2.2 Hydrology of urban freshwater wetlands

Prof Paul Boon
Institute for Sustainability & Innovation
Victoria University
Footscray Park, Melbourne VIC
Australia

Abstract

Hydrology is a critical factor to consider when defining, delimiting, and classifying wetlands. A number of criteria are often invoked, but common to all approaches is the presence of shallow and still or slowly moving waters for at least some of the time. Indeed, hydrological criteria dominate many of the typologies used to classify wetlands, both in the Northern Hemisphere and in Australia. Three typologies are provided as examples: the hydrogeomorphic approach used widely in the USA; a system used in an early Australia-wide classification of wetlands; and the Norman and Corrick typology used in Victoria as the State-endorsed system. Not surprisingly, the biota of Australian wetlands is mostly well adapted to a variable hydrology, in other words to fluctuating water levels and to periodic wetting and drying, and the most common of these adaptations are outlined. Hydrological factors are important also in mediating many of the ecological processes that take place in wetlands, including rates and sources of primary production, the structure of food webs, and the rates of nutrient and biogeochemical cycling. Hydrological manipulations are a crucial feature in many wetland management regimes, and ecological benefits can often accrue by introducing alternating wet and dry phases (of appropriate duration) and fluctuating water levels in natural and man-made wetlands. These ecological benefits, however, need to be offset against other management imperatives that operate in urban settings, including aesthetic considerations, and a small number of potentially significant hazards when attempting to manipulate water levels in wetlands, be they natural or constructed.

Introduction: why is hydrology so important in wetlands?

Wetlands are notoriously difficult to define, but in every definition that I know of there is mention of the pivotal role played by some aspect of hydrology. Let's look at a few definitions as examples.

In their introduction to a widely used textbook on freshwater and estuarine wetlands, Batzer and Sharitz (2006) began by noting the Oxford English Dictionary defines a wetland as:

'*... an area of land that is usually saturated with water, often a marsh or swamp*' (underlining mine).

The highly influential report by the US Fish and Wildlife service on American wetlands (Cowardin *et al*. 1979; see also Cowardin and Golet 1985) defined wetlands as:

'*... lands transitional between terrestrial and aquatic systems where the water table is usually at or near the surface or the land is covered by shallow water...*' (underlining mine).

On a more international scale, Article 1.1 of the Ramsar Convention on wetland conservation (Ramsar Convention Secretariat 2006) defined wetlands as:

'*... areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres*' (underlining mine).

The origin of the 6 m depth limit is obscure, but it is thought to be based on the maximum depth to which sea ducks can dive whilst feeding (Ramsar Convention Secretariat 2006).

In one of the first nation-wide surveys of types and distributions of wetlands across Australia, Paijmans *et al*. (1985) defined wetlands as:

'*...land permanently or temporarily under water or waterlogged. Temporary wetlands must have surface water or waterlogging of sufficient frequency and/or duration to affect the biota. Thus the occurrence, at least sometimes, of hydrophytic vegetation or use by waterbirds are necessary attributes*' (underlining mine).

Howard-Williams (1985, p. 393), in an old but well-cited article, defined wetlands as:

'*an area where the water table is at or above the land surface for long enough each year to promote the formation of hydric soils and to support the growth of aquatic vegetation much of which is emergent (photosynthetic organs above the water surface)*' (underlining mine).

More recently, Keddy (2010, p. 3) adopted the following definition:

'*... an ecosystem that arises when inundation by water produces soils dominated by anaerobic processes, which in turn, forces the biota, particularly rooted plants, to adapt to flooding*' (underlining mine).

What can we draw from this brief survey of how wetlands can and have been defined? I think it is that, despite all the problems with their formal definition, wetlands commonly share four characteristics:

1. Shallow water is present, either at the surface or within the root zone, for at least some of the time;
2. The water moves very slowly or is static (i.e. wetlands are lentic environments), unlike the case with flowing-water (lotic) aquatic systems;
3. Water-logging produces wetland soils that are reducing or at the least anaerobic, quite unlike 'normal' terrestrial soils that are oxic; and
4. Vegetation is adapted to water-logging and/or flooding, and plants not tolerant of inundation are largely absent. Plants adapted to wet conditions are often called 'hydrophytes', and may be emergent or submerged.

This small sample of definitions is surveyed because it shows it is impossible to define, let along delimit, classify or understand, wetlands unless we first understand the hydrological setting in which they occur. In every example, we see a requirement for water to be present, for at least some of the time, and that when it is present the water is i) shallow and ii) not moving.

Both these fundamental characteristics are shared by wetlands and by shallow lakes. What then differentiates these two sorts of aquatic environment? Here there is a great advantage in using a definition like Howard-Williams' (1985), which includes a vegetative component, over one like Paijmans *et al.* (1985), which does not. In other words, wetlands and shallow lakes share a set of underlying hydrological similarities, but can be pragmatically differentiated on the basis of the former possessing emergent plants and the latter not possessing them. To many people, a 'classic' wetland is one that has at least some emergent vegetation, whereas a shallow lake possesses only submerged vegetation, with any emergent vegetation mostly limited to the edges. Even so, the overlap is considerable. Figure 2.2.1 shows three different types of wetlands that may be found in urban settings, and illustrate the distinctions I am trying to draw.

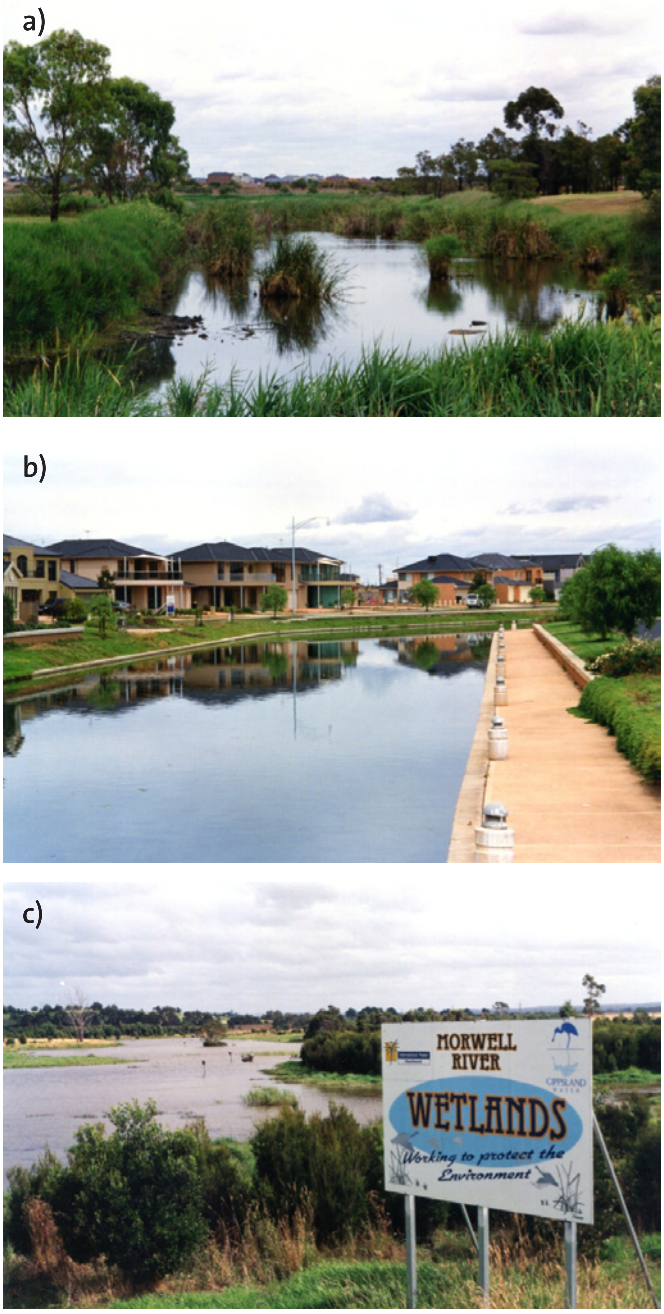

Figure 2.2.1. Three types of shallow aquatic systems common in urban settings: a) a 'classic' constructed wetland, with abundant emergent vegetation, used to treat land run-off; b) a shallow lake system, lacking emergent vegetation but possessing abundant submerged vegetation (in this case mostly *Ruppia* spp.), built mostly for aesthetic reasons; and c) a wetland constructed in the bend of a translocated waterway to replace a wetland lost through development. Photographs a) and b) come from the Sanctuary Lakes development on the south-western peri-urban fringe of Melbourne; photograph c) on the Morwell River, near the outskirts of Traralgon, eastern Victoria.

Characteristics of shallow aquatic systems

Because wetlands and shallow lakes are now so common in urban environments, especially in newly created housing developments in the peri-urban fringe of our cities, it is worth looking briefly at how they differ from other types of aquatic systems, and in particular how wetlands and shallow lakes collectively differ from other, much better studied aquatic systems such as deep lakes and reservoirs.

On a global basis, shallow aquatic systems are far more common than are deep lakes or reservoirs. Wetzel (1990), for example, showed that on a global scale there are only about 10 lakes with a mean depth of >100 m and between 10^4 and 10^5 lakes with a mean depth of 10 m; in contrast, there are between 10^7 and 10^8 lakes with a mean depth of 1 m. Similarly, Cooke *et al.* (2005) noted that the mean depth of the 309 lakes managed by the US Army Corps of Engineers in the USA was only 4.5 m, and the average maximum depth was only 10.7 m.

Despite their ubiquity, shallow aquatic systems have often been neglected as a focus of limnological research and, instead, much of our historical understanding of lake ecology has come from studies on deep, permanent lakes in the temperate zone (Williams 1988). Nevertheless, considerable progress has been made with shallow-lake research over the past 20–30 years, and this research has revealed consistent differences in the physico-chemical properties of shallow and deep aquatic systems (e.g. see Moss 1990, 1998, 2003; Nixdorf and Deneke 1997; Scheffer 1998; Håkanson 2004; Phillips 2005). The research has indicated that shallow and deep systems differ fundamentally in two ways:

1. Type of thermal stratification in the water column, which is often one of polymixis in shallow systems and dimixis or other less complex patterns in deeper systems (e.g. see Lewis 1983 versus Ford *et al.* 2002)

2. Degree of light attenuation and the degree to which light penetrates to bottom sediments, which in turn is linked to differences in the presence of submerged (rooted) aquatic plants (e.g. Chambers and Kalff 1985).

These underlying primary differences translate into a number of important secondary ecological differences, for example:

- submerged aquatic macrophytes have the potential to cover the entire sediment surface, even the sediments in the deepest parts of a shallow lake or wetland;
- emergent aquatic macrophytes have the potential to grow across the entire wetland, unlike the case in deep lakes where they are limited to the shallow fringes;
- the macrophyte communities are relatively stable to the effects of nutrient enrichment because of the strong feed-forward processes operating in stable-state communities;
- once critical thresholds have been exceeded, however, plant communities can shift rapidly and unexpectedly from a macrophyte-dominated system to a system dominated by algae (either in the water column as periphyton, on the sediments as algal mats, or attached to plant surfaces as epiphytes) or by floating plants such as the fern *Azolla;*
- sediment-water column interactions are intense in shallow lakes, and affect patterns of nutrient regeneration and release, sediment resuspension, and water-column turbidity; and
- nutrient budgets in wetlands and shallow lakes are often dominated by internal loadings from the sediments, rather than external loads derived immediately from the catchment.

Are wetland hydrology and water regime the same thing?

Hydrology is such a powerful controller of wetland ecosystems that there are many excellent – if now old – reviews on the topic (e.g. Gosselink and Turner 1978; Orme 1990; Mitsch and Gosselink 1993; Vymazal 1995; Wheeler 1999; Jackson 2006). In particular, there is a robust and growing literature on the effects of hydrology on wetland and riparian plants, including a number of valuable Australian syntheses (e.g. Deil 2005; Colmer and Voesenek 2009; Bornette and Puijalon 2011; Roberts and Marston 2011; Rogers 2011; Webb *et al.* 2012).

Linked inextricably to the term ‘wetland hydrology’ is the term ‘wetland water regime’. What is a wetland’s water regime? Water regime is clearly not a single characteristic but, instead, covers a multitude of attributes. Bedford (1996) argued that there were three core variables to understand when describing the hydrology of a given system:

1. Source of the water, specifically the relative inputs from precipitation, surface water and groundwater;
2. Quality of water, primarily the ionic content (salinity and cation/anion ratios), supplemented by other water-quality variables such as nutrient concentration, pH and suspended load (or turbidity); and
3. Spatial and temporal characteristics of the wetland’s wetting and drying cycle. This is a complex variable which includes the frequency, duration and timing of inundation, the rate of water rise and fall, the maximum water depth and so on. Bedford (1996) termed this factor “wetland hydrodynamics”.

The first of these variables – the source of water – has long been recognised as a critical factor in controlling the type of wetland that develops under a given climatic regime (Brinson 1999; Cronk and Fennessy 2001). Bogs, for example, are peat-accumulating wetlands with no discernable inflows or outflows of surface water, whereas fens are peat-accumulating wetlands that receive some drainage from their surrounding catchment (Mitsch and Gosselink 1993). Other wetland types receive water from their parent river: wetlands on the floodplains of large (usually lowland) rivers commonly receive their water when their parent river is in flood. Many wetlands in semi-arid and mediterranean regions are surface expressions of shallow unconfined groundwaters, and changes in the level of the surrounding watertable have a large impact on the level of water in these types of wetland that are hydraulically connected with the groundwater.

The central role played by the source of water is acknowledged in the hydrogeomorphic approach to wetland classification, delimitation and functional assessment. This approach, first devised in the USA in the late 1990s by the late Mark Brinson (e.g. see Brinson 1999), is used across the USA to classify wetlands and to make quantitative assessments of their condition and integrity (e.g. see Shaffer *et al.* 1999; United States Department of Agriculture 2008; Brooks *et al.* 2011). Figure 2.2.2 shows how the hydrogeomorphic scheme conceptualizes

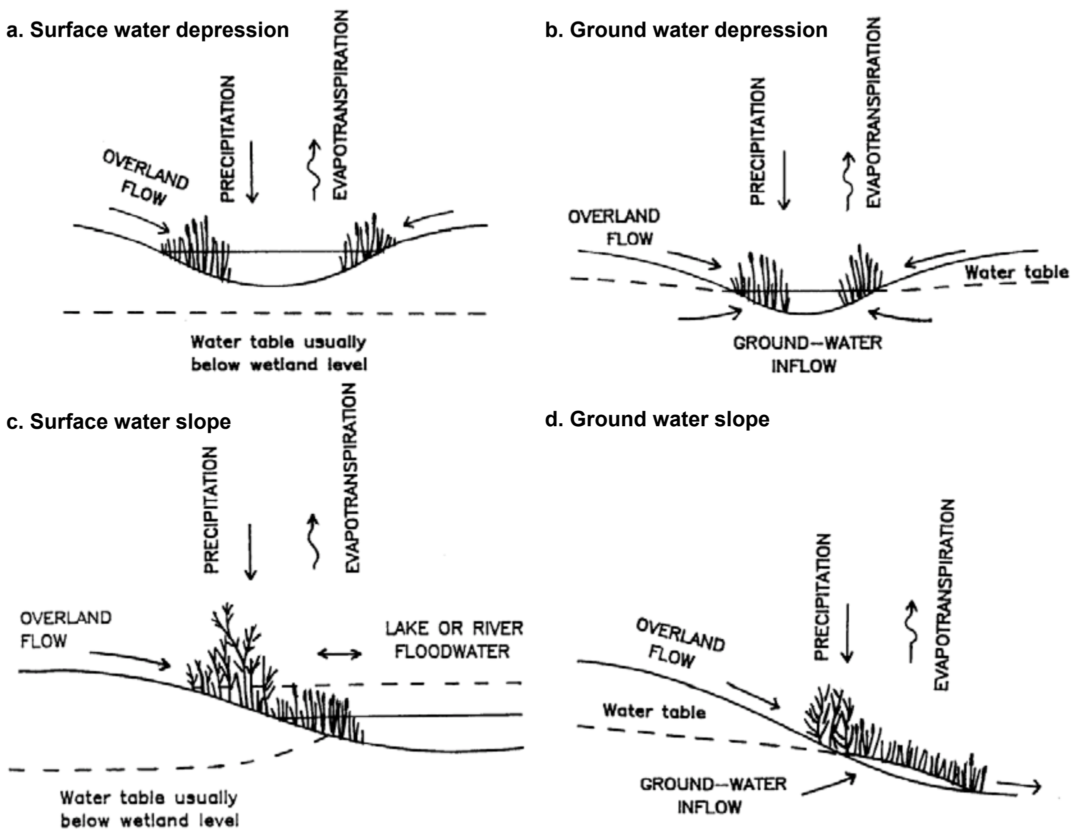

Figure 2.2.2. Development of four wetland types on the basis of the relative importance of surface water versus ground water inputs. Source: modified from Brinson (1999, Figure 2). Reproduced with permission of the US Army Corps of Engineers (permission pending).

surface-water and groundwater flows in controlling the development of four distinct classes of wetlands.

The source of inundating water plays a significant role in influencing the second of Bedford's (1996) variables – water quality. Moore (1990) outlined the ways in which energy patterns, nutrient cycling and water quality were controlled by wetland hydrology; this early treatment has been updated by Boon (2006). As an example of the importance of water source, since raised ombrotrophic bogs receive all their water via precipitation they have no external supply of nutrients other than that provided by rainfall. As a result they are relatively nutrient-poor wetlands, characterised by low concentrations of plant nutrients and exchangeable cations. In contrast, mineralotrophic fens, which lay at lower points in the landscape, receive water that has previously passed through mineral soil and so are relatively enriched in plant nutrients (Mitsch and Gosselink 1993). Water quality plays a critical role in structuring the ecology of wetlands and shallow lakes in urban settings, where high loads of plant nutrients can have adverse impacts on the vegetation (e.g. see Boon and Bailey 1998; Morris *et al.* 2003 a, b; 2004; 2006 for Australian examples).

As shown in Figure 2.2.2, not all natural wetlands receive their water (or their nutrients) as surface flows. Wetlands subject to movements of the groundwater are called 'rheotrophic', and in these cases the groundwater can be responsible for the importing of organic carbon and of nutrients as well as supplying almost all the wetland's water inputs (Moore 1990). Clearly, groundwater-fed wetlands may experience vastly different ionic and nutrient relationships to wetlands that are inundated by surface-water flows and, in fact, constancy in water quality is a factor that can be used to infer, at least at a preliminary stage, the relative importance of surface-water (highly variable) to groundwater (less variable) inputs to wetlands and shallow lakes. It is unlikely that urban wetlands would be sited in areas of groundwater discharge (as footings would then be near-impossible to dig, and houses and gardens would often become flooded), but the potential for significant groundwater inputs to wetlands still deserves mention.

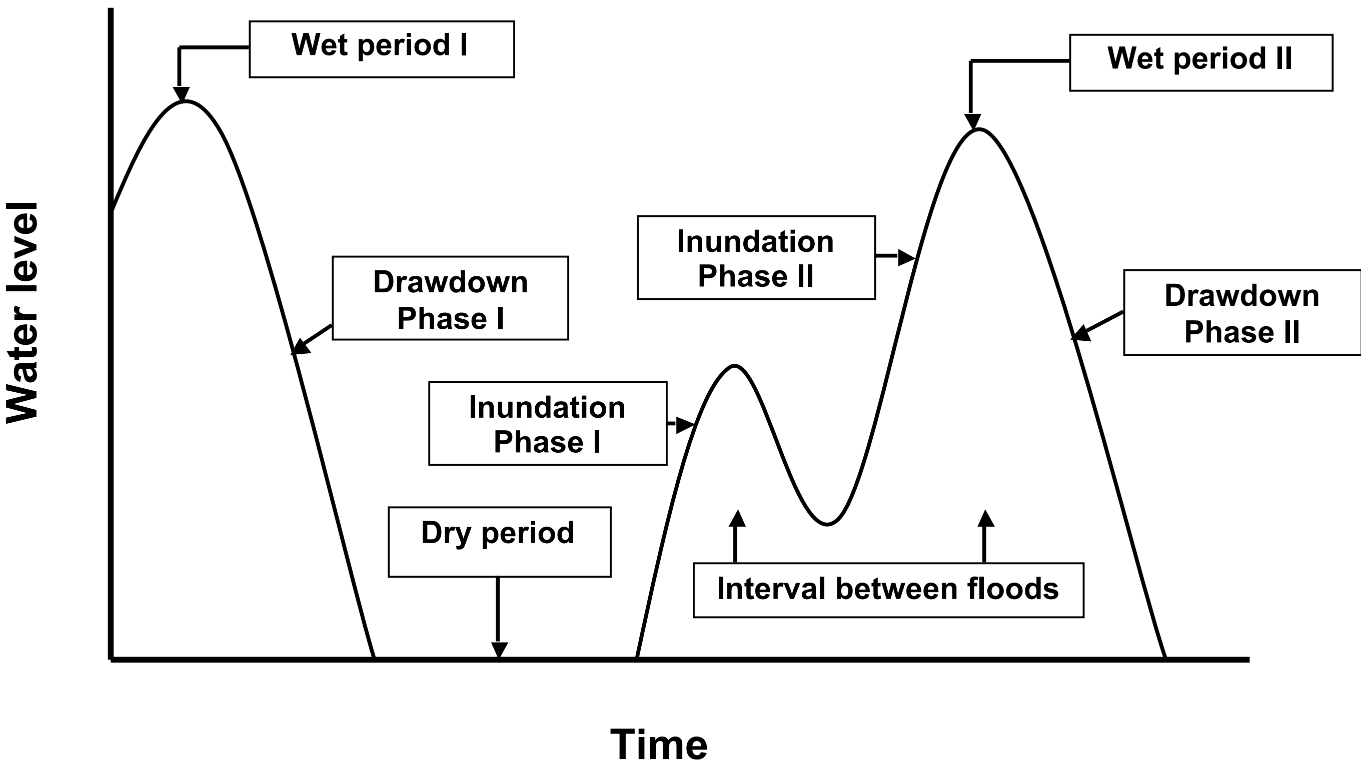

Figure 2.2.3. Critical components in wetland hydrology. Source: redrawn from Boon (2006).

The relative availability of surface waters and groundwaters is important also for the performance of vegetation that surrounds wetlands and other shallow bodies of water, especially given that the two water sources often vary greatly in their salinity and nutrient content. A good example is provided by the Swamp Paperbark (*Melaleuca halmaturorum*, Myrtaceae), which in South Australian wetlands uses both the relatively fresh surface waters and the more variably saline ground waters, the ratio to which each is utilized being dependent on the prevailing weather, the presence of surface waters, and the position of the water table (Mensforth and Walker 1996). Similar relations hold for the widely distributed River Red Gum (*Eucalyptus camaldulensis*) and Black Box (*Eucalyptus largiflorens*: both in the family Myrtaceae): see Eamus *et al.* (2006).

The third of Bedford's (1996) three hydrological variables – wetland hydrodynamics – is perhaps the most complex. Figure 2.2.3 shows some the critical components of wetland hydrodynamics and the way in which the water levels vary with time in a temporary wetland. The shape of the annual hydrograph is important (e.g. rise times and fall times) but so too are long-term characteristics, such as the frequency and reliability of floods over the period of decades-to-centuries (Boulton and Brock 1999). Long-term hydrodynamics are especially significant for long-lived species that might recruit only rarely, such as clonal plants (e.g. see Robinson *et al.* 2006). The length of time for which a wetland is inundated in sometimes called the 'hydroperiod', but this terminology has a strongly Northern Hemisphere ancestry where seasons are distinct and largely reliable, and it is by no means clear how well it can be applied to the highly variable systems that are common across much of Australia.

Hydrological criteria and wetland classification

Hydrology is also an important factor to consider when attempting to classify wetlands. As Table 2.2.1 shows, wetlands have often been classified with reference to their hydrological regime, generally with reference to terms such as 'permanent', 'seasonal' and 'episodic' to provide a very high level classification. The system shown in Table 2.2.1 groups wetlands largely on the basis of the predictability and duration of filling.

Hydrological criteria are used also in the State-endorsed scheme used to classify wetlands in Victoria. This classification system – known as the Norman and Corrick scheme – has its origins in 1975 in a detailed survey of wetlands in the Gippsland region (Corrick and Norman 1980; Norman and Corrick 1988). Although the original classification system was specific to Gippsland

Table 2.2.1. Simplified classification of temporary wetlands. Adapted from Boulton and Brock (1999) and Paijmans *et al.* (1985).

Wetland type	Predictability and duration of filling
Ephemeral	Filled only after unpredictable rainfall and runoff. Surface water dries within a couple of days of filling and seldom supports macroscopic aquatic life.
Episodic	Dry most of the time, with rare and very irregular wet phases that may persist for months. Annual inflow is less than minimum annual loss in 9 years out of 10.
Intermittent	Alternately wet and dry, but less frequently and less regularly than seasonal wetlands. Surface water persists for months to years after filling.
Seasonal	Alternately wet and dry every year, according to season. Usually fills in the wet part of the year and dries predictably every year during the dry season. Surface water persists for months, long enough to support macroscopic aquatic life. Biota adapted to desiccation.
Permanent	Predictably filled although water levels may vary across seasons and years. Annual inflow is greater than minimum annual loss in 9 years out of 10. May dry during extreme droughts. Biota generally cannot tolerate desiccation.

and orientated towards the use of wetlands by waterbirds (Pressey and Adam 1995), it provided the basis of the State-wide wetlands inventory held by the Victorian Department of Primary Industries and Environment.

Wetlands are classified under the Norman and Corrick scheme in a pseudo-hierarchical classification, based initially on salinity and the depth and permanence of water (Table 2.2.2). At the coarsest level (Category), wetlands are classified on the basis of three water depths (0.3 m, 0.5 m and 2 m) and the duration of inundation (4 months, 8 months and permanent). It is acknowledged that even 'permanent' wetlands can dry out every 4–5 years. Subcategories are then defined, within Categories, on the basis of a mixture of structural and floristic descriptors (Department of Conservation, Forests and Lands 1988; see also Department of Sustainability and Environment 2005). A similar system based initially on hydrological criteria was used much earlier to classify waterbird habitat in Victoria (Cowling 1977).

Wetland hydrology and its ecological implications

The water regime shown in Figure 2.2.3 is clearly a simplified and idealized model. Figure 2.2.4 shows how this information can be translated into a real-life situation, where the zonation of different types of plants is controlled in large part by fluctuations

Table 2.2.2. Summary of the Norman and Corrick scheme used for wetland classification in Victoria. Source: Department of Conservation, Forests and Lands (1988).

Wetland category	Water depth (m)	Inundation criteria
Flooded river flats	< 2	
Freshwater meadow	< 0.3	< 4 months per year
Shallow freshwater marsh	< 0.5	< 8 months per year
Deep freshwater marsh	< 2	Permanent (but may dry out every 4-5 years)
Permanent open freshwater	< 2 (shallow) > 2 (deep)	Permanent Permanent
Semi-permanent saline*	< 2	< 8 months per year
Permanent saline*	< 2 (shallow) > 2 (deep)	Permanent Permanent

**'saline' is defined as >3 g L^{-1} throughout the year*

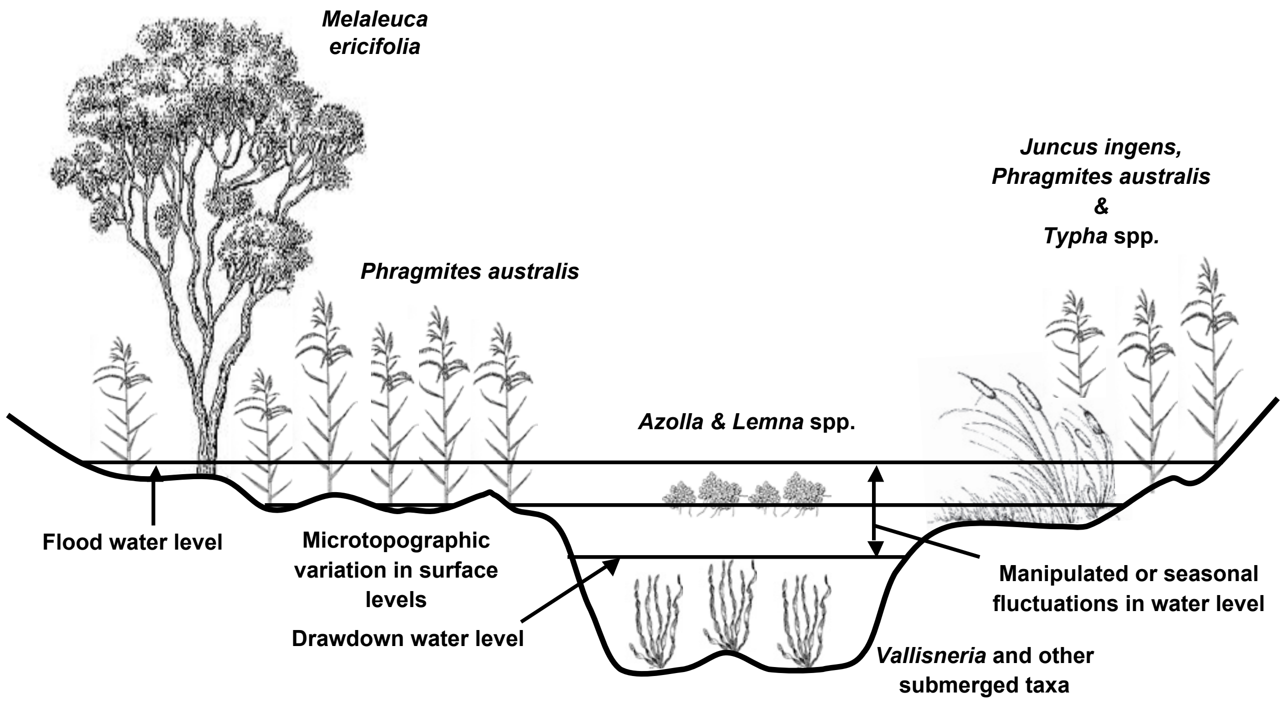

Figure 2.2.4. Ecological consequences of water regime for the growth and zonation of wetland plants. Plant taxa shown are ones that occur commonly in wetlands in south-eastern Australia.

in water levels and by the resultant different wetting and drying regimes experienced in different parts of a wetland.

Although hydrology is probably the single most important determinant for the establishment and maintenance of specific wetland types and is critical in determining the range of plants that occur, it is a mistake to consider wetlands as merely passive players in their relationship with water. Figure 2.2.4 tends to suggest such a one-way interaction but, as Vymazal (1995) has pointed out, the biotic components of a wetland may control its hydrology through a variety of mechanisms, including the generation of peat, trapping of sediment, vegetative shading, and altered rates of evapotranspiration as a result of the presence of large beds of emergent macrophytes.

Over the past five years or so it has become increasingly apparent that emergent vascular plants in wetlands modify not only the water regime they experience but also many of the biogeochemical processes that operate in a wetland. They do this by creating microtopographical relief in the profile of the bed of the wetland and in its sediments. As an example of the process, we have shown recently that large emergent plants (mainly Common Reed, *Phragmites australis,* and Swamp Paperbark, *Melaleuca ericifolia*) create a complex series of hummocks and hollows in wetlands that fringe the Gippsland Lakes in eastern Victoria, and that this small variation in relief then results in small-scale variations in water regime across the wetland (Raulings *et al.* 2010, 2011). In turn, these small-scale variations in topography help structure vegetation mosaics, perhaps via a process of positive feedback whereby certain plants grow better on the slightly elevated mounds (e.g. Peach and Zedler 2006; Raulings *et al.* 2010). The importance of microtopographic variations to wetland rehabilitation is now acknowledged (Larkin *et al.* 2006; Boon 2011), and there is growing awareness of the role that it plays also in biogeochemical processes such as sulphur metabolism (Stribling *et al.* 2007) and nitrogen cycling (Wetzel *et al.* 2011; Wolf *et al.* 2011) in wetlands.

The existence of microtopographic relief in wetlands has implications also for wetland classification. The classification schemes shown in Tables 2.2.1 and 2.2.2 are really 'whole-of-wetland' approaches that require an assumption that the entire wetland has just one, overarching water regime. This is, of course, unrealistic; different parts of the wetland will experience different wetting and drying cycles, according to whether they are on the edges (and thus shallowest) or in the middle (and thus deepest) parts of a wetland. The recent identification of the role played by microtopographic relief further modifies this simplistic approach, by recognizing that depressions (causing deep areas) and hummocks (causing shallow or even exposed areas) can exist across all parts of a given wetland. Thus it is not merely the edges that are shallow and the centre that is deep: incredibly complex mosaics of deep and shallow regions can occur across a wetland at scales of centimetres to tens or even hundreds of metres. The implication of these newest findings is that the water regime experienced in a wetland is scale-specific and, except for the broadest attempts at classification, it is very hard – perhaps impossible – to describe a wetland's water regime with a single descriptor such as 'permanent', 'seasonal' and 'episodic'. Raulings *et al.* (2010) shows clearly how different parts of a large (1500 ha) wetland can posses quite different water regimes, in spite of the wetland being classified in the Victorian classification scheme as 'permanent freshwater'.

Hydrology also influences – and even may control – many other aspects of the ecological structure and function of wetlands. As shown in Figure 2.2.3, for example, water regime strongly affects the growth of submerged and emergent plants. Hydrology also influences the way plant material is entrained into wetlands from the surrounding catchment, and the relative importance of vascular plant tissue versus algal biomass to wetland (metazoan and microbial) food webs. This process is particularly important in arid-zone, mediterranean, and wet-dry tropical systems, where episodic, intermittent or seasonal inundation creates pulsed inputs of vascular plant material (e.g. of leaf litter) from the floodplain, followed by pulses of autochthonous algal productivity in the pools that remain as water levels drop (Burford *et al.* 2008; Warfe *et al.* 2011). These pulses of inundation and of organic matter inputs are then reflected in pulsed rates of organic matter decay and methane emission (Mitsch *et al.* 2010; Harms and Grimm 2012). They are reflected too in the activity rates of extracellular enzymes in floodplain soils (Burns and Ryder 2001).

One element of a wetland's water regime – the much slower rise and fall of water levels with rain and evaporation – creates a shifting mosaic of ponded, wet, damp, and dry sediments from the centre of the wetland to its periphery. These different environments represent ecotones: zones of transition between adjacent ecological systems (see van der Valk 2012 for a good review of wetland vegetation and ecotones). Where there is a strong hydrological shift across different spatial or temporal scales, a series of ecotonal environments is often created. Brock and Casanova (2000), for example, identified three different types of wetland environments in Australian systems on the basis of these differences in water level:

1. Submerged environments, vegetated with plants that do not tolerate drying out;
2. Amphibious environments, where plants tolerate flooding and desiccation; and
3. Terrestrial environments, vegetated with plants that are intolerant of inundation.

On intermediate time scales, a seasonal rise and fall in groundwater levels has a crucial role in mediating biogeochemical processes in wetlands, including rates of methane uptake and release, as well as in nitrogen and sulfate cycling (Boon 2006; Bougon *et al.* 2011). These processes are quite well understood for boreal wetlands, but are likely to operate also in many of the climates that occur in Australia. Temporary wetlands in areas of the Swan Coastal Plain in Western Australia, for example, experience a semi-arid or mediterranean type of climate with very clear distinctions between wet and dry periods; these wetlands are overwhelmingly surface expressions of a shallow unconfined ground-water aquifer (Froend *et al.* 1993). Water levels in these wetlands vary markedly according to changes in the level of the watertable, the latter being controlled by climate and, increasingly, the degree to which groundwater is abstracted for human use (Balla and Davis 1993).

Finally, it is worth pointing out that the biota of Australian wetlands is often well adapted to a variable hydrology, in other words to fluctuating water levels and to periodic wetting and drying. Table 2.2.3 shows some of the adaptive responses of different biotic groups to variable water regimes in Australian wetlands.

There are good reasons why the Australian biota is so well adapted to fluctuating water levels and to periodic wetting and drying; droughts and flooding rains. As I write this chapter, Melbourne is experiencing only its second 'wet' winter since the recent, decade-long drought that covered most of eastern Australia; meanwhile, large parts of Queensland are flooded and northern Australia is experiencing unseasonable rain during its supposed 'dry' period. The drought we lived through in Victoria commenced in 1997 and broke only in 2010; it officially started later – 2002 – in New South Wales and in southern Queensland, but broke

Table 2.2.3. Responses of different biotic groups to variable water regimes in Australian wetlands.

Biota	Inundation phase	Drawdown phase	Dry phase
Bacteria	• Shift to anaerobic metabolism • Production of methane and S^{2-} • Decomposition of organic material • Increased rates of nutrient cycling	Production of resistant stages (e.g. spores)	• Death of some taxa and release of nutrients • Shift to aerobic metabolism • Possible replacement by fungi as primary decomposers
Phytoplankton	• Growth and reproduction • Uptake of nutrients	Production of resistant stages (e.g. akinetes)	Death of vegetative stages and release of nutrients
Aquatic and riparian angiosperms	• Germination • Growth and reproduction • Uptake of nutrients	• Death of obligately aquatic taxa • Continued growth of semi-terrestrial taxa • Production of seed bank (in soil or plant canopy)	• Death and nutrient release from aquatic taxa • Survival of riparian taxa (possibly utilizing groundwater as water source) • Seed bank as dormant and desiccation-resistant stage
Terrestrial angiosperms	• Death • Decomposition		Colonisation, growth and reproduction
Zooplankton	• Imported with flood waters and/or germination from resting stages in sediments • Growth and reproduction • Consumption by fish and other aquatic grazers	Production of desiccation-resistant stages (e.g. spores)	Death of adult stages
Aquatic macroinvertebrates	• Deposition of eggs by aerial adult stage • Emergence from refugia in sediments • Growth and reproduction • Processing of terrestrial organic material	Production of dormant and desiccation-resistant stages	• Survival as terrestrial (aerial) adult phase • Survival in sediments as dormant and desiccation-resistant stage
Terrestrial invertebrates	• Death via drowning • Consumption by waterbirds		• Colonization of dry wetland soil • Growth and reproduction
Waterbirds	• Migration for feeding and breeding • Consumption of terrestrial plants and animals	• Breeding complete • Feeding on green herbage along receding water	• Death • Dispersal to other wetlands

at roughly the same time as it did down here (see Bureau of Meteorology 2012). So droughts, even prolonged droughts that can last over a decade, are not uncommon.

It's important to remember also that much of Australia is arid or semi-arid, and that large parts of it receive vanishingly little rainfall. On a nation-wide basis, Australia's average rainfall is only 455 mm per year; the value for Europe is 640 mm, and for North America is 660 mm. Even the average annual rainfall for the continent of Africa – 690 mm – is markedly greater than Australia's. To make matters worse in terms of water resources, the percentage of the meagre rainfall that eventually ends up as stream flow is also smaller in Australia than elsewhere in the world. Here only 11% of the total precipitation (i.e. rain and snow) that falls on the land (on a nation-wide scale) finds its way into rivers: 88% is lost via evaporation and 1% goes to recharge groundwaters. In North America and Europe, only about 60% of total precipitation is lost to evaporation (Smith 2998; Pigram 2006). What these two processes – low rainfall and high evaporation – translate into is that mainland Australia has the smallest annual river run-off of any continental land mass; and this affects the hydrology of wetlands.

It is also nearly a truism that rainfall in much of Australia is notoriously unreliable. In 1992, Professor Tom McMahon and co-workers at the University of Melbourne collated a global database of annual runnoff data for nearly 1,000 catchments across the world. Their analysis showed two things. First, annual rainfall in Australia was far less predictable than that in most other regions of the world. A similar finding had been reported five decades ago by Professor G.W. Leeper in The *Australian environment*, which was first published in 1949. Building on earlier work by Griffith Taylor, he showed that only the extreme south-west of Western Australia, the southern-most parts of Victoria and western Tasmania had more-or-less reliable rainfall; the rest of the continent was characterized by alarmingly unpredictable rainfall. Newsome *et al*. (1996) have explained what this variability means for the terrestrial and aquatic plants and animals that live in central Australia, away from the more predictable maritime coast where most of our population is based. Second, the variability in rainfall detected by McMahon *et al.* (1992) translates quickly into enormous variability in river flow. The statistical term used to describe inter-annual variability is the coefficient of variation; the higher the value, the more variable, or the less predictable, the process being quantified. McMahon *et al*. showed that the coefficient for runoff (C_{vr}) in Australia was 0.70; for Europe it was only 0.29. The global average was 0.43.

What are the management implications of these climatic and hydrological characteristics? The first is that we should consider periods of low rain cum drought and periods of high rain, sometimes resulting in floods, as a natural part of our environment. Our wetlands have developed in such a climatic and hydrological milieu, and the biota is well adapted to the resultant physico-chemical conditions. The second is that we can – and probably should – manage wetlands in ways that are consistent with this climatic and hydrological variability. I come back to this matter at the end of the chapter, with some specific recommendations. The third is that we should consider hydrological variability within a historical perspective. Some recent and lovely research has shown that the colony of 1788 at Sydney Cove was, in fact, established during a period of highly variable weather and that water scarcity profoundly shaped the development of early Sydney (Gergis *et al*. 2009, 2010). My favourite river – the Hawkesbury, just to the north of Sydney – experienced a sequence of dry periods and of catastrophic floods throughout the 18th and 19th centuries, which again controlled agricultural activities and urban expansion (Nichols 2001; Barkley-Jack 2009). In fact, the very location of now populous towns such as Windsor, Richmond, Pitt Town and Wilberforce, was predicated by Governor Lachlan Macquarie on their being not subject to flooding (Bowd 1994). To conclude: floods and drought are a natural feature of most of the Australian landscape, and we would do well to manage our wetlands in the light of this fact.

Quantifying the hydrological budget of a wetland

At its simplest, the amount of surface water in a wetland at a given time is merely a reflection of the amount of water entering the wetland versus the amount leaving, taking into account the volume at time zero (Jackson 2006):

Volume = Storage + Inputs – Outputs

A hydrological budget is a simple model of the inputs and outputs of water to a wetland.

Typically, there are three sources of water to a wetland:

1. Precipitation falling onto the wetland (P)
2. Surface water flowing into the wetland (S_{in})
3. Groundwater flowing into the wetland (G_{in}).

Losses of water typically occur because of three processes as well:

1. Evaporation from open water and evapotranspiration from emergent plants (E)
2. Surface water flowing out of the wetland (S_{out})

3. Groundwater flowing out of the wetland (G_{out}).

We can combine these various components into a single equation that describes changes in water level (WL) within a wetland (modified from Gippel 1996):

$$WL = P + S_{in} + G_{in} - E - S_{out} - G_{out} \ (+ e)$$

An error term (e) is included in the above formula in recognition of the uncertainties associated with each component of the water budget. In many calculations it is not included, but this would be a mistake as the result would then be assumed to be 'correct'. Of course, all our estimates of water budgets are just that: estimates, with a suite of possible errors. Interestingly, many textbooks showing similar equations for hydrological budgets do not include an error term: naughty of them!

Direct inputs of water to wetlands via precipitation are mostly easy to estimate, provided that long-term rainfall data are available from a nearby meteorological station. Average annual precipitation inputs are then calculated simply as the average annual rainfall multiplied by the area of the wetland. Unlike the case with terrestrial systems, where stems, bark and foliage can trap rainfall and allow it to evaporate before reaching the ground, no correction needs to be made for interception losses in wetlands.

Difficulties can arise in those cases where there is no nearby meteorological station, but this is unlikely to arise in most urban areas. A cryptic problem can occur, however, if the record is short or is biased by covering only unusually wet or unusually dry periods. There is also a growing awareness that historical records may not accurately reflect current or future climates.

The amount of water that enters a wetland via direct precipitation therefore can be calculated as:

$$\text{Precipitation (ML)} = \frac{\text{Area of wetland (m}^2\text{)} * \text{Average annual rainfall (m)}}{1000 \text{ (conversion to ML)}}$$

Surface-water inflows to wetlands can come from point sources or from diffuse sources. Point-source inputs are easily estimated if the stream is gauged, but are less certain if it is not. In the case of diffuse surface-water inflows, inputs must be estimated on the basis of average rainfall multiplied by catchment area, corrected with a factor known as a 'volumetric run-off co-efficient' for each of the different land uses in the catchment. In urban areas, a large portion of incoming rainfall ends up as run-off, because of the large proportion of impermeable surfaces. In agricultural areas, by contrast, there is more open soil to absorb the water and so run-off co-efficients are smaller. Catchments dominated by industrial land uses have a high proportion of impermeable surfaces, and large run-off co-efficients.

The amount of water that enters a wetland via surface flows therefore can be calculated as:

$$\text{Surface inflow (ML)} = \frac{\text{Area of catchment (m}^2\text{)} * \text{Average annual rainfall (m)} * C_V}{1000 \text{ (conversion to ML)}}$$

Table 2.2.4 shows the range of volumetric run-off co-efficients recommended for use in the commonly used MUSIC modeling program. The volumetric run-off co-efficient is denoted by the symbol C_V. It should not be confused with the co-efficient of discharge (C), which is the factor used in the Rational Method to calculate short-term run-off from single storms with a known ARI. Values of C are usually much greater than those of C_V, and can commonly be as high as 0.8–0.9 for industrial areas (Department of Natural Resources and Water 2007).

Table 2.2.4. Values for the volumetric run-off co-efficient (C_V) used in the MUSIC modelling package. Source: Gold Coast City Council (2006, Table 3).

Land-use category	Volumetric run-off co-efficient
Forest	0.15
Rural – agricultural	0.20
Rural – residential	0.20–0.24
Urban	0.38–0.40
Commercial/industrial	0.57

Surface water flowing out of a wetland is rarely gauged, and so is usually difficult to quantify this loss term. One case where determination is straightforward is when water is extracted for a given purpose, such as for irrigation, and where good and accurate records should (in principle at least) be available. Especially problematic, however, is the case when the wetland is on a floodplain and subject to regular overbank flooding, where surface-water inflows and outflows are practically un-measurable and, in any case, dubious whether a single 'annual' figure would adequately represent large year-to-year variations in flow.

Groundwater inputs and outputs and evapotranspirational losses are also often hard to quantify reliably. Apart from detailed studies where they are specifically measured or modeled, groundwater fluxes are often determined as the 'remainder' when all the other (more easily measured) hydrological components have been accounted for. There are two drawbacks with this approach:

1. It conflates groundwater fluxes with the overall error term 'e'; and
2. It often requires the subtraction of one large and variable number from another large and variable number, with the result that individual errors are compounded.

In one of the few Australian case studies where groundwater fluxes were monitored intensively, Raisin *et al*. (1999) showed that groundwater inputs and outputs were critical for both the water balance and the nutrient balance of wetlands of north-eastern Victoria. In the case of one wetland, groundwater fluxes were responsible for ~50% of the total export of nitrogen and phosphorus. In more detailed studies, groundwater fluxes can be modeled with Darcy's Law, but only if there are good data on a range of factors, including groundwater dynamics and soil porosity.

Evaporative losses from open bodies of water can be derived from evaporation-pan data. A correction term (the 'pan co-efficient') of ~0.7 is commonly applied to account for the slower evaporation that occurs from large, open water bodies in comparison with that measured from small evaporation pans at meteorological stations (Gippel 1996).

The amount of water that is lost from a wetland via evaporation therefore can be calculated with a formula analogous to that used to estimate P and Sin:

$$\text{Evaporation (ML)} = \frac{\text{Area (m}^2\text{)} * \text{Average pan evaporation (m)} * \text{Pan co-efficient}}{\text{1000 (conversion to ML)}}$$

In this case, 'area' refers to area of the open water in the wetland, and not the area of the catchment or the total wetland. Pan co-efficients can vary widely, but typically values of 0.7 or 0.8 are used as first estimates.

As so often seems to be the case with wetlands, there are added complications to this simple model. There are very real problems with estimating evaporative losses from wetlands because of the presence of large beds of emergent macrophytes. These plants not only lose water by transpiration from aerial organs (e.g. leaves) but also modify the rate of evaporation from the surface water that surrounds the plants. Emergent plants, for example, shade the water around them and shelter it from wind, both processes tending to lower evaporative losses.

Because of these contrasting influences, it is often considered more accurate to refer to the total loss of water from the surface of wetlands vegetated with dense beds of emergent macrophytes as 'evapotranspirational losses' rather than simple 'evaporative losses'. Although evapotranspirational losses from wetlands are usually higher than simple evaporation-pan data would indicate (Gippel 1996), there are few or no data available to reliably estimate losses from well-vegetated wetlands in the Southern Hemisphere. Vymazal (1995) collated information on evapotranspirational losses from vegetated wetlands in the Northern Hemisphere, an analysis which showed that evapotranspirational losses ranged from <1 to >10 mm day^{-1}. For wetlands dominated by emergent macrophytes such as sedges and rushes, the relationship between evapotranspiration and simple evaporation ranged between 0.7 and 2.5, but if an average needed to be estimated, a value of ~1.5 is not unreasonable (see Vymazal 1995, Table 3-6). Other values have been reported by Idso (1981) and Dolan *et al*. (1984) but, even so, I find it rather surprising that the research community has not addressed the topic of evapotranspirational losses from vegetated wetlands in sufficient detail to make hydrological budgets more accurate.

Hydrology and the management of urban wetlands

What have we achieved so far in this chapter? First, I hope that you have recognized that hydrology is a critical factor to consider when studying wetlands. Second, we have seen how it informs, and sometimes is the keystone behind, many systems of wetland classification. Third, the biota of Australian wetlands shows many adaptations to living in hydrological variable environments, perhaps not surprisingly given the variability of our climate. Fourth, a method has been advanced to show how to calculate a hydrological budget for a wetland, and its limitations outlined.

It is now time to bring all these topics together and look at how hydrological considerations can be used to enlighten the management of wetlands in urban settings. The stylised hydrograph shown in Figure 2.2.3 allows us to raise a number of questions about the ecological and management significance of different wetland water regimes. Some salient issues and questions might include:

1. There can be near-overwhelming pressures to maintain high water levels in urban wetlands. This is because urban wetlands are often constructed, in part, to provide an attractive vista to residents, and people generally prefer looking over full water bodies than over drying mud flats. The management problem is that alternating wetting and drying periods are often ecologically beneficial to Australian wetlands (e.g. see Briggs 1998; Navanteri and Kambouris 2008). In fact, it is generally thought that all but the most permanent wetlands in south-eastern Australia should be drained for at least 6 months every couple of years, in order to allow soils to dry fully and biogeochemical processes to attain their end points (Boon 2006). Unless other factors intervene, drying should occur over the summer to autumn period, when the wetlands would naturally experience the high temperatures and high evaporative losses typical of summers in south-eastern Australia with a temperate or mediterranean-type of climate. Complete desiccation may be required also to control noxious fish species, such as carp. Such regimes of alternating wet and dry periods generally seek to mimic natural wetting and drying cycles, which was one of the first recommendations made by Gippel (1996) for managing water regimes in high-value wetlands in Victoria: '*Ideally, reinstate the natural hydrological regime by removing disturbing factors*' (Gippel 1996, p. 136, italics in original).
2. If it is not possible to draw-down wetlands such that they periodically dry out, at the very least water levels should be allowed to fluctuate. Fluctuating water levels allow a wide range of vegetation types to develop, as rising and falling water alternately exposes and inundates different parts of the shoreline (e.g. see Brock and Casanova 2000; Smith and Brock 2007). The well-established 'intermediate disturbance hypothesis' posits that under environmentally constant conditions one or a few well-adapted and competitively superior species will eventually become dominant: the frequent invasion by and dominance of *Typha* spp. in wetlands with constant water levels is an example. With too frequent and severe disturbance, however, no species can establish itself long enough to survive and reproduce before the next disturbance eradicates it. In the broad intermediate zone of periodic disturbance, a range of temporary habitats are created which allow a wide suite of animals and plants to co-exist in a dynamic equilibrium.

If wetlands are allowed to dry out completely or their water levels to fluctuate substantially, another set of management questions may then arise:

3. How long is the wetland dry between wet periods? The activity of wetland microbes and the biogeochemical cycling of nutrients and other elements is almost always closely linked with the growth and decay of aquatic plants, and their performance in turn is similarly linked closed with wetland hydrodynamics. Obligately submerged plant taxa, for example, can survive only short periods of drying; conversely, prolonged and complete drying may be required for sediment oxidation, liberation of potential electron acceptors, and changes in crystal mineralogy that affect phosphorus adsorption and release (Boon 2006; see also McComb and Qiu 1998).
4. How quickly does the water rise? Does it rise so quickly that submerged aquatic plants cannot extend their leaves fast enough to remain in the photic zone, maintain a positive carbon balance, and continue to oxygenate their rhizosphere?
5. How deep is the water? If water is deeper than about 2 m, it is difficult for even the tallest emergent aquatic plants to keep some aerial organs exposed to permit root aeration, and the plants may drown if the water remains at this level for appreciable periods. With the death of emergent plants the aeration of below-ground organs will fail, and oxic zones around the rhizosphere may disappear. Submerged plants may become light limited if the water is too

deep, especially if it is also turbid. The death of these plants may also affect the supply of oxygen to the roots and rhizomes, and ultimately the oxygen status of the sediments and the survival of obligately aerobic bacteria in otherwise highly reducing sediments.

6. How long does the wetland retain water? If the wet period (the so-called 'hydroperiod': see above) is too short, aquatic plants will not achieve their maximum biomass or lay down long-lived desiccation-resistant propagules. This will have implications for the supply of organic substrates to wetland bacteria. If the sediments are submerged for only short periods, anoxia may not develop and plant material will be degraded primarily by oxic decay processes rather than anoxic ones.
7. How quickly does the water level drop during the dry period? Does it recede slowly, allowing fringing herbage to remain green for an appreciable time and provide a food resource for waterbirds, or does it drop rapidly and the wetland dry quickly due to evaporative losses?
8. In what season does the wetland fill with water? Is filling a natural event over the rainy season (e.g. in winter-spring in temperate and mediterranean climates, in the 'wet' in monsoonal climates) or an unnatural and out-of-season event related to the anthropogenic maintenance of high river water levels for irrigation supply? Aseasonal filling in autumn or winter may create conditions too cold for wetland plants, animals and bacteria, and biogeochemical processes may be far slower than had the wetland filled in spring or summer (e.g. Boon *et al*. 1997).
9. Can we use water-level fluctuations to control noxious fish, especially carp? An excellent reason for introducing a dry phase into chronically flooded wetlands is to control carp, *Cyprinus carpio*. Near-complete drawdown of water levels in wetlands is a common and effective means of controlling the abundance and size of undesirable fish species such as carp, as it periodically rests the population and limits the size to which adult fish can grow. The implementation of water regimes aimed at maintaining or improving biodiversity values is often consistent with a program of carp control. Local eradication of exotic fish species from a wetland may not be achievable, but a good degree of control is often possible if the rate at which carp are removed exceeds the rate at which they increase, that immigration from external sources is minimal, and if reproductive members of the population are removed (Koehn *et al*. 2000). Water-level manipulations, therefore, may be seen as one tool – albeit a critical tool – in the armory for controlling carp in wetlands.

Notwithstanding the generally strong ecological and biogeochemical reasons for allowing wetlands to dry out and their water levels to fluctuate, there are sometimes risks in implementing a draw-down phase in inland (Boon *et al*. 2009) or coastal (Raulings *et al*. 2011) wetlands. The greatest ecological hazards are when:

- potential or active acid sulfate soils are present;
- there is a risk of salinization, either when the wetland lies over shallow saline groundwater or when it is located adjacent to a saline creek or water body; and
- a high-value wetland system has evolved in response to chronic inundation and is at risk of being lost if major hydrological changes are implemented.

Of course, there is also the significant management risk of introducing a dry phase in urban wetlands of creating an aesthetic or odour problem when drawing down a previously full wetland.

Let's finish by looking at the three ecological hazards in turn:

1. Acid sulfate soils are soils that contain sulfidic materials and produce sulfuric acid when exposed to the air. For a long time it was thought that they were limited to coastal areas, and their extent determined mostly by the extent of mid-Holocene sea levels that were 1.0 to 1.5 m higher than at present (Department of Sustainability and Environment 2009; Department of Primary Industries 2011). Although over the past decade it has become apparent that they can occur in inland areas as well, probably as a result of progressive salinization (Lamontagne *et al*. 2006), it is unlikely that urban wetlands would be created in such inland environments. The same argument may not hold for developments on the coast though, where there can often be a considerable risk of disturbing (and activating) potential acid sulfate soils.

Acid sulfate soils generally do not present a management problem as long as they remain undisturbed and waterlogged. They become problematic when susceptible wetlands are disturbed and especially when they are drained, for example in attempts to re-instate more natural wetting and drying regimes, or when drainage ditches cause the watertable to drop rapidly and surface soils to dry out and oxidise (Department of Sustainability and Environment 2009). The laying of pipes, for example for the supply of gas or water in urban areas, across susceptible land can activate potential acid sulfate soils. In most cases reverting to the earlier hydrological regime is not sufficient to cure the problem, as large volumes of acid may have accumulated in the sediments and there may have been irreversible changes to the soil structure due to drying, acidification and oxidation.

2. Altering a wetland's water regime can induce an intrusion of saline water, either from nearby saline waterbodies (mostly, but not always, in coastal situations: Raulings *et al*. 2010) or from shallow saline groundwater (Bailey *et al*. 2006; Boon *et al*. 2009). The hydrostatic head maintained by a permanently inundated wetland limits the ability of saline groundwater to penetrate the wetland bed. Should such wetlands be drained, however, the hydrostatic head is lost and saline groundwater can flow into them, upwards from underneath the wetland or sideways from around it. Even if the wetland bed were above the watertable or if soil porosity were such that it did not permit mass flow of saline groundwater, the watertable may be so close to the surface (i.e. <~3 m) that capillary action could draw salts to the soil surface (Eamus *et al*. 2006). Salts then accumulate on the surface and in the soil profile, and the wetland becomes progressively more saline over time.

3. Significant ecological risks may be associated with implementing a drying phase in wetlands that have been chronically inundated and in which a particular and valued biota has established itself. This process is likely to be most relevant with long-established wetlands in inland Australia that have had their water regimes modified as a consequence of irrigation practices rather than in newly created urban wetlands, but the principle is worth stating, just in case the situation does arise. For example, it is possible that over time permanently inundated wetlands have evolved ecological communities that are now of high ecological value. Even though the re-instatement of a more natural wetting and drying regime may seem theoretically desirable, in such cases any hydrological change from existing conditions may have undesirable ecological consequences. Boulton & Brock (1999, p. 150) noted that 'Drying of a permanent wetland usually extinguishes most of the aquatic biota and recovery is much slower than in nearby naturally temporary wetlands'. For example, long-established populations of native fish and amphibians could be compromised by the re-introduction of a drying phase in wetlands that have long been filled with water.

Conclusions

There is no element of wetland ecology and management that is not affected by hydrological factors. At the most fundamental level, even the system we decide to use to classify wetlands (e.g. 'permanent' versus 'temporary'; 'fresh' versus 'saline') has a hydrological component. The ecological processes that take place in a given wetland (e.g. primary and secondary production, competition, herbivory and predation, food-web interactions, nutrient cycling and energy flow) form and control its ecological structure (i.e. the presence of particular species of plant and animal), and all the important ecological processes are controlled to some extent by hydrology. It is now abundantly clear that wetlands cannot be managed effectively unless we first understand their ecological structure and function and the degradative forces that we subject them to; it is hubris to assume otherwise (e.g. see Turner 2009). Acknowledging and working with a site's hydrology is, therefore, crucial to better managing natural and constructed wetlands in both urban and non-urban settings.

References

Bailey, P. C. E., Boon, P. I., Blinn, D., and Williams, W. D. (2006). Salinity as an ecological perturbation to rivers, streams and wetlands of arid and semi-arid zones. In 'Ecology of desert rivers'. (Ed R. Kingsford.) pp. 280–314. (Cambridge University Press: Cambridge.)

Balla, S. A., and Davis, J. A. (1993). 'Wetlands of the Swan Coastal Plain. Volume 5. Managing Perth's wetlands to conserve the aquatic fauna.' (Water Authority of Western Australia: Perth.)

Barkley-Jack, J. (2009). 'Hawkesbury settlement revealed: a new look at Australia's third mainland settlement 1793–1802.' (Rosenberg: Dural.)

Batzer, D. P., and Sharitz, R. R. (2006). Ecology of freshwater and estuarine wetlands: an introduction. In 'Ecology of freshwater and estuarine wetlands'. (Eds D. P. Batzer and R. R. Sharitz.) pp. 1–6. (University of California Press: Berkeley.)

Bedford, B. L. (1996). The need to define hydrological equivalence at the landscape scale for freshwater wetland mitigation. *Ecological Applications* **6**, 57–68.

Boon, P. I. (2006). Biogeochemistry, ecology and management of hydrologically dynamic wetlands. In 'Ecology of freshwater and estuarine wetlands'. (Eds D.P. Batzer and R.R. Sharitz.) pp. 115–176. (University of California Press: Berkeley.)

Boon, P. I. (2011). The rehabilitation of coastal wetlands: why small-scale variations in topography are critical to success. *Australasian Plant Conservation* **20**, 9–10.

Boon, P. I., and Bailey, P. C. (1998). Implications of nutrient enrichment for management of primary productivity in wetlands. In 'Wetlands in a dry land: understanding for management'. (Ed W. D. Williams.). (Environment Australia: Canberra.)

Boon, P. I., Holmes, A., Holmes, S., Mozina, M., and Wills, T. (2009). Which wetlands should be flooded and which ones dried? A decision-making tool for wetlands of the Murray-Darling Basin. Refereed paper 25: Presentation 223. *OzWater 09*. (Melbourne, 16-19 March, 2009.)

Boon P. I., Mitchell, A., and Lee. K. (1997). Effects of wetting and drying on methane emissions from ephemeral floodplain wetlands in south-eastern Australia. *Hydrobiologia* **357**, 73–87.

Bornette, G., and Puijalon, S. (2011). Responses of aquatic plants to abiotic factors: a review. *Aquatic Sciences* **73**, 1–14.

Bougon, N., Auterives, C., Aquilina, L., Marmonier, P., Derrider, J., *et al*. (2011). Nitrate and sulphate dynamics in peat subjected to different hydrological conditions: batch experiments and field comparison. *Journal of Hydrology* **411**, 12–24.

Boulton, A. J., and Brock, M. A. (1999). 'Australian freshwater ecology.' (Gleneagles Press: Adelaide.)

Bowd, D. G. (1994). 'Macquarie country: a history of the Hawkesbury.' (Library of Australian History: Sydney.)

Briggs, S. V. (1988). Guidelines for management of inland wetlands in southern New South Wales. *Wetlands (Australia)* **8**, 1–2.

Brinson, M. M. (1999). 'A hydrogeomorphic classification for wetlands.' US Army Corps of Engineers. Wetlands Research Program, Technical Report WRP–DE–4.

Brock, M. A., and Casanova, M. T. (2000). 'Are there plants in your wetland? Revegetating wetlands.' (Land and Water Resources R&D Corporation: Canberra.)

Brooks, R. P., Brinson, M. M., Havens, K. J., Hershner, C. S., Rheinhardt, R. D., *et al*. (2011). Proposed hydrogeomorphic classification for wetlands of the mid-Atlantic region, USA. *Wetlands* **31**, 207–219.

Bureau of Meteorology (2012). 'Drought statement archive: short-term deficiencies in east but long-term rainfall deficits remain.' Available at http://www.bom.gov.au/climate/drought/archive/20100408.shtml [Accessed 12 June 2013].

Burford, M. A., Cook, A. J., Fellows, C. S., Balcombe, S. R., and Bunn, S. E. (2008). Sources of carbon fuelling production in an arid floodplain river. *Marine and Freshwater Research* **59**, 24–234.

Burns, A., and Ryder, D. S. (2001). Responses of bacterial extracellular enzymes to inundation of floodplain sediments. *Freshwater Biology* **46**, 1299–1307.

Chambers, P. A., and Kalff, J. (1985). Depth distribution and biomass of submersed aquatic macrophyte communities in relation to secchi depth. *Canadian Journal of Fisheries and Aquatic Sciences* **45**, 1010–1017.

Colmer, T. D., and Voesnek, L. A. C. J. (2009). Flooding tolerance: suites of plant traits in variable environments. *Functional Plant Ecology* **36**, 665–681.

Cooke, G. D., Welch, E. B., Peterson, S. A., and Nichols, S. A. (2005). 'Restoration and management of lakes and reservoirs.' 3rd edn. (Taylor & Francis: Boca Raton.)

Corrick, A. H., and Norman, F. I. (1980). Wetlands and waterbirds of the Snowy River and Gippsland Lakes catchment. *Proceedings of the Royal Society of Victoria* **91**, 1–15.

Cowardin, L. M., Carter, .V., Golet, F., and LaRoe, E. T. (1979). 'Classification of wetlands and deepwater habitats of the United States.' Fish and Wildlife Service, Report OBS-79/31.

Cowardin, L. M., and Golet, F. C. (1995). US Fish and Wildlife Service 1979 wetland classification: a review. *Vegetatio* **118**, 139–152.

Cowling, S. J. (1977). Classification of the wetland habitats of waterbirds. *Australian Marine Science Association* **58**, 15–16.

Cronk, J. K., and Fennessy, M. S. (2001). 'Wetland plants. Biology and ecology.' (Lewis Publishers: Boca Raton.)

Deil, U. (2005). A review of habitats, plant traits and vegetation of ephemeral wetlands – a global perspective. *Phytocoenologia* **35**, 533–705.

Department of Conservation, Forests and Lands (1988). 'Wetlands conservation program for Victoria.' (State Government of Victoria: Melbourne.)

Department of Natural Resources and Water (2007). 'Queensland urban drainage manual.' Volume 1. 2nd edition. (Department of Natural Resources and Water: Brisbane.)

Department of Sustainability and Environment (2005). 'Index of Wetland Condition. Conceptual framework and selection of measures.' (State Government of Victoria: Melbourne.)

Department of Sustainability and Environment (2009). 'Victorian coastal acid sulfate soils strategy.' (Department of Sustainability and Environment: East Melbourne.)

Department of Primary Industries (2011). 'Coastal acid sulfate soils.' Available at: http://www.dpi.vic.gov.au/dpi/vro/vrosite.nsf/pages/soil_acid_sulfate_soils.

Dolan, T. J., Hermann, A. J., Bayley, S. E., and Zoltek, J. Jr. (1984). Evapotranspiration of a Florida, U.S.A., freshwater wetland. *Journal of Hydrology* **74**, 355–371.

Eamus, D., Hatton, T., Cook, P., and Colvin, C. (2006). 'Ecohydrology: vegetation function, water and resource management.' (CSIRO Publishing: Collingwood.)

Ford, P. W., Boon, P. I., and Lee, K. (2002). Methane and oxygen dynamics in a shallow floodplain lake: the significance of periodic stratification. *Hydrobiologia* **485**, 97–110.

Froend, R. H., Farrell, R .C. C., Wilkins, C. F., Wilson, C. C., and McComb, A. J. (1993). 'Wetlands of the Swan Coastal Plain.' Volume 4. (Water Authority of Western Australia: Perth.)

Gergis, J., Garden, D., and Fenby, C. (2009). A climate reconstruction of Sydney Cove, New South Wales, using weather journal and documentary data, 1788–1791. *Australian Meteorological and Oceanographic Journal* **58**, 83–98.

Gergis, J., Garden, D., and Fenby, C. (2010). The influence of climate on the first European settlement of Australia: a comparison of weather journals, documentary data and palaeoclimate records, 1788–1793. *Environmental History* **15**, 485-507.

Gippel, C. (1996). Water management. In 'Manual of wetlands management.' Wetlands Conservation Report Series, Number 4. pp. 101–137. (Department of Conservation and Natural Resources: East Melbourne.)

Gold Coast City Council (2006). 'MUSIC modelling guidelines.' (Gold Coast City Council: Gold Coast.)

Gosselink, J. G., and Turner, R. E. (1978). The role of hydrology in freshwater wetland ecosystems. In 'Freshwater wetlands. Ecological processes and management potential.' (Eds R. E. Good, D.F. Whigham and R.L. Simpson.). pp. 63–78. (Academic Press: New York.)

Håkanson, L. (2004). Internal loading: a new solution to an old problem in aquatic sciences. *Lakes & Reservoirs: Research and Management* **9**, 3–23.

Harms, T. K, and Grimm, N. B. (2012). Responses of trace gases to hydrologic pulses in desert floodplains. *Journal of Geophysical Research – Geosciences* **117**, G01035.

Howard-Williams, C. (1985). Cycling and retention of nitrogen and phosphorus in wetlands: a theoretical and applied perspective. *Freshwater Biology* **15**, 391–431.

Idso, S. B. (1981). Relative rates of evaporative water losses from open and vegetation covered water bodies. *Water Resources Bulletin* **17**, 46–48.

Jackson, C. R. (2006). Wetland hydrology. In 'Ecology of freshwater and estuarine wetlands'. (Eds D.P. Batzer and R.R. Sharitz.). pp. 43–81. (University of California Press: Berkeley.)

Keddy, P. A. (2010). 'Wetland ecology. Principles and conservation.' 2nd edn. (Cambridge University Press: Cambridge.)

Koehn, J., Brumley, A., and Gehrke, P. (2000). 'Managing the impacts of carp.' (Bureau of Rural Sciences: Canberra.)

Lamontagne, S., Hicks, W. S., Fitzpatrick, R. W., and Rogers, S. (2006). Sulfidic materials in dryland river sediments. *Marine and Freshwater Research* **57**, 775–788.

Larkin, D., Vivian-Smith, G., and Zedler, J. B. (2006). Topographic heterogeneity theory and ecological restoration. In 'Foundations of restoration ecology'. (Eds D.A. Falk, M.A. Palmer, and J.B. Zedler.). pp. 142–164. (Island Press: Washington.)

Leeper, G. W. (1960). Climates. In 'The Australian environment.' 3 edn. pp. 19–28. (CSIRO and Melbourne University Press: Melbourne.)

Lewis, W. M. (1983). A revised classification of lakes based on mixing. *Canadian Journal of Fisheries and Aquatic Sciences* **40**, 1779–1787.

McComb, A. J., and Qiu, S. (1998). The effects of drying and reflooding on nutrient release from wetland sediments. In 'Wetlands in a dry land: understanding for management'. (Ed W.D. Williams.). pp. 147–159. (Environment Australia: Canberra.)

McMahon, T. A., Finlayson, B. L., Haines, A. T., and Srikanthan, R. (1992). 'Global runoff – continental comparisons of annual flows and peak discharges.' (Catena: Cremingen-Destedt.)

Mensforth, L. J., and Walker, G. R. (1996). Root dynamics of Melaleuca halmaturorum in response to fluctuating saline groundwater. *Plant and Soil* **184**, 75–84.

Mitsch, W. J., and Gosselink, J. G. (1993). 'Wetlands.' 2nd edn. (Van Nostrand: New York.)

Mitsch, W. J., Nahik, A., Wolski, P., Bernal, B., Zhang, L., *et al*. (2010). Tropical wetlands: seasonal hydrological pulsing, carbon sequestration, and methane emissions. *Wetlands Ecology and Management* **18**, 573–586.

Moore, P. D. (1990). Soils and ecology: temperate wetlands. In 'Wetlands: a threatened landscape.' (Ed M. Williams.). pp. 95–114. (Basil Blackwell: Oxford.)

Morris, K. L., Bailey, P. C., Boon, P. I., and Hughes, L. (2003a). Stable states in the aquatic vegetation of shallow lakes. II. A novel mechanism for the catastrophic loss of macrophytes in shallow lakes. *Marine and Freshwater Research* **54**, 185–200.

Morris, K. L., Boon, P. I., Bailey, P. C., and Hughes, L. (2003b). Stable states in the aquatic vegetation of shallow lakes. I. Effects of nutrient enrichment and plant harvesting on primary producers. *Marine and Freshwater Research* **54**, 201–215.

Morris, K. L., Harrison, K. A., Bailey, P. C., and Boon, P. I. (2004). Domain shifts in the aquatic vegetation of shallow urban lakes: the relative roles of light limitation and anoxia in the catastrophic loss of the submerged angiosperm *Vallisneria americana*. *Marine and Freshwater Research* **55**, 749–758.

Morris, K. L., Bailey, P. C., Boon, P. I., and Hughes, L. (2006). Effects of plant harvesting and nutrient enrichment on phytoplankton community structure in a shallow urban lake. *Hydrobiologia* **571**, 77–91.

Moss, B. (1990). Engineering and biological approaches to the restoration from eutrophication of shallow lakes in which aquatic plant communities are important components. *Hydrobiologia* **200/201**, 367–377.

Moss, B. (1998). 'Shallow lakes – biomanipulation and eutrophication.' SCOPE Newsletter 29.

Moss, B. (2003). Cinderella goes to the ball. A story of shallow lakes. *LakeLine Spring* **2003**, 11–16.

Navanteri, M., and Kambouris, P. (2008). 'Interim water management guidelines for key wetlands of the Gippsland Lakes Ramsar site.' (Parks Victoria: Bairnsdale.)

Newsome, A., Nicholls, N., and Hobbs, T. (1996). The uncertain climate and its biotic consequences. In 'Exploring central Australia: society, the environment and the 1894 Horn Expedition'. (Eds S.R. Morton and DJ Mulvaney.). pp. 287–304. (Surrey Beatty and Sons: Chippendale.)

Nichols, M. (2001). 'Disastrous decade: flood and fire in Windsor 1864–1874.' (Deerubin Press: Berowra Heights.)

Nixdorf, B., and Deneke, R. (1997). Why 'very shallow' lakes are more successful opposing reduced nutrient loads. *Hydrobiologia* **342/343**, 269–284.

Norman, F. I., and Corrick, A. H. (1988). Wetlands in Victoria: a brief review. In 'The conservation of Australian wetlands'. (Eds A.J. McComb and P.S. Lake.). pp 17–34. (Surrey Beatty & Sons: Sydney).

Orme, A. R. (1990). Wetland morphology, hydrodynamics and sedimentation. In 'Wetlands: a threatened landscape'. (Ed M. Williams.). pp. 42–94. (Basil Blackwell: Oxford.)

Paijmans, K., Galloway, R. W., Faith, D. P., Fleming, P. M., Haantjens, H. A., *et al*. (1985). 'Aspects of Australian wetlands.' CSIRO Australia Division of Water and Land Resources Technical Paper 44, 1–71.

Peach, M., and Zedler, J. B. (2006). How tussocks structure sedge meadow vegetation. *Wetlands* **26**, 322–335.

Phillips, G. L. (2005). Eutrophication of shallow temperate lakes. In 'The lakes handbook. Volume 2. Lake restoration and rehabilitation'. (Eds P.E. O'Sullivan and C.S. Reynolds.). pp. 261–278. (Blackwell: Oxford.)

Pigram, J. J. (2006). 'Australia's water resources: from use to management.' (CSIRO Publishing: Collingwood.)

Pressey, R. L., and Adam, P. (1995). A review of wetland inventory and classification in Australia. *Vegetatio* **118**, 81–101.

Raisin, G. W., Bartley, J., and Croome, R. (1999). Groundwater influence on the water balance and nutrient budget of a small natural wetland in northeastern Victoria, Australia. *Ecological Engineering* **12**, 133–147.

Ramsar Convention Secretariat (2006). 'The Ramsar Convention manual: a guide to the Convention on Wetlands (Ramsar, Iran, 1971).' 4th edn. (Ramsar Convention Secretariat: Gland.)

Raulings, E., Morris, K., Roache, M., and Boon, P.I. (2010). The importance of water regimes operating at small spatial scales for the diversity and structure of wetland vegetation. *Freshwater Biology* **55**, 701–715.

Raulings, E., Morris, K., Roache, M., and Boon, P. I. (2011). Is hydrological manipulation an effective management tool for rehabilitating chronically flooded, brackish-water wetlands? *Freshwater Biology* **56**, 2347–2369.

Roberts, J., and Marston, F. (2011). 'Water regime for wetland and floodplain plants. A source book for the Murray-Darling Basin.' (National Water Commission: Canberra.)

Robinson, R. R., Boon, P. I., and Bailey, P. C. (2006). Germination characteristics of Melaleuca ericifolia Sm. (Swamp Paperbark) and their implications for the rehabilitation of coastal wetlands. *Marine and Freshwater Research* **57**, 1–9.

Rogers, K. (2011). Vegetation. In 'Floodplain wetland biota in the Murray-Darling Basin'. (Eds K. Rogers and T.J. Ralph.). pp 17–82. (CSIRO Publishng: Collingwood.)

Schaffer, P. W., Kentula, M. E., and Gwin, S. E. (1999). Characterization of wetland hydrology using hydrogeomorphic classification. *Wetlands* **19**, 490–504.

Scheffer, M. (1998). 'Ecology of shallow lakes.' (Chapman and Hall: London.)

Smith, D. I. (1998). 'Water in Australia; resources and management.' (Oxford University Press: Oxford.)

Smith, R. G., and Brock, M. A. (2007). The ups and downs of life on the edge: the influence of water level fluctuations on biomass allocation in two contrasting wetland plants. *Plant Ecology* **188**, 103–116.

Stribling, J. M., Cornwell, J. C., and Glahn, O. A. (2007). Microtopography in tidal marshes: ecosystem engineering by vegetation? *Estuaries and Coasts* **30**, 1007–1015.

Turner, R. E. (2009). Doubt and the values of an ignorance-based world view for restoration: coastal Louisiana wetlands. *Estuaries and Coasts* **32**, 1054–1068.

United States Department of Agriculture (2008). 'Hydrogeomorphic wetland classification scheme: an overview and modification to better meet the needs of the Natural Resources Conservation Service.' Natural Resources Conservation Service, Technical Note 190–8–76.

van der Valk, A. G. (2012). 'The biology of freshwater wetlands.' 2nd edn. (Oxford University Press: Oxford.)

Vymazal, J. (1995). 'Algae and element cycling in wetlands.' (Lewis Publishers: Boca Raton.)

Warfe, D. M., Pettit, N. E., Davies, P. M., Pusey, B. J., Hamilton, S. K., *et al*. (2011). The 'wet-dry' in the wet-dry topics drives river ecosystem structure and processes in northern Australia. *Freshwater Biology* **56**, 2169–2195.

Webb, J. A., Wallis, E. M., and Stewardson, M. J. (2012). A systematic review of published evidence linking wetland plants to water regime components. *Aquatic Botany* **103**, 1–14.

Wetzel, P. R., Sklar, F. H., Coronado, C. A., Troxler, T. G., Krupa, S. L., *et al*. (2011). Biogeochemical processes on tree islands in the Greater Everglades: initiating a new paradigm. *Critical Reviews in Environmental Science and Technology* **41**, 670–701.

Wetzel, R. G. (1990). Edgardo Baldi memorial lecture: Land-water interfaces: metabolic and limnological regulators. *Verhandlungen Internationale Vereinigung Limnologie* **24**, 6–24.

Wheeler, B. D. (1999). Water and plants in freshwater wetlands. In ‘Eco-hydrology. Plants and water in terrestrial and aquatic environments’. (Eds A.J. Baird and R.L. Wilby.). pp. 127–180. (Routledge: London.)

Williams, W. D. (1988). Limnological imbalances: an antipodean viewpoint. *Freshwater Biology* **20**, 407–420.

Wolf, K. L., Ahn, C., and Noe, G. B. (2011). Mictotopography enhances nitrogen cycling and removal in created mitigation wetlands. *Ecological Engineering* **37**, 1398–1406.