetlands International, 2003). This can be problematic when former agricultural soils are used in constructed wetlands if previous fertiliser application was high (Kadlec, 2006). Constructed wetland substrate typically has low sorption capacity, despite the benefits of a high adsorption capacity (associated with aluminium- or iron-rich media with high clay or humus content) to phosphorus and metal removal (O'Sullivan et al., 2004, Wetlands International, 2003, Vymazal, 2007).

High organic carbon and nitrogen content, gravimetric soil moisture and low bulk density is correlated with denitrification (Ahn and Peralta, 2012, Wolf et al., 2011a).

In addition to composition, substrate depth is also critical to plant rooting depth and vegetation survival.

In turn, the successful establishment of vegetation is vital to prevent erosion of the topsoil, which, once commenced, impedes plant survival further, forming a negative feedback cycle (Greenway and Polson, 2007). Ideally, the substrate would accommodate the entire root system of the macrophyte community (Heers, 2006). However, the majority of plant roots generally occur in the top 200 mm, although roots down to 300 mm are also common and some species may have even deeper root systems (Kadlec and Wallace, 2008, Lieffers and

Shay, 1981). While deeper topsoil is more desirable, a minimum of 200 mm appears reasonable for the majority of roots.

Characteristics of the substrate are expected to change over the wetland life. Initially, hydric soil properties develop rapidly (e.g. within 7 years) from former terrestrial soils in constructed wetlands (Mitsch et al., 2005b). Additionally, sedimentation and the action of plant roots and microbes over time increases porosity and organic content (Stottmeister et al., 2003). However, soils may take a long time, if ever, to mimic characteristics of natural wetlands (Campbell et al., 2002).

The choice of lining for constructed wetlands is critical for interactions with groundwater. Most constructed wetlands are lined with a compacted clay layer and groundwater interactions are largely ignored – either assumed to be negligible or for inputs and outputs to cancel (Williams 1985). However, as wetlands are typically low-lying areas, groundwater interactions are frequently an important hydrological component for natural wetland function (Williams 1985)(Hunt Factsheet 2) (Hunt et al., 1999). In constructed wetlands, infiltration of stormwater to surrounding soils would reduce outflow volumes and benefit pollutant removal. However, a permeable wetland base is accompanied by risks of flooding and salinization from groundwater intrusion into the wetland, or infiltration could produce excessive drying out of the wetland or contamination of the underlying aquifer. Hence, if limited data on underlying hydrogeology is available, use of a liner in constructed wetlands is most appropriate.

4.7 Hydrology and Hydraulics

Effective wetland treatment requires the well-established hydraulic principles of retention, high contact between water and wetland components, flow distribution and plug flow. A number of design features can be utilised to promote these conditions, but their placement is critical - orientation perpendicular to the flow, most often across the entire width, is necessary to yield an advantage.

The hydrological regime and its resulting hydraulics are the defining characteristic of wetlands (Heers, 2006, Wong, 1999), leading to the development of other key features including macrophyte vegetation and hydric soils. Essentially, the flow dynamics define the degree of contact between the stormwater and wetland elements, and therefore the extent of processing (Bodin et al., 2012).

Inappropriate hydrology leads to vegetation loss in wetlands (discussed in Section 4.3.5) (Vanderbosch and Galatowitsch, 2011, Greenway et al., 2007, Gawne and Scholz, 2006, Raulings et al., 2010, Webb et al., 2012). Once erosion, scouring and topsoil loss commences, vegetation survival and establishment is further impeded, leading to a negative feedback (Greenway and Polson, 2007). Achieving the desired hydrology relies upon multiple design elements including inlet and outlet configurations, invert levels and gradients, bathymetry, wetland shape and vegetation zones. The importance of hydrology and hydraulics has already been outlined in the context of wetland configuration (Section 4.4). In the following sections the hydraulics of flow paths and short circuiting, and their interaction with wetland design features, are discussed. Suitable hydrologic regimes to support healthy vegetation are also discussed.

4.7.1 *Flow paths*

Even distribution of flow across the width of the wetland is essential for effective water treatment. Hydraulic efficiency represents the degree of flow distribution and extent of idealised plug flow (or conversely mixing) occurring within the wetland. High hydraulic efficiency is promoted by wide inflow distribution and features that entire span the width of the wetland such as shallow bars and bands of vegetation. Deep water zones, placed intermediately within the wetland, help to break short circuits by dissipating flow momentum and promoting mixing. However, as deep zones provide only minimal treatment processes, their proportional area must be limited and they can provide no net benefit in undersized wetlands.

Flow pathways determine the travel time for stormwater and its interaction with different wetland components on its journey. Hence, hydraulics is closely tied to treatment performance (Wong et al., 2006, McClain et al., 2003). Ideally, all water parcels entering the wetland will experience uniform retention and even distribution throughout the wetland cross-section (Persson et al., 1999). This requires plug flow, maximised use of the wetland area and minimal stagnant zones, short circuiting or mixing between incoming and resident water (Wong et al., 2006, Persson et al., 1999).

Hydraulic efficiency, an indice proposed by Persson et al. (1999), quantifies the degree of flow distribution (represented as the effective volume) and plug flow (represented by the pollutant hydraulic residence time distribution). The effective volume is the wetland volume overlying the substrate (excluding submerged plant components, litter and sedimentation), while the hydraulic residence time represents 'average' retention time, determined by dividing the wetland volume by average flow rate (Heers, 2006). The hydraulic efficiency can be determined from outflow concentration-time graphs (Persson et al. 1999). A spike in contaminant/tracer concentrations at the outflow represents ideal plug-flow, whereas a prolonged recession over time indicates poor flow dynamics with mixing and short-circuiting (Wong et al. 2005).

In reality mixing will invariably occur, as will velocity heterogeneity, both horizontally and vertically within the wetland (Kadlec, 2010). While hydraulically undesirable, mixing can act to dilute incoming contaminant concentrations with cleaner resident water (Holland et al., 2004). These complex dynamics can generate concentration-retention time graphs that deviate substantially from the ideal single spike, in which case an average retention time (i.e. time of peak outflow concentration) may be misleading and provides scant information on the wetland hydraulics (Holland et al., 2004). Beneficial mixing is also promoted by intermediate deep zones, aligned perpendicular to the flow, which allow wind-driven mixing and reduce the flow velocity (Lightbody et al., 2009). While these zones may contribute minimally to direct treatment mechanisms, their hydraulic benefits can be realised as long as a wetland is not undersized. Their optimal number and size will depend on each system. Modelling results suggest intermediate deep zones with length greater than 10 m and comprising between 5-37% of the total wetland area, while a comparison of field systems found 0-20% of the wetland area as deep zones maximises nitrogen removal (Lightbody et al., 2009).

Parameters that influence wetland flow paths include vegetation (zonation, type/species, density), system layout (placement of treatment zones) and wetland bathymetry (Greenway, 2010, Min and Wise, 2009). Flow paths may shift over time due to the accumulation of sediment or plant litter, which can encourage channel flow (Wood, 1995, Keefe et al., 2010).

Ideally designs will promote the broad distribution of flows across the width of the wetland. This optimises use of the entire area and facilitates a more uniform distribution of oxygen and organic material (Akratos and Tsihrintzis, 2007). Wide flow distribution is assisted by multiple inlets, wide inlets, gentle bottom gradients, features aligned perpendicular to the flow across the width including shallow bars, limited deep zones (including intermediate, inlet and outlet) and consistent bands of vegetation and an island placed centrally to disperse inflows (but avoiding narrow channels on either side) (Greenway, 2010, Akratos and Tsihrintzis, 2007, Hammer, 1992, Persson and Wittgren, 2003, Persson et al., 1999, Lightbody et al., 2009, Koskiaho, 2003, Kadlec, 2011). Conversely, designs should avoid promoting concentrated inflow, channelized flow paths and fringing vegetation, all of which facilitate short-circuiting and dead zones (Kadlec, 2008, Greenway, 2010, Persson et al., 1999, Jenkins and Greenway, 2005).

4.7.2 *Short-circuiting and stagnant zones*

Short-circuit flow paths can transport a high proportion of the flow at high velocity, leading to minimal retention time and contact with vegetated wetland zones. This invariably reduces water treatment. Short-circuits can be difficult to detect visually and their significance will change with the flow.

Beneficial design features include consistent vegetation bands across the wetland, low gradients, baffles, bathymetry that promotes flow distribution rather than channelisation and the careful placement of features to facilitate hydraulic efficiency.

Short-circuit flow paths are typically either non-vegetated or sparsely vegetated, narrow and deep channels that carry a sizeable proportion of the flow at relatively high velocity (e.g. 20-70% of the flow) (Dierberg et al., 2005, Lightbody et al., 2008, Min and Wise, 2009). They occur due to channelisation of the flow, which may be promoted by zones of sparse vegetation, erosion, existing deep channels parallel to the flow, shorter flow path length on one side of the wetland, or the funnelling of flow between features (e.g. islands, discontinuous bars, fringing vegetation, accumulated plant litter at the end of the growing season, V-notch weirs) (EDAW, 2008, Storm Consulting, 2013, Heers, 2006, Kadlec, 2008, Greenway, 2010, Keefe et al., 2010). Multiple short-circuit pathways can occur across a wetland but they are not always readily visible (Lightbody et al., 2008).

Short-circuits lead to reduced retention times and minimal contact with vegetation, sediment and microbes. This reduces water treatment, including lower nitrogen and phosphorus removal, and increased TSS concentrations (Kadlec, 2008, Dierberg et al., 2005). An interesting comparison was noted by Dierberg et al. (2005) with exponential decline in phosphorus concentrations in shallow vegetated zones but only a linear decline within a short-circuit.

Short-circuits will invariably be accompanied by stagnant zones away from the fast-flowing channel/s. These may be promoted by discontinuous shallow benches across the wetland width, corners, sheltered zones (i.e. protected by features obstructing the flow) and fringing vegetation (Thackston et al., 1987, Jenkins and Greenway, 2005). Stagnant zones can provide effective treatment due to the prolonged retention (Persson 2004), but this benefit will not always compensate for the channelling of the majority of flow rapidly in non-vegetated short-circuit channels (Dierberg et al., 2005).

The potential for short-circuit development will be highly sensitive to the design and construction of wetland levels, gradients and bathymetry (Hammer, 1992, Min and Wise, 2009). Deep zones oriented across the wetland (located at the inlet, intermediate and outlet), low gradients, flow distribution, multiple inlets and outlets, well distributed vegetation and carefully designed hydraulics are some design elements that can reduce short-circuiting (Lightbody et al., 2008, Lightbody et al., 2009, Koskiaho, 2003, Kadlec, 2011) (Figure 7). However, they can still be problematic even when designs have incorporated these features (Lightbody et al., 2008). The degree of short-circuiting is likely to differ across the flow regime. Poor hydraulic efficiency (i.e. high degree of short-circuiting) may occur at low flows but improve at higher flows (Braskerud, 2001), but conversely others report greater short-circuiting at higher water levels (Holland et al., 2004). Detecting short-circuits can be difficult – they may not be visually apparent, but Lightbody et al. (2008) suggested thermal imagery could be applied to detect and map short-circuits due to cooler water in the densely vegetated slow-velocity areas.

Wind is also a key factor that can both help promote or mitigate short-circuiting and interact with treatment processes. Wind can be beneficial to mixing when the direction is perpendicular to the flow, and acts to re-oxygen of the water column (Brix, 1997, Lightbody et al., 2008), but wind can also have a detrimental influence by promoting short-circuits (Persson and Wittgren, 2003, Thackston et al., 1987) and resuspending sediments. Its influence will change as wind direction and strength shifts (Thackston et al., 1987). In general, designs that minimise the wind fetch in the typical wind direction are recommended, and baffles are particularly effective for this (Thackston et al., 1987).

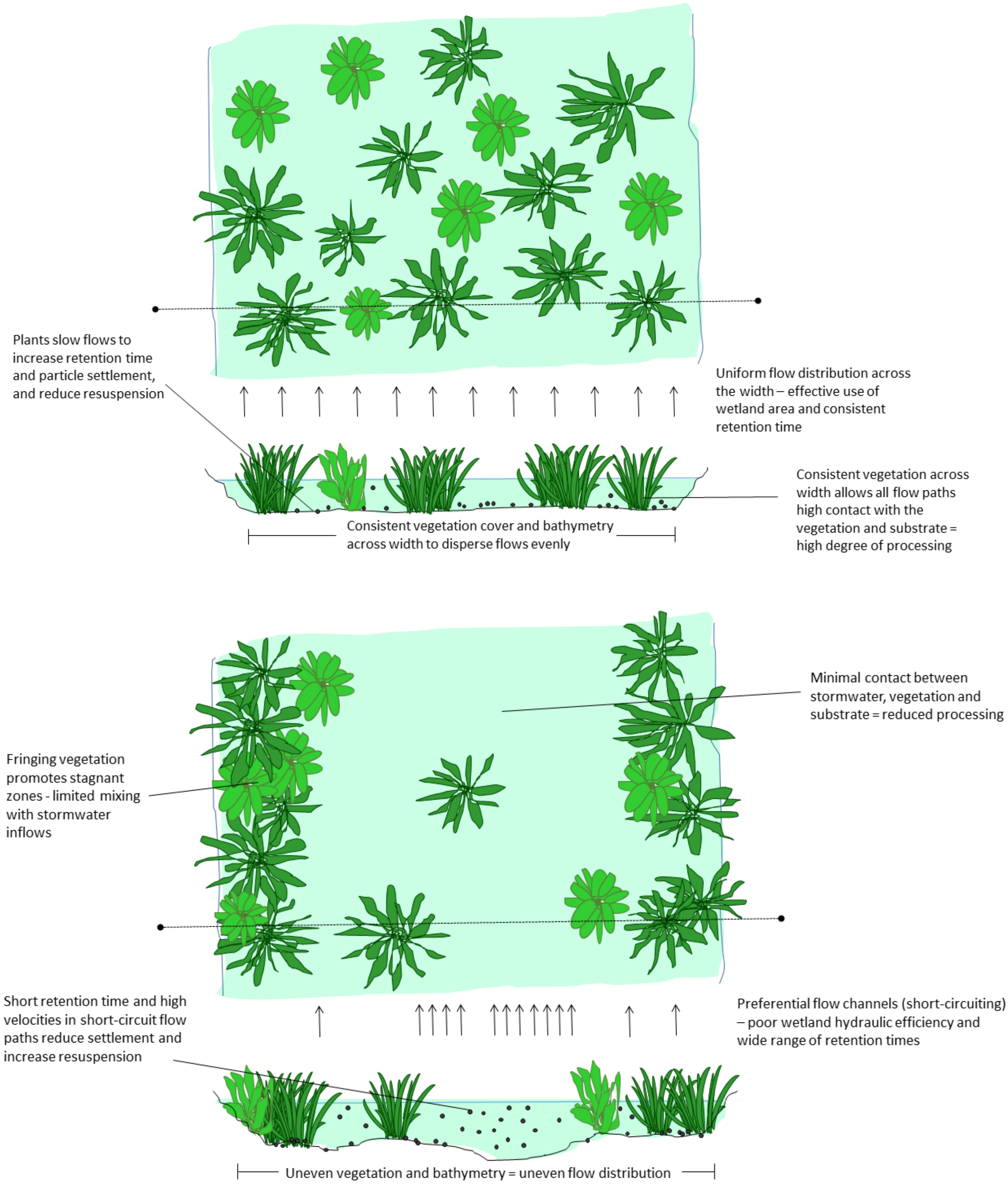


Figure 7 Wetland configuration with consistent vegetation and good flow distribution (above) and poor vegetation distribution and short-circuit flow paths (below)

4.7.3 Supporting healthy vegetation

Growth of emergent macrophytes is highly sensitive to water depth. Plants will change their morphology and physiology to try to cope with increased water depths, but growth becomes difficult beyond a threshold depth. Tolerances vary between species but the majority of species survive best in water depths < 0.3 m. Constructed wetlands therefore require extensive shallow areas providing a range of water depths with zones < 0.2 – 0.25 m, and others < 0.3 - 0.4 m.

However, water level fluctuations are a key characteristic of constructed wetlands. Designers must consider not only the normal water depth but also the range of extended detention depths experienced across the wetland zones. Plant growth will be sensitive to the depth, frequency and duration of water level changes as the extended detention volume is engaged and slowly drains following an inflow event. Species will vary widely in their preference or tolerance for a relatively static water level or varying degrees of fluctuations in water level. Plants may be best able to adapt their growth when water levels either change slowly or very rapidly, whereas growth may be compromised most by an 'intermediate' frequency of change which prevents the plant from adopting an appropriate growth strategy.

Increasing water depth has a negative influence on emergent plant growth (Dugdale and Ede, 2013, Webb et al., 2012). As depth increases, plants must spread resources and biomass across an increasing distance between the substrate (for anchorage, nutrient uptake and storage) and atmosphere above the water (for efficient gas exchange and photosynthesis) (Vretare et al., 2001, Perata et al., 2011, Hoban et al., 2006). Plants generally decrease allocation to roots and rhizomes and rapidly increase stem length when faced with an increase in water depth (Blanch et al., 1999, Miller and Zedler, 2003, Perata et al., 2011). Hence, excessively deep water presents a clear threat to the establishment and survival of vegetation in constructed wetlands.

Species will vary significantly in their preferred water regime and it is often difficult to identify optimal conditions for select species (Brock and Casanova, 2000, Greenway and Polson, 2007, Webb et al., 2012). Plants display non-linear growth responses to changes in water depth (Sorrell et al., 2012, Webb et al., 2012), and emergent macrophytes will experience growth difficulties as depth exceeds a species-specific threshold (Sorrell et al., 2012, Blanch et al., 1999). Most emergent species require water depths less than 0.3 – 0.4 m (Kadlec, 2006, Mitsch and Gosselink, 2000b, Sorrell et al., 2012, Lieffers and Shay, 1981).

Some species will require water shallower than 0.2 or 0.25 m, while a few select species may tolerate depths beyond 0.50 m (e.g. *Typha orientalis*, (Sorrell et al., 2012)) (Blanch et al., 1999). In line with this, constructed wetlands are commonly reported to contain shallow vegetated zones less than 0.4 m deep (Malaviya and Singh, 2011, Mitsch et al., 2005a), but generally including extensive zones below 0.2 or 0.3 m maximum depth (Hammer, 1992, Woods et al., 2004, Kadlec, 2011, Environment Heritage and Local Government Ireland, 2010).

However, constructed wetlands do not simply experience static water levels; depths will increase beyond the normal water level as the extended detention volume fills and drains in response to inflow events (Wong, 1999). Melbourne Water wetlands are designed for the extended detention storage to drawdown over 72 hours following an inflow (and minimum 48 hour drawdown period) (Carew, 2012). However, an investigation using data from water level loggers found < 20% of wetlands had both i.) water levels drawing down to the normal water level and ii.) this occurred over the appropriate drawdown period (Carew, 2012). The majority of wetlands instead experienced deeper water levels than the design intended.

Plant growth will be influenced not only by the normal water depth, but also the depth, duration and frequency of water levels within the extended detention capacity of the wetland. Identifying an appropriate hydrological regime is challenging given the complexities of plant responses to different aspects of the flow regime. Multiple studies identify contradictory responses between plant species to changes in the flow regime (Dugdale and Ede, 2013, Webb et al., 2012, Greet et al., 2011, Miller and Zedler, 2003), differing with depth and the amplitude and frequency of fluctuations (Miller and Zedler, 2003, Vretare et al., 2001, Deegan et al., 2007). Some species will grow better and uptake more nutrients under static water levels, while other species are adapted to water level variation and require some fluctuation for optimum growth (and the ideal extent of fluctuation will vary) (Deegan et al., 2012, Deegan et al., 2007). High variation in water levels present a challenging environment as species are constantly seeking to re-allocate biomass to optimise growth at each water depth (Vretare et al., 2001). Vretare et al. (2001) suggest if the frequency of change is low plants are able to adjust their growth strategy, and conversely a high frequency of water level change means plants will not have time to adapt to either extreme but will compromise with an 'in-between' strategy. The greatest negative influence may result from an 'intermediate' rate of change in water levels, which provides enough time for a response, but the water level changes again before the plant can gain much advantage and a different strategy is required.

Despite the variation in optimal hydrological regime between species, is clear that excessive inundation depth is detrimental to plant growth, and reduced water levels, up to the point of mud, benefits species recruitment (Dugdale and Ede, 2013, Webb et al., 2012). Water depth and fluctuations in depth have a strong influence on plant growth. While species vary in their tolerance, deep water or an 'intermediate' frequency of high water level variation place plant growth under stress (Vretare et al., 2001, Miller and Zedler, 2003). Further work is required to ascertain beneficial water level manipulation across a broad range of species and environmental conditions, for both plant growth and nutrient retention (discussed in detail in Section 4.2.2).

4.8 Temperature

Due to its influence on microbial processes and plant growth, temperature is a key determinant of wetland efficiency. Nitrogen removal is particularly sensitive to temperature while sediment removal is not. Cold temperatures may require longer retention time for effective nitrogen removal, although a higher degree of oxygen saturation can help to offset the slower process rates. Plant shading acts to reduce water temperatures and temperature differentials within the wetland may be useful for identifying hydraulic pathways.

Temperature dictates the rates of biochemical processes, and therefore influences treatment efficiency and the necessary retention time (Akratos and Tsihrintzis, 2007, Kadlec, 2011). Due to its reliance upon plant and microbial processes, the retention of nitrogen is particularly sensitive to temperature, and longer retention is generally required during colder periods as reaction rates slow (Akratos and Tsihrintzis, 2007, Kadlec, 2011). However, cold water temperatures allow a higher degree of oxygen saturation, which will benefit aerobic processes, providing some offset for slower rates (Kadlec, 2008, Faulwetter et al., 2009, Environment Heritage and Local Government Ireland, 2010). Physical processes are relatively insensitive to temperature, so retention of sediments and particulate-bound contaminants, including phosphorus, will not be influenced to the same extent (Akratos and Tsihrintzis, 2007, Bodin, 2013).

Water temperature has been reported to both decrease (Kadlec, 2008) and increase (Mitsch et al., 2005b) across wetland systems depending upon system characteristics. Shading by vegetation tends to reduce the water temperature in vegetated zones (Kadlec, 2008, Mitsch et al., 2005b). Such temperature differentials can be used to indicate wetland function, particularly hydraulics and the identification of short-circuits, which may be associated with warmer water (Lightbody et al., 2008).

5 Effective methods for monitoring wetland performance

5.1 Performance evaluation against objectives

Assessing wetland performance is vital to improve future design and direct operation and maintenance of systems. Constructed treatment wetlands need objectives that are realistic in light of their urban context, potential for contaminant re-release, high variability in performance, inevitable trade-offs between multiple contaminants, and changing structure and function over time.

Given the sensitivity of wetland function to design parameters, and the investment required to design, construct and maintain constructed treatment wetlands, a method to assess system performance is vital. This requires clear objectives and well defined monitoring protocols. The outcomes are vital to inform future designs and direct the prioritisation of maintenance and remediation activities. Without well-defined objectives significant resources can be wasted (Grayson et al., 1999). Nevertheless, objectives are useless unless they are relevant. This means they need to account for the context and reality of constructed wetlands, taking into consideration:

- Differences between systems in the hydrology and pollutant received from the catchment (Malaviya and Singh, 2011).
- The dynamic and complex nature of wetlands with changes in structure and function expected over time (Mitsch et al., 2005a). Trends can shift and reverse direction (Grayson et al., 1999).
- The initial establishment phase, during which time the structure and functions within the system will change and likely differ from longer-term patterns (Kadlec, 2006). The timeframe for this is unknown, particularly for the development of

functions, but in terms of vegetation establishment may be between one to three growing seasons (Hammer, 1992, Brock and Casanova, 2000).

- Performance is commonly highly stochastic and poorly represented by 'averages' – some indication of performance variation about the mean is required (Koch et al., 2013, Kadlec, 2011).
- Some trade-offs are required across the multiple water treatment objectives as not all objectives can be simultaneously met (Koch et al., 2013, Grayson et al., 1999). For example, the removal of dissolved and particulate contaminants is often uncoupled (Goonetilleke et al., 2005, Bodin, 2013) and treatment capacity requires compromise against provision of sufficient retention and shallow treatment zones.
- Many processes only provide temporary attenuation and at some point the pollutant will be re-released (Koch et al., 2013). Sediment re-suspension, erosion and nutrient release from decomposing organic matter can lead to poor performance, even if removal processes are relatively effective (Braskerud, 2001).
- Wetlands integrate a number of fields including civil engineering, ecology, geotechnical science, hydrology and hydraulics. Objectives and performance indicators should be equally multi-disciplinary.

These considerations suggest the objectives for constructed treatment wetlands should differ substantially from those developed for remediation of natural wetlands and be system-specific. In addition, long monitoring periods are required to assess objectives in the face of shifting wetland structure and function (Grayson et al., 1999) and the potential for subsequent re-release of contaminants.

5.2 Requirements for functional indicators

The search for proxy indicators of system function is a challenge across many natural and constructed environments. Structural parameters are commonly used to infer system function, but this approach requires caution. Wetland structure is not always correlated with function. Indicators should be quantifiable, closely and consistently related to processing, and comparable against objectives. Monitoring must be conducted over a sufficient frequency and time frame to capture seasonal and long-term shifts in wetland performance.

Determining reliable and cost-effective indicators for wetland treatment performance is something of a holy grail. Typically, wetland structural measures are quantified and used to infer wetland function (Grayson et al., 1999). However, it is vital to understand the potential differences between structure and function. While they may be correlated (Knight, 1992), achieving the desired wetland structure does not universally imply effective function (Kadlec, 2006, Grayson et al., 1999, Español et al., 2013). Structural measures characterise the system at a point in time, whereas functions occur over time (Grayson et al., 1999). As a result, monitoring over time is necessary to characterise function (Grayson et al., 1999). In addition, wetland functional capacity may develop across differing time scales for different contaminants, and over much longer timescales than wetland structure (Español et al., 2013).

Ideally, monitored parameters and their sampling require the following characteristics (Fennessy et al., 2007)(Spencer et al. 1998)(Grayson et al., 1999):

- Clear and quantifiable using rapid and easy methods
- Closely related to wetland function
- If possible, indicators able to aggregate with each other allows an overall assessment and comparison between sites
- Provide consistent response to system function across systems
- Non-destructive sampling methodology
- Quantification methods should be consistent and applied across a clearly defined area
- Sampling methodology that sufficiently captures temporal and spatial variation in wetland structure and function – in terms of events/inter-event periods, seasons and long-term shifts
- Sampling across a sufficient time period to surpass the establishment phase and identify long-term performance
- Defined objectives to assess performance

Importantly, the capacity of a wetland to process contaminants is not necessarily related solely to the quality of its effluent. For example, Bourgues and Hart (2007) found denitrification potential was highest in a wetland with the highest contamination of the wetlands studied with high nutrients, hydrocarbons and metals.

5.3 Potential indicators of wetland performance

Monitoring water levels, in situ dissolved oxygen, plant cover, distribution, vegetation type and substrate characteristics are particularly useful indicators of wetland performance.

Alone many of these indicators can have ambiguous or indirect relationships with water treatment processes, but in conjunction with a suite of monitored parameters their value increases.

Excluding measurement of contaminant concentrations, there are a range of potential alternate measures of wetland performance (Table 2). These variously indicate hydrology, water quality, a measure of the system physical condition or other characteristic. Some are more closely related to the primary water treatment objective of constructed wetlands than others, and these relationships have been discussed in more detail in the following sections. The most promising indicators have been highlighted. Indicators are likely to have the greatest predictive value when considered amongst a suite of other monitored parameters.

The indicators also vary in their mode of measurement including ground-based sampling within plots, along transects, on-water and various points within the wetland. Capturing regular images using both ground-based photography from consistent points and various aerial imagery, from satellites or chartered flights, are also valuable tools for documenting shifts in wetland vegetation characteristics (e.g. cover, broad plant type, density) and hydrology over time (Greenway, 2010, Brock and Casanova, 2000, Baschuk et al., 2012).

Table 2 Potential indicators of wetland performance (note – highlighting indicates the most promising indicators)

Potential indicator		Method	Relationship to function
Hydrology (see Section 5.4)	Inflow / Outflow	Automatic water level monitor and stage-discharge relationship at control point	Inflow and outflow measurements directly measure wetland function for flow modification and allow calculation of a water balance (Greater Wellington Regional Council, 2005, Environment Heritage and Local Government Ireland, 2010, Kayombo et al., 2005). This data indicates volume and flow peak attenuation in the wetland – useful for comparison against hydrological wetland objectives. If combined with water quality measurements, contaminant load reductions can be calculated.
	Water level variation	Automatic water level monitor	Provides a direct measure of function (National Research Council, 2001) in terms of the hydrological regime (or hydroperiod). This indicates if conditions are suitable for plant growth within each treatment zone (Wetlands International, 2003) and whether design objectives are met for water depth in each zone and the duration and frequency of extended detention following inflows (Somes et al., 2000, Wong et al., 1999). The data will indicate if action is required to correct unsuitable hydrology that would otherwise lead to plant death. Recommendations for suitable water depths in vegetated zones are provided in Section 4.7.3.

Water quality (see Section 5.5)	Dissolved oxygen (DO)	*In situ* probe	Levels of DO in the water column indicate aerobic or anaerobic conditions, which in turn control the extent of biochemical processes. Low oxygen will restrict nitrification and oxidation of organic matter (decomposition). It is important to note that DO in the water column is independent of oxygen concentrations in the substrate, i.e. does not indicate conditions for denitrifiers (Bachand and Horne, 1999). Hence, aerobic conditions (i.e. sufficient DO) in the water column are desirable for water treatment processes. Changes in DO across a wetland can reflect changes in nutrient availability (Wu and Mitsch, 1998). Low DO suggests less effective processing, and high nutrient concentrations or organic matter (Morris and Papas, 2012, Moreno-Mateos et al., 2012, Thullen et al., 2002). However, diurnal variation of DO can be high due to photosynthesis and respiration patterns (Kadlec, 2010).
	pH	*In situ* probe	Many biochemical processes are pH-sensitive. pH values that deviate from neutral indicate conditions that suppress certain processes. Denitrification and nitrification are reduced or completely suppressed at low pH, and nitrification and ammonia volatilisation are enhanced by a relatively high pH (Kadlec and Wallace, 2008). pH can display a diurnal pattern with an increase due to photosynthesis in the daytime (Mitsch et al., 2005b) and this pattern can indicate algal abundance (Kadlec, 2008, Greenway, 2010).
	Turbidity	*In situ* probe	Measures light scattering within the water reflecting particulate quantity, size and characteristics (includes algae, clay, silt, colloids). It is important to note turbidity cannot indicate suspended solids unless the instrument and particle characteristics are consistent, i.e. within the same system (NLWRA, 2008). If a change is detected across the wetland, turbidity can indicate wetland functioning (Spencer et al., 1998, Environment Heritage and Local Government Ireland, 2010) for settlement and retention of particulates, although caution is required as the response can instead be driven by algal abundance (Kadlec, 2008, Greenway, 2010).
	Conductivity	*In situ* probe	Measure of salinity and if excessive indicates conditions that can disrupt processes and impact negatively upon plant health.
	Transparency	Secci disk	Measurement with a secci disk only possible in deep water and requires a boat. Not particularly useful as an indicator in wetlands which are predominantly shallow.
	Water temperature	*In situ* probe	Temperature is closely related to wetland function indicating the rate of a wide range of biochemical processes including photosynthesis, respiration, nitrification and denitrification (discussed in Section 4.8) (Español et al., 2013, Kadlec, 2011). It will vary seasonally and day-to-day. High temperatures indicate rapid processing, but also reduced oxygen saturation within the water column (Faulwetter et al., 2009). Temperature differentials across the wetland can result from differences in vegetation shading and flow paths. It may indicate short-circuit flow paths if they have higher water temperatures than shallow vegetated areas with more stagnant flow (Lightbody et al., 2008). However, data interpretation should note that water temperature can variously increase or decrease across wetlands depending upon inflow hydrology, flow paths and vegetation cover (Mitsch et al., 2005b, Kadlec, 2008). Hence, temperature data must be considered in light of system-specific conditions and a range of data points are required to detect patterns across the wetland.

	Colour	Visual assessment of colour, photograph	Changes in water colour across the wetland can indicate functioning if a noticeable contrast is apparent (Spencer et al., 1998). Water colour will indicate suspended particulates, including sediment but also algae. Increased clarity between inflows and outflows will roughly indicate settlement of particles within the wetland. It is a simple measure, but also subjective and does not quantify performance.
	Chlorophyll	Lab or *ins situ* probe	Indicates algal abundance (Kadlec, 2008, Greenway, 2010). In turn algae reflect the degree of vegetation cover with highest algal abundance typically in unvegetated zones with increased light penetration (McCormick et al., 1997). Algae are particularly sensitive to nutrients and can be a useful indicator of changing nutrient availability (Van Dam et al., 1998). Excessive algae (i.e. an algal bloom) indicates high nutrient concentrations and can lead to anoxic conditions (YSI, Moreno-Mateos et al., 2012). However, at other times algae contribute positively to processing via nutrient assimilation and photosynthesis (see Section 4.5.2). Hence, the relationship between chlorophyll in the water column and water treatment is not straightforward, and it will also vary seasonally (Wu and Mitsch, 1998).
Physical condition (see Section 5.6)	Vegetation cover	Aerial photography or ground-based (plots along transects)	Plant cover is a commonly used indicator of wetland function and a targeted % plant cover is often a key objective. Changes in cover are particularly useful in in early wetland life to track the survival of the initial planting and expected increase in plant cover (Spieles, 2005). Changes in cover also indicate the suitability of system hydrology (Aronson and Galatowitsch, 2008), although the response relationship lags by up to 3 years (Squires and Valk, 1992). Changes in plant cover have been directly related to changes in water quality (Mitsch et al., 2005b). Plant cover indicates zones of organic matter input to the substrate and to microbial processes (Aronson and Galatowitsch, 2008, Cole, 2002), although its relationship with function is indirect and not always clear or useful (Cole, 2002, Matthews and Endress, 2008). Specific categories, such as % cover of hydrophytic plants (Greenway, 2010, Spieles, 2005) or weed cover can indicate the state of the wetland environment and can be compared against quantifiable objectives. However, as cover increases additional performance indicators become necessary (Shuman and Ambrose, 2003, Kihslinger, 2008). In addition, systems with similar cover can differ markedly in other key parameters such as species composition (Campbell et al., 2002). Hence, cover is a useful indicator but in conjunction with other performance measures. Further discussion is provided in Section 5.6.1.
	Vegetation biomass – above-ground	Sampling and extrapolation	Plant biomass indicates productivity and positively correlates with nutrient removal efficiency (Yu et al., 2012). However, biomass can vary in response to nutrient concentrations and water level fluctuations (Moreno-Mateos et al., 2012, Knight, 1992). Biomass indicates carbon provision from plants to drive microbial processes and so is linked to nutrient retention (Wetzel, 2001) and the assimilation and storage of nutrients in plant tissues. Biomass changes substantially across the seasons and needs to be estimated multiple times across a growing season to determine the magnitude of change (Cole, 2002, Gottschall et al., 2007). Also note plant cover does not necessarily = biomass (Drinkard et al., 2011). Plant biomass is a useful indicator of wetland function but it is more challenging to determine than plant cover. Seasonal variation and

			belowground biomass are frequently excluded from its measurement.
	Vegetation distribution	Aerial photography or ground-based	The configuration of vegetation within the wetland strongly influences hydraulic efficiency and therefore water treatment. This measure is readily assessed visually, although it is subjective and difficult to quantify. If the consistency of vegetation bands across the wetland width, or conversely, the patchiness of vegetation, can be characterised this is a vital indicator of performance.
	Vegetation species composition / diversity	Ground-based sampling (e.g. plots along transects) and photographic recording	Species composition can indicate wetland function, reflecting the habitat available for microbes, biofilms and fauna (Fennessy et al., 2007). In particular, plant type (i.e. emergent, submerged, free-floating) or functional group (e.g. obligate wetland, facultative wetland or terrestrial) can indicate wetland functions (e.g. carbon provision, assimilation) more successfully than considering individual species (Drinkard et al., 2011, Español et al., 2013). In addition, the composition of the plant community in downstream receiving waters, particularly in relation to the upstream community, may indicate water quality changes within the wetland (Environment Heritage and Local Government Ireland, 2010). The dominance of certain species can indicate either nutrient-deficiency or eutrophic conditions depending upon species preference and local conditions (Kadlec, 2006). However, species composition is expected to change substantially over time and this occurs independently of water quality processing, which can remain consistent (Kadlec, 2006, Spieles, 2005). In addition diversity is not a reliable indicator of wetland function. This is further discussed in Sections 4.3.3 and 5.6.3. Hence, species composition is most useful when considered in aggregate terms i.e. plant types or functional groups, and should be accompanied by other indicators of wetland performance. The influence of different plant types is discussed in Section 4.3.2.
	Vegetation density	Ground-based sampling e.g. plots, or assessment from aerial images	Similarly to biomass, vegetation density will indicate the extent of microbial processing and plant assimilation. Vegetation density will also closely interact with hydraulic efficiency. Very dense vegetation may be detrimental to aerobic processes and facilitate short circuit development. The number of species per unit area may also be a useful indicator for comparison between wetland systems (Spieles, 2005) and to indicate the success of planting. Vegetation density is only useful in conjunction with other measures of the extent of vegetation. The influence of vegetation density on hydraulics is discussed in Section 4.3.4.
	Spatial heterogeneity	Ground-based sampling or categorisation of aerial photographs	This is reported to be a potential indicator of wetland function (Spencer et al., 1998). High heterogeneity can indicate the range of conditions available for biogeochemical processing and increased hydraulic efficiency. However, it is difficult to characterise or quantify as a parameter. Mapping configuration, cover and plant type may substitute for a measure of spatial heterogeneity.
	Bioassays and rapid assessment methods	Range of commercial toxicity tests available e.g. Microtox®, E-screen, Ames test, umu test method	Bioassays and rapid assessment tools indicate toxicity of a combination of compounds. As they measure a toxicity response, they can indicate the presence of compounds that are unknown, possibly not otherwise detectable by laboratory analysis (Poulsen et al., 2011), can detect micropollutants and the influence of a mixture of contaminants within a system (Allinson et al., 2012). These methods are currently being developed for greater application and require further research. More information can be found at from

		Rapid assessment tools include gas chromatography-mass spectrometric techniques	the CAPIM (Centre for Aquatic Pollution Indentification and Management) research group, among others.
	Plant health	Visual signs or accumulated H_2O_2 (stress response)	Plant stress response can be measured by H_2O_2 (Shelef et al., 2013) or visual indicators of yellowing, spotting, increased senescence or poor growth. Poor plant health can result from inappropriate hydrology, although a lag effect (Squires and Valk, 1992) means signs of die-back do not always provide effective early-warning. Irrespectively, poor plant health must be investigated to determine the cause (Wetlands International, 2003).
	Substrate characteristics e.g. TN, gravimetric soil moisture, TOC, soil organic matter, bulk density or redox potential, sediment accumulation	Lab analysis, *in situ* sampling or sediment depth measurements	The substrate is not typically measured, but it is a key contributor to water quality processes (Fennessy et al., 2007, Kihslinger, 2008). When grouped together into soil categories, characteristics of the substrate serve as useful indicator of denitrification potential (although high spatial variation in denitrification within sites should be noted) (Ahn and Peralta, 2012). As the substrate develops characteristics of a mature wetland soil (i.e. high TOC, TN and low bulk density) nitrogen processing capacity increases (Wolf et al., 2011a). Properties of the substrate can also dictate vegetation characteristics, including cover, diversity and biomass (Dee and Ahn, 2012). Overall, changes in sediment characteristics represent slow changes in wetland function/water quality over time (Mitsch et al., 2005b). Measurements of sediment accumulation or erosion indicate sediment settlement and suspension respectively, and provide direct measures of wetland function (National Research Council, 2001, Carpenter and Lodge, 1986). Redox potential indicates conditions for aerobic or anaerobic processing, but it may not clearly relate to ecological functions (Mitsch et al., 2005b). If sampling is feasible, sediment characteristics are closely related to function, particularly for nitrogen removal, and sediment accumulation indicates sediment and phosphorus retention.
	Substrate colouring/ mottling	Collect sample cores, photograph and visual assessment	Mottling and orange colouring indicates hydric wetland soils (Turner et al., 2001) which develop over the initial years of wetland life alongside the development of wetland function (Mitsch et al., 2005b). Orange mottling results from oxidised iron on root surfaces and indicates root oxygen release. This suggests the occurrence of coupled nitrification-denitrification and organic matter decomposition under aerobic conditions (Brix, 1997).
	Similarity in design to other systems	Visual assessment of configuration, treatment zones, size from aerial photographs and site plans	Features of wetland design can indicate their performance. Similar designs have been found to have relatively consistent water quality improvement (Kuehn and Moore, 1995). Wetland area relative to its catchment area is correlated with long-term removal of nitrogen and phosphorus (Carleton et al., 2000).
Secti	Fauna in general (vertebrates	Sampling (e.g. trapping) and visual	Wetland fauna are not typically directly involved in pollutant treatment, but play other beneficial roles and can visually indicate system health or poor function (Hammer, 1992, Environment

	& invertebrates)	observations noted during field visits (include notes on conditions in wetland, date, time of day etc.)	Heritage and Local Government Ireland, 2010). High faunal abundance may indicate high nutrient availability and high DO in the water column (Knight, 1992). However, it should be noted that some species tolerate high nutrient levels (Greenway, 2010, Hammer, 1992). In addition, faunal abundance may not be coupled to plant community condition which can lag in development or recovery from disturbance (Moreno-Mateos et al., 2012).
	Indicator species	As above	Some species can act as indicator species due to a high sensitivity to water quality, for example trout or salmon (Environment Heritage and Local Government Ireland, 2010) or amphibians (Kihslinger, 2008). Conversely, some species can thrive in poorly functioning systems (Hammer, 1992). For example, carp can promote sediment re-suspension. Hence, any relationships need to be relevant in the local environment and amongst local species.
	Downstream fauna	As above	The composition of fauna in downstream receiving waters, possibly relative to upstream, may indicate wetland function and a change in water quality (Environment Heritage and Local Government Ireland, 2010)

5.4 Hydrology

Water level data provides valuable information that will indicate if the wetland hydrology reflects design. The depth, duration and frequency of flooding can be compared to the desired retention within the wetland and vegetation depth tolerance.

Some aspects of wetland hydrology can be directly and economically measured through the logging of water levels and calculation of inflow and outflow. Measurement of inflow and outflow is useful for comparison against flow modification objectives, but water levels within the wetland are highly relevant to vegetation, biochemical processes and retention time. In addition, all aspects of wetland monitoring should be recorded with reference to water level at the time (Woods et al., 2004).

Water level logging over time will provide data on water depth and inundation duration and frequency within wetland zones. Inundation frequency curves and time series of water levels will identify the suitability of the water regime in each zone to support the desired vegetation (Greenway et al., 2007, EDAW, 2008) and indicate if adjustment is required. Vegetation death will result from excessive inundation (EDAW, 2008, Dugdale and Ede, 2013) or even a single isolated event (Greenway and Polson, 2007). However, identifying inappropriate hydrology is complicated by a lagged response by plants – while there may be indications of a response to flooding (e.g. reduced biomass, stem elongation), plants may survive for 3 years or more before declining (Dugdale and Ede, 2013, Squires and Valk, 1992). Plant species will also display contradictory responses to different water regime adjustments (Webb et al., 2012, Dugdale and Ede, 2013). This highlights the importance of monitoring hydrology and making timely adjustments to inflow or outflow structures as required. The characteristics of an appropriate hydrological regime to support vegetation are discussed in Section 4.7.3.

5.5 Water quality

These in situ measurements indicate conditions that influence processing, but unless extreme conditions are observed, may provide no clear indication of wetland performance. Interpretation of measurements and relationships with target pollutants are complex. Hence, in situ proxy measurements of environmental conditions are not recommended on a widespread basis. Instead, any sampling program should be carefully designed to maximise useful output and occur alongside measurement of parameters.

In situ measures of water quality such as dissolved oxygen, pH, temperature and conductivity indicate the prevalent biochemical conditions for processes (Van Dam et al., 1998, Kayombo et al., 2005). Their measurement is cheap relative to direct measurement of contaminant concentrations. Isolated measures may not provide useful information unless conditions are extreme, such as non-neutral pH, high salinity or anaerobic conditions. However, when sampled over space and time these parameters may reveal patterns indicative of functioning. For example, temperature differentials can indicate short-circuiting and changes in turbidity across the wetland may indicate settlement or resuspension. However, a high level of expertise is required to interpret and relate measurements to the target pollutants. In addition, many parameters vary diurnally and seasonally. Hence, unless objectives and the sampling protocol are well defined, investment in measuring these parameters could yield little valuable insight into wetland function.

5.6 Physical condition

Measurement of a suite of characteristics is recommended including vegetation cover, distribution, type, health and substrate development.

Measures of system physical condition include assessment of vegetation, substrate and wetland configuration characteristics. The most promising indicators appear to be vegetation cover, vegetation distribution, vegetation type, bioassays and other rapid assessment methods under development, and characteristics of the substrate as it develops. Observation of plant health is also vital, but may not provide an early warning of the problem.

The following sub-sections provide further information on the most commonly monitored parameters – vegetation cover, density and species composition.

5.6.1 Vegetation cover

In summary, plant cover appears a relevant, useful and practical indicator, but it should be assessed in light of vegetation distribution, which has a crucial role for effective hydraulics (Sections 4.3, 4.4 and 4.7) and also plant type, which influences processing efficiency (Section 1.1.1).

Plant cover, or some subset of cover (e.g. % herbaceous cover), is one of the most commonly used indicators of wetland function (Cole, 2002, Kihslinger, 2008). In light of the multiple beneficial roles of plants in constructed wetlands (Section 4.3.1), it is reasonable to expect plant cover to infer nutrient retention. Effective nutrient removal is related to plant biomass (Yu et al., 2012) and multiple studies relate vegetation presence or cover to effective nitrogen removal (Kadlec, 2008, Zhu and Sikora, 1995, Yu et al., 2012, Ruiz-Rueda et al., 2009). However, there are some contradictory reports for select nitrogen species and noise in the relationship (Kadlec, 2008). Ammonia removal may be improved by reduced cover when microtopography is introduced, as a result of greater oxygen availability (Thullen et al., 2002, Thullen et al., 2005).

High cover may also indicate effective sedimentation or reduced resuspension (Bodin et al., 2012, Kadlec, 2008, Wong et al., 2006), although the relationship is likely to be less straightforward and consistent, given the benefits of deeper open water for sedimentation, the potential for short-circuiting around vegetation and the potential for re-suspension in shallow zones (Fennessy et al., 1994, Braskerud, 2001). Additionally, vegetation may only be beneficial to sediment removal up to a point – under high hydraulic loading it is not significant (Brueske and Barrett, 1994) or once cover exceeds approximately 50% the influence hydraulic and sediment loading may dominate (Braskerud, 2001).

The extent of plant cover can indicate the distribution of soil organic matter and TN within the substrate (Bai et al., 2005). Plant cover may be used to infer the accumulation of organic matter, which in turn indicates the development of wetland function (Ahn and Dee, 2011).

Cover may also correlate with macrophyte productivity, which positively influences many wetland functions (van der Valk, 2012), and the provision of plant-derived carbon, which promotes denitrification (Ingersoll and Baker, 1998). Plant cover will also influence physiochemical conditions in the water column, including temperature and light availability (Carpenter and Lodge, 1986).

Despite the importance of vegetation to wetland function (Kadlec et al., 2005, Kadlec, 2008), the extent of vegetation cover should not be used as a stand-alone indicator of wetland performance. The relationship between the retention of total nitrogen or organic matter with plant cover can be noisy with no clear pattern evident (Kadlec, 2008, Akratos and Tsihrintzis, 2007). In such cases plant type or vegetation configuration may be important (Cole, 2002, Akratos and Tsihrintzis, 2007, Jenkins and Greenway, 2005). Patchy cover may be more undesirable than a configuration with lower total cover but even distribution across the wetland (Fennessy et al., 1994).

5.6.2 Vegetation density

Very dense vegetation can negatively impact upon water treatment due to shading, litter accumulation, a reduction in effective wetland volume and reduced oxygen levels. However, up to a point dense vegetation provides benefits to water treatment including high carbon to fuel microbial processes, high assimilation and transpiration rates and reduced flow velocity. Hence, vegetation density can provide a useful indication of wetland function, but is not likely to be informative on its own without an indication of total cover and configuration.

Despite the multiple positive benefits of vegetation, mature dense vegetation can negatively influence treatment processes. This results from reduced growth, high levels of shading (which reduces photosynthesis and dissolved oxygen but may foster anaerobic conditions for denitrification) and litter accumulation, which leads to lower productivity within the water column, restricted nitrification and photolytic reactions, slower decomposition and preferential flow path formation (Thullen et al., 2002, Thullen et al., 2005, Weisner and Thiere, 2010, Bachand and Horne, 1999). Dense vegetation also reduces the effective treatment volume of the wetland at low flows (Bodin, 2013, Jadhav and Buchberger, 1995) and may increase the export of organic nitrogen (Thullen et al., 2002). It can also promote short-circuits if cover is not consistent across the width (Fennessy et al., 1994), yet other reports suggest short-circuits are more prevalent in low density vegetation (Dierberg et al., 2005).

On the other hand, the benefits of dense vegetation include high carbon to promote anaerobic conditions, drive denitrification and other microbial processes, high transpiration to draw contaminants towards the rhizosphere and high surface area to support biofilms (Thullen et al., 2002, Weisner and Thiere, 2010). High density can benefit TSS, TN and TP removal (Bodin, 2013, Yu et al., 2012, Dierberg et al., 2005) and sorption of herbicides/pesticides in conjunction with shallow water (Lange et al., 2011). At high flows dense vegetation can also slow velocity and increase the retention time (Jadhav and Buchberger, 1995, Brueske and Barrett, 1994).

5.6.3 Species composition

Changes in plant species are an inherent characteristic of wetlands and to be expected in constructed systems. However, effective water treatment processes can continue independent of changes in vegetation composition. Hence, species composition is not a reliable indicator of wetland performance, unless grouped into plant type or functional group.

The initial plant selection cannot be expected to survive intact over the longer term, nor may it be suitable for it to as conditions change (Kadlec, 2006). Numerous studies detail substantial changes in vegetation composition over time in constructed wetlands (Kadlec, 2011, Mitsch et al., 2005b). These may result from variable seasonal inflows, nutrient accumulation and, as with all ecosystems, the forces of competition, succession, disturbance and invasion act on wetland plant communities (Kadlec, 2011). Disturbance from construction may dictate plant composition in early wetland life, before its effects dampen and successional processes act (Kadlec, 2006). However, wetlands can remain functional for nutrient reduction despite changes in vegetation composition over the long-term (Kadlec, 2009). Hence, constructed wetland design and monitoring programs must expect plant communities to be dynamic; changing spatially and temporally over a range of scales (Brock and Casanova, 2000, Aronson and Galatowitsch, 2008).

5.7 Other surrogates

Fauna can provide an indirect signal of water quality either through abundance, species composition or select indicator species. However, relationships must be interpreted with care and with reference to local conditions.

Although not generally directly involved in wetland processing, wetland fauna can indicate water quality through abundance or species composition (Hammer, 1992, Knight, 1992). If species with high sensitivity to water quality can be identified, these can serve as useful indications of system state (Kihslinger, 2008, Hammer, 1992). However, these relationships will be specific to local conditions and species. In addition, wetland function can be directly influenced by high numbers of certain species which are detrimental to vegetation survival, such as birds or carp (Greenway and Polson, 2007, Morris and Papas, 2012).

6 Predicting likely lifespan and maintenance requirements

6.1 Factors influencing lifespan

Multiple factors contribute to wetland lifespan including loading, wetland size, the contribution of permanent removal pathways and the survival of healthy vegetation across the long-term. There are promising examples in the literature of sustained long-term function in constructed wetlands for nutrient removal, but some wetlands export sediments and nutrients before their nominal end-of-life.

Many constructed wetlands have a nominal life of 20 to 25 years (Hammer, 1992, Shutes, 2001). However, while this timeframe is frequently stated, it is rarely justified. This may be because data on long-term wetland performance is extremely limited (Koch et al., 2013). Long-term and large-scale experiments are difficult to conduct, particularly with robust statistical design (i.e. replication) (Mitsch et al., 2005b). However, extrapolating results from laboratory experiments across spatial and temporal scales is fraught with uncertainty (Mitsch et al., 2005b).

Factors with critical influence on wetland lifespan include:

- **Loading and wetland area** – contaminant loading (in terms of amount per unit area per year) is vital to long-term performance (Mitsch et al., 2005b). This may form a key difference between stormwater and wastewater treatment wetlands. However, a performance review across multiple stormwater treatment wetlands found removal rate constants on an areal basis for phosphorus and nitrogen species were similar with reported values for wastewater treatment wetlands (Carleton et al., 2000). Further, the same study found wetland surface area relative to its catchment area is indicative of its long-term performance for nitrogen and phosphorus removal.
- **Extent of permanent removal** – is particularly relevant to the long-term sustainability of nitrogen removal, i.e. the contribution from denitrification relative to temporary attenuation processes (Koch et al., 2013, Kadlec et al., 2005). The contribution of plant uptake relative to microbial processing (and particularly denitrification) is central to long-term performance and debated within the literature (Gottschall et al., 2007, Stottmeister et al., 2003{Bachand, 1999 #201)}.

The division can depend upon nutrient loading and the plant growth phase (Kadlec, 2008, Vymazal, 2007, Brix, 1994a). The benefits of plant harvesting for permanent removal are also part of the debate (Stottmeister et al., 2003, Gottschall et al., 2007).

In the case of phosphorus, which has no permanent removal pathway, accumulation over time is even more problematic, and depending upon loading, can be the critical factor dictating wetland lifespan (Environment Heritage and Local Government Ireland, 2010).

- **Initial planting** – while the vegetation composition and cover can change significantly over time, the initial state (in terms of planted versus unplanted) can have long-term ramifications on the system. For example, Mitsch et al. (2005b) found a planted versus unplanted system did initially converge towards a state of similar vegetation, but subsequently diverged again, which implications on productivity, water quality, carbon provision. Weisner and Thiere (2010) similarly found the initial planting had long-term consequences on system function.
- **Plant establishment** – successful establishment is critical to ensure resilience in the vegetation in the long-term (Dugdale and Ede, 2013).
- **Hydrological regime** – an appropriate regime is closely linked to plant growth and survival (Webb et al., 2012, Dugdale and Ede, 2013).
- **Feedback between vegetation cover and substrate condition** – both vegetation and the substrate influence each other; plants provide organic matter and reduce erosion/resuspension, while the substrate supports plant growth. If plant cover is sparse and erosion commences, re-establishing plant cover will be even more challenging or impossible if insufficient topsoil is available. Ensuring adequate topsoil depth, plant cover and protecting the wetland from high flow events is important to prevent erosion and vegetation loss.
- **Accumulation of toxic compounds** – for example, metals or organochlorine compounds may be particularly prone to preservation and accumulation within the system (Knight, 1992). Some of these may bioaccumulate (e.g. mercury, lead, DDT and dioxins) or build up within the plants and soil (Knight, 1992).

What are the implications of maturity on wetland processing? Of the limited data set, a study by Mitsch et al. (2005b) found field-scale wetlands treating high nutrient river water remained effective for nitrate and phosphate removal after ten years of operation. However, sediment retention had declined to the point of net export, which led to poor retention of TP and TSS. Another study, conducted in an 8-year-old wetland by Gottschall (2007) also indicated that wetland function for nutrient removal from dairy wastewater can continue under long time periods of high loading (in this case treating agricultural wastewater). Using 30 years of phosphorus removal data in a peatland treating wastewater, Kadlec (2009) noted sustained performance despite structural changes to vegetation and shifting removal pathways. Similarly, Walker (1995) observed consistent retention of P in a natural peatmarsh over 26 years, sustained across changed vegetation composition and attributed to long-term incorporation into the peat. These results are consistent with the reported long-term nature of natural wetland function (Hammer, 1992).

There is an argument that at some stage nutrient loading will detrimentally impact upon the wetland and storages will reach saturation, likely near the inlet zone first (Howard - Williams, 1985). However, contrary to initial predictions, Kadlec (2009) were surprised to find saturation did not occur – a zone of high phosphorus concentrations was present and initially grew rapidly, but slowed and reached a dynamic equilibrium across the seasons.

The efficiency of wetlands over time may be contradictory, subject to multiple driving forces other than age, and contaminant-specific. Some authors note higher nutrient process rates in early life (Dunne et al., 2013), while others observed increasing retention of suspended solids across the first 5 years, attributed to lower resuspension (Mitsch et al., 2005a) or an increase in nitrogen process rates (Wolf et al., 2011a). There is also some suggestion the development of wetland function can be independent of system maturity, instead dictated by soil and plant characteristics (Dee and Ahn, 2012).

6.2 Sediment hotspots

Wetland management is facilitated by an understanding of sediment accumulation zones. Deposition will be a function of flow, vegetation and particle characteristics. Sediment will accumulate in zones of low velocity and high retention, and heavy particles, along with the associated contaminants, will settle first. However, the dynamics of re-suspension and erosion must not be ignored to determine net retention within the wetland.

Identifying contaminant accumulation zones and monitoring the build-up can be a valuable tool in wetland management. Beyond a point, contaminant concentrations will exceed the legislated thresholds/'background concentrations', which restricts reuse options and increases the cost of disposal (MacMahon, 2013a, Weinstein, 2008).

The accumulation of sediment, and its associated contaminants, across the wetland will depend upon flow velocity, water depth, vegetation, particle size and chemical characteristics, each of which may influence settling, flocculation and filtration processes (Leira and Cantonati, 2008, Verstraeten and Poesen, 2000). In general, heavier particles such as sand will settle nearer to the inlet relative to clay particles (Weinstein, 2008) and the highest accumulation will occur off the main flow path (Storm Consulting, 2013). In the same way, particulate-bound contaminants may distribute differentially according to their affinity to adsorb to mineral or organic particles. For example, Weinstein et al. (2008) found high concentrations of aluminium and cadmium near the inlet associated with heavier sand particles, but copper and zinc in the centre where clay tended to settle. In addition, contaminant concentrations near the surface of accumulated stormwater sediment deposits in Melbourne Water's sediment ponds are reported to reflect the peak concentration, which facilitates sediment testing (MacMahon, 2013b).

Processes also act to re-suspend, erode and transport sediment and these are also dependent upon hydrology, particularly variation in water level and velocity (Leira and Cantonati, 2008). Relationships may be unpredictable. For example, higher sedimentation rates have been reported at high flows relative to low flows (Fennessy et al., 1994), but this may reflect system-specific hydraulics and loads rather than sediment concentrations.

Vegetation also plays a fundamental role in minimising sediment re-suspension (Braskerud, 2001). When vegetation is present, a higher velocity is required to erode sediments. A positive feedback can develop if erosion reduces the capacity for vegetation to survive, and once erosion has commenced the sediments become increasingly susceptible to further erosion (Storm Consulting, 2013). Hence, none of the associated processes are static in time and either accumulation or erosion will alter flow paths and system bathymetry over time (Storm Consulting, 2013). The relationship between plant distribution, sediment and flow is cyclical and expected to change with time. Vegetation may not always benefit sedimentation – particularly if the shallow vegetated zones are prone to disturbance by wind, birds or fish; in this case deep water zones may facilitate greater sedimentation (Fennessy et al., 1994). It should also be noted that processes may change with particle size and hydraulic loading. Fine particle sedimentation and re-suspension may not be particularly sensitive to vegetation, instead responding to hydraulic load (Braskerud, 2001). Others suggest the role of vegetation in sediment dynamics is greatest at low hydraulic loading (Brueske and Barrett, 1994).

7 The potential impacts of modelling uncertainty on wetland design and performance

7.1 Introduction

The objective of this study is to investigate uncertainties associated with the wetland treatment node in MUSIC, particularly in relation to the prediction of nitrogen removal for various sizes and design configurations.

There are many steps in the complete design of a stormwater wetland (see for example: Melbourne Water, 2005), but the process typically begins with approximate sizing and configuration using the Model for Urban Stormwater Improvement Conceptualisation (MUSIC). MUSIC cannot (and thus should not) be used for detailed configuration but it can show the effect of basic design parameters such as the wetland's:

- Dimensions (surface area, extended detention depth, inlet pond volume)
- Detention time (governed by equivalent pipe diameter and extended detention volume or by a custom outflow and storage relationship)
- Re-use properties
- Hydraulic configuration (Figure 8).

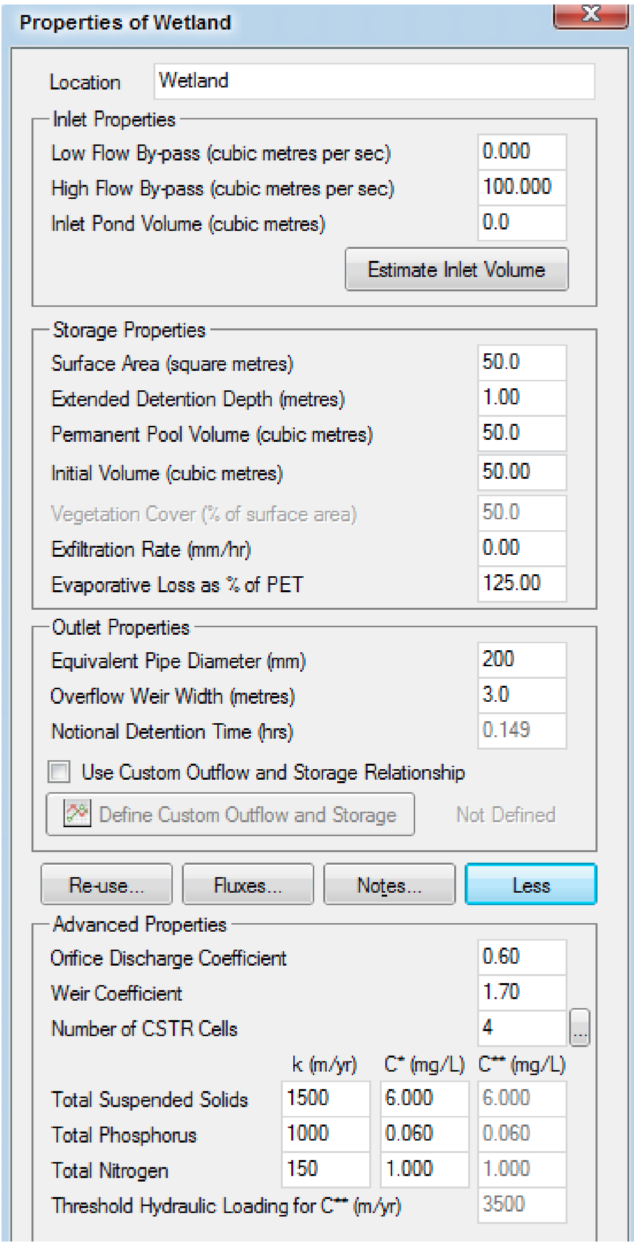


Figure 8 MUSIC (v6) wetland configuration dialogue box.

There are many areas of potential uncertainty between the conceptual modelling of a wetland and the actual performance of the system that is finally built. Such uncertainties include:

1. Uncertainties in the appropriateness of the model's default treatment performance parameters (k and C*), which describe, respectively, the pollutant decay (or removal) rate and the background concentration. The default parameters may not be appropriate for a given wetland, due to a range of influences.
2. Potential discrepancies between the hydraulic efficiency of the constructed system and the system as it was modelled (as expressed by the number of Continuously Stirred Tank Reactors (CSTR cells in MUSIC).

3. Potential differences between what was modelled, designed (detail design) and what was actually built.

The third of these uncertainties can be resolved by appropriate procedures to ensure that what is built accurately reflects the original (and thus modelled) intent and design.

However, the first two uncertainties relate specifically to whether the default parameter values in MUSIC can be used with confidence in the design of stormwater wetlands in Melbourne. The aim of this study is therefore to assess the potential impacts of differences between the default values of k (pollutant removal rate), C* (background concentration) and the hydraulic efficiency (N_{CSTR}) and those which might apply to a given wetland in practice. While the MUSIC User Manual provides default values for these three parameters, there is a need to determine how much these parameters might vary between individual wetlands and to assess the potential consequences of this variation on differences between predicted and actual wetland performance.

It is important to note here that this study is limited to investigating the uncertainties relating to the use of the wetland node in MUSIC. Other potential sources of error or uncertainty also exist, such as in the rainfall-runoff model (and the choice of rainfall data) and the pollutant generation from the catchment (again, the default pollutant concentrations may differ from those of the specific catchment being modelled).

7.2 Overview of MUSIC's representation of pollutant removal in wetlands

MUSIC represents stormwater treatment using the Universal Stormwater Treatment Model (USTM). This combines a simulation of hydraulic efficiency through the treatment system (defined by parameter N_{CSTR}) with a first-order decay model which describes the change in contaminant concentrations exponentially with time (at a rate k), towards a background concentration (C*). First order decay provides a reasonable approximation to treatment by sedimentation, biological processes, and chemical reaction, although the underlying processes are very different. The model has been calibrated for TSS, TP and TN, which differ in the degree to which physical, chemical and biological processes play a role in treatment. One possible source of uncertainty is that these biological and chemical processes are not explicitly represented by the model; instead, the lumped effect of all processes is taken into account by the calibration of the k, C* and N_{CSTR} parameters.

Wetlands are represented in MUSIC as shallow systems with dense vegetation, while ponds are characterised by deeper water, with fringing vegetation and a central open water zone; each has their own recommended default N_{CSTR}, k and C* values, derived from empirical studies (details are provided in Appendix F of the MUSIC User Guide).

7.2.1 Stormwater treatment within a wetland

Key model inputs that describe a particular treatment system include the following.

Storage-discharge relationship - this is utilised by MUSIC to route flows through the treatment system and describes the discharge rate as a function of the depth of water in storage. The relationship may be supplied directly as a table, or alternatively, users can input storage dimensions and outlet physical data and allow MUSIC to calculate the S-Q relationship; it is likely the majority of MUSIC users take this approach (entering data about the surface area, extended detention depth and the equivalent outlet diameter).

C* – is the background or equilibrium concentration for a contaminant, which is assumed to be relatively consistent across the inter-event periods. C* is a measure of the proportion of pollutants not typically removed within the wetland (or other treatment measure). If inter-event water quality data are not available, this may be determined from the outlet concentration monitored at the very beginning of an event, before inflow has first reached the outlet. Low C* reflects effective wetland treatment, but values of zero are not realistic given the inevitability of some resuspension or release. As noted in the MUSIC user guidelines, C* is assumed to be constant, but in reality is likely to vary possibly with seasons, system maturity and flow velocity; all of which may influence particulate re-suspension and potential recycling of nutrients within the water body.

k – the rate at which contaminant concentrations move exponentially towards C*. Calibration of the k value requires concentrations of TSS, TP and TN measured at short time-steps during a storm event. Default values of k and C* are shown in Table 3, below.

Table 3. Default k and C* values for MUSIC (Source: CRC, 2012).

Pollutant	TSS		TP		TN	
Parameter	k	C*	k	C*	k	C*
Default value	1500	6.0	1000	0.06	150	1
Default range	500 - 5,000	5-6	300 - 2,800	0.03 - 0.09	50 - 500	0.7 - 1.3

N – the number of Continuously Stirred Tank Reactors (CSTRs). CSTRs are used to model the degree of mixing or turbulence as flow moves through the wetland. The flow dynamics will be influenced by wetland configuration (e.g. l:w ratio) and components that may act to either disperse or, conversely, channel flows (e.g. vegetation density and layout, islands, submerged bars). N can be fitted to the shape and lag of the pollutograph. A high N approximates plug flow and high hydraulic efficiency (optimal for contaminant removal), while low N represents a high degree of mixing or short-circuiting with the rapid passage of some inflow through the wetland to the outlet (Persson et al., 1999). A significant uncertainty in modelling wetlands is that while the default value of N is used, the constructed wetland may be quite a different shape and thus have very different hydraulic efficiency than the default value (N=5).

7.2.2 Uncertainty in parameterisation

The default values of C* and k are provided in ranges in the User Guide for each treatment node. These default ranges are based upon a combination of theory and limited calibration studies (both of which are specified in Table F-5 in Appendix F of the MUSIC User Guidelines).

The theoretical estimation of k and C* utilises sedimentation equations, which require input of particle size distribution (typical values for Australia are default but use of local data is recommended if available). The MUSIC User Guide notes that theoretical values derived from these equations are not conservative, representing simplified conditions without turbulence and a uniform velocity distribution. While turbulence and velocity heterogeneity can be reflected in the parameter N, the theoretical values of k should be considered an upper-limit, most critically for settling of fine particles. It is for this reason that the preferred approach to calibration of k and C* is through the use of short timestep concentration data for each of TSS, TP and TN. Given this, it is important to note the limited extent of data for calibration – extensive data sets are simply not available and two stormwater ponds (Blackburn Lake and Lake Annan) and one large constructed wetland (Hampton Park) were used to develop default values of k and C* for ponds and wetlands. Since this time, further calibration has been undertaken by Duncan and Fletcher (2006) and Watson (2014), but calibrated wetlands remain scarce. The findings of these studies have been summarised below and are used to provide insights into the degree of uncertainty or confidence in the default parameters.

7.3 Methods

We investigated the potential impacts of uncertainty in the three principal calibration parameters in the MUSIC stormwater treatment model (k, C* and N_{CSTR}). The investigation was undertaken using two separate approaches.

Calibration study: Firstly, we drew on two calibration studies (Duncan and Fletcher, 2006, Watson, 2014) which focus on the k, C* and N_{CSTR} values. We use these to examine the range of values that these parameters may span and to investigate the potential impacts on predicted versus actual treatment performance. In the Duncan and Fletcher study, the MUSIC parameters N, C* and k were calibrated using monitoring data from three systems – Blackburn Lake, Hampton Park (both in Melbourne) and Lake Annan (Sydney). A further calibration – using the Ruffey Creek wetland in Doncaster (east of Melbourne CBD) – was undertaken by Watson (2014) under guidance from Fletcher and Duncan. A copy of Watson's report is attached as Appendix A.

Watson undertook a rigorous approach to calibration, first calibrating the source node (using observed rainfall and flow data, as well as measured event and dry weather pollutant concentrations). The physical properties of the Ruffey Creek wetland used in calibrating the model are shown in Table 4.

Cell	Surface Area (m^2)	Permanent Pool depth (m)	Permanent Volume (m^3)	Extended Detention depth (m)	Extended Volume (m^3)
Inlet zone	753	1.2	499	0.70	751
Ephemeral	239	0.5**	119**	0.55	510
Macrophyte	1272	0.54 (av.)	687	1.00	1660
Total	**2264**		**1305**		**2921**
** Refers to the low-flow channel through the Ephemeral Cell					

Inlet Properties

Low Flow By-pass (cubic metres per sec)	0.000
High Flow By-pass (cubic metres per sec)	1000.000
Inlet Pond Volume (cubic metres)	499.0

Estimate Inlet Volume

Storage Properties

Surface Area (square metres)	2264.0
Extended Detention Depth (metres)	1.00
Permanent Pool Volume (cubic metres)	1305.0
Initial Volume (cubic metres)	1305.00
Vegetation Cover (% of surface area)	50.0
Exfiltration Rate (mm/hr)	0.00
Evaporative Loss as % of PET	125.00

Table 4. Physical properties of Ruffeys Creek wetland and their application for parameterisation of the MUSIC wetland dialogue box (Source: Watson, 2014).

Having calibrated MUSIC to nine events in terms of hydrographs and pollutographs through Ruffeys Creek wetland, he then undertook a sensitivity analysis, to understand the potential difference between the calibrated treatment parameters and the default parameter values in MUSIC. It should be noted that Ruffeys Creek wetland was found to have an average detention time of only 4.3 hours, which is roughly an order of magnitude lower than current design standards. This poor design (by today's standards) needs to be taken into account in interpreting the results.

We use the resultant parameter values from the two studies to derive a range of treatment scenarios with which to assess the potential impact of differences between actual and default k, C* and N_{CSTR} values on pollutant removal. The scenarios assess the impact of the lowest and highest values of k and C* from the Duncan & Fletcher (2006) and Watson (2014) studies with the default values on (i) removal performance and (ii) wetland area required to meet the current BPEM targets.

Sensitivity to the N_{CSTR} parameter: In a second component, we specifically undertake a sensitivity analysis to determine the potential impact of incorrect representation of a wetland's hydraulic efficiency (represented in MUSIC by the N_{CSTR} parameter). The motivation for this component of work is to identify whether Melbourne Water should consider adopting recommended values of the N_{CSTR} parameter for given wetland configurations, given that anecdotal evidence suggests most designers are accepting the default N_{CSTR} value, despite often sub-optimal hydraulic configuration. It is important to remember that unlike k and C*, N_{CSTR} is not a calibration parameter, but a parameter chosen to represent the physical layout and thus hydraulic efficiency of the wetland, according to the schema provided in MUSIC (Figure 9).

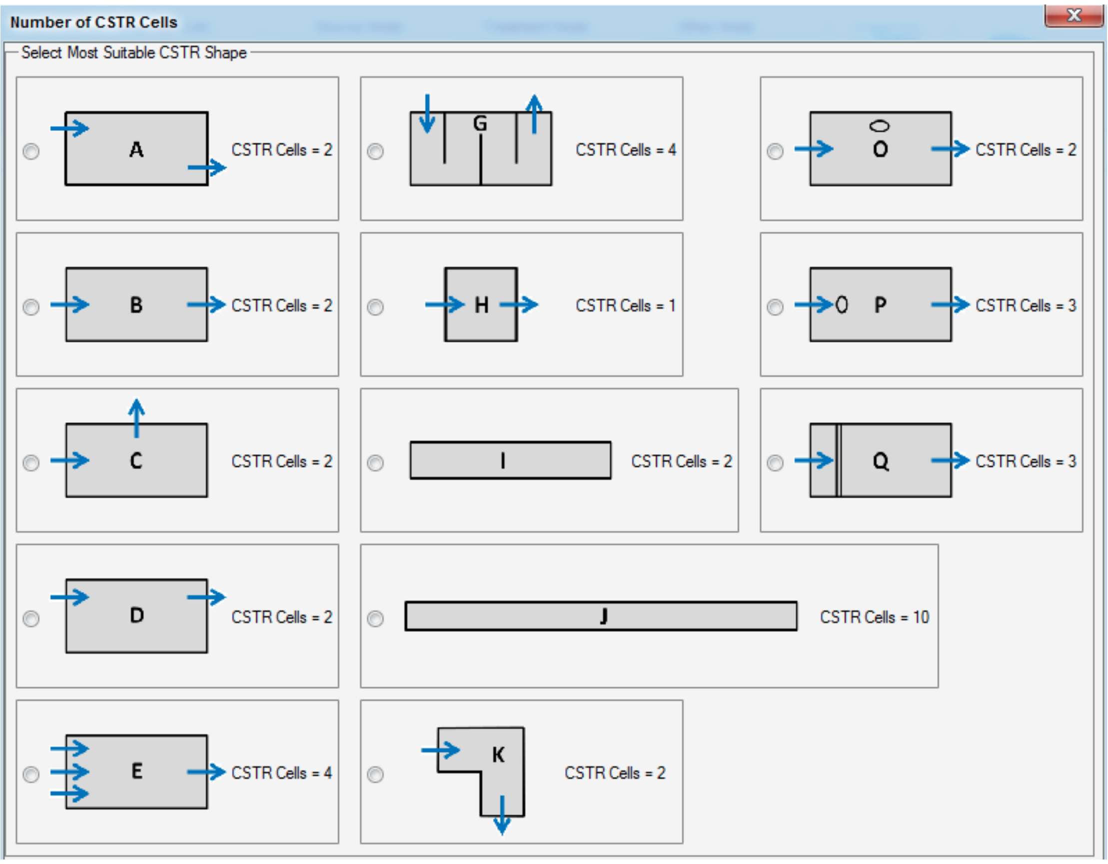


Figure 9. Selection of N_{CSTR} value based on wetland layout.

The uncertainty analysis for N_{CSTR} is undertaken by creating a wetland optimised to meet the (current) BPEM targets of 80, 45 and 45% reductions in the mean annual loads of TSS, TP and TN respectively. This optimal design is based on the default N_{CSTR} parameter value of 4. This value is then adjusted through the range down to 1, and the following results reported:

1. Removal of TSS, TP and TN if the wetland design parameters otherwise remain the same

2. The required increase in area for the wetland to still meet the BPEM targets with the reduced N_{CSTR} value.

7.4 Results and Discussion

7.4.1 Calibration study

Based on their calibrations, Duncan and Fletcher (2006) presented (Table 5) a summary of k, C* and N_{CSTR} values for their three investigated sites. It is apparent that the default k and C* values for TSS are <u>non-conservative</u> (in other words they over-represent the treatment observed in the three wetlands studied by Duncan and Fletcher), while the values for TP and TN are generally very close or conservative (ie. over-estimate treatment). An exception is Lake Annan (which has a k value of only 30 m/yr); this lower value may represent its open-water nature, being a lake. Given that TN is generally the limiting pollutant (ie. the final sizing of the wetland depends on TN removal), the non-conservative nature of TSS removal (k) and background concentration (C*) are unlikely to be of concern, but the potential for much lower N removal rates in open water (the Lake Annan case) is potentially more significant. The observation that TN removal rates in Blackburn Lake (another open water body) are quite high reinforces the potential for variability between individual systems. It is apparent that there may be variation in the k and C* values between winter and summer (data for seasonal analysis were only available at Blackburn Lake). MUSIC does not account for such variability, but this is unlikely to be of significant concern in estimating *long term mean annual loads*, since the default k and C* values represent the overall average.

Table 5. Derived values against default k and C* values (Source: Duncan and Fletcher, 2006)

Pollutant		**TSS**			**TP**			**TN**		
Parameter		**k**	**C***	**N**	**k**	**C***	**N**	**k**	**C***	**N**
Site	Blackburn Lake	200 - 900	10 (winter) - 20 (summer)	2	50 - 500	0.055	2	80 - 300	0.9 (summer) - 1.2 (winter)	2
Site	Lake Annan	600	5	1	80	0.02	1	30	0.6	1
Site	Hampton Park	300 - 400	5	10	100 - 200	0.03	10	100 - 200	0.9	10
MUSIC Default		1500	6	4	1000	0.06	4	150	1	4

Watson's (2014) study showed significant variation in the k values for TSS, TP and TN. For TN, values varied between 150 and 1000 m/yr, with an average of around 500 m/yr (considerably higher than the default value in MUSIC of 150 m/yr. The mean C* value of 1.2

mg/L is around the same value as the MUSIC default (1.0 mg/L). It is important to note that the N_{CSTR} value averages 2, but with 4 out of 7 events having a value of 1.

Table 6. Derived k and C* values from Watson (2014).

Event	TSS			TP			TN		
	k	C*	N	k	C*	N	k	C*	N
4	500	4	1	150	0.1	1	150	1	3
5	500	6	1	1000	0.06	1	1000	2	1
6	1500	6	1	1000	0.12	1	700	1.4	4
7	100	6	1	1000	0.06	1	150	0.8	1
8	700	6	1	1000	0.2	1	700	1	1
10	100	6	1	1000	0.06	1	150	1	1
11	1500	6	1	1500	0.1	1	700	1.2	4
12	300	6	1	500	0.075	1			
13	100	6	1	700	0.12	1			
14	1500	6	1						
Mean (μ)	680	5.8	1	872	0.099	1	507	1.2	2
s.d	599.63	0.63	0**	381.7	0.045	0	350.51	0.4	1.4
μ+ 1s.d	1280	6.4	1	1253.9	0.145	1	858	1.6	3*
μ − 1s.d	80*	5.17	1	490.5	0.054	1	157	0.8	1*

7.5 Summary

A summary of the results of both the Duncan and Fletcher (2006) and Watson (2014) studies is shown in **Table** 7. To understand the potential impact of these variations, we applied the following values for k and C*:

1. The default set
2. "Low treatment" (the lowest k and highest C* from all studies)
3. "High treatment" (the highest k and lowest C* from all studies)
4. "Crossed low": the lowest k with lowest C*
5. "Crossed high": the highest k and highest C*

to (i) a wetland optimised to meet the BPEM 80/45/45 targets (the wetland had an area of 3% of its impervious catchment and a detention time of 30 hours) and (ii) a wetland designed simply (without any attempt at optimisation) at 1% of its impervious area with a detention time of 48 hours (typical of recent practice). The results of this analysis are presented Table 8 and Table 9. As k and C* behave in an "opposite way", scenarios 4 and 5 test to what extent errors in k might be cancelled out by changes in C*, and vice versa.

The results are very encouraging, particularly for total nitrogen. They demonstrate that in the case of the "optimised wetland" (Table 8) even for the "low treatment scenario", TN removal only drops from 46 to 43%. Similar results are observed for TSS and TP. In the case of the "default size" wetland (1% of catchment area), performance remains poor regardless of the choice of k and C* values. This suggests appropriate wetland sizing is much more critical to a sound MUSIC model than modifications to k, C* and N_{CSTR} values.

> We can conclude that from the calibrations undertaken of stormwater wetlands, the default k and C* values in MUSIC seem to match relatively well with the (small number of) wetlands to which calibrations have been undertaken. This means that predicted pollutant removal using the default k and C* parameters are likely to provide a reasonable estimate of the real performance for a modelled wetland.

Table 7 Comparison between default and calibrated MUSIC parameters from Duncan and Fletcher (2006) and Watson (2014).

Data source		TSS		TP		TN	
		k	C*	k	C*	k	C*
Defaults & MUSIC manual	**Constructed wetland**						
	Theoretical	5,000	6	2,800	0.09	500	1.3
	Calibration (Hampton park) in user manual	500	5	300-500	0.03	50-100	0.9
	Recommended	**500 – 5,000**	**5-6**	**300-2,800**	**0.03-0.09**	**50-500**	**0.7-1.3**
	Default (N=4)	**1500**	**6**	**1000**	**0.06**	**150**	**1**
Defaults & MUSIC manual	**Pond**						
	Theoretical	1,000	12	500	0.13	50	1.3
	Calibration (Blackburn Lake) in user manual	200-300	15	150-300	0.05	30-50	0.7
	Recommended	**200-1,000**	**12-15**	**150-500**	**0.05-0.13**	**30-50**	**0.7-1.3**
Calibration study Duncan & Fletcher (2006)	**Lake Annan** (lake, fringing vegetation, small island, l:w ~2) N=1	~600	5	~80	0.02	~30	0.6
	Blackburn Lake (lake, fringing vegetation, dammed in natural valley, l:w ~5) N=2	200 - 900	10 (winter) 20 (summer)	50 - 500	0.055	80 - 300	0.9 (summer) 1.2 (winter)
	Hampton Park wetland (long, multicelled constructed wetland, l:w > 10)	300 – 400	5	100 - 200	0.03	100 - 200	0.9

	N=10						
Calibration study Watson et al. (2014)	**Ruffey Creek Wetland** (vegetated constructed wetland, ephemeral cell and macrophyte cell) N=1 (TSS and TP) N= 1 - 4 (TN) varied between events	Mean 680 Range 100 - 1500	Mean 5.8 Range 4 - 6	Mean 872 Range 150 - 1500	Mean 0.1 Range 0.06 – 0.12	Mean 507 Range 150 - 1000	Mean 1.2 Range 0.8 – 2
Summary	***Derived from wetlands only in Duncan & Fletcher (2006) & Watson (2014)***						
Low value		***350***	***5.0***	***150***	***0.03***	***150***	***0.9***
High value		***680***	***5.8***	***850***	***0.10***	***500***	***1.2***

* Note: blue colouring represents lakes/ponds and green colouring represents vegetated wetlands

Table 8 Impact of variations in k and C* on pollutant removal by a wetland designed to meet the BPEM targets (area = 3% of impervious catchment, detention time = 28 hours).

	Reduction in mean annual loads		
	TSS	**TP**	**TN**
Default	85	74	46
Low treatment	79	54	43
High treatment	83	79	57
Crossed low	77	64	47
Crossed high	81	66	50

Table 9 Impact of variations in k and C* on pollutant removal by a wetland with an area of 1% of catchment area and a detention time of 48 hours.

	Reduction in mean annual loads		
	TSS	**TP**	**TN**
Default	49	40	25
Low treatment	44	31	23
High treatment	47	43	30
Crossed low	47	37	26
Crossed high	43	37	26

7.5.1 Sensitivity to the NCSTR parameter

Table 10 shows the impact of changing the N_{CSTR} value from its default value of 4 all the way down to 1 (representing a system with a high level of short-circuiting). The results suggest that the impacts on model performance are quite small, with the size of the wetland (which controls the proportion of water bypassed during high flow events) having the dominant influence. The results show that users cannot compensate for a poorly sized wetland by adjusting the N value. Even adjusting the N value up to its maximum (10) for the wetland of 1% cannot approach the current BPEM targets, with TN removal being around 26% compared with 25% for the default setting. This gives confidence to Melbourne Water that the user cannot easily "trick" the model by changing the N value.

Table 10 Impact of variations in N_{CSTR} on pollutant removal by a wetland with an area of 1% of catchment area and a detention time of 48 hours.

	Reduction in mean annual loads (%)		
Value of N	**TSS**	**TP**	**TN**
Wetland sized at 3% of catchment area with 30 hour detention time			
Default (N=4)	85	74	46
N=3	84	73	44
N=2	83	73	44
N=1	82	69	40
Wetland sized at 1% of catchment area with 48 hour detention time			
Default (N=4)	46	39	25
N=3	46	39	25
N=2	45	39	22
N=1	45	39	22
N=10	46	39	26

The analysis undertaken for this study has shown that the MUSIC model is quite robust in terms of its calibration parameters (k and C*) and the representation of the hydraulic efficiency of a constructed wetland (through the N_{CSTR} value). However, it must also be remembered that other design factors which may significantly affect treatment performance, such as vegetation density and configuration, type and stability of substrate, and the presence of resuspension factors, are not explicitly accounted for in MUSIC.

7.6 Conclusions and Recommendations

The analysis undertaken for this study has shown that the MUSIC model is quite robust in terms of its calibration parameters (k and C*) and the representation of the hydraulic efficiency of a constructed wetland (through the N_{CSTR} value). However, it must also be remembered that other design factors which may significantly affect treatment performance, such as vegetation density and configuration, type and stability of substrate, and the presence of resuspension factors, are not explicitly accounted for in MUSIC. Given the insensitivity of MUSIC outputs to 'typical' model adjustments found here, undertaking further monitoring to calibrate and validate the use of the MUSIC model does not seem to be the highest priority for investigation and investment in the next few years. Instead, Melbourne Water should:

1. Strengthen assessment and project control procedures to ensure that when a wetland is constructed the end result matches accurately to the conceptual and detailed design and to the original model of the system. Where there is a variation along the way, this must be reflected and recorded by revising the model.
2. Investigate the influence of other design and construction factors such as (i) vegetation cover and configuration, (ii) substrate type and stability, (iii) poor bathymetry and hydrodynamics (which may not be being well represented by MUSIC).
3. Investigate the influence of wetland age (ideally using a longitudinal study, but if not, using comparisons between wetlands of varying ages).

8 Identifying the most promising management options

8.1 Comparison of key Melbourne Water design assumptions against the literature review

The assumptions underpinning the design and construction of Melbourne Water's wetlands (documented in Melbourne Water (Melbourne Water, 2005) and the latest revised guidelines (Melbourne Water, 2014)) were compared against the findings from this literature review. The findings have been tabulated and divided in the following sections into design, maintenance and missing aspects.

In general, the recent guidelines (Melbourne Water, 2014) have soundly justified principles behind the requirements, and also address many issues shown to be problematic in constructed wetland design and maintenance in the literature review. Importantly, the guidelines require wetlands to have extensive shallow zones with varied water depth up to a maximum of 0.35 m at normal water level (NWL), vegetated with 80 % emergent macrophyte cover in bands across the wetland and open water limited to less than 20 % of the area and vegetated with submerged vegetation. In addition, specifications are given for topsoil depth and composition, the wetland is required to be offline, with high l:w, the maximum extended detention is limited to 0.35 m above NWL, with a retention period of 72 hours, and the design process incorporates an inundation frequency analysis relating the results to plant height. Most critically, the specifications insist on the capacity to adjust water levels which is vital to ensure appropriate water levels and capacity for drawdown, either for plant establishment, long-term viability, maintenance or to introduce wetting and drying variation.

However, the literature is lacking some practical and quantitative guidance, which makes it challenging to assess some aspects of the guidelines. For example, data is not available to determine suitable thresholds for the frequency, depth and duration of vegetation inundation resulting from extended detention.

Some aspects of Melbourne Water's guidelines require clarification or additional considerations, such as:

- Undertaking spells analysis in design and based upon monitoring data to determine if individual events threaten wetland function
- Conducting inundation frequency analyses based upon water level logging when the wetland is operating, with comparisons to vegetation height

- Avoid promotion of dense fringing vegetation (other than for safety concerns), instead focus on emergent vegetation distributed in continuous bands within the wetland
- Further consideration of the potential for stormwater harvesting and infiltration into underlying soils to achieve water treatment objectives – should harvesting be limited to a 100 mm drawdown from NWL and could the clay liner be removed from some designs?
- Consideration of the differences between a gradual and sharp transition into the deep zone – can the gradual transition recommended in the guidelines achieve the same hydraulic benefits demonstrated for an intermediate deep zone with a sharp drop in the literature?
- Clarify the use of the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (Water by Design, 2009). Users of the Melbourne Water guidelines are referred to this document for plant establishment guidance, but no details are provided of its content and scope. The Water by Design (Water by Design, 2009) document provides a high level of detail for plant establishment in particular, and given the sensitive nature of plant establishment, some of the recommendations in the report should be mandatory for wetland construction. In addition, there is some contradiction between the Water by Design (Water by Design, 2009) document and Melbourne Water guidelines (Melbourne Water, 2005, Melbourne Water, 2014), such as the depth of substrate (300 mm and 200 mm respectively) and construction tolerances.
- Provide greater specification surrounding the use of the MUSIC model. This is further discussed in Section 7.
- A greater frequency of monitoring is required than the 3 months and 6 months recommended for establishment and operation, especially if the design is prone to blockage, such as riser pipe configurations. Given the implications on plant health and wetland function, designing to minimise blockages should be a key focus.
- Expected wetland lifespan is not specifically detailed in the guidelines but a nominal lifespan of 20-25 years is not specifically supported in the literature, with suggestions it could be longer or shorter for different wetlands.

There are several aspects which stand out as important to wetland function but are not given a great deal of attention in the guidelines. These include protocols surrounding the timing of planting, water level manipulation during establishment, inclusion of wetting and drying, design characteristics to promote hydraulic efficiency, initial filling and avoidance of substrate erosion, definition of measurable objectives and monitoring. The design guidelines may not be the place to outline operational procedures, but they should direct readers towards the relevant protocol. The

‘Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands’ by Water by Design (Water by Design, 2009) do detail protocols for plant establishment and water level adjustment, but they are only mentioned sparingly and not enforceable.

8.1.1 Design principles

Assumption in MW Design Guideline	Supported by literature review?	Comment
<u>Treatment Zones</u>		
Normal water levels Normal Water Level (NWL) depths: Shallow marsh ≤0.15 m deep, Deep marsh 0.15 – 0.35 m, Minimum 80% of macrophyte zone at NWL must be ≤ 350 mm i.e. shallow and deep marsh. Submerged marsh 0.35 – 0.7 m.	Yes – NWL depths Possibly – EDD depths require further analysis	The sensitivity of emergent vegetation to inundation depth is well established. Similarly the importance of emergent vegetation to water treatment processes, particularly nutrient removal, has been demonstrated by multiple studies. Therefore, a clear definition of the maximum depth of each treatment zone is essential. Plant species will differ in their tolerated range of water depths. However, in general, dense vegetation is commonly restricted to water depths below 0.4 m. Growth difficulties for emergent macrophytes are typically reported to occur for water depths > 0.3 m, and some species require water shallower than 0.2 or 0.25 m. On this basis, most of the shallow and deep marsh in the guidelines falls within a sufficiently shallow range when at NWL in the new guidelines. Permanent inundation exceeding 0.4 m deep will severely restrict the growth of most plant species, leading to plant death, low plant cover, biomass and productivity. This results in poor wetland function. **-> The selected depth range for the establishment and continuation of healthy plant cover in the macrophyte zone at NWL appears to be < 400 mm and likely < 300mm. The 350 mm 'middle ground' in the MW constructed wetlands manual is consistent with current scientific literature.**
Extended detention Extended detention depth (EDD) in the macrophyte zone must be ≤ 350 mm (previously 0.25-0.75m in MW 2005). Water Level depths at maximum EDD assuming 350 mm extended detention: Shallow marsh ≤ 0.5 m, Deep marsh 0.5 – 0.7 m, Submerged marsh 0.7 – 1.05 m.	Dependent upon depth, frequency and duration of inundation	The information available to assess the impact of periodic inundation beyond the NWL depth (i.e. when the extended detention is engaged) on plant growth is limited. Given a maximum extended detention depth of 0.35 m, this increases the shallow marsh depth range to 0.35 - 0.5 m water depth and up to 0.7 m in the deep marsh. Based on the research compared against the recommended normal water levels (above), these depths predominantly lie outside the maximum depth typical of healthy and dense vegetation. The suitability of these EDD depths is conditional upon the frequency, duration and depth when the extended detention is engaged. Repeated events can lead to prolonged inundation and even a single flood event can cause significant loss of vegetation. This is addressed by Melbourne Water's requirement of an inundation frequency analysis during design, and further strengthened by relating the results to plant height, also specified within the guidelines – see below. **-> At the top of an extended detention of 350 mm water depths are increased beyond the range typical of healthy and dense vegetation. Hence, the health of emergent macrophyte vegetation will be compromised if the duration of extended detention**

		becomes protracted. An inundation frequency analysis is important to assess this risk (see below).
Water depth relative to plant height Effective water depth (permanent pool depth + EDD) must not exceed half the average plant height for more than 20% of the time (demonstrated during design using inundation frequency analysis)	Yes - vital to assess No - no clear guidance for setting threshold in literature, also consider longest individual event.	Incorporating this requirement is well justified given the significant increase in depth and risk to plant health during engagement of the extended detention storage (and discussed above). The frequency and duration of water level variation can have crucial implications on plant growth and survival. Despite this importance, the literature provides no clear guidance for quantifying an appropriate threshold. Plant responses are complex and contradictory between species. Referencing plant height and water depth to determine plant exposure above the water is a sound approach. However, as individual events can cause long-term damage, the duration of the longest period of extended detention should also be noted. One study does suggest that slow and rapid variations in water level are better for plant growth than 'intermediate' variations which do not allow plants to adopt an appropriate growth strategy (Vretare et al., 2001). This analysis should not only be undertaken during design, but the expected hydrological regime in each vegetation zone should be verified once the system is constructed using water depth loggers and frequent site visits. This is critical to indicate wetland function and allow appropriate modification to help prevent widespread plant death, particularly as a Melbourne Water study (Carew, 2012) found < 20% of constructed wetlands surveyed with water level loggers experienced both the appropriate drawdown period of the extended detention volume and drawdown to the designed normal water level. **-> Conducting an inundation frequency analysis relative to plant height is well supported by the literature and essential to facilitate plant survival. Additional monitoring and analysis, such as verifying water levels once constructed, and determining the longest duration of extended detention, would provide further certainty to wetland function.**
Treatment zone proportions and depth variation Approximately equal amounts of shallow (≤0.15 m deep) and deep marsh (0.15-0.35m) in the macrophyte zone. Range of depths within each zone along the line of the flow path (but consistent depth across the flow path – see below)	Yes	These requirements provide a diversity of hydrological conditions for different plant species and ensure a substantial portion of shallow marsh, promoting a high degree of contact between the stormwater and wetland. The approach is supported in the literature and should provide a wide range of conditions to promote a variety of biogeochemical treatment processes. **-> Ensuring sufficient shallow zones and a range of depths across these zones is supported in the literature for effective wetland function.**
Positioning of treatment zones Submerged, shallow and deep marsh arranged in bands perpendicular to the flow path with transitions between zones and minimum grade 1:150 to allow free	Yes	Vegetation bands and flat bathymetry perpendicular to the flow path are beneficial for even flow distribution and reduced flow short-circuiting. This enhances hydraulic efficiency and the benefits for water treatment are well established. A uniform depth across the wetland width minimises areas of fringing vegetation, which negatively influence water treatment by

drainage. Uniform depth perpendicular to the flow path. Good performance = dense vegetation and distributed flow path, Poor performance = clumped vegetation and multiple open water flow paths (New Guidelines, Figure in Part A1)		enhancing dead zones. Shallow gradients within the wetland provide a wide zone to facilitate successful vegetation establishment. While a maximum grade has not been specified in the guidelines, the maximum depth and proportion of treatment zones will indirectly limit the slope of the wetland. This guideline does exclude the possibility of small-scale variation in topography (or 'microtopography) – a heterogeneous feature in natural wetlands. A number of studies have noted the benefits of microtopographic variation for water treatment processes and vegetation survival. However, microtopography may only provide advantages during low flows (within the height range of the topography). Hence, based on current knowledge its inclusion does not appear essential for good design at present. **-> These design guidelines are well supported by the literature and benefit hydraulic efficiency. Further benefits could be gained from inclusion of microtopography, but this is not essential and other design elements exert greater influence on wetland function.**
Open water deep zones Open water must include submerged vegetation and comprise <20% of wetland area, present as inlet and outlet pools (<= 1.5 m), plus intermediate pools (<= 1.2 m) in large systems (Melbourne Water, 2014) Open water > 20% must be present as a separate water body Consistent grading towards open water (Melbourne Water, 2005)	Yes – in large part	Inclusion of a limited proportion of open water is supported in the literature. Importantly, deep zones help to break short-circuits when placed as intermediate pools. Placement at the inlet or outlet also helps to distribute flows and decrease the likelihood of blockages. Deep zones at the inlet will also accumulate heavy sediment particles, which will reduce sediment accumulation in shallow zones. The benefits of intermediate deep zones to hydraulic efficiency suggest a greater emphasis should be placed on intermediate deep zones in the guidelines. However, while deep zones harbour some water treatment processes, the magnitude is inferior relative to vegetated shallow zones. As a result, their area must be limited to avoid a reduction in wetland function. If a wetland is undersized, the hydraulic benefits of deep zones are outweighed by the negative effect of a reduction in shallow vegetated treatment zones. Hence, the guidelines correctly place a limit on the proportion of deep zones and suggest wetland size should be considered for their inclusion. The optimal number and size of deep zones will depend upon each system. The recommended limit of 20% is supported by modelling results from Lightbody et al. (2009) suggesting between 5-37% of the wetland area is optimal and a comparison of field systems which found 0-20% was most beneficial for nitrogen removal. Much remains unknown about flow dynamics in different wetland configurations, but the benefits of deep zones for wetland hydraulics were noted in studies with relatively sharp declines from a shallow vegetated zone into the deep zone (e.g. slopes of 1:3 - 1:5). It is uncertain if the same benefits result from gradual and consistent declines in the order of

		1:150 specified by the Melbourne Water guidelines. A steeper decline at the interface from the deep marsh to submerged marsh may be warranted if it induces a rapid drop in momentum and greater mixing. It may also reduce the proportion of wetland area occupied by zones too deep for most emergent macrophytes. **-> Deep zones can contribute to wetland treatment function in wetlands that are of sufficient size for their catchment. Intermediate zones benefit hydraulic efficiency, although it is unclear if a relatively steep drop in the bathymetry is required for this function (contrary to the gradual decline in the guidelines). If a wetland is undersized, then fully vegetated shallow wetlands are likely more beneficial.**
Batter slopes Vegetated batters ≤ 1:5 slope before 2.4 m wide vegetated bench located at 1:8 between NWL and 350 mm below NWL and maximum 1:3 slope beyond 350 mm below NWL OR batters ≤ 1:4 slope between TED and 250 mm below NWL with dense impenetrable vegetation a minimum of 2.5 m wide and 1.2 m high.	In part – but driven by safety	For the purposes of hydraulic efficiency it is desirable to minimise fringing vegetation and potential stagnant zones, which suggest steeper batter slopes are beneficial. Conversely, steep slopes produce rapid wetting and drying and a limited ephemeral zone, which differs from the gradual slopes typical of natural wetlands. Variable wetting and drying over a wider zone will benefit vegetation survival and the diversity of treatment processes. Irrespective of these competing considerations, these guidelines are primarily designed for safety. Steep continuous batter slopes are avoided, although the slope is likely to exceed natural wetlands. **-> In terms of treatment, both shallow and steep batter slopes may be beneficial to vegetation survival and hydraulic efficiency respectively. However, the recommended slopes are most importantly guided by safety.**
Vegetation		
Emergent macrophytes should comprise at least 80% cover in the macrophyte zone (Melbourne Water, 2014) Aim for 70-80% cover after two growing seasons (Melbourne Water, 2005)	Yes	The strong focus on high macrophyte cover is warranted given the numerous interactions between plants and wetland function for water treatment (both direct and indirect). Numerous studies report more effective water treatment, particularly for nitrogen, in the presence of vegetation relative to unvegetated sediments. Hence, high plant cover is essential for treatment. While open water zones contribute hydraulic benefits and some degree of processing, their proportion must be limited as retention in deep zones does not benefit water treatment significantly. A sufficient area of shallow vegetated zones is required to justify inclusion of a deep zone. In addition, the use of high cover to infer function is widespread and justified by studies relating cover or plant biomass to effective nitrogen removal. However, plant cover should not be the sole indicator. Configuration can be equally, and at times, more important. The guidelines incorporate this by specifying continuous vegetation bands across the wetland width but monitoring programs should assess vegetation configuration on an ongoing basis. **-> High plant cover is a good precursor for effective water treatment in a wetland in conjunction with other design considerations**
Plant species selection, planting density and	Yes	Dense planting is beneficial for the quick establishment of plant cover.

seedling requirements Plant densely and use a minimum of 3 of the listed species, these species must comprise 80% of plants used in the macrophyte zone and 90% of plants used in the deep and submerged marsh. Planting density specified for each zone. Pot volume, minimum stem height and well developed root system specified for seedlings (Melbourne Water, 2014)		There is no quantitative guidance in the scientific literature for the optimal number of species to plant. However, in specifying only 3 species as a minimum, the guidelines are not enforcing highly diverse systems, which is consistent with reports that diversity does not necessarily equate to water treatment. Conversely, planting a number of species provides scope for 'self-selection' of the most suitable species. A minimum of 3 species in each zone may provide a suitable balance. It should however, be expected that the species composition will inevitably change over time in the wetland, but this may occur independently of wetland function. Nutrient removal can continue despite substantial changes in vegetation. There is also a possibility that invasive species can provide effective water quality treatment, despite other environmental shortcomings, although their overall benefits are debateable and little quantification has been undertaken. Specifying seedling size requirements will benefit vegetation establishment, helping plants to grow and aim for sufficient emergence from the water. **-> Focusing on a relatively small number of species in each zone is supported by the literature – this provides a degree of diversity without forcing high diversity, which is not essential for effective nutrient removal. Guidance towards ensuring robust size and health of seedlings will benefit establishment.**
Submerged vegetation Open water areas must include submerged vegetation (Melbourne Water, 2014)	Yes	Submerged vegetation will enhance water treatment in the deeper areas relative to unvegetated sediments. Submerged macrophytes may be less productive than emergent macrophytes, but provide some differing benefits, such as the provision of simple carbon compounds to drive denitrification. In addition, studies have found systems with a mixture of vegetation types can provide superior performance. In spite of this, the benefits of submerged or mixed vegetation do not override the requirement for extensive shallow areas with emergent vegetation. **-> This is well supported from evidence of the benefits for water treatment provided by submerged vegetation and reports of superior treatment when a range of vegetation types are present. However, large shallow areas with emergent vegetation remain essential.**
Ephemeral fringing vegetation Dense planting on ephemeral batters (NWL to 200 mm above) (Melbourne Water, 2014)	Yes for safety only	Dense planting will benefit safety and reduce erosion. The vegetation will also benefit water treatment processes when inundated, although this zone above NWL will only be engaged during extended detention. However, dense fringing vegetation acts to reduce hydraulic efficiency if it is not part of a continuous vegetation band. . Healthy fringing vegetation is not a substitute for high cover of emergent vegetation within the wetland. **-> Densely vegetated batters provides benefits but also some disadvantage if the vegetated band does not extend across the entire wetland. Other than for safety**

		reasons, fringing vegetation should not be a focus in design and maintenance. Instead, promoting high and consistent vegetation cover across the wetland (below the NWL) should be a priority.
<u>**Flow dynamics**</u>		
Drawdown of extended detention Slow release from extended detention across 72 hour retention period (90^{th} percentile retention time in the macrophyte zone) (Melbourne Water, 2005, Melbourne Water, 2014)	Yes in conjunction with an inundation frequency analysis	Melbourne Water (2005) notes this detention time is based upon settling velocity equations for fine particles (but ignoring the influence of turbulence). Consistent with this, the literature suggests 2-3 days is the timeframe for contaminant concentrations to return to 'background' levels within a treatment wetland. The necessary period will depend upon contaminant characteristics, particularly the proportion of very fine particulates. However, extended detention poses a risk to vegetation health. The impact will vary depending upon wetland size (relative to catchment) and local climate. If a wetland is undersized or in high rainfall areas frequent storm events may engage the EDD for long consecutive periods. An inundation frequency analysis (also recommended by the guidelines and discussed further above) and spells analysis will help determine the time period of extended detention engagement. Once constructed, water level logging and analysis should be conducted to verify this and the drawdown time period. Further understanding the impact of extended detention dynamics on vegetation health would be beneficial for design and management, but the literature does not provide clear guidance. **-> Supported by the literature, but implementation needs careful analysis of inundation patterns to ensure vegetation survival**
Offline location Macrophyte zones must be offline (i.e. have a bypass route) Exception when wetland located in a retarding basin, but this inundation should be of short duration (hours) (Melbourne Water, 2005)	Yes, although placement in a retarding basin is undesirable	Well supported given the critical influence of hydrology on wetland function and vegetation survival – excessive flows through the wetland will drown vegetation, reduce the degree of contact between the stormwater and wetland components and may scour the topsoil. This compromises wetland performance well into the future. It is understood that Melbourne Water prefer wetlands are not located within retarding basins. **-> Offline wetlands are well supported by principles in the literature and provide practical benefits for maintenance.** **If a wetland must be located in a retarding basin, design should ensure relatively rapid drainage of the basin and an inundation frequency analysis is required to understand the additional flooding.**
Configuration High l:w ratio (Melbourne Water, 2005)	Yes	A high l:w ratio is a well-established and demonstrated principle to promote hydraulic efficiency, but must be used in conjunction with design elements that promote low velocity and wide flow distribution, such as bands of vegetation, shallow gradients and consistent

		bathymetry perpendicular to the flow. These principles are also largely incorporated into the guidelines. **-> Well supported and should be used in conjunction with other principles of good design**
Hydraulic efficiency indice Hydraulic efficiency should be > 0.5 with design aiming for > 0.7 (Engineers Australia 2003 and noted in Melbourne Water (2005))	In part	0.5 is not a particularly high hydraulic efficiency, hence > 0.7 is a more appropriate objective. The designs modelled by Persson et al. (1999) indicate that a large improvement in hydraulic efficiency can be made using design elements that ensure good use of the wetland area, such as distributed inflows, a submerged bar or an island near the inlet or baffles, without changing the wetland configuration. Similarly, consistent bands of high vegetation cover and a mixture of vegetation types also greatly improve hydraulic efficiency. Hence, even when land area is limited, a high hydraulic efficiency should be possible with good design. **-> Hydraulic efficiency is not specifically stated in the new guidelines, but aiming for > 0.7 should be recommended and design principles which promote this encouraged.**
Flow velocities Aim for velocities < 0.05 m/s (Melbourne Water, 2005) for the peak 3 month ARI flow and < 0.5 m/s for the peak 100 year ARI flow (Melbourne Water, 2014)	Yes	Low velocities through the wetland are critical to prevent scouring, erosion and vegetation loss – hence they are critical to wetland function in the long-term. The design of a high flow bypass, wetland sizing, configuration and gradients will be key influences on velocities. Once erosion has started vegetation growth is compromised, which further promotes erosion in a negative feedback cycle. Hence it is vital to protect the wetland across its entire life, including the infrequent large events. From the Hjulström-Sundborg diagram of sediment erosion, transport and deposition, the velocities in the guidelines appear sufficiently conservative, although some erosion may result to unconsolidated fine sediments during the 3 month ARI. **-> Low velocities are critical to wetland function and lifespan. Based on limited information, these values appear reasonable.**
Tolerance for levels Construction tolerance of 50 mm is generally accepted (Melbourne Water, 2005)	No	A tolerance of 50 mm is not sufficiently accurate for critical invert levels such as the outlet. Different tolerances may be acceptable for different features in the wetland, with invert levels that influence water levels in the shallowest zones particularly crucial. Tolerance levels do not appear to be stated in the latest guidelines but the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009) note that hydraulic structures (e.g. inlet, outlet, bypass, maintenance pipe components) require small tolerances < 25 mm but a 50 mm tolerance is acceptable for earthworks and topsoil levels. **-> Not sufficiently low for flow control structures**
Stormwater harvesting If stormwater is harvested from the permanent pool, extraction must not occur if the water level is > 100 mm below NWL (Melbourne Water, 2014)	No	The intention of this guideline is to prevent rapid drawdown and drying within the wetland. However, given the benefits of wetting and drying to contaminant processing and plant growth, and the challenges to introduce sufficient variability in constructed stormwater wetlands, water harvesting could provide useful fluctuation in water level. Depending upon the season, further drawdown could be justified as long as excessive drying was not likely, the

		pattern reflected natural wetting and drying (e.g. relatively slow drawdown in stages and in the right season) and did not expose submerged plant species or those requiring permanent inundation. **-> Not necessarily justified as drawdown could potentially provide benefits, but care must be taken to avoid excessive drying.**
Inlet location Major inlets (i.e. drain > 10 % of catchment) must be located within the first 20 % of the macrophyte zone (Melbourne Water, 2014)	Yes	This is necessary for hydraulic efficiency and effective treatment within the wetland. Inlets close to the outlet will result in little, if any, treatment. For best use of the wetland area inlets must be located the furthest distance from the outlet. **-> Soundly justified by the theory of hydraulic efficiency and wetland function**
MUSIC modelling Must use MUSIC to model the treatment and flow regime ((Melbourne Water, 2014), GN11 Part B2)	Yes	This represents best practice. However, a clear understanding of applications that are suitable for MUSIC and appropriate parameter selection is required to avoid use of the model beyond its design intent. This is discussed in detail within Section 7. **-> Sound practice when MUSIC is used for its designed purposes and appropriate parameters are selected**
Drawdown Outlet structure provides linear storage-discharge relationship across the full extended detention range	Likely yes	A linear drawdown would be expected to provide a consistent rate of drying and macrophyte exposure as the water levels falls. The advantage of this is not readily apparent from the literature except that it might provide the most gradual drawdown, possibly with greatest similarity to natural wetlands, and provides an even retention time across the extended detention depth. This straightforward approach may be better than a more complicated drawdown relationship that may instead be too difficult to model and verify in the field. **-> Appears justified in the absence of much guidance**
<u>Substrate and groundwater interaction</u>		
Liner If groundwater interaction likely or saline soils are present, must use impermeable liners. Exfiltration rate must be accurately represented from the base and sides of the wetland in the modelling. Clay liner or geotechnical testing to determine exfiltration rate (Melbourne Water, 2014)	Partly yes	Some infiltration of stormwater to surrounding soils could be beneficial and mimic interaction in natural wetlands. Loss of flows to the soil will reduce flow volumes and provide contaminant treatment. Conversely, if there was a net movement of groundwater into the wetland flow volumes would increase. In addition, if the groundwater is shallow it is at risk of contamination and the wetland should support a permanent presence of water – a high water loss may stress the wetland environment. Given the unknowns generally associated with groundwater and its variability between sites, standard use of liners is a conservative option. **-> While natural wetlands interact with groundwater and there may be some benefits to stormwater infiltration into surrounding soils, use of a liner is a safer option if groundwater conditions are unknown.**
Top soil composition Top soil standards must adhere to requirements of AS4419 Soils for landscaping and garden use.	Yes	Specifying standards for the wetland substrate is well justified. The substrate is an active component of wetland function. A vital role is to support a healthy vegetation community – poor substrate composition, insufficient depth or erosion of the substrate are common causes for vegetation loss. A wide range of microbial processes, including denitrification, also

		occur within the substrate. These plant and microbial communities require sufficient nutrients and organic material. Hence, specifying minimum soil requirements is important and the AS4419 standards do this with the objective of supporting plant growth. However, excess nutrients are not desirable as nutrient release may occur instead of nutrient uptake from the stormwater. The use of soils that have received large amounts of fertiliser, such as some former agricultural soils, should also be avoided. Other properties of the substrate may be desirable, such as higher aluminium or iron content to promote phosphorus binding, or lime can be added to achieve a desirable pH. It is to be expected that the composition of the substrate will change over time, likely with increasing organic matter and increased porosity. This may accompany an increase in function – e.g. higher denitrification potential. **-> This specification is important as the substrate is a vital component of the wetland and must support healthy plant growth. While the standards specify minimum requirements, excess nutrients in the substrate should also be avoided to reduce the chance of nutrient release. In addition, there may be other beneficial substrate properties that can increase contaminant retention.**
Topsoil depth A 200 mm depth of topsoil must be laid across the wetland.	Yes	Along with composition, substrate depth is a critical design parameter. It dictates the rooting depth of vegetation and will influence plant survival. In turn, the establishment of high plant cover is vital to prevent erosion of the topsoil, which, once commenced, impedes plant survival further – forming a negative feedback cycle. The majority of plant roots generally occur in the top 200 mm, although roots down to 300 mm are also common and some species may have even deeper root systems (Kadlec and Wallace, 2008). While deeper topsoil is more desirable, a minimum of 200 mm appears reasonable for the majority of roots (Scholz and Lee, 2005). However, this does contradict the 300 mm specification in 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009). **-> Important parameter to specify, critical to successful plant establishment and 200 mm appears reasonable, although deeper topsoil is desirable**

8.1.3 *Maintenance principles*

Assumption in MW Design Guideline	Supported by literature review?	Comment
<u>Monitoring</u>		
Monitoring frequency Inspect every 3 months in the first year and every six months following (Melbourne Water, 2014)	No	During early operation and establishment visits more frequent than 6 month intervals are required to ensure the wetland is functioning as expected and to adjust the flow regime as plants establish. Frequent site visits during wetland filling and early operation will also be needed to verify the designed water regime is occurring, or re-adjust water levels as required. Water-level logging should also be employed to verify the flooding regime matches the design across different seasons and flow events. In the long-term, more frequent site visits to check the outlet will be required if the outlet design is prone to clogging (e.g. orifice plates, riser pipes). If clogging increases water levels for a prolonged period between site visits, this can lead to widespread vegetation loss, with long-term implications on wetland function from a single event. Even a partial blockage is unacceptable if it increases the water level in the wetland. **-> More frequent inspections are required during establishment to monitor and adjust water levels as plants grow and in the long-term frequent visits will be required if the outlet design is prone to blockage.**
Visual water level marker Include a marker to indicate wetland water level relative to NWL and EDD – must be visible from the bank and attached to the wall of the submerged outlet pit (Melbourne Water, 2014)	Yes	Monitoring water levels is vital to ensure the wetland is functioning as designed and action can be taken to prevent plant drowning. Even if automatic water level logging is used, a permanent visual marker allows quick assessment during site visits and can be visually compared against inundation of the vegetation. All wetland assessments should be considered in the context of the current water level. **-> Important feature for monitoring and verifying wetland function.**
<u>Maintenance</u>		
Flexibility to adjust water levels for plant establishment Design outlet to allow drawdown from NWL of up to 150 mm for plant establishment and maintenance (Melbourne Water, 2014) Outlet configuration must allow water level control within the macrophyte zone – including flow low enough for plant establishment (Melbourne Water, 2005)	Yes but further drawdown capacity is warranted	The hydrological regime during establishment is particularly crucial for successful vegetation. Flexibility to lower the water level is essential to achieve this. A drawdown of 0.15 m will lead to water levels from 0 – 0.2 m at NWL across the shallow and deep marsh macrophyte zone, and 0.2 – 0.55 m in the submerged marsh. This drawdown may be sufficient given seedling stem heights have been specified in the New Guidelines as a minimum of 300 mm or 500 mm (depending upon pot size). It is also consistent with depths of 150 – 200 mm recommended for the deep marsh in the first 6-8 weeks of plant growth in the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009). However, muddy

		conditions are preferred for plant establishment (Water by Design, 2009), hence further drawdown capability would be beneficial, and would also assist drawdown for maintenance using minimal pumping or pumping points. **-> Reduced water levels during plant establishment are essential for plant survival and long-term wetland function. However, capacity for even greater drawdown of the wetland – to fully drain the emergent macrophyte zone – is warranted.**
Flexibility to permanently adjust water levels Design to allow permanent adjustment of NWL by ± 100 mm to respond to changes in wetland hydrology (e.g. in response to future climate) (Melbourne Water, 2014)	Yes	Note this capacity also provides flexibility to correct for uncertainty in the flow modelling – which may even simply stem from a lack of local rainfall data. Flexibility to adjust flows is crucial to long-term wetland efficiency and may prevent widespread plant loss. **-> Well justified to allow flexibility in wetland operation and facilitate healthy long-term plant growth.**
Flexibility to adjust outflows Allow flexibility to alter the stage-discharge relationship for changes to the residence times or inundation as required (Melbourne Water, 2014).	Yes	As above, the capacity to alter the drawdown dynamics for the extended detention storage is important if adjustments are required to maintain a healthy plant community. Such flexibility in wetland operation is vital and may be used to correct for uncertainties from limited data availability during design and modelling. **-> Critical to ensure wetland functionality and adjust for uncertainties in design**
Sediment pond drainage Must be able to drain the sediment pond while maintaining the macrophyte zone at NWL.	Yes	This requirement is important to allow sediment pond maintenance, particularly dredging, which is required much more frequently sediment removal within the wetland cells. **-> Important requirement for sediment removal**
Riser outlet Riser outlet used to achieve desired retention period for settlement of very fine particles (Melbourne Water, 2005)	No other designs may be superior	Riser outlets are beneficial to water treatment in theory, but if a blockage occurs to impede drainage the benefits can be negated by vegetation loss. To reduce the need for high monitoring frequency, outlet designs should not be prone to blockages. The repercussions on vegetation survival and wetland function are critical. The New Guidelines do not state the type of outlet, which allows more design flexibility, but preventing the likelihood of blockages should be a key design objective. **-> Riser pipes can be prone to blockage. Blockages severely compromise wetland function and preventing their occurrence should be integral to outlet design. Alternate designs may be less prone to blockages.**
<u>Lifespan</u>		
Nominal lifespan **Expected lifespan 20-25 years** (not specifically listed in the guidelines but a general assumption for MW wetlands)	No	This nominal lifespan is commonly stated for constructed wetlands. However, no clear justification for this lifespan was apparent in the literature. Long-term studies, particularly on replicated field systems, are rare. There are promising reports of long-term wetland efficiency. Nutrient saturation is not inevitable across the wetland lifespan and an equilibrium may instead establish. Wetland function can remain relatively stable, despite changes in wetland structure and reported shifts in key processing pathways. There are

		promising examples of ongoing wetland function for 10-30 years of operation, although sediment and phosphorus export can become problematic in some systems. **-> No clear evidence to justify this timeframe. It may be shorter or longer in different systems.**

8.1.4 Aspects missing from the guidelines that may be worth considering

Assumption in MW Design Guideline	**Comment**
Water level operation during wetland establishment **-** specific operating instructions have not been stated – for example, targets for water depth in different zones, duration of establishment period, plant growth monitoring requirements, triggers for water level management, influence of season – these are instead discussed in the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009).	Both MW 2005 and the New Guidelines state the need for designs to allow a reduction in water levels during plant establishment, and the New Guidelines specify that a change of 0.15 m must be possible. Further guidance, such as the duration of the establishment period or direction for monitoring and adjusting water level is not given. Instead designers and operators are referred to the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009). These guidelines provide information on the timing of planting, signs of mature and healthy tubestock, conditions for establishment, watering requirements and water level control. This includes a preference for plant establishment in muddy conditions with watering providing each plant with 2.5 – 5.0 L/week to maintain the mud. If wetland flooding is unavoidable, water levels should aim to be half the plant height and should not be > 2/3 the plant height especially in winter. However flooding will prolong the duration of plant establishment by 2-3 times. Guidance for slowly increasing water levels, dependent upon stem height, is also given. **-> Most of the key details are given in Water by Design (2009), but the New Guidelines from Melbourne Water should provide more reference to this document and point to its contents.**
Timing of planting – no clear guidance given, other than noting it depends upon season, capacity to irrigate and construction timeframes (Melbourne Water, 2005).	The seasonal timing of planting can be crucial to plant establishment, at times being a dominant factor in the success of plant survival. Wetland planting may depend on many factors unrelated to the appropriate season, including the timing of development. Some guidance on the ideal planting window, and the implications of planting outside of this (e.g. a prolonged establishment period) is required. The Establishment Guidelines (Water by Design, 2009) only discuss timing in relation to the lead-time required for nurseries to supply tubestock. **-> Little guidance is provided in the guidelines but it can be critical to successful plant establishment**
Wetting and drying – only mentioned briefly in the guidelines. Noted to disturb the mosquito breeding cycle in MW 2005 Guidelines. The New Guidelines suggest design should allow for occasional drawdown of the permanent pool to replicate wetting and drying in natural wetlands, but no	Wetting and drying can benefit wetland performance. A degree of water level fluctuation will facilitate vegetation survival and may provide a diversity of conditions for enhanced contaminant processing and retention. Varied water levels are a key characteristic in natural wetlands. In systems with fluctuating water levels, the water depth can be less important than the frequency and duration of dry and wet periods in shaping the vegetation community. However, other than the 3 day drawdown of the extended detention volume, oscillating wet and dry

clear guidance is given.	conditions are not promoted by wetland design. While the flashy nature of stormwater inflows makes wetting and drying challenging, the importance of this variation should be noted in guidelines for possible future incorporation. It is important to also caution that re-introduction of wetting and drying, or a single drawdown in isolation, does not necessarily lead to an immediate improvement in vegetation health. **-> Water level variation providing wetting and drying oscillations can improve plant health and benefit some wetland functions. Very little detail is currently incorporated into guidelines.**
Optimal design characteristics – The guidelines include little mention of optimal design of the configuration for hydraulic efficiency e.g. wide inlet, structures to distribute flows etc.	The paper by Persson et al. (1999) illustrates the improvements in hydraulic efficiency that can result from flow distribution, even with the same wetland shape. Although design flexibility must be maintained, the guidelines could benefit from a stronger focus on promoting hydraulic efficiency. **-> Further discussion of features that contribute positively and negatively to hydraulic efficiency is warranted.**
Procedure for initial filling and management of the substrate – not stated in the guidelines. Water level management is discussed in detail in the Establishment Guidelines (Water by Design, 2009), but not in the context of minimising erosion of the freshly laid substrate.	This is a period when the wetland is vulnerable to erosion which may permanently influence its efficiency. In particular, zones of higher velocity flow, such as over banks or weirs, are susceptible to erosion. This creates channels which counteract flow distribution and may produce permanent short-circuits. Design mechanisms and a protocol for the initial filling of the wetland are needed to minimise flow velocities during filling. Stabilisation of the substrate, such as an initial grass covering, may also help prevent erosion of the fresh topsoil, particularly in the time period between construction and planting. Other options could include roughening the surface of the clay liner to increase cohesion with the overlying topsoil. **-> Discussion of means to minimise substrate erosion during early wetland life are required in the guidelines.**
Measurable objectives and monitoring – little guidance given, and where monitoring has been discussed (e.g. checklist in Melbourne Water (2005)) many items are not readily measurable with insufficient information and checks e.g. simply asking if the condition of various components is OK and to look for weeds or litter.	The guidelines provide little direction for wetland monitoring or definition of measurable objectives. If sufficient information is available, state in terms of parameters that can be measured e.g. vegetation cover (80% cover is given), distribution, change in water turbidity across the wetland etc. **-> Robust, descriptive and practical maintenance inspection sheets are required. These should allow different users, with little presumed knowledge, to understand the necessary checks to be made. The assessment should be detailed to minimise subjective differences between assessors.**

8.2 List of options which are 'worth a shot' to improve performance

A critical issue for Melbourne Water is allocating limited resources to wetland restoration – efforts need to be based on sound knowledge/scientific evidence, with a strong chance of a sufficient improvement to justify the expenditure. The knowledge from this literature review is intended to guide these efforts.

Management actions with potential to improve performance and expand lifespan include:

- **Define and record clear objectives** for wetland performance, not only in terms of water quality but also readily monitored parameters. These should be system-specific and regularly assessed against monitoring outcomes.
- Implement clear protocols surrounding the **plant establishment phase** during this critical time in early wetland life. It is vital to plant early in the growing season and implement an appropriately shallow hydrological regime as the seedlings grow (Vanderbosch and Galatowitsch, 2011). Long-term constructed wetland performance can hinge on this stage. Key details for planting, watering requirements and management of the water regime and its gradual increase over time, are provided in the 'Construction and Establishment Guidelines for Swales, Bioretention systems and Wetlands' by Water by Design (2009). However, this content is only given passing reference in Melbourne Water Guidelines (2005, 2014). Studying the appropriate hydrology and time period for plant establishment is also recommended by Dugdale (2013).
- Similarly, carefully manage and develop protocols surrounding **substrate management**, including initial filling procedure, management to reduce erosion (including before planting) and checking that sufficient depth and quality of substrate are provided.
- Ensure **consistency between constructed wetlands and their conceptual and detailed design elements**, including characteristics incorporated into the underlying MUSIC model. If variation is detected, investigate the ramifications on the model output.
- **Monitoring water levels and emergent macrophyte height, cover, distribution and health** to assess wetland hydrological function and its appropriateness for plant survival Raulings, 2010 #178;Greenway, 2007 #174}. Manipulations of water regime to investigate the optimal regime for plant survival and reproduction were suggested and discussed in more detail by Dugdale (2013).
- **Analyse time series of water level and the depth/duration/frequency (i.e. hydroperiod) across different vegetation zones** – conduct the analysis using model output during the design process, and monitored data following construction. Compare the findings against

design objectives, plant height and growth observations. Comparison between modelled and actual water levels is important to ensure the hydrology and bathymetry interact as expected. There are multiple reports of wetlands with mis-alignment between bathymetry and hydrology (EDAW, 2008, Alluvium, 2010, Greenway et al., 2007).

- **Observe wetland functioning** under high flow conditions, noting the degree of emergence of plants above the water, functioning of the inlet, outlet and bypass structures, and any obvious short-circuiting. Also assess the time required for water levels to return to normal. This could be undertaken by capturing time lapse video or photo images from a camera, possibly mounted on a pole or high position above the wetland.
- **Actively modify** the inlet or outlet structures **as required to adjust water levels in wetlands** in response to data analysis and monitoring outcomes. Note that a one-off drawdown event will not fix problems with vegetation survival in the long-term – changes to the hydrological regime need to be sustained (Raulings et al., 2011, Baschuk et al., 2012). Wetland operation should include **drawdown events** to increase resilience and expansion of the emergent vegetation, as recommended in the detailed investigation undertaken by Dugdale (2013).
- **Develop monitoring protocols** to assess plant cover, configuration, plant type, substrate characteristics, plant health and possibly *in situ* water quality parameters. Additional monitoring should include the conditions of the inflow and outflow structures, signs of erosion, short-circuiting/stagnant zones, abundance of weeds and sediment accumulation (Hammer, 1992, Woods et al., 2004, Brock and Casanova, 2000).
- **Incorporate microtopographic variation** into design. Multiple studies have noted the benefits to nutrient removal. The literature debates the extent of these benefits and other design elements have greater influence on performance, but with good design microtopography will exert a positive influence.
- Design wetlands to **include greater heterogeneity** in biogeochemical conditions – i.e. varied topography, water regime, treatment zones and vegetation types. Heterogeneity is continually raised as an integral component of natural wetlands and related positively to diverse biogeochemical processing and plant survival (Van Dam et al., 1998, Gawne and Scholz, 2006). No single set of conditions are ideal for achievement of all treatment objectives (Goonetilleke et al., 2005, Bodin, 2013) and a diversity of treatment zones is recommended in multiple studies (Mitsch et al., 2005a, Greenway, 2010).
- However, there are also contrasting examples in the literature to suggest beyond a certain point species diversity is not optimal for water treatment. Hence, constructed wetland

design must **strike a balance** – incorporate enough variation for a resilient system, supportive of a wide array of processes under changing conditions, but also dominated by aspects 'successful' design from a water treatment perspective.

- Such **'successful' characteristics** most likely include –
  - Extensive zones with emergent macrophyte vegetation with water depths primarily < 0.2 – 0.3 m and not exceeding 0.4 m, with some variation in water depths across this zone
  - Inclusion of zones with submerged vegetation and possibly also free-floating vegetation
  - High plant cover and biomass, arranged in consistent bands across the width of the wetland (and not solely fringing vegetation). However plant density should not be excessive to the point of creating anaerobic conditions in the water or litter accumulation severely impeding the flow.
  - Presence of algae (but not excessive abundance, i.e. a 'bloom') and biofilms
  - A degree of wetting and drying, while avoiding extreme drying
  - If the wetland is sufficiently sized for its catchment, the inclusion of limited deep zones located at the inlet, outlet and intermediate within the wetland.
  - Wetland shaped with high l:w ratio – either narrow and long or designs with baffles or a sinuous shape
  - Wide distribution of inflows
  - Bathymetry that is either flat or with random microtopographic variation across the width of the wetland

9 Knowledge gaps and future work

9.1 Prioritised knowledge gaps

The outcomes of this review are intended to guide future research programs at Melbourne Water. Some of the key questions raised include:

- Limited understanding of the effect of **extended detention** on vegetation growth – what depth, duration and frequency combinations are acceptable? Is plant growth sensitive to one hydrological parameter more than others?
- Can **wetting and drying** variation benefit performance in constructed wetlands or are benefits confined to the restoration of natural systems? While much literature focuses on the benefits of wetting and drying to diversity, survival and recruitment, some studies report reduced plant nutrient uptake and productivity, and extreme drying causes nutrient pulses following re-wetting. Further work is required to identify optimal water regimes for constructed wetlands.
- What are the **key processes and fates** for each key contaminant? How much is stored within the system, what is the timeframe of this retention and how much is permanently lost or incorporated into long-term compartments? Can the availability of organic matter, water dynamics, aeration and use of different plant species be manipulated to optimise processes, as suggested by Faulwetter et al. (2009)?
- **Limited quantitative data** is available to determine objectives for design. For example, what proportion of plant cover is necessary for efficiency? What mixture of emergent, submerged or free-floating vegetation is optimal? How important is vegetation configuration relative to overall cover? The literature indicates the important influences but does not provide quantification to assist designers with trade-offs or answers to questions on how to best manage or prioritise remediation activities.
- Following from this, what is the actual **performance of wetlands identified to be in poor condition**?
- Data relating performance for water quality with **readily quantified characteristics** of wetland structure and function is scarce.
- What is the **long-term performance** that can be expected from stormwater treatment wetlands, and how does this relate to indicators that can be quantified?
- Indicators of **wetland end-of-life** or management actions to prolong lifespan are not a focus in the literature.

9.2 Program of research/trials/investigations/best-guesses

The knowledge gaps give rise to a number of potential research studies:

- **Collect monitoring data and compare characteristics** of wetland structure and function across Melbourne Water's network of constructed treatment wetlands. This information can be used to investigate many of the key questions and suggested investigations below.
- Seek **relationships and correlations between design parameters (e.g. vegetation cover and configuration, substrate type and stability, bathymetry, hydrology and hydraulics) and function/performance for a variety of wetland ages**, aiming to identify characteristics of 'successful' wetlands across the wetland lifespan. For example, relate data on water depth/frequency/duration with plant cover and type in each treatment zone and for wetlands of different ages. Also relate characteristics of wetland configuration to flow characteristics including retention time and short-circuiting.
- Investigate the **optimal water regime** for water treatment processes in constructed wetlands. This should include:
    - an assessment of wetting and drying – what degree of variation is beneficial in the context of constructed wetlands, both for plant resilience and contaminant retention, and how this can be incorporated into wetland operation?
    - studying the influence of extended detention upon the vegetation community using inundation frequency curves and data on plant cover and biomass. Is there an optimal depth/duration/frequency and what are the thresholds beyond which most species cannot sustain growth?
- Investigate the practicalities of incorporating **microtopography** into wetland designs. Also conduct a desktop study of the characteristics (e.g. roughness, tortuosity, elevation) that may optimise the benefits and minimise any adverse effects (e.g. significantly reduced plant cover, potentially greater short-circuiting).
- Conduct **direct water quality monitoring in select wetlands** to confirm relationships between water treatment and surrogate measures of performance (i.e. the indicators outlined in Section 5.3).
- Study **changes in wetland characteristics and function and performance over time**, including comparison between mature systems approaching their nominal lifespan and younger systems. Investigate indicators and the timeframe of wetland life, including limited water quality monitoring within mature wetlands. Studies should include a variety of ages and a long-term water quality monitoring program should also be undertaken in select systems (even if only outflows or inflow and outflows are monitored).

- Investigate the **performance of invasive or dominant plant species** to determine if they can out-perform other species to meet water treatment objectives.

10 Conclusions

Based upon the literature review and modelling study the key conclusions from this investigation are as follows:

- Effective designs promote high contact between stormwater and wetland components, wide flow distribution across the wetland width and provide suitable hydrology to support the desired plant configuration.
- Constructed wetland performance is highly sensitive to poor design. This is illustrated by the feedbacks and interdependencies in the relationships between vegetation, hydraulics and bathymetry, all of which strongly dictate water treatment.
- The calibration parameters (k and C*) in the MUSIC model are relatively insensitive to adjustment within their default range in a well-designed system. The model output is also relatively robust to adjustment of the parameter representing hydraulic efficiency (N_{CSTR}). However, the representation of performance in MUSIC for wetlands in poor condition remains unknown.
- To avoid short-circuiting and poor hydraulic efficiency, wetland bathymetry and vegetation should be consistent in pattern across the width of the wetland. Features should be either continuous and uniform (e.g. vegetation bands), or with small-scale and randomised variation (e.g. microtopography).
- Plants influence multiple aspects of wetland function and their beneficial role has a firm basis, despite some negative influences reported for dense vegetation or in association with seasonal senescence.
- Algae and biofilms also contribute significantly to water treatment alongside vegetation. However, little data is available to quantify the potential downsides from nitrogen fixation or export of suspended organic particulates downstream, nor to guide proactive management of algae and biofilms in constructed wetlands.
- While much can be gained from understanding the structure and function of natural wetlands, not all characteristics are beneficial given the context and specific objectives of constructed treatment wetlands. For example high plant species diversity is not a prerequisite for effective nutrient removal. In addition, wetting and drying fluctuations benefit overall vegetation resilience, but may reduce the productivity of some species and lead to substantial nutrient release, if an optimal regime is not first identified.

- Microtopographic variation is a characteristic of natural wetlands that has demonstrated treatment benefits in constructed wetlands. However, further understanding is required to determine its optimal form and the practicalities of its incorporation during construction.
- The substrate provides a vital function supporting vegetation and harbouring microbial processes. Good designs ensure sufficient depth and quality of substrate, and management protocols should protect the substrate from erosion.
- Wetland functionality can be maintained despite significant changes in the vegetation over time.
- Retention within shallow vegetated zones provides the majority of water treatment, but inclusion of limited open water zones benefit hydraulic efficiency and can provide a minimal degree of treatment from sediment processes and algal assimilation.
- Stormwater treatment wetlands may function effectively up to and beyond 20 – 25 years, but some designs will export sediments and nutrients much earlier in their life. Long-term storage in the sediments or gaseous losses (e.g. denitrification) are critical to sustain net retention of contaminants. Overall, good design and operational practices, based upon the principles outlined above, are key to long-term performance efficiency.

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Appendix A - Watson 2014 – *MUSIC stormwater modelling; a calibration study*