supervising scientist report 147

Techniques for enhanced wetland inventory and monitoring

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Environment Australia
Department of the Environment and Heritage

supervising scientist

The work supporting this publication was funded by the National Wetlands Program of the Natural Heritage Trust.

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This report should be cited as follows:

Finlayson CM & Spiers AG (eds) 1999. *Techniques for enhanced wetland inventory and monitoring*. Supervising Scientist Report 147, Supervising Scientist, Canberra.

The Supervising Scientist is part of Environment Australia, the environmental program of the Commonwealth Department of Environment and Heritage.

Supervising Scientist
Environment Australia
GPO Box 787, Canberra ACT 2601 Australia

ISSN 1325-1554

ISBN 0 642 24350 6

Printed in Darwin by NTUniprint.

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Preface

In recent years Australian governments have directed more and more attention towards the wise use and conservation of wetlands. This has resulted in a number of international initiatives such as hosting the 1996 Conference of the Ramsar Wetlands Convention, supporting the Ramsar Scientific and Technical Review Panel and initiating an Asia/Pacific wetland management training program. At a national level it has resulted in the development of specific federal and state wetland policies and a National Wetlands Program. The latter has provided support for a number of wetland projects, including the development of management plans for individual Ramsar sites and a directory of nationally important wetlands.

During the 1996 Ramsar Conference Australia strongly supported the adoption of Resolution 6.1 'Working Definitions of Ecological Character, Guidelines for Describing and Maintaining the Ecological Character of Listed Sites, and Guidelines for Operation of the Montreux Record'. This resolution called for a greater effort in wetland monitoring and a review of early warning systems for detecting adverse ecological change in wetlands.

In order to further develop the National Wetland Program and abide by Resolution 6.1 serious consideration has been given to the development of national approaches for wetland inventory and monitoring. As a consequence, the ANZECC Wetlands and Migratory Shorebirds Taskforce, consisting of representatives from all state/territory and the federal conservation/environment agencies, issued a recommendation supporting the development of a draft protocol for a national wetland inventory. In response, the Environment Australia (EA) Biodiversity Group obtained funding under the National Wetlands Program for a project aimed at developing a draft national wetland inventory proposal. The project, *Technique Development and Databases for Enhanced Wetland Inventory in Northern Australia – Designing the Scope of the National Wetlands Inventory*, is currently being undertaken by the Environmental Research Institute of the Supervising Scientist.

The scope of this ambitious project included four major tasks for wetland inventory and monitoring with a particular emphasis on northern Australia. These tasks are paraphrased below

1. Review the information provided in the national wetland directory
2. Assess the usefulness of remote sensing techniques for wetland inventory and monitoring
3. Review the usefulness of early warning systems for wetland monitoring
4. Draft protocols for a national approach to wetland inventory and monitoring

Reports on these four tasks are provided in this volume. As such they provide a basis for further decisions on the development and implementation of wetland inventory and monitoring programs in Australia and elsewhere.

In these reports it is stressed that whilst a great deal of wetland monitoring and inventory of Australian wetlands has occurred this has been uneven and fragmentary, and, in too many instances, poorly done. Over the same period a large but possibly indeterminate proportion of Australian wetlands has been degraded or lost. If this situation is to be reversed and we move forward into an era of not only preventing further loss but also recouping past losses, we will require a greatly enhanced inventory and monitoring effort. This effort is not beyond the technological expertise and experience that currently exists within Australia.

An assessment of the extent of wetland inventory data held in Australia

Abbie G Spiers & C Max Finlayson[1]

Abstract

In order to undertake effective research and to manage the extensive wetlands of Australia, a comprehensive and easily accessible collation of wetland information is required. As this is the case for all states and territories, the concept of a coordinated approach to wetland inventory at the national level has received support. To date, an inventory of Australian wetlands does not exist, although a national overview of wetlands was provided in the 1980s. Much of the existing information base for wetlands was recently compiled in the *Directory of Important Wetlands in Australia* (2nd edn) that was coordinated by the Australian Nature Conservation Agency (1996). Collation of material for the Directory has resulted in the development of a federally-funded project to draft the scope of a national wetland inventory. The scope for such an inventory will be developed through liaison with wetland experts and by assessing various existing information sources. This paper provides a first step in this process by summarising the extent of wetland data provided by each state/territory in the Directory.

Based on the above analysis it is recommended that a national approach to wetland inventory is developed and implemented. This should include standardised techniques to systematically collect, collate, store and disseminate data and information and provide mechanisms for identifying priorities for national funding. Addressing these issues should overcome apparent short-comings in the current national wetland directory and provide state and territory jurisdictions with a basis for making decisions on wetland inventory.

1 Introduction

In recent years a large amount of attention has been directed towards managing, including restoring (Steever 1997), Australian wetlands. Part of this effort has centred on collecting and collating information for wetland directories and inventories. However, this effort has been uneven and fragmentary and is incomplete. Further, a national inventory has not been compiled. An inventory is now often seen as a means of providing at least part of the information base that managers need for strategic planning (Dugan 1990, Finlayson 1996). Thus, at a national scale the management of Australian wetlands is proceeding without having recourse to a comprehensive and current information base for planning purposes.

Though a national wetland inventory has not been undertaken, an overview of the distribution of Australian wetlands was conducted in the 1970s by CSIRO (Paijmans et al 1985). This was a mapping exercise based on the distribution pattern of wetlands as shown on 1:250 000 topographical maps. Whilst an invaluable exercise, this overview was not comprehensive and did not contain information on individual wetlands. The database and maps produced from this overview were not published.

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The extent of information on Australian wetlands has been augmented by several reviews prepared in the 1980s (McComb & Lake 1988, Finlayson & von Oertzen 1993, Jacobs & Brock 1993). These collated and presented summaries of a great deal of information on wetlands and further exposed the unevenness and fragmentary state of wetland knowledge. Seemingly, much of the better quality information had been collected a decade earlier by dedicated field recording teams with access to aerial photography and early Landsat imagery.

The recent *Directory of Important Wetlands in Australia* (ANCA 1996), while not intended as a comprehensive inventory, provides an invaluable description of wetlands of national importance, based on information held by government agencies in each state/territory. The format of the Directory is based on those undertaken at a continental or regional scale by various international organisations (eg for Oceania and Asia – Scott 1989, 1993). Whilst the Directory is a major step forward it is not a complete record of all information held by the states/territories (Briggs 1996, Whitehead & Chatto 1996, Blackman et al 1996). Specific data collecting exercises were funded by the National Wetland Program to augment the data and information already held, but this does not appear to have been conducted in a systematic manner.

The information base provided by the Directory of Wetlands of National Importance could lead to a national wetland inventory. However, for this to be realised a more strategic approach at a national level is required and major decisions on goals, techniques and funding need to be taken. The current approach has undoubtedly been a valuable exercise, but as it is not systematic or comprehensive its continuation should be seriously examined. The current edition of the Directory has merit as a compilation of existing information, but as it is uneven, fragmentary and incomplete it does not substitute for a national inventory. Importantly, the international models on which the Directory was based have themselves been found to be less useful than anticipated (Finlayson & Spiers 1999).

2 Aim

In this paper we aim to determine the extent of wetland data held by each state/territory, including the information contained in the *Directory of Important Wetlands in Australia* (2nd edn) (ANCA 1996), hereafter described as 'the Directory'. As such it provides an information base to assist in the development of a draft national wetlands inventory proposal. Research for this paper involved liaison with representatives of the ANZECC Wetlands and Migratory Shorebirds Taskforce and other wetland experts, and extraction of information from the Directory (ANCA 1996).

3 State/territory review of wetland information

In order to review the extent of wetland information held by the states/territories, representatives of the ANZECC Taskforce and other wetland experts around Australia were contacted for information relating to wetland inventory in their state/territory.

Further information was obtained from the National Wetland Program (NWP) agreements between Environment Australia and the states and territories. This information was combined with that contained in the introductory sections of the Directory (ANCA 1996) and used to present a state/territory summary of the extent of wetland inventory information. The summary assessment of each state and territory may not be comprehensive, but it does represent that provided by the above-mentioned sources. The contributions of the authors for each state/territory section in the Directory are acknowledged (ACT – Lintermans &

Ingwersen 1996, NSW – Briggs 1996, NT – Whitehead & Chatto 1996, Queensland – Blackman et al 1996, SA – Morelli & De Jong 1996, Tas – Blackhall et al 1996, Vic – Hull 1996, WA – Lane et al 1996, External Territories – Usback 1996).

Unfortunately the information held in the Directory on individual wetlands has not been interrogated and it is not possible to report on the information resource itself. Interrogation of the information in the Directory, especially that covering wetland area, uses and threats, would provide a major resource for managers and policy makers. The interrogation could be done on a state/territory or biogeographical region basis and provide information for a national overview of wetlands.

In Table 1 we present the total area of wetland listed in each state/territory. However, we strongly caution against extrapolating from these data as it is not an accurate estimation of the total wetland in each state/territory. For example, the Directory is incomplete with some states/territories not having supplied complete datasets. Further, the Directory only lists wetlands of national importance, it does not list all wetlands, and much of the data that is supplied is not accurate or complete and some is missing. There are also inconsistencies in classification and delineation of wetland areas (eg the inclusion of estuarine open water areas). Much greater accuracy in data collection and analysis is required before we can state with any degree of certainty the extent of wetland across large parts of the continent.

Table 1 Total area of wetland listed by each state/territory in the Directory of Important Wetlands in Australia (Phillips 1996)

State/Territory	Area (ha)
Australian Capital Territory	670
New South Wales	2 171 740
Northern Territory	2 912 790
Queensland	11 453 560
South Australia	4 100 290
Tasmania	20 830
Victoria	395 100
Western Australia	2 056 250
External Territories	1 090 580
Total	**24 201 810**

Thus, the data presented in Table 1 should be used only as a relative indication of the extent of wetland in each state/territory. On this basis the Directory entries for Queensland cover a far greater area than the other states/territories. The Queensland data include the Great Barrier Reef whereas some other states/territories have not supplied information on estuarine or marine wetlands. The Directory uses the definition of a wetland adopted by the Ramsar Wetland Convention:

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

However, it would appear that some states/territories may have used a much narrower definition that does not include marine wetlands. If this is the case it is a major inconsistency in the Directory and greatly undervalues its usefulness.

The bibliographic references given in the Directory provide a useful information resource for a national wetland inventory. This could form the basis of a comprehensive wetland bibliography.

Descriptive information provided in the Directory for each state/territory includes notes on information gaps and recommendations for future wetland inventory and research. This information adds considerable weight to the proposal for a coordinated national approach to wetland inventory, including standardised data management systems. In undertaking this review of the Directory it very quickly became apparent that standardised formats for data collation and presentation had either not been agreed or not followed if they had been agreed! The collation and presentation of all existing information (incorporating spatial data and a bibliographic and meta-database) according to standardised procedures would be a major goal for a national wetland inventory (Finlayson 1998). Information collated and presented in such a manner would provide a valuable resource for management and planning purposes and also provide a base for satisfying national obligations under the Ramsar Convention on Wetlands.

3.1 Australian Capital Territory (ACT)

A list of nationally important wetlands in the ACT was prepared by Lintermans and Ingwersen (1996) and is presented in Appendix 1.1. This includes 13 individual wetland sites or complexes from 2 biogeographical regions. The ACT has a diverse range of aquatic habitats, ranging from small sub-alpine bogs to the larger riverine systems such as the Murrumbidgee River. However, the geographic location and altitude of the ACT preclude some wetland types that are common elsewhere in Australia. For example, there are no marine, estuarine or brackish wetlands in the ACT due to its inland location. Likewise, with the location of the ACT in the south-eastern highlands, large lowland floodplain systems are absent.

The first substantial review of aquatic ecosystems in the ACT was conducted by Hogg and Wicks (1989). This dealt mainly with lotic systems and did not attempt to cover the high altitude wetlands such as fens and bogs. Evans and Keenan (1993) remedied this when they reviewed published and unpublished literature on high altitude wetlands in the ACT.

The ACT is located within two of the biogeographical regions defined by Thackway and Cresswell (1995) – the Australian Alps, and the South Eastern Highlands. The majority of the important wetlands are found in the Australian Alps biogeographical region and are located above 1000 m altitude with the highest being Snowy Flats at 1610 m.

3.2 New South Wales (NSW)

A list of nationally important wetlands in NSW was prepared by Briggs (1996) and is presented in Appendix 1.2. This includes 94 individual wetland sites or complexes from 14 biogeographical regions. Wetland inventory has been undertaken in a number of regions of NSW. Mapping of the Lower Darling wetlands was undertaken in 1992 using existing black and white aerial photography at scales ranging from 1:50 000 to 1:84 600, dating from 1981–1982 (King & Green 1993). This project aimed to systematically catalogue all natural and man-made wetlands along the Barwon-Darling Rivers and their tributaries (in NSW). The natural and cultural values, vegetation and human impact at each wetland site was described, along with a cursory fauna survey.

Similar mapping projects were undertaken in the Gwydir valley and the Namoi valley (Green & Bennett 1991, Green & Dunkerley 1992). The Gwydir valley wetlands were mapped from 1:50 000 scale black and white aerial photographs, dated June/July 1985. The data were limited in detail, but did provide the basis for a water management plan for the valley (Green & Bennett 1991). The Namoi Valley wetlands were mapped using 1:40 000 to 1:80 000 scale aerial photographs, dating from 1978 to 1989. Mapping units were based on the dominant perennial wetland vegetation, and wetland characteristics entered on a database. Areas of open water were identified using Landsat satellite imagery and defined by a flood in August 1984. As with the Gwydir wetlands, this was the first systematic inventory of the wetlands in the Namoi valley (Green & Dunkerley 1992).

A large project is underway to map all the wetlands (>5 ha) in the Murray-Darling Basin, using Landsat imagery (MSS). This is one of the largest wetland mapping projects in Australia and requires a supercomputer to cope with the size of the dataset that includes 58 Landsat MSS images (Kingsford et al 1997). All images used in the project were rectified and analysed and wetlands identified and broadly classified. The mapping was done using expert knowledge and ancillary information from other mapping projects. It is envisaged that the methodology will be extended to map wetlands throughout NSW, creating maps that are readily comparable.

Associated with the Murray-Darling wetlands inventory a database is being developed to link published information to each wetland, river and basin. This will contain a series of key fields and the database will link at different levels of information. A comprehensive design phase has been completed and the project has been trialed on the catchment of the Macquarie River. The database will be produced on CD-Rom for McIntosh and IBM platforms and user feedback will be encouraged during the development phase. Ultimately, it is envisaged that the database will contain information on all wetlands in NSW within a framework of individual catchments. The project was been endorsed as an all-Government project in NSW and the Murray-Darling Basin Commission is interested in extending it to the entire Basin (R Kingsford pers com).

An inventory of lakes 100 ha or larger, as shown on 1:100 000 topographical maps, in western NSW has recently been completed (Seddon et al 1997). Attributes such as source of water, salinity, water regime, and area were recorded and overlain with information on lakebed cropping. In the course of this inventory a large amount of literature was collated.

An extensive review of wetlands literature for the south coast of NSW was undertaken in 1994, producing data sheets, in a format consistent with the Directory of Important Wetlands in Australia, for coastal and upland wetlands in the study area (Winning & Brown 1994). These specific studies coupled with past and ongoing inventory effort will provide an invaluable framework for developing a national approach to wetland inventory.

3.3 Northern Territory (NT)

A list of nationally important wetlands in the NT was prepared by Whitehead and Chatto (1996) and is presented in Appendix 1.3. This includes 30 individual sites or complexes from 11 biogeographical regions. A further 21 supplementary sites are also listed, but are not described in any detail. Whilst wetlands are well represented in the NT landscape, knowledge of the flora and fauna of these environments is patchy, and understanding of their ecological functioning often little better than rudimentary (Storrs & Finlayson 1997).

The constraints that this knowledge deficit places on the robust discrimination of the relative conservation significance of different sites was explicitly recognised in the first edition of this directory (Usback & James 1993). Rather than create a potentially misleading list of ‘best’ sites, a small number of wetlands, thought to represent a reasonable sample of the range of wetland environments existing in the NT, were identified and their better-known characteristics summarised.

In the period of time between the first and second editions of the Directory there have been some improvements in the knowledge base, particularly in regard to the coast and the sub-humid wetlands of the middle latitudes of the NT. In this edition, most revisions of prior listings derive from enhanced information on coastal sites, and the few additions are for new sites in the sub-humid tropics, for which a useful knowledge base was provided by the extensive surveys in the 1990s (Jaensch 1994, Jaensch & Bellchambers 1995). An overview of the conservation status of wetlands of the NT, based on existing information, was provided by Storrs and Finlayson (1997) and included a listing of all known data sets of relevance to wetlands.

Whitehead and Chatto (1996) regard the NT contribution to the Directory, and the resultant lists, as insignificant in themselves, but rather as small steps in a larger and much more important process. That is, to derive conservation strategies which embed the conservation of the region's extraordinary wetlands in sustainable management arrangements encompassing entire landscapes and supported by a comprehensive inventory of all substantial wetlands. Thus, whilst the NT has not fully reported on its wetland data holdings in the Directory it has offered very public support for a strategic national wetland inventory.

3.4 Queensland (Qld)

A list of nationally important wetlands in Queensland was prepared by Blackman et al (1996) and is presented in Appendix 1.4. This includes 165 individual sites or complexes from 17 biogeographical regions. With the exception of several biogeographical regions, sites listed in the first edition of the Directory (Usback & James 1993) were mostly chosen arbitrarily, based on available information. In the second and revised edition, selection of new sites has been much less arbitrary for the Cape York Peninsula, Channel Country, Gulf Plains and Wet Tropics biogeographical regions, and for coastal areas of the two Brigalow Belt biogeographical regions, because of new broadly based information becoming available for these.

The current listings partially define the geographical distribution of both the major areas of wetland development across the state, as well as the strengths and weaknesses of current information on which to make assessments of wetlands. For example, the better representation of Cape York Peninsula, Gulf Plains, Wet Tropics and Channel Country biogeographical regions reflects that these regions also contain the most extensive areas of wetland development in Queensland and are amongst the best known. Other regions are poorly represented because they are little known, particularly the Mitchell Grass Downs and the Mulga Lands.

On a statewide basis, coastal freshwater, estuarine and intertidal marine wetlands are now reasonably well represented, but other marine wetlands, notably coral reefs, are poorly represented as individual sites. Of the 162 terrestrial wetlands, 119 (totalling almost 6.2 million hectares) lie north of the Tropic of Capricorn, while 43 (totalling 1.8 million hectares) lie south of this latitude. While this partially reflects real differences in natural occurrence of wetlands, the southern areas are none the less clearly under represented.

The present work underscores the relative paucity of regional scale primary data derived from systematic field surveys, as well as the lack of overall comparative information throughout the state. Completion of the field surveys necessary to provide such data is a priority, but also a considerable undertaking because of the huge areas involved. In this respect the biogeographical regions have proved to be a suitable framework for inventory of wetlands in Queensland's terrestrial environments, and this should now be extended to corresponding marine environments. The major priority is regional scale identification and delineation of all wetland aggregations to allow a state-wide assessment at the resolution of the present directory. At the same time this will identify areas which require additional systematic field surveys to complete this assessment.

Thus, whilst Queensland has added considerable data for specific regions the coverage and reporting of information is still very uneven. Nevertheless, the information that is being reported is being addressed in a systematic manner and gaps identified. It is expected that the systematic approach taken in Queensland will provide a valuable framework for developing a national approach to wetland inventory.

3.5 South Australia (SA)

A list of nationally important wetlands in South Australia was prepared by Morelli and De Jong (1996) and is presented in Appendix 1.5. This includes 68 individual sites or complexes from 8 biogeographical regions. South Australia contains an array of significant wetlands despite being the driest of the Australian states. In the first edition of the Directory (Usback & James 1993), 43 wetlands were listed. Most of these wetlands have been retained and updated while others, such as Serpentine Lakes, Ooldea Soak and Warbla Cave Lakes, have been omitted from the second edition of the Directory (ANCA 1996) mainly because they either lack site information or no longer meet the criteria for inclusion. By reviewing the wetlands listed in the Directory the volume of information for all of the wetlands has been greatly improved. At present, knowledge of the Riverland, South East, and coastal wetlands is relatively adequate. The least known wetland areas occur within the southern Mt Lofty Ranges, Flinders Ranges, Great Victoria Desert and far north eastern desert and gibber plains.

Data for each wetland site were primarily derived from published reports, unpublished material, from databases held by the Nature Conservation Society of South Australia (NCSSA) and Western Australian Royal Australasian Ornithologists Union (WARAOU), and from consultations with wetland managers, scientists and others with expert knowledge. The report by Lloyd and Balla (1986) was used as a major source of reference for collating information on important wetlands. It provides a detailed review and description of the State's wetlands for each watershed region and discusses impacts, threats and identifies conservation values and management requirements.

In the course of compiling the first edition of the Directory a database was prepared and currently holds information for 120 wetland sites. However, the database was not upgraded to include the extra data presented in the second edition of the Directory. Further information is contained in the RAOU database for wetlands of ornithological importance for the Register of the National Estate in southern Australia.

The absence of systematic broadscale surveys in the Directory program is obvious by the information gaps, in particular the omission of some poorly known, yet potentially important, sites. It is recommended that a statewide survey be conducted and compared with the results of Lloyd and Balla (1986). Special attention should be given to the Great Victoria Desert, Flinders and Olary Ranges and Nullarbor biogeographical regions since present survey information is severely inadequate. Any further editions of the Directory should also include a supplementary list of wetlands that meet one or more of the criteria, but remain too poorly known for inclusion at this stage.

3.6 Tasmania (Tas)

A list of nationally important wetlands in Tasmania was prepared by Blackhall et al (1996) and is presented in Appemdix 1.6. This lists 91 wetlands from 8 biogeographical regions and a further 58 wetlands of state importance. These were drawn from a list of 800 sites on an inventory of Tasmanian wetlands. This inventory is made up largely of data from Kirkpatrick and Harwood (1981) and other surveys and contains very little recent information. Although much of the information is now dated, it is an important starting point for future projects. However, so far only about half of the State's land area has been surveyed. There is an urgent need to update and expand the coverage. Ten wetlands in Tasmania have been designated under the Ramsar Convention as being of international importance. These sites have been investigated in greater detail than most of the other wetlands, and management plans for each are currently being written.

As with other states, Tasmania has lost many of its original wetlands, primarily to agricultural land clearing and urban development. Little information is available on the original extent of wetlands and it is difficult to determine an accurate picture of what has been lost. Unfortunately the loss has continued up to the present and the inventory shows that in 1981, some 51% of wetlands recorded were disturbed, and 12% were severely disturbed.

The purpose of the inventory of Tasmanian wetlands was to consolidate and reformat existing information and develop an open-ended database to allow future updating and revision (Atkinson 1991). The wetlands presented here were drawn from this inventory and it is anticipated that more sites that fit the criteria will be added to the list in the future. Thus, a more strategic approach to wetland inventory in Tasmania is being developed and many gaps and priorities have been identified.

3.7 Victoria (Vic)

A list of important wetlands in Victoria was prepared by Hull (1996) and is presented in Appendix 1.7. This includes 121 individual wetland sites or complexes from 10 biogeographical regions. A GIS-based inventory of wetlands in Victoria was completed in 1994 (CNR 1995), having originated from mapping work undertaken in the 1970s (Corrick & Norman 1980; Corrick 1981, 1982). There are two layers in the GIS; the first indicates pre-settlement wetland areas (wetlands >1 ha), divided into categories based on salinity and water regime; and the second details current wetland distribution, divided into categories and sub-categories based largely on subjective description of vegetation types recognisable from aerial photographs (ie reedbeds, redgums, black box, open water). As the layers are linked to a GIS it is possible to gain other information about each area, including demography, jurisdictional boundaries etc.

The Victorian wetlands inventory (CNR 1995) has recorded in excess of 16 000 naturally-occurring wetlands (>1 ha) to date. In addition to these, there are approximately 2000 artificial impoundments in Victoria which contribute to permanent open freshwater wetland habitat in the state. Nine wetland categories ranging from shallow freshwater wetlands, high altitude fens to marine embayments and tidal flats have been identified by the wetlands survey and inventory investigations (CNR 1995).

The impact of non-Aboriginal settlement and development on Victorian wetlands has been severe. About one-third of the state’s wetlands have been lost, and many of those remaining are threatened by continuing degradation from salinity, drainage problems and agricultural practices. Over 90% of the wetlands lost have been on private land. Most of the wetlands losses occurred after 1860 when the land selection process began. In the past 150 years the total area of the State's wetlands has decreased from an estimated 782 000 to 629 000 ha (CNR 1995). Present challenges in wetland conservation in Victoria are the management of wetland reserves on public land, protection and management of wetlands on private land, allocation and secure tenure for environmental water allocations.

The second edition of the Directory (ANCA 1996) completes the representation of all Victorian wetland categories (see Norman & Corrick 1988). The representation of regional wetland types is, however, incomplete for the highlands, East Gippsland, Wimmera plains and north-central Victoria. A review of Victorian biogeographical regions is now required to ensure that all Victorian wetland sites potentially meeting the national criteria are identified. The completion of this work will contribute to the management of Victoria's important wetlands and to the protection and maintenance of their identified values. Thus, wetland inventory and data management in Victoria is well advanced and the gaps identified. The approach taken in Victoria could well provide valuable guidance for further national effort, in particular for smaller regions with many small wetlands.

3.8 Western Australia (WA)

A list of nationally important wetlands in Western Australia was prepared by Lane et al (1996) and is presented in Appendix 1.8. This lists 110 wetlands from 23 biogeographical regions. A systematic survey of wetlands or wetland values across WA has not yet been conducted. In the Directory this is manifest in gaps in the information presented and in the omission of some poorly known, yet potentially important, sites. This edition of the Directory is therefore not definitive.

Accounts of WA wetlands were compiled mainly from published reports, from databases held by WADCALM and from consultations with wetland scientists, managers and others with relevant knowledge. The terminology and categories of Semeniuk (1987) and Semeniuk et al (1990) have been used to describe certain physical, hydrological (salinity) and structural (vegetation) characteristics of the wetlands. The 110 sites described only represent a small fraction of the total (and unknown) number of wetlands in WA. Because most is known about south-west wetlands, more than half (58) of the sites included are from this region.

In preparing the second edition of the Directory, most effort was directed towards increasing the representation of wetlands in biogeographical regions (Thackway & Cresswell 1995) from which few or no wetlands had previously been selected. In the main, these were in remote arid areas. This was a time-consuming process as much of the information was found only in the knowledge, notebooks and unpublished reports of an array of people from across WA.

With the completion of the second edition, all but three of the 26 biogeographical regions of WA now have wetlands included in the Directory. Representation is generally limited to two to four sites per region, however, and more field work is needed to ensure that the great diversity of wetlands in this western third of the continent is truly represented in future editions. There are no apparent plans to develop a comprehensive inventory to further address deficiencies in the information base. However, a database has been designed for WA mangroves, with classification based on geomorphology at a regional and local scale (V Semeniuk pers comm).

Thus, wetland inventory across WA is incomplete and extremely uneven; however, there are no plans to rectify this situation. Given the vast areas of WA and the aridity of much of the landscape further inventory could be a costly and difficult exercise unless suitable remote sensing techniques are adopted. The detailed analyses for wetlands in the south-west of the state could provide a framework for similar relatively small areas, but may not be that useful for larger extensive areas.

3.9 External territories

A list of nationally important wetlands in Australia's external territories was prepared by Usback et al (1996) and is presented in Appendix 1.9. One of the significant gaps identified in the first edition of the Directory was the absence of important wetland areas located on lands managed by the federal government, including some of Australia's island territories such as Norfolk Island, the Cocos (Keeling) Islands, Christmas Island, the Coral Sea Islands, Ashmore and Cartier Islands, Heard Island and McDonald Island. The federal government is responsible for land management matters on these island territories.

Islands and their surrounding marine environment provide habitat for a variety of wildlife including species which may be endemic to a particular island and those which utilise the area during a critical stage in their life cycle. However, many islands are subject to some degree of human interference, and are threatened by the same factors as the Australian mainland, particularly habitat destruction, exploitation, tourism, recreation and human habitation. Over the past decade, in recognition of their significance, many of the Australian territorial islands have been afforded a high level of protection under the *National Parks and Wildlife Conservation Act 1975*. Federal agencies together with relevant State and Territory agencies are working cooperatively to effectively manage these isolated areas.

Wetland components of six island territories are described, all of which fall within the management jurisdiction of Parks Australia. In order to complete the review of the external territories future assessment of the glacial lakes and pool complexes located on Heard and McDonald islands is required.

4 Knowledge gaps

Gaps in wetland inventory and knowledge, as outlined by each state/territory in the Directory, are summarised below. The publication of the Directory has highlighted the large gaps in knowledge of wetlands in some biogeographical regions of all states/territories, and the need for further effort towards inventory of wetlands in the less studied, more remote and/or less protected areas of Australia. These gaps extend across a wide range of information fields – wetland occurrence and extent, threats and management issues, ecological character, management processes and actions.

It is also apparent that the Directory has been primarily the preserve of conservation agencies with variable levels of input from resource development agencies. This is a disappointing feature given the national commitment to ecologically sustainable development. For the latter to be achieved far greater cooperation between governmental sectors and local communities is required. Under a scenario of ecologically sustainable development of wetlands it is imperative that all sectors that influence the management of wetlands contribute to the design and implementation of a national approach to wetland inventory. Wetland inventory is a tool for managing wetlands – it is not just a catalogue of wetland conservation information.

4.1 Australian Capital Territory (ACT)

The vast majority of wetlands in the ACT are protected in nature reserves or national parks. The largest reserved area is Namadgi National Park, which contains all the wetlands within the Australian Alps. The majority of the larger lowland aquatic habitats in the ACT are also protected in nature reserves. There has been no systematic survey of the distribution and importance of the smaller lowland wetlands in the ACT, with information often being collected in an *ad hoc* fashion. These are potentially the wetlands at highest risk as they lie outside the nature reserve system.

4.2 New South Wales (NSW)

The amount of information available on wetlands in NSW has increased, albeit in an uneven manner, with the advent of various inventory programs and additons specifically undertaken for the second edition of the Directory (Briggs 1996). This has resulted in listing of all known wetlands that meet the criteria for inclusion in the Directory. In this repect, the knowledge base for NSW wetlands is more complete than that in many other states/territories. Gaps in information are being sought and will be covered by various ongoing inventory programs (eg for the Murray-Darling Basin – Kingsford et al 1997) and supplemented by further community consultation. The value of the latter source of information has been acknowledged by Briggs (1996) and could assist in the documentation of further information on specific sites. This process is assisted by formal links between governmental agencies, academic institutions and community groups.

4.3 Northern Territory (NT)

The NT regards its contribution to the Directory, and the resultant lists, as small steps in the larger and more important process of deriving conservation strategies that embed the conservation of NT wetlands in sustainable management arrangements encompassing entire landscapes. The NT looks forward to recognition and further development of the Directory as a comprehensive inventory of all substantial wetlands, its presentation to reflect functional wetland groupings, better indicate the role of wetland systems in the regional ecology, and the management actions needed to maintain that role.

4.4 Queensland (Qld)

The Directory listings for Queensland partially define the geographical distribution of the major areas of wetland development across the state, and the strengths and weaknesses of current information on which to make assessments of wetlands. For example, the better representation of wetlands in the Cape York Peninsula, Gulf Plains, Wet Tropics and Channel Country biogeographical regions reflects that these regions contain the most extensive areas of wetland development in Queensland and are amongst the best known. Other biogeographical regions are poorly represented because they are little known, although some are expected to contain a range of very significant wetlands.

On a statewide basis, coastal freshwater, estuarine and intertidal marine wetlands are reasonably well represented in the Directory, but other marine wetlands, notably coral reefs, are poorly represented as individual sites. Of the 162 terrestrial wetlands, 119 (totalling almost 6.2 million hectares) lie north of the Tropic of Capricorn, while 43 (totalling 1.8 million hectares) lie south of this latitude. While this partially reflects real differences in natural occurrence of wetlands, the southern areas are none-the-less clearly under-represented.

4.5 South Australia (SA)

In South Australia, knowledge of the Riverland, South East, and coastal wetlands is relatively adequate. The least known wetland areas occur within the southern Mt Lofty Ranges, Flinders Ranges, Great Victoria Desert and far north eastern desert and gibber plains.

While the establishment of a reserve system provides a basis for wetland protection and management, the lack of data available for some wetlands highlights the need for systematic inventories, biological surveys and research programs in many areas of South Australia. In the Directory the lack of systematic broadscale surveys show in the gaps of information presented and in the omission of some poorly known, yet potentially important, sites.

It is recommended that a statewide survey be conducted to compare with the results of Lloyd and Balla (1986). Special attention should be given to the Great Victoria Desert, Flinders and Olary Ranges and Nullarbor biogeographical regions as present survey information is severely inadequate. Future revisions of the Directory should also include the supplementary list of wetlands that meet one or more of the criteria but remain too poorly known for inclusion at this stage.

4.6 Tasmania (Tas)

There are 91 Tasmanian wetlands listed in the Directory, drawn from a list of 800 sites on an inventory of Tasmanian wetlands made up largely of data from Kirkpatrick and Harwood (1981) and other surveys. As this inventory contains very little recent information, some sites already on the list may change or have already changed in value in the time since the original information was gathered more than ten years ago. However, the inventory is an important starting point for future projects. So far only about half of Tasmania’s land area has been covered, and there is an urgent need to update and expand the coverage as sufficient funding becomes available. The wetland inventory covers about one quarter of the state's natural wetlands and contains a high proportion of data on shallow lentic waters. There are many flowing, artificial and marine waters still awaiting investigation.

4.7 Victoria (Vic)

All Victorian wetland categories (see Norman & Corrick 1988) are represented in the Directory. However, representation of regional wetland types is incomplete for the highlands, East Gippsland, Wimmera plains and north-central Victoria. A review of Victorian biogeographical regions is required to ensure that all Victorian wetland sites potentially meeting the national criteria are identified and described.

4.8 Western Australia

The Directory presents a summary of existing knowledge of important wetland sites in WA and of their values. No systematic survey of wetlands or wetland values across the State has been conducted. In the Directory this is manifest in gaps in the information presented and in the omission of some poorly known, yet potentially important, sites. The WA chapter of the Directory is not definitive.

There are 26 biogeographical regions in WA, eight shared with South Australia and/or NT. All but two, Hampton and Nullarbor, have wetlands included in the Directory. However, representation is generally limited to two to four sites per region, and more field work is needed to ensure that the great diversity of wetlands in WA is truly represented in future editions. Government funding for formal wetland inventory and evaluation is limited and information will continue to be collected by other means.

4.9 External territories

Wetland components of six island territories are described in the Directory, all of which fall within the management jurisdiction of Parks Australia. In order to complete the review of the external territories future assessment of the glacial lakes and pool complexes located on Heard and McDonald islands is required.

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Appendix 1 Estimates of area for important wetlands of Australia

A list of wetlands contained in the *A directory of important wetlands in Australia* (ANCA 1996) is presented for each state and territory of Australia. The information is presented on a biogeographical basis and includes the name of the wetland, an estimate of area, reference number and the page number in the hardcopy volume.

Area estimates collated below are in hectares unless otherwise specified. Wetland area of rivers and estuaries, for example, is recorded in kilometres or square kilometres. Where the Directory provides estimates of area for both the wetland and associated open flats or forest, the wetland estimate only is recorded here. All recorded areas are estimates only, as i) for many wetlands the extent of seasonal inundation may vary dramatically each year; and ii) there is limited current and accurate scientific data available for many wetlands. Where no information is recorded in the Directory, the wetland area is specified 'NIA' (no information available). Wetland areas listed here as 'variable' were described in the Directory as highly variable in area, due to the particular flooding regime of each wetland and/or lack of adequate data. Area calculations for estuaries is difficult, due to the need to incorporate coastline, islands and areas of water up to 6 metres deep. Rivers also present problems, being long, winding and of variable width, and intricately associated with other wetland types, eg billabongs, floodplains and estuaries. Extra information from the Directory has been provided where necessary, to clarify an entry.

Appendix 1.1 Important wetlands in the Australian Capital Territory

Australian Alps (estimated wetland area 244 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Big Creamy Flats	15	AA001AC	33
2	Cotter Flats	41	AA004AC	34
3	Ginini and Cheyenne Flats	50	AA006AC	35
4	Rock Flats	12	AA010AC	36
5	Rotten Swamp	30	AA011AC	37
6	Scabby Range Lake	5	AA012AC	38
7	Snowy Flats	35	AA014AC	38
8	Upper Cotter River	15	AA015AC	39
9	Upper Naas Creek	56	AA016AC	40

South Eastern Highlands (estimated wetland area 348 ha)

10	Bendora Reservoir	81	SEH002AC	41
11	Horse Park Wetland	40	SEH007AC	42
12	Jerrabomberra Wetlands	174	SEH009AC	43
13	Nursery Swamp	53	SEH018AC	44

Appendix 1.2 Important wetlands in New South Wales

Australian Alps (estimated wetland area 90 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Blue Lake	14	AA002NS	52
2	Kosciusko Alpine Fens, Bogs and Lakes	30	AA007NS	52
3	Rennex Gap	45	AA009NS	53
4	Snowgum Flat	1	AA013NS	54

Brigalow Belt South (estimated wetland area 6000 ha)

5	Lake Goran	6000	BBS007NS	54

Channel Country (estimated wetland area 179 065 ha)

6	Bulloo Overflow / Carypundy Swamp	~178 500	CHC003NS	55
7	Salisbury Lake (Lake Altibouka)	565	CHC023NS	55

Darling Riverine Plains (estimated wetland area 357 120 ha)

8	Lower Gwydir Wetlands	102 120	DRP002NS	56
9	Macquarie Marshes	~200 000	DRP003NS	57
10	Menindee Lakes	45 000	DRP004NS	58
11	Narran Lakes	10 000	DRP005NS	58
12	Talyawalka Anabranch and Teryawynia Creek	variable	DRP006NS	59

Mulga Lands (estimated wetland area 758 939 ha)

13	Green Creek Swamp	variable	ML001NS	60
14	Lake Burkanoko	271	ML002NS	60
15	Lake Nichebulka	348	ML003NS	61
16	Murphys Lake	1000	ML007NS	61
17	Paroo Overflow	720 000	ML008NS	62
18	Willeroo Lake	120	ML009NS	63
19	Yantabulla Swamp (Cuttaburra Basin)	37 200	ML010NS	63

Murray Darling Depression (estimated wetland area 469 000 ha)

20	Darling Anabranch Lakes	269 000	MDD005NS	64
21	Lowbidgee Floodplain	200 000	MDD021NS	64

New England Tableland (estimated wetland area 920 ha)

22	Little Llangothlin Lagoon	120	NET001NS	66
23	New England Wetlands	0.5–500*	NET002NS	66
24	Round Mountain	300	NET003NS	67

* Total area over 300 sq. km. Individual wetlands 0.5–500 ha.

NSW North Coast (estimated wetland area 104 663 ha)

25	Barrington Tops Swamps	~1 500	NNC001NS	68
26	Bundjalung National Park	17 738	NNC002NS	68
27	Clarence River Estuary	10 300	NNC003NS	68
28	Clybucca Creek Estuary	1 817	NNC004NS	69
29	Crowdy Bay National Park	8 022	NNC005NS	69
30	Everlasting Swamp	244	NNC006NS	70
31	Lake Hiawatha and Minnie Water	367	NNC007NS	70
32	Limeburners Creek Nature Reserve	9 083	NNC008NS	71
33	Myall Lakes	10 193	NNC009NS	71
34	Port Stephens Estuary	30 253	NNC010NS	71
35	Swanpool/ Belmore Swamp	1 250	NNC011NS	72
36	The Broadwater	1 550	NNC012NS	72
37	Upper Coldstream	1 400	NNC013NS	73
38	Wallis Lakes and adjacent estuarine islands	8 556	NNC014NS	73
39	Wooloweyah Lagoon	2 390	NNC015NS	73

* 103 km^2 of water, 22 400 km^2 of catchment

NSW South Western Slopes (estimated wetland area 121 000 ha)

40	Lake Cowal/ Wilbertroy Wetlands	29 000	NSS002NS	74
41	Tomneys Plain	90	NSS004NS	75

Riverina (estimated wetland area 146 584 ha)

42	Black Swamp	350	RIV002NS	76
43	Booligal Wetlands	5 000	RIV004NS	76
44	Cuba Dam	200	RIV007NS	77
45	Great Cumbung Swamp	50 000	RIV010NS	77
46	Koondrook and Perricoota Forests	31 150	RIV015NS	78
47	Lachlan Swamp (Part of mid Lachlan Wetlands)	6 600	RIV017NS	78
48	Lake Brewster	6 114	RIV019NS	79
49	Lake Merrimajeel/ Murrumbidgil Swamp	300	RIV023NS	79
50	Lower Mirrool Creek Floodplain	Variable	RIV028NS	80
51	Merrowie Creek	2 500	RIV029NS	80
52	Mid Murrumbidgee Wetlands	Variable	RIV030NS	81
53	Millewa Forest	33 636	RIV031NS	81
54	Tuckerbil Swamp	~400	RIV039NS	82

Riverina continued

55	Wakool-Tullakool Evaporation Basins	2 100	RIV040NS	82
56	Werai Forest	11 234	RIV042NS	83

Simpson-Strzelecki Dunefields (estimated wetland area 5816 ha)

57	Sturt National Park Wetlands	*	SSD003NS	84
58	The Salt Lake	5 816	SSD004NS	84

* Large number of small, highly variable wetlands

South East Corner (estimated wetland area 9920 ha)

59	Clyde River Estuary	~4 500	SEC001NS	85
60	Cullendulla Embayment	~220	SEC002NS	85
61	Merimbula Lake	5 200	SEC006NS	86

South East Highlands (estimated wetland area 17 648 ha)

62	Bega Swamp	23	SEH001NS	86
63	Big Badja Swamp	106	SEH003NS	87
64	Coopers Swamp	15–20	SEH006NS	87
65	Jacksons Bog	150	SEH008NS	87
66	Lake Bathurst	1 350	SEH010NS	88
67	Lake George	15 000 full	SEH012NS	89
68	Micalong Swamp	526*	SEH015NS	89
69	Monaro Lakes	0.5–215	SEH016NS	90
70	Yaouk Swamp	258	SEH024NS	90

* Whole reserve, including surrounding forest.

Sydney Basin (estimated wetland area 59 598 ha)

71	Bicentennial Park	56	SB001NS	91
72	Blue Mountains Sedge Swamps	*	SB002NS	92
73	Botany Wetlands	64	SB003NS	92
74	Boyd Plateau Bogs	variable	SB004NS	93
75	Budderoo National Park Heath Swamps	1 150	SB005NS	93
76	Coomoderry Swamp	670	SB006NS	93
77	Eve St. Marsh, Arncliffe	4	SB007NS	94
78	Jervis Bay	41 044	SB008NS	94
79	Killalea Lagoon	NIA	SB009NS	96
80	Kooragang Nature Reserve	3 000	SB010NS	96
81	Lake Illawarra	3 000	SB011NS	97
82	Long Swamp	88	SB012NS	98
83	Longneck Lagoon	24	SB013NS	98
84	Minnamurra River Estuary	200	SB014NS	99
85	Newington Wetlands	26	SB015NS	100

Sydney Basin continued

86	O'Hares Creek Catchment	~900	SB016NS	101
87	Pitt Town Lagoon	46	SB017NS	101
88	Shoalhaven/ Crookhaven Estuary	2 500	SB018NS	102
89	Shortland Wetlands Centre	45	SB019NS	102
90	St Georges Basin	4 400	SB020NS	102
91	Thirlmere Lakes	50	SB021NS	103
92	Towra Point Estuarine Wetlands	1 161	SB022NS	103
93	Wingecarribee Swamp	320	SB023NS	104
94	Wollumboola Lake	850	SB024NS	104

* numerous small wetlands

Appendix 1.3 Important wetlands in the Northern Territory

Daly Basin (estimated wetland area < 1650 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Daly River Middle Reaches	*	DAB001NT	123

* 165 km of river up to 100m wide, also billabongs each 5–40 ha.

Finke (estimated wetland area 10 ha)

2	Finke River Headwater Gorges System	10	FIN001NT	125

Gulf Fall and Uplands (estimated wetland area 100 ha)

3	Mataranka Thermal Pools	<100	GFU002NT	127

Great Sandy Desert (estimated wetland area 133 700 ha)

4	Karinga Creek Palaeodrainage System	30 000	GSD002NT	128
5	Lake Amadeus	103 700	GSD003NT	129

Gulf Coastal (estimated wetland area 303 980 ha)

6	Borroloola Bluebush Swamps	80–90	GUC001NT	131
7	Limmen Bight (Port Roper) Tidal Wetlands System	184 800*	GUC002NT	132
8	Port McArthur Tidal Wetlands System	119 000	GUC003NT	134

* excluding subtidal seagrass areas

Mitchell Grass Downs (estimated wetland area 372 900 ha)

9	Corella Lake	15 000	MGD002NT	136
10	Eva Downs Swamp	~11 000	MGD004NT	137
11	Lake de Burgh	~35 000	MGD005NT	138
12	Lake Sylvester System	~41 000	MGD006NT	139
13	Lake Woods	50 900	MGD007NT	140
14	Tarrabool Lake	~220 000	MGD008NT	142

Ord-Victoria Plains (estimated wetland area 25 000 ha)

15	Birrindudu Waterhole and Floodplain	19 000	OVP001NT	144
16	Nongra Lake	6 000	OVP002NT	145

Pine-Creek Arnhem (estimated wetland area 234 950 ha)

17	Kakadu National Park	233 450*	PCA001NT	146
18	Katherine River Gorge	< 1500	PCA002NT	148

* Wetland area only. Total park area is 1 375 940 ha.

Tanami (estimated wetland area 800 ha)

19	Lake Surprise (Yinapaka)	800	TAN002NT	150

Top End Coastal (estimated wetland area 872 300 ha)

20	Adelaide River Floodplain System	134 800	TEC001NT	151
21	Arafura Swamp	71 400	TEC002NT	154
22	Blyth-Cadell Floodplain and Boucaut Bay System	35 500	TEC003NT	155
23	Cobourg Peninsula System	84 000	TEC004NT	157
24	Daly-Reynolds Floodplain-Estuary System	159 300	TEC005NT	159
25	Finniss Floodplain and Fog Bay System	81 300	TEC006NT	161
26	Mary Floodplain System	127 600	TEC007NT	162
27	Moyle Floodplain and Hyland Bay System	48 100	TEC008NT	165
28	Murgenella-Cooper Floodplain System	81 500	TEC009NT	166
29	Port Darwin	48 800*	TEC010NT	168

* Deepwater area not deducted; includes mangroves at least 16 000 ha.

Victoria Bonaparte (estimated wetland area 9000 ha)

30	Legune Wetlands	9 000	VB003NT	169

Supplementary sites for the Northern Territory
(no area records in the Directory for these wetlands)

Wetland Name	Latitude	Longitude	Principal Known Values
Anson Bay and associated Islands	13°20'	130°15'	Extension of Daly-Reynolds Floodplain-Estuary System. Substantial shorebird numbers. The Perron Islands are site of very large Australian Pelican rookery and important turtle breeding sites (especially for Flatback turtles *Natator depressus*).
Arnhem Bay System	12° 27'	136° 14'	High diversity of mangrove species; saltwater crocodile breeding on floodplain.
Bathurst and Melville Islands	11° 25' – 11° 32'	130° 15' – 130° 46'	Occurrence of uncommon mangrove species; swamps suitable for saltwater crocodile breeding.
Blue Mud Bay System	13° 10'	136° 01'	High numbers of Brolga and whistling-ducks on associated floodplains; large areas of seagrass; occurrence of dugongs.

Supplementary sites for the Northern Territory continued

Buckingham Bay System	12° 14'	135° 40'	High numbers of migratory shorebirds on mudflats; saltwater crocodile breeding on floodplain.
Castlereagh Bay System	12° 08'	134° 58'	High numbers of migratory shorebirds on mudflats; numerous Aboriginal sacred sites on islands.
Cutta Cutta Caves Subterranean Wetlands	14° 35'	132° 28'	Good example of subterranean karst wetland; cave wetland fauna.
Darwin Peninsula Swamps (Knuckey's Lagoon – McMinn's Lagoon)	12° 26'	130° 57'	High numbers of Magpie Goose and Little Curlew; relatively high numbers of Garganey occur regularly; passive recreational use by local residents.
Dum-in-Mirrie Island	12° 38'	130° 23'	Major breeding site for sea turtles.
East Alligator River Middle Reaches	12° 49'	133° 22'	Scenic entrenched river; habitat for freshwater fishes and turtles of upland.
Fitzmaurice River Middle Reaches	14° 49'	130° 38'	Undisturbed scenic river.
George Gill Range Rockholes	24° 20'	131° 48'	Research on aquatic invertebrates in desert region.
Hale River Floodout	24° 33'	135° 42'	Good example of (Simpson) desert wetland type.
Hay River Floodout	23° 43'	137° 16'	Good example of (Simpson) desert wetland type.
Lake Angurugubira, Groote Eylandt	13° 57'	136° 43'	Good example of saline coastal lagoon on an island; has seagrass beds.
Little Moyle Floodplain	13° 46'	129° 50'	Breeding colony of herons and allies in mangroves; saltwater crocodile breeding.
McKinlay River Floodplain	13° 04'	131° 41'	Important area for breeding by Brolga; international reference area for studies on freshwater crocodiles.
Quail Island and other islands associated with the Fogg Bay region			Large numbers of shorebirds and breeding of marine turtles.
Robinson-Calvert Swamps System	16° 29'	137° 55'	Good example of seasonal swamps behind coastal ridges; significant habitat for waterbirds in wetter years.
Shoal Bay and Leanyer Wetlands System	12° 21'	131° 01'	High numbers of migratory shorebirds on mudflats; research on barramundi nursery; intensive observations on migrant waterbirds.
Victoria River Middle Reaches	15° 31'	130° 46'	Habitat for fishes not occurring elsewhere in NT.

Appendix 1.4 Important wetlands in Queensland

Brigalow Belt North (estimated wetland area 475 398 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Abbot Point – Caley Valley	5 154	BBN001QL	187
2	Bowling Green Bay	32 541	BBN002QL	189
3	Broad Sound	212 042	BBN003QL	191
4	Burdekin Delta Aggregation	31 723	BBN004QL	192
5	Burdekin – Townsville Coastal Aggregation	149 197	BBN005QL	194
6	Lake Dalrymple	30 570	BBN006QL	196
7	Lake Elphinstone	300	BBN007QL	197
8	Ross River Reservoir	2 782	BBN008QL	198
9	Southern Upstart Bay	11 089	BBN009QL	199

Brigalow Belt South (estimated wetland area 240 973 ha)

10	Boggomoss Springs	*	BBS001QL	201
11	Fairbairn Dam	15 397	BBS002QL	202
12	Fitzroy River Delta	70 254	BBS003QL	203
13	Fitzroy River Floodplain	19 502	BBS004QL	204
14	Hedlow Wetlands	11 101	BBS005QL	205
15	Lake Broadwater	215	BBS006QL	206
16	Lake Nuga Nuga	2 070	BBS008QL	207
17	Northeast Curtis Island	9 537	BBS009QL	208
18	Palm Tree and Robinson Creeks	50 274	BBS010QL	209
19	Port Curtis	31 264	BBS011QL	210
20	The Gums Lagoon	343	BBS012QL	211
21	The Narrows	20 906	BBS013QL	212
22	Yeppoon – Keppel Sands Tidal Wetlands	10 110	BBS014QL	213

* Several hectares of wetland, scattered over 400 ha area.

Channel Country (estimated wetland area 898 310 ha)

23	Birdsville – Durrie Waterholes Aggregation	32 656	CHC001QL	215
24	Bulloo Lake	83 227	CHC002QL	216
25	Cooper Creek Overflow swamps – Windorah	124 853	CHC005QL	217
26	Cooper Creek Swamps – Nappa Merrie	106 311	CHC006QL	218
27	Cooper Creek – Wilson River Junction	63 925	CHC007QL	219
28	Diamantina Lakes Area	393	CHC008QL	220
29	Diamantina Overflow Swamp – Durrie Station	29 196	CHC009QL	221
30	Georgina River – King Creek Floodout	138 347	CHC011QL	222
31	Lake Bullawarra	1 287	CHC012QL	222
32	Lake Constance	1 841	CHC013QL	223
33	Lake Cuddapan	1 704	CHC014QL	224

Channel Country continued

34	Lake Mipia Area	69 691	CHC015QL	225
35	Lake Phillipi	16 086	CHC016QL	226
36	Lake Torquinie Area	15 242	CHC017QL	227
37	Lake Yamma Yamma	86 548	CHC018QL	228
38	Moonda Lake – Shallow Lake Aggregation	14 738	CHC019QL	229
39	Mulligan River – Wheeler Creek Junction	17 014	CHC020QL	229
40	Muncoonie Lakes Area	88 767	CHC021QL	230
41	Nooyeah Downs swamps Aggregation	6241	CHC022QL	231
42	Toko Gorge and Waterhole	243	CHC025QL	232

Central Mackay Coast (estimated wetland area 249 220 ha)

43	Corio Bay Wetlands	6 909	CMC001QL	233
44	Dismal Swamp – Water Park Creek	1 000*	CMC002QL	236
45	Edgecumbe Bay	4 593	CMC003QL	237
46	Eungella Dam	797	CMC004QL	238
47	Four Mile Beach	7 130	CMC005QL	239
48	Island Head Creek – Port Clinton Area	27 042	CMC006QL	240
49	Iwasaki Wetlands	646	CMC007QL	241
50	Propserpine – Goorganga Plain	16 851	CMC008QL	242
51	Sand Bay	10 182	CMC009QL	243
52	Sandringham Bay – Bakers Creek Aggregation	7 372	CMC010QL	245
53	Sarina Inlet – Ince Bay Aggregation	27 945	CMC011QL	246
54	Shoalwater Bay	122 672	CMC012QL	247
55	St. Helens Bay Area	16 081	CMC013QL	249

* Wetlands spread over catchment area of 11 700 ha.

Cape York Peninsula (estimated wetland area 2 429 936 ha)

56	Archer Bay Aggregation	29 911	CYP001QL	251
57	Archer River Aggregation	149 761	CYP002QL	254
58	Bull Lake	26	CYP003QL	257
59	Cape Flattery Dune Lakes	44 034	CYP004QL	258
60	Cape Grenville Area	7 304	CYP005QL	259
61	Cape Melville – Bathurst Bay	5 480	CYP006QL	261
62	Harmer River – Shelburne Bay Aggregation	31 751	CYP007QL	262
63	Jardine River Wetland Aggregation	81 740	CYP008QL	264
64	Lloyd Bay	15 682	CYP009QL	266
65	Marina Plains – Lakefield Aggregation	392 333	CYP010QL	268
66	Newcastle Bay – Escape River Estuarine Complex	42 307	CYP011QL	270
67	Northeast Karumba Plain Aggregation	182 418	CYP012QL	272
68	Northern Holroyd Plain Aggregation	1 114 324	CYP013QL	274

Cape York Peninsula continued

69	Olive River	17 609	CYP014QL	277
70	Orford Bay – Sharp Point Dunefield	17 239	CYP015QL	279
71	Port Musgrave Aggregation	52 685	CYP016QL	281
72	Princess Charlotte Bay Marine Area	87 835	CYP017QL	284
73	Silver Plains – Nesbitt River Aggregation	44 834	CYP018QL	285
74	Skardon River – Cotterell River Aggregation	63 194	CYP019QL	287
75	Somerset Dunefield Aggregation	8 095	CYP020QL	289
76	Temple Bay	4 424	CYP021QL	290
77	The Jack Lakes Aggregation	35 054	CYP022QL	292
78	Violet Vale	1 896	CYP023QL	293

Desert Uplands (estimated wetland area 50 160 ha)

79	Aramac Springs	*	DEU001QL	295
80	Cauckingburra Swamp	782	DUE002QL	296
81	Doongmabulla Swamp	399	DEU003QL	297
82	Lake Buchanan	23 201	DEU004QL	297
83	Lake Galilee	25 778	DEU005QL	298

* Wetland area ‘small’ but scattered over 400 ha.

Darling Riverine Plains (estimated wetland area 200 ha)

84	Balonne River Floodplain	*	DRP001QL	300

* Actual wetlands only several hundred hectares, spread over a larger floodplain of approx. 24 000 ha.

Einasleigh Uplands (estimated wetland area 132 173 ha)

85	Blencoe Falls – Blencoe Creek	87	EIU001QL	302
86	Great Basalt Wall	100 253	EIU002QL	303
87	Herbert River Gorge	21 536	EIU003QL	306
88	Innot Hot Springs	78	EIU004QL	307
89	Lake Lucy Wetlands	1 078	EIU005QL	307
90	Laura Sandstone	1 090	EIU006QL	308
91	Minnamoolka Area	589	EIU007QL	309
92	Poison Lake	785	EIU008QL	310
93	Spring Tower Complex	78	EIU009QL	311
94	Undara Lava Tubes	1 254	EIU010QL	312
95	Valley of Lagoons	3 645	EIU011QL	313
96	Wairuna Lake	152	EIU012QL	314
97	Walters Plains Lake	1 548	EIU013QL	315

Great Barrier Reef (estimated wetland area 3 488 468 ha)

98	Cairncross Reef Complex	238	GBR001QL	317
99	Clack Reef Complex	1 230	GBR002QL	318
100	Great Barrier Reef Marine Park	3 487 000	GBR003QL	319

Gulf Fall and Uplands (estimated wetland area 1133 ha)

101	Lawn Hill Gorge	1 133	GFU001QL	321

Gulf Plains (estimated wetland area 2 221 612 ha)

102	Bluebush Swamp	879	GUP001QL	323
103	Buffalo Lake Aggregation	1 909	GUP002QL	324
104	Dorunda Lakes Area	6 801	GUP003QL	325
105	Forsyth Island Wetlands	6 388	GUP004QL	326
106	Lignum Swamp	282	GUP005QL	327
107	Macaroni Swamp	258	GUP006QL	328
108	Marless Lagoon Aggregation	167 009	GUP007QL	329
109	Mitchell River Fan Aggregation	714 886	GUP008QL	330
110	Musselbrook Creek Aggregation	45 157	GUP009QL	332
111	Nicholson Delta Aggregation	63 640	GUP010QL	333
112	Smithburne – Gilbert Fan Aggregation	250 320	GUP011QL	334
113	Southeast Karumba Plain Aggregation	336 233	GUP012QL	336
114	Southern Gulf Aggregation	545 353	GUP013QL	338
115	Stranded Fish Lake	67	GUP014QL	341
116	Wentworth Aggregation	82 430	GUP015QL	341

Mitchell Grass Downs (estimated wetland area 69 395 ha)

117	Austral Limestone Aggregation	69 395	MGD001QL	344
118	Elizabeth Springs	*	MGD003QL	345

* Actual wetland area very small, springs scattered over 400 ha.

Mount Isa Inlier (estimated wetland area 329 204 ha)

119	Gregory River	26 639	MII001QL	346
120	Lake Julius	1 935	MII002QL	347
121	Lake Moondarra	1 742	MII003QL	348
122	Thorntonia Aggregation	298 888	MII004QL	350

Mulga Lands (estimated wetland area 26 422 ha)

123	Lake Numalla Aggregation	10 724	ML002QL	351
124	Lake Wyara	6 021	ML003QL	353
125	Lakes Bindegolly and Toomaroo	9 677	ML004QL	354

South Eastern Queensland (estimated wetland area 642 475 ha)

126	Burrum Coast	15 128	SEQ001QL	356
127	Bustard Bay Wetlands	21 850	SEQ002QL	357
128	Carbrook Wetland Aggregation	329	SEQ003QL	359
129	Colosseum Inlet – Rodds Bay	24 307	SEQ004QL	361
130	Conondale Range Aggregation	1 983	SEQ005QL	362
131	Fraser Island	163 294	SEQ006QL	363
132	Great Sandy Strait	93 160	SEQ007QL	366
133	Lake Weyba	2 860	SEQ008QL	367
134	Moreton Bay	300 177	SEQ009QL	369
135	Noosa River Wetlands	9 945	SEQ010QL	371
136	Pumicestone Passage	9 442	SEQ011QL	373

Wet Tropics (estimated wetland area 173 477 ha)

137	Alexandra Bay	841	WT001QL	375
138	Alexandra Palm Forest	146	WT002QL	377
139	Bambaroo Coastal Aggregation	5 360	WT003QL	378
140	Birthday Creek	43	WT004QL	379
141	Bromfield Swamp	63	WT005QL	380
142	Cowley Area	8 344	WT006QL	381
143	Edmund Kennedy Wetlands	11 083	WT007QL	383
144	Ella Bay Swamp	1 315	WT008QL	385
145	Eubanangee – Alice River	1 991	WT009QL	386
146	Herbert River Floodplain	44 496	WT010QL	388
147	Hilda Creek Headwater	5	WT011QL	390
148	Hinchinbrook Channel	30 682	WT012QL	392
149	Innisfail Area	1 220	WT013QL	393
150	Kurrimine Area	754	WT014QL	394
151	Lake Barrine	99	WT015QL	396
152	Lake Eacham	43	WT016QL	396
153	Licuala Palm Forest	232	WT017QL	397
154	Lower Daintree River	5 276	WT018QL	398
155	Missionary Bay	11 227	WT019QL	400
156	Nandroya Falls	19	WT020QL	401
157	Port of Cairns and Trinity Inlet	6 389	WT021QL	402
158	Russell River	2 377	WT022QL	406
159	Russell River Rapids	235	WT023QL	408
160	Sunday Creek, Broad-leaved Paperbark Site	39	WT024QL	409
161	Tully River – Murray River Floodplains	39 154	WT025QL	410
162	West Mulgrave Falls	7	WT026QL	413
163	Wyvuri Swamp	1 492	WT027QL	414
164	Yuccanbine Creek	529	WT028QL	415
165	Zillie Falls	16	WT029QL	416

Appendix 1.5 Important wetlands in South Australia

Channel Country (estimated wetland area 1 980 000 ha)

No.	Wetland Name	Area (ha)	Wetland Reference No.	Page
1	Coongie Lakes	1 980 000	CHC004SA	440
2	Diamentina River Wetland System	NIA	CHC010SA	442
3	Strzelecki Creek Wetland System	NIA	CHC024SA	444

Eyre and Yorke Blocks (estimated wetland area 17 048 ha)

4	Baird Bay	200–300	EYB001SA	445
5	Barker Inlet & St Kilda	NIA	EYB002SA	446
6	Big Swamp	200	EYB003SA	448
7	Clinton	1 964	EYB004SA	449
8	Coffin Bay Coastal Wetland System	NIA	EYB005SA	450
9	Davenport Creek	NIA	EYB006SA	452
10	Franklin Harbor	1 500	EYB007SA	453
11	Lake Hamilton	2 000	EYB008SA	454
12	Lake Newland	8 448	EYB009SA	455
13	Point Davenport	181	EYB010SA	456
14	Point Labatt	147	EYB011SA	457
15	Port Gawler & Buckland Park Lake	>434**	EYB012SA	458
16	Streaky Bay	NIA	EYB013SA	459
17	Tod River Wetland System	*	EYB014SA	460
18	Tumby Bay	1 000	EYB015SA	461
19	Wills Creek	874	EYB016SA	463

* Tod River catchment (197.4 km^2), Poonindie Swamp and Tod River estuary (15 km^2)

** Port Gawler 434 ha.

Flinders and Olary Ranges (estimated wetland area NIA)

20	Upper Spencer Gulf Mangrove System	NIA	FOR001SA	464

Lofty Block (estimated wetland area 50 740 ha)

21	American River Wetland System	~2 000	LB001SA	467
22	Birchmore Lagoon	~150	LB002SA	468
23	Busby and Beatrice Islets	~1 525	LB003SA	469
24	Cygnet Estuary	~1 300	LB004SA	470
25	Cygnet River	NIA	LB005SA	471
26	D'Estrees Bay	140	LB006SA	472
27	Flinders Chase River Systems	~40 450	LB007SA	472
28	Grassdale Lagoons	135	LB008SA	474
29	Lake Ada	994	LB009SA	474
30	Lanacoona Road Swamps	~30	LB010SA	475

Lofty Block continued

31	Lashmar Lagoon	130	LB011SA	476
32	Murrays Lagoon	~2 200	LB012SA	477
33	Onkaparinga Estuary	60	LB013SA	478
34	Tookayerta & Finniss Catchments	~300	LB014SA	479
35	Upper Hindmarsh River Catchment	6.2	LB015SA	480
36	Upper Tunkalilla Creek Swamps	50	LB016SA	481
37	Waidrowski Lagoon	~530	LB017SA	481
38	White Lagoon Wetland System	750	LB018SA	482

Murray-Darling Depression (estimated wetland area 37 001 ha)

39	Banrock Swamp Wetland Complex	1 220	MDD001SA	483
40	Gurra Lakes Wetland Complex	660	MDD006SA	484
41	Irwin Flat	50	MDD010SA	485
42	Loch Luna Wetland Complex	1 905	MDD019SA	486
43	Loveday Swamps	479	MDD020SA	488
44	Lower Murray Swamps	155	MDD022SA	489
45	Marne River Mouth	~40	MDD024SA	490
46	Noora Evaporation Lakes	~500	MDD027SA	491
47	Pike-Mundic Wetland Complex	410*	MDD028SA	492
48	Riverland Wetland Complex	~30 600	MDD032SA	494
49	Spectacle Lakes	~427	MDD034SA	496
50	Stockyard Plain	305	MDD035SA	497
51	Swan Reach Wetland Complex	250	MDD036SA	498

* 410 ha permanent water in a 6700 ha section of the floodplain

Naracoorte Coastal Plain (estimated wetland area 169 073 ha)

52	Bool & Hacks Lagoons	3 221	NCP001SA	499
53	Butchers & Salt Lakes	40	NCP002SA	501
54	Deadmans Swamp	545*	NCP003SA	502
55	Ewens Ponds	~5	NCP004SA	503
56	Honans Scrub	842	NCP006SA	504
57	Lake Frome & Mullins Swamp	3 216	NCP007SA	505
58	Marshes Swamp	665	NCP010SA	507
59	Naen Naen Swamp & Gum Lagoon	335	NCP012SA	508
60	Piccaninnie Ponds	300	NCP013SA	509
61	Poocher & Mundulla Swamps	300	NCP014SA	510
62	South East Coastal Salt Lakes	13 744	NCP015SA	511
63	The Coorong, Lake Alexandina & Lake Albert	140 500	NCP016SA	512
64	Watervalley Wetlands	5 660	NCP017SA	515

* Includes the swamp and surrounding native forest

Simpson-Strzelecki Dunefields (estimated wetland area 1 798 000 ha)

65	Inland Saline Lakes	829 000	SSD001SA	517
66	Lake Eyre	969 000	SSD002SA	518

Stony Plains (estimated wetland area NIA)

67	Dalhousie Springs	*	STP001SA	520
68	Lake Eyre Mound Springs	**	STP002SA	522

* Springs spread over 19 000 ha.
** Some springs have associated wetlands of several ha, while others are soaks of a few m^2.

Appendix 1.6 Important wetlands in Tasmania

Ben Lomond (estimated wetland area 226 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Blackmans Lagoon	28	BEN001TA	538
2	Jocks Lagoon	10	BEN002TA	539
3	Little Waterhouse Lake	10	BEN003TA	539
4	Surveyors Creek	10	BEN004TA	540
5	The Chimneys	90	BEN005TA	541
6	Tregaron Lagoons 1	16	BEN006TA	541
7	Tregaron Lagoons 2	20	BEN007TA	542
8	Unnamed Wetland	1	BEN008TA	542
9	Unnamed Wetland	7	BEN009TA	543
10	Unnamed Wetland	2	BEN010TA	543
11	Unnamed Wetland	10	BEN011TA	544
12	Unnamed Wetland	5	BEN012TA	545
13	Unnamed Wetland	12	BEN013TA	545
14	Unnamed Wetland	2	BEN014TA	546
15	Unnamed Wetland	3	BEN015TA	546

Central Highlands (estimated wetland area 2421 ha)

16	Allwrights Lagoons	6	CH001TA	547
17	Clarence Lagoon	100	CH002TA	548
18	Dublin Bog	1	CH003TA	549
19	Eagle Tarn Sphagnum	1	CH004TA	549
20	Great Lake	1 400	CH005TA	550
21	Interlaken Lakeside Reserve	520	CH006TA	551
22	Kemps Marsh	230	CH007TA	552
23	Lake Kay	60	CH008TA	552
24	Lake Lea	100	CH009TA	553
25	Maggs Mountain Sphagnum	<1	CH010TA	554
26	Mt Rufus Sphagnum	<1	CH011TA	554
27	Shadow Lake Sphagnum	>1	CH012TA	555

D'Entrecasteaux (estimated wetland area 61 ha)

28	D'Arcys Lagoon	26	DE001TA	556
29	Oyster Cove	25	DE002TA	557
30	South East Cape Lakes	10	DE003TA	557

Freycinet (estimated wetland area 6576 ha)

31	Apsley Marshes	~700	FRE001TA	558
32	Douglas River	100	FRE002TA	559
33	Earlham Lagoon	220	FRE003TA	560
34	Freshwater Lagoon	14	FRE004TA	561
35	Hardings Falls Forest Reserve	100	FRE005TA	561
36	Maria Island Marine Reserve	1 500	FRE006TA	562
37	Moulting Lagoon	3 930	FRE007TA	563
38	Unnamed wetland	12	FRE008TA	564

Furneaux (estimated wetland area 2257 ha)

39	Fergusons Lagoon	75	FUR001TA	565
40	Flyover Lagoon 1	18	FUR002TA	566
41	Flyover Lagoon 2	24	FUR003TA	566
42	Hogans Lagoon	85	FUR004TA	567
43	Little Thirsty Lagoon	30	FUR005TA	568
44	Logan Lagoon	700	FUR006TA	568
45	Sellars Lagoon	1 200	FUR007TA	569
46	Stans Lagoon	20	FUR008TA	570
47	Syndicate Lagoon	1	FUR009TA	570
48	Thompsons Lagoon	55	FUR010TA	571
49	Unnamed wetland	25	FUR011TA	571
50	Unnamed wetland	4	FUR012TA	572
51	Unnamed wetland	2	FUR013TA	573
52	Unnamed wetland	18	FUR014TA	573

Tasmanian Midlands (estimated wetland area 2145 ha)

53	Bells Lagoon	80	TM001TA	574
54	Blackman River	1	TM002TA	575
55	Calverts Lagoon	46	TM003TA	575
56	Cataract Gorge	<1	TM004TA	576
57	Elizabeth River Gorge	1	TM005TA	577
58	Folly Lagoon	17	TM006TA	577
59	Glen Morey Saltpan	15	TM007TA	578
60	Glen Morriston Rivulet 1	1	TM008TA	578
61	Goulds Lagoon	3	TM009TA	579

Tasmanian Midlands continued

62	Lake Dulverton	200	TM010TA	580
63	Lake Tiberias	900	TM011TA	580
64	Macquarie River 2	<1	TM012TA	581
65	Macquarie River 4	1	TM013TA	581
66	Mona Vale Saltpan	26	TM014TA	582
67	Near Lagoon	15	TM015TA	583
68	Orielton Lagoon	265	TM016TA	583
69	River Derwent	550	TM017TA	584
70	South Esk River	<1	TM018TA	585
71	Tin Dish Rivulet	<1	TM019TA	585
72	Township Lagoon	10	TM020TA	586
73	White Lagoon	10	TM021TA	586

Woolnorth (estimated wetland area 7075 ha)

74	Bungaree Lagoon	11	WOO001TA	588
75	Lake Flannigan	150	WOO002TA	688
76	Lavinia Nature Reserve	6 800	WOO003TA	589
77	Pearshape Lagoon 1	6	WOO004TA	590
78	Pearshape Lagoon 2	2	WOO005TA	590
79	Pearshape Lagoon 3	1	WOO006TA	591
80	Pearshape Lagoon 4	2	WOO007TA	591
81	Rocky Cape Marine Area	100	WOO008TA	592
82	Unnamed wetland	3	WOO009TA	593

West and South West (estimated wetland area 67 ha)

83	Hatfield Sphagnum	<1	WSW001TA	594
84	Lake Ashwood	12	WSW002TA	595
85	Lake Bantick	5	WSW003TA	595
86	Lake Chisholm	5	WSW004TA	596
87	Lake Garcia	8	WSW005TA	596
88	Lake Surprise	25	WSW006TA	597
89	Lake Sydney	10	WSW007TA	598
90	Little Bellinger	<1	WSW008TA	598
91	Unnamed wetland	0.3	WSW009TA	599

Supplementary wetlands of state significance

(no area estimates available in the Directory for these wetlands)

Ben Lomond Bioregion

Wetland Name	Latitude	Longitude	Wetland type	Criteria for inclusion
Hardwickes Lagoon	40° 55' 34.43"	147° 55' 47.90"	B15	1, 5
Medeas Cove	41° 19' 23.76"	148° 14' 20.72"	A6	1
Tamar Saltmarshes	41° 14' 10.48"	146° 57' 8.16"	A8	1, 3
Windmill Lagoon	41° 19' 50.07"	148° 18' 39.34"	A11	5
Unnamed wetland	41° 1' 21.92"	146° 57' 21.57"	A11	5
Unnamed wetland	41° 4' 29.89"	147° 5' 51.40"	A11	5
Unnamed wetland	40° 50' 36.81"	148° 12' 1.64"	A10	5
Unnamed wetland	40° 50' 32.20"	148° 8' 57.94"	A11	5
Unnamed wetland	40° 51' 10.42"	148° 10' 6.93"	A11	5
Unnamed wetland	40° 51' 27.92"	147° 55' 52.99"	A11	5
Unnamed wetland	40° 54' 29.95"	147° 54' 59.98"	B15	5
Unnamed wetland	40° 51' 12.26"	147° 38' 43.46"	A11	5
Unnamed wetland	40° 52' 13.97"	147° 38' 26.97"	A11	5
Unnamed wetland	41° 1' 47.69"	147° 28' 11.57"	B15	5
Unnamed wetland	41° 1' 44.09"	147° 29' 37.19"	A11	5
Unnamed wetland	40° 56' 54.36"	147° 33' 38.83"	A11	5

Central Highlands Bioregion

Robinsons Marsh	42° 4' 9.63"	147° 7' 58.68"	B15	5
Unnamed wetland	42° 6' 51.84"	147° 6' 31.92"	B15	5

D'Entrecasteaux Bioregion

Southport Lagoon	43° 29' 15.95"	146° 57' 24.18"	A10	1, 5
Port Cygnet	43° 10' 21.29"	147° 5' 10.04"	A1	1, 3

Freycinet Bioregion

Bryans Lagoon	42° 15' 8.26"	16' 52.89"	A10	5
Charlie Dilgers Hole	42° 2' 15.86"	148° 13' 4.22	B15	5
Hazards Lagoon	42° 10' 45.22"	148° 17' 31.16"	A11	1, 5
Little Punchbowl	42° 3' 31.69"	148° 11' 3.84"	A11	1, 5
Old Mines Lagoon	41° 50' 14.48"	148° 15' 30.93"	A10	5
Turners Lagoon 4	42° 59' 18.96"	147° 41' 43.81"	A11	5
Yorkys Lagoon	42° 25' 55.74"	147° 40' 50.90"	B10	5

Furneaux Bioregion

Badger Corner	40° 14' 37.52"	148° 11' 2.10"	A4, A5	5
Curves Lagoons	39° 55' 45.99"	149° 4' 25.32"	A10	5
Green Lagoon	40° 26' 35.81"	148° 8' 29.12"	A11	5
Halfmoon Lagoon	40° 4' 11.34"	148° 16' 28.96"	A10	5
Sandy Lagoon	40° 30' 50.65"	148° 10' 45.17"	A10	5
Scotts Lagoon	40° 12' 27.92"	148° 15' 55.95"	A11	5
Stony Lagoon	40° 0' 16.75"	148° 13' 6.37"	A10	5
Walters Lagoon	40° 12' 27.68"	148° 11' 12.53"	A11	5
Unnamed wetland	39° 45'	149° 7'	A10	5
Unnamed wetland	39° 45'	149° 7'	A10	5
Unnamed wetland	39° 53' 6.01"	148° 0' 46.15"	B15	5
Unnamed wetland	39° 59' 11.90"	148° 13' 5.22"	A11	5
Unnamed wetland	40° 1' 25.64"	148° 11' 51.66"	B15	5
Unnamed wetland	40° 12' 30.19"	148° 17' 24.82"	A10	5
Unnamed wetland	40° 20' 10.50"	148° 12' 45.39"	A11	5
Unnamed wetland	40° 19' 22.96"	148° 20' 51.83"	A10	5
Unnamed wetland	40° 27' 56.22"	148° 24' 13.15"	A11	5
Unnamed wetland	40° 26' 52.81"	148° 12' 31.39"	A11	5

Tasmanian Midlands Bioregion

Crayfish Point Marine Reserve	42° 57' 8.45"	147° 21' 11.03"	A4	1, 6
Gullivers Lagoon	42° 18' 45.23"	147° 27' 39.96"	B15	1, 5
Middle Lagoon	41° 35' 50.82"	147° 2' 31.19"	B6	5
Rushy Lagoon	42° 56' 46.79"	147° 30' 53.40"	A11	5
Smiths Lagoon	41° 48' 43.08"	147° 25' 3.95"	B10	5
Unnamed wetland	42° 45' 46.43"	147° 26' 59.14"	A8	5

Woolnorth Bioregion

Bob Lagoon	39° 41' 7.49"	143° 57' 6.79"		5
Homestead Lagoon	40° 31' 39.11"	144° 43' 51.22"	A11	5
Lily Lagoon	39° 58' 29.11"	143° 55' 30.04"	A11	5
Three Tree Lagoon	39° 38' 16.70"	143° 57' 47.85"	A11	5
Unnamed wetland	39° 56' 40.10"	143° 54' 14.88"	B15	5
Unnamed wetland	39° 53' 1.39"	144° 5' 50.84"	B15	5
Unnamed wetland	39° 37' 26.68"	143° 56' 55.53"	A11	5

West and South West Bioregion

Unnamed wetland	42° 7' 27.62"	145° 17' 52.08"	A11	5

Appendix 1.7 Important wetlands in Victoria

Australian Alpine (estimated wetland area 12 ha)

No.	Wetland Name	Area	Wetland Reference No.	Page
1	Caledonia Fen	5.6	AA003VI	611
2	Davies Plain	NIA	AA005VI	612
3	Mount Buffalo Peatlands	7.1	AA008VI	612

Murray-Darling Depression (estimated wetland area 78 319 ha)

4	Belsar Island	2 500	MDD002VI	614
5	Beveridge Island	1 018	MDD003VI	615
6	Bunguluke Wetlands, Tyrrell Creek & Lalbert Creek	530	MDD004VI	616
7	Hattah Lakes	1 018	MDD007VI	617
8	Heards Lake	135	MDD008VI	618
9	Heywoods Lake	228	MDD009VI	619
10	Kings Billabong Wetlands	502	MDD011VI	620
11	Lake Albacutya	5 700	MDD012VI	621
12	Lake Hindmarsh	15 600	MDD013VI	623
13	Lake Lalbert	500	MDD014VI	624
14	Lake Ranfurly	265	MDD015VI	625
15	Lake Tyrrell	20 860	MDD016VI	626
16	Lake Wallawalla	828	MDD017VI	627
17	Lindsay Island	15 000	MDD018VI	628
18	Major Mitchell Lagoon	9	MDD023VI	630
19	Mitre Lake	784	MDD025VI	631
20	Natimuk Lake, Natimuk Creek & Lake Wyn Wyn	1 170	MDD026VI	632
21	Pink Lake (Lochiel)	106	MDD029VI	633
22	Pink Lakes	393	MDD030VI	634
23	Raak Plain	550	MDD031VI	635
24	Saint Marys Lake	113	MDD033VI	636
25	Wallpolla Island	9 200	MDD037VI	637
26	Wargan Basins (Meridian Lakes)	690	MDD038VI	638
27	White Lake	620	MDD039VI	639

Narracoorte Coastal Plain (estimated wetland area 4679 ha)

28	Glenelg Estuary	98	NCP005VI	640
29	Lindsay-Werrikoo Wetlands	1 785	NCP008VI	641
30	Long Swamp	764	NCP009VI	642
31	Mundi-Selkirk Wetlands	2 032	NCP011VI	643

NSW South West Slopes (estimated wetland area 19 065 ha)

32	Lake Hume	18 465	NSS001VI	644
33	Ryans Lagoon	60	NSS003VI	645

Riverina (estimated wetland area 82 427 ha)

34	Barmah-Millewa Forest	29 500	RIV001VI	647
35	Black Swamp	176	RIV003VI	649
36	Broken Creek	2 500	RIV005VI	650
37	Cemetery Swamp	89	RIV006VI	652
38	First Marsh (The Marsh)	780	RIV008VI	652
39	Fosters Swamp	219	RIV009VI	653
40	Gunbower Island	19 500	RIV011VI	654
41	Hird's Swamp	344	RIV012VI	655
42	Johnson's Swamp	411	RIV013VI	656
43	Kanyapella Basin	2 581	RIV014VI	657
44	Kow Swamp	2 724	RIV016VI	658
45	Lake Bael Bael	648	RIV018VI	659
46	Lake Charm	520	RIV020VI	661
47	Lake Cullen	632	RIV021VI	661
48	Lake Kelly & Stevenson Swamp	320	RIV022VI	662
49	Lake William	96	RIV024VI	663
50	Little Lake Charm, Kangaroo Lake & Racecourse Lake	1 332	RIV025VI	664
51	Lower Broken River	1 268	RIV026VI	665
52	Lower Goulburn River Floodplain	13 000	RIV027VI	666
53	Muckatah Depression	2 909	RIV032VI	668
54	Second Marsh (Middle Marsh)	233	RIV033VI	669
55	Tang Tang Swamp	103	RIV034VI	670
56	Third Marsh (Top Marsh)	946	RIV035VI	671
57	Third, Middle & Reedy Lakes	598	RIV036VI	672
58	Town Swamp	80	RIV037VI	673
59	Tragowel Swamp	262	RIV038VI	674
60	Wallenjoe Wetlands	303	RIV041VI	675
61	Woolshed Swamp	353	RIV043VI	676

South East Coastal Plain (estimated wetland area 154 284 ha)

62	Anderson Inlet	2 230	SCP001VI	677
63	Bald Hills Wildlife Reserve	1	SCP002VI	678
64	Billabong Reserve	23	SCP003VI	679
65	Bosses/Nebbor Swamp	235	SCP004VI	680
66	Corner Inlet	51 500	SCP005VI	681
67	Deep Water Morass	30	SCP006VI	682

South East Coastal Plain continued

68	Edithvale-Seaford Wetlands	215	SCP007VI	683
69	Jack Smith Lake State Game Reserve	2 730	SCP008VI	685
70	Lake Connewarre State Game Reserve	3 100	SCP009VI	686
71	Lake King Wetlands	7 100	SCP010VI	689
72	Lake Victoria Wetlands	10 850	SCP011VI	689
73	Lake Wellington Wetlands	18 000	SCP012VI	690
74	Lindenow Wildlife Sanctuary	26	SCP013VI	692
75	Lower Merri River Wetlands	146	SCP014VI	693
76	McLeods Morass	520	SCP015VI	694
77	Mud Island Marine Reserve & State Wildlife Reserve	656	SCP016VI	695
78	Powlett River Mouth	NIA	SCP017VI	696
79	Russells Swamp	125	SCP018VI	697
80	Shallow Inlet Marine & Coastal Park	1 342	SCP019VI	697
81	Swan Bay & Swan Island	2 800	SCP020VI	699
82	Tambo River (Lower Reaches) East Swamps	33	SCP021VI	700
83	Western Port	52 325	SCP022VI	701
84	Yambuk Wetlands	297	SCP023VI	702

South East Corner (estimated wetland area 3646 ha)

85	Lake Bunga	460	SEC003VI	704
86	Lake Tyers	1 186	SEC004VI	705
87	Lower Snowy River Wetlands System	~2 000	SEC005VI	706

South East Highlands (estimated wetland area 6305 ha)

88	Central Highlands Peatlands	33	SEH005VI	708
89	Lake Dartmouth	5 990	SEH011VI	710
90	Lake Tali Karng	16.2	SEH013VI	711
91	Lower Aire River Wetlands	84	SEH014VI	712
92	Nuniong Plateau Peatlands	10	SEH017VI	713
93	Princetown Wetlands	119	SEH019VI	714
94	Rooty Break Swamp	1	SEH020VI	715
95	Tea Tree Swamp (Delegate River)	52	SEH021VI	715
96	Wongungarra River	NIA	SEH023VI	716

Victorian Midlands (estimated wetland area 981 ha)

97	Creswick Swamp	16	VM001VI	718
98	Lake Muirhead	330	VM002VI	719
99	Mt William Swamp	635	VM003VI	720

Victorian Volcanic Plain (estimated wetland area 45 925 ha)

100	Banongill Network	59	VVP001VI	721
101	Cobden-Terang Volcanic Craters	613	VVP002VI	722
102	Cundare Pool/Lake Martin	3 730	VVP003VI	723
103	Kooraweera Lakes	427	VVP004VI	724
104	Lake Beeac	662	VVP005VI	725
105	Lake Bookaar	500	VVP006VI	726
106	Lake Colongulac	1 400	VVP007VI	727
107	Lake Corangamite	23 300	VVP008VI	728
108	Lake Cundare	395	VVP009VI	729
109	Lake Gnarput	2 350	VVP010VI	730
110	Lake Linlithgow Wetlands	1 432	VVP011VI	731
111	Lake Milangil	125	VVP012VI	732
112	Lake Murdeduke	1 550	VVP013VI	733
113	Lake Terangpom	208	VVP014VI	734
114	Lower Lough Calvert & Lake Thurrumbong	878	VVP015VI	735
115	Middle Lough Calvert	578	VVP016VI	736
116	Point Cook & Laverton Saltworks	900*	VVP017VI	737
117	Red Rock Lakes & The Basins	223	VVP018VI	738
118	Stonyford-Bungador Wetlands	NIA	VVP019VI	739
119	Tower Hill	311	VVP020VI	740
120	Upper Lough Calvert	824	VVP021VI	741
121	Werribee-Avalon Area	5 460**	VVP022VI	742

* 900 ha plus 5 km of coastline

** 5460 ha plus ~13 km of coastline

Appendix 1.8 Important wetlands in Western Australia

Avon Wheatbelt (estimated wetland area 7274 ha)

No.	Wetland Name	Area (ha)	Wetland Reference No.	Page
1	Coyrecup Lake	500	AW001WA	767
2	Dumbleyung Lake	5 561	AW002WA	768
3	Toolibin Lake	437	AW003WA	770
4	Yealering Lakes System	775	AW004WA	771
5	Yorkrakine Rock Pools	<1	AW005WA	773

Carnarvon (estimated wetland area > 537 500 ha)

6	Cape Range Subterranean Waterways	175 000	CAR001WA	774
7	Exmouth Gulf East	120 000	CAR002WA	775
8	Hamelin Pool	90 000*	CAR003WA	777

Carnarvon continued

9	Lake MacLeod	150 000	CAR004WA	778
10	McNeill Claypan System	2 500	CAR005WA	779
11	Shark Bay East	250 km**	CAR006WA	780

* approx. two thirds is known to be >4.5 m deep at low tide and a substantial portion is probably more than 6 m deep at low tide.

** 250 km of coastline; width of intertidal flats is 1–10 km on east side of site, 1–3 km on west side, 1–2 km at Faure Island and 1–5 km at Pelican Island.

Central Kimberley (estimated wetland area > 20 ha)

12	Tunnel Creek	500 m*	CK001WA	783
13	Windjana Gorge	20	CK002WA	783

* The cave is 500 m long.

Central Ranges (estimated wetland area < 1 ha)

14	Rock Pools of the Walter James Range	<1*	CR001WA	785

* Two pools, each ~9 m in diameter, 4 m deep.

Coolgardie (estimated wetland area 550 ha)

15	Rowles Lagoon System	550	COO001WA	787

Dampierland (estimated wetland area 168 512 ha)

16	Bunda Bunda Mound Spring	22	DL001WA	789
17	Camballin Floodplain (Le Lievre Swamp System)	30 000	DL002WA	791
18	Eighty Mile Beach System	40 000	DL003WA	792
19	Geikie Gorge	130	DL004WA	794
20	Roebuck Bay	50 000	DL005WA	795
21	Roebuck Plains System	48 340	DL006WA	797
22	Willie Creek Wetlands	~20	DL007WA	799

Esperance Plains (estimated wetland area 19 911 ha)

23	Balicup Lake System	1 400	ESP001WA	801
24	Culham Inlet System	11 300	ESP002WA	802
25	Fitzgerald Inlet System	1 200	ESP003WA	803
26	Lake Gore System	1 500	ESP004WA	805
27	Lake Warden System	1 200	ESP005WA	807
28	Mortijinup Lake System	750	ESP006WA	808
29	Pink Lake	1 061	ESP007WA	810
30	Yellilup Yate Swamp System	1 500	ESP008WA	811

Gascoyne (estimated wetland area 153 752 ha)

31	Kookhabinna Gorge	< 250	GAS001WA	813
32	Lake Carnegie System	153 100	GAS002WA	814
33	Windich Springs	~2	GAS003WA	815
34	Yadjiyugga Claypan	~400	GAS004WA	817

Geraldton Sandplains (estimated wetland area 3529 ha)

35	Hutt Lagoon System	3 000	GS001WA	818
36	Lake Logue/Indoon System	529	GS002WA	819
37	Murchison River (Lower Reaches)	125 km	GS003WA	821

Gibson Desert (estimated wetland area > 500 ha)

38	Gibson Desert Gnamma Holes	~12 m^2	GD001WA	822
39	Lake Gruszka	500*	GD002WA	823

* Mapped area 500 ha, reported potential area 2000 ha.

Great Sandy Desert (estimated wetland area 112 005 ha)

40	Dragon Tree Soak	5	GSD001WA	825
41	Lake Dora (Rudall River) System	>32 000*	GSD004WA	826
42	Mandora Salt Marsh	80 000	GSD005WA	827
43	Rock Pools of the Breaden Hills	~70 m^2	GSD006WA	829

* Lake Dora & Eva Broadhurst Lake total 32 000 ha, Rudall River is ~120 km long.

Great Victoria Desert (estimated wetland area 71 000 ha)

44	Yeo Lake/Lake Throssell	71 000	GVD001WA	831

Jarrah Forest (estimated wetland area 16 148 ha)

45	Avon River Valley	~64 km	JF001WA	833
46	Byenup Lagoon System	5000	JF002WA	834
47	Chittering-Needonga Lakes	248	JF003WA	836
48	Lake Muir	4600	JF004WA	838
49	Lake Pleasant View System	550	JF005WA	839
50	Moates Lake System	750	JF006WA	840
51	Oyster Harbour	~5000	JF007WA	842

Little Sandy Desert (estimated wetland area > 150 000 ha)

52	Lake Disappointment (Savory Creek) System	>150 000*	LSD001WA	844
53	Pools of the Durba Hills	~2	LSD002WA	846

* Lake Disappointment 150 000 ha, Savory Creek 280 km long, max. 150 m wide (floods to 2 km)

Mallee (estimated wetland area 13 213 ha)

54	Lake Cronin	13	MAL001WA	848
55	Lake Grace System	13 200	MAL002WA	850

Murchison (estimated wetland area 412 630 ha)

56	Anneen Lake (Lake Nannine)	120 000	MUR001WA	851
57	Breberle Lake	750	MUR002WA	852
58	Lake Ballard	~60 000	MUR003WA	853
59	Lake Barlee	194 380	MUR004WA	854
60	Lake Marmion	~35 300	MUR005WA	856
61	Wooleen Lake	2 200	MUR006WA	857

North Kimberley (estimated wetland area NIA)

62	Drysdale River	~170 km	NK001WA	858
63	Mitchell River System	126 km	NK002WA	860
64	Prince Regent River System	>14 300*	NK003WA	861

* Mangrove 14 300 ha, river 100 km incl. tidal part 47 km long and 50–1500 m wide.

Pilbara (estimated wetland area 117 730 ha)

65	De Grey River	> 4 500*	PIL001WA	863
66	Fortescue Marshes	100 000	PIL002WA	864
67	Karijini (Hamersley Range) Gorges	~80	PIL003WA	866
68	Leslie (Port Hedland) Saltfields System	13 000	PIL004WA	867
69	Millstream Pools	150	PIL005WA	869

* Tidal wetlands 4500 ha, plus river above tidal influence ~160 km long

Swan Coastal Plain (estimated wetland area 28 382 ha)

70	Barraghup Swamp	25	SWA001WA	871
71	Becher Point Wetlands	10	SWA002WA	872
72	Benger Swamp	572	SWA003WA	873
73	Booragoon Lake	13	SWA004WA	874
74	Brixton Street Swamps	30	SWA005WA	876
75	Chandala Swamp	100	SWA006WA	877
76	Ellen Brook Swamps System	~20	SWA007WA	878
77	Forrestdale Lake	199*	SWA008WA	879
78	Gibbs Road Swamp System	70	SWA009WA	881
79	Guraga Lake	350	SWA010WA	882
80	Herdsman Lake	250	SWA011WA	884
81	Joondalup Lake	530	SWA012WA	885
82	Karakin Lakes	600	SWA013WA	887
83	Lake McLarty System	400	SWA014WA	888

Swan Coastal Plain continued

84	Lake Thetis	7	SWA015WA	889
85	Loch McNess System	255	SWA016WA	891
86	McCarley's Swamp (Ludlow Swamp)	~25	SWA017WA	892
87	Peel-Harvey Estuary	14 000	SWA018WA	893
88	Perth Airport Woodland Swamps	~22	SWA019WA	896
89	Rottnest Island Lakes	180	SWA020WA	897
90	Spectacles Swamp	141	SWA021WA	899
91	Swan-Canning Estuary	3300**	SWA022WA	900
92	Thomsons Lake	213	SWA023WA	902
93	Vasse-Wonnerup Wetland System	1000	SWA024WA	904
94	Wannamal Lake System	470	SWA025WA	906
95	Yalgorup Lakes System	5600	SWA026WA	908

* Potentially 250 ha.

****** Less ~20% for waters >6 m deep at low tide in Swan Estuary.

Tanami (estimated wetland area 38 700 ha)

96	Lake Gregory System	38 700	TAN001WA	910

Victoria Bonaparte (estimated wetland area 180 200 ha)

97	Lake Argyle	74 000*	VB001WA	913
98	Lake Kununurra	2 500	VB002WA	915
99	Ord Estuary System	94 700	VB004WA	917
100	Parry Floodplain	9 000	VB005WA	918

* Area at cessation of spillway flow. Area usually greater (>100 000 ha) for several months after Wet season floods, exceptionally 200 000 ha.

Warren (estimated wetland area 10 836 ha)

101	Blackwood River (Lower Reaches) and Tributaries	530*	WAR001WA	921
102	Broke Inlet System	>4 800**	WAR002WA	923
103	Cape Leeuwin System	~20	WAR003WA	924
104	Doggerup Creek System	>2 500	WAR004WA	926
105	Gingilup – Jasper Wetland System	>1 600	WAR005WA	928
106	Maringup Lake	286	WAR006WA	929
107	Mt. Soho Swamps	~50	WAR007WA	931
108	Owingup Swamp System	1 050	WAR008WA	932

* Blackwood River 53 km long, 50–150 m wide. Each tributary 15–20 km long, up to 10 m wide.

** Broke Inlet 4800 ha, Shannon River 65 km long, 5–50 m wide.

Yalgoo (estimated wetland area 585 ha)

109	Thundelarra Lignum Swamp	135	YAL001WA	934
110	Wagga Wagga Salt Lake	~450	YAL002WA	935

Appendix 1.9 Important wetlands in Australia's External Territories

(estimated wetland area 1 032 282 ha)

Number	Wetland name	Area (ha)	Wetland Reference No.	Page
1	Ashmore Reef	58 300	XT001CO	946
2	Coringa Islet, Magdelaine and Herald cays	160	XT002CO	948
3	Elizabeth and Middleton Reefs	188 000*	XT003CO	949
4	Hosnie Springs	0.33	XT004CO	950
5	Lihou Reef	844 000*	XT005CO	951
6	Pulu Keeling	122	XT006CO	952

* Includes marine waters.

An assessment of the usefulness of remote sensing for wetland inventory and monitoring in Australia

Stuart Phinn[1], Laura Hess[2] & C Max Finlayson[3]

Abstract

The usefulness of remote sensing applications for wetland inventory in Australia is reviewed. Past reviews of remote sensing applications in wetland environments for inventory purposes have been confined to the continental United States. With few exceptions, these reviews have focused on approaches to inventory and mapping species composition, mainly in inter-tidal wetlands dominated by grasses and sedges. In the Australian context there are very few publications in the refereed literature providing a comprehensive review of the suitability of remote sensing techniques for monitoring wetlands.

Existing and planned commercial remote sensing systems are reviewed in relation to potential wetland inventory and monitoring purposes. Each system is described in terms of its purpose, type of information, and status (research, operational). Examples of camera systems, radiometers, airborne platforms, and digital multi-spectral imaging systems on polar orbiting satellite platforms are discussed. Consideration is then given to the 'next generation' of commercial resource monitoring satellites as potential sets for an Australian wetland inventory because of their high spatial resolution, large area coverage, multi- to hyper-spectral configuration, radiometric precision, availability and cost. The usefulness of Synthetic Aperture Radar is specifically addressed with reference to example applications.

The data types and applications reviewed indicate that remotely sensed data have the potential to act as a major data source for a national wetland inventory and monitoring program for Australia. Several constraints to the extent and form of this application should be recognised. To ensure remotely sensed data are selected wisely and applied to appropriate questions, the following issues must be addressed: i) selection of an appropriate classification scheme or means to incorporate all existing schemes; ii) incorporation of existing wetlands inventory data sets; and iii) identifying the intended product(s) of the inventory and their intended applications.

Selection and application of remotely sensed data for use in a national wetland inventory should take place within the context of existing inventory data sets both statewide and nationally and an appropriate classification system. Every attempt should be made to utilise remotely sensed data at the appropriate scales to complement and extend existing inventory data and 'fill the appropriate gaps.'

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1 Introduction

Wetland inventory and monitoring in Australia has been fragmentary, uneven and poor (Finlayson & Mitchell 1999, Spiers & Finlayson 1999). A national wetland inventory does not exist, although in recent years a great deal of information has been compiled in the Directory of Important Wetlands in Australia (Usback & James 1993, ANCA 1996). However, this does not constitute a national inventory and it is far from comprehensive (Spiers & Finlayson 1999). Further, other inventory and monitoring data sources for wetlands are often scattered and poorly documented (Storrs & Finlayson 1997, Bayliss et al 1997, Finlayson & Mitchell 1999). The need for a national wetland inventory has been supported by the ANZECC Wetlands and Migratory Shorebirds Taskforce. It is noted, however, that the concept of a national inventory has not been universally supported in the past and previous efforts have floundered well short of achieving their goal despite some first class data collation and analysis, for example, that conducted in the 1970s by Paijmans et al (1985).

Past inventory effort was also plagued by technical and logistical problems in obtaining, interpreting and reporting data, as mentioned by several contributors to the *Directory of important wetlands in Australia* (eg Blackman et al 1996, Hull 1996). Recognising that wetland inventory is nowadays seen as a necessary tool for wetland management (Dugan 1990, Finlayson 1996) and with the advent of improved data handling capacity for spatial information (eg GIS – Blackman et al 1995, Hess & Melack 1995, Kingsford 1997, Kingsford et al 1997) we have been requested by the National Wetland Program to review the technological options for obtaining inventory information and to recommend further priority research for inventory techniques. In undertaking this review we are well aware of the diversity of opinion on the usefulness of remotely sensed imagery for wetland inventory both nationally and internationally. Within Australia various imagery based inventory programs have been successfully undertaken (eg Blackman et al 1995) and others proposed or initiated (eg Milne 1997). Opposition is not as well publicised, but seems to revolve around problems of scale, costs and unfamiliarity with the technological options.

Reviews of remote sensing applications in wetland environments for inventory, mapping species composition and estimating biophysical properties have been confined to passively imaged data in the continental United States (eg Carter 1977, 1978, Bartlett & Klemas 1981, Butera 1983, Hardisky et al 1986, Gross et al 1989, Dobson et al 1995, Zhang et al 1997) and actively imaged data in forested and tropical wetlands (eg Hess & Melack 1995). With the exception of Hardisky et al (1986), Gross et al (1989) and Hess and Melack (1994), these reviews have focused on approaches to inventory and mapping species composition, mainly in inter-tidal wetlands dominated by grasses and sedges. Hess and Melack (1994) and Hess et al (1990) conducted a review of synthetic aperture radar remote sensing of wetland environments, noting their ability to establish a range of structural and biophysical variables in forested wetlands. At the 1996 meeting of the International Geosphere and Biosphere Program (IGBP) wetlands monitoring taskforce, a series of reviews were completed on the applicability of current remote sensing techniques to questions concerning a global wetlands inventory (Sahagian & Melack 1997).

In the Australian context, apart from Johnston and Barson (1993), Blackman et al (1995), Hess and Melack (1995) and Wallace and Campbell (1998), there are no publications in the refereed literature providing a comprehensive review of the suitability of the current range of remote sensing data types and techniques for monitoring Australian wetlands. Given the advances in sensor technology and processing routines since Gross et al (1989), it is time for an up-to-date review.

To develop a suitable review covering the types of information able to be extracted from remotely sensed data for a wetland inventory in Australia the following structure was applied. Prior to identifying the range of data sets, considerations for selecting remotely sensed data for environmental inventory or monitoring are outlined. It is also useful to categorise the information produced on wetland environments from remotely sensed data into three types: i) delineation and inventory; ii) mapping wetland types and species composition; and iii) mapping biophysical properties. The data sets identified as capable of providing this information included, aerial photography, hand-held instruments, airborne imaging sensors – optical, satellite imaging sensors – optical, hyperspectral sensors, airborne imaging sensors – radar/microwave, satellite imaging sensors – radar/microwave, and GPS surveys.

A range of processing techniques, capable of producing the three main types of information produced were identified – manual interpretation and digitising, spectrometry and radiometry, spectral mixture analysis, image classification, landscape pattern analysis and spatial statistics, and deterministic and empirical biophysical models. A summary of findings is presented, including a table containing a listing of potentially applicable remotely sensed data sets and an assessment of their scales, availability and suitability to wetland inventory, compositional mapping or estimation of biophysical properties.

2 Considerations for selecting remotely sensed data for wetland inventory and monitoring

Remotely sensed data are now used in almost all terrestrial ecosystems as a source of information to identify land cover types and make inferences or estimates about the condition or structure of the surface and vegetation cover. Acquiring and processing remotely sensed data to address resource monitoring questions for wetland environments presents more of a challenge as opposed to other terrestrial ecosystems (Gross et al 1989, Hess & Melack 1994). Perhaps the main difference and complicating factor is surface water in wetlands and its movement due to tides, flood events and storm surges. In many coastal wetlands, intertidal vegetation will be completely covered at some stage in the tidal cycle, restricting the ability to detect its presence and condition. Although the fluctuation in water levels may create a problem for mapping vegetation, the extent and flow patterns of water in wetlands is an important variable able to be remotely sensed. Other problems pertain to coastal locations and their potential for extensive cloud cover in the tropics and mid-latitudes, and fog cover on coasts with cold offshore currents. Other general problems encountered in previous applications of remote sensing techniques to wetland environments have been: requirements for a multi-disciplinary approach; variability of flooding depth and duration; transitional nature of wetland ecotones; range of scales (vernal pools-estuaries-catchments); and variety of wetland types (Gross et al 1989, Jensen et al 1993, Hess & Melack 1994, Mertes et al 1995).

Remotely sensed data and spatial analytic techniques are capable of providing information on vegetation structures from local to regional scales. Two problems limit the application of these techniques: i) identifying suitable spectral, radiometric, spatial and temporal data resolutions; and ii) defining analytic techniques to provide appropriate information for specific monitoring objectives and wetland environments. Both of these 'scaling' problems result from not utilising prior knowledge on the forms and processes controlling an environment's spatial structure, to select and interpret data (Graetz 1990, Ustin et al 1993).

Addressing these problem requires that systematic consideration is given to characteristics of the environment(s) to be examined and the type of information required. The fundamental types of information able to be extracted from remotely sensed data are inventories of wetland location, maps of the internal composition and estimates of their biophysical parameters (Asrar 1989). A summary listing of processing techniques applicable to each type of information is provided in Table 1. Spatial resolution concerns ground resolution element (GRE) dimensions and image extent. GRE should be selected based on the smallest feature or minimum mapping unit required for a specific wetland. Image extent will depend on the area to be monitored and availability of georeferenced image mosaics. For example, the ANCA (1996) Directory of Important Wetlands provides a basis for estimating the minimum and maximum extent of typical wetland environments in Australia (see Spiers & Finlayson 1999). Spectral dimensions concern the spectral wave-bands from which image data can be acquired to maximise the probability of discriminating wetland from non-wetland vegetation, differentiating different species and estimating biophysical properties. For example, colour infrared (CIR) aerial photographs and images acquired in the near-middle IR have been found to be most useful for delineating wetlands and identifying their internal composition (Federal Geographic Data Committee 1992, Taylor et al 1995). Radiometric resolution pertains to the precision with which an imaging sensor records reflected or emitted electromagnetic radiation (EMR). Assessment of radiometric resolution requires the magnitude and variation of reflected EMR from different types of wetland vegetation and ground cover to be measured (Phinn & Stow 1996 a,b). Temporal resolution pertains to the optimal time of phenological cycle, hydrological cycle, tidal cycle or diurnal cycle at which to acquire image data to maximise potential for discriminating the target from the surrounds.

Table 1 Processing techniques applicable to remotely sensed data to obtain information on wetland environments.

Wetland delineation or composition	Wetland Configuration	Biophysical parameters
Classification	Semivariance	Empirically derived relationships between spectral and ground data
Segmentation	Spatial covariance	Deterministic modelling of a physical process using remotely sensed data as input
Spectral mixture analysis	Spatial frequency transformations	
Air photo interpretation	Wavelet transformations	
	Scale variance	
	Spatial filtering	
	Texture measures	
	Landscape structure metrics	
	Individual patch metrics	

3 Applications of remotely-sensed data in wetlands

3.1 Wetland delineation and inventories

Wetland delineation involves identifying the spatial and temporal extent of areas that meet one of the ‘established’ definitions of a wetland. Several of these definitions are:

US Fish and Wildlife Service (Cowardin et al *1979)*

> Wetlands are lands transitional between terrestrial and aquatic systems where the water table is at or near the surface, or the land is covered by shallow water...Wetlands must have one or more of the following three attributes: at least periodically, the land supports predominantly hydrophytes; the

substrate is predominantly undrained hydric soil; and the substrate is non-soil and is saturated with water or covered with shallow water at least some time during the growing season of each year.

Ramsar Convention (Davis 1994)

Wetlands are areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish, or salt, excluding areas of marine water, the depth of which at low tide does not exceed six metres.

Paijmans et al (1985)

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

For the purposes of this work, the Ramsar Convention definition is accepted, as it also provides the basis for the classification in *Directory of Important Wetlands in Australia* (ANCA 1996). However, as illustrated by Spiers and Finlayson (1999) the Paijmans et al (1985) definition better reflects the jurisdictional responsibilities of the management agencies.

Wetland inventory occurs after delineation and involves applying a classification system to determine the extent and composition of a wetland based on the selected classification system. The hierarchically structured classification system of Cowardin et al (1979) revised in Cowardin and Golet (1995), provides the basis for most wetland classification systems in use, including the Ramsar Convention (Scott & Jones 1995), Paijmans et al (1985) and ANCA (1996). A geomorphic classification system for Australian wetlands has been proposed by Semeniuk (1987) and recently extended to the Ramsar Convention (Semeniuk & Semeniuk 1998).

Delineation and inventory projects have been implemented at a number of spatial scales, with specific objectives at each scale:

- Global – focus on presence/absence in specific continents and islands (eg Mathews 1990)
- Continental – distribution of regions within continents or islands dominated by wetlands (eg north slope of Alaska – Wilen & Bates 1995)
- Regional – scale of coastal wetlands predominance (ie dominant regional scale ecosystems – Jensen et al 1986, 1993, Harris 1994, Johnston & Barson 1993)
- Local – individual wetlands (eg Phinn and Stow 1996a, 1996b, Phinn et al 1997)
- Site – variability within wetlands or micro-scale wetland features such as billabongs and coastal plain lakes.

The majority of approaches to wetland delineation and inventory have been at regional to local scales using air-photo interpretation techniques and standardised classification systems. Previously, satellite image data were considered too coarse for accurate local-regional scale delineation and inventory. However, continued refinements in spatial resolution of satellite sensors, coupled with more refined spectral resolution and integration with active data sets, have enabled some detailed inventories to be completed (eg Dobson et al 1995).

Examples of operational wetland delineation and inventory programs include:

- National Wetlands Inventory of the United States Fish and Wildlife Service (Wilen & Bates 1995) – a regional scale approach, with a 10 year cycle, based on manual interpretation and digitising from 1:60 000 to 1:130 000 colour infrared air photos to produce maps using the Cowardin et al (1979) scheme on 1:100 000 topographic map sheets.
- The CCAP (Coastal Change Analysis Program) of the United States National Oceanic and Atmospheric Administration is also a regional scale approach, with a five year cycle, to map land cover change (including wetlands and submerged aquatic vegetation) along selected coastal catchments of the coastline using Landsat Thematic Mapper data and a modified version of the Cowardin et al (1979) scheme (Klemas et al 1993, Dobson et al 1995).

3.2 Mapping wetland types and their internal composition

Once the extent of a wetland has been defined, the next stage in providing information essential for resource managers is to determine the internal composition within the wetland (eg vegetation, landform and hydrology) and then map their boundaries. Mapping approaches typically employ a classification scheme developed specifically for wetlands (eg Cowardin et al 1979, ANCA 1996) or for the environments within which wetlands occur (eg Dobson et al 1995).

The spatial scale of this activity depends on extent of the wetland, detail of information required on vegetation (eg individual plants, associations or communities) and the minimum mapping unit. For very small wetlands, eg coastal plain lakes and billabongs, the extent of wetlands is less than 1 km^2 and the minimum mapping unit necessary to delimit patches of vegetation types useful for resource monitoring is 1.0 m^2. Larger wetlands can extend from 100–1000 km^2 and require minimum mapping units greater than 10^4 m^2 (1 ha).

Aerial photographs, mainly IR and false colour CIR, were the first and continue to be most extensively obtained data source for producing maps of the vegetation composition within small (< 1 ha) to medium (>10 000 ha) size wetlands (Johnston & Barson 1993, Taylor et al 1995). Photographs for each wetland are typically joined in a mosaic and then subjected to manual interpretation to produce an overlay of vegetation class polygons. The final map overlay is often digitised and tagged with attributes for each polygon to produce a coverage for use within a geographic information system (GIS). At larger scales (> 100 km^2) optical image data from sensors such as the Landsat Multi-Spectral Scanner and Thematic Mapper, SPOT HRV, Indian Resource Satellite – LISS, and Japanese Earth Resources Satellite have been used in image classification procedures to produce regional to continental scale maps of vegetation composition (eg Dobson et al 1995). Airborne multi-spectral imaging systems and now digital cameras are being used in the same context for local to regional scale applications (Jensen et al 1986, Stow et al 1996, Jupp et al 1986). Most recent developments include: i) application of hyperspectral imaging sensors (eg Zhang et al 1997) with spectral un-mixing algorithms to map fractional cover; and ii) applying active imaging sensors (eg Hess & Melack 1995, Imhoff et al 1997) alone or with optical image data to produce vegetation composition maps based on structural and water coverage differences.

3.3 Mapping biophysical properties

Because the amount of EMR (electromagnetic radiation) reflected or emitted from land surfaces and plants depends on their biological, chemical and physical properties, remotely sensed data may be ‘inverted’ to estimate some of these properties (Asrar 1989). Some of these properties include: mineral fractions in soils; sediment content in waters; bathymetry; water surface temperature; and vegetation structure, chemical content and growth. Approaches taken to estimating biophysical properties from remotely sensed data range between two types, empirical or deterministic. Empirical approaches rely on regression based relationships established between image values (calibrated radiance or reflectance) and the property of interest, eg live biomass (eg Gross et al 1989, 1990, Phinn & Stow 1996b). Deterministic approaches use physically based models, eg for evapotranspiration, CO_2 flux, and ecosystem function, where remotely sensed data provide one of the input parameters (eg Bartlett et al 1990, Costanza et al 1990, Sklar & Costanza 1991, Jacquemond & Baret 1990).

The spatial scales at which biophysical properties have been estimated in wetland environments range from individual plants or stands (height, cover, live/dead biomass, N content) to entire wetland complexes (biomass, CO_2 flux, methane emissions). Estimation of biophysical properties closely follows the ‘scaling’ problem in earth systems science, ie how to scale up or link plot scale processes/structures to landscape and biome scales. Initial biophysical work in wetland environments began with hand-held radiometers in saltmarsh environments where plot level relationships were established between spectral vegetation indices and live/dead biomass and cover amounts in different structural groups of wetland plants (eg Hardisky et al 1983 a,b). These provided a basis for application of airborne and spaceborne imaging sensors using the same spectral bands to estimate these quantities over larger areas. Later developments included application of canopy/leaf models, to entire wetlands. Similarly, models based on hyperspectral data for estimation of canopy chemical content and transpiration purposes are still research orientated (Ustin et al 1993, Zhang et al 1997). Biophysical remote sensing has received an increased impetus of late from the focus of global change research and role of wetlands as sources/sinks of greenhouse gases, especially methane. Most recent developments are application of processing models to radar data to estimate structure and hydrodynamic properties of wetlands (eg Hess & Melack 1994, Mertes et al 1995, Imhoff et al 1997). Related models using remotely sensed data as an input source, with a GIS environment providing database for all input/output display and analysis, and actual model execution (eg by ARC AML’s or by program scripts in C/Fortran), eg ecosystem productivity or functions, and wetland dynamics, such as spatial explicit dynamic models operating on raster cells or landscape units (Sklar & Costanza 1991).

4 Available data sets

In the following paragraphs a representative sample of existing and planned commercial remote sensing systems are reviewed in relation to potential wetland inventory and monitoring purposes. Each system is described in terms of its purpose, type of information, status (research, operational), and includes examples.

4.1 Aerial photography

Camera systems used for acquiring photographs of wetland environments range from standard 35 mm and metric cameras to large format and panoramic cameras. Differences between these systems affect the field of view and geometric integrity of photos. Further variations in photographic data depends on the altitude at which photos are acquired and the type of film and filters. Lower altitude photographs provide greater spatial resolution, down to scales of 1: 1000 (eg 0.235 km and 0.05 km^2) for examining individual stands or plants, and can extend to 1:50 000 high altitude photographs, that provide regional coverage (eg 11.75 km by

11.75 km, 138 km^2). Different film types add a spectral dimension, enabling panchromatic (black and white) or colour photos of visible wavelengths, and black and white near-infrared and colour infrared (green, red and NIR). Photographic prints or transparencies may be scanned (at a suitable resolution, eg 200 microns) to produce digital format images, able to be geometrically corrected and subjected to image processing operations.

Digital multi-spectral cameras are now commercially available and being used extensively for airborne imaging operations in the United States and Europe (Stow et al 1996). If processed appropriately these systems have the geometric integrity of aerial photographs and the spectral and radiometric capabilities of multi-spectral image data. Their main advantage in the context of wetlands applications is that they have all the characteristics of analogue aerial photographs, but are already in digital format. In addition, digital camera images may be subject to radiometric processing operations commonly limited to digital satellite data. Image data can be acquired by these systems for GRE dimensions from 0.5 m to 5.0 m. Individual frames can be processed to provide a seamless mosaic for an area.

The main purpose of camera systems has been to collect analogue data for use in manual interpretation work that may later be digitised as a vector coverage or scanned in as raster. Such operations provide a basis for discriminating different surface cover types, vegetation communities or landforms and processes based on established interpretation cues at specific scales.

Historically, aerial photography has been the predominant source of remotely sensed data used in wetland inventory and monitoring applications (Anderson & Wobber 1973, Wilen & Bates 1995). Prior to the widespread availability of commercial satellite data, aerial photographs provided the basis for wetland inventory work undertaken from local to regional and national scales in a number of different wetland environments (eg Stanton 1975, Carter 1977, Cowardin et al 1979). Due to the relatively coarse scale of Landsat and Spot multispectral image data and the ecological-local scale basis of most wetland classification schemes, aerial photographs have remained dominant in wetland inventory. Earlier image data sets (eg Landsat) could not be used to consistently identify and classify wetlands, in both the US (Federal Geographic Data Committee 1992, Taylor et al 1995) and Australian contexts (Johnston & Barson 1993). Other advantages of aerial photography for wetland inventory include: control over the timing of data acquisition (clouds, tidal and flood cycles); ability to acquire repeat coverages; ability to extract information on relief; control over scale of acquisition; and access to existing and updated aerial-photography data bases of local, state and federal government bodies in Australia.

There has been limited systematic consideration of the potential role(s) that the next generation of high spatial resolution satellites and digital camera systems would perform in a national wetland inventory. Aerial photography is: time consuming to process; insensitive to structural and sub-canopy properties; has limited application for quantitative estimates of biophysical properties or their change over time; and is not considered cost effective for a national wetland inventory (Dobson et al 1995, Wilen & Bates 1995, Taylor et al 1995, Stow et al 1996). Hence, consideration should be given to the type of wetland classification system applied to remotely sensed data along with high spatial resolution, multispectral satellite and digital camera data. Wetlands and their internal composition are best detected through reflectance features in the infra-red portion of the spectrum according to the Federal Geographic Data Committee (1992) and Gross et al (1990) and in combination with microwave images to provide data on structural and sub-canopy elements (Hess & Melack 1994, 1995). With the spatial resolution of new satellite sensors approaching resolution used in aerial photography, consideration could be given to a hierarchical approach, in inventory and classification, utilising coarse scale data at the broadest level and moving down to finer scale digital data, and analogue if required (Blackman et al 1995, Dobson et al 1995, Taylor et al 1995).

4.2 Hand-held instruments (radiometers and spectrometers)

A radiometer is any instrument recording the strength of electromagnetic radiation incident upon its collection optics. 'Radiometer' normally refers to broad-band radiometer, which can be fitted with various interference or absorption filters to determine the wavelengths of light incident on the sensor. 'Spectral radiometers' or 'spectrometers' are narrow band radiometers, recording the strength of reflected EMR from 10 to 256 narrow bandwidths. If the response of a sensor can be calibrated to a known source of EMR at different levels, output can be produced in spectral radiance and reflectance for targets.

Radiometers are used to acquire information on the spectral reflectance characteristics (radiance or reflectance) of surface cover types in the field or in the laboratory (Curtiss & Goetz 1994). This enables acquisition of spectral reflectance information under controlled atmospheric and surface conditions. By controlling acquisition parameters, several important advantages are gained:

- atmospheric interference effects are minimised and/or can be measured
- data can be from different view angles
- the structural, condition and biophysical characteristics of surface cover type can be collected at the same time as spectral information
- data can be acquired from pure or mixed cover types
- repeated visits to same site in the field over time
- laboratory measurements can be used with precise control on illumination and other factors to acquire data coincident with airborne or spaceborne imaging of a site.

For the purposes of wetland monitoring these data provide a basis for determining spectral reflectance characteristics of different surface cover types and factors that control variation in these characteristics (Gross et al 1989, Phinn & Stow 1996b). Specifically, collecting ground radiometric data enables control of the surface cover structural, condition and biophysical characteristics and its spectral reflectance characteristics can be established. This provides an initial assessment of the utility of remotely sensed data to discriminate between wetland vegetation cover types and to estimate biophysical properties of these environments (Ustin et al 1993). Hand-held radiometer and spectrometer data also provide information necessary to fine-tune remotely sensed investigations of wetland environments. By measuring atmospheric conditions at the time of data acquisition the effect of varying amounts of cloud cover, water vapour and illumination geometry on the spectral reflectance characteristics of different surface cover types can be established. Acquiring spectra at different viewing angles enables the effect of off-NADIR views and interaction with illumination geometry and surface cover type to be established. Acquiring reflectance spectra from pure and mixed cover types provides a basis to test the spectral band(s) in which they exhibit significant differences. Repeated visits to the same site in the field over a day or growing season may help to determine the time to best acquire image data to maximise the potential for discriminating different cover types or estimating a biophysical property. Finally, by acquiring radiometer or spectrometer data coincident with airborne or spaceborne imaging of a site, ground data provide a basis for atmospheric correction and calibration of image data.

Hand-held radiometry and spectrometry is a fully operational activity, with several different types of radiometers and spectrometers being made commercially (eg Curtiss & Goetz 1994). Specific applications have focused on the applications outlined above, mainly for individual plants – from one to hundreds of square metres. Disadvantages associated with this approach pertain to the small area covered on the ground and the ability to scale measurements made at this scale to minimum sample units in satellite imaging systems.

4.3 Airborne Imaging Sensors – Optical

Airborne platforms including piloted aircraft, remotely piloted vehicles, helicopters and balloons contain a scanning or framing sensor, capable of acquiring images with GRE between 0.5 m and 30 m, over areas of one to hundreds of square kilometres, in a limited number of spectral bands. A scanning sensor utilises a laterally oscillating field of view (FOV) to provide across flight line coverage and platform movement provides along flight path movement. Multi-spectral capability is provided by different sensor elements for each pixel. In framing sensors an array of CCDs instantaneously acquires an image line and is displaced to the next line by movement along a flight path. Further details on the status of this type of image data are provided in the context of a global wetland inventory (see Sahagian & Melack 1997).

Multi-spectral scanners provide high to medium spatial resolution multi-spectral image data in visible, short wavelength IR and TIR bands. Image data are processed using ground information and laboratory tests to produce radiance and reflectance images. With geometric and radiometric processing these data may be joined together to produce image mosaics for larger areas then subject to image processing algorithms to delineate cover types or examined in other ways to estimate biophysical and biogeochemical properties (eg macrophyte production in Jensen et al 1986 and projective foliage cover in Phinn et al 1997).

A similar set of criticisms may be established for airborne scanner systems, as were identified for aerial photography. Specifically, the spatial resolution and multi-spectral data able to be achieved by these sensors will soon be available from the next generation of commercial small satellites. In addition, the new satellites will provide much larger area coverage, and permit construction of regional to national scale mosaics.

Advantages of airborne scanner data for wetlands applications include: scale specificity for smaller wetlands; an ability to obtain data when requested and when suitable atmospheric or tidal conditions become available; minimal atmospheric interference; and a capability for calibration to ground data reference data as a basis for scaling between plant/patch/community/wetland scales and multi-temporal analyses.

Due to the reliance of these sensors on reflected sunlight limitations to their applications are caused by cloud cover, atmospheric moisture and haze. Data acquisition may be restricted for wetlands in areas subjected to continual cloud cover or fog during specific times of the year. This may be offset by their ability to be mobilised for image acquisition at short notice. Inherent problems with the scanning geometry and 'hotspot' effects limits the geometric and radiometric utility of these sensors for producing mosaics of larger wetland sites. Due to the nature of reflectance from wetland vegetation types, these sensors portray canopy structure, chemical and moisture content and provide limited ability to penetrate the canopy to establish volumetric information or sub-canopy information, eg detection of flooded forests.

4.4 Satellite optical imaging sensors

Digital multi-spectral imaging systems on polar orbiting satellite platforms provide regional to global scale coverage at repeat cycles from twice daily to approximately once monthly.

These sensors (eg Landsat multispectral scanner [MSS] and Thematic Mapper [TM], SPOT-MSS and Indian Resource Satellite [IRS]-1C) deliver medium (10–30 m) to coarse (30–80 m) spatial resolution multi-spectral image data in visible, short wavelength IR and thermal IR bands. Image data are processed using ground information, satellite ephemeral data and atmospheric conditions to correct for geometric and atmospheric distortions to the spatial and radiometric integrity of the data. As with airborne multi-spectral sensors these data are then subject to image processing algorithms to delineate cover types or examined in other ways to estimate biophysical and biogeochemical properties. Refer to Sahagian and Melack (1997) for a more detailed review in relation to global wetland inventory capabilities.

Dominant controls on the type of information able to be extracted from satellite images is dependent on their GRE and the type of classification selected. Spatial resolution refers to minimum dimensions of the sensor's sampling element on the ground, ie the area from which reflected or emitted EMR is measured, referred to as GRE or pixel dimensions. Interaction with landscape features determines the smallest feature visible on an image. Trial applications of these sensors for wetland inventory, mapping internal composition and biophysical properties have been carried out in most wetland types around the world (eg Johnston & Barson 1993, Blackman et al 1995, Dobson et al 1995, Mertes et al 1995) with a consensus that they may only be useful for regional overview and delineation, but not for mapping species composition unless used in association with aerial photography or ground calibration (Federal Geographic Data Committee 1992, Taylor et al 1995). This may be in part due to the classification systems being used being based on characteristics of wetland and sub-divisions that aren't able to be detected in image data, eg species composition. Klemas et al (1993) established a modified version of the Cowardin et al (1979) and Wilen and Bates (1995) classification to enable Landsat Thematic Mapper data to be applied in the national Coastal Change Analysis Project (Dobson et al 1995). The CCAP project aims to establish the condition and changes in coastal watersheds over a five year basis and the classification scheme enables delineation of cover types including wetlands to the sub-system level.

Applications of Landsat TM data to Australian wetlands have been evaluated by Johnston and Barson (1993) in inland wetlands in Victoria and New South Wales, and by Blackman et al (1995) in north Queensland. Results from these studies and other less extensive reviews highlight the utility of Landsat TM data for reconnaissance mapping, delineating wetland extent, monitoring water regimes and classification of internal variation based on structural and functionally based classification approaches. Both manual and digital image processing methods were considered applicable for these tasks. Collection of remotely sensed data should be designed to complement existing wetland inventory data sets where possible, and be applied to the most appropriate level in a classification scheme (Blackman et al 1995, Taylor et al 1995, Zoltai & Vitt 1995).

Limitations to using satellite based image data for national wetlands inventory will be related to the availability of suitable image data sets at required times of the year, cloud cover effects and price. In terms of appropriate spectral bands to utilise, the Federal Geographic Data Committee (1992), established that wetland delineation (for saltmarshes, mangroves and forested wetlands) was most effective when using Thematic Mapper bands 4 and 5. This concurs with findings of Johnston and Barson (1993) for inland wetlands in New South Wales and Victoria. Digital images are not the only data source found to be effective for wetland delineation, hard copy plots of appropriate image bands were also considered useful means of manually delineating wetlands. An existing cloud-free Landsat MSS mosaic of Australia composed of images acquired between 1990 and 1992 images is available in digital and hardcopy format. Cloud free TM data sets are available for Queensland in 1988, 1991, 1995 and 1997. Cloud free TM images may be purchased from the Australian Centre for Remote Sensing, or as data exchange for other state government offices. Either the Landsat TM or MSS image base would provide a suitable basis for wetland delineation and reconnaissance (for wetlands > 1.0 – 5.0 ha). Restrictions may apply based on the time of year they were collected if information on inundation extent is required.

Table 2 Characteristics of existing and 1997/1998 launch satellite optical sensors (based on Kramer 1994, Fritz 1996, Morain & Budge 1994)

Sensor application	Pixel size (m)	# Bands	Revisit	Wetlands
Landsat TM	30	7	16	D C B
SPOT PAN	10	1	26(1–4)	D
SPOT XS	20	3	26(1–4)	D C B
SPOT PAN/XS 20		3	26(1–4)	D C B
NOAA AVHRR	1100/4000	5	0.5	D B
CZCS	825	6	?	D B
SMRR	27–105 km	5	Varies	D C B
IRS 1C	23.5	4	24	D C
JERS-OPS VNIR	18	3	24	D C
JERS-OPS SWIR	24.2	4	24	D C
RESURS-0	45 x 35	3	16	D C
Planned satellite sensors				
Landsat ETM+PAN	15	1	16	D C
Landsat VNIR/SWIR	30	5	16	D C B
Landsat LWIR	60	1	16	D C B
Smallsats				
Space imaging				
PAN	1	1	14 (1–3)	D B
MS	4	4	14 (1–3)	D C B
Earthwatch				
PAN	3	1	< 5	D B
MS	15	3	20	D B
Clark				
Worldview	3	3	20	D B
Earth observation stations				
ASTER VNIR	15 x 25	3	5	D C B
ASTER SWIR	30	6	16	D C B
ASTER TIR	60	5	16	D C B

Sensor = type/name of imaging system; **Pixel size** = G.R.E dimensions; **#bands** = Number of spectral bands images area collected in, (eg green, red and infra-red); **revisit** = Minimum time between successive image acquisitions for the same area; **Wetlands application** = Listing of the wetlands monitoring techniques the image data have been applied to in an operational (c/f. research) basis, where D refers to delineation and inventory, C = mapping internal composition and B= estimating biophysical properties.

Documentation of the extent and success of remote sensing applications for wetlands monitoring in Australia has been limited to the ‘grey-literature’ of internal publications, conference proceedings and several articles in scientific/technical journals. Pressey and Adam (1995) present several of these applications in a review of wetland inventory and classification in Australia. Manual interpretation of aerial photography provided the basis for

Stanton's (1975) reconnaissance of significant wetlands in Queensland, and also for the CSIRO maps of the Australian coastline and its natural resources (Galloway et al 1984, Paijmans et al 1985, Wood & Cocks 1990). In each application using aerial photography a different interpretation scheme was applied. Apart from the afore-mentioned work of Johnston and Barson (1993), Landsat TM data have also been used extensively in Queensland and the Northern Territory for inventory and composition mapping over a wide range of different wetland types. Blackman et al (1992, 1995) presents a hierarchically structured classification scheme for Queensland wetlands based on the Cowardin et al (1979) scheme. This scheme is stratified by biogeographic zones and relies on TM data to provide a basis for wetland delineation and mapping internal community composition. Aerial photographs and field checks then provide more detailed checks. The establishment of statewide Landsat TM coverage for Queensland in 1988, 1991, 1995 and 1997 as part of the statewide land and tree survey (SLATS) by the Department of Natural Resources provides a logical image base to apply this classification to the entire state. Mangrove communities along the coast of Queensland are currently being mapped by applying image classification techniques to Landsat TM data (Danaher & Luck 1991). Coarse scale NOAA-AVHRR data have been collected in the Northern Territory and processed to monitor the extent of ephemeral wetlands in the sub-humid tropics (Pressey & Adam 1995). Landsat TM data continue to be used by the Parks and Wildlife Commission of the Northern Territory to map the internal composition of coastal floodplain wetlands and assess changes to their condition over time associated with removal of buffalo and invasive weeds (Whitehead & Chatto 1996). The variety of different data sets and approaches was noted by Pressey and Adam (1995), as being necessary in light of the variety of wetland types and scales of their spatial and temporal variability.

The 'next generation' of commercial resource monitoring satellites should be given serious consideration as potential sets for the Australian wetland inventory because of their high spatial resolution (GRE <= 15m), large area coverage, multi- to hyper-spectral configuration, radiometric precision, availability and cost. Sensors to be launched from August 1997 and into 1998 include the Lewis hyperspectral instrument, Earthwatch Earlybird, Space Imaging Systems and Orbview. With the exception of Lewis these sensors are part of commercial groups designed to provide high quality image data for environmental monitoring applications on a global scale. Of particular concern to the wetland inventory is that these sensors will provide image data down to the scales able to be obtained from aerial photography, hence they address one of the primary limitations for applying satellite image data in wetland inventories. The high spatial resolution satellite data may still not be able to separate vegetation communities with similar spectral responses, but delimiting smaller patches and structures will be possible. These sensors may provide aerial photographic scales and temporal resolution with satellite multi-spectral and large area coverage, enabling smaller wetlands to be detected (< 1 ha) and their internal composition to be estimated. Test data sets for these sensors have been generated from multispectral digital camera systems and applied in several wetland environments (over much smaller areas than a typical satellite scene). Successful geometric and radiometric calibration of these data sets demonstrated their utility for mapping cover types within them and estimating their biophysical properties in saltmarshes and mangroves (Phinn and Stow 1996a, 1996b, Jupp et al 1986).

4.5 Hyperspectral imaging sensors

Imaging spectrometer systems are currently carried on aircraft and will soon (as of 1999) be carried on a satellite. These systems operate in the same mode as optical sensors discussed in the previous sections, but collect reflected and emitted EMR in at least 20 narrow spectral bandwidths. The large number of spectral bandwidths enables a complete spectral signature to be established for each pixel element within an image. Hence, detailed analyses can be conducted on the atmospheric column constituents of each pixel, surface composition and surface biogeochemical elements (Goetz 1992, Vane 1993, Curtiss & Goetz 1994). Data sets from imaging spectrometers occupy much larger volumes, as image cubes, ie instead of having 4-8 spectral bands per pixel there may be up to 240 spectral bands. Geometric distortions are similar to other scanning and solid state sensor systems, and may be corrected from aircraft/satellite ephemeral data and GCPs. Radiometrically, image values may be converted to sensor and to surface radiance and reflectance using modelled atmospheric parameters (to extract interference absorption/scattering, eg MODTRAN) (Vane 1993). Due to the increased data dimensionality different image processing and analysis procedures have been applied to hyper-spectral data sets (c/f. multi-spectral). The most commonly applied algorithms are for spectral-unmixing, to provide information on the type(s) of feature present at the surface and its fractional cover of each element within each pixel (Roberts et al 1993, Adams et al 1995). Sahagian and Melack (1997) contains a review of hyper-spectral imaging applications for global wetlands inventory and monitoring.

Operational monitoring applications in wetland environments are not common for hyperspectral imaging sensors due to their limited availability and coverage of existing data sets. The majority of hyperspectral data for wetland monitoring applications in Australia have been collected from the NASA-AVIRIS (airborne visible and infra-red imaging spectrometer) sensor and the Itres Inc. CASI (compact airborne spectrographic imager). The AVIRIS sensor is limited to pre-scheduled flights, mainly in the continental USA, and typically acquires images with 20 m GRE. The CASI sensor provides images with pixels < 0.5 m and up to 10 m, but only for narrow width images, but has been used in a variety of wetland environments (MacCleod et al 1995, Held et al 1998, Green et al 1997, Zhang et al 1997). With the launch of the EOS, ENVISAT and ARIES satellites and their hyperspectral imaging sensor projected for 1999–2000, hyperspectral data may be available over more geographic areas and more readily. Due to the experimental nature of the processing and data acquisition involved with Hyperspectral sensors, further assessment is required to determine their suitability to operational wetlands monitoring and inventory.

Table 3 Characteristics of operational and planned hyperspectral sensors

Sensor	Platform	Pixel size (m)	#Bands	Range (nm)
Operational sensors				
AVIRIS (NASA)	Air	20	224	380–2500
CASI (Itres) image	Air	0.5–10	19	418–926
CASI (Itres) spectrometer	–	Varies	288	418–926
DAIS (GER)	Air	2–30	79	400–12700
HYDICE (US Navy)	Air	Varies	206	400–2500
Planned				
LEWIS HIS (TRW/NASA)	Air	30	256	900–2500
MODIS (NASA-EOS)	Satellite	250	2	600–900
		500	5	460–2200
		1000	29	0.4–14.3µm

4.6 Airborne and satellite radar

Synthetic aperture radars (SARs) are active sensors operating in the microwave region (roughly 1 mm to 1 m in wavelength). Unlike passive sensors which measure radiation from natural sources such as reflected sunlight, SARs both transmit and receive pulses of specific wavelength and polarisation; they thus operate independently of solar illumination. Operating at much longer wavelengths than optical sensors, imaging radars can penetrate clouds and smoke and are sensitive to structural elements of vegetation canopies such as leaves, branches, and boles. They are particularly well suited to wetland inventory and monitoring because of their ability to remotely detect flooding beneath vegetation canopies. The following sections will briefly review SAR data sources, microwave scattering mechanisms, and results of SAR studies of wetlands in Australia and elsewhere.

4.6.1 SAR system characteristics

SAR instruments operate from both airborne and spaceborne platforms and are characterised by their band and polarisation (Table 4). Satellite SAR sensors are currently limited to single-frequency, single-polarisation systems, either C-band (5.6 cm) or L-band (23.5 cm); airborne systems also operate at X-band (3 cm) and P-band (65 cm). Radars transmit plane-polarised waveforms, oriented either horizontally (H) or vertically (V), and then receive one or both polarisations. The satellites listed in Table 4 all record a single polarisation, either HH (horizontal send, horizontal receive) or VV. Horizontal send, vertical receive (HV) is currently available only from airborne SARs. Incidence angle refers to the imaging geometry of the radar. It is equal to the angle between the radar beam and a line perpendicular to the ground surface, and may be fixed or variable.

Table 4 Synthetic aperture radar (SAR) systems

Platform	Satellite			Space shuttle		Aircraft
Sensor	ERS-1/2	Radarsat	JERS-1	SIR-C/X-SAR		JPL AIRSAR
Operator	Europe	Canada	Japan	USA/Germany/Italy		USA
Radar band	C	C	L	C L	X	C L P
Polarisation	VV	HH	HH	HH VV HV	VV	HH VV HV
Pixel spacing (m)	12.5	6.25–50	12.5	12.5	12.5	3–12
Swath width (km)	100	50–500	75	15–40	15–40	6–12
Repeat cycle (d)	35	1–24	44	*	*	< 1
Incidence angle	23	20–50	35	20–50	20–50	15–60
Launched	1991	1995	1992	1994		1988

Bands refer to wavelength: X (3 cm), C (5.6 cm), L (23.5 cm), and P (65 cm). H and V are horizontal and vertical polarisations. Nominal resolution is generally 1.5 to 2.5 times larger than pixel spacing. Asterisks denote 11-day SIR-C missions flown in April and October 1994.A planned third SIR-C mission will generate digital elevation models for most of the earth's land surfaces using interferometry. Airborne SAR systems are too numerous to list; the Jet Propulsion Lab AIRSAR is given as an example.

After pulses transmitted by a SAR sensor are reflected, scattered, and/or absorbed at the earth's surface, the intensity and timing of the energy scattered back toward the sensor (backscattering) are received and recorded. The brightness of an object in a SAR image corresponds to its radar backscattering coefficient $\sigma°$. Because of the large dynamic range of SAR systems, the unitless $\sigma°$ is normally expressed in decibels ($\sigma°_{dB} = 10 \log \sigma°$). The signal detected by SAR is the coherent sum of signals from randomly distributed scatterers within an image pixel. Random constructive and destructive interference in the addition of these signals causes variability in $\sigma°$ among pixels, even for homogeneous targets. The resulting salt-and-pepper appearance, called speckle, poses problems in digital classification due to the high within-class variance of targets. Speckle is reduced during signal processing by multiple-look summing and can be further reduced during image processing by median or other filters.

4.6.2 Microwave interaction with water, soil and vegetation

SAR wavelengths are very long compared with atmospheric constituents, so they are not significantly scattered or absorbed by the atmosphere as are visible and infrared wavelengths. The longer SAR wavelengths (L- and P-bands) are virtually unaffected by clouds or rain, while the shorter wavelengths can penetrate all but the densest cloud (C- and X-bands) and rain (C-band). Scattering from most earth surfaces usually involves a combination of surface scattering, where the medium encountered by the radar wave is homogeneous or nearly so (eg a water surface, and to a first approximation, a soil surface), and volume scattering, where the medium is inhomogeneous (eg a vegetation canopy). For surface scattering, the roughness of the surface determines the angular radiation pattern of the scattered wave, while the relative complex dielectric constant of the surface determines the strength of the scattered wave (Ulaby et al 1981). The smoother the surface relative to the radar wavelength, the greater the coherent specular component reflected away from the radar. The rougher the surface relative to the wavelength, the greater the diffuse component backscattered to the radar.

The dielectric constant of a material is a measure of how absorptive or reflective it will be of an incident wave; for most natural surfaces, dielectric constant is a function of water content. Because of the high dielectric constant of liquid water, moist soils, for example, are more reflective than dry soils. In volume scattering, the density and dielectric constant of scatterers within the volume, such as leaves and branches within a forest canopy, determine the scattering strength, and the angular scattering pattern is a function of the boundary surface roughness, the average dielectric constant of the medium, and the sizes of the scattering objects in the volume (Ulaby et al 1981). The contrast between herbaceous and woody vegetation is greater at longer wavelengths.

Two smooth surfaces oriented perpendicular to one another, such as a paved surface and a building, constitute a corner reflector: the specular reflection from the first surface is directed back toward the radar by the second surface, causing a strong return. These double-bounce returns are the mechanism for enhanced backscattering from flooded trees or macrophytes (Richards et al 1987). Specular reflections from the smooth, highly reflective water surface are bounced back toward the radar by vertically oriented trunks, branches, or stalks. Double-bounce reflections also occur in unflooded situations, but returns are much weaker because scattering from an unflooded soil surface has a much greater diffuse than specular component, and is less reflective because of its lower dielectric constant.

Trunk-ground or canopy-ground double-bounce returns can occur only when the radar penetrates the canopy to reach the ground; extinction of the radar signal by absorption and scattering within the canopy volume can prevent this if the canopy layer is sufficiently dense or deep. Longer wavelengths penetrate further into canopies than shorter ones, so L-band is more likely than C-band to penetrate a forest canopy.

4.6.3 Wetlands studies using SAR

Smooth water surfaces specularly reflect SAR pulses away from the sensor, resulting in very low backscattering. Open water surfaces can thus usually be delineated accurately with any of the systems in Table 4. The principal source of error is non-specular returns caused by wave-induced surface roughness, which in the worst cases can cause confusion between land and water surfaces. Mapping of open water area of rivers and lakes has been demonstrated for the Amazon River (Sippel et al 1992) and the Mississippi River (Brakenridge et al 1994).

Many studies (reviewed in Hess et al 1990) have found increases in L-band returns from flooded forests, for a wide range of stand densities. Using Shuttle Imaging Radar-B data to study *Eucalyptus camaldulensis* forests of the Murray River floodplain, Richards et al (1987) found increases of about 10 dB in LHH returns due to flooding. The increase in backscattering caused by forest flooding is greater at LHH than LVV, and greater at L-band than at C-band (Wang et al 1995). The effect of flooding on SAR returns is more variable for herbaceous than for woody vegetation. Pope et al (1997) found SIR-C backscattering from Yucatan marshes increased due to flooding at both CHH and LHH for tall, dense stands but decreased for short sparse stands. The increased backscattering from flooded vegetation, and the wavelength-dependent differences in returns between woody and herbaceous vegetation, can be used to classify SAR images into categories useful for wetlands mapping. SIR-C data has been used to delineate flooded and nonflooded vegetation and open water on a reach of the Amazon floodplain with accuracies greater than 90%, and to quantify the change in inundated area accompanying a change in river stage (Hess et al 1995). LHH was found optimal for separating flooded from nonflooded forests (about a 3 dB difference), CHH for inundated versus upland grasses, and LHV for woody versus nonwoody vegetation. Rosenqvist et al. (1998) used multi-temporal JERS-1 imagery to model floodplain inundation and methane emissions for a central Amazon black water river.

4.6.4 Australian AIRSAR campaign

The Joint NASA/Australia AIRSAR deployment in November 1993 acquired airborne SAR data over 55 Australian sites, some of which involved wetlands (Milne 1997). A second AIRSAR deployment in November 1996 covered many of the wetland sites that were flown in 1993. Changes in backscatter across a gradient in vegetation structure at a floodplain site near the South Alligator River in the Northern Territory were documented by Imhoff et al (1997). Multi-polarised C-, L-, and P-band data were acquired over vegetation transects that extended from perennially wet *Melaleuca cajuputi* woodland through seasonally flooded *M. cajuputi* and *M. viridiflora* woodland to mixed *Eucalyptus* woodland. At all wavelengths, VV- and HV-polarised returns were more highly correlated with structural parameters than were HH returns, and scattering from crown elements was dominant relative to scattering from boles or bole-ground interactions. The dominance of crown scattering was probably due to the long path length through the crown layer, a function of the very shallow incidence angles used (52° to 57°).

Taylor et al (1996) were able to clearly delineate the distribution of saline soils in the Tragowel Plains Irrigation Area of Victoria, by identifying areas with anomalous dielectric constants using an inversion technique. Identification of saline versus nonsaline soils was better at L-band than at C- or P-bands. These results are relevant to identification of saline wetlands, and indicate that algorithms for wetland delineation using SAR need to take salt-affected soils into account. However, Acworth (1997) reported that due to variability in surface roughness and soil moisture variations, efforts to map saline sites in the Yass Valley Catchment of New South Wales were not successful. In one of the few published SAR studies of grasslands, Hill et al (1997) found strong relationships between $\sigma°$ and herbage cover.Total fresh biomass was highly correlated with the combination of $\sigma°_{CVV}$, $\sigma°_{LVV}$, $\sigma°_{PHV}$, and $\sigma°_{PVV}$. Discrimination of vegetation differences was poor with all HH channels.

Magela Creek SIR-C study

The Shuttle Imaging Radar-C, or SIR-C (Table 4) was the first spaceborne multi-frequency, multi-polarisation SAR. Hess (1998) studied wetland communities of the Magela Creek floodplain, Northern Territory, using multi-frequency HH- & HV-polarised SIR-C data. Because that study focused exclusively on Australian wetland vegetation, it will be reviewed in detail. The acquisition dates of the two SIR-C missions (April and October 1994) corresponded to the late-Wet and late-Dry seasons. It was therefore possible to characterise backscattering both when the floodplain was completely inundated with most aquatic macrophytes at peak biomass, and when floodwaters had receded and macrophytes were largely senescent. Helicopter-based, oblique video surveys carried out on 20 April 1994 and 18 October 1994 (Devonport et al 1994, Waggitt et al 1995) were used to verify ground conditions during the two SIR-C acquisitions. A 27 x 38 km scene centred on Magela floodplain was extracted from SIR-C data takes 117.6 (April) and 117.52 (October). The two scenes were coregistered and filtered using using two interations of a 5x5 median filter. Centre incidence angles were similar for the two data takes: 42^{o} for April and 39^{o} for October.

Backscattering signatures consisting of $\sigma°$ at CHH, CHV, LHH, and LHV were extracted for training and test polygons, using the video record as a basis for polygon location. A decision-tree model was used to generate rules for classifying both scenes into five classes: open water, non-flooded grassland or bare ground, flooded aquatic macrophyte, non-flooded *Melaleuca* and eucalypt woodland, and flooded *Melaleuca* woodland. In addition, sub-types of macrophyte and *Melaleuca* were identified on SIR-C colour composites and interpreted using both the video survey and the Magela floodplain vegetation map of Finlayson et al (1989).

Evaluation of backscattering signatures and classification accuracies strongly indicated the usefulness of multifrequency SAR in three areas:

1. Mapping of woody and herbaceous vegetation in flooded and nonflooded states

Misclassification rates for test pixels were < 10% for all five vegetative/hydrologic categories, on both dates. Correspondence between areas classified as *Melaleuca* with mapped areas of *Melaleuca* on the vegetation map supported the accuracy of the woody/herbaceous classification. Flooded herbaceous and woody vegetation were easily separable using LHH: with the exception of *Nelumbo nucifera*, median $\sigma°_{LHH}$ for macrophytes was at least 6 dB lower than for *Melaleuca* on both dates. The difference in median $\sigma°_{LHH}$ between nonflooded woodland and flooded *Melaleuca* ranged from 2.8 to 6.7 dB. These large differences, consistent under both late Wet and late Dry season conditions, suggest that mapping of vegetative/hydrologic classes can be accomplished with high accuracy using multifrequency SAR.

2. Discriminating among aquatic macrophyte communities

Subtypes of flooded woodland and macrophyte could be discriminated with accuracy on both dates. If the two lowest-biomass macrophyte types are grouped together, all subtypes had accuracy rates greater than 96% in April, with the exception of *Nelumbo* (83% accuracy). In several cases there was a strong correspondence of macrophyte types with mapped communities, particularly with *Nelumbo nucifera, Pseudoraphis* grassland, and *Hymenachne-Eleocharis* swamp. The correspondence between classified and mapped communities resulted from differences in canopy structure related both to phenologic state and to species morphology. *Pseudoraphis* grasslands were distinctive because they were in an early emergent state in April, with a shorter and sparser canopy relative to the other communities. *Nelumbo* was distinctive because of its large stalks and leaves (lower accuracy rates for *Nelumbo* resulted from confusion with *Melaleuca,* not with other macrophyte types). On both dates, the ratio of CHV to CHH returns was greater for *Hymenachne-Eleocharis* swamp than for other macrophyte types; this difference was probably related to stem orientation.

3. Monitoring phenologic change in macrophyte communities

Several communities exhibited striking differences in SAR response between April and October. Median backscattering from *Pseudoraphis* grasslands south of Leichhardt Billabong increased by 7.4, 9.2, 11.1, and 7.3 dB at CHH, CHV, LHH and LHV; for a typical *Hymenachne-Eleocharis* swamp, CHH returns were unchanged between April and October, while CHV returns decreased by 2.6 dB, and LHH and LHV returns increased by 8.1 and 7.7 dB; and for an area of *Oryza* grassland near the East Alligator floodplain, σ° decreased by 5.7, 7.2, and 3.7 dB at CHH, CHV, and LHV but increased by 2.1 dB at LHH. In general, multi-temporal variability between macrophyte communities was higher at C-band than at L-band. The exception was *Nelumbo*, for which median $\sigma°_{LHV}$ was 5.6 dB lower in October. The change for *Pseudoraphis* resulted from increased height and canopy cover. Changes in the other communities seemed to be related mainly to senescence, which caused differences in stem and stalk angles, and affected the balance of canopy attenuation by leaves and reflections from stems and stalks.

Until multi-frequency SAR satellites become available, satellite monitoring of wetlands using SAR will rely on data from the single-frequency and polarisation instruments ERS-1/2 (CVV), Radarsat (CHH), and JERS-1 (LHH), used singly or in combination. To evaluate the effect of eliminating one or more of the four band/polarisation combinations used in the study, classifications were carried out on the April data using subsets of the complete set of radar parameters (Table 5). The results indicate that either Radarsat or JERS-1, used singly, could distinguish with fair to good accuracy between flooded and non-flooded woodlands, but would have a very poor ability to discriminate flooded macrophytes. With the combination of Radarsat and JERS-1, however, expected accuracies would be fair (70%) for flooded macrophytes and very good (> 90%) for the other categories. Performance for flooded macrophytes using Radarsat + JERS-1 could be further improved by using multi-date sequences. Although CVV (the ERS-1 configuration) was not part of the Magela SIR-C dataset, past studies have found CVV to be significantly less useful than CHH for wetlands studies, either alone or in combination with LHH (Hess et al 1994).

4.6.5 Considerations for SAR inventory of Australian wetlands

The findings described above, of high classification accuracies and good correspondence of SIR-C vegetation patterns with mapped floodplain vegetation, indicate the suitability of multi-frequency SAR data for detecting flooding, phenologic state, and in some cases plant community for floodplains such as those in the Kakadu region. In judging whether such results are applicable on a wider scale, several factors must be considered.

Sensor and platform

Multi-frequency, multi-polarisation SARs are currently available only on airborne systems. While airborne SARs are useful for small-area studies and for research, their narrow swath width and large cross-swath incidence angle effects make them unsuitable for inventories at a regional or countrywide scale. Although multi-temporal coverage can greatly improve the accuracy of classifying with single-frequency data, the results in Table 5 indicate severe limits to single-frequency data for some classes. Combining data from different SAR satellites approximates a space-based multi-frequency capability, although it should be noted that the process of accurately co-registering large datasets from different satellites is non-trivial and can be time-consuming (Kellendorfer et al 1996). High accuracies have been achieved using ERS-1/JERS-1 composites for land-cover classification (Dobson et al 1996). Because the difference in backscattering between flooded and nonflooded vegetation is generally greater at HH than VV polarisation, however, the combination of Radarsat and JERS-1 is preferable for inundation monitoring. Accuracies using combinations of satellite SAR datasets would be lower than for the simulation in Table 5, owing to calibration and misregistration errors, and to non-simultaneous acquisition.

Spatial and temporal resolution

SAR satellites have spatial resolutions comparable with or better than optical satellites. Repeat coverage can be obtained every 24–44 days at high resolution. The Global Rain Forest Mapping project (GRFM) of Japan's NASDA has acquired JERS-1 data sets of the earth's major rain forest areas, demonstrating that it is feasible to image even very large areas such as the Amazon basin at two different seasons and to mosaic the thousands of scenes involved into a single dataset at 100 m resolution (Hess et al 1998). A large portion of Australia north of 20^{o} S was also imaged as part of the GRFM in 1996–97. The Radarsat SAR is able to image wide (up to 500 km) swaths at 50 m resolution; using the RADARSAT ScanSAR mode, frequent coverage (every 1 to 4 days) is possible for large regions. This capability is particularly useful in inventorying seasonal and ephemeral wetland types.

Wetland type

Good classification results from the Magela, the Amazon, and other floodplain wetlands indicate that a broad range of wetland types can be mapped at a structural level (herbaceous/woody, flooded/ nonflooded) using at least CHH and LHH. However, it has yet to be determined how broadly a single classification algorithm can be applied, and regional algorithms may be necessary. Some wetland types present in Australia such as ephemeral lakes have not been studied with SAR. Potential problems include distinguishing between open water and flat dry sandy areas. Although some studies have found it possible to map inundation beneath mangroves (Imhoff et al 1986), others have noted mangrove forests that were indistinguishable from upland forests. Owing to the variability in tree height and inundation regime for mangroves, further studies are required to determine possible limitations of SAR for mangrove mapping.

Terrain effects

Because of effects such as shadowing and layover caused by radar imaging geometry, digital terrain models should be used with SAR data in areas with significant relief.

Complementarity of SAR and optical data

While SAR can penetrate clouds and vegetation canopies, it cannot detect some features that are easily distinguishable with optical sensors such as Landsat Thematic Mapper, eg sediment in water, and submerged vegetation such as seagrasses. Because SAR and optical sensors together span a large range of the electromagnetic spectrum, when used in combination they respond to a much greater range of vegetation characteristics than either sensor alone.

Table 5 Correctly classified test pixels (%) for radar variable subsets derived from Magela Creek SIR-C data, April 1994 (Hess 1998)

	Open water	Bare ground	Flooded macrophyte	Non-flooded woodland	Flooded Melaleuca
CHH	98	26	22	94	77
LHH	100	99	32	82	90
CHH+LHH	98	100	70	97	90
LHH+LHV	100	89	55	94	95
CHH+CHV+LHH+LHV	98	98	92	97	92

4.7 Global Positioning System (GPS) surveys

Global Positioning System (GPS) data are an integral part of any project applying remotely sensed data for mapping the location of wetlands, defining their internal composition and estimating some of their biophysical properties. For each application the GPS unit is required to record the location of a point as horizontal coordinates and vertical displacement from an established geodetic system and datum, ie its geographic coordinates and elevation. For inventory and cover type mapping applications GPS surveys provide data on the location of wetland boundaries and polygons of specific wetland vegetation cover types. Assuming the image data set or existing wetlands map are georeferenced the GPS data can be viewed at the same time to visually assess accuracy, or perform automated data extraction and checking. Development of biophysical models, especially empirical models requires accurate spatial referencing of field data collection points, to co-locate with image data from the same area and develop appropriate models.

5 Processing techniques for wetlands monitoring

Processing remotely sensed data to extract further data or information relevant to defining wetland boundaries, mapping their internal composition or estimating biophysical properties requires application of the appropriate technique and considerations of their input requirements and limitations. The following sections provide an overview of the range of techniques that have been successfully applied to remotely sensed data to produce information for micro to global scales for environmental monitoring. These techniques may also be applied in a multi-temporal context to detect change or map dynamic properties, and requirements are discussed for implementing them as such.

5.1 Manual interpretation and digitising

Visual interpretation of aerial photographs has been the most frequently applied methodology for delimiting wetlands and mapping their internal composition over a wide range of spatial scales and types of environments (Gross et al 1989, Finlayson & van der Valk 1995, Green et al 1996). Pre-defined wetland classification schemes are used to provide a basis for a series of interpretation keys, usually only applicable to a set range of wetland types, and specific scales and types (eg colour or infra-red) photographs (Cowardin & Golet 1995, Blackman et al 1992). Reconciling this approach with the range of wetland types available in Australia would necessitate a very general classification scheme, one capable of being implemented using the next generation of satellite data. At large scales, ie areas of limited spatial extent, aerial photographs still provide optimal data sets for establishing topographic and vegetative boundaries in wetlands, as well as their internal composition, often down to a species level (Federal Geographic Data Committee 1992). Specific scales of photographs may be selected from existing coverages generated by federal, state and local agencies, corresponding to appropriate levels within a hierarchically structured classification system (eg Blackman et al 1992, Scott & Jones 1995, Paijmans et al 1985).

Interpretation practices vary depending on the type of film used for interpretation, with infrared, colour and colour-infrared being the most successfully applied from 1:100 to 1:50 000 scales. Two types of interpretation procedures are commonly followed. In the first, standard photographs (23.5 cm x 23.5 cm) or enlargements are analysed by trained interpreters using a pre-defined classification scheme (and field notes), polygons delimiting wetlands or relevant classes of cover are traced onto mylar film, prior to digitising into a GIS for final map composition. The second approach utilises aerial photographs that have been scanned into digital format (at high spatial resolution, eg 300 um). By displaying the scanned photographs using image processing or GIS software, polygon boundaries can be digitised directly from the photograph (heads up digitising). This approach still uses an interpretation key, but also enables the scanned photographs to be subject to correction processes to remove geometric distortions inherent in aerial photographs and to construct mosaics for the area of interest (Jensen 1996).

Limitations of aerial photography for wetland inventory and monitoring concern the cost of extensive photo-acquisition runs, the time required and errors introduced in manual delimitation, and problems of normalising photos from different dates (removing variations in solar geometry and intensity) to quantify changes in wetland extent, composition or biophysical properties (Johnston & Barson 1993, Jensen 1996, Stow et al 1996, Green et al 1996). Manual delineation and interpretation of high spatial resolution digital camera data and next generation satellite data, may provide information equivalent to that for 1:5000 photographs for digital cameras (0.5 m pixels) and 1:125 000 photographs for high spatial resolution satellites. These data sets can also be obtained for extensive areas in georeferenced mosaics, may be resampled to larger pixel sizes, and are capable of radiometric calibration for estimating biophysical properties and their changes over time (Haines-Young et al 1993, Kramer 1994).

5.2 Hand held spectrometry and radiometry

Processing techniques applied to radiometer and spectrometer data sets provide information on the spectral reflectance characteristics (radiance or reflectance) of surface cover types in the field or in the laboratory (Asrar 1989). Most successful applications to wetland environments have been based on hand-held measurements made in saltmarshes and observations from light planes in mangroves and forested wetlands. In both cases plot level results provided relationships capable of ‘scaling-up’ to larger pixels of satellite sensors, hence testing the types of vegetation and cover types able to be spectrally discriminated or estimate biophysical properties for (Gross et al 1989, Jensen 1996, Phinn et al 1996b, Zhang et al 1997). In relation to monitoring wetlands environments several specific questions can be addressed:

1. The control of the surface cover type’s structural, condition and biophysical characteristics on its spectral reflectance characteristics can be established (determine spectral bands for discrimination or estimation of a biophysical parameter).
2. Repeated visits to same site in the field over a day or growing season may help to determine the time to best acquire image data to maximise the potential for discriminating different cover types or estimating a biophysical property.
3. By acquiring radiometer or spectrometer data coincident with airborne or spaceborne imaging of a site, these ground data provide a basis for atmospheric correction and calibration of image data.

Output from radiometers and spectrometers is processed with sensor gain/offset and calibration coefficients to produce spectral radiance and spectral reflectance from calibration panels. Useful information may then be extracted for radiometer data from graphical plots of signatures for cover type, accumulated statistics for multiple measurements to define cover type variance and statistical analysis in association with solar geometry or biophysical data. For spectrometers, extraction of information is facilitated by graphical plots of voltage, radiance or reflectance for each spectral band produces a spectral signature curve; visual comparison of spectral curves; automated curve matching routines for use with spectral libraries for discrimination of surface cover type; spectral unmixing of component signals to provide fraction of sample area occupied by each cover type, mineral or chemical composition; statistical measures of curve separability in different spectral bandwidths using analysis of variance, variance measures and derivative analysis; and statistical analysis in association with solar geometry or biophysical data

5.3 Spectral mixture analysis

Spectral mixture analysis (SMA) or spectral unmixing was developed to address the ‘mixed pixel’ problem. Because the size of the ground sampling element on imaging systems is often large in relation to surface cover patches and these patches are not internally homogenous, a mixture of surface cover types produces pixel response (digital number). The goal of SMA is to apply reflectance or radiance spectra obtained from homogeneous areas of each cover type (endmember) to determine the fraction of each pixel occupied by a cover type. SMA was developed from factor analytic inversion techniques in chemistry and optics to identify independent sources of variability (Adams et al 1995). Initial remote sensing applications were in semi-arid environments by Pech et al (1986), Huete (1986) and in forested to wetland environments by Ustin et al (1993), Adams et al (1995), Mertes et al (1995) and Sippel et al (1992).

The principle of the SMA approach (for linear mixing) is presented below:

1. Define endmembers (scene structure and number of bands)

2. Aim is to solve for the fraction of each endmember in a pixel

Fraction images provide more intuitive assessment of scene structure and applicability for mapping.

$DN_c = \Sigma^N_{i=1} F_i DN_{i,c} + E_c$

where,

$\Sigma^N_{i=1} F_i = 1$

DN_c = uncalibrated radiance in channel c of image pixel

N_i = Number of endmembers

F_i = Fraction of endmember i (parameter to solve for)

$DN_{i,c}$ = radiance/ reflectance of endmember i in band c

E_c = Residual or error for channel based on the fit of N spectral endmembers

SMA techniques have only recently been applied to wetland environments in a number of published research projects. Forested wetlands, inundation and turbidity levels have been examined using this technique and Landsat TM data (Mertes et al 1995) and microwave data (Sippel et al 1992). Results from these studies demonstrate the utility of SMA for single and multi-date mapping of the fractional cover of end-members (eg vegetation species, communities, live/dead biomass, surface moisture, inundation, and turbidity levels), as well as biophysical and biogeochemical information.

5.4 Image classification approaches

The common goal of the following algorithms, loosely grouped as classification approaches, is to identify groups of pixels with similar spectral reflectance values and assign a label to each group as a type of landcover. That is, their end goal is to produce a thematic map of surface cover types. By compiling image maps of the same areas based on a common classification scheme, but using images collected on successive dates in time (days, weeks, months, stages in tidal/flooding or phenological cycles), maps of change and wetland dynamics may also be produced (Graetz 1990).

Per-pixel classification routines use both parametric and non-parametric classification algorithms to evaluate whether each pixel is assigned to an image class (eg parallelepiped, minimum distance to means, maximum likelihood). Application of the routines is either by a supervised approach where the analyst identifies groups of pixels to be used as training sites, or an unsupervised approach where a data clustering routine is used to identify groups of similar pixels in spectral space. This approach is the most widely applied, simple, flexible, applicable to different data types, computationally non-intensive, and able to be fine tuned to an appropriate image data set and environment. However, its principal disadvantage relates to input data requirements (normal distributions), mixed pixel problems, mis-classification, minimum mapping unit size. Classification algorithms have provided the basis for delimiting wetlands and mapping their internal composition from Landsat TM data (eg Klemas et al 1993, Johnston & Barson 1993, Harris 1994, Blackman et al 1995), airborne scanner data (Jensen et al 1986) and digital camera data (Phinn and Stow 1996a, 1996b).

Image segmentation applies region growing routines that examine pixel digital numbers and texture values to grow segments up to specified dimensions (Woodcock & Harward 1992, Shandley et al 1996). Segments are labelled using a per-pixel classification and dominance/plurality rules. This approach does require knowledge of the spatial structure of existing ground cover types, ie, typical patch size and/or hierarchy of sizes. No examples were found of wetland applications for these approaches in the literature, although they may provide a useful approach to mapping wetlands with complex internal structures.

Each classification procedure requires multi-spectral digital image data or fraction images (produced from SMA) and varying degrees of information on the number of image classes required and their spectral variability and spatial extent. Non-remotely sensed data may also be used as input in the classification process, if it is in a conformal coordinate system and spatial resolution. For example, digital elevation and soils data have been used to improve the accuracy of wetland delineation and separation of low, middle and high marsh vegetation zones. Multi-sensor data sets, eg, optical and radar data sets may also be subjected to image classification approaches, as successfully demonstrated by Hess and Melack (1994, 1995). Output from these applications are thematic maps used as input into GIS database for multi-temporal analyses and also as the basis for further modelling, using the image data in each cover type or models that require information on the area of each cover type.

There are several essential considerations to be made before applying classification techniques to wetland environments. First, the size of wetland vegetation and landscape elements (eg patches and communities) should be able to be defined by the image sampling element dimensions (pixel or GRE size). Definition of landscape features within an image requires the GRE to be a least 1/10th the linear dimensions of a feature. The number and placement of available spectral bands should be sufficient to detect differences between wetland cover types. Finally, is it possible to produce a map of the required covered types within acceptable error levels, taking into account the nature of the wetland landscape and the number of image classes required.

Multi-temporal analyses of changes in extent, composition or biophysical properties of wetland environments may be achieved by several modified classification approaches. Direct differencing of radiometrically normalised images acquired at two dates for the same area can be used to produce a difference image (Jensen 1996). A classification approach may then be applied to group areas with similar changes and assign them labels. The most commonly applied approach, based on images subject to the same classification systems, is post classification comparison (Jensen et al 1993, Jensen 1996). Other approaches based on multi-temporal classification work that have been successful include examining trajectories to produce maps of landscape dynamics (Graetz 1990).

5.5 Landscape pattern analysis and spatial statistics

Applying landscape pattern analyses and spatial statistics can yield quantitative information on the spatial structure of the landscape (ie its configuration) from either an unprocessed multi-spectral image or from an image map of cover types (Turner & Gardner 1991, Rossi et al 1992). To define the size, shape, adjacency, frequency and connectivity of different landscape elements. Algorithms in this area can be broken into two groups, those that define dimensions of landscape elements based on image data (spatial structure functions) and those that define dimensions and patterns based on raster or vector based digital maps raster (pattern metrics).

Algorithms grouped under spatial structure functions include spatial statistics such as semi-variance, scale-variance and power spectrum analyses. Scale variance analyses establish the total variance at increasing block (pixel window) sizes and presents the results on a plot of variance versus block size. This enables the effects of varying GRE size to be established in terms of the pixel size or feature size at which most variation occurs on average in the landscape (Woodcock & Strahler 1987). Semi-variance analysis is based on regionalized variable theory and examines variance levels between pixels separated at increasing distances to determine at what distances these values are similar or dis-similar. Output from semi-variance analysis at each distance interval (lag) is plotted on a semi-variogram. Like scale variance analysis, this approach facilitates an assessment of the dominant scales of spatial variation, ie feature dimensions, in a landscape (Curran 1988, Woodcock et al 1988). Output from power spectrum analyses can be used to identify scale(s) of repeated patterns in the landscape. In these approaches two dimensional Fourier transforms are applied to decompose data by spatial frequency, rather than just dominant patterns or structure (Smith et al 1988).

Pattern metrics have been developed in landscape ecological applications to provide quantification of landscape structure dimensions, particularly the dimensions of patches of individual cover types and their arrangement in the landscape and in relation to each other (Turner & Gardner 1991, Turner et al 1991, McGarigal & Marks 1994). Examples of patch dimensions, commonly calculated for individual patches of a specific cover type include: area (mean and variance), core area; perimeter; shape (perimeter:area, fractal dimension); density; edge; and diversity (compositional variation within patches). Spatial statical functions provide the basis for measures of pattern, including contagion, interspersion (scale of aggregation/dispersion) and clustering. A review by Riitters et al (1995) of 55 different landscape metrics applied to 85 USGS air-photo interpreted land use maps established redundancy between many indices. Up to 87% of the variance in land-use pattern was able to be accounted for by the following six metrics: average perimeter-area ratio; contagion; standard patch shape; patch perimeter area scaling; number of attribute classes; and patch density area scaling.

To date there have only been several published results of landscape structure analyses in wetland environments based on spatial statistics and pattern metrics (Mertes et al 1995, Phinn & Stow 1996b). Spatial statistics and pattern metrics have been applied extensively in non-wetland environments (Turner & Gardner 1991, Haines-Young & Chopping 1996) and warrant consideration for providing quantitative dimensions of landscape pattern in wetlands. However, attention should be paid to the limitations of these approaches before applying them. Specifically, statistical assumptions for their application and significance testing (stationarity, sinusoidal variation, gridded data, regular periodicities) and the fact that many of the measures of spatial association were not developed for data dense and contiguous data sets (eg remotely sensed images). Results will also be dependent on how classification units were derived and the scale at which analyses are conducted.

5.6 Deterministic and empirical biophysical models

The common goal of the following approaches is to provide estimates of biophysical or biogeochemical properties over an area for output as a thematic map or as input into a dynamic model. Biophysical properties able to be estimated from remotely sensed data include: vegetation density (Gross et al 1989); vegetation cover (Gross et al 1989); plant basal area and height (Phinn et al 1997); plant biomass (live, dead, above, below ground) (Ustin et al 1993); plant productivity (Hardisky et al 1983 a,b, Gross et al 1989); vascular versus non-vascular plants (Roberts et al 1993); and soil cover versus non-photosynthetic vegetation.

Complete inversion of remotely sensed data relates the measured reflectance, absorption and transmittance characteristics of the scene element to its physical dimensions or biophysical properties. For vegetation patches this may include estimating the horizontal and vertical structure of plants along with the amount of live and dead biomass present. Two approaches are used to invert the data, the first is a statistical or empirical approach whereby spectral data and corresponding physical data are collected and a mathematical form of relationship is derived using regression analysis (eg NDVI and biomass). Applications of airborne and satellite sensor data to estimate biomass in wetlands was provided by Gross et al (1989). In the physical or deterministic approach an existing understanding of the physical interaction between EMR and the property of interest is used specify a model of their relationship (eg latent heat transfer). Goel (1989) and Strahler and Jupp (1991) provide detailed reviews of the components, applications and limitations of various types of geometric-optical, turbid-medium and simulation models for estimating plant structural characteristics. Franklin et al (1993) applies geometric-optical models to estimate shrub canopy sizes, while Morris (1989) uses a turbid-medium model to examine light diffusion in the canopy of wetland grass.

The role of GIS in providing an environment for model development, testing, execution and display and analysis of results should also be established (Haines-Young et al 1993). These roles include data storage and retrieval (graphic and database); functioning as a 'repository' of knowledge, able to be continually updated; providing functional capabilities for executing models if operating on a raster cell or polygonal basis for computations (ie simple AML – C script). Specific advantages include their ability to implement spatially explicit dynamic models to examine spatial variations in model output, eg for sea-level rise, coastal subsidence and/or other ecosystem dynamics and to facilitate integration with other non-remotely sensed data sets.

To assess biophysical characteristics such as, height, density, cover, biomass and productivity, hand-held radiometers were initially used to determine spectral characteristics of wetland vegetation and their controlling factors (Gross et al 1989). Once the nature of these controls was established, empirical relationships at the scale of the radiometer footprint were established between a structural characteristic of the plant and its spectral reflectance characteristic (Drake 1976, Hardisky et al 1983 a,b). Work by Hardisky established the main controls on wetland vegetation's spectral reflectance characteristics to be the amount of live and dead leaf area in the horizontal and vertical planes. Empirical relationships have been difficult to apply and obtain sound results due to complicating factors of: solar elevation; amount of live/dead plant matter; substrate type; standing water and wind stress (Bartlett et al 1988). More success in providing stable estimates of biophysical parameters has come from use of deterministic approaches in canopy reflectance models for examining light decay in canopies (Morris 1989) and the leaf area and biomass in canopies (Jacquemond & Baret 1990), with limited application beyond plot scales. Although the majority of these modelling application have been in saltmarsh environments (forbs, grasses and shrubs) with passive data sets, results from radar based estimates of structural parameters in forested wetlands suggest the range of wetland environments may be monitored and modelled from remotely sensed data.

6 Conclusions and recommendations

The data types and applications reviewed in the preceding sections indicate that remotely sensed data has the potential to act as major data source for a national wetland inventory and monitoring program for Australia. Several constraints on the extent and form of this application should be recognised. To ensure remotely sensed data are selected wisely and applied to appropriate questions the following issues must be addressed: i) selection of an appropriate wetland classification scheme or means to incorporate all existing schemes; ii) how to incorporate existing inventory data sets; and iii) identifying the intended product(s) of the inventory and their potential applications.

The wetland classification systems developed for Australian environments and reviewed in Pressey and Adam (1995) and Finlayson (1997) exhibit certain commonalities that provide directions for appropriate scales/types of remotely sensed data to use in a national wetlands inventory. Paijmans et al (1985) system was intended to provide a generalised overview of Australian wetlands using a hierarchy of categories-classes (hydrology and climate) and sub-classes (geologic and geomorphic context, position in basin). The system applied in *A directory of important wetlands in Australia* (ANCA 1996) is a simplification of a scheme derived by Scott and Jones (1995), drawing originally from the US National Wetland Inventory approach of Cowardin et al (1979) and Wilen and Bates (1995). For the Australian context this classification employs only three systems: marine and coastal; inland; and human made. A similar hierarchical approach has been adopted for the Queensland wetland inventory by Blackman et al (1992, 1995). A different approach is taken by Semenuik and Semenuik (1995, 1997) to provide a geomorphic basis to classifications focusing on landform setting and hydroperiod (water availability). In a global context the geomorphic classification consists of 13 wetland categories, subdivided by landform and then hydroperiod, with further modifications by wetland shape, size, soils, water salinity and consistency over time. Similar hydro-geomorphic approaches have been applied to a wetland inventory for central and southern California (Ferren et al 1995). With the exception of Blackman et al (1995) and Semenuik and Semenuik (1995), to a lesser extent, no specifications are provided for applying the currently available range of airborne and satellite data sets with each classification system to wetland environments in Australia.

A step towards resolving the classification systems, and preserving suitable detail for the variety of Australian wetland types, is to establish commonalities in each of the classification systems and utilise this as a basis for recommending appropriate remotely sensed data sets. Each classification scheme has a hierarchical structure, with the major criteria for subdivisions, including vegetative and geomorphic structure and position, able to be derived from remotely sensed data. Each classification system may be retained and modified to include processing or interpretation cues for each subdivision, relevant to an appropriate form of remotely sensed data. In this way a similar approach is taken as in the modification of the Cowardin et al (1979) approach for application with Landsat TM data (Klemas et al 1993, Dobson et al 1995). However, in this case the classification systems will be linked to appropriate remotely sensed data set at each hierarchical level.

Table 6 contains a summary of the remotely sensed data sets reviewed in this report and an evaluation of their suitability for providing information on wetland delineation/inventory, internal composition and biophysical parameters. Taking the classification system of Scott & Jones (1995), modified for Australian wetlands, its main categories and subdivisions define the levels of detail of information to extract. The levels within the classification hierarchy do not correspond to a spatial hierarchy, as there are significant variations in minimum and maximum extent of wetlands within each level of the classification (eg coastal, inland and human). The following recommendations are based on the extent of a monitoring or inventory area and the minimum size or minimum mapping unit (MMU) required. Accurate mapping of a MMU requires the image GRE dimensions are less than 1/10 the dimensions of a feature. Data sets were selected as suitable if they received medium-high rankings for previous applications in this area. Applications from regional to national scales, where the MMU > 100 ha are suited to Landsat MSS data. At an intermediate to national scales, applications where the MMU > 9 ha can be met by Landsat TM, SPOT XS, IRS-1C and merged imaging radar from Radarsat and JERS-1. Trade-offs between optical and radar data will be made in tropical and subtropical regions due to cloud cover effects and needs for detecting sub-canopy flooding. Consideration should also be given to the complementarity of optical and radar data sets in terms of their ability to provide a much greater range of information on vegetation characteristics than if they were examined separately. For individual wetlands to national scale applications, with MMU > 1 ha, image data sets from the small, high spatial resolution commercial satellites to be launched within the next 12 months may provide valuable sources of data. Simulation data acquired for these sensors have demonstrated their potential for multi-scale land-cover mapping and estimation of biophysical properties. Finally, at MMU scales < 1 ha, in individual wetlands to regional scales, hardcopy and scanned aerial photography, along with digital camera data from federal and state programs provides an essential longer term high resolution data source.

Selection and application of remotely sensed data for use in a national wetland inventory should take place within the context of existing inventory data sets both statewide and nationally (eg ANCA 1996, Blackman et al 1995) and an appropriate classification system. Every attempt should be made to utilise remotely sensed data at the appropriate scales to complement and extend existing inventory data and ‘fill in the appropriate gaps’. Multiple data types will have to be utilised due to range of different wetland environments and their characteristics. Specific attention should be paid to the approach developed in Blackman et al (1995), in terms of ensuring appropriate local, regional and national data sets are incorporated with remotely sensed data in the process of wetland monitoring and inventory. Selection of remotely sensed data should concern matching the spatial characteristics of the data to the type of environment (and its track record) and to ensuring the ‘best’ processing technique is selected to produce the required information.

Table 6 Summary of Remotely Sensed Data Sets Aplicable to Elements of a National Wetlands Inventory

Data Type	**Coverage**	**Spatial Dimensions**	**Temporal Resolution**	**Wetland Inventory**	**Wetland Composition**	**Wetland Biophysical**
Hand-held radiometers & spectrometers	N/A	0.1–0 m	User defined	N/A	HIGH	HIGH
Airborne Photographs • Colour • Panchrom • IR • Colour IR	Australia (Scale dependent)	1:2 500 1:5 000 1:7 500 1:10 000 1:15 000 1:20 000 1:25 000 1:30 000 1:40 000 1:50 000 1:75 000 1:80 000 1:100 000	Product dependent	HIGH	HIGH	LOW
Airborne Digital Cameras	Selected sites	0.5–5.0 m	User defined	MEDIUM	HIGH	HIGH
Airborne Daedalus Scanner	Not able to establish	2.5–25m	User defined	MEDIUM	HIGH	HIGH
Airborne Imaging Spectrometer	Selected sites	0.1–10 m	User defined	LOW	MEDIUM	MEDIUM
Airborne SAR	Selected sites	5–20 m	User defined	MEDIUM	MEDIUM	MEDIUM
Landsat MSS	Australia Mosaic Individual scenes	 100 m 79 m	 1990-1992 User defined	MEDIUM	MEDIUM	LOW
Landsat TM	Individual scenes Mosaics: QLD NSW/ACT ? VIC TAS SA WA NT	30 m	User defined 16 days min. 1988,91,95,97 ? ? ? ? ? ?	HIGH	MEDIUM	MEDIUM

Table 6 continued

SPOT XS XS/PAN	Individual scenes Individual scenes	20 m 10 m	User defined 1–26 days	MEDIUM	MEDIUM	LOW
IRS 1-C	Individual Scenes	23.5–100 m	User defined	HIGH	MEDIUM	MEDIUM
JERS	Individual Scenes	18–24.2 m	User defined 44 days	HIGH	MEDIUM	MEDIUM
NOAA AVHRR	Australia – mosaic	1.1 km	Monthly	MEDIUM	LOW	LOW
Small-sat Earthwatch	Individual scenes (not active)	4–15m	5 days	MEDIUM*	HIGH*	MEDIUM*
Small-sat Space Imaging	Individual scenes (not active)	4 m	14 days	MEDIUM*	HIGH*	MEDIUM*
Small-sat Lewis-HIS	Individual scenes (not active)	30 m	14 days	MEDIUM*	MEDIUM*	MEDIUM*
Small-Sat Clark	Individual scenes (not active)	15 m	20 days	MEDIUM*	HIGH*	MEDIUM*
ERS-1/2	Individual scenes	12.5 m	35 days	MEDIUM	MEDIUM	N/A
JERS 1	Individual scenes	18 m	User defined 44 days	MEDIUM	MEDIUM	HIGH
RADARSAT	Individual scenes Mosaic for the dry/wet tropics	25 x 28 m 100 x 100 m	1–24 days	MEDIUM	MEDIUM	LOW

Data Type: Lists the available remotely sensed data sets in the categories covered in Section 4.

Coverage: Indicates whether complete data sets are available for Australia or as individual scenes.

Spatial Dimensions: G.R.E. dimensions for georeferenced data sets

Temporal Resolution: Time period over which a complete data set has been collected or the frequency at which images can be collected over a site.

Wetland Inventory/Composition/Biophysical: A ranking is assigned to each data set based on results presented in refereed publications evaluating its suitability for producing one of the three types of information required on wetlands. A similar set of rankings was derived in the IGBP report (Sahagian & Melack 1997).

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Wetland risk assessment

A framework and methods for predicting and assessing change in ecological character

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Abstract

The working definitions of *ecological character* and *change in ecological character* adopted by the Ramsar Convention on Wetlands are discussed. These are used as the basis for addressing early warning indicators for the major types and causes of wetland loss and degradation (changes to the water regime, water pollution, physical modification, exploitation and loss of production, introduction of exotic species). A framework for *wetland risk assessment* encompassing six basic steps (identification of the problem, identification of the effects, identification of the extent of the problem, identification of the risk, risk management and reduction, and monitoring) is presented. This introduces a framework under which management decisions on hazards and risks to wetlands can be assessed. The framework also provides a basis for choosing appropriate early warning systems for monitoring change in the ecological character of wetlands. The latter requires careful attention and linking to the guidelines for describing and maintaining the ecological character of wetlands.

Definitions of *early warning indicators* are presented along with a list of ideal attributes. Examples of such indicators for wetland loss and degradation due to water pollution are described and their relative advantages and limitations are presented.

1 Introduction

The usefulness of early warning indicators for detecting adverse change in the ecological character of wetlands has received increased attention in recent years by the Ramsar Convention on Wetlands. This effort has principally been directed towards sites listed as internationally important, but the concepts can equally be applied to all wetlands. Thus, for convenience in developing these concepts we have focused on the approaches being promulgated under the Ramsar Convention and linked them to a framework for wetland risk assessment. The development of the wetland risk assessment framework also contributes to the elaboration of the strategic directions for wetland management promoted under the Ramsar Convention. Thus, we present the wetland risk assessment framework as an integral component of the management planning processes for wetlands listed as internationally important. In this respect the framework is considered to be inseparable from the wise use, environmental impact assessment and monitoring guidelines already developed by the Convention (Davis 1994, Finlayson 1996a).

The Ramsar processes for assessing and maintaining the ecological character of wetlands comprise many facets. These initially centred on establishing criteria for listing sites as

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internationally important. Criteria were first drawn up in 1974, revised and accepted in 1980 (Conference of the Contracting Parties Recommendation 1.4), with further revisions and additions in 1987 (Recommendation 3.1), 1990 (Recommendation 4.2) and 1996 (Recommendation 6.2). Further, the Convention's Scientific and Technical Review Panel (STRP) has been requested to review the criteria and take note of cultural values and/or benefits that may not be currently included (Resolution 6.3).

Thus, the importance of a site can now be adjudged by using the criteria and completing the Ramsar Information Sheet (Recommendation 4.7 and Resolution 5.3) which is used by Wetlands International as a basis for maintaining the Ramsar site database. The Convention has also addressed the concept of maintaining the ecological character of wetlands with working definitions and guidelines (Resolution 6.1). This resolution encouraged the development of early warning indicators for detecting and initiating action in response to change in ecological character.

In this paper we revisit the definitions of *ecological character* and *change in ecological character* adopted in 1996 (Resolution 6.1), and outlines a framework for using wetland risk assessment to predict and assess change in ecological character. Thus, the wetland risk assessment framework is intricately linked with the monitoring guidelines and framework described in Resolution 6.1. We then describe the attributes of *early warning* indicators that can be used within the risk assessment process, give examples of such indicators, and place them, along with the risk assessment framework, within the context of management planning for the wise use of wetlands.

2 Ecological character and change in ecological character

Article 3.2 of the Convention text, agreed in 1971, introduces the importance of maintaining the ecological character of wetlands listed as internationally important. This is expressed as:

> Each Contracting Party shall arrange to be informed at the earliest possible time if the ecological character of any wetland in its territory and included in the List has changed, is changing or is likely to change as the result of technological developments, pollution or other human interference.

However, it was not until the 1990s that Wetlands International and the Scientific and Technical Review Panel of the Convention attempted to articulate the concepts of ecological character and change in ecological character.

Dugan and Jones (1993) defined the ecological character of a wetland as:

> The sum of the wetland's functions, products and attributes that are derived from the individual biological, chemical, and physical components of the ecosystem and their interactions.

They also defined change in ecological character as:

> The alteration of the biological and/or physical components of the ecosystem, and/or the interaction between them, in a manner which results in a reduction in the quality of those functions, products and attributes which give the wetland value to society.

The STRP recommended similar definitions to the Conference of the Convention in 1996 (see Finlayson 1996a); however, the Conference did not accept these definitions, and instead, adopted different 'working' definitions (Resolution 6.1). In this case, ecological character was defined as:

> The structure and inter-relationships between the biological, chemical, and physical components of the wetland. These derive from the interactions of individual processes, functions, attributes and values of the ecosystem(s).

Change in ecological character was defined as:

> The impairment or imbalance in any of those processes and functions which maintain the wetland and its products, attributes and values.

In line with Resolution 6.1 it is noted that these definitions are interim and may be changed at a later date. The definitions of processes, functions and attributes are the same as those used in Resolution 6.1 and reported in Finlayson (1996a) based on information from several sources.

3 Types of change in ecological character

Finlayson (1996a) presented a summary of the causes of adverse change in the ecological character of a wetland based on an analysis done by Dugan and Jones (1993). This comprised three broad general groups – changes in water regime, physical alteration, and biological change. This grouping is now considered to be too broad and has been modified to also emphasise water pollution, exploitation of biological products and invasion by exotic species. Thus, the major causes of change in ecological character are summarised as being: *changes to the water regime*; *water pollution*; *physical modification*; *exploitation of biological products*; and *introduction of exotic species*.

It should be noted that the relative importance of these causes varies regionally and even from site to site. In addition, it should be recognised that the above causes of change are often highly inter-linked. Thus, identification of the separate effects of each of them can be difficult. A simpler way to view change in ecological character is by the *type* of change as opposed to the cause of change. In accordance with the definition of change in ecological character described above (see Section 2), the following types of change in ecological character can be defined:

- Biological
- Chemical
- Physical

Causes of change can be grouped within these three types. For example, biological change might involve the introduction of exotic species; chemical change could encompass water pollution and also deterioration of water quality due to altered water regimes, physical modification or exploitation; while physical change could include aspects of altered water regimes, physical modification, and possibly exploitation and the introduction of exotic species. It is also recognised that chemical and physical change will ultimately be expressed in terms of biological change. In outlining an appropriate framework and associated methods for the prediction of change in ecological character of wetlands, this paper is primarily concerned with *types* of change. In addition, the types of change of concern are those that are anthropogenic and adverse in nature.

4 Wetland risk assessment: A framework for predicting and assessing change in ecological character

The major goal of this paper is to provide information and guidance on early warning indicators of change in ecological character. A large range of such indicators has been developed for predicting effects due to various stressors, particularly chemical stressors. Many of these techniques have potential for predicting and assessing change in ecological character, and their applications, advantages and limitations are discussed below (see Sections 5 and 6). However, in order to ensure the appropriate application of early warning indicators, it is considered essential that the processes of selecting, assessing, analysing and basing decisions on indicator responses be contained within a structured but flexible form of assessment framework. Rather than design a new framework we propose the use of a modified ecological risk assessment framework, termed *wetland risk assessment*, to support this need.

The following section outlines the general process of ecological risk assessment, and discusses how wetland risk assessment can act as the 'vehicle' for driving the process of predicting and assessing change in ecological character. Following this, we discuss examples and applications of various early warning techniques for use within the wetland risk assessment framework.

4.1 Ecological risk assessment

The last two decades have seen a growing emphasis towards improved, or sustainable management of wetland environments, whereby both ecosystem health and the quality of human life are maintained (Cairns & van der Schalie 1980, Stortelder & van de Guchte 1995). For effective environmental management, an understanding of the type and magnitude of anthropogenically-related stressors that an environment can, or cannot, tolerate is required. In addition, potential effects of such stressors on the environment need to be characterised and weighted against economical and/or societal benefits. A process that serves to achieve these complex objectives is known as *ecological risk assessment* (US EPA 1992).

Ecological risk assessment is related to established approaches to environmental impact assessment (EIA) (D Pritchard pers com). EIA may be defined as a process of predicting and evaluating the effects of an action or series of actions on the environment, then using the conclusions as a tool in decision-making. Its relationship to risk assessment can take two forms. The first involves a continuous (risk assessment) process of identifying vulnerable elements of the environment, and the types of hazards to which they may be exposed. This forms part of the background baseline information which helps with screening and scoping decisions when a detailed EIA is drawn up for a particular proposal. Methods developed in the context of risk assessment, for quantifying the likelihood of impacts, may also be drawn on in the EIA process. In this sense, risk assessment forms a component or components of EIA. The second form of relationship is when information on hazards/stressors and associated risks for a given project or event, built up by a risk assessment process, is drawn from a variety of sources including EIA. In this sense, EIA forms a component of risk assessment. The scope of an EIA is always driven by a definition of the activity or activities being assessed. The scope of a risk assessment may be driven either by a definition of an activity or by a definition of the environmental endpoints of concern.

4.1.1 Definition and structure

Ecological risk assessment can be defined as:

> a structured process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors (US EPA 1992)

Although highly structured, ecological risk assessment is a flexible process for collecting, organising and analysing data, information, assumptions and uncertainties in order to estimate the likelihood of adverse ecological effects (US EPA 1998). As such, it provides a framework that allows effective analysis and decision making based on the analysis, while also providing an adequate mechanism of feedback if and when required. The process of risk assessment can be divided into six steps:

i) Identification of the problem

ii) Identification of the effects

iii) Identification of the extent of the problem

iv) Identification of the risk

v) Risk management and reduction

vi) Monitoring

The details of these steps are explained below (see Section 4.2), with specific application to wetlands. It is important to note that ecological risk assessment has primarily been applied to pollutant effects on ecosystems, and even more particularly on single chemical effects. This issue is also further addressed below (see Sections 4.1.2 and 4.2).

4.1.2 Application of ecological risk assessment for predicting and assessing change in ecological character of wetlands

Ecological risk assessment evolved from human health risk assessment following recognition that protection of human health did not automatically protect non-human health (Suter 1993). The development of risk assessment frameworks for environmental/ecological issues focussed initially on risks of single chemical stressors, most likely because that was also a major focus of human health risk assessment. However, the generalised risk assessment paradigm can most likely also be applied to complex chemical mixtures as well as physical and biological stressors; it is the detail within the steps that will require careful consideration depending on the type of issue being assessed.

In addition to its ability to provide a means of estimating the likelihood of adverse ecological effects, the major benefit of using an ecological risk assessment framework for predicting and monitoring change in the ecological character of wetlands lies in its structured nature. Effective monitoring involves a series of steps, including identification of the issue and the values to be protected, setting objectives, selecting indicators, making conclusions/decisions, and subsequently auditing the effectiveness of the decisions (Finlayson 1996a,b). Ecological risk assessment provides a framework that ensures all these processes are carried out. In addition, and just as important, it provides a mechanism of feedback so that if new information arises or management decisions are not effective, appropriate action can be taken. Thus, ecological risk assessment is an iterative process, and this is represented by the broken arrows in figure 1.

The general ecological risk assessment paradigm is suitable for predicting and monitoring the likelihood of change in ecological character of wetlands. However, in order for this to be realised, the details within the general structure must be appropriate for assessing the types of change experienced in wetlands. This not only includes the types of change in ecological character (ie chemical, biological and physical), but the scales (spatial and temporal) over which they occur, and also how they relate to the Ramsar concept of *wise use* (see Davis 1993, 1994). The following section presents an ecological risk assessment framework, termed wetland risk assessment, which could be used to predict and assess change in the ecological character of wetlands.

4.2 Wetland risk assessment: A framework for predicting and assessing change in ecological character

Wetland risk assessment is not a new term or process. The US Environmental Protection Agency (US EPA) defined wetland ecological risk assessment as a quantitative or qualitative evaluation of the actual or potential adverse effects of stressors on a wetland ecosystem (US EPA 1989). In addition, Pascoe (1993) discussed the concept of wetland risk assessment, outlining two case studies to demonstrate its use, while the US EPA (1998) are currently developing *Watershed ecological risk assessment* frameworks similar to that required for wetland risk assessment. Further, the US EPA's recently revised guidelines for *ecological risk assessment* incorporate detailed information on the prediction and assessment of physical and biological stressors as well as chemical stressors (US EPA 1998). They are very broad, and generally embody the concepts of wetland risk assessment that are discussed below.

A basic model for wetland risk assessment, modified from a generalised ecological risk assessment paradigm by van Leeuwen (1995) is shown in figure 1. It outlines the steps described in the preceding section, with specific examples of approaches for predicting and assessing change of ecological character in wetlands. Each of the steps is briefly described below.

i) Identification of the problem

This is the process of identifying the nature of the stressor and the receptor (ie environment of interest), and developing a plan for the remainder of the risk assessment based on this information. It defines the objectives and scope of, and provides the foundation for, the entire risk assessment (Pascoe 1993, US EPA 1998). In the case of a chemical stressor, identification of the problem would include obtaining and integrating information on the stressor characteristics (eg properties, known toxicity) and source, what is likely to be affected and how is it likely to be affected, and importantly, what is to be protected. Such information is then used to determine the structure and complexity of the remaining steps of the risk assessment. This includes selection of *assessment* and *measurement* endpoints: assessment endpoints are explicit expressions of the actual environmental value(s) to be protected, while measurement endpoints are measurable responses to a stressor that can be correlated with or used to predict changes in the assessment endpoints (Solomon et al 1996). Thus the selection of *ecologically relevant* measurement endpoints is essential, and is discussed further in section 5.3.

ii) Identification of the effects

In this step the effects of the stressor on the measurement endpoints selected during problem formulation are evaluated (van Leeuwen 1995, US EPA 1998). For wetland risk assessment, data for identifying the effects should preferably be derived from field studies, as field data are more appropriate for assessments of multiple stressors (US EPA 1998), and wetlands are known to be exposed to multiple stressors. Depending on the stressor(s) and available resources, such studies can range from quantitative field experiments to qualitative observational studies (Pascoe 1993, US EPA 1998). For chemical stressors, *in situ* or on-site ecotoxicological bioassays constitute the most appropriate approach (Pascoe 1993); however, this does not exclude the use of laboratory experiments if they are considered of particular use (eg for single chemicals or when particular environmental conditions need to be controlled). The criterion that measurement endpoints be ecologically relevant (ie correlated to assessment endpoints; see *Problem formulation*, and section 5.3) is also of paramount importance, and effectively prevents the use of biomarker-type responses for this purpose. Potential techniques for use in effects characterisation are discussed in sections 5 and 6.

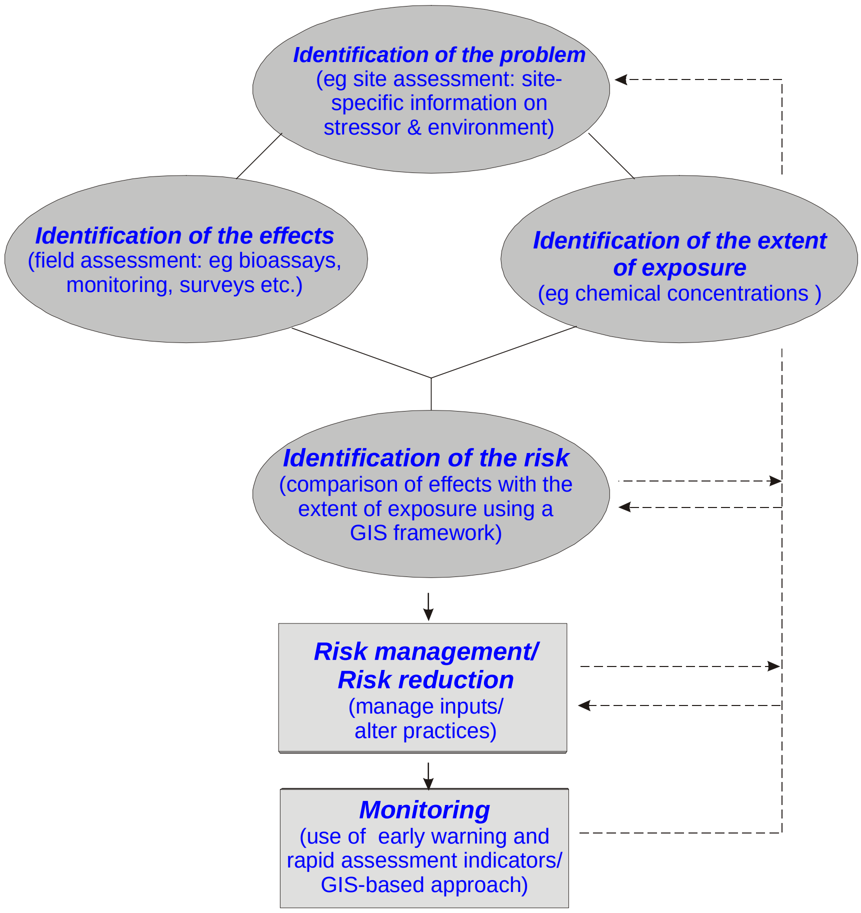

Figure 1 Suggested model of wetland risk assessment (modified from van Leeuwen 1995)

iii) Identification of the extent of the problem

Data on the effects of a stressor to an organism, plant, or ecosystem provide little useful information without knowledge on the actual level of exposure. Identification of the extent of the problem involves estimating the exposure of a stressor to the receptor, by utilising information gathered about its behaviour and extent of occurrence. In the case of a chemical stressor, this includes information on processes such as transport, dilution, partitioning, persistence, degradation, and transformation (Suter 1993), in addition to general chemical properties, and data on rates of chemical input into the environment. For multiple chemical contamination of a wetland, analyses of particular chemical residues throughout the site (eg in water, sediment and/or biota, depending on the chemical), based on knowledge of the pollutant source(s) (obtained during problem formulation) would represent an important component of identifying the extent of the problem. In the case of an invasive weed, it might include detailed information on its entry into an ecosystem, rate of spread and habitat preferences. While field surveys of stressor exposure most likely represent the ideal approach for wetland risk assessment, use of historical records, simulation modelling, and field and/or laboratory experimental studies all represent alternative or complementary methods of characterising exposure.

Identification of the effects and extent of the problem form the overall analysis phase of an ecological risk assessment. They are generally inter-related, and thus, usually carried out concurrently and in an iterative fashion: simple assessments are often performed initially, followed by more comprehensive assessments if considered necessary.

iv) Identification of the risk

This involves integration of the results of the two previous steps in order to estimate the likely level of adverse ecological effects resulting from the exposure to the stressor (Pascoe 1993, US EPA 1998). There exist a range of techniques for estimating risks, often depending on the type and quality of effects and exposure data. The US EPA (1998) describes a number of these techniques. A potentially useful technique for identifying risks in wetlands is via the use of a GIS-based framework, whereby the results of effects and exposure assessments are overlaid onto a map of the region of interest in order to link effects to exposure. In addition to estimating risks, such an approach would also serve to focus future assessments and/or monitoring on identified problem areas. It is important to emphasise that the output of this step need not be a quantitative estimate of risk. However, sufficient information should, at the very least, be available for appropriate experts to make judgements based on a weight of evidence approach. In the event of insufficient information being available, it is possible to proceed with another iteration of one or more phases of the risk assessment process in order to obtain more information (US EPA 1998). Regardless of the approach, uncertainty associated with the risk assessment must always be described, while interpretation of the ecological significance of the conclusions must also be carried out (Pascoe 1993, US EPA 1993). In addition, the risks must be sufficiently well defined to support a risk management decision, as discussed below.

v) Risk management and reduction

Risk management is the final decision-making process that utilises the information obtained from the risk assessment (the processes described above), and attempts to minimise the risks without compromising other societal, community or environmental values. In the context of the Ramsar Convention, risk management must also consider the concept of *wise use* and the potential effects of management decisions on this. If the risks associated with a chemical stressor are considered significant, risk reduction is implemented. This process would include management of inputs and the alteration of practices resulting in such inputs into the ecosystem. The result of the risk assessment is not the only factor that risk management considers; it also takes into account political, social, economic, and engineering/technical factors, and considers the respective benefits and limitations of each risk-reducing action (van Leeuwen 1995). It is a multidisciplinary task requiring communication between risk manager, the risk assessors, and experts in the other relevant disciplines (US EPA 1998).

vi) Monitoring

Monitoring is the last step in the risk assessment process, and one that has largely been ignored as a formal one. In the context of wetland risk assessment and the prediction and assessment of change in ecological character, the monitoring phase should represent or include a major early warning component, as outlined further in the following section. Monitoring should be undertaken to verify the effectiveness of the risk management decisions. It should incorporate components that function as a reliable early warning system, detecting the failure or poor performance of risk management decisions prior to serious environmental harm occurring. The risk assessment will be of little value if effective monitoring is not undertaken. As with effects characterisation, the choice of endpoints to measure in the monitoring process (ie what will be monitored?) is critical. Depending on the nature of the risk assessment and available resources, endpoints may or may not be the same as those used for effects characterisation. As with risk characterisation, a GIS-based approach will most likely be a useful technique for wetland risk assessment, as it incorporates a spatial dimension that useful for monitoring effects on wetlands.

4.3 Ensuring early warning through wetland risk assessment

The concept of early warning is relevant to two parts of the wetland risk assessment process, as outlined below.

The identification of effects in the wetland risk assessment process described above will provide the opportunity to assess or determine the effects of the stressor to regionally relevant biological, physical, and/or chemical processes, using types of approaches described below, in section 6. If the risk assessment is predictive (ie the stressor is yet to act upon the waterbody of interest), then the detection of potential adverse effects provides early warning that, if the stressor enters, or acts upon the aquatic environment, serious harm may result. Estimations of likely discharge rates and environmental concentrations (in the case of pollutants) and other estimates of the likely exposure of the stressor to the environment of interest will provide further information on the likelihood and extent of impact. Again, outcomes of this, when compared with those from the assessment of likely effects will potentially provide early warning of adverse effects.

Management decisions arising from an ecological risk assessment (ie *risk management*) require ongoing monitoring to assess their effectiveness. The purpose of the monitoring is to ensure that the quality of the waterbody of interest is not altered more than the level chosen during the risk assessment process (Sortkjaer 1984). Such a monitoring program should include a range of appropriate early warning indicators, having been selected according to information obtained during the risk assessment process on the stressor and its potential effects, the habitat characteristics, and the assessment and management objectives. It should be noted that most of this information will have been obtained during the problem formulation phase. The original risk management and reduction decisions are then reassessed, and remedial action implemented accordingly. Remedial action could include implementing new or altered management decisions, or even proceeding with another iteration of one or all the phases of the risk assessment. Early warning indicators for monitoring purposes are discussed in sections 5 and 6.

5 Early warning indicators for predicting and assessing change in ecological character

5.1 Definition and types of early warning indicators

We have defined early warning indicators as:

> measurable biological, physical or chemical responses to a particular stress, preceding the occurrence of potentially significant adverse effects on the system of interest.

The actual event, or cause of the stress is usually termed the *stressor*. An early warning indicator need not be directed exclusively at the biological level. That is, subtle changes in water quality, or physical parameters, such as erosion or saline intrusion can act as early warning indicators of more widespread environmental degradation. However, in the context of this discussion paper, only biological and some physico-chemical indicators of pollution are considered. The underlying concept of early warning indicators is that effects can be detected, which are in effect, precursors to, or indicate the onset of actual environmental impacts. While such ‘early warning’ may not necessarily provide firm evidence of larger scale environmental degradation, it provides an opportunity to determine whether intervention or further investigation is warranted.

In aiming to protect wetland ecosystems from anthropogenically-related stressors, it is desirable that effects are detected and acted upon before significant environmental impacts occur. Both Finlayson (1996b) and Bunn et al (1997) have emphasised the need to develop assessment techniques that would provide advanced warning of significant wetland stress or degradation. To do this we need guidelines and processes to assist in evaluating the feasibility and potential of using early warning indicators, preferably those which provide rapid yet environmentally relevant results.

Of the three major types of change in ecological character described above (see Section 3), chemical change has received by far the most attention regarding its environmental impacts and their prediction. As a result, the vast majority of biological and chemical early warning techniques have been developed to assess the impacts of chemical stressors on aquatic ecosystems. Of these techniques, some also have potential applications for assessing biological and physical change in ecological character. While these are acknowledged and addressed as far as possible in this paper, techniques for the early detection of biological and physical change in ecological character are generally not nearly as well defined or developed as those for chemical change. Therefore, it is recommended that further assessments be carried out, independent of the present overview, to identify appropriate indicators for the other major types of change in ecological character. This is of prime importance considering the inter-relationships between the three types of change in ecological character, and the limitations of primarily focussing on one. Nevertheless, the importance of chemical change in wetland ecosystems should not be underestimated, as it is considered to be an increasingly significant cause of wetland degradation. Thus, the examples of early warning indicators discussed in this paper mostly represent biological and physico-chemical assessment approaches to predict or forewarn of important chemical changes (ie pollutant impacts) on wetland ecosystems.

A number of early warning indicators have been developed for the assessment of aquatic ecosystems. For the purposes of the present discussion, these are placed into three broad categories: i) *Rapid response toxicity tests*; ii) *Field early warning tests*; and iii) *Rapid assessments*. A general description of these, including potential limitations, is outlined in table 1. As described in table 1, each of the techniques may meet different objectives in water quality assessment programs. Although the majority of early warning indicators are of a biological nature, physico-chemical indicators do exist and often form the initial phase of assessing water quality. Such techniques are discussed independently of the biological indicators (see Section 6.4).

Features and examples of each of these three broad categories are described below.

5.2 Assessment objectives and selection of early warning indicators

At this point we reflect on the setting and framework in which biological indicators are chosen for employment in monitoring programs. The choice of indicators follows a hierarchy of other decisions required by managers in setting up monitoring programs to assess ecosystem health (Finlayson 1996a,b).

Table 1 Role and possible limitations of types of early warning indicators

Type of response and role	Potential limitations
Rapid response toxicity tests	
Laboratory toxicity assessment of sensitive whole organism responses (eg growth, reproduction) with rapid turn-around of results. They are predictive tests that potentially enable timely and flexible management actions (eg determining a safe dilution for discharge of effluents of changing composition) to be implemented.	Ecological relevance of measured sub-lethal responses (eg growth, reproduction) has generally not been established.
Field early warning tests	
Field measurement of sensitive sub-lethal organism responses through monitoring or assessment. They can provide pre-emptive or preventative information so that substantial and ecologically important impacts are avoided.	Ecological relevance of measured responses (especially biochemical biomarkers) has generally not been established.
Rapid assessments	
Standardised, rapid and cost-effective monitoring of various forms can provide 'first-pass' assessment of the ecological condition of sites over large areas. Broad coverage has potential to identify 'hot spots' and hence pre-empt and prevent similar occurrences elsewhere.	Output is usually coarse and generally only detects relatively severe impacts.

Thus, after identifying the water quality issue of concern or potential concern and determining the environmental values to be protected, managers should then be concerned with identifying *assessment objectives* for protection of the water resource. As an example the following could be used:

- *Early detection of acute and chronic changes*, providing pre-emptive information so that ecologically important impacts are avoided.
- *Assessing the ecological importance of impact* through measurement of biodiversity, conservation status and/or population, community or ecosystem-level responses.

To determine effects upon the ecosystem as a whole – or the ecological importance of effects that are observed – measurement of ecosystem 'surrogates' is usually required. Typically these surrogates are communities or assemblages of organisms, or habitat or keystone-species indicators where these have been closely linked to ecosystem-level effects. The Ramsar Convention currently uses habitat area extent as its surrogate indicator. Information on the ecological importance of effects is best met in programs that have regional coverage and that encompass a full disturbance gradient, ie programs that can provide some context to the gathered data. Rapid assessment methods can in many situations provide this context.

In addition to the assessment objectives, there are a range of other factors that must be considered when selecting appropriate indicators and protocols for biological assessment. Economic and ecological considerations will always limit the number of indicators that can be assessed. As such, they must be selected in order to maximise relevant information and minimise redundant information (Cairns et al 1993). Prior information on this will be of great use when selecting suitable indicators, as is outlined below

In selecting an indicator we need to keep in mind the definition of the ecological character of a wetland and its emphasis on the *biological*, *chemical* and *physical* components of the ecosystem. Therefore, it may be useful to select early warning indicators according to which of the above three components is/are considered more susceptible to change. The three components are intricately linked; chemical change can lead to biological change; physical change can lead to biological and chemical change; and biological change could potentially lead to chemical or physical change. Although these interactions exist, the wetland risk assessment framework provides a process to assist in identifying the most appropriate indicators to assess or predict change. If sufficient information exists to determine this, satisfactory early warning of large scale wetland degradation may be achieved.

In addition, in listing a wetland site as internationally important under the Ramsar Convention, a contracting party must describe the ecological character of the wetland (Finlayson 1996a). Part of this process requires the provision of information on human-induced factors that have affected or could significantly affect the benefits and values of international importance. Such information on stressors could be utilised to aid in determining which of the biological, physical or chemical components of an ecosystem will be affected. Finally, as discussed below and by Grillas (1996), the more precise the assessment objectives of the program, the easier and clearer the selection of indicators. Based on the above information, a range of suitable early warning indicators should be selected, to form an adequate 'suite' of indicators, as part of an *early warning system*.

Thus, the choice of indicators is intricately linked to the nature of the stressor and the assessment objectives, and should be well articulated and documented in much the same way as recommended for wetland monitoring (Finlayson 1996a,b). Further, the feasibility of successfully using a particular indicator should be carefully assessed. In this respect, the logistics of implementing a particular technique, interpreting the information, and reporting in a timely manner require as rigorous assessment as does the technical relevance. For this to occur, a well concurred management scenario should be in place, preferably through an agreed management plan, with well documented procedures for ensuring the early warning information is acted upon. The success of an indicator in detecting early warning of change could be lost if the responses to the early warning are inadequate.

5.3 Ecological relevance of early warning indicators

In discussing the wetland risk assessment framework (see section 4.2), the term *ecological relevance* was used with respect to measurement endpoints. This is discussed further in terms of the implications for the use of early warning indicators, which are, in effect, measurement endpoints, for identifying effects and monitoring, and linking to the assessment objectives (as described above). The ecological relevance of an early warning indicator can be considered as the ability to directly link the observed response to effects at the population, community and/or ecosystem level. However, the concepts of early warning and ecological relevance can conflict.

The types of biological responses that can be measured, and their relationship to ecological relevance and early warning capability is generalised in figure 2. As an example, biomarker responses (see section 5.4.3) can offer exceptional early warning of potential adverse effects as they indicate exposure to a pollutant(s), but there exists very little evidence that observed responses result, or culminate in adverse effects at an individual level, let alone the population, community or ecosystem level. Therefore, they cannot be considered ecologically relevant. If the primary assessment objective is that of early detection, then it is likely that it will be at the expense of ecological relevance, while the opposite would probably apply if knowledge of the ecological significance of effects was considered paramount. It is important that these distinctions are clear for wetland risk assessment: when identifying the effects of a stressor, ecological relevance will take precedence over early warning capability, while the opposite will often apply during the monitoring phase.

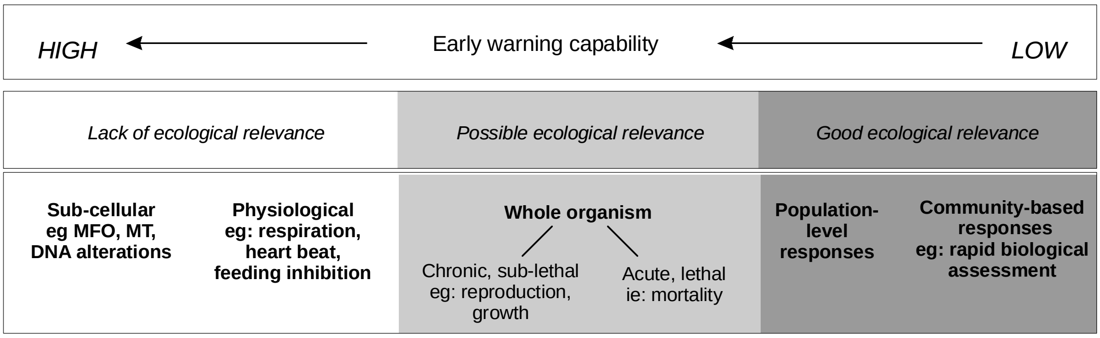

Figure 2 Relationship of ecological relevance and early warning capability to measurable biological responses

Thus, while chemical effects on individuals are well known through biochemical and physiological changes, present understanding and prediction of effects on populations (or communities) are poor (Clements & Kiffney 1994). Further, for some ecosystems, it has been argued that causal links between early detection methods and those effects occurring in the wider ecosystem can never be demonstrated (Underwood & Peterson 1988). Munkittrick & McCarty (1995) maintained that the only changes which could be transferred from the individual level (ie sub-cellular, physiological, whole organism) to the population level involve information on growth, reproduction and survival. Nevertheless, many researchers simply take a precautionary view to a contaminant-related change in sub-lethal effects. For example, Livingstone (1993) viewed changes in sub-lethal effects as indicating 'cause for concern' and being 'sufficient in itself to merit attention and action.'

5.4 Ideal attributes of early warning indicators

Ideal attributes of early warning indicators have previously been discussed (Cairns & van der Schalie 1980, Cairns et al 1993, McCormick & Cairns 1994), and have recently been summarised in a modified form by van Dam et al (in press), as presented below.

To have potential as an early warning indicator, a particular response should be:

i. *anticipatory*: should occur at levels of organisation, either biological or physical, that provide an indication of degradation, or some form of adverse effect, before serious environmental harm has occurred,

ii. *sensitive*: in detecting potential significant impacts prior to them occurring, an early warning indicator should be sensitive to low levels, or early stages of the stressor,

iii. *diagnostic*: should be sufficiently specific to a stressor, or group of stressors, to increase confidence in identifying the cause of an effect,

iv. *broadly applicable*: should predict potential impacts from a broad range of stressors,

v. *correlated to actual environmental effects/ecological relevance*: knowledge that continued exposure to the stressor, and hence continued manifestation of the response, would usually or often lead to significant environmental (ecosystem-level) effects (see section 5.3 above),

vi. *timely and cost-effective*: should provide information quickly enough to initiate effective management action prior to significant environmental impacts occurring, and be inexpensive to measure while providing the maximum amount of information per unit effort,

vii. *regionally relevant*: should be relevant to the ecosystem being assessed,

viii. *socially relevant*: should be of obvious value to, and observable by stakeholders, or predictive of a measure that is,

ix. *easy to measure*: should be able to be measured using a standard procedure with known reliability and low measurement error,

x. *constant in space and time*: should be capable of detecting small changes, and clearly distinguishing that a response is caused by some anthropogenic source, not by natural factors as part of the natural background (ie high signal to noise ratio),

xi. *nondestructive*: measurement of the indicator should be nondestructive to the ecosystem being assessed.

The importance of the above attributes cannot be over-emphasised, since any assessment of actual or potential change in ecological character will only be as effective as the indicators chosen to assess it (Cairns et al 1993). However, an early warning indicator possessing all the ideal attributes cannot exist, as in many cases some of them will conflict, or will simply not be achievable. For example, a biochemical biomarker might provide an excellent indication of potential impacts due to a particular chemical, but might not detect serious effects of other chemicals. Subsequently, decisions are required as to which attributes are more important for a particular purpose, and appropriate indicators chosen based on those attributes (in accordance with the aspects discussed in section 5.2 above). Further, the 'diagnostic' and 'broadly applicable' attributes are mutually exclusive, with their relative importance being directly related to the nature of the problem and early warning required.

6 Examples of early warning indicators

This section provides several examples of the types of early warning indicators for predicting and assessing change in ecological character due to chemical stressors, and where applicable, biological and/or physical stressors. In addition to the examples discussed in the text, Appendixes A and B present a wider range of techniques available for the prediction and assessment of change in ecological character.

6.1 Rapid response toxicity tests

Rapid response toxicity tests represent laboratory toxicity bioassays designed to provide rapid and sensitive responses to one or more chemical stressors. They provide an indication that there may be a risk of adverse effects occurring at higher levels of biological organisation (eg communities and ecosystems). Laboratory toxicity tests are of particular use for a chemical or chemicals yet to be released into the aquatic environment (eg a new pesticide or a pre-release waste water). They provide a basis upon which to make decisions about safe concentrations or dilution/release rates, thereby eliminating, or at least minimising adverse impacts on the aquatic environment.

The use of sensitive test species, and measurement of sensitive responses further enhances the capacity of the assessment to predict potential adverse ecological effects, while the use of rapid tests, lasting no longer than 3 to 4 days, allows for rapid processing of the information for feedback towards subsequent management decisions. The types of organisms, or species to test can vary from simple organisms such as bacteria and phytoplankton, to aquatic vertebrates, such as fish and amphibians. The choice of organisms depends on a number of factors, including their sensitivity, their regional relevance, and the type of contaminant.

Types of responses measured can vary markedly, from sub-lethal, physiological effects such as feeding inhibition, respiration and heart rate, to whole organism effects such as growth, reproduction and mortality. However, there are major differences in the *ecological relevance* of responses that can be measured, and this must be considered when selecting measurement endpoints during the problem formulation stage of the wetland risk assessment framework. Generally, physiological or sub-cellular responses are not used for the identification of effects (Solomon et al 1996).

The duration of rapid response toxicity tests depends on the species being assessed, the response being measured, and on the objective of the test. For example, where the composition of an effluent discharge is continually and unpredictably changing, adequate responses should be sought within 24 hours. Alternatively, where inputs into, and changes to natural ecosystems are more subtle, and occur over a longer time scale, experiments lasting up to, but no longer than 4 days are usually sufficiently rapid to allow timely processing of the information for consideration in subsequent management decisions.

A range of examples of rapid response toxicity tests is provided in Appendix A. In addition, van Dam et al (in press) provide an overview of potential toxicity testing and monitoring techniques for assessing wetland degradation.

6.2 Early warning field tests

This group of early warning indicators comprises a range of techniques that are grouped because they are used to measure responses or patterns in the field (*in situ*) and thus, provide a more realistic indication of effects in the environment. In contrast to laboratory rapid response toxicity tests, early warning field tests predict and/or assess (or monitor) the effects of existing chemical stressors. Some of the techniques can also be applied to biological and physical stressors.

6.2.1 Direct toxicity assessment

The use of toxicity tests to assess and monitor the consequences of chemical stressors in aquatic ecosystems (eg waste water releases, contamination of waterways with pesticides and other agrochemicals), is known as *direct toxicity assessment (DTA). In situ* toxicity assessment of a waterbody receiving a pollutant input serves to monitor the effectiveness of predictions based on the rapid response toxicity tests described above. However, assuming the measured responses are sensitive, results can also provide early warning of potential impacts at higher levels of biological organisation. In wetland risk assessment for contaminated sites, one must consider the local abiotic conditions and the sensitivity of local species, to obtain ecologically relevant information from standard testing. Thus *in situ* toxicity assessment is of prime importance for the identification of effects. Again, assuming adequate sensitivity, such tests could also be applied during the monitoring phase of the risk assessment, when the emphasis is on early warning.

Selection of species for DTA is similar to that described above for rapid response toxicity tests, obviously with an emphasis on sensitivity and use of local species.

Early warning of water quality problems using field assessment has been achieved with considerable success in Europe, using shell and valve closure in freshwater mussels as an indicator of long-term water quality of heavily polluted rivers (Kramer et al 1989, de Zwart et al 1995). Similarly, a ‘creek-side’ toxicity testing program has also been successfully developed and used for the Alligator Rivers Region (ARR) in northern Australia, using a local gastropod and fish species to assess the effects of mine waste water releases (the dilution/release rates of which have been set by pre-release rapid response toxicity testing) into a nearby wetland system (Humphrey et al 1990, 1995). These two approaches differ in that the former is a continuous, longer-term assessment, while the latter involves toxicity tests of four days duration.

6.2.2 Phytoplankton monitoring

Due to their nutritional requirements, their position at the base of aquatic food webs (dominant primary producer), and their ability to respond rapidly and predictably to a broad range of pollutants (McCormick & Cairns 1994, Lewis 1995, Stauber 1995), phytoplankton represent perhaps the most promising early warning indicators of change in ecological character of wetlands due to chemical stressors. In addition, their sensitivity to changes in nutrient levels makes them ideal indicators for assessing eutrophication. Therefore, it is considered worthwhile to discuss the use of phytoplankton for early warning purposes separately.

Phytoplankton can be used in the types of toxicity bioassays described above, for rapid response toxicity tests and direct toxicity assessment. Such methods are rapid, inexpensive and sensitive, and can be carried out in the laboratory or *in situ*, using either laboratory cultured algae, or natural phytoplankton assemblages. For example, algal fractionation bioassays (AFB) assess the effects of pollutants, or natural waters on functional parameters (eg ^{14}C uptake, biomass) of various size fractions of a natural assemblage of algae. Structural indicators, such as species composition and size assemblage shifts have also been found to be particularly sensitive (Munawar et al 1989, Munawar & Legner 1993, Munawar et al 1994).

Biological monitoring using phytoplankton represents a promising tool for detecting early pollutant impacts, including eutrophication. McCormick and Cairns (1994) suggested that taxonomic analyses of diatom assemblages could provide useful early warning of pollutant effects, as diatoms are known to be abundant, and to respond rapidly to changes in water quality (McCormick & Cairns 1994, Schofield & Davies 1996). Alternatively, monitoring could involve assessment of the 'microbial loop', incorporating bacteria, picoplankton, nanoplankton, microplankton, and the larger phytoplankton, as proposed by Munawar and co-workers (Munawar & Weise 1989, Munawar et al 1994).

The use of phytoplankton as an early warning indicator certainly appears to have potential for assessing all three types of change in ecological character. Field-measured phytoplankton assemblages could provide a reliable indication of the state of an ecosystem, so long as sufficient baseline data exist to differentiate between natural variations and perturbations caused by anthropogenic stress. However, a major limitation, as with all such biological monitoring programs, is that much time and effort is required to i) obtain sufficient baseline data initially, and ii) to continue monitoring to the extent that will provide the required sensitivity to be useful as an early warning indicator.

6.2.3 Biomarkers

Biomarkers can be defined as biochemical, physiological, or histological indicators of either exposure to, or effects of particular stressors (usually xenobiotic chemicals) at the sub-organismal or organismal level (Huggett et al 1992). The underlying concept is that changes to the biochemistry, physiology or histology of individual organisms often precede effects at the organismal, and therefore, potentially population, community and ecosystem level. Briefly, aquatic animals (usually fish but also invertebrates) are collected from the site(s) of interest and a reference site, and the biomarkers assessed and compared. A modification of this is to place 'caged' organisms in the environment of interest, and measure biomarker responses following a pre-determined period of time. Biomarkers have been used to predict potential adverse effects of a number of pollutant types, including organic chemicals such as pesticides and petroleum hydrocarbons, heavy metals, and complex mixtures (eg industrial effluents). A number of reviews describing useful biomarkers have previously been published (eg Haux & Förlin 1988, Goksøyr & Förlin 1992, Hugget et al 1992; Roesijadi 1992, Everaats et al 1993, Livingstone 1993, Förlin et al 1995, Holdway et al 1995, Walker 1995). Three potentially useful types of biomarkers are discussed below, while additional biomarkers, and their attributes and potential applications, are listed in Appendix B.

Mixed function oxidases

The cytochrome *P*-450 - linked mixed function oxidase (MFO) system is a group of enzymes involved in the metabolism of poorly water-soluble compounds, which includes many pollutants. Exposure to such a compound results in an increase in activity (induction) of MFOs compared to non-exposed animals, and this can be used as a measure of chemical exposure, and hence potential adverse effects. The existence and use of MFOs in aquatic organisms, particularly fish, has been well documented (Payne et al 1987, Haux & Förlin 1988, Ahokas 1990, Goksøyr & Förlin 1992). Although limited in terms of indicating actual toxic effects, evidence of exposure provides early warning of potential effects, and allows the implementation of further research or remedial action if considered necessary. Due to their specificity, MFOs are most useful as biomonitors in cases where the pollutant(s) is known, and known to result in induction. In addition, the detection of MFO induction can indicate the presence of a particular compound in natural waters. Some common pollutants that induce MFOs include persistent organochlorine pesticides, polycyclic aromatic hydrocarbons (PAHs), polychlorinated biphenyls (PCBs), as well as many complex effluents containing many thousands of unidentified compounds (Haux & Förlin 1988, Holdway et al 1995). MFO induction can be measured at both a biochemical and molecular level, and is both time- and cost-effective (Payne 1984).

Vitellogenin – a biomarker of potential endocrine disruption

Endocrine disruption has emerged as a current and potentially major threat to aquatic (as well as terrestrial) ecosystems. It is known that certain anthropogenic and natural compounds have the ability to modulate the endocrine system in both humans and wildlife (Kendall et al 1998). In many cases such effects might be reversible or non-adverse, however, there is growing evidence that particular chemicals exert irreversible, adverse effects on wildlife (Kendall et al 1998). While there are many possible mechanisms for such effects, the majority of research has focussed on hormone mimics, particularly those acting as estrogen agonists (Giesy & Snyder 1998).

Due to the fact that manifestation of effects are often not observed until the onset of sexual maturity, well beyond the period of chemical exposure, there exist no standard aquatic bioassays for specifically detecting effects of endocrine-disrupting chemicals (Lamb et al 1998). However, there are a number of biomarkers that might be used to at least detect exposure to and potentially effects of, estrogenic compounds in aquatic vertebrates. Of these, the egg phospholipoglycoprotein, vitellogenin (VTG), has received a great deal of attention, particularly in male fish (Giesy & Snyder 1998). VTG, a precursor to egg yolk components, is synthesised in the liver of oviparous vertebrates (egg layers) (Mommsen & Walsh 1988). VTG synthesis is induced in response to estrogen. While both male and female fish have the ability to produce VTG, males generally have undetectable or very low levels because they lack sufficient levels of estrogen to induce its production (Mommsen & Walsh 1988, Ankley et al 1998). Estrogenic compounds have been shown to induce VTG production in male fish, both in the laboratory and field (see review by Giesy & Snyder 1998). Thus, the presence of VTG in male fish (and also elevated levels in female fish) can be used as an indicator of exposure to estrogenic compounds in aquatic environments. Routine methods of analysis exist, primarily by radioimmunoassay (RIA) or enzyme-linked immunoabsorbent assay (ELISA) (Ankley et al 1998). Some further potential assays and endpoints for detection of exposure to endocrine disrupting chemicals in fish are detailed by Ankley et al 1998.

Endocrine disruption has also been extensively studied in birds, with several notable effects being eggshell thinning, feminisation of embryos, and crossed-bill deformities, as a result of exposure to organochlorine pesticides (eg DDT), polycyclic aromatic hydrocarbons (PAHs) and/or polychlorinated biphenyls (PCBs) (Ankley & Giesy 1998, Ankley et al 1998, Lamb et al 1998). VTG in birds has rarely been used as an indicator of exposure to estrogenic compounds (Ankley et al 1998), however, Lamb et al (1998) suggested that current avian toxicity test protocols for endocrine disrupting chemicals include assessment of VTG in parental adults, as well as first generation hatchlings and adults. However, Ankley et al (1998) suggested that a further understanding of VTG induction and xenobiotic binding to hormone receptors be gained before recommending them as potential screening assays for estrogenic compounds.

Bioaccumulation

Certain aquatic biota, particularly fish and molluscs, possess an inherent capacity to accumulate chemicals, both organic and inorganic, to very high concentrations compared to ambient water concentrations. The ‘time integrated’ accumulation of higher than baseline levels of a compound in an organism may provide a better representation of overall ambient environmental/bioavailable concentrations than instantaneous grab samples of water, soil or sediment (Richardson et al 1992). Thus, measuring bioaccumulation provides an indication of exposure and potential effects.

Of the many compounds that have the potential to bioaccumulate, those of particular concern are the organochlorines (or chlorinated hydrocarbons), including many pesticides and PCBs, petroleum-derived hydrocarbons (eg PAHs), and trace metals and radionuclides (Connell 1990, Phillips 1992, Rainbow 1992, Martin et al 1995). Of the organisms suitable for bioaccumulation studies, fish and molluscs (eg bivalved mussels) have by far been the most extensively used, in both freshwater and marine ecosystems (Martin 1992, Richardson et al 1992, Jeffree et al 1995,). While also providing an indication of bioavailable contaminant concentrations, the assessment of bioaccumulation in, for example, fish, is also important for predicting potential indirect effects on other wildlife and humans via biomagnification (Hawker 1990). This is particularly important when considering fisheries, or more broadly, food resources of wetlands. For example, Martin et al (1995), assessed bioaccumulation of radionuclides in organisms belonging to the diet of Aboriginal people living downstream of a uranium mine (including shrimp, fish, turtle, magpie goose, and freshwater crocodile), with the concern being that to humans.

Summary of biomarkers

Many biomarkers have been demonstrated to give early warning of potential environmental effects of particular chemicals or complex effluents (Walker 1995; see Appendix B). They provide the most advanced form of biological early warning. As such, it is likely that, in conjunction with other ecotoxicological and biological monitoring techniques described in this section (also see Appendix A), appropriately selected biomarkers could be usefully applied during the monitoring phase of the wetland risk assessment model. In addition, as they provide an indication of exposure to particular chemical stressors, many biomarkers can also be useful tools for exposure characterisation.

6.3 Rapid assessments

Rapid assessments are being increasingly used for water quality monitoring, having the appeal of enabling ecologically-relevant information to be gathered over wide geographical areas in a standardised fashion and at relatively low costs. The trade-off in these virtues is that rapid assessment methods are usually relatively ‘coarse’ and hence are not designed to detect subtle impacts. As such, they would not normally be considered as useful indicators of early environmental change. Nevertheless, under some circumstances they can serve a very useful early warning role: they are well suited to identifying and detecting problem locations and stressors that occur across large areas and which could pass unnoticed by more specific early warning indicators (such as discussed above). At this broad scale, they could provide managers with forewarning of problems that, if not detected earlier, could have led to irreversible and unwanted change.

Desired or essential attributes of rapid assessment include: i) measured response is widely regarded as adequately reflecting the ecological condition or integrity of a site, catchment or region (ie ecosystem surrogate); ii) approaches to sampling and data analysis are highly standardised; iii) response is measured rapidly, cheaply and with rapid turnaround of results; iv) results are readily understood by non-specialists; and v) response has some diagnostic value. Resh and Jackson (1993) and Resh et al (1995) elaborate further upon features of these approaches as applied to rapid biological assessment of stream macroinvertebrate communities, a group for which most progress has been made worldwide in rapid-assessment development.

A range of rapid assessment methods is now being developed for aquatic habitats worldwide. In Australia, protocols are being developed for riverine benthic algae (diatoms) and fish, as well as for macroinvertebrate communities in streams, wetlands and estuarine sediments (Schofield & Davies 1996). Amongst promising approaches that might be applied specifically to wetlands is the use of remote sensing to detect changes in wetland vegetation (Finlayson 1994). Perhaps the most basic and most fundamentally important information provided in broad survey methods such as these is habitat area extent. As discussed above, most of these methods will have limitations when applied to site-specific situations requiring enhanced sensitivity to detection of change.

6.3.1 Rapid biological assessment

As stated above, the most advanced of the rapid biological assessment (RBA) techniques developed worldwide are those that have been applied to stream macroinvertebrate communities. Typical of the approach are the RIVPACS and AUSRIVAS methodologies developed for Britain (Wright 1995) and Australia (Schofield & Davies 1996), respectively. Both approaches are based on comparing macroinvertebrate community composition at sites of interest against the composition at unimpacted or least impacted reference sites, using predictive modelling procedures. The outcome is a measure of actual or observed community composition compared with the composition that could be expected (predicted) at a site if it was not disturbed; predictions are based upon key environmental characteristics of the site. The model output is a simple ratio of observed:expected taxa; the taxa lists included in the model output can also be applied to any number of biotic indices of water quality.

The efficacy in applying the AUSRIVAS methodology, in particular, across the vast expanse of the Australian continent is achieved through reduced effort and cost of sampling and sample processing over those normally associated with biological surveys of macroinvertebrate communities. Standardised sampling, sample processing and data analysis are integral to the approach. Thus, single replicate samples (only) of individual habitats are taken, and in many instances, sorting of the sample is carried out in the field for no longer than one hour per sample. The effort involved in identification of samples in the laboratory is also reduced substantially by identifying the fauna to family level. Once predictive models have been developed, it could be expected that the quality of a site based upon its macroinvertebrate composition could be provided to managers on the same day that sampling occurred.

6.3.2 Monitoring of birdlife

Waterbirds are prominent and often easily observable and quantifiable features of wetland biodiversity, and have considerable potential as tools for rapid assessment of wetland character and changes in it. Several relatively simple-to-collect categories of waterbird information offer possibilities for employing rapid assessment to detect change in the quality of the waterbird assemblage. These include

- The presence, number and status (eg breeding, ‘overwintering’) of globally or regionally threatened species (globally threatened species are listed by Collar et al (1994) and species of regional conservation concern in Europe by Tucker and Heath (1994)).
- The presence, number and status of characteristic species of a particular wetland habitat or habitats – often called ‘indicator species’ or ‘keystone species’.
- The diversity of the waterbird assemblage (at its simplest, the number of species present).
- The size of the total waterbird population at a particular time of year or stage in their annual cycle (eg breeding, moulting, migratory staging, non-breeding).
- The population size of species or biogeographic populations for which the wetlands are of particular significance (eg as supporting internationally important numbers of a particular population or species).

Perhaps the simplest is identifying the presence of threatened species on a wetland, since such information provides an absolute indication that the site is of major significance to waterbirds. For the other measures, assessing the significance of a particular wetland also requires understanding its quality relative to standards derived from wetlands of similar type and within the same geographical region.

Few such standards are, however, readily available and their derivation and application can be contentious and complex. Some examples from United Kingdom (UK) coastal wetlands demonstrate how such standards can be used. For each estuary in the UK, Davidson (1991) assessed the number of breeding wader (shorebird) species present, permitting identification of estuaries supporting relatively high species diversity. Assessing characteristic species, Craddock and Stroud (1996) reported numbers of confirmed breeding species characteristic of coastal wet grassland for each part of each UK region, so identifying locations of regionally high diversity.

Population size, both of the total assemblage and of individual species of waterfowl, derived from co-ordinated counting schemes is regularly reported for many wetlands and can be used in several ways to assess wetland character. Each, however, requires information additional to population size at the wetland for appropriate interpretation. Population size can be used in combination with knowledge of the size of each flyway population (Rose & Scott 1997) to identify wetlands of international importance using the Ramsar Convention criteria of regular presence of 1% or more of a flyway species’ population and/or a minimum of 20 000 waterfowl. In an analogous way nationally important sites are identified where national population totals are known (eg Cranswick et al 1997 for the UK).

Population size and species diversity are strongly correlated with wetland area, so comparing size or diversity in an individual wetland with a size-area or species-area curve permits identification of wetlands with above or below average density or diversity for their size – see Davidson (1991) for an example for British estuaries.

There are three important points to consider in appropriately applying such assessment methods.

- Several years of information on population size at a site is ideally needed to allow for annual variations since in many, notably arctic-breeding, species breeding success and hence population size and distribution is known to vary substantially between years. Application of the Ramsar criteria for waterbirds recommends assessing the regular occurrence for a minimum of five years. A rapid assessment based on a single survey needs, therefore, to be interpreted with caution.
- The standard against which rapid site assessment comparison is made should be derived from the same wetland type only (eg estuaries, marshes, lacustrine wetlands) since species-area curves are very different for different wetland types.
- It is essential to select an appropriate geographical region for such comparisons: even within a flyway both species diversity and density are known to vary latitudinally (eg Hockey & Barnes (1997) for non-breeding coastal shorebirds on the East Atlantic flyway) and longitudinally (eg Alerstam et al (1986) for diversity of arctic-breeding shorebirds).

6.3.3 Remote sensing

Satellite imagery can be used to assess the status of particular features of wetlands (eg areal extent, vegetation and water coverage) and possibly even identify changes in land use that detrimentally affect wetlands. By comparing with historical records, the extent of ecological change can be identified. In this instance, the baseline or reference state is provided by previously taken images and/or maps (Finlayson 1994, Taylor et al 1995).

The use of satellite data (remote sensing) as an environmental monitoring tool commenced in 1972 with the launch of the US Landsat satellite. Data are now also available from a variety of satellites (Taylor et al 1995, Sahagian & Melack 1997) and satellite remote sensing has been used for monitoring wetlands and lakes in a number of parts of the world (eg Nakayama 1993, Yates et al 1993, Muller et al 1993). However, even with improvements in the detail and reliability of information derived from satellites the accuracy and reliability may not match that from conventional aerial photography using manual photo-interpretation, such as that used in the USA National Wetlands Inventory Project (see Wilen & Bates 1995) and recommended for use in Australia by Johnston and Barson (1993).

Phinn et al (1998) have reviewed the use of remote sensing for wetland inventory and monitoring and noted several problems that need consideration in any applications. Remotely sensed data and spatial analytic techniques are capable of providing information on vegetation structures from local to regional scales. Two problems limit the application of these techniques: i) identifying suitable spectral, radiometric, spatial and temporal data resolutions; and ii) defining analytic techniques to provide appropriate information for specific monitoring objectives and wetland environments. Addressing these problem requires that systematic consideration is given to characteristics of the environment(s) to be examined and the type of information required. They provide a summary listing of processing techniques applicable to each type of information and discuss their advantages and limitations.

From an early warning perspective it is important to realise that the cost and processing time for such data also need to be considered. However, despite these limitations remote sensing is increasingly being used to monitor wetlands and can provide high quality spatial and temporal data.

6.4 Physico-chemical indicators

Only direct measurements of the biota of a waterbody is sufficient to fully characterise its status, or reliably detect adverse impacts. However, physico-chemical monitoring has been recognised as being a vital component of an integrated assessment program that also utilises biological measures for assessing a waterway's condition (Schofield & Davies 1996).

The monitoring of standard physico-chemical parameters such as pH, dissolved oxygen (DO), biochemical oxygen demand (BOD), total organic carbon (TOC), hardness, conductivity, salinity, and nutrients such as nitrogen (N) and phosphorus (P) can be of use in several ways. Firstly, it provides a record of the physico-chemical characteristics of the waterbody, which when continued over an extended period, provides a record of the variation in the characteristics over time. Unusual changes in any of the parameters will provide an indication that the characteristics of the water, and therefore potentially the water quality, are changing. Secondly, many physico-chemical parameters have the ability to alter the toxicity of particular pollutants. For example, pH, hardness and TOC are all known to modify the toxicity of a range of heavy metals. Subsequently, knowledge of the behaviour of such parameters is of great importance, particularly in regions where elevated levels of pollutants already exist. The majority of standard physico-chemical water quality parameters are simple, inexpensive and quick to measure, and should be used to complement any ecotoxicological or biological monitoring study.

The measurement of priority, suspected and/or known pollutants in a waterbody will also provide potentially useful information, particularly for identifying the extent of the problem in the wetland risk assessment model. Chemical monitoring will generally assist in identifying the toxic components in a waterbody, but is unlikely to be able to provide adequate early warning, unless gradual increases at very low levels (ie below the toxic threshold), due to processes such as seepage from contaminated soils or groundwater, can be detected. Nevertheless, the use of speciation and chemical equilibrium models, incorporating chemical measurements and physico-chemical data will certainly assist in the prediction of bioavailable fractions of pollutants and hence potential toxicity, and thus may provide a form of early warning. Chemical monitoring is relatively simple, and rapid, with standard methods existing for most major organic and inorganic chemicals, but unlike physico-chemical monitoring, is relatively expensive. Unless pollutant inputs are suspected, or known to be entering a wetland system, chemical monitoring may not be a regular requirement.

7 Responsiveness to changes in an early warning indicator

Acceptance of the need for, and inclusion of, an early warning indicator in a monitoring program implies that information on early change is acted upon by management; for this to occur, an agreed management plan must be set in place at the onset of the investigation (see section 5). Normally, the initial stages of this management plan would entail a series of iterations amongst negotiating stakeholders about the type and size of the change that are deemed important, as well as the relative costs of inferring that there is an impact when in fact there is none (Type I error) and of failing to detect a real impact (Type II error) (Mapstone 1995). These 'decision criteria' are important statistical parameters that must be decided upon as they stipulate the confidence with which the null hypothesis (eg of ‘no impact’) is either accepted or rejected. It is important to note that this type of discussion and decision-making are all aspects of the problem formulation phase of the wetland risk assessment model.

Inclusion of an early warning indicator in a monitoring program implies a *precautionary* management approach, ie intervention before real and important ecosystem-level changes have occurred. As stated above (section 5.3), for most of the sublethal group of early warning indicators, information about the ecological importance of any observed change is lacking; links have rarely been established to higher-order level effects. Management intervention in response to changes in an early warning indicator, therefore, occurs as some conservative and generally arbitrary threshold or trigger value in the measured response (variously called ‘effect size’ or ‘alert level’) is approached. In addition, because an early warning indicator is being employed, it would normally be assumed that the costs of failing to detect a real impact are regarded as ‘high’; statistically, this translates to reducing the Type II error rate by increasing the amount of replication in a baseline data collection phase and/or by accepting a higher Type I error rate. Thus, in the latter case, there is acceptance of a greater rate of ‘false positives’ – a cost burden for developers/industry at the expense of providing greater protection of ecosystems.

The most powerful impact assessment programs will generally be those that include two types of indicator, namely those associated with early warning of change and those (regarded as) closely associated with ecosystem-level effects. The ‘ecosystem-level’-type indicator might include ecologically important populations (eg keystone species) or habitat, or communities of organisms that serve as suitable ecosystem ‘surrogates’ (as described in section 5.2). Indicators used in rapid assessment would also normally serve this role. With both types of indicator measured in a monitoring program, information provided by ‘ecosystem-level’ indicators may then be used to assess the ecological importance of any change observed in an early detection indicator.

Just as for early warning indicators, thresholds of change and other statistical decision criteria for the ‘ecosystem-level’ indicators must also be negotiated and decided upon up front. The main difficulty in deriving such thresholds is that, unlike the situation for early warning indicators, there is generally a lack of information about indicator/pollutant relationships, complicated by the many indirect effects that may occur in animal and plant communities as a consequence of exposure to contaminants (Clements & Kiffney 1994). Whilst there are a number of approaches that might be considered in deriving thresholds of change for ‘ecosystem-level’ indicators, such a discussion is beyond the scope of this paper. Specific decisions on thresholds of change are an issue that can only be dealt with effectively on a site-specific basis, whilst taking account of the ecological values and wise use of the site. We reiterate that this is a site-specific issue and that it should be addressed before the monitoring regime is implemented.

8 Conclusions

A theoretical basis for change in ecological change has been developed and refined under the precepts of the Ramsar Wetlands Convention. This is based on a framework for wetland risk assessment which provides a holistic mechanism for making decisions on the choice of early warning indicators for predicting and assessing adverse change in the ecological character of wetlands. It is recommended that this framework is adopted and the early warning techniques instituted.

The framework encompasses the Ramsar Convention definitions of ‘ecological character’ and ‘change in ecological character’. The attributes of ‘early warning indicators’ for wetland loss and degradation are described. Examples of early warning indicators for wetland loss and degradation are provided and placed within the context of the framework for wetland risk assessment. The framework encompasses six basic steps (identification of the problem, identification of the effects, identification of the extent of the problem, identification of the risk, risk management and reduction, and monitoring). In conclusion we discuss the responsiveness of management systems to changes detected by early warning systems.

9 Acknowledgments

This paper was developed by the Environmental Research Institute of the Supervising Scientist, Jabiru, NT, Australia, in response to parallel requests from the Environment Australia National Wetland Program and the Ramsar Convention Scientific and Technical Review Panel (STRP). In developing the paper we have drawn heavily on the expertise and comments provided by our colleagues on the STRP and, in particular, from participants at a specialist workshop organised by the Ramsar Convention Bureau specifically to discuss these issues. David Pritchard and Nick Davidson graciously supplied further information following the workshop. We acknowledge the support given by Environment Australia and the Ramsar Convention STRP and Bureau.

The latter will use the paper as the basis of a recommendation to be considered at the 1999 Conference of the Convention. The material for this recommendation will be a summary of the main issues raised in the paper.

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Appendix A Summary of some potential methods of rapid response toxicity tests, field early warning tests (excluding biomarkers), and rapid assessments for use as early warning indicators of chemical change in ecological character

Method	**Organism**	**Measurable response(s)**[1]	**Laboratory/ Field (L/F)**	**Test duration**	**Attributes** (see section 5.4)	**Key references**
Laboratory and/or field toxicity assessment[2]	bacteria	luminescence, glucose U-^{14}C mineralisation,	L,F	5 – 60 minutes	i, iv, vi, vii, ix, x(?)[3], xi	Reteuna et al (1989); Belkin et al (1996)
	phytoplankton	population growth rate, biomass, ^{14}C uptake, respiration, fluorescence, species composition/size assemblage shifts, picoplankton distributions	L,F	2 hours – 4 days	i, ii, iv, v(?), vi, vii, viii, ix, x(?), xi	Munawar et al (1989); Munawar & Legner (1993); McCormick & Cairns (1994); Stauber (1995)
	macrophytes	frond necrosis/chlorosis, plant/frond number, root length, dry biomass, growth	L,F	4 – 7 days	i, ii, iv, v (?), vi, vii, viii, ix, x , xi	Smith & Kwan (1989); Lewis (1995)
	Mussels (*biological early warning systems*)	valve movements (opened/closed)	L,F	continuous	i, ii, iv, vi, vii, ix, x (?), xi	Kramer et al (1989); Borcherding & Volpers (1994); de Zwart et al (1995)
	invertebrates	swimming behaviour, swimming activity, respiration, feeding inhibition, reproduction, survival, population growth rate	L,F	1 – 21 days (depending on test and endpoint)	i, ii, iv, v(?), vi, vii, viii(?), ix, x, xi	Cairns & van der Schalie (1980); Humphrey et al (1990; 1994); Hendriks & Stouten (1993); Balk et al (1994)
	fish	avoidance, rheotaxis, ventilatory behaviour/ respiration, larval survival, embryo survival	L,F	2 – 14 days (depending on test and endpoint)	i(?), ii(?), iv, v(?), vi, vii, viii, ix, x, xi	Cairns & van der Schalie (1980); Sprague (1971); Hendriks & Stouten (1993); Balk et al (1994); Humphrey et al (1990; 1994)

Appendix A continued

Method	Organisms	Measurable response(s)	Laboratory/ Field (L/F)	Test duration	Attributes (see section 6.1)	Key references
Biological monitoring[4]	Bacteria	taxonomic analyses (see phytoplankton)	F	(see phytoplankton)	(see phytoplankton)	Munawar et al (1994)
	phytoplankton	assemblage structure, particularly for diatoms, assessment of the microbial loop	F	non-rapid	i, ii, iv, v, vii, viii, ix(?), x(?)	McCormick & Cairns (1994); Munawar et al (1994)
	macrophytes	species diversity, vegetation cover, trophic status	F	non-rapid	i(?), iv, v, vii, viii, ix(?)	Haslam (1982); Sortkjaer (1993); Lewis (1995)
	macroinvertebrates	community structure	F	non-rapid	i, iv, v, vii, viii, ix	
	fish	community structure, mean age, fecundity, condition factor	F	non-rapid	i(?), ii(?), iv, v(?), vii, viii, ix(?), x(?)	Munkittrick & Dixon (1989)
Rapid assessments	Rapid biological assessment (RBA)	Community structure of macroinvertebrates (methods for other groups being developed)	F	~ 24 h	Iv, v, vi, vii, viii, ix, x	Chessman (1995); Resh et al (1995); Wright (1995)
	Monitoring of birdlife	presence/absence, population alert limits	F	~ 24 h	Iv, v, vi, vii, viii, ix, x	
	Remote sensing	habitat extent,	F	On-going	Iv, v, vi, vii, viii, ix, x	Johnston & Barson 1993; Sahagian & Melack 1997; Phinn et al 1998

[1] The listed responses are examples, and are not intended to represent the full range of endpoints that can be measured.

[2] *Laboratory and field toxicity assessments* incorporates Rapid response toxicity tests (as in section 6.1), and *in situ* toxicity tests described in Early warning field tests (as in section 6.2).

[3] (?) indicates uncertainty about some, or all of the measurable responses possessing the attribute.

[4] Biological monitoring techniques are generally time consuming, however, sampling and sorting effort can vary according to the objectives of the program.

Appendix B Summary of a range of biomarkers used to predict and assess exposure and potential effects of chemical stressors to organisms[1]

Biomarker	Organism(s)	Measurable response(s)	Potential application(s)	Attributes (see section 5.4)	Key references
mixed function oxidases (cytochrome *P*-450 system)	mammals, fish, invertebrates, bacteria	enzyme activity/amount; mRNA; protein,	PAHs, PCBs, organochlorine pesticides, other chlorinated compounds	i, ii, iii, vi, vii, ix, x(?)[2], xi	Haux & Förlin (1988); Goksøyr & Förlin (1992); Stegeman et al (1992); Livingstone (1993)
vitellogenin	mammals, birds, fish, invertebrates	amount in plasma	Endocrine disruptors (eg aluminium, cadmium, mercury, lead, TBT, organochlorine pesticides, hexa-chlorocyclo-hexane, PCBs, PAHs)	i, ii, iii, vi, vii, ix, x(?), xi	Hontella et al (1993); Ankley et al (1998; Giesy & Snyder (1998); Arcand-Hoy & Benson (1998)
metallothionein (MT)	mammals, birds, fish, invertebrates, bacteria	amount of protein; amount of bound metal	cadmium, copper, zinc, mercury, silver, platinum, gold, bismuth	i, ii, iii, vi, vii, ix, x(?), xi	Haux & Förlin (1988); Stegeman et al (1992); Livingstone (1993)
serum-sorbital dehydrogenase (S-SDH)	mammals. fish	amount	general	i, ii, iv, vi, vii, ix, x, xi	Dixon et al (1987); Holdway et al (1994)
phase II conjugating enzymes: glutathione transferases, glucuronosyltransferases, sulfotranferases	mammals, birds, fish, invertebrates, bacteria	enzyme activity/amount	PAHs, PCBs, organochlorine pesticides, cadmium	i, iii, vi, vii, ix, x (?), xi	Stegeman et al (1992)
anti-oxidant enzymes: catalase, superoxide dismutase	all aerobic organisms, including plants	enzyme activity/amount	air pollutants (eg O_3, SO_2), direct acting oxidants (eg H_2O_2, chlorine), redox-active compounds (eg transition metals, bypyridyl herbicides)	i, ii, iv, vi, vii, ix, x, xi	Stegeman et al (1992)
stress proteins/heat shock proteins	mammals, birds, fish, invertebrates, bacteria	protein concentration	general	i, ii, iv, vi, vii, ix, x (?), xi	Stegeman et al (1992)
esterases: acetylcholinesterase (AChE), butyrylcholinesterase (BChE)	mammals, birds, fish	activity in blood/plasma/ brain	organophosphorus and carbamate insecticides, mercury	i, ii, iii, vi, vii, ix, x (?), xi	Mayer et al (1992); Johnston (1995); Walker (1995)

Appendix B continued

Indicator	Organisms	Measured response(s)	Potential application(s)	Attributes (see section 6.1)	Key references
haem/porphyrin pathway (eg δ-aminolevulinic acid dehydratase, δ-aminolevulinic acid synthase)	mammals, birds, fish	enzyme activity	general (eg lead, arsenic, mercury, PCBs, dioxins, hexachlorobenzene, alcohols, peroxidising herbicides)	i, ii, iv, vi, vii, ix, x (?), xi	Conner & Fowler (1994)
bioaccumulation	Fish, invertebrates (particularly molluscs)	Heavy metal and certain persistent organic chemical content in body tissues	Many heavy metals	i(?), ii(?), iii, vi(?), vii, ix	Martin (1992); Phillips (1992); Miskiewicz (1992); Jeffrees et al (1995)
DNA alterations	mammals, birds, fish	adducts (^{32}P-postlabelling); strand breaks (alkaline unwinding assay)	PAHs, PCBs, PCDDs	i, ii (?), iii, vi (?), vii, ix (?), x (?), xi	Shugart et al (1992); Livingstone (1993); Walker (1995); Depledge (1996)
chemical metabolites	mammals, birds, fish, invertebrates	presence/content in tissues/organs/fluids (eg muscle, liver, bile)	chlorinated organic compounds	i, ii, iii, vi (?), vii, ix (?), x, xi	Melancon et al (1992); Haritos et al (1995); Brumley et al (1996)
haematological parameters	mammals, birds, fish, invertebrates	haemoglobin content; haematocrit; white blood cell count	general	i, ii, iv, vi, vii, ix, x (?), xi	Everaats et al (1993); van Dam et al (1995)
ascorbic acid	fish	content in liver	general	i, ii (?), iv, vi, vii, ix, x (?), xi	Thomas et al (1982); Andersson et al (1987)
immune responses	mammals, birds, fish, bivalves	lymphocyte mitogenesis; natural cytotoxic cell activity/viability	immunotoxicants – eg organotins, heavy metals, pesticides, polycyclic and halogenated aromatic hydrocarbons	i, ii, iv (?), vi, vii, ix, x (?), xi	Weeks et al (1992); Holdway et al (1995)
tissue indices : condition, liver somatic index (LSI), gonad somatic index (GSI)	mammals, birds, fish	tissue weight as % of total body weight	general	i, ii (?), iv, vi, vii, ix, x (?), xi	Everaats et al (1993);
histopathology	mammals, birds, fish	incidence and type of lesions	general	i (?), ii, iv, vi (?), vii, ix (?), x (?), xi	Hinton et al (1992)
RNA:DNA ratio	mammals, birds, fish, invertebrates	ratio of RNA and DNA concentration	general	i, ii, iv, vi, vii, ix, x (?), xi	Mayer et al 1992; Holdway et al (1995); Reid (1997)

1 Note that all biomarkers lack the ecological relevance to be used for effects characterisation. Their major role will be for exposure characterisation and in monitoring the effectiveness of particular risk management decisions.

2 (?) indicates uncertainty about the biomarker possessing the attribute.

Protocols for an Australian national wetland inventory

CM Finlayson[1]

Abstract

It is intended to compile a National Wetland Inventory from existing datasets and inventory projects where possible. To assist in the collation and collection of data and information that can specifically contribute to the national inventory a number of protocols are presented. These have been derived from national and international reviews of inventory practices and developed with expert advice. A national program could be conducted under the auspices of the ANZECC Wetland and Migratory Waterbird Taskforce which contains representatives from all states/territories.

The National Wetland Inventory will contribute information to the ongoing Directory of Important Wetlands and provide a hitherto incomplete information source for wetland resource managers. Techniques for collecting and assessing the information are not provided; a separate technical manual is required.

The individual components of the protocols are summarised. They allow for different objectives and practices under a single goal. Further, they are in line with internationally accepted concepts and recommendations.

Goal

Provide the information essential for the conservation, wise use, management and restoration of Australia's wetland resource.

Objectives

The inventory encompasses a number of objectives:

- Establish a basis for wetland assessment and establishment of priorities for monitoring
- Provide input to national State of the Environment Reporting
- Support bioregional planning of land and water resources
- Fulfill national obligations under the National Strategies for Ecologically Sustainable Development and Conservation of Australia's Biological Diversity
- Fulfill national obligations under the Ramsar Convention on Wetlands and the Convention on Biological Diversity
- Provide an information base for increasing general awareness about the status and values of wetlands

Definition

The inventory covers coastal and inland wetlands. It does not cover marine wetlands. As such, two definitions are presented as alternatives, with the preferred being:

> Land permanently or temporarily under water or waterlogged. Temporary wetlands must have surface water or waterlogging of sufficient frequency and/or duration to affect the biota. Thus, the

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occurrence, at least sometimes, of hydrophytic vegetation or use by waterbirds are necessary attributes.

Classification

A number of classifications are currently being used and will continue in line with local objectives and established programs. The recommended classification is hierarchical and based on an initial matrix of landform and water regime features. Further separation is provided by the addition of descriptors for water salinity, size, plan shape and vegetation.

Water longevity	Landform				
	Basin	Channel	Flat	Slope	Highland
Permanent inundation	Lake	River	–	–	–
Seasonal inundation	Sumpland	Creek	Floodplain	–	–
Intermittent inundation	Playa	Wadi	Barlkara	–	–
Seasonal waterlogging	Dampland	Trough	Palusplain	Paluslope	Palusmont

Further comparisons with other classifications are required.

Review

Existing inventory programs that will support the National Wetland Inventory include:

- Queensland wetland inventory
- Murray-Darling Basin inventory and database
- Inland lakes used by waterbirds

In addition, the ongoing Directory of Important Wetlands has resulted in several projects that will contribute further information. Further technique development, such as that with radar imagery for tropical wetlands will also provide information for the inventory.

Delineation and mapping

In order to meet differences in objectives the following three scales of delineation and mapping are proposed

i Wetland regions with maps at a scale of 1:5 000 000

ii Wetland aggregations with maps at a scale of 1:250 000

iii Wetland sites with maps at a scale of 1:50 000 or 1:25 000

The boudaries of a wetland can be determined by assessing the extent of flooding and the vegetation and soil characteristics. It is important to record the variation in flooding.

Ecological characterisation and core data

Given the objectives for a national Wetland Inventory core data elements are required to provide a basis for:

- delineating wetland habitats
- describing the basic ecological character of the delineated habitats.

Delineation and characterisation based on a time series of data is the preferred situation, however, this may not be possible. Critically, sufficient information (core data) should be derived to enable the major wetland habitats (at least) to be delineated and characterised.

The ecological character of a wetland is defined as:

> The sum of the individual biological, chemical, and physical components of the ecosystem and their interactions which maintain the wetland and its products, functions and attributes.

For this to be determined the following core set of biological, chemical and physical data is needed:

- Area and boundary* (size and variation, range and average values)
- Location* (coordinates, map centroid, elevation)
- Geomorphic setting* (where it occurs within the landscape, linkage with other aquatic habitats, biogeographical region)
- General description (shape, cross section and plan view)
- Soil (structure and colour)
- Water regime (periodicity, extent of flooding and depth)
- Water chemistry (salinity, pH, colour, transparency)
- Biota (vegetation zones and structure, animal populations and distribution, and special features including characteristic or rare/endangered species)

The core data elements marked with an asterisk (*) could normally be derived from aerial photographs and/or satellite images as could some aspects of the general description, water regime and vegetation features.

The core data elements could be supplemented with further information from bibliographic and administrative sources.

Further data elements, more associated with wetland assessment than inventory, could also be collected, but not at the expense of making the inventory program too complex. These include:

- Landuse – local and in the catchment
- Impacts and threats to the wetland – within the wetland and in the catchment
- Land tenure and administrative authority – for the wetland and critical parts of the catchment
- Conservation and management status of the wetland – including legal instruments and social or cultural factors
- Climate and groundwater features – noting that catchment boundaries may not correspond with those of groundwater basins
- Management and monitoring programs – in place and planned

Data management

As the inventory will be composed of many individual data sets it is not necessary that all data is maintained within a single national dataset. As a minimum a national meta-database that meets national specifications should be maintained and be generally accessible through the World Wide Web.

Review

Progress with the inventory could be reviewed after an initial period of five years by an independent panel appointed by the national wetland forum that represents all states/territories (ANZECC Wetlands and Migratory Waterbird Taskforce). This could be done in time for reporting to the 2005 meeting of the Ramsar Convention on Wetlands and would entail regular interim reports through the forum.

1 Introduction

Spiers and Finlayson (1999) assessed the extent of wetland inventory and assessment information in Australia and concluded that it was uneven, fragmentary and incomplete. Overall, there was little evidence of a coherent approach to developing a truly national assessment of the location and ecological status of wetlands. They further asserted that current short-comings could only effectively be addressed through the development of a national approach to wetland inventory with an agreed goal and standardised techniques to systematically collate, collect, store, and disseminate data and information. The technical basis for such an approach is presented in this document. It is stressed that this is not a detailed manual or handbook for wetland inventory; it presents a set of steps and decisions that combined can be used as protocols for planning further wetland inventory as part of a national program.

Wetland inventory is considered a basic requirement for effective wetland management (Dugan 1990, Costa et al 1996, Finlayson 1996a). An inventory can take many forms (Finlayson & van der Valk 1995a) and contain an array of data. Essentially, an inventory provides a list of sites with information on location and size, biophysical features, values and benefits derived from the wetland, uses made of the wetland and its resources, threats to the wetland, status of the wetland, and management procedures (Costa et al 1996). The nature of the information contained within the inventory is very much related to the goal of the inventory program; although Finlayson and Davidson (1999) have pointed out that in many inventories the goal or objectives have not been clearly stated!

Costa et al (1996) provide the following statement about wetland inventory:

> An inventory should be undertaken within set objectives over a given time-period or as an ongoing project, with a final aim of publishing/disseminating the information or making this readily available in a database system.

A recent review of the extent of global wetland inventory (Finlayson & Davidson 1999) has illustrated the dismal state of wetland inventory with few examples meeting the expectations of Costa et al (1996). The recommendations produced by Finlayson and Davidson (1999) have been incorporated into the protocols presented below.

It is emphasised that the protocols provide a *technical basis* for such an approach. It is incumbent on relevant agencies and institutions to plan and implement the actual programs that are necessary to realise the agreed goal and purpose of a national, or indeed, any inventory. The protocols do not provide a policy basis for developing a national approach to wetland inventory.

It is further added that the protocols do not provide a basis for ongoing monitoring of wetlands. In line with international approaches wetland inventory is treated separately from monitoring (Costa et al 1996, Finlayson & Davidson 1999). However, it is recognised that an inventory can supply the basis for the assessment of wetlands and hence the development of monitoring strategies (eg to determine the overall health of sites listed under the Ramsar Convention on Wetlands). Wetland inventory is thus treated according to the internationally agreed definition provided by Finlayson et al (1999).

> *Wetland Inventory* is the collection and/or collation of core information for wetland management, including the provision of an information base for specific assessment and monitoring activities.

(The same authors have also defined wetland assessment and monitoring and these are presented in Appendix 1.)

2 Background

Under the Natural Heritage Trust the Commonwealth and State/Territory governments have agreed to develop a National Wetland Inventory assembled from complementary State/ Territory and Commonwealth databases. Thus, the inventory will build on past and existing wetland inventory where possible. However, a word of caution is issued at the outset as Spiers and Finlayson (1999) have found many inconsistencies in the recent wetland inventory effort in Australia, and Finlayson and Davidson (1999) have drawn attention to the inadequate state of much of the global wetland inventory effort.

The latter warning is particularly poignant given that much of the global wetland inventory was based on a similar model to that used for the *Directory of important wetlands in Australia* (ANCA 1996). At an international level such inventories have produced a vast amount of information, but they have not generally produced a comprehensive basis for further assessment and monitoring of the wetlands covered. This is, in part, a reflection of the difficulties associated with developing and implementing ambitious wetland inventories without adequate funding. Further, many funding bodies seem to have had overly high expectations from such exercises.

The review by Spiers and Finlayson (1999) and a subsequent review of the usefulness of remote sensing techniques for wetland inventory by Phinn et al (1999) were used as resource documents for a Specialist Workshop (see Appendix 2) to develop the basis of a national approach to wetland inventory in Australia. The results of these discussions have been incorporated in the protocols presented below. In doing this due notice was taken of recent inventory effort and every attempt made to incorporate these into the national protocols.

It is anticipated that these protocols will provide the basis for undertaking a more strategic and consistent approach to wetland inventory. They will also assist in the fulfillment of Australia's obligations under the Ramsar Convention on Wetlands and assist with implementation of the Wetlands Policy of the Commonwealth Government of Australia (Commonwealth of Australia 1997). As such the National Wetland Inventory could be overseen by the ANZECC Wetland and Migratory Waterbird Taskforce which contains representatives of all states/territories.

3 Protocols

The protocols are presented under headings that provide both background information and direction for further wetland inventory. The protocols are an amalgam of comments taken from the Specialist Workshop (Appendix 2), the recommendations provided by Finlayson and Davidson (1999) (Appendix 3) and information from the five volume Mediterranean Wetland Inventory manual (Costa et al 1996, Hecker et al 1996, Farinha 1996, Zalidis et al 1996, Tomas Vives et al 1996).

3.1 Goal

The goal of the National Wetland Inventory, as agreed in the specialist workshop, is to provide the information essential for the conservation, wise use, management and restoration of Australia's wetland resource.

3.2 Objectives

The goal presents a generic overview for the National Wetland Inventory and can support a number of key activities. These activities are contained within the following objectives:

- Establish a basis for wetland assessment and establishment of priorities for monitoring
- Provide input to national State of the Environment Reporting
- Support bioregional planning of land and water resources
- Fulfill national obligations under the National Strategies for Ecologically Sustainable Development and Conservation of Australia's Biological Diversity
- Fulfill national obligations under the Ramsar Convention on Wetlands and the Convention on Biological Diversity
- Provide an information base for increasing general awareness about the status and values of wetlands

These objectives are wide-ranging and cover many aspects of conservation and land/water regional planning. Hence they cross many sectoral divides between and within governmental agencies. They also address issues that cross the jurisdictional bounds of local, state/territory and federal authorities. Thus, the focus within the inventory process is to provide a basic set of core data that can be used by the various responsible agencies and interest groups to promote the wise use of wetlands across Australia. As such, the information obtained from the National Wetland Inventory will be useful for and used by a wide audience.

3.3 Definition of wetlands

Given the plethora of wetland definitions it is important to first determine the broad bounds of the habitats or landscape units being considered. For this inventory we are including all inland and coastal wetlands, with the latter being constrained by those that occur within the intertidal zone. Habitats that extend into the marine realm beyond the intertidal zone are not included, although it is noted that on occasions information on these marine habitats may be collected along with that from the coastal habitats.

A commonly used definition in Australia in recent years is that provided by the Ramsar Convention on Wetlands (Barson 1992) and which is given below (Davis 1994)

> Areas of marsh, fen, peatland or water, either natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres.

This is a broad definition and includes coastal and marine wetlands as well as those that occur inland. The inclusion of truly marine wetlands such as offshore reefs and seagrass beds provokes much contention (Pressey & Adam 1995; Finlayson & van der Valk 1995b). In many instances agencies responsible for 'marine wetlands' do not readily accept these as being covered by the relatively recent term 'wetland'.

Given the sectoral non-acceptance of the Ramsar definition other definitions have come into vogue. The definition used by Paijmans et al (1985) is well accepted and is given below. This includes coastal wetlands, such as mangroves and seagrasses within estuarine areas, but not marine habitats. It also includes areas of land that are dry for a substantial period; a necessary feature in a dry continent with strong seasonal precipitation over most of its area.

> Land permanently or temporarily under water or waterlogged. Temporary wetlands must have surface water or waterlogging of sufficient frequency and/or duration to affect the biota. Thus, the

> occurrence, at least sometimes, of hydrophytic vegetation or use by waterbirds are necessary attributes.

The Paijmans et al (1985) definition was used in a recent review of national wetland research and development priorities (Bunn et al 1997).

A further definition is given by Semeniuk (1987), see below, and includes areas with waterlogged (saturated) soils, ponds, lakes, swamps, rivers and their tributaries, as well as marine and coastal wetlands such as tidal flats and estuaries.

> Areas of seasonally, intermittently or permanently waterlogged soils or inundated land, whether natural or otherwise, fresh or saline.

Whilst this definition could be seen as being similar to that provided by Paijmans et al (1985) it has not been used as widely.

It is recommended that where possible the definition provided by Paijmans et al (1985) be used for the National Wetland Inventory. In making this recommendation it is recognised that this definition more closely reflects the common and even governmental, as indicated by sectoral divides within agencies and the like (Finlayson & Spiers 1999), view of wetlands. The Ramsar definition does not have this.

Adoption of this recommended definition should not detract from plans to extend the *Directory of Important Wetlands in Australia* which in many instances does not contain extensive listings of marine wetlands (Spiers & Finlayson 1999). Thus, the National Wetland Inventory will provide a subset of the information that may be reported within the format currently used for the Directory. This decision also reflects the sectoral divisions within Environment Australia where marine issues are handled by a separate section to that which handles wetland issues.

3.4 Classification

Wetland classification has consumed an almost inordinate amount of time and controversy (Finlayson and van der Valk 1995b). Many systems have been developed and used within Australia (see review in Finlayson and von Oertzen 1993). Three classifications are outlined in this discussion. All are currently in use in Australia and this situation may continue. Thus, whilst the logically reasoned and consistently constructed classification system originally presented by Semeniuk (1987) is recommended for the National Wetland Inventory, a tabulated partial comparison with other systems is provided. This classification system is particularly amenable to a scalar approach to wetland inventory and facilitates rapid assessments at the national scale under an agreed scheme of bioregionalisation. It is anticipated that more detailed classifications may suit inventory at the sub-national or localised scales.

Before describing the various classification schemes commonly used in Australia a word about the need for classification. As shown in Section 6 below, wetland inventory can go ahead without recourse to an agreed classification, given that a standardised and logical process of data collection or collation is undertaken. However, at some stage during the assessment phase of wetland management it becomes necessary to make comparisons and then choices for actions and funding. At this stage an agreed set of terms is not only desirable but possibly mandatory to ensure conformity of comparisons and hence decisions. Thus, the importance of classification can not be overstated, but it equally needs to be remembered that classification is a tool within a larger set of tools that are designed to provide an adequate information base for the wise use, conservation and management of all wetlands.

3.4.1 Directory of Important Wetlands

The Directory of Important Wetlands (ANCA 1996) has adopted a modified version of the classification of wetland types used by the Ramsar Convention (1989, Davis 1994). The classification agreed for the Directory is given in Table 1. As with the Ramsar classification, that used for the Directory has an initial separation of wetlands into three major classes – marine/coastal, inland, and artificial. At the next stage the level of detail differs with the Ramsar classification being further divided whereas that used for the Directory reverts to a simple listing of habitat types.

The Ramsar classification was not developed as an all encompassing classification system. The categories provided under the Ramsar classification present a broad framework to aid rapid identification of the main wetland habitats represented at a site regarded as internationally important (Davis 1994). It was not anticipated that it would be the precursor, let alone the template for an international classification scheme (Scott & Jones 1995). However, it has continued to be used as the basis for many wetland inventories that possibly failed to fulfill their potential due to the acknowledged limitations of the typology (see papers in Finlayson and Spiers 1999).

The system presented by Scott (1989) and further modified (Davis 1994, Ramsar Convention Bureau 1997) has possibly been adopted as it seems easy and straight forward compared to other classifications. However, this advantage could soon be lost if the inconsistencies etc (Semeniuk & Semeniuk 1997) detract from the objectives and purpose of the inventory program.

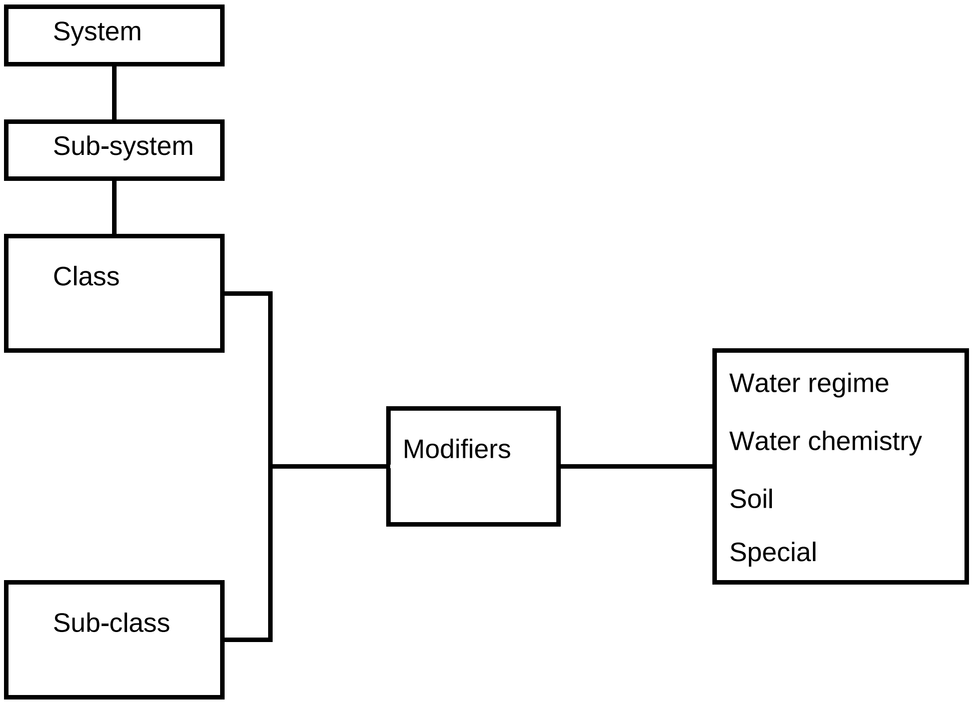
Table 1 Wetland classification used for A directory of important wetlands in Australia (ANCA 1996)

Marine and coastal wetlands	Inland wetlands	Peatlands
Marine waters	Permanent rivers and streams	Alpine and tundra wetlands
Subtidal aquatic beds	Seasonal and irregular rivers and streams	Freshwater springs, oases and rock pools
Coral reefs	Inland permanent deltas	Geothermal wetlands
Rocky marine shores	Riverine floodplains	Inland subterranean karst wetlands
Sand, shingle or pebble beaches	Permanent freshwater lakes	Human-made wetlands
Estuarine waters	Seasonal/intermittent freshwater lakes	Water storage areas
Intertidal mud, sand or saltflats	Permanent saline/brackish lakes	Farm/stock ponds, and small tanks
Intertidal marshes	Seasonal/intermittent saline lakes	Aquaculture ponds
Intertidal forested wetlands	Permanent freshwater ponds, marshes and swamps on inorganic soils	Salt pans
Brackish-saline lagoons	Seasonal/intermittent freshwater ponds and marshes on inorganic soils	Excavation pits
Freshwater lagoons	Permanent saline/brackish marshes	Wastewater treatment ponds
Non-tidal freshwater forested lagoons	Seasonal saline marshes	Irrigated land and channels
	Shrub swamps	Seasonally flooded arable land
	Freshwater swamp forest	Canals

3.4.2 Queensland wetland and deepwater habitats

The Ramsar classification was itself loosely based on that developed for the wetland and deepwater habitats in the USA (Cowardin et al 1979, Cowardin and Golet 1995) and this has been developed further and in great detail for the Mediterranean Wetland Inventory (Farinha et al 1996). Similarly, Blackman et al (1992) have modified the Cowardin et al (1979) scheme for use in Queensland deepwater and wetland habitats.

This has been successfully applied in Queensland (Blackman et al 1995) and uses a hierarchical approach from systems and subsystems at the most general levels, to class, subclass and dominance types at the lowest levels (Figure 1). At the lower levels of the classification modifiers for water regime, water chemistry, and soils are applied to all classes and sub-classes. Special modifiers describe artificial wetlands or those that have been substantially altered by human activity.



System – Complex of wetlands and deepwater habitats influenced by similar hydrologic, geomorphic, chemical or biologic factors.

Class – General appearance of the habitat based on features that can be recognised without detailed measurements, such as physiography and composition of the substrate, or the dominant vegetation life form

Modifiers – Provide the means of actually describing the wetland habitats

Figure 1 Hierarchical framework for regional classification of Queensland wetlands and deepwater habitats (from Blackman et al 1992)

The manual provided by Blackman et al (1992) provides detailed instruction and guidance to use this classification system. An overview of the classification system is shown in Table 2. Whilst Blackman et al (1992) provide a detailed manual for using this classification it is not assured that such a detailed scheme would be widely accepted or used. The latter will be in part related to the specific objective(s) of the inventory and the adequacy of the resources. Blackman et al (1996) have certainly shown that a detailed description of wetlands in parts of Queensland can be based on this classification given time and effort.

Table 2 Wetland classification used by Blackman et al (1992) as modified from that used in the USA by Cowardin et al (1979)

Systems	Marine – comprising two subsystems	Subtidal Intertidal
	Estuarine – comprising two subsystems	Subtidal Intertidal
	Riverine – comprising four subsystems	Tidal Lower perennial Upper perennial Intermittent
	Lacustrine – comprising two subsystems	Littoral Limnetic
	Palustrine – comprising no subsystems	
Classes	Substrate and flooding regime – comprising six classes	Rock bottom Unconsolidated bottom Rocky shore Unconsolidated shore Streambed Reef
	Vegetative life form – comprising five classes	Aquatic bed Moss/lichen Emergent wetland Scrub-shrub wetland Forested wetland
Modifiers	Water regimes – comprising two headings	Tidal water regimes Non-tidal water regimes
	Water chemistry – comprising two headings	pH – acid, circumneutral, alkaline Salinity – coastal fresh to hyperhaline Salinity – inland fresh to hypersaline
	Soil – comprising two headings for wetland habitats only	Mineral Organic
	Special – comprising six headings	Excavated Dyked/impounded Partially drained/ditched Artificial Farmed Spoil

3.4.3 Geomorphic landform and water characteristics

Semeniuk (1987) has presented a wetland classification that is based fundamentally on the two features that determine the existence of all wetlands – namely, landform and water regardless of the climatic setting, soil type, vegetation cover, or origin. This classification brings out the underlying and unifying features of wetlands that occur across the spectrum of climatic and physiographic settings. In extending this classification Semeniuk and Semeniuk (1995, 1997) point out that this system also overcomes major inconsistencies in other systems that have been primarily based on vegetation characteristics, either by themselves or in association with soil/substrate patterns or inundation patterns.

By classifying wetlands initially on the basis of five landform attributes and four water characteristics some 13 categories have been identified (Table 3). These are mutually exclusive categories and provide a more consistent basis for identifying wetlands. Semeniuk and Semeniuk (1997) have illustrated that classifications systems that use a jumble of vegetation, soil, inundation and landform features are not consistent and have added to the confusion in typology. Adoption of this system and its basic landform and water characterisation facilitates a scalar approach to classification and hence to wetland inventory without being entrapped initially by, for example, vegetation features that are not independent of climatic or soil characteristics.

The categories used in the geomorphic system have been given single-word terms that avoid confusion with existing commonly used names for wetlands types, such as bog or marsh. This approach brings out the underlying similarity of wetlands across a wide range of climatic, geomorphic, soil, and vegetation settings based on the rationale that landform and water characteristics are the dominant and/or common feature for all wetlands, regardless of their setting.

Table 3 Schematic presentation of the classification of wetland habitats based on landform and water characteristics (from Semeniuk & Semeniuk 1997)

Water longevity	Landform				
	Basin	Channel	Flat	Slope	Highland
Permanent inundation	Lake	River	–	–	–
Seasonal inundation	Sumpland	Creek	Floodplain	–	–
Intermittent inundation	Playa	Wadi	Barlkara	–	–
Seasonal waterlogging	Dampland	Trough	Palusplain	Paluslope	Palusmont

The classification can be extended to a further level by the addition of descriptors for salinity – fresh, brackish, saline or hypersaline. These are general terms and can be used as such. Seasonal variability is further described by the terms stasohaline (the salinity is fairly consistent throughout the year) and poikilohaline (the salinity fluctuates markedly throughout the year). The shape of the landform can also be described as linear, elongate, irregular, fan-shaped, ovoid or round for basins, slopes and hills/highlands, and straight, sinuous, anastomosing, or irregular for channels. Further vegetation can be described in terms of the cover (peripheral, mosaic and complete) and complexity (homogeneous, zoned and heterogeneous) with specific names being provided by Semeniuk and Semeniuk (1997), but not necessarily needed. The size of a wetland can also be described by five terms – megascale (very large, 10 km x 10 km), macroscale (large, 1000 m x 1000 m), mesoscale (medium, 500 m x 500 m to 1000 m x 1000 m), microscale (small, 100 m x 100 m to 500 m x 500 m) and leptoscale (very small, < 100 m x 100 m).

The classification has been successfully applied in a number of locations and a comparison made with the inland wetland category in the Ramsar classifications (Semeniuk and Semeniuk 1997) and is shown in Table 4. This tabulation illustrates the inconsistent landform categorisation used in the Ramsar classification. In assessing the inland categories for the Ramsar classification mention is made of a number of inconsistencies and gaps that subtract from its suitability as a globally applicable system. The gaps are particularly relevant to the Australian National Wetland Inventory. In contrast, the landform and water characteristics used by Semeniuk and Semeniuk (1997) were derived from Australian analyses of wetland types. As such they allow classification at a variety of tiers that can accommodate the need for either a generic overview or more detail according to one or other set of descriptors, as based on the individual objective(s) of the inventory.

Table 4 Comparison between the inland wetland categories used by Semeniuk and Semeniuk (1997) and the Ramsar Convention on Wetlands

Semeniuk & Semeniuk subdivisions		Ramsar subdivisions
Basins	Permanent inundation – lake	Permanent freshwater lake
		Permanent saline lake
		Permanent freshwater pool
		Open peat fens
		Shrub dominated swamp
		Freshwater swamp forest
		Peat swamp forest
	Seasonal inundation – sumpland	Seasonal freshwater lake
		Permanent saline lake
		Seasonal saline lake
		Permanent saline lake
		Seasonal saline marsh
		Permanent freshwater marsh
		Peat swamp forest
		Oases
	Intermittent inundation – playa	Not covered
	Seasonal waterlogging – dampland	Permanent freshwater marsh
		Seasonal freshwater marsh
Channels	Permanent inundation – river	Permanent river
		Permanent stream
	Seasonal inundation – creek	Intermitten river
		Intermittent stream
	Intermittent inundation – wadi	?Intermittent river
		?Intermittent stream
	Seasonal waterlogging – trough	Not covered
	Seasonal inundation – floodplain	Floodplain wetland
		Inland delta (part of)
	Intermittent inundation – barlkarra	Not covered
	Seasonal waterlogging – palusplain	Floodplain wetland (part of)
Slopes	Seasonal waterlogging – paluslope	Open peat bog (part of)
		Alpine/tundra wetland (part of)
		Freshwater spring (part of)
Hills/Highlands	Seasonal waterlogging – palusmont	Open peat bog (part of)
		Alpine/tundra wetland (part of)
		Geothermal wetland
		Karst and cave systems

The acquisition of data for classifying wetlands by this system can be made by the following four steps:

1. Assessment of geomorphic setting from aerial photographs to provide the landform setting (ie basin, channel, flat, slope, or hill/highland) and possibly extended to cover wetland shape and size, vegetation and hydroperiod (using time series data).
2. Preliminary field survey to determine hydroperiods, soils, etc to classify as one of the 13 wetland types, and possible application of modifiers for soil, vegetation and water quality.
3. Field survey to determine more detailed hydroperiods, water chemistry, soils and biota.
4. Field survey to determine more detailed information on seasonal and long-term dynamics to discriminate further, if required..

The flexibility of the approach is highlighted in that classification is useful at all steps, and is refined as more detailed information becomes available.

3.5 Collation and review of existing information

Substantial wetland inventory effort is already underway and it is anticipated that these initiatives will contribute information to the National Wetland Inventory, in line with the Natural Heritage Trust agreements. A complete review of the extent of wetland inventory in Australia, both current and present, has not been undertaken. Further, a gaps analysis of the *Directory of important wetlands in Australia* (ANCA 1996) has not been undertaken (Spiers & Finlayson 1999).

Spiers and Finlayson (1999) have also pointed out that the Directory is not an even or complete record of wetlands across Australia. Watkins (1999) undertook a preliminary review only of national wetland effort in Australia and noted many gaps or inconsistencies. Given this situation it is recommended that a thorough review of wetland inventory information is undertaken and a central repository of inventory sources established. The results of the review could be used to assist in setting priorities for further inventory effort at the national, state/territory and local scales. It could further be linked to the national Directory program.

A review and collation exercise would build on existing sources of information (eg ANCA 1996, Spiers and Finlayson 1999, Watkins 1999) plus specific projects such as that in the Murray-Darling Basin (Kingsford et al 1997), in Queensland (Blackman et al 1992, 1996) and inland Australia (D Roshier pers comm). Further, projects designed to develop techniques for more effective inventory, such as that using radar imagery for tropical wetlands (Milne 199) can also include information. Other sources of information are presented in Finlayson and Spiers (1999) and Watkins (1999). It will probably also be beneficial to access documents non-conventional sources of information on wetlands and their resources, such as fisheries and land-use analyses (Finlayson & Davidson 1999).

3.6 Delineation and mapping

The principal purpose of the National Wetland Inventory is to delineate and map Australia's wetland resource taking in wetland habitats across the intertidal zone to the inland. It is intended that this would occur without prejudice of size of wetland, but would be influenced by conservation and management priorities within each state/territory. Further, this is likely to occur on totally different scales given the objectives and management priorities within each jurisdiction.

Initially, however, it is necessary to delineate the extent of each wetland. Given a landscape basis for classifying wetlands this is not considered a difficut task, especially if it is coupled with an analysis of the soil and vegetation characteristics. The classification provided by Paijmans et al (1985) provides some initial indicators for delineation. First, the water regime is a key indicator – if the land is permanently or temporarily under water or waterlogged it is considered to be a wetland, with the proviso that surface water or waterlogging is of sufficient frequency and/or duration to affect the biota. Thus, the area being considered contains, at least sometimes, hydrophytic vegetation. Variation in the extent of flooding can be reported and interpreted, using information derived from either ground or remote sensing surveys, on the basis of the area inundated and the type of vegetation present, or if vegetation is not present, on the type of soils. Paijmans et al (1985) and Semeniuk (1987) provide further information on these features. When delineating a wetland it is important that the extent or variation of flooding should be recorded regardless of the vegetation as wetlands can not be managed in isolation of the surrounding environment.

In order to meet differences in objectives the following three scales of delineation and mapping are proposed:

1. Wetland regions with maps at a scale of 1:5 000 000
2. Wetland aggregations with maps at a scale of 1:250 000
3 Wetland sites with maps at a scale of 1:50 000 or 1:25 000

To provide a ‘foundation statement’ on the occurrence and status of wetlands in Australia it is proposed that delineation and mapping is undertaken at the regional scale. The regions will correspond to those within the interim biogeographical regions of Australia (Thackway & Cresswell 1995) to maintain consistency with other natural resource programs.

The production of the 1:5 000 000 maps will provide a basis for more detailed delineation and mapping of wetland aggregations. An aggregation is determined on the basis of land systems that are united by function and/or origin (Blackman et at 1995). These provide the basis for lower order delineation of wetlands within the biogeographical scales that have been delineated as a basis for national summary and priority setting purposes. Thus, by assessing the basis of the landforms and water regimes likely to occur within a region an initial outline of the wetland types present can be made. This is comparable with the determination of consanguineous suites of wetlands as undertaken by Semeniuk (1988).

Whilst this will provide a logical progression in scale it is expected that delineation and mapping at the wetland site scale will occur in tandem as state/territory agencies address specific management priorities. This is seen as advantageous as the addition of information at the site level should assist with maintaining the goal of the National Wetland Inventory. Specific management actions will occur at the site level, but without a strategic outline as provided by the ‘aggregational’ and/or ‘regional’ levels of mapping it is difficult to see a national perspective being developed and maintained.

3.7 Ecological characterisation and core data

Given acceptance of the habitats that will be covered by the National Wetland Inventory some attention is required to determine the basic core data that will comprise the inventory. Such data could be collected independently of the classification system and hence provide a basis for further comparison and elaboration as required. This is implicit in the four-step process outlined above from Semeniuk and Semeniuk (1997). Thus, delineation and characterisation of wetland habitats is not constrained, as the focus of the inventory is the core data elements and not the specific features of the classification.

Given the above mentioned objectives for a national Wetland Inventory the core data elements are required to provide a basis for:

- delineating wetland habitats
- describing the basic ecological character of the delineated habitats.

These tasks could be done with or without a seasonal or long-term context. Delineation and characterisation based on a time series of data is the preferred situation. However, the reality may be that an inventory is based on survey data with little opportunity for more comprehensive surveillance to provide time series information. The critical point is that sufficient information should be derived to enable the major wetland habitats (at least) to be delineated and characterised (corresponding to steps 1 and 2 of the process outlined by Semeniuk and Semeniuk 1997).

3.7.1 Ecological character of a wetland

Whilst wetland delineation has attracted a large amount of technical attention and argument (Finlayson & van der Valk 1995b), far less attention has been directed to determining what is meant by the ecological character of a wetland (as opposed to measuring the ecological health of a wetland). In recent years this has been addressed by the Ramsar Convention on Wetlands and the following working definition accepted (see Recommendation 6.1 – *Working definition of ecological character, guidelines for describing and maintaining the ecological character of listed sites, and guidelines for operation of the Montreux Record*, as outlined by Finlayson 1996b).

> The structure and inter-relationship between the biological, chemical, and physical components of the wetland. These derive from the interactions of individual processes, functions, attributes and values of the ecosystem(s).

(Definitions of processes, functions, attributes and values are provided in Appendix 1.)

However, the Convention's Scientific and Technical Review Panel has rejected this definition in favour of the following (www.ramsar.org/key_res_vii.10e.htm).

> The sum of the individual biological, chemical, and physical components of the ecosystem and their interactions which maintain the wetland and its products, functions and attributes.

For the purpose of the National Wetland Inventory is not critical which of these two definitions is accepted. The main difference between them is the manner in which they relate the biological, chemical and physical components of a wetland to the values and benefits derived from the wetland; this is essentially a management issue that can be pursued independently of an inventory exercise. For the ecological character of any wetland to be determined a core set of biological, chemical and physical data is needed.

3.7.2 Core data elements

It is initially assumed that all data collection will be accompanied by a discrete description of the person or persons responsible for collecting the data. Thus, the recorder(s) should be identified in all instances by name and contact details, including, if applicable, the agency name and general contact details. Further, the date and time of all data collection should be recorded. These basic data should head all data records.

The collection of core data elements will provide a basis for comparing the basic features of wetlands at a national and a sub-national level. The data elements will provide a basis for future classification and even reclassification of wetland habitats. They will also provide a basis for describing the ecological character of the wetland. The core date elements are given below along with some guidance on the type of information that would preferably be collected.

- Name of the wetland and location* (coordinates, map centroid, elevation)
- Area and boundary* (size and variation, range and average values)
- Geomorphic setting* (where it occurs within the landscape, linkage with other aquatic habitats, biogeographical region)
- General description (shape, cross section and plan view)
- Soil (structure and colour)
- Water regime (periodicity, extent of flooding and depth)
- Water chemistry (salinity, pH, colour, transparency)
- Biota (vegetation zones and structure, animal populations and distribution, and special features including characteristic or rare/endangered species)

The core data elements marked with an asterisk (*) could normally be derived from aerial photographs and/or satellite images (see Phinn et al 1999) as could some aspects of the general description, water regime and vegetation features. This corresponds to stage 1 of the data collection scheme outlined by Semeniuk and Semeniuk (1997) and presented above. The core data elements not marked with an asterisk require field surveys and possibly some further basic analyses and would come under stage 2 of the Semeniuk and Semeniuk (1997) scheme.

The core data elements could be supplemented with further information from bibliographic and administrative sources. However, it should be borne in mind that Finlayson and Davidson (1999) note that making the inventory more complex through the collection of further data fields can detract from the effort required to obtain the basic core data. Thus, a basic approach that focuses on the core data elements is recommended with further effort being held in abeyance until this is complete. This is particularly important as the core data elements provide the inventory whilst the additional data elements (as given below) are better considered within the assessment phase of wetland management.

However, given that inventory and assessment could be considered as components of a data collecting continuum these further data elements are listed below. Thus, further data elements, more associated with wetland assessment than inventory, include:

- Landuse – local and in the catchment
- Impacts and threats to the wetland – within the wetland and in the catchment
- Land tenure and administrative authority – for the wetland and critical parts of the catchment
- Conservation and management status of the wetland – including legal instruments and social or cultural factors
- Climate and groundwater features – noting that catchment boundaries may not correspond with those of groundwater basins
- Management and monitoring programs – in place and planned

The note of caution issued by Finlayson and Davidson (1999) is based on an analysis that has shown that many countries are still lacking even the most basic data on their wetland. Without this basic data it is nigh impossible to improve the decision making processes that have hitherto led to a worldwide loss of wetland resources (1999). Further, in separate analyses Spiers and Finlayson (1999) and Watkins (1999) report that the basic wetland inventory effort in Australia is far from complete.

3.8 Data management

The need to manage wetland inventory data has been outlined by Finlayson and Davidson (1999). As the wetland inventory is a national program and will be composed of many individual data sets it is not considered necessary that all data is maintained within a single national dataset. This consideration is underpinned by the assumption that at the minimum a national meta-database is maintained. Further, this would meet national specifications and be generally accessible via the World Wide Web.

Thus, all data sources used in the compilation of the National Wetland Inventory should be entered into a nationally accessible meta-database that incorporates national standards for data management. One such format is provided in table 5.

3.9 Review and gaps analysis

As it is the stated intent to compile the National Wetland Inventory from individual datasets held by jurisdictions across Australia a process of review and indeitifcation of gaps should be undertaken. This could most conveniently be organised through the existing national forum for wetland planning, the ANZECC Wetlands and Migratory WaterbirdsTaskforce. This includes representatives from all states/territories and is augmented by invited experts. The review could best be undertaken by an expert panel appointed by this forum after an initial five years after adoption and in time for reporting to the 2005 meeting of the Ramsar Convention on Wetlands.

Table 5 Meta-database format proposed by Finlayson and Davidson (1999) for recording details of individual wetland inventory projects as components of the National Wetland Inventory

Data	Description	Data Currency	Data Status	Access	Data Quality	Contact Information	Metadata Date	Additional Metadata
Title	Abstract	Begin date	Progress	Data format	Lineage	Contact organisation	Metadata date	Additional metadata
Jurisdiction	Search words	End date	Update Frequency	Available format	Positional accuracy	Contact position		
Custodian	Extent			Access constraint	Attribute accuracy	Mail address		
					Logical consistency	Place		
					Completeness	State		
						Country		
						Postcode		
						Telephone		
						Facsimile		
						Email		

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Appendix 1 Internationally agreed definitions used in these protocols

The following definitions for wetland survey, surveillance and monitoring were provided in a paper presented by Finlayson (1996b) using information adopted from Hellawell (1991). The definitions were accepted by the Conference of the Contracting Parties to the Convention and incorporated in Recommendation 6.1 – Working definition of ecological character, guidelines for describing and maintaining the ecological character of listed sites, and guidelines for operation of the Montreux Record, as outlined by Finlayson (1996b).

> **Survey** is an exercise in which a set of qualitative observations are made but without any preconception of what the findings ought to be.
>
> **Surveillance** is a time series of surveys to ascertain the extent of variability and/or range of values for particular parameters.
>
> **Monitoring** is based on surveillance and is the systematic collection of data or information over time in order to ascertain the extent of compliance with a predetermined standard or position.

Definitions for wetland inventory, assessment and monitoring were agreed during the 2nd International Conference on Wetlands and Development in Dakar, Senegal, November 1998 (Finlayson et al 1999).

> **Inventory** is the collection and/or collation of core information for wetland management, including the provision of an information base for specific assessment and monitoring activities.
>
> **Assessment** is the identification of the status of, and threats to, wetlands as a basis for the collection of more specific information through monitoring activities.
>
> **Monitoring** is the collection of specific information for management purposes in response to hypotheses derived from assessment activities, and the use of these monitoring results for implementing management. (Note that the collection of time-series information that is not hypothesis-driven from wetland assessment should be termed surveillance rather than monitoring.)

The following definitions for wetland values and benefits, encompassing products, functions and attributes, were provided in a paper presented by Finlayson (1996b) using information adopted from Davis (1994), Dugan (1990) and Maltby et al (1994). The definitions were accepted by the Conference of the Contracting Parties to the Ramsar Convention on Wetlands and incorporated in Recommendation 6.1 – Working definition of ecological character, guidelines for describing and maintaining the ecological character of listed sites, and guidelines for operation of the Montreux Record, as outlined by Finlayson (1996b).

> **Functions** performed by wetlands include the following: water storage; storm protection and flood mitigation; shoreline stabilisation and erosion control; groundwater recharge; groundwater discharge; retention of nutrients, sediments and pollutants; and stabilisation of local climatic conditions, particularly rainfall and temperature. These functions are the result of the interactions between the biological, chemical and physical components of a wetland, such as soils, water, plants and animals.
>
> **Products** generated by wetlands include the following: wildlife resources; fisheries; forest resources; forage resources; agricultural resources; and water supply. These products are generated by the interactions between the biological, chemical and physical components of a wetland.
>
> **Attributes** of a wetland include the following: biological diversity; geomorphic features; and unique cultural and heritage features. These have value either because they induce certain uses or because they are valued themselves.
>
> The combination of wetland functions, products and attributes give the wetland **benefits and values** that make it important to society.

Appendix 2 Specialist workshop to outline approaches for a national wetland inventory

This workshop was held in Darwin on 31 October and 1 November 1998 to discuss protocols for a national approach to wetland inventory. The workshop was organised by the Environmental Research Institute of the Supervising Scientist with funding support from the Environment Australia National Wetland Program.

The workshop comprised two components. In the first, examples of current inventory effort and techniques were presented and discussed. In the second component a number of specific topics were introduced and discussed as the basis for the draft protocols.

Informal presentations were made by

- Vic Semeniuk – Inventory and classification of wetlands in Western Australia
- Gavin Blackman – Inventory and classification of wetlands in Queensland
- Richard Kingsford – Inventory and databases for wetland inventory in the Murray-Darling Basin
- David Roshier – Inventory of inland lakes used by waterbirds
- Joanna Ellison – Inventory of mangroves and associated wetland habitats
- Doug Watkins – Review of wetland inventory in Australia
- Stuart Phinn – Usefulness of remote sensing for wetland inventory
- Tony Milne – Use of radar imagery for wetland inventory in tropical Australia

Following these presentations participants addressed each of the following topics and provided guidance for drafting the protocols for a national wetland inventory.

- Objective of a national wetland inventory and regional needs for inventory information
- Extent of existing wetland inventory and classification efforts
- Habitats that would be encompassed within a national wetland inventory
- Techniques for undertaking a national wetland inventory, including issues of scale and spatial and temporal variability
- Techniques for managing the data generated from a national wetland inventory, including storage and analysis but also ownership and access
- Means of updating and utilising the inventory as a component of a national wetland monitoring approach

It was envisaged that a number of approaches would be necessary for a national wetland inventory given differences in objectives, existing data and information, current inventory programs, and major constraints on resources. A single approach was not expected. The discussion in the workshop was designed to outline the approaches currently in use and to recommend common ground or priority approaches. The discussion was also used to identify major gaps in techniques and areas that required priority attention.

Participants in the workshop

BLACKMAN Gavin	Queensland Department of Environment, Townsville
CHURCHILL Ben	Wetland and Migratory Waterbird Unit, Biodiversity Group, Environment Australia, Canberra
CORRICK Andrew	Department of Conservation and Natural Resources, Melbourne
ELLISON Joanna	University of Tasmania, Launceston
FINLAYSON Max	Environmental Research Institute of the Supervising Scientist, Jabiru
HANDLEY Michelle	WWF Australia, Perth
KINGSFORD Richard	National Parks and Wildlife Service, New South Wales, Sydney
MILNE Tony	University of New South Wales, Sydney
PHINN Stuart	University of Queensland, Brisbane
ROSHIER David	Charles Sturt University, Wagga Wagga
SEMENIUK Vic	V&C Semeniuk Consulting Group
SPIERS Abbie	Environmental Research Institute of the Supervising Scientist, Jabiru
WATKINS Doug	Wetlands International Oceania, Canberra

Appendix 3 Recommendations for future wetland inventory

The executive summary and recommendations are taken from the summary report presented by Finlayson and Davidson (1999) from a project to review the extent of global wetland resources and to identify priorities for wetland inventory.[1] This project was undertaken by Wetlands International under contract to the Bureau of the Ramsar Convention on Wetlands and with funding from the Government of the United Kingdom. The review was coordinated by the International Coordination Unit from Wetlands International in conjunction with the Environmental Research Institute of the Supervising Scientist (Jabiru, Australia) in support of the Wetland Inventory and Monitoring Specialist Group.

Executive summary

This summary is based on reviews of the extent of wetland inventory in each Ramsar region. These were supplemented by a review of regional and international wetland inventories. Standardised data collation and recording formats were used in each of the reviews.

It is important to note that these reviews were limited by available funds and time, and that further effort will unearth more information.

It was not possible to make reliable overall estimates of the size of the wetland resource globally or regionally. Some good examples of wetland inventory processes exist (eg the MedWet program), but many inventories allowed only a cursory assessment of the extent of wetland area or condition. Whilst not undermining the value of individual inventories, this highlights wetland inventory as being incomplete and difficult to undertake.

Recommendations are made to improve the accuracy of quantifying and describing the wetland resource through wetland inventory, and to provide the basic information required for managing the wetland resource.

Recommendations focus on the need to conduct national inventory programs, and the inclusion of basic information on the location and extent of each wetland and its major ecological features as a forerunner to collecting further management-oriented information.

Development of standardised methods for data collection, collation and storage are called for. These methods should address the use of relatively new techniques for collecting and interpreting remotely-sensed data; storing this in electronic formats, including Geographic Information Systems (GIS); and recording key information in a meta-database.

The key conclusion of this review is that little is still known about the extent and condition of the global wetland resource. On a regional basis only parts of North America and Western Europe have adequate past and current inventory. Without good inventory it is difficult to promote the wise use of the wetland habitats covered by the Ramsar Convention.

Priority habitats for future inventory are identified. These are seagrasses, coral reefs, salt marshes and coastal flats, mangroves, arid-zone wetlands, peatlands, rivers and streams and artificial wetlands.

The Ramsar Convention should play a pivotal role in implementing these recommendations.

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[1] Excerpt from Finalyson CM & Davidson NC (1999). Global review of wetland resources and priorities for wetland inventory: Summary report. Supervising Scientist Report 144, Supervising Scientist, Canberra.

Recommendations

This review makes many critical comments on the state of global wetland inventory. In summary, global wetland inventory is incomplete and inadequate for most management purposes. From our many comments eight are recommended for priority action. These reflect the effort required to implement an effective inventory program as the basis for wise use of the global wetland resource. Not all recommendations are, however, relevant to all geographic situations or inventory programs,

- All countries lacking a national wetland inventory should undertake one, using an approach that is comparable with other wetland inventories and for which the Ramsar Convention should provide guidance (see below). These inventories are needed to underpin national planning, policy development and all efforts directed at wetland conservation and wise use promoted by the Ramsar Convention, and other related conventions. The inventories will assist in identifying wetlands of national and international importance, and through this to contribute to the Ramsar Convention achieving its vision for the List of Wetlands of International Importance (Ramsar COP7 Doc. 15.11 – Proposal No.11).
- Quantitative studies of wetland loss and degradation are urgently required for much of Asia, Africa, South America, the Pacific Islands and Australia.
- Further inventory should focus on a basic data set describing the location and size of each wetland, and its major biophysical features, including variations in area and the water regime. This information should be made available in both hardcopy and electronic formats.
- After acquisition of the basic data further information oriented to management, on wetland threats and uses, land tenure and management regimes, benefits and values, should be collected. Source(s) of information should be clearly recorded along with comments on its accuracy and availability.
- Each inventory should include a clear statement of its purpose and the range of information that has been collated or collected. This extends to defining the habitats covered and the date the information was obtained or updated.
- The Ramsar Convention should support the development and dissemination of models for improved globally applicable wetland inventory. These should be derived from existing models, for example the MedWet program, that are capable of using both remote sensing and ground techniques, as appropriate. Models should cover appropriate habitat classifications (eg those based on landform categories), information collation and storage, in particular Geographic Information Systems for spatial and temporal data that can be used for monitoring purposes.
- The Ramsar Convention should support development of a central repository for both hard-copy and electronic inventories. The meta-data that describe the inventories should be published on the World Wide Web for greater accessibility.

Further support is required for completion of the global review of wetland resources and priorities for wetland inventory; and to develop procedures for regular updating and publishing of inventory information on the World Wide Web. Regular updating (eg in conjunction with the triennial national reporting to the Ramsar Convention) may require restructuring the format and style of the current databases and bibliographic materials supplied by this project.