Development and testing of a rapid appraisal wetland condition index in south-eastern Australia

C. Spencer†, A. I. Robertson‡ and A. Curtis†

The high costs and lengthy time commitments associated with traditional monitoring, and the remoteness of many wetlands have necessitated the development of techniques for rapid wetland appraisal. A rapid appraisal condition index based on four attributes of wetlands, soil, fringing vegetation, aquatic vegetation and water quality, was developed for assessing the health of permanent floodplain wetlands in the Murray-Darling Basin of south-eastern Australia. The index, composed of 13 indicators related to wetland function, was tested in the field for scientific validity relative to an independent long-term monitoring data set, replicability of indicator scores by different investigators and the responses to the seasonality in wetland processes. Indicator values were based on a mixture of visual estimates and measurements using simple instruments or procedures and all data could be collected in the field in less than 3 h. There was a significant positive correlation between rankings of the condition of 10 wetlands based on an independent long-term monitoring data set and the wetland condition index. There were also highly significant positive correlations between indicator scores collected by different investigators. Indicator scores for physical factors and fringing vegetation did not differ between autumn and winter, but winter rainfall had a significant impact on aquatic vegetation and water quality indicators. The results indicate that the wetland condition index is a valuable and reliable tool for the rapid surveying of the condition of permanent floodplain wetlands.

Keywords: wetland condition index, ecosystem health, rapid assessment, indicators, validation.

Introduction

Wetlands are amongst the most degraded of ecosystems and losses have been estimated at 50% of the original global wetland area (Gardiner, 1994; Jones et al., 1995). Significant resources are currently focused on monitoring the status of the wetlands remaining in many countries (McKenzie et al., 1992; Clough, 1993; Henry and Amourou, 1995). Most of the large government or intergovernmental wetland monitoring projects such as the Environmental Monitoring and Assessment Program (EMAP)—wetlands, in the United States (Peterson, 1994; Novitzki, 1995) rely on scientists or management agency staff working within an organized project framework to collect ecological data. These data are then used via Government channels to influence decisions regarding wetland conservation policy (e.g. Anon, 1996, 1997). However, for a variety of reasons (e.g. Schrader-Frechette and McCoy, 1994; Christensen et al., 1996; Robertson, 1997) research and highly structured scientific monitoring efforts do not necessarily result in wetland conservation (Robertson, 1992, 1997). There are two important reasons for the lack of transfer of scientific and monitoring data into wetland conservation action. First, natural resource managers often do not have the resources to spend on detailed and lengthy ecosystem monitoring. Secondly, a large proportion of wetlands occur on private property, and despite Government policy, landowners do not have the simple and reliable monitoring tools available to assess the influence of their land-management actions on wetlands.

For these and other reasons rapid appraisal methods have been developed for assessing the condition of wetlands and other ecosystems relative to natural conditions (e.g. Morgan and Bray, 1982; Ainslie, 1994; Larson

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and Mazzarese, 1994). Most rapid assessment methods are conceptually similar, being based on indicators of important wetland function (e.g. Mitsch, 1992). However, little effort has been devoted to evaluating rapid assessment methodologies (Cable et al., 1989; WWF, 1992). The spatial variability in wetland ecosystems, the paucity of information on the natural temporal variance for wetland variables and the scarcity of unimpacted reference sites (Turner et al., 1995) all contribute to concerns about whether rapid appraisal methods can provide scientifically valid measures of wetland condition (e.g. Kantrud and Newton, 1996).

In south-eastern Australia, the floodplain wetlands of the Murray-Darling Basin have been severely degraded since European settlement by grazing and land clearing, and by significant river regulation and alterations to the frequency, timing and duration of hydrologic connections between river channels and floodplain habitats (Walker, 1985; Bacon et al., 1994). There has been significant progress made in Government policy on wetland conservation in the region (e.g. MDBMC, 1990; Sharley and Huggan, 1994; Anon, 1996, 1997). However, the wetlands occur over a vast area (10⁶ km²), precluding detailed ecological monitoring and while many of the largest wetlands are now listed as being of international importance and are actively conserved (ANCA, 1996), more than 90% of all wetlands occur on private property where directives regarding wetland conservation may have little value.

For these reasons we have developed a wetland condition index for permanent wetlands in the Murray-Darling Basin that is easy to use and is based on a rapid field survey approach. Rapid wetland survey methodologies have been used in Australia to identify wetlands of conservation value for particular taxa (e.g. Pressey, 1984) but have not been developed or tested as tools for assessing wetland condition.

Here the choice of ecological indicators used in the wetland condition index is discussed and the performance of the index evaluated in assessing wetland conditions relative to longer-term scientific data collected for a number of wetlands. The authors also assess the replicability of index scores by different investigators and the index responses to the seasonality in wetland processes.

Methodology

Definition of condition

Extension of the concept of health to ecosystems has not been without controversy (Okey, 1996), and there is a continuing debate regarding all aspects of the concept (e.g. Suter, 1993; Wicklum and Davies, 1995; Rapport et al., 1996). In the context of this study, condition refers to ecosystem health. This incorporates the stability and sustainability of the system to withstand environmental stress (Rapport, 1995). It also includes the capacity of the ecosystem to support a diverse community of organisms and perform functions comparable to that of a local unimpaired site (Karr and Dudley, 1981).

Indicators

Breckenridge et al. (1995) defined an indicator as any expression of the environment that estimates the condition of ecological resources, magnitude of stress, exposure of a biological component to stress, or the amount of change in a condition.

To be useful in rapid assessment, indicators should possess most of the following attributes: (1) show low natural temporal and spatial variability; (2) be highly responsive to condition change; (3) not be ambiguous in their interpretation; (4) be cost effective and simple to apply; (5) have regional applicability; (6) be biologically relevant, that is be an indicator of ecosystem condition; (7) be a simple or commonly measured parameter—reference data on healthy thresholds is more likely to be available for simple indicators; (8) be non-destructive of the ecosystem; and (9) be able to have results summarized so as to be understood by non-experts (Kent et al., 1992; Cairns et al., 1993; Breckenridge et al., 1995 Turner et al., 1995).

In order to choose appropriate indicators for floodplain wetlands in the Murray-Darling Basin, the authors focused on indicators of wetland function (Mitsch, 1992; Mitsch and Gosselink, 1993), including the ability of wetlands to act as habitats for terrestrial and aquatic fauna, as “filters” of materials in overland flows and as sources of primary...
Table 1. Justification of the indicators used in the present study and their relationship to floodplain wetland attributes and function in the south-east of the Murray-Darling Basin

<table>
<thead>
<tr>
<th>Attribute</th>
<th>Function(s)</th>
<th>Indicators</th>
<th>References</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil</td>
<td>Interception of overland flows; nutrient storage; supports growth of vegetation; habitat for fauna</td>
<td>Bank stability—unstable banks are detrimental to fringing vegetation and water quality</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Pugging by livestock—reflects damage to soil structure and chemistry</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>Soil organic content—reflects organic inputs to wetlands</td>
<td>Laperre (1964), van de Graaff (1988)</td>
</tr>
<tr>
<td>Fringing vegetation</td>
<td>Interception of overland flows; nutrient storage; habitat for fauna; carbon source for aquatic food-chains</td>
<td>Width—reflects the ability of fringing vegetation to act as a faunal habitat and in material filtering</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Continuity—as above</td>
<td>Campbells (1993), Catterall (1993), Quinn et al. (1993)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Height diversity—indicates habitat availability for fauna</td>
<td>As above</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Spatial heterogeneity—indicates habitat availability for aquatic fauna</td>
<td>As above and Calms et al. (1993), Peterson (1994), De Jalon et al. (1996)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Attached algae—important food source, indicative of trophic status of water bodies</td>
<td>Costa et al. (1992), Rosas et al. (1993), Goldsborough and Robinson (1996)</td>
</tr>
<tr>
<td>Water</td>
<td>Habitat for biota; medium for biogeochemical processes</td>
<td>Turbidity—high turbidities can reduce light penetration for autotrophic community</td>
<td>Mackay et al. (1988), Maltland (1990)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Conductivity—increased salinity will affect natural biotic species composition</td>
<td>Hart et al. (1991)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Colour—indicates input of organic matter, can reduce oxygen concentrations</td>
<td>Mackay et al. (1988), Detenbeck et al. (1996)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Algal bloom frequency—indicates eutrophication</td>
<td>Steinberg and Hartmann (1988)</td>
</tr>
</tbody>
</table>

production for floodplain river systems (Table 1).

The wetlands

All wetlands considered in the study occurred on the floodplains of four rivers in south-eastern Australia (Figure 1). In this region, westward flowing rivers fed by spring snow melt and local rainfall drain the Great Diving Range. All wetlands on the Kiewa, Ovens, Murray and Murrumbidgee floodplains were meander cutoffs or depressions, both of which are filled by overbank flows. Soils on the floodplains are black earths and minimal prairie soils developed on alluvial deposits dominated by fine sand, silt and clay. The dominant vegetation on the floodplains of all four rivers is river red gum (Eucalyptus camaldulensis), which formed extensive forests prior to thinning for European agriculture. Most native understorey vegetation in the sections of the rivers surveyed has now been replaced by introduced grasses (Smith...
and Smith, 1990) and the floodplains are now used to support beef cattle and sheep as well as some cropping (Crabb, 1997).

Measurement and the scoring of indicator values

The values obtained from field and laboratory measurements of each indicator were normalized by allocating scores based on: (1) published information, or (2) the authors' own observations, on relatively undisturbed reference wetlands and a range of damaged wetlands in the south-east of Australia. Scores were allocated on a scale from 0 to 4, and in such a way that highest scores reflected best conditions and lowest scores the most degraded condition (Table 2). Below field and laboratory measurement procedures have been outlined along with the basis for score allocation.

Soil attributes

Bank stability. Assessment of the degree of bank erosion was made following a walk around the complete wetland. Scores were allocated as follows (DCNR, 1995): 5 = Stable—stable banks protected by good vegetation cover; 4 = Good—some minor spot erosion occurring; 3 = Moderate—some erosion occurring, spot erosion linked, structural and vegetative damage occurring due to little
erosion; 2 = Poor — significant erosion occurring, little vegetation present; 1 = Unstable — extensive erosion occurring unchecked, bare banks, steep banks may be present due to erosion (Table 2).

Degree of pugging. Pugging of soil by livestock was measured as the mean of the number of cattle hoof marks in 10 1-m² quadrants placed randomly on the sediment at the water's edge. Through pugging, livestock cause soil compaction, erosion, lowered water infiltration rates and a reduction in the water storage capacity of floodplain soils (Bacon et al., 1994). Observations on a variety of wetlands throughout the Murray-Darling Basin (authors' unpub. data) indicated that severely degraded wetlands had more than 20 hoof marks per square metre (Table 2).

Organic content of soil. Soil samples were collected from five random sites at the land-water interface around the perimeter of each wetland. The mean percentage of organic matter (loss on ignition at 500 °C for 2 h) was then calculated for the five samples.

Wetland soil organic matter content will naturally vary with soil type, degree of flooding and input of primary production. With the exception of regions with peat accumulations, where organic contents of soil may be up to 75%, most floodplain soils will have organic matter contents greater than 5% (Mitch and Gosselink, 1993). More frequently flooded soils will have higher levels of organic matter (Leeper and Uren, 1993). Based on this, and observations in relatively pristine floodplain regions in the region of the study scores were allocated for organic matter content as outlined in Table 2.

Fringing vegetation

Width of fringing vegetation strip. The mean width of the native vegetation fringing was based on visual estimates of the riparian strip made at the four major compass points at each wetland. In the case of wetlands where sides differed in steepness, the maximum flood height in the wetland was used as a guide to determining the boundary between the wetland riparian strip and other floodplain flora. Recent reviews of riparian buffers for watercourses have indicated that there are no general minimum acceptable widths (e.g. Bron, 1993; Castelle et al., 1994; Spackman and Hughes, 1995). However, it appears

See text for discussion.

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Score</th>
<th>4</th>
<th>3</th>
<th>2</th>
<th>1</th>
<th>0</th>
</tr>
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<tbody>
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<td>Bank stability</td>
<td>Stable</td>
<td>0</td>
<td>1-6</td>
<td>7-12</td>
<td>13-19</td>
<td>&gt;20</td>
</tr>
<tr>
<td>Degree of pugging (mean pugs per m²)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Organic content of soil (% organic matter)</td>
<td>20-30</td>
<td>6-19</td>
<td>&lt;5 and &gt;30</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Width of fringing vegetation strip (mean width m)</td>
<td>&gt;30</td>
<td>8-30</td>
<td>3-8</td>
<td>0-5-3</td>
<td>&lt;0-5</td>
<td></td>
</tr>
<tr>
<td>Continuity of fringing vegetation (%)</td>
<td>&gt;90</td>
<td>66-90</td>
<td>36-65</td>
<td>10-35</td>
<td>&lt;10</td>
<td></td>
</tr>
<tr>
<td>Height diversity (%)</td>
<td>&gt;25</td>
<td>17-24</td>
<td>9-16</td>
<td>1-8</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td>Cover of aquatic vegetation (%)</td>
<td>&gt;76-95</td>
<td>76-95</td>
<td>76-95</td>
<td>76-95</td>
<td>76-95</td>
<td></td>
</tr>
<tr>
<td>Attached algae</td>
<td>Little</td>
<td>&gt;4</td>
<td>3</td>
<td>2</td>
<td>1</td>
<td>0</td>
</tr>
<tr>
<td>Spatial heterogeneity (no of layers present)</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
</tr>
<tr>
<td>Turbidity (NTU)</td>
<td>0-40</td>
<td>41-80</td>
<td>81-120</td>
<td>121-160</td>
<td>&gt;160</td>
<td></td>
</tr>
<tr>
<td>Conductivity (μS cm⁻¹)</td>
<td>0-292</td>
<td>292-833</td>
<td>833-2500</td>
<td>2500-5833</td>
<td>&gt;5833</td>
<td></td>
</tr>
<tr>
<td>Colour (hazen units)</td>
<td>&lt;15</td>
<td>16-80</td>
<td>81-140</td>
<td>141-300</td>
<td>&gt;300</td>
<td></td>
</tr>
<tr>
<td>Frequency of algal blooms</td>
<td>Never</td>
<td>&lt;5 Yearly</td>
<td>2-5 Yearly</td>
<td>Every 2 yrs</td>
<td>Annually</td>
<td></td>
</tr>
</tbody>
</table>

Table 2. Score allocation for indicator values in the south-eastern Murray-Darling Basin.
that buffer strips less than 5 m provide minimal protection to aquatic resources under most conditions (Castelle et al., 1994). Bren (1993) indicated that buffer strips greater than 20 m in width were most frequently recommended as providing the best protection for the physical, chemical and biological components of wetlands (also see Castelle et al., 1994). In Australia, the most common minimum buffer widths recommended are 20–30 m (Riding and Carter, 1992). Using this information as a guide, scores were allocated for fringing vegetation width as outlined in Table 2.

**Continuity of fringing vegetation.** Continuity of the fringing vegetation around the wetland perimeter was estimated by eye for each of the main vegetation layers including trees, rushes/sedges and grasses. Following Pressey (1987), a score was given for the continuity of trees, emergents and grass around the wetland perimeter (Table 2). These scores were then combined to give an overall score on the continuity of fringing vegetation, i.e., continuity of fringing vegetation = (continuity of trees + continuity of rushes/sedges + continuity of grasses) / 3.

**Height diversity of fringing vegetation.** A measure of height diversity was based on the sum of the scores for the continuity of each of seven recognized vegetation layers in the fringing vegetation strip.

Three overstorey layers were recorded, following Gillison and Anderson (1981); viz, trees greater than 30 m, trees 10–30 m, trees less than 10 m. There were four possible understorey layers (Smith and Smith, 1990; Sainty and Jacobs, 1981); viz reeds, rushes/sedges, reeds, grasses and herbs (lichens, mosses, ground covers).

The scores out of four for the continuity of each of the seven vegetation layers were then added to obtain a total score for height diversity. The maximum possible score for height diversity was therefore 7 x 4 = 28. All height diversity measures obtained from summing the scores for individual layers of vegetation were then normalized to scores out of four as shown in Table 2.

**Aquatic vegetation**

**Cover of aquatic vegetation.** For each wetland, the percentage of the water surface that was covered with aquatic vegetation was estimated by eye. Aquatic plants included emergents, submerged and floating plants (see below). Score for the aquatic vegetation cover (Table 2) were adapted from Pressey (1987). A wetland totally covered by aquatic vegetation, and thus without open water, may result from nutrient enrichment or by overgrowth of exotic taxa (Sainty and Jacobs, 1981; Henry and Amoros, 1995). Such wetlands are considered to be in poor condition (Mitchell, 1990) and were thus allocated a low score (Table 2). An estimate of cover between 25% and 75% was allocated the highest score under this scheme.

**Attached algae.** Most surfaces within wetlands support a film of microalgae, bacteria, other micro-organisms and organic matter (= biofilm). While biofilms and their components can be used as useful indicators in fresh waters (Burns et al., 1994), they are not appropriate for a rapid appraisal index. The presence and relative abundance of macroalgae was used to indicate the trophic status of wetlands (Goldborough and Robinson, 1996). Following a circuit of the wetland the amount of macroalgae was recorded according to the following categories: little = no obvious macroalgae present (score of 1, Table 2); medium = clumps of significant macroalgae present (score of 2, Table 2); and abundant = macroalgae present over at least one-third of water area (score of 3, Table 2).

**Spatial heterogeneity of aquatic vegetation.** The number of layers of aquatic vegetation occurring in each wetland were noted following Williams (1983) who identified the following five layers of aquatic vegetation: free-floating at surface; free-floating beneath surface; emergent; in substrate with floating leaves; and submerged (anchored in substrate). Scores were based on the number of layers observed (Table 2).

**Water quality**

**Turbidity.** Mean turbidity was calculated based on readings taken at four positions around the perimeter of the wetlands. Turbidity was measured in natural turbidity units (NTU) using a turbidity tube developed by Water Ecoscience of Australia. The highest
score for turbidity (Table 2) was based on data collected from five billabongs along the Murray River from 1977–1982, considered to be in excellent condition and which had turbidities of less than 40 NTU (Hillman, 1986). Lowest scores were allocated based on data from oxbow lakes subject to severe disturbance by stock and introduced carp (Cyprinus carpio) (King et al., 1997).

Conductivity. Mean conductivity was based on measurements taken with a Hanna pocket conductivity meter at four positions in each wetland. The conductivity ratings are based on the classes used by the Victorian Soil Conservation Authority (1980) following their conversion from mg/l to electrical conductivity using the relationship, TDS ppm = 0.6 x EC in microhms cm⁻¹ (GHD, 1992). The scores allocated to conductivity measurements were based on data for pristine wetlands (Hillman, 1986) and known ranges of salinity for wetlands affected by human-derived salinity (Crabb, 1997).

Colour. Water samples were collected from the upper half metre of water, approximately 2 m from the edge at four locations in each wetland. The true colour of each sample was then determined by placing the filtered sample in a Lovebond Colorimeter visual comparator, using distilled water as a reference. The mean of the four colour measures was determined.

Colour scores (Table 2) were based on Hillman’s (1986) observations on relatively pristine wetlands in the region and on recommendations by NHMRC (1994).

Frequency of algal blooms. Although algal blooms occurred naturally in the river systems of the Murray-Darling Basin in the early phase of European settlement (Codd et al., 1994) they now occur more frequently throughout the Basin. A variety of factors have resulted in increased nutrient loadings in floodplain wetlands in the region (Cullen et al., 1993). The frequency of algal bloom occurrence (green slicks covering at least 50% of the water surface) was therefore considered as a good indicator of anthropogenic disturbance in these wetland types. The frequency of algal blooms was determined through questioning of the owner of the wetland and scores allocated as shown in Table 2.

Calculating index scores

After completion of field and laboratory measurements of indicators and allocation of scores, sub-index values for soil, fringing vegetation, aquatic vegetation and water attributes were calculated as shown in Table 3. Note that each sub-index value was normalized to produce a score out of 10. In the case of soil, fringing vegetation and aquatic vegetation for which there were three indicators, the maximum sum of indicator scores was 12. This was normalized to an overall score out of 10 by dividing by a factor of 1.2. In the case of water attributes where there were four indicators, the maximum total score for the sub-index was 16 and this was normalized to a score out of 10 by dividing by a factor of 1.6. For each of the four sub-indices, the range of scores was 0 = extremely poor condition, to 10 = excellent condition. The total index score was the average of the four sub-index scores (Table 3).

Evaluating the condition index

Agreement with long-term monitoring data

A major issue with rapid-appraisal indices based on indicators is how well they might reflect differences relative to ‘natural’ habitats and conclusions based on long-term, highly structured monitoring data (Hart, 1988; WWF, 1992). The authors chose to test the agreement between the wetland condition index and monitoring data available from an independent, 18 month long study of the effects of farming and river regulation on the ecology of over 30 wetlands in the Ovens and Murray River floodplains (Figure 1) conducted by Ogden (1996).

Ten of the 30 wetlands studied by Ogden (1996) were ranked by Ogden (pers. comm.) from best to worst condition based on water quality, vegetation and zooplankton data. The rankings were based on total phosphorus, total nitrogen, turbidity and the species diversity of cladoceran skeletal remains in surface sediments. These chemical parameters
are commonly employed in long-term water quality monitoring programmes (ANZECC, 1992), and the use of biota to assess the condition of aquatic systems has received widespread acceptance (e.g. Norris et al., 1995).

The skeletal remains of cladocera were used to assess differences in the diversity of zooplankton because the use of recent skeletal remains damps the short-term variation observed in net zooplankton data. Cladocera are valuable monitoring taxa because they are responsive to changes in water quality (Anderson-Carnahan et al., 1995) and also reflect alteration in submerged aquatic vegetation because chydorid cladocerans are littoral zone inhabitants and are phyetal associates (Prey, 1986). Total phosphorus, total nitrogen and turbidity measures were all standardized to 100 cm to remove the effect of wetland depth. The turbidity and cladoceran diversity data was given double the weighting of the nutrient data because nutrient data are naturally highly variable at many time scales. Billabongs which were ranked equal based on the above data, were separated using the data on aquatic macrophyte cover for each wetland. Wetlands without macrophytes were ranked lower than those with macrophytes (R. Ogden, pers. comm.). Without prior knowledge of the rankings based on the long-term data, the 10 wetlands were

<table>
<thead>
<tr>
<th>Attribute and indicator</th>
<th>Measure</th>
<th>Score</th>
<th>Sub-index score</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bank stability</td>
<td>Stable</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Degree of pugging</td>
<td>4-9 per sq metre</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Percent organic matter</td>
<td>15-9</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Soil sub-index</td>
<td>9/1.2</td>
<td>7.5</td>
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<tr>
<td>Fringing vegetation</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean width of fringing vegetation</td>
<td>3-75 m</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Continuity of fringing</td>
<td></td>
<td></td>
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</tr>
<tr>
<td>vegetation</td>
<td>Trees—50%</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Rushes—90%</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Grasses—100%</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Height diversity of fringing vegetation</td>
<td>Trees &gt;30 m 50%—2</td>
<td>2</td>
<td>(2 + 3 + 4)/3 = 3</td>
</tr>
<tr>
<td></td>
<td>Trees &lt;10 m 10%—1</td>
<td>1</td>
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</tr>
<tr>
<td></td>
<td>Rushes/sedges 90%—3</td>
<td>3</td>
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</tr>
<tr>
<td></td>
<td>Grass 100%—4</td>
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</tr>
<tr>
<td></td>
<td>Herbs 100%—4</td>
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<tr>
<td>Fringing vegetation sub-index</td>
<td>7/1.2</td>
<td>5.8</td>
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</tr>
<tr>
<td>Aquatic vegetation</td>
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<td></td>
<td></td>
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<tr>
<td>Percent cover of aquatic vegetation</td>
<td>20%</td>
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</tr>
<tr>
<td>Attached algae</td>
<td>Little</td>
<td>4</td>
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<tr>
<td>Spatial heterogeneity of aquatic vegetation</td>
<td>Layers present:</td>
<td>3</td>
<td>Free floating at surface</td>
</tr>
<tr>
<td></td>
<td>Emergent</td>
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<tr>
<td></td>
<td>Submerged</td>
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<td>9-7/1.2</td>
<td>8.0</td>
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<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Turbidity</td>
<td>36 NTU</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Conductivity</td>
<td>75 µS cm⁻¹</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Colour</td>
<td>62-5 hazen units</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Algal bloom frequency</td>
<td>Never</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Water quality sub-index</td>
<td>15/1-6</td>
<td>9.3</td>
<td></td>
</tr>
<tr>
<td>Total Index</td>
<td>(7.5 + 5.8 + 8.0 + 9.3)/4</td>
<td>7.7</td>
<td></td>
</tr>
</tbody>
</table>
surveyed using the wetland condition index over a 3 day period in 1996.

### Influence of the season on index values

Ten wetlands on the floodplain of the Kiewa Valley (Figure 2) were surveyed in autumn (April to May). Although the Kiewa Valley is a relatively wet area (710 mm annual average rainfall), the autumn surveys were done in the peak of the dry period in 1996. During the wettest part of the year (August to October) the floodplain is frequently flooded. To determine whether index scores differed significantly in the two season extremes, the same 10 wetlands were re-surveyed in the period 12–14 August 1996.

### Influence of investigator on index values

Rapid appraisal indices need to be robust to be used by different investigators (Kent et al., 1992). To determine the effect of the investigator on the index, a set of 11 wetlands on the Murrumbidgee River near Wagga Wagga (Figure 3) were surveyed by two independent groups. Both groups worked independently, and obtained index scores without knowledge of the rankings obtained by each other. All surveys were completed between 26 June and 23 July 1996. The index applied to the Murrumbidgee wetlands did not include the frequency of algal bloom or colour indicators. Many of the Murrumbidgee wetlands were located on Government-owned reserves and stock routes, so it was not possible to obtain information on the frequency of algal blooms from land owners. Information on colour was not available owing to a mix-up with samples.

### Results

#### The index and long-term monitoring data

The rapid index results compared very favourably with the rankings of the 10 wetlands based on long-term monitoring data (Figure 2), suggesting the results obtained by the wetland condition index provide a valid estimate of the health of wetlands.

Many of the wetlands used in this part of the study were located in reserves and were thus relatively unaffected by floodplain development. For this reason the range of wetland condition index scores of the wetlands was small (6.3–8.3).

#### The influence of season on index scores

##### Soil

Scores for the soil attribute sub-index (consisting of the indicators: bank stability, degree of pugging and organic content of soil) were very similar in the two seasons (Figure 3(a)), with slightly higher scores recorded in winter. In the Kiewa Valley, the floodplain is subjected to severe flooding during winter and grazing herds do not always use the river flats. The cattle hoof marks that exist around wetlands in autumn are covered by water in winter, as are sections of wetland banks that were visible in autumn. The result is a lower average number of pug marks per square metre and a subsequent higher score for this indicator. Seven of the 10 wetlands surveyed in winter had fewer pugs per square metre than in autumn.

##### Fringing vegetation

The fringing vegetation sub-index scores were very similar across the two seasons (Figure
3(b)]. Slightly higher scores were obtained in the autumn survey because increased water levels in water cover portions of the fringing vegetation, resulting in lower scores for the width of fringing vegetation.

Aquatic vegetation

There was little agreement between scores for the aquatic vegetation sub-index which were much lower in the winter survey [Figure 3(c)]. Local rainfall and winter floods increased water levels in the wetlands, removing floating vegetation. Of the 10 wetlands surveyed, seven showed a decreased score for vegetation cover and also a decreased score for spatial heterogeneity.

Water quality

There was no correlation between water quality sub-index scores obtained from wetlands in the two seasons [Figure 3(d)]. However, the sub-index scores all fell within a very narrow range and were mostly very high, indicating flood water quality regardless of time of year.

Total index

Despite seasonal differences in the soil (high scores in winter) and aquatic vegetation (lower scores in winter) sub-indices, the total index scores from the two seasons were very similar [Figure 3(d)].
Robustness of the index to the influence of investigators

Soil

There were minor differences in soil sub-index scores obtained by different investigators [Figure 4(a)]. When broken down into indicator scores, the main differences resulted from assessments of bank stability. This is the most subjective of the three indicators used for soil. Variations in bank stability scores occurred for five of the 11 wetlands, though this was only one scoring bracket difference in all cases.

Fringing vegetation

Despite the high correlation between scores obtained by different investigators [Figure 4(b)], scores from investigator one were often one unit higher than those from investigator two. All three of the indicators used to assess fringing vegetation are based on visual assessments and are subject to some investigator error. Inspection of the raw data indicated that the width of fringing vegetation was the indicator that caused most variation in sub-index scores given to the wetlands by the two independent groups.
Aquatic vegetation

There was little evidence of consistent differences between the two investigators in obtaining aquatic sub-index scores (Figure 4(e)). However, the slightly lower correlation coefficient indicates some variation in the scoring of indicators. The scores given for the spatial heterogeneity indicator were consistent in all but one case. Scores given for cover of aquatic vegetation varied in four of the 11 wetlands. Several wetlands had a significant cover of the floating plant *Azolla* pinnata and variation in the spread of *Azolla* with wind may have affected results, as the surveys by the two groups were not performed on the same day. Scores for attached algae differed by one unit in seven of the 11 wetlands surveyed by the two investigators.

Water quality

The water quality sub-index scores obtained by the two groups showed a strong correlation (Figure 4(d)). This was anticipated as the sub-index scores were based solely on direct measurements of conductivity and turbidity for these surveys. However, two of the wetlands showed differences in turbidity ratings, and this was likely due to rainfall that occurred in the 3 weeks between visits by the two investigators.

Total index

There was a very high correlation in total index scores obtained by the two investigators (Figure 4(e)). However, in seven of the 11 wetlands, investigator one obtained scores that were slightly higher than those obtained by investigator two.

Discussion

The monitoring of aquatic systems has evolved in three directions. Ambitious programmes based on formal ecological principles have been developed by national and international agencies to assess the condition of aquatic systems over large temporal and spatial scales (e.g. Novitski, 1995; Resh et al., 1995). Such programmes depend on rigorous and time-consuming data collection and analyses by scientists and target professional resource managers and policy makers (Hart, 1988).

More inexpensive and rapid assessment tools, which integrate condition over various temporal and spatial scales, have been developed to enable smaller teams of scientists to perform assessments of wetlands and waterways (e.g. Karr, 1981; Kent et al., 1992; DCNR, 1995). For instance, the index of biological integrity (Karr, 1981), which uses aspects of fish community structure to assess aquatic ecosystem health, has been tested and used successfully in the United States (e.g. Steedman, 1988) and modified for use elsewhere (e.g. Harris, 1995). Such approaches also target professional resource managers and policy makers.

In many countries a variety of simple and rapid monitoring tools have also been developed for use by management agencies to obtain information on wetland conservation status (e.g. Pressley, 1984; Cable et al., 1989). In addition, simple tools are available to the general human community to collect information on the health of aquatic systems. These programmes often target community-based conservation efforts (Courtemanch, 1988).

In the case of rapid assessment tools it is essential that methods are validated and that results from these methods reflect real changes in ecological condition (Hart, 1988; Courtemanch, 1988; Resh et al., 1995). Recent reviews of rapid appraisal methods for wetlands indicate that the validity of these methods is rarely assessed (Cable et al., 1989; WWF, 1992).

The results of the present study have shown that the wetland condition index is a rapid and easily used assessment tool which produced results on wetland condition that were similar to those based on longer-term ecological data. It was not surprising that there was not 100% agreement between the two sets of condition ratings for the wetlands on the Ovens and Murray floodplains. Ogden (1996) completed his monitoring of the wetlands 1 year before they were assessed using the wetland condition index. Factors such as water quality, fringing vegetation biomass and the degree of soil disturbance can change
rapidly in the region in response to alterations in grazing pressure by domestic stock and introduced fish (e.g. Bacon et al., 1994; King et al., unpubl.).

The wetlands used for the comparisons of long-term data with the condition index were all relatively protected from severe degradation by farming practices (R. Ogden, pers. comm.). The total condition scores for these wetlands fell within a narrow and relatively high range (6.3-8.3). A closer relationship between the rankings based on rapid index scores and long-term data would have been expected over a greater range of wetland conditions.

Parts of the wetland condition index are sensitive to seasonal changes in environmental factors. In the Kiewa Valley, the major environmental variable which changed between autumn and winter was flooding, which increased water levels in the floodplain wetlands and reduced the access of stock to floodplain habitats. These changes were reflected in the higher winter scores for the soil sub-index resulting from a reduction in soil pugging by cattle, and the lower winter score for the aquatic vegetation sub-index resulting from the removal of floating vegetation by floodwater. While such sensitivity is a prerequisite for indicators of ecosystem health (e.g. Kent et al., 1992; Breckenridge et al., 1995), it means that natural seasonal changes will need to be accounted for when using the wetland condition index to assess changes in condition following alterations to land or water management practices.

An important consideration in the use of any wetland condition index is access to reference wetlands that are in pristine or relatively pristine condition (Turner et al., 1995; Brinson and Reinhardt, 1996). Reference wetlands will be required in each major bioregion where the wetland condition index is used to ensure that the scoring of indicator data reflects regional variation. For instance, indicators or 'metric' scoring has been adjusted against regional reference data for other assessment tools, such as the index of biological integrity, so that it provides a relevant measure of stream health (e.g. Steedman, 1988). In addition, if the wetland condition index is to be used for long-term monitoring, rather than as a tool to assess the relative condition of a number of wetlands at any one time, then the index will need to be used in a study design which employs several reference wetlands as control sites. In this way natural variation in condition can be separated from that arising from land and water management practices (e.g. Turner et al., 1995).

The wetland condition index appears to be relatively robust to use by different investigators, and thus should be useful for comparisons between sites assessed by different groups. However, there are potential problems with the allocation of scores for some of the more qualitative indicators (e.g. attached algae), which were responsible for most of the variation in scoring between groups. This implies that some joint training may be necessary to acquaint investigators with the variations in indicator measurements before the condition index is used by a number of different people.

A rapid, easily used, and scientifically valid wetland condition index should have wide application in the south-eastern Murray-Darling Basin of Australia. More than 80% of wetlands in the Basin are on private property and the index will allow property managers and management agency staff to assess the condition of wetlands in a cost-effective manner. The exact version of the index described here may only be applicable in the wetlands used in this study, i.e. floodplain meander cutoffs. Extension to other regions of the Basin, and other floodplain wetlands, may be possible if reference wetlands are available to adjust indicator scores to reflect local natural conditions.

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Evaluating a wetland condition index


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