Wetland vegetation of inland Australia

For *Australian vegetation*, ed. D.A. Keith, 3rd edn.

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This is the final author-version (not type-set or copy-edited) of the following book chapter:

Catford, J.A., Roberts, J., Capon, S.J., Froend, R.H., *Windecker, S.M. & Douglas, M.M. (2017) Wetland vegetation of inland Australia. *Australian Vegetation* (ed. D.A. Keith). Cambridge University Press.p. 490-515. DOi: 10.1017/9781316339701

Abstract

Wetlands fed by rainwater, surface flows and groundwater occur throughout Australia, even in arid areas. This chapter focuses on temporary wetlands and permanently wet systems that are dominated by non-woody macrophytes, and uses nine case studies that span Australia to illustrate their biogeography, dynamics and key threats. The type and distribution of wetland vegetation reflects hydrology, climate and geomorphology, from the annually flooded wetlands of northern Australia, to saline lakes of arid and semi-arid Australia, to groundwater-dependent systems of the southwest, to bogs and fens of the Alps and Tasmania. Wetland plants have developed a range of adaptations and life histories to tolerate the dynamic water regimes characteristic of Australian wetlands, and can be grouped into seven categories that reflect these adaptations. Waterbirds and water can connect spatially isolated systems, and seedbanks that last for decades allow species to disperse through time. Water regime is a strong driver of species composition and abundance, thus hydrological modification through water extraction, flow regulation or reductions in rainfall is a significant threat to wetland flora, and arguably the principal threat for Australian wetland vegetation. The displacement of native macrophytes by exotic and terrestrial species is both a symptom and cause of ecological change, with exotic plants often being better adapted to modified flooding and fire regimes, livestock grazing and eutrophication than natives. Introduced livestock and feral fauna eat, trample and uproot native plants, and degrade their habitat. These types of threats are expected to intensify, increasing the challenge for wetland management and policy.

Introduction

Wetlands are among the most variable and productive of Earth's ecosystems. They are highly valuable to humans and of crucial importance for ecosystem health. Being at the interface of land and water, wetlands are home and host to numerous taxa, from phytoplankton to macroinvertebrates, to birds and mammals. Wetlands occur throughout Australia, varying from permanently wet to almost permanently dry. Fed by rainwater, rivers, surface flows and groundwater, the boundaries of many wetlands shift seasonally, or in response to longer climatic cycles, as inundation varies. Waterbirds and water can connect seemingly isolated systems, allowing plants and their propagules to disperse widely. The diversity of these systems is mirrored by the diversity of the flora that inhabits them.

This chapter explores the types of wetlands that occur in Australia, using case studies to illustrate their biogeography and dynamics, and the key drivers that shape their vegetation. Over half of Australia's wetlands have been destroyed since European settlement (Bennett 1997); we outline the main drivers of this loss, as well as some management and policy initiatives designed to arrest and ameliorate it. We provide an overview of the major factors and processes affecting wetland vegetation, and give examples of wetland plant adaptations.

We focus on temporary and permanent wetlands that are dominated by non-woody macrophytes (or hydrophytes, i.e. plants adapted to living in water, saturated soil or very moist soil) and can be characterised by water regimes involving periodic or permanent inundation or waterlogging. The high variability of rainfall and runoff in Australia means that some Australian wetlands may only flood once every several years, and possibly less often in ephemeral arid zone river systems (Capon 2003). Wetlands fed by groundwater are buffered against this variability but being usually quite small and relatively few account for only a small percentage of Australia's wetland. Although much of the material discussed in this chapter can apply to vegetation in flowing waters, we focus on vegetation of lentic (standing or slow flowing water) systems.

Drivers of wetland vegetation

Numerous abiotic, biotic and human factors affect the environmental conditions, dispersal opportunities and biotic interactions that shape wetland vegetation (Boulton *et al.* 2014). The influence of these interacting factors determines the biogeography and ecology of wetland vegetation (Fig. 1).

Hydrology is the key force structuring wetland and riparian vegetation. Hydrology refers to the distribution and quantity of water in the landscape. Hydrology at the wetland level is characterised by the water regime, which encapsulates the duration, frequency, magnitude, timing (seasonality), rate of change, and variability of wetting and drying. Wetlands may experience extremes in physico-chemical conditions as a result of changing flooding conditions. Because of different hydrological conditions associated with wetland bathymetry (underwater topography), wetland vegetation typically forms zones (or mosaics) demarcated by subtle changes in water depth or inundation duration (Fig. 2). The species that occur in each zone usually have physiological, morphological or life history adaptations for those particular hydrological and environmental conditions (Brock and Casanova 1997).

Wetland hydrology is integrally linked with geomorphology (the physical form of the catchment, wetland and associated waterways) and hydraulics (flow dynamics and velocity), and these three factors interact closely to affect other key drivers. Together with water quality (including nutrient fluxes, sediment loads and ionic composition) and soil and sediment characteristics (lithology), these factors determine the key environmental conditions affecting wetland vegetation (Fig. 1). Plants and animals may affect the abiotic characteristics of a site by altering its physical or chemical characteristics. For example, fallen trees affect hydraulics and channel form, and common carp disturb the sediment, uprooting plants and increasing turbidity.

Wetland biota may tolerate changing hydrological conditions, but many complete their life cycles during periods in which environmental conditions are suitable. The ability to colonise, grow and reproduce quickly during such times, and persist in a dormant state when conditions are unfavourable, is particularly important under boom-and-bust conditions (Bunn *et al.* 2006). Dispersal in time and space is therefore crucial for species persistence. Plants can effectively disperse through time by storing propagules (or diaspores: seeds, spores and vegetative fragments that can develop into an individual plant) in seed banks in the soil, litter and held in plant canopies. In desert wetlands that may only be inundated a few times a century, seeds must be able to survive intervening dry periods, and germinate when appropriate flooding conditions arise (Brock *et al.* 2003). Like terrestrial plants, wetland plants disperse by a range of mechanism, including various means, including via wind, animal and self-propulsion, and many also spread clonally through aboveground stolons and belowground rhizomes. However, a large fraction of wetland plants also use water as a means of dispersal (hydrochory), and some may be dispersed by fruit-eating fish (ichtyochory; though there appears to be no evidence of ichtyochory in Australia: Jane Roberts, pers. comm. 2015) (Catford and Jansson 2014). Water-borne dispersal enables propagules to reach sites that are hydrologically connected and likely to have suitable environmental conditions. Hydrological connectivity refers to permanent and temporary connections among wetlands and other (mostly aquatic) systems; it includes lateral, longitudinal (in the cases of riverine wetlands) and vertical (surface to groundwater) connections (Fig. 1).

Waterbirds are effective in dispersing propagules over long distances to hydrologically disconnected systems. Such long distance dispersal facilitates species- and population-mixing among wetlands, and contributes to the wide geographic distributions, spanning continents and beyond, of many wetland plant species. Wetlands that are isolated, like small discharge springs fed by groundwater, offer fewer opportunities for long distance dispersal and levels of endemism are subsequently higher than in more connected systems, like floodplain wetlands.

Like any vegetation, wetland vegetation is affected by interactions with other trophic levels and with other types of vegetation. Terrestrial vegetation may compete with (or facilitate) wetland vegetation under drier conditions, and herbivores may modify the abundance of certain species. As discussed below, humans affect wetland vegetation directly and indirectly by altering environmental conditions, dispersal opportunities and biotic interactions. Owing to the interplay of so many factors, the dynamics of wetlands can be quite diverse, yet there are commonalities as the case studies below illustrate.

Ecology and floristics of wetland plants

Adaptations of wetland plants

Wetland plants have adapted to specific flooding and drying cycles, and this is particularly evident in regeneration and dispersal.

Germination and recruitment directly affects plant community composition. Water regimes can influence plant phenology (the timing of periodic events, such as flowering and seed set) by providing cues for germination and dispersal, as well as suitable conditions for seedling growth (Pettit and Froend 2001). Flooding can break dormancy by causing a sudden increase in salinity after drought in arid or semi-arid systems (Nielsen *et al.* 2007).

Although not all wetland plants disperse by water and hydrochorous species rarely disperse exclusively by water, water is important for dispersal in two major ways – as a vector and as a trigger for dispersal. Attributes of propagules that facilitate hydrochory include: buoyancy; corky or spongy tissue of low density; waxy, cuticularized epidermis; and surfaces with furrows, pits or hairs that can enhance floating ability (Catford and Jansson 2014). In climates with predictable seasonal patterns, some riparian plants time the release of propagules to coincide with particular phase of the river hydrograph. This adaptation increases the likelihood that propagules will be transported and deposited on recently flooded soils, providing conditions favourable for germination and recruitment. If the seasonal flow patterns are changed, for example by river regulation, native plants may no longer release propagules at optimal times. The summer-autumn timing of peak flows in regulated rivers in Victoria coincides with the seed release of exotic species, whereas native riparian species release seeds in winter-spring when flows were historically at their highest (Greet *et al.* 2012).

Functional types of wetland plants

Macrophytes can be classified into groups based on their adaptations to, and tolerance of, flooding. The most commonly used wetland plant classification scheme in Australia is that developed by Brock and Casanova (1997). This hierarchical scheme classifies wetland plant species into three main groups, Submerged, Amphibious and Terrestrial (with seven subgroups) based on their life history and response to flooding (Table 1). Casanova (2011) updated the scheme, but we prefer the relative simplicity of the original one.

The Terrestrial group comprises two sub-groups, Terrestrial-dry and Terrestrial-damp. Terrestrial-dry are unable to tolerate flooding, so occur above the high water mark (Terrestrial dry species, Table 1, Fig. 2b). Terrestrial-damp species are found around wetlands, reflecting their need for damp soil and their tolerance of infrequent inundation (see also species described in Chapter XX on Floodplain forests and woodlands).

Plants in the Amphibious group are either Tolerators, plants that tolerate fluctuations in water level, or Responders, plants that respond to them. Amphibious Tolerator-emergent are often the dominant and most conspicuous type of wetland vegetation (Fig. 2b, c). These species are able to tolerate fluctuations in water levels by ensuring that a sufficient portion (roughly two thirds of their aboveground biomass) is above the water, allowing for respiration. Amphibious Tolerator low growing species generally contribute a relatively small fraction to the plant biomass in a wetland.

There are two types of Amphibious Responders: plastic and floating (Fig. 2a, d, e, f). Plastic species change their morphology and form, and may have heterophyllous leaves, which differ whether submerged or emergent (e.g. *Sagittaria platyphylla*, Fig. 2f). Waterlilies are a classic example of floating responders, as their long internodes or petioles allow leaves to float to the surface of the water even if its depth fluctuates (Brock 2003). Floating responders includes some of the smallest wetland plants (*Spirodela, Lemna* and *Wolffia* species; see Fig. 2e) and invasive exotic plants (Fig. 2d), including *Eichhornia crassipes* (water hyacinth) and *Salvinia molesta* (salvinia, see Chapter XX on Invasive plants and plant pathogens).

Submerged plants grow below the surface of the water and have adaptations that enable them to germinate, grow and reproduce entirely underwater. Submerged plants cannot tolerate exposure to air. As well as angiosperms like *Vallisneria* species, many of the submerged taxa are charophytes (freshwater green algae), e.g. *Chara* and *Nitella* species.

Trends in Australian wetland vegetation science

Wetland plant ecology and wetland vegetation science in Australia were in their infancy prior to the 1970s, mirroring global trends (Fig. 3). Early study of Australian wetland plants was principally concerned with species and habitat descriptions (including taxonomy), and aquatic weed management in irrigation channels.

The main emphasis on emergent macrophytes in the 1970s and 1980s reflected scientific and management priorities at that time. Emergent macrophytes were recognised as drivers of lake ecology and appealed to management interests because of their use as a "green" alternative to treating wastewater in constructed wetlands (Brix 1987). Australian research, on *Eleocharis sphacelata* and *Typha domingensis* for example, also contributed to international interest in macrophyte adaptations to anoxic and hypoxic substrates (e.g. Sorrell *et al.* 1997).

Research on salinity and water regime dominated wetland vegetation research from the 1990s onwards. Salinity experiments in the 1990s built on research from the 1970s and 1980s by asking more complex questions, such as identifying tolerances of macrophytes using concentration gradients in growth trials. Experiments in the 2000s considered interactions between salinity and drying, explored geographic variability and interpreted findings in terms of biodiversity (Goodman *et al.* 2011).

Investigations into water regimes have progressed from assessing the response of few species to a single-hydrologic variable (e.g. emergent macrophytes along a depth gradient, Rea and Ganf 1994), to comparing multiple species, to the interaction of the whole water regime with salinity, sedimentation or plant community composition (e.g. Catford *et al.* 2011). In parallel, perspective has shifted from short-term questions (such as within-year patterns of growth, resource allocation and reproduction) to establishing the role of water regime throughout the plant life cycle and over large spatial scales. Years of research have revealed that wetland plants employ many strategies to cope with variable hydrology. More recently, scientists have built on the body of knowledge of altered water regimes to test theoretical questions, for example of invasion biology (Catford *et al.* 2011).

Australia has made a significant contribution to wetland ecology through seed bank studies. Initially seedbank studies were descriptive, and intent on characterising seed density and species richness, understanding spatial and temporal variability, and establishing the role of hydrology as a driver of this variability. They have since become a powerful tool allowing comparison among wetlands, evaluation of antecedent conditions, scenario testing, impact assessment, and a means of establishing recovery potential and resilience. One outcome of seedbank research in Australia led to the development of a macrophyte classification system (Brock and Casanova 1997).

Consistent with global trends and terrestrial plant ecology, wetland science has seen growing emphasis on characterising species based on their morphology, responses to environmental stimuli (e.g. flooding) and effects on ecosystem function and services (Catford and Jansson 2014). This shift is motivated by the greater generalisation that trait-based classifications can offer to ecology and management: understanding plant responses and effects based on species characteristics rather than species identity allows findings gathered in one study and one region to be applied elsewhere.

The focus on salinity and water regime has necessarily resulted in fewer studies dedicated to other environmental factors: for example, effects of fire and temperature on wetland vegetation are poorly understood outside of alpine and sub-alpine areas. The short-term effects of native herbivores (mainly waterbirds such as black swans and magpie geese), exotic livestock (cattle, horses, pigs) and exotic benthivorous fish (common carp) on wetland vegetation and seed banks may be locally known, but the cumulative and differential long-term effects of biota on floristic composition and structure need further study. Plant-plant interactions, factors maintaining dominance or facilitating invasion, and the nursery effects of structurally dominant species (James *et al.* 2015), have been investigated only rarely, and consequently their importance in structuring wetland plant communities is likely underappreciated.

Case studies of wetland types

Wetlands are found throughout Australia, with their type and distribution reflecting their hydrology, climate (particularly rainfall and evaporation) and topographic and morphological features (Fig. 1, Brock 2003). Northern Australia has wet summers, southern coastal parts of Australia that are temperate have wet winters, and the arid and semi-arid interior, which makes up two thirds of the continent, has highly unpredictable rainfall. If fed by rainfall, surface runoff or river flows, wetland hydrology reflects these climatic patterns. Flooding is highly predictable and seasonal in the tropics, whereas the saline and episodic wetlands that dominate arid and semi-arid regions can be characterised by their (wet) boom and (dry) bust cycles. Seasonal or intermittent wetlands are more common near the coast, and wetlands that are more persistent often occur where rainfall exceeds 1000 mm annually (Boulton *et al.* 2014; Brock 2003). In contrast to wetlands fed by surface water, many groundwater-fed systems naturally have a permanent supply of water. Springs fed by groundwater can seemingly occur in unlikely places, providing rare oases in the middle of the desert, but their distribution reflects the location of cracks and fissures in the aquitards that confine groundwater or where sediments that form the aquifer project above the ground (Fensham and Fairfax 2003). Wetlands flooded by rivers are necessarily found along river channels. The distribution of wetlands is changing as surface and groundwater is used and regulated by humans, as climatic patterns change, and as wetlands are infilled and their catchments modified. The nine case studies presented below illustrate some of this diversity.

Floodplain wetlands of the wet-dry tropics

One of the most striking features of northern Australia is the extensive areas of floodplain associated with many of its large river systems (Fig. 4a). When in flood, the whole expanse of a river floodplain can be inundated; wetlands are not just restricted to lower elevation areas of the floodplain. In the lower rainfall regions, such as the Gulf of Carpentaria and the Kimberley, floodplains may only be inundated for a few weeks a year and contain very few vascular plants. In contrast, wetlands in the higher rainfall regions may be inundated for up to half of the year and support expansive areas of aquatic macrophytes, grasses and sedges (Pettit *et al.* 2011). These latter systems have been the focus of much of the past research on tropical floodplain wetlands, particularly in the Alligator Rivers Region in Kakadu National Park (Finlayson 2005). Formed only 1,500–6,000 years ago, these tropical floodplains are relatively recent geomorphological features and contain a flora that is common to many other tropical regions of the world with less than 25% of the plant species endemic to northern Australia (Cowie *et al.* 2000).

The pronounced seasonality of the rainfall, and associated patterns of annual flooding and drying, is the primary driver of vegetation dynamics in the region. Local variation in vegetation community composition reflects microtopography and local differences in inundation (Finlayson 2005). The predictability of the timing and magnitude of floods that drive wetland plant dynamics is positively correlated with fish species richness, bird population stability and riparian forest production in both Australian and South American floodplains (Jardine *et al.* 2015). A few species dominate large areas of Australian tropical floodplains; for example, *Oryza* species (wild rice), *Eleocharis* species (spike-rush), *Hymenachne acutigluma* (native hymenachne), *Pseudoraphis spinescens* (water couch) and water lilies (*Nymphaea* species and *Nymphoides* species) (Finlayson 2005). Although these systems are relatively low in plant species richness compared with the surrounding savanna landscape, they are hot-spots of primary production and support an abundance of fauna, particularly fish and waterbirds (Finlayson *et al.* 2006). Recent research on Australia's tropical river ecosystems has highlighted the importance of primary production in floodplain wetlands, particularly by epiphytic algae, for supporting aquatic ecosystems of the floodplain, main river channel and estuary (Jardine *et al.* 2012).

In contrast to many floodplain systems in more populated regions, the hydrology of Australia's tropical floodplains remains largely intact. However, they have been listed as one of Australia's ten most vulnerable habitats due to exotic plant invasions, which transform the structure and function of these systems and the risk of sea level rise (Laurance *et al.* 2011). Invasive perennial grasses such as *Urochloa mutica* (para grass, Fig. 4d) and *Hymenachne amplexicaulis* (olive hymenachne) were introduced for cattle grazing and have since spread long distances assisted by floods and waterbirds and now cover increasingly large areas of some floodplains, including those in Kakadu National Park (Setterfield *et al.* 2013). Although successfully managed in Kakadu, the South American woody shrub *Mimosa pigra* (Mimosa), is a serious weed on several floodplains in the Northern Territory (Setterfield *et al.* 2013). Their low elevation (most are < 2 m above sea level) makes these coastal floodplains highly susceptible to sea-level rise (Catford *et al.* 2013), which has already transformed areas of freshwater wetland to saline swamp, including more than 17,000 ha on the Mary River in the Northern Territory (Mulrennan and Woodroffe 1998).

Desert floodplain wetlands in arid and semi-arid Australia

Desert floodplains occur throughout inland Australia but are particularly well-developed in the Channel Country of the eastern Lake Eyre Basin and, to a slightly lesser extent, in the lowlands of the western Murray-Darling Basin (Capon *et al.* 2016). Characterised by extremely low topographic gradients with mean annual rainfall below 250 mm, these eastern desert floodplains typically encompass complex channel networks that distribute floodwaters over vast areas during periods of high river flows. In contrast, the ephemeral floodplains of Australia's central and western deserts are usually associated with poorly defined drainage systems and are typically inundated by flashy overland flows. In either case, the defining characteristic of desert floodplains is their highly variable and unpredictable patterns of wetting and drying, which have an overriding influence on the composition and structure of their vegetation (Capon 2005).

Desert floodplain wetlands encompass the whole floodplain and are not restricted to depressions within the floodplain. They support a diverse range of short grass, sedge and forb associations that shift in composition and structure both temporally, in response to wetting and drying, and spatially in relation to flood history (Capon 2005). Plant species inhabiting these environments tend to be widely distributed, many having cosmopolitan distributions extending beyond the arid zone (Box *et al.* 2008). Common families represented include Asteraceae, Amaranthaceae, Chenopodiaceae, Cyperaceae, Euphorbiaceae, Fabaceae, Goodeniaceae, Malvaceae, Nyctaginaceae, Poaceae and Polygonaceae. Many species maintain large, persistent soil seed banks, which enable plants to avoid unfavourable conditions (e.g. drought) and re-establish when suitable conditions arise, typically following the drawdown of floodwaters (Capon 2007). In the arid zone, woody vegetation is mostly limited to watercourses and their banks or along flood strandlines. Further to the east however, large areas of semi-arid floodplain woodlands also occur (see Chapter XXX on Floodplain forests). Low-lying swampy areas also frequently support wetland shrublands (e.g. of *Duma florulenta*, *Chenopodium auricomum*).

Water resource development, of both surface and ground waters, probably represent the greatest threat to Australia's desert floodplains, governed as they are by hydrology. Land use is predominantly livestock grazing. Although overgrazing may have significant effects on vegetation during dry periods (e.g. severe vegetation cover reduction, soil compaction), cattle have limited access to vegetation when it is at its most productive and diverse (i.e. following flooding) due to surface instability. Recent intensification of mining exploration and extraction also pose a threat to desert floodplain vegetation as a result of infrastructure development, water use and the disposal of excess ground water. Climate change has the potential to further influence vegetation, especially where water regimes are altered.

Inland saline lakes

Saline lakes and wetlands, including lakes, claypans and rain-filled depressions, are a major feature of inland Australia, accounting for over 80% of total wetland area on the continent (Timms and Boulton 2001, also see Chapter XX on Chenopod shrublands and salt lakes). Additionally, many freshwater lakes become saline as they dry. Most saline lakes occur in Western Australia and South Australia (e.g. Lake Eyre) but there are also many in the eastern mainland states. Most are ephemeral or temporary, but some wetlands, especially in southern Australia, can be seasonal or permanent. Saline lakes also vary in wave action and turbidity (Timms and Boulton 2001).

Vegetation composition in saline lakes is driven by salinity, water regime and turbidity (Porter *et al.* 2007). Species richness typically declines with increasing salinity and is usually greater in temporary, rather than permanent, lakes. However, only a few algal and angiosperm species are capable of persisting under both high levels of salinity and widely fluctuating water levels. Nevertheless, even highly saline lakes can support very productive submerged vegetation (Brock 1994). Rapid growth of submerged plants can occur under conditions of low turbidity when salinities are below ~60 g L^{-1} (Timms and Boulton 2001). Charophytes (e.g. *Nitella* species and *Lamprothamnium* species) are typically well represented and aquatic forbs are usually dominated by species of *Ruppia* and *Lepilaena*. Lakes are often fringed by salt-tolerant samphires (e.g. *Halosarcia* species, *Tecticornia* species), succulents (e.g. *Gunniopsis* species) and sedges (e.g. *Cyperus* species). Forbs and grasses may colonize some ephemeral and temporary saline lakes during dry periods, but some lakes will form salt crusts when dry and remain free of vegetation (Porter *et al.* 2007).

Threats to inland saline lakes include dryland salinization, which has been a widespread result of altered hydrology (from water extraction and river regulation) and vegetation clearing in southern Australia, particularly south-west Western Australia (Halse *et al.* 2003). Dryland salinity, along with a drying climate, has the potential to significantly alter the salinity and water regimes of inland saline lakes in much of Australia's south. Increased nutrient loads in runoff from surrounding farms may also pose a threat (Timms 2005). Extraction of materials from saline lakes, as well as further changes to hydrology resulting from mining infrastructure and water resource development, are also likely to have significant effects (Timms 2005).

Groundwater dependent springs of the Great Artesian Basin

Spring wetlands are permanent wetlands fed by groundwater from underlying aquifers. Spring wetlands occur throughout Australia but are particularly distinctive in arid environments where, as in other countries, they are one of the few sources of permanent water (Fig. 4b, Fensham and Fairfax 2003). Spring wetlands can occur in areas where sediments that form the aquifer project above the ground (outcrop springs). These include springs in aquifer recharge areas where rates of rainfall-derived inflow exceed throughflow and where water-bearing sediments are close to the ground surface, which often occurs at the edges of an aquifer. Groundwater feeding outcrop springs can have short residence times (i.e. water has not been in the aquifer for long), so these types of springs may be ephemeral. Discharge springs occur where water flows upwards through confining beds (aquitard) via faults or conduits in the overlying sediments (Fensham and Fairfax 2003). Discharge springs may sustain small permanent wetlands because groundwater feeding these systems has a long residence time in the aquifer and thus shows minimal fluctuations in flow rates.

The Great Artesian Basin is an aquifer system that underlies 22% of north-eastern Australia, extending over parts of Queensland, NSW, SA and NT (Powell *et al.* 2015). Water percolates through the sandstone aquifer generally in a south-westerly direction, with sandstone intake beds located mainly along the eastern margin in north Queensland (Powell *et al.* 2015). Springs in the Great Artesian Basin are often referred to as mound springs, but only a small fraction of spring wetlands have visible mounds associated with them. Mounds may be formed by upwelling of subsoil in the artesian water, from vegetation-derived peat accumulation, from the expansion of surface clays and from accretion of aeolian sand or calcium carbonate (Fensham and Fairfax 2003). Unlike spring wetlands in arid areas of WA and NT, which are thought to have greater connectivity, the spring wetlands of the Great Artesian Basin have high levels endemism in both flora and fauna and are thus of high conservation value. Of 325 native vascular plants recorded in the Great Artesian Basin spring wetlands, at least eight appear to be endemic, with presumably little opportunity for dispersal outside of these isolated, and environmentally unique systems. These include *Myriophyllum artesum, Eriocaulon carsonii* and related subspecies, and *Sporobolus pamelae*, along with described species from the families Poaceae, Cyperaceae and Scrophulariaceae. At least 30 exotic plant species have also been recorded in these wetlands (Fensham and Price 2004).

Spring wetlands in the Great Artesian Basin are severely threatened by groundwater extraction and aquifer drawdown. Between 1880 and 1990s, water levels were drawn down by as much as 100 m in parts of Queensland. As a result, 89% of spring-groups have become inactive throughout much of Queensland over the last century (Fensham and Fairfax 2003). The discharge springs are most affected by groundwater extraction and 87% of springs in the discharge area have become partly or completely inactive due to groundwater drawdown, as opposed to only 8% in the recharge area. Of the springs that are still active, 26% have been severely degraded through wetland excavation, which is intended to increase access for livestock. Other threats include exotic species used for ponded pasture (e.g. *Hymenachne* *amplexicaulis, Urochloa mutica, Echinochloa polystachya*), livestock trampling, exotic animals (e.g. pig rooting) and fire (Fensham and Fairfax 2003). To protect the remaining spring wetlands, no more bores should be developed near springs, bores close to springs should be capped, and spring excavation and use of exotic ponded pasture should be prohibited (Fensham and Fairfax 2003). Greater protection of spring wetlands, especially in discharge areas, through the reserve system would be valuable.

Freshwater wetlands on coastal floodplains of south-eastern Australia

Coastal freshwater wetlands occur around the coast of Australia on sandy loam soils of waterlogged or periodically inundated alluvial flats. Primarily fed by rivers, coastal freshwater wetlands are similar in vegetation structure to inland floodplain wetlands, but tend to be located at elevations less than 20 m above sea-level, have sandier soils, higher rainfall, and periodic saline water input (Whitehead et al. 1990). Refer to Chapter XX for Halophytic vegetation, including coastal mangroves and saltmarshes.

Vegetation communities in coastal floodplain wetlands are largely determined by frequency, duration and depth of inundation, but are being increasingly affected by human activities that include livestock grazing, catchment land clearing and pollution (Pressey & Middleton 1982). Freshwater wetlands in coastal plains typically have low cover of woody species, though will occasionally have scattered trees (e.g. *Casuarina glauca, Melaleuca ericifolia*) that are common in neighbouring swamp floodplain forests (de Jong 2000). Wetlands that are dry most of the time are typically dominated by dense grassland, sedgeland, or rushland under 0.5 m tall. Common species include *Paspalum distichum, Leersia hexandra, Pseudoraphis spinescens,* and *Carex appressa.* Wetlands with more regular cycles of inundation and drying may support taller emergent species, such as *Bolboschoenus* species and *Schoenoplectus* species up to 1 m tall. Common herbs in these areas include *Hydrocharis dubia, Philydrum lanuginosum, Ludwigia peploides, Marsilea mutica,* and *Myriophyllum* species. As water levels become deeper or more permanent, these systems begin to develop more floating vegetation such as *Azolla* species, *Lemna* species, *Hydrilla verticillata, Ceratophyllum demersum, Nymphoides indica, Ottelia ovalifolia,* and *Potamogeton* species.

Land development is a particular threat for freshwater wetlands along the coast because of the high value of the land, and because high organic matter in the sediments of freshwater wetlands make them ideal for agricultural development, including exotic pasture. Koo Wee Rup Swamp, historically located along the coast between Melbourne and Gippsland, was once one of the largest freshwater wetlands in Victoria. Extending over 40,000 hectares, this swamp was drained for agriculture in 1876, destroying dense stands of swamp paperbark (*Melaleuca* species) and giant bulrush (*Typha* species) (Yugovic and Mitchell 2006). Coastal wetlands filter overland flow, improving water quality before it enters the ocean. The degradation and loss of wetlands can result in increased sediment and nutrient concentrations in runoff, which may lead to algal blooms, increased turbidity, and the degradation of marine ecosystems (Johnson et al. 1999). Coastal wetlands are themselves threatened by eutrophication, especially by nitrogen and phosphorus that originates from urban and agricultural areas, which results in the displacement of native species by exotic species.

In coastal areas, where most of the population resides, many wetlands were either 'reclaimed', via drainage or infilling, or regulated by control structures (e.g. barrages) to permit urban and agricultural development. This led to the fragmentation, degradation and loss of a large proportion of Australian coastal wetlands, which continue to be very poorly researched ecosystems. In south-western Western Australia, around 70% of coastal wetlands are estimated to have been lost since European settlement (Davis *et al.* 2015). In many coastal regions, wetland drainage and land use change have also led to the exposure of underlying sulfidic sediments (i.e. acid sulfate soils) with a wide range of detrimental outcomes including diminished plant growth (White *et al.* 1997).

Billabongs and lakes of the southern Murray-Darling Basin floodplain

The floodplains of the Murray, Murrumbidgee and Goulburn Rivers are inset with two major types of wetlands that differ in geomorphology – billabongs and lakes (see Chapter XX on Floodplain forests that describes the drier parts floodplains). These fill, drawdown and dry in response to river inundation, which varies across the floodplain resulting in a mosaic of water regimes, ranging from near permanent to episodic.

Billabongs are former river channels, typically small (surface area < 10 ha), mostly clay-based, and very abundant, numbering thousands per floodplain. Though billabongs may be referred to as oxbow lakes, billabongs can take many shapes and forms, and often simply feature as depressions in the floodplain (Fig. 4c). The diversity of wetland plants within a billabong is strongly influenced by wetland form: deeper wetlands have littoral vegetation on their slopes and submerged herbland on their bed when flooded, whereas shallower wetlands may have sedgeland throughout. Floodplain lakes are deflation basins with a clay or sand lunette on one margin, typically large (surface area 100-600 ha), and are gilgai, clay or sand-based (if not in-filled by sediment): they number tens to hundreds per river system

(Pressey 1986). Their compact round shape and uniform floor result in low environmental heterogeneity, so individual lakes typically have a single dominant vegetation type.

The mosaic of water regimes occurring across a floodplain means floodplain wetlands may carry a complete spectrum of growth forms, such as river red gum (*Eucalyptus camaldulensis)* woodland, lignum (*Duma florulenta*) or chenopod shrubland, sedgeland, various types of grassland, and various types of aquatic herbland, and the number of species is correspondingly large. The species pool is a mix of Australian families and species (Myrtaceae and Chenopodiaceae for trees and shrubs), cosmopolitan or widely-distributed species (e.g. *Phragmites australis*, *Potamogeton crispus*), Australian aquatic and amphibious representatives of cosmopolitan or widely-distributed families (Cyperaceae, Poaceae, Typhaceae, Juncaceae, Juncaginaceae, Haloragaceae, Ranunculaceae, Elatinaceae, Polygonaceae), and a few species with restricted distributions (e.g. *Amphibromus fluitans*, *Myriophyllum porcatum*). Although the species composition of floodplain wetlands is highly variable, wetlands with similar water regimes have functionally similar plant communities (Casanova 2011).

Over the last 200 years, floodplains of the southern Murray-Darling Basin have been dramatically altered through vegetation clearing, cultivation, livestock grazing, sediment fluxes, altered water quality, river regulation, salinization and acidification. The widespread decline of submerged macrophytes and high abundance and richness of exotic species exemplify this (Catford *et al.* 2011). River regulation is a major (but not sole) factor determining the abundance of exotic species. In billabongs, 20-40% of species may be exotic, and most of these are short-lived terrestrial species such as pasture grasses or agricultural weeds. Their presence is a legacy of past agricultural enterprises (Lunt *et al.* 2012) but their abundance is boosted by the generally drier conditions that river regulation imposes. These conditions have the dual effect of disadvantaging the already established native amphibious species and of providing conditions suitable for exotic terrestrial species, but not native terrestrial species (Catford *et al.* 2011). Regulation disrupts the match between native species phenology and floods, and seasonal flow inversion (high flows in summer) favours the dispersal and establishment of exotic species generally (Greet *et al.* 2012) thus river regulation exacerbates the threat of invasive exotics such as amphibious *Nymphaea mexicana and Sagittaria platyphylla* (Fig. 4d & f).

Submerged macrophyte abundance and diversity has been adversely affected by i) increased sediment flux from eroded uplands, which has increased turbidity and reduced photic depth; ii) exotic common carp (*Cyprinus carpio*) that can dislodge macrophytes through their benthivorous feeding strategy, a strategy not used by native fish; and iii) the prolonged Millenium Drought in the early 21st century. Tolerance to such pressures requires capacity to grow under low light, bulky rhizomes and relatively long-lived seed (6-9 years) that can aid recolonisation (Brock 2011). Only the robust perennial, *Vallisneria australis*, is known to have these characteristics.

Bogs and fens of the Australian Alps and Tasmania

The Alpine Sphagnum Bogs and Associated Fens ecological community predominantly occurs in the Australian Alp Bioregion, which spans 375 km from Mount Baw Baw in Victoria to southern New South Wales and the western margin of Australian Capital Territory. These communities are also found in Tasmania, and typically occur above the treeline in alpine, subalpine, and montane environments. There is no strict altitude requirement, however, since local climatic conditions can enable these communities to flourish at lower altitudes and below the climatic treeline. Alpine bogs and fens shelter endangered plants, such as the Bogong Eyebright (*Euphrasia eichleri*), and are generally less than 1 hectare in size contributing to their protection under the Environmental Protection and Biodiversity Conservation Act of 1999. For a broader description of alpine vegetation, see Chapter XX.

Alpine bogs and fens are both peatland ecosystems. Peat is terrestrial soil composed of at least 20% organic matter, which is at least 30 cm deep. Peatlands occur in areas with higher water inflow than outflow, or where there is a large supply of groundwater. Waterlogged soil produces reduced, anaerobic conditions that inhibit activity of microbial decomposers, allowing carbon-rich undecomposed plant remains to accumulate. Accumulation of plant detritus into peat can produce small dams, which further modify hydrology in favour of continuing peat formation. Dynamic growth of peatlands can produce a mosaic of heterogeneous habitat that supports a range of vegetation communities (Camill, 1999).

Although alpine bogs and fens are both characterised by presence of peat soils, and often co-occur in the landscape, they differ in nutrient levels and the vegetation communities they support. Fens are primarily fed by surface flow and groundwater, which is nutrient rich, and support sedge communities dominated by species such as *Carex gaudichaudiana.* Sphagnum mosses are generally absent from these areas as they are outcompeted by plants that are better able to exploit high nutrient environments. Bogs, in contrast, have very low nutrient levels because they mostly rely on rainwater (though streams can run through “valley bogs”). Bog soils are generally more acidic and support sphagnum mosses such as *Sphagnum* *cristatum* and *Sphagnum novozelandicum* and sclerophyllous shrubs up to 1 m tall such as *Bossiaea foliosa*, *Epacris glacialis*, and *Grevillea australis.*

Alpine bogs and fens face a number of threats related to their restricted geographical extent and small patch size. There are more than 11,000 bogs and fens throughout the Australian Alp Bioregion, the high altitude areas of south-eastern Australia's Great Dividing Range. Their small size and patchy distribution make them difficult to protect against trampling from livestock and feral animals (e.g. horses, deer, pigs). Exotic ungulates (e.g. cattle, sheep, horses, pigs) trample bog and fen vegetation and also create incisions in peat soil that result in wetland drainage.

Presence of water is vital for the persistence of hydrophytic bog and fen vegetation, and so threats that impact water supply or peat structural integrity endanger these largely endemic communities (Wahren *et al.* 1999). Climate change is predicted to increase occurrence and severity of drought and fire, which damage peat. Fires can destroy carbon-rich peat, changing the physical structure of bog and fen soils and inhibiting their ability to capture and store water. Models predict that alpine bogs and fens around the world are among the ecosystems most threatened by climate change (Schuur et al., 2008).

Groundwater dependent wetlands of the southwest

Characteristics of freshwater wetland vegetation communities in south-western Australia are largely determined by wetland water source, sediment type and duration of inundation (DEC 2012). There is a high number of permanently and seasonally inundated, shallow (<3 m) wetlands in the region for which groundwater is the dominant source of water, either from local and regional superficial groundwater or from perched groundwater held by an impeding sediment layer that is recharged directly by rainfall.

Despite having a hot, dry Mediterranean summer, the southwest is home to flora that represents the world centre for diversity in a number of wetland-associated plant families (e.g. Droseraceae, Restionaceae, Juncaginaceae, Centrolepidaceae and Hydatellaceae). The vegetation communities include a range of endemic wetland genera (e.g. *Reedia, Cephalotus, Tribonanthes, Epiblema, Schoenolaena, Cosmelia, Euchilopsis, Acidonia, Homalospermum, Pericalymma* and *Taxandria*) and high levels of local and regional endemism. The southern Swan Coastal Plain alone has approximately 445 wetland-associated plant taxa, 74% of which are endemic to Western Australia and 3% endemic to the southern Swan Coastal Plain (DEC 2012).

The dominant growth forms in Swan Coastal Plain wetlands appear to reflect the seasonally dry Mediterranean climate. The flora is rich in shrubs, but herbs and sedges make up 77% of taxa (Fig. 4e). Although 78% of species are perennial, 22% of these regrow their aboveground biomass from underground storage organs each year (DEC 2012) – a likely adaptation to seasonal drought and disturbance such as fire.

Seasonally waterlogged wetlands are typically maintained via perched groundwater and have the greatest diversity (DEC 2012). This floristic diversity reflects spatial and temporal patterns in waterlogging (e.g. duration), sediment stratigraphy (e.g. distribution of organic fraction) and composition of impeding layer (e.g. ironstone, calcrete or granite). These systems are dry in summer, with the plant diversity not becoming apparent until late winter and spring when the sedgelands and herblands regrow from their underground storages. They typically support a mosaic of vegetation units, especially woodlands (dominated by *Eucalyptus rudis* and *Melaleuca* species), shrublands (dominated by a diverse suite of species of Myrtaceae e.g. *Calothamnus lateralis, Melaleuca teretifolia, M. viminea, Astartea* species, *Pericalymma ellipticum, Kunzea* species, *Hypocalymma angustifolium,* and some Proteaceae), sedgelands, herblands and rarely grasslands.

Seasonally waterlogged and perched groundwater dependent wetlands were once extensive in the southwest, particularly on the Swan Coastal Plain, but due to land development and hydrological changes, many are now only represented by remnant portions (DEC 2012). Groundwater levels on the Swan Coastal Plain have been declining since the 1970s due to changes in climate, land use and groundwater extraction. Declining groundwater levels have affected vegetation in groundwater-dependent wetlands in a number of ways, including: reduced surface water levels and in some cases drying and loss of wetland habitat; peat fires; declining health and death of some groundwater-dependent vegetation; changes in flow from springs/seeps; and acidification of groundwater and wetlands (Sommer and Froend 2014). With changing hydrological conditions, many permanent/semi-permanent wetlands are developing vegetation characteristics of seasonally waterlogged wetlands, whilst formerly waterlogged areas are being encroached by dryland species. Longer periods of exposed, dry sediments have also lead to increased damage from illegal vehicle access and disturbance of sediment surfaces. The waterlogged fringes of more permanent wetlands have also been extensively reclaimed as water levels gradually fall and urbanisation encroaches.

Artificial wetlands

Artificial wetlands are waterbodies that are wholly or largely anthropogenic in origin, in contrast to natural wetlands that have been formed solely by hydrological and/or geomorphic processes over time. They are typically used for stock watering (farm dams, bore-fed wetlands), water storage (irrigation water storages, weirpools, upland reservoirs, lowland storages), amenity (urban lakes and ponds, Fig. 4f), water treatment (waste water treatment ponds, ponds and meadows as part of water-sensitive urban design) or production (rice fields). Artificial wetlands differ from natural wetlands in being younger, and having differing combinations of size, shape and water regime: reservoirs and weirpools are considerably deeper; storages and amenity lakes are permanent; weirpools and amenity lakes have stable water levels; ricefields provide shallow clear water in spring-summer in a semi-arid region.

A wetland assemblage develops (or is planted) that is dominated by native species (Dugdale *et al.* 2009) but distinct from its regional equivalents (Badman 1999). Riverine storages and amenity lakes are notable exceptions to native dominance; these are often dominated by floating or submerged macrophytes, including highly invasive exotic species like *Egeria densa*, *Cabomba caroliniana* and *Nymphaea mexicana* (Fig. 2d, Dugdale *et al.* 2009). Characteristics of artificial wetlands that may constrain vegetation assemblages from developing are: high turbidity, which excludes submerged plants; water deeper than 1.5 m with steep vertical banks, which precludes the development of a littoral zone; permanent water or extreme fluctuations which preclude successful establishment and persistence (Casanova *et al.* 1997).

Hydrological connectivity is variable. A few farm dams and all bore-fed wetlands are hydrologically isolated, and their plant assemblages have developed independently of hydrochory. Some, notably meadows in urban contexts, are only intermittently connected. The continuous hydrological connectivity of riverine storages such as Lake Mulwala on the River Murray means a continual threat of downstream distribution of vegetative fragments of undesirable species. Although utilitarian in intent, artificial wetlands may develop ecological value as habitat for fauna, such as waterbirds: an example is the Ramsar-listed Western Treatment Plant at Werribee, Victoria.

Summary

The nine case studies presented illustrate commonality in the forces that structure wetland vegetation across Australia, as well as the processes that threaten them (Fig. 1). The primacy of hydrology and wetland water regimes for determining the distribution and characteristics of wetland vegetation is particularly apparent. It is the variation in each of these structuring forces, and interactions among them, that give the plant communities of individual wetlands their own character.

Threats

Practically all wetland systems, with the notable exception of tropical floodplain wetlands and some springs, are threatened by changes in water supply and water regimes. Such changes are caused by the extraction and regulation of surface and groundwater, with modification of surface water being particularly pronounced in southeastern Australia and groundwater extraction affecting wetlands in central and western Australia. Depending on the system in question, the chief threat may involve an increase in water persistence (i.e. for wetlands that are naturally ephemeral or intermittent) or a decrease (i.e. for wetlands that are naturally always wet, like bogs, fens and some springs). Increased water persistence (e.g. around dams and weir pools) can lead to tree death and shifts towards species favoured by stable water levels, e.g. *Typha* species (Blanch *et al.* 2000), whereas reductions in flooding can lead to wetland loss or result in native macrophytes being replaced by woody species (Bren 1992) and exotic terrestrial herbs (Catford *et al.* 2014). Climate change also threatens wetland hydrology, as illustrated in alpine bogs threatened by declining rainfall and runoff, and in coastal systems at risk of saltwater intrusions from rising sea levels. Overall, climate change has the capacity to reduce native wetland plant diversity, promote exotic invasions, accelerate wetland fragmentation and homogenise wetland ecosystems at local and landscape scales (Capon *et al.* 2013).

Exotic species are threats throughout Australia, and are both a symptom (e.g. in response to flow regulation along the River Murray, see Case Studies) and a cause of changes in native flora (e.g. exotic pastures that outcompete native vegetation, Fensham and Fairfax 2003). Fifteen of the 32 Weeds of National Significance invade riparian or aquatic systems and were intentionally introduced (see Chapter XX on Invasive plants). Ponded pasture and other exotic pastures continue to be planted, and new varieties that will likely increase their invasion success continue to be developed (Driscoll *et al.* 2014). Exotic animals also threaten native plants directly through grazing, trampling and uprooting (e.g. ungulates, carp) and indirectly through soil compaction or increased gully erosion, for example. Changes in the land use of wetlands (e.g. livestock access to alpine and Murray-Darling Basin wetlands) and their catchments for agriculture, urbanisation and increasingly mining have led to sedimentation, pollution and eutrophication, salinization, acidification, changes to hydrology and connectivity among wetlands.

Management and policy

Policy relating to wetland vegetation

The first national policy on wetland management was released in 1997 (The Wetlands Policy of the Commonwealth Government of Australia). Whilst all states and territories have legislation to protect the environment and conserve natural resources (e.g. environment protection, land use planning, protected areas, water and vegetation management), state-level policies rarely address wetlands and wetland vegetation directly. In 2013, the National Wetlands Policy for Australia was initiated and is designed to improve alignment of Australian Government, state/territory and community action and investment, and provide consistency in decision-making across government agencies (WetlandCare Australia 2013).

Who is responsible for the management of wetland vegetation?

The management of wetlands and associated vegetation in Australia is primarily the responsibility of landowners and land managers. Individual states or territories have the legislative and policy responsibility for natural resource management (Finlay-Jones 1997). The Australian Government administers the Ramsar Convention on Wetlands of International Importance by providing national wetland policy, leadership and direction. It also implements the Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act), and develops programs to improve the management of wetlands. The EPBC Act specifies that any activities within or outside designated Ramsar wetlands that are likely to have a significant impact on the ecological character of listed wetland, including the vegetation, must be approved. Managers may also need approval from the relevant agencies under state legislation for activities potentially impacting Ramsar listed and non-Ramsar listed wetlands. Indigenous management of wetland vegetation continues in much of the country and has been revived in some places as a result of native title.

Management approaches

Like all plant communities, wetland vegetation is shaped by the composition of the regional species pool, local dispersal, environmental conditions and interactions with other biota (Fig. 1, Catford and Jansson 2014). Alteration to any one of these four elements through e.g. exotic species introduction, reduced connectivity among wetlands and flow regulation can alter wetland vegetation. Effective management requires a comprehensive understanding of the factors and processes that influence wetland vegetation so that the chief threats to wetland vegetation can be identified and targeted (Fig. 1).

Threats to wetland vegetation along the River Murray include cattle grazing, eutrophication, drought, human activities around towns and farms, and flow regulation. Of these, flow regulation was found to be primarily responsible for an observed decline in native vegetation, and particularly native amphibious vegetation (Catford *et al.* 2014). By altering historical water regimes, and reducing flood magnitude in particular, flow regulation inhibits native plants that are poorly adapted to the modified flooding regimes. With less competition from native plants, exotic plant species (most of which are terrestrial) have increased in abundance. Environmental flows can reduce some of these negative effects, but it is rarely possible to provide the full spectrum of needs for wetland plants, and also service other wetland needs. For example, environmental flows are commonly used to extend the duration of natural flood events because flood duration is crucial for fish spawning and waterbird breeding, but extended duration into summer months favours competitive emergent macrophytes such as *Typha* or *Juncus ingens* and can lead to perverse outcomes (Box 1).

Once a threat is identified, varied approaches can be used to ameliorate it. For example, for systems with an insufficient supply of propagules, seeds can be sown directly or hydrological connectivity can be re-established among wetlands to facilitate dispersal. When various management options are available and the most effective one unknown, modelling can aid decision-making by predicting the likely response of vegetation communities to different management actions. Based on ecological understanding and empirical data, a state and transition model has been developed for the Ramsar-listed Macquarie Marshes to model effects of flood frequency, distance to river and fire frequency on vegetation dynamics (Bino *et al.* 2015).

Modelling is a valuable tool for anticipating ecological responses to management actions, but can also be used to anticipate effects of environmental conditions not previously experienced (Catford *et al.* 2013). Novel ecosystems that arise from global environmental change and exotic species invasions present new challenges for environmental policy and management. In cases where maintenance or restoration of historical vegetation is unrealistic, wetland managers may instead focus on the maintenance of key ecosystem services or functions (e.g. water filtering, carbon sequestration, flood attenuation). This could involve facilitating the establishment of salt-tolerant species for wetlands affected by secondary (human-induced) salinization (Sim *et al.* 2006).

Conclusion

Wetlands and wetland vegetation are of critical ecological, social, cultural and economic value, exemplified by their likely role as refugia and stepping stones for dispersal in a changing climate (Capon *et al.* 2013). Over half of Australia's wetlands have been lost in the last 200 years (Bennett 1997). Those remaining are severely degraded yet they are facing increasing threat from rising water demand, land use change, mining, climate change, agricultural development in new regions, rising sea levels and associated saltwater intrusion, and the introduction and spread of invasive species. Expansion of irrigated agriculture in northern Australia would increase pressure on freshwater ecosystems in the north (Douglas *et al.* 2011) and continued extraction of groundwater will exacerbate wetland degradation in central and western Australia. Wetland vegetation is resilient by its very nature with plant physiological and life history adaptations to cope with a highly variable environment. However, they can only withstand so much. To increase protection of wetland ecosystems, greater inclusion in the national reserve system is required and increased policy initiatives that protect them and the water on which they rely. The loss and degradation of wetland ecosystems throughout the Great Artesian and Murray-Darling Basins provide sobering lessons for other parts of the country.

Acknowledgments

We thank David Keith for inviting us to write this chapter, Patrick Driver for the idea and information for Box 1, and Margaret Brock and Greg Keighery for feedback on the chapter plan, and Margaret Brock and an anonymous reviewer for comments on a chapter draft. JAC and SW acknowledge financial support from the ARC Centre of Excellence for Environmental Decisions.

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Display items

Table 1 Hierarchical classification scheme used to categorise wetland species based on their response to water regime. Classification was devised by Brock and Casanova (1997) based on field and germination trials.

Box 1 Perverse outcomes of environmental watery delivery.

Figure 1 Key abiotic, biotic and human factors that shape the biogeography and ecology of wetland vegetation in Australia. Arrows indicate major causal relationships (e.g. climate affects hydrology; lithology affect geomorphology; hydrology, geomorphology and hydraulics collectively affect water quality and hydrological connectivity). Feedbacks (not shown to avoid clutter) occur among and within the abiotic and biotic drivers, between vegetation and abiotic drivers (e.g. vegetation can modify hydraulics, water quality, microclimates) and between external factors (e.g. bushfires, land use practices) and the abiotic and biotic components of wetlands. Humans affect wetland vegetation directly and indirectly by altering environmental conditions, dispersal opportunities and biotic interactions.

Figure 2 Range of common Australian wetland plant species highlighting the diversity of wetland plant growth forms: a) *Ludwigia peploides* subsp. *montevidensis* and *Myriophyllum* species; b) outer, darker band of *Eleocharis acuta* and inner band of *Pseudoraphis spinescens* illustrate characteristic zones of wetland vegetation; c) *Typha orientalis*; d) exotic *Nymphaea mexicana,* Jerrabomberra Wetlands, ACT; e) floating *Azolla pinnata* with submerged exotic *Egeria densa*, Euston Lakes, NSW; f) emergent form of exotic *Sagittaria platyphylla* with an inset showing its submerged form, which is just starting to bolt into the emergent form. Photo credits: Jane Catford (a, b, c, f: all taken in River Murray wetlands in Victoria and NSW); Jane Roberts (d, e).

Figure 3 a) Number of publications in ISI Web of Science on wetland vegetation globally and for the ten countries with the greatest number of publications [grey bars are total publications – note the disjointed y-axis; black bars are publications with applied angle; % indicates percentage of total studies that have an applied angle]; year of first publication per country, followed by year of first applied publication in parentheses: global 1900 (1968), USA 1900 (1981), Canada 1928 (1983), China 1991 (1995), Germany 1980 (1991), England 1973 (1975), Australia 1974 (1990), Spain 1985 (1992), France 1972 (1990), Netherlands 1973 (1984), Sweden 1972 (1992); **b)** Number of publications over five year periods focusing on wetland vegetation in Australia [grey line indicates total publications (<16 per five year interval until 1991-1995 period); black line indicates publication with applied angle (0 before 1990); number of 2015 publications assumed to be the same as 2014]. Search terms on Web of Science (15 October 2015): [TOPIC: ((Wetland* OR Bog* OR Marsh* OR Swamp* OR Fen* OR Mire* OR Riparian* OR Lake* OR Littoral* OR Spring*) AND (Vegetation OR Flora OR Floristic* OR Plant* OR Macrophyt* OR Hydrophyt*)); RESEARCH AREAS: (ENVIRONMENTAL SCIENCES ECOLOGY OR PLANT SCIENCES OR MARINE FRESHWATER BIOLOGY OR WATER RESOURCES OR PHYSICAL GEOGRAPHY OR BIODIVERSITY CONSERVATION); Timespan: All years.; Search language=Auto] *plus* [TOPIC: ((manag* OR restor* OR rehabilitat* OR applied OR policy OR policies))] *plus* [Country: Australia].

Figure 4: Examples of wetland ecosystem types: a) tropical coastal floodplain, NT; b) spring wetland in central Australia; c) floodplain wetland of the River Murray, Victoria; d) monoculture of exotic *Urochloa mutica* (para grass) in Kakadu, NT; e) groundwater dependent wetland in southwest WA; and f) constructed stormwater treatment wetland in Canberra, ACT. Photo credits: Michael Douglas (a, d); Jane Catford (b, d, f); Ray Froend (e).