

DETECTING EFFECTS OF ENVIRONMENTAL WATER ALLOCATIONS IN WETLANDS OF THE MURRAY–DARLING BASIN, AUSTRALIA

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ABSTRACT

River regulation has significantly altered the hydrology and ecology of floodplain wetlands throughout the Murray–Darling Basin, Australia. Management plans proposed for rivers in the Murray–Darling Basin incorporate ‘environmental water allocations’ (EWAs) designed to redress some of the damage caused by regulation, via partial restoration of the natural hydrological regime that used to be experienced by associated floodplain wetlands. Monitoring and scientifically rigorous adaptive management practices are the key to the long-term success of EWAs, and successful monitoring relies on the well-informed selection of a variety of hydrologically sensitive indicators. In this paper, we recommend a range of physical, chemical and biological indicators for use for monitoring change in wetland health in response to EWAs. Physical and chemical variables suggested include wetland depth, wetland area and salinity. Aquatic macrophytes and macroinvertebrates are recommended as the primary biological indicators for monitoring change within the Murray–Darling Basin, although the indicator potential of macroinvertebrates still has to be confirmed by planned and ongoing research. Information is also presented for a variety of other components of wetland ecosystems, including biofilms, zooplankton, birds, fish, mammals, reptiles, amphibians and fringing vegetation. Our current knowledge of the relationships of these variables with wetland hydrology and ecosystem health is relatively limited. Further research is required to investigate the nature of these relationships and determine the utility of these parameters as indicators within wetlands of the Murray–Darling Basin. Copyright © 2000 John Wiley & Sons, Ltd.

KEY WORDS: ecosystem health; flow regime; hydrology; indicators; management; monitoring; Murray–Darling Basin; salinity

INTRODUCTION

The hydrological regime experienced by floodplain wetlands, in particular the timing and duration of the river flood pulse, is the major factor responsible for the composition, structure and function of floodplain wetland communities (Junk *et al.*, 1989). The life cycles of wetland biota are related to the flood pulse in terms of its timing, its duration and the rate of rise and fall of the floodwaters. The timing of breeding cycles of fish, and seed production by fringing vegetation, such that they capitalize on periods of floodplain inundation, offer two examples of the dependence of wetland flora and fauna on the timing and duration of river floodwaters (Junk *et al.*, 1989).

The regulation of rivers affects all features of the hydrological regime experienced by floodplain wetlands. Depending on the form of regulation, river flow may become more or less temporally variable, may increase or decrease in magnitude, and may peak at times of the year when flow would naturally be at its lowest (Maheshwari *et al.*, 1995). These changes to river flow conditions have direct consequences for the timing, magnitude and duration of floodwaters received by river floodplains, which can result in significant, lasting changes to the ecology and health of associated wetlands.

The Australian government has allocated water ‘for the environment’ in an attempt to re-establish more natural hydrological regimes and improve the ecosystem integrity of floodplain wetlands within the

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Murray–Darling Basin (Figure 1). The key to the long-term success of these ‘environmental water allocations’ (EWAs) is a well-established monitoring programme; monitoring must involve ecological indicators that are sensitive to hydrological variation, whose selection has been based on scientific criteria. To this end, this paper reviews potential indicators that might be used to detect ecological responses to hydrological changes in wetlands of the Murray–Darling Basin. Our aim in conducting this review is to provide wetland managers and scientists with sufficient information to select physical, chemical and biological indicators for use when monitoring wetlands of the Murray–Darling Basin to assess the effectiveness of EWAs.

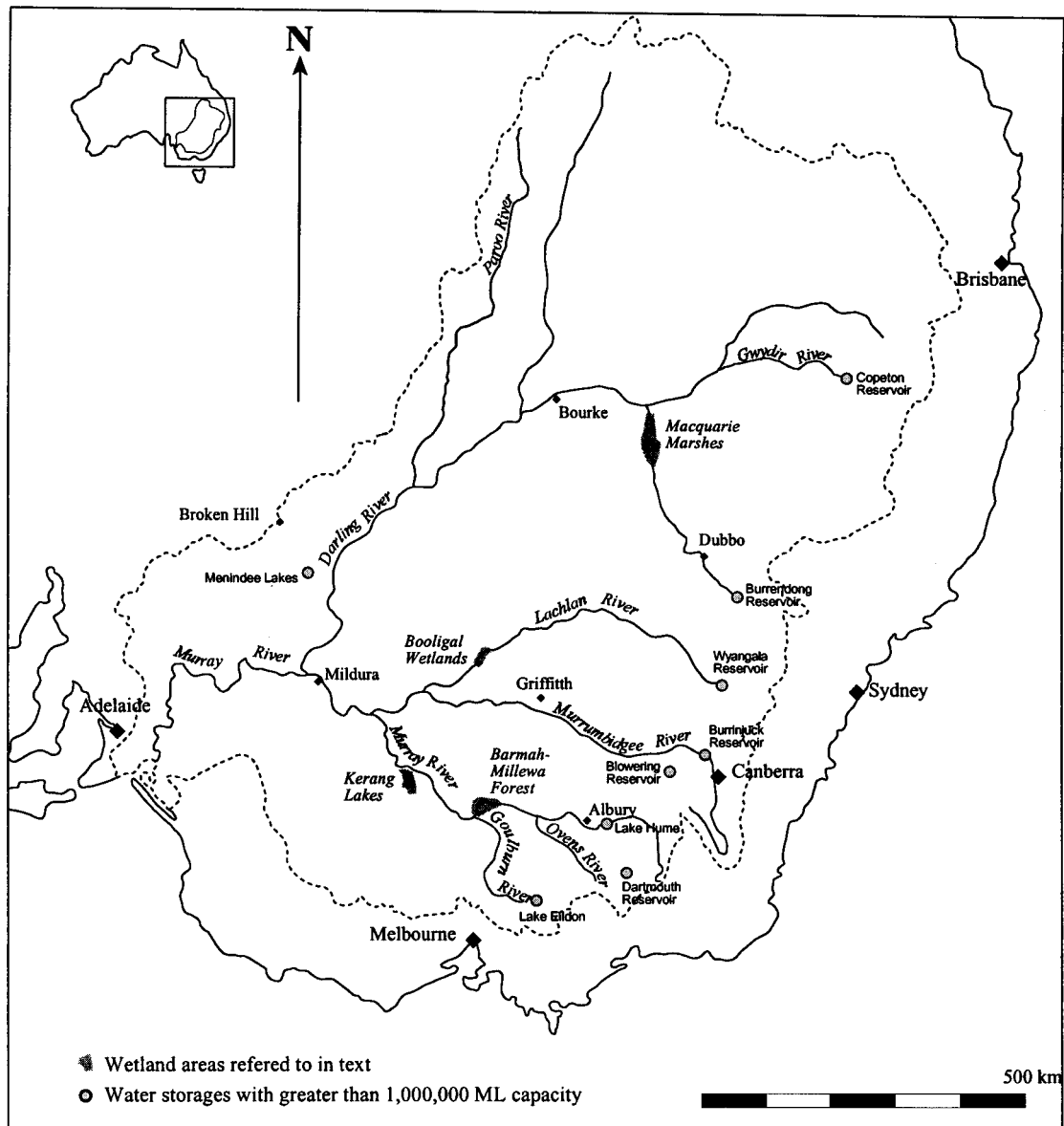


Figure 1. The Murray–Darling Basin

HISTORY OF RIVER REGULATION WITHIN THE MURRAY–DARLING BASIN

The Murray–Darling river system extends over most of southeastern Australia, covering nearly 14% of Australia's land area (Figure 1). The combined lengths of the River Murray (2560 km) and the Darling River (2740 km) make the system the fourth longest in the world. In total catchment area (1.073 million km²), the Murray–Darling ranks sixth largest in the world (Walker, 1992); however, most of this land is arid or semi-arid and only a small part contributes significant runoff (Walker and Thoms, 1993).

Prior to regulation, the River Murray and its tributaries were characterized by highly variable discharge, slow flows and gentle land gradients. In combination, these features resulted in the inundation of adjoining land tracts during flooding, causing the establishment of extensive floodplains and wetlands throughout the Murray–Darling Basin. The great size of the Murray–Darling Basin means that it extends across several major biogeographical regions and incorporates substantial precipitation and temperature gradients. These features have contributed to the great variety of floodplain wetlands located within the Murray–Darling Basin, including abandoned channel sections (typified by the classic ox-bow lake or billabong), ephemeral streams and anabranches (e.g. the Edward–Wakool system on the River Murray), sub-terminal distributary deltas (e.g. the Macquarie Marshes), lake–lunette complexes (e.g. the Kerang Lakes) and palaeo-channels (e.g. Lake Moodemere and Croppers Lagoon on the mid-Murray). These wetlands range from temporary to permanent, and from freshwater to hypersaline.

River regulation in the Murray–Darling Basin has a long history. The use of river water for irrigation first began in an *ad hoc* manner in the Kerang region of Victoria during the 1870s (Close, 1990). By the late 1880s, large-scale irrigation districts had been established in northern Victoria. These areas were supplied with water from weirs constructed on the Goulburn, Loddon and McKenzie Rivers. Irrigation subsequently commenced in South Australia in 1894 and in New South Wales (NSW), centred on the Murrumbidgee River, during the 1910s (Crabb, 1997; see also Powell, 1993).

Until the 1920s, regulatory structures consisted primarily of weirs designed to maintain river levels at irrigation supply off-takes. Large capacity reservoirs were constructed in headwater regions of the Murray, Murrumbidgee and Goulburn rivers during the 1920s (Erskine, 1996; Crabb, 1997) (Figure 1). Construction of new dams on other rivers in the Murray–Darling Basin, and modifications to the older dams, has continued throughout this century. Today, the only major rivers that remain relatively unregulated are the Ovens River in the south, and the Warrego and Paroo rivers in the north.

The combined effects of water extraction and flow regulation have had a dramatic effect on rivers and wetlands within the Murray–Darling Basin. Close to the major impoundments, the most dramatic changes to river flows have been to the timing and magnitude of high and low flows. Under natural conditions in the south of the basin, where most of the large reservoirs are located, lowest flows would occur during the summer months. However, this season coincides with the period of peak irrigation demand when water must be released from storage. Accordingly, the annual hydrographs of rivers immediately below these large impoundments are virtually reversed. Summer water releases have the added effect of reducing the frequency of very low flows in the middle reaches of rivers such as the Murray, Murrumbidgee and Goulburn. In some instances, irrigation releases can also lead to so-called 'rain-rejection' floods. These floods are usually of small magnitude and occur when downstream irrigators cancel water ordered previously, after local rains have made its use unnecessary (Chesterfield, 1986; Ward *et al.*, 1994). The large impoundments also have a flood mitigating capacity: while very large floods still occur at around the same frequency, mid-sized floods are much less common (Maheshwari *et al.*, 1995).

The supply of water to the floodplain wetlands within the Murray–Darling Basin is largely via input from the associated main stream during high flow, rather than by local runoff (Maheshwari *et al.*, 1995); therefore, alterations to river hydrology are of direct consequence to both the hydrology and the ecology of associated floodplain wetlands. In addition to the broad scale hydrological changes arising from flow regulation, some larger lakes, such as the Menindee Lakes and several of the Kerang Lakes (Figure 1), have been actively modified for use as irrigation water supply basins. Many smaller wetlands have also been affected (both incidentally and intentionally) by the construction of irrigation channels (or the enlargement of existing, natural channels) and levees. Pressey's (1986) hydrological classification system

Table I. Hydrological classification of River Murray wetlands (Pressey, 1986)

Category	Description	Effect of regulation on hydrology
1	Wetlands connected to the river at minimum regulated flow or at weir pool level, or potentially connected at these levels but separated by a regulator or a block in the inlet channel	More frequent inundation
2	Wetlands actually or potentially connected to the river above minimum regulated flow but at, or below, maximum regulated flow	More frequent inundation
3	Wetlands above maximum regulated flow, filled only with surplus flows	Less frequent inundation
4	Wetlands above maximum regulated flow and which receive water (often saline) from adjacent irrigated areas via drainage, runoff or seepage or, in a few cases, effluent water from various sources	Variable, but with declining water quality

provides an excellent summary of the current hydrological regimes experienced by many of the Murray–Darling Basin wetlands, and of the way in which these regimes differ from pre-regulation conditions (Table I).

A range of changes in wetland ecosystems has been attributed to these alterations in hydrology. For example, it has been suggested that a combination of water extraction, drainage and levee construction has resulted in a 75% reduction in total wetland area within the Gwydir River valley (Keyte, 1992; Wettin *et al.*, 1994) (Figure 1). There is also evidence that the composition, diversity and abundance of wetland plant assemblages have been altered by changes in river hydrological regimes (Chesterfield, 1986; Beovich and Lloyd, 1992; Beovich *et al.*, 1992; Bren, 1992; Ward *et al.*, 1994; Lugg and Lloyd, 1995). Unfortunately, much of the evidence for an effect of river regulation on wetland ecosystems is anecdotal, and few systematic surveys have been carried out. Nevertheless, the weight of evidence presents a compelling case to suggest that river regulation and irrigation have had dramatic and widespread impacts on wetland ecosystems within the Murray–Darling Basin. In light of this, the need for a formal allocation of water for ecological management purposes is clear.

EWAs

The earliest formal EWA within the Murray–Darling Basin was for the Kerang Lakes in 1968. This allocation was made in an effort to maintain habitat for avifauna (principally ducks) following modifications to Pyramid Creek for irrigation supply (Wettin *et al.*, 1994). Elsewhere in the basin, water allocations have followed the trend set at Kerang and have been used to support bird populations. Specifically, stored water is released to prolong inundation of wetlands where bird-breeding events have been initiated in response to natural floods. Allocations have been used in this manner for the Macquarie Marshes (Wettin *et al.*, 1994; DLWC, 1997), the Barmah-Millewa Forest (K. Ward, personal communication, 1997) and the Booligal wetlands (Wettin *et al.*, 1994) (Figure 1).

Water management plans have been or are being developed for most of the regulated rivers in the Murray–Darling Basin; however, it may be some time before these plans are truly operational. Current plans tend to have a broader focus than their fowl-oriented predecessors, and may be characterized as having the general aim of mimicking the natural flooding regime experienced prior to regulation. Clearly this is a difficult task, being limited by both the amount of water available after irrigators' demands have been met, and by the need to minimize property damage, which may result from flooding.

In some areas, notably the Kerang Lakes (see Figure 1), EWAs are also used to flush wetlands in an attempt to combat the problem of rising surface and groundwater salinity (Kelly, 1997a,b).

INDICATORS OF ECOSYSTEM CHANGE FOR WETLANDS OF THE MURRAY–DARLING BASIN

The term 'ecological indicator' is used in this paper to refer to any of a range of response variables (be they physical, chemical or biological) that may be used in a monitoring programme designed to detect ecological changes. Ecological indicators vary in their capacity to register short- and long-term change in ecosystem structure or function. Parameters that change relatively slowly through time, or exhibit high year-to-year variability, are more suited to long-term monitoring, with sampling occurring annually or biannually (e.g. birds, fringing trees). Conversely, variables that change rapidly through time are suited only to very short-term monitoring programmes (e.g. zooplankton); high short-term variability necessitates a high frequency of sampling, which introduces logistical constraints for long-term monitoring. The assessment of short-term change in wetland health may require monitoring programmes which are separate from programmes that aim to assess change over a longer time frame, and which use different sets of indicators. To assist decisions regarding the selection of indicators appropriate for different project aims, we offer recommendations on the utility of various indicators for detecting short- and long-term changes in wetland health.

In order to be deemed a suitable indicator, a response variable should be sensitive to the putative stressor. In the case of EWAs, this means changes in hydrological regime (both short-term responses to inundation and, more importantly, longer-term responses to changes in periodicity). In some instances (e.g. the Kerang Lakes), this also means changes in salinity. Moreover, in order to be of practical use to managers, the suite of indicators used in a monitoring programme should provide information relevant to management goals. These goals may be specific (e.g. to reduce salinity); however, in most cases there is also the broader goal of maintaining or improving the 'health' of wetland ecosystems. The term 'ecological health', widely used with regard to ecosystem management, is difficult to define, but most definitions stress the need to make comparisons with ecosystems that are relatively free from human influence (e.g. Karr and Dudley, 1981; Schaeffer *et al.*, 1988; Adamus, 1992; Wright *et al.*, 1994).

Accordingly, the remainder of this paper examines potential physical, chemical and biological indicators, with reference to their sensitivity to changes to hydrology and salinity and their value as indicators of ecosystem health. Details are provided only for variables whose potential as indicators is highly regarded. Notable exceptions include turbidity, pH, dissolved oxygen, nutrients and ionic composition; the spatial and temporal variability of these parameters is so high that they are unlikely to provide useful information regarding changes in wetland health.

Physical and chemical parameters: (i) water depth and surface area

Water depth and area of inundation, though directly linked, influence wetland ecosystems by different mechanisms. Changes in depth affect light attenuation, temperature and water chemistry. Changes in wetland area affect factors such as the availability of water, nutrients and organic matter, soil properties in boundary areas, and connectivity between habitats.

Before establishing the effects of EWAs and hydrological management, changes in wetland depth and area, in response to inundation, must be quantified. The measurement of depth and surface area of wetlands subject to EWAs will allow managers to refine calculations that relate river flow volumes and wetland flooding, simplifying decisions regarding the volume of water required to inundate wetlands.

Where surface water is absent, measurement of soil moisture content should be made. This is particularly important for rooted aquatic macrophytes, which are affected not only by the depth of inundation above the soil surface, but also by the depth below (Froend *et al.*, 1993).

Physical and chemical parameters: (ii) salinity

The salinity of the water column and substratum has an important influence over the community composition and functioning of wetland ecosystems (Bayly and Williams, 1973; Hart *et al.*, 1990). Not only does ionic concentration affect a range of chemical and physical processes (Bayly and Williams,

1973), but the distributions of wetland organisms are frequently determined, in part, by their level of tolerance to salinity (Hart *et al.*, 1990; Anderson, 1991).

The salinity of surface waters in wetlands within the Murray–Darling Basin ranges from very dilute to hypersaline, in areas such as the Kerang Lakes (Table II, Figure 2). The ecological importance of salinity in wetlands of the Murray–Darling Basin is highlighted by the fact that, as noted previously, EWAs are being used in the Kerang Lakes area with the specific aim of addressing problems of increased salinity

Table II. Conductivity measurements for selected wetlands within the Murray–Darling Basin

	Upper River Murray billabongs	Middle River Murray billabongs	Kiewa River billabong	Ovens River billabongs	Goulburn River billabongs	Barmah- Millewa Forest wetlands	Kerang Lakes
	46	61	319	54	89	75	1800
	51	70	—	75	160	75	3000
	52	71	—	79	204	92	4000
	93	76	—	80	208	94	4000
	131	78	—	90	285	94	5000
	165	78	—	96	312	103	9400
	185	83	—	110	362	134	20 000
	187	85	—	327	384	137	35 000
	220	85	—	—	398	—	36 000
	236	92	—	—	—	—	60 000
	246	95	—	—	—	—	75 000
	—	98	—	—	—	—	75 000
	—	103	—	—	—	—	90 000
	—	104	—	—	—	—	102 000
	—	119	—	—	—	—	103 000
	—	121	—	—	—	—	122 000
	—	126	—	—	—	—	135 000
	—	127	—	—	—	—	176 000
	—	132	—	—	—	—	—
	—	230	—	—	—	—	—
	—	266	—	—	—	—	—
	—	355	—	—	—	—	—
Minimum	46	61	—	54	89	75	1800
Maximum	246	355	—	327	398	137	176 000
Mean	147	121	—	114	267	101	58 678
Median	165	97	319	85	285	94	48 000

All figures expressed as $\mu\text{S/m}$; based on data from KLA WG (1992), Ogden (1996) and Reid (1997).

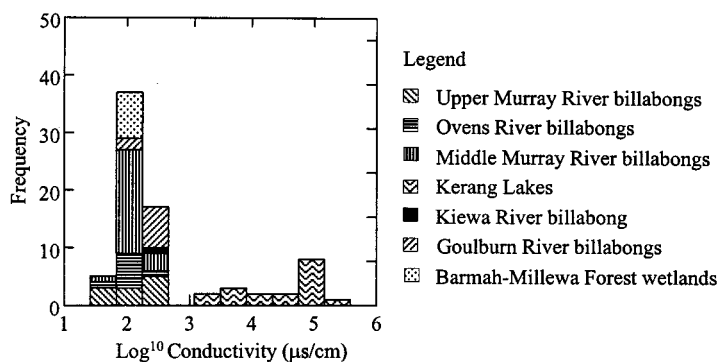


Figure 2. Frequency histogram for mean conductivity at selected wetlands within the Murray–Darling Basin (all figures expressed as $\mu\text{S/cm}$; based on data from KLA WG, 1992; Ogden, 1996; Reid, 1997)

(KLA WG, 1992; Kelly, 1997b). These problems have resulted from local and regional rises in saline groundwater tables, which have been linked to irrigation and the clearance of deep-rooted native vegetation in recharge areas. Increasing salinity has been blamed for loss of species, reduced biodiversity and declining agricultural yields (KLA WG, 1992).

The ionic concentrations of wetland water and sediments are likely to be directly influenced by the frequency of flushing and the ionic concentration of incoming water. In view of this, 'flushing flows' are one of the most common types of EWA selected for wetland regions affected by high salinity (Kelly, 1997a,b). Naturally, where the aim of water management is to control salinity, changes in salinity must be closely monitored. However, the ease and economy of salinity measurement, coupled with the concern that salinity-related problems may become more extensive in the Murray–Darling Basin, contribute to our recommendation that salinity be monitored routinely for all wetlands.

Biological parameters: (i) aquatic macrophytes

Aquatic macrophytes affect the structure and function of freshwater ecosystems in a variety of ways (Carpenter and Lodge, 1986). In brief, rooted macrophyte species link the sediments with the overlying water and provide an important conduit for the transfer of oxygen to the sediments during growth, and nutrients to the water column during decay (Carpenter, 1980, 1980; Carpenter and Lodge, 1986; Barko *et al.*, 1991; Boon and Sorrell, 1991). Macrophytes also provide substrata on which epiphytic algae and bacteria can grow. Indeed, macrophytes do not simply provide a surface for colonization, but also secrete dissolved organic compounds, which can be consumed by heterotrophic bacteria (Carpenter and Lodge, 1986). It has also been demonstrated that some forms of algae derive a large portion of their phosphorus (up to 60%) from the host macrophyte (Moeller *et al.*, 1988). In turn, epiphytes provide food for a wide range of macroinvertebrate and fish species (Cattaneo, 1983; Underwood and Thomas, 1990; Underwood *et al.*, 1992; Bunn and Boon, 1993; Rankin, 1996).

Macrophytes contribute greatly to the structural diversity of wetland environments. In doing so, they facilitate niche differentiation (Boon *et al.*, 1990; Boulton and Lloyd, 1991), and provide important refuge areas for insect larvae and small fish (Keast, 1984; Suren and Lake, 1988; Lillie and Budd, 1992). Annual cycles of growth and dieback of macrophytes also have important consequences for the aquatic system. The decay of large beds of macrophytes releases carbon and other nutrients into the water column and sediments, which can lead to dramatic changes in chemical parameters such as dissolved oxygen concentration and pH (Carpenter and Lodge, 1986; Hillman, 1986). Partially decayed macrophyte beds also contribute to lake sedimentation, which can have important consequences for plant succession (Carpenter, 1981; Wetzel, 1983). Changes in macrophyte assemblages, including removal or growth of macrophyte biomass or changes in plant assemblage composition, could potentially affect all of these processes (Carpenter and Lodge, 1986; Lodge *et al.*, 1988).

Although a large number of factors and interactions influence wetland plant assemblages, hydrological regime is the most important abiotic factor determining their distribution and composition (Froend *et al.*, 1993; Brock and Casanova, 1997). Aquatic macrophytes live in the ecotone between the terrestrial and aquatic environments. Their sensitivity to the hydrological regime is best demonstrated by the existence of distinct vegetation zones across this ecotone (Brock, 1986; Froend *et al.*, 1993; Mitsch and Gosselink, 1993).

On a smaller scale, the hydrological sensitivity of macrophytes is demonstrated by the response of individual taxa and assemblage composition to hydrological change. Variations in hydrology may lead to an overall reduction in the spatial extent of aquatic macrophyte assemblages (both at the terrestrial and at the aquatic ends of the scale), or to dramatic shifts in assemblage composition. Changes in the macrophyte assemblages of the Barmah-Millewa Forest may provide examples of these kinds of responses (cf. Chesterfield, 1986; Beovich and Lloyd, 1992; Beovich *et al.*, 1992; Bren, 1992; Leslie and Lugg, 1994; Ward *et al.*, 1994; Lugg and Lloyd, 1995), although conclusions from these studies are based on correlative data only. Experimental evidence for the sensitivity of macrophytes to hydrological change was provided by Nielson and Chick (1997), who showed that characteristics of the macrophyte assemblages

which developed in artificial 'billabongs' were strongly determined by the imposed hydrological regime (effects of permanent, spring and summer flooding were compared).

Wetland hydrology is also thought to affect the productivity of macrophyte assemblages. Briggs and Maher (1985) found that the biomass generated by macrophytes was generally greater in areas that dried in the previous season than in those areas that had been continually inundated over several years. They attributed this increased productivity to the release of nutrients following aerobic decay of accumulated organic material.

Although the hydrological sensitivity of aquatic macrophytes makes them ideal indicators for monitoring change in response to EWAs, high spatial and temporal variability in plant assemblage structure and composition may reduce the indicator capabilities of these plants. Variation in the patterns of distribution of different species may cause further problems. While some plant species may be sparsely but evenly distributed, others may be distributed in patches, the sizes of which vary according to species. This means that a sampling regime that provides an adequate survey for one species may not do so for several other important taxa.

Other factors that may interfere with the indicator potential of aquatic macrophytes include pre-existing differences in macrophyte communities, grazing pressure from cattle, sediment type (Froend *et al.*, 1993; Spencer *et al.*, 1998) and the presence of carp (*Cyprinus carpio*), which contribute to high turbidity, high nutrient concentrations and the loss of submerged macrophytes (Roberts *et al.*, 1995; King *et al.*, 1997; Robertson *et al.*, 1997). The inertia of established macrophyte assemblages might also confound results. Several researchers have noted that the actual distribution of macrophytes across hydrological gradients rarely reflects their full physiological range (Froend *et al.*, 1993). This phenomenon can result from factors such as interspecific competition, as well as stochastic events associated with seed production, germination and establishment of plants on bare areas (Rea and Ganf, 1994). Once a particular species becomes established in an area, it is not uncommon for monospecific or near monospecific stands to develop as a result of competitive exclusion (Froend *et al.*, 1993). Under these circumstances, dramatic changes in hydrology may not lead to equivalent changes in assemblage structure, simply because the dominant species is able to tolerate a wide hydrological range and prevent the growth of species that are better adapted to the new hydrological regime.

In light of these potentially confounding factors, we propose that a range of variables be monitored. Data on species presence and abundance will provide information on the spatial distribution of taxa, temporal changes in composition and distribution, species associations, the ratio of introduced to native species, the ratio of open water to areas with macrophyte cover, and the ratio of emergent to submerged macrophytes within each wetland. Macrophyte biomass and growth measures should also be included when monitoring. Changes in plant vigour may occur sooner than changes in composition; measures of plant biomass and growth rates may increase the chances of detecting short-term change in response to EWAs.

Other variables that should be recorded include grazing intensity, the presence of carp, and macrophyte seed-bank viability. As noted, each of these variables has the potential to confound the response of the macrophyte assemblage to altered hydrology.

Biological parameters: (ii) aquatic macroinvertebrates

Macroinvertebrates have been shown to be effective indicators of ecosystem health in stream environments (Cranston *et al.*, 1996), and their use as indicators in wetlands has been advocated (Davis *et al.*, 1987, 1993; Hicks, 1997). They are important primary and secondary consumers within wetland ecosystems, occupying a range of niches, including those of grazers (of both macrophytes and biofilms growing on macrophytes and other substrata), detritivores, filter feeders and predators. They also provide the principal food source for many vertebrates, such as fish, birds and amphibians (Boon *et al.*, 1990; Bunn and Boon, 1993; EPA NSW, 1994).

Spatial and temporal variation in macroinvertebrate assemblage characteristics are influenced by a wide range of biological, chemical and physical variables, including the hydrological regime. Several studies

within the Murray–Darling Basin have shown that lotic and lentic environments each have distinct macroinvertebrate communities, and that species richness is generally higher in lentic environments (Boulton and Lloyd, 1991; Hillman and Shiel, 1991; Bray, 1994). Distinctions are also apparent between the macroinvertebrate assemblages of lentic water bodies subject to different hydrological regimes. Boulton and Lloyd (1991), Davis *et al.* (1993) and Balla and Davis (1995) all found that temporary wetlands generally have a more diverse macroinvertebrate fauna than nearby permanent waterbodies; these differences have been found to persist for several months after a dry wetland is filled (Hillman and Quinn, *in press*).

Despite the apparent potential of macroinvertebrates as indicators of change in response to EWAs, the amount of spatial and temporal variability in wetland macroinvertebrate assemblages, together with the high level of taxonomic expertise required to process samples, will reduce the cost-effectiveness of using macroinvertebrates as indicators. Further, the abundance and diversity of wetland macroinvertebrate assemblages are strongly influenced by the composition and distribution of aquatic macrophytes, upon which some invertebrates depend for food and shelter (Boulton and Lloyd, 1991; Downing, 1991; Bray, 1994; Balla and Davis, 1995). Therefore, there is likely to be a degree of redundancy in the information provided by macroinvertebrate surveys (that is, patterns of spatial and temporal distribution in macroinvertebrates may reflect similar patterns detected in macrophyte assemblages). Finally, our knowledge of the relationships between wetland macroinvertebrate assemblages and hydrology is relatively poor, and further study is required before causal links between pattern and process can be confirmed.

In light of these concerns, macroinvertebrates are only provisionally recommended as indicators. The major cost involved in monitoring changes in macroinvertebrate assemblages lies not in the sampling but in the processing of samples; therefore, we suggest that samples be collected in current monitoring programmes, and stored for processing at a later date if required. Planned and ongoing studies are examining sampling protocols and developing techniques (including rapid bioassessment techniques) for using wetland macroinvertebrate communities for monitoring purposes (Norris and Thoms, 1999). Results from these studies should prove invaluable when monitoring groups are deciding whether or not to process stored macroinvertebrate samples. Further, decisions regarding the selection of appropriate variables for analysis should be based on the results of these studies; our understanding of the links between macroinvertebrate assemblage structure, hydrology and wetland health will be greatly improved by the information they provide.

Biological parameters: (iii) biofilms

Biofilms, which consist of a mixture of algae, bacteria, protozoa, fungi and detritus, characteristically form on the surfaces of submerged objects, such as woody debris, stones, silt and macrophytes. They are increasingly being seen as an important food source for a range of aquatic animals (Bunn and Boon, 1993; Sheldon and Walker, 1993). The algal component of biofilms is widely used in the monitoring of rivers and streams in the Northern Hemisphere, mostly as indicators of trophic status and organic pollution (Descy, 1979; Watanabe *et al.*, 1988; Round, 1991, 1993). They have been less widely used in Australia (Reid *et al.*, 1995), although several recent studies have used benthic diatoms as indicators, particularly in urban streams (Fluin, 1995; Reid, 1996).

Biofilms are affected by a range of hydrological factors. These include depth (which affects light availability and hence primary production), frequency and duration of aerial exposure, and flow velocity (high flows can limit biofilm development and cause sloughing; Burns *et al.*, 1994). Each of these factors is affected by the imposition of regulated flow regimes, leading several researchers to suggest that the quality of biofilms in the Murray–Darling Basin has altered since regulation (Sheldon and Walker, 1993; Burns *et al.*, 1994; Walker *et al.*, 1995). In particular, it has been argued that these conditions allow biofilm communities to develop more fully, leading to a predominance of later successional stages (Walker *et al.*, 1995) and dominance by filamentous green algae (Sheldon, 1994).

Biofilms may prove to be important indicators of changing hydrology; however, our understanding of their relationship with wetland health is relatively poor, and, therefore, further studies are required before

they can be fully recommended for use in monitoring change in response to EWAs within wetlands of the Murray–Darling Basin.

Biological parameters: (iv) zooplankton

Zooplankton common to wetlands in the Murray–Darling Basin include a range of micro-crustaceans (cladocerans, copepods and ostracods), rotifers and the juvenile forms of many species of macroinvertebrates. These organisms occur in open water or in association with macrophyte beds in the littoral zone, and feed principally on phytoplankton, bacteria, detritus, higher plants and other zooplankton (Shiel, 1990). In doing so, zooplankton play an important role in regulating algal biomass and, more generally, nutrient pathways. Zooplankton provide a major food source for macroinvertebrates, fish and waterfowl, and can be particularly abundant in temporary wetlands immediately following inundation, because of the ability of many forms to persist in a resting stage in the dry sediment and to emerge rapidly following inundation (Boulton and Lloyd, 1992; Jenkins and Briggs, 1997).

Studies of the relationships between hydrology and zooplankton abundance and diversity have centred on the capacity of zooplankton to emerge from wetland sediments (Boulton and Lloyd, 1992; Jenkins and Briggs, 1997). In a study based on the Chowilla floodplain on the lower River Murray, Boulton and Lloyd (1992) found that the abundance and diversity of the emergent zooplankton assemblage was greater for sediments taken from more frequently flooded areas than from those flooded more rarely. Jenkins and Briggs (1997) found a similar pattern in the number of invertebrates emerging from the sediments of wetlands on the Darling River. In this case, the pattern was largely a result of the smaller number of rotifers emerging from infrequently flooded sediments. In contrast, a study of mid-Murray wetlands found that the diversity of zooplankton emerging from sediments taken from an area that had been continuously flooded for the previous 15 years was lower than from sediments taken from a shallow lake that underwent regular cycles of flooding and drying (Cole and Fenton, 1995).

Notably, the life cycles of zooplankton are relatively short, extending over a period of days to months (Shiel, 1995); therefore, assemblage characteristics are highly variable and require a high frequency of sampling. Although such a high level of variability may limit the usefulness of zooplankton as indicators of change in response to EWAs, it may be possible to monitor the abundance and diversity of emerging invertebrates as an indicator of wetland health. This analysis could be incorporated into any seed-bank studies initiated as part of the macrophyte surveys. Further studies are required before such a technique can be recommended for widespread application.

Biological parameters: (v) fish

Fish are widely used as biological indicators in inland waters, particularly streams (e.g. Karr, 1981, 1991; Anderson, 1991; Bishop *et al.*, 1995; Harris, 1995; Paller *et al.*, 1996). The value of fish as indicators stems from their position in the trophic structure of aquatic environments, and from their demonstrated response to a variety of environmental factors, extending from water chemistry to complex structural features, across a range of spatial scales (Harris, 1995). Importantly, the relatively long life cycles of fishes means that they integrate conditions over time-scales that are relevant to hydrological management (Harris, 1995).

There is ample evidence that fish communities have been dramatically affected by the suite of human-induced changes experienced by the Murray–Darling Basin over the last two centuries. Several formerly abundant species of native fish are now rare or threatened, and most have undergone severe restrictions in range (Cadwallader and Lawrence, 1990; Walker and Thoms, 1993; Harris and Gehrke, 1997). In the lower reaches of rivers such as the Murray, Murrumbidgee and Lachlan, the dominance of introduced carp (*Cyprinus carpio*) is such that they made up more than half the total number of fish caught in these rivers during a recent NSW fish survey (Gehrke *et al.*, 1995; Gehrke, 1997a,b; Harris and Gehrke, 1997).

River regulation appears to have been an important factor contributing to the expansion of the range of exotic species and the concurrent decline of native fish species. In the upper reaches of major rivers

such as the Mitta Mitta River and the Goulburn River, for example, cold water discharges from large dams have lowered spring and summer water temperatures to the point that native fish no longer breed (Cadwallader and Lawrence, 1990; Koehn *et al.*, 1997). These dams, along with weirs in downstream areas, are also believed to act as barriers to the migration of native species (Cadwallader and Lawrence, 1990; Jackson and Jenkins, 1997).

It has also been argued that changes in the frequency and timing of floods may have contributed to the decline in native fish populations by affecting the dispersal of fish from main river channels to floodplain wetlands; reduced flood frequency reduces fish access to the food and habitat resources provided by floodplain wetlands. Further, fish-breeding success is likely to depend to some extent on the synchronicity between flooding and food availability in floodplain habitats; changes in the timing of floods have the potential to affect this. Notably, while native fish appear to commence migratory behaviour in response to rising water levels, carp appear to commence migration in response to rising water temperatures (Mallen-Cooper *et al.*, 1997). The behavioural response of carp would seem to be better suited to the summer flooding regime that prevails in many regulated river reaches.

Assuming that regulation of river flow has contributed to declining native fish numbers, attempts to introduce more 'natural' hydrological regimes should favour native fish. This expectation, along with the suggestion that fish are good indicators of ecological health (Karr, 1981), could be tested by including fish in programmes that monitor the effectiveness of hydrological management measures. However, to conduct full surveys of wetland fish assemblages would be expensive and time-consuming, and would, in all likelihood, require specialist field staff. This would necessitate the allocation of resources over and above those that are usually available.

We recommend, however, that fish should be included when monitoring certain types of wetlands. For example, in wetlands where regulators have been installed to restrict the inflow of summer irrigation flows and rain-rejection floods, fish surveys may be necessary to gauge whether the movement of fish between river and wetland is being restricted (e.g. Moira Lake, Croppers Lagoon; Leslie and Lugg, 1994; Lloyd, 1996).

Biological parameters: (vi) birds

As noted previously, EWAs had their genesis in attempts to promote bird-breeding in wetlands affected by river regulation. This fact, along with strong community concerns regarding avifauna, ensures that the success of hydrological management will be partly gauged by the abundance and diversity of birds living and breeding in wetlands.

It is reasonable to assume that, in the long-term at least, abundant and diverse native bird populations are indicative of wetland health; birds require both sufficient food resources and nesting habitat, which generally equate to a healthy wetland (Cable *et al.*, 1989; Weller, 1996). Techniques for surveying and monitoring bird populations in wetlands are well-established (Caughley *et al.*, 1976; Jaensch *et al.*, 1988; Harper, 1990; Weller, 1996; Kingsford *et al.*, 1997), and there is a good deal of baseline information regarding the distribution and abundance of species within the Murray–Darling Basin (Kingsford *et al.*, 1997).

There is, however, some doubt over the sensitivity of birds to hydrological change in the short-term. The great mobility of birds means that they are able to exploit resources from both local and distant water bodies. The presence or absence of birds within a given wetland may, therefore, depend upon conditions hundreds of kilometres away, rather than the condition of the wetland under consideration. Moreover, birds possess a high degree of behavioural complexity, which may further confound responses to changing hydrological conditions. For example, there is concern among managers that successive breeding failures within a wetland, as a result of shortened flood duration, may cause birds to shun that wetland during subsequent floods, even if hydrological management has since ensured that inundation occurs for a period sufficient for successful breeding (J. Reed, personal communication). In short, it is unlikely that birds will be useful indicators of short-term changes in wetland ecosystems in response to hydrological management; however, they may prove to be valuable indicators of wetland ecosystem changes over longer periods.

Biological parameters: (vii) mammals

Few species of mammals depend on the inland wetlands of NSW and Victoria, and only two species, the platypus (*Ornithorhynchus anatinus*) and the eastern water rat (*Hydromys chrysogaster*), are truly amphibious. Platypuses are restricted to fresh water, inhabiting both streams and lentic wetlands in eastern mainland Australia and Tasmania. In NSW, they are common in the tributaries of the Darling, Lachlan, Murrumbidgee and Murray Rivers (Grant, 1992). Water rats are widely distributed throughout Australia, inhabiting many inland water systems (including both fresh and saline wetlands), estuaries and marine beaches (Woollard *et al.*, 1978).

Knowledge of the ecological requirements of platypus and water rats in wetland ecosystems is relatively limited. While their distributions provide insight into their relative salinity tolerances (Hart *et al.*, 1991), little is known of their responses to altered hydrological regimes. Similarly, virtually nothing is known of the ecological effects of platypuses and water rats within southeastern Australia's wetlands; however, given the opportunistic feeding habits of *H. chrysogaster* (Woollard *et al.*, 1978), high densities of this species could have important consequences for ecosystem structure. In light of available knowledge, it would be inappropriate to recommend the use of these species as key indicators; however, their potential significance in future programmes should not be discounted.

Biological parameters: (viii) reptiles and amphibians

Of the many reptiles associated with inland wetlands in NSW and Victoria, only freshwater turtles are considered here, because they are the group most likely to be affected by changes in salinity or hydrology.

The eastern long-necked turtle (*Chelodinia longicollis*) is the species most commonly found in inland wetlands. Other species common to southeastern Australia (*C. expansa* and *Emydura macquarii*) are generally restricted to permanent lakes and streams (Chessman, 1978). The identification of *C. longicollis* from saline localities suggests a reasonable tolerance of high salinity levels (Hart *et al.*, 1991), a suggestion supported by the possible identification of lachrymal glands in *C. longicollis* (Rogers, 1966). However, further research is required to confirm these results.

Chessman (1978) found evidence for relatively high levels of terrestrial activity and aestivation by *C. longicollis*. The ability of this species to conserve water is also high relative to *C. expansa* and *E. macquarii* (Chessman, 1984). Both aestivation and water conservation are likely to be important to the ability of *C. longicollis* to inhabit remote temporary waters and migrate over land. These features may make *C. longicollis* a useful indicator of extreme changes in wetland hydrology; however, subtle changes in hydrology are unlikely to be easily detected using this species.

Amphibians are relatively good indicators of water quality (Cranston *et al.*, 1996). Characteristics such as breeding success are often quite sensitive to changes in abiotic variables, such as toxic metal concentrations, pH, alkalinity, cation concentrations and dissolved organic carbon levels (Horne and Dunson, 1995a,b,c). Effects of hydrological regimes on amphibian distributions and breeding success are less well known, although evidence from recent studies suggests sensitivity by a number of amphibians to a variety of hydrological characteristics (Joly and Morand, 1994; Richter and Azous, 1995; Rowe and Dunson, 1995). For example, Richter and Azous (1995) showed that fluctuations in the mean water level of wetlands affected amphibian species richness. Interestingly, though, numbers of lentic breeding pairs and total species richness showed little variation between permanent and semi-permanent wetland sites.

Knowledge of the response of amphibians to varying salinity levels is based purely on information from overseas studies. The salinity tolerances of Australian tadpoles and adult frogs are unknown (Hart *et al.*, 1991). Overseas studies suggest that amphibians are rarely good regulators of salt, with many adult frogs being completely intolerant of salinity (Hart *et al.*, 1991). It is likely, therefore, that Australian frogs are also relatively salt-intolerant. This characteristic may make them suitable indicators of increasing salinity levels within wetlands.

The significance of freshwater turtles and frogs to the structure and function of wetland ecosystems has yet to be properly established. Spatial and temporal variability of these species may restrict their usefulness as indicators; however, further investigation is required before they are discarded as possible parameters for monitoring.

Biological parameters: (ix) fringing vegetation

Fringing vegetation of the Murray–Darling Basin wetlands is dominated by the river red gum, *Eucalyptus camaldulensis*. Other tree species found co-occurring with red gum include black box (*E. largiflorens*), silver wattle (*Acacia dealbata*), river bottlebrush (*Callistemon paludosus*), alpine bottlebrush (*C. sieberi*), river cooba (*A. stenophylla*), coolibah (*E. microtheca*) and prickly bottlebrush (*C. brachyandrus*) (Costermans, 1989; ANCA, 1996; J. Frankenberg, personal communication).

River red gum forests typically form areas of open woodland in regions receiving frequent flooding. Where flooding is less frequent, black box woodlands are usually more common (Smith and Smith, 1990; ANCA, 1996). The understoreys of most red gum forests contain a variety of grasses. In highly flood-prone areas, red gum is more sparsely distributed, and extensive grasslands containing moira grass (*Pseudoraphis spinescens*) and a variety of reed species predominate (Smith and Smith, 1990).

Fringing vegetation communities affect the health of associated wetlands in a variety of ways. First, the inundation of floodplain soils, grasses and accumulated tree debris is thought to provide a pulse of nutrients to floodplain wetlands when floodwaters are rising (Junk *et al.*, 1989; Ward, 1989a,b). Patterns of nutrient cycling within wetlands are likely to be altered, at least to some extent, by the influx of this nutrient pulse. Second, direct litter input from trees and shrubs provides an important source of allochthonous organic material within wetlands. The decomposition of this material, including leaves, sticks, bark and logs, is likely to have considerable influence on patterns of nutrient cycling within wetlands. Third, both leaves and wood provide an important food source for wetland detritivores (mainly aquatic invertebrates; Mitsch and Gosselink, 1993). Fourth, accumulated wood debris in wetlands also provides important habitat for fish and invertebrates (Boulton and Lloyd, 1991; Walker, 1992). Fifth, growth of epiphytes is often prolific on the surfaces of leaves and wood (personal observation). Lastly, fringing trees are an important nesting site for water birds (Briggs and Thornton, 1997), and form an important link between wetlands and groundwater (Carter and Novitzki, 1988).

Effects of hydrological regimes on the fringing zone of lentic wetlands have been examined for only a few Australian vegetation assemblages (e.g. Bren and Gibbs, 1986; Bren, 1992; Denton and Ganf, 1994; Bacon, 1996). Most studies have focused on vegetation associated with river floodplains, and the majority of these have focused on the response of river red gum woodlands to different wetting–drying cycles. These studies have been based on a combination of interpretation of historical records and maps (e.g. Bren and Gibbs, 1986; Dexter *et al.*, 1986; Bren, 1992), and a small amount of experimental work, much of which remains unpublished (exceptions include Dexter, 1970; Bacon, 1996).

Results from this research suggest a relatively close association between the health of mature trees, the establishment of red gum seedlings, and the duration, frequency and timing of flooding. As a consequence, the extent and overall health of entire red gum forests has also been assumed to be related to flood duration, frequency and timing (Bren and Gibbs, 1986; Dexter *et al.*, 1986; Bren, 1992). Notably, conclusions drawn from studies relying on historical and photographic records have involved a considerable amount of speculation (a point acknowledged by Bren, 1992). Nevertheless, these studies provide at least some insight into the relationship between hydrology and red gum forests.

The effects of salinity on fringing vegetation are better known than the effects of hydrological change, although there is still much to learn. Hart *et al.* (1990, 1991) provided extensive reviews of the current knowledge of this topic, including details of the physiological causes of salt sensitivity (toxicity versus water deficit), and information on the salt sensitivity of a variety of non-halophytic Australian tree species (i.e. species that achieve best growth in non-saline conditions, and whose growth is reduced as salinity increases (Hart *et al.*, 1991).

In light of available evidence, future use of fringing vegetation communities as indicators of an effect of EWAs on wetland health may be possible. Because of the life history of fringing tree species, their usefulness as indicators will be restricted to long-term monitoring programmes; however, other fringing plant species (e.g. grasses, sedges and reeds) may be more responsive over a short time period, enabling their use in short-term programmes.

SUMMARY

Monitoring and scientifically rigorous adaptive management practices are the key to the long-term success of EWAs, which are designed to partially restore the natural hydrological regime to floodplain wetlands in the Murray–Darling Basin. Successful monitoring relies on the well-informed selection of a variety of hydrologically sensitive indicators. Physical, chemical and biological indicators that appear suitable for use in wetland monitoring include wetland depth, wetland area and salinity, aquatic macrophytes and associated assemblage characteristics and variables, and aquatic macroinvertebrates. Current knowledge is limited concerning the relationships between wetland health and other potential indicators, including biofilms, zooplankton, birds, fish, mammals, reptiles, amphibians and fringing vegetation.

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