An ecological basis for water resource management

WD Williams
Editor
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Preface

There are many reasons why editors indulge in the masochism of putting books like this one together. Without elaborating on all those that induced me to undertake the task, let it be said that the main one was the obvious need for the book. Almost every Australian lake, river, stream and pond now bears the imprint of man, or will shortly do so. And, like it or not, the future of the Australian inland aquatic environment is for man to determine. It follows that the more that is known about how to manage this environment, the greater the chance that its present and future quality will be optimised. Moreover, as perceived over a century ago, wise environmental management is a most important measure of man's capacity to live less dangerously as a dominant ecological agent. This book attempts to aid management by drawing together some useful ecological knowledge into readily accessible form.

A number of colleagues and organisations have helped produce the book. They are thanked most sincerely. Foremost, of course, are those colleagues who contributed. It is a source of genuine pleasure for me to record that every colleague approached accepted the invitation to contribute, and all but two, and they with impeccable excuses, produced contributions. The pleasure is not in the least lessened by the slight amount of editorial agitation accompanying the final squeeze on some few authors; editors take pleasure in vicarious ways!

Also acknowledged is the Research and Publications Committee of the University of Adelaide who provided financial support to enable the inclusion of certain illustrations. The staff of the Australian National University Press, who have been most courteous and helpful, and various members of the ancillary staff of the Department of Zoology, University of Adelaide, especially Miss Ruth Altmann who prepared all drawings, and Mr Philip Kempster who was responsible for all photography are thanked too.

Separate acknowledgment and thanks is accorded Miss Sandra Lawson, Secretary to the Department of Zoology, for without her help the book simply would not be.

Finally, I acknowledge the benefits of my association in various ways with the engineering firm of Gutteridge, Haskins and Davey Pty Ltd; their attitude to environmental matters has been a source of both stimulation and enlightenment.

It remains to note that the opinions of authors expressed in this book are not necessarily those of the organisation to which they are attached.

W. D. W.

Adelaide, 1978
Contents

Preface v
About the authors xiv

1 Introduction
W. D. Williams 3

I The nature of the resource
2 Distinctive features of Australian water resources
W. D. Williams 6
3 Chemistry of Australian waters: the basic pattern with comments on some ecological implications
R. T. Buckney 12
4 The benthos of Australian Lakes
B. V. Timms 23

II Management problems
5 Limnological problems in the management of Tasmanian water resources
P. A. Tyler 43
6 Ecological considerations in the management of reservoir phytoplankton
G. G. Ganf 67
7 Algal problems in water supplies
I. C. Smalls 74
8 Aquatic weeds
D. S. Mitchell 81

III Water usage
9 Managing urban lakes
P. Cullen and R. S. Rosich 90
10 Catchment management
W. D. Williams 100

11 Australian water quality criteria
B. T. Hart 117
12 Public health aspects of water usage
J. R. L. Forsyth 128
13 Medically important diseases with aquatic vectors and hosts
T. Petr 137
14 Wastewater as a resource
A. G. Strom 149
15 Water as a waste transport and treatment mechanism: an ecological evaluation
W. D. Williams 155

IV The impact of man on inland waters
16 Conservation
P. S. Lake 163
17 Environmental impact of reservoir construction: the Dartmouth Dam invertebrate survey: a case history
J. D. Blyth 174
18 The downstream influence of Lake Hume on the River Murray  
K. F. Walker 182

19 Biological monitoring  
W. D. Williams 192

20 Environmental surveys  
B. T. Hart 205

V Aquatic fauna

21 Aquaculture  
N. M. Morrissy 215

22 Management of freshwater fish and fisheries  
D. A. Pollard, L. C. Llewellyn and R. D. J. Tilzey 227

23 Introduced fish  
R. D. J. Tilzey 271

24 Introduced amphibians: the cane toad  
M. J. Tyler 280

25 Waterfowl resources and their management  
L. W. Braithwaite 287

26 Ostracoda and water resources: diagnosis and prognosis  
K. G. McKenzie 295

VI Man-made lakes

27 Lake Hume  
R. L. Croome 305

28 North Pine Dam  
C. R. King and R. G. Everson 311

29 Lake Burley Griffin  
T. J. Hillman 317

30 Some limnological features of the Sydney water supply system  
L. D. Bowen and I. C. Smalls 324

31 Limnological features of some Victorian reservoirs  
I. J. Powling 332

VII Some special aquatic environments

32 Farm dams  
B. V. Timms 345

33 Waste stabilisation ponds  
B. D. Mitchell 360

34 Billabongs of the Murray-Darling system  
R. J. Shiel 376

35 Estuaries and coastal lakes  
I. A. E. Bayly 391

Index 399
# Plates

<table>
<thead>
<tr>
<th>Plate</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>I</td>
<td>Manganese deposits inside a hydroelectric pipeline</td>
</tr>
<tr>
<td>II</td>
<td>An experimental recirculatory system</td>
</tr>
<tr>
<td>III</td>
<td>Detail of a recirculatory apparatus</td>
</tr>
<tr>
<td>IV</td>
<td>A larger-scale experimental system</td>
</tr>
<tr>
<td>V</td>
<td>The anastomosing network of cells and hyphae of the manganese-oxidising budding bacterium <em>Pedomicrobium</em></td>
</tr>
<tr>
<td>VI</td>
<td>Applied limnology in action</td>
</tr>
<tr>
<td>VII</td>
<td>The densely forested Mersey Valley, Tasmania</td>
</tr>
<tr>
<td>VIII</td>
<td>Lagoon of Islands, Tasmania, in 1963</td>
</tr>
<tr>
<td>IX</td>
<td>Lagoon of Islands, Tasmania, in 1963</td>
</tr>
<tr>
<td>XI</td>
<td><em>Potamogeton tricarinatus</em> impeding flow in an irrigation supply channel</td>
</tr>
<tr>
<td>XII</td>
<td><em>Salvinia molesta</em> covering the surface of a sheltered reservoir</td>
</tr>
<tr>
<td>XIV</td>
<td><em>Bufo marinus</em></td>
</tr>
<tr>
<td>XV</td>
<td>Dartmouth Dam site before impoundment</td>
</tr>
<tr>
<td>XVI</td>
<td>Aerial photograph of a 30 km section of the Goulburn River</td>
</tr>
</tbody>
</table>
## Tables

<table>
<thead>
<tr>
<th>No.</th>
<th>Title</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Species diversity and mean biomass in some Australian lakes and impoundments</td>
<td>24</td>
</tr>
<tr>
<td>2</td>
<td>Benthos of four saline maars in western Victoria</td>
<td>27</td>
</tr>
<tr>
<td>3</td>
<td>Benthic biomass and organic matter in sediments</td>
<td>34</td>
</tr>
<tr>
<td>4</td>
<td>Percentage contribution to total biomass by species</td>
<td>35</td>
</tr>
<tr>
<td>5</td>
<td>Some commonly recorded algae in reservoirs in south-eastern Australia</td>
<td>74</td>
</tr>
<tr>
<td>6</td>
<td>Effluent disposal from Australian sewage treatment plants</td>
<td>108</td>
</tr>
<tr>
<td>7</td>
<td>Some examples of water-borne infective agents causing infection by contact</td>
<td>128</td>
</tr>
<tr>
<td>8</td>
<td>Some examples of water-borne infective agents causing infection when swallowed</td>
<td>129</td>
</tr>
<tr>
<td>9</td>
<td>Medically significant arboviruses and their vectors in Australia</td>
<td>139</td>
</tr>
<tr>
<td>10</td>
<td>Potential resources in wastewater from a population of 100,000</td>
<td>149</td>
</tr>
<tr>
<td>11</td>
<td>Number of examples of effluent reuse in Australia and geographical pattern</td>
<td>150</td>
</tr>
<tr>
<td>12</td>
<td>Abundance of principal macroinvertebrate fauna of Mitta Mitta River before and after construction of Dartmouth Dam</td>
<td>178</td>
</tr>
<tr>
<td>13</td>
<td>Number of species in Mitta Mitta River before and after construction of Dartmouth Dam</td>
<td>179</td>
</tr>
<tr>
<td>14</td>
<td>Information and environmental indicators needed to study algal problems</td>
<td>209</td>
</tr>
<tr>
<td>15</td>
<td>Commercial catches of Australian freshwater fishes</td>
<td>228</td>
</tr>
<tr>
<td>16</td>
<td>Freshwater fish of coastal rivers and streams</td>
<td>231</td>
</tr>
<tr>
<td>17</td>
<td>Freshwater fishes of the Murray-Darling drainage system</td>
<td>240</td>
</tr>
<tr>
<td>18</td>
<td>Freshwater fishes of the Lake Eyre Internal and Central Western Desert drainage systems</td>
<td>258</td>
</tr>
<tr>
<td>19</td>
<td>Endemism and distribution of Australasian continental ostracode genera</td>
<td>296</td>
</tr>
<tr>
<td>20</td>
<td>Salinity ranges of representative saline and freshwater ostracode genera</td>
<td>297</td>
</tr>
<tr>
<td>21</td>
<td>Some morphometric and hydrologic data for Lake Hume</td>
<td>306</td>
</tr>
<tr>
<td>22</td>
<td>Morphometric data for North Pine Dam</td>
<td>312</td>
</tr>
<tr>
<td>23</td>
<td>Typical major ionic composition in North Pine Dam</td>
<td>312</td>
</tr>
<tr>
<td>24</td>
<td>Morphometry of Lake Burley Griffin</td>
<td>318</td>
</tr>
<tr>
<td>25</td>
<td>Water budget of Lake Burley Griffin</td>
<td>319</td>
</tr>
<tr>
<td>26</td>
<td>Analysis of zinc pollution sources</td>
<td>320</td>
</tr>
<tr>
<td>27</td>
<td>Nutrient budgets for Lake Burley Griffin</td>
<td>321</td>
</tr>
<tr>
<td>28</td>
<td>Some salient morphometric infor-</td>
<td></td>
</tr>
</tbody>
</table>
Tables

29 Zooplankton Crustacea in some water supply reservoirs for Sydney 325
30 Major chemical features of older reservoirs 327
31 Annual mean PO₄-P levels in Prospect Reservoir and Lake Burragorang 327
32 Summary data for some important features of Prospect Reservoir 328
33 Phytoplankton in Prospect Reservoir 329
34 Some nutrient and phytoplankton population data for Wingecarribee and Fitzroy Falls Reservoir during initial filling phases 330
35 Algal counts in Wingecarribee and Fitzroy Falls Reservoirs during initial filling phases 331
36 Major morphometric and other physical features of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam 334
37 Major chemical characteristics of Lake Eildon, Lake Eppalock, Tarago Reservoir, Mitta Mitta River and Dartmouth Dam 335
38 Zooplankton of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam 336
39 Secchi disc readings in some farm dams and reservoirs 348
40 Some physico-chemical features of six dams at Copping, Tasmania 348
41 Surface temperature ranges in farm dams in southern Australia 349
42 Spatial variation in pH in St Lucia Pond, Queensland 352
43 Invertebrates in dams of different ages at Boscabel, W.A. 354
44 Waste stabilisation pond use throughout Australia 361
45 Species of microcrustacea occurring in waste stabilisation ponds 367
46 Macrophytes associated with billabongs at Wodonga and Alexandra 382
47 Rotifera and microcrustacea recorded from billabongs of the Murray and Goulburn Rivers 384
48 Macroinvertebrates frequently collected from the reedbeds of billabongs 386
49 Fish species recorded from Albury-Wodonga billabongs 387
50 Waterbirds recorded from Hawksview Wildlife Refuge, Albury 388
Figures

1 Schematic representation of the effects of climate, geology and sea salt aerosols on the ionic dominance of the world's inland waters 13
2 The main pattern of pH distribution over the salinity range recorded for Australia's sodium/chloride-dominated waters 15
3 Numbers of benthic species according to depth in three south-eastern maar lakes 25
4 Numerical abundance and dry-weight biomass of *Tanytarsus barbitarsis* in Lake Werowrap 26
5 Depth/biomass profiles for seven lakes in eastern Australia 36
6 Map showing location of major hydroelectric storages and power developments 44
7 Hydroelectric developments on Tasmania's Central Plateau 46
8 Diagram showing occurrence of troublesome manganese deposits in hydroelectric pipelines in Tasmania 46
9 Temperature and dissolved oxygen profiles of Lake Barrington 53
10 Diagram showing features of V-shaped reservoir 55
11 Depth profiles of temperature and dissolved oxygen for Lake Gordon 56
12 Variation with depth of various chemical parameters in Lake Barrington 57
13 Down among the sapropel 63
14 Diagram for water weed management 85
15 Overall movement of phosphorus in lakes 92
16 Interchange of phosphorus between various compartments in the sediment and water of a lake 93
17 Relationship between water quality criteria and standards 118
18 Main factors to consider when formulating water quality criteria for heavy metals 119
19 Uptake of mercury by previously unexposed brook trout 121
20 Pictorial comparison of natural and unnatural nutrient pathways 157
21 Study area 175
22 The study area showing stations sampled 182
23 Stream flows at various sampling stations 185
24 Temperatures at various combinations of sampling stations 186
25 Oxygen saturation data for sampling stations 188
26 Approach to the design of a water quality study program 206
27 Management scheme for a water quality program 210
Figures

28 Main Australian drainage systems 230
29 Relationships of annual catches of three important commercial fishes in the River Murray with river height 242
30 Relationships of annual catches of three important commercial fishes in the Murrumbidgee River with river height 243
31 Distribution of European carp in Australia 254
32 Geographic range of Bufo marinus in Australia 281
33 Distribution of waterfowl and relative abundance of species according to wet or dry conditions 288
34 Wetland habitats of waterfowl 289
35 Numbers of stock dams and reservoirs 290
36 The relationship between reproductive condition of non-moulting adult grey teal and mean hunter success 293
37 Temperature, dissolved oxygen and turbidity profiles for Lake Hume 307
38 Areal day rates of phytoplankton productivity 308
39 Lake Burley Griffin 317
40 Schematic representation of water supply system for Sydney 326
41 Dissolved oxygen and temperature profiles in Prospect Reservoir 330
42 Location of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam 332
43 Thermal patterns in Lake Eildon 337
44 Farm dam types 346
45 Density of all types of farm dam and number of large farm dams in N.S.W. 347
46 Seasonal changes in a small reservoir 349
47 Temperature stratification in a farm dam 350
48 Seasonal abundance and species periodicity in a small reservoir 353
49 Stages in billabong formation 378
50 Some physio-chemical features of a billabong near Alexandra 380
51 Seasonal changes in the composition and abundance of microcrustacea in a billabong of the Goulburn River 385
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PART I

The nature of the resource
The major problems of water resource management that faced modern Australia related largely to water quantity. The paramount questions were: how much, when, and where? Many problems of that ilk remain. There were certainly also problems of quality, as Powell (1976) has so well documented, but in general they were not so pressing. In the more recent past, that situation has altered; water quality—in Australia as elsewhere (Lvovitch, 1977)—has now become a most important matter for consideration in water resource management.

Both physico-chemical and biological events are involved in determining water quality, but it is the latter that have assumed greater prominence. Freshwater pollution, eutrophication, the introduction of noxious plants and animals (e.g. the water fern *Salvinia*, carp, cane toad), and water-borne diseases spring readily to mind as examples. Many more exist. It is now clear that water resource management must take serious account of freshwater ecology since much of it is of significant economic and environmental importance.

Many biological problems have arisen or assumed more significance in only the past two or three decades. In part, this is because it is within this time span that many barriers which formerly served to isolate the Australian continent—and thus protect it—have become less effective. In part, it is because of long-continued demographic trends and the increasing divergence of the Australian community from sensible ecological ethics. Whatever the cause, biological problems there are, and Australia is peculiarly ill-equipped to deal with them. Such programs of research as exist are largely fragmentary, there is no strong tradition of freshwater biological studies in higher educational institutions (though its makings are present), and Australia lacks any cohesive national organisation specifically charged to undertake water resource investigations in the broad sense (the Australian Water Resources Council and the Commonwealth Scientific and Industrial Research Organisation do not fill that niche).

It is mainly in response to considerations of this sort that the present book has been brought together. The intention is to focus attention on some of the more important biological events involved in water resource management in Australia, to provide authoritative summary accounts of significant topics in the area (together with useful bibliographies), and to stimulate the future work felt necessary in most cases for satisfactory resolution of problems.

It has not been possible to cover all areas of interest: in a multi-authored book on so broad a topic that is in the nature of things. Nevertheless, a conscious effort has been made to provide accounts covering all the most pressing problems and important areas of
interest. To that end, the book covers seven ‘subject’ areas. The first includes accounts about the nature of the resource. By no means do these provide a comprehensive coverage, but they do emphasise the distinctive nature of Australian inland waters and deal with two most important compartments of the aquatic environment; the chemistry of waters and the composition of benthic communities in large standing water bodies. The second subject area is basically concerned with management problems. Within that framework, a variety of topics is discussed ranging from the management of bacterial populations in pipelines to the management of catchments. A third section considers water usage with particular reference to water quality criteria, water supplies and wastewater. Two non-congruent chapters in this section deal with public health problems associated with water usage and the aquatic environment. A fourth section deals broadly with the impact of man on inland waters, and the way this impact has been or can be assessed. Some useful or undesirable components of the aquatic fauna receive consideration in a fifth section. The last two sections provide descriptive accounts of various inland waters. The sixth consists of a series of case studies of man-made lakes scattered throughout south-eastern Australia; in selecting these, an effort was made to provide a variety of reservoir type and purpose. The seventh and final section considers four special aquatic environments that have received rather less attention from biologists than would seem merited. Thus, there is a chapter on farm dams, for these, despite their number and ubiquity in Australia, are little known biologically. There is a chapter on waste stabilisation ponds (sewage lagoons), an even more neglected set of aquatic environments. And there are chapters on billabongs of the Murray-Darling river system, and estuaries and coastal lakes.

A prominent feature of almost all chapters is an emphasis upon the need for further research. The basic message is that we still stand a considerable distance from any thoroughgoing understanding of the biology of Australian inland waters, and therefore are not yet in any position to be able to manage inland waters to maximum advantage. This situation is likely to prevail for some time (though it is to be hoped that this book will somewhat shorten the period) largely because of inadequate funding of appropriate research— which is the same as stating that there is an inadequate amount of research being pursued. To be sure, a good deal of money is spent annually on what may be broadly termed ‘water research’, but relatively little of it supports the sort of basic ecological work necessary for the proper management of Australia’s water resources. The recent Birch report (Birch, Looker and Madigan, 1977) put the case for research well:

We have a unique continent of which we are at the same time custodians and exploiters. We have to find out about it, even if the objective is only to make use of it to our advantage. We have problems which depend closely on our geology, our weather, our geography and even on our social structure. These are all unique: world science and technology give guidelines but cannot solve our particular problems or delineate our particular opportunities. Moreover the situations and problems change contin-
uously, and what may be an Australian advantage disappears, but new advantages arise... Research is needed to make best use in the long-term of our assets, and in the short-term to overcome as many as possible of our difficulties. Without such work our capabilities will stagnate and our social structure and our place in the world will be prejudiced. As Alice said in *Through the Looking Glass*: ... 'it takes all the running you can do to keep in the same place. If you want to get somewhere else, you must run at least twice as fast as that!'

References


2 Distinctive features of Australian water resources

W. D. Williams

Introduction
Australian inland waters are characterised by many distinctive features. Any book purporting to deal with the biological basis of Australia's water resources needs—at the very least—to outline these features, and the aim of the present chapter is to do that. The account is brief because the objective is simply to highlight the features, not consider them in detail; this procedure is regarded as adequate bearing in mind overall objectives. Readers dissatisfied with the brevity of the discussion below are referred for further details to references cited and to Williams and Wan (1972). An expanded version of this discussion is given in Williams (in press a).

An important prefatory and general point should be made. It is that, because of its distinctiveness, caution is required in any extrapolation to the Australian aquatic environment of limnological or ecological generalisations based only upon investigations conducted outside Australia, or worse, drawn only from north temperate investigations. There is no question, of course, but that overall environmental principles are the same worldwide; it is their detailed regional operation that may differ in Australia.

As for the reasons behind the distinctiveness, it is noted only that the special character of Australia's geological history, topography, climate (particularly its aridity), and isolation are major factors implicated.

Distinctive physico-chemical features
Chapter 3 considers many of the chemical characteristics of Australian inland waters, and little elaboration is needed here apart from noting distinctive features. Ionically, Australian inland waters diverge widely in composition from the so-called 'standard composition' of fresh water (Rodhe, 1951); instead of a dominance of divalent cations and bicarbonate, Australian inland waters frequently display a predominance of sodium and chloride ions, particularly those of elevated salinity. Additionally, salinity is frequently higher than is found elsewhere in comparable climates and topography. In Victoria, for example, the best watered mainland state, most natural standing bodies of water have over 3000 mg l⁻¹ of total dissolved salts, and even many rivers—including the Murray—have elevated salinities. Although high salinity is a natural phenomenon, it has been exacerbated as a problem by human impact upon the environment. High phosphate and nitrate values also appear to be a feature of many Australian inland water bodies. The values often exceed the 'critical' value which, it is frequently claimed, will cause problems of eutrophication if exceeded, though such problems are not as evident as might be expected (cf. Wood, 1975).

Physically, one important feature distinguishing large standing bodies of water in Australia is their pattern of
thermal stratification. By far the most common pattern is the warm monomictic one; complete mixing takes place once a year, in winter. Polymictic lakes, those which never stratify or which have no persistent periods of stratification, are also common. Notably, there are no examples known from Australia of lakes which are dimictic, that is have two periods—one in spring and one in autumn—when mixing takes place. Such lakes are typical for most northern temperate regions, and it is from such lakes that a good deal of limnological knowledge has been derived. Additionally, Australia has a few lakes which freeze over in winter, but circulate in summer (warm thermomictic lakes sensu Bayly and Williams, 1973); this thermal pattern is unusual.

Another distinct physical feature relates to river discharge values. These are highly variable both seasonally and annually, and far more variable than is generally the case for rivers with similar average discharges. As a direct result of this it has been necessary to build relatively more impoundments on Australian rivers than might be expected based on average rainfall data. In any event, the point should be stressed that as a whole the total annual average discharge for Australian rivers is exceedingly small on an areal basis. High turbidity is another distinguishing physical feature of many Australian inland waters.

**Distinctive biological features**

A great number of features distinguishes the biota of Australian inland waters, though the extent of distinctiveness varies from group to group.

Those invertebrates that are minute, easily dispersed, and which can resist desiccation at some time during their life exhibit the least number of distinctive features and are often widespread if not cosmopolitan forms. All species of the Protozoa, many species and most genera of nematodes, most species and all genera of the Rotifera and Polyzoa, and probably most genera if not species of the Gastrotricha appear to be of this sort. Most coelenterates and nemer­teans, too, are cosmopolitan or widespread. However, the other invertebrate groups show increasing degrees of distinctiveness, from those in which species and perhaps some genera are endemic, to those in which major taxonomic categories are confined to Australia (Williams, 1968). As a rule, most species though not genera of Australian sponges, turbellarians and nematomorphs appear to be endemic, and most species and genera of Australian temnocephalids are endemic. The gastropod molluscs have many endemic species and genera, and there are also a few families which are either endemic to or particularly well-developed in Australia. The bivalve molluscs, too, have many distinctive features; most (probably all) Australian freshwater mussels belong to the Hyriidae which is found only in Australasia and South America (the dominant northern hemisphere families, the Margaritiferidae and Unionidae, are absent). The other truly freshwater bivalve families, the Corbiculidae and Sphaeriidae, are not so distinctive. Within the Annelida, special note of the polychaete worm *Striatodrilus* (Histogramellidae) should be made. In Australia this has two endemic species, both ectoparasites of freshwater decapods, and elsewhere is known only from fresh waters in South America and Madagascar. Native Australian oligochaetes are distinguished at the family level by paucity, but endemlicity is high.
at the generic level, and is almost complete at the species level. About half the genera and most species of Australian freshwater leeches (Hirudinea) are endemic.

Of the Crustacea, the Malacostraca is of greatest interest because relatively poor means of dispersal and the lack of resistant stages in the life-cycle are the rule (Williams, in press b). Within the Syncarida, the relictual and very primitive Anaspidacea occur only in Australia. In the Isopoda, the aberrant suborder Phreatoicidea reaches its greatest diversity and abundance in Australia (it occurs elsewhere only in New Zealand, South Africa and India). Notably, the dominant freshwater isopods of the northern hemisphere, the Asellidae, are absent from Australia; such Asellota as do occur (Janiridae) are closely related to marine forms. There is also an aquatic oniscoid isopod apparently uniquely inhabiting salt lakes. It is probable that all species and most if not all genera of freshwater amphipods will prove endemic. Finally, all Australian freshwater crayfish belong to the Parastacidae, a family restricted to southern hemisphere land-masses; representatives of the northern Astacidae are absent.

Entomostracan crustaceans, though frequently possessing resistant stages and more easily dispersed than the Malacostraca, also do not lack distinctive features. The calanoid copepods, for example, are highly characteristic. Of the dominant genera, *Boeckella* and *Calamoecia*, the first is basically a southern circumpolar freshwater group, and the second restricted to Australasian inland waters. Both genera are in the Centropagidae, a family of no great importance in the northern hemisphere where the quite unrelated Diaptomidae can be regarded as its ‘replacement’. In Australia, diaptomids occur only in the north (two species only). Recent investigations have shown that the cyclopoid copepods, contrary to expectations, also exhibit many distinctive features in Australia (Morton, 1978).

Even entomostracan groups classically regarded as cosmopolitan display endemism. The worldwide notostracan *Triops* is specifically distinct in Australia. And *Artemia salina*, the brine shrimp (Anostraca), often regarded as the cosmopolite par excellence, is rather doubtfully regarded as indigenous to Australia; the unrelated *Parartemia* is invariably found in non-coastal Australian salt lakes.

Because most insect orders can fly, it could be assumed that aquatic insects display fewer distinctive features than crustaceans. Although this is true for some insect groups, there are others, nonetheless, with well-marked regional features (CSIRO, 1970, 1974).

Almost all species and genera of Plecoptera (stoneflies) are endemic, and about one-quarter of the species and a third of the genera belong to the Eustheniidae, one of the most primitive extant stonefly families. Eustheniids are known only from Australia, New Zealand and South America. The other Australian families, Austroperlidae, Grippopterygidae and Notonemuoridae, also display many regionally distinctive features. Within the Ephemeroptera (mayflies), all families present are cosmopolitan, but one of them, the Leptophlebiidae, has undergone a pronounced adaptive radiation so that leptophlebid species occupy in Australia many niches characteristically occupied elsewhere by species from other fami-
lies. The rich Australian dragonfly fauna includes several very primitive members. *Hemiphlebia mirabilis* (in the endemic Hemiphlebiidae), is amongst the most primitive of living odonatans. The Mecoptera contains the family Nannochoeristidae, the only known aquatic section of this insect order, and found only in Australasia and South America. There are many features of the Trichoptera, also, which distinguish this order in Australia: the absence of the Phryganeidae, common in the northern hemisphere, the dominance and diversity of the Leptoceridae, a well-marked species endemcity throughout the order, and the occurrence, even, of a few endemic families. For the remaining aquatic insect orders, it is noted only that almost all include many endemics.

The vertebrate fauna of Australian inland waters is no less distinctive. Thus, the fish fauna is remarkably depauperate with only some 130 species restricted to fresh waters throughout their life (Lake, 1971). Most are endemic. Another 70 species are found which either require estuarine or marine environments to breed (20), or are marine itinerants (50). Species diversity is greatest amongst the Galaxiidae, a family confined to southern circumpolar regions. Another important feature is that some major fish groups characteristic of fresh waters outside Australia do not occur naturally here, e.g. the carp family (Cyprinidae), freshwater salmonids, and true percids. Further, with three exceptions, the freshwater fish fauna has a closer evolutionary relationship with marine forms than occurs in other continents. The exceptions are the archaic Queensland lungfish, *Neoceratodus*, and two species of barramundi, *Scleropages jardini* and *S. leichhardtii*.

The dominant frog families are the Hylidae and Leptodactylidae with over 120 species between them; there are six more species in two other families (Tyler, 1976). Conspicuously sparse are species of Ranidae (1 species), the dominant frog family of the northern hemisphere. *Bufo marinus* remains the only introduced frog (Chapter 24), and there are no Amphibia in Australia other than the Anura (salamanders and caecilians are absent).

Almost all Australian tortoises belong to the suborder Pleurodira, and there are four main genera, *Chelodina*, *Emydura*, *Elseya* and *Pseudemydura*. Tortoises are found throughout Australia wherever suitable aquatic environments occur. Apart from Australia, the Pleurodira survive only in New Guinea and South America. Of the two Australian species of crocodile, one, *Crocodylus johnstoni*, is endemic to northern Australia; the other is widespread in south-east Asia. Many other Australian aquatic reptiles (lizards, snakes) are endemic (Cogger, 1975).

Finally, for the vertebrates, mention of the avifauna and the few aquatic mammals is required. A notable feature of the avifauna is the absence of the Gaviformes (loons) and Phoenicopteridae (flamingos), both of which orders are widespread beyond Australia. The lack of a precisely timed, and directionally predictable migration in the avifauna is also notable (Frith, 1967). The uniqueness of the duck-billed platypus, *Ornithorhynchus anatinus*, requires no emphasis; it is the only aquatic monotreme mammal known. *Hydromys* and *Xeromyx*, both placental water-rats, constitute the remaining fraction of the aquatic mammalian fauna; both are
members of the highly interesting Muri-
dae.

The flora of Australian inland waters is
not as distinctive as the fauna. There
are, nevertheless, several features of
interest. Thus, since algal species are
usually cosmopolitan and easily trans­
ported and dispersed, there is little
which distinguishes them. However, the
presence of a few species is notable, e.g.
Micrasterias hardyi, an endemic desmid.
The macrophytes are more distinctive,
and some 40 per cent of the 200-odd
Australian species is endemic (this does
not take into account most species of
aquatic sedge and grass) (Sculthorpe,

To summarise and conclude, it is
noted that the fauna is distinguished by
its high degree of endemicity, the occur­
rence of several groups not found or at
least not common outside Australia, and
by adaptive radiation within Australia of
several groups which elsewhere are not
as diverse. The aquatic flora, though
perhaps less distinctive in this way, also
displays considerable endemicity. Addi­
tionally, the lack of a well-defined period
of leaf-fall in riparian (bank-side) veget­
ation requires noting. It is known that
such leaf fall contributes a substantial
input to the energy budget of most small
streams (Hynes, 1970). In the northern
hemisphere, riparian leaf fall has a
well-defined periodicity and mainly oc­
curs in autumn; in Australia, on the other
hand, there is no analogous autumnal
shedding of leaves. Some repercussions
of this appear to be that the life-cycles of
stream faunas are less precisely timed,
and the fauna is also less diverse than is
the case in northern hemisphere streams.
However, the full implications of the
difference have not yet been rigorously
explored.

As a final comment in this chapter, it
is noted that the biota of Australian
inland waters is subject to invasion
pressures from introduced species to a
degree probably unparallelled on other
continents. The pressures principally
involve introduced fish and macrophy­
tes; Chapters 23 and 8 have further
details. This phenomenon can be con­
sidered as yet one more distinctive
feature, particularly in view of the
apparent ease with which many exotics
become established and multiply.

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3 Chemistry of Australian waters: the basic pattern, with comments on some ecological implications

R. T. Buckney

Introduction
The aridity of the Australian continent expresses itself in innumerable ways, and the broad chemical pattern of Australian inland waters is one of them. This fact presents special problems of study and of management. Some facets can be discussed in depth; inevitably others require further research. However, some data provide interesting bases for speculation on the nutritional ecology of aquatic species. It is appropriate here to discuss the following main areas; major ions ($Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-}, HCO_3^-$, and $CO_3^{2-}$); nitrogen, phosphorus and other nutrients; chemical variability; ecological considerations.

The major ions
To a very large extent, the review by Williams (1967), which asserts the predominance of sodium and chloride, remains an accurate picture of the major ion chemistry of Australian waters. Comments on a few features of more recent studies are appropriate, nevertheless.

Firstly, the importance of calcium (and to a lesser extent, magnesium) and bicarbonate seems to be greater in some areas than early data suggested. Some examples are to be found among lentic sites in central and near-coastal New South Wales (Timms, 1970a), various waters in the Central Plateau and Midlands of Tasmania (Buckney and Tyler, 1973a), the plains reaches of the rivers in the Murray-Darling system (Williams et al., 1970; Croome et al., 1976), and in some of the waters draining the Kimberley Ranges of Western Australia (Williams and Buckney, 1976b). No doubt similar waters occur in other areas. Despite these examples, it is probably still true to say that sodium-chloride dominance is most usual in Australia.

Secondly, the only glacial lakes on the Australian mainland, those on the Kosciusko plateau, are easily the freshest in the country, with salinities of less than 5 mg l$^{-1}$ (Williams et al., 1970). For each of these, total concentration and relative ionic proportions (stoichiometry) are comparable to the local rainwater.

Thirdly, like the mainland, Tasmania contains salt lakes in its driest region, the Midlands. These lakes are chemically similar to most of those on the mainland (Buckney and Tyler, 1976). Some appear to be permanent, and at least one or two are highly saline, i.e. near-saturated.

Finally, the chemistry of Australian rivers remains relatively unknown. As noted, they appear to be enriched in bicarbonate relative to lentic waters (Buckney, 1977, 1978; Croome et al., 1976).

Perhaps the most useful scheme in which to place the major ion chemistry of waters is that of Gibbs (1970). This recognises sea salt aerosols and the catchment rock as the main sources of ions, and evapotranspiration as the main control on total concentration; it pro-
poses that the majority of the world’s waters fall within a distribution like that shown in Fig. 1. Few attempts have been made to relate geological and climatic factors to the chemistry of Australian waters, but the surveys by Buckney and Tyler (1973a, 1976) applied Gibbs’s scheme to Tasmanian waters. The review by Williams (1967), and much later work, can be interpreted as pointing to the fact that most of Australia’s waters occupy the bottom third of the shaded area in Fig. 1; the majority of waters would appear to receive very little geochemical modification, and this fact raises the question of the origin of dissolved material in them.

A number of early papers (e.g. Anderson, 1945) stressed the importance of sea salt aerosols (cyclic salts) in determining the chemistry of Australian inland waters. These observations have formed the basis for a continuing theme in Australian limnology. Bayly and Williams (1973) summarised much of the evidence for and against the proposition, which largely rests on the observation that a wide range of waters in geographically diverse areas have a similar stoichiometry—one like seawater; the main arguments against it (e.g. Currey, 1970) propose that the salt is of geological origin or relict sea salt. In any event, there would appear to be a remarkable consistency in the ratio of sodium concentration to chloride concentration, as shown by Buckney and Tyler (1973a) and by Bayly (1964); this relationship was found to hold even for Ca-HCO₃ waters in Tasmania, pointing to the similarity of sources of sodium and chloride for all of those waters.

It should be pointed out that this theory on the atmospheric origin of salts is not confirmed until it can be shown

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**Fig. 1** Schematic representation of the effects of climate, geology and sea salt aerosols on the ionic dominance of the world's inland waters. (After Gibbs, 1970: modified according to Tyler, 1972.)
that atmospheric input to a catchment (measured in tonne year\(^{-1}\)) is adequate to account for the gross annual load in the streams draining that catchment. A recent study (Buckney, 1979) indicated that atmospheric supply of chloride, phosphorus and nitrogen was probably enough to account for loads in the Onkaparinga River, South Australia. The same may not be true for other ions or catchments. In arid areas where evaporative concentration of waters is significant, the marked insolubility of calcium and magnesium carbonates could well be enough to explain the depression in concentration of alkaline earth ions and carbonate-bicarbonate in saline waters (some solid material which had settled from a salt lake sample over several months was recently examined by this writer and found to contain a great deal of carbonate, suggesting that CaCO\(_3\) precipitation is at least partly responsible for the stoichiometry, and is probably supplied in larger quantities than the water analysis suggests). Timms (1970b) has demonstrated a relative enrichment of sodium and chloride during evaporative concentration in some lakes. The definitive experiments—the slow evaporative concentration of calcium and bicarbonate-rich waters with appropriate monitoring of stoichiometry—have yet to be performed.

Whether the consistency of stoichiometry has its causes in the air, the earth or the sun’s fires (or, as is more likely, all three), the use of the world average freshwater (Rodhe, 1949; Livingstone, 1963) as a reference composition has little to recommend it in Australia. Seawater would be a more appropriate reference, and has been used as such in a numerical description of Tasmania’s waters (Buckney, 1976a).

Since waters like seawater in stoichiometry are so common in Australia, it is instructive to examine some of the trends their analyses display. In what follows, consideration is given to waters for which sodium constitutes 70 per cent or more of the total cation concentration (in equivalents \(l^{-1}\)) and chloride at least 80 per cent of the anion concentration. The freshest waters of this type have been identified by Bayly (1964) as characteristically confined to sandy exposed coastal lowlands in areas around much of the country. These are acidic waters which are coloured brown (up to 400 Pt units of colour) by allochthonous organic material. Chemically similar waters occur in siliceous areas of south-west Tasmania (Buckney and Tyler, 1973a, 1976); some small perched lakes in old dunes near Lake Eliza (south-eastern South Australia) appear to be of the same sort. These waters are restricted to siliceous heathland, sedgeland or to bogs; indeed they appear similar to the bog waters of the northern hemisphere (Tolpa and Gorham, 1961). Usually they have relatively high concentrations of iron and manganese, and low dissolved silicon concentrations.

The most saline waters of this sodium-chloride type are confined to areas in which annual evaporation exceeds precipitation (Langbein, 1961); they are closed salt lakes and ephemeral saline pans and are usually found inland and to the west of the Eastern Highlands, although they occur also in several small coastal areas where the climate is suitable, such as the Tasmanian north-east and Bass Strait Islands (Buckney and Tyler, 1976). Some are, indeed, marine hypersaline waters (Bayly, 1967). These saline waters are quite often very turbid and usually
alkaline. The water may be coloured yellow, but never to the extent encountered in the acidic waters; the dissolved organic material is probably autochthonous.

These two types of waters have often been recognised and considered as separate entities. In fact, they form the two ends of a continuum, which is most easily illustrated by the relationship between pH and salinity for these Na-Cl waters (data for colour, silicon and iron concentrations and so on, are somewhat scarce for Australian waters). In Fig. 2, results for 350 analyses have been summarised (Sources: Bayly, 1964; Buckney, 1976a; Buckney and Tyler, 1973a; Timms, 1970a, 1972, 1973; Williams, 1967; Williams and Buckney, 1976b; Williams et al., 1970). The variation of pH with salinity is shown by the data envelope—all but the three waters shown fall within the enclosed area. This figure is a more extensive treatment of that presented by Buckney (1976a).

Clearly, a salinity of about 10,000 ppm (10°/oo) is chemically significant in these waters, representing as it does the salinity at which the highest pH can be attained. While it is tempting to consider this salinity as some sort of demarcation point, other chemical considerations suggest otherwise. Some few data for colour, iron and silicon concentrations (mainly for Tasmanian waters) show complementary changes in the range 1°/oo-10°/oo, which is the salinity range chosen by Buckney and Tyler (1976) on stoichiometric grounds to delineate ‘fresh’ from ‘saline’ waters in Tasmania. It is also the range of salinities over which a variety of ‘freshwater’ Crustacea reach the upper limits of their salinity tolerances. Bayly (1967) provides ample evidence that this salinity range is one of ecological and physiological transition. The salinity of 3°/oo was chosen by Williams (1964) to distinguish ‘fresh’ from ‘saline’ waters; it was chosen on the basis of that author’s ability to taste salt in solution and has been used widely by other workers. Since the use of this arbitrary value could distract attention from the important physico-chemical
and biological transitions which occur in the range 1°/00 to 10°/00, the practice could well be discouraged, and the terms 'fresh' and 'saline' used in a much broader sense. There is no substitute for the precise (and relatively simple) procedure of quoting the salinity value.

For the broad chemical features of dilute waters not dominated by sodium and chloride, the reader is referred to the paper by Buckney (1976a).

**Nitrogen, phosphorus and other nutrients**

Rather little is known about the concentrations of and movement of nitrogen and phosphorus in the waters of Australia, this despite the escalation of interest in these elements as limiting factors for primary production and as the usual causes of man-made eutrophication. Even less is known of the occurrence of trace metals.

Early observations on nutrient levels suggested that nitrogen and phosphorus concentrations in Australian waters were somewhat higher than would have been expected from the plant growth they supported (Williams and Wan, 1972). This raised the question as to whether nitrogen and phosphorus were limiting plant growth or whether some other factor (light, micronutrients, etc.) was responsible. Little has been done to resolve this question. Wood (1975) has gathered together many of the results so far obtained for Australia and has summarised work being conducted. Croome et al. (1976) reported on phosphorus and nitrogen concentrations in Lake Hume and Lake Mulwala on the Murray River, and Buckney (1979) has reported results for the Onkaparinga River, S.A. To date, one study on sediment phosphorus in an Australian lake (Lake Mulwala) has been published (Hart et al., 1976).

Published trace metal data seem to be restricted to the Murray River (Hart, 1976, 1977), to a description of cadmium concentrations in N.S.W. waters (Doolan and Smythe, 1973), and to one of iron and manganese in Tasmanian waters (Buckney and Tyler, 1973a, b). In addition to these, studies on contamination of rivers by mine effluents have appeared (Tyler and Buckney, 1973; Weatherley et al., 1967).

With so few data available, it is not possible to present even a basic survey of Australian waters in these respects. However, it may be useful to record some expectations and relevant thoughts. Primary industries in many areas of Australia have relied on trace element additions for success (Evans, 1977), and it would be useful to know whether such micronutrient deficiencies have restricted the growth of aquatic primary producers. Deficiencies of that sort could explain the observation of Williams and Wan (1972) mentioned above, and it is possible that the application of trace elements to catchments or their waters would stimulate the growth of aquatic producers in many waters rich in N and P (but unproductive). Considerations such as this make it imperative that each water is considered separately in respect to the element limiting production. Furthermore, it is important to distinguish, as did Vallentyne (1974), between that element which limits production and that element whose addition to a water is controllable to the extent that it can be made to limit production; the former may be any of several elements, while the latter is most usually phosphorus. Wood (1975) recommended that some research into identifying limiting nutrients be undertaken in Australia.
If the importance in Australia of atmospherically derived salts is accepted, and if the contribution from the catchment rock to a water is small, then the atmospheric loading of nutrients must be assumed to be of great significance for Australian water bodies. Such would appear to be the case for the Onkaparinga River, South Australia (Buckney, 1979). In any case, atmospheric accretion of nutrients represents the minimum possible loading to a catchment, and a knowledge of the atmospheric loading could be used to set broad limits to the size of planned lakes. Thus, if the atmospheric loading of phosphorus is 500 mg ha⁻¹ yr⁻¹, and the catchment area is 80,000 ha, then the load transmitted to the water body would be at least 40 kg yr⁻¹. (This assumes that all P from the atmosphere ultimately appears in the streams of the catchment.) Since Vollenweider (1968) has shown that for a lake of 10 m mean depth, an annual P loading of about 2 μg per hectare of lake surface results in eutrophic conditions, a minimum area of such a lake would be 20 ha (volume—2 ML). Anything smaller than this could be expected to be eutrophic.

A great many Australian inland waters appear very turbid—the inland rivers and many salt lakes are good examples—and this means that a high proportion of all nutrients is likely to be in the particulate fraction of a water sample. Two consequences of this are: high nutrient concentrations may not be utilised by plants which are shaded by the suspended matter, and settling of the particles under still conditions or when the major ion chemistry promotes aggregation of particles could be a major component of the nutrient flux in some systems. Buckney (1979) has demonstrated the importance of the settling of seston in regulating nutrient concentrations in laboratory experiments. It would be possible to find a situation where attempts to limit production by removal of nutrients—in particulate form—were thwarted by the effects of improved light conditions which resulted from the nutrient removal; consideration would have to be given to reduction of the ‘soluble’ levels of the nutrients.

**Chemical variability**

All inland waters exhibit chemical changes which are relatively large in comparison to those occurring in marine situations, with the possible exception of near-coastal localities. Two main sorts of approaches have been taken to the study of chemical changes in waters; one consists of attempts to identify a broad seasonal pattern, the other is, in a wide sense, statistical.

In Australia, reporting of the seasonal patterns of chemical change has usually accompanied a biological study on the same water. It would be difficult to summarise all such reports in a small space, and so an approach is made here towards presenting a broad scheme—one which applies precisely nowhere but which identifies the main annual changes to be expected.

Williams (1967) reported that seasonal fluctuations in salinity appeared to be more or less in rhythm with climatic fluctuations, maximum concentrations occurring in the dry season and minimum in the wet, and that non-seasonal changes were associated with climatic irregularities. Timms (1970a, b, 1973) has emphasised that floods or showers may obscure regular seasonal changes in salinity. Changes in stoichiometry are more complex; very fresh waters may vary
considerably in stoichiometry (see Cheng and Tyler, 1973a; Croome and Tyler, 1972), while to date the Australian experience is for salt lake stoichiometry to be almost invariable despite considerable changes in salinity (Bayly and Williams, 1966; Walker, 1973; Williams and Buckney, 1976a). In a great number of waters, a photosynthetic increase of pH could be expected during the summer months.

Large areas of Australia experience long dry periods during which catchments may desiccate considerably. The onset of rain may not, therefore, result in river flows for some time (that is, until the catchment is saturated). Thus, catchment soils may be in contact with water for a considerable time before any transport of material via streams is achieved. The first discharges during the wet season are often chemically unusual for that reason, and also because they carry the dry season accumulation of litter, manure and cyclic salts. Typically, such discharges have high colour, low pH, sodium and chloride predominance with (often) high levels of bicarbonate and of most nutrients. Similar conditions apply during the early stages of flood flows, which may also be highly turbid (see Buckney, 1977). These early wet season influxes, which have been recognised in a semi-permanent stream in South Australia (Buckney, 1979), are the only Australian analogue of the northern hemisphere autumnal leaf litter contribution to streams (Hynes, 1970). Since these early wet season flows contribute a high proportion of the annual nutrient load to streams, the practice of using reservoirs to store as much of the wet season discharge as possible (see Timms, 1970b) could well contribute as much to the trophic conditions in the water as land use activities in the catchment; most of such reservoirs are farm dams.

Statistical approaches to chemical variability have been used either to indicate the extent of chemical change which can be expected in water or else to relate chemical changes to physical variables. The gross pattern seems to be that variability of concentration of an ion increases as the mean concentration increases.

The observed range of concentration has been used to specify the chemical variability of a water (Bayly and Williams, 1966; Buckney and Tyler, 1976; Timms, 1970b; Williams and Buckney, 1976a), as has the ratio of maximum value obtained to minimum obtained (Timms, 1970b). Fluctuations of total dissolved solids in some fresh waters have been shown (Timms, 1970b) to be correlated with the 'water renewal ratio' (catchment area/lake surface area). Bayly and Williams (1966) related salinity changes in closed lakes to the interaction of precipitation and evaporation; they further claimed that salt lakes were both absolutely and relatively more variable in salinity than fresh waters. This latter observation has been confirmed for Tasmanian waters (Buckney, 1976b); the standard deviations of ionic concentrations are better related to the means by logarithmic (rather than linear) equations. Climatic variation is probably the most important determinant of changes in both absolute concentration and in stoichiometry, although the water renewal ratio is also important (Buckney, 1976b); as mentioned above, salt lakes seem to be stoichiometrically stable.

Ionic concentrations can be related to water availability. Hynes (1970) re-
corded an equation of the general form:
\[ c = aQ^b \]
relating concentration, \( c \), to stream discharge, \( Q \), by means of the constants \( a \) and \( b \). Hall (1970) has provided a theoretical derivation for this equation and for some others. Equations of this sort may be highly descriptive of the chemical environment of a stream, but often are of little more use than as indicators of trends (Buckney, 1977, 1978). The magnitude and sign of \( b \) is a useful index of stream conditions; it will show whether the concentration of an ion is likely to increase (\( b \) +ve) or decrease (\( b \) -ve) with increasing discharge and whether the ionic concentration is sensitive (magnitude of \( b \) large) or insensitive (\( b \) near zero) to changes in discharge. Flood flows and the first discharges of the wet season usually deviate most from these equations.

Some ecological considerations

Ranges of salinity in many athalassic saline waters (e.g. Bayly, 1970; Williams and Buckney, 1976a,b) quite often exceed the salinity tolerance ranges of a wide variety of organisms (see Bayly (1972) for many of these data). In consequence, a great deal of the ecology of such waters is concerned with the topic of osmoregulation. The stoichiometric stability of these waters simplifies somewhat the application of laboratory data to field experience, and perhaps even acts to select for certain types of organisms—Bayly (1969) described the occurrence of the typically ‘freshwater’ Boekella triarticulata at a salinity of 22.3\%/oo in a lake for which bicarbonate-carbonate was enriched relative to what is normally found in Australia.

Chemical conditions in the dilute waters have long been regarded as indicators of nutritional status; the wide interest in nitrogen and phosphorus is commensurate with their importance to water users, but the presence or absence of any of at least nineteen elements (Vallentyne, 1974) could conceivably have similar effects. Gerloff (1969) demonstrated potassium limitation in aquatic macrophytes growing in some Wisconsin lakes. The importance of silicon in controlling diatom population size is well known (e.g. Cheng and Tyler, 1973b).

Lund (1965) summarised the distribution of the major freshwater phytoplankton groups among water types. The Chlorophyceae, for example, are most common in calcium-poor waters, while diatoms and blue-green algae seem to prefer harder waters. These observations have led to the idea that various algal types can be classified as calciphilic or calciphobic. Moss (1972) has shown that this concept is probably too simplistic and that factors (other than the major ions) which differ between the calcium-rich and calcium-poor fresh waters could be important. Buckney (1976a) has demonstrated that some such differences do exist in Tasmanian waters—in contrast to calcium-rich waters, the sodium-chloride fresh waters are acidic, coloured and enriched in iron and manganese—and other differences could be demonstrable. According to Gibbs (1970), waters in similar climates with different water chemistries are displaying differences in geochemical inputs; it would be surprising if such differences did not extend to a variety of nutrient species. All this points to a need for closer study of the nutritional ecology of a wide range of organisms.

Aside from the above static observations, the effects of chemical variability need to be considered. The persistence
of a particular species in a water presupposes some degree of persistence in the availability of its essential elements, so that changes in water chemistry may be reflected in changes of community composition. This would be particularly so for organisms (e.g. the unicells) with rapid reproductive cycles. The chemical changes associated with the first discharges after the dry season (see above) might qualify as 'shock' factors which regulate algal growth and succession (Round, 1971). The well-known phenomenon of luxury consumption of nutrients (Gerloff, 1969; Vallentyne, 1974) is perhaps the response of organisms to ephemeral appearances and disappearances from solution of essential nutrients.

Environmental variability is thought to reduce species diversity; Whittaker, (1972) and Williams (1970) have placed this idea in the context of salinity instability in salt lakes. It would be of interest to examine its applicability to freshwater systems in the light of these nutritional considerations.

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Chemistry of Australian waters


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4 The benthos of Australian lakes

B. V. Timms

Introduction

Strictly the term 'benthos' refers to organisms of the solid-liquid interface (Haekel, 1891), but when most authors use this term they mean 'herpobenthos', that is animal inhabitants of sediments (Warming, 1923). This usage is adopted here.

Data on the benthos of some Australian lakes have become available only recently. Two reasonably detailed studies have been completed, one on Lake Werowrap, a saline lake near Colac, Victoria (Paterson and Walker, 1974a,b), the other on three nearby deep lakes ranging in salinity from fresh to 58°/00 (Timms, 1973a). Significantly, three of these four lakes are saline, indicating an early interest in the unique fauna of Australia's salt lakes. Two further studies are in progress.

Useful as these studies are, much of this review needs to rely on surveys, many as yet unpublished. These now cover some seventy lakes, most of which are either deep volcanic lakes, highland glacial lakes, or coastal dune lakes; almost all are in eastern Australia. This bias is unfortunate, for besides problems of inadequate coverage of lake types, there are few data on lakes in the western half of Australia, on inland billabongs, and on reservoirs and small lakes elsewhere.

Almost all available information relates to macrobenthos, i.e. benthic animals visible to the naked eye and retained by a sieve of mesh size ~ 0-4 mm. However, mesobenthic species (semi-microscopic animals such as small ostracods, nematodes, copepods, cladocerans and water mites) have been recorded in many surveys. No data are available on microbenthos, i.e. protozoans, minute nematodes, gastrotrichs, etc.

Review of benthic studies

Volcanic lakes in the south-east. Data are available on seven lakes; Purrumbete (Timms, 1973a), Mumblin and Surprise (Timms, 1975) and Burrumbete (Environment Protection Authority, 1976) in western Victoria, and Valley, Leake and Edward near Mt Gambier, South Australia (Timms, 1974a). Lake Purrumbete is the largest and most thoroughly studied; it contains 38 species (Table 1). Important ones include the oligochaetes Branchiura sowerbyi, Lumbriculus variegatus and Limnodrilus hoffmeisteri, the amphipod Austrochiltonia subtenuis, the chironomids Coelopynia pruinosa and Chironomus duplex and the gastropod Potamopyrus nigra. Most species are restricted to the littoral and sub-littoral areas (Fig. 3) with only four—Phaenocora sp., L. hoffmeisteri, Tubifex tubifex, and Coelopynia pruinosa—known from the profundal. Annual average biomass was 5-47 g m⁻² during the period of study with a pronounced peak in the littoral and little in the profundal.
Table 1  Species diversity and mean biomass (g m$^{-2}$) in some Australian lakes and impoundments

<table>
<thead>
<tr>
<th>Lake</th>
<th>Purumbete</th>
<th>Bullenmerri</th>
<th>Gnotuk Valley</th>
<th>St Clair</th>
<th>Great Tas. Highland</th>
<th>Kasco, Glacial dune</th>
<th>coastal dune</th>
<th>modified dune</th>
<th>north Qld Maar</th>
<th>north Qld Lagoon</th>
<th>Tasmanian Reservoir</th>
</tr>
</thead>
<tbody>
<tr>
<td>Comment</td>
<td>S.E. Aust. Maar (8o/oo)</td>
<td>Saline (38o/oo)</td>
<td>S.E. Aust. Maar</td>
<td>Tas. Highland</td>
<td>Tas. Highland Large</td>
<td>Kosciusko Glacial</td>
<td>Coastal Dune</td>
<td>Modified Dune</td>
<td>North Qld Maar</td>
<td>North Qld Lagoon</td>
<td>Tasmanian Reservoir</td>
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<tr>
<td>Porifera</td>
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<td>1</td>
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<tr>
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<td>1</td>
<td>1</td>
<td>1</td>
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<td>1</td>
</tr>
</tbody>
</table>

| Total number | 38 | 12 | 6 | 24 | 38 | 45 | 10 | 6 | 11 | 29 | 8 | 21 |
| Mean biomass | 5.47 | 8.33 | 1.77 | 16.8 | 4.7 | 14.87 | 0.7 | 1.91 | 0.78 | 8.66 |


The nature of the resource
Both curves are typical of most deep lakes in Australia.

Of the six chironomids present in Purrumbete there is some separation of species populations by depth. *Coelopynia pruinosa* and *Ch. duplex* have one generation per year with emergence protracted over summer-autumn while *Cryptochironomus griseidorsum* has two or more generations annually with spring the most important emergence period.

The other deep lakes studied are smaller and less deep than Purrumbete and hence theoretically present less heterogeneous environments and so fewer species. This is exemplified in the Mt Gambier series: Valley Lake (16 m deep) has 24 species, Lake Edward (7 m) 17 species, and Lake Leake (4 m) 12 species (Timms, 1974a). In Valley Lake the influence of substrate type and location is readily apparent. Grit supports much lower diversity than mud, especially when the latter is near littoral weed beds. The most common or widespread species in the Mt Gambier lakes and in Mumblin and Surprise are the oligochaete *Antipodrilus davidis*, the amphipods *Austrochiltonia* spp., the odonatan *Hemicordulia tau*, the chironomids *Procladius villosimanus*, *Chironomus duplex* and *Cladopelma curtivalva*, and *Chaoborus* sp. Considerable seasonal variation in biomass and some in species composition has been noted in Mumblin and Surprise (Timms, 1975) and at least

![Fig. 3 Number of benthic species according to depth in three south-eastern Australian maar lakes.](image-url)
in Edward, there is secular variation in the composition and abundance of dominant chironomids (J. Martin, unpublished).

The large shallow lakes of this region seem to have a particularly depauperate fauna. For instance only chironomids and molluscs have been reported from Lake Purrumbete, although numbers are high (Environment Protection Authority, 1976). Wind-induced stirring of the substrate probably inhibits colonisation by many benthic species, as it is thought to in shallow Lake Crescent in Tasmania (Timms, 1978).

Most of the freshwater lakes on the volcanic plains of western Victoria and south-east South Australia have standing crops > 5 g m⁻² and often > 10 g m⁻² and so are classed as productive (see p. 34).

**Saline lakes.** Salt lakes are many and varied in Australia. Those that are ephemeral and highly saline (salinity > 10⁰/₀₀) seem to lack a macrobenthos, though mesobenthic-planktonic ostracods such as *Platycypris* may be abundant (Geddes, 1976, and personal communication). Shallow, semi-permanent to permanent lakes with lower salinities (< 10⁰/₀₀) may support chironomids; often populations of these are abundant. The best studied example, Lake Werowrap (Paterson and Walker, 1974a), had an extremely high average annual wet biomass of 53.75 g m⁻² during the period of investigation; most of the biomass was the chironomid *Tanytarsus barbitarsis*, but small numbers of the beetle *Necterosoma penicillatus* were also present (Fig. 4). Slightly saline lakes support large biomasses too, but with greater numbers of species. For instance, Lake Coragulac when surveyed in January 1975 had a salinity of 5⁰/₀₀, and contained *Mytilocypris splendidida*, *Austrochiltonia subtenuis*, oligochaetes, *Procladius villosimanus*, and *Chironomus duplex* (dominant) with a combined wet biomass of 20.7 g m⁻² (Timms, unpublished). Lake Modewarre (4⁰/₀₀) contains about twice as many species as Coragulac and these are also abundant (Pollard, 1971). In deeper salt lakes, the decrease in number of species with depth (Fig. 3) becomes less at
higher salinities. This is thought to result from saline lake species being less restricted in microhabitat (Timms, 1973a).

The above examples indicate that there is a general decrease in species diversity with increasing salinity. In theory this relationship should be more precise for lakes which are physiographically and physico-chemically similar. Four permanent maar lakes in western Victoria form such a series (Table 2); they have a markedly lower diversity than that which obtains at lower salinities (possibly 30+ in fresh water) but then only a slight further drop in diversity with increasing salinity. More marked is the depression of standing crops with increasing salinity (or salinity related factor), a phenomenon also recorded elsewhere (e.g. Rawson and Moore (1944) on Canadian lakes). However, the relative importance of species groups at differing salinities is unique to Australia. At lower salinities gastropods and oligochaetes are important, but at higher ones ostracods, isopods and ceratopogonids are of some consequence. Chironomids may dominate at any salinity (Chironomus duplex at lower salinities and Tanytarsus barbitarsis at higher ones). The life cycles of some benthic species have been elucidated. Tanytarsus barbitarsis in Lake Werowrap (Paterson and Walker, 1974a) has seven generations per year, five during late October to March each taking about a month, one in March-May, and an overwintering one. Numbers fluctuate greatly because of synchronous emergence early in the season, summer predation by Necterosoma penicillatus, and seasonal variation in food availability. Necterosoma penicillatus has one generation per year with free-swimming larvae during summer (Paterson and Walker, 1974a). In Lake Bullenmerri, Chironomus duplex typically has one to two generations per year, a long overwintering one and perhaps a shorter summer one, though at greater depths development may take longer than twelve months (Timms, 1973a).

### Table 2  Benthos of four saline maars in western Victoria

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Bullenmerri&lt;sup&gt;1&lt;/sup&gt;</th>
<th>East Basin&lt;sup&gt;2&lt;/sup&gt;</th>
<th>Gnotuk&lt;sup&gt;1&lt;/sup&gt;</th>
<th>West Basin&lt;sup&gt;2&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td>Salinity (°/oo)</td>
<td>8</td>
<td>44</td>
<td>60</td>
<td>88</td>
</tr>
<tr>
<td>Number of species</td>
<td>12</td>
<td>5</td>
<td>6</td>
<td>5</td>
</tr>
<tr>
<td>Biomass g m&lt;sup&gt;-2&lt;/sup&gt;</td>
<td>8.3</td>
<td>3.1</td>
<td>1.8</td>
<td>0.9</td>
</tr>
<tr>
<td>Percentage contribution by:</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Oligochaeta</td>
<td>37</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Ostracoda</td>
<td>—</td>
<td>60</td>
<td>11</td>
<td>75</td>
</tr>
<tr>
<td>Isopoda</td>
<td>—</td>
<td>33</td>
<td>19</td>
<td>—</td>
</tr>
<tr>
<td>Trichoptera</td>
<td>2</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Chironomidae</td>
<td>48</td>
<td>—</td>
<td>65</td>
<td>1</td>
</tr>
<tr>
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<td>7</td>
<td>5</td>
<td>24</td>
</tr>
<tr>
<td>Gastropoda</td>
<td>13</td>
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</tr>
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</table>

<sup>1</sup> Data from Timms (1973b).

<sup>2</sup> Data from Timms and Brand (1973).
Daily and seasonal variations in spatial distribution are obvious in salt lakes. In Werowrap (Paterson and Walker, 1974a) a considerable proportion of the *Tanytarsus barbitarsis* population becomes free-swimming on sunny days. Also, other factors such as rough water conditions lead to a variable percentage of the population being in the sediments, in the water column, or among littoral stones. There are seasonal changes in depth distribution of *Haloniscus searlei* in Gnotuk and of *Coxiella striata* in Bullenmerri associated with dispersal of juvenile stages (Timms, 1973a).

**Tasmanian lakes.** Tasmania has thousands of lakes, but the benthos of only a few has been surveyed. Of the larger highland lakes, Lake St Clair, Dove Lake, Lake Sorrell, Lake Crescent (Timms, 1978), the Great Lake, Arthurs Lakes (W. Fulton, unpublished) have been surveyed; their benthos comprises 20-45 species with oligochaetes, chironomids and sometimes amphipods and sphaerid bivalves dominant. Small mountain tarns such as Lake Rhona (D. Coleman, unpublished) and Hartz Lake (Knott, Suter and Richardson, 1978) contain < 15 benthic species with chironomids and phreatoicid and gammarid crustaceans dominant. Common, widespread species present include *Limnodrilus hoffmeisteri*, *Phreodrilus* spp., the crustaceans *Neoniphargus* spp. and *Metaphreatoicus* sp., the chironomids *Coelopynia pruinosa*, *Procladius villosimanus*, *Chironomus oppositus* and *Rieithia* spp., and *Sphaerium* spp. About one third is endemic to Tasmania. The most diverse groups are chironomids, crustaceans, and particularly oligochaetes in lakes such as Lake St Clair and the Great Lake (Table 1). There are also representatives of many groups absent or unimportant in continental Australia. These include nematomorphians, nemerteans, neoniphargian, phreatoicid and anaspidacean crustaceans, mecopterans and plecopterans.

Although St Clair (*z_m = 167 m*) and Dove (*z_m = 60 m*) are deep, their profundal fauna lacks any special characteristics. Oligochaetes, *Procladius villosimanus* and *Chironomus oppositus* form a sparse community, which in the case of Lake St Clair probably does not extend much beyond 75 m, thus leaving the predominately blue clays of the deep profundal uncolonised.

Of the many factors which affect benthic spatial variation (see Brinkhurst, 1974), the influence of substrate type is particularly evident. Thus species distributions are often clumped and biomass between stations variable; for example, in Lake Sorell numbers varied from 950 to 4,580 m⁻² at similar depths, but on different substrates. In Lakes Sorrell and Crescent the benthos near weedbeds had greater biomass and species diversity than elsewhere. Lakes St Clair and Dove receive considerable inputs of allochthonous organic detritus, with the result that there is greater biomass and some differences in species composition or relative abundance at that end of the lake into which the river enters.

Mean standing crops are typically < 5 g m⁻², and in some, e.g. Dove Lake, are quite low (1.7 g m⁻²). Though these are spot values subject to errors associated with seasonal fluctuations, they indicate a general oligotrophy to mesotrophy. Lake Dobson with a standing crop value of 6.9 g m⁻² in October 1972 had an unusually high value which may partly reflect its artificial enrichment in
the early 1950s, or a rich supply of allochthonous organic matter.

**Highland lakes of mainland Australia.** In contrast to Tasmania, there are very few highland lakes in continental Australia. All have been surveyed (Timms, 1974b, and unpublished). The alpine lakes near Mt Kosciusko and Lake Tali Karng in Victoria are all small and as expected their diversity is lower, averaging 10 species per lake. Common species include *Antipodrilus davidis*, the phreatoicid *Metaphreatoicus australis*, a gammarid amphipod, the chironomids *Procladius villosimanus* and *Chironomus oppositus*, and *Pisidium tasmanicum*. Some of the less common species, e.g. the snail *Glacidorbis hedleyi* and the flatworm *Spathula truculenta* seem to be restricted to the area.

Biomass values are variable; they are predictably low in oligotrophic Lakes Tali Karng (2.25 g m$^{-2}$) and Club (2.4 g m$^{-2}$), but remarkably high in Cootapatamba (9.5 g m$^{-2}$), Albina (14.9 g m$^{-2}$) and Blue (12.3 g m$^{-2}$), lakes considered unproductive by other criteria. Phreatoicids are almost restricted to and are dominant by weight in those lakes without fish (Cootapatamba and Albina), but chironomids and oligochaetes dominate where fish (*Galaxias findlayi*) are present. Predation pressure from fish is apparently responsible for this difference, and the absence of benthic amphipods in Club and Blue.

**Coastal dune lakes.** Typically, these lakes lie on sand, contain acid waters with high concentrations of humic material and low concentrations of total dissolved solids, usually have an abundant but monospecific zooplankton, and possess a variety of littoral invertebrates (Bayly, 1964; Bayly *et al.*, 1975; Bensink and Burton, 1975). However, their benthos is unproductive and depauperate (Bensink and Burton, 1975). Some, for example Freshwater Lagoon at Myall Lakes, N.S.W., have apparently no macrobenthos at all (Timms, 1972); others, for example Cooks Lagoon near Woodburn, N.S.W., have only *Chaoborus* sp. (Timms, unpublished). Perhaps most have a few species of oligochaete, chironomids and a *Chaoborus* species: Brown Lake on Stradbroke Is., Queensland, has oligochaetes, three species of chironomid, a mayfly species and a species of shrimp (Bensink and Burton, 1975); Lake Ainsworth near Ballina, N.S.W., has an oligochaete, three species of chironomid, a *Chaoborus* sp., and a species of water mite. In both lakes, most species are restricted to shallower areas, so that there is no or very little benthos at depths > 5 m. In Brown Lake numbers averaged 244 m$^{-2}$, and in Lake Ainsworth 900 m$^{-2}$ with a wet biomass of 0.4 g m$^{-2}$.

The extremely low species diversity reflects the lack of biological adaptation to their extreme acidity and low salinity (as in bog lakes in the northern hemisphere: Wetzel, 1975), and possibly the lack of food derived from the sparse phytoplankton and the general tendency for the highly organic sediments to be reducing.

Dune lakes that abut rock substrata or receive inflows from such regions are a little different from typical dune lakes in chemical and zooplanktonic characteristics (Timms, 1969). Further, their benthos is more diverse, though not much more productive. Lake Elusive in eastern Victoria has ten species including *Chironomus oppositus*, *Tubifex tubifex*, ceratopogonids, and *Sphaerium* sp. of which the first two dominate (Timms,
The nature of the resource

Lake Hiawatha near Grafton, N.S.W., has eleven species with *Chironomus* sp. and *Chaoborus* sp. dominant and ceratopogonids, trichopterans and water mites. Average biomass in the latter was $< 1.0 \text{ g m}^{-2}$.

**Coastal lakes of marine derivation.** In south-eastern Australia, and possibly elsewhere, there are some freshwater lakes located coastally that had a marine phase in their development. The fauna is basically of freshwater origin, but some species are of marine derivation. The best documented example, Lake Barracoota in East Gippsland (Timms, 1973b), has 14 benthic species including the polychaete *Boccardia limnocola*, an anthurid isopod apparently new to science (*Cyathura* sp.), and the sphaeromatid isopod *Cymodetta gambosa*. The freshwater component comprises a nematode, a mayfly, six species of chironomid (*Procladius villosimanus* and *Chironomus nepeanensis* dominate), and a ceratopogonid species.

Other known ex-marine coastal lakes include the Bridgewater Lakes and Lake Bong Bong near Nelson, western Victoria, which have *Boccardia limnocola* (Timms, 1976), and a series of lakes between Narooma and Bega, southern N.S.W., which have *Cymodetta* sp. (Timms, unpublished).

**Shallow freshwater lakes of Western Australia.** Only Lake Monger, near Perth, has been studied in any detail (Edward, 1957, 1964). It contains 11 macrobenthic species including 6 chironomids. The latter are generalist feeders though each species has preferences. Some scant survey data on some other lakes (J. Martin, unpublished) indicate a similar low diversity of chironomids.

**North Queensland lakes.** The only deep lakes in north Queensland are the maars on the Atherton Tableland, but there are many shallow lagoons on floodplains and in upland areas associated with lava flows. The maars and some lagoons have been examined (Timms, 1979).

Lakes Barrine and Eacham on the Atherton Tableland contain some 25 and 19 macrobenthic species respectively. The chironomids *Procladius* sp., *Ablabesmyia* sp., and *Chironomus nepeanensis* dominate in both lakes whilst *Dictrotendipes* sp. is important in Eacham and *Conochironomus* sp. and a variety of gastropods in Barrine. An interesting occurrence in Barrine is *Cyclostheria hislopi*, an aberrant circumtropical conchostracan living in permanent waters, unlike most members of this group. Almost all of these species are restricted to littoral and sub-littoral regions with only *Chaoborus* and the chironomid dominants in Barrine occurring sparsely in the profundal. Mean weighed biomass is low for both lakes; 1.9 g m$^{-2}$ in Barrine and 0.8 g m$^{-2}$ in Eacham. Curiously, Lake Euramoo has no macrobenthic fauna, but its substratum is extremely unstable and the hypolimnion is deoxygenated for long periods (McCarrach and Russell, unpublished).

The lagoons associated with the lower and upper Burdekin River have a depauperate fauna with an average of six macrobenthic species each. Almost all are dipterans with *Procladius, Coelopynia, Cryptocladopelma, Chaoborus*, and a ceratopogonid species in most. It is thought that physico-chemical instability together with clay substrates contribute to this reduced diversity and the exclusion of other groups. Biomass is $< 1 \text{ g m}^{-2}$ in most lagoons.
**Artificial impoundments.** Despite the large and growing number of reservoirs in Australia almost nothing is known of their benthos. It seems, according to isolated dredgings from a few mainland reservoirs, that their benthos is generally sparse with but few species represented (J. Martin, unpublished). However, some, particularly the older ones, may support a richer benthos, though diversity is still lower than in lakes. For instance, Lake Leake and Tooms Lake in Tasmania are 90 and 140 years old, have standing crops of 8-7 and 7-2 g m⁻² and contain 19 and 12 benthic species respectively (Timms, 1978). Both lack some of the groups which occur in Tasmanian highland lakes such as nematomorphians, plecopterans, *Nannochorista*; there are different species of *Rietha* and *Metaphreatoicus*, and in addition there are odonatans and *Chaoborus*. It is not known if these differences are due to altitude-related factors, to the geological youth of the reservoirs, or to the lack of habitat heterogeneity.

Comments on the benthos of farm dams are to be found in Chapter 32.

**Past faunas.** Although cores have been obtained from many maars, most studies on these have been botanical in emphasis, and information on past faunas is limited to one small site, namely Lake Werowrap in western Victoria (Paterson and Walker, 1974b). In this, wide fluctuations in salinity in the past are indicated; when saline, *Tanytarsus barbitarsis* and often *Procladius paludicola* were present, but during fresher phases these species were replaced by *Chironomus duplex*. The abundance of *Tanytarsus barbitarsis* increased during the saline phase, perhaps indicating a change in trophic status of the lake.

**Discussion**

The information reviewed above needs to be interpreted with care, as it is largely based on single visit surveys. Although many samples per lake may have been obtained (e.g. 60 in Lake Barrine), seasonal influences and chance no doubt mean that a lake’s full quota of species was not recorded. For instance, the instantaneous number of species each month in Purrumbete over a year averaged 22, whereas the total number for the year was almost twice that at 38 (Timms, 1973a). Also biomass figures fluctuate with season, so that mean values based on a single visit can easily differ from the grand yearly mean by up to a factor of two. In this context it should be noted that most lakes surveyed by the author were sampled in winter/spring when biomass values were likely to be seasonally high. Despite these problems there are a number of unifying themes and evident trends.

**Diversity.** There is no doubt that species diversity is low by northern hemisphere standards. Even the most studied lakes, Purrumbete and the Great Lake, have only 38 and 45 species respectively. Other deep lakes such as St Clair, Dove, Valley, Barrine and Eacham have 20 to 38 species. In contrast, most moderately deep Holarctic lakes have 75 or more benthic species. Comparative data are provided by Great Slave Lake with 95 (Rawson, 1953), Saginaw Bay, Lake Huron, with 44 genera and twice as many species (Schneider, Hooper and Beeton, 1969), and Llyn Tegid, Wales, with 50+ species in the profundal alone (Hunt and Jones, 1972).

Other Australian lakes studied have even fewer species. As indicated, various factors contribute to this. They
include lake size (cf. lakes near Mt Gambier), harsh general physico-chemical conditions (cf. the floodplain lagoons and maars of north Queensland), and, as particular aspects of physico-chemical conditions, salinity (cf. Victorian maar lakes, Table 2), and humic acid conditions (cf. dune lakes of eastern Australia). Reservoirs and farm dams also have relatively few species; this is probably because of habitat homogeneity, unsuitable substrates, and the adverse influence of drawdowns. Biological factors such as fish predation may depress diversity, as in some of the lakes near Mt Kosciusko (Timms, unpublished) and some Tasmanian farm dams (Walker, 1974), but the evidence is equivocal.

Curiously, Tasmanian lakes tend to have higher species diversities than mainland lakes. In one study there was a mean of 21.3 species in Tasmanian lakes and 16.6 in mainland ones (Timms, 1978). This may be an expression of the oft-cited claim (e.g. Margalef, 1964) that oligotrophic lakes have a more diverse fauna than eutrophic ones, or it may be that for some reason opportunities for speciation have been greater. Certainly there are more endemic species in the benthic fauna of Tasmanian lakes, particularly among the oligochaetes and crustaceans, than there are in the benthos of lakes elsewhere in Australia.

Lack of diversity is evident within most taxa. These include the Nematoda, Ostracoda, Ephemeroptera, Plecoptera, Trichoptera, Megaloptera, Neuroptera, Gastropoda, Bivalvia and particularly the Chironomidae. Concerning the latter, more than 30 species are known from Saginaw Bay, Lake Huron (Schneider, Hooper and Beeton, 1969), 20 from Llyn Tegid, Wales (Hunt and Jones, 1972), and 13 from the profundal alone of Lake Washington (Thut, 1969). In contrast, the total for almost all Australian lakes is < 7, the exceptions being Lake St Clair with 11 and the Great Lake with 8 - (Table 1). Forsyth (1976) noted a similar condition for New Zealand lakes. Exceptions to the above trend are provided by the Turbellaria, Ceratopogonidae, and in Tasmania by the Isopoda and Oligochaeta. The latter group is depauperate in mainland lakes, but St Clair and the Great Lake have > 10 species, many endemic.

The reasons for the low diversity of the benthic fauna in Australian lakes are inadequately understood. Four can be suggested:

(i) The diversity of Australia's benthic fauna was restricted by zoogeographic isolation. However, although this perhaps applies to the even more depauperate benthic fauna of New Zealand (Forsyth, 1976), it seems unlikely to have been important for Australian lakes since almost all known benthic groups have been recorded. Moreover, other faunal communities restricted by isolation have undergone an adaptive radiation and are diverse (cf. terrestrial mammals).

(ii) There has been insufficient time for speciation as most lakes studied (volcanic maars, glacial lakes, dune lakes) are young, that is are less than 30,000 years old. However, northern hemisphere lakes of similar age have larger numbers of benthic species. Further, the Great Lake in Tasmania has an origin which predates Pleistocene glaciation (Davies, 1974), and hence is one of the oldest and largest of freshwater lakes in Australia. It is noteworthy that its fauna is relatively
Benthos

...
The nature of the resource

tarsus barbitarsis and Coxiella striata is characteristic. Very few species are restricted to the volcanic lakes in southeastern Australia. These include the rhabdocoel Phaenocora sp. and the neuropteran Sisyra sp. As expected, north Queensland lakes have some tropical species, e.g. the oligochaetes Aulodrilus remex and Branchiodrilus hortensis, the crustaceans Cyclestheria hislopi and Cherax waselli, and the chironomids Nilodorum biroi and Clinotanypus crus, but most are widespread southern species.

Coastal dune lakes and ex-marine coastal lakes (except for species of marine derivation) apparently lack restricted benthic species. This contrasts sharply with the situation for the crustacean zooplankton (Bayly, 1964), littoral cladocerans (Smirnov and Timms, unpublished), and some littoral groups such as Odonata (Arthrington and Watson, personal communication) and higher crustaceans (Timms, unpublished). A possible reason for this has been discussed above.

Production. Although benthic productivity of only one lake in Australia has been determined, the result is noteworthy. Lake Werowrap, a shallow saline lake in western Victoria, has a net annual production of 66.4 g dry weight m⁻² (320 kcal m⁻²), which is the highest figure recorded for the benthos of any inland water (Paterson and Walker, 1974a). Expressed as an average standing crop biomass, this is 8.07 g dry weight m⁻² or 53.75 g wet weight m⁻² (all values henceforth are expressed as wet weight).

Most other data on benthic production in Australian lakes are spot determinations of standing crops. Values range from 0 to 22.3 g m⁻². Despite some variation within each lake group, they can be arranged in a series of ascending values (Table 3). Following Lundbeck (1936), Deevey (1941) and others, lakes with low values (< 2 or 3 g m⁻²) are classed as oligotrophic, and those with high values (> 6 or 8 g m⁻²) as eutrophic. According to various authors working in the northern hemisphere, these assignments should correlate with sediment organic matter content. Additionally, with increasing trophic status there should be a change in the relative proportion that various taxa contribute to the biomass at a given depth (e.g. Ahren and Grimas, 1965), and a change in the species of

<table>
<thead>
<tr>
<th>Lake group</th>
<th>Number of lakes</th>
<th>Mean biomass for group</th>
<th>Range of biomass</th>
<th>Mean percent organic matter for groups</th>
<th>Range in percent organic matter</th>
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</thead>
<tbody>
<tr>
<td>Dune lakes</td>
<td>3</td>
<td>0.1</td>
<td>0.0-0.2</td>
<td>65</td>
<td>24-94</td>
</tr>
<tr>
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<td>3</td>
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<td>0.0-1.9</td>
<td>62</td>
<td>31-79</td>
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<tr>
<td>North Qld lagoons</td>
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<td>1.2</td>
<td>0.1-4.7</td>
<td>13</td>
<td>12-13</td>
</tr>
<tr>
<td>Dune contact lakes</td>
<td>3</td>
<td>2.8</td>
<td>0.4-7.4</td>
<td>35</td>
<td>22-49</td>
</tr>
<tr>
<td>Tasmania</td>
<td>5</td>
<td>3.6</td>
<td>1.7-6.9</td>
<td>21</td>
<td>13-28</td>
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<tr>
<td>Aust. highlands</td>
<td>5</td>
<td>8.3</td>
<td>2.2-14.8</td>
<td>18</td>
<td>17-32</td>
</tr>
<tr>
<td>South-east volcanic</td>
<td>6</td>
<td>10.4</td>
<td>2.6-22.3</td>
<td>46</td>
<td>32-51</td>
</tr>
</tbody>
</table>
dominant chironomids present (e.g. Thienemann, 1920).

For the benthos of Australian lakes, however, there is no ready correlation between sediment organic matter content and trophic status, although unproductive lakes (e.g. north Queensland lagoons wherein sediment organic matter content (OM) is 13 per cent of total matter) tend to have low values, productive lakes (e.g. the deep volcanic lakes of south-eastern Australia wherein OM is 46 per cent) high values, and humic lakes (e.g. dune lakes wherein OM is 65 per cent) even higher ones (Table 3). These figures are not much different from those (16, 34-2, 80, respectively) given by Rybak (1969) for lakes of comparable trophic status in Europe. Some exceptions can be explained partially by the fact that in them the amount of organic matter is increased by allochthonous input (e.g. in Dove Lake; Timms, 1978), or by the well-known inverse relationship between depth and production (e.g. Rawson, 1953), but for many the explanation remains a puzzle.

In the northern hemisphere, the relative contribution to total biomass by different taxa can give some indication of trophic status: amphipods dominate in many oligotrophic lakes, chironomids in meso-eutrophic lakes, and oligochaetes in eutrophic waters (see, for example, Ahren and Grimas, 1965; Carr and Hiltonen, 1960; Schneider, Hooper and Beeton, 1969). Such a scheme is inappropriate for Australian lakes. Unproductive lakes in Tasmania and near Mt Kosciusko are dominated by chironomids, and, though crustaceans are present, oligochaetes are relatively important, molluscs unimportant, and chaoborids absent (Table 4). In two of the Kosciusko lakes, crustaceans (albeit phreatoicid isopods) are dominant, but their unproductive status is questionable (Timms, unpublished). In the productive

<table>
<thead>
<tr>
<th>Lake</th>
<th>Taxonomic groupings</th>
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<tr>
<td>Dove</td>
<td>-</td>
</tr>
<tr>
<td>Blue</td>
<td>-</td>
</tr>
<tr>
<td>Barrine</td>
<td>-</td>
</tr>
<tr>
<td>Valley</td>
<td>-</td>
</tr>
<tr>
<td>S.A.</td>
<td>-</td>
</tr>
<tr>
<td>Mean for 7</td>
<td>-</td>
</tr>
<tr>
<td>Tas.</td>
<td>-</td>
</tr>
<tr>
<td>Mean for 4</td>
<td>-</td>
</tr>
<tr>
<td>Kosciusko</td>
<td>-</td>
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<tr>
<td>lakes</td>
<td>-</td>
</tr>
<tr>
<td>Mean for 4</td>
<td>-</td>
</tr>
<tr>
<td>S.E. Aust.</td>
<td>-</td>
</tr>
<tr>
<td>maars</td>
<td>-</td>
</tr>
</tbody>
</table>

<table>
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<tr>
<th>Trophic status</th>
<th>Oligo-trophic</th>
<th>Oligo-trophic?</th>
<th>Oligo-trophic</th>
<th>Eu-trophic</th>
<th>generally unproductive</th>
<th>supposedly unproductive</th>
<th>generally productive</th>
</tr>
</thead>
</table>
maars of south-eastern Australia, chironomids or sometimes oligochaetes are dominant, but, perhaps more significantly, crustaceans are unimportant, and chaoborids and particularly molluscs make some contribution (Table 4). This situation may suggest a scheme for Australia, but a better one involves zoogeographical considerations for the generally unproductive cold water lakes are readily distinguishable from others on such grounds. For warm water lakes, the importance of chironomids and insignificance of oligochaetes in unproductive Barrine (Table 4) and Eacham, and the decreased importance of chironomids in productive lakes, suggests that at least part of the northern hemisphere scheme has merit.

Deevey (1941) and Lundbeck (1936) have successfully related biomass:depth profiles to trophic status for many northern hemisphere lakes, each proposing four types. However, most deep Australian lakes have similar profiles irrespective of trophic status; there is a large peak in the littoral diminishing rapidly to low values in the profundal (Fig. 5). For two lakes, Hiawatha and Dove, there is little variation in biomass with depth (Fig. 5). The latter appropriately fit Deevey's and Lundbeck's oligotrophic class, but the remainder would be classed as eutrophic. Such a classification is most inappropriate for unproductive lakes like Tali Karng and Barrine. If then biomass:depth profiles are to be of any use in assignment of trophic status, a new scheme is necessary. A reasonable one would seem:

Oligotrophic (a)—no littoral peak; total biomass < 2 g m⁻²

Oligotrophic (b)—littoral peak; profundal biomass < 2 g m⁻²; total biomass < 3 g m⁻²

Mesotrophic —littoral peak; profundal biomass < 4 g m⁻²; total biomass < 6 g m⁻²

Eutrophic —littoral peak; profundal biomass > 5 g m⁻²; total biomass > 8 g m⁻²

This proposed scheme has several weaknesses. Firstly, it is based on data from few lakes. Additionally, edaphic conditions, allochthonous organic matter

Fig. 5 Depth:biomass profiles for seven lakes in eastern Australia. Boxed values on figure represent mean biomass values (g m⁻²).
input, and mean depth can greatly influence biomass levels. And seasonal inaccuracies undetected by spot determinations add a need for caution. The use of the scheme will require the exercise of care.

Finally, the relationship between trophic status and dominant chironomids remains for consideration. This relationship has certainly been useful overseas (e.g. Thienemann, 1920; Lundbeck, 1936; Brundin, 1956) and a generalised scheme (criticised by Deevey (1941) among others) which relates Orthocladius or Tanytarsus to oligotrophic lakes, Stictochironomus or Sergentia to mesotrophic lakes, and Chironomus benthophilus or C. plumosus to eutrophic lakes is reported to apply to most northern hemisphere lakes. Brundin (1958) goes so far as to claim world-wide applicability, but New Zealand (Forsyth, 1976) and Australia (Timms, 1973a, 1978) are exceptions.

The argument for Australia is that some of the genera and most of the species are absent. Although there may be some ecological equivalents (e.g. the eutrophic Chironomus benthophilus or C. plumosus = the Australian C. duplex), Australian unproductive lakes are also dominated by species of Chironomus. A species of Tanytarsus is dominant in productive saline lakes, yet this genus supposedly indicates oligotrophy. In addition, many common species, including Procladius villosimanus, Ablabesmyia notabilis, and Coelopynia pruinosa, are almost ubiquitous to all classes of lake. Nevertheless, as implied above, there are some relationships: Chironomus oppositus is generally dominant in cold water unproductive lakes (but it may also be present in warm water productive lakes) and Chironomus nepeanensis or, rarely, C. oppositus is important in warm water unproductive lakes. Riethia spp. are important in many mesotrophic cold water lakes, while Chironomus duplex is the dominant species in productive and slightly saline lakes. Tanytarsus barbitarsis dominates saline lakes. Although it may seem worthwhile to establish a scheme based on these data, this is not attempted because the data base is regarded as insufficient at present and there remain perhaps too many exceptions.

In overview, Australian lakes do not fit into trophic categories according to northern hemisphere observations concerning relative contributions by different taxa to biomass, biomass:depth profiles, or dominant chironomids. Moreover, it is difficult to order Australian lakes into any distinctly and complementary regional scheme. Perhaps this is because there are too few lakes involved, or because limited species diversity means that many species have 'generalist' niches, or because zoogeographical differences between the cold water lakes of Tasmania and highland Australia and other sorts of Australian lakes obscure trophic differences.

Acknowledgments
I sincerely thank Dr J. Martin of Melbourne University for his assistance in many ways to my benthological studies and specifically in the preparation of this review, for access to his unpublished data, for further chironomid identifications, and for comments on the manuscript. I also thank Mr D. Coleman, Dr B. Knott, and Mr W. Fulton for access to data, and Mr P. De Deckker for an ostracod identification.
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Bentos


PART II

Management problems
5 Limnological problems in the management of Tasmanian water resources

P. A. Tyler

Introduction
Tasmania, that detached Eden of Australia, differs from other states in its equitable climate, lack of deserts, and greater ecological diversity within a small area. It is bountifully endowed with a commodity most rare in mainland states—a labyrinth of natural lakes of great beauty and sweetness of water (Tyler, 1974). Moreover, most lakes were formed during Pleistocene glaciation (Derbyshire et al., 1965; Derbyshire, 1966) so that their morphology differs from all but a few (Peterson, 1968) on the mainland. This, and the prevailing oligotrophy of those pristine, montane regions gives Tasmania a characteristic limnological aspect, sharply contrasting with the torrid, salt pan face of greater Australia.

Only in the Midlands, in the rainshadow of the Central Plateau, do we catch a glimpse of Australia's misfortune in the salt lagoons of Ross and Tunbridge (Buckney and Tyler, 1976), and for a kindred limnology we turn eastwards, across the Tasman Sea. Nor do we have serious problems of eutrophication, though slightly modified, natural eutrophic lakes do occur (Cheng and Tyler, 1973, 1976a,b). Like mainland states, however, Tasmania has many artificial water bodies formed by raising levels of pre-existing lakes or swamps, or by damming de novo. These range from the multitude of farm dams, which extend aquatic floras and faunas into previously untenanted areas, through small reservoirs built for municipal water supply, to the large lakes dammed for hydroelectric power production. It is in these, rather than in natural lakes, that limnological problems have developed or could develop, and in which biologists may be of assistance. This chapter provides examples of these problems and their biological implications.

Despite the wealth of lakes and rivers and the great interest of their fauna (Bayly and Williams, 1973), local academic institutions have, until recently, neglected aquatic ecology. Recent work has had notable result in the discovery of a new genus of syncarid crustacean (Swain et al., 1970, 1971), and the basic limnology of sufficient lakes of diverse type has now been investigated to predict the nature of many others (Tyler, 1974 and references; Buckney and Tyler, 1973; Croome and Tyler, 1975; Tyler and Buckney, 1974; Cheng and Tyler, 1973, 1976a,b; Buckney and Tyler, 1976; Buckney, 1976, 1977). However, that important aspect, the biota, is in much less satisfactory state. Major taxonomic hurdles remain and, probably, many new species await description. The problem is more acute with the invertebrate fauna, in which there is considerable endemism (Williams, 1968, 1974), than with the phytoplankton which tends to cosmopolitanism.

The principal Tasmanian agencies
managing natural lakes and constructing reservoirs are the Hydro-Electric Commission (HEC), the Rivers and Water Supply Commission, the Inland Fisheries Commission, the Metropolitan Water Board in Hobart, and various regional water bodies. Only the Inland Fisheries Commission employs biologists; other bodies have access to the advice and services of the Government Analyst. The Hydro-Electric Commission, in particular, conducts considerable limnological investigation, and is much the largest body concerned with damming and management of reservoirs. The average output of stations installed or under construction is 1073 MW, considerably more than the 445 MW of the more publicised Snowy Mountains scheme. Naturally, the hydroelectric schemes are located on rivers draining the Central Plateau, or in the south-west, where high rainfall occurs (Fig. 6). Over the last 25 years the Commission has investigated a number of problems, some of considerable economic consequence (McFie, 1973).

**Manganese problems**

A nagging problem is the occurrence of biogenic deposits inside the pipelines (Plate I). Though they increase to only a few millimetres maximum thickness inside pipelines, together with moss and

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**Fig. 6** Map showing location of major hydroelectric storages and power developments. The Pieman power development is under construction. Stippled area shows the maximum area of two alternative schemes under investigation for the Lower Gordon Power development, which has not yet been submitted to Parliament for approval. Actual areas flooded would be somewhat less than the stippled area.
algal growths on concrete canals they cause a 10 per cent loss of effective capacity, necessitating periodic manual cleaning with its attendant disruption of power production. For the four main power stations affected, this amounted by 1973 to a capitalised loss of $3 \times 10^6 (McFie, 1973).

The H.E.C. investigated this problem through the 1950s, and a joint submission with the University of Tasmania resulted in 1962 in the establishment of a Water Research Foundation of Australia Fellowship financed jointly by the Foundation, the Commission and the university.

At that time the hydroelectric network had five components, (1) waters of the upper Derwent from Lake St Clair through Lake King William and the power stations of Butlers Gorge and Tarraleah, (2) the Lake Echo and Upper Nive catchment developed through Lake Echo and Tungatinah, (3) the conjoined waters from (1) and (2) developed through Liapootah, Wayatinah and Catagunya, (4) the Great Lake catchment, developed southwards through Shannon and Waddamana stations (Figs. 7, 8), and (5) Trevallyn Power Station on the South Esk at Launceston.

It was clear that not all pipelines suffered to the same extent. Those carrying King William water, serving Butlers Gorge, Tarraleah, Liapootah and Wayatinah stations (Fig. 8) experienced rapid build-up of deposits in about six months after cleaning. The Waddamana system, on the other hand, had remained virtually free of deposit during forty years of operation. Tungatinah pipelines did develop deposits, but were less sig-

Plate I  Manganese deposits, with characteristic ripple pattern, inside a hydroelectric pipeline.
Fig. 7 Hydroelectric developments on Tasmania's Central Plateau in 1964. Waddamana and Shannon power stations have since been closed and further stations have been installed on the Derwent downstream from Catagunya. The positions of recent lakes Big Jim and Samuel are shown. P.S. = power station.

Fig. 8 Diagram showing occurrence of troublesome manganese deposits in hydroelectric pipelines in Tasmania, and the effects of diverting Arthurs Lake into Great Lake. P.S. = power station.
nificant economically. Therefore, early attention focused in comparative vein on Great Lake and King William waters, and the reasons for presence or absence of deposits.

A limnological explanation was sought in stratification and consequent build-up of manganese in anoxic hypolimnia and, to be sure, Great Lake was a shallow windswept expanse, unlikely to stratify, whereas Lake St Clair was glacially deepened and King William occupied a gorge with a bottom outlet from the dam. However, though Lake St Clair stratified thermally with no diminution in oxygen, Lake King William, like Great Lake, appeared not to stratify, and attention turned to surface water chemistry.

All three lakes have soft water with low total dissolved solid concentrations (< 50 mg l\(^{-1}\)), but whereas Great Lake water is colourless, St Clair and especially King William are slightly humic (5-15 Hazen\(^{*}\)). Preliminary observations suggested that this subtle difference is accompanied by differences in zooplankton and phytoplankton, and results from different surrounding vegetation. Great Lake is surrounded by Restio-Epacrid heathland, whose runoff is colourless, whereas King William and St Clair are bordered by temperate rainforest and button grass (\textit{Gymnoschoenus sphaerocephalos}) plains which give south-western Tasmania its dystrophic face (Buckney and Tyler, 1973). Perhaps an answer could be found in chelation of manganese by humic acids, so holding it in solution in an oxidising environment. Analyses by the inadequate techniques then available showed only that though more abundant in King William (10-70 \(\mu\)g l\(^{-1}\) Mn) than in Great Lake (trace—13 \(\mu\)g l\(^{-1}\) Mn), manganese was scarce in the feed water of both systems (Tyler and Marshall, 1967a). Therein lay the clue—the concentration may be low but much manganese in solution would pass a point over a period of time and if extracted could cause a build-up of deposits.

There were then two prevailing views on how this might happen; first, by chemical oxidation and, second, by microbiological agency, and there was considerable overseas experience of iron and manganese oxide deposits reputedly caused by 'iron bacteria'. Hindsight shows that the former is possible but the latter more likely. That micro-organisms were to blame in Tasmania was demonstrated by a simple apparatus (Tyler and Marshall, 1967a).

Lake water was recirculated for days through tubes (Plate II) containing microscope coverglasses. With King William water, manganese-rich deposits developed on the glasses (Plate III). The H.E.C. installed similar equipment at Tarraleah, drawing water from a pipeline, and deposits were produced on microscope slides (Plate IV). Autoclaving or poisoning the water with azide prevented deposition, but inoculation of the autoclaved system with fresh King William water initiated deposition, leaving no doubt that micro-organisms were involved. These techniques also indicated that Great Lake water had the micro-organisms but insufficient or unavailable manganese, for when manganese was added to the recirculating water deposits would form, so demonstrating Beijerinck's dictum \textit{Alles is overal maar het milieu selecteert}—everything is everywhere and the environment selects. The equipment was also used to predict the likelihood of deposits developing in new systems. Redesigning the Great Lake power
Plate II  An experimental recirculatory system in which bacterial manganese deposits were produced in the laboratory.
scheme called for water to be redirected northwards through Poatina (Fig. 8). By circulating mixtures of Great Lake and Arthurs Lakes waters it was predicted that the associated proposal to pump Arthurs Lakes water into Great Lake could lead to deposits forming in Poatina pipelines (Tyler, 1967). Inspections have shown this to be the case.

The identification of causative microorganisms required considerable effort. Almost all reports to date implicated either various 'iron bacteria', members of the taxonomically recalcitrant group of sheathed bacteria (Chlamydomonadales), or cocci and bacilli morphologically indistinguishable from a host of heterotrophic forms. Sheathed bacteria could not be responsible since they were absent or rare in the deposits, and for several months a fruitless search for a causative organism was mounted. Isolation techniques employing selective media revealed manganese-oxidising fungal colonies. The discovery that fungal oxidation was carried out by the enigmatic symbiont, *Metallogenium symbioticum*, of the fungus was an expensive, six month red herring, for fungal hyphae were not present in fresh deposit in sufficient density to account for the deposit.

The answer came in 1965 when Aristovskaya's (1961) paper appeared in the university library. The text was in Russian but the pictures were in English, and at once showed that the 'unidentified thread-like structures' common in fresh...
Tasmanian deposit were hyphae of a bizarre, manganese-oxidising budding bacterium, *Pedomicrobium*, isolated by Aristovskaya from Leningrad soil. Refined microscopical techniques revealed that fresh deposit was densely ramified by cells of this organism, connected by anastomosing networks of cellular hyphae whose diameter (0.2-0.3 μm) approached the limits of resolution of light microscopy (Plate V). New selective media were developed and soon the organism could be isolated at will, in large numbers, by plating ground fresh deposit. It could also be isolated from deposits formed in the recirculatory apparatus and, in fact, direct microscopy showed it to initiate the deposits (Tyler and Marshall, 1967a). In pure culture it displayed highly pleomorphic characteristics and could be regarded as either *Pedomicrobium* or its relative *Hyphomicrobium* (Tyler and Marshall, 1967b). There were no previous reports of budding bacteria being involved in pipeline deposits.

By now the Fellowship's course had run. It remained only to announce the findings to the water supply industry (Tyler and Marshall, 1967c) and argue with bacteriologists about the proper name of the organism (Bauld and Tyler, 1971). Subsequent investigation of pipeline deposits from many parts of the world showed considerable chemical uniformity and widespread occurrence...
of budding bacteria as causative agents. Fading sepias in dusty journals suggest that they may have been present in deposits long ago ascribed to 'iron bacteria' (Tyler, 1970).

While this was academically satisfying, the H.E.C. was still cleaning pipelines manually. Recognition of budding bacteria as culprits did little to alleviate the problem. However, while academics pondered diffuse double layers as a mechanism for attachment of bacteria to pipeline walls (Tyler and Marshall, 1967d), the H.E.C. tested linings with possible bacteriostatic action. This gave little encouragement, and the next approach was to apply intermittent chlorination to a clean pipeline. This had a firm scientific basis in that it should kill bacteria, preventing colonisation of pipeline walls. After a twelve month field trial (Plate VI) the H.E.C. believed that chlorination was a completely effective, inexpensive solution to the problem (McFie, 1973). However, it was not adopted partly because of possible deleterious action downstream, and cleaning is still performed mechanically. Nonetheless, McFie’s (1973) proposals for further testing with careful monitoring has considerable merit.

Water quality in large storages
A widespread problem encountered with deep reservoirs is deterioration of water quality consequent upon stratification. Hypolimnetic waters may become
Plate VI  Applied limnology in action—the Hydro-Electric Commission investigates control of manganese deposits by intermittent chlorination of pipelines.

anoxic and foul, with build up of reduced substances such as hydrogen sulphide, methane and soluble species of iron and manganese.

Recent hydroelectric developments in Tasmania have formed new, deep reservoirs. In the Mersey-Forth scheme (Fig. 6) the hypolimnion of Lake Rowallan, formed by flooding the well-forested Mersey Valley, contained considerable concentrations of iron, manganese and hydrogen sulphide soon after closure, and, when discharged through a low level offtake, coated the river bed with iron oxides for several kilometres. Hydrogen sulphide discharged to atmosphere could be smelled several kilometres downstream, and minor corrosion of power station equipment took place. Artificial destratification was considered (McFie, 1973) but not adopted.

The possible reduction in deleterious effects on hydraulic structures could not justify expensive destratification, particularly since water quality at downstream supply intakes remained adequate (H. McFie, personal communication).

Lake Barrington, formed by the 70 m Devils Gate dam, impounded the Forth River in a deep-V, densely vegetated valley. The dam was closed in July 1969 and by October the lake was almost full. The outlet is near the top. The lake was severely stratified, and over the bottom 6 m of water a temperature rise of 1.9°C occurred (Thomas 1976; Tyler and Buckney, 1974). This phenomenon is diagnostic of density-stabilised, stagnant layers as in meromictic lakes, and this was later confirmed by the pronounced drop of redox potential and oxygen con-
centrations over this stratum, accompanied by equally dramatic increases in many chemical species, including iron, manganese, and hydrogen sulphide (Fig. 9). However, this layer of dense water was not present throughout the length of the long reservoir. It extended as a pool for a few kilometres upstream of the dam face (Tyler and Buckney, 1974) and it seemed appropriate to regard the reservoir not as meromictic but as a warm monomictic lake in which a local monimolimnetic pool had accumulated in the deepest contours. It seems that damming forested valleys (Plate VII, see also Chapter 31, Plate XV), especially, but not only, if of deep-V section, virtually predisposes the resultant reservoir for such an event.

First, deep-V shapes concentrate debris into former valley floors so that even meagre epilimnetic production produces heavy sediment loads per unit volume of bottom water. Moreover, flooding dense forest ensures a heavy and immediate oxygen demand so that anoxia develops rapidly from microbiological oxidation of organic matter. Further, in deep-V reservoirs most of the total depth will be hypolimnetic and probably devoid of oxygen. In extreme cases the epilimnion may also have low oxygen concentrations. Under these conditions areas of contact between water and anoxic mud are large, and minerals mobilised in that reducing milieu cause miniature density currents to creep downslope and accumulate in the lowest strata (Fig. 10). Because iron is relatively easily mobilised (Mortimer, 1941, 1942) it is frequently the primary factor in rendering monimolimnia dense (Kjensmo, 1962, 1967, 1968; Meriläinen, 1970; Tyler and Buckney, 1974). Finally, the shelter of steep valley sides and the limited fetch of dendritic or sinuous reservoirs limits vertical turbulence so that at autumnal overturn the work available to erode the monimolimnion will be less than in large exposed lakes.
Plate VII: The densely forested Mersey Valley, Tasmania. This is the type of valley flooded to form Lake Barrington.
It will thus persist if not disturbed by low level offtakes or artificial aeration.

It is likely that the monimolimnetic pool in Lake Barrington formed in this biogenic way, perhaps aided initially by turbid density currents from construction upstream. In either case, maintenance of the pool relies on processes and conditions outlined above, and Lake Barrington is likely to be the pattern for forested valley reservoirs of the Pieman Scheme (Fig. 6) now under construction, all having high level offtakes and, depending on design, for reservoirs of the proposed Lower Gordon Scheme (Fig. 6) if approved. It is also the pattern which was predicted for Lake Gordon, Tasmania (Tyler and Buckney, 1974), and Dartmouth Reservoir, Victoria (Tyler, 1975). Data supplied by the Hydro-Electric Commission (Fig. 11) show that Lake Gordon did develop a pool of dense water, complete with hydrogen sulphide and a bottom temperature rise. Initially, the epilimnion also had a high oxygen demand. Dartmouth dam was sealed early in November 1977. Six weeks later the reservoir had developed a monimolimnetic pool with a temperature rise, high concentrations of \( \text{H}_2\text{S} \) in the hypolimnion, and severe oxygen depletion in the epilimnion (see Chapter 31). It is possible that formation of monimolimnetic conditions may be avoided or ameliorated if the reservoir begins filling in winter when low water temperatures retard biological oxidations and facilitate complete mixing. If the organic load is great, however, this strategy may simply delay partial mermixis until the following spring.

If the existence of a monimolimnetic pool relies on maintenance of highly reducing conditions, then as flooded vegetation is consumed and not replaced the oxygen demand will drop. Then, the severity of the chemocline may be eroded until eventually the pool is mixed with the rest of the lake at an overturn. This appears to be happening in Lake Barrington. A pronounced monimolimnion was present in Barrington from soon after sealing until July 1970 (Tyler and Buckney, 1974). It was destroyed by floods in August 1970 (Fig. 9) but had reformed by May 1971. It persisted through 1973 and 1974 but by 1976 it was displayed weakly (Fig. 11). In March 1978 oxygen was present right to the bottom and consequently hydrogen sulphide was absent. A slight temperature rise at the mud-water interface may be a remnant of the characteristic pool of warmer water, but chemical conditions
suggestion that it has been mixed with the hypolimnion. The dramatic changes across the chemoclines of previous years were absent, and the magnitude of increases in various chemical parameters (Fig. 12) in the bottom strata are no more than for many hypolimnia.

The events in Lake Barrington, Lake Gordon and Dartmouth Reservoir suggest a natural evolutionary sequence for new reservoirs of this type. The enormous oxygen demand of flooded forest leads to early anoxia and downslope migration of solutes to a pool of dense water. The pool is maintained if accretion of solutes exceeds losses, but as inundated forest is consumed the reservoir adjusts to an equilibrium state. Maintenance of the pool then relies on autochthonous production, and if this is low the monimolimnetic density gradient declines to the point where an overturn can destroy it. This notion accords entirely with the classical view of a peak of primary, secondary and tertiary productivity, followed by decline, in new reservoirs. Dams recently constructed, or presently planned, in south-eastern Australia offer a splendid opportunity for testing this scheme. It would require preparation of sample sites before flooding, intensive monitoring during the filling phase, and careful surveillance thereafter. This appears to be planned for Dartmouth dam.

Whether limnological phenomena influence dam design depends on proposed uses. Monimolimnetic pools behind dams need cause no problem and in fact may be beneficial in locking up undesirable substances. Creation of artificial monimolimnia for just that purpose
Limnological problems

Fig. 12 Variation with depth (measured from the bottom) of various chemical parameters in Lake Barrington. In A-D the solid line represents maximum values obtained from 1969 to 1973 (see Tyler and Buckney, 1974). The dotted line represents the values for March 1978. The figures contrast the dramatic increases in solute concentrations near the bottom while a monimolimnetic pool existed (1969-73 data), with the slight increases in 1978 when the pool no longer existed. In E redox values for 1978 are contrasted with those for specific dates during the existence of the pool. In August 1970 a flood destroyed the pool and redox values were high throughout the depth (cf. Fig. 11).

has been proposed (Lund, 1966). However, if bottom water is discharged then effects on downstream water quality may be significant, and local destratification near the hypolimnetic drawoff was planned for Dartmouth (Burns, 1977; see also Chapter 31).

For the H.E.C., engineering dictates far outweigh limnological considerations in dam design (H. McFie, personal com-
Management problems

Since limnological conditions are not critical for safety or structural life of the dam, the level of the offtake is determined entirely by such factors as operating range of the storage and the need to avoid vortices. Nonetheless, the certainty of anoxic, aggressive water influences choice of materials, stainless steel being substituted for other metals and high quality concrete being specified. These precautions and use of proven protective paints avoid costs in the range $0.5-1.0 \times 10^6$ for multi-level outlets (H. McFie, personal communication).

Municipal water supplies

In Hobart, management of domestic and industrial supplies has no biological basis. Two principal sources are Lake Fenton, a montane reservoir on Mt Field, and small reservoirs fed by the Mt Wellington Range behind the city. In addition water abstracted from the Derwent River above New Norfolk is treated and pumped to storages. In most cases, water quality is so high that the Metropolitan Water Board need do little more than distribute it. Only at Risdon Brook Reservoir, serving eastern suburbs, have slight problems arisen. Pumped water suffered deterioration on storage consequent upon thermal stratification. The reservoir stratifies annually for about six months (Tyler, 1974) and latterly the hypolimnion becomes anoxic, with iron, manganese and hydrogen sulphide in the water. Undesirable levels of colour and turbidity occasionally result from intrusion of local water during flash storms which overflow the peripheral, diversionary leat. The board carried out limited monitoring and commissioned the university to study water quality so that water was withdrawn from most favourable levels. The modest problems encountered with Risdon Brook will disappear when water is no longer stored for long periods. In summer, constant draw down and constant replenishment will introduce high quality treated water at any appropriate level to ameliorate water quality conditions. The abundance of excellent water, and lack of biological problems, make Hobart a favoured city compared with such places as Adelaide.

Water supplies beyond Hobart are concerns of municipal or city councils, or of the Rivers and Water Supply Commission. No clear picture of biological problems is available. Most suppliers operate elementary treatment plants for flocculation, filtration and chlorination. Neither the commission nor the councils employ biologists and it is assumed that problems are minimal, occasional trouble being investigated via the Government Analyst.

Large tracts of Crown land are controlled by the Forestry Commission or the Lands Department. While neither organisation carries out research in aquatic ecology, the need to protect water quality influences management practices in catchments. Thus the value of pristine catchments was a cardinal consideration of the Lands Department's Central Plateau Management Plan, and presence of a water supply feeder may dictate modified logging practices.

Pollution of water resources

Despite a short history of occupation by industrialised man, and a small population (c. 400,000), Tasmania has not escaped abuse of its water resources. The Tamar and Derwent estuaries, on
which stand the two major cities, receive pollutants ranging from barely-treated sewage to industrial effluents. Nor have inland waterways remained unscathed. The King River, on the west coast, receives a legendary torrent of metaliferous particles and solutes from mining operations at Queenstown (Senate Select Committee on Water Pollution, 1970). Other west coast rivers (Whyte, Savage and Pieman) are also affected by heavy metals. In the north-east, alluvial tin sluicing delivers a heavy load of turbidity to a number of rivers, and zinc or wolfram mines poison a stretch of the South Esk River (Tyler and Buckney, 1973; Thorpe and Lake, 1973).

Under the Environment Protection Act (1973), the Director of Environmental Control is empowered to impose conditions upon operations of scheduled premises. These include mines, quarries, manufacturing and processing industries. Emission standards have been promulgated and monitoring by the Department of the Environment is revealing cases where legislation needs to be enforced. Gradually the degree of pollution is being reduced as licence conditions are enforced and made progressively more stringent. Nonetheless, pollution from a number of mines is likely to continue for many years, even though mining may have ceased (Lake et al., 1977).

Inland fisheries
The Inland Fisheries Commission, receiving revenue from government and from licence fees of c. 30,000 anglers, promulgates and polices regulations concerning taking of fish. Lack of suitable native sporting fish prompted introductions of several salmonid species only two of which—brown trout and rainbow trout—were successful. From a hatchery in Tasmania they were introduced widely in Australasia to form recreational fisheries of high quality (see Chapter 23).

Trout are highly regarded, and have been liberated in remote lakes beyond the many colonised by natural spread. In the fastness of mountains there are still lakes with only native fish, or no fish, and there, endemic mountain shrimps, *Anaspides tasmaniae*, elsewhere mainly confined to small pools and creeks, are almost pelagic, hunting chaoborid larvae which abound in the dystrophic waters.

The commission has undertaken short-term studies tailored to specific needs of management and development of inland fisheries. These are reported in the commission’s annual reports. To counter demands for restocking, management principles deduced from Nicholls (1957, 1958a,b,c, 1961) are backed by specific investigations by electrofishing, by observations during spawning runs, by creel census or by tag and release methods. Hatcheries have played a minor role since the 1950s, but fish are transferred from overstocked lakes to those with inadequate spawning grounds. Sometimes, stocking with trout is prohibited to protect native fish such as blackfish, *Gadopsis tasmanicus*. Further, the commission has eradicated unwanted, illegal introductions such as European carp (*Cyprinus carpio*). The commission also has interest in water quality and, where routine analysis is not carried out by other agencies, undertakes surveillance of waters subject to pollution, using the services of the Government Analyst.

There are signs that the commission is now undertaking more fundamental research since increased governmental
funding has allowed appointment of two biologists. The results of this policy are already evident in recent publications (Fulton, 1978; McDowall and Fulton, 1978; Brinkhurst and Fulton, 1978).

**Biological conservation of aquatic ecosystems**

In 1961 the Inland Fisheries Commission engaged a biologist to carry out a specific piece of research. He was to determine the depth of flooding necessary to curtail growth of emergent vegetation of a marsh. Adoption of the recommendations of the report led to destruction of a unique piece of aquatic ecology as a deliberate management policy.

Lagoon of Islands, on the Central Plateau, was a lagoon of outstanding scientific interest, upon whose surface the phenomenon of vegetational succession was graphically enacted. The basin was overlain by interlacing rhizomes of two emergent reeds, *Baumea rubiginosa* and *Chorisandra cymbaria*, forming a floating carpet on which it was possible to walk, with care, over the 3 m of water below. Randomly distributed over the surface were islands, ranging from mere tussocks of the sedge *Carex appressa*, through tussocks colonised by a few bushes, to larger islands weighed down with a forest of *Leptospermum lanigerum* and *Callistemon viridiflorus* bushes, and even trees of *Eucalyptus* (Tyler, 1976a,b). As young, small islands grew by spread and colonisation of tussocks, so the accumulating mass ensured the island’s destruction from waterlogging as it progressively sank, depressing the mat and forming a moat (Plate VIII). The numerous islands at various stages of

![Plate VIII](image_url) Lagoon of Islands, Tasmania, in 1963. Bush covered islands are dotted on the floating reed mat overlying 3 metres of water. Larger islands depress the mat, forming a moat.
development (Plate IX) or decline recapitulated in space the temporal sequence of any one island.

Unfortunately, two commissions had designs on damming the lagoon. For the H.E.C. it represented riparian water to substitute for Great Lake water about to be sequestered northwards (Figs. 6, 8). At the time it was policy of the Inland Fisheries Commission to convert marshes into 'new waters' (Hobbs, 1961), allaying in the first fine flush of productivity the clamour of the angling fraternity for bigger and better fisheries. The two commissions shared costs of the dam built in 1964, and agreed on the minimum water level to be maintained in

the flooded lagoon. This level was that deemed, by the biologist engaged by the Inland Fisheries Commission for the purpose, necessary to limit growth of emergent vegetation (Hobbs, 1962). Not surprisingly, conservation efforts made after construction were of no avail, and the success of the management policy became evident as the system died beneath rising waters (Plate X). Suggestions that water levels be regulated empirically to ensure survival of the system yet still supply water were not accepted. The lagoon has become an excellent fishery.

Failure of biologists to champion Lagoon of Islands before the dam was

Plate IX Lagoon of Islands, Tasmania, in 1963—a young island formed of Carex appressa (light colour, in foreground) and Restio tetraphyllus (left) colonised by bushes of Leptospernum lanigerum. The reed mat and other islands are in the background.
constructed highlights the lack of a biologically-based conservation rationale or policy in the 1960s. To be sure, recommendations had been made to 'Project Aqua' (Luther and Rzoska, 1969, 1971) but these were largely *ad hoc* recommendations, as they had to be with the prevailing ignorance of aquatic ecosystems in Tasmania.

The loss of Lagoon of Islands was a sad one, but it went largely unsung. Pleadings were swamped by the bitter rancour which accompanied the flooding of a lake of incomparably greater beauty and of unknown scientific merit. That flooding, of Lake Pedder, sparked the greatest conservation debate in Tasmania, and had far-reaching repercussions throughout Australia and other parts of the world. The H.E.C. became the butt of widespread vituperation and criticism (Burton *et al.*, 1974).

With the mid-reaches of the Gordon River harnessed for power production, the H.E.C.'s attention turned to the Lower Gordon (Ashton, 1977). A number of alternative proposals for the Gordon, Olga, Franklin and King Rivers (Fig. 6) has recently been released. The area is part of the south-west wilderness, a sensitive conservation issue. Some of the sober criticism levelled at the H.E.C. over Lake Pedder was not directed so much at its activities but at the cavalier way an area of outstanding beauty had been flooded with only the most meagre effort to determine the scientific values of this unexplored and intractable area.

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*Plate X*  Lagoon of Islands, Tasmania, in 1974—open water and a dead island testify to the success of a biological investigation carried out to determine the depth of flooding necessary to destroy the reeds.
The same cannot be said of the H.E.C. 's present investigations in the Lower Gordon. The H.E.C. has instituted a scientific survey in which biologists, geologists and others have been invited to study, with heavy logistic support from the commission. The provision of helicopters has allowed people and equipment to reach areas otherwise accessible only by arduous expeditions. At the same time the H.E.C. is releasing information unstintingly to the South West Tasmania Resources Survey, an independent planning exercise financed by the Australian government. Whether the findings of these two surveys will in any way influence water resources management in the area remains to be seen, but the fact remains that they are being carried out prior to parliamentary debate and they promise to reveal considerably more about south-western Tasmania than has hitherto been known.

Concluding remarks
The foregoing review indicates that management and development of water resources in Tasmania, for whatever purpose, has a scant limnological basis. In part, this results from the small population and therefore small municipal undertakings, but in part it also results from the lack of need. Whereas the Engineering and Water Supply Department in South Australia employs a large team of scientists, probably because good quality natural water is lacking, and whereas in Sydney limnological surveillance improves the quality of water to the city (see Chapter 30), in Tasmania water is minimally treated and pumped. Likewise, the Inland Fisheries Commission has had little pressure placed upon it to carry out extensive investigations because there are dozens of lakes to fish and new ones ever forming in the wake of the H.E.C. In fact, there are so many

Fig. 13 Down among the sapropel—a biologist's view of the importance of biologists in water resources management hierarchy (see text).
trout-infested lakes that one marvels at the recent creation of Lakes Barber and Big Jim (Fig. 8), close to existing fisheries, by a private angling organisation. Lagoon of Islands was in part sacrificed upon the same altar.

Because of its scale of operations, the H.E.C. has encountered limnological problems, and investigations by its own engineers using the services of the Government Analyst have given insight into them and influenced design and materials. Events in their Lake Barrington are of great limnological interest and may be the pattern for a natural sequence of reservoir evolution. In its scientific survey of the Lower Gordon area, the H.E.C. has sought assistance outside its own ranks, and with its heavy logistic support will be the progenitor of the largest body of biological data yet assembled for the south-west.

That biologists' influence on water resource management to date has been slender is rather poignantly reflected in a cartoon devised by a colleague when asked why she took so long to reply to letters (Fig. 13). Perhaps the engagement of biologists by the H.E.C. and Inland Fisheries Commission, and the increasing recourse to consultation on environmental matters in mainland states (Croome et al., 1976), is a good omen. Whatever the case, it is to be hoped that it will always be that mainland states have more need of waterworks apothecaries than will Tasmania.

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References


6 Ecological considerations in the management of reservoir phytoplankton

George G. Ganf

Introduction
Water management policy must include an understanding of phytoplankton ecology because phytoplanktonic algae are major nuisance organisms in inland waters. Of particular relevance is the manner by which phytoplankton have overcome three problems inherent in living in the aquatic medium. First, there is the problem that phytoplankton cells are generally denser than water; second, that the aquatic habitat is a shade environment; and third, that water masses are continually replaced, with a replacement time which may be as short as a few days.

The first problem resolves itself into an understanding of the significance of morphology, physiology and the nature of water movement in combating the laws of gravity; in a word, suspension (Lund, 1959; Hutchinson, 1967; Smayda, 1970; Fogg, 1975; Smith, 1975). The occurrence of water-blooms and dense (> 100 µg chl-1) phytoplankton populations is a measure of how well these minute organisms have overcome the problem of suspension.

The second problem—how to exploit a shade habitat successfully—overlaps with the first. It is also of special importance in Australia where the majority of inland waters are very turbid (Kirk, 1976a, 1977; Ganf, 1976), and the balance between photosynthesis and respiration is often precarious (Ferris, 1977). The practical implication of this precarious balance may be used to the advantage of the water manager. Water impoundments should be so constructed as to produce an unfavourable habitat for the growth of phytoplankton by ensuring that the depth of the illuminated zone is small compared with the mixed depth (Talling, 1971).

The third problem is of special significance in water bodies where the retention time is short (0-5-6 weeks). To combat the continual loss via outflow, phytoplankton are characterised by fast (2-10 days) doubling times. Any process which increases either retention time or growth rate (e.g. favourable nutrient concentration and mixed depth) should be minimised if dense phytoplankton crops are to be avoided.

This brief account of phytoplankton ecology will discuss specific points of algal ecology and physiology which bear upon these problems. Furthermore, suggestions will be made about how a knowledge of these problems can be turned to the benefit of water management.

Suspension
Hutchinson (1970) claimed that the solution to the problem of suspension facing phytoplankton communities was equivalent to that faced by terrestrial plants during their invasion of land. Obviously, in order to grow, phytoplankton populations must reside within the euphotic zone for a period sufficient to overcome
Management problems

(via photosynthesis) the losses occurring due to respiration, sedimentation, washout and grazing. Since the constituents of phytoplankton cells are heavier than water, a buoyancy mechanism is implied. This buoyancy may be provided either by some morphological or physiological adaptation or by water movement.

**Morphology.** As a starting point the relationship between cell size and sinking rates may be considered. It is described by Stokes Law,

\[ V = \frac{2}{9} g r^2 \left( \rho' - \rho \right) \eta^{-1} \]

where \( V \) is terminal velocity; \( g \), the gravitational constant; \( r \) the radius of a spherical cell; \( \rho' \), the density of a cell; \( \rho \), the density of the bathing medium; and \( \eta \), the viscosity of the liquid. From this equation it may be concluded that sinking rates are proportional to cell size and excess density \( (\rho' - \rho) \), but inversely proportional to viscosity. Commonly, phytoplankton cells have bizarre shapes, this departure from the spherical shape increasing the surface/volume ratio, and thereby increasing the frictional drag so that the cell sinks more slowly.

Experimental evidence tends to confirm that cells of approximately 100\( \mu \) diameter sink at rates between one and ten metres per day. Smaller cells (5-50\( \mu \)) sink more slowly (-2-1 m d\(^{-1}\)). An excellent review of this subject is given by Smayda (1970). However, there is a number of anomalies. For instance, Smayda and Boleyn (1965) have demonstrated that the sinking rates of *Skeletonema costatum* cells decrease with increasing size. As a general rule, sinking rates in the vicinity of 2 m d\(^{-1}\) for non-motile phytoplankton in non-turbulent water appear to be reasonable. However, it must be noted that very little quantitative fieldwork has been done on this subject, and none in Australia. For the calculation of silting rates and nutrient budgets of water systems this information is essential and attempts should be made to close this gap in our knowledge of phytoplankton ecology in Australia.

The possession of flagella by dinoflagellates (e.g. *Ceratium*) and other planktonic algae renders them motile. They are capable of quite astonishing vertical migrations of a million times their cell length per day—the equivalent of a man travelling 200 km d\(^{-1}\). The significance of these rapid vertical migrations in the selection of outlet sites on dam walls is obvious.

Blue-green algae (e.g. *Microcystis*) possess gas-vacuoles. The walls of these vesicles contain protein bands which when erected form a space within the protoplast. This space fills with gas in equilibrium with the gases in the cell solution (Walsby, 1972), thus acting as an internal buoyancy mechanism. Reynolds (1972) has demonstrated how bloom formation of the blue-green alga *Microcystis* is a direct result of redistribution within the water column. Deep-living populations rise to the surface under the influence of their gas vacuoles, and form surface scums. A similar phenomenon is thought to operate in Lake Burrunjuck, N.S.W. (Stella Humphries, personal communication), and undoubtedly occurs in Mount Bold Reservoir, South Australia. Because of these rapid flotation rates (1m min\(^{-1}\); Ganf, 1976), blooms may appear overnight, but equally may disperse as quickly as they form when their gas-vacuoles collapse and cells lose buoyancy. As well as the loss of buoyancy by natural means...
Ecological considerations

(Reynolds and Walsby, 1975), sharp increases in hydrostatic pressure also cause gas-vacuoles to collapse, and this may prove a practical method for the rapid dispersement of blue-green algal blooms.

**Physiology.** Much has been written on the adaptive physiological mechanisms of cells to counteract sinking. However, little that has been written is of direct relevance to water management; Smayda (1970) has again provided an excellent review. One mechanism, the accumulation of fat by *Botryococcus braunii*, however, has recently attracted considerable interest in Australia. This alga contains 30-40 per cent of its dry weight in the form of lipoids, and large-scale, commercial production of it could provide fuel in the future. Unfortunately the organism is not easy to culture.

**Water movement.** Buoyancy generated by water movement is probably the most important single factor for the suspension of phytoplankton. Unfortunately it is also the least researched. The classical work of Stommel (1949) and Munk and Riley (1952) remains amongst the few serious attempts to investigate this matter. Smith (1975) has pointed out that the conditions for turbulent flow are normally satisfied in unstratified lakes with a depth more than 10 m and a mean horizontal water velocity of 10 cm s⁻¹. For stratified lakes the interaction between laminar and turbulent flow may be expressed in terms of Richardson's Number ($R_i$)

$$R_i = g \left( \frac{1}{\rho} \frac{\delta \rho}{\delta \eta} \right) \left( \frac{\delta \eta}{\delta \eta} \right)^{-2}$$

where $g$ is the gravitational constant; $\rho$, the density of water, $\delta \rho/\delta \eta$ the change of water density with depth and $\delta \eta/\delta \eta$ the change of horizontal water velocity with depth. Numerical values of $R_i$ of less than 1 indicate turbulent flow. Under these conditions it is likely that the magnitude of up- and down-welling water velocities would negate any independent plankton movement. Smayda (1970) in his examination of Langmuir circulation concluded that turbulent flow is the normal condition in both lakes and the sea where horizontal water velocities of more than 2 cm s⁻¹ occur. Since these rates of movement are far in excess of those measured for independent plankton movement it seems likely that water movements are primarily responsible for the suspension of plankton. P. A. Tyler (personal communication) has made the interesting observation that only motile phytoplankton inhabit a deeply stratified lake in Tasmania which is sheltered from the wind and therefore is unlikely to have turbulent flow.

Water movement, and therefore the distribution and suspension of non-motile phytoplankton, is of prime importance to water management policy. In many reservoir systems, algal blooms develop in the upper reaches (e.g. Malpas Dam; Weir, 1976) and are subsequently swept down towards the outlet. Frequently, localised accumulations occur in sheltered bays where water exchange with the main body is minimal, so increasing localised retention times and allowing significant growth of the bloom. Ganf (1976) has explained how artificially increasing the mixing depth can significantly reduce the density of algal communities. Before reservoir construction, attempts should be made to predict the pattern of water movement. Reservoirs should then be constructed and sited so as to minimise the number of sheltered bays, and avoid the
occurrence of shallow areas. The overall objective of reservoir planning should be
to provide an unfavourable underwater climate for phytoplankton growth.

The shade habitat
One way of illustrating that the aquatic environment is a shade habitat is to
compare how far one can see on land and in water. This comparison may be more
effectively made by the use of extinction coefficients, which are derived from the
exponential decrease in light intensity with increasing depth and turbidity
according to the Beer Lambert Law:

\[ I_z = I_o \cdot e^{-ez} \]

where \( I_z \) is the intensity (I) at depth \( z \) metres; \( I_o \), the light intensity at the surface,
and \( e \), the extinction coefficient, which can readily be determined as the
slope of the line relating the \( \ln I \) to depth. The greater the value of \( e \) the
greater the turbidity of the water. As a starting point, the extinction coefficient
may be subdivided into that portion due to algal material \( (e_a b) \) and that due to
non-algal material \( (e q) \). In Australian waters, \( e q \) constitutes a large percentage
of the total extinction, indicating extremely turbid water, and causes a
marked reduction in the light available for photosynthesis (Kirk, 1976a; Ganf,
1976).

Intuitively, it is obvious that there
must be a relationship between increasing turbidity and decreasing phytoplankton
density. Providing the water column is homogeneously mixed, a quantitative
estimate of the maximum quantity of chlorophyll-\( a \) (an index of phytoplankton biomass) that may be contained
within the euphotic zone, may be made. Taking as a starting point the Beer Lam-
bert Law for light penetration

\[ I_z = I_o \cdot e^{-ez} \]

the depth of the euphotic zone \( (z_{eu}) \) is
defined as the depth at which 1 per cent
of the surface irradiance remains. Therefore

\[ z_{eu} = \frac{1}{e} \ln \left( \frac{100}{1} \right) = \frac{4.61}{e} \]

Since \( e = e_q + e_a b \)

\[ z_{eu} = \frac{4.61}{\frac{e_q + e_a b}{4.61}} \]

The maximum chlorophyll-\( a \) content of
the euphotic zone \( (B_{eu}^{max}) \) will be achieved
when \( e_q \rightarrow 0 \) and \( e \rightarrow e_a b \) (the extinction
due to phytoplankton). The amount of
chlorophyll-\( a \) in the euphotic zone is
given by the product of the euphotic depth \( (z_{eu}) \) and the chlorophyll-\( a \) con-
centration \( (b, \text{ mg m}^{-3}) \).

\[ B_{eu} = z_{eu} \times b \]

Substituting for the condition when \( e \rightarrow e_a b \)

\[ B_{eu}^{max} = \frac{4.61}{e_a b} \times b = \frac{4.61}{e_a b} \]

Reasonable values of \( e_a \) for South Aus-
tralian waters range from 0.01-0.02 (Ganf,
these values, \( B_{eu}^{max} \) shows a range from
4.61 to 230 mg chl-\( a \) m\(^{-3}\). A theoretical
analysis for variations in \( e_a \) are given by
Kirk (1975a,b, 1976b). From Kirk’s dis-
cussion it is clear that small cells (nan-
oplankton) trap light more efficiently
than larger cells, and the tendency
towards small sized phytoplankton in the
turbid Australian waters may be an
adaptation to these adverse conditions.
A more complete analysis of the relation
between phytoplankton biomass and the
underwater light climate is given by Tal-
ling (1971), Steel (1973) and Ganf (1976).

The practical implications of the rela-
tionships between light penetration,
chlorophyll-a concentration, mixing depth and production are of interest to water management policy. In their simplest forms these relationships indicate that algal crops will decrease as the turbidity, mixing depth, ratio between mixed and euphotic depth, and respiration rates increase. Large-scale mixing devices—such as those used by Burns (1976) on Tarago Reservoir (see Chapter 31)—are a convenient and cheap method for artificially mixing water bodies; the results are dramatic and beneficial if the oxygen content of the water is increased and algal concentrations decreased. A recent review of techniques for reservoir destratification/aeration has been given by Tolland (1977).

Retention times
There are usually both riverine inflows and outflows to reservoirs and lakes. Both serve to dilute the concentration of algal material. In many natural water systems retention times are greatly in excess of doubling times of phytoplankton cells. However, in storage reservoirs retention times may be as low as a few days, and consequently greatly influence the concentration of algal material present. Phytoplankton loss via the outflow in homogeneously mixed water bodies is analogous to the wash-out that occurs in chemostat cultures. Unfortunately, most water bodies are not homogeneously mixed, and the direct application of chemostat theory to lake systems is dubious. Despite the lack of a sound theoretical basis it is obvious, nevertheless, that phytoplankton crop density may be controlled by inflow/outflow rates. Ferris (1977) working on Mt Bold Reservoir, South Australia, has concluded that the bloom of Ceratium hirundinella (approx. 250 mg chl-a m⁻³) which occurred during April-May 1977 was a result of terminating the inflow of water pumped from the River Murray. Before this date, River Murray water was pumped continuously from late in 1976 to March 1977 at a rate exceeding 60 Ml d⁻¹ and phytoplankton density remained low (less than 10 mg chl-a m⁻³) despite favourable temperatures and nutrient conditions. After cessation of pumping, flow rates fell to less than 15 Ml d⁻¹, and the phytoplankton population explosion followed. The demise of the bloom was correlated with both a general draw-down of water in the reservoir and the onset of autumnal rains, which increased flow rates to more than 100 Ml d⁻¹. Similarly M. C. Geddes (personal communication) has concluded that a decline in the population density of Tribonema sp. in Lake Alexandrina, South Australia, was the result of River Murray floodwaters.

For practical purposes, long retention times in shallow water systems should be avoided as they would allow maximum exploitation of the available nutrients by phytoplankton, giving rise to large algal densities.

Conclusions
In this review of algal ecology, emphasis has been placed on three problems: suspension, light and retention times. By intensifying any of those characteristics that contribute to these problems a degree of control over algal growth can be achieved without the necessity of using undesirable algicides and expensive filtration plants. Many of the suggestions made require implementation during the construction phase of water impoundments. Others require that during the siting of impoundments biological
as well as engineering and geological criteria be considered.

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7  Algal problems in water supplies

I. C. Smalls

Introduction
Microscopic algae are the major primary producers in reservoirs; the only macroscopic algae present are the stoneworts (Charales) and these play a much less significant role in production. Two sorts of microscopic algae are involved: the phytoplankton or free-floating forms, and those generally attached to submerged surfaces. A few algae have both free-floating and attached stages, e.g. the diatom *Achnanthes*. The water supply manager is normally only concerned with phytoplanktonic algae, which affect water quality in various ways, but attached algae in multipurpose reservoirs can sometimes lower the recreational value of water resources. Table 5 indicates some of the commonest algae recorded in south-east Australian reservoirs.

Biological studies of water supplies—especially with regard to the algal flora of reservoirs—are by no means recent phenomena. In Europe, hydrobiological studies during the nineteenth century included several on water supplies, and by 1900 various laboratories in the United States were undertaking regular monitoring of water supplies. Copper sulphate as an algicide was developed in the United States by Moore and Kellerman in 1904. In Australia, West (1909) provided an early and comprehensive report on the phytoplanktonic algae of Yan Yean Reservoir, a supply reservoir for Melbourne, and Playfair (1913) did likewise for waters supplying Sydney. Chamberlain (1948), based in Brisbane, broadly considered

<table>
<thead>
<tr>
<th>Phylum</th>
<th>Class</th>
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<th>Genus</th>
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<tbody>
<tr>
<td>Cyanophyta</td>
<td>Cyanophyceae</td>
<td>Chroococcales</td>
<td><em>Microcystis</em></td>
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<td></td>
<td></td>
<td>Nostocinales</td>
<td><em>Anabaena</em></td>
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<td></td>
<td>Chrysomonadales</td>
<td><em>Dinobryon, Ochromonas, Synura</em></td>
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<td></td>
<td></td>
<td>Centrales</td>
<td><em>Melosira, Cyclotella</em></td>
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<td></td>
<td>Volvocales</td>
<td><em>Chlamydomonas, Volvox</em></td>
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<td></td>
<td></td>
<td>Chlorococcales</td>
<td><em>Ankistrodesmus, Keratococcus, Oocystis</em></td>
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<td>Conjugatophyceae</td>
<td>Chaeotophorales</td>
<td><em>Stigeoclonium</em></td>
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<td>Oedogonales</td>
<td><em>Oedogonium</em></td>
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<td>Desmidiales</td>
<td><em>Cosmarium, Staurastrum, Closterium</em></td>
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<td>Zygymemales</td>
<td><em>Zygnema, Spirogyra</em></td>
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<tr>
<td>Euglenophyta</td>
<td>Euglenophyceae</td>
<td>Euglenales</td>
<td><em>Euglena</em></td>
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<tr>
<td>Pyrrophyta</td>
<td>Dinophyceae</td>
<td>Peridinales</td>
<td><em>Ceratium, Peridinium</em></td>
</tr>
<tr>
<td>Cryptophyta</td>
<td>Cryptophyceae</td>
<td>Cryptomonadales</td>
<td><em>Cryptomonas, Chroomonas</em></td>
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</table>
the effects of algae on water supply. Currently, most major Australian cities regularly monitor water supplies, a clear indication of the extent to which biological organisms in supplies are viewed as a matter for concern. This chapter discusses one sort of biological organism involved, namely the microscopic algae. The emphasis is upon the nature of problems arising from algal growths. Australian examples are largely drawn from the author's first-hand experience with the water supply system for Sydney.

The nature of algal problems
The development of algae in water supplies may have a number of deleterious effects. Most of these have been reviewed recently by Mackenthun and Keup (1970) and Smalls (1973). The following discussion documents the major effects.

pH modification. The source of carbon for algal photosynthesis is usually free carbon dioxide (although bicarbonate and carbonate ions and dissolved organic compounds may sometimes serve as a source—Goldman et al., 1972). A side effect of the removal of carbon dioxide is the elevation of pH, and this may cause problems. Thus, Collingwood (1971) noted that an algal-induced pH of 9.5 was the probable cause of problems in a water treatment plant; the high pH led to a breakdown of the alum sludge blanket in a sedimentation tank and this in turn released previously trapped algae. Problems are also caused by algal-induced fluctuations in pH. Under darkness, algal photosynthesis stops and respiration becomes more important so releasing carbon dioxide and depressing pH values. Such pH fluctuations are obviously disquieting to the water treatment chemist and engineer, especially where processes such as coagulation and sedimentation requiring precise pH control are involved.

Colour and turbidity. When algae are present in high numbers they cause increased turbidity in water supplies. An example is provided by events which took place in Prospect Reservoir, a major water supply for Sydney, in November 1969; at that time, a total cell count of 17,139 cells ml\(^{-1}\) was recorded for the diatom *Achnanthes*, and, concurrently, turbidity values (SiO\(_2\) scale) increased from 2 to 7. Excessive numbers of algae constitute so-called 'blooms', but exactly what algal numbers are excessive is open to debate. Kilkus et al. (1975) used a cell count of 500 cells ml\(^{-1}\) as indicative of bloom conditions on the basis of a literature survey, but, as Vollenweider (1968) pointed out, this may represent a cell volume of 0.01 mm\(^3\) l\(^{-1}\) for small green algae or 40 mm\(^3\) l\(^{-1}\) for a large one such as *Ceratium*. If an algal count of 500 cells ml\(^{-1}\) is accepted, most reservoirs near Sydney have permanent algal blooms. A more realistic approach is to adopt Vollenweider's (1968) value of 10 mm\(^3\) l\(^{-1}\) for algal cell volume as indicative of bloom conditions, though noting that in some cases bloom conditions (as a surface scum) may occur at smaller algal volumes. Typical algal cell volumes for Prospect Reservoir are about 1.5 mm\(^3\) l\(^{-1}\), but maxima reach 4.5 mm\(^3\) l\(^{-1}\). These values equate with 3 mg chl-\(a\) m\(^{-3}\) for typical algal biomasses and 9 mg chl-\(a\) m\(^{-3}\) for maximal biomasses. Vollenweider's criterion suggests that a chlorophyll-\(a\) level of 20 mg m\(^{-3}\) is the bloom level criterion for Prospect.
Reservoir. However, bloom conditions with attendant aesthetic unacceptability are largely a matter of subjective decision, and whereas 10 mg chl-a m$^{-3}$ derived from the blue-green alga *Anabaena* may provide an aesthetically displeasing scum, 50 mg chl-a m$^{-3}$ derived from the diatom *Cyclotella* may go unnoticed except by the trained observer and the treatment plant operator for whom large biomasses of *Cyclotella* may cause trouble. Finally, it may be noted that algal growths only rarely give rise to dissolved colouring matter—as distinct from colours imparted by particulate material—although algal decomposition products will increase coloured organic acid levels.

**Screen and filter clogging.** If present in sufficient numbers, algae will clog or block water treatment screens or filters. The extent to which they do this also depends on the type of algae involved; amongst the most notorious in this respect are the diatoms *Melosira*, *Asterionella* and *Synedra*. On the other hand, the more liable an alga is to block filters, the easier it is to remove from supplies by treatment. Many examples of cases where algae have blocked filters may be given, for virtually all filtration works drawing water from algal-rich sources experience drops in filter runs. Several have been given by Mackenthun (1969), Mackenthun and Keup (1970) and Palmer (1959) including, notably, one relating to a treatment plant in Washington in which *Synedra* at a density of 4,800 cells ml$^{-1}$ cut filter runs down to 4.5 hours.

**Taste and odour.** The most common algal problem for water supply authorities concerns unpleasant tastes and odours. Both Palmer (1959) and Mackenthun (1969) have discussed this matter at length and listed algae responsible for specific tastes and odours. Broadly, the odoriferous materials of algal origin are essential oils, although decomposition products may also be important. A committee of the American Water Works Association (1970) implicated geosmin as a principal compound involved. Major taste and odour problems encountered in Sydney's water supplies have been caused by the growth of *Anabaena* in Prospect Reservoir (Cannon *et al.*, 1970); large numbers of complaints arose when cell numbers of this alga exceeded 1,000 ml$^{-1}$. Active management of Prospect Reservoir now controls the problem. It may be noted, in passing, that the Australian practice of showering rather than bathing presents an efficient way of monitoring the presence of odoriferous essential oils from algae in water supplies.

**Industrial and domestic process problems.** Several problems may arise in particular industrial and domestic processes as a result of the presence of algae in water used in the process. Thus, household and industrial filters may be clogged. Household washing may be stained (it appears that the spin-rinse cycle of certain washing machines is highly successful in centrifuging out particulate matter in water supplied onto clothes being washed). Diatoms in water supplies cause difficulties to the electroplating industry and in the manufacture of television tubes. Soft drink manufacture and film processing may experience problems of a like sort.

**Variation in quality.** The concentrations of algae in raw water supplies are never constant, but vary according to environmental conditions. To the water
supply industry this presents the problem of dealing with a product of variable quality. This, in itself, may evince consumer complaints; generally, consumers prefer a constant quality, even if the quality is low, and changes, even when from poor to good, may bring complaints.

**Toxins.** That toxins are produced by algae has been known for many years. Interestingly, the first record of the phenomenon referred to an Australian example; Francis (1878) found that a number of animal (stock) deaths were associated with a blue-green algal bloom at Lake Alexandrina, South Australia. More recent Australian examples are provided by May (1970, 1972) and D. Garman (personal communication). May (1970) gave some information pertinent to New South Wales, and recorded (1972) the history of a specific case at Braidwood and its control. D. Garman (personal communication) noted the death in 1977 of 100 sheep and 6 cattle at Carcoar Reservoir, central New South Wales, following imbibition of surface scum from water with a bloom of blue-green algae. Algal toxins have also been implicated in several mass kills of fish in Australia. Medical aspects of algal toxins have been dealt with adequately by Schwimmer and Schwimmer (1968), though with a general, not Australian, emphasis. These authors noted that known cases of algal poisoning arose from the use of waters other than those specifically meant for drinking. However, of recent note is the case in New England (New South Wales) in which a blue-green algal bloom was associated with a severe outbreak of gastro-enteritis in areas served by an 'infected' reservoir. More data on sub-

acute effects of algal toxins are clearly required.

**Organic load in distribution systems.** Most of the organic load in water supplies from reservoirs consists initially of live algae. When these reach the distribution system they die because of the absence of light and the presence of chlorine (if the supply is chlorinated). This may cause a variety of problems. The decomposition of the algae may release products which discolour water supplies; actinomycetes (branching, filamentous ‘bacteria’) may develop on the dead algal cells; iron and later sulphur bacteria may appear and cause problems; and the water may tend to anaerobic conditions following algal decomposition. Problems of this type are most likely to occur in quiescent areas of supply systems such as in the dead end regions of water mains. Additionally, it may be noted that with increasing algal loads chlorine demand can increase and the ability of the system to maintain a residual amount of chlorine can decrease, and troublesome animal growths within pipes can be stimulated. Smalls (1973) has discussed the sorts of animal found in water supplies.

**Algal growths on waterworks structures and corrosion problems.** Algal growths on the sides of water treatment basins and channels may sometimes give rise to an aesthetically displeasing appearance. Ordinarily this is of no moment, but can give rise to comment if the area is visited by consumers. Occasionally such growths can cause the pitting of concrete and the corrosion of iron and steel structures (Palmer, 1959).
**Reservoir chemistry.** When algae die they sink from the upper layers (epilimnion) of reservoirs to the lower ones (hypolimnion) where their decomposition brings about oxygen depletion. Depletion may be so intense that anaerobic conditions ultimately occur with the result that ferrous iron and manganous salts are released from sediments together with plant nutrients, particularly phosphorus (see Chapter 9). Furthermore, in anaerobic hypolimnion, sulphates are reduced to sulphides and the resultant hydrogen sulphide (plus ammonia from reduced nitrate) can give rise to malodorous conditions (Lund, 1966). Oxygen depletion is usually accompanied by an increase in carbon dioxide concentrations, too, with the result that pH values are lowered. Thus, hypolimnial waters in which considerable algal decomposition has occurred frequently provide an undesirable source for water supplies.

**Biological growths associated with algae.** The development of algal blooms may stimulate the subsequent development of other organisms of which some are a nuisance *per se*. An example is provided by Silver's (1963) observation that a growth of actinomycetes following a bloom of blue-green algae caused taste and odour problems in water supplies.

**Algal problems in impoundments other than those used for supplying water.** Algal problems are not, of course, restricted to waters used as sources of supply. In recreational waters, algae may give rise to unaesthetic conditions when present in unsightly masses or smelling unpleasantly. The development of thick algal mats may hinder navigation, restrict flows, and interfere with recreational pursuits in many waters.

**Concluding remarks**

Lest it be thought from the above comments that the effects of algae in water supply reservoirs are entirely disadvantageous, it should be remarked that the presence of algae also conveys benefits. Algae are powerful oxygenating agents and this property aids the natural purification processes within reservoirs; indeed it is this ability which is the basic mechanism in the purifying action of waste stabilisation ponds (see Chapter 33). In certain waters with high pH levels, algae will precipitate significant amounts of calcium carbonate as bicarbonate is withdrawn for photosynthesis, and this causes a distinct drop in the total hardness of the water—a welcome feature for most water supply managers. Soluble phosphorus, as calcium phosphate, may also be precipitated in such circumstances and thus be made less readily available as a plant nutrient.

As to what are desirable standards (or criteria) for algal concentrations in water supplies, as much debate prevails on this subject as does on the subject of what algal densities constitute blooms (see above). The only certain fact is that levels which cause problems in water usage are undesirable. Ridley (1970) has suggested that pre-treatment levels below 3,000 cells ml⁻¹ should be aimed for, and the Thames Water Authority in Britain manages London's raw water supplies to maintain algal cell numbers below this value. West (1971) suggested that an algal biomass level (expressed as a chlorophyll-α concentration) of 1.0 mg m⁻³ after treatment was a reasonable standard, but values one tenth of this have also been proposed. In dealing with
water supplies for Sydney, knowledge of specific taste and odour problems associated with given algal species has enabled control levels to be set for different species. Thus, control action is initiated against *Anabaena* when the cell count for this alga exceeds 600 cells ml⁻¹.

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References


Introduction
In principle, aquatic vegetation should be managed in the context of the overall management of water resources in relation to their quantity, quality and biology. In practice, a pragmatic approach is generally adopted with some attention being paid to chemical aspects of water quality but with relatively little regard being given to the biology of a particular system, the nature of plant growth, and the ecological causes of problem situations. While the pragmatic approach (usually control with herbicides) leads to satisfactory short-term management, it is not appropriate, nor does it provide relevant experience when, for example, multipurpose water bodies have to be managed to promote the growth of certain native plants and/or to prevent the invasion of undesirable aliens. Also, regular applications of herbicides over long periods will alter the structure of aquatic environments—often with adverse consequences. Sometimes, however, there is no alternative to herbicides being used in a system which, without this action, would be more adversely affected, say by the explosive growth of an alien weed.

Ideally, the rational formulation of management programs for aquatic plants, whether short-term or long-term, should be ecologically based, take account of the purposes for which the water body is used, and be sufficiently flexible to allow for the application of new knowledge and techniques, or for changes in water use. In any event the impacts of the program should be monitored so that unforeseen adverse effects can be detected at an early stage. The type of information needed to fulfil these requirements will be briefly reviewed in this chapter.

Weed status and water use
Aquatic plants can both enhance and interfere with different forms of water use in Australia. Native species are important components of Australian ecosystems. They contribute to primary production and play an essential role in many food chains. Their uptake and release of oxygen, carbon dioxide, and nutrient chemicals have both macro- and micro-climatic effects on the characteristics of aquatic ecosystems and in this way they diversify the habitat. They also provide a substrate for communities of microscopic organisms and shelter for larger animals. Rooted aquatic plants stabilise sediments and decrease turbidity. Many communities of aquatic vegetation improve water quality by decreasing bacterial counts and by absorbing pollutants, such as cadmium, phenol, and excessive nutrients (Seidel, 1971; Wolverton et al., 1976). Against these benefits, aquatic plants interfere with the flow of water in irrigation systems and flood mitigation drains (Plate XI), impede boat movement, promote siltation, interfere with recreation, compete
Management problems

with crops, such as rice, impede the access of stock to water, degrade water quality when they decay, interfere with floodgates, locks and weirs, and promote the spread of diseases, such as those borne by mosquitoes. Thus a plant such as cumbungi (Typha spp.) can, on the one hand, provide an important wildfowl refuge on the margins of lakes and in swamps and, on the other, interfere with flow in channels and drains. The weed status of a native plant therefore depends on careful evaluation of all the uses made of the particular water resource in which it is growing, with the emphasis being placed on showing why the plants are undesirable. Alien plants, by contrast, often replace native species, can disturb the structure and function of an ecosystem, and seldom provide a favoured habitat for native animals. In these cases an assessment of a plant’s status as a weed should place emphasis on showing why it should not be controlled.

The problems caused by aquatic weeds in Australian inland waters has been reviewed recently (Mitchell, 1978), and it was conservatively estimated that the cost of control by government and local authorities of water weeds in this country amounted to approximately 1½-2 million dollars annually. Total economic losses will amount to substantially more than this and would have to include the cost of control of weeds in rice (more than 1 million dollars) and losses due to uncontrolled weeds. The cost is likely to become higher in the future as problems increase with increasing demands for water.

Ecological knowledge required for weed management

Two basic causes of water weed problems can be identified: the introduction

Plate XI  Potamogeton tricarinatus F. Muell. & A. Benn. ex A. Benn. impeding flow in an irrigation supply channel near Griffith, N.S.W. (Photographed by D. S. Mitchell; November 1977.)
Aquatic weeds

and spread of alien plants, such as water hyacinth (*Eichhornia crassipes*) and the modification of water resources by nutrient enrichment or the construction of facilities, such as farm dams and irrigation systems. These latter provide permanent water often with a stable level, where before water was transitory or fluctuated rapidly and extensively. Recognition of these factors is essential for the planning of preventive measures against weed problems. Furthermore, most of the plants causing water weed problems can be regarded ecologically as pioneer or opportunist species that are adapted to colonise areas which are largely or entirely free of plant growth. Most short-term weed control programs that are successful in removing the plants therefore generate the conditions the plants are best adapted to exploit. A long-term control program should thus include features, in addition to the control techniques themselves, to render the environment less hospitable for the weed species and/or to prevent its spread or reinvasion. This cannot be done unless the structure and function of the affected ecosystem is understood, at least in broad outline, and the ecology of the plant or plants concerned is known. Consequently, the control of undesirable water plants (or the promotion of desirable ones) demands a knowledge of the mechanisms of reproduction, dispersal, and perennation of the species concerned, as well as of the nature of their requirements for ecological factors such as substrate, nutrients, temperature, and light. Of paramount importance in the last category is a knowledge of the factors, or combination of factors, which promote rapid rates of plant growth. Plant populations are generally only a nuisance when they are excessive in size as a result of rapid growth over a relatively short period. A knowledge of the rates of growth that can be attained by a particular plant in certain conditions can be used to predict the course of a particular outbreak. Mitchell and Tur (1975) showed that salvinia (*Salvinia molesta*), for example, doubled every 3.5 days in laboratory culture at 24°C. Growth rate was increased at 30°C but at 35°C was reduced to a doubling time of the order of 50 days. Low temperatures will also decrease growth rates. Nutrient concentrations and light are other factors that affected growth rates. There was a marked response to nitrate-nitrogen (Mitchell, 1970) but recent unpublished work has shown that the plant responds more slowly to phosphorus deficiencies or additions because it is capable of luxury uptake and storage of this nutrient. In Lake Kariba, Central Africa, where these conditions were continuously variable (although water temperatures were seldom below 20°C), salvinia doubled every 8 days in early summer and every 14 days in winter to spring. The plant can be expected to grow at the same rates in similar conditions in Australia. Musil and Breen (1977) recently applied growth kinetic studies of water hyacinth under differing conditions of supply of nitrate-nitrogen to the development of a predictive model of growth for that weed. The model could be used to estimate the frequency of harvesting of the plant in certain situations in order to control its excessive growth, thereby also removing the nutrients which often contribute markedly to the problem. Salvinia and water hyacinth are free floating plants and are therefore adversely affected by wind, waves and currents. Large populations generally occur in calm sheltered habitats where these
Management problems

factors are minimal (Plate XII). Bottom substrate conditions have little direct effect on their growth. This is obviously not so, however, for rooted plants, such as elodea (*Elodea canadensis*), or cum-bungi. For example Brown (1975) noticed that the former prefers silty rather than sandy substrates and that, as the depth of the silt layer increases, the height of the plant increases, although water depth and clarity also affect plant height.

Knowledge of the reproductive biology of a weed is clearly necessary for its management and it is surprising how often that is deficient in important aspects. In the case of water hyacinth, for example, more work is needed on initiation of flowering, requirements for successful formation of seeds, and seed viability, even though this plant has been called the world’s worst water weed and has been extensively studied in other respects (see Pieterse, 1974). The situation can be exemplified by the conflicting statements which appear in the literature in respect of the length of time for which seeds remain viable: a factor of essential importance in the long-term management of the weed. This period is often given as 15 years, presumably based on the observation of Matthews (1967), who noticed that seedling germination had occurred annually for up to 15 years at a number of sites in New Zealand ‘even though no plant has been allowed to flower once an infestation is discovered’. However, the flowering spikes are short-lived and it seems possible that the plants could have flowered and not been seen. In comparison, Pieterse’s (1974) review of

*Plate XII* Salvinia molesta D. S. Mitchell covering the surface of a sheltered reservoir on the Ross River near Townsville, Queensland. The reservoir formed part of the water supply for the City of Townsville at that time. Grasses are colonising the weed mat to form floating islands. (Photographed by D. S. Mitchell: November 1975.)
Aquatic weeds

water hyacinth refers to Das (1969) who in turn cited Biswas and Calder (1954) in stating that the seeds remain viable for 5 to 7 years. The period could be even shorter in certain conditions and most seeds seem to germinate once requirements for this are fulfilled.

The formulation of weed management programs

The procedure for formulating a program to manage a particular water weed problem typically falls into six phases:

(1) Identification of the weed and assessment of the seriousness of the problem in terms of the water use.

(2) Evaluation of alternative methods for handling the problem in terms of their effectiveness and cost (both economic and environmental). One alternative that should be evaluated is that of taking no action.

(3) Selection of the most appropriate control method or combination of methods.

(4) Implementation of program.

(5) Evaluation of the success of the program and of its impact on the environment by means of short-term and long-term monitoring procedures.

(6) Reassessment of the problem leading to possible modification or alteration of control measures and the establishment of a flexible long-term program of management.

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Fig. 14 Flow diagram for water-weed management, showing inputs to the various stages.
This sequence can be depicted as a flow diagram which requires certain inputs to each stage (Fig. 14). Critically important parts of the process occur in those stages where decisions have to be made. These call for the careful objective presentation of all relevant information to the decision-maker in a concise comprehensible form. Several procedures could be used here. For example, Mitchell (1978) has outlined the way in which cost-effectiveness analysis could be employed to choose between alternative strategies in water weed control; and the New South Wales State Pollution and Control Commission (1974) have documented principles and procedures for environmental impact assessment in New South Wales. The various inputs to the flow diagram illustrate the range of information and expertise that is required for weed management. Naturally, the magnitude of the problem will determine the extent to which any of the inputs are relevant or useful. It is significant that ecological and biological information is required for four of the six phases. Texts such as those by Aston (1973) and Sainty (1973) are useful for the identification of Australian aquatic plants, and that by Bayly and Williams (1973) for general ecological information. However, in most cases it is essential that people with expertise and experience are consulted as well.

The management of water weeds in Australia

There are three main categories of methods used for controlling unwanted vegetation: chemical (Burkhalter et al., 1973; Weed Science Society of America, 1974; Audus, 1976; Graham, 1976; Swarbrick, 1976); mechanical (Robson, 1974); and biological (Andres and Bennett, 1975; Harley, 1977). In addition, environmental factors can be manipulated to make the habitat inhospitable for the weeds, and preventive measures taken to minimise the risk of spread of a noxious plant (Mitchell, 1974). There is a growing tendency for several of the above methods to be combined into an integrated control program (Fryer and Matsunaka, 1977).

Also, increasing interest is being shown in the utilisation of weed plants for various purposes (Boyd, 1974; National Academy of Sciences, U.S.A., 1976). The most widely employed method in Australia is chemical control (Bill and Graham, 1970). This replaced the more traditional mechanical methods which had become prohibitively expensive as labour costs rose. Herbicides are generally effective though they can be expensive. Three main problems arise from their use: persistent chemicals inhibit the re-use of treated water; non-target organisms are often affected; and the decay of killed vegetation can degrade water quality. These problems can be avoided or minimised if the full effect of the chemical is known. Bartley and Gangstad (1974), Blackburn (1974) and Newbold (1975) have briefly listed the effects of the more commonly used herbicides, while Brooker and Edwards (1975) have outlined the biological and ecological consequences of applying herbicides to aquatic ecosystems. The effects of herbicides in Australian waters have been discussed by Bill and Graham (1970), Bill (1971), and Dunk (1974), and valuable studies of the behaviour of herbicides in irrigation systems have been carried out by Bowmer and O'Loughlin and their colleagues (O'Loughlin, 1975; O'Loughlin and Bowmer, 1975; Bowmer
Aquatic weeds


Recently, increasing attention has been paid to the biological control of aquatic weeds. This generally calls for the introduction of an organism (usually an insect) to feed specifically on the unwanted species of plant. Success has been achieved with this method in the control of the water weed, alligator weed (*Alternanthera philoxeroides*). Considerable work has also been carried out in the control of water hyacinth and control organisms have been released in Australia and the United States after food specificity criteria had been satisfied (K. L. S. Harley and I. W. Forno, personal communication). Research in this field in Australia by K. L. S. Harley and his colleagues now includes studies on the control of three notorious weeds: alligator weed, water hyacinth and salvinia.

The greatest shortcoming in the present management of aquatic weeds in Australia is the inadequacy of measures being taken to prevent the spread of alien aquatic plants. While it is recognised that this is difficult and in the end may not succeed, more effort must be made to acquaint Australians with the problems caused by such plants and to enforce legislation against selling, transporting and cultivating declared noxious weeds.

The most serious constraint to improving the management of aquatic vegetation in Australia is lack of basic ecological knowledge about the systems, the plants concerned, and their inter-relationships in this country. It is significant that much of the information of this nature given in this chapter has come from work done outside Australia, although the notorious alien weeds that have provided the main examples mostly behave in a similar way throughout the world. Nevertheless there are important characteristics of Australian aquatic ecosystems that are unique (see Chapter 2) and the detailed response of the plants to these may differ in a significant way from their responses elsewhere. It is in these fields that knowledge must be synthesised and research carried out and applied in current control practices if aquatic plants in Australia are to be managed for maximum benefit.

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Managing urban lakes

P. Cullen and R. S. Rosich

Introduction
The value of water in the urban landscape has long been appreciated by planners and urban residents, and in a dry country like Australia it is hardly surprising that ponds, fountains and lakes have been built in many cities. Lake Burley Griffin in Canberra is one example of a lake designed specifically as an integral landscape feature of the urban environment, and further lakes are planned for this city including one in the new development of Tuggeranong. It is worthwhile noting the summary by McCoy (1976) of the role that planners see for this new lake: the lake is seen as providing:

(i) a strong visual element creating a macro-setting for the Town Centre and spatially linking nearby suburbs;
(ii) a means of collecting urban runoff that is preferable to the more conventional concrete channels; and
(iii) a metropolitan recreational water feature in a dry inland city that needs all the water sports it can get.

Another function of urban lakes is in water pollution control. Urban runoff carries significant amounts of sediment, nutrients and organic material (e.g. Loehr, 1974), though this fact has been recognised only relatively recently. Its quality will improve if the water is impounded before release downstream.

But there are problems. Excessive growth of water plants is a major one.

The occurrence of this problem in Albert Park Lake, Melbourne, was reported as early as 1954, and similar problems have been reported in Lake Wendouree at Ballarat (Williams, 1969) and in Lake Burley Griffin at Canberra (Cullen, Rosich and Bek, 1977; see also Chapter 29). Weed problems were reported in Lake Burley Griffin only a few months after filling in 1964 and have recurred annually. In 1977-78 there were some 50 ha of weed beds, dominated by Potamogeton crispus and Vallisneria spiralis. Chemical control methods were used from 1965 until 1973, but have been replaced by mechanical cutting because the chemicals used were expensive and poisoned fish (Bill, 1969).

This chapter discusses the reasons for excess plant growth problems in lakes, outlines some processes operating within lakes with regard to phosphorus, a major plant nutrient, and considers various management strategies. The emphasis is upon urban lakes.

Sources of plant nutrients in urban lakes
Aquatic plant growth (both algal and macrophyte) is controlled primarily by light, temperature and nutrient availability. Of these, only nutrient availability can be readily manipulated. Plants require a wide range of nutrients to grow, but the macro-nutrients of phosphorus and nitrogen are most often the limiting ones. Both of them can enter urban lakes from point sources such as
sewage treatment works, garbage tips and industrial premises, or from diffuse non-point sources, mainly runoff.

A typical urban area has a variety of surfaces contributing nutrients to runoff. Impervious ones such as roofs, footpaths and paved areas around houses collect nutrient material from atmospheric fallout, animal droppings, litter, and car-washings (as detergent). Streets and car parks, in addition, receive material from vehicles. Vegetated areas (house gardens, nature strips, parks) receive substantial amounts of fertilisers that may later enter drains. Vegetation itself directly contributes nutrients when leaves and grass enter drains. An intermittent but probably significant additional source of nutrients is the overflow from sewerage systems in very wet times. These systems are designed to overflow into stormwater drains once sewer design capacity is exceeded, but even when it is not, they frequently contribute to drainage flows because they leak. Finally, nutrients associated with sediments enter drains from soil erosion, vehicular deposition, atmospheric fallout, and construction sites.

The sum effect is that large concentrations of plant nutrients (as well as organic and suspended matter) are present in urban runoff. Some recent reviews of this subject are those by Loehr (1974) and Cullen (1976) (see also Chapter 10). Of particular studies, specific mention may usefully be made of that by Kluesner and Lee (1974). These authors studied runoff quality in Madison, Wisconsin, and found that nutrient (and suspended solid) concentrations were usually highest during the early stages of a runoff event, and that rainfall appeared the major source of inorganic-N; phosphorus generally came from plant litter and possibly vehicle exhausts. They reported an annual export of 1.1 kg ha\(^{-1}\) of phosphorus, a value comparable to that of 0.91 kg ha\(^{-1}\) reported for an old established urban area of Canberra (nearby rural areas export only a third of this amount) (Cullen, Rosich and Bek, 1977).

**Eutrophication**

The information outlined above indicates that even if there are no point sources of nutrients to urban lakes, the composition of non-point drainage is such that high inputs of plant nutrients are received. If, then, light and temperature are adequate, excessive aquatic plant growth follows. Plants most often involved are macrophytes, large plants, which trap debris and block recreational access, but microscopic plants, algae, are frequently involved too. When abundant, that is in 'bloom' conditions, algae may cause water discolouration, formation of toxicants, and odour problems.

The phenomenon of excessive increase in plant growth is broadly referred to as eutrophication, but it is important to distinguish two sorts, natural and cultural. Natural eutrophication is not a result of increased nutrient loading; normally no sustained change in the catchment that would lead to increased export of nutrients occurs during the lifetime of the lake. Rather it is the result of a reduction in lake depth through sediment build-up. This allows more sediment phosphorus to be recycled in the water column and hence increases productivity. Cultural eutrophication, on the other hand, results from increases in lake nutrient levels caused by increased nutrient export from the catchment through altered land use, especially for
agricultural purposes, and from the input of nutrients in sewage, industrial effluents and urban runoff. The subject of eutrophication with particular regard to Australian waters has been reviewed recently by Wood (1975) and Cullen, Rosich and Bek (1977).

The dynamics of phosphorus in lakes
Many nutrients are required for aquatic plant growth. Most are present in waters which drain into lakes, but some, such as carbon and nitrogen, can also enter from the atmosphere. Of nutrients, phosphorus is the one which most frequently limits plant growth; phosphorus concentrations become critical before concentrations of other nutrients do. However, whether limiting or not, phosphorus is certainly the nutrient most amenable to control. Golterman (1975) has suggested that there is only one feasible method of controlling excessive plant growth in lakes, and that is to remove phosphorus from waters entering lakes: ‘It is not important whether phosphorus is currently the limiting factor or not, or even that it has ever been so. It is the only essential element that can easily be made to limit algal growth.’ Because of this significance, it is important to consider in more detail the dynamics of phosphorus in lakes.

Figure 15 indicates schematically the major pathways of phosphorus in a lake. Noteworthy is the overall net movement of phosphorus to the sediments so that phosphorus outflow is less than inflow (in Lake Burley Griffin, for example, depending on climatic conditions 12 to 91 per cent of phosphorus entering the lake is retained according to Cullen, Rosich and Bek, 1977). Phosphorus enters the system in either dissolved or particulate form. Hitherto, dissolved forms were regarded as perhaps the most significant, but it is now known that particulate forms may also be important. Thus, again considering Lake Burley Griffin, Cullen, Rosich and Bek (1977) found that during an 18 month period 64 per cent of the phosphorus entering the lake was in particulate form.

Once in a lake, the various forms of phosphorus are interchangeable both within the water column and the sediments (Fig. 16). Of particular importance is the fact that while phosphorus can be lost from the water column by precipitation and the settling of plant material, it can also enter the water column by release from sediments. That large amounts of phosphorus can be so released if the overlying water column becomes anaerobic is commonly accepted, but recent work by Lean et al.,
Managing urban lakes

Fig. 16 Interchange of phosphorus between various compartments in the sediment and water of a lake. Solid arrows indicate flux between compartments without change in the chemical form; broken arrows indicate flux involving a change in chemical form.

(1975) and Lean and Nalewajko (1976) has shown high release rates (up to 18 mg m\(^{-2}\) day\(^{-1}\)) even under aerobic conditions. In a situation where there was not a net release of sediment phosphorus, Lean et al. reported that up to 7-7 mg m\(^{-2}\) day\(^{-1}\) still passed each way across the sediment-water interface. It seems therefore that the internal loading of phosphorus to the lake from sediments may be as great or greater than the loading from external sources. This has major implications if attempts are to be made to divert nutrients from the lake, or to provide treatment of urban runoff; a reduction in the external sources of phosphorus may have a relatively minor effect if internal sources are large, or can compensate for reduced external input. In Lake Burley Griffin, Cullen, Rosich and Bek (1977) reported that 42-52 per cent of the sediment phosphorus could be utilised by algae.

The biological compartments within a lake also play an active role in phosphorus dynamics. Phosphorus is extracted from the water column and sediments and is subsequently recycled back into the water as cells break down, or into the sediments as organic debris accumulates. The exchange of phosphorus between living algal cells and the water column is very rapid (a matter of minutes), and of course cellular phosphorus is released when cells die. This rapid recycling is one of the reasons why many control measures such as spraying or cutting have such a short-lived effect unless weeds are removed. Macrophytes also release large amounts of phosphorus to the water column and sediments as their leaves die off. However, masses of macrophyte material apparently do not contain a large proportion of the phosphorus within the lake system. Cullen, Rosich and Bek (1977) have calculated that during 1976 the 20 ha of macrophytes in Lake Burley Griffin contained some 50 kg of phosphorus out of a total of some 2,000 kg within the lake system. Later results suggested that this may have been an underestimate, but it does
Management problems

appear that in the Lake Burley Griffin system the macrophyte compartment contains less than 8 per cent of the total amount of phosphorus within the system at any one time.

Management of urban lakes

In Hart's (1974) compilation of water quality criteria for Australia it is suggested that lakes designed for aesthetic or recreational purposes should be free of algal blooms and heavy growths of macrophytes. If not, aesthetic and recreational values are diminished, water flow may be impeded (with the possibility of causing flooding), and decaying aquatic weeds may give rise to unfavourable oxygen deficits. On the other hand, plants have some advantageous features; they act both as a physical filter for suspended debris (including organic matter) and as a nutrient sink. Lake management decisions, therefore, should carefully consider the ranking of the various possible management objectives, namely, the use of the lake for aesthetic purposes, and for recreation, wastewater purification, and flood mitigation. All things considered, however, the balance is generally against the presence of excess plant growth. It is appropriate, therefore, for the remainder of this chapter to consider management strategies to combat this particular problem. Such strategies involve reducing nutrient inflows, removing nutrients from lakes, manipulating growth factors other than nutrient concentrations, or reducing the impacts of excessive plant growth.

Strategies to reduce nutrient inflow

(i) Divert sewage effluents. Sewage effluent from conventional secondary sewage treatment works is the major point source of phosphorus and nitrogen to waterways in Australia. It may be possible simply to divert effluents around the lake to be protected. The most quoted example of the use of this strategy relates to Lake Washington, where some 99 per cent of secondary effluents have been diverted from the lake. This has led to a dramatic reduction in phosphorus concentration (to about 25 per cent of earlier levels), and a consequent reduction in plant growth (Edmondson, 1970; Sonzogni and Fitzgerald, 1975). This strategy, of course, simply transfers nutrients and the potential problem they may cause to some other location.

(ii) Treat sewage to a tertiary level. It is technologically possible to build sewage treatment plants which can remove phosphorus and nitrogen from sewage. One is being built in Canberra, the Lower Molonglo Water Quality Centre (National Capital Development Commission, 1974). But capital and running costs are high.

(iii) Remove phosphorus from detergents. Modern detergents generally have high concentrations of phosphorus. If these are banned or heavily taxed, people will substitute soaps or nitrogen-based detergents. Sze (1975) reported that when there was a total ban on detergents containing phosphorus in New York State, phosphorus concentrations in Onondaga Lake dropped to less than half previous levels with the result that the composition of the phytoplankton changed, green algae replacing the more troublesome blue-green algae.

(iv) Restrict urban development. Tuffey and Baker (1975) have described the tendency to overdevelop reservoir catchments with consequent deterioration in water quality. Planners have to
accept that when waste treatment facilities are overloaded continued urban development will be costly in terms of later reclamation of degraded water bodies.

(v) Reduce phosphorus inputs from raw sewage overflows. Most Australian cities have separate sewage and stormwater drainage systems. The sewerage system is normally designed to cope with somewhere near ten times normal dry weather flow. When the sewage flow exceeds this in wet weather because of pipe leakage and illegal connections of stormwater drains, the system becomes overloaded and raw sewage is discharged directly into receiving waters. Raw sewage overflow can be reduced by stopping illegal drainage connections and by renewing leaky pipe systems. Another approach is to discharge such overflows into holding basins rather than directly into receiving waters.

(vi) Reduce phosphorus inputs from urban runoff. It is possible to treat urban runoff before it enters receiving waters (Cordery, 1977). Shock loadings of nutrients and other pollutants in urban runoff can be reduced by improving the efficiency of street sweeping and hence reducing the build-up of contaminants. Sartor and Boyd (1972), however, showed that 56 per cent of phosphorus was transported on small particles (< 43 μm) which are not effectively removed by street sweepers.

(vii) Restrict the use of phosphate fertilisers. Most urban lakes are surrounded by parklands and superphosphate applied to them may, under some conditions, wash into the lakes (Cullen, 1974). Restricted use may decrease phosphorus loadings from this source.

(viii) Treat waters entering the lake. Where nutrient inflows are channelled into urban lakes it may be possible to treat them before they enter the lake. However, although conventional treatment units are available to remove phosphorus from water, these are costly and can only be expected to cope with dry weather flows. Biological methods of nutrient removal, on the other hand, do offer exciting prospects for cheap and effective purification of such inflows. Toth (1972) and Weir (1976) reported on the effects of passing inflow waters through a swamp before they entered a main lake, and showed that swamp plants not only acted as a physical filter for sediment but were also an effective sink for plant nutrients. This was mainly due to algal mats on the stalks of rooted plants; these acted like a trickling filter in a conventional sewage treatment plant. Whilst there are many hydrologic and design considerations still to be resolved this does appear a most promising approach and has been recommended for implementation at Lake Burley Griffin (Cullen, Rosich and Bek, 1977).

Strategies to remove nutrients from lakes. While the best long-term solution to the problem of eutrophication is to control nutrient inflow, this is not always possible, especially if much of it is from diffuse sources. Moreover, even if nutrient inflow is reduced, lake water quality may not improve quickly since lacustrine nutrients are effectively recycled and improvement will depend on losses from the lake. Despite this, techniques of lake restoration have been developed aimed at exporting nutrients from the system by physical removal or by isolation to prevent or minimise recycling.

(ix) Remove sediments. Removal of sediments will clearly remove nutrients.
Its effectiveness can be gauged by referring to work at Lake Trummen in Sweden. This lake was eutrophic and did not recover when waste water inflows were stopped. Investigations showed that sediments were supplying nutrients to the water column and so were removed by dredging. Material was pumped to settling ponds and the drained water was treated to remove phosphorus. This treatment led to a reduction of nutrients in the water and the replacement of nuisance algae with more desirable species (Björk, 1972).

(x) Harvest living organisms. Nutrients can be exported from a lake by harvesting algae, macrophytes or fish. However, algal harvesting is still in the development stages since the problem of separating algal cells from water on a large scale has not yet been overcome. To harvest macrophytes is easier, but harvesting is commonly done largely for cosmetic reasons. Nevertheless, if plant material is physically removed from the lake, then nutrients are exported. Peterson et al. (1974) reported on the effect of physically removing some 429,000 kg (wet weight) of plants from Lake Sallie in Minnesota; they showed that this removed only 100 kg of phosphorus from the lake, merely 1.4 per cent of total input. The removal of 57,200 kg of fish represented an export of 320 kg. Obviously harvesting techniques do not remove much available nutrient material.

(xi) Flush and dilute. If lakes with high nutrient and algal content can be flushed with ‘cleaner’ waters this will result in the discharge of nutrient and algal rich waters and the dilution of remaining waters. Oglesby (1969) reported that flushing Green Lake in Washington over five years suppressed nuisance blue-green algae.

(xii) Inactive nutrients. Attempts to inactivate nutrients involve leaving them in situ but making them unavailable to plants, precipitating them from the photic zone, or preventing their release. Higgins et al. (1976) reported on pilot studies using fly ash (the product of burning pulverised coal); they showed that it may remove phosphorus from the water column, lead to the sealing of sediments, and thus inhibit nutrient release. Sanville et al. (1976) described field trials using sodium aluminate to precipitate phosphorus from the water column. These trials indicated a reduction in phosphorus concentration and nuisance plant growth.

(xiii) Seal lake sediments. Another way to isolate phosphorus in sediments is to erect a physical barrier to release into the water column. Materials used include plastic or rubber sheets, sand layers, fly ash, clay, hydrous metal oxides and gels (see review by Dunst et al., 1974). Results are encouraging but questions remain about the permanency of such barriers and the ecological consequences of isolating the benthic community.

(xiv) Oxidise sediments. Ripl (1976) reported that the application of oxidising agents, e.g. nitrate salts, can oxidise surface sediments and so prevent the release of phosphorus to the water column.

Strategies involving manipulation of physical factors

(xv) Increase turbidity. Turbidity determines the depth to which light penetrates and hence the volume available for aquatic plant growth. An increase in turbidity will decrease plant
growth. Although turbidity is largely influenced by catchment landuse, it may, in certain situations, be amenable to manipulation. Some fish, such as European carp, may increase lake turbidity, and turbidity may also be influenced in shallow lakes by motor boats. Note, however, that the feeding of carp has been reported to convert sediment phosphorus to more readily available phosphorus (Lamarra, 1975).

(xvi) Deepen lakes. Many nuisance plants grow on lake bottoms. If the lake is made deep (or deepened) so that light does not penetrate to the bottom, these plants cannot grow. However, lakes become shallower as sediments and organic debris accumulate, and the proportion of the lake bottom shallow enough to support rooted plants increases with time unless dredging is carried out.

**Strategies reducing the impact of excessive plant growth**

(xvii) Harvest aquatic plants. Various weed cutters have been developed so that by regular cutting the impact of excessive macrophyte growth can be reduced. Ideally, cutting should be accompanied by removal of cut material: some nutrients are removed and oxygen depletion as plants decay is prevented. Removal is essential if the plants reproduce by vegetative means for otherwise each cut portion will grow and thus increase rather than reduce the problem.

(xviii) Control chemically. Certain chemicals can control algal blooms and macrophyte growth. However, they may have impacts on non-target species, though information on this matter is relatively scanty. Additionally, the death of plant cells from herbicide application leads to a rapid release of nutrients. These re-enter the water column and may stimulate further plant growth.

(xix) Control by biological means. Attempts are being made to find ‘predators’ and diseases to use against nuisance aquatic plants. The introduction or use of such biological controls should not be undertaken, however, until long-term and detailed studies of possible environmental impacts are complete. Ctenopharyngodon idella, the grass carp, is a herbivorous fish presently being investigated by New Zealand and many other countries in this capacity.

**Concluding remarks**

A thoroughgoing evaluation of management objectives for urban lakes requires a knowledge of the limnological characteristics of the lakes in question. In particular, detailed nutrient budgets are needed, nutrient sources require identification, and those factors limiting plant production need isolation. Once all this information is available, it is then possible properly to evaluate the various options and select the most appropriate for any given situation. The options should be evaluated in terms of their likely effectiveness in restoring the lake, the practicability of implementing them, and the environmental and socio-economic impacts arising from the implementation.

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10 Catchment management

W. D. Williams

Introduction
This chapter discusses correlations between land-use and water resources. It indicates the basic nature of the relationship. It illustrates the way in which catchment usage influences water resources. And it draws conclusions about catchment management. The approach is restricted; the main, though not exclusive emphasis is upon catchments as water producers.

The basic nature of the relationship between catchments and inland waters
Although early geologists and geographers recognised the importance of the drainage basin on water chemistry (Elster, 1974), the earliest limnologists tended to regard lake processes and metabolism as self-contained phenomena. The classic expression of this viewpoint is that by Forbes (1887) in his essay 'The lake as a microcosm': 'It (the lake) forms a little world within itself—a microcosm within which all the elemental forces are at work and the play of life goes on in full...'

No doubt it was the preoccupation of those limnologists who immediately succeeded Forbes with descriptive and comparative features of lakes that led to the survival of Forbes's 'microcosmic' view for so long. Eventually, of course, lakes became recognised as less independent of surrounding peripheral events than Forbes thought, and, as pointed out by Hutchinson (1964), Birge's (1915) studies on the heat budgets of lakes were among the first to draw attention to the lake as a natural system with an input and output. It is now abundantly apparent that lakes and their drainage systems and sediments function as parts of single systems. Eutrophication is an obvious example of a lake phenomenon which explicitly expresses the close causal relationship between what goes on in lake catchments and what goes on in lakes. Palaeolimnological studies provide further examples.

Though hydrological studies have long demonstrated the close physico-chemical links between running water systems and the terrestrial environment, it is only relatively recently that the close ecological links have been firmly demonstrated. It is also only relatively recently that stream ecologists have recognised the ecological implications of many hydrological, pedological and forestry studies: as Hynes (1975), with considerable courage and honesty, admitted: 'one could say that some of our most important recent discoveries (by stream ecologists) have been of the existence of hydrologists, foresters and soil scientists!' In any event, it is now known that, unlike most terrestrial and lake ecosystems, streams derive a substantial part of their energy requirements from allochthonous sources; that is, a good deal or most of the energy necessary for their operation is derived from outside...
the system. For the most part such systems are predominantly heterotrophic (e.g. Fisher and Likens, 1973; Hynes, 1970, 1975). But the relationship between the catchment and its streams goes much further; to quote Hynes (1975) again:

We may conclude that in every aspect the valley rules the stream. Its rock determines the availability of ions, its soil, its clay, even its slope. The soil and climate determine the vegetation, and the vegetation rules the supply of organic matter. The organic matter reacts with the soil to control the release of ions, and the ions, particularly nitrate and phosphate, control the decay of the litter, and hence lie at the root of the food cycle.

The influence of catchment usage
Four major catchment types may be recognised: forests, grasslands, crop-lands, and urban catchments.

Forested catchments. Our understanding of interactions in forested catchments is not complete, but the broad features are now known (Bougton, 1970; Pereira, 1973). In general, the clearing of forests and their replacement by more shallowly-rooted grassland vegetation usually results in an increase in water yield (and vice versa in grassland afforestation). Extensive clearing of riparian (riverbank) vegetation frequently has the same effect, as the large-scale removal of the tamarisk shrub or salt cedar (Tamarix pentandra) in south-western U.S.A. has clearly shown. Differences between the amounts of water used by different sorts of tree, however, remain more equivocal. Thus, if catchments are regarded primarily as sources of water, deforestation could be regarded as an advantage.

But, deforestation has many disadvantages; these more often than not amply counterbalance the benefits of extra water yields. An increase in the salinity of runoff is frequently one such disadvantage. Perhaps the most obvious local example of this is in south-western Western Australia where the increasing salinity of soils, streams and rivers over a large area in which forests have been replaced by crops is now the cause of concern. The explanation for the increased salinity is that the excess water resulting from deforestation has caused the saline groundwater table to rise (Bettenay, Blackmore and Hingston, 1964; Peck and Hurle, 1973). Salinities in some rivers of the area now even show a longitudinal salinity gradient the reverse of normal; their upper reaches are most saline, their lower ones least (Morrissy, 1974).

Deforestation may also result in loss of inorganic nutrient material. The work of Likens and collaborators in the Hubbard Brook Experimental Forest (a mixture of deciduous and coniferous species) in New Hampshire, U.S.A., provides good evidence of this phenomenon (Likens et al., 1967, 1970, 1971, 1975, 1977). Considering only nitrate and phosphorus outputs from forested and deforested catchments, Likens and co-workers found that the overall loss of phosphorus was about an order of magnitude greater as a result of deforestation (Hobbie and Likens, 1973), and the nitrate loss was even greater with concentrations in runoff water often in excess of recommended values for drinking water! The implications for nutrient balances in those forested catchments in Australia presently being clear-felled for
woodchips are clear, though it should be noted that the experimental procedures followed by Likens and co-workers do not really reflect current clear-felling practice. An Australian example of the effect of afforestation on the chemistry of runoff water is perhaps provided by Lakes Leake and Edward in South Australia. The waters of these adjacent lakes differ in their proportions of anions, with more sulphate in Lake Edward; Bayly and Williams (1966) explained the differences on the basis that a greater proportion of the catchment of Lake Edward is planted with conifers and these contribute more sulphate as a result of an intercept effect on atmospheric salt.

Yet another major disadvantage associated with deforestation is the disturbance to the natural runoff pattern. There is a great deal of evidence—including much from Australia—which indicates that catchment runoff is more rapid and less evenly spread in time following deforestation. Such effects are usually undesirable. The same effects often occur after forest fires (e.g. Melbourne and Metropolitan Board of Works, 1974); essentially these can be regarded as temporary deforestations. Floods are more frequent and intense; further, sediment transportation is considerably increased (that is, turbidity of runoff water increases). Road making, grazing, logging, and other human activities in a catchment have similar though more localised effects.

In summary, when catchments are forested, water yields are likely to be lower than from non-forested catchments, but runoff is usually less saline, contains fewer dissolved nutrients and suspended matter, is more even in discharge periodicity, and floods are less intense.

Grasslands. The way grassland catchments influence water resources varies greatly because of the importance of grassland management or mismanagement (Boughton, 1970). Of particular importance is the nature of grazing regimes, as has been shown by much Australian work. Thus, in central New South Wales, long-term experiments by the Soil Conservation Service of that state have shown that heavily grazed but improved pasture can grow five times as many sheep—with negligible storm-flow or erosion—as heavily grazed, but unimproved pasture; however, the latter yields some thirty times more water (unpublished data of Knowles and Scurlow quoted by Pereira, 1973).

One most important feature of grazing management relates to overgrazing. This is especially so in semi-arid regions. There, rainfall is usually not only small in quantity, but erratic in appearance, and, consequently, ‘good’ seasons more often than not result in overstocking vis-à-vis succeeding ‘bad’ seasons, with predictable results: erosion, sheet runoff with little soil infiltration, and degraded pasture. It might be added that the pathetic situation in which many thousands of African pastoral nomads now find themselves, and attributable as much to blind adherence to traditional cultural attitudes on grazing as to any secular climatic change (so-called desertification), has many parallels in more sophisticated societies. Ratcliffe (1963) provided some for South Australia. The history of the catchment of the Eppalock impoundment in Victoria before the rescue operation mounted by the Soil Conservation Authority of Victoria provides
others (Downes, 1961). In the Snowy Mountains of New South Wales it proved necessary to prohibit grazing altogether so that the appropriate amount of alpine vegetation could redevelop and maximise water yield (Costin, 1971).

In high rainfall areas, on the other hand, where a large water loss from the catchment may occur via transpiration from a vigorous grass cover, heavy grazing pressure—though not over-grazing—will probably increase water yields (though stimulated plant growth may partly offset this). Grazing of drained marshlands also increases water yields from such areas.

There are, of course, many factors involved in the relationship between grazing and catchment behaviour. Brief mention should be accorded the nature of the pasture, the number of stock watering points, and the prevalence of fire. However, it seems that there are few if any significant differences in effects caused by different species; sheep and cattle appear, in general, to have the same impact and it is absolute numbers that are the important criteria in determining the extent of the impact. With regard to the nature of the pasture, profound differences in yield may occur according to whether deep- or shallow-rooted perennial or annual grasses are involved. In dry regions, deep-rooted perennial grasses may, in fact, prevent any effective runoff. Inadequate numbers of stock watering places may cause local trampling and then erosion and flood flow. Erosion is also a major deleterious effect of fire, particularly in dry regions.

Other activities not directly concerned with grazing frequently take place on grassland catchments that may greatly influence the quality of catchment runoff. The spreading of animal wastes, the (over)use of fertilisers, and the application of biocides are among the more important. The construction of terraces and farm dams may have an important effect on quantity if not quality in local areas. None of these requires much further discussion here. Suffice it to say that the first two may contribute excess plant nutrients (especially nitrates and phosphates) to water-courses the end result of which may be manifested in the phenomenon of eutrophication in lakes or impoundments downstream—though it is not claimed that agricultural contributions are the only causes of eutrophication (see, for example, Vollenweider, 1968; Armstrong and Rohlich, 1970; Tomlinson, 1970; Cooke and Williams, 1970, 1973; White, 1972; McColl, White and Waugh, 1975). The effect of biocides is usually less obvious—and thereby the more insidious—but is no less important (e.g. see Rudd, 1964). Terrace construction, though often mitigating erosional loss, appears to have no significant effect on total yield; farm dams, on the other hand, if numerous and large, may decrease total yield.

**Croplands.** Discussion of the influence that the cultivation of arable crops has upon water resources falls tidily into a consideration of situations where crops are grown without and with irrigation. Where catchments include areas of non-irrigated crops, water yields are nearly always higher than would be the case were the same areas forested or grassed; though arable crops have seasonally heavy water demands, average demands are less than most types of natural vegetation. Logan's (1960) work at Wellington, N.S.W., amongst many
such experiments, neatly showed this; plots subject to a wheat-fallow or wheats-oats-fallow rotation gave runoffs >20 cm, whereas plots growing lucerne or retired from cultivation gave runoffs of <10 cm. However, associated with higher water yield is a lowered ability of cropped catchments to absorb rainfall without damage. Thus, without preventive measures, areas of dry-land arable cropping are frequently subject to undesirable flooding (and immediate loss of valuable water so far as the crop is concerned), erosion, and, concomitantly, degradation of the nutrient status of soils and decrease in runoff water quality because of high turbidities. Such damage is of critical importance in tropical developing countries. Indeed, its prevention there ‘remains one of the major large-scale problems facing mankind’ (Pereira, 1973). Damage is especially likely to occur in areas formerly covered by rain-forest, where the nutrient status of soils is fragile and easily degradable, and in low rainfall areas with erratic patterns of precipitation.

Many techniques for the mitigation of damage exist, including contour-tilling, terracing, the provision of grassed channels, crop rotation, ‘strip’ cropping and shallower cultivation (Webster and Wilson, 1966). American research, above all, has shown the marked improvement in catchment behaviour (and crop yields) which can occur following successful institution of techniques of this sort on damaged arable land. Lessons are hard learned, however, and even in the so-called developed countries, modern techniques of soil conservation are accepted by the farming community all too slowly. Pauli (1974) reported how attempts have been made since 1946 to encourage sugar-cane growers in Queensland to adopt soil conservation measures; yet in spite of lowered soil fertility following erosion and an alarming decline in crop productivity, only 9 per cent of landholders in one area (Isis District: Childers) have adopted soil conservation measures in any form and only 2 per cent of the total cane producing area has been protected.

The problems of catchments on which crops are grown under irrigation are of a different genre altogether. First there is the problem of excessive use of water, which, irrespective of locale (developed country or otherwise), is usually taken as being inefficient. Second, there is the problem of inadequate drainage. And third there is the related problem of increasing salinity.

Little further comment is offered on the first problem except to add that extremely large volumes of water indeed may be committed for purposes of irrigation. In Australia, the present annual commitment for consumptive irrigation use is of the order of $26,000 \times 10^6 \text{m}^3$ (Australian Water Resources Council, 1976). With regard to inadequate drainage, the problem is not so much one of waterlogging—though this may occur too—as elevation of the groundwater table to a level from where capillary action can move groundwater salts to the surface. There, evaporation brings about concentration, and thus salinisation. These problems are sharply illustrated by events now taking place in irrigated areas of the Murray Valley, the more sharply because the underlying groundwater is frequently highly saline. The problems are well-understood and well-documented (e.g. Gutteridge, Haskins and Davey, 1970), but solutions, nonetheless, are not easily arrived at within present economic and political
Catchment management

reality. A major difficulty is that the River Murray is the principal source of water for Adelaide, and this acts as a constraint upon the disposal of saline drainage water directly back to the Murray. The Engineering and Water Supply Department, the water supply authority for Adelaide, is presently investigating all available options to decrease at least the current rate of salinity increase in the lower reaches of the Murray (Engineering and Water Supply Department, 1977). The River Murray Commission is also mounting an investigation of options available. Present methods for control of salinity include the provision of evaporation basins for disposal of highly saline drainage water, and regulation of river flows so as to control water quality by means of dilution. Davidson's (1969, 1974) views concerning the uneconomic nature of irrigation in Australia, and the views of those who oppose them (e.g. Munro, 1969, 1974), are not considered appropriate for discussions within the present framework, though it is appropriate to mention them in passing.

Urban catchments. In a sense, many of the immediate hydrological impacts of urban development on catchments can be regarded as the continuation of trends evident in forested, grassed, and cropped catchments. Thus, compared with vegetated catchments, as a rule total yields are large, direct runoff (and its rate) is high, and annual peak discharges show marked elevation. Such characteristics are more or less predictable from even the briefest consideration of the nature of urban catchments; many paved roads and surfaces, extensive areas of roofing, large amounts of bare land, and—in Australia at least—the application of great volumes of imported water to gardens. Also involved in determining the nature of urban hydrology, but not quite as obviously so, is the fact that water in urban gutters and drains has higher velocities than it does in natural water-courses. In any event, these, under urban conditions, come to have smaller capacities for the disposal of runoff water because of their frequent enclosure within storm drains, silting, and the encroachment of urban development onto flood plain land. Aspects of urban hydrology have been discussed by Jens and McPherson (1964), McPherson (1975) and Douglas (1976) amongst many. Though a good deal of information—at least for developed countries apart from Australia—is now available, much of its collection, as pointed out by Cullen (1975), has basically been for civil engineering purposes rather than for catchment management objectives. Even for urban hydrology of that sort, however, much more research is required in Australia according to Aitken (1973). He noted that although some $50 million was spent annually on storm water drainage alone (this figure is for pre-1973 estimated values), Australian urban hydrological studies and models left much to be desired.

Closely associated with the quantitative aspect of urban runoff is the amount of sediment contained; those activities largely responsible for the special character of urban hydrology also include erosion, particularly in the initial phases of construction. A number of authors have pointed to the importance of road making in this respect, though the construction of new buildings is by no means unimportant as a source. In older urban areas, sources include soil and plant debris from gardens and other areas of vegetation, pavement materials,
Management problems

atmospheric fall-out, motor vehicles, litter and spilt material (Cullen, 1975). Values of suspended sediment in urban runoff, even in settled urban catchments, may actually exceed values for this parameter in raw sewage (~ 200 mg l⁻¹); for example, Weibel (1969) recorded 227 mg l⁻¹ from an area of Cincinnati, and S. Toshach and P. Cullen (personal communication) measured a peak value of 5,677 mg l⁻¹ from a newly developed urban area in Canberra. Factors involved in the determination of concentrations are principally the frequency of rain and street-sweeping, and the nature of traffic and urban development (Sartor and Boyd, 1972). Predictably, there is much variation in concentration with time, and intervals over which great variation may occur are often quite short—a matter of minutes. The work of Wood and Cullen (unpublished) on a small urban catchment in Canberra neatly illustrates this. Following a storm, they recorded a turbidity reading in runoff water which went from <20 to >150 NTU in the space of less than 30 minutes. This result accords with the findings of others (e.g. Weibel et al., 1966; Kluesner and Lee, 1974; Cordery, 1976) that suspended sediment concentrations are usually greatest in urban runoff during the early phase of a catchment event and decrease in time. It may be stressed, however, that the relationship between suspended sediment concentrations and discharge is by no means a simple one; it is, for example, complicated by the spatial distribution of rainfall as well as time (cf. Douglas, 1975).

Though sediment is the major contaminant in urban runoff in terms of mass (Sartor and Boyd, 1972), other important contaminants are present. Especially important are forms of phosphorus and nitrogen since both are significant plant nutrients. Values derived by Weibel (1969) from an urban catchment in Cincinnati were 3.1 mg l⁻¹ or 875 kg km⁻² yr⁻¹ for total nitrogen and 0.36 mg l⁻¹ or 105 kg km⁻² yr⁻¹ for total phosphorus. Much smaller and greater values have been recorded for one or other of these nutrients, including some recorded by Australian investigators (e.g. Melbourne and Metropolitan Board of Works, 1973; Campbell, 1975; Cordery, 1976, 1977; Cullen, Rosich and Bek, 1977). Care is required in the interpretation of such values, for some are based on dry weather flows and computed annual loads are clearly underestimates. The careful work of Cullen, Rosich and Bek (1977) on phosphorus may be quoted as reliable, however; they reported that the phosphorus output from an urban catchment in Canberra was 91 kg km⁻² yr⁻¹.

The important point about all nutrient values is that they exceed those usually found in natural waters, and as a result urban runoff may be a potent contributor to eutrophication problems in downstream lakes and impoundments. Moreover, many determinations of both concentration and mass per unit area and time underestimate the total contribution likely because the determinations usually neglect to take into sufficient account the large temporal variations which occur in the nutrient content of urban runoff. Similar neglect, incidentally, is a feature of many rural catchment studies, as the work of Holmes (1977) on nitrogen in catchments near Adelaide has shown. Like suspended sediment, nutrient concentrations in urban runoff may undergo extreme variation within short time intervals. The work of Wood and Cullen (unpublished)
in Canberra is again illustrative. After a storm, and during a single hour, concentrations of total phosphorus went from <0.1 to >0.6 and then to ~0.2 mg l\(^{-1}\). The published work of Cordery (1976) showed a similar pattern in runoff from an urban area of Sydney. Nutrient sources seem principally to be atmospheric (rain) for inorganic nitrogen, and terrestrial for phosphorus and other forms of nitrogen, though rainwater is also a substantial source of phosphorus according to Campbell (1975) and others.

Rather less is known about other chemical contaminants of urban runoff. There is no question, however, that some are important. Such contaminants include lead, mercury, and a variety of organic compounds (e.g. oil) as well as the major inorganic ions found in all natural waters. Traffic routes apparently contribute significantly to the total load (e.g. Söderlund, Lehtinen and Friberg, 1971). The dynamics of the major ions in an urban stream have recently been discussed by Caesar et al. (1976).

Finally, brief mention should be made of the fact that urban development contributes large numbers of coliform bacteria to runoff. Thus, Weibel (1969) reported that 90 per cent of storm-water samples examined from a sewered urban catchment in Cincinnati exceeded bacterial population counts of \(2.9 \times 10^9\) 100 ml\(^{-1}\). This greatly exceeds the limit on total coliforms for recreational waters in much of the United States (\(1 \times 10^8\) 100 ml\(^{-1}\)). Higher values have been recorded for some urban areas in Australia, particularly during short periods after storms.

A most useful (and comprehensive) summary of the impacts of land-use changes and modifications during urbanisation on hydrology and river systems has been given recently by Douglas (1976). The impacts are categorised as: modifications of land-cover or landform (e.g. removal of plant cover), construction (e.g. housing, airports), water supply, waste disposal, and channel modification. The type of effect each impact has on various parameters of water quality, quantity, and fluvial geomorphology is considered in terms of significance (major, minor), location (on site, downstream), and direction (positive, negative).

Other catchment activities of importance. The preceding sections have considered catchment events in terms of very broad regional usage. Also to be considered, however, are several 'cultural' phenomena which have a significant influence upon water resources but which are not specifically restricted to a given type of regional catchment. Little if any expansion of what was intended by the phrase 'catchment usage' is involved in order to fit a consideration of these phenomena into the plan of this chapter. Only brief comments are warranted, and only water pollution, mining, and drainage are considered.

The most important of these phenomena is of course water pollution sensu lato. It may be claimed with conviction that the use of water as a transport and purification mechanism for wastes is almost as important as its industrial and consumptive use. Indeed a significant proportion of water supplied to domestic—if not industrial—users is supplied so that it can serve as a waste transport mechanism (Chapter 15). Whatever the case, a large number of sewage treatment plants discharge their effluents to inland waters in Australia (Table 6). The present unsatisfactory situation in most Australian inland
waters, then, is that of a balance between the amount of ecological degradation suffered as a result of pollution, and the amount of assimilation and purification of wastes taking place subsequently. In many waters that balance was reported by the Senate Select Committee on Water Pollution (1970) to result in a resource of low quality. It is fortunate that no further pollution of consequence is caused to catchments by rain in Australia, as is the case in many Western European countries and North America; there, rain and snow are now increasingly characterised by acidity as a result of air pollution. The long-term ecological effects of this are not yet known, but are potentially very great (e.g. Likens, 1972; Likens et al., 1972).

Mining is another 'cultural' phenomenon that frequently affects water resources to a significant degree irrespective of type of catchment. It may do this not only by the addition of pollutants to water courses but also by altering the very nature of the catchment. Mining effects, moreover, may persist for many years after mining operations have ceased, and the costs of renovation (if possible) may be very high [the estimated capital cost of the remedial measures recommended for the prevention of further pollution of the Molonglo River above Canberra was over $2,500,000 (Joint Government Technical Committee on Mine Waste Pollution of the Molonglo River, 1974)].

With regard to drainage, the overall hydrological effect of draining wetlands is usually a gain in water yield from a catchment, but with a decreased regularity of flow and an increased liability for stormflow peaks. The simple removal of vegetation from wetlands to give a free-water surface was formerly thought to be another way of increasing water yield when complete drainage was either not possible or not desired, but recent work indicates that the opposite may apply; the shading effect of emergent macrophytes decreases evaporation (Linacre, Sainty and Grange, 1970). With regard to the influence wetlands have on the water quality of runoff, one important and desirable effect is to decrease suspended solids and plant nutrients in flow-through water. Thus, another effect of wetland drainage is usually to increase the turbidity of and the concentrations of plant nutrients in waters from the catchment involved (Lee, Bentley and Amundson, 1975). Lakes and impoundments have many additional effects on water quality (e.g.
Management

Many management conclusions will be apparent from the preceding section. This section aims merely to organise them into more explicit and briefer form. First, however, the question of catchment objectives needs consideration: clearly no strategy can be formulated properly until management objectives have been defined. For all catchments the ultimate aim is ecological stability (sensu Slatyer, 1970) since the long-term survival of man depends upon this. However, at a more immediately pragmatic level, seven penultimate objectives can be identified. For the four broad catchment types discussed above, in simple matrix form they are:

<table>
<thead>
<tr>
<th>Catchment type</th>
<th>Timber production</th>
<th>Recreation</th>
<th>Water supply</th>
<th>Conservation</th>
<th>Grazing</th>
<th>Crop production</th>
<th>Urban development</th>
</tr>
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<tbody>
<tr>
<td>Forested</td>
<td>+</td>
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<td>+</td>
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<tr>
<td>Grasslands</td>
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<td>Crops</td>
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<tr>
<td>Urban</td>
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</tbody>
</table>

For forested catchments, it is clear that all four objectives are most likely to be achieved by maintenance of the forest cover, prevention of wild fire, restriction (or prohibition) of grazing, and care with logging procedures and road making to minimise erosion (e.g. Lynch, Corbett and Hoopes, 1977). Since many water supply reservoirs are located in forested catchments it is appropriate to add here that denial of public access to these catchments and impoundments (supplying both irrigation, domestic and other needs) is a benefit to the water authority that under modern pressures for recreational facilities may be exceeded by the extent of the disadvantages to those seeking recreation (Clawson and Knetsch, 1967; Simmonds, 1975). Many Australian water supply authorities recognise this and permit controlled access to both catchment and reservoir, e.g. the Irrigation and Water Supply Commission of Queensland (Ross, 1974); others, e.g. the Melbourne and Metropolitan Board of Works, are much more protective. The argument advanced by such authorities is usually that their impoundments are mostly for domestic urban supplies, and prevention of possible contamination of water by visitors on the catchment is of paramount importance. Involved in this argument is the recognition that all ratepayers would have to pay for any extra treatment of supplies, that more accessible catchments would increase the risk of catchment degradation by fires, and that water supply treatment techniques remain imperfect (the removal of viruses from supplies is a particularly refractory problem). The application of systems analysis techniques to water resource management in Australia would be of much interest on this issue. Watt (1968), Walters (1975), and Coppock and Duffield (1975) have discussed such application in a general sense.

For grassland catchments, whatever the objective, it is clear that the priority strategy is for grazing and pasture management of high order. Prevention of overgrazing, fire protection, the provision of an adequate number of stock watering points, and judicious application of fertilisers, animal wastes and biocides are all important strategies.

For catchments growing crops without...
irrigation, major concern centres upon the prevention of erosion. As indicated, many techniques exist to do this. In the case of irrigated pastures, adequate drainage and the correct application of water to prevent salinisation are the major strategies involved. Suggestions advanced to minimise present difficulties in areas irrigated by the lower reaches of the River Murray are many. One of the most obvious, albeit most difficult to effect, is to decrease the area being irrigated.

With regard to urban catchments, a major strategy is the prevention of pollutants entering water courses. As much urban development as possible should be sewered with sewage treatment being taken to as high a level as is consistent with adequate environmental protection; further, erosion should be minimised (e.g. by performance controls on building sites), litter not allowed to accumulate, and streets cleaned regularly. Because of the special character of urban hydrology, urban development on floodplains should be carefully controlled, and more local hydrological knowledge sought. An especially difficult problem relates to stormwater runoff; this runoff is of diverse origin, highly variable in quality, contains significant concentrations of pollutants, and may be large in volume. One strategy for dealing with such water is to direct it into urban lakes with subsequent treatment in sewage works or simple release after an adequate diminution in unfavourable characteristics as a result of temporary retention (cf. Cordery, 1976). This strategy is being pursued in part by the National Capital Development Commission in Canberra (Higgins, Henkle and Lawrence, 1975). Finally, it may be added that strategies for managing urban water resources *sensu lato*—more than for any other sort of catchment—should involve serious input from disciplines, such as sociology, generally unrelated to hydrology. As McPherson (1975) put it: ‘in sum, urban hydrology necessarily confronts a higher ratio of art to science than traditional or classical hydrology’. A less flamboyant statement of Douglas (1976) made the same important point: ‘to decide on the best type of water management for a given urban area requires an evaluation, not only of the technical data, but also of the aspirations and perceptions of the wide variety of private, local and national interests involved’.

Finally, a brief general comment on the question of water pollution is appropriate. Although, in the short term, the best strategy involves preventive measures and better sewerage and sewage treatment, the best long-term strategy is to phase out natural waters as means of transporting wastes. This strategy certainly makes the greatest ecological sense, for not only would it affect in an advantageous way the quality of rivers, lakes and other natural waters, it would recirculate the high amounts of valuable material presently thrown to waste (Valentyne, 1973; Westman, 1974; Gutteridge, Haskins and Davey, 1976). And it is the only way that the ecological stability mentioned earlier and so important for the survival of man can be arrived at. This matter is considered further in Chapter 15.

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Management problems


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Management problems


PART III

Water usage
11 Australian water quality criteria

B. T. Hart

Introduction
Australian water resources management has been mainly concerned—understandably—with factors relating to water quantity. Recently, however, there has been increasing concern with the quality of water resources. The concern has resulted largely from increased awareness that the quality of much of Australia's water resources has deteriorated significantly in the past twenty to thirty years. This awareness has led in most Australian states to wide-ranging environmental legislation to counter the degradation. Thus far, legislation has concentrated on the control of pollutant discharge, but several states and the federal government have introduced additional legislation which requires the inclusion of environmental consideration in plans for major projects and for those proposals likely to affect significantly environmental quality.

With the introduction of broader environmental legislation, it has become apparent that informed decisions need more scientific information to enable the proper assessment of long- and short-term consequences of pollution and other possible environmental threats. Recognising this, the Australian Water Resources Council (A.W.R.C.) sponsored the production of a Compilation of Australian Water Quality Criteria (Hart, 1974). This chapter considers these criteria in a general way and with the benefit of some hindsight. It considers the formulation of criteria, their current status, uses, and strategies for the future in this area of water resource management.

Formulation of water quality criteria
Water quality criteria have been developed to introduce more objectivity into decisions concerning water quality management. In particular, their development has involved the collection of all relevant scientific information in an accessible and usable form. A number of formal definitions exist. A useful one is a slight modification of that suggested by the U.N. Conference on the Human Environment in 1972:

Criteria (or exposure-effect relationships) are the quantitative relationships between the exposure to an agent (pollutant) and the risk or magnitude of effects under specified circumstances defined by environmental and target variables (the target is the organism, population or resource to be protected from specific risk).

Water quality criteria, then, represent the scientific information on which to base decisions or judgments about the suitability of water quality for a designated use.

A clear distinction between criteria and standards should be made. Figure 17 will help make this distinction. It shows that whereas criteria are necessary to set standards, standards also take into account classifications of water uses and
plans for implementation and enforcement. It follows that standards are levels derived from subjective consideration of economic, social, political and other consequences of possible regulatory strategies, and are legally enforceable. Most documents detailing criteria contain the information needed to protect a number of designated beneficial uses for water resources (National Technical Advisory Committee on Water Quality Criteria, 1968; Canada Department of Environment, 1972; Committee on Water Quality Criteria, 1972; Hart, 1974). Designated uses generally include domestic purposes, preservation of aquatic ecosystems, agriculture (including irrigation, stock watering and farmstead use), industry, and recreational and aesthetic purposes. The greatest effort has been in the development of adequate criteria for domestic supplies and the preservation of aquatic ecosystems. In this chapter,
the latter criteria will receive most attention since in many respects they are the most difficult to develop.

Because current publications on criteria include information about a variety of beneficial uses of water bodies, they are voluminous and require much effort to prepare. As information accrues it will be impracticable to keep such single compendia up to date: a more viable procedure will be to prepare monographs, each dealing with a specific group of pollutants. Progress in this direction has been made in Canada, where the National Research Council of Canada (Associate Committee on Scientific Criteria for Environmental Quality) has prepared monographs on, for example, criteria for chromium, lead, fluoride, picloram, chlordane, endosulfan, methoxychlor, NTA, eutrophication and pulp and paper wastes (National Research Council of Canada, 1977).

In the actual formulation of water quality criteria it is essential to adopt a rational methodology to collate the most relevant information. Information is needed in three areas (Fig. 18):

1. possible sources of the agent, including both point and non-point sources, and the adequacy of treatment technologies for controlling concentrations and amounts discharged;
2. the fate of the agent in the aquatic environment, including its pattern of distribution, transport mechanisms, levels of biologically active forms, possible transformations between forms, and critical pathways to sensitive and critical organisms;
3. the effects of the agent on critical organisms, including acute, chronic, cumulative and synergistic effects, the influence of modifying factors on these effects, and possible ecological consequences.

Of course, in practice very few criteria include this ideal amount of information; it is only rarely that all the necessary information is available.

Status of water quality criteria
For toxic materials, such as heavy metals, the development of comprehensive quality criteria for the protection of aquatic ecosystems is still in a very early stage. The usual methodology is to derive ‘safe’ levels from acute toxicity data. The technique is twofold. Firstly, bioassays are conducted, preferably on important and sensitive species from the ecosystem, to determine the acute toxicity (generally quoted as the 96 hr LC$_{50}$—the concentration at which 50 per cent of the test organisms are killed in 96 hours). Secondly, an application (or safety) factor is then applied to obtain an estimate of ‘safe’ environmental levels. Simply stated, the philosophy is that there is a threshold level below which the ecosystem is protected from deterioration, and this level can be estimated from acute toxicity data on selected species in the ecosystem. This
Water usage

approach has a number of limitations. Since it is important for those using water quality criteria to be aware of these, some of the major problems associated with the derivation of current criteria for aquatic ecosystems will be briefly discussed. There are many more.

The problem of threshold concentrations. For some pollutants, it is not valid to apply the concept of a threshold concentration below which there are no adverse effects. This is particularly so for some new synthetic organic chemicals. It is more likely to be valid for such pollutants as heavy metals since most of these are naturally present in the aquatic environment.

The problem of extrapolation of lethal levels. There is no scientific basis for the extrapolation of lethal levels to levels where there is 'no effect'. Recent evidence, however, indicates that, in a few cases, the ratio of the concentration producing acute effects to that producing chronic effects in certain fish is often similar to generally quoted application factors. This gives some confidence on the possible validity of the technique. However, major problems of extrapolation of these laboratory data remain.

The problem of testing single species. Data derived from the testing of single species, such as occurs in most bioassay tests, do not guarantee adequate protection of the ecosystem. This is true even if the test species are from the system in question. There would be more confidence in this respect if information were available on critical organisms—those essential to ecosystem function. However, this implies more knowledge of aquatic ecosystems than is available. Most bioassay data relate to acute and chronic testing of indicator species. These are generally species which breed easily in the laboratory and whose life-cycles are well-known. They may well be quite unimportant in ecosystem function. The inevitable problems of extrapolation of laboratory data to the field follow. Some progress is being made overseas using small microcosms in tests rather than single species (Cairns et al., 1977).

The problem of chemical form. Until recently, little concern existed about the influence of chemical forms of toxic pollutants on their biological activity. Yet the chemical forms of heavy metals, for example, can be significantly affected by such factors as hardness, alkalinity, pH and organic content. The evidence suggests that the form of heavy metals can have a considerable influence on their biological activity (Brown et al., 1974; Andrew, 1976; Davies et al., 1976); methyl mercury (MeHg⁺), for example, is much more toxic than mercuric salts. It appears that the active form of a number of heavy metals is the free ion. This has been shown by Andrew (1976) for copper using Daphnia (a microcrustacean), and by Davies et al. (1976) for lead using rainbow trout as a test animal.

The problem of bio-accumulation. Direct toxicity is not always of most concern; pollutant accumulation within organisms can also be important. Mercury, for example, can accumulate markedly in fish, even though concentrations in water are not directly toxic (Fig. 19). Molluscs also accumulate heavy metals. Thrower and Eustace (1973) have reported that oysters in a polluted area of the Derwent estuary, Tasmania, had
concentrations of zinc, cadmium and copper as high as 21,000 μg g⁻¹, 63 μg g⁻¹ and 450 μg g⁻¹ (wet weight), respectively.

**The problem of synergism.** Certain mixtures of pollutants have greater toxicities than added individual toxicities (the phenomenon of synergism). Anderson and Weber (1976) found that a mixture of zinc and copper was 2.5 times more toxic to fish (guppy, *Poecilia reticulata*) than was predicted by adding together the individual metal toxicity values. Tests with mixtures of copper and nickel, however, showed no such synergistic effect. Thus caution is needed in using bioassay results from single pollutant tests. Where possible, all major waste dischargers should be required to test the toxicity of their total waste discharge. Cairns and Dickson (1978) have discussed a protocol for conducting such tests.

**The problem of pollutant transformation.** Many changes to the biological activity of pollutants may occur between the discharge point and the point of biological exposure. Both physicochemical (Hart and Davies, 1978) and biological (Ridley *et al.*, 1977) changes
may occur. While the principles involved in most transformations are reasonably well known, detailed knowledge of which processes are important is almost non-existent. Thus, almost no current water quality criteria contain information on this aspect.

Uses of water quality criteria
Broadly, there are two main uses for water quality criteria: in assessment of short- and long-term consequences of pollution (the development of standards); and to provide scientific information to assist in environmental planning. An additional use of some significance is in the development of water quality management strategies.

Development of standards. Two types of water quality standards are in general use: receiving water quality standards and effluent (or discharge) standards. Considerable debate has taken place on the relative merits of each type. Much of it has taken place in the United States where, recently, the U.S. Environmental Protection Agency has changed its legislation from one with emphasis on receiving water standards to one with emphasis on effluent standards and involving the requirement that discharges are treated to a specified degree. Receiving water standards were apparently found to be unworkable or at least unenforceable. In Australia, the legislative power for establishing water quality standards rests with the state governments and the federal government with regard to the Australian Capital Territory. Predictably, there are many regional differences in pollution control legislation, but all states, nevertheless, appear to be tending towards a system for licensing discharges.

It is, of course, possible to develop environmental protection systems which make effective use of both receiving water and effluent standards. This, in fact, has been done by the Victorian Environmental Protection Authority (E.P.A.). Under the provisions of the Victorian Environmental Protection Act (1970) (Victorian State Government, 1973), the E.P.A. must produce environmental policies for each major water body in the state. At present one policy, that for Port Phillip Bay (Victorian State Government, 1975), has been enacted, but others are in draft form. Each policy contains the following: a definition of the area covered by the policy, and its main features; documentation of existing beneficial uses, discharges and water quality conditions; a statement of the beneficial uses to be protected, and the standards necessary to achieve this protection; and the specifications of a program to attain upgraded quality. An essential part of each policy document is the water quality standards chosen to protect selected beneficial uses. Available water quality criteria are most important in the development of such standards. In fact, without the benefit of scientific information contained in the criteria, the standards would be extremely arbitrary and would carry little guarantee of effecting adequate environmental protection.

The Victorian Act also requires that all waste dischargers be licensed. One important consequence is that the Victorian E.P.A. has information on the types and quantities of all wastes discharged, and, further, it can impose restrictions on the concentration and load of any pollutant discharged. Such powers are essential to meet and maintain the
receiving water quality standards outlined in each policy.

**Environmental planning.** The federal and several state governments have recently introduced legislation requiring the consideration of environmental consequences in all major developments. This legislation is often manifest in the requirement that developers produce an environmental impact statement (EIS). Hart and Cullen (1976) considered that there are four main steps in the process of environmental assessment (see also Chapter 20):

- identify the potential environmental effects;
- predict the magnitude of possible environmental impacts that may be produced by the identified effects;
- evaluate the significance of all impacts; and
- communicate the above information in such a form that it can be used by the decision-maker in assessing the various alternatives available.

The third step is often the most difficult, since it must involve value judgments on what is significant. While these may reflect the bias of the assessor(s), they are likely to be more objective if the estimated magnitudes of the possible impacts are based on sound scientific information. Such information is provided by criteria compilations—such as those by Hart (1974) and the Committee on Water Quality Criteria (1972).

In passing, it may be noted that the development of water quality criteria is far more advanced than is the development of criteria for maintaining the quality of other sectors of the environment. It is also noted that the use of documented criteria, while useful, in most cases does not reduce the need for site-specific environmental investigations to be carried out also.

**Development of water quality management strategies.** An additional example of the use of water quality criteria, and one which includes elements of both standard setting and environmental assessment, is provided in the methodology adopted to develop a water quality management strategy for the Albury-Wodonga Growth Centre (Cities Commission, 1974; Albury-Wodonga Development Corporation, 1975). Although the example is specific, the methodology is applicable to other situations where such a management strategy is needed and is described on that basis.

In 1973 a decision was made to develop the twin cities of Albury and Wodonga as a growth centre. A population of 300,000 by the year 2000 was projected. Albury and Wodonga are situated on opposite banks near the headwaters of Australia's most important river, the River Murray. The Murray is about 2,500 km long, and provides water for some 500,000 ha of irrigated land along its length; it is also the source of domestic water for many towns, e.g. it provides over half of Adelaide's supply and all Whyalla's, Port Pirie's and Port Augusta's. The river also serves as a focal point for recreation, and is aesthetically important for a vast area of dry inland Australia. Additionally, much of the ecology of the river is unique.

It was obvious, therefore, that detailed planning and strictly controlled development were essential in order to
minimise the impact of this growth centre on the river water quality and hence its beneficial uses. The detailed planning has already taken place, and the first stages of development are now in progress.

An integral part of the detailed planning was the development of a water quality management strategy to maintain (and perhaps enhance) River Murray water quality, such that no designated beneficial use would be adversely affected in the short or long term. The strategy was seen as a dynamic process, sensitive and responsive to any defined need for change, and consisting of three fundamental and interrelated activities:

1. the formulation of water quality objectives (standards) to protect identified beneficial uses;
2. the formulation of a management plan to include rational and enforceable policies for both effluent discharge and land use;
3. a continuing water quality survey and monitoring program to provide information for decisions on the efficacy of the water quality objectives and management plan.

A central need in this strategy was for water quality criteria, particularly in the formulation of the water quality objectives. There was also a need to use documented criteria to assist the development of rational effluent and land-use policies.

A close similarity will be noted in the information needed for the environmental assessment process briefly described previously, and that needed for the development of the water quality objectives for Albury-Wodonga. An extensive survey was undertaken to provide a baseline description of the ecological characteristics of the River Murray in

the study area (see Cities Commission, 1974; Croome et al., 1976). The likely major environmental impacts due to the development project were also identified and their likely magnitude estimated.

Strategies for the future

Information needed for future criteria. Because of the complexity of the interrelationships between the fate and effects of any aquatic pollutant, it is important to define more precisely the most important sorts of information to be collected in order properly to develop the next generation of water quality criteria. The need for information on sources, fate and effects of agents in aquatic systems has already been suggested. This is of course an extremely broad statement, and in order to be somewhat more specific those areas where good scientific information is needed for the further development of heavy metal criteria are listed below by way of example. Information is needed on:

the important pathways by which the metals will reach the target organisms;
the critical organisms;
the biologically active forms of the heavy metal;
the levels of these forms within the system;
the exposure-effect relationships between the active metal forms and the critical organisms;
the natural processes that will influence the biologically active forms.

Obviously some of this information depends on the nature of the aquatic system. For example, the critical organisms, the levels of the biologically active
form of heavy metal, and the processes influencing these (which will be considerably different in an estuary and a soft-water lake). Some of it does not.

A particularly serious deficiency in Australia has been, until recently, the almost complete lack of work on exposure-effect relationships for critical Australian organisms. A few data are now available (Thorp and Lake, 1974; Cassidy and Lake, 1975; Negilski, 1976; Ahsanullah, 1976).

It is naive to assume that all the information required properly to develop quality criteria, particularly for toxic pollutants, will be forthcoming in the near future. It will not, and for some considerable time there will be a need to make do with less than adequate information. It is therefore essential to couple the use of water quality criteria with programs in which the receiving environment is continually monitored, chemically and biologically.

**Updating criteria.** In a field as broad, changeable and important as that concerned with water quality, considerable thought must be given to the best methodology for updating criteria. This is particularly so in Australia with its small population and limited financial and social resources to support a large local research program. The extent to which overseas information can be used requires especial care for, although it would be unwise for Australia to ignore appropriate overseas information, there are sufficient differences between the Australian inland aquatic environment and that elsewhere to preclude the direct application of non-Australian data (see Chapter 2).

An equally important problem is the lack in Australia of any one organisation whose task it is to collect and assess relevant information. There is certainly a very real need for one. There is also a need for an organisation to prepare state-of-the-art reports and review documents, to organise information for updating criteria, to disseminate information, and to undertake long-term research of national importance generally in the field of water quality (at present, the small amount of research underway is conducted in fragmented fashion in state water authorities, universities and colleges of advanced education, and CSIRO; some small measure of co-ordination is provided by the A.W.R.C.). The establishment of a national water research centre is long overdue (see Chapter 1).

**References**


Australian water quality criteria


Introduction
Even beyond the minimal daily amount for physiological requirements, the supply of water for human needs is now so much a part of man's domestic and working life—not to mention the aesthetics of his existence—that he has become totally dependent on adequate systems of water supply and storage. It follows that public health is equally dependent, for many parasites of medical significance have exploited the opportunities thereby provided. Dissolved chemical substances may form an additional hazard for public health. These two aspects of water resource management are discussed in this chapter.

Infective disease organisms in water supplies
Water-borne disease in the strict sense is transmitted by contact or ingestion. Organisms transmitted by contact enter the body through the broken or unbroken skin or through susceptible mucous membranes, e.g. the conjunctivae. A wide range of organisms of this type exists: some inhabit water naturally; others live in water for part of a life cycle which involves man as a final host; yet others can be regarded as contaminants or pollutants, that is as infective agents discharged from a host into water and merely dispersed by water to a new host. Table 7 indicates some of the major parasites of this sort and the infections they cause. Of those listed, it may be noted that schistosomes are generally unimportant in Australia (see Chapter 13), and that although meningo-encephalitis is almost invariably fatal and is cosmopolitan in distribution, its frequency of occurrence is not great.

Table 7 Some examples of water-borne infective agents causing infection by contact

<table>
<thead>
<tr>
<th>Agent</th>
<th>Presence in water</th>
<th>Infection</th>
</tr>
</thead>
<tbody>
<tr>
<td>PLATYHELMINTHES</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Schistosoma spp.</td>
<td>life stage</td>
<td>bilharzia</td>
</tr>
<tr>
<td>PROTOZOA</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Naegleria aerobia (N. fowleri)</td>
<td>free-living</td>
<td>amoebic meningo-encephalitis</td>
</tr>
<tr>
<td>BACTERIA</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Aeromonas hydrophila</td>
<td>free-living</td>
<td>wound infection</td>
</tr>
<tr>
<td>Leptospira interrogans</td>
<td>contaminant</td>
<td>febrile disease</td>
</tr>
<tr>
<td>Chlamydia trachomatis</td>
<td>contaminant</td>
<td>conjunctivitis</td>
</tr>
<tr>
<td>VIRUSES</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Adenovirus</td>
<td>contaminant</td>
<td>conjunctivitis</td>
</tr>
</tbody>
</table>
Diseases transmitted when water is swallowed are more important. Again a variety of organisms is involved (Table 8), but in general they are not found free-living in water. Although the number of organisms of this sort is very large (Geldreich, 1972), the common ones are relatively few. Of those listed in Table 8, only *Dracunculus medinensis*, a parasitic worm of limited distribution, is dispersed by water alone, and this is due to the nature of its life cycle. In other cases, dispersal involves contamination of water supplies by excreta from man and other animals. The water acts as the transport mechanism for infective forms of the parasite and invasion primarily takes place via the intestinal tract.

That water could spread infection was discovered, and convincingly demonstrated, well before micro-organisms were shown to cause disease. William Budd very early correlated the spread of typhoid in Britain with specific supplies of water (Budd, 1856, 1873). Similarly, Snow (1855) related rates of cholera mortality in a part of London with particular supplies of water. However, the exact nature of the relationship between disease organisms and water supply is now known not to be simple. It is not easy to set up infections, and the actual size of an infectious dose of organisms is often considerable (infectious dose is usually measured as the I.D.₅₀, i.e. that dose causing infection in 50 per cent of recipients). Experiments on human volunteers (Hornick et al., 1966, 1971) have given an I.D.₅₀ for typhoid (*Salmonella typhi*) of $10^7$ bacteria, and for cholera (*Vibrio cholerae*) of $10^5$, although smaller doses may give rise to minor symptoms. Such results explain Budd’s (1873) observation that fairly simple precautions serve to stop the spread of typhoid, a fact that would be incomprehensible if the infectious dose was low.

There is a paradox involved, however. If the dilution factor involved in the contamination of water supplies is taken into account, it is unlikely that any individual would receive a large enough dose of infective agents since these do

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**Table 8** Some examples of water-borne infective agents causing infection when swallowed

<table>
<thead>
<tr>
<th>Agent</th>
<th>Presence in water</th>
<th>Site of disease</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>NEMATODA</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Ascaris lumbricoides</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><em>Dracunculus medinensis</em></td>
<td>life-stage</td>
<td>sub-cutaneous regions</td>
</tr>
<tr>
<td><strong>PROTOZOA</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Entamoeba histolytica</em></td>
<td>contaminant</td>
<td>gut and liver</td>
</tr>
<tr>
<td><em>Giardia lamblia</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><strong>BACTERIA</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td><em>Salmonella spp.</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><em>Shigella spp.</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><em>Vibrio cholerae</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><em>Escherichia coli</em></td>
<td>contaminant</td>
<td>gut</td>
</tr>
<tr>
<td><strong>VIRUSES</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hepatitis A</td>
<td>contaminant</td>
<td>liver</td>
</tr>
<tr>
<td>Rotavirus (?)</td>
<td>contaminant</td>
<td>gut</td>
</tr>
</tbody>
</table>
not multiply significantly in water. Yet individual persons do contract infections from water supplies. The paradox is partly resolved if the I.D.₅₀ is considered as inappropriate in epidemics. Thus, in the outbreak of typhoid in London in 1937, when some 300 cases occurred among a population of 30,000 to 40,000, an I.D.₁ is more appropriate. This represents either that proportion of the population particularly susceptible, or those individuals who swallowed an aggregate of bacteria or a sample at the extreme end of the normal distribution curve. Mathematically, Kehr and Butterfield (1943) suggested that 1.5 per cent of a population can be infected by a single individual of *Salmonella typhi*.

The paradox is still not fully resolved, and resembles the related problem of the survival of intestinal bacteria in water. Here, too, separate experiments have yielded markedly different results.

**Chemically induced diseases**

All natural supplies of water contain dissolved substances. Normally, these are not hazardous to public health, but a few natural solutes can cause problems if present in excessively high (or low) concentrations. For example, high concentrations of nitrate may give rise in babies to methaemoglobinaemia, a blood disorder. Waters with excessive amounts of fluoride may cause teeth to mottle. Conversely, low concentrations of fluoride may cause other dental problems. In many places, fluoride concentrations in water supplies are artificially regulated in the interests of dental health (see, for example, Duckworth, 1966). The aggregate sum of certain ions appears to be important too; there is evidence that ‘hard’ and ‘soft’ water supplies have differential effects on public health—and perhaps especially so with regard to coronary heart disease, though it should be added that conflicting evidence exists (Meyers, 1975). Above a certain salinity (the sum of all dissolved salt concentrations), of course, supplies of water become non-potable.

Man now discharges a great many exotic, and largely unnatural substances to the environment, and many of these eventually appear in water supplies. A number are deleterious to public health. Heavy metals, organic poisons (e.g. pesticides), and potentially carcinogenic compounds are some of the more significant deleterious substances. Their effects on public health, and those of other substances, are currently the subject of intensive investigation (though perhaps not intensive enough). Somewhat arbitrarily, effects can be considered as those that develop acutely, and those that give rise to health problems only after long periods (as happens, for example, with cumulative poisons).

**The point of origin of health problems in large water supply systems**

Every part of a system for the storage and supply of water can be the source of a problem affecting public health. It is convenient to discuss this on a *seriatim* basis.

**Engineering works.** Pits, quarries, wheel-ruts and other cavities created during the construction of dams, aqueducts and channels may be the cause of health problems, since they provide a multiplicity of habitats in which aquatic vectors of disease (such as mosquitoes) can multiply. Sound engineering practice recognises this. The potentiality for health problems from the water storages themselves has also been recognised
Public health aspects

131

(see, for example, Stanley and Alpers, 1975; also Chapter 13).

**Water sources.** It has long been axiomatic that water supplies should come from the least polluted source. However, the nature of the source can change adversely, either by slow deterioration in quality, or by sudden catastrophe. As a result, the problem may escape notice, or be too swift for effective preventive measures. The hepatitis epidemic involving nearly 30,000 cases that occurred in Delhi in 1955 resulted from a change in the source of supplies; when the water-level of the River Jumna fell, local currents directed the grossly polluted effluent of the Najafgarh Nallah into supply intakes and not raw river water (Viswanathan, 1957).

**Water treatment.** It is possible to treat adequately source water of almost any quality. The cost of treatment, however, may be such as to make the use of a particular supply impracticable. There is every reason, therefore, to tailor with care the extent of treatment with the extent required. If too little treatment is given, the safety margin may be insufficient to maintain public health; on the other hand, increasing complexity in treatment techniques increases the demands on equipment, processes and operators. Frequently, the training and motivation of plant operators is the weakest link in the chain (Bodily, 1976), and incorrect operation of plant or inadequate monitoring has often been the cause of health problems from water supplies.

**Service reservoirs and aqueducts.** Uncovered aqueducts and service reservoirs are no longer as common as they were. Where they still occur, however, they remain open to contamination by disease organisms. This has often occurred from birds or other wild animals, although there has been the tendency to dismiss the bacteriological signs of such pollution on the grounds that typhoid (*Salmonella typhi*), Hepatitis A and shigellosis are diseases which for all practical purposes are confined to man. This argument should be resisted, for it may lead to an ignoring of pollution from human excreta (particularly in rural areas), and the argument itself ignores the contribution to human disease of organisms found in other animals (zoonoses). Salmonellosis is endemic in populations of wild and domestic animals and birds. Wild animals are a likely source for water-borne outbreaks of giardiasis (*Giardia lamblia*) (Anon., 1977a). And the rotaviruses have recently been discovered to be an extremely important cause of gastroenteritis in young animals of many species including our own (Anon., 1977b). Rotaviruses from different animal species have been shown to be very similar and, in some cases, able to cause disease in heterologous species (Snodgrass *et al*., 1976). Animal rotaviruses could, therefore, pose a threat to humans.

**Distribution system.** Breakages in the distribution system for water supplies may give rise to health problems by allowing access of polluted material to the system. This is particularly so when water mains pass near sewers. Furthermore, mains inevitably become contaminated whenever they are removed from service for maintenance or repair; disinfection and testing before re-use are needed (Boniface and Taylor, 1968). Occasionally, problems arise from the employment of disease carriers by
waterworks authorities. This has led to the formulation of schedules for screening waterworks employees. The number of carriers actually reported, however, is low. Nevertheless, there is a need to persist with this sort of screening; the Microbiological Diagnostic Unit in Melbourne, for example, detected two disease carriers employed in the construction of the Upper Thompson River Scheme, an extension of Melbourne's water supply system (Commission of Public Health, 1973, 1974/75).

**Domestic reticulation.** Health problems originating in domestic reticulatory systems generally relate to the leaching of toxic materials from water pipes, but plumbing defects may cause other problems. The problem of excessive lead consumption as a result of lead plumbing in old houses—especially when the water involved is 'soft'—has been well-documented (e.g. Taylor, 1958; Lauwerys et al., 1977). This particular problem, however, is not important in Australia which has few buildings with lead plumbing. Lead (as well as tin and cadmium) may also leach from unplasticised polyvinylchloride pipes (see, for example, World Health Organisation, 1973). Problems may arise too as a result of the leaching of copper from plumbing; such problems have certainly occurred in Australia (hot water services are particularly implicated). Plumbing defects cause infectious diseases, rather than problems of poisoning. An example is provided by Ritchie and Davis (1948) who reported the occurrence of an outbreak of amoebiasis (*Entamoeba histolytica*) in a Tokyo apartment as a result of defective plumbing.

**Health problems in small water supply systems**
Small-scale water supply systems include those of single households drawing supplies from roof catchments or bores, and small rural water trusts. Many problems are associated with these small systems, and it is perhaps significant that most cases of waterborne disease reported to the Center for Disease Control in North America originate from small supply systems (Center for Disease Control, 1976, 1977). In any event, it is clear that in Australia many small rural water supply systems are defective with regard to the source of their supply and the state of reticulation and treatment (with respect to both equipment and standards of maintenance). This is perhaps understandable in that many rural communities have declining populations and cannot or are unwilling to pay for improvements when most ratepayers rely on personal supplies from roofs or bores.

Even bore supplies are not immune to problems; contamination may occur through ground fissures, at the borehead, in storage tanks, or in the reticulation system (e.g. Neefe and Stokes, 1945). Small-scale supplies in national parks and other recreational areas in the United States have had many problems (Center for Disease Control, 1976, 1977), and a wide range of infective agents have been incriminated. The same situation probably prevails in Australia though goes largely undocumented; certainly the water supply of at least one ski resort has been shown to be grossly defective.

**Health problems in the manufacture of foodstuffs**
The use of water in various processes involved in the manufacture of food and
drink for human consumption provides several opportunities for the development of health problems. Even the use of ice may cause problems if contaminated water is used in its manufacture, since most pathogenic micro-organisms can survive being frozen. Commonly, problems arise when water used in the manufacture of foodstuffs is contaminated with pathogenic bacteria at levels well below those necessary to cause infection directly; then, within the foodstuff, the bacteria multiply to highly infective levels. Thus, an outbreak in Victoria in 1977 of salmonellosis in young children fed on powdered milk was caused by *Salmonella bredeney* which was probably present initially in small numbers, but which developed a large population in the presence of warmth, moisture and food. Contaminated channel water was implicated in this outbreak (Microbiological Diagnostic Unit, unpublished).

**Health problems in the recreational use of water**

There is a number of well-characterised diseases associated with the recreational use of inland waters. Schistosomiasis is one, but only schistosomes parasitising birds are of any significance in Australia (Rohde, 1978; see also Chapter 13); the parasitic larvae may cause irritating rashes in man, a condition known as benign schistosome dermatitis or bather's itch. Leptospirosis, a bacterial infection, may be caused by swimming in infected waters, but is more usually caused by industrial exposure; it is a considerable problem in some countries, though not in Australia (see Wannan, 1975). Most cases of amoebic meningoencephalitis have followed swimming in lakes (Carter, 1970) or, even, in chlorinated town water supplies (Anderson and Jamieson, 1972). In some cases mere contact with water in puddles was sufficient to cause the disease (Apley et al., 1970). As pathogenic amoebae are favoured by high temperatures, it is surprising that conclusions from investigations of protozoans in thermal discharges are not in agreement (Willaert and Stevens, 1976; Wellings et al., 1977); clearly several factors remain for resolution (Cursons et al., 1977).

Other diseases that may on rare occasions be acquired from swimming in fresh waters include typhoid, shigellosis, and conjunctivitis resulting from infections by *Chlamydia* and adenoviruses. Several causes of typhoid and shigellosis have been ascribed to recreational swimming in polluted waters (Center for Disease Control, 1972, 1974). Both *Chlamydia* and adenoviruses have been shown to be transmitted in swimming pools with inadequate standards of disinfection (although McLean's (1967) experiments with poliovirus indicated that the chances of transmission in well-maintained pools is remote). *Pseudomonas aeruginosa*, an organism which causes rashes, can also be transmitted in inadequately maintained swimming pools.

Finally, it should be noted that bacterial infections of eyes, sinuses and the middle ear after swimming usually follow the mechanical flushing of bacteria from the nasopharynx to these parts of the body; they are not caused by contaminated water.

**Concluding remarks**

There are two areas of potentially great significance and current interest and controversy with regard to public health problems associated with water usage.
One is the question of the importance of viruses in drinking water supplies; the other concerns the importance of toxic chemicals, especially those likely to have carcinogenic properties.

Until recently there was no good evidence that any virus infection was directly transmitted by water with the exception of Hepatitis A (infectious hepatitis) (Mosley, 1967). However, with the discovery by electron microscopy of several viral agents that cause gastro-enteritis, ideas on this matter need revision (Bishop et al., 1973; Blacklow et al., 1972; Appleton et al., 1977). Moreover, the water-borne outbreak of disease at Robinvale, Victoria, in June 1973 has retrospectively been shown to have probably been caused by rotaviruses. 'Sewage poisoning', that is gastro-enteritis associated with a mixed flora suggestive of sewage contamination but without recognised pathogens, is almost certainly of viral aetiology.

Shuval (1976) and others have pointed out that virological techniques can now detect viruses in large volumes of town supplies. Indeed, the infectious dose of those viruses which can now be investigated approximates to a single particle. Viruses of the groups in question appear to be stable and are relatively resistant to chlorine. Thus, it is hypothesised that the water supply system of a large community transmits viruses at a low frequency and causes scattered cases of disease not recognisable as water-borne but each capable of further spread by contact or food contamination. In this way, viral disease is maintained in the community. It follows that the treatment and monitoring of water supplies should be more oriented towards viruses than is presently the case.

Conversely, it has been argued that recognised water-borne outbreaks of hepatitis are associated with known failures in conventional treatment processes. As Hepatitis A cannot presently be isolated the situation in which water supplies are regarded as transmitting at low frequencies is merely an extrapolation from a picornavirus (e.g. poliovirus) for which there is no good evidence that dispersal is by water. It is argued that stronger evidence is required before expensive preventive measures can be justified (Gamble 1979).

Controversy concerning the importance of toxic chemicals runs along similar lines. As the techniques of detection and analysis improve, so are more potentially toxic compounds isolated in lower and lower concentrations in more and more sources of water. The question is: to what extent are these compounds a serious threat to community health? There is no evidence at present of any epidemics of cancer resulting from drinking contaminated water. The question is an important one nevertheless, and needs careful investigation.

References

Appleton, H., Buckley, M., Thom, B. T.


13 Medically important diseases with aquatic vectors and hosts

T. Petr

Introduction

Australia is fortunate in that, compared to many tropical and subtropical countries, it has a relatively small number of parasitic diseases directly associated with the aquatic environment. Its relative closeness to south-east Asia has apparently not yet led to the occurrence of many such diseases common in that part of the world—as had been feared by some authors. Even malaria, a common disease in neighbouring Papua New Guinea and Irian Jaya, has been eradicated from Australia (although its aquatic vectors are common here). In Australia, the only relatively common human diseases transmitted by aquatic vectors are those caused by viruses, and these are not restricted to tropical and subtropical regions; the Murray Valley encephalitis virus and the Ross River virus are common in New South Wales and Victoria.

At present, exotic parasitic diseases are imported into Australia mainly by tourists, migrants, and Australians returning from overseas. However, the imported exotic parasites such as Plasmodium and Wuchereria stand very little, if any chance to enter the available vector pool, or, as with Schistosoma, to establish themselves in some alternative vectors. In addition, good hygienic conditions and low human population density in the tropics and subtropics offer little chance for imported parasites such as Schistosoma, or some other trematodes, to survive or become established. Further, the eating habits of Australians differ from those in south-east Asia, where raw snails and fish and some aquatic plants are often consumed together with their parasitic inhabitants.

There remains cause for concern, however, over the possibility of the natural introduction of parasites and their vectors into tropical and subtropical Australia. Birds may carry parasites and in some cases even their hosts, and they may cross the short distance between the Indonesian islands and north-western Australia relatively easily. The Ord reservoir (Lake Argyle) with its associated irrigation system is considered vulnerable to such introduction. However, it has yet to be proved whether such a transfer is taking place, and a large amount of experimental work on the suitability of Australian aquatic invertebrates for exotic parasites is still required. Thus, the ability of Australian molluscs to accommodate larval stages of Schistosoma haematobium and S. mansoni is not known. Nor, at present is it known whether aquatic birds are able to transport the host of S. japonicum alive to Australia from its nearest area of distribution, Sulawesi. The ability of Australian mosquitoes to vector diseases other than those caused by viruses also requires further investigation.

These and other important questions related to water-borne disease remain for resolution. The answers may well be
significant in view of the fact that large numbers of immigrants from south-east Asia have recently been arriving in Australia; there is always the possibility that some may introduce parasites to Australia despite careful screening by health authorities.

This chapter discusses medically important diseases vectored by animals which live in inland waters for the whole (e.g. snails) or part (e.g. mosquitoes) of their life, and which have significance for Australia. For ease of discussion, such diseases are discussed as: diseases endemic to Australia, with aquatic vectors here; diseases occasionally imported into Australia, and with aquatic vectors present here; and diseases occasionally imported into Australia, but without known aquatic vectors here. Other important trematode infections of humans are also discussed briefly; though present in Asia, none occurs in Australia. The only other trematode of possible medical significance in Australia is the common liver fluke, *Fasciola hepatica*, which sometimes infects man in Europe. This fluke, it may be noted in passing, is responsible for large economic losses in the sheep and cattle industry in Australia. It is very briefly referred to because it is not otherwise discussed in this book—yet its occurrence is an important factor to be considered in water resource management. The chapter concludes with a discussion of the impact of man himself on the spread of diseases vectored by aquatic organisms.

There are, of course, other human diseases associated with the aquatic environment, but not vectored by aquatic organisms; they are considered in Chapter 12.

**Diseases endemic to Australia: arboviruses and their vectors**

Arboviruses (ARthropod-BOrne VIRUSES) are viruses which are transmitted to vertebrate hosts by arthropod vectors. The majority use adult mosquitoes as natural vectors, and therefore control depends almost exclusively on eradication of local mosquito transmitters. The mobility of non-human reservoirs such as aquatic birds may be an important factor in the spread of arbovirus diseases such as Murray Valley encephalitis, where the bird is the main host in the arbovirus-mosquito-host cycle (Stanley and Alpers, 1975).

There are more than 300 recognised arboviruses forming 41 groups based on antigenic relationships (Surtees, 1975). In Australia at present a small number of viruses is important from the point of view of human disease (Table 9). Among them two are more serious: the Murray Valley encephalitis (MVE) virus which causes irregular outbreaks of encephalitis, and the Ross River virus (RRV) which affects many more people and causes epidemic polyarthritis. Sindbis and Kunjin viruses cause a mild febrile illness and are considered to be medically less important than the MVE and RRV viruses. Dengue, which is now of historical interest only as there is no transmission going on in Australia (Prof. R. L. Doherty, personal communication), in some outbreaks leads to haemorrhagic form in children which may be fatal.

In epidemic areas of MVE, much effort has been spent on identification of arthropod vectors. Reeves *et al.* (1954), who sampled the Wentworth-Mildura-Red Cliffs area along the Murray River, tested 17,833 mosquitoes of 16 species for MVE virus, but all tests were nega-
Medical importance diseases

In a study by Doherty *et al.* (1963) in the endemic area in north Queensland, seven strains of MVE virus were isolated from 25,901 mosquitoes of 31 species collected at Mitchell River, Cairns and Normanton. *Culex (Culex) annulirostris, Aedes (Ochlerotatus) normanensis, Anopheles (Cellia) amictus* and *Culex (Culex) bitaeniorhynchus* contained Sindbis virus: *C. (C.) annulirostris* and *Aedes (Ochlerotatus) vigilax* contained MVE virus. *C. (C.) annulirostris* was the commonest mosquito collected and the source of 44 of the virus strains, of which 8 were new. In a second study by Doherty *et al.* (1963/64) on mosquitoes collected from mangroves near the Ross River at Townsville, RRV was isolated from *A. (O.) vigilax*. In laboratory studies by McLean (1953), 11 species of mosquitoes tested were infected with MVE virus, and from 7 to 41 days later the mosquitoes could transfer the infection to chickens.

The vertebrate hosts for most of the viruses isolated in Australia have been reviewed by Doherty (1972, 1974, 1977). Among the vertebrate reservoir hosts are aquatic and marsh birds. The RRV is believed to survive in nature by cycles of infection in which man may incidentally become involved (Anon., 1976a). More direct evidence that the RRV can infect man was shown in 1972 when Doherty *et al.* (1972) isolated this virus from an Aboriginal boy with a febrile illness. In 1971, an epidemic of polyarthritis due to this virus occurred in towns along the River Murray in South Australia as well as in towns situated on the Murray River in Victoria and New South Wales and towns in the Murrumbidgee and Coleambally irrigation areas (Seglenieks and Moore, 1974). There was general agreement among the residents that there was a greater than usual prevalence of mosquitoes, but no information about the species was available. Between October 1970 and June 1971 extensive shallow flooding probably led

Table 9 Medically significant arboviruses and their vectors in Australia

<table>
<thead>
<tr>
<th>Virus</th>
<th>Antigenic group</th>
<th>First isolated</th>
<th>Vectors</th>
<th>Non-human reservoir</th>
</tr>
</thead>
<tbody>
<tr>
<td>Murray Valley Encephalitis</td>
<td>B</td>
<td>1951</td>
<td><em>Culex (C.) annulirostris</em></td>
<td>wild birds, domestic fowls, cattle, horses</td>
</tr>
<tr>
<td>Ross River</td>
<td>A</td>
<td>1961</td>
<td><em>Aedes (O.) vigilax</em></td>
<td>wallabies, kangaroos, cattle, horses, dogs, birds</td>
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<tr>
<td></td>
<td></td>
<td></td>
<td><em>Aedes (O.) normanensis</em></td>
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<td></td>
<td></td>
<td></td>
<td><em>C. (C.) annulirostris</em></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td><em>Mansonia (M.) uniformis</em></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td><em>Anopheles (C.) amictus</em></td>
<td></td>
</tr>
<tr>
<td>Sindbis</td>
<td>A</td>
<td>1961</td>
<td><em>C. (C.) annulirostris</em></td>
<td>wild birds, domestic fowls, cattle, dogs, wallabies</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td><em>C. (C.) squamosus</em></td>
<td></td>
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<td></td>
<td></td>
<td></td>
<td><em>C. (C.) fatigans</em></td>
<td></td>
</tr>
<tr>
<td>Kunjin</td>
<td>A</td>
<td>1974</td>
<td><em>C. (C.) fatigans</em></td>
<td>domestic fowls, cattle</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td><em>C. (C.) squamosus</em></td>
<td></td>
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<tr>
<td></td>
<td></td>
<td></td>
<td><em>C. (C.) annulirostris</em></td>
<td></td>
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<tr>
<td></td>
<td></td>
<td></td>
<td><em>Aedes (O.) tremulus</em></td>
<td></td>
</tr>
<tr>
<td>Dengue</td>
<td>B</td>
<td>1943</td>
<td><em>Aedes (S.) aegypti</em> (key vector)</td>
<td></td>
</tr>
</tbody>
</table>
Water usage
to the establishment of a large number of suitable mosquito breeding sites.

_Culex (C.) annulirostris_, the key vector of MVE virus, breeds over wide areas in shallow freshwater pools such as those present after any major floods, and it has been suggested that high spring rainfall in northern Australia and the Darling River watershed, such as occurred before the four major epidemics of MVE in 1917, 1918, 1925 and 1951, could be used as an index of epidemic risk.

In the Ord River area of north-west Australia, Liehne et al. (1976) isolated MVE virus from _C. (C.) annulirostris_, and other arboviruses from _C. (C.) annulirostris_, _Aedeomyia catasticta_ and _Aedes (Macleaya) tremulus_. Since then, MVE virus has been isolated from _C. (C.) fatigans_ (N. F. Stanley, S. Anderson and M. P. Alpers, personal communication).

Although the area of incidence of MVE is principally the Murray Valley (Anon., 1974), during epidemics before 1974 cases of MVE were reported elsewhere in Australia. Therefore, it appears that there is a possibility of MVE outbreaks elsewhere, and new water resource developments such as irrigation schemes and dam building could provide suitable habitats for MVE virus invertebrate vectors and for non-human vertebrate reservoirs such as aquatic and wetland birds. Recent work on migration and dispersal of banded aquatic and semiaquatic birds may assist in our knowledge of the spread of viruses.

_Aedes_ species, such as _A. (Stegomyia) aegypti_ which breed in small collections of clean water (empty food tins, flower vases, old tyres), carry dengue virus. Infection by this virus may cause epidemics which are normally not fatal.

The most serious dengue epidemic in Australia was in Brisbane in 1898 (Waddy, 1975), and the most recent in Queensland in 1953-55 (Doherty, 1957, 1974). The disease was a problem in Papua New Guinea during World War II and a number of small outbreaks, based in most cases on clinical findings without virological confirmation, have been reported from northern Australia. But, despite epidemics in south-east Asia and the western Pacific, dengue has been conspicuously absent in Australia since 1958. The declining populations of _Aedes (O.) aegypti_ make further epidemics unlikely (R. L. Doherty, personal communication).

Diseases imported into, and with vectors present in Australia

_Malaria_. In Australia, malaria epidemics used to occur in the northern coastal areas (Black, 1972). The disease has now been eradicated for over 15 years, the last indigenous case being recorded in 1962. However, the northern part of the country remains receptive to malaria because suitable vectors are still present. In Australia, the protozoan _Plasmodium_ parasites were transmitted mainly by _Anopheles (C.) farauti_ (Colless and McAlpine, 1970), which is a member of the _A. (Cellia) punctulatus_ group. _A. (C.) farauti_ breeds in a wide range of temporary and permanent waters, natural and man-made, both fresh and brackish, shaded and sunlit. Lee and Woodhill (1944) suggested that in exceptional circumstances _A. (C.) annulipes_ could be responsible for brief epidemics of malaria. This species is likely to be the principal vector of malaria in the north-west of Australia (O'Gower, 1968). It shows a preference for bovine blood and
it rarely enters houses. Currently, malaria is imported by travellers (see below).

**Filariasis.** Although the nematode *Wuchereria bancrofti*, the main cause of elephantiasis, is widely distributed throughout the tropical and some subtropical areas of the world, and is present in Papua New Guinea, it is rarely found in Australia. In Papua New Guinea and Irian Jaya, the parasite has been found in 10 mosquito species (Ewers & Jeffrey, 1971). The main potential vector of filariasis in Australia is *Culex (C.) fatigans* (Colless and McAlpine, 1970). Lee and Woodhill (1944) mentioned for Australia also *Anopheles (Anopheles) bancroftii*, *A. (Cellia) farauti*, and *A. (C.) amictus* as vectors of *Wuchereria bancrofti*. The breeding sites of mosquito vectors are well-described: *C. (C.) fatigans* breeds in any polluted water such as open drains, septic tanks and water closet bowls (Waddy, 1975). *Anopheles (A.) bancroftii* breeds in shaded swamps and pools containing plants and *A. (C.) amictus* in non-polluted fresh water of a great variety of character, shaded or fully exposed to sun.

**Diseases occasionally imported into Australia, without known aquatic hosts there**

**Schistosomiasis.** Benign schistosome dermatitis is ubiquitous throughout the world, including Australia. The infection is probably unavoidable and of little importance to public health. Cercariae of certain non-human blood flukes cause human dermatitis, due to skin penetration by the schistosome cercariae. Operculate and non-operculate snails serve as intermediate hosts, and the adult flukes are usually parasitic in migratory birds. *Austrobilharzia terrigalensis* is transmitted by a prosobranch, *Velacumantus australis*.

Very few cases of urinary and intestinal schistosomiasis in Australia have been notified (Anon., 1972), mostly in war refugees, migrants or tourists. However, cases of African-acquired schistosomiasis caused by *Schistosoma mansoni* and *S. haematobium* continue to be recorded in Western Australia. Charters (1975) mentioned eight patients with urinary and intestinal schistosomiasis over a period of seven years. Snail hosts of these schistosomes, the pulmonate snails *Bulinus* and * Biomphalaria* respectively, have not yet been reported as exotic genera from Australia. Both schistosomes are responsible for debilitating diseases. A third medically important schistosome, *S. japonicum*, is the closest in its geographical distribution to Australia and therefore of particular interest. At present it is confined to China, Japan, the Philippines and the island Sulawesi in Indonesia. Another so-called Mekong schistosome was discovered in Thailand, Laos, Cambodia and Malaysia.

*S. japonicum* has many mammalian reservoirs, yet only one intermediate host: the snail *Oncomelania hupensis.* The host of the Mekong schistosome appears to be confined to the Mekong lower region (Liang and Van Der Schalie, 1975). The principal source of

*In some publications, different specific names for *Oncomelania* are used for geographically isolated areas: *O. quadrasi* in the Philippines, *O. hupensis* in China, *O. formosana* in Taiwan, *O. lindoensis* in Sulawesi. Burch (1975) has suggested that all of them are the same species, *O. hupensis.*
infection in the snail is faecal contamination by man. *S. japonicum* is most difficult to control because of the numerous reservoir hosts (Faust et al., 1975).

Schistosomiasis japonica is a chronic disease characterised by fever, dysentery, hepatosplenomegaly and a terminal cirrhosis of the liver. As in urinary and intestinal schistosomiasis, the eggs are responsible for the disease, being retained in the tissue of the host where they cause reactions leading to pathological changes. These changes are related to the number of eggs, which in turn is dependent on the number of adult worms (Jordan, 1975). Central nervous system involvement is more common with *S. japonicum* than with the other two species. As a rule, Schistosomiasis japonica has the worst prognosis of the three types of human schistosomiasis.

*O. hupensis* is common in rice fields, as was found for example in Sulawesi (Carney et al., 1973). There appears to be a correlation between the distribution of the snail and the annual rainfall pattern: the snail is confined to areas that do not exhibit a marked dry season (Anon., 1976b). This might preclude it from most of Australia. However, Burch (1975) mentioned that *O. hupensis*, as an operculate snail, can more easily withstand desiccation and aestivate during the dry season than the pulmonate hosts of *S. haematobium* and *S. mansoni*.

The snail-parasite relationship of *O. hupensis—S. japonicum*, their population dynamics, effects of molluscicides, snail control methods etc. are thoroughly studied. Such information, however, is unavailable for the *Lithoglyphopsis aperta*-Mekong schistosome relationship. At present no suitable host for the *S. japonica* and Mekong schistosome is known from Australia and it has been suggested that the presence of such hosts is highly unlikely. Wright (1971) has pointed out that there is a very marked restriction of the intermediate host range of most species, and that in some species this restriction extends well below the species level so that a particular strain of a parasite will only develop satisfactorily in certain races of snail host. Larvae of a particular parasite in contact with exotic snails may meet with total resistance which prevents their distribution in such an exotic environment.

Such protection is a distinct possibility for water bodies such as the Ord River scheme, where the introduction of schistosomes has been feared by some authors. It is known from experiments that a number of non-host snails, bivalves, *Daphnia pulex* (Cladocera), and larvae of *Corethra* sp. (Diptera) exercise significant interfering effects by protecting the 'target' snail *Lymnaea truncatula* from infection by *Fasciola hepatica* (Christensen et al., 1977). It has been pointed out by some authors (e.g. Charters, 1975) that the high temperatures of the Ord River would be harmful to both cercariae of urinary and intestinal schistosomes and their mollusc hosts if these were accidentally introduced. It still needs to be ascertained whether high temperatures would also be lethal for *S. japonicum* and its host.

The prevention of schistosomiasis is to a great degree a matter of hygiene. It is important to emphasise that the definitive host, man, is responsible for the dissemination of schistosomiasis by contaminating the aquatic environment. Education of people to break the parasite's life cycle by improved personal hygiene will probably prove in the long run to be more effective than any attack
on the host (Jackson, 1975). In the ordinary course of events the hygienic patterns in Australia are such that large-scale outbreaks of schistosomiasis are not to be expected. However, the ability of *S. japonicum*—but not the other human schistosomes—to parasitise rodents and other feral hosts makes it very difficult to eradicate, once established, even when rigorous attention is paid to human hygiene.

**Other trematode infections.** It has been suggested recently that molluscan hosts of trematodes present in several tropical and subtropical east Asian countries could be accidently introduced to Australia. The Ord River has a potential for infestation of *Paragonimus, Clonorchis, Heterophyes* and *Fasciolopsis* (Alpers et al., 1972). *F. buski* is present amongst other countries in Indonesia, *C. sinensis* in Vietnam, *P. westermani* in Indonesia, Malaysia and the Philippines, *Heterophyes* in Japan and the Philippines. Introduction of *Paragonimus* to Australia is feasible if culinary habits which make use of raw freshwater crabs are imported; the crabs may harbour encysted metacercariae. In the Far East, the snail hosts of *Paragonimus westermani* are operculate snails such as a number of species of *Semisulcospira*, one species of *Tarebia*, and one of *Assiminea*. Human infection with *Clonorchis sinensis* is also due to eating raw products—in this case fish. The resulting disease may culminate in heterophyid myocarditis which, for example, has been responsible in the Philippines for 14.6 per cent of fatal heart cases. Manson-Bahr (1966) listed 37 operculate snail hosts of this parasite present in the Far East. The snail host of *Heterophyes heterophyes* in Japan is *Tympanotomus micropterus*.

A widespread parasite of south-east Asia and the Far East is *Fasciolopsis buski*. It is acquired by eating water chestnut (*Eleocharis tuberosa*) whose bulbs are eaten raw and whole, and water caltrop (*Trapa natans*), whose fruits are traditionally peeled with the teeth, a process which dislodges adherent cysts of the fluke (Wright, 1971). The hosts are the planorbid snails *Segmentina trochoidea*, *S. hemisphaerula*, *Gyranoulus convexisculus* (*saigonensis*), and *Hippeutis cantori* (Manson-Bahr, 1966).

Another trematode infection caused by the consumption of raw snails is common in the Philippines, Java and China; the snails contain the encysted metacercarial stage of *Echinostoma ilocanum*. The hosts are planorbid and lymnaeid snails from which the released cercariae enter large edible snails such as *Pila conica* in the Philippines and *Viviparous javanicus* in Java (Faust et al., 1975).

The common liver-fluke (*Fasciola hepatica*) of sheep and cattle occurs frequently in man in Europe. In Australia, about one quarter of all sheep and cattle graze on potentially endemic pastures, and fascioliasis causes severe economic losses (Boray, 1969). In Australia, the pulmonate snail *Lymnaea tomentosa* is the intermediate host of the fluke. The main habitats of this snail are springs, backwaters, billabongs and swamps. Large permanent waters are not suitable and usually harbour only a few individuals. However, these are frequently responsible for the recolonisation of temporary waters, especially streams. The snails are common in irrigation channels in New South Wales, Victoria
and South Australia. Persistently high water temperature may have an adverse effect on the snail. Also, *Fasciola hepatica* metacercariae may not tolerate the hot climate of tropical Australia. Human infection in Europe nearly always results from eating water-cress grown in the wild where no control over freshwater snails is exercised (Wright, 1971).

**The impact of man on the spread of diseases with aquatic vectors and hosts**

The increasing number of tourists may import diseases such as schistosomiasis and malaria but also some other rarer trematode diseases. In 1975 there were 247 cases of malaria reported of which 242 were imported, 4 were introduced and one induced (congenital); 44 cases were reported from the malaria-receptive area of Australia (Black, 1977). The recent incidence of malaria in Western Australia results mainly from the number of Australians travelling and working abroad; the influx of migrants and temporary visitors plays a minor role (Charters and Davis, 1974). The immigration of refugees from south-east Asia, and the presence of *Anopheles* species capable of carrying the malaria parasite, make Australia a suitable area for malaria re-introduction, and this obviously demands close surveillance. However, it has been stressed by a number of authors (e.g. Dobrotworsky, 1965) that a capacity to harbour the malarial parasite does not necessarily mean that a mosquito is an important vector in nature. Its importance will be influenced by its feeding preferences, the frequency with which it comes into contact with man (especially indoors), the abundance of the species, and its longevity. For example, the mosquito *Anopheles (C.) annulipes*, widespread through Australia and a potential vector of malaria, prefers bovine blood to that of man. It rarely enters houses. In fact, in Australia no natural infection has been found in this species.

Formation of impoundments, both small and large, is initially associated with simplification of the habitat, which reduces the diversity of the aquatic fauna; on the other hand, it frequently produces an environment conducive to the establishment of a species of medical importance, which spreads over a considerable area in accordance with the size of the impoundment. The creation of large man-made lakes in Africa resulted in an enormous spread of urinary, and to some extent also intestinal schistosomiasis among fishermen who settled down on shores of new lakes. The distribution of schistosome mollusc hosts is usually closely related to the distribution of aquatic plants which invade the new water body. The Ord River scheme, with its impoundment and irrigation canals, and some other water developments elsewhere in Australia, has given rise to considerable arthropod and vertebrate reservoirs which, if infected, could result in the establishment of endemic areas for some diseases borne by aquatic animals. The proximity of the Ord to endemic areas of south-east Asia has led to concern over the possible hazard of introducing such diseases to the Ord from where they might spread to some other parts of Australia. At present the Ord River environment is free of endemic schistosomiasis, malaria and filariasis, and there is a low human population density in the area.

The following mosquitoes associated with various vector borne diseases favour new impoundments and irrigation works in Australia: the freshwater *Aedes*
(O.) normanensis, and brackish water A. (O.) vigilax (both vectors of RRV), and Culex (C.) annulirostris, which is a vector of MVE, RRV, and Sindbis viruses. A. (O.) vigilax is believed to be best controlled chemically in its larval stage, but care should be taken not to affect the prawns and fish inhabiting the same habitat (Sinclair, 1976). It is important to emphasise, however, the complexities of mosquito ecology. Anophelinae, for example, breed in a variety of conditions, and more than one species may be present in any one habitat. Before starting control of breeding, it is, therefore, of vital importance to determine which species in that locality is the main vector of the disease being studied (Waddy, 1975). Surtees (1975) suggested the following changes in mosquito populations in tropical Australia due to alteration of habitat: formation of an impoundment will result in reduction of the shaded swamp breeder, Anopheles (A.) bancroftii, and its replacement by A. (C.) amictus, both vectors of malaria. Loss of brackish waters would eliminate Aedes (O.) vigilax, vector of RRV. Increased areas of shallow water will increase the abundance of Culex (C.) annulirostris, vector of the MVE, RRV and Sindbis viruses, C. (C.) bitaeiiorhynchus, vector of the MVE, and Anopheles (C.) farauti, vector of both malaria and Wuchereria bancrofti. Springett (1975) suggested management of insect vectors by manipulation of the environment in order to render the area unsuitable for the vectors. Such measures would include manipulation of water levels in reservoirs, replanting or removal of vegetation, and salinity modification. Not all of these may be possible.

The impact of man is manifest in several other ways; space permits consideration of only two more.

Nearly every crop requiring irrigation supports mosquito populations of medical significance, as found in the Ord River scheme (Surtees, 1975). Rice, as the major crop related to water storage and irrigation, contributes to mosquito production on a global scale; Culex (C.) tritaeniorhynchus, associated with rice fields, is present throughout tropical Asia, India and Japan.

Secondly, the introduction and spread of exotic aquatic plants is a factor of significant impact on aquatic vectors of human diseases. Of some importance in this respect are Salvinia molesta and Eichhornia crassipes (see Chapter 8). Both plants are now widespread in Australia, and both are known to be associated with snail and mosquito transfer of parasitic and viral diseases on other continents. For example, the number of mosquitoes greatly increased on the White Nile in the Sudan with the appearance of E. crassipes in that area (Davies, 1958/59). In India, Mansonia mosquitoes are closely associated with E. crassipes, Pistia and Salvinia (Burton, 1959, 1960), and in West Africa with Pistia (Petr, 1968). Chemical control of water hyacinth in Indonesia, which resulted in opening the water surface, led to an explosion of Anopheles (C.) hyrcanus which was followed by spread of malaria (Leentvaar, 1975, quoting Van Thiel).

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Medically important diseases


Introduction
In developed countries the financial and environmental costs of further exploitation of remaining natural water resources are becoming increasingly unattractive. This unquestionably applies in Australia, yet the use of wastewater as an alternative water resource (and as a source of energy and fertiliser) forms no significant part of Australian sewerage schemes. The approach to sewage collection, treatment and disposal has essentially been one involving minimal capital and operating costs and use of the least offensive site. Reuse has not been a major planning consideration. There is, nevertheless, considerable interest in the matter as indicated by several recent Australian studies (Gutteridge, Haskins and Davey, 1976, 1977a,b). This chapter is based on and briefly summarises part of the findings of these studies.

Table 10 gives an estimate of the potential value of wastewater derived from an Australian city of some 100,000 people. The values ignore costs of treatment, transport to reuse site, seasonal storage, and other associated costs, and are for the products of a properly managed secondary treatment plant. The table indicates that possible values range from $1.4 to $8.2 per capita per annum. Yet less than 17 per cent of wastewater resources is presently used. Increased reuse of wastewater from the total Australian population at present sewered ($\sim 10 \times 10^6$) would clearly result in considerable resource reclamation and savings.

Water as a resource from wastewater
As Table 10 indicates, water represents one of the most important resources available within wastewater, particularly

<table>
<thead>
<tr>
<th>Resource</th>
<th>Amount</th>
<th>Value</th>
<th>Form</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water</td>
<td>7000 ML</td>
<td>$700,000</td>
<td>As town water supply replacement for parks, etc. Average charges</td>
</tr>
<tr>
<td>Phosphorus</td>
<td>100 tonne as P</td>
<td>$40,000</td>
<td>As equivalent superphosphate</td>
</tr>
<tr>
<td>Nitrogen</td>
<td>400 tonne as N</td>
<td>$17,000</td>
<td>As equivalent commercial fertiliser</td>
</tr>
<tr>
<td>Methane</td>
<td>600,000 m³</td>
<td>$20,000</td>
<td>As town gas</td>
</tr>
<tr>
<td>CO₂</td>
<td>300,000 m³</td>
<td>$20,000</td>
<td>Cost of oil (if prepared from fuel oil)</td>
</tr>
<tr>
<td>Sludge</td>
<td>200 tonne</td>
<td>$2,000</td>
<td>Soil conditioner</td>
</tr>
</tbody>
</table>
if used as a replacement for town supplies in parks, gardens and sports grounds. Its nutrient content enhances its suitability for such purposes, but limits its use in ornamental lakes. Pathogen and salt content may be further limits. The potential health problems from the possible pathogenicity of wastewater certainly requires care in treatment, disinfection and application of such water. The dissolved salts in reclaimed water—additional to those present in original supplies—originate from a variety of household contributions, from industry, and from groundwater infiltration into sewerage schemes. If present in large amounts, or if excessive concentrations of certain elements are involved, the reclaimed water can deleteriously affect soils and crops.

Table 11 indicates the present pattern of reuse in Australia. Not surprisingly perhaps there is little interest in reclaimed water in Tasmania and the populous, but well-watered, coastal areas of N.S.W. and Queensland. There is also little interest, however, in dry inland areas of N.S.W. where some 400,000 people reside in 88 towns with an annual rainfall < 760 mm. Reclaimed water has been more readily accepted for use in South Australia and Western Australia, Australia’s driest states (though not in the vicinity of Perth which normally has a good winter rainfall, substantial storages, and reasonable underground supplies). Victoria, too, though the best-watered mainland state, has displayed considerable interest in reclaimed water, and through the Reclaimed Water Committee (Ministry of Water Resources and Water Supply), has undertaken specific reuse projects and has plans for further reuse.

A more detailed examination of the pattern of reuse in Australia will give substance to the above general comments.

**Reuse as water for irrigation.** Agriculture, silviculture, parks and other recreational areas, and domestic gardens provide the most usual outlets for reclaimed water at present. In this way, irrigation constitutes the commonest form of water reuse.

Reclaimed water is used for agricultural purposes in over two hundred localities, though specific practices differ from state to state. In Queensland, effluent is

Table 11  Number of examples of effluent reuse in Australia and geographical pattern*

<table>
<thead>
<tr>
<th>Type of reuse</th>
<th>N.S.W</th>
<th>Victoria</th>
<th>Queensland</th>
<th>Western Australia</th>
<th>South Australia</th>
<th>Federal Territories</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pasture</td>
<td>5</td>
<td>30</td>
<td>5</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Vine growing</td>
<td>1</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trees</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sportsground (including golfcourses)</td>
<td>12</td>
<td>8</td>
<td>10</td>
<td>18</td>
<td>16</td>
<td>1</td>
</tr>
<tr>
<td>Mine waste revegetation</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>House garden (two pipe system)</td>
<td>1</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>2</td>
</tr>
</tbody>
</table>

* No examples known from Tasmania.
usually spread by falling contour channels so as to grow rank grass, and grazing by sheep or cattle is permitted after the grass has dried. In Victoria, even quite small towns apply secondary effluent by flood irrigation on to prepared fields which are then grazed, mostly by sheep; whilst primarily a method for disposing of effluent, undoubtedly greater numbers of sheep are grown on such fields than would otherwise be the case. Additionally, the Victorian Health Department has licensed the grazing of beef cattle by eighteen of the larger Victorian sewerage authorities. The best-known example of the use of sewage effluent in agriculture in Victoria (and Australia) is provided by Melbourne’s Werribee Sewage Farm. There, in summer, over 200 ML day\(^{-1}\) of raw sewage is applied to 4,300 ha of redbrown silty-clay loam (Kirby, 1973). Although treatment is the main objective, 15,000 head of cattle are bred and maintained annually and up to 50,000 sheep are fattened. Sales largely pay for the cost of the operation. South Australia, too, provides several instances of successful agricultural use. A substantial proportion of Adelaide’s milk supply comes from a farm where 300 ha of fodder crops are irrigated with 900 ML yr\(^{-1}\) of chlorinated secondary effluent from the Bolivar sewage treatment plant (Cooper et al., 1974; Phillips, 1977). Wine grapes at Angle Vale are irrigated from Bolivar effluent pumped about 11 km on to 280 ha of vines. The same effluent has been used to grow vegetables without the addition of fertilisers (Cooper et al., 1974). However, an investigation by Kinnaird Hill (1976) indicated that the large-scale use of secondary effluent from Bolivar for agricultural irrigation of the North Adelaide Plains was impractical because of the high cost of the chlorination deemed essential.

Only minimal use is presently made of reclaimed water in silviculture. To this writer, however, such a use appears one of the most promising, despite the presence of a number of practical problems. Examples of this use are provided by the wide variety of hardwoods grown reasonably successfully since 1972 under irrigation by secondary effluent at Darwin. Eucalypts have been grown successfully at Alice Springs using effluent to irrigate them. Additionally, brief mention may be made of several experiments underway: experimental forest plots have been planted at Werribee, Horsham, Robinvale and Mildura in Victoria; an existing stand of *Pinus radiata* is being irrigated from the Dutson Downs sewage treatment plant in the Latrobe Valley; and CSIRO have been experimenting since 1973 with the growth of *Pinus radiata* irrigated by secondary effluent at Beenup in Western Australia.

There are rather more examples of the use of effluent in irrigating parks and other recreational areas. Thus, there are at least sixty sportsgrounds (including golfcourses) in Australia which are spray-irrigated with effluent from nearby sewage treatment plants. While more prevalent in hotter, drier areas, they also occur in more climatically favoured areas. For such landscape irrigation, the water used is secondary effluent, usually chlorinated to a level required by the state health authorities. Other health requirements may limit play on sportsgrounds until a fixed time has elapsed, or until the grass is dry, or restrict watering to times when there is little or no play in progress. The most extensive and signifi-
cantal landscape watering scheme is at Adelaide where chlorinated secondary effluent from the Glenelg activated sludge plant is used (Cooper et al., 1974); almost all the plant's output, some 54 ML day\(^{-1}\), is allocated in this way. The appearance of the surrounding irrigated recreational areas indicates the success of the venture despite an effluent salinity above 1,500 mg l\(^{-1}\) and a sodium absorption ratio above 9. Public acceptance appears complete and no disease problems have been reported. With regard to examples elsewhere, there are now some eighteen Western Australian towns with populations > 80,000 people that use secondary sewage effluent on public or school sportsgrounds. Again, results have been successful and no health problems reported (G. Prgomet, personal communication). At Broken Hill, N.S.W. over 40 ML day\(^{-1}\) is spray-irrigated on to parks, tree-belts, and sportsgrounds. Effluent has been successfully used at Broken Hill in these ways for the past thirty years. Effluent is also drip-irrigated on to nearby spoil dumps to aid in the difficult task of re-establishing vegetation cover.

Finally, it may be noted that several examples exist of schemes involving the use of effluent in watering domestic gardens. For the most part these occur in hot and arid areas and are under the control of single mining or other commercial companies (that is, the reclaimed water is not supplied to privately owned houses). Thus, at Broken Hill, N.S.W., the gardens of 44 houses owned by the local mining company are supplied with separately reticulated secondary effluent. Dual municipal water-supply systems, also, could well operate in more arid areas, but, though the construction of a two-pipe system is practicable, such systems would have greater problems of acceptance, management and responsibility than is the case for non-municipal systems.

**Reuse to recharge aquifers.** At present there are no schemes in which any Australian aquifer is deliberately recharged from reclaimed water. However, considerable casual recharge occurs to the aquifer on the Perth coastal plain from sullage and effluent from septic tanks, and water from this aquifer is extensively used to water gardens in Perth. The local authorities are currently monitoring the situation and investigating the possibility of further recharge using secondary effluent from the Westfield sewage treatment plant. The Victorian Mines Department, likewise, are investigating the possibility of recharging the Carrum aquifer near Melbourne with secondary effluent. If the investigation proves successful, a pipeline some 20 km long may be constructed to recharge the extensive but failing Koo-Wee-Rup aquifer nearby (as originally proposed by Bird and Lang, 1968).

**Reuse to maintain streamflows.** At present there are no schemes in which any Australian stream is deliberately flushed or has its flow maintained by reclaimed water. Many examples are known, however, particularly in N.S.W. and Queensland, of streams which have dry weather flows supplemented by the direct or indirect discharge into them of secondary effluents. In some streams (especially in certain city areas), water quality may actually improve from such discharges, but the situation is a matter for concern if the same stream is used downstream as a source of water supplies. In the opinion of this writer, effluents should not be discharged to streams
Wastewater as a resource

at times of low flow in many cases, but impounded or disposed of by land irri-
gation.

**Industrial reuse and reuse as potable water.** Apart from minor usage by the Western Mining Company at Kambalda, Western Australia, where treated efflu-
ent is used for general process treatment purposes, there is at present no known industrial use of reclaimed sewage efflu-
ent in Australia. There are no examples of reuse for potable water supplies, a situation which is unlikely to change rapidly, if ever.

**Energy as a resource from sewage**
Not a great deal of use is presently made of the energy content of sewage. It is true that in Australia most large and many small sewage treatment plants collect methane derived from treatment processes and burn it to heat digesters, and that, in addition, eight large plants generate enough electricity from methane to satisfy their own power needs, but a good deal more energy is available in sewage than is presently used. As energy shortages increase with the accelerating depletion of fossil fuel sources, more of this energy is likely to be utilised. Thus, it is likely that sewage treatment authorities will generate sup-
plies of electricity *surplus* to their own requirements from available methane, digesters will be used to dispose not only of sewage material but also other organic wastes in order to increase methane output, and, instead of using the energy demanding activated sludge process in treatment, there will be a trend towards such low energy systems as secondary treatment in high rate algal ponds or in macrophyte ponds—the crops from which would be added to digesters to increase methane production further. A planned and large-scale development of methane could lead to the development of on-site ‘cracking’ processes to yield methanol, or at least to the direct sale of methane to oil refineries.

One possible form that a future sewage treatment plant could take to maximise the energy resource in sewage is as follows. There would be primary treatment designed for maximum methane production followed by sec-
ondary treatment in macrophyte ponds. The effluent from these would be used to irrigate trees which when harvested would be fed to a pyrolysis plant for hydrocarbon production, or converted to ethanol by acid hydrolysis and fer-
mentation or by enzyme action. Instead of trees, sugar-beet or corn could be grown, the former then being fermented directly to alcohol, the latter converted first to sucrose using enzymes and then to alco-
hol.

**Digested sludge as a resource from sewage**
Thus far in Australia, only minimal use has been made of anaerobically digested sludge as a resource. Yet sludge is a good soil conditioner and new soil ingre-
dient and has significant nutrient con-
tent. It can best be used in semi-liquid form by soil injection, but can also be used after air-drying in mixtures with sand to topdress new recreational areas, repair landscapes damaged by construc-
tional activities such as road-making, and in similar ways. However, as sludge may contain viruses, helminth eggs, protozoan cysts and pathogenic bacteria, prolonged storage before use or care in its use is essential. Care is also required when the use of sludges derived from
certain industrial wastes is being contemplated; the heavy metal content of this sort of sludge, e.g. cadmium, may limit use and, at the very least, may require monitoring after use.

Conclusions

A substantial increase in the use of wastewater as a resource can be predicted. This will arise from greater recognition of the value of the resource and of the environmental advantages involved. The development of clear-cut, sensible guidelines is therefore called for, particularly on a national basis, together with the development of acceptable techniques of treatment, disinfection, control for industrial wastes, and analysis. As part of a revision of attitudes to wastewater treatment and disposal, alternatives to the traditional approach involving centralised sewage treatment plants should now be contemplated and in particular the alternative of regional plants serving to supply reclaimed water for specific purposes. A change in the present approach of sewerage authorities to energy generation should also be contemplated; the likely future situation with regard to energy availability in Australia certainly invites investigation of low energy treatment processes and the deliberate generation of methane in treatment plants.

Further research, the development of demonstration projects, and the formation of a secretariat able to collect, coordinate, collate and disseminate information on reuse activities are seen as priorities in this area.

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15 Water as a waste transport and treatment mechanism: an ecological evaluation

W. D. Williams

Introduction
If the threat posed by the equation $E = Mc^2$ is ignored, the major problems facing mankind are broadly ecological: overpopulation, pollution, increasing shortages of resources, and decreasing food production per capita. In various ways, water resource management has a role to play in resolutions to all these problems. And sewage disposal, as one component of water resource management, has a significant part in this role: increasing human numbers means increasing sewage production; the discharge of sewage to inland waters is a direct cause of pollution in the aquatic environment; and, as Chapter 14 indicates, it is possible to use sewage to alleviate shortages of energy, water and other resources (including food). That water is of central importance in sewage disposal per se scarcely needs stating; suffice it to state only that water is the means whereby almost all human waste in developed countries is carried from its source to treatment facilities, and it is into natural waters that nearly all treated waste is disposed (see Chapter 10, Table 6). The present chapter considers these matters from an ecological standpoint.

Ecosystems. The complexity of the interrelationships between non-living and living material is now recognised by ecologists as the normal order of natural events. Such interrelationships can usually be partitioned into relatively discrete spatial units known as ecosystems or ecological systems. A working definition of these systems, therefore, is that an ecosystem is any reasonably discrete area of the world's surface wherein operate causal and interdependent relationships between living and non-living material. Alternatively, an ecosystem is any functional area of nature with characteristic pathways for energy (bio-energetic pathways) and matter (biogeochemical cycles), and with a characteristic structure.

Though relatively discrete, no ecosystem is completely so; to various degrees, all ecosystems are interlinked and the only truly valid ecosystem boundary is that of earth itself. In a sense, the world is one large ecosystem with many sub-systems. It is not possible, therefore, for man to modify one ecosystem (or sub-system)—e.g. by disposing large volumes of sewage into it—without changes reverberating in others. Moreover, since earth is finite, there is a limit to the number of ecosystems that can be modified without significant impact upon the totality of world ecosystems (the ecosphere).

Ecosystems are not modified by man alone; ecosystem modification is also a
natural phenomenon, and, indeed, ecosystems display many long-term natural changes of a general sort. In brief, and simply, these (1) represent an orderly process of biological development which is directional (i.e. predictable), (2) result from biologically induced modifications of the non-living environment, and (3) terminate in an ecosystem that is stable, maintains the maximum amount of living material per unit of energy flow, and has complex interrelationships (Odum, 1969). The more specific features of such changes are that in the developmental stages of ecosystems ('immature' stages) the amount of energy fixed diverges markedly from the amount actually used in situ, food chains are linear and involve many grazing forms, community structure is simple, nutrient cycles are open, rapid and detritus plays an unimportant role in them, and, finally, overall homeostasis is not well-developed. Odum (1969) stressed the importance of recognising these features since many of them conflict with man's effect on ecosystems. Thus, the overall effect of pollution on ecosystems is to simplify them by reversing the long-term direction of ecosystem development (Woodwell, 1970).

A clear example of man-induced ecosystem modification is seen in aquatic environments receiving sewage effluent; the discharge of significant amounts of plant nutrients, organic matter, and other material to such systems imposes stresses often evidenced by excessive growth of nuisance plants, simplification of biological communities, and other features of ecosystem degradation. As pointed out by Commoner (1970), although sewage treatment is one of the better-developed technologies, failure to appreciate the ecological repercussions of the technology has resulted in gross pollution of many surface waters. The basic flaw has been an inability to appreciate (or an unwillingness to accept) that the end problem, that is the recycling of the waste, is simply transferred from one ecosystem to another—and usually to one unable to cope in adequate fashion. The solution is that waste discharge must largely become an internalised process that is not allowed to extend into other ecosystems. More directly, the most ecologically sensible way to deal with sewage is to recycle it on land. The same point has been made by several other authors (e.g. Westman, 1974; Lvovitch, 1977) but perhaps most succinctly by Vallentyne (1974) in his diagrammatic comparison of ecological and technological patterns of waste disposal (Fig. 20). It is stressed, however, that the point is a general one, for, obviously, the ecological effectiveness of land disposal of wastes depends upon local soil types, climate, wastewater composition, and other factors. In a given local situation it may be that land treatment is not ecologically the best method of treatment. Nevertheless, land treatment is ecologically the most desirable goal overall.

**Biogeochemical cycles.** An integral feature of ecosystems is the movement within them of essential elements (elements needed by living material). This movement is along pathways that are characteristic for a particular ecosystem but in all ecosystems are basically circular; were they not so, then clearly it would only be a matter of time before the sources of essential elements became depleted. As it is, essential elements are recycled, the pathways followed being
termed biogeochemical cycles. Rates of natural recycling vary according to element and ecosystem, as also does the sort of reservoir within which a major stock of the element is retained. However, for most though not all essential elements the pathways are relatively short and the rate of circulation rapid.

When man modifies ecosystems he usually disrupts biogeochemical cycles so that rates of element movement are altered and cycles broken. The long-term effects of such disruption need little ecological knowledge to predict: the ecosystem is ultimately destroyed. This, then, is a second important point when considering the fundamental ecological nature of present methods of sewage treatment and disposal. The main questions are to what extent does present sewage technology breach biogeochemical cycles, how important are the breaches, and how may they be ameliorated or, better, plugged. Questions of this sort are infrequently, if ever, posed by those directly involved in sewage treatment and disposal, yet are clearly questions of considerable consequence.

It is true that ecologists are aware of the important role now played by sewage as a biogeochemical agent, but a comprehensive evaluation of this role has yet to be attempted. In fairness, it should be added that it is a matter of concern amongst many ecologists that adequate quantitative data on flow rates and pool sizes within the biogeochemical cycles of all essential elements are still largely absent. Ecologists recognise the need to pursue this matter (e.g. Institute of Ecology, 1972), but seem to accord less recognition to measures such as land disposal of sewage which, even in the absence of precise data, clearly represent practical techniques of maintaining unbroken biogeochemical cycles for several elements.

Effects of sewage on the aquatic environment
Having considered the ecological impact of sewage disposal in general fashion, the more immediate ecological effects of sewage in the aquatic environment should now receive brief attention. Before giving this, however, it may be
Water usage

noted that the crux of the problem concerning the disposal of sewage into rivers, lakes, estuaries and the sea is that these environments belong to no one person or group of persons, and thus the 'tragedy of the commons' applies (Hardin, 1968). That is, so far as waste disposal is concerned, the shared cost to any particular discharger of the results of communal waste discharge is less than the cost of purifying wastes before discharge or of not discharging them at all.

**Effects on rivers.** The principal environmental effect of sewage discharge to rivers is to decrease dissolved oxygen concentrations. This results from the heterotrophic bacterial breakdown of organic matter in sewage. Other effects are mostly corollaries. The most important is that the normal, non-pollutional biota is eliminated or impoverished because of its inability to withstand low oxygen concentrations. The lost biota is replaced by a more tolerant but less diverse one which, however, has a much greater biomass because of increased nutrient input.

An important feature of the discharge of sewage effluents to rivers is that within limits rivers are capable of assimilating the material discharged. That is to say, from a point source of sewage addition, there is first some environmental degradation but then recovery downstream occurs until conditions similar to those originally present re-occur. This phenomenon lies at the basis of the various zonations of polluted rivers derived by freshwater biologists (see Bayly and Williams, 1973; Alabaster, 1977). Under certain conditions, therefore, little ecological damage would result from sewage discharge into rivers. However, the vast amounts of sewage, the diffuse nature of sewage discharge, and the inability of rivers to cope with excessive volumes of organic material now preclude rivers in most cases from serious consideration as ecologically acceptable sinks. It is true that standards (see Chapter 11) have been erected to protect rivers from environmental damage (cf. Hart, 1974)—and these are certainly welcome as short-term attempts at protection—but their adoption should not be allowed to camouflage the fundamental ecological truth that rivers do undergo environmental degradation when abnormal amounts of organic matter are added, and that the use of rivers as a major waste disposal mechanism is an unnatural one. Ideally from the ecological viewpoint, therefore, sewage treatment and disposal technology should aim to eliminate the use of rivers as 'inexpensive' natural drainpipes or, when such use is absolutely necessary, to add to them water of a quality not worse than that already present.

**Effects on lakes.** Similar environmental effects from sewage discharge occur in lakes, but of course their spatial distribution is different. Generally more important and widespread in lakes are effects resulting from the presence of plant nutrients in sewage discharges, particularly phosphates and nitrates. These nutrients frequently stimulate algal and macrophytic growth to such an extent that a number of deleterious features develop. The phenomenon is known as eutrophication, or cultural eutrophication, and is more fully discussed in Chapters 7 and 9. Wood (1975) has given a recent and comprehensive review of the phenomenon in Australia.

Measures for controlling eutrophica-
Water as a waste transport and treatment mechanism

159

tion are basically of two kinds: the prevention of nutrients from entering a lake and remedial measures within the lake (see Chapter 9). Clearly the former is ecologically the most sound, but its implementation is by no means simple. The complete removal of plant nutrients from sewage effluents by physico-chemical means is complicated and expensive and biological removal methods are at present imperfect. Since the water transport system of waste disposal is the root means whereby the causative agents of eutrophication are allowed expression (Vallentyne, 1974), it is logical to suggest that a further control measure for eutrophication bears examination—the abandonment of water as a waste transport medium, or at least its abandonment beyond the point of sewage treatment.

Effect on estuaries. Because estuaries, almost by definition, are aquatic environments regularly subject to tidal flushing, the development of conditions akin to lake eutrophication in estuaries seems hardly possible. However, such is not the case, and there are many estuaries throughout the world into which sufficient sewage is now discharged to overwhelm even the daily flushing of those that are 'typical' estuaries (and it may be noted that there are many estuaries where daily flushing is more an assumption than a fact).

'Eutrophication' is not, of course, the only effect that the discharge of sewage may have upon estuaries; others include the development of polluted conditions similar to those which develop in rivers receiving sewage effluents. No account is needed here, but again the point emerges that the discharge of sewage effluent to estuaries is not a satisfactory ecological solution to the problem of human waste disposal and treatment (Cronin, 1967; Thomson, 1973). Such discharge is certainly a convenient, inexpensive and usually sanitary way of dealing with wastes, but it should be seen unequivocally as the sideways movement of a problem from one ecosystem to another.

The extent of the ecological damage so far caused to estuaries by their use as waste sinks has yet to be determined. Indeed, so little is known of the basic functional ecology of estuaries (see, for example, Lauff, 1967; Perkins, 1974)—in spite of their known value as nutrient traps and therefore as very productive ecosystems—that we are far from any position from which to determine the damage. The impact of man other than as a source of organic waste is a further complicating factor, and one which can scarcely be over-rated (see, for example, Australian UNESCO Committee for Man and the Biosphere, 1974).

Effect on ocean. Part of the rationale behind the discharge of sewage to estuaries is that they provide natural 'pumps' to the 'ultimate' environmental sink, the ocean. There was a time when the ability of that environment to assimilate and deal with sewage and other wastes added was regarded as limitless, but that time has now gone; the majority of marine ecologists and oceanographers recognise that there are limits. Thus, in the recent symposium on marine pollution held in London (GESAMP, 1975), at which the scientific criteria for the selection of sites for dumping wastes into the sea were thoroughly discussed, there was a clear recognition that marine disposal was to be regarded as a last resort rather than a favoured one.

The philosophy of the symposium
coincides with the ecological views expressed in this chapter, and may usefully be reproduced as an appropriate conclusion:

Ideally, the only ultimate method of eliminating waste disposal of conservative substances is recovery and re-utilisation of the materials presently considered to be wastes; other disposal operations merely move material from one part of an environment to another. The decision to consider a substance a 'waste' rather than a potential 'natural resource' is based on economic rather than on scientific principles, because the technology to recover the material in useful form is either not available, or is more costly than the value of the recovered product. For certain wastes, and under particular circumstances, the cost of disposal at sea may be less than that of recycling or of disposal on land, but the cost must be assessed against the risk and cost of damage to marine resources.

References


PART IV

The impact of man on inland waters
Conservation

P. S. Lake

Introduction

In this chapter, conservation is defined as the management of natural, man-influenced or man-made components of the environment such that essential features are not degraded or destroyed. The oft-expressed view that conservation is not preservation is rejected, in that to conserve many features effectively, for example a rare species or a unique ecosystem, preservation is often the essential and only strategy. Of course, management does not necessarily mean active intervention by man; it may simply involve a 'leave alone' strategy. Management of components of the environment to allow exploitation of a particular resource as a commodity is here regarded not as conservation but rather as resource husbandry. Thus, conservation aims to maintain an ecosystem in its natural or intact state so that the essential attributes of the original system are not degraded or destroyed by exploitation of a particular resource.

Because Australia is the world's driest inhabited continent, it is not surprising that the need for 'water conservation' has provided powerful motivation for the creation of numerous artificial storages and river impoundment schemes. This, however, is resource husbandry for it deals with water as a commodity. Indeed, the construction and operation of many impoundments has often been at odds with the conservation of valuable environmental components; it was the need to 'conserve' water for hydroelectricity generation that led to the destruction of the unique ecosystem of Lake Pedder (Australian Conservation Foundation, 1972; Burton et al., 1973, 1974), and the construction and operation of the Eildon reservoir to 'conserve' water for irrigation, likewise, has jeopardised the mobility of native fish populations in the Goulburn River system—both upstream (Cadwallader and Rogan, 1977) and downstream (Pollard and Scott, 1966; Williams, 1967).

Although arid, Australia has a great variety of inland waters, with a diverse and distinctive aquatic biota (see Chapter 2). Many features of this aquatic environment—both abiotic and biotic—are well worth conservation. However, comprehensive knowledge of Australian inland water bodies and their biota is lacking, and often the only available information on a particular water body is related to exploitative aspects. For many an Australian river there is considerable information on flow volumes and pattern over long periods, but biological knowledge remains anecdotal. Frequently, in deriving environmental impact statements for a particular project, knowledge of the biota is accumulated while the project is actually in course of construction, not in the planning stages (cf. Chapter 17). Moreover, whereas some species of fish and waterfowl have attracted attention from a conservation viewpoint, the creation of reserves has
largely been to conserve terrestrial features—especially those of high scenic and/or recreational value. It is true that water bodies are included in a number of Australian National Parks, but their conservation was rarely the reason or even one of the reasons for creating the reserves, and often the management of such water bodies leaves much to be desired. Bonython and Frith (1974) have pointed out the poverty of national parks and reserves associated with the Murray-Darling system, and noted the poor representation of many riverine plant communities, e.g. red gum forests, in those few areas so far set aside for conservation. It is to be hoped that with increasing public concern for the environment, the need to conserve inland water bodies effectively will be properly realised.

The preceding remarks will indicate that much basic knowledge required for a proper basis to the conservation of Australian inland waters and their biota is lacking, and the public (and governmental) attitudes towards conservation leave considerable room for improvement. In order to be constructive in this area, this chapter will discuss the values of conservation, indicate broadly the nature of those phenomena inimical to the conservation of Australian inland waters and their biota, and provide some guidelines for effective limnological conservation.

The case for conservation of inland waters and their biota

**Economic value.** Inland water bodies and their biota have an immense array of realised and potential economic values and uses. Many are dealt with in other chapters of this book. The economic benefits derived from the use of water bodies may be either direct or indirect. Thus to keep a water body unpolluted allows use of the water for human economic needs—the benefit is direct. However, keeping a water body unpolluted may maintain aesthetic values and thus promote economic activities such as tourism—an indirect benefit. Conserving stocks of certain biota, e.g. fish, may be pragmatic (Myers, 1976) and of direct economic benefit now, but conservation of some presently non-commercial species is not without benefit for many will have future economic value. Currently in Australia only three species of parastacid crayfish have commercial value, but given the likely expansion of aquaculture in Australia (see Chapter 21), and the prodigious parastacid radiation into a greater variety of habitats, it is likely that in the future more species of parastacid crayfish will have an economic value.

**Scientific value.** By conserving biota in natural ecosystems, a rich store of genetic material is maintained for scientific study and possible future use (Frankel and Bennett, 1970). The ecosystems themselves provide systems that can be studied to derive ecological guidelines which can form the basis of sound environmental management (Moir, 1972). They also serve the vital role of acting as baselines for environmental monitoring, and thus allow an assessment of man's environmental impact (Moir, 1972; Jenkins and Bedford, 1973). Additionally, natural ecosystems have many features of immense intellectual curiosity. Although the curiosity of professional
scientists usually receives public recognition and support, it should not be forgotten that we all possess some intellectual curiosity about the natural world and that the curiosity of amateurs is as viable as and important as that of professional scientists.

**Educational value.** The teaching of the natural sciences at all levels relies on first hand experience of study material. In the earth and life sciences, this means access to natural areas. Clearly, therefore, the teaching of such subjects and their value as an educational experience declines if areas or features of interest are damaged or destroyed. This applies, above all, in the teaching of ecology, a subject which is increasingly part of a liberal education (Ratcliffe, 1976); ecology simply cannot be taught without access to natural field situations. In a wider sense, it is important to note that conservation of particular areas serves to heighten public awareness of the natural world and threats to it. It may also serve to educate in a political sense: Smith (1977a) made the point that attempting to set aside an area for conservation (whether successful or not) often serves to educate the public in the ways that decisions are made in our society. There is no doubt that the Australian public gained great insight into the deficiencies of decision-making procedures when the chain of events which led to the destruction of Lake Pedder became known.

**Recreational value.** The recreational value of inland water bodies is considerable and is realised in a multitude of activities (Bayly and Williams, 1973). These activities range from rather passive ones, e.g. painting, photography, to active ones, e.g. fishing, swimming, water skiing. Many recreational activities depend upon the water bodies being relatively intact, while others, such as speedboat racing, may take place on any suitable expanse of water, natural or otherwise. Some, it should be pointed out, may actually damage ecosystems. Thus power boating may greatly disturb waterfowl. Even certain sorts of fishing, indirectly, can be destructive. At the behest of the angling fraternity, trout have been introduced into many remote water bodies. There is evidence that this has had a deleterious impact upon the native fish fauna (see Chapter 23).

**Spiritual, enlivening and inspirational values.** There is no doubt that the contemplation of aesthetically pleasing water bodies is emotionally evocative for many people. Lakes and rivers, ponds and streams have served to inspire painters, photographers, poets, writers and the religious. While such a value is not readily quantifiable or discrete, it should not be dismissed. Areas of natural landscape, natural waterscape, because of their low population density and their diversity of form and mood, can fulfil the need to soothe people under stress in urban-industrial complexes (Hutchinson, 1963). Moreover, because many natural water bodies are either remote or in natural enclaves in man-affected land, they represent an approach to wilderness. Being on or around such water bodies may engender some of the feelings or values outlined by Smith (1977a) and termed by him ‘individually perceived values’. These include ‘freedom’, ‘challenge’, ‘romance’, ‘solitude’, ‘perception’, ‘inspiration’ and ‘spontaneity’.
Cultural and identity-enforcing values. Plants and animals, as Myers (1976) correctly noted, are part of our heritage. The extinction of any one of them represents the loss of part of our cultural heritage. The same applies to water bodies considered as a whole. Because human settlements tended to develop along water bodies, many lakes and rivers have considerable historical and cultural value. This is not so obvious perhaps in Australia where cultural and historical links with particular water bodies are not strong, but it is there nonetheless, especially if the cultural values of the Australian Aboriginal people are recognised. Sadly, many Australian water bodies of sacred and historical significance to the Aboriginal people have been desecrated and damaged by European man (Committee of Enquiry into the National Estate, 1974).

Events detrimental to the conservation of inland waters and their biota

The construction of impoundments. The need to conserve water for human use has resulted in the creation of a large number of impoundments and reservoirs in Australia. These invariably change the nature and flow pattern of rivers on a daily and seasonal basis, and such changes may adversely affect the distribution and breeding of native fish and waterfowl (Williams, 1967; Wharton, 1969; J. S. Lake, 1971, 1975; Bonython and Frith, 1974; Frith, 1973). For example, differences in thermal regime caused by release of hypolimnial water can adversely affect the spawning and distribution of fish; in summer, the outflow from Eildon reservoir may be 15°C lower than inflow water and this has favoured the replacement of native warm-water fish species (e.g. Murray cod) in the Goulburn River by introduced cold-water species (e.g. trout). The impact of Lake Hume on the fish fauna of the River Murray downstream is discussed in Chapter 18.

Impoundments have also caused a reduction in the extent and frequency of major floods. A reduction in the area of temporary wetlands has followed with consequent deleterious effects on native fish and waterfowl. This reduction, as well as the disruptions to the natural thermal regime, has disadvantaged the breeding of such native fish as Murray cod, golden and silver perch according to several authors. Additionally, the creation of impoundments may destroy unique or valuable freshwater habitats. The destruction of the unique Lake Pedder ecosystem by the Serpentine dam is a case in point.

Further impacts involving impoundments relate to the imposition of physical barriers to migration, the occurrence of unnatural water-level fluctuations, and alterations to natural drainage patterns. With regard to the imposition of physical barriers it is noted that in the Murray-Darling system extensive fish movements have been found to coincide with flooding (Reynolds, 1976), yet there are but two fishways in the entire system (Wharton, 1969). Unnatural water-level fluctuations occur both within impoundments and downstream. They may depress production of macrophytes and invertebrates (e.g. Hunt and Jones, 1972) and damage fish populations which breed in shallow water over submerged vegetation (e.g. western carp gudgeon; J. S. Lake, 1975).
The biological effects of alterations to drainage patterns have not been studied in Australia, but it is obvious that they provide a mechanism whereby natural zoogeographic barriers may be broken down (Butcher, 1967).

**River improvement and drainage works.** A list of techniques for river improvement and drainage applied in Australia is given by Strom (1962) and Turnbull (1973, 1974). They include channelisation, canalisation, levee construction, wetland drainage, bank stabilisation and desnagging. While such techniques have been accepted in the past, there is now considerable resistance to their continuation (e.g. Wharton, 1969; Rogan, 1977; Conservation Council of Victoria, 1977) because many have been carried out with little or no consideration for detrimental effects on stream biota. Kendrick (1976), for instance, described the transformation of a stable, pooled, section of the Avon River, Western Australia, into an unstable sandy section by so-called river improvement. In urban areas, likewise, streams are often needlessly degraded into concrete-lined drains, yet, as outlined by Brotchie (1977), it is possible by careful planning to have urban development and maintain conservation values for many urban creeks and their environs.

**Mining operations in rivers and sand mining.** Extraction of gravel and sand from river beds is carried out in many parts of Australia. In Victoria, past gold dredging and hydraulic sluicing has had ‘a totally destructive effect on the stream bed’ (Wharton, 1969). Besides disturbing the stream bed at the site of operations, the discharge of highly turbid water adversely affected downstream biota. Coastal dune lakes are particularly susceptible to damage by sand mining. While sand mining enterprises pay considerable attention to the ‘rehabilitation’ of vegetation on mining areas, no techniques have evolved for the restoration of damaged dune lakes. It is to be hoped that the high dune lakes of Fraser Island (Bayly, 1974; Bayly et al., 1975) remain safe from mining and that the lagoons of the Cooloola sand mass (Bayly, 1975) and of the Myall Lakes area are protected.

**Changes in catchment usage.** Chapter 10 discusses in detail the nature of the relationship between catchment usage and inland water resources. Here, it is necessary to note only that this relationship is close and that the impacts of many changes in catchment usage on inland waters are significant and usually deleterious. Unnatural hydrological changes, increased siltation, elevated salinities, higher stream temperatures, and eutrophication are amongst the more obvious deleterious impacts. There are many more.

**Pollution.** Pollution has many definitions, but an appropriate one here is that pollution is ‘the addition by the operation of human activities of materials and/or energy into an ecosystem in amounts above those to which that ecosystem is adapted to accept or deal with’ (P. S. Lake, 1974). The overall result of water pollution, transitory or long-term, is to change aquatic ecosystems in a detrimental way. Changes include the development of a ‘pollution biota’, the decline or elimination of populations of naturally occurring species, a loss in the species richness and
diversity of the ecosystem, and the creation of inefficient pathways for energy and nutrient transfer in the ecosystem. Many documented examples of water pollution in Australia have been given by Weatherley (1967), Senate Select Committee on Water Pollution (1970), Bayly and Williams (1973) and Connell (1974).

**Introduction and establishment of exotic biota.** Numerous species of freshwater fish have been introduced into and established viable populations in Australian inland waters (see Chapter 23 for details). The effects on the native biota are largely unknown, but undoubtedly include (1) a reduction in native fish and invertebrate populations by predation, (2) a reduction in native fish populations by competition for habitat and food, (3) habitat alteration, and (4) introduction of diseases. Circumstantial evidence indicates the actual occurrence of some of these effects, as outlined in Chapter 23.

Numerous species of aquatic plants, too, have been introduced (see Chapter 8). Of these the water hyacinth *Eichhonia crassipes* and the water fern *Salvinia auriculata* are regarded as serious pests. Both are noted for their ability to grow rapidly under favourable conditions and to block waterways and blanket lentic water bodies. In addition, many water bodies, especially streams, have been altered by the growth of introduced terrestrial plants (P. S. Lake, 1974; Douglas, 1977): many streams are clogged by willows or cloaked by dense growths of blackberries.

**Exploitation of native species.** In only a few cases where a native species has declined in numbers is there direct evidence that over-exploitation was a cause of the decline. There is a number of cases, however, where circumstantial evidence suggests this. In the Murray-Darling system overfishing may have contributed to the decline of some native fish species (Pollard and Scott, 1966). For Macquarie perch stocks in Eildon reservoir, Cadwallader and Rogan (1977) concluded that angling pressure during the upstream spawning migration of the fish did markedly impair the stocks. In Tasmania, overfishing has been blamed for the decline in stocks of the Tasmanian whitebait, *Lovettia seali* (Blackburn, 1950), though undoubtedly a contributory factor has been the increasing damage, especially by pollution, inflicted by man upon the rivers concerned. The decline in stocks of the giant Tasmanian freshwater crayfish *Astacopsis gouldi* has also been partly attributed to overfishing (Lynch, 1969). Finally, both the widely distributed saltwater crocodile (*Crocodylus porosus*) and the endemic freshwater crocodile (*Crocodylus johnstoni*) have been greatly over-exploited in northern Australia (Worrell, 1966; Frith, 1973; Webb, 1974). Unfortunately, whereas both species have been afforded protection in the Northern Territory and Western Australia (Frith, 1973), no protection is given in Queensland (Webb, 1974).

**Some guidelines for limnological conservation**

The following guidelines for conservation as applied to the Australian inland aquatic environment are offered in an attempt to be constructive in an area frequently clouded by emotion, vested interest and subjectivity. They are certainly not exhaustive but if followed would provide a more rational basis for conservation of Australian aquatic
ecosystems and their biota than has frequently been used in the past. The order of listing has no connotations about priority.

Inland water bodies with unique biological or abiotic features should be conserved. Where an inland water body or ecosystem is inhabited by a unique species or community, or has unique abiotic features, all efforts should be made to conserve the system. The need for such a guideline is exemplified by the comparatively recent loss of two unique lacustrine ecosystems in Tasmania. Lake Pedder in south-west Tasmania with its unusual beach of white quartz sand and with eighteen endemic species was destroyed by impoundment. The interesting ecosystem of nearby Lake Edgar (see Knott and Lake, 1974) was lost at the same time. Although the effects of the flooding on the biota of Lake Pedder have been described as exaggerated (Spry, 1976), investigations of the Serpentine impoundment by the author for the past three years have revealed only three of the eighteen endemic species. In like manner, the Lagoon of Islands on the Tasmanian Central Plateau harboured a unique community of floating mats (islands) of vegetation (Tyler, 1976), but is now being destroyed by impoundment.

Greater attention should be paid to the conservation of rivers and streams. In general, the conservation of Australian rivers and streams has received little attention. Many have already been badly damaged by European man, and yet they do have distinctive attributes. This neglect applies particularly to natural, permanent rivers. Indeed, in the list of
water bodies proposed for conservation in *Project Aqua*, no rivers were listed (Luther and Rzoska, 1971). The situation is that so far in Australia the conservation of wild and scenic rivers has been almost totally ignored. In the United States, on the other hand, the need to conserve rivers has been clearly and forcibly recognised (U.S. Congress, 1968). Of Australian rivers presently threatened by impoundments, brief mention may be made of the Colo, Wollang and Crab Tree Rivers in the Colo Wilderness, New South Wales (Dixon *et al*., 1977), and the dramatically beautiful and biologically interesting Franklin River in Tasmania (Ashton, 1977; Rolls *et al*., 1977; Smith, 1977b). Conservationists are struggling to conserve all these rivers, for the number of wild Australian rivers left to conserve is rapidly decreasing.

**River management should be compatible with the aims of conservation.** There has been a tremendous emphasis in Australia on ‘water conservation’ with a proliferation of impoundments and irrigation schemes. In the major river system for which such schemes apply, the Murray-Darling system, the operation of impoundments has not been compatible with conservation of the system’s native biota; operations have changed natural flow and temperature regimes with adverse effects on such biota as native fish, waterfowl and river red gums (e.g. Australian Conservation Foundation, 1968; Wharton, 1969; Frith, 1973). In this system (and others) some attempts should be made to allow pre-regulation conditions of flow and thermal regime occasionally to re-occur. And overall there is a clear need in Australia for river management to become multi-objective and thus allow satisfactory resolution between various competing needs for water.

**Wetlands and dune lakes need more effective conservation.** Wetlands, both permanent and temporary, are of limited extent in Australia. Since most occur in well-watered, fertile areas, they have been subjected to considerable human interference (Frith, 1973; see also Chapter 25). Flood mitigation, ‘river improvement’, and drainage have all taken their toll. Thus, in coastal New South Wales Goodrick (1970) estimated that 60 per cent of prime wetland has now been drained, and in Western Australia Riggert (1966) estimated that 55 per cent of the wetlands of the Swan coastal plain has also been lost. Further loss should be prevented.

Finally, it should be noted that the coastal dune lakes of Australia are of considerable limnological interest (e.g. Bayly and Williams, 1973; Bayly, 1975; Bayly *et al*., 1975). They appear to be particularly susceptible to damage, yet many have already been interfered with by man’s activities—ranging from sand mining to drainage (see Timms, 1977). They require special care from the viewpoint of conservation.

The threat to rivers of the Colo Wilderness has gone. The N.S.W. Government in 1979 declared the area a National Park. In October 1979, the Hydro-Electric Commission, Tasmania, sought approval by the Tasmanian Parliament of the Gordon River Power Development Stage Two. This scheme, if allowed to proceed, will flood most of the Lower Gordon River and Franklin River valleys.
References


Introduction
Decisions on building reservoirs are based essentially on economic and political considerations. However, water resource managers and the public have begun to accept the view that projects for increased water storage should meet certain environmental standards, and should cause minimal damage to aquatic communities. Biologists have an important role to play in encouraging this view for it is essential to recognise natural aquatic communities as an integral part of water resources, and the preservation of such communities as an important management priority.

In this context, investigation and monitoring of the aquatic biota carries its own justification. The benthic invertebrates are particularly important in this regard, as discussed at length in Chapter 19. Here it is noted only that because benthic invertebrates are relatively sedentary, their community structure and composition reflects both short-term, acute conditions, and long-term chronic ones. Monitoring such communities therefore helps to give an integrated picture of overall river conditions. However, as noted also in Chapter 19, a better understanding of the taxonomy and ecology of aquatic species and communities in Australia is needed in order to be able to make full use of their potential as biological indicators. Moreover, accurate predictions concerning the effects of certain practices upon the biota need an understanding of species requirements and limiting factors. Well-designed field monitoring programs can be invaluable in adding to this taxonomic and ecological understanding.

The present chapter describes, as a case history, the background to and the results of a survey of aquatic invertebrates in an area to be profoundly modified by reservoir construction: a major reservoir, Dartmouth Dam, is being constructed in the area. This account is necessarily brief; more details are given by Smith, Malcolm and Morison (1978) and State Rivers and Water Supply Commission (1978).

Background to the survey
The decision to build Dartmouth Dam, on the Mitta Mitta River in north-eastern Victoria, was made in March 1972, and construction of access roads commenced immediately. There was no environmental impact study at the time of the decision but, in June 1972, the River Murray Commission agreed to fund a wide-ranging environmental survey. Its objects were to document pre-impoundment conditions in the inundation area, and to establish base-line conditions downstream to Lake Hume (~100 river km) (Fig. 21). A study of aquatic invertebrates by the National Museum of Victoria formed part of this survey, along with related surveys by other groups (vegetation and terrestrial vertebrates of area to be inundated, fish
and water quality of Mitta Mitta river system, terrestrial weeds and vermin of Dartmouth catchment).

First stage studies of the aquatic biota were largely descriptive. However, because physico-chemical and biological monitoring began before significant construction activity, the continuing program will provide a unique and continuous record of the impact of various stages in the building of a reservoir upon aquatic biota.

The objectives of the total aquatic invertebrate survey were basically threefold: to describe existing invertebrate communities; to provide predictive advice on the impact of particular operational and constructional practices upon the aquatic invertebrate community; and to monitor the actual effect of these practices. The monitoring program itself had two aims—in the short-term to detect and advise construction authorities of adverse effects of particular activities, and in the long-term to accumulate information on the biological res-
The impact of man on inland waters

Responses of species and communities to given physical and chemical conditions.

Results thus far

Description. Despite changes to the river fauna caused by various past human activities, the base-line study showed that the Mitta Mitta River above Mitta Mitta township (~25 km downstream of the dam) still supported a rich and diverse aquatic fauna, and well over 250 species were collected (Smith et al., 1978). This fauna was similar to that in most relatively unmodified rivers of the Victorian foothills. Numerical dominance of the aquatic invertebrate communities, within and below the inundation area, was shared by a large number of species, with many others represented by a very few individuals. Many of the originally dominant species are known to be favoured by either summer-warm but well-oxygenated water, or fast currents, or clean, solid, surfaces or unsilted gravel to cobbles (or any combination of these factors).

Downstream of the Mitta Mitta township, the river runs from forested hill country into cleared farmland, and an appreciable drop in diversity was then evident, as was the tendency for more tolerant, widespread species to become numerically more important. Still-water habitats, such as lagoons and swamps, yielded an even less diverse collection; about 80 species were recorded of which some 30 were common to river collections. Many of the shared species were not abundant in river samples.

Prediction. Some basic predictions are obvious. Terrestrial communities in the inundation area will be destroyed and, except for some still-water species, so will aquatic communities within the same area—a river distance of about 50 km. Thus, over the area as a whole a marked reduction in diversity will occur. Two diverse and totally unlike communities will be replaced by the much less diverse community which inhabits deep lakes and reservoirs. This is an absolute effect and will largely be unaffected by reservoir management. However, in the case of still-water faunas, management practices can still be important: the prevention of rapid fluctuations in water level, for example, would allow the development of a rich and stable fringing vegetation, and thus the presence of a favourable habitat for many invertebrates.

Management practices are even more important in determining the impact of the impoundment upon the downstream biota. This biota will be affected by changes to physical regime (particularly temperature and flow patterns, including current speed), water quality (e.g. levels of dissolved oxygen, turbidity, plant nutrients), substratum type (additional sediment loads during construction and a changed flow regime during operation may alter this), and diversity in the types of organic food material (especially a reduction in this). All of these changes can be influenced by the way the impoundment is operated or managed.

The changes downstream will have a variety of effects, many of which thus far are not well-defined in physical and chemical terms, let alone in biological terms. Nevertheless, clearly many selective species, with specific environmental requirements, will be adversely affected. Such species will include many of those originally common in the Mitta Mitta River and which favour summer-warm and well-oxygenated water, fast
currents, and clean solid surfaces or unsilted gravel to cobbles, or any combination of these conditions. Conversely, many of the more tolerant species will be favoured by some or all of the new conditions, and by the reduction of competition and predation.

Predictably, therefore, the overall effect of riverine changes downstream of impoundments will be a reduction in faunal diversity. Such an effect is welldocumented in several studies overseas (e.g. Pfitzer, 1954; Hilsenhoff, 1971; Ward, 1974; see also Hynes, 1970a,b; Cummins et al., 1972; and Ward, 1976), and appears also to be common in Australia according to Aldenhoven (1975) and observations of the present author.

In many cases, particularly when hypolimnial water has been released, changes in the temperature regime have been proposed as the main cause of the extirpation of many species (Hilsenhoff, 1971; Lehmkuhl, 1972; Ward, 1974). In any event, Nebeker (1971) on the basis of laboratory studies, and Aldenhoven (1975) who worked on a small natural lake in Victoria, have shown that changes to long-term temperature patterns of only a few degrees can be critical in disrupting the normal life cycles of many insect species. Even water released from dams with surface or mixed level outflows may moderate natural seasonal extremes of temperature. This may disrupt thermal signals required by some species to trigger particular steps in the life cycle. Because large summer releases from Dartmouth are planned in very dry years, the depth of the release point, the degree of mixing, and the resulting temperature are likely to be critical in determining impact on the invertebrate fauna. At the very least, those stoneflies requiring summer-warm conditions and other insects with similar requirements are likely to be eliminated by high summer flows of cool water.

Release of hypolimnial water can also have very adverse effects on water quality (Pfitzer, 1954; Ingols, 1957, 1959; Hilsenhoff, 1971), with marked fluctuations in dissolved oxygen and addition of nutrients creating effects similar to those of organic pollution on downstream biota. The constructing authority at Dartmouth has made provision for possible future construction of a multi-level outlet, and is also experimenting with destratification as a possible means of overcoming some of the problems created by hypolimnial water.

The overall trend created by the other likely changes is towards decreasing environmental diversity available to the invertebrate fauna, with a resulting decrease in diversity of the fauna itself (Hynes, 1970a,b; Cummins et al., 1972; Ward, 1976).

**Monitoring.** Baseline conditions were assessed from a series of quantitative samples collected in the summer of 1973-74 and from several qualitative samples obtained at various times over eighteen months.

Quantitative and semi-quantitative collections during the construction period showed that by spring 1976 a dramatic change in the benthic fauna had occurred; this was maintained until dam closure in November 1977 (see Table 12). The change involved a considerable reduction in species diversity, and a marked change in the species composition of invertebrate communities below the dam site. Thus, many previously
Table 12 Abundance of principal macroinvertebrate fauna of Mitta Mitta River before and after construction of Dartmouth Dam. Data do not represent any single set of results, but are an approximate summary. Abundance shown as: +, rare; ++, uncommon; ++++, moderately common; +++++, common; ++++++, abundant and dominant.

<table>
<thead>
<tr>
<th>Fauna</th>
<th>1974*</th>
<th>1977 upstream of dam site</th>
<th>1977 downstream of dam site</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nematoda</td>
<td>++</td>
<td>++</td>
<td>+++</td>
</tr>
<tr>
<td>Oligochaeta</td>
<td>++</td>
<td>++</td>
<td>+++++</td>
</tr>
<tr>
<td>Mollusca</td>
<td>Lymnaea tomentosa</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Acarina</td>
<td>Frontipoda sp.</td>
<td>++</td>
<td>++</td>
</tr>
<tr>
<td>Odonata</td>
<td>Aeshnidae</td>
<td>+++</td>
<td>+++</td>
</tr>
<tr>
<td>Ephemeroptera</td>
<td>Centroptilum sp.</td>
<td>++++</td>
<td>+++</td>
</tr>
<tr>
<td>Plecoptera</td>
<td>Trinotoperla nivata</td>
<td>++++</td>
<td>+++</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>Helminthidae</td>
<td>many spp.</td>
<td>many spp.</td>
</tr>
<tr>
<td>Diptera</td>
<td>Antocha sp.</td>
<td>++++</td>
<td>+++</td>
</tr>
<tr>
<td>Chironomidae</td>
<td>many spp.</td>
<td>many spp.</td>
<td>few spp.</td>
</tr>
<tr>
<td>Ceratopogonidae</td>
<td>++</td>
<td>++</td>
<td>+++</td>
</tr>
<tr>
<td>Empididae</td>
<td>+++</td>
<td>+++</td>
<td>+</td>
</tr>
<tr>
<td>Trichoptera</td>
<td>Agapetus sp.</td>
<td>+++</td>
<td>+++</td>
</tr>
<tr>
<td>Hydropsychidae</td>
<td>+++</td>
<td>+++</td>
<td>+</td>
</tr>
</tbody>
</table>

* At series of stations down to Mitta Mitta township and including some in area subsequently inundated in 1977.

common species, especially in the Lep-totaphlebiidae, Gripopterygidae, Helmin-thidae, and Glossosomatidae and other trichopteran families, became rare or disappeared. A few other species, previously uncommon or only moderately common, became dominant over much of the riverbed. These were an aquatic mite, Frontipoda sp., a tiny species of hydroptilid caddisfly, and one or two species from the Naididae, Caenidae and Chironomidae; all were small to minute. Although the number of taxa decreased, the number of individual organisms increased greatly (Table 13).

The cause of such faunal change was the development from spring 1976 onwards of a dense carpet of silt and algae in which small burrowing forms throve. This carpet appeared over most of the river bed downstream of the dam site, and was particularly dense and solid in the faster flowing areas upstream of the Mitta Mitta township, and rather lighter and less solid in pools and below the Mitta Mitta township. No such changes in substratum or fauna occurred either above the construction area or in Snowy Creek, a main tributary downstream of the dam site.

When the dam was finally closed totally for two weeks, a significant reduction of invertebrate numbers occurred. The reduction was due largely to a loss of covered bed—50 to 80 per cent of all riffle surfaces were exposed
Table 13  Number of species in Mitta Mitta River before and after construction of Dartmouth Dam.  Summary of results.

<table>
<thead>
<tr>
<th></th>
<th>1974*</th>
<th>upstream of dam site</th>
<th>downstream of dam site</th>
</tr>
</thead>
<tbody>
<tr>
<td>Approximate total number of species per full sample</td>
<td>100</td>
<td>100</td>
<td>50</td>
</tr>
<tr>
<td>Number of species classed as moderately common to common</td>
<td>55</td>
<td>50</td>
<td>20</td>
</tr>
<tr>
<td>Number of species classed as abundant</td>
<td>0</td>
<td>0</td>
<td>6</td>
</tr>
</tbody>
</table>

* At series of stations down to Mitta Mitta township and including some in area subsequently inundated in 1977.

for about 25 km downstream—and to much reduced current speeds. The most affected groups were those requiring fast currents and clean solid surfaces. Many such species, originally common, and including *Trinotoperla nivata*, a small *Centroptilum* species, and several species of helminthid beetle, had previously been reduced to low numbers by silting; their numbers were even further affected.

In summary, the benthic invertebrate fauna of much of the Mitta Mitta River downstream of the dam at the beginning of filling was essentially similar to that reported in the literature for streams below operating dams with hypolimnial releases. Some of the few mayfly, stonefly and caddisfly species which are still present in reduced numbers may be finally extirpated by changes in flow and temperature regime during filling and operation.

**Concluding remarks**

Results to date point to several recommendations of broad management significance with regard to the construction and operation of impoundments. A brief listing of them is an appropriate conclusion for a contribution to a book such as the present.

**Planning for environmental protection.**

Protection of natural communities should be an integral part of the duties of bodies responsible for the management of water resources. Environmental considerations should be included with economic, social, and political factors when deciding if a particular development is necessary or desirable. Moreover, once a decision concerning development is taken, the project design should aim to minimise environmental impact at the earliest opportunity. An obvious example from the Dartmouth survey was the need for better planning of roads and earthworks to prevent erosion and to capture and treat sediment laden runoff.

**Environmental supervision.** The construction authority should bear full responsibility for the supervision of construction practices and for minimising adverse environmental effects. To this end each major project should involve at least one senior on-site environmental officer during both planning and construction.
Prediction and monitoring of physical and chemical changes. Project designers and construction authorities should provide full information on physical events under their jurisdiction and definitive, biologically relevant, predictions concerning physical and chemical impacts. Since biological communities are ultimately controlled by the physico-chemical regime, good biological predictions require accurate and detailed predictions of physical and chemical inputs. Better monitoring of these inputs would allow correction of potentially damaging practices before significant deleterious impact occurs.

Operational procedures. Once operational, the position within the impoundment from which water is released, the volume released, and the timing of releases is critical in the ecology of biota downstream of the dam. The decreased water quality and greatly reduced summer water temperatures resulting from the release of hypolimnial water would almost certainly reduce further the diversity of an invertebrate fauna already badly stressed by construction activities. The frequency and rate of water-level fluctuations within the impoundment are also important in the ecology of littoral communities. These matters should be strongly borne in mind when providing releases or otherwise managing impounded waters.

Scientific studies. Considerably more information about the ecology of aquatic invertebrates should be sought. In particular there is an urgent need for combined field and laboratory studies to examine such questions as the interactions between sediment load, current speed and algal growth, the detailed requirements of particular species in terms of current, substratum structure and temperature, and interactions between various critical species. The present dearth of such information prevents the proper discharge by biologists of their responsibility to provide accurate predictions concerning environmental impacts.

Acknowledgments
The Dartmouth Invertebrate Survey was co-ordinated and funded by the State Rivers and Water Supply Commission of Victoria, as constructing authority, with funds from the River Murray Commission. Many officers within this commission have been helpful throughout the study and are thanked. The survey would have been impossible without the work of special project officers Ms C. Kohlman, Ms P. Morison, Mr E. Savage and Ms R. St Clair, and contributions from many members of the National Museum, particularly Dr A. Neboiss, Dr B. J. Smith and members of the Survey Department.

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Environmental impact of reservoir construction


Introduction
Although the technology of river impoundment remains substantially a province of civil engineers, ecologists in recent times have played an important complementary role. This is illustrated elsewhere in this book by discussions of a number of Australian man-made lakes. One important aspect, however, has been omitted from these discussions, namely the ecological relationship between an artificial impoundment and its parent stream. An analysis of one such situation forms the basis of this chapter.

Attention here is focused upon Lake Hume, a large irrigation-supply reservoir that impounds flows from the headwaters of the River Murray in south-eastern Australia (Fig. 22). The general character of the reservoir is described in Chapter 27, and an account of the regional limnology is provided by Walker and Hillman (1977). The latter account, on which the present chapter is based, represents the conclusion of a three-year (1973-76) survey of the Murray and associated waters undertaken with regard to proposed urban expansion at the cities of Albury and Wodonga. The survey was administered by the engineering firm of Gutteridge, Haskins and Davey, and designed and implemented by a group including R. L. Croome, T. J. Hillman, P. A. Tyler, W. D. Williams and the present author.

In this chapter, following a review of...
some salient features of the study area, it is shown how flow, temperature and oxygen regimes in the Murray near Albury-Wodonga are affected by the near proximity of Lake Hume. Some consideration then is given to associated changes in the composition of the local fish fauna. A summary account of matters raised here is given by Walker, Hillman and Williams (1978).

Study area
The Murray rises some 50 km south of Mt Kosciusko (2,228 m SL), and at first flows northward, gathering the waters of the Swampy Plain and Ghee Rivers, including diversions from the Snowy Mountains Hydroelectric Scheme. Farther on, near the confluence with the Tooma River, the Murray turns westerly toward Lake Hume and the junction with the Mitta Mitta River (Fig. 22). This, the Murray’s headwaters section, extends for some 350 km through steep and later gently undulating forested country, with the water often flowing swiftly over rock and coarse gravel substrates.

Hume Dam is sited near the former confluence of the Murray and Mitta Mitta Rivers, so that water in Lake Hume is banked up along the two stream valleys. The reservoir attained its present full storage capacity (3.07 million ML) in two stages, completed in 1936 and 1961 respectively. The dam (Plate XIII) includes a mass concrete spillway and outlet structures, with lateral earthen embankments extending, in all, about 1.6 km across the river floodplain. When required, water is released to the Murray below the dam either through regulating valves or through the turbines of a small (50 MW) power station. In each case

Plate XIII A view of Hume Dam, showing the spillway and outlet valves. The power station is seen on the left. (Photographed by Dr T. J. Hillman, Albury-Wodonga Development Corporation.)
The impact of man on inland waters

Water is drawn from near the bottom of the lake (maximum depth 41.4 m). There is no mechanism for varying the depth of withdrawal, although gates on the dam spillway may be opened to release surplus water, providing some amelioration of local flooding. Further information about the dam may be found in River Murray Commission (1965), Burton (1974), and Harrison and Johnson (1975).

Below Hume Dam the Murray spreads across the floodplain as a maze of meandering waterways with numerous billabongs (see Chapter 34). About 10 km downstream, it is joined by the Kiewa River, and shortly beyond, in its passage through Albury-Wodonga, gives rise to a short anabranch (Wodonga Creek) which soon rejoins the main stream (Fig. 22). Because of this anabranch, the waters of the Murray and Kiewa may not be completely mixed for some distance downstream of the city area. The Murray continues on a meandering course for about 70 km (200 river km) below Hume Dam, to a point where it is joined by the Ovens River at the uppermost reach of Lake Mulwala, the next downstream impoundment.

Lake Mulwala was formed in 1939 by the construction of Yarrawonga Weir, and provides a hydraulic head for two large irrigation canals (combined capacity 9,000 Ml day$^{-1}$). The lake generally is shallow (mean depth 2.6 m), with a relatively small volume (0.116 million Ml) and a rapid rate of water renewal (average 6-4 days).

Additional information about the study area is provided by Department of National Development (1947), Rowe (1969), Frith and Sawer (1974), Gutteridge, Haskins and Davey (1974), Lawrence and Smith (1975), and River Murray Commission (1977).

Seasonal regimes

Flow. In an average year, more than 30 per cent of the Murray's total discharge (mean 12 million Ml) is from the combined inflows of the upper Murray (incorporating the Snowy-Murray diversion) and Mitta Mitta Rivers to Lake Hume. In these mountain catchments most precipitation falls in winter and spring and, after the snows melt, stream runoff peaks during spring. Stream flows decline from November until the following March, when the cycle resumes (Burton, 1974). Although water from the Snowy-Murray diversion is primarily a byproduct of power generation, it averages about 12 per cent of the inflow to Lake Hume, and may increase to 44 per cent in very dry years (Tisdall, 1974).

During 1973-76, the principal effect of the diversion was to maintain higher-than-natural flows over 1976, an unusually dry year (Fig. 23).

The release of water from Lake Hume is determined mainly by irrigation demand. From June to November, when downstream tributaries usually are flowing strongly, releases from storage are generally not necessary. Demand increases progressively after September, but it may not be until December that water from Lake Hume is required. Releases then are made as required until May, as tributary flows decline. A provision in management is that flows in the Murray immediately below the Kiewa confluence are maintained above 1,200 Ml day$^{-1}$.

From Lake Hume to Lake Mulwala, the only significant inflows to the Murray are the Kiewa and the Ovens, with the former averaging about one-third the annual discharge of the latter (means 0.632 cf. 1.580 million Ml, respectively).
Neither tributary is subject to significant flow regulation.

The general effect of flow regulation at Hume Dam is to reduce seasonal flow variations by taking headwater flows into storage in winter and spring, and by maintaining higher-than-natural flows over the dry summer. However, stream flows in the region remain liable to large variations, in accord with local weather conditions. During the survey period stream flows were markedly higher than average in 1974 and 1975, and well below average in 1976 (Fig. 23).

**Temperature.** Temperature records at various sampling stations during 1974-76 are shown in Fig. 24. At each station there was a consistent seasonal pattern, corresponding with the seasonal pattern of air temperatures (cf. Fig. 24F). It is noteworthy that stream temperatures followed a similar trend in each year despite the flow variations that occurred. The seasonal regimes of the upper Murray and Mitta Mitta Rivers were
Fig. 24 Temperatures (°C) at various combinations of sampling stations, 1974-76. Air temperatures at Bonegilla, near Wodonga, are shown for comparison.

The influence of Lake Hume is shown by comparison of data for the upper Murray (station 1, Fig. 22) and for the Murray 1 km below Hume Dam (station 6). Records at station 1 are taken to be indicative of the river temperatures that would have prevailed at station 6 in the absence of the reservoir. The comparison (Fig. 24C) shows that Lake Hume had a dual effect. Firstly, the amplitude of the seasonal temperature cycle was reduced by about 6°C, so that the annual range below the dam was 10-21°C rather than 7-24°C. Secondly, the lake displaced the seasonal pattern such that trends at station 6 in spring, summer and autumn were delayed by about one month relative to those at station 1.

The thermal effects of Lake Hume are
The downstream influence of Lake Hume

explained by seasonal thermal stratification (see Chapter 27), and the fact that in summer and autumn water in the Murray at station 6 is drawn from the bottom (hypolimnion) of the reservoir. Although slight stratification was observed at two stations on the arms of the reservoir (3 and 4), it was at the station nearest the dam wall (5) that the temperature gradient was most pronounced. During spring and summer each year a thermocline forms at about 10-12 m depth and gradually deepens through summer and autumn, hastened by the withdrawal of hypolimnetic water for irrigation. The peak temperature differential between top and bottom waters at the station nearest the dam wall in different years has varied from 3° to 8°C.

In considering how far downstream the thermal influence of the reservoir extends, the temperature patterns of the Kiewa and Ovens Rivers must be taken into account. In fact the regime of the Kiewa (Fig. 24E) was similar to those of the upper Murray and Mitta Mitta Rivers, having a seasonal range of about 7-23°C. In summer, therefore, the Kiewa inflow was slightly warmer than the Murray near the junction of the two rivers.

As mentioned earlier, the Wodonga Creek anabranch complicates mixing of the Murray and the Kiewa, and hence records for stations 8 and 9 (see Fig. 22) are best omitted from the present discussion. At station 10, some 25 km downstream of the Murray-Kiewa confluence, temperature patterns (Fig. 24D) were more like those at station 6 than those of the Kiewa River, although some compensation of the temperature lag was evident. At station 12, about 67 km farther downstream, summer temperatures attained 22-23°C (Fig. 24D), significantly warmer than at 10, but still marginally below the maximum expected from extrapolation of data for the upper Murray. In addition, there was further compensation of the temperature lag introduced by Lake Hume. A similar pattern was shown by records at another station located on the Murray immediately upstream of Lake Mulwala.

The Ovens River inflow was, in spring and summer, generally about 2°C warmer than that from the Kiewa, and slightly warmer than the Murray near the point of entry to Lake Mulwala. Summer maxima for the Ovens River were in the range 25-26°C, and the corresponding range for the Murray near the confluence was 22-23°C.

Surface temperatures in Lake Mulwala (station 13, Fig. 24D) and in the Murray below Yarrawonga Weir (station 14) were similar. Lake Mulwala does not show persistent thermal stratification, primarily because of its shallowness and rapid flushing rate.

The temperature data suggest that, although progressively diminished to near extinction, the thermal effects of Lake Hume extend right along the reach to Lake Mulwala. The inflow of the Ovens River, possibly supplemented by heat storage in Lake Mulwala, completely restores a 'natural' thermal regime.

Oxygen. At all sampling stations other than those on the mainstream of the Murray below Hume Dam, oxygen values generally were near saturation. In Lake Hume, however, complete hypolimnetic oxygen depletion occurred in each summer during the survey, except 1973-74. Water discharged to the Murray in summer, therefore, is usually anoxic, and although this is rapidly compensated
by turbulence, there remains a distinct oxygen ‘sag’ which is apparent some distance downstream.

Oxygen data for certain stations downstream of the dam are shown in Fig. 25. At station 6, about 1 km downstream, summer oxygen levels tended to be about 50 per cent saturation, and at station 10, 37 km downstream of the dam, values were typically about 70 per cent saturation. The oxygen sag, however, was not detectable at station 12, a little more than 100 km below the dam.

Changes in the fish fauna

Certain species of native fish that formerly were common along the Hume-Mulwala reach of the River Murray have apparently declined in abundance, and some may be locally extinct. The evidence linking the apparent decline with the environmental changes caused by Lake Hume inevitably is partly circumstantial, but nonetheless warrants attention. The evidence is considerably strengthened by the recent publication of observations first made in 1948-50 by Langtry (Cadwallader, 1977).

A similar decline in the abundance of native fish, and a corresponding increase in numbers of introduced fish species, has been recorded for several man-made lakes in Australia, including Lake Hume. Weatherley and Lake (1967) considered that the major contributing factors are associated with environmental changes, although there is increasing evidence (e.g. Jackson, 1975) that some native species are adversely affected by interactions with introduced fish (see also Chapter 23). A recent review of this topic, emphasising changes in the Murray-Darling system, has been provided by Cadwallader (1978).
The downstream influence of Lake Hume

Two native species that have apparently disappeared from the Murray near Albury-Wodonga are the golden perch, *Macquaria ambigua*, and the freshwater catfish, *Tandanus tandanus*. According to Lake (e.g. 1971), golden perch require water temperatures above c. 23°C before spawning is initiated, although spawning may occur at lower temperatures in times of flood. Catfish favour warmer, sluggish waters and spawn in gravel nests in shallow water; short-term variations in water levels, such as caused by management for irrigation, may have obvious ill effects. The disappearance of these two species, at least, may be explained by environmental changes due to Lake Hume, as their habitat requirements are unlikely to be met under the present river regime.

Both Lake Hume and Lake Mulwala are impounded by barriers that effectively exclude any upstream migration of fish, and in the absence of artificial stocking there is little chance that native species such as golden perch could become re-established locally. It is not generally known that some native species, golden perch included, undertake extensive upstream migrations (e.g. Anon., 1977), although the reasons for such migrations are obscure.

Several introduced fish species, notably the golden carp (*Carassius auratus*), the redfin (*Perca fluviatilis*), the tench (*Tinca tinca*) and brown trout (*Salmo trutta*) are well-established in the Albury-Wodonga area. From observations made during the survey, it is clear that whatever the reasons for the decline of native fish species, the introduced species are becoming increasingly dominant. This situation may be further enhanced once the European carp (*Cyprinus carpio*) becomes fully established; this species has become widespread throughout the River Murray following its introduction in the early 1970s.

**Concluding remarks**

Because of limited space, a more complete analysis of the ecological relationships between Lake Hume and the River Murray cannot be given; more details pertinent to such an analysis are provided by Walker and Hillman (1977) and Walker (1979). Nor has it been possible to compare the Albury-Wodonga study with other studies; for relevant information the reader is referred to papers by Fraser (1972), Hannan and Young (1974) and Ward (1974, 1976a,b,c).

As Butcher (1966) has shown, reservoir management in Australia rarely has taken account of the ecological implications of manipulation. It would be encouraging to find that the most recent projects are making allowances in this regard, but appearances are to the contrary. The River Murray Commission recently has completed construction of Dartmouth Dam (storage capacity 4 million Ml, depth 180 m) on the Mitta Mitta River, some 90 km upstream of Lake Hume. The new dam has only a single offtake (near 60 m), with provision for later installation of multi-level outlets should temperature control be considered necessary (Burns, 1977). As a result of modified flow, temperature and oxygen regimes the ecology of the Mitta Mitta River below the dam will be changed irrevocably. In particular, the local population of Macquarie perch (*Macquaria australasica*), regarded as a seriously endangered species (Lake, 1971), probably will be eradicated. It may well be that engineers are able to contemplate the construction of multi-
level offtakes at later stages, but biologists have not the same degree of freedom.

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19 Biological monitoring

W. D. Williams

Introduction
This chapter is concerned with the use of living organisms as indicators of environmental conditions ('quality') in inland waters. The phrase often used to describe such use is biological monitoring. The chapter firstly outlines the values of biological monitoring, goes on to describe the principal methods used, and then discusses in some detail Australian applications. Although Australian work is considered, the aim of the chapter is less to act as a review of this than to emphasise the usefulness of biological monitoring techniques. A minimum aim is to alleviate in constructive fashion the apparent obsession with numerical data derived from physico-chemical investigations that seems to characterise so many—though not all—managers of Australian water resources!

Values of biological monitoring
Before considering specific values, it is appropriate to note that all values reflect the fact that pollution of inland waters is primarily a biological phenomenon. As such, biological phenomena would seem a priori to provide the salient characters to be assessed in determining the degree of pollution. There will be some chemists and engineers who might debate this point, and who doubtless regard water quality as basically an integration of chemical and physical factors, but there will be a few, even amongst those, who would disagree with the widely accepted definition of water pollution as a degradation in the natural character of a water body. The use of the word 'natural' connotes a close if not exclusive linkage with contained aquatic life. In any event, the value of aquatic organisms as environmental indicators was certainly recognised at a very early time, e.g. by Forbes (1913) and Purdy (1923).

The values of biological monitoring usually advanced may be summarised as follows:

(1) a single series of biological samples may give a summation of environmental conditions during the preceding period (physico-chemical methods would need to use multiple samplings over a long period to arrive at a similar result);
(2) biological monitoring is better able to detect intermittent pollution than are physico-chemical methods;
(3) biological monitoring is better able to provide an indication of the extremes of stress during critical periods than are physico-chemical methods: that is, the biota represents the results of the most extreme conditions to have occurred;
(4) certain sorts of organisms involved in biological monitoring may be extremely sensitive to given pollutants, and thus biological monitoring is an especially valuable tool in cases of slight pollution. An example may be cited in the use of certain animals to monitor pesticide levels: these, because of the process of biological magnification, may accumulate easily detectable amounts of
pesticide residues when the surrounding water may contain scarcely discernible levels;

(5) biological monitoring provides a more complete (integrated) picture of environmental stresses at a given point of time than physico-chemical methods. Synergistic and antagonistic effects of pollutants in particular are especially difficult to assess from purely chemical tests.

Several of these values have been considered by Wuhrmann (1974) who took a decidedly pessimistic view of the use of living organisms in pollution monitoring. In part, his criticisms have some validity, but no more than that, and there is little doubt that time will bear out the usefulness of biological monitoring as a valuable complementary exercise to physico-chemical monitoring. Whatever the validity of Wuhrmann's criticisms, there has certainly been no lessening of interest in the subject as the result of them (cf. Cairns, Dickson and Westlake, 1977; Alabaster, 1977). Conceptual shortcomings in the approaches adopted to biological monitoring there may well be, but none suggests that biological monitoring should therefore be less, and not more used. Doudoroff's (1977) recent and sensitive critique of bioassays makes the point well: he documented many shortcomings in the approaches presently followed by biologists, but ultimately emphasised the value of bioassays over and above physico-chemical investigations in isolation.

The values listed relate principally to assessment within water bodies, that is, they mainly relate to in situ effects of external impacts. Biological monitoring also has an important predictive role, however; certain sorts of biological monitoring may help in forecasting the degree to which a given water body will be useful (or otherwise) for a specific purpose, e.g. for domestic supplies. The usual predictive form biological monitoring takes is that of bioassays (though bioassays may also, of course, be used in other ways, e.g. in determining the amount of allowable waste for discharge into waters (Tarzwell, 1971)).

Having argued the case for biological monitoring, the question must be answered as to why more use has not been made of biological monitoring in water pollution studies and control. Doubtless the answer is partly because physico-chemical methods require less expertise, provide superficially more acceptable quantitative measures, and are not subject to the same degree of variation in interpretation besetting biological data interpretation—for which there is, in any case, a confusing variety of methods of derivation with few or no standard methods. None of these reasons is strong enough for those monitoring aquatic environments to continue to ignore the contribution that biological monitoring can make.

Methods
No detailed comments on this subject are required because several recent and comprehensive accounts exist (e.g. Goodnight, 1973; Cairns and Dickson, 1973; Wilhm, 1975; Cairns, Dickson and Westlake, 1977; Hellawell, 1978). The briefest of surveys is all that is needed for present purposes.

Methods of biological monitoring range from those involving biochemical reactions, and cellular and tissue responses, to those based on species and community effects. Biochemical reactions that have been used include changes in haemoglobin concentrations.
in body fluids as a measure of environmental oxygen tension (Fox, 1954), and in brain acetylcholinesterase concentration in fish as a measure of organic phosphorus insecticide levels. Fish serum analysis by paper electrophoresis has been proposed as a method for measuring industrial pollution (Fujiya, 1961), and changes in the energy content of whole organisms has been suggested as indicative of both general and particular environmental conditions (Spoehr and Milner, 1949). Biological magnification (bioaccumulation) of certain pollutants is a particularly important metabolic response in many organisms.

At the cellular and tissue level, several features have been recorded as responses to pollutants. Mostly they take the form of aberrant changes. Thus, changes in the number and size of erythrocytes occur in fish exposed to various chemical pollutants. Gill epithelial changes, a reduction in gonadal development, and liver damage are other responses recorded for fish.

At the species level, living organisms have been used in several ways. Toxicity testing of individual species is one. In such testing, the lethal concentration to 50 per cent of a test population (LC₅₀) is probably the widest used criterion. Behavioural responses in animals is another way. Mostly the animals used in this sort of test have been fish (see review by Warner, 1965; also Kleerekoper, 1977), but invertebrates have also been used (see Hart and Fuller, 1974). The straight compilation of species lists is yet another method. However, perhaps the most common way in which individual species are used in biological monitoring is as ‘ecological indicators’: the occurrence of a species at a particular location indicates that pollution has not exceeded a critical value during the lifetime of the species at that point (highly mobile species such as fish are, of course, exempt here). Whilst the use of a species in this way may provide considerable reliability, full interpretation requires ecological knowledge of such matters as environmental tolerances (including potential temporal variation), adaptations to pollutants, and life cycles (Hart and Fuller, 1974).

Closely related to the use of species as ecological indicators is use at the community level, that is to say in associations of species. The classic method which used species like this is the Saprobiensystem of Kolkwitz and Marsson (1908, 1909), but many similar approaches subsequently appeared as well as various refinements and modifications to the original Saprobiensystem. Because of the numerous difficulties involved in the Saprobiensystem and similar approaches, several attempts have been made to quantify community relationships. A variety of mathematical expressions has been proposed. They range from those which require little taxonomic expertise such as the sequential comparison index of Cairns and Dickson (1971) to those whose computation requires taxonomic knowledge of all species and the numbers of individuals in each (Shannon and Weaver, 1963). Other mathematical indices which deserve mention are those of Pantle and Buck (1955), Burlington (1962), Zelinka and Marvan (1963), and Stoneburner, Smock and Eichhorn (1976). Finally, a few mathematical expressions which involve relationships between community-based parameters but not taxonomic units per se should be noted. The ratio between production and respiration is one example. The ratio between production and
Biological monitoring

195

Biomass has been proposed as another (Margalef, 1958).

Many taxonomic groups have been made use of in biological monitoring, in both in situ and in vitro (i.e. bioassay) methods, when the total aquatic community, or some community-based parameter, is not involved. The most important are bacteria (e.g. Liebmann, 1951), Protozoa (e.g. Sramek-Husek, 1958; Cairns, Lanza and Parker, 1972; Cairns, 1974), algae (e.g. Patrick, 1954; Archibald, 1972; Cairns, Lanza and Parker, 1972), benthic macro-invertebrates (e.g. Hynes, 1959; Goodnight, 1973; Gaufin, 1973; Hart and Fuller, 1974), and fish (e.g. Sprague, 1969). No one group seems pre-eminently suited as a biological monitoring tool: each has advantages and disadvantages. Fish are useful because they are macroscopic, vertebrate, and usually a good deal is known of their ecology; however, they are very mobile and may migrate. Algae have been claimed as particularly useful; unfortunately, their taxonomy requires expert training, as does also that of the Protozoa. The same difficulty often applies to macro-invertebrate taxonomy, but probably not to the same extent. Thus, since few invertebrates are as mobile as fish, many workers regard macro-invertebrates as amongst the more useful groups. Much work remains, but already a number of interesting and useful systematic differences between macro-invertebrate groups have appeared. Oligochaetes, molluscs and turbellarians are less tolerant to heavy metal pollution than many insects; stoneflies are generally less able to tolerate lowered oxygen tension than are mayfly nymphs; certain oligochaetes appear more tolerant to some insecticides than other aquatic invertebrates; and a number of interesting generic and specific differences occur within a given taxonomic group as responses to changing environmental conditions.

Australian applications

Most work on the use of the aquatic biota in water pollution assessment and control has concerned northern hemisphere waters and biota. But the principles and values of biological monitoring are, of course, as applicable in Australia as elsewhere. Their application, however, requires caution. There is a number of reasons for this, but two are salient. First, most methods of biological monitoring were conceived in particular regions and therefore, considering the differences in the nature of the distribution and abundance of living plants and animals, are fundamentally regional in application. And, second, there is a number of distinctive, indeed unique, features which characterise Australian inland waters. The first of these points requires no further elaboration here; the second is fully discussed in Chapter 2.

Related to the differences of the Australian inland aquatic environment from aquatic environments elsewhere is another major difficulty in the use of biological monitoring techniques in Australia, namely, the difficulty caused by inadequate taxonomic knowledge of the aquatic biota. This is less a regional problem when bacteria, Protozoa and algae are being considered than when less cosmopolitan groups are, but, as indicated previously, these three groups have taxonomic problems of their own, and, in any event, other groups (fish, benthic macro-invertebrates) have several advantages over micro-organisms as biological monitors. Williams (1976) has recently surveyed the extent
of systematic knowledge of the Australian inland aquatic fauna (invertebrate and vertebrate). The conclusion was that comparatively few groups have been ‘relatively well-worked taxonomically’, a majority are ‘not well-worked taxonomically’, and for several there is ‘little taxonomic information’. Unfortunately in the second category are many groups that have proved to be important in biological monitoring in other countries, e.g. Bivalvia, Gastropoda, Oligochaeta, isopods, amphipods, various decapods, Ephemeroptera, Chironomidae (Hart and Fuller, 1974). And, irrespective of the extent of taxonomic knowledge of the Australian aquatic fauna, ecological knowledge for almost all groups is grossly deficient.

Some examples of biological monitoring in Australia. Despite the difficulties inherent in any attempt to use biological methods in pollution investigations in Australia, there have been several studies which have attempted to do so. Perhaps because of the difficulties it is not surprising they are few; or perhaps, conversely, their paucity is surprising in view of the frequency of references in much current Australian environmental protection authority regulations to the ‘preservation of the natural aquatic biota’ as a stated management aim (e.g. Environment Protection Authority, Victoria, 1972; see also Hart, 1974). The principal studies are discussed below seriatim; a few less comprehensive ones are omitted from discussion.

Two confluent rivers in New South Wales (Coxs River and Farmers Creek) were studied by Jolly and Chapman (1966). These waters were organically polluted by sewage, and the authors studied attached algae and macro-invertebrates at a series of stations downstream of the sewage outfall and at one station upstream. They found that immediately below the outfall there was an association of oligochaetes, chironomids, gastropods, and sewage fungus together with small growths of the alga Stigeoclonium and diatoms. This was rapidly followed by extensive growths of Stigeoclonium, high protozoan densities, and increased gastropod numbers. The authors took recovery from pollution as indicated by the reappearance of stonefly and mayfly nymphs, odonate, trichopteran and psephenid larvae, and a dominance of snails, limpets and chironomids. No ‘indicator species’ were noted.

The Molongo River near Canberra was studied by Weatherley, Beevers and Lake (1967). A major pollutant in this river was (and is) zinc. The authors showed that this pollutant markedly depressed both the number and variety of benthic macro-invertebrates. Some group differences were apparent, however. Relatively speaking, the Crustacea, Mollusca, Annelida, Ephemeroptera, Megaloptera and Turbellaria proved intolerant and were largely not found in badly polluted stretches; Hemiptera, certain Plecoptera (Dinopterla) and cased-caddis larvae were amongst the more tolerant forms.

The Mountain River, a small Tasmanian river polluted by pesticides from apple orchards, was investigated by Steen (1969). His principal conclusion was that fish collected from this river had significantly higher concentrations of DDE and DDT in their bodies than was the case for fish collected from another Tasmanian river, the Russell River, which has no orchards on its catchment.
Another Tasmanian River, the South Esk was studied by Thorp and Lake (1973, 1974). The major pollutants in this river were cadmium and zinc (Tyler and Buckney, 1973), and the biota studied was the macro-invertebrate. The most intolerant groups were Crustacea, Mollusca and Annelida; the most tolerant were Hemiptera, Arachnida and leptocerid trichopteran larvae. Decreased species diversity and abundance was also noted (Thorp and Lake, 1973). Toxicity bioassays on selected invertebrates were involved in these studies (Thorp and Lake, 1974).

It is remarkable that there were no biological monitoring studies of the River Murray, Australia's most significant inland waterway, until a recent study undertaken by Gutteridge, Haskins and Davey (Cities Commission, 1974; Croome, Tyler, Walker and Williams, 1976). This study involved physical, chemical and biological monitoring of the River Murray upstream of and for some distance downstream of Albury and Wodonga, two townships located on the upper Murray. The study aims were to (1) monitor changes in the natural character of the River Murray consequent upon an increased growth of the townships, and (2) provide engineers with predictive advice on permissible discharge loadings for certain pollutants (especially phosphates and nitrates). Most fieldwork of a biological sort was concerned with algae. Additionally, however, fish and freshwater bivalves were assayed for heavy metals and pesticide residues. The major conclusion was that unless the entry of plant nutrients to the River Murray was restricted from Albury-Wodonga sewage treatment plants, eutrophication would eventually occur in the first downstream river impoundment (Lake Mulwala). Pesticide and heavy metal residues were not considered significant. Following publication of the initial report, a decision was made to continue monitoring the river using biological and other parameters, and a second more comprehensive report has recently been published (Walker and Hillman, 1977). Its major conclusions are not significantly different from those of the first report. Long-term biological monitoring continues.

A shorter study of the Dandenong Creek, a small river on the outskirts of Melbourne, also merits noting. It was studied by Campbell (1975), and is organically polluted by sullage from unsewered suburban properties. The macro-invertebrate fauna was investigated. No true indicator species were found, but seven groups of common animals could be arranged in order of increasing tolerance to organic pollution. The order was as follows, commencing with the least tolerant group: *Atalophlebioides* (Ephemeroptera), Trichoptera (excluding Hydropsychidae and Hydroptilidae), a chironomid species, *Potamopyrgus* (Gastropoda), *Bulinus* (Gastropoda), Tubificidae, and Psychodidae.

Finally, brief attention may be given to a study of the King River in western Tasmania (Lake, Coleman, Mills and Norris, 1977). This river is polluted by copper, and to less extent by lead and zinc; the pollution was reflected in the distribution of the riffle-dwelling macro-invertebrates and trout investigated by Lake and his colleagues. The pollution also affected the drift fauna in that there was a significant drop in its abundance with the entry of pollution.

Several questions may be asked of these Australian studies. What facts do they illustrate about pollution; how do
these facts compare with similar studies overseas; and do the studies show any features better than would have physico-chemical monitoring methods? Before attempting answers, it should be made quite clear that the Australian studies relate to four distinct types of pollution—by organic material, heavy metals, pesticides and plant nutrients—and that any answers derived from them are based on only one or at most few studies for each type of pollution. Thus, though answers are attempted here, their largely premature nature should be recognised.

Firstly, what facts may be gleaned from the Australian studies directly? The following seem to be the major ones:

1. Zinc, zinc and cadmium, or copper pollution results in decreased diversity and abundance of macro-invertebrates;
2. The most intolerant macro-invertebrates to zinc or zinc and cadmium pollution are the Crustacea, Mollusca and Annelida, and the most tolerant, the Hemiptera, certain caddis and, in one study, certain Plecoptera;
3. The nymphs of an Atalophlebioides sp. (Ephemeroptera) appear to be partly tolerant of copper pollution;
4. Copper pollution depresses the abundance of stream drift;
5. There appear to be no ‘indicator species’ for organic pollution;
6. The most intolerant macro-invertebrates to organic pollution are the Ephemeroptera and Trichoptera, and the most tolerant, oligochaetes, gastropods, and various dipteran larvae;
7. Fish and bivalve molluscs accumulate pesticides and heavy metals when these are present in the external environment;
8. The addition of plant nutrients to rivers results in increased algal biomass downstream.

Comparing these facts, despite their limited data base, with studies pursued outside Australia points to several similarities. Thus, molluscs and oligochaetes elsewhere have also been shown to be intolerant to heavy metals and at least some insect groups to be tolerant to such pollution; likewise, oligochaetes have been shown to be tolerant to organic pollution and most insects except several dipteran groups intolerant. That fish and bivalves accumulate pesticides and heavy metals, and that plant nutrients stimulate plant growth outside Australia requires no comment. Beyond such gross comparisons, however, it is scarcely possible to proceed at present. The question may be left with the simple statement that Australian work to hand indicates parallels with work elsewhere.

As for the intrinsic value of Australian studies of biological monitoring, we should note that several were designed as no more than descriptive exercises of localities regarded on non-biological criteria as polluted. The value of such descriptive work lies more in potential uses elsewhere, that is, in extrapolation and application, than in static evaluation of pollution in the study stream itself (though this of course is not without value). It is therefore difficult to ascribe a value to such studies in terms of those given at the beginning of this chapter. With the exception of the work on the River Murray, a study deliberately designed to predict and monitor changes in that river, data from no other Australian study can be advanced as evidence of the greater usefulness of biological monitoring vis-à-vis physico-chemical monitoring. Indeed, apart from the River Murray study and that by Jolly and
Chapman (1966), the primary objective of all studies was to describe the effects of pollution, not assess its extent. Yet, as noted, it is basically as a tool for assessment that biological monitoring has most to contribute. Of course, any ability to assess pollution by biological means can only be derived from an adequate descriptive base, so there is no implication here of any criticism of the fundamental direction of the studies performed thus far—certainly, many similar studies are greatly needed. It is simply pointed out that biological monitoring techniques as methods for assessing the degree of pollution, mitigating the need for multiple physico-chemical sampling, detecting intermittent pollution, indicating extreme stress values during critical periods, integrating environmental stress at a given time, and for detecting slight pollution (e.g. by pesticides or heavy metals), have scarcely been made use of in Australia.

**The future for biological monitoring in Australia.** There are four basic areas in which further work is required before full advantage can be taken of biological monitoring as a descriptive, assessment and predictive tool for water pollution control and management in Australia. The areas are those of taxonomy, autecology, toxicity and bioassays, and environmental description. The actual marriage of results from these areas perhaps constitutes a fifth area of research: how best to integrate the results to provide a system for assessment and prediction.

Regarding taxonomy, the importance of precision in identifying the living units (species) of natural systems can scarcely be over-emphasised. Unless the constitution of the biota of rivers, streams and lakes is known precisely, biologists will never be in the position of being able to indicate what should be present and why it is or is not. Fosberg (1972) has informally discussed this matter further. Since it is impossible for various reasons—and in any event is hardly necessary—to be taxonomically exact about all units involved, the question is: what groups constitute the significant members of the biota. The indications are that within the context of usefulness as biological monitors the following do: gastropod and bivalve molluscs, oligochaetes, amphipods, isopods, atyid and palaemonid prawns, stoneflies, mayflies, dragon- and damselflies, caddis flies, certain bug families (Hemiptera), most dipteran families, some beetle families (Coleoptera), fish, and certain algal groups. Further taxonomic study of most of these groups is required in Australia (Williams, 1976).

Turning to the area of autecology, it is obvious that unless details of the ecology of a particular species or taxonomic group are known, little insight can be gained into the nature of those environments wherein it occurs simply from occurrence data alone. At present very little indeed is known about the ecology of those groups enumerated in the preceding paragraph. More knowledge is needed if they are to be of use as biological monitors.

Part of the autecology of any species is its tolerance of and reaction to various environmental parameters (including stresses). It is convenient, however, to treat the subject of toxicity testing and bioassays separately. Notwithstanding the various criticisms that can be and have been raised against laboratory experimentation of this type and subsequent extrapolation of results to field
situations, there is little doubt that toxici
ty trials and bioassays have played and will continue to play a useful role in biological monitoring as both assessment and predictive devices (Doudoroff, 1977). They are capable of playing a particularly important role in Australia if for no other reason than that their taxonomic and autecological demands are minimal. It is worthwhile therefore to consider briefly what species might be most useful as subjects. Firstly, however, some consideration needs be given to the sort of criteria that should be met by taxa useful in this context.

The principal criteria would seem to be the following: suitable taxa should be:

1. geographically widespread (there would be little point in using species of restricted distribution);
2. easy to identify and not part of a taxonomically difficult species complex;
3. not restricted to a very special sort of habitat;
4. easy to maintain under laboratory conditions;
5. not microscopic (that is, should be easy to see and therefore handle);
6. native to Australia (state regulations could well restrict the use of exotic species, and, in any event, the possibility of further spread of many exotic forms in Australia should be guarded against);
7. common (difficulties would clearly arise if the taxa could only be collected rarely);
8. neither markedly resistant to environmental stresses or changes nor yet overly sensitive to them;
In the case of animals, they should also be:
9. unspecific in dietary requirements (that is, with a diet which includes a broad composite of other taxa present);
10. reasonably restricted to partic-
cular microhabitats for food sources but with feeding mechanisms which sample widely from these microhabitats; and
11. relatively immobile in the field.

Not many species are likely to meet all these criteria. However, four animals appear to meet most if not all. They are *Paratya australiensis* (an atyid prawn), *Austrochiltonia australis* (or *subtenuis*) (an amphipod), *Velesunio ambiguus* (a bivalve mollusc), and *Galaxias maculatus* (a fish). Undoubtedly some plants would also meet most of the criteria; none, however, is proposed here since toxicity testing has in general—though not entirely—tended to use animals as subjects, and bioassays involving plants have almost without exception used algae, a group with more or less cosmopolitan species (cf. Middlebrooks, Falkenberg and Maloney, 1977).

There is little profit in documenting at length how and to what extent these four animals meet the criteria; in a chapter of this sort, it suffices simply to say that to various degrees they do. There is some advantage, nevertheless, in drawing attention to certain aspects of the ecology of each. Thus, *Paratya australiensis* is a free-swimming form which in common with all atyid shrimps has a feeding mechanism which includes an efficient set of scrapers (the first two pairs of chelipeds with their terminal brush setae) that undoubtedly act as broad samplers of the detritus and other food material on submerged surfaces. *Austrochiltonia australis* (or *subtenuis*) is also free-swimming but is rather more closely associated with the benthos than *P. australiensis*; like *P. australiensis* it also is probably mainly detritivorous, but it has a quite different feeding mechanism and no doubt draws its food from different sources therefore. *Vele-
Sunio ambiguus is a filter feeder, and its food consists of organic matter sieved from material and water drawn into its mantle cavity. Since V. ambiguus lives on the bottom of rivers and lakes it is particularly suited to the assaying of pollutants associated with sediment particles. Galaxias maculatus is an omnivore and appears to be the native fish which best meets the criteria for a suitable subject for toxicity tests or bioassays.

It is noteworthy that even in the absence of a detailed rationale and without reference to the sort of criteria enumerated above, all four species proposed have already been used in toxicity and bioassay experiments in Australia. There is already available therefore a base on which to build in this area. Velesunio ambiguus was used to assay pesticides and heavy metals in the study of the River Murray by the Cities Commission (1974) and Croome, Tyler, Walker and Williams (1976). It clearly proved satisfactory in this capacity. Another Australian freshwater mussel, Hyridella australis, has also been found to be an excellent monitor of certain pesticide (endrin) concentrations in fresh waters (Ryan, Bacher and Martin, 1972). Paratya australiensis (as P. tasmaniensis: see Williams, 1977) and Austrochiltonia subtenuis were used by Thorp and Lake (1974) in toxicity experiments involving zinc and cadmium. These authors also used other macro-invertebrates but found that (p. 98): 'Paratya proved to be the most satisfactory species.' They experienced difficulty with A. subtenuis only because of the difficulty of distinguishing it from A. australis, although the two species are reasonably distinct (Williams, 1962). Of other animals used, the trichopterans were too resistant and responded irregularly, the zygopteran nymphs interacted aggressively, and several of the ephemeropteran nymphs emerged to sub-imagos thus disrupting the experiments. A. australis was the main animal used by Lake, Mills and Swain (1976) to determine the effect of cadmium on freshwater crustaceans. Finally, Galaxias maculatus has been subjected to acute toxicity tests for cadmium by Cassidy and Lake (1975).

The fourth area where research is needed to enable effective use of biological monitoring in Australia is that of 'environmental description'. By that is meant the adequate provision of comprehensive descriptions of a variety of aquatic environments with differing degrees of pollution stress for each of the major pollutants. Some data are available already, as noted, but no study thus far is either fully adequate or comprehensive. It should be emphasised that such studies should include environments where minimal pollution is occurring; obviously the extent of pollution as indicated by a particular biological description cannot be gauged in isolation.

It will be clear from the above account that many difficulties remain before biological monitoring can be used to full account in Australia. None, however, is regarded as so insuperable as to make redundant any attempt to apply the techniques of biological monitoring. The situation is that biological methods provide a valuable complementary means of monitoring water pollution, that some difficulties exist in applying methods of non-Australian origin, but that with reasonable care, caution and some further investigation these difficulties can be minimised.
Acknowledgments
Several colleagues have seen a draft manuscript of this chapter and made comments. In this regard I am particularly indebted to Mr A. Haughey (formerly, Engineering and Water Supply Department, Adelaide; now in New Zealand), Dr P. S. Lake (Monash University), Miss Joan Powling (State Rivers and Water Supply Commission, Victoria). An early version of the chapter provided the basis of a lecture given at the International Association on Water Pollution Research course on biological monitoring held in Melbourne in October 1976.

References


* Not seen


* Not seen
Introduction
Owing largely to the greater environmental awareness of the public and government, the number of environmental surveys in Australia has considerably increased over the past decade (e.g. Cities Commission, 1974; Hodgkin, 1975). Most Australian states, in fact, now have legislation requiring that new developments consider environmental consequences. Nevertheless, it is arguable whether we are now in any better position than before to predict the impact of man upon natural ecosystems. Certainly the effects of gross pollution are well-known, but there is still much to learn before the magnitude and significance of more subtle effects can be predicted. Additionally, since the early 1970s, legislation has been enacted in most states to control waste discharge. This has meant that relevant authorities (and often the dischargers too) need to undertake surveys of the systems being affected.

Although there are often severe restrictions in the application of traditional scientific methods in environmental surveys, it is still possible to employ scientific practice in devising and designing hypotheses. Nevertheless, a large number of modern surveys appear merely to be inventories which make no attempt to understand system interrelationships. The adoption of scientific practice is particularly important since environmental data are inherently variable; there are extremely complex interrelationships between cause and effect, and there are difficulties in identifying the major (important) variables associated with these interrelationships.

This chapter identifies and briefly discusses fundamental features of environmental surveys in an attempt to emphasise to environmental scientists the importance of adopting a sound scientific framework for the planning and implementation of future surveys. Within the context of this book, studies of the aquatic environment are emphasised.

Survey design
The initial design of any survey is the vital step in ensuring success, yet is often paid scant regard. The following comments relate mostly to large-scale surveys such as that of the Gippsland Lakes Regional Survey (Ministry for Conservation, 1977; Hart and Snow, 1976), but the principles are equally applicable to smaller surveys. The general design pathway for an environmental survey is shown schematically in Fig. 26.

Problem-orientated approach. A ‘problem-orientated’ approach for environmental surveys is advocated since it has a number of advantages. The definition of a series of system ‘problems’ also provides a focus for the establishment of hypotheses or conceptual models of the way the system functions, a result of
Fig. 26 Approach to the design of a water quality study program.

particular use in both setting objectives and assessing data. However, before one can define 'problems'—existing or potential—there is an obvious need for information on the study system. This requires the collection of all relevant information before the main study commences. This approach also generally requires the establishment of a multidisciplinary team to define problems and determine study objectives. If there is no information available from which the system problems can be defined, there may well be a need for a small initial survey with restrictive and extremely specific objectives.

To provide substance to these general remarks, an example may usefully be considered. The case of lake systems that receive excessive plant nutrient inputs is discussed.

In a lake receiving excessive nutrient inputs, inevitably there are phytoplankton blooms and/or excessive weed growth. Both conditions reduce water quality, seriously impair recreational use of the lake, and generally indicate adverse effects due to man's activities. If this situation were presented to a resource manager, he would probably seek answers to six questions, namely:

1. What is the extent of the existing problem—that is, where are the present blooms and weed beds to be found?
(2) What causes the problem—that is, what are the relationships between nutrient supply and algal growth, and how do environmental variables affect this relationship?

(3) What are the important sources and sinks for nutrient material?

(4) What development activities are most likely to accentuate the problem?

(5) How can existing problems be eradicated or reduced?

(6) What controls are needed on possible development activities to ensure problems do not arise in the future?

The consideration of questions of this type are fundamental in any attempt to define study objectives.

Desk study. An interesting example of the benefits from an organised collection and assessment of existing information is the 'desk study' recently conducted by the Victorian Ministry for Conservation prior to the commencement of the full-scale Gippsland Regional Environmental Study (Ministry for Conservation, 1977). This desk study took one year and involved all identifiable parties with a statutory concern or other interest in the Gippsland Lakes catchment. Six working groups were formed. Each produced a final report and a considerable number of working papers. The groups were concerned with: existing systems; resource uses and demand pressures; interrelationships; critical parameters; options; and comparison of options. From the information provided it was then possible to identify gaps in data and to make more rational recommendations on major projects to be undertaken. Although the desk study was a large undertaking and went far beyond normal requirements for similar environmental surveys, the concept of rigorously collecting and assessing all available information is nevertheless sound and should be more frequently implemented.

Study objectives. The definition of study objectives follows the identification of real or potential problems. It should preferably be done by a multidisciplinary team, the actual expertise depending somewhat upon the nature of the survey. It should be noted that there are many advantages in involving the public from the earliest planning stage (Cullen, 1977).

It is important that objectives be realistic. Consideration should be given to constraints such as finance, manpower, time and expertise in developing study objectives. It is wasteful to attempt a three-month study of a problem that realistically requires a twelve-month study to solve. Of course it is impossible to foresee all the pitfalls and problems that arise in the actual implementation of a study program, but realistic appraisal can often mean that these are acknowledged and appropriate contingency plans developed.

The formulation of sensible study objectives can only be achieved if the major interrelationships associated with the problem have been identified. One way in which this can be done is to develop a conceptual model of the problem. Such a model will be of great assistance in the definition of the best system indicators to measure. However, a major weakness of many past environmental studies has been the insufficient emphasis placed on the development of conceptual models (or working hypotheses) before the development of objectives. In some cases those involved
in the formulation of study programs would no doubt have either discussed a model or had a mental picture of the major interrelationships involved, but, whatever the case, it is only rarely that the major elements are written down. In our opinion this is an extremely important omission.

Study objectives can often usefully be separated into long- and short-term objectives. The former generally relate to information necessary to provide sufficient understanding of the problem to ensure it is avoided or, at the least, minimised in the subsequent management of the system. Depending upon the information required this may need one to five years of study. Short-term objectives are generally aimed at providing small pieces of information to make the long-term objectives more specific or permit the data collection program to be modified if necessary.

Whatever the nature of objectives, however, the aim of well thought out and realistic study objectives cannot be stressed highly enough. Without a clear and concise set of objectives, environmental studies tend to degenerate into aimless data-gathering exercises.

Assessment methods. The fourth step in Fig. 26 relates to the selection of assessment methods. These are the techniques of data analysis within the stated objectives. They may include the development of a descriptive approach to the problem, the fitting of the data to a quantitative mathematical model, the comparison of data with literature values obtained for similar situations, or the use of factor or multiparameter regression analysis to obtain relationships between the major variables and the use of scientific insight to formulate hypotheses for the relationships. An excellent example of the practical use of a number of these assessment techniques is provided by the U.S. Geological Survey study of the Willametre River basin in Oregon (Rickert et al., 1975; Geological Survey, 1977).

The development of a basic working hypothesis is an essential element in any assessment method. The following was suggested as a hypothesis for algal problem investigations in the Gippsland Lakes (Hart and Snow, 1976). It will serve as a useful example.

Algal problems will develop, provided the major nutrient loads to the system remain constant or increase, and certain environmental conditions exist.

Excessive algal growth is prevented by relatively high summertime turbidities.

Excessive algal problems will exist in poorly flushed embayments.

In shallow, poorly flushed embayments, a major proportion of the nutrient supply is from the sediments.

The major nutrient inputs are via the rivers, with the sediments acting as substantial sinks. Thus, a significant proportion of the input nutrients are trapped within the lakes system.

Selection of indicator parameters. The most difficult task in designing an environmental study program is the selection of relevant and specific indicators of the system's behaviour. Only rarely can the measurement of a single parameter be used to define completely a situation occurring in the system. Table 14 contains a list of the possible information needed and corresponding base indica-
Environmental surveys

Table 14 Information and environmental indicators needed to study algal problems in a lake

<table>
<thead>
<tr>
<th>Information</th>
<th>Indicators</th>
</tr>
</thead>
<tbody>
<tr>
<td>Existing levels of nutrients in water</td>
<td>PO₄-P, total-P, total-N, NO₂-N, NO₂-N, NH₃-N, Si(OH)₄-Si conc. as above x flow</td>
</tr>
<tr>
<td>Existing nutrient loads entering and leaving</td>
<td>PO₄-P, total-P, total-N, NO₂-N, NO₂-N, NH₃-N, Si(OH)₄-Si, total-C, inorganic-C</td>
</tr>
<tr>
<td>Existing nutrient levels in sediments</td>
<td>selection of sample sites</td>
</tr>
<tr>
<td>Variability in concentrations and loads both spatially and temporarily</td>
<td>selection of sampling frequency</td>
</tr>
<tr>
<td>Algal species present</td>
<td>sampling, identification</td>
</tr>
<tr>
<td>Algal biomass present</td>
<td>algal counting, chlorophyll-a</td>
</tr>
<tr>
<td>Environmental variables of importance</td>
<td>light penetration, turbidity, salinity, temperature, depth, wind speed and direction</td>
</tr>
</tbody>
</table>

tors to be measured in a lake subject to algal problems as the result of excess nutrient input. It need hardly be added that there are many examples where measurement of well-known parameters contribute little to the knowledge of system problems. Additionally, there have been situations where measurements have been made of particular parameters (e.g. total dissolved solids and conductivity) that are highly correlated. Considerable savings can often be effected by establishing their relationship statistically, subsequently measuring only the minimum number of relevant parameters, and using the derived relationship to predict others. Other considerations to be made at this stage relate to the number and position of sampling sites and sampling frequency.

Data collection program

The design of any data collection program is inevitably a compromise between the ideal program and that possible within financial and time constraints. However, all too often environmental programs are constrained to produce answers within a totally insufficient time frame. There are two ways to organise the data collection program: (1) A sequential approach in which a short-term but wide-ranging study is used initially to obtain information sufficient to define the broad trends and conditions. The identification of existing or possible problem areas for more extensive study follows. (2) A program involving the simultaneous initiation of a long-term broad-based study and a relatively short-term but intensive study of potential or existing problem areas. This method is very dependent upon the quality of existing data and inevitably involves some judgment of areas for intensive study and the degree of intensity of these studies. Whatever alternative is chosen, it is vital that an effective review and feedback mechanism be set up to ensure regular data assessment so that program modifications can be initiated swiftly. In too many environmental surveys, data assessment is terminal and program modifications are impossible.

Figure 27 shows a possible management scheme for a water quality program. A small (4-5 person) assessment group is envisaged: its responsibility is to
assess all data at, say, three-monthly intervals and to recommend program modifications if necessary. An essential member is the person responsible for day-to-day sampling operations since he is in a position to vary sampling according to local variables such as weather and equipment breakdowns. Obviously the more informed this person is about the implications of data already collected, the better the position he is in to make meaningful changes.

To enable the assessment group to function effectively, analytical data should be placed on computer files as soon as possible. It would then be available for instant recall and for statistical analysis to assist interpretation.

The present organisation of most government analytical laboratories represents a major constraint to rapid modifications in environmental programs. Most government analytical laboratories find it considerably more convenient to analyse samples collected regularly than to undertake analyses of samples collected irregularly (e.g. daily in the summer and at much longer intervals at other times). In the interests of more effective data collection from environmental surveys it is necessary to solve this problem urgently.

**Data interpretation**

Interpretation of data is conducted on the assumption that the data are both *relevant* and *valid*. If not, then the initial study design or the data collection program or both are faulty. In general terms the methods available for interpretation are: comparison with 'standard' data; use of statistical methods; use of models or hypotheses, both qualitative and quantitative; and use of multi-disciplinary teams. Earlier, the advantages of models and the multi-disciplinary team approach have been emphasised. Here brief comment on the use of statistical methods is made.

An example where use of a statistical analysis of data assisted in the understanding of a system is provided by a recent study of the total-P levels in the sediments of Lake Mulwala (Hart et al., 1976). A multi-regression analysis of the data showed that the lake could be segmented into two sections and that the total-P levels could be explained in terms...
of position in lake, organic matter and depth. A simple explanation of this model would be that the main phosphorus input was from the two rivers, the Murray and the Ovens, which deposited a significant proportion of their suspended load in the upper section of Lake Mulwala, thus resulting in the higher total-P levels here. No such trends were apparent from a visual assessment of the tabulated data.

References


PART V

Aquatic fauna
21 Aquaculture

N. M. Morrissy

Introduction
Recently, increasing interest has been shown in Australia in fish-farming or aquaculture, an interest stimulated by the possibility of being able to apply scientific principles to meet the world's food demands and by commercial motivations (Pownall, 1969; Wells, 1969; Anon. 1971a). Smith (1971) captured early Australian enthusiasm when he stated that aquaculture was 'what must be one of the most exciting, challenging and potentially valuable areas of fisheries work'. Public interest has been fanned by the media, and as a result government departments have been flooded by requests for advice. This, as provided, was largely based on ad hoc local knowledge and information from standard texts (cf. Butcher and Thompson, 1947; Anon. 1959a,b; Lake, 1962a,b, 1965; Cowling, 1967; Springall, 1972). However, MacLean's (1975) recent review provides a modern, regional and comprehensive source of information.

The present chapter considers the subject in the light of more recent knowledge; it discusses the present status of Australian aquaculture and a variety of associated topics.

The present status of Australian aquaculture
Trout remain the only fish yet farmed commercially in Australia. Farming of them was first attempted on a limited scale at Buxton, Victoria, in 1960 (Pownall, 1966). Since then, some fifteen commercial trout farms have been developed in south-eastern Australia, although only two are of any size by North American standards (the Hume Weir Trout Farm near Albury [Anon., 1972] and the Blowering Trout Farm (Snowy Mountains Trout Ltd) near Tumut, N.S.W. [Anon., 1975c]). The Blowering Trout Farm processes about 20 tonnes of fish per month from 0.6 ha of water. The Sevrup Trout Farm at Bridport, Tasmania, was the first major venture (Anon., 1964), opening in 1964 after commercial trout farming was made legal in that state (Anon., 1963). As well as marketing processed pan-sized fish, most of these trout farms also cater for tourism (some with pay fishing), sell eyed-ova to other hatcheries and farms and fingerlings for stocking private farm dams, and export disease-free eyed-ova during the off-breeding season of the northern hemisphere.

It is as well to note here that the trout is not native. It has, however, been cultured in Australian hatcheries for over one hundred years in the interests of sport fishing. Nevertheless, the technological and managerial skills needed for trout farming were based upon overseas commercial experience (e.g. Marchant, 1966; Hudson, 1972a,b; Wingate, 1974), and most equipment needed is imported. It should also be noted that special legislation was required to permit the
farming of trout because wild stocks of it (and others of potential significance) have been and continue to be protected from commercial exploitation.

With regard to the present status of aquaculture research in Australia, all current investigations operate on a state basis, although each of the five species most highly rated for inland aquaculture is common to three or more states, viz. trout, the short-finned eel, Murray cod, giant perch and golden perch. Important support for this research is obtained from the Commonwealth Fishing Industry Research Trust Account; this, in turn, is supported from federal funds and monies levied on the fishing industry by state authorities.

For introduced species, research into the production of trout and tench in Tasmanian farm dams has been carried out by CSIRO (Weatherley, 1958a,b, 1959). Considerable practical experience in the hatching of trout is possessed by all fisheries authorities in those states where this species occurs, but the Snobs Creek Freshwater Fisheries Research Station and Hatchery of the Victorian Fisheries and Wildlife Division is the major Australian and southern hemisphere hatchery and station for research on trout (e.g. see Affleck, 1952; Wharton, 1957). This station is also the sole Australian centre with expertise in identification and treatment of fish diseases, notably those of pond-held trout and, more recently, of native fish (Butcher, 1947; Ashburner, 1970, 1973, 1975; Ashburner and Ehl, 1973).

Pertinent research on native fish has concentrated on how to stimulate pond-breeding on a reliable, large-scale and annual basis. The N.S.W. Department of Fisheries has led this field with work on several of the Murray-Darling species at the Narrandera Inland Fisheries Research Station (Dakin and Kesteven, 1938; Lake 1967a; Wyse, 1973; Bowerman, 1975) and on Australian bass (Perca latues colonorum) (Anon., 1975b). Macquarie perch (Macquaria australasica) has been investigated at Snobs Creek (Wharton, 1973), and Victoria has recently established a government warm-water (pilot) fisheries station at Lake Charlegrark in the north-west of the state where introduced Murray cod spawn naturally (Buckmaster, 1974, 1977). Other research by the Victorian Division of Fisheries is currently directed at estimating the availability of elvers for both intensive and extensive eel culture (Beumer, 1976).

The aquaculture potential of freshwater crayfish has been a subject of some interest in Victoria (Beattie, 1972), South Australia and Western Australia. In South Australia, both the Fisheries Department and the University of Adelaide have displayed interest (Fradd, 1974; Coombe, 1976; Lewis, 1976). In Western Australia, reliable annual pond spawnings of the crayfish marron have been obtained since 1969 (Morrissy, 1976b) and experiments to define levels of feeding, density and pond conditions in order to realise high yields from domesticated stocks (Morrissy, 1975) are nearing completion. Considerable research has also been carried out to explore the potential of farm dams for marron culture (Morrissy, 1970, 1974, in press) and on site selection for intensive pond culture (Morrissy, 1976a).

Finally, brief mention is made of some investigations by Hamar Midgely on the breeding of native species in ponds in northern Australia, particularly the spotted barramundi (Scleropages), and of some on-going investigations by the
Queensland Fisheries Service of native fish to stock storages (M. R. MacKin­non, personal communication).

**Extensive or intensive aquaculture?**

The basic aim of aquaculture is to produce more fish from a given body of water than the water would produce naturally. Increased yields may be brought about in a number of ways. These include by stocking young fish (‘seed’) from separately maintained brood stocks (hatcheries), by increasing food availability through fertilisation to increase the amount of natural food available or through direct provision of food, by protection from pollution, predators or pathogens, by enhancing reoxygenation and removal of waste material from the water body, and by rearing single-year classes of fish in ponds which can be drained and thus efficiently harvested. In general, the tighter the control of fish production in these ways, the lower are land and lab­our costs leading to even greater effi­ciency in the total operation.

Intensive aquaculture refers to those situations where yields are high, controls are tight, and production takes place in man-made ponds or other structures (Neal, 1973). When yields are low, and aquaculture is simply the exploitation of an unconfined species in a natural body of water with little control, the practice is referred to as extensive aquaculture.

MacLean (1975) recommended the development of intensive aquaculture for private commercial ventures in Australia, as did Neal (1973) for similar ventures elsewhere. However, certain characteristics of important Murray-Darling fish are disadvantageous in intensive aquaculture (further details are given below). Trout do not have such disadvantages, but even they could only be intensively cultured in Australia on a commercial basis because the return on the large financial investment needed was assured by proven feasibility studies elsewhere. Thus, the development of intensive aquaculture using native Australian fish is unlikely in the near future.

As predicted by Buckmaster (1973), an awareness of the high investment risk of intensive aquaculture has led to proposals from commercial interests to use existing lakes or reservoirs for aquacul­ture, with monopoly control by the entrepreneur involved. No doubt such proposals will continue to be made. However, in lakes where the species to be exploited occurs naturally, the proposals should be strenuously resisted since the principles of aquaculture apply only marginally. It is also undesirable to permit introductions of species of fish to lakes or to permit other manipulations such as the destruction of unmarketable but natural fish competitors and of bird predators. And the introduction of desirable species to public man-made storages, probably requiring continuing hatchery maintenance, should be government sponsored, or at least some proportion of public interest retained. In summary, private monopolistic use of common property resources (sensu Hardin, 1976) should be resisted and responsibility for their management should remain with government.

The question of aquaculture in farm dams remains for discussion since these are neither a common property resource nor require any special investment (having been built for other, agricultural, purposes). Weatherley (1967) suggested they had considerable scope for aquacul­ture. However, since most are filled only seasonally and only limited yields
Aquatic fauna

are possible, it is doubtful whether even the high yields of detritivorous crayfish (Woodland, 1967; Morrissy, 1974, in press) can be used commercially. Nevertheless, in isolated areas the production of fish for home consumption is an important incentive for aquaculture in these small water bodies.

The legalities of exploitation of wild species with scientific, cultural or recreational value can be resolved—at least in theory—by confining exploitation to man-made ponds on lease or freehold property. In the latter, the species can be considered as domestic stock, that is, as privately owned. The application of this philosophy has been successful in intensive marron aquaculture in Western Australia where extensive aquaculture is not permitted.

The requirements for aquaculture

Water needs for intensive aquaculture. An important concern for Australian aquaculture is the volume and source of water needed. Volumes required are large for intensive aquaculture, and even a ‘small’ trout farm, producing say 100,000 kg yr\(^{-1}\), on a harvest rate of 0.005 kg m\(^{-3}\) requires 20 × 10\(^8\) m\(^3\) of water annually or 55,000 m\(^3\) daily. Volumes required may be somewhat reduced by introducing water recirculation following treatment but only routine mechanical aeration is generally practicable. Further reduction may be effected by using a hardy species, by domesticating species to crowded pond conditions, and by lessening food wastage. But there is clearly a limit to which these measures can be taken.

The nature of the water source is of concern because of the unpredictability and uncertainty of rainfall over much of Australia (Australian Water Resources Council, 1976; Heathcote, 1973). Supplies are most cheaply and assuredly obtained by interposing fish farms between irrigation projects and their upstream storages. There appears to be a largely untapped potential for such integration with established irrigation projects in south-eastern Australia, in coastal eastern Queensland, and at Lake Argyle in the Kimberleys. In some places, however, integration is not possible because irrigation is markedly seasonal and no off-season discharge is permitted from partially filled dams in years of drought (for example, in south-western Australia).

Temperature and site selection. Ideally, sites for aquaculture should be where temperatures favour rapid growth in both winter and summer. The ideal is usually difficult to achieve, and site selection is more likely to be a compromise between the desirable and practicable. There are, nevertheless, a number of constraints with regard to temperature and site selection that cannot be ignored. High summer temperatures promote disease, favour excessive activity (and thus the unnecessary consumption of energy), are correlated with low oxygen levels, and may directly cause mortality. Low winter temperatures, on the other hand, may involve long periods of uneconomically slow or negligible growth.

Initial tendencies are usually to select the optimum temperature for maximum growth rate, which is closer to the extreme of high than low temperature in the favourable range for survival of a species. However, this decision neglects the increased risk of mass mortality because of closer proximity to the range of sublethal high temperatures and the
increased likelihood of disease and oxygen deficiencies for crowded pond fish. In addition, food conversion rates may be poor. Without detailed study of these factors, the mean temperature desirable for aquaculture is best estimated from the mean annual water temperature at the centre of the natural range of distribution of a species, i.e. where the species is most successful. Least daily and annual variation in temperature is obtained in coastal rather than in far inland waters.

Buckmaster (1973) has suggested the higher growth rates and production in warmer areas will lure fish farming ventures in Australia to the north. If this happens it would obviously be beneficial for the much needed local breeding and supply of ornamental pond and aquarium fish for hobbyists in Australia: Carassius spp. (goldfish and carp) cease pond growth during winter in southern Australia, while the cost of the heated water supply required for large-scale exotic tropical fish breeding in the south is prohibitive. Such a venture is currently being set up in the Ord irrigation area near Kununurra.

The market

Trout. It is significant that the only fish to be successfully farmed on a commercial basis in Australia, namely trout, is a luxury product and does not compete in markets with fish from wild stocks (MacLean, 1975). Moreover, products from the first Australian trout farm (Bridport, Tasmania) used the market developed by Japan with imported frozen fish. Even so, between 1967 and 1969, capital expenditure, drought and competition from imported products prevented profitable operation of the Tasmanian farm (Anon., 1971b), and it was necessary to seek government assistance. It received this in September 1971 in the form of an imposition of a small import tariff on fresh, chilled or frozen trout. Later, in 1975, price competition from imports was effectively eliminated by a quarantine proclamation prohibiting imports of unprocessed salmonids for disease control reasons (Ashburner, 1976a,b).

Crayfish. Large markets for imported freshwater crayfish have developed in Europe. This is particularly so in Sweden (for processed products) and France (for live animals) where local supplies have been reduced by introduced diseases (Unestam, 1973), over-exploitation, and other factors. Interest in the market has been stimulated by the high prices commanded by the local product: Adelkräftor, the native Swedish crayfish (Astacus astacus), for example, cost about $14 per kilogram in 1976. However, a number of problems relate to penetration of European markets. Thus, for Sweden, there is the problem of the present disparity in price between the scarce but preferred Astacus astacus and imported crayfish, the problem of assuring a firm annual supply, difficulties in maintaining a high degree of quality control in cooking, and variation in size.

Giant perch (and freshwater prawns). Although giant perch (barramundi) is a well-known and highly marketable fish in Australia, intensive aquaculture is unlikely to compete with conventional fishing for the species in the Gulf of Carpentaria and Papua New Guinea. Giant perch and the long-legged freshwater prawn (unknown on Australian markets) of northern Australia also
Aquatic fauna

occur in South-east Asia where extensive aquaculture is widespread on both species and intensive aquaculture of the prawn is being developed overseas, notably in Hawaii (Shang and Fujimura, 1977). Intensive aquaculture of the prawn was first proposed in Australia by Fish Farms International Ltd (see Pownall, 1975), but this venture, at Kununurra in the Ord irrigation area, appears to have lapsed.

Eels. Exploitation of wild stocks of eels in Australia was largely stimulated in the 1960s by overseas demands, particularly for live elvers for fattening in Japan, and by local demand from European immigrants. The first Australian attempt to pond culture eels appears to have been at the Racovolis Prawn Culture Farm at Port Broughton, South Australia.

Native fish other than giant perch. The Australian predilection for luxury ‘sea-foods’ together with increasing local affluence (Doyle, 1976) may permit the economic marketing of expensive native fish from fish farms. At present, however, most Murray cod, callop and silver perch are sold to local inland markets near the point of capture and only a trickle comes through to metropolitan markets.

Giant perch (silver barramundi) and northern water storages
The creation of dams on northern and north-eastern Australian rivers effectively eradicates upstream stocks of barramundi because this species spawns in estuaries. Since there is no other northern fish which has the recreational and edible qualities of barramundi, the result has been that man-made water storages in these areas are virtually ‘fishless’ so far as local anglers are concerned. It is not surprising, then, that pressure has been exerted for the introduction of a ‘worthwhile’ fish to these storages.

Recognising this, in 1969 the Queensland government proposed the introduction of Nile perch (*Lates niloticus*), a look-alike congener of barramundi which, however, spawns in fresh waters. The proposal met with considerable concern (Williams, 1970) and was subjected to an intensive review—the first of its kind in Australia for a fish species—by a Commonwealth Advisory Committee on Imports of Live Food and Sport Fish, reporting to the Commonwealth Standing Committee on Fisheries and thence to the Australian Fisheries Council. The committee prepared synopses for both species and an impact statement on the value of the proposal and the possible ecological consequences of a new predatory fish species. A long-term, extensive program of investigation was stipulated for a trial shipment of Nile perch to be held under quarantine conditions at Walkamin, Queensland. Basically, the proposed investigation, also involving barramundi, can be viewed retrospectively as aquaculture research. However, Queensland withdrew the proposal in 1975 before the documentation was brought into the public arena for final debate on the issue (Anon., 1975d).

The introduction of Nile perch was not favoured by authorities in the Northern Territory, but those in Queensland viewed the introduction initially as the cheapest and most expeditious means of stocking northern storages. Other alternatives considered for stocking northern storages were by planting native fish and by building fish ladders to permit migra-
tion of barramundi. However, no suitable native substitutes for barramundi were found, and fish ladders were impracticable since barramundi do not move upstream with salmonid zeal. The remaining alternative was to culture barramundi in hatcheries and stock storages artificially.

The last alternative has now been taken up by a private company in Western Australia, and to permit it to do so the state government has passed the *Fish Farming (Lake Argyle) Development Agreement Act, 1976*, allowing the company to breed, rear and farm barramundi for commercial purposes near and in the lake (Anon., 1975a). The company proposes to breed and rear barramundi to a fingerling stage and release not less than 200,000 in the lake annually. Fishing by tourists will be permitted to a limit of 10 per cent of the company’s annual catch or 100,000 kg annually. The proposed commercial arrangements are similar in concept to that of ‘ocean ranching’ of anadromous salmonids, in which hatchery-bred fish migrate to the sea, grow in the oceans, and are harvested when they return to the natal hatchery stream. In the case of ‘lake ranching’ of barramundi, harvesting is proposed when two- or three-year old fish move to the outlet of the lake in order to migrate downstream to estuaries to spawn.

It is to be noted that a number of uncertainties are associated with the success of this proposed venture. Firstly, hatchery production of young barramundi is not yet an established routine procedure, although progress has been made in Thailand (FAO, 1975) and eventual success with this highly fecund species seems assured using hormones. Secondly, the productivity of the lake itself remains unevaluated. And thirdly, appropriate stimuli for downstream migration of barramundi may not be generated in a lake so large (5.720 x 10^6 m^3); river flooding is probably the normal stimulus. Notwithstanding these uncertainties, it remains true that barramundi is a valuable resource in northern and north-eastern waters, tolerates close confinement, is highly fecund, and can grow rapidly in suitable conditions.

**Aquarium fish and the problem of exotic disease**

The problems posed by the importation to Australia of vast numbers of aquarium fish are discussed at length in Chapter 23. Here, only those aspects of this matter which relate to aquaculture need consideration.

At present, most aquarium fish sold in Australia are imported rather than bred locally, but because of the immense number of fish involved (> 12 million in 1975-76), absolute numbers of locally bred fish remain high. Thus, in 1971-72, 1,127,000 goldfish were bred on four fish farms in Victoria, N.S.W. and Queensland (800,000 on the Boolara fish farm alone). This number represented only about 25 per cent of the total annual market demand for this fish. Apart from goldfish, many of the popular aquarium species could also be bred in Australia (24 of the 30 most popular species according to a survey in 1967 by the Commonwealth Advisory Committee on Imports of Live Aquarium Fish). There is, therefore, a clear opportunity in this area for local culture of aquarium fish. It does not appear to have been taken for the simple reason that fish can be imported extremely cheaply from South-east Asia (Singapore and Taiwan), where many are cultured as a cottage industry, or from the Pacific region.
Aquatic fauna

(Philippines and Fiji). Moreover, retailers can import directly without involving Australian wholesalers.

Although it may be more profitable to import aquarium fish, there are several dangers associated; the principal one is the alarming opportunity which the present haphazard import procedure offers for the introduction of highly undesirable fish and associated or incidental pathogens. The matter has been under the scrutiny of the Commonwealth Advisory Committee for the past decade or so but without satisfactory resolution. This chapter is not the place to enter the debate, but it may be noted here, with regard to crayfish aquaculture, that haphazard importation poses enormous dangers to the Australian crayfish fauna in the form of the northern hemisphere crayfish plague to which Australian species are known to be extremely susceptible (Unestam, 1973). Already, live specimens of north American crayfish have been confiscated at Perth airport from one of many shipments of aquarium fish. Their discovery, it may be noted, was more by chance than design.

Marron aquaculture

The marron is a large species of freshwater crayfish endemic to south-western Western Australia and held in high local esteem. It supports a significant amateur fishery protected from commercial exploitation (1975-76 season: total catch, 470,000 kg; licences issued, 20,000) (Morrissy, 1978a). However, investigations have shown that stocks have disappeared from the upper and middle reaches of the four major south-western rivers due to agricultural pollution (Morrissy, 1978b). And stocks in the lower reaches are threatened by a variety of events (dam construction, forest clearing, irrigation, mining). Half the present total catch, in fact, is from a few large water supply storages between Perth and Bunbury. Moreover, catching effort has risen recently to unacceptable levels. Marron aquaculture may have an important role to play in this situation for it could provide alternative stocks of (domesticated) marron for fishing and sale to tourists and city people and thus relieve fishing pressure on wild stocks. To aid this, in December 1976 the marron was scheduled by the Western Australian government as a ‘farm fish’, and thus permitted to be cultured for commercial purposes on registered fish farms. By mid-1977 there were sixteen such farms.

Prior to this legislation, marron had been available (since 1966) from the Pemberton Fish Hatchery (Department of Fisheries and Wildlife). They were sold to stock farm dams over a wide area of the south-west (from Geraldton to Esperance) to provide for local domestic consumption (Morrissy, 1970). Research on the dams provided the basis for advice on water quality management problems and stocking techniques, and, when the advice is followed, this form of extensive aquaculture could realise harvests of 400 kg ha⁻¹ on average every two years (Morrissy, 1974; in press). Research on intensive aquaculture of marron has produced useful information for site selection and breeding methods; it also has indicated that crops of 3,000 kg ha⁻¹ are achievable in one or two years (Morrissy, 1976a,b). Nonetheless, whereas the social incentives for commercial aquaculture of marron are high, the profitability and feasibility of private ventures remain doubtful (Morrissy, 1977). The whole question of mar-
ron aquaculture is considered at greater length by Morrissy (1978a).

Aquaculture of Murray-Darling fish
Early attempts to culture Murray-Darling fish, particularly the Murray cod, aimed to produce fish to stock the Murray-Darling system to combat overfishing and excessive fluctuations in population density between years (Dakin and Kesteven, 1938). This aim is no longer pursued, and, following the success of culturing experiments (Lake, 1967a,b), emphasis is now placed on stocking circumscribed waters such as farm dams and public fishing lakes.

The greatest production of fingerlings is at the N.S.W. State Fishery hatchery at Narrandera where a semi-commercial indigenous fish breeding program began in 1970 (Anon., 1970). Hormone treatment, rather than pond stimulation of natural spawning stimuli, is used to induce breeding (Wyse, 1973; Anon., 1977). In total, N.S.W. State Fisheries have been able to provide since 1973 about 150,000 fish for over 1,000 property owners and 3,000 dams. In the dams, fish growth is about twice that expected in natural waters.

Despite the evident success of this extensive aquaculture program, there are several problems in the more intensive culture of Murray-Darling fish (Murray cod, golden perch, silver perch, and catfish). The fish are carnivorous with low food conversion rates; it is difficult to get cod and golden perch to accept artificial food; none of the species takes well to crowded pond conditions and all become more subject to disease when crowded; it takes at least two years for fish to grow to marketable size; and, finally, fingerlings are expensive (20 to 40¢ each; cf. 2.5¢ per catfish in the U.S.). Nonetheless, these problems, there is a ready market for native fish fingerlings, and a number of hatcheries in addition to that at Narrandera are operational. The Tinbeerwhah Hatchery near Nambour, Queensland, produces mainly golden and silver perch fingerlings; 6,000 golden perch were produced in 1976 (Anon., 1976). The Tabbita Hatchery near Griffith, N.S.W., breeds golden perch and sold its whole production in 1976 (5,000 fish) without the need to advertise (Bowerman, 1977). The Murray Cod Hatcheries of Australia near Wagga Wagga, N.S.W, produced 30,000 fish in 1975 (Parry, 1973, 1977; McLaren, undated). And the Warm-Water Fisheries Research Station of the Victorian Fisheries and Wildlife Division at Lake Charlegrark in north-west Victoria produces Murray cod fingerlings for stocking purposes.

Finally, brief mention should be accorded the European carp for, while not native to the Murray-Darling system, it is certainly widespread there now (see Chapter 22). Some form of carp aquaculture may be viable in Australia since the current market price as pet food is 20¢ kg⁻¹. Conjunctive culture of carp and rice is being investigated at Yanco, N.S.W. (Woodward et al., 1977), and annual yields could reach 1,000 kg ha⁻¹.

Acknowledgments
H. Midgley is acknowledged for unpublished information.

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22 Management of freshwater fish and fisheries

D. A. Pollard, L. C. Llewellyn and R. D. J. Tilzey

Introduction
By world standards Australia has only one moderately large river system, a fact reflected in the depauperate freshwater fish fauna of the continent. Only in this system, the Murray-Darling, and in some of the larger rivers of the tropical north, are there significant stocks of large indigenous fishes; most of Australia’s 180 or so freshwater fish species are small and of little economic or recreational importance. The entire Australian commercial freshwater fish and crustacean catch (excluding giant perch or silver barramundi, which is caught mainly in estuarine waters) currently averages only some 1,000 or so tonnes live weight per annum (cf. the total national annual fish and crustacean catch of almost one hundred thousand tonnes); almost all of this freshwater catch (the only significant exceptions being eels and European carp in the south-eastern coastal drainage) is from the Murray-Darling river system drainage (see Table 15).

For recreational fisheries (angling), again the main indigenous species, with the exception of silver barramundi in tropical northern rivers and Australian bass in south-eastern coastal rivers, are the larger and predominantly percichthyid species of the Murray-Darling river system. In addition to native fish, a significant segment of the recreational fishery is based on a few introduced salmonid species, mainly in the cooler upland waters of the south-eastern corner of the mainland and in Tasmania.

In spite of its small size in terms of species numbers and the relative paucity of commercially and recreationally important forms, the Australian freshwater fish fauna includes a number of species of considerable scientific and aesthetic interest. Fisheries management should of necessity principally concern itself with those species of economic and recreational importance but must also involve the conservation and protection of the freshwater biota as a whole. Popular accounts of some of the more biologically interesting freshwater fish and some of the threats facing their aquatic habitats have been given by Whitley (1955) and Pollard and Scott (1966).

Lake (1971) listed 231 species of fish as occurring in the fresh waters of Australia and commented that 49 of these are essentially marine forms which only rarely or occasionally enter fresh water, another twenty require estuarine or marine conditions for spawning or completion of some other stage of their life cycles, and sixteen are introduced forms. Allowing for approximately another 20 which may not be distinct

The authors dedicate this chapter to the memory of their friend and colleague the late John Lake, whose contributions to the fields of Australian freshwater fisheries management and the conservation problems of our native freshwater fish fauna are immeasurable.
Table 15 Commercial catches of Australian freshwater fishes (figures from Australian Bureau of Statistics, Canberra)\(^1\)

<table>
<thead>
<tr>
<th>Species</th>
<th>Five-year means (metric tonnes)</th>
<th>Percentage catch by state or territory(^2)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1961/65(^3)</td>
<td>1966/70(^3)</td>
</tr>
<tr>
<td>Silver barramundi</td>
<td>195</td>
<td>263</td>
</tr>
<tr>
<td>Short-finned eel(^5)</td>
<td>69</td>
<td>160</td>
</tr>
<tr>
<td>Australian bass(^6)</td>
<td>1-2</td>
<td>0-2(^7)</td>
</tr>
<tr>
<td>Tasmanian whitebait</td>
<td>15</td>
<td>33</td>
</tr>
<tr>
<td>Golden perch</td>
<td>282</td>
<td>164</td>
</tr>
<tr>
<td>Murray cod</td>
<td>73</td>
<td>45</td>
</tr>
<tr>
<td>Silver perch</td>
<td>21</td>
<td>24(^8)</td>
</tr>
<tr>
<td>Freshwater catfish</td>
<td>15</td>
<td>27(^8)</td>
</tr>
<tr>
<td>Bony bream</td>
<td>—</td>
<td>238(^9)</td>
</tr>
<tr>
<td>Macquarie perch</td>
<td>2-1</td>
<td>0-6(^10)</td>
</tr>
<tr>
<td>English perch</td>
<td>45</td>
<td>44(^8)</td>
</tr>
<tr>
<td>European carp and goldfish(^12)</td>
<td>6</td>
<td>15</td>
</tr>
<tr>
<td>Tench</td>
<td>14</td>
<td>72(^8)</td>
</tr>
<tr>
<td>Rainbow trout</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Others</td>
<td>3</td>
<td>13</td>
</tr>
<tr>
<td>Total freshwater fishes</td>
<td>545</td>
<td>731</td>
</tr>
<tr>
<td>Yabby</td>
<td>—</td>
<td>25</td>
</tr>
<tr>
<td>Murray crayfish</td>
<td>3-5</td>
<td>—</td>
</tr>
</tbody>
</table>

\(^1\) All freshwater catch figures exclude Queensland catches, for which no figures are available (n.a.) except those for silver barramundi (which are caught in estuarine waters).


\(^3\) South Australian catches of species other than golden perch and Murray cod were not recorded before 1968-69.

\(^4\) 1971-75 means exclude Victorian catches, as no Victorian catch figures are available from 1972-73 onwards.

\(^5\) Estimated catches (of 15 metric tonnes per annum, from 1971-72 and 1972-73 figures) of marine conger eels, which were included under this category in the official statistics, have been extracted from the 1966-70 figures.

\(^6\) These catches are probably mainly estuary perch.

\(^7\) Mean for first year (1966-67) only.

\(^8\) Means for 4 years only, excluding 1967-68 figures.

\(^9\) Mean for last 2 years only.

\(^10\) Mean for first 3 years only.

\(^11\) Mean for last 3 years only.

\(^12\) Figures for these two species (and for crucian carp, which may not occur in Australia) have been lumped due to taxonomic confusion. Figures for percentage catches by state are for 1971-72 only; estimates for the 1972-76 period would be closer to 45% for Vic., 35% for S.A. and 20% for N.S.W.

\(^13\) Mean for 1 atypical year (1974-75) only (previous catches of this species were insignificant).

\(^14\) From 1973-74 onwards New South Wales has accounted for approximately 30% of the total yabby catch.

\(^15\) Most of this catch was comprised of yabbies.
species, he estimated that only 130 or so species spend their entire life cycles in Australian fresh waters. More recent estimates (Hoese, 1977, unpublished information) put the number of species which may be found in Australian fresh waters at approximately 235, of which 65 are primarily marine forms. Between 140 and 150 of the remaining species may complete their entire life cycles in fresh water.

Of these 140 or so ‘true’ freshwater fish, almost all belong to essentially marine families, and Australia possesses almost no indigenous ‘primary division’ freshwater fish (sensu Myers, 1938 and Darlington, 1957) such as the ostariophysian carps (Cypriniformes) and catfishes (Siluriformes) which make up approximately seven-eighths of the primary division freshwater fish of the other four continents.* The exceptions here are the Queensland lungfish Neoceratodus forsteri (Family Ceratodidae) and the two species of spotted barramundi Scleropages spp. (Family Osteoglossidae) which must be regarded as primary division freshwater fish on the criteria of Myers and Darlington.

In the present chapter the most important elements of the fish faunas of the main drainage systems are discussed in terms of their management for commercial, recreational and conservation purposes. To facilitate discussion, three main types of aquatic environment are recognised, namely coastal rivers and streams, inland rivers and lakes, and mountain streams and lakes. When a species inhabits more than one of these environments, it is discussed under the heading of that environment type where it is most abundant or important. Brief discussion of freshwater crayfish (crustaceans) is also made in this chapter since the fisheries based on a few of them are now of some significance.

The main drainage systems

Lake (1971) recognised fifteen separate drainage systems, but in this chapter these are grouped as follows (see also Fig. 28):

A. Tropical northern coastal (including Lake’s north-eastern slopes, Gulf of Carpenteria, Timor Sea and Indian Ocean drainages).
B. Temperate south-eastern coastal (including Lake’s south-eastern slopes, No. 13 unco-ordinated, and Adelaide coastal drainages).
C. Tasmanian (as in Lake).
D. Temperate south-western coastal (Lake’s south-western slopes drainage).
E. Murray-Darling (including Lake’s Murray-Darling, Wimmera, and Bulloo drainages).
F. Lake Eyre internal (Lake’s Lake Eyre, and No. 14 unco-ordinated drainages).
G. Central Western Desert (Lake’s No. 15 unco-ordinated drainage).

Coastal rivers and streams comprise systems A, B, C and D; inland rivers and lakes comprise E, F and G; and mountain streams and lakes occur mainly along the boundaries of B and E, in C, and partly in D.

The largest number of freshwater fish species occurs in the tropical northern coastal drainage system, followed by the

* The ariid (fork-tailed) and plotosid (eel-tailed) catfishes found in Australian fresh waters belong to the only two ostariophysian families which are predominantly marine and, like most of the continent’s other freshwater fish, these species have been relatively recently derived from marine ancestors.
temperate south-eastern coastal, Murray-Darling, Lake Eyre internal, Tasmanian and temperate south-western coastal drainage systems. Although it covers almost a third of the continent, very few freshwater fish are known to occur in the Central Western Desert area. A general description of each of the main drainage systems and a listing of their major rivers and fish faunas has been given by Lake (1971).

Coastal rivers and streams

A. Tropical northern coastal drainage system. About 110 to 120 species of native freshwater fish occur in this system but only one, the catadromous silver barramundi, is of any commercial or really significant recreational importance. Another one or two species have some potential commercial value, and another 10 to 15 are of actual or potential angling interest. Some 15 to 20 species may be in need of protective environmental management if their stocks are to be conserved, though very little is known about most of the species in this tropical area (see Table 16). Only the barramundi needs detailed discussion.

Silver barramundi (Lates calcarifer: Family Centropomidae)
The silver barramundi, or giant perch, is most closely related to the strictly freshwater Nile perch (Lates niloticus) of Africa, though its life cycle more closely resembles that of the related North American snook (Centropomus undecimalis) in that it must migrate from fresh to saline waters to spawn. It occurs in rivers and estuaries over much of the tropical Indo-Pacific area between south China and the Persian Gulf, growing to
### Table 16 Freshwater fishes of coastal rivers and streams

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Edible value</th>
<th>Angling value</th>
<th>Commercial value</th>
<th>Conservation status</th>
<th>Occurrence in drainage systems</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Geotria and Mordacia spp.</strong></td>
<td>Lampreys</td>
<td>(+)</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>BCD (1-2, 7)</td>
</tr>
<tr>
<td><strong>Neoceratodus forsteri</strong></td>
<td>Queensland lungfish</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D (protected)</td>
<td>A (3)</td>
</tr>
<tr>
<td><strong>Anguilla australis</strong></td>
<td>Short-finned eel</td>
<td>+</td>
<td>(+)</td>
<td>—</td>
<td>—</td>
<td>BC (1-2)</td>
</tr>
<tr>
<td><strong>A. reinhardti</strong></td>
<td>Long-finned eel</td>
<td>+</td>
<td>(+)</td>
<td>(+)</td>
<td>—</td>
<td>BA (2-3)</td>
</tr>
<tr>
<td><strong>A. obscura</strong></td>
<td>South-Pacific eel</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>A (3)</td>
</tr>
<tr>
<td><strong>A. bicolor</strong></td>
<td>Northern eel</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>A (4-6)</td>
</tr>
<tr>
<td><strong>Potamalosa richmondia</strong></td>
<td>Freshwater herring</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>C B A (2)</td>
</tr>
<tr>
<td><strong>Scleropages leichhardti</strong></td>
<td>Spotted barramundi</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>D</td>
<td>A (3)</td>
</tr>
<tr>
<td><strong>S. jardini</strong></td>
<td>Northern spotted bar-ramundi</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>D</td>
<td>A (4-5)</td>
</tr>
<tr>
<td><strong>Retropinna spp.</strong></td>
<td>Australian smelts</td>
<td>(+)</td>
<td>—</td>
<td>(+)</td>
<td>?</td>
<td>CBA (1-3)</td>
</tr>
<tr>
<td><strong>Galaxiidae spp.</strong></td>
<td>Galaxias</td>
<td>(+)</td>
<td>—</td>
<td>(+)</td>
<td>D</td>
<td>CBAD (1-3, 7-8)</td>
</tr>
<tr>
<td><strong>Prototroctes maraena</strong></td>
<td>Australian grayling</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>A</td>
<td>BC (1-2)</td>
</tr>
<tr>
<td><strong>Lovettia shell</strong></td>
<td>Tasmanian whitebait</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>B</td>
<td>C (1)</td>
</tr>
<tr>
<td><strong>Hexanematichthys spp.</strong></td>
<td>Fork-tailed catfishes</td>
<td>+</td>
<td>+</td>
<td>(+)</td>
<td>—</td>
<td>AB (2-6)</td>
</tr>
<tr>
<td><strong>Anodontiglanis dahl</strong></td>
<td>Toothless catfish</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>?</td>
<td>A (4-5)</td>
</tr>
<tr>
<td><strong>Porochilus obbesi</strong></td>
<td>Obbes catfish</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>A (4-5)</td>
</tr>
<tr>
<td><strong>Tandanus bostocki</strong></td>
<td>Freshwater cobbler</td>
<td>+</td>
<td>(+)</td>
<td>—</td>
<td>?</td>
<td>D (7)</td>
</tr>
<tr>
<td><strong>Neosilurus spp.</strong></td>
<td>Tandans</td>
<td>(+)</td>
<td>(+)</td>
<td>—</td>
<td>?</td>
<td>A (3-6)</td>
</tr>
<tr>
<td><strong>Strongylura krefft</strong></td>
<td>Freshwater longtom</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>A (3-5)</td>
</tr>
<tr>
<td><strong>Melanotaeniidae spp.</strong></td>
<td>Rainbow fishes</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>AB (2-6)</td>
</tr>
<tr>
<td><strong>Pseudomugil spp.</strong></td>
<td>Blue eyes</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>AB (2-5)</td>
</tr>
<tr>
<td><strong>Atherinidae spp.</strong></td>
<td>Silversides</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>AB (2-6)</td>
</tr>
<tr>
<td><strong>Lates calcarifer</strong></td>
<td>Silver barramundi</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>B</td>
<td>A (3-6)</td>
</tr>
<tr>
<td><strong>Ambassidae spp.</strong></td>
<td>Glassfishes</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>AB (2-5)</td>
</tr>
<tr>
<td><strong>Bostockia porosa</strong></td>
<td>Nightfish</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>D (7)</td>
</tr>
<tr>
<td><strong>Percalates colonorum</strong></td>
<td>Estuary perch</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>?</td>
<td>C BC (1-2)</td>
</tr>
<tr>
<td><strong>P. novaemaculeatus</strong></td>
<td>Australian bass</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>B BA (2-3)</td>
</tr>
<tr>
<td><strong>Pingalla Gilberti</strong></td>
<td>Gilberts grunter</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D A (4)</td>
</tr>
<tr>
<td><strong>Other Theraponidae spp.</strong></td>
<td>Grunters</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>? A (3-6)</td>
</tr>
<tr>
<td><strong>Kuhlia rupestris</strong></td>
<td>Jungle perch</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>C</td>
<td>A (3)</td>
</tr>
<tr>
<td><strong>Other Kuhliidae spp.</strong></td>
<td>Pigmy perch</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>BCAD (1-3, 7)</td>
</tr>
<tr>
<td><strong>Kurandapogon blanchardi</strong></td>
<td>Blanchards perchlet</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (3)</td>
</tr>
</tbody>
</table>

---

1 Excluding predominantly estuarine or marine forms entering fresh water and species introduced from inland drainages.

2 Conservation status code according to Lake (1971): A. seriously threatened species; B. species whose distribution and/or abundance have been considerably reduced; C. species not yet in a serious position but threatened by changes now taking place; D. species restricted to one or two river systems and/or with very patchy or discontinuous distributions; ?, conservation status unknown.

3 See text and Fig. 28 for explanation to code of drainage systems.

+, of significant value; (+), of marginal or potential value only; —, of no economic value.
Table 16 (continued)

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Edible value</th>
<th>Angling value</th>
<th>Commercial value</th>
<th>Conservation status</th>
<th>Occurrence in drainage systems</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Toxotes chatareus</em></td>
<td>Archer fish</td>
<td>(+)</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>A (3-5)</td>
</tr>
<tr>
<td><em>Protoxotes lorentzi</em></td>
<td>Primitive archer fish</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>BC (1-2)</td>
</tr>
<tr>
<td><em>Pseudaphritis urvilli</em></td>
<td>Congolli</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>?</td>
<td>BA (2-6)</td>
</tr>
<tr>
<td><em>Hypseleotris spp.</em></td>
<td>Carp gudgeons</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>A (3-5)</td>
</tr>
<tr>
<td><em>Oxyeleotris lineolatus</em></td>
<td>Sleeper</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td><em>Ophiocara aperoros</em></td>
<td>Snakehead gudgeon</td>
<td>+</td>
<td>+</td>
<td>—</td>
<td>?</td>
<td>A (3-5)</td>
</tr>
<tr>
<td><em>Lindemanella iota</em></td>
<td>Bumblebee</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (3)</td>
</tr>
<tr>
<td>Other Eleotridae and Gobiidae spp.</td>
<td>Gudgeons and gobies</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>?</td>
<td>ABD (2-7)</td>
</tr>
<tr>
<td><em>Kurtus gulliveri</em></td>
<td>Nursery fish</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (4-5)</td>
</tr>
<tr>
<td><em>Brachirus salinarum</em></td>
<td>Salt-pan sole</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (4)</td>
</tr>
<tr>
<td><em>B. selheimi</em></td>
<td>Freshwater sole</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (3-4)</td>
</tr>
<tr>
<td><em>Aseraggodes klunzingeri</em></td>
<td>Tailed sole</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>D</td>
<td>A (5)</td>
</tr>
</tbody>
</table>

Adults generally live in the brackish and freshwater reaches of large, slow-flowing coastal rivers and associated billabongs and lagoons and migrate downstream in late spring and summer during floods to spawn in coastal waters near estuaries. Small juveniles migrate upstream in late summer around the end of the wet season. This species cannot complete its life cycle in fresh water, and no self-propagating landlocked populations are known. Adult fish are predatory, feeding mainly on smaller fish and crustaceans; small juveniles also consume some insects and occasionally some plant material in fresh waters.

The silver barramundi is a magnificent sporting fish because of its size, strength, speed, and often spectacular performance when hooked. It can be taken on large artificial lures and on live prawn and fish baits in both fresh and brackish waters (Pollard, 1969). It is also probably the finest food fish in northern Australia, and although it is not (at least legally) fished commercially in fresh water, it supports significant commercial mesh-net and trap fisheries in the estuarine river mouths, especially in the Northern Territory.

Australia wide, the commercial catch of silver barramundi is currently around 1,500 tonnes per annum, the bulk coming from the Northern Territory, most of the remainder from north and central Queensland, and a very small proportion from the northern part of Western Australia. The fishery for silver barramundi in Queensland was discussed by Dunstan (1959), in the Northern Territory by Cunningham (1965), in Western Australia by Morrissy (1969), and in New Guinea by Dunstan (1962) and Reynolds and Moore (1973). Although commercial pond culture of barramundi is not yet practised in Australia, pond culture is of some economic importance in parts of Asia, particularly Thailand (Yingthavorn, 1951).

While the fishery in the Northern Ter-
teritory is expanding at a rapid rate, the proportion of the catch taken from Queensland waters (particularly on the east coast) is declining. (The Queensland catch fell from 46 per cent of the Australian total in 1972-73 to 25 per cent in 1976-77 and averaged only 32 per cent during the five-year period ending 1976-77, cf. figures in Table 15.) This decline is no doubt mainly due to the increasing number of coastal rivers being dammed for water storages, and thus resulting in the disruption of the barramundi’s catadromous life cycle. Prevention of the upstream migration of juvenile fish results in the species’ subsequent exclusion from the waters above these barriers. The control of flooding and diversion of water by these dams probably also affects populations of barramundi downstream of these barriers. Dunstan (1955) recorded considerable fish mortality, and in particular of barramundi, from the release of effluents by sugar mills located on Queensland coastal rivers. At the present time the fishery is virtually unmanaged, with no regulations such as bag-limits or closed seasons being applied, other than a size limit of 50 cm. Research is clearly needed in this area.

The decreasing abundance of barramundi in Queensland rivers, and the lack of suitable sport fish in the impoundments created by the damming of these rivers, led the Queensland government to consider the introduction from Africa of Nile perch to stock these waters in the late 1960s. In the event, the proposal was not proceeded with. Chapter 21 considers this matter in more detail. More recently the CSIRO Division of Fisheries and Oceanography, in conjunction with Northern Territory and Queensland fisheries agencies, has initiated a commercial fishery-orientated biological study of barramundi in Queensland and Northern Territory waters. The Northern Territory work is being carried out in the Alligator Rivers area where this species is particularly abundant and could be affected by future uranium mining and milling operations (see Pollard, 1974).

**B. Temperate south-eastern coastal drainage system.** Some 40 or so native freshwater species occur in this drainage system, but of these only the catadromous short-finned eel is of any commercial significance. The long-finned eel has some potential commercial value. Moreover, only a single species, the Australian bass, is of any significant angling interest, though several others (e.g. estuary perch) are of marginal interest to anglers. Several species, and particularly the Australian grayling, are in need of protective environmental management (see Table 16).

**Short and long-finned eels (Anguilla australis and A. reinhardtii: Family Anguillidae)**

These two species of eels, which occur in the coastal rivers and streams of south-eastern Australia, are replaced by *A. obscura* on the north-eastern and *A. bicolor* on the northern and north-western coasts of the continent. *Anguilla australis* occurs in the temperate south-eastern (from eastern South Australia to northern New South Wales) and Tasmanian coastal drainages, where it is fished commercially, especially in Victoria and Tasmania. It also occurs in New Zealand. *Anguilla reinhardtii* occurs in the temperate south-eastern and part of the tropical northern coastal drainages (from eastern Victoria to northern...
Aquatic fauna

Queensland), where it is fished to a small extent in Victoria and New South Wales. It also occurs in New Caledonia.

The life cycles of all species of anguillid eels are similar in that they are strictly catadromous, the adults living in coastal fresh waters and migrating to sea as ‘silver eels’ to spawn, sometimes thousands of miles away from their home streams. (In the case of the two Australian species in question, the adults probably spawn in the Coral Sea at 8 to 20 years of age (Ord, 1978)). After spawning the adults presumably die and the pelagic, flattened, leaf-like ‘leptocephalus’ larvae drift on the ocean currents until they reach the coast, when their bodies take on a cylindrical shape and they enter the freshwater rivers and streams as elvers. Anguilla australis grows to over a metre in length and about 3 kg in weight and A. reinhardtii to over 1.5 metres and almost 20 kg; their growth rates are apparently faster in still or slow-flowing rather than fast-flowing waters.

Although they are rarely sought by anglers, the flesh of freshwater eels is generally of good eating quality. Most of the commercial catch, which is taken mainly by means of fyke nets, is frozen for export, though a small amount (about 5 per cent) is smoked for local consumption. Victoria is the largest producer, accounting for approximately 87 per cent of the total catch of short-finned eels (mainly from western Victoria) while Tasmania accounts for the remaining 13 per cent (Table 15). Very small quantities (generally averaging less than one tonne per annum) of long-finned eels are taken from some of the northern coastal rivers of New South Wales, and about 5 per cent of the Victorian catch is composed of this species.

Although the published national catch figures for short-finned eels showed a marked fall in the 5-year period to 1975 (Table 15), this is presumably mainly due to the absence of Victorian catch figures in the official statistics. Ord (1978) stated that there were 22 full-time eel fishermen in Victoria in 1976-77 and that they caught approximately 200,000 kg of eels valued at approximately $1 per kg to the fisherman.

As in the case of bass and grayling, the construction of dams and weirs on the coastal streams has prevented or limited the upstream movements of elvers in some areas (e.g. Bishop and Bell, 1978a), and this must have a significant effect on eel stocks in the upstream reaches. In Victoria young elvers and glass eels are also captured from estuarine waters for both rearing to market size in concrete ponds and transfer to natural waters for later harvesting. Scott, Glover and Southcott (1974), in referring to the rare occurrence of short-finned eels in South Australia, pointed out that ‘This species was once thought to be common in our freshwater rivers and streams, but nowadays specimens are rarely seen’, and concluded that ‘The blocking of the outlets of our main rivers with weirs and barrages has no doubt interfered seriously with the life cycle of this species’. D. Buckmaster (personal communication), however, believes that elvers could certainly negotiate the barrages on the lower Murray and that the scarcity of eels in this area is probably because it is near the limit of their westwards distribution. The capture, culture, marketing and processing of eels on a world-wide basis is discussed by Forrest (1976), their culture by Usui (1974), and biological knowledge of New Zealand freshwater eels is summarised by Skrzynski (1974).
Australian bass and estuary perch (*Percolates novemaculeatus* and *P. colonorum*: Family Percichthyidae)
The Australian bass is an important angling species and closely resembles its near relative the estuary perch, which is of lesser angling interest. The two species are sympatric in the coastal streams of south-eastern Australia, where the bass generally inhabits the freshwater reaches and the estuary perch the brackish upper estuarine reaches of these rivers (Williams, 1970). In winter, however, the bass, which is catadromous, migrates downstream to spawn in saline estuarine waters.

The distribution of the bass extends from small streams flowing into Tin Can Bay (approximately 240 km north of Brisbane) in Queensland (it is common in the Noosa River) to East Gippsland in Victoria. The estuary perch extends from the Richmond River in northern New South Wales to northern Tasmania and occasionally as far west as the eastern part of South Australia (rarely to the Murray River mouth). Reputedly, bass have been introduced into coastal streams in south-western Australia, but there are no recent records of them from this area. Both species may reach a weight of around 4 kg, most of those fish caught which are larger than about 25 cm or 1 kg usually being females and most of the smaller fish males (Williams, 1964).

Bass are the most valuable angling fish of south-eastern coastal rivers and streams. They are aggressive feeders and readily attack a lure or live bait; this and their vigorous fighting when hooked make them one of the most important freshwater game species (Pollard, 1969). In fresh waters, bass feed mainly on insects falling on the water surface, but a variety of organisms living near to or on the bottom (such as aquatic insect larvae, tadpoles, shrimps and small fish) are also eaten; in estuarine waters, they feed on prawns, small crabs, fish, amphipods and worms. The diet of estuary perch consists mainly of small fish and prawns living on or near the bottom.

Bass were apparently abundant in most south-eastern coastal streams during the last century, and well into the present century both bass and estuary perch were freely available commercially at fish markets during winter. However, this commercial harvest from the estuaries probably removed large numbers of prospective spawners each year, which may have been one of the factors leading to a population decline. Present regulations in New South Wales do not allow commercial capture of or the use of nets to fish for bass or estuary perch, and only the use of a rod and/or line by amateur anglers is now permitted. A bag limit of 10 fish per person per day also applies, though there is no closed season or minimum legal length. The minimum size limit was removed following the discovery of differential growth rates between sexes (Williams, 1964) which led to male fish being virtually unexploited.

Bass and estuary perch spawn at approximately the same time, namely in winter, when estuarine water temperatures are between 14.5 and 16°C. Bass must migrate from fresh into saline water to spawn because their eggs and sperm do not survive in fresh water. Although some eggs might hatch and some larvae develop at relatively low salinities (between 8 and 10°/oo), present evidence suggests that for successful propagation salinities of at least 14°/oo are needed. Estuary perch are believed
Aquatic fauna

to spawn in a different part of the estuary from that used by bass, and probably closer to the sea. If so, this may explain why these species rarely appear to hybridise in nature, though hybrids can be readily obtained by artificial fertilisation in the laboratory (Chamberlain and Ehl, unpublished data).

The start of the spawning season, signalled by the beginning of the downstream migration of adult bass, seems to depend on the weather. Generally, bass start migrating from about mid-June, although cold and dry weather may delay this downstream movement. The return of adult fish to fresh water extends over several months, from mid-August to mid-October. Once in saline water, male bass often congregate in large shoals numbering several hundreds; females have never been caught in such large aggregations. Hatching takes up to several days, depending on water temperature (e.g. 3 to 4 days at 15°C). After some time in the estuaries the juveniles migrate upstream. Although it is not known when this upstream migration begins, young bass may be found in freshwater from the end of November, when they are generally between 2-5 and 5 cm in length.

The migratory nature of the life cycle of the bass means that the species is particularly vulnerable to the impact of man's activities on its environment. In this respect, the construction of high level dams will terminate the movements of bass and other fish in rivers unless suitable fish ladders are constructed. However, where such fish ladders are provided, there may be a need to conserve water during the dry season and this is usually done by stopping the flow through the ladders, possibly at the very time when the young bass are on their way upstream. As in the case of barramundi, this interference with their migrations is probably the most serious single adverse effect man has had on the abundance of bass in coastal rivers. Other deleterious impacts include land clearing and other changes in the nature of catchments (often causing erosion and silting-up of riverine pools), flood mitigation works, removal of snags and shoreline vegetation from rivers, and the general effects of water pollution.

Current research in New South Wales is chiefly oriented towards the artificial propagation of bass with the eventual aim of stocking coastal impoundments and farm dams (in which bass grow well but do not reproduce); in addition, further research on the efficacy of fish ladders for bass and other migratory species is planned.

**Australian grayling (Prototroctes maraena: Family Prototroctidae)**

The Australian grayling is the largest of the native Australian salmoniform fishes occurring in the coastal drainages of south-eastern Australia (its range extending from the Macquarie Rivulet about 100 km south of Sydney to western Victoria and including Tasmania). The Prototroctidae is closely related to the Retropinnidae (southern hemisphere 'smelts') and the Aplochitonidae (represented in Australia by the Tasmanian whitebait). Although once common (Tunbridge, 1972) and often taken by anglers (Pollard, 1969), it is now considered the Australian freshwater fish most seriously threatened with extinction (Lake, 1971). Already, the closely related New Zealand grayling (*Prototroctes oxyrhyncus*) is probably extinct; it was last recorded in 1926, and there is no evidence that it is still extant (McDo-
The Australian grayling began to decline in abundance soon after European settlement. The reasons for its decline are unknown, though several have been proposed; they include habitat disruption caused by deforestation, introduction of brown trout and/or carp, a widespread epidemic (perhaps causing post-spawning mortalities), and the effects of dam construction (Lake, 1971).

Features of the biology of the grayling have been noted by Lake (1971), Tunbridge (1972), McDowall (1974, 1976) and Jackson (1976). McDowall (1976) reviewed the status and distribution of both *P. maraena* and *P. oxyrhyncus*. A sample of over 300 specimens from below Tallowa Dam on the Shoalhaven River, New South Wales, taken in November 1976 (Bishop and Bell, 1978a, b), provided the first large sample of *P. maraena* for study since the need for conservation measures has become more generally realised.

Grayling may reach a length of 33 cm (Whitley, 1957) and a weight of 350 g; however, they are more commonly taken at lengths of between 17 and 19 cm. The species rises to artificial flies and may also be caught using baits. McDowall (1974) speculated from an examination of the grayling's specialised dentition that *P. maraena* was either an omnivore or a partial herbivore. However, if this were so, the grayling would be unique amongst salmoniform fish, for these are generally predatory. Jackson (1976), after examining the stomach contents of 22 specimens of *P. maraena* from the Mitchell River in eastern Victoria, concluded that, like other salmoniforms, it is mainly insectivorous. Stomach contents analysis of specimens taken below Tallowa Dam by Bishop and Bell (1978b) indicated that cladocerans were the major food item, though algae comprised approximately a quarter of the contents by volume. Grayling collected at other times and sites within the Shoalhaven Valley were found to be feeding only upon aquatic insects. These observations confirm McDowall's speculation that *P. maraena* is an omnivore.

The spawning season appears to extend from January to March. Tunbridge (1972) concluded that grayling in the Mitchell River spawn in fresh water, but in Tasmania ripe grayling have been found in brackish water. The hypothesis that grayling are diadromous (Johnston, 1891; Lake, 1971; Frankenberg, 1974) appears viable since, although they have only been recorded from coastal rivers and streams on the mainland, they are often found in estuaries in Tasmania. The occurrence of large numbers of grayling (Bishop and Bell, 1978a, b) immediately below Tallowa Dam indicated upstream migratory behaviour. Nevertheless, as long ago as 1870, Allport artificially fertilised grayling ova and showed that they could develop in fresh water. The matter remains unresolved.

The future survival of *P. maraena* depends on how soon effective conservation measures can be implemented, and the basis of a sound conservation policy for this species depends on reliable biological information. However, basic biological data are scarce, and a research program should be initiated to elucidate the species' distribution and abundance, the nature and location of spawning sites and juvenile habitats, and the nature and extent of migratory activity. The impact of such factors as dams and weirs on coastal streams could then be determined, and, it is hoped,
appropriate measures implemented to ensure the survival of this endangered species.

**C. Tasmanian drainage system.** Some 20 indigenous freshwater fish occur in Tasmania. Apart from the short-finned eel (already discussed), only one, the Tasmanian whitebait, has previously been of commercial significance. Freshwater angling is based almost entirely on introduced salmonids, but the management of these and the conservation problems of the smaller forms (such as the galaxiids) are best discussed later (see *mountain streams and lakes*).

**The Tasmanian whitebait (**Lo**vettia seali**: Family Aplochitonidae)**
The Tasmanian or Derwent whitebait is a small salmoniform fish in the family Aplochitonidae, which is confined to the southern hemisphere and fairly closely related to the Galaxiidae (galaxias or native ‘trouts’), Prototroctidae (southern ‘graylings’), and Retropinnidae (southern ‘smelts’). It is found only in Tasmania, where the one year old adult fish (averaging only 4.5 to 5 cm in length) migrate in spring from the sea to spawn in freshwater reaches of coastal rivers and streams. Its life cycle is thus anadromous, like that of many northern hemisphere salmonids. Having spawned, the adults die, and the newly-hatched larvae are washed downstream to the sea.

The commercial whitebait fishery, which began in the early 1940s, was based on migrating adults and was carried out mainly in the freshwater spawning areas of northern (and some southern) Tasmanian rivers using scoop nets (simple, movable fly-wire traps). The catch was canned and marketed as ‘whitebait’ in competition with New Zealand whitebait (mainly post-larval *Galaxias maculatus*). However, by 1948 excessive removal of spawning fish caused a serious decline in catches (Blackburn, 1950), and by 1956 the total catch had declined to less than 2,000 kg (from a maximum catch in 1947 of almost 500,000 kg) (Lynch, 1965). By the early 1950s the canning of whitebait in Tasmania ceased to be a commercial proposition and the canneries closed. The proportion of *Lovettia seali* in the whitebait catch had dropped from over 95 per cent between 1944 and 1948 (the rest being mainly post-larval galaxiids, with a few retropinnids and gobies; Blackburn, 1950) to less than 25 per cent *Lovettia* by 1964 (the other 75 per cent being post-larval galaxiids with a few retropinnids). By 1975 the few remaining small whitebait ‘runs’ were closed to commercial fishing and will remain so until the condition of the stocks has been assessed and found to be improved.

**Tasmanian giant crayfish (**Astacopsis gouldii**: Family Parastacidae)**
*Astacopsis gouldii* is the largest known freshwater crustacean, with some individuals reaching over 6 kg in weight. It is found in the north-east and north-west of Tasmania where it supports a small recreational fishery for local communities. Concern for stocks has led to an export ban and the imposition of a minimum legal length for capture. Little is known of its biology. Its large size has provoked much interest overseas, but a small ratio of tail to body weight (hence a relatively small quantity of flesh) and a slow growth rate indicate that the species is not likely to be suitable for intensive culture.
D. Temperate south-western coastal drainage system. Only 10 or so native freshwater fish are known from the south-western corner of the continent, and almost all are small and of no commercial or recreational interest. The only possible exception is the freshwater catfish or 'cobbler', *Tandanus bostocki*, which grows to only half the size of its close relative in the Murray-Darling and south-eastern coastal drainages. What little freshwater angling takes place in this area is based mainly on introduced salmonids. Mention should also be made, however, of the native freshwater crayfish, the marron. This species is of considerable interest to recreational fishermen (see Chapter 21).

General conclusions concerning management of coastal stream fish and fisheries. The preceding accounts indicate that by far the most important threat to fish in coastal rivers and streams involve habitat disruption. This is true for all the species discussed in detail with the possible exception of the Tasmanian whitebait; the latter appears to provide a clear example of the result of overfishing a limited stock of spawning fish with a one-year life cycle, although Lake (1971) considered that environmental changes had also played some part in its decline. The main threat to the other species discussed appears to have been the interference by dams and weirs with the upstream movements of juveniles, resulting in the eventual exclusion of adult populations from waters above these obstacles (e.g. the silver barmahundi, Australian bass, Australian grayling, and to a lesser extent the short-finned eel). A good illustration of the effects on upstream movements of fish of such a dam on a south-eastern coastal river system is given by Bishop and Bell (1978a).

With regard to possible management strategies in such cases, the only effective approach—other than not building dams and weirs on coastal rivers—is the construction of suitable fish ladders. The construction of these, however, first requires adequate studies of the biology and life cycle movements of those fish involved, and then studies on their swimming speeds and abilities in relation to different fish ladder designs. Unfortunately, no fish ladders have been constructed on any major Australian coastal river dam, and no adequate research has yet been carried out on the design of fish ladders suitable for Australian species. New South Wales State Fisheries has plans to carry out this essential research.

Inland rivers and lakes

E. Murray-Darling drainage system. The Murray-Darling, by far Australia's largest river system, drains over one million square kilometres of land. Since its waters originate in the highest mountain range in Australia, where precipitation is high, its main rivers are relatively permanent, though they flow for most of their courses through arid plains. Nevertheless, extensive variations in discharge occur from year to year. There are four main rivers in the system, of which the Darling is probably the least affected by man; it is thus the stronghold for many native species.

The south-eastern area is the least arid, particularly that between the Murrambidgee and Murray Rivers, which consists of a ramifying riverine network with numerous anabranches. The main rivers of the Wimmera area in western
<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Abundance&lt;sup&gt;1&lt;/sup&gt;</th>
<th>Economic Commercial value&lt;sup&gt;2&lt;/sup&gt;</th>
<th>Angling value&lt;sup&gt;3&lt;/sup&gt;</th>
<th>Distribution&lt;sup&gt;4&lt;/sup&gt;</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Mordacia mordax</em></td>
<td>Short-headed lamprey</td>
<td>2?</td>
<td>E</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Anguilla sp.</em></td>
<td>Freshwater eel&lt;sup&gt;5&lt;/sup&gt;</td>
<td>1</td>
<td>B</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Nematolosa erebi</em></td>
<td>Bony bream</td>
<td>4</td>
<td>A</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Salmo trutta</em></td>
<td>Brown trout</td>
<td>4</td>
<td>C!</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>S. gairdneri</em></td>
<td>Rainbow trout</td>
<td>4</td>
<td>C!</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Salvelinus fontinalis</em></td>
<td>American brook trout</td>
<td>3</td>
<td>C!</td>
<td>+</td>
<td>?</td>
</tr>
<tr>
<td><em>Oncorhynchus tschawytzschla</em></td>
<td>Quinlan salmon&lt;sup&gt;6&lt;/sup&gt;</td>
<td>2</td>
<td>C!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Retropinna semoni</em></td>
<td>Australian smelt</td>
<td>4</td>
<td>E!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Galaxias maculatus</em></td>
<td>Common galaxias</td>
<td>4</td>
<td>E!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>G. planiceps</em></td>
<td>Flat-headed minnow</td>
<td>2</td>
<td>E!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>G. olidus</em></td>
<td>Inland mountain trout</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Cyprinus carpio</em></td>
<td>European carp</td>
<td>4</td>
<td>C!</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Carassius auratus</em></td>
<td>Goldfish</td>
<td>4</td>
<td>D!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>C. carassius</em></td>
<td>Crucian carp</td>
<td>1</td>
<td>D!</td>
<td>—</td>
<td>(?</td>
</tr>
<tr>
<td><em>Tinca tinca</em></td>
<td>Tench</td>
<td>3</td>
<td>C!</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Tandanus tannanus</em></td>
<td>Freshwater catfish</td>
<td>4</td>
<td>A!</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><em>Neolisurus argenteus</em></td>
<td>Silver tannan</td>
<td>1</td>
<td>B</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Gambusia affinis</em></td>
<td>Mosquito fish</td>
<td>4</td>
<td>F!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>Nematocentris fluviatilis</em></td>
<td>Rainbow fish</td>
<td>2</td>
<td>E!</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td><em>N. maculatus</em></td>
<td>Checkered rainbow fish</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>+</td>
</tr>
</tbody>
</table>

1 Abundance code: 1, rare; 2, occasional; 3, common; 4, abundant. N.B. Applies only to individual distribution areas listed above.

2 A. native species of economic importance; B, large native species but of limited or no economic importance; C, introduced species of economic importance; D, large introduced species but of limited or no economic importance; E, small native species of no economic importance; F, small introduced species of no economic importance; !, species in which breeding biology is known.

3 +, of significant commercial or angling value; —, of no economic value.

4 +, present; (+), doubtful occurrence; —, not recorded; ?, position uncertain.

5 Eels occasionally occur as a result of their traversing the Great Dividing Range.

6 Non-breeding populations sustained by stocking.
<table>
<thead>
<tr>
<th>Species</th>
<th>Habitat Description</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Craterocephalus fluviatilis</strong></td>
<td>Mitchellian freshwater hardyhead</td>
<td>3</td>
</tr>
<tr>
<td>C. eyresi</td>
<td>Lake Eyre hardyhead</td>
<td>2</td>
</tr>
<tr>
<td><strong>Ambassis castelnaui</strong></td>
<td>Western chanda perch</td>
<td>2</td>
</tr>
<tr>
<td>Macquaria australasica</td>
<td>Macquarie perch</td>
<td>2</td>
</tr>
<tr>
<td>Percalates colonorum</td>
<td>Estuary perch</td>
<td>1</td>
</tr>
<tr>
<td>Plectroplites ambiguus</td>
<td>Golden perch</td>
<td>4</td>
</tr>
<tr>
<td>Maccullochella peeli</td>
<td>Murray cod</td>
<td>4</td>
</tr>
<tr>
<td>M. macquariensis</td>
<td>Trout cod</td>
<td>1</td>
</tr>
<tr>
<td>Leiopotherapon bidyanus</td>
<td>Silver perch</td>
<td>4</td>
</tr>
<tr>
<td>L. unicolor</td>
<td>Spangled perch</td>
<td>4</td>
</tr>
<tr>
<td>Hephaestus welchi</td>
<td>Welchs grunter</td>
<td>1</td>
</tr>
<tr>
<td>Nannoperca australis</td>
<td>Southern pigmy perch</td>
<td>2</td>
</tr>
<tr>
<td>Perca fluviatilis</td>
<td>English perch</td>
<td>4</td>
</tr>
<tr>
<td>Gadopsis marmoratus</td>
<td>River blackfish</td>
<td>2</td>
</tr>
<tr>
<td>Mugil cephalus</td>
<td>Bully mullet</td>
<td>2</td>
</tr>
<tr>
<td>Aldrichetta forsteri</td>
<td>Yellow-eye mullet</td>
<td>3</td>
</tr>
<tr>
<td>Pseudaphritis urvilli</td>
<td>Congolli</td>
<td>2</td>
</tr>
<tr>
<td>Hypseleotris klunzingeri</td>
<td>Western carp gudgeon</td>
<td>4</td>
</tr>
<tr>
<td>H. gali</td>
<td>Fireetail gudgeon</td>
<td>2</td>
</tr>
<tr>
<td>Hypseleotris n.sp.1</td>
<td>—</td>
<td>2</td>
</tr>
<tr>
<td>Hypseleotris n.sp.2</td>
<td>—</td>
<td>2</td>
</tr>
<tr>
<td>Mogurnda striata</td>
<td>Purple spotted gudgeon</td>
<td>2</td>
</tr>
<tr>
<td>Philypnodon grandiceps</td>
<td>Flat-headed gudgeon</td>
<td>3</td>
</tr>
<tr>
<td>Philypnodon n.sp</td>
<td>—</td>
<td>2</td>
</tr>
</tbody>
</table>
Victoria, on the other hand, drain internally and terminate in ephemeral salt lakes; their waters seldom reach the Murray. Likewise, the Bulloo drainage system only rarely discharges into the Darling River.

Ephemeral swamps and broad floodplain areas are typical of the Murray-Darling drainage system, and are important habitats for fish survival and breeding. Additionally, there are now 84 man-made water storages on the system. Although these have substantially increased available fish habitats, subtle accompanying environmental changes have proved detrimental to some species, and especially to those with special adaptations to the original environment (see Chapters 17 and 18). Many changes of this sort favour introduced species. Few permanent lakes occur in this drainage system.

According to Lake (1966), 27 species occur in the Murray-Darling system within New South Wales. Nineteen are native, the rest exotic. Including those species which spend most of their life at sea or in estuaries, but part of it in fresh

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**Fig. 29** Relationships of annual catches of three important commercial fishes in the New South Wales section of the River Murray with river height (at Euston).
waters (e.g. *Mugil cephalus, Percalates colonorum, Aldrichetta forsteri*), the total complement for the entire system is about 44 species (Table 17). This total includes a number of species of *Philypnodon* and *Hypseleotris* not yet described. Eight of the 44 have been introduced. At present, 9 species, 6 of which are native, are exploited commercially. They include the bony bream (until recently the largest tonnage caught), golden perch (economically the most important), Murray cod (the prime table species), European carp (the largest tonnage caught at present), silver perch,
freshwater catfish, English perch and tench. Yellow eye mullet, although it may occur in fresh waters, is principally exploited in brackish estuarine areas. Information on the commercial fisheries for and distributions of a number of the more important species in the Murray-Darling system, from a survey carried out by J.O. Langtry in 1949-50, is presented by Cadwallader (1977). Figs. 29 and 30 indicate total annual fish catches and annual catches of golden perch, Murray cod and silver perch in the New South Wales sections of the Murray and Murraybidgee Rivers. Also shown are river heights at Euston (Murray) and Balranald (Murraybidgee).

Fourteen species are of some angling interest, of which 8 are native (Table 17). Three of the eight, namely the trout cod, Macquarie perch and river blackfish, are relatively uncommon and are therefore of little real angling value. Of the six introduced species only three, the English perch, European carp and quinnat salmon, occur in the lowland waters of the Murray-Darling drainage. The European carp is regularly angled but seldom eaten, and the quinnat salmon, which was stocked in a few selected lakes near the Murray River by the Victorian Fisheries Department, is now rare, if it still exists in this drainage system. Another three salmonids (see detailed discussion later) support a substantial recreational fishery in the highlands, but are only rarely caught elsewhere. Although Welch’s grunter, estuary perch, bully mullet and silver tandan are large enough to be angled, they are too uncommon to be considered as angling species in the drainage system. The freshwater eel, an infrequent invader from across the Great Dividing Range, is also in this category. The goldfish, whilst occasionally reaching 2 kg in weight, is rarely angled, and there is some doubt as to whether its close relative the crucian carp still exists in the system.

Finally, there are two species of freshwater crayfish of limited commercial value, 19 small native and one introduced forage species of no direct economic importance, and the short-headed lamprey which cannot be classed as a forage species and which has no commercial value.

Native species
Golden perch (*Plectroplites ambiguus*; Family Percichthyidae)
This species lives entirely in fresh waters and is the most important wholly freshwater native fish commercially and recreationally exploited in Australia. Also known as the callop, yellowbelly and Murray perch, its closest relatives are the Macquarie perch and Australian bass, although its breeding requirements and egg and larval development are closest to those of the silver perch (Family Theraponidae).

Golden perch occur throughout most of the inland areas of eastern Australia, but not in the arid Central Western Desert area and at high altitudes. They are also reported to occur in a few easterly flowing streams such as the Fitzroy and Dawson Rivers in Queensland and the Clarence and Richmond Rivers in northern New South Wales, and they have been introduced to a number of reservoirs and numerous farm dams in several of the eastern states. There was reputedly also some early stocking of this species in Western Australia. Individuals may grow to 23 kg in weight and 76 cm in length, but a fish of 5 kg is generally considered a good size. Males
have been known to mature at 19 cm when 2 to 3 years old, but females are closer to 40 cm (1-4 kg) and 4 years old when mature.

The golden perch primarily lives in warm, turbid and sluggish inland rivers, where it is well-adapted to fluctuating water temperatures and river heights; these conditions, in any event, appear essential for successful breeding. This species moves considerable distances upstream in floods, and during such floods it normally spawns in spring or early summer at temperatures greater than 23.5°C (Lake, 1967a). Females can retain ova at an advanced stage of development until conditions are suitable for spawning; if these do not occur, resorption takes place. A 2.5 kg fish may produce 500,000 semi-buoyant ova (Lake, 1967b). The early larvae are buoyant and start to swim and feed on zooplankton when five days old; they develop juvenile features by 25 days, when they are around 12 mm long. Adult food is essentially suitably sized crustaceans, but insect larvae are occasionally also taken. Fish can be readily handled and can survive temperatures between 4 and 37°C.

The golden perch is a substantial part of the amateur and commercial catch from the Murray-Darling drainage system. Available statistics indicate that 54 per cent of the commercial catch is taken from South Australia, 43 per cent from New South Wales and 3 per cent from Victoria (Table 15). Professional fishermen use nets and occasionally crosslines. Golden perch are readily angled but are spasmodic 'biters'. Techniques have been developed by New South Wales State Fisheries to breed them in hatcheries to supply the considerable demand for fry to stock farm dams for, although they certainly survive and grow well in dams, they are unlikely to breed there without some means of water level control (see Chapter 21). The commercial catch in the New South Wales part of the Murray River has varied considerably over the past 20 years from 192.3 tonnes in 1956-57 to 6.7 tonnes in 1971-72 (Fig. 29). This decline in the commercial catch may be interpreted as giving at least some indication of population trends over this period, although accurate information on the variation in angler effort over the same period is unavailable. However, 74,000 amateur and 127 commercial fishing licences were sold in New South Wales in 1965-66 and by 1972-73 these numbers had grown to 106,800 and 291 respectively. Both angling and commercial fishing effort thus probably increased significantly between 1956 and 1972.

Amateur fishermen are allowed to fish throughout the year in New South Wales but professional fishing is limited to December to August inclusive. No bag or size limits are enforced, but commercial fishermen may not use gill or drum nets with mesh sizes below 12.7 cm (5 in).

In general, larger than average catches occur about three years after reasonable river flooding (Llewellyn, 1978; Reynolds, 1976b). Thus, the drought years between 1961 and 1969 (except for 1964) caused a significant decline in catches (and populations), and the floods of 1970, 1973, 1974 and 1975 a significant increase. Overall, however, populations of the species appear to have been declining gradually. Continued environmental change, particularly alterations to water regimes following dam construction, is probably the cause. The decline
Aquatic fauna

is particularly obvious at the extremity of the species' distributional range; it is slowly disappearing from waters which it formerly occupied at higher altitudes.

**Murray cod and trout cod** (*Maccullochella peeli* and *M. macquariensis*: Family Percichthyidae)
The totally freshwater Murray cod is one of the most prized native sport and commercially exploited species in Australia. Its closest relative, the rare trout cod, has only recently been positively established as a separate species (Berra and Weatherley, 1972). A Murray cod weighing 113.5 kg (approximately 1.8 m long) has been recorded and specimens of around 36 kg are not uncommonly caught by anglers. They mature when about 56 cm in length and between 1.8 and 2.7 kg in weight.

The Murray cod is widespread throughout the Murray-Darling river system, though rarer in its headwaters. It also occurs in the Mary and Dawson Rivers of southern Queensland, the Richmond and Clarence Rivers of northern New South Wales, and Lake Charlegrark in Victoria. It is absent from the Central Western Desert and the Lake Eyre and Bulloo drainage systems, but has been introduced into Western Australia (apparently unsuccessfully) and also into many lakes and farm dams in New South Wales. Typically, it inhabits sluggish, turbid rivers in which hollow logs and stumps afford hiding and spawning sites. According to Lake (1967a), spawning occurs in early spring when a rise in river level is associated with rising water temperatures between about 16.5 and 20.5°C. Female fish may produce 400,000 or more adhesive demersal eggs which are probably deposited in hollow logs. Newly hatched larvae (6-9 mm long) possess a very large yolk sac, and attain a juvenile body form by about 27 days. Larvae feed on zooplankton, and adults on fish, crustaceans and bivalve molluscs of appropriate size.

In the Murray-Darling drainage system, the only region where the species is exploited, available statistics indicate that 59 per cent of the commercial catch is caught in South Australia, 39 per cent in New South Wales, and 2 per cent in Victoria (Table 15). Professional fishermen mainly use drum nets, gill nets and occasionally crosslines, and the species is readily angled. Fishing regulations in New South Wales are identical to those for golden perch. It is also in considerable demand for stocking purposes, particularly for farm ponds, and techniques are now being developed by New South Wales State Fisheries for its large-scale artificial propagation (see Chapter 21).

Murray cod populations appear to have shown a more marked decline (judging from commercial catch figures from the Murray and Murrumbidgee Rivers in New South Wales—see Figs. 29 and 30) than have those of golden perch. In the Murray, catches declined steadily from 84.4 tonnes in 1955-56 to 4.6 tonnes in 1967-68, which was towards the end of the 1961-69 drought, and have remained relatively low since that time. It thus appears that this species is not able to recover after such a population 'crash' with the same ability as the golden perch. Introduced fish such as European carp and English perch are often blamed for the decline in this species, but decreasing population size was evident well before the spread of at least the European carp (Reynolds, 1976b). Nevertheless, factors other than drought appear to be implicated; they probably
include destruction of spawning sites, silting up of deep holes, desnagging and channelisation. An unfavourable environment may greatly influence fry survival, and if this is so subsequent over-exploitation could certainly contribute to this species' decline.

The trout cod, which is also a strictly freshwater species, is now rare, having declined steadily over the past twenty years. In recent years it has been caught only in the upper reaches of the Murrumbidgee River near Queanbeyan, N.S.W., and in Seven Creeks near Euroa and Lake Sambell near Beechworth in Victoria (in the latter it was obviously stocked). Earlier records are from the Goulburn River, Murray River, elsewhere on the Murrumbidgee River, and the Macquarie River. Specimens larger than 16 kg have not been reported, and fish taken from Lake Sambell weighed around 11 kg and averaged approximately 70 cm in length.

Little is known about this species except that it was generally restricted to the cooler waters of the Murray system. Specimens die readily when handled, particularly at temperatures exceeding 20°C (Lake, 1966). Trout cod ova are reported to be larger than those of Murray cod. The trout cod is too rare to be of any economic value, and the main management concern in relation to this species is with its conservation. However, when present it is readily angled, and when it was more plentiful it was regularly caught by commercial fishermen.

**Silver perch** (*Leiopotherapon bidyanus*: Family Theraponidae)

This native species (previously known as *Bidyanus bidyanus*) is probably the fourth most important commercially exploited species in the inland region. Its edible qualities, however, are not as good as those of golden perch or Murray cod. It lives entirely in fresh waters and its breeding cycle is very similar to that of golden perch. In some areas it is known as bream, black bream, grunter, bidyan and tcheri. Relationships are with the spangled perch and northern grunters. The largest specimen recorded weighed 7.7 kg and was 61 cm long. However, fish of around 3 kg are not uncommon, and in ponds specimens may reach a length of 28 cm in one year.

The silver perch is common in the Murray-Darling river system in Queensland, New South Wales, Victoria, and South Australia. Although absent from higher altitudes, it extends further upstream into colder waters than does the golden perch. Silver perch also occurs in a few coastal streams in New South Wales where it may have been introduced (Lake, 1971). Recently, it has been introduced into Glenbawn Dam on the Hunter River in New South Wales. Streams and waterholes of inland areas are frequented by this species, and in these it often schools and is occasionally seen near the surface. It can survive at least for short periods at temperatures as high as 37°C and as low as 2°C. Breeding normally occurs in October and November when the upper metre of water reaches a temperature of 23°C and this is accompanied by at least a 30 cm rise in water level (Lake, 1967a). Spawning may be delayed by unfavourable conditions. A 2 kg fish may yield c. 500,000 ova which, although semi-pelagic, sink in still water. Newly hatched larvae (about 3-6 mm long) live on the bottom and feeding begins 6 days after fertilisation. The larvae develop the characteristics of juvenile fish at 18 days when
about 11 mm long. Larvae feed on phytoplankton and zooplankton, and adults on zooplankton, worms and shrimps (Paratya and Macrobrachium spp.); filamentous algae and aquatic plant remains are also frequently found in the stomachs of adult fish.

Available statistics indicate that 68 per cent of the commercial catch is from New South Wales, 28 per cent from South Australia and 4 per cent from Victoria (Table 15). Anglers, however, appear not to be attracted to the species although it can often be caught in large numbers, provides fine sport, and has good edible qualities. It has considerable potential for fish farming and handles well at cooler temperatures. Techniques for its artificial propagation have recently been developed by New South Wales State Fisheries.

Commercial catches in the Murray River have slowly declined over the last 20 years from 23-2 tonnes in 1958-59 to 2-3 tonnes in 1975-76. Fluctuations in catch are not influenced by floods to the same extent as in the case of golden perch, and the decline in catches of this species has not been as marked as that for golden perch (Figs. 29 and 30). Lake (1971) suggested that the reduced extent and duration of flooding in backwaters has adversely affected juvenile survival. He also stated that the eggs are not quite as buoyant as those of golden perch and will sink in very quiet water; as a result ova and larvae will not survive in waters with anoxic lower layers, a feature which is more prevalent towards the northern limits of this species’ distribution. This may explain why the silver perch does not have the same extensive northward distribution as the golden perch (Lake, 1971).

**Freshwater catfish (Tandanus tandanus: Family Plotosidae)**

This freshwater species is exploited commercially, particularly in swampy areas where exploitation of the three species just discussed is not practicable. It is also called tandan, freshwater jew-fish, eeltail and kenaru, and its closest relatives are the neosilurid catfishes which occur to the north of its range. Specimens up to 90 cm long and weighing 6-8 kg have been recorded; however, 2 kg is considered a good weight. Despite popular prejudice, catfish flesh is of excellent eating quality.

The species is widespread throughout the Murray-Darling river system, although not upstream of Wagga Wagga and Mulwala in the Murrumbidgee and Murray Rivers (probably due to cool water discharges from large dams). It also occurs in the coastal rivers of eastern Australia from the Fitzroy River in Queensland to south of Sydney. As a general rule, however, it prefers lakes and sluggish rivers.

According to Davis (1977), most males and females mature after four years. Spawning takes place during spring or summer when water temperatures rise above 24°C (Lake, 1967a). Courtship involves nest building using pebbles or gravel up to 2 weeks before spawning. The large demersal eggs settle within the interstices of the gravel without actually adhering, and one parent remains in attendance until after hatching. Fecundity varies from 2,800 ova at 39 cm (0-8 kg) to 20,600 ova at 53 cm (2-3 kg) (Davis, 1977). Hatching takes about 7 days at temperatures between 19 and 25°C, at which time the larvae are 7 mm long; their barbels appear in three days and by three weeks they have most of the characteristics of adult fish. Juve-
niles feed on zooplankton, but most adults feed on dipterans (mainly chironomid larvae) and decapod crustaceans (mainly *Macrobrachium*). Forage fish (mainly *Hypseleotris*) are also important in the diet of small catfish. Additional items in the diet include molluscs, other crustaceans, and several other aquatic and terrestrial insect groups. Catfish are bottom feeders. Available statistics (Table 15) indicate that 46 per cent of the commercial catch is from New South Wales, 50 per cent from South Australia and 4 per cent from Victoria, with most fish being caught by gill nets in swamps. The species provides good sport and is frequently angled throughout its range. It breeds and grows well in ponds, and shows potential for artificial propagation and farming purposes. It does not survive well at water temperatures below 4°C.

The five-year means for commercial catches between 1961 and 1975 have risen from 15 to 38 tonnes (Table 15). This possibly indicates an increasing commercial interest in this species. In any event, there is no indication of a decline in populations at present. However, Lake (1971) pointed out that short-term fluctuations in water levels caused by irrigation requirements could adversely affect breeding by interfering with nest building.

**Bony bream** (*Nematalosa erebi*: Family Clupeidae)

This native freshwater fish (previously included in the genus *Fluvialosa*) is the most important species taken commercially for crayfish bait, although its exploitation is confined to the River Murray in South Australia. Previously, several species of bony bream were recognised throughout Australia, with considerable taxonomic confusion; most have recently been synonymised in a single species. Apart from bony bream, its common names include hairback herring, melon fish, pyberry and tukari. Specimens are usually no more than 20 cm long, but may occasionally reach 46 cm in length and weights of up to 2 kg (Lake, 1966). The species occurs in the Murray-Darling drainage system below 200 m altitude (rarely east of Narrandera in the Murrumbidgee River), in rivers and lakes in central Queensland, in the Northern Territory, in Western Australia, and in the Lake Eyre drainage system.

Little is known about its biology, except that spawning occurs in spring in fish that are at least one year old, and that fecundity is high. Aquatic plants, insects and sand have been found in stomachs. The bony bream is rarely angled but frequently entangles itself in nets. On the basis of South Australian commercial catches, the Murray River population remained relatively stable until 1974-75, but then sharply declined. Tench catches displayed a similar but more pronounced pattern. It has been suggested that the increase in the numbers of European carp may have been involved in the decline in both species, either by competition for food or by interference with eggs or larvae, although changing fishing patterns have also contributed to the decreased catches of bony bream (Reynolds, 1976b). However, marked reductions in range have also occurred, particularly below dams discharging cold water. Fluctuations in predatory waterbird populations may also be implicated in causing dramatic population fluctuations in this species.
Macquarie perch (*Macquaria australasica*: Family Percichthyidae)
This species has almost vanished from commercial catches over the past ten years and Lake (1971) regarded it as one of the four most seriously threatened Australian freshwater species. It is also known as silvereye, white eye, Murray perch, or mountain perch. Its closest relative is the golden perch although the breeding behaviour of this species is quite different. Specimens weighing up to 3-5 kg have been recorded, but most fish caught weigh not much above 1 kg.

The distribution of the species is mainly restricted to the upper reaches of the Murray, Murrumbidgee and Lachlan River systems, although not to the sources of these rivers. The Macquarie perch is also recorded from the Barmah Lakes area south of Deniliquin in New South Wales. The single record from South Australia is doubtful. It is also reasonably common in a number of man-made lakes and some coastal streams in New South Wales and Victoria (probably in all cases by introduction). Nowhere, however, is it now very abundant, though it was previously common in large man-made lakes with suitable inflowing spawning streams.

Males breed when two years old and 21 cm long, and females when three years old and 30 cm long. In Victoria spawning occurs in October and November when river temperatures reach about 16°C (Wharton, 1968). Fish then move into rivers within a few kilometres of their lake habitats and deposit adhesive, demersal eggs amongst stones and gravel in shallow riffle areas at depths of between 0.5 to 0.75 metres. Ripe Macquarie perch have been collected from the Mongarlowe River, New South Wales, from October to early January (Bishop and Tilzey, 1978). Fecundity has been calculated at 32,000 ova per kilogram of fish. Larvae feed on zooplankton, older fish mainly on adult and larval insects.

Despite its declining abundance, this species is of distinct value as a sportfish (Harrison, 1977; Bishop and Tilzey, 1978). Moreover, preliminary investigations indicate that it may be amenable to pond culture (Wharton, 1973). Although the available catch figures are not very reliable, the mean annual commercial catch for 1961-65 was given as 2.1 tonnes and that for 1973-75 as only 0.3 tonnes, with all fish being caught in New South Wales (Table 15). This catch decline may be at least partly due to the closure of the upper reaches of many eastward-flowing streams inhabited by these fish to professional fishing during this period. These rough figures and the observations of reliable anglers, however, indicate a fairly marked recent decline for a species that in any case was probably never very abundant. Cadwallader and Rogan (1977) claim that the Lake Eildon population of Macquarie perch in Victoria appears to have been adversely affected by overfishing and competition with introduced fish, and river siltation and a reduction in the number of suitable spawning sites by dam construction has probably also had an effect on populations of this species.

Spangled perch (*Leiopotherapon unicola*: Family Theraponidae)
This native freshwater species (previously known as *Madigania unicola*) is not exploited commercially but is regularly angled in more arid regions. Although closely related to the silver perch, its angling value is not as great and is similar to that of several other
closely related central Australian thera­ponids. The species is also known as spangled grunter, jewel perch, bobby cod and nicky. Adults commonly reach 20 cm in length and 110 g in weight; occasionally they may reach weights of up to 570 g.

Lack of tolerance to temperatures below 5.3°C (Llewellyn, 1973) restricts its distribution to northern Western Australia, the Northern Territory, Queensland, the extreme north of South Australia, and north of a line between Menindee and Condobolin in inland New South Wales. It also occurs in coastal drainages in Queensland, but has only once been reported from a coastal drainage in New South Wales (the Nepean River at Castlereagh). On the whole, spangled perch is the most abundant species throughout much of its range, generally frequenting warm turbid waters. Sudden appearances of this species after rains in dams which were previously dry have been explained in terms of an ability to aestivate (live buried in mud), but there is little scientific support for this explanation. Upper and lower lethal temperatures are 39 and 5.3°C.

Some males may mature at 58 mm in length and 2.7 g in weight, and some females when 78 mm long and 8.7 g in weight. Under optimum conditions such fish could mature in one year. Breeding begins in November, generally assisted by flooding, when surface temperatures reach 26°C and bottom temperatures 20°C (Llewellyn, 1973). Fecundity was found to vary from 24,000 eggs in a fish weighing 24 g to 113,200 eggs in a fish of 65 g. Eggs are demersal and are randomly dispersed over the bottom. Typical adult characters are developed at 35 days when scales first appear. These fish are aggressive feeders and eat shrimps, insects and worms.

The spangled perch is a good eating species and an excellent angling fish in spite of its small size. It grows well and breeds prolifically in farm dams, but unless these are heavily cropped, dense populations of small fish result. Since it mostly occurs in arid, unpopulated and undeveloped regions, it is not yet in any danger. However, there is no means at present of monitoring any future decline in its populations.

**River blackfish** (*Gadopsis marmoratus*: Family Gadopsidae)

This native freshwater species is not exploited commercially but is angled throughout its range. Its edible qualities are reported to be excellent. Also called a slippery, slimy, tailor, or marbled river cod, it is the only species yet described in the family Gadopsidae. Size varies considerably; on the mainland it is largest in southern Victoria where it occasionally grows up to 60 cm in length and almost 5 kg in weight. In northern Victoria and New South Wales it usually reaches only 30 to 35 cm in length and about 0.25 kg in weight. It grows largest in Tasmania where a length of 62.5 cm and a weight of 5 kg have been recorded. There is some controversy as to whether the species should be split into three species, one each for the Tasmanian, Victorian and New South Wales populations.

It occurs in many westerly-flowing streams in New South Wales, but in northern New South Wales only at altitudes from 600-900 metres above sea level; in southern New South Wales it occurs much further west and down to altitudes as low as 140 metres above sea level. In Victoria it is common in many
southerly flowing streams and its range used to extend nearly as far west as the Murray River mouth in South Australia. In Tasmania the natural distribution was in the north-east, but the species has now been introduced into rivers elsewhere. The species has also been reported from southern Queensland. Although still common in some regions, it appears that the distribution of this species has become considerably reduced, particularly near the periphery of its range. Because of this, some now regard it as an endangered species (Lake, 1971). Data on its abundance are sparse, though populations are known to fluctuate dramatically in any one area over a period of a few years for reasons which are not known. Small, sluggish streams with plenty of cover, such as snags and boulders, are favoured as habitats. It can, however, survive and breed in dams. It is normally confined to fresh waters but has been caught in tidal and brackish waters in Tasmania. Preferred temperatures are below 21°C.

The life cycle of the river blackfish is poorly understood but breeding is known to begin in spring, the exact time no doubt varying with latitude. In inland New South Wales, blackfish are found to breed at temperatures above 16°C. Large females may produce up to 2,500 large demersal eggs which are deposited amongst stones and hollow submerged logs (Jackson, 1975, 1978). Victorian studies have indicated that 60 per cent of its food consists of caddis fly larvae, but that beetle larvae (12 per cent), flies (4 per cent), yabbies and shrimps (4 per cent) and some other foods are also eaten (Butcher, 1946).

The species can be readily angled, but may not be able to withstand heavy angling pressure. It can live in aquaria though more than one specimen per aquarium often initiates fighting.

**Introduced species**

**English perch** (*Perca fluviatilis*: Family Percidae)

This European freshwater species was introduced into New South Wales in 1888 and into Tasmania in 1862. It is exploited both commercially and as a sportfish, and is also known as redfin, European perch and perch. Its closest relative is *Perca flavescens*, the yellow perch of North America. Specimens may reach 10-4 kg in weight, but usually vary between 0-1 and 1-0 kg.

It occurs throughout most of the Murray-Darling system south of Wilcannia, with the exception of the Beardy River on the New England plateau. Its spread is restricted by an upper thermal death point of 31°C and unsuitable fast flowing streams in the high country. It also occurs in Tasmania, the south-eastern and south-western coastal drainages of the mainland, and the Wimmera district of Victoria. It is abundant in sluggish, weedy streams, and in billabongs and swamps, but its abundance fluctuates markedly in the main streams when movements from its more favoured habitats are stimulated by floods.

Spawning, which is unlike that of any of the native species, takes place in early spring (around August) when water temperatures reach c. 12°C (Lake, 1966). Even one-year old fish are capable of spawning, and a 2.5 kg fish can produce 100,000 eggs. Floods are not required to initiate spawning although they are significant in that they increase larval food supplies. Breeding generally occurs in quiet backwaters. Ova are spawned in a ribbon-like gelatinous mass, which is often wound around aquatic weeds, and
hatch after approximately 8 days. Newly hatched larvae are about 5 mm long and both they and the adults are carnivorous, the larvae often feeding on copepods and cladocerans and the adults on crustaceans, insect larvae and fish (often cannibalistically).

With regard to the commercial catch from the Murray-Darling river system, available statistics indicate that 50 per cent is from Victoria, 28 per cent from New South Wales and 22 per cent from South Australia (Table 15). Five year mean commercial catches over the 15 years to 1975 have been relatively stable at around 45 to 50 tonnes per annum and they have not been influenced by flood frequency. The English perch is a reasonable sportfish, provides good eating, and its flesh is less oily than that of most native species. It survives well in farm dams but rapidly overpopulates and becomes stunted.

English perch have been blamed for the decline of many native species. However, although it may compete with native species such as Murray cod and golden perch near the top of the trophic pyramid, competition with others appears to be minimal. Competition may also occur with other introduced species: an accidental introduction into Lake Canobolas, Orange (New South Wales), led to a rapid decline in the lake's trout population over 3 years. This was probably because of heavy exploitation of the limited food resources within a very restricted habitat. In any event, the spread of English perch by stocking is generally discouraged, and is illegal in New South Wales.

The introduced European carp has recently become one of the most commercially important freshwater species, being exploited for crayfish bait and pet food. Also called common, German, or Asian carp, it can hybridise with the related goldfish and wild hybrids have been found (L. F. Reynolds, personal communication). Recent work by Shearer and Mulley (1978) has shown that there are three varieties of carp in Australia. One variety has been present in irrigation canals north of Narrandera, New South Wales, for many years and has shown no tendency to spread and another is confined to Prospect Reservoir near Sydney. However, a third variety, introduced into Lake Hawthorn near Mildura in Victoria in 1964, later entered the Murray and has since spread throughout virtually the entire Murray-Darling system (Fig. 31). Major dams such as Burrinjuck, Hume and Wyangala are the only obstacles that have checked its spread. It was also once present in a number of farm dams in Tasmania but was successfully eradicated. It is present in Lake Burley Griffin in Canberra and many streams in northern Victoria.

Individuals can weigh up to 45 kg but seldom reach weights of more than 10 kg in Australia. They are found primarily in sluggish waters, with spawning occurring from spring to early summer in shallow areas when temperatures reach about 18°C. The small adhesive eggs are usually scattered over plants. A female weighing 1 kg can produce some 100,000 eggs. European carp are omnivorous and feed by roiling the bottom and then selecting food items exposed. Specimens are hardy, can be readily handled, and survive low oxygen tensions. Tagging experiments indicate that the species is rather sedentary; the furthest distance a

European carp (Cyprinus carpio) and Goldfish (Carassius auratus): (Family Cyprinidae)
tagged fish moved was 100 km (Reynolds, 1976c).

Large quantities are now netted by professional fishermen fishing for other species. Some selective electro-fishing has also been permitted. Angling and edible qualities are only fair and most of the commercial catch is utilised for pet food or bait. The commercial catch in South Australia rose from 2 tonnes in 1967-70 to 235 tonnes in 1975-76, indicating a very rapid increase in population size as the species spread (Reynolds, 1976a). There are indications that population numbers are now stabilising. Although the apparent decline in the population sizes of many native species has been attributed to the spread of European carp, these do not coincide with the sudden appearance and sharp increase in carp populations since 1970; many native species appear to have been declining over a much longer period. Nevertheless, population trends thus far do indicate that carp has at least had significant impacts on tench and bony bream, if not other species. There is also growing concern about its possible

Fig. 31 Distribution of European carp (Cyprinus carpio) in Australia. A. Prospect Reservoir; B. Murrumbidgee Irrigation Area (Riverina); C. initial distribution of the most recent (1960) introduction. Broken line indicates extent of distribution in 1977.
long-term effects on yabbies. However, it now constitutes an important dietary item for golden perch and Murray cod and may be of benefit to these species. European carp have affected the aquatic environment in other ways too; many inland swamps (e.g. in the Narrandera area) which were formerly heavily weeded are now almost entirely open, turbid water bodies. This change appears to affect the potential of these swamps to support waterfowl.

The goldfish (sometimes simply known as 'carp') was introduced in 1876. It has little economic value other than for bait and in the aquarium fish industry. Although they may reach weights of up to 2 kg, most individuals normally weigh less than 0.5 kg. This species has similar breeding habits to the European carp.

Widespread in southern Australia, it occurs as far north as the Fitzroy River in Queensland. It is absent from high altitudes and prefers slowly flowing streams or still waters; it is thus common in dams, water holes and billabongs. Maturity is reached after one year and spawning takes place in spring and early summer when water temperatures reach 15°C. The adhesive eggs are approximately 1 mm in diameter and generally adhere to plants; a female weighing only 250 g can lay tens of thousands of eggs. Fry feed on plankton; adults are omnivorous and feed on insects, algae, other plants, and detritus. Numerous goldfish farms exist in south-eastern Australia to supply the aquarium fish trade (see Chapter 21), but unfortunately goldfish previously imported may have already introduced a number of fish diseases. The importation of goldfish is now banned in several states for this reason. Until the recent expansion of European carp populations, commercial catches of goldfish were around 5 tonnes per annum.

**Tench (Tinca tinca: Family Cyprinidae)**
The tench or doctor fish was introduced from England in 1876. Until 1974, when European carp became the dominant crayfish bait species, it was the second most important commercial bait fish after bony bream. Closely related to other introduced cyprinids, tench have been known to reach a weight of 9 kg in New South Wales (even higher weights are known overseas).

The species occurs in the more sluggish waters of the Murray, Murray-bidgee and Lachlan Rivers, and in lakes, lagoons and ponds with abundant weed growth. Its distribution, however, is generally rather patchy. It was once common in lakes near the mouth of the River Murray, and remains common in Tasmania. Eventually, it is expected that its distribution will approach that of the English perch.

Spawning occurs in late spring to early summer when water temperatures reach 16°C (Lake, 1966). Females produce c. 400,000 ova per kg body weight, these eggs being small (1 mm diameter), adhesive, and generally attached to plants. The diet is essentially carnivorous, but plants are also eaten. Angling qualities are good but the fish is seldom angled in Australia. Commercially, this species is caught with gill nets.

Available statistics indicate that South Australia contributes 57 per cent, Victoria 27 per cent and New South Wales 16 per cent of the commercial catch (Table 15). Five year mean commercial catches between 1961 and 1975 showed a marked increase from 14 to 140 tonnes. However, in 1974 the catch plummeted,
Aquatic fauna

a phenomenon that has been associated with the presence of large numbers of European carp (Reynolds, 1976b). In any event, the European carp is displacing tench, probably because it is a direct competitor for the same food resources.

**Yabby** (*Cherax destructor* and **Murray crayfish** (*Euastacus armatus*): (Family Parastacidae)

*Cherax destructor*, the yabby, is the most heavily commercially exploited of the Australian freshwater crayfishes. It occurs in the Murray-Darling and Lake Eyre drainage systems and the coastal drainages of Victoria, and has been introduced into Western Australia. Maximum lengths (excluding chelipeds) of about 20 cm are reached. The species lives primarily in static or very slowly flowing waters (dams, lakes, swamps) where it forms a major food item for many large native fish species and also birds such as cormorants and ibis. A large commercial fishery has been established in South Australia, mainly in the lakes at the mouth of the River Murray, principally through the development of an export market to Scandinavia; annual catches have risen from 42.7 tonnes in 1969-70 to 127.3 tonnes in 1974-75 (Table 15). However, the stability of the fishery may be threatened by the rapid increase in European carp populations; both species are omnivorous, and the yabby may be affected not only by direct competition for food but also by habitat disturbance caused by the carp’s feeding habits (roiling). Overfishing (i.e. exploitation resulting in total mortality greatly exceeding recruitment) could possibly temporarily deplete local populations. Attempts have been made to farm yabbies, but sound techniques have been difficult to establish (see Chapter 21).

*Euastacus armatus*, also known as the spiny lobster or Murray crayfish, the second largest known freshwater crayfish in the world (maximum weight 3 kg), occurs in the Murray and Murrumbidgee Rivers where it is exploited by both professional and amateur fishermen. It is also found in the Goulburn River and the upper reaches of the Darling River system in the western Blue Mountains of New South Wales. Its distribution, however, is apparently shrinking and for unknown reasons it has now vanished from considerable stretches of the Murrumbidgee River and from the Murray River in South Australia. *Euastacus* species typically occur in a wide range of habitats from cold, highly oxygenated, fast-flowing mountain streams to warmer, slower-flowing waters such as the inland Murray-Darling system. Most (80 per cent) of the commercial catch of *E. armatus* is taken in New South Wales, the remaining 20 per cent in Victoria (Table 15). It is sold as a table delicacy.

**Forage fish**

There are approximately 19 native species which can be classed as forage species in the Murray-Darling system (Table 17). Generally they are small, often colourful fish which play an important role in the general ecology of the system as an important food source for larger fish. They are of no direct economic significance except that several species are of minor importance to the aquarium fish trade. The breeding biology of only 10 of the species is known, and of these only one (*Galaxias maculatus*) requires flooding to stimulate spawning. Egg deposition is of various sorts: four species of gudgeon deposit dense batches of adhesive eggs (these would probably not
survive marked water-level fluctuations); rainbow fish randomly disperse eggs with long adhesive strands amongst aquatic weeds; and the five other species with known breeding habits randomly disperse demersal eggs on the bottom. The mosquito fish, *Gambusia affinis*, is the only introduced forage species.

**F. Lake Eyre Internal drainage system.** This system is one of the world's largest internal drainage systems, slightly larger than that of the Murray-Darling catchment. Rivers generally flow south to terminate in the Lake Eyre or Lake Frome systems, and permanent natural waterholes are sparse and generally elongated in shape (3 to 30 km long). Scattered bores, artesian springs, reservoirs and dams also occur. Although occasionally huge floods inundate vast tracts of land, most water is ephemeral. Thus, Lake Eyre was filled in 1974 for only the second time this century, though it is likely to be dry again in 1979. Its drying has caused heavy losses of fish in the area (Ruello, 1976).

There are 29 species of fish recorded from the system (Table 18; Glover and Sim, 1978a,b) including a number of undescribed ones (spp. of *Neosilurus*, *Hephaestus* and *Hypseleotris*). Overall, none is of much economic value with the possible exception of the golden perch, which is rarely exploited. Six species are of some angling interest, of which the golden perch, spangled perch and silver perch are the most important (in that order). There are 20 native forage species including 8 which are also known from the Murray-Darling system. The breeding biology of only two is known—the Australian smelt and the Mitchellian freshwater hardyhead.

Many species in the system are well-adapted to the harsh environment. The desert goby, for example, tolerates rapid fluctuations in water temperature and salinity; it also has behavioural responses which mitigate the effects of high temperatures (it shelters in weeds and silt beds) and can respire aerily when dissolved oxygen levels become low.

Two introduced forage species occur, both mosquito fishes (*Gambusia affinis* and *G. dominicensis*). The latter is absent from the Murray-Darling system.

**G. Central Western Desert drainage system.** This extremely dry region covers around two and a half million square kilometres of unco-ordinated drainage catchment. Creeks are short, flow only after heavy local rains, and permanent surface water is almost completely absent. Recent collections have recorded only five species (Glover and Sim, 1978a,b) of which only one, the spangled perch, has any potential angling value. Commercial exploitation and angling in this area is presently nil, though some local Aboriginal communities may catch spangled perch for food. Three of the five known species are forage species.

**General conclusions concerning management of inland warmwater fish and fisheries.** Man has had a tremendous impact on the nature of the aquatic environment over large areas of the Murray-Darling system. In the other two systems discussed in this section the impact has been very much less. Man-made modifications in the Murray-Darling system include, in particular, the construction of 84 water storages and innumerous small farm dams (see
Table 18  Freshwater fishes of the Lake Eyre Internal and Central Western Desert drainage systems (distribution ranges from Glover and Sim, 1978a,b)

<table>
<thead>
<tr>
<th>Scientific name</th>
<th>Common name</th>
<th>Abundance¹</th>
<th>Economic importance²</th>
<th>Commercial value³</th>
<th>Angling value³</th>
<th>Distribution⁴</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nematalosa erbi</td>
<td>Bony bream</td>
<td>4</td>
<td>A</td>
<td>?</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Retropinna semoni</td>
<td>Australian smelt</td>
<td>4?</td>
<td>E!</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Neosilurus argenteus</td>
<td>Silver tandan</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>N. hyrtli</td>
<td>Hyrtis tandan</td>
<td>1</td>
<td>B</td>
<td>—</td>
<td>+</td>
<td>?</td>
</tr>
<tr>
<td>Neosilurus n.sp.1</td>
<td>—</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Neosilurus n.sp.2</td>
<td>—</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>?</td>
</tr>
<tr>
<td>Neosilurus n.sp.3</td>
<td>—</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>?</td>
</tr>
<tr>
<td>Neosilurus n.sp.4</td>
<td>—</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>+</td>
<td>—</td>
</tr>
<tr>
<td>Gambusia affinis</td>
<td>Mosquito fish</td>
<td>4</td>
<td>F</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>G. dominicensis</td>
<td></td>
<td>2</td>
<td>F</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Nematocentris maculata</td>
<td>Checkered rainbow fish</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>N. tatei</td>
<td>McDonnell Ranges rainbow fish</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Craterocephalus fluviatilis</td>
<td>Mitchellian freshwater hardyhead</td>
<td>2</td>
<td>E!</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>C. eyresii</td>
<td>Lake Eyre hardyhead</td>
<td>3</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>C. dalhousiensis</td>
<td>Dalhousies hardyhead</td>
<td>3</td>
<td>E</td>
<td>—</td>
<td>+</td>
<td>—</td>
</tr>
<tr>
<td>Ambassis castelnaei</td>
<td>Western chanda perch</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Denariusu bandata</td>
<td>Penny fish</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Plectroplites ambiucus</td>
<td>Golden perch</td>
<td>3</td>
<td>A</td>
<td>+</td>
<td>+</td>
<td>—</td>
</tr>
<tr>
<td>Letopotherapon bidyanus</td>
<td>Silver perch</td>
<td>2</td>
<td>A</td>
<td>+</td>
<td>+</td>
<td>—</td>
</tr>
<tr>
<td>L. unicolor</td>
<td>Spangled perch</td>
<td>4</td>
<td>B</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Amniliata percoides</td>
<td>Black striped grunter</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Hephaestus welchii</td>
<td>Welch's grunter</td>
<td>1</td>
<td>B</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Hephaestus n.sp.</td>
<td>—</td>
<td>1</td>
<td>B</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Scortum barcoo</td>
<td>Barcoo grunter</td>
<td>1</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Hypseleotris n.sp.1</td>
<td>—</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Hypseleotris n.sp.2</td>
<td>—</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>?</td>
</tr>
<tr>
<td>Mogurnda mogurnda</td>
<td>Purple striped gudgeon</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Glossogobius gius</td>
<td>Flathead goby</td>
<td>2</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Chlamydogobius eremius</td>
<td>Central Australian goby</td>
<td>3</td>
<td>E</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
</tbody>
</table>

¹ Abundance code: 1, rare; 2, occasional; 3, common; 4, abundant.
² A, native species of economic importance; B, large native species but of limited or no economic importance; E, small native species of no economic importance; F, small introduced species of no economic importance; !, species in which breeding biology is known.
³ +, of significant commercial or angling value; —, of no economic value.
⁴ +, present; —, not recorded; ?, position uncertain.
Chapter 32), gross alterations to the nature of catchments (by clearing and grazing), the addition of pollutants to waterways, and the introduction of exotic biota (especially fish, and above all the European carp). Fishing may have an additional impact on certain species. The extent of the total impact and the importance of each modification cannot be precisely determined, nor ever will be since too little is known about the distributions and abundances of native fish in these inland systems before man began to change their nature. Only one thing is certain and that is that the impact of man on native fish in the Murray-Darling system has been profound and generally deleterious.

Fisheries management in this context largely revolves around the accrual of more knowledge concerning individual species and their environments, attempts to mitigate known deleterious impacts (e.g. by preventing over-exploitation, eliminating pollution, etc.), and conservation (protection) of endangered and economically valuable species. A special difficulty for management, however, is the distinctive nature of Australia’s inland lakes and rivers, a fact which renders management experience gained elsewhere not easily transferable to Australian conditions.

One possible specific management technique deserves brief mention. To overcome the problem of disruptions to annual breeding cycles and recruitment in fish populations, it may be desirable to arrange for releases of water of a suitable temperature from large impoundments at times appropriate for fish breeding in localised areas, and to retrieve this water further downstream. To facilitate such releases future impoundments should incorporate multiple level outlets. However, it is not known whether such releases would be adequate to provide a suitable environment for optimum survival of the resulting fish larvae. Alternatively, at times suitable for fish breeding, floods could be simulated in previously stocked billabongs by directing water into them. After breeding, the water could be emptied back into the river system. Techniques of this sort would demand close liaison with water conservation bodies.

Mountain streams and lakes
Australia’s eastern highlands and southern states, including Tasmania, encompass many streams and lakes in which water temperatures rarely exceed 25°C and are well below 10°C for much of the year. Such cool waters typically occur at above 600 m elevation in the temperate south-eastern coastal, Murray-Darling, and throughout the Tasmanian drainage systems. The continuing creation of impoundments, moreover, is extending the mainland distribution of cooler waters to lower elevations. These cool waters do not usually contain many native fish species of economic or recreational importance. For example, Costin (1954) recorded only two species of eel (Anguilla australis and A. reinhardtii) and several small galaxiid species from the montane tract (914 to 1,524 m elevation) of the Snowy Mountains area of New South Wales and Victoria. Some larger fish species of recreational importance, such as the silver perch (Leiopotherapon bidyanus), Macquarie perch (Macquaria australasica), Murray cod (Maccullochella peeli), trout cod (M. macquariensis) and river blackfish (Gadopsis marmoratus), do occur in tableland waters between 600 and 900 m altitude. However, there are few records
of their breeding in such waters and populations are usually much smaller than in warmer waters at lower altitudes. In Tasmanian waters only three indigenous species achieve significant size: the short-finned eel (Anguilla australis), the river blackfish and the sandfish or congolli (Pseudaphritis urvilli). Although all of these species are caught and, in some instances, sought by anglers, none constitutes a major fishery in cooler waters, except in a few scattered habitats where one or other is comparatively abundant.

Several galaxiid species inhabit cold waters also but all are too small to be of recreational importance. Their distribution and abundance has become much reduced in recent years, probably largely through competitive pressure from introduced salmonids (cf. Chapter 23). Eels are commercially exploited in Victoria and Tasmania (Table 15) where eel-fishing is confined to coastal streams.

Australia's cold fresh waters, therefore, contain few indigenous fish populations of commercial or angling importance. However, such cool waters have generally proven to be an ideal habitat for introduced salmonid species.

**Introduced salmonids** (Salmo spp., Salvelinus fontinalis, Oncorhynchus tshawytscha: Family Salmonidae)

The five species which have been introduced are the brown trout (Salmo trutta), the rainbow trout (S. gairdneri), the Atlantic salmon (S. salar), the brook trout (Salvelinus fontinalis) and the quinnat salmon (Oncorhynchus tshawytscha). Both brown and rainbow trout have adapted well to Australian conditions, now have wide distributional ranges (Weatherley and Lake, 1967), and dominate the cold water fisheries. Populations of brook trout and the two salmon species are maintained in government hatcheries but have had only limited success in wild habitats. Although there is some recent evidence of self-maintaining populations of brook trout in a few waters in New South Wales, there is no positive evidence for either of the two salmon species breeding in the wild. Brown and rainbow trout therefore constitute the principal angling species. Rainbow trout are also of increasing importance in aquaculture in Australia (see Chapter 21). Roughley (1951) described the history of trout introductions, and Weatherley and Lake (1967) and Tilzey (1977b) summarised the major reasons for their success in Australia.

Trout are undoubtedly the most popular freshwater recreational fish in Australia, in terms of angler numbers. This popularity stems from their sporting and edible qualities, their occurrence in proximity to the comparatively densely populated south-eastern seaboard, their dominance in most Tasmanian fresh waters, and the northern hemisphere origins and fixations of most Australian inland anglers. Collins (1976), in an economic survey of inland angling in New South Wales, observed that 66 per cent of the fishing effort was directed towards trout, and only 34 per cent towards native species. Although no comparable estimates are available, the proportions are probably similar in Victoria, and trout certainly dominate freshwater angling in Tasmania.

There have been a number of studies on trout in Australian waters (e.g. those of Lake, 1957; Nicholls, 1957, 1958a, b,c,d, 1961; Morrissy, 1967; Jackson, 1975; Tilzey, 1977a). Several observations which deal only with trout food have also been published (Butcher, 1945,
1946; Evans, 1942; McKeown, 1934a,b, 1936, 1937, 1955; Lake and Bennison, 1977). However, despite this abundance of local knowledge, and, of course, a vast amount of overseas knowledge on trout biology and management, it is mainly very basic management techniques which are currently applied in Australia. At first sight this seems somewhat surprising in view of the recreational importance of trout fisheries; it is explained, however, by the relative absence in Australia of many problems that beset North American and European trout fisheries and which prompted the development there of sophisticated management policies. The majority of Australian trout fisheries are not subject to many of these problems, such as very high angling pressure, over-exploitation, or excessive pollution. On the other hand, the healthy condition of most local trout fisheries has produced several local dogmas. Of these, stocking, closed seasons, bag limits and minimum legal lengths merit discussion.

Whereas there is no doubt that stocking is essential for trout fisheries where fish cannot breed, there is also little doubt that most stocking effort is wasted. Nicholls (1958c) studied the effectiveness of stocking in a Tasmanian stream and concluded that less than 2 per cent of hatchery-reared fish reached the angler. Lake (1957) arrived at essentially the same conclusion for trout waters in New South Wales. More recent quantitative studies are lacking, but there is no reason to suppose that current stockings of lotic trout waters are any more successful. Stocking, however, still remains the backbone of Australian trout management policy.

Apart from its ineffectiveness, stocking is also an expensive short-term procedure. American studies such as that by Rawstrom (1973) have shown that even when trout are planted at a legally-takeable size and the harvest in most cases exceeds 40 per cent, the cost of each fish caught is nearly one American dollar. The cost of each hatchery-reared trout reaching the Australian angler is probably much higher in view of the extremely low harvest rates in our waters. An economic appraisal of Australian trout stocking practices is definitely needed, and research along these lines is presently being carried out in New South Wales. The futility of rearing brown trout in hatcheries for later release into waters containing self-maintaining populations of trout has been realised by New South Wales State Fisheries; this authority discontinued stocking streams with brown trout after 1964. Continuous dissatisfaction with this decision on the part of angling groups has recently resulted in a few streams in New South Wales again being stocked with brown trout ‘on an experimental basis’.

It may further be added that although no evidence is yet available, there are good grounds for the belief that continual stocking actually damages existing trout stocks by interrupting the process of natural selection. Trout have been in Australia for over a century and they are undoubtedly adapting to Australian conditions. Morrissy (1973) demonstrated that a Western Australian strain of *S. gairdneri* possesses a higher temperature tolerance than fish from Victoria and New South Wales. A recent North American study on the swimming ability of *S. gairdneri* (Tsuyuki and Williscroft, 1977) found that a New Zealand strain could outswim the progeny of ancestral
Aquatic fauna

262

stocks. Also, selective breeding programs (e.g. Donaldson, 1971) have clearly demonstrated the genetic plasticity of trout stocks. Thus, if stocking has to be carried out it should be on a selective basis with a sound knowledge of the genetic background and physiological abilities of the stocks used. For example, the so-called ‘marginal trout waters’ of New South Wales are usually streams in which periodic thermal trout-kills occur, and it is in these waters that natural selection towards higher temperature tolerances is most likely to be occurring. It is also for these waters that stocking is most often demanded by anglers. Ideally, they should not be restocked unless the genetic background of the released stock matches that of the original stream stock, or unless a complete kill has occurred. Mostly, restocking is not in fact required, as the progeny of even a small number of survivors rapidly repopulate the affected stream. Numerous researchers (e.g. Neave, 1953; Fraser, 1969; McFadden, 1969) have shown that the survival of salmonid juveniles decreases markedly with increasing population density; the low trout population densities typically following periods of drought or thermal stress therefore favour the survival of the next generation.

The above criticisms should not be taken to infer that stocking serves no purpose in Australian waters. That is not so; it is essential to some salmonid fisheries and a most useful management technique when correctly deployed. The extremely successful quinnat salmon fishery of Lake Purrumbete, Victoria (Barnham, 1977), is maintained solely by stocking, as are numerous small lentic trout fisheries in New South Wales and Victoria. Releases of brook trout and Atlantic salmon into Lake Jindabyne, New South Wales, have provided much sport for the angler. However, existing stocking practices undoubtedly squander the bulk of released stock. Funds so wasted could be more usefully employed in financing much-needed fisheries research. It should be stressed that the fault lies not with the fisheries authorities responsible for government hatcheries, but with the angling public. Any attempt to reduce numbers of fish released would be met by indignation from most trout fishermen. Thus, education of the angling public is needed before progressive management policies can be implemented.

The enforcement of closed seasons, bag limits and minimum legal lengths for fish retention are other local dogmas. They undoubtedly serve little purpose in all but the most heavily-fished trout streams. Many Australian trout waters are certainly overpopulated, and therefore have a low fish growth rate and a small mean fish size. This fact was clearly recognised by the Victorian Fisheries and Wildlife Department which in 1970 took the bold step of removing the closed season and size and bag limits on all trout waters. Unfortunately, there has been little follow-up research. Many cold water impoundments in New South Wales have been open to year-round fishing since the early 1960s.

With regard to the management of salmonids in Australia, the most effective strategies can best be devised through study of individual fisheries since each usually has distinctive characteristics. Obviously, this is justifiable only when the fishery is important. This is certainly so in Lake Eucumbene, New South Wales, and a discussion of some
management techniques applied in this fishery is apposite.

Brown trout overpopulation is the major problem in this fishery. As a management strategy, therefore, anglers have been permitted exploitation of the brown trout spawning run since 1972. The decision to permit this was prompted by several reasons. The lake contains both brown and rainbow trout but tagging experiments showed that the latter was more readily caught by anglers. Rainbow trout stocks were therefore better utilised and of greater benefit to lentic trout anglers than brown trout. Further, catch-per-effort analyses showed that whereas rainbows were much more readily caught in summer, browns were more readily caught in winter. The year-round angler obviously benefited from having both species present. However, studies into the competitive interactions between the two species revealed that browns depressed rainbow survival and consequently depressed catch rates during the major, summer tourist season. Moreover, the increase in brown trout numbers was resulting in a decline in the mean weight of browns caught, as well as a decline in rainbow numbers. The removal of spawning browns, a relatively simple management strategy, was therefore aimed at checking the decline in brown trout size and easing competitive pressures on rainbow trout. The exploitation of spawning browns did not depress the catch-per-effort for lake browns throughout the period 1972 to 1978, and there are strong indications for the 1977-78 season that the mean size of brown trout caught has now ceased declining (R.D. J. Tilzey, unpublished data). Additionally, the decline in rainbow catch-per-effort has slowed since 1974-75, although it is too early to assess if any increase in rainbow numbers is occurring.

Continual monitoring of the catch is an essential part of the management of the Lake Eucumbene fishery. It is, however, one of the few fisheries which is so monitored. The lack of catch data for other water bodies means that effective management of them is significantly hampered, as it is only when catch data for several years exist that confident assessments concerning the health of a fishery is possible.

**General conclusions concerning the management of coldwater fish and fisheries.** The first and absolutely essential principle of fisheries management is to maintain desirable stocks. In this regard, the majority of Australian trout fisheries are in a very healthy condition, especially when compared with many ancestral northern hemisphere stocks, or some native fish stocks. Several Australian trout fisheries undoubtedly rank amongst the best in the world. The success of trout in Australia is largely because of favourable physico-chemical and biological conditions in many of the habitats into which they have been introduced (Tilzey, 1977b) rather than the result of the application of enlightened fisheries management policies. Moreover, the continuing creation of upland impoundments is providing yet more suitable habitats. The future of reservoir trout fisheries in Australia, therefore, seems reasonably assured, even considering mounting recreational and environmental pressures on them. The same cannot be said with such confidence, however, of trout fisheries in some of our streams.
Water conservation schemes (particularly dam construction), catchment despoliation (see Chapter 10), and pollution are adverse factors of major importance to stream fisheries in mountainous regions. Overall there has been a considerable loss of stream habitat so that whilst lake trout fishermen are increasingly well catered for, this is not so for stream fishermen.

Although trout dominate coldwater fisheries, the conservation of native fish is also a matter of concern for fisheries management in mountainous regions, despite the negligible economic value of these native fish. Several coldwater galaxiids in particular have extremely limited distributions and can thus be considered as endangered species (some are confined to only one lake). In order to bring about effective conservation of these and other coldwater native species considerably more research is needed. In any event, there is increasing evidence (Tilzey, 1976; see Chapter 23) that trout have had a direct and deleterious effect on some indigenous species, especially galaxiids. The management of coldwater fisheries should include setting aside trout-free non-fishing reserves where needed; trout fishermen are extremely well catered for and little justification exists for placing trout into any more natural waters in which they do not presently occur. This policy already applies at least in New South Wales and to some extent in Victoria.

The major factor limiting research into, and therefore effective management of, coldwater fisheries is inadequate funding. This is largely because most coldwater fisheries are recreational in nature with economic values not readily assessable (although of immeasurable benefit in aesthetic, spiritual and psychological terms). Consequently, whilst federal monies aid research on many commercial fisheries, the financial responsibility for recreational fisheries research lies mainly with state governments. Care should be taken not to underestimate the economic value of recreational fisheries, despite the difficulty of arriving at a value. An investigation by New South Wales State Fisheries is of interest in this connection. This organisation conducted a survey of inland angling in New South Wales during the 1969-70 financial year (Collins, 1976). A detailed questionnaire was distributed to anglers and on the basis of 1,018 returns it was estimated that the average inland angler spent $493 on fishing tackle, travel, accommodation and so on in relation to his sport during the financial year in question. Simple extrapolation from the number of inland licences sold (78,724) showed that approximately $39 million was spent on inland angling in New South Wales during 1969-70, of which nearly $26 million was directed towards trout fishing. Considering inflation and the increased number of licences sold (119,168 in New South Wales during 1976-77), these amounts are undoubtedly much larger now. Whatever its real economic value, it is clear that recreational inland fishing, and especially coldwater trout fishing, is a multi-million dollar industry.

Endangered species and main conservation problems
Lake (1971) pointed out that, 'in Australia, a country considered by many to be underdeveloped and underpopulated, approximately one-third (about fifty species) of our freshwater fishes, most of them endemic to Australia, are in jeopardy'.

Aquatic fauna
He classified these species into four groups as follows: A. Seriously threatened species (Australian grayling, Macquarie perch, trout cod and river blackfish); B. Species whose distribution and/or abundance have been considerably reduced (Tasmanian whitebait, freshwater catfish, silver barramundi, Australian bass, golden perch, Murray cod and Tasmanian blackfish); C. Species not yet in a serious position but threatened by changes which are taking place now (freshwater herring, estuary perch and jungle perch; several additional species whose life histories are not well known but which probably need to move between fresh and brackish or salt water for completion of their life cycles could be added here); D. Species which are restricted often to one or two river systems and/or with very patchy or discontinuous distributions (37 species, the most notable being the Australian lungfish, the two species of spotted barramundi and several galaxiids; many more species could probably also be added to this latter list).

Lake concluded that the main threats to our native freshwater fishes were factors causing a reduction in or failure of their reproductive capacity. These factors were listed as siltation, fluctuating water levels due to irrigation demands, removal of snags, dam construction affecting flooding and water temperature (inland species), dam construction preventing life cycle movements (coastal species), insecticides washed from agriculture and forestry, aquatic weedicides, and increased nutrient loads from fertilisers and sewage, etc.

While one could argue with some of the details of Lake's analysis in relation to the exact threat status of different species and the relative importance of the different factors mentioned, his analysis is basically sound and highlights the main conservation problems facing our freshwater fishes today.

**Concluding remarks**

It is appropriate to conclude this chapter by listing in summary form a number of recommendations concerning basic management strategies. Several have already been mentioned in relation to particular species and/or their habitats, but can usefully be reiterated in a general synthesis. Some of these recommendations are to some extent already being implemented in several states.

Further research should be undertaken on the basic biology, distribution and environmental requirements of native fish. This applies particularly to those considered to be endangered species.

More attention should be accorded the requirements of fish and fisheries in the operations of water resource agencies. This will entail close liaison between fisheries and water resource authorities.

The effects of so-called 'river improvement' schemes, and forestry and agricultural practices, should be further investigated and appropriate measures taken to incorporate the results of this research into appropriate management policies. A number of actions should be implemented without further research: greater control is needed of channelisation, desnagging, aerial pesticide spraying near inland waters, and the clearing of vegetation near watercourses.

The effectiveness of fish ladders for Australian species under Australian conditions should be investigated.
Aquatic fauna

Where appropriate, suitably designed fish ladders should be provided for all present and future dams.

An analysis should be undertaken of the effectiveness and economics of all planned and existing hatchery and stocking operations (both for salmonids and warm-water species).

The economic value of both recreational and commercial freshwater fisheries should be determined with greater precision than presently exists.

More reliable data should be collected on catch-per-effort for both commercial and recreational fisheries. In the case of recreational fisheries, more effort should be directed towards angler education to facilitate the collection of such data.

The transfer of fish to drainage systems in which the species does not naturally occur should only be carried out when 'resident' species are not adversely affected.

More careful control of the importation of aquatic organisms, especially of ornamental aquarium fishes, is clearly needed (see Chapter 23).

To prevent the introduction of fish diseases and parasites, strict quarantine regulations should be enforced in relation to all imports of exotic fish and other aquatic organisms.

The scientific basis of existing fishing restrictions (bag limits, minimum legal lengths, closed seasons, etc.) should be justified.

Habitat reserves should be created for rare or endangered species, and for those with restricted distributions or of particular scientific interest. The giant Tasmanian crayfish and the Queensland lungfish are already protected in this fashion.

Finally, the most crucial point in the management of freshwater fish and fisheries is the need for a thorough consideration of effects on the freshwater biota when formulating water resource management policies. Although one cannot deny the need for irrigation works and the construction of some new dams and impoundments, the effects of such works on fisheries need to be carefully evaluated. Where such works are carried out, construction should and must be done in such a manner as to minimise the impact on the natural aquatic environment and ensure the maintenance of freshwater fish populations.

To cite the National Water Policy Statement of the Australian Water Resources Council in this regard:

the importance of water as a basis for recreation is already high and will undoubtedly increase. Social objectives which may not be directly compatible with economic efficiency must be given proper weight. Hence it is particularly appropriate in the case of water reservoir projects that they are planned and assessed not only on the basis of the extent to which they influence economic growth but also on the basis of their impact on social well-being, on regional development and on the environment generally.

These are fine sentiments but they contrast strongly with more factual statements such as 'the effects of dam release on river ecology have received little attention and require further
research’ (State Pollution Control Commission of New South Wales, 1978).

Providing reasonable implementation of the above recommendations occurs, the future of both recreational and commercial freshwater fisheries in Australia should be assured. If inadequate implementation occurs, increasing insecurity for freshwater fish and fisheries is the most predictable scenario. In short, if we look after the environment, our native freshwater fishes will look after themselves.

Acknowledgments
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23 Introduced fish

R. D. J. Tilzey

Introduction
In his valuable review of the Australian freshwater fish fauna, Lake (1971) noted that many people were surprised to learn that inland waters possess fish other than introduced species such as trout! A cursory glance at Australia's colonial history possibly explains the origin of such public ignorance. Early settlers were quick to import the well-known animals of their homelands and the prime concern of freshwater anglers was to establish familiar fish species here. Comparatively little attention was paid to the native fish fauna, an attitude that has regrettably persisted. The biology of introduced fish species is, in most cases, more clearly understood than that of indigenous species.

Introduced fish have certainly flourished in Australian inland waters. Weatherley and Lake (1967) summarised available information on the biology, distribution, population characteristics, food and growth of introduced freshwater fish species and examined the probable reasons for their success in Australia. Tilzey (1977) suggested the key factors in the establishment and success of trout in Australia were the physico-chemical and biological similarities between Australian habitats and those of ancestral stocks, the abundance and availability of native food species, the minimal competition from indigenous fauna, and the virtual absence of parasites and diseases. Such factors undoubtedly played varying roles in the establishment and success of other introduced fish species but the brevity of this chapter precludes more detailed examination.

Weatherley and Lake's (1967) review is an essential background against which to view more recent information on introduced fish species. This chapter attempts to supplement their review without reiteration except where absolutely necessary.

The species and their distribution
Weatherley and Lake (1967) listed nine self-maintaining species:

- *Salmo trutta* L. brown trout
- *Salmo gairdneri* Richardson rainbow trout
- *Cyprinus carpio* L. European or common carp
- *Carassius carassius* (L.) crucian carp
- *Carassius auratus* (L.) goldfish
- *Tinca tinca* (L.) tench
- *Rutilus rutilus* (L.) roach
- *Perca fluviatilis* L. European perch, redfin
- *Gambusia affinis* (Baird and Girard) mosquito fish

They further noted the presence of quinnat salmon (*Oncorynchus*...
tschawytscha) and Atlantic salmon (Salmo salar) populations for which no evidence of natural reproduction was available.

All of the above species still persist with the possible exception of Carassius carassius. A recent (1974-75) freshwater fish survey of south-eastern Australia failed to yield this species although the closely related C. auratus or goldfish was found to be widespread. It appears probable that many early records are unreliable as, in many instances, any Carassius specimen devoid of gold colouration was considered to be C. carassius. In fact, few wild goldfish possess this classic colouration. The continued existence of C. carassius in Australia therefore needs verifying.

The 1975 distribution ranges of Salmo trutta, S. gairdneri, C. auratus, Tinca tinca, Rutilus rutilus, Perca fluviatilis and Gambusia affinis in south-eastern Australia did not significantly differ from those described by Weatherley and Lake (1967) although the continuing creation of inland impoundments is slowly expanding the distribution of trout species, and Rutilus rutilus now occurs in Lake Eildon, Victoria (P. L. Cadwallader, personal communication). There is also some evidence that trout are becoming increasingly adapted to Australian conditions; Morrissy (1973) found Western Australian strains of S. gairdneri had a higher temperature tolerance than those from the cooler waters of south-eastern Australia. Such evolution could slowly extend trout distribution as the major zoogeographic barrier is maximum seasonal water temperature. The status of Onchorynchus tschawytscha and Salmo salar has remained virtually unaltered, although occasional spent specimens of the latter species have been reported from the Thredbo River, N.S.W., indicating that spawning may be occurring therein.

The distribution range of Cyprinus carpio, the European carp, has, however, undergone a rapid expansion since the early 1960s. According to Weatherley and Lake (1967), this species was confined mainly to the Riverina area of New South Wales with a population in Prospect Reservoir, near Sydney, N.S.W. They further noted that in 1960 C. carpio was released into many farm dams in Victoria after having been imported by a commercial fish farmer. This latter introduced population now inhabits virtually the entire Murray-Darling system, having spread through New South Wales and into Queensland (Fig. 31). Recent electrophoretic and taxonomic studies strongly suggest that there are three distinct strains of C. carpio in Australia, namely 'Riverina', 'Prospect' and 'Victorian' (Shearer and Mulley, 1978). If this suggestion is true it does much to explain why a species which occupied a comparatively restricted Australian distributional range for a considerable period suddenly expanded boundaries. Although the Prospect strain is confined to this reservoir, no such physical barrier prevented the Riverina strain from gaining access to the Murray-Darling system; the inference is that the recently introduced 'Victorian' strain, of German origin, is better equipped for survival under Australian conditions.

In the past decade wild populations of at least another six exotic fish species have become established in Australian inland waters. These are:
Introduced fish

Lake (1971) also recorded the presence of the belut (*Fluta alba*) and cuchia (*Amphipnous cuchia*), two eel-like fishes, in Queensland waters and concluded that both had probably been introduced from Asia. Only isolated reports of these two species exist and it is not known if self-maintaining populations occur. *Gambusia dominicensis* has also been reported from the Eyre drainage, in South Australia, and was probably introduced along with *G. affinis*. Presumably, self-maintaining populations exist.

It is significant, and disturbing, that only one of the species tabled immediately above was deliberately introduced. The brook trout (*Salvelinus fontinalis*) has been widely distributed in N.S.W. and placed in several Tasmanian waters with the intention of providing an additional sportfish for trout anglers. In N.S.W. its distributional range matches that of brown and rainbow trout, but its success has been limited, probably as a result of competition from the latter species. However, natural reproduction has occurred in some waters (L. C. Llewellyn, personal communication) and wild populations existed in 1977.

Feral populations of the swordtail, platy, guppy, sailfin molly and rosy barb have been established via the aquarium fish trade. All were apparently confined to south-eastern Queensland in 1977 (R. J. McKay, personal communication) although isolated sightings of swordtails have been reported from waters in N.S.W. Available data suggest that the major zoogeographic factor limiting the Australian distributions of these aquarium species is that of minimum seasonal water temperature, so a northwards spread of such species can be expected, regardless of the introduction of more stringent controls on the importation and maintenance of aquarium fish. The mobility of man renders classic geographic barriers obsolete. A 1977 survey of the freshwater fish fauna of the Moreton Bay region of south-eastern Queensland by R. J. McKay of the Queensland Museum found the swordtail, guppy, and platy to be present in 14, 4 and 3 catchments respectively.

**Ecological impact**

The introduction of exotic fish has been closely accompanied by other human activities adversely affecting the aquatic environment. Thus, the effects such fish have had on indigenous aquatic organisms are difficult to delineate, and there is divided opinion about the effect of introduced fish upon native fish. One school maintains that the effect has been minimal, habitat alteration, changes in hydrological regimes, pollution and so on, having a more disastrous effect than any competitive interaction between native and introduced species. This is certainly the position of Weatherley and Lake (1967) and Lake (1971). The other school maintains that introduced fish have had considerable impact (Whitley, 1959; Frankenberg, 1966; Morrissy, 1967; Jackson, 1975; Tilzey, 1976).
McDowall (1968) also adopted the latter stance for New Zealand.

There is no question, of course, that man's environmental modifications have had a deleterious effect upon native fish; they clearly have, and it is also true that some introduced fish have had at least a minimal effect upon native fish. However, there are at least indications that some introductions, such as trout, have displaced native fishes.

Frankenberg (1966) noted that on the Australian mainland trout have apparently replaced galaxiids in the mid-sections of certain streams, galaxiids persisting in only the upper reaches which are either inaccessible to or unsuitable for trout, and in lowland reaches where weed cover is more abundant. Some firmer evidence for this has recently been provided by Jackson (1975) and Tilzey (1976). Jackson, in a study of trout ecology in Victoria, surveyed several streams in detail. His results indicated that when trout were present, galaxiids were either absent or not abundant. Tilzey arrived at essentially the same conclusion for streams in the Kosciusko area in southern New South Wales and also observed that the invasion of one stream by rainbow trout caused the disappearance of galaxiids below a natural barrier to trout. Trout are carnivores, and Evans (1942), Butcher (1945) and Bishop and Tilzey (1978) have recorded galaxiids being ingested. Further, an examination of the stomach contents of a few brown trout caught in the 'new' (sic) Lake Pedder, Tasmania, in November 1977 revealed that *Galaxias pedderensis* dominated the contained food (Tilzey, unpublished). Although this galaxiid was abundant at the time of sampling, it is endemic to the lake and could thus possibly become an endangered species if unable to withstand predation by the newly introduced trout. Cadwallader (1977) cited Langtry's observations of 1948 that blackfish (*Gadopsis marmoratus*) formed a large part of the diet of trout in the Ovens River, Victoria. However, Jackson (1975) found marked differences in the habitat preferences of trout and blackfish in another Victorian stream, suggesting little direct competition between the two species. The paucity of information on interactions between trout and native fishes other than galaxiids renders any generalised statements defending or indicting trout speculative, to say the least.

There are some indications also that trout have had an effect upon freshwater fauna other than fish. From the distribution of the Tasmanian mountain shrimp (*Anaspides tasmaniae*), Williams (1974) drew the implication that it survived only in localities where trout predation is not unduly heavy, or which are inaccessible to trout.

The impact of other introduced fish is even less discernible, and it is doubtful if it will ever be otherwise. It seems unlikely that it will be possible to assess the full extent of the impact of exotics because too little is known of what the native fish fauna was like before it was supplemented by introduced species. Indeed, the impact of even 'recent' introductions has not yet been assessed. The impact of the European carp, which has exploded in population spread and abundance in south-eastern Australia in the past decade or so, remains unknown in terms of quantitative and well-documented evidence. There is some evidence, largely circumstantial at present, that this fish damages the environment by its habit of roiling; it sucks
Introduced fish

up water and material from the bottom, ejects it and selects edible material from the suspended matter. This behaviour causes the water to become turbid. Additionally, the carp disturbs the bases of submerged plants, eventually dislodging them, and its vigorous spawning habits further muddy the water and dislodge vegetation. However, many Australian rivers are by nature muddy. Carp are also becoming an important dietary item to golden perch (*Plectroplites ambiguus*) and Murray cod (*Macullochella peeli*) in the Murray-Darling system and may be of benefit to these species.

Thus far, definite, well-documented evidence of interference is lacking. Active research on the carp is presently underway in all three south-eastern mainland states; this may well provide the evidence. In the meantime the carp has been declared a noxious animal in Victoria with penalties for persons keeping, hatching, or conveying live specimens. Commercial exploitation of carp is being encouraged in Victoria, South Australia and New South Wales. Branson (1977) summarised events related to the introduction of carp to the U.S.A. and noted that populations had increased dramatically, usually at the expense of desirable fish species.

The impact of the mosquito fish (*Gambusia*), another relatively recent introduction, likewise remains uncertain, though for this fish too there is accumulating circumstantial evidence that the impact has been of considerable significance (Myers, 1965). The species is now widespread throughout Australia. It is hardy, able to tolerate salinities up to that of seawater and high water temperatures (Ahuja, 1964), voracious, and a live-bearing breeder. It would be surprising indeed if such a fish had not had an impact upon the Australian fish fauna.

R. J. McKay (personal communication) notes that the native fish most severely affected by introduced live-bearing fish are the surface-feeding genera such as *Melanotaenia*, *Pseudomugil*, *Craterocephalus* and *Retropinna*. He further observes that the more recently introduced swordtail exceeds *Gambusia* in voracity and finds all surface-feeding native fish to be rare or extinct where swordtails occur.

What of the future?

In a discussion of the problems related to introduced fish it is appropriate to bring into focus the wider and more general subject of the procedures that should be followed whenever a fish introduction is being contemplated. In the past the majority of introductions were made without public debate. This situation no longer holds true. For example, the proposed introduction of Nile perch (*Lates niloticus*) was the subject of real public debate, resulting in withdrawal of the proposal. However, there is still room for improvement in debate procedure.

The sort of procedure that should be mandatory before any fish is introduced into a country has been clearly spelt out recently by Courteney and Robins (1973). A short summary of this procedure is of sufficient interest to bear repetition here.

1 A rationale should be stated. Reasons for seeking an introduction should be clearly given and demonstrated.

2 Bearing in mind the reasons for the proposed introduction, a search of possible candidates should be made as well as a list of the most suitable, together with their favourable and unfavourable characteristics.
3 There should be a preliminary assessment of impact. If this assessment is to have meaning it should include the opinion of scientists not specifically involved in the proposal.

4 There should be adequate publicity and review: 'no importation of a species for release purpose is so urgent that its biological implications should not be severely reviewed by a broad panel of experts'.

5 A research program should be initiated by an independent organisation.

6 The resultant research report should be publicised and evaluated by a broad spectrum of interested parties.

7 Only if this last review is favourable should release be effected.

Hubbs (1977) also stated that 'a control mechanism must be available to prevent over-population from the final release'. Would that such a mechanism were available for the European carp, cane toad, etc.!

The most sensible position to adopt in this area is surely to acknowledge that introduced fish can serve useful ends, but to acknowledge too that attendant disadvantages may outweigh those ends. The guiding thought should be that good intentions are no excuse for mistakes.

Introduction of the grass carp (*Ctenopharyngodon idella*) is often suggested. This is said to be a useful fish in that it eats nuisance aquatic plants (many of which are exotics introduced via the aquarium trade), and this is the basis for its suggested introduction. However, there are some likely unfavourable side-effects (Courtenev and Robins, 1973), and many other problems to be considered also. Continuing efforts to introduce the Nile perch must also be viewed with concern unless it is clear that this species will not have any deleterious effect on the indigenous fauna or the local environment. The European carp serves as a striking example of an apparently benign introduction which, after maintaining a fixed distribution for a considerable time, suddenly expanded its boundaries because of the illegal introduction of a few specimens of a more vigorous strain. This demonstrated that if a decision to introduce a particular species is made after impact studies have been conducted, care must be taken to ensure that the genetic background of introduced stocks matches that of the study fish. There is an increasing awareness in freshwater fisheries management circles that within the one species different stocks often possess widely varying biological, physiological and behavioural characteristics.

Open minds need to be retained on these and other questions if Australia is not to become the ‘biological cesspool’ of introduced fish that the State of Florida is said now to have become (Courtenev and Robins, 1973), and perhaps Papua New Guinea is well on the way to becoming (Berra et al., 1975), with obvious dangers for Australia. On this point, a special note should be made of the presence in New Guinea of the introduced Mozambique mouth-brooder (*Sarotherodon mossambica*). It already infests parts of New Guinea (Glucksman et al., 1976) and reports exist of its presence in Australia. Hubbs (1968) described the detrimental effects of *Sarotherodon* (formerly *Tilapia*) introductions into North America. We should now regard New Guinea somewhat as an
example of what northern Australia could well become without due care and attention.

Finally, this subject should not be left without noting one legislative loophole that urgently requires blocking. It relates to the importation, breeding, and transport of aquarium fish already approved for entry. These matters seem not to be subject to the same controls as for non-aquarium fish, but, of course, frequently the only real difference between the two sorts is that one is captive, the other not. More control is clearly needed at federal level. Already, several species of aquarium fish have established feral populations in Queensland as discussed earlier. In any event, the aquarium fish trade in Australia is now a million dollar industry involving several million individual fish per year. At the latest available annual estimate (1975-76) 12,148,252 live fish were imported representing a value of $1,186,505. Including the supply of aquaria, auxiliary equipment, fish food and so on, a multi-million dollar enterprise is involved. Moreover, interest in ornamental fish is growing rapidly so that future values are likely to exceed considerably present ones (in 1963-64, the number of live fish imported to Australia was only 278,000 worth $21,000). The current world retail value of the ornamental fish trade exceeds $3,000,000,000 per annum (Conroy, 1975). Most fish involved come from tropical and subtropical countries, and Singapore, Hong Kong and Thailand are prominent sources for Australia. At present almost 700 species of aquarium fish are permitted to enter Australia.

Predictably, such a vast trade in fish is not without dangers, dangers which, moreover, are exacerbated by the less rigorous control accorded aquarium fish than is given non-ornamental fish. The principal dangers are the possibilities of undesirable aquarium species establishing viable feral populations, and the possibilities of adding to the number of fish diseases already present in Australia. The latter aspect is perhaps especially worrying because of obvious loopholes in Australian quarantine regulations.

It should be stressed that the problem lies at the federal level. For example, under the terms of the South Australian Fisheries Act 1917-62, authorities in South Australia are empowered to impose a quarantine ban on fish imported from abroad, but not from other states, and not on 'aquarium fish'! The importation of goldfish from overseas is prohibited in N.S.W. yet not in the Australian Capital Territory. Thus N.S.W. aquarists obtain goldfish from the latter source. Efforts by state governments to channel all aquarium fish imports through one central, quarantine clearing-house have been frustrated by federal officials. Control systems, such as they are, are largely voluntary on the part of dealers. Several fish diseases remain unrecorded from Australia, but already it appears that at least one has been introduced via aquarium fish which subsequently communicated it to certain native fish.

It should be remembered that recreational freshwater fishing in Australia is also a multi-million dollar 'industry'. Its worth certainly dwarfs that of the aquarium trade. An across-the-board survey commissioned by N.S.W. State Fisheries in 1977 found 30 per cent of New South Wales residents to fish at least once a year of which 13 per cent (4 per cent overall) fished exclusively in
fresh water. The estimated gross monetary turnover generated by caravan parks, marinas, tackle shops, etc., centring on a single recreational fishery, Lake Eucumbene in southern N.S.W., in 1975-76 exceeded the value of all the aquarium fish imported in that year (R. D. J. Tilzey, unpublished data). Such natural resources must not be jeopardised by inadequate quarantine regulations.

The possibility also exists, of course, of introducing diseases other than those of fish. We should remember that live imported fish bring with them imported water. It is the disposal of this water wherein lies danger. A large number of animal and human pathogens are transmitted by water, and a number of disease vectors are aquatic.

Australia has suffered greatly from introduced pests such as rabbits, pigs, lantana, prickly pear, and many others. The list is a long one. Many are controlled only at enormous expense. Strict quarantine measures are employed to prevent further introductions of many undesirable terrestrial organisms but a glance at the statistics of the aquarium industry shows that, in actual fact, less than one per cent of all vertebrate animals entering Australia are subject to adequate quarantine supervision.

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------ 1977. Possible rationale and protocol


24 Introduced amphibians: the cane toad

Michael J. Tyler

In now permitting the liberation of *Bufo* in the cane areas of Queensland, we may be confident that no element of casualness has been involved in the decision, but that every biological aspect known to modern science has been carefully considered. We can only hope that the anticipation of his sponsors, that *Bufo* will prove a useful predator against harmful pests and that he will develop no bad habits, will be fully realised.

(Editorial, *The Australian Sugar Journal*, 10 October 1936)

Introduction
Since the turn of the century several different species of amphibians have been introduced into Australia. These include at least two species of European newts of the genus *Triturus*, the salamander *Ambystoma mexicanum*, the African clawed frog *Xenopus laevis*, and the cane toad *Bufo marinus*. Of these the cane toad is the only amphibian deliberately released.

The cane toad was originally a neotropical species with an extensive geographic range from Brazil to Mexico. As a potential means of biological control it has been introduced into numerous countries on other continents, and it was imported into Queensland and released there in 1935. Recently the historical background to these introductions has been documented by Covacevich and Archer (1975), and by Tyler (1975, 1976).

Accordingly I do not record such data here.

In terms of the overall number of individuals the cane toad is now by far the most abundant anuran in Australia. It already occupies the entire east coast of Queensland, and extends into northeastern New South Wales (Fig. 32).

Unlike the situation with many other species of exotic animals there is periodically generated in the media public controversy about whether in Australia the species is a 'pest', and whether any actions to inhibit its dispersal or effect control are merited. This is a ridiculous state of affairs because it influences politicians and others able to take action, and so, within the ambit of interest of the present volume, I will draw particular attention to evidence that is relevant to resolving such a controversy. In addition I propose to indicate lines of research likely to aid any future control or management decisions.

Morphological and physiological attributes
The exceptionally warty skin, enlarged parotoid glands and characteristic resting stance of elevating the body at 45° to the ground (Plate XIV) are a combination of external features readily distinguishing the cane toad from native species. In their survey of the species in Queensland Covacevich and Archer (1975) attributed many central Australian records (received in response to their
postal circulars) to be a native species then named *Cyclorana australis*; these Queensland individuals have since been shown by Tyler and Martin (1976) to represent the broader-bodied *C. novae-hollandiae*.

The upper limit of size varies with population densities, duration of colonisation of the districts and possibly other factors as well. The maximal sizes attained are in the vicinity of 230 mm snout to vent length, and the weight 1-25 kg. However, these outstanding individuals are fortunately rare.

Physiologically the species is exceptionally labile. Its critical thermal minimum was examined by Johnson (1972) working in Australia, and was reported to range from 5°C to 7.5°C with a mean of 7.1°C. In Guatemala Stuart (1951) recorded a mean of 15°C, but following the release of toads at Perth Airport (discussed below) toads were active at air temperatures of 8°C. Similarly in Florida the introduced toads survive in areas where there are several nights when temperatures as low as 0°C are recorded. It would clearly be hazardous to predict survival capabilities in Australia exclusively on the basis of minimal temperatures likely to be experienced at particular localities.

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**Fig. 32 Geographic range of Bufo marinus in Australia. (After Covacevich and Archer, 1975.)**
According to Brattstrom (1970) upper and lower limits of temperature tolerance, and the extent of the range between these extremes, greatly influence geographic dispersal potential of native species. It is therefore relevant that the upper tolerance limits of the cane toad are particularly high, separate investigators placing the physiological endpoint at more than 42°C.

Tolerance to dehydration is similarly remarkably effective, and comparable to Eyrean arid adapted native species (Warburg, 1966).

Tadpoles can tolerate high water temperatures and in Florida Krakauer (1968) places the critical thermal maximum at slightly above 43°C.

**Biological attributes**

There are relatively few publications providing any significant details of the biology of *B. marinus* in Australia. Apart from a short account by Straughan (1966), most publications are exceptionally brief. Hence to obtain basic details
it becomes necessary to extrapolate from the relative wealth of published data on populations of the species occurring overseas.

Within its natural range the cane toad breeds in ephemeral waters and noteworthy features of its reproductive strategy are:

(a) The existence throughout the year of individuals in breeding condition.

(b) Deposition of very high numbers of eggs on each occasion of spawning.

(c) An exceptionally abbreviated metamorphic span.

(d) Reproduction by any individual occurring on more than one occasion in each year.

The number of ova laid on each occasion is reported to range from 8,000 to 35,000 per individual; the larval life span may be as brief as 17 days, although the upper limit is stated to be six months. The tadpoles commonly aggregate in vast numbers and may swim in gigantic schools.

The age structure of populations of adults is unknown. In captivity a *Bufo* species has been credited with an age of forty years, and there are two records of *B. marinus* alive after 15 years in captivity.

The formula for successful colonisation

The dramatic population explosion of the cane toad in Australia, Papua New Guinea, Fiji and elsewhere, is attributable to a number of factors including the absence of any comparable terrestrial amphibian. *Bufo marinus* filled a largely unoccupied niche, and the attributes that in the Neotropical region ensured survival in areas of unreliable rainfall, enabled it to exploit the permanent freshwater dams, lakes and lagoons within the area that it was permitted to colonise. Furthermore the initial practice of the sugar industry of transferring thousands of juveniles to new areas limited the degree of intraspecific competition for food and shelter resources amongst these juveniles.

Of the four attributes listed above no Australian species lays more than one-half of the minimum number of eggs cited for the cane toad, and comparable abbreviation of the free larval life span is confined to species largely, if not entirely, restricted to deserts.

Added to these factors is the absence of any significant predators, and the fact that the nature of the toad skin secretions produces the rapid death of native vertebrates that might play a minor role in culling the toad population.

Consequences of dispersal by man

The use of adult toads for dissection purposes in schools and universities has led to thousands of toads being transported to areas where they do not otherwise occur. In Australia, these have led directly to three releases:

(a) Darwin. In June 1974, 24 toads were released accidentally in the suburb of Nightcliff. The release occurred at the height of the dry season, with daily shade temperatures averaging 33°C. The majority of the specimens were recaptured within the immediate vicinity of the site of release. Several of the remainder entered totally dry, concrete stormwater drains and travelled approximately 500 m to a freshwater culvert. Tyler (1975) documents the way in which one toad was accidentally transported a further 3 km from that culvert.
Had the release at Darwin occurred during the monsoon season there is a high probability that the species would have become established there.

(b) Perth. In July 1974, 50 to 55 toads escaped at an airline freight office located at Perth Airport. The majority of specimens were recaptured within 100 m of the release site, but the task of recapture was exceptionally difficult because of the variety of refuges provided in a busy and congested terminus.

(c) Sydney. On 24 December 1976, 40 to 50 toads were deliberately released by someone who broke into the Physiology Department at the University of Sydney. Whereas some individuals were recovered on the campus others were taken several kilometres away.

Effect of the cane toad upon invertebrates

There is a vast body of literature upon the diet of the cane toad collectively producing the conclusion that this species is an indiscriminate feeder (e.g. Dexter, 1932; Hinckley, 1962; Alexander, 1965; Pippet, 1975; Bailey, 1976; Zug, Lindgren and Pippet, 1976). Accordingly, the toads will feed upon economically beneficial insects if they come into contact with them. For example in New Britain Bailey (1976) found that in cacao plantations toads ate ants such as Oecophylla smaragdina, abundant in such situations. O. smaragdina is considered to control several cacao insect pests and is itself worthy of management according to Room (1973).

All of the dietary studies listed above have been undertaken on samples of toads collected in areas where it was well established and already abundant. It would be valuable to examine diet at the periphery of the toads' range.

Effect of the cane toad upon other vertebrates

Until recently evidence of the effect of the cane toad upon other vertebrates was confined to accounts of the action of toad parotoid venom upon domestic animals. Many of these reports were poorly documented and were commonly subject to reasonable doubt. Assessments by veterinarians of the severity of the action of the toxins upon cats and dogs are provided by Knowles (1964) in South Florida, and by Otani, Palumbo and Read (1969) in Hawaii.

Covacevich and Archer (1975) were the first authors to collate, and critically assess data upon the effect of the cane toad on the Australian reptile, bird and mammalian fauna. From their exhaustive survey they were able to establish that deaths involving twelve native vertebrates were attributable to ingestion or 'mouthing' (taking into the mouth but later dropping) *Bufo marinus*. The fatalities involved two species of lizards, six snakes, two birds including the kookaburra *Dacelo gigas* and two native mammals. The mammals included the western native cat, *Dasyurus geoffroii*, a captive individual being observed to bite a toad during an accidental encounter, and to die thirty minutes later following tetanic contractions and convulsions. The authors argued that in the wild *Dasyurus* would be equally at risk. Two additional reports of fatalities involving *Dasyurus* sp. (Lear, 1970; Pippet, 1975) provide confirmation.

Of the reptilian predators of *B. marinus*, populations of three Australian frog-eating snake species and a fourth in New Guinea are reported to be declining following the arrival at certain localities of the toad (Covacevich, 1974; Pippet, 1975; Covacevich and Archer, 1975).
Options for control

It is difficult to envisage a comparable confusing anomaly to that presented by attitudes to the cane toad. Here we have a demonstrably harmful animal being transported around the continent, for use by secondary and tertiary studies in the natural sciences, whilst nothing is being done to establish control techniques to cope with any population that may be accidentally established in a new area. Legislation has an effective role to play principally in minimising the chances of establishment in new areas, but it also provides more time to develop control methods before the range is grossly extended by natural dispersal. In the absence of legislation the cane toad was deliberately released in 1967 80 km south of Darwin (correspondent in *The Australian*, 20 July 1974); fortunately it apparently did not become established at the release site.

Control measures do not seem ever to have been seriously contemplated. Hence the question of selective biological control cannot be dismissed. There are several features of niche selection, feeding habits, adult anatomy, larval anatomy, and larval behaviour that are unique to this species. Populations of toads with a high incidence of emaciated individuals dying in a state of extreme muscular atrophy have been observed at Negros Island, Philippines by Alcala (1957), at a locality near Mackay, Queensland (J. Covacevich, pers. comm.) and by me at Kerevat, New Britain. Pippet (1975) attributes such fatalities to population increases exceeding the available food resources, but the possibility of the cause being attributable to a viral disease cannot be excluded, and is certainly worthy of investigation.

There are various avenues of research that could be investigated as part of a control feasibility study. Unfortunately no governmental agency with the necessary resources has yet directed its attention to even the most elementary of them.

Acknowledgments

I am indebted to Mr B. Stankovich-Janusch for providing the plate and Miss R. Altmann for the figure.

References


Introduction
Waterfowl have a symbolic importance in an arid continent. Yet despite this, their economic worth to hunters, and occasional significance as crop pests, coherent national management policies are lacking. A number of reasons explains this. The principal one is the paucity of information on waterfowl distribution, population dynamics, specific habitat requirements, and the way waterfowl find, use and share resources of food and space. There is also a dearth of information on the influence of man on waterfowl habitats and on populations through hunting. In addition—and in part because of this lack of information—it is difficult to derive a consensus on possible management requirements. Opinions held on this subject are often divergent, emotional, or biased. There is also the problem of administering and controlling a mobile resource.

The present chapter examines a number of problems and aspects of waterfowl management in Australia, and indicates areas believed to merit investigation. Certain parts of this subject matter have been commented on earlier (Braithwaite, 1975). These earlier comments are now supported or revised on the basis of new information.

The species and their habitats
There are nineteen species of Australian waterfowl (Fig. 33). Each has a different set of biological characteristics; differences occur in distribution, patterns of movement, times of breeding and moulting, and in habitat (Frith, 1967; Braithwaite, 1975). For all but grazing species, ecological succession—dependent on cycles of partial or entire inundation and drying of wetlands or on salinity changes in wetlands—is a principal factor in the development of suitable habitats. The distribution of many species extends into regions where the occurrence of suitable habitat is governed by widely varying climatic conditions; this presumably confers some safety for continued survival in that droughts tend to be localised phenomena (Gibbs, 1967).

The effect of man on waterfowl habitats
There are two main areas of concern in waterfowl management. The first is the extent to which man influences waterfowl habitats, the second, the effect of exploitation by hunters. The influence on habitats is considered in this section, exploitation in the following one. Of course, the identification of these areas is by no means new. The game management philosophy developed by Leopold (1933) clearly recognised that the harvesting of renewable game resources could continue only if two conditions were met: habitats were preserved, and hunting pressure could be regulated. The present situation with regard to habitats in various regions of Australia is discussed below.
Northern Australia. In northern Australia, man's direct impact on waterfowl habitats appears generally beneficial because of water storage and control activities (Lavery, 1975). The storage of water, in particular, provides refuges during the dry season that would otherwise be unavailable. However, Frith

Fig. 33 Distribution of waterfowl and relative abundance of species according to wet or dry conditions (after Frith, 1967).
(1973) has pointed out that in the Northern Territory the presence of the introduced water-buffalo is a cause for concern. This species has extensively damaged much coastal wetland, though local relief has been brought about by shooting.

**Coastal and southern habitats.** Habitats near densely populated coastal and southern regions are endangered by agricultural and urban expansion. The loss of habitat in these regions has been extensive and is serious, and especially so in particular regions (Riggert, 1966; Goodrick, 1970). The total loss on a national scale remains unknown. The extent to which coastal paperbark (*Melaleuca*) swamps of marginal use to waterfowl have been cleared since settlement is also unknown. However, these former swamps now provide freshwater meadows of high value to waterfowl and this may partly offset losses of coastal wetland.

**The arid inland.** No comprehensive survey exists of the distribution and extent of the typically ephemeral wetland habitats of the arid inland. The best assessment available is given in Fig. 34. This excludes, however, the numerous salt lakes shown in topographic maps and of known importance to waterfowl when full. The frequency of flooding in the inland rivers is highly variable, and ranges, for example, from once each six to eight years in the Georgina and Diamantina Rivers to once per year in the Cooper Creek or its tributaries (see Davidson, 1969).

Many wetlands of the arid interior provide food and water for stock. Pastoralists enhance this by retaining water behind dams and levees in minor catchments and across the beds of intermittent streams. The large number of waterfowl using such restrained water is an indication of the positive contribution man has made to waterfowl management.

![Fig. 34 Wetland habitats of waterfowl illustrating the floodplains and those portions of rivers providing an important habitat. The figure is based on field knowledge of habitat requirements and examination of topographic details on 1:250,000 maps.](attachment:image)
as a by-product of agricultural practices. Certainly for some species, e.g. grey teal, wood duck, black duck and plumed tree duck, the likely size of this contribution is probably reflected by the number of dams built (Fig. 35).

**Major inland rivers and their floodplains.** Possibly the most controversial problem of wetland preservation and management relates to the effect of large water storages on the major inland rivers of eastern Australia. One important impact has been to alter river flow and flood patterns (e.g. Butler *et al.*, 1973), and thus change the nature of floodplain 'ecosystems' (e.g. Robinson, 1967; see also Chapter 18). Concern about the possible effects of this on waterfowl habitats is repeatedly expressed (Frith, 1967, 1973, 1974), but exact information is difficult to obtain. Summarising available evidence, however, there seems little doubt that on the inland floodplains man has generally had a beneficial effect on waterfowl habitats. He has minimised losses of water which otherwise would not have provided habitat. Further, there is no evidence that large water storages *per se* have been detrimental to the occurrence of floodwater breeding habitats, and storages do at least provide marginally useful habitats. In times of water surplus (but not flood), minor control structures distribute water on floodplains, and thus provide habitats similar to those produced by natural flooding. Finally, in dry to drought times, lakes and swamps used as regulating facilities provide refuges that would also not have been available.

**Exploitation by hunters**
The exploitation of any game resource in a modern community has not only biological but also sociological problems. The fundamental sociological questions are: should exploitation be permitted, and if so, why? Although many people are antipathetic to hunting, the answer to the first question is usually in the affirmative, and only the second requires

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**Fig. 35 Numbers of stock dams and reservoirs according to grid references for 1:1,000,000 international world map.** The data were compiled from features on 1:250,000 maps of Australia. Shaded portions represent areas for which details of dams and reservoirs are not available. Since mapping was based on aerial photography dating from 1948, and only the larger dams and reservoirs were mapped, numbers may be underestimates by as much as 60 to 70 per cent.
Waterfowl resources and their management

consideration. The answer is possibly because society recognises the need for outlets to man's innate aggressiveness, itself a reflection of his evolution from the role of hunter and killer (Ardrey, 1976). The alternative viewpoint—man's desire to hunt is not innate but an unnecessary and expensive brutality—is rather unpalatable. Whatever the case, society undoubtedly condones hunting: sporting goods are aggressively advertised, favourable parallels are drawn between a man's excellence as a hunter and as a soldier, and hunting is made a socially acceptable recreation, as evidenced by the number of Heads of State and members of Royalty who participate. Moreover, in Australia, the political system gives substantial voting rights to farming communities and these as a rule jealously guard privileges to use firearms against pests. Thus, the elimination of hunting by social reform seems unlikely to occur in the foreseeable future. Government authorities have needed, therefore, to assume the responsibility of seeing that exploitation is restricted to the limits the resource can support. The hunter, in turn, contributes towards the cost of the necessary research and administration through licence fees and taxes.

Effective regulation of hunting pressure depends not only on wise control, of course. Effective enforcement is also needed. But there is a number of practical difficulties involved in enforcing game laws in Australia: hunters are highly mobile and travel far (Braithwaite and Norman, 1974, 1976, 1977); probably most hunting occurs on private properties and these are often widely dispersed and difficult of access; and the size of territories for which single game enforcement officers are responsible may be extremely large (perhaps 100,000 to 200,000+ km²). Thus, to achieve success in the observance of game regulations, hunter education is essential, and regulations have to appear rational and need to be formulated not only with reference to biological data, but to hunter psychology too.

The hunters themselves can usually be assigned to one of three wide social categories: Australian, southern European, or 'traditional'. For the Australian hunter, often a rural resident, hunting is primarily a social occasion at which skill is demonstrated by the number of ducks shot. Usually little concern is evinced for conservation: if ducks are scarce, they are elsewhere or have not been able to breed in the absence of floods. Southern European hunters are mainly of Italian or Greek descent, and all birds shot are usually eaten. Their philosophy to wildlife is consistent with an upbringing in countries where there has been little concern for migratory bird conservation: annual migrations of birds seemingly provided an inexhaustible resource. In Australia this group finds the wide variation in abundance of game and the need for conservation difficult to understand: if ducks are scarce, the wrong place has been selected for hunting. 'Traditional' hunters are typically business or professional people or migrants from northern Europe or North America. They emphasise the aesthetics of hunting, and thought and effort go into assembling and preparing equipment, selecting hunting sites, building hides, and training and using dogs for game retrieval. Game laws, if not strictly adhered to, are kept well in mind. Conservation is perennially discussed. There is usually a strong social discipline. This group, above all, sets the example that
Aquatic fauna

all hunters must follow if effective control over hunting pressure is ever to be achieved.

With regard to the actual assessment of the effects of hunting, knowledge of three population characteristics is mandatory: the size of the population being hunted, its reproductive rate, and its mortality rate (natural or otherwise). None of these parameters is easily studied. A brief examination of some data on the population dynamics of Australian waterfowl indicates, by way of example, the complexity of even simple calculations of mortality rates. The analysis by Caughley (1977) of data obtained by Frith (1963) may be considered. The analysis showed that on average approximately 34 per cent of the population of grey teal died each year. But the usefulness of such a value is highly questionable for analysis was based on the assumptions that marked (banded) birds always move rapidly through the species population, and that all birds, marked and unmarked, have the same average death risk. Both assumptions seem most unlikely to be true.

Research requirements for management
The nature of research necessary for effective management of waterfowl falls naturally into two categories, preliminary and general. A major function of preliminary research is the development of rational guidelines for government policies in this area, and the formulation of plans for general research. The core question in the development of guidelines is whether hunting is significantly reducing the value of waterfowl populations of all the different species shot to an unacceptable level as a resource. The principal preliminary research requirement is thus obvious: information is needed on the size and distribution of waterfowl populations, and the way these change with changes in environmental conditions. Such information is best gained by survey techniques, despite the difficulties of these and the criticisms that may be levelled at them. Light aircraft have an essential role to play in this regard, but the value of assessments by non-airborne but experienced and competent biologists should also not be underestimated.

Fully effective management, of course, needs rather more information than can be provided by surveys. The provision of this more detailed information constitutes the rationale for general research. Techniques of use here include colour marking of birds with patagial tags. Using these, a good deal of information can be gained on waterfowl movement and its relationship to food availability. Individual marking can also give information on social organisation and other behavioural characteristics, on the way food, space and nesting sites are used, shared or responded to, on the relationship between age and breeding, and on the availability and nutritional value of food resources. Direct examination of birds shot by hunters is another technique of value. This provides routine data on the species, age, and sex of ducks shot, as well as information on hunting success. Data on reproductive and moulting condition may also be provided. Data from this source indicate that at least for the grey teal there is a significant inverse relationship between reproductive condition and population density (Fig. 36); that is, at high population densities there is a suppression of reproductive potential. The implications for management are clear. Fewer data on
Examination of hunters' bags, it may be added, has values other than that of producing data of scientific interest. It provides the opportunity for contact with hunters and thus of educating them. Liaison and the exchange of information between various government departments is also fostered.

Finally, it is evident that European man has brought substantial changes to the distribution, extent and permanency of habitats utilised by waterfowl in Australia. Fortunately the changes are of substantial benefit and seem likely to assure our waterfowl of habitat into the future. If problems for the conservation of waterfowl exist, then their solution will rest in the hands of the hunters and those who set the regulations on hunting.

Acknowledgments
In producing this chapter I have been considerably helped by information or expressions of opinion from a number of colleagues and government departments. I acknowledge in particular help from the National Parks and Wildlife Service (Australia, South Australia, and N.S.W.), Fisheries and Wildlife Division, Ministry for Conservation (Victoria), Conservation and Agriculture Branch (Department of the Capital Territory, A.C.T.), Department of Environment, Housing and Community Development (Canberra), Division of Wildlife Research (CSIRO), Messrs F. H. J. Crome, M. Clayton, F. D. Knight and B. K. Brown, and many hunters (particularly from the Victorian Field and Game Association). Special thanks go to Dr H. J. Frith for encouragement and the provision of facilities.

Fig. 36 The relationship between reproductive condition (testes weight) of non-moulting adult male grey teal and mean hunter success, an assumed measure of duck density for all species present at hunting sites in south-eastern Australia on the annual opening day of hunting for the years 1972-77 inclusive. Data from Braithwaite and Norman (1974, 1976, 1977 and unpublished).
References


Ostracoda and water resources: diagnosis and prognosis

K. G. McKenzie

Introduction
Ostracoda are minute bivalved crustaceans which occur in every variety of continental aquatic environment. In Australia, they were first studied by King (1855), but the foundations for systematic research were laid by Sars (1889, 1894, 1896) who worked with cultures raised from dried muds sent to his Norwegian laboratory by several antipodean correspondents. Henry (1923) published a useful review and Chapman (1967) provided the first key to the Australian fauna, including reference to 21 genera. All known Australian taxa are listed by De Deckker and Jones (1978).

In this chapter, comments are made upon the endemicity of the Australian ostracode fauna, its geographical distribution, and the nature of its occurrence in various major sorts of aquatic environment. Fossil assemblages are also briefly commented upon. The aim of the chapter is to demonstrate the utility of ostracode studies in water resource investigations, and to provide an introductory basis for such studies.

Endemicity of the fauna and its distribution
Over 30 genera are now known to occur in Australian continental aquatic environments and of these about one-third are endemic to the Australasian region. A few other genera, such as Mesocypris and Gomphocythere, are restricted to southern continents, but the remainder are widely distributed (Table 19).

Sampling in Australia has been almost wholly restricted to the coastal and southern parts of the continent and large gaps remain, particularly in central Australia. It is premature, therefore, to generalise on the provinciality of individual taxa. Some evidence exists, nevertheless, to indicate that a few genera are restricted to northern Australia and that several species differences occur between south-western and south-eastern faunas.

Ecology
Ostracoda are known to be sensitive environmental indicators and this sensitivity is usually documented with respect to depth, temperature, salinity and substrate. McKenzie (1971a) was able to demonstrate a relationship between salinity and carapace length for species restricted to protected environments on Aldabra, and in Australia research rapidly indicated that some genera are halophilic while others are usually restricted to fresh water. The known salinity ranges of several typical Australian genera are given in Table 20, which shows that halophilic genera can tolerate salinities up to about five times the concentration of seawater. Parenthetically, it does not appear to matter whether these saline environments are coastally located or inland—the same Ostracoda characterise them.
Table 19  Endemism and distribution of Australasian continental ostracode genera

<table>
<thead>
<tr>
<th>Genus</th>
<th>Endemic</th>
<th>Southern continents</th>
<th>Cosmopolitan</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mytilocypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Australocypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Trigonocypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>'Isocypris'</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Platocypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Herpetocypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Stenocypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Ilyodromus</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Candonocypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>'Chlamydotheca'</td>
<td>?+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Heterocypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Hemicypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Cyprinotus</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Cypretta</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>'Cypridopsis'</td>
<td>-</td>
<td>-</td>
<td>?+</td>
</tr>
<tr>
<td>Diacypris</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Newnhamia</td>
<td>+</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Mesocypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>'Cyclocypris'</td>
<td>-</td>
<td>-</td>
<td>?+</td>
</tr>
<tr>
<td>Cypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Candona</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Candonopsis</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Ilyocypris</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Darwinula</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Limnocythere</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>'Gomphocythere'</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>'Metacypris'</td>
<td>-</td>
<td>-</td>
<td>?+</td>
</tr>
<tr>
<td>Cytheroma</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Microcytherura</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
<tr>
<td>Cyprideis</td>
<td>-</td>
<td>-</td>
<td>+</td>
</tr>
</tbody>
</table>

The broad division of Australian Ostracoda into halophilic and halophobic taxa is supported by analyses of the faunas of major habitat types.

**Rivers and billabongs.** Ostracodes are rarely important in river communities according to most authorities, but this may be a result of inadequate sampling (too coarse a net, poor benthic collections) rather than a statement of fact. Thus, in Australia, very few workers have attempted to make ostracode collections from rivers and there is only one satisfactory published record (involving about 80 samples from Farmers Creek and Cox's River, near Lismore, N.S.W.; Jolly and Chapman, 1966). Jolly and Chapman noted three genera, *Candonocypris, Ilyodromus* and *Cypretta*, as 'occasionally found in the marginal vegetation at station CD on Cox's River'.

Other records relate to spot collections and personal observations. The earliest of these is by Brady (1886) who described species in the genera *Ilyodromus, Cypretta, Newnhamia, Candona*
and *Candonopsis* from ‘Condong District, Tweed River, New South Wales’, and a *Heterocypris* species from ‘brackish pools in a dry creek at Adelaide’. Of these records, the generic status of *Candonina lutea* may be doubtful (see Henry, 1923: 283-4). McKenzie (1966) identified the genera *Ilyocypris*, *Heterocypris*, *Hemicypris*, ‘Isocypris’, *Newnhamia* and *Cypretta* in collections from small intermittent creeks and waterholes in creek beds in the Kimberleys, Western Australia, and *Newnhamia* from Kings Creek, Kings Canyon, Northern Territory. Chapman (1966) described a species of *Mytilocypris* from Oldfield River, Western Australia. McKenzie (1968) described a *Paracypris* species from a small, scarcely flowing, brackish creek, near Inverloch, Victoria. In addition, *Newnhamia* has been collected from creeks around Canberra (P. J. Jones, personal communication), and *Herpetocypris* has been noted in a creek in the hills behind Adelaide (McKenzie, unpublished).

Hitherto unpublished generic records from rivers in Western Australia comprise *Cyprinotus* and *Mytilocypris* from Murchison River, *Platocypris*, *Mytilocypris* and *Diacypris* from the Phillip River, west of Ravensthorpe, *Diacypris* from a pool in the river-bed east of Ravensthorpe, and *Cyprinotus*, *Diacypris* and *Mytilocypris* from a pool in the Juddacuttup River, east of Ravenshorpe. The water in every case was brackish.

Ostracoda are common around the margins of billabongs associated with rivers but there is a dearth of published records. McKenzie (1966) identified juveniles of *Cyprinotus*, *Hemicypris* and *Cypretta* and adults of *Limnocythere* from billabongs in the Kimberleys, Western Australia. In addition, *Heterocypris*, *Cypretta*, *Newnhamia*, *Ilyodromus*, *Candonocypris* and *Ilyocypris* have been collected thus far from billabongs associated with the Lachlan, Murrumbidgee and Murray Rivers (see Chapter 34), and Shiel (1976) noted nine species in eight genera in a billabong associated with the Goulburn River, Victoria (*Cypretta*, *Diacypris*, *Paracypris*, *Stenocypris*, *Ilyocypris*, *Newnhamia*, *Potamocypris*, *Ilyodromus*).

### Freshwater ponds, temporary pools, reservoirs and lakes

The ostracodes of such habitats in central Australia are

<table>
<thead>
<tr>
<th>Genus</th>
<th>Salinity range (°/00)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Platocypris</td>
<td>9 - 180</td>
</tr>
<tr>
<td>Diacypris</td>
<td>11 - 175</td>
</tr>
<tr>
<td>Australocypris</td>
<td>4 - 132</td>
</tr>
<tr>
<td>Mytilocypris</td>
<td>&lt;1 - 36</td>
</tr>
<tr>
<td>Cypretta</td>
<td>&lt;1 - 4</td>
</tr>
<tr>
<td>Newnhamia</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Candonocypris</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Ilyodromus</td>
<td>&lt;1</td>
</tr>
<tr>
<td>Heterocypris</td>
<td>&lt;1</td>
</tr>
</tbody>
</table>
unknown. In Western Australia, however, collections have been made in the Kimberleys and in the south-west as far inland as Balladonia. In the eastern states, the first descriptions were by King (1855), and subsequent work, mainly by Sars and Brady, was reviewed by Henry (1923) who incorporated her own data from N.S.W. Sars (1889) cultured several genera from Gracemere Lagoon, Queensland. Timms (1970a,b), in his two surveys of reservoirs and freshwater lakes in New South Wales, recorded six ostracode genera (*Candonocypris*, *Cypretta*, *Eucypris*, *Ilyodromus*, *Cypridopsis* and *Newnhamia*). *Newnhamia* was the most commonly found. Recently, De Deckker (1976) described a new large genus, *Trigonocypris*, from a freshwater lagoon in Queensland. Hussainy (1969a,b) recorded the presence of several taxa from a freshwater lake in the western district of Victoria. McKenzie (1971b) figured an atypical *Newnhamia* from Bridport, Tasmania, and in 1966 collected *Herpetocypris*, *Ilyodromus*, *Darwinula* and ‘*Metacypris*’ from periglacial tarns and mountain ponds in south-western Tasmania. At least 13 species (in 9 genera) were recorded by Morton and Bayly (1977) from temporary freshwater pools in Victoria.

Summarising the results of this work it seems that the following genera are widespread in freshwater ponds, lakes and similar environments: *Ilyocypris*, *Newnhamia*, *Cypretta*, *Diacypris*, *Candonocypris*, *Ilyodromus*, *Herpetocypris*, ‘*Cypridopsis*’, ‘*Chlamydotheca*’, ‘*Isocypris*’ and *Gomphocytthere* particularly in southern Australia, while a few genera, such as *Hemicypris* and *Stenocypris*, occur only in tropical northern parts.

**Estuaries and open coastal lagoons.** Estuarine wetlands are highly productive natural systems (Collett and Hutchings, 1977) and it is regrettable that in Australia their microfaunas are little known. This comment is particularly apposite for the Ostracoda, which are not only very diverse in estuaries but also highly efficient scavengers, and consequently play an important role in estuarine ecological cycles. McKenzie (1969) listed over 30 species from Oyster Harbour, near Albany, Western Australia, an embayment formed by the drowned lower valleys of the King and Kalgan Rivers. In 1966, *Callistocythere* and *Microcytherura* were collected from a tributary creek at the head of Joe Page Bay, Bathurst Harbour, south-western Tasmania (McKenzie, unpublished).

The most typical estuarine ostracode, in Australia as elsewhere, is the genus *Cyprideis*. This was first recorded from Australia in Oyster Harbour (McKenzie, 1964). Subsequently, it has been identified in Western Australia in subfossil material from the Swan River banks near Fremantle, from about 1 m depth in a bore at Lake Cooloongup, and inshore on the marine floodplain of the Fitzroy River, near Derby, west Kimberleys. It has also been collected from surface waters in Lakes Cooloongup and Preston, Western Australia. The absence of records from eastern Australia is almost certainly due to a lack of sampling.

**Athalassic or non-marine saline environments.** The athalassic lakes of southern Australia have been of considerable interest to limnologists, and collections have been made from a large number of Victorian and South Australian localities (Bayly and Williams, 1966; De Deckker, 1974, 1975). The most
common genera in these environments are *Mytilocypris*, *Australocypris*, *Platycypris* and *Diacypris*.

Collections are also available from some athalassic lakes of south-western Australia. Many of these lakes are thought to be relicts of a former river system (Clarke, Prider and Teichert, 1971). Chemically, however, they are similar to other athalassic bodies of different origin such as the lakes in the volcanic western district of Victoria (Williams and Buckney, 1976), and the typical genera remain *Mytilocypris*, *Australocypris*, *Platycypris* and *Diacypris*. In addition, *Cyprinotus* was collected from Wagin Lake, single specimens of *Cyprinotus* and a new genus with reticulated valves were found in a collection from Lake Gidgi, and the latter genus was also identified from a lake near Lort River. The typical *Diacypris* from these Western Australian lakes differs from south-eastern Australian representatives of the genus. Finally, Chapman (1966) described a species of *Australocypris* from a saline pond on West Wallabi Island, Houtmans Abrolhos, Western Australia, and there are several unpublished records of this genus from similar environments on Rottnest Island, Western Australia, where it is usually associated with *Mytilocypris* and *Diacypris*.

**Fossil assemblages**

The first description of an Australian fossil continental ostracode was by Chapman (1914). Little has since been published although, when they occur, ostracodes are abundant in continental sediments, and environmental interpretations, at least of Tertiary faunas, can be made with confidence. It is unfortunate, therefore, that so much material remains uninvestigated and so few observations published. The well-preserved Early Miocene assemblage from Cape Hillsborough, Queensland, is of particular note in this context. This assemblage includes abundant material of 7 or 8 species (5 of these represented by both sexes).

Some Late Miocene material from Lake Frome, South Australia, has been reported on briefly and is of interest because one species is ornamented with large tubercles. For continental and polyhaline ostracodes, nodding of this sort is usually associated with variation in environmental parameters such as salinity, \(\text{CaCO}_3\) concentration (Vesper, 1975), or temperature (Carbonnel, 1975). The living South African species, *Sclerocypris tuberculata*, from alkaline Lake Chrissie in the Transvaal, is an example of the phenomenon. In a recent examination of further South Australian Miocene assemblages, *Diacypris*, a new genus and *Mytilocypris* were identified in a fauna also characterised by abundant 'Chara' oogonia and fossil rootlets (McKenzie, unpublished). The organic environment again has a parallel in southern Africa since it resembles sediments collected from Etosha Pan, Namibia. Taken together, these records indicate that central Australia had a lacustrine environment generally similar to that of modern Transvaal lakes (Hutchinson, Pickford and Schuurman, 1932) during the Late Miocene. An equivalent modern Australian environment could be the lakes in the Coorong area.

In Queensland, the Miocene faunas of the Narrows Basin and other sequences were described by Beasley (1945). The genera include *Diacypris*, *Newnhamia*, *Mytilocypris* (or *Trigonocypris*) and five others which can no longer be identified.
From the Murray River Valley, N.S.W., several Plio-Pleistocene assemblages have been examined by McKenzie and Gill (1968). The genera included *Ilyocypris, Diacypris, Cypretta, Candonocypris* and 'Eucypris', and the depositional environment was interpreted as similar to the semi-permanent lagoons and billabongs which still occur in the area.

Other fossil records are fragmentary. Chapman (1936, 1937) described a *Mytilocypris* from the Victorian Mallee, and identified *Newnhamia* from Mt Elder, Western Australia, De Deckker (1976) identified *Trigonocypris* from the Old Cork Beds, central Queensland, and McKenzie and Hussainy (1968) recorded *Gomphocythere* from south-eastern Australia, near Myaring Bridge, Victoria.

**Prognosis**
The environmental interpretations advanced in the last section provide a satisfying rationale for ostracode studies. In the context of an arid continent for which the primary resource is water, however, the correlations between Ostracoda and the quality of surface or subsurface water resources have a more immediate utility. That such correlations exist is well-known, and has been exploited in research on the subsurface deposits of Saskatchewan, Canada (Delorme, 1971).

For Australia, the evidence adduced in this chapter makes it clear that saline or alkaline waters of poor quality are the environment for *Australocypris, Platycypris, Diacypris* and often *Mytilocypris*, whereas good quality waters are characterised by such genera as *Ilyocypris, Newnhamia, Cypretta, Candonocypris, Ilyodromus, Herpetocypris, 'Isocypris', 'Chlamydotheca' and Gomphocythere*. The typical polyhaline genus is *Cyprideis*. In the long-neglected study and classification of Australia’s water resources (e.g. Woolley and Williams, 1977), such correlations have obvious relevance.

**Acknowledgments**
Thanks are extended to many colleagues who have made material available for study and which has been commented on in this chapter. Material from Western Australia has been provided by P. J. Coleman, D. H. D. Edward, G. W. Kendrick, D. Merrilees and W. D. Williams, from billabongs of the Murray-Darling system by R. Shiel, and from South Australia by R. Callen and W. D. Williams. The research was supported in part by A.R.G.C. Grant No. D 76/15127.

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Ostracoda and water resources


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PART VI

Man-made lakes
Introduction
Lake Hume is the most important of several reservoirs on the Murray-Darling river system. This chapter briefly describes it, and comments on certain features of its management. Additional information, with particular regard to the downstream influence of Lake Hume, is given in Chapter 18.

Basic ecological knowledge of Lake Hume has largely been derived from a limnological survey of the River Murray undertaken by the consultant engineering firm of Gutteridge, Haskins and Davey and begun in 1973 (Gutteridge, Haskins and Davey, 1974). Instigated by the Cities Commission, and originally intended to span ten months, the study has continued under the aegis of the Albury-Wodonga Development Corporation. The fundamental objective of the survey was to assess and predict the impact on the River Murray and associated waters of expansion of Albury-Wodonga. A comprehensive summary account of the major findings of the first stage of the survey has been given by Croome et al. (1976), and information to 1977 has been reported recently by Walker and Hillman (1977). Detailed accounts of particular studies are in course of preparation. The survey has established bench-mark information on waters of the area, including Lake Hume. Much of the information given in this chapter is derived from data collected during the first two years of the study. High rainfalls were recorded in both years; rainfall at Bonegilla, near the shore of the reservoir, was 1,298 and 900 mm in 1974 and 1975 respectively, against an annual mean of 565 mm. The data collected were therefore those of a very wet period.

Background information
Lake Hume impounds the River Murray 10 km east of the rural growth centre of Albury-Wodonga. It was formed in 1936 by the construction of a dam immediately downstream of the convergence of the Murray and Mitta Mitta Rivers. Much of the area inundated was floodplain, lightly timbered with the river red-gum, *Eucalyptus camaldulensis*, and the remains of many of these trees still stand in the reservoir. In 1961, works raising the dam by some 10 m were completed, doubling the storage capacity to $3.07 \times 10^6$ Ml. The old town of Tallangatta (population 800) was inundated in the rise and the new town of the same name was located on the shore of the reservoir some 5 km from its original site.

The reservoir was formed to ensure supplies of water to towns and irrigation areas along the River Murray, and is operated by the River Murray Commission (River Murray Commission, 1965). Eventually water will be drawn from the reservoir to supply the Albury-Wodonga growth centre; at present, although Tallangatta and Bellbridge take their water
from the reservoir, both Albury and Wodonga draw water directly from the River Murray. Water is released to the river either through valves in the concrete wall of the dam or through the small power station adjacent to the dam wall on the New South Wales bank. During floods spillway gates can be opened to release surplus water at controlled rates of flow.

The reservoir is situated at a transition point along the profile of the River Murray. It separates the headwaters section, in which the river and its many tributaries are swift-flowing, from the billabong section, in which the topography is gently undulating and the river enters a maze of waterways, billabongs, and swamps (Burton, 1974). Lake Hume acts as a buffer between the two sections, storing water from winter run-off for summer irrigation releases. The headwaters section, the catchment, is constituted mostly of older Palaeozoic sediments with intrusive granitic masses. Elevations within the catchment range from 150 to 2,100 m above sea-level, and topography has been aptly described as rugged; permanent, fast-flowing streams in steep valleys, and a thick cover of natural forest characterise much of the catchment.

The reservoir is a focal point for people in Albury-Wodonga, and many also journey to it from Melbourne, three hours away by car. In addition to the towns of Tallangatta on the Mitta Mitta arm of the reservoir, and Bellbridge opposite the dam wall, there are caravan parks, camping grounds and holiday cottages to cater for holiday-makers and tourists. The reservoir itself is the centre of most holiday activities and is used for boating, fishing, swimming and numerous other water sports. Adjacent mountains and river valleys help make Lake Hume a popular recreation area.

Basic descriptive features

**Physico-chemical features.** Selected morphometric and hydrologic data for the reservoir are given in Table 21. Retention times for 1974 and 1975 were short, 0-3 and 0-4 years respectively. The water was of low salinity, with a conductivity range of 30-72 μmho cm⁻¹ at 25 °C, and the order of ionic dominance was Na > Mg ≃ Ca > K : HCO₃ > Cl > SO₄. The pH was usually 6-8-7-5, but occasionally rose as high as 8-5 at the surface during high photosynthetic activity. The maximum surface turbidity recorded near the dam was 14 NTU, but values up to 50

| Table 21. Some morphometric and hydrologic data for Lake Hume. Data from River Murray Commission (1965). |
|-------------------------------------------------|------------------|
| Elevation (m a.s.l.)                             | 193.8            |
| Volume (10⁶ Ml)                                 | 3.07             |
| Surface area (ha)                               | 20,250           |
| Length (km)                                     |                  |
| Murray arm                                      | 90               |
| Mitta Mitta arm                                 | 55               |
| Mean depth (m)                                  | 15.2             |
| Max. depth (m)                                  | 41.4             |
| Catchment area (km²)                            | 15,300           |
| Renewal time (yr)                               | range 0-3-3.7    |
|                                                | mean 0.75        |
NTU were recorded in the upper reaches of the reservoir. Nutrient concentrations were moderate; up to 60 and 170 µg 1\(^{-1}\) of total phosphorus and nitrate-nitrogen respectively near the dam wall, and 150 and 185 µg 1\(^{-1}\) respectively in the upper reaches.

The reservoir thermally stratified from December to April in both 1973-74 and 1974-75. Previous records available indicated that stratification was an annual occurrence, with anoxia and H\(_2\)S production occurring in the hypolimnion. In 1975, however, anoxic conditions persisted for a few days only, and H\(_2\)S was not detected for the first time in five years. The stratification was probably less severe in 1975 because of rapid exchange of water during the high-flow conditions.

Profiles typical of those found in the main body of the reservoir are shown in Fig. 37. The depth of the thermocline (19.5 m) is a reflection of the amount of turbulence and wind action on the exposed surface of the reservoir, and the turbidity of the hypolimnion illustrates the difference in water quality usually seen between the surface and bottom waters.

**Biological features.** Diatoms, particularly three species of *Melosira*, dominated the algal standing crop. Other algal contributors included *Anabaena*, *Cryp­tomonas*, *Trachelomonas* and *Sphaero­
cystis. An algal bloom of the species *Anacystis cyaneae* occurred in October 1973, and a heterotrich ciliate, possibly *Climacostomum*, was also present. In August 1965 the latter organism was said to be present in such large numbers as to colour the littoral water black (J. Rhodes, personal communication). Macrophytes were very sparse and contributed little to primary production in the reservoir. Equally insignificant were the diatoms and filamentous green algae growing on the dead river red-gums littering the shallower areas.

The major zooplankton species present were *Calamoecia ampulla*, *Boeckella triarticulata*, *Bosmina meridionalis*, *Daphnia carinata* and *Ceriodaphnia* sp. Other species occurring intermittently included the copepods *Microcyclops* and *Mesocyclops* and the rotifers *Polyarthra*, *Keratella* and *Conochilus*. The freshwater medusa, *Craspedacusta sowerbyi*, was seen in the early months of both years.

The most prevalent species of fish in the reservoir are those which have been introduced, the English perch, *Perca fluviatilis*, being the fish most commonly caught by anglers. Brown trout, *Salmo trutta* are taken occasionally. The European carp, *Cyprinus carpio*, has not been detected in the reservoir to December 1977, but has been caught (in March 1976) from waters immediately below the dam.

Much of the work done on Lake Hume was aimed towards a determination of its trophic status. The algal species present, their biomass, the rate of fixation of...
inorganic carbon (as measured by the C\textsuperscript{14} method), and the general biological activity in the reservoir, all indicated that it was of mesotrophic status. The indication was confirmed by the physico-chemical behaviour of the waters, their nutrient status, and by calculations of nutrient loadings on the system (Gutteridge, Haskins and Davey, 1974; Croome \textit{et al.}, 1976). Some indication of the vertical and seasonal pattern of phytoplanktonic production is given in Fig. 38 in which data on areal day rates of production at three stations on the reservoir are figured. It will be seen that the amount of production proceeding at depths more than 4 m below the surface is insignificant, and that maximum rates of production occur below ~0.5 to 1 m rather than at the surface. During the period of observation covered by Fig. 38, at two of the stations (A, B) there were two seasonal maxima in productivity (midsummer and late autumn), and at one station (C), a single maximum (late autumn/early winter). This latter maximum is interesting because the lower light intensities and temperatures of that time of year are often associated with low rates of phytoplankton production.

Some remarks on future management
As indicated, at some time in the future Albury-Wodonga will probably take water supplies directly from Lake Hume, rather than from the River Murray as at present. The water offtakes should be constructed to take advantage of the variations in water quality with depth. The highest quality water is usually at the surface, but, during summer, diurnal variations in temperature and pH and, more especially, the occurrence of algal blooms may make the surface waters less suitable than those at greater depth.

Dartmouth Reservoir has been constructed above Lake Hume on the Mitta Mitta River (see Chapter 17). Given the distance of the new reservoir from Lake Hume, and the volume and quality of the receiving water, it is probable that the water released from Dartmouth Reservoir will not affect adversely the water quality in Lake Hume. However, conditions in Lake Hume may be altered by the mode of operation of the two reservoirs. It should be possible in future, for instance, to use Lake Hume for flood control, lowering the reservoir secure in the knowledge that Dartmouth Reservoir contains sufficient water to satisfy the demands of the River Murray irrigation areas. At the same time water which previously spilled over the Hume Dam could be retained as additional storage. It is thus possible that the level of the reservoir will fluctuate more than in the past. While this might have important consequences in other situations, it will probably not interfere greatly with the present uses of Lake Hume. It will not mean a loss of amenity to recreational users of the lake, who have already adapted to large long-term changes in water level.

References


Introduction
This chapter describes salient features of North Pine Dam (N.P.D.), Queensland, a relatively shallow, subtropical reservoir storing water primarily for reticulation to Brisbane and with an extensively grazed catchment. With this combination of features, N.P.D. differs from other reservoirs discussed in this book, and, consequently, has a somewhat different set of management problems. A successful solution to these is of considerable importance for N.P.D. currently supplies some 15 per cent of Brisbane’s water needs. In addition to providing a brief description of N.P.D. as an example of events in a shallow, subtropical reservoir, this chapter also sketches the nature of changes that have taken place in N.P.D. since first impounded. Such changes have special predictive significance not only for future operations involved in the management of N.P.D. but also in relation to future water storage projects in this region and elsewhere.

Some background features
North Pine Dam is a recent impoundment on the North Pine River, 30 km north of Brisbane, and 15 km from the coast. Water storage began in December 1973 and full supply level was reached in 1977. The climate of the area is subtropical with mean summer and winter temperatures of 25° and 15°C. Average annual rainfall on the catchment is 1,168 mm, average annual pan evaporation about 1,400 mm. Approximately 27 per cent of rainfall forms runoff (yielding an average annual inflow of about $100 \times 10^6$ m$^3$), but rainfall and runoff pattern is erratic and variable: annual runoff has varied from $5 \times 10^6$ m$^3$ (1922-23) to $> 400 \times 10^6$ m$^3$ (1949-50, 1973-74), and in 1937-38 the highest monthly runoff for the year was in May although this is normally part of the dry season (Webber, 1955, 1973). Usually some 55 per cent of runoff occurs in January to March.

Catchment area is 347 km$^2$, with highlands of the D’Aguilar Range forming northern, western and southern borders. The soil is mostly a shallow, stony, lithosol on metamorphic rocks, some rich in iron and manganese. Some deep soils occur on igneous rocks and in alluvial valley deposits. The land is extensively grazed, with natural and improved pastures occupying over half the catchment and carrying some 6,000-13,000 cattle. About half the catchment has been cleared of timber and about one tenth improved by sowing or irrigation. Horticulture (mainly pineapples) occupies about 0.7 per cent of the catchment, and state forest and national parks close to one fifth. The main forest types are wet sclerophyll (Eucalyptus pilularis, Tristania conferta), dry sclerophyll (E. maculata, E. signata), and vine scrub in small patches usually in gullies and along creek banks. Approximately a thousand
Table 22  Morphometric data for North Pine Dam

<p>| | |</p>
<table>
<thead>
<tr>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Lake surface area</td>
<td>21-63 km²</td>
</tr>
<tr>
<td>Catchment surface area</td>
<td>347-00 km²</td>
</tr>
<tr>
<td>Lake volume</td>
<td>203 x 10⁶ m³</td>
</tr>
<tr>
<td>Maximum depth</td>
<td>35-1 m</td>
</tr>
<tr>
<td>Average depth</td>
<td>9-4 m</td>
</tr>
<tr>
<td>Location of maximum depth</td>
<td>just upstream from dam wall</td>
</tr>
<tr>
<td>Ratio average depth: maximum depth</td>
<td>~0-27</td>
</tr>
<tr>
<td>Volume of epilimnion (E)</td>
<td>~116 x 10⁶ m³</td>
</tr>
<tr>
<td>Volume of hypolimnion (H)</td>
<td>~71 x 10⁶ m³</td>
</tr>
<tr>
<td>Volume of metalimnion</td>
<td>~16 x 10⁶ m³</td>
</tr>
<tr>
<td>Ratio E:H</td>
<td>1-63</td>
</tr>
<tr>
<td>Time and duration of stagnation</td>
<td>~Oct.-April (approx. 180 days)</td>
</tr>
<tr>
<td>Depth range of metalimnion</td>
<td>~6-0 - 9-0 m</td>
</tr>
<tr>
<td>Shoreline at full supply level</td>
<td>151 km</td>
</tr>
<tr>
<td>Shoreline development</td>
<td>9-2</td>
</tr>
<tr>
<td>Retention time</td>
<td>~2-0 yrs.</td>
</tr>
<tr>
<td>Secchi transparency</td>
<td>min. 1-6 m wet season</td>
</tr>
<tr>
<td></td>
<td>max. 4-5 m dry season</td>
</tr>
</tbody>
</table>

people reside in the catchment, with 400 in Dayboro (situated on Terrors Creek) the only town in the catchment. Sewage disposal is by septic systems (half of rural dwellings) or earth closets with subsequent transport beyond the catchment.

A basic description

**Physico-chemical features.** Morphometric data for N.P.D. are summarised in Table 22. At full supply level, the lake occupies 6-1 per cent of the catchment. The storage is designed to yield an average of 150,000 m³ day⁻¹ (max. 230,000 m³ day⁻¹) of water. Supplies leave via ten screened valves at 2-44 m intervals, each separately controlled. Water is fully treated by flocculation, sedimentation, filtration and chlorination, although of good quality and soft with an average total hardness of 67 ppm CaCO₃. The ionic order of dominance is consistently Na⁺ > Ca²⁺ > K⁺ and HCO₃⁻ > Cl⁻ > HSiO₃⁻ > SO₄²⁻ (Table 23), in agreement with the general observations of Williams and Wan (1972) for Australian waters.

North Pine Dam is a shallow (z = 9-4 m), warm monomictic lake stratifying from October to April, with a poten-

Table 23  Typical major ionic composition in North Pine Dam. Data from 31 samples collected January-April 1975. Values as mg 1⁻¹.

<table>
<thead>
<tr>
<th>Cations</th>
<th>Mean</th>
<th>Range</th>
<th>Anions</th>
<th>Mean</th>
<th>Range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Na⁺</td>
<td>23-5</td>
<td>(19-31)</td>
<td>HCO₃⁻</td>
<td>48-0</td>
<td>(36-102)</td>
</tr>
<tr>
<td>Ca²⁺</td>
<td>21-7</td>
<td>(16-42)</td>
<td>Cl⁻</td>
<td>44-3</td>
<td>(34-80)</td>
</tr>
<tr>
<td>Mg²⁺</td>
<td>8-3</td>
<td>(3-18)</td>
<td>SO₄²⁻</td>
<td>4-3</td>
<td>(1-5-8-5)</td>
</tr>
<tr>
<td>K⁺</td>
<td>2-0</td>
<td>(0-3-12-0)</td>
<td>HSiO₃⁻</td>
<td>10-9</td>
<td>(0-4-15-2)</td>
</tr>
</tbody>
</table>
tial for partial mixing during periods of high inflow; high intensity inflows of short duration are characteristic of the wet season. Average summer temperature in the epilimnion is 24-25°C, and in the surface 2-3 metres often > 26°C. Mid-winter temperature is nearly a uniform 14-15°C throughout but with the surface 2-3 metres near 16-18°C.

**Biological features.** Since storage began the abundance of phytoplanktonic indicators of eutrophication such as *Microcystis* and the diatom *Melosira* has consistently increased. Apart from these genera, various other phytoplanktonic algae have been identified with abundances around $10^6$-$10^7$ cells m$^{-3}$. Chlorophyll-a concentrations from June 1976 to July 1977 have ranged from 3-20 mg m$^{-3}$ (King, 1978). The limnetic crustacean zooplankton appears to behave like that of Borumba Dam, Queensland (Timms and Midgley, 1969). It is dominated by *Boeckella minuta* and *Mesocyclops leuckarti*; both species are continuously present, with densities varying around 13,000 and 10,000 animals m$^{-3}$ respectively, values similar to those reported by Bayly (1962) for *Boeckella propinqua* and *Mesocyclops leuckarti* in some New Zealand lakes. Three rotifer species have maintained a more or less continuous presence with population densities averaging 3-6,000 m$^{-3}$. The species are *Keratella valga* form *tropica*, *Asplanchna priodonta* and *Polyarthra longiremis*; of these, *A. priodonta* maintains the greatest standing crop (biomass). Other species are present in significant but often fluctuating densities (King, 1978).

Initially the dominant macrophytes colonising the lake margin were *Hydrilla verticillata*, *Potamogeton crispus* and *P. ochreatus*, giving a standing crop estimated as 140 tonnes dry weight in July 1975 (0.8 tonnes Tot-P, 6.0 tonnes Tot-N) (Dodd *et al.*, 1975). This vegetation was summarily drowned in 1976. However, *H. verticillata* reappeared and by 1977 was entirely dominant, in almost pure stands, and covered large areas of the lake margin. Additionally, *Eichhornia crassipes* has become firmly established in small pockets around the entire 151 km of shoreline. The area of one small bed was observed to double within 7 days in November 1977. The presence of numerous gullies sheltered by forested banks (see Table 22, shoreline development = 9.2) could make future control of this weed extremely difficult.

**Post-impoundment changes**

Two distinct stages occurred in the filling of N.P.D. The first was during the floods of January 1974, after which N.P.D. maintained a fairly constant size until the beginning of 1976. The second stage followed, when N.P.D. filled rapidly to almost full supply level, increasing sevenfold in volume, fourfold in surface area and doubling in depth. Both stages have been monitored.

Although trees were felled and burned before inundation, subsequent pasture growth resulted in the inundation of some 4,400 tonnes (dry wt) of grasses during the first stage filling. Eighteen months later it was estimated that 87 per cent of this material had broken down to fragments smaller than 1 mm long. Oxygen stratification during this period was very marked, with a gradient of 5-0 ppm dissolved oxygen for each 0.5 m change in depth; 20 per cent of the lake floor and 10 per cent of its volume had oxygen concentrations < 0.3 ppm. Following the second stage of filling, in 1976, more
than 10,000 tonnes dry weight of terrestrial grasses were then submerged. During subsequent stratification, 35 per cent of the new lake volume and 47 per cent of its floor were again depleted of oxygen. Under these anoxic conditions, release of nutrients and mobilisation of iron and manganese proceeded. High concentrations of the latter caused some serious operational problems for the treatment plant during this period.

Considerable amounts of iron (~13,000 tonnes) and manganese (~400 tonnes) were initially present in submerged vegetation and the top 2 cm of soil on the bottom of N.P.D. Further, it was estimated that 10 tonnes of total phosphorus and 60 tonnes of total nitrogen were present in the vegetation alone prior to submergence (King and Everson, 1978). These values represented a significant potential internal loading for these nutrients. Total phosphorus in the soil ranged from about 100-2,000+ mg kg⁻¹ dry wt, a range comparable to that (350-1,188) reported by Hart et al. (1976) for Lake Mulwala sediments. During a period of holomixis, it is estimated that 4 tonnes of total phosphorus and 11 tonnes of total nitrogen were released from the sediments into the first stage lake (and when the surface area of this was only one quarter the present area of N.P.D.).

Oxygen depletion rates are estimated to have exceeded 500 mg O₂ m⁻² day⁻¹ but not, however, to have approached the high values of 1,000 mg O₂ m⁻² day⁻¹ reported for Lake Washington (Edmondson et al., 1956).

**Trophic status**

Mean summer concentrations of chlorophyll-α in excess of 10-15 mg m⁻³ have been regarded as indicative of eutrophic conditions (Vollenweider, 1968). Concentrations during 1976-77 in the epilimnion of N.P.D. ranged between 3-4 mg Chl-α m⁻³ in the dry season to 8-20 mg Chl-α m⁻³ (mean 10 mg Chl-α m⁻³) in the wet. These figures suggest that the trophic status lies between mesotrophy and eutrophy. Similarly, scores of 50 on Carlson's indices (Carlson, 1977) indicate mesotrophic conditions.

Zooplankton production is estimated as ~500 KJ m⁻² yr⁻¹ (~13 Kcals m⁻³ yr⁻¹ whole lake, ~23 Kcals m⁻³ yr⁻¹ for the epilimnion); this is a moderate level of productivity and also suggests mesotrophy (Wetzel, 1975). Zoobenthic production, not regarded as a reliable indicator of trophic status, is estimated as ~100 KJ m⁻² yr⁻¹.

The importance of nitrogen and particularly phosphorus in determining trophic status is now widely accepted. Exports of total nitrogen and total phosphorus from the N.P.D. catchment are estimated to be in the order of 1.8 and 0.4 kg ha⁻¹ yr⁻¹ respectively, yielding annual loadings of ~2.3 g Tot-N m⁻² and ~0.5 g Tot-P m⁻² for the lake. The rate of nitrogen export is considered low; no estimates of nitrogen entering the system through thunderstorm activity are available. The nitrogen loading, on the basis of Vollenweider's (1968) function, would suggest that N.P.D. is at present between mesotrophy and eutrophy. Phosphorus loading is moderate (cf. Wood, 1975), but would still place N.P.D. in the potentially eutrophic category in a number of models (Vollenweider, 1968; Vollenweider and Dillon, 1974; Dillon, 1975). Thus, although the lake may not yet be thoroughly mixed and steady state conditions do not yet prevail, phosphorus and nitrogen loadings point to a future eutrophic status.
It is very difficult to quantify the extent of internal nutrient loading, although data (King and Everson, 1978) support Vollenweider's (1968) view that this source of nutrients is significant, at least in the initial stage. Similarly, it is impossible to predict the extremely erratic and variable inflow; for example amounts larger than the average annual loading of nitrogen and phosphorus can be added in one month's runoff under flood conditions (e.g. in 1974 and 1976). Such spike inputs, plus additional internal loading from partial mixing, could produce nuisance blooms of algae. Furthermore, inputs from various subcatchments vary. Nitrogen is exported from the Terrors Creek catchment at about double the rate that it is from the other subcatchments (Dodd et al., 1975; King and Everson, 1978); similarly, higher export rates for calcium and magnesium are also found in this relatively developed sub-catchment. The most heavily forested sub-catchment (Kobble Creek: 90 per cent forested) exports magnesium, calcium and sulphate at the lowest rates, but chloride at the highest (King and Everson, 1978). Variation in sub-catchment inputs may reflect past land use, and it is not clear precisely what effect further deforestation, increased grazing or more urbanisation would have on N.P.D. It is reasonably certain that nitrogen and phosphorus loadings will increase and calcium, magnesium and sulphate losses from the catchment will rise if uncontrolled development of the catchment occurs.

**Conclusion**
At present N.P.D. has good quality water. With the predicted trend towards eutrophication, however, water quality can be expected to deteriorate and treatment costs gradually to increase. Moreover, extensive development in the catchment would certainly disturb the present nutrient balance and bring about an even more rapid deterioration in water quality. Unfortunately, short of direct land acquisition, the Brisbane City Council has no control over the catchment, and therefore has inadequate protection for a valuable resource. *This is a very clear cause for concern.* If this chapter does no more than draw attention to the need for catchment control in water resource management, its preparation will have been well worthwhile.

**Acknowledgments**
The investigation of N.P.D. has been undertaken by staff of the Department of Biology and Environmental Science, Queensland Institute of Technology. The project group consists of Mr C. R. King, Dr A. Bailey, and Dr R. G. Everson (Chairman). Finance has come from the Queensland Co-ordinator General's Department and the Australian Water Resources Council (A.W.R.C. Project 75/91). The technical assistance of J. Wilson and P. Lather, and the co-operation of the Queensland Department of Local Government and the Brisbane City Council are gratefully acknowledged.

**References**


Introduction
Lake Burley Griffin (L.B.G.) differs from other reservoirs discussed in this book in that its primary role is that of an urban resource. Its main purpose is to provide a unifying, landscape-architectural feature (McCoy, 1976) as part of the city of Canberra, and an aesthetic

Fig. 39  A (left), Lake Burley Griffin in relation to the upper Molonglo-Queanbeyan River system. Stipple represents urban catchments. B (right), Bathymetric map of Lake Burley Griffin; contours in metres. Arrows represent major stormwater inflows apart from Sullivan’s Creek.
and recreational component of the urban environment.

Two distinctive characteristics of L.B.G. are a high level of abiogenic turbidity (far from unique amongst Australian fresh waters), and chronic zinc pollution derived from an abandoned mining operation in the catchment. Both characteristics impinge on the lake's value as an urban resource.

Lake Burley Griffin: the resource

Physical environment and morphology. L.B.G. is situated in the Australian Capital Territory, 35° 17' S, 149° E, and 556 m above sea level. It was first filled in 1964 following the construction of Scrivener Dam on the Molonglo River, west of Canberra. The mean annual rainfall at Canberra is 664 mm, but during the past 40 years rainfall has varied from 254 mm to 1,092 mm in a year. The lake has a total catchment of 183,700 ha. Approximately 4,500 ha of this is in urban Canberra. The main streams of the catchment are shown in Fig. 39A.

L.B.G. consists of four sequentially connected basins followed by a long narrow reach immediately above Scrivener Dam (Fig. 39B). Although the lake increases in depth along its length, a feature typical of dammed rivers, it is generally very shallow. Over some 60 per cent of its surface area, including virtually all of the first two basins, it is less than 5 m deep. Morphometric data are presented in Table 24 and Fig. 39B.

Hydrology. Mean annual flow measured over 44 years at a site on the Molonglo now inundated by L.B.G. was 160,600 Ml. Assuming that increases in local runoff through urbanisation tend to compensate for the lake's increased evaporative surface, this figure can serve to estimate the average water replacement time: 0.21 years. A detailed hydrological budget for 1972-73 (Hillman, 1974) is summarised in Table 25. Rainfall for much of the period was below average, particularly late in 1972 and early 1973. Consequently, local streamflow was depressed until the latter part of 1973.
Table 25  Water budget of Lake Burley Griffin. Data in Megalitres.

| Changes in lake volume | Input |  | Output |  |  | Unaccounted residual |
|-----------------------|-------|-------|------|-------|---------|
| Year                  | Molonglo R. | Jerrabomberra Ck | Stormwater NorthSouth | Rainfall | Total Outflow at dam | Evaporation | Total | |
| 1972                  | 50685  | 2069  | 2971  | 2426  | 3272    | 61164  | 56150 | 8008 | 64158 | -1768 |
| 1973                  | 93870  | 6256  | 5361  | 4057  | 5738    | 176446 | 86947 | 6919 | 93866 | 23842* |

* Discrepancy attributed to bed erosion at outflow gauging station.

**Major ion chemistry.** Results of fortnightly major ion determinations made by the Department of the Capital Territory during 1972-73 showed no clear-cut order of dominance amongst the cations (Hillman, 1974). The order is represented as Ca, Mg, Na > K : HCO₃ > Cl > SO₄. This order is not usual for Australian waters, but approximates that of Warragamba Dam: 31, 28, 38, 3: 49, 43, 8 (as per cent equivalent sum of total cations) in the order listed above (Williams, 1967).

**Biology.** L.B.G. usually supports light to moderate densities of planktonic algae, often dominated by the diatom genus *Melosira*, although other algae including the blue-green, *Anacystis cyaneae*, and the colonial greens, *Volvox* sp. and *Botryococcus braunii*, have appeared in large numbers occasionally. Field measurements in 1972-73 showed that planktonic primary production was limited through the restriction of light penetration by turbidity, and by the depressant effect of zinc on photosynthesis (Hillman, 1974). These two factors also influenced species composition of the phytoplankton.

Submergent aquatic plants in L.B.G. include *Potamogeton crispus*, *P. tricir- natus*, *Vallisneria spiralis*, and less commonly *Elodea canadensis*, *Triglochin procera*, *Myriophyllum verrucosum*, and *M. propinquum*. Establishment of new weed beds is probably limited by turbidity. During periods of low turbidity in summer 1972-73 and 1977, extensive beds (mainly of *Potamogeton crispus*) appeared in shallow areas, particularly in East Basin.

The main zooplankton species in L.B.G. are *Daphnia carinata*, *Ceriodaphnia quadrangula*, *Bosmina meridionalis*, *Boeckella triarticulata*, and *Calamoecia lucasi*. In the laboratory these species showed varying degrees of susceptibility to zinc (Hillman, 1974) and levels of zinc pollution may influence species composition and density of the zooplankton community. Macroinvertebrate fauna in the Molonglo and probably the lake itself is depauperate, presumably resulting from zinc pollution (Weatherley et al., 1967). In particular, molluscs and macrocrustaceans are poorly represented.

The lake has been stocked regularly with fish (including rainbow trout, *Salmo gairdneri*, brown trout, *S. trutta*, and the native golden perch, *Plectroplites ambiguus*), successfully maintaining a supply
Man-made lakes

for anglers. The European carp, *Cypri­nus carpio*, has recently invaded the lake, but the level of urbanisation, traffic, etc., probably prevents the establishment of aquatic mammal populations. A community of eastern water dragons, *Physignathus lesueurii*, has established on a steep, rocky shore of West Lake.

Problems in managing Lake Burley Griffin

**Zinc pollution.** The ultimate source of zinc pollution is a mining operation at Captains Flat on the Molonglo River some 60 km upstream from L.B.G. The mine was abandoned in 1962. The history of heavy metal pollution from Captains Flat has been detailed elsewhere (e.g. Weatherley *et al.*, 1967; Anon., 1974; Hillman, 1974). For the present discussion the salient points are:

A small collapse of mine wastes in 1939 and a larger one (approximately 31,000 m³) in 1942 resulted in a flash flood of highly polluted acid water and deposited large quantities of heavy metals along the bed of the Molonglo and in parts of the floodplain. Extensive natural flooding in 1945 contaminated the floodplain approximately 20 km downstream from the mine, killing extensive areas of vegetation.

The steep-sided tailings dumps, directly adjacent to the river, have eroded extensively.

Since closure of the mine, groundwater and a small natural stream have inundated the mine shafts, providing a source of low-pH water contaminated by heavy metals.

Thus there are four main sources of zinc pollution of L.B.G.: two point sources (the tailings dumps and water from the mine), and two diffuse sources (sediments in the Molonglo and L.B.G. and deposits on the Molonglo floodplain).

Table 26 provides some estimates of the potential and pattern of pollution from these sources. Since the lake's completion, zinc concentrations in the water have been at a fairly constant level, usually ranging between 0.05 and 0.25 ppm. The level appears to vary directly with stream flow and turbidity. During 1972-73 an estimated 33,120 kg entered the lake in water from the Mol-

<table>
<thead>
<tr>
<th>Source</th>
<th>Quantity of Zn stored (kg)</th>
<th>Input characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Waste dumps</td>
<td>$31,600 \times 10^3$*</td>
<td>Surface erosion and percolation following local rain</td>
</tr>
<tr>
<td>Mine water</td>
<td>unknown, but large</td>
<td>Input virtually continuous—dependent on seepage and creek flow</td>
</tr>
<tr>
<td>Molonglo floodplain</td>
<td>unknown</td>
<td>From groundwater input and erosion following local rain</td>
</tr>
<tr>
<td>Sediment</td>
<td>120,000†</td>
<td>Bed load dependent on flow, elution dependent on water chemistry, particularly pH</td>
</tr>
<tr>
<td></td>
<td>(100,000 in LBG, 20,000 in Molonglo)</td>
<td></td>
</tr>
</tbody>
</table>

† Per cm depth from Hillman (1974).
Table 27 Nutrient budgets for Lake Burley Griffin over the 2-year period: Jan. 1972 to Dec. 1973 (kg). Figures in parentheses are weighted average concentrations (mg/l). Data from Hillman (1974).

<table>
<thead>
<tr>
<th>Nutrient</th>
<th>Inputs from:</th>
<th>Stormwater</th>
<th>Rain</th>
<th>Total</th>
<th>Output at:</th>
<th>Balance</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Rivers</td>
<td></td>
<td></td>
<td></td>
<td>Scrivener Dam</td>
<td></td>
</tr>
<tr>
<td>Total-P</td>
<td>15038</td>
<td>3566</td>
<td>43</td>
<td>18647</td>
<td>12314</td>
<td>6333</td>
</tr>
<tr>
<td></td>
<td>(-0.99)</td>
<td>(-2.41)</td>
<td>(-0.05)</td>
<td>(-1.06)</td>
<td>(-0.76)</td>
<td></td>
</tr>
<tr>
<td>Ortho-P</td>
<td>2024</td>
<td>3414</td>
<td>43</td>
<td>5483</td>
<td>1802</td>
<td>3681</td>
</tr>
<tr>
<td></td>
<td>(-0.13)</td>
<td>(-2.03)</td>
<td>(-0.05)</td>
<td>(-0.31)</td>
<td>(-0.11)</td>
<td></td>
</tr>
<tr>
<td>Nitrate-N</td>
<td>16940</td>
<td>5801</td>
<td>722</td>
<td>23463</td>
<td>7681</td>
<td>15782</td>
</tr>
<tr>
<td></td>
<td>(-1.11)</td>
<td>(-3.91)</td>
<td>(-0.8)</td>
<td>(-1.33)</td>
<td>(-0.47)</td>
<td></td>
</tr>
</tbody>
</table>

Ongoing. Of this, 10,940 kg left in the water flowing from L.B.G. (Hillman, 1974).

**Turbidity.** Turbid waters may be considered to have less aesthetic value than clear waters. The water of L.B.G. almost always has a high level of abiotic turbidity, resulting in rapid attenuation of light through the water column. Measurements throughout 1972-73 indicated that the 1 per cent light attenuation level (the depth at which light intensity is 1 per cent of that at the surface) was deeper than 4 m only during four months, December to March (a period marked by extensive colonisation by Potamogeton crispus). For more than half the period the 1 per cent level was less than 2 m deep (Hillman, 1974).

Munro and Hattersley (1969) stated that fine sediment was evenly distributed throughout the lake and that in waters of 3 m depth or less—about 30 per cent of the area of the lake—this material would be resuspended by wind-induced turbulence. Although, as previously noted, turbidity does tend to decline during extended dry periods, indicating an important contribution by influent streams to turbidity in L.B.G. (strong winds may also be less frequent at such times), internal resuspension is also likely to be a significant source.

**Nutrient enrichment.** A budget for phosphorus and nitrogen in 1972-73 is presented in Table 27. During the same period several trials, using samples of lake water, indicated that, under laboratory conditions, additions of phosphorus significantly stimulated algal growth, whereas nutrient combinations lacking phosphorus did not (Hillman, 1974). Enrichment trials in the field were not undertaken, but ambient concentrations of readily available phosphorus (Table 27) appear usually to have been considerably in excess of the needs of existing algal populations. In relation to northern hemisphere values (e.g. Vollenweider, 1968; Dillon, 1975), L.B.G. would be rated as bordering between mesotrophy and eutrophy. From available evidence it appears that all nutrient elements occur in L.B.G. at concentrations which could support substantial plant populations.

**Ecological management of Lake Burley Griffin**

Left unchecked, the sum of all ecological forces in any ecosystem will drive it
towards a steady state condition referred to as the *ecological climax* (thus in general terms, an abandoned farm might return to natural scrub, burnt forests regenerate, etc.). It appears that under conditions of chronic zinc pollution and high nutrient loads, with occasional drops in turbidity, the climax state of L.B.G. could include a depauperate fauna, extensive weed beds, and possibly occasional algal blooms.

For an ecosystem to be maintained in other than its natural climax condition, inputs of energy are necessary. These may be in the form of continuous or recurrent inputs to prevent the system from reaching its climax state, or discrete inputs to change the forces determining the climax and thereby achieve a new, more desirable, final state. Examples of the first type are routine restocking of fish and application of herbicide and/or mowing of weed beds. The second form of ecological management is being attempted in control measures against zinc pollution.

Options available for zinc pollution control were analysed by Weatherley and Dawson (1973) and later, in some detail, by Weatherley *et al.* (1975), who provided strategies through which cost/benefits might be assessed. An unrelated study (Anon., 1974) recommended measures to limit two of the four sources of zinc pollution: runoff from waste dumps and groundwater from the mine shafts. By controlling these two major point sources, it is hoped to reduce zinc concentrations in the system to safe levels. An extensive sampling program has been undertaken to assess the outcome (Fitzgerald and Haldane, 1977). To produce real ecological improvement it will be necessary not only to reduce the base-line levels of zinc in L.B.G. but also to avoid sudden 'spikes' of zinc pollution of the sort which have often followed rain in the past. The success of the proposed measures, at least in the short term, will depend on the relative contribution of the uncontrolled zinc sources (see Table 26) to the overall zinc load. In the long term, if inputs from Captains Flat are effectively stemmed, deposition of 'low-zinc' sediment should produce a gradual improvement in the situation.

At present no procedures are planned to change the climax state with regard to algal and plant growth. Despite the high level of nutrients, weeds and algae are usually kept in check by turbidity. Weed beds are mown or poisoned when irregular drops in turbidity levels allow them to detract from the aesthetic or recreational value of L.B.G. These recurrent measures are probably the most cost-effective available whilst the turbidity regime remains unchanged. Should other operations (e.g. stormwater quality control, sedimentation to restrict zinc pollution) reduce levels of turbidity, discrete changes to the system may need to be envisaged. These might include deepening shallow areas of the lake (to limit weed colonisation) or nutrient removal. Studies in progress (Higgins, Henkle and Lawrence, 1976; Cullen, Rosich and Bek, 1977) should provide the data on which to base future management effort.

It appears certain that L.B.G. will remain a highly managed ecosystem and it may well be that some solutions developed for management problems on L.B.G. can be incorporated into the design of future urban water resources.
References


Introduction
The most significant use of Australian water resources is for domestic and industrial supplies in cities. Elsewhere in this book, detailed consideration is given to specific problems in providing these supplies, or to specific impoundments from which supplies are derived. This chapter has a different emphasis, and considers the supply system to a single city, Sydney, and discusses this more generally but as a whole. Differences between the water supply systems of the major Australian cities are so great that a detailed consideration of one supply system is unlikely to be useful on a comparative basis. Such a consideration does serve, however, to emphasise the complexity and extent of present supply systems for large Australian cities.

The supply and its development
The first major storage of water for supply to the Sydney area occurred with the commissioning of Prospect Reservoir in 1886. This reservoir was not a river impoundment scheme but drew water from weirs on the Upper Nepean and Cataract Rivers via a 57 km conduit which later became known as the Upper Canal. The reliability of the Upper Nepean supply was enhanced in 1907 by the completion of Cataract Reservoir, impounding the Cataract River, and subsequently by Cordeaux, Avon and Nepean Reservoirs which with their contiguous catchments form the Upper Nepean or Metropolitan supply. The final development on the Metropolitan catchment occurred in 1941 with the completion of Woronora Dam. However, its supply is a local one largely independent of the Prospect system.

Sydney's increasing population and its concomitantly increasing demand for water required an augmentation of supply from the Metropolitan catchment and this was achieved in 1940 with the pumping of water from a weir on the Warragamba River. In 1959, gravity feed from Warragamba Dam (Lake Burragorang) into Prospect Reservoir began. Thus, in 1974 Prospect was drawing 393 x 10^6 m^3 of water supplied from Lake Burragorang (by pipeline; 74.2 per cent), Nepean River Reservoirs (by canal; 25.5 per cent), and natural catchment (0.3 per cent). Apart from the natural inflow, inputs were chlorinated.

Further supplementation of supplies has been needed and has led to the development of the Shoalhaven scheme. This will become fully operational in 1978. An impoundment on the Shoalhaven River at Tallowa will supply pump-lifted water to a reservoir at Fitzroy Falls which in turn will supply pumped water to another storage at Wingecarribee. From this point, water will be released either via the Wingecarribee and Wollondilly Rivers to enter Lake Burragorang, or via Glenquarry Cut to Nepean Reservoir—which itself has been recently connected to the Avon
Some limnological features of the Sydney water supply system

Table 28 Some salient morphometric information on Sydney water supply reservoirs

<table>
<thead>
<tr>
<th>Reservoir</th>
<th>Total capacity (10^6 m³)</th>
<th>Max. depth (m)</th>
<th>Year completed</th>
<th>Area (km²)</th>
<th>Full storage level (m.a.s.l.)</th>
<th>Catchment area (km²)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Prospect</td>
<td>50-1</td>
<td>24</td>
<td>1886</td>
<td>5-1</td>
<td>60</td>
<td>9-7</td>
</tr>
<tr>
<td>Cataract</td>
<td>94-3</td>
<td>46</td>
<td>1907</td>
<td>8-5</td>
<td>290</td>
<td>130</td>
</tr>
<tr>
<td>Cordeaux</td>
<td>93-6</td>
<td>52</td>
<td>1926</td>
<td>7-8</td>
<td>304</td>
<td>90</td>
</tr>
<tr>
<td>Avon</td>
<td>214-4</td>
<td>66</td>
<td>1927</td>
<td>10-6</td>
<td>320</td>
<td>140</td>
</tr>
<tr>
<td>Nepean</td>
<td>81-4</td>
<td>69</td>
<td>1935</td>
<td>3-6</td>
<td>321</td>
<td>320</td>
</tr>
<tr>
<td>Woronora</td>
<td>71-8</td>
<td>61</td>
<td>1941</td>
<td>3-8</td>
<td>169</td>
<td>85</td>
</tr>
<tr>
<td>Burragorang*</td>
<td>2057-0</td>
<td>105</td>
<td>1960</td>
<td>75-0</td>
<td>117</td>
<td>9000</td>
</tr>
<tr>
<td>Wingecarribee</td>
<td>34-5</td>
<td>12</td>
<td>1976¹</td>
<td>6-2</td>
<td>678</td>
<td>40</td>
</tr>
<tr>
<td>Fitzroy Falls</td>
<td>23-5</td>
<td>10</td>
<td>1976²</td>
<td>5-2</td>
<td>663</td>
<td>31</td>
</tr>
<tr>
<td>Tallowa</td>
<td>110-0</td>
<td>35</td>
<td>1977</td>
<td>9-3</td>
<td>56</td>
<td>5750</td>
</tr>
</tbody>
</table>

* = Warragamba Dam.
¹ Period of first significant fill, October 1973.
² Period of first significant fill, March-April 1974.

Reservoir by tunnel. A further reservoir proposed for the Upper Shoalhaven near Braidwood will complete the current scheme.

Some salient information on the various storage reservoirs involved is given in Table 28, and a schematic representation of the arrangement of the system is provided in Fig. 40. Further information on the system has been given by Jolly

Table 29 Zooplankton Crustacea in some water supply reservoirs for Sydney. Data based on monthly or more frequent samples for one year, and irregular observations 1960-65 (after Jolly, 1966).

<table>
<thead>
<tr>
<th>Taxon*</th>
<th>Avon</th>
<th>Cataract</th>
<th>Cordeaux</th>
<th>Nepean</th>
<th>Burragorang</th>
<th>Woronora</th>
</tr>
</thead>
<tbody>
<tr>
<td>CLADOCERA</td>
<td></td>
<td></td>
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<td></td>
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<tr>
<td>Bosmina meridonialis</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
</tr>
<tr>
<td>Bosminopsis deitersi</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ceriodaphnia quadrangularis</td>
<td>++</td>
<td>++</td>
<td>++</td>
<td>++</td>
<td>+</td>
<td>++</td>
</tr>
<tr>
<td>Daphnia carinata</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>D. lumholtzii</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Moina tenuicornis</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diaphanosoma excisum</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>COPEPODA</td>
<td></td>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Boeckella fluvialis</td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>B. minuta</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+</td>
<td>+++</td>
</tr>
<tr>
<td>Calamoecia ampulla</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>C. lucasi</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mesocyclops leuckarti</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
<td>+++</td>
</tr>
<tr>
<td>Tropocyclops prasinus</td>
<td>++</td>
<td>++</td>
<td>++</td>
<td>++</td>
<td>++</td>
<td>++</td>
</tr>
</tbody>
</table>

Key: ++++, always present; ++, periodically common; +, occasional occurrence; —, never present.
* Nomenclature directly after Jolly (1966).
Table 29 summarises her data on crustacean zooplankton which otherwise are not considered in this chapter.

Chemical features of the supply reservoirs
The major chemical features of the older reservoirs have changed little over the years. Table 30 indicates typical values. One feature, however, that has changed in Lake Burrarorang is the extent of the input of phosphorus; it has greatly increased. The increase can be correlated with catchment events—especially bushfires and stormwater flows. Thus, studies have shown enhanced phosphorus concentrations in stormwater flows from the catchment; the correlation between phosphate concentrations and inflow volume is high, and for the period 1972-1974 was $+0.63$ ($P < 0.001$) between outflow volume ($=\text{inflow} - \text{evaporation}$) and phosphate concentrations at the dam wall. It is important to note that 50 per cent of the catchment is covered by only sclerophyll forest, 30 per cent by depauperate forest with some grazing, 19 per cent by improved pasture, and < 1 per cent by urban development. Total population number on the catchment is about 120,000. The average concentrations of $\text{PO}_4-P$ in Lake Burrarorang and
Some limnological features of the Sydney water supply system

Table 30 Major chemical features of older reservoirs (from Jolly, 1966). Data relate to period January-November 1961 and are mean values. Data (except for pH and conductivity) as mg l⁻¹.

<table>
<thead>
<tr>
<th></th>
<th>Nepean</th>
<th>Avon</th>
<th>Cordeaux</th>
<th>Cataract</th>
<th>Woronora</th>
<th>Burragorang</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total dissolved solids</td>
<td>52</td>
<td>36</td>
<td>45</td>
<td>46</td>
<td>58</td>
<td>114</td>
</tr>
<tr>
<td>HCl insol. residue</td>
<td>5-1</td>
<td>1-9</td>
<td>2-0</td>
<td>2-0</td>
<td>2-4</td>
<td>7-2</td>
</tr>
<tr>
<td>Sodium</td>
<td>8-1</td>
<td>6-8</td>
<td>8-7</td>
<td>9-1</td>
<td>11-0</td>
<td>14-2</td>
</tr>
<tr>
<td>Potassium</td>
<td>0-9</td>
<td>0-7</td>
<td>0-7</td>
<td>0-7</td>
<td>0-9</td>
<td>1-9</td>
</tr>
<tr>
<td>Calcium</td>
<td>1-6</td>
<td>0-7</td>
<td>0-9</td>
<td>0-6</td>
<td>0-8</td>
<td>11-4</td>
</tr>
<tr>
<td>Magnesium</td>
<td>2-1</td>
<td>1-1</td>
<td>1-6</td>
<td>1-5</td>
<td>1-8</td>
<td>6-5</td>
</tr>
<tr>
<td>Chloride</td>
<td>13-4</td>
<td>10-1</td>
<td>12-6</td>
<td>15-4</td>
<td>19-8</td>
<td>25-0</td>
</tr>
<tr>
<td>Sulphate</td>
<td>2-5</td>
<td>2-0</td>
<td>2-6</td>
<td>2-6</td>
<td>2-0</td>
<td>6-9</td>
</tr>
<tr>
<td>Bicarbonate</td>
<td>8-8</td>
<td>3-0</td>
<td>5-7</td>
<td>2-7</td>
<td>2-5</td>
<td>41-1</td>
</tr>
<tr>
<td>Total hardness</td>
<td>13-1</td>
<td>6-2</td>
<td>9-6</td>
<td>7-8</td>
<td>9-7</td>
<td>56-4</td>
</tr>
<tr>
<td>Iron as Fe</td>
<td>0-3</td>
<td>0-1</td>
<td>0-15</td>
<td>0-25</td>
<td>0-25</td>
<td>0-1</td>
</tr>
<tr>
<td>Conductivity*</td>
<td>71</td>
<td>47</td>
<td>65</td>
<td>68</td>
<td>83</td>
<td>174</td>
</tr>
<tr>
<td>pH</td>
<td>6-6</td>
<td>6-2</td>
<td>6-5</td>
<td>6-2</td>
<td>6-0</td>
<td>7-2</td>
</tr>
<tr>
<td>Free CO₂</td>
<td>4</td>
<td>3-5</td>
<td>3</td>
<td>4</td>
<td>5</td>
<td>5-5</td>
</tr>
</tbody>
</table>

* µS cm⁻¹.

Prospect Reservoir are shown in Table 31.

Prospect Reservoir
At present, Prospect Reservoir serves almost entirely as a storage through which most (not all) of Sydney's water supply passes. A description of it, therefore, is a better indication of the nature of Sydney's supply than would be an indication based on any single 'feed' reservoir. For this reason, the principal features of Prospect Reservoir merit summary description.

Some salient data are given in Table 32. For the most part these have been obtained during routine water quality monitoring including part undertaken as a eutrophication investigation for O.E.C.D. (Organization for Economic Cooperation and Development). Phosphorus appears normally to limit

Table 31 Annual mean P0₄-P levels in Prospect Reservoir and Lake Burragorang. Data as mg m⁻³, and based on a minimum of 26 samples per annum taken from the surface to a depth of 4.5 m.

<table>
<thead>
<tr>
<th>Year</th>
<th>Prospect</th>
<th>Burragorang</th>
</tr>
</thead>
<tbody>
<tr>
<td>1970</td>
<td>8-9</td>
<td>8-0</td>
</tr>
<tr>
<td>1971</td>
<td>8-1</td>
<td>8-5</td>
</tr>
<tr>
<td>1972</td>
<td>8-5</td>
<td>7-9</td>
</tr>
<tr>
<td>1973</td>
<td>8-2</td>
<td>6-0</td>
</tr>
<tr>
<td>1974</td>
<td>9-8</td>
<td>16-1</td>
</tr>
<tr>
<td>1975</td>
<td>10-5</td>
<td>15-2</td>
</tr>
<tr>
<td>1976</td>
<td>9-7</td>
<td>11-0</td>
</tr>
<tr>
<td>1977</td>
<td>7-9</td>
<td>8-5</td>
</tr>
</tbody>
</table>
plant production, but changes in phosphorus input have occurred in recent years such that the status of the reservoir has tended to change from an oligotrophic one to a mesotrophic one. This has been reflected by changes in the nature of the algal crop; Table 33 presents some qualitative and quantitative data for 1964 and 1977 which illustrate this. The quantitative values are annual means, and though these take no account of periodicity (and are therefore an artificial method of describing the algal population), they do serve to identify gross changes. In 1964, the annual mean count for total algal cells (including a small number of unidentified forms) was 908 cells ml⁻¹. Melosira was dominant in 25 of the samples, Dinobryon in the other one. The most common sub-dominant was Staurastrum. In 1977, on the other hand, the annual mean count for total algal cells was 1,675 cells ml⁻¹. Melosira was dominant in 20 of the samples, Cyclotella in 4, and Rhizosolenia in 2. The most common sub-dominants were Synedra and Sphaerocystis. Between 1964 and 1977, there has been, in general, a decline in desmids with the notable exception of Cosmarium, and an increase in diatoms, especially Cyclotella and Rhizosolenia.

Concomitant with increased algal crops has been increased hypolimnial oxygen depletion rates. In 1976 these were so great that to avoid the development of anaerobic conditions in supplies leaving Prospect, the reservoir was overturned artificially. Figure 41 illustrates the sequence of events as indicated by temperature and dissolved oxygen profiles. Other control measures for hypolimnion oxygen control such as the introduction of Burragorang hypolimnion water were shown to be not suitable. The only observed changes in algal crop following the artificial overturn was an early development of the autumn Melosira granulata bloom, following the circulation of filaments from the hypolimnion.

### Changes associated with new impoundments

Few data exist which indicate the nature of the changes taking place in Australian reservoirs during the initial phases of their existence (cf. Chapter 31). Some data which bear upon this subject and are derived from studies of two new reservoirs in the Sydney supply system are of interest, therefore, and may use-
### Table 33 Phytoplankton in Prospect Reservoir 1964 and 1977. Data based on 26 fortnightly samples for each year obtained from depths of 0 to 4.5 m.

<table>
<thead>
<tr>
<th>Alga</th>
<th>Status and occurrence</th>
<th>Annual mean cell no. ml⁻¹ 1964</th>
<th>1977</th>
<th>Alga</th>
<th>Status and occurrence</th>
<th>Annual mean cell no. ml⁻¹ 1964</th>
<th>1977</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>CHLOROPHYTA</strong></td>
<td></td>
<td></td>
<td></td>
<td><strong>BACILLARIOPHYCEAE</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ankistrodesmus</td>
<td>++++</td>
<td>1.5</td>
<td>5.3</td>
<td>Achnanthes</td>
<td>+++(++++)</td>
<td>—</td>
<td>13.8</td>
</tr>
<tr>
<td>Actinotaenium</td>
<td>++</td>
<td>—</td>
<td>38.8</td>
<td>Amphora</td>
<td>+</td>
<td>—</td>
<td>5.8</td>
</tr>
<tr>
<td>Botryococcus</td>
<td>+</td>
<td>&lt;1-0</td>
<td>—</td>
<td>Asterionella</td>
<td>+++(++++)</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Chlamydomonas</td>
<td>++</td>
<td>4-0</td>
<td>—</td>
<td>Attheya</td>
<td>+</td>
<td>—</td>
<td>0-2</td>
</tr>
<tr>
<td>Closteriopsis</td>
<td>+</td>
<td>0-8</td>
<td>—</td>
<td>Cyclotella</td>
<td>+++(++++)</td>
<td>18.9</td>
<td>171-3</td>
</tr>
<tr>
<td>Closterium</td>
<td>+</td>
<td>0-2 &lt;0-1</td>
<td>—</td>
<td>Cymbella</td>
<td>+++</td>
<td>5.8</td>
<td>9-1</td>
</tr>
<tr>
<td>Coelastrum</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>Gomphonena</td>
<td>+</td>
<td>—</td>
<td>0-2</td>
</tr>
<tr>
<td>Cosmarium</td>
<td>+++(++++)</td>
<td>15.8 173.7</td>
<td>—</td>
<td>Gyrosigma</td>
<td>+</td>
<td>—</td>
<td>0-3</td>
</tr>
<tr>
<td>Crucigenia</td>
<td>+</td>
<td>1-4</td>
<td>—</td>
<td>Melosira distans</td>
<td>+</td>
<td>—</td>
<td>0-7</td>
</tr>
<tr>
<td>Desmidium</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td>Melosira granulata</td>
<td>+++(++++)</td>
<td>682-0</td>
<td>799-4</td>
</tr>
<tr>
<td>Dictyothespaerium</td>
<td>+++(++++)</td>
<td>1-8 8-6</td>
<td>6</td>
<td>Navicula</td>
<td>+++</td>
<td>—</td>
<td>23.9</td>
</tr>
<tr>
<td>Elakatothrix</td>
<td>++</td>
<td>1-5</td>
<td>6-2</td>
<td>Nitzschia</td>
<td>+</td>
<td>—</td>
<td>0-5</td>
</tr>
<tr>
<td>Eudorina</td>
<td>+</td>
<td>1-1</td>
<td>—</td>
<td>Rhizosolenlia</td>
<td>+++(++++)</td>
<td>1-1</td>
<td>126-2</td>
</tr>
<tr>
<td>Gloecystis?</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td>Surirella</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
</tr>
<tr>
<td>Hylatochla</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td>Synedra</td>
<td>+++(++++)</td>
<td>28-4</td>
<td>49-5</td>
</tr>
<tr>
<td>Kirchneriella</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td>Tabellaria</td>
<td>+</td>
<td>0-2</td>
<td>1-7</td>
</tr>
<tr>
<td>Keratococcus?</td>
<td>++</td>
<td>—</td>
<td>1-1</td>
<td>Thalassiosira?</td>
<td>+</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Lagerheimia</td>
<td>+</td>
<td>—</td>
<td>0-3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mougeotia</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nephrocytium</td>
<td>++</td>
<td>—</td>
<td>3-9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Oocystis</td>
<td>++</td>
<td>0-5</td>
<td>1-0</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pediastrum</td>
<td>+</td>
<td>&lt;0-1</td>
<td>0-6</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Quadrigula</td>
<td>++</td>
<td>—</td>
<td>7-1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Selenastrum</td>
<td>+</td>
<td>—</td>
<td>3-0</td>
<td>Ceratium</td>
<td>++</td>
<td>0-3</td>
<td>0-2</td>
</tr>
<tr>
<td>Scenedesmus</td>
<td>++</td>
<td>0-2</td>
<td>3-6</td>
<td>Gymnodinium</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
</tr>
<tr>
<td>Sphaerocystis</td>
<td>+++(++++)</td>
<td>25-3</td>
<td>—</td>
<td>Peridinium</td>
<td>+++</td>
<td>4-5</td>
<td>22-4</td>
</tr>
<tr>
<td>Spirogyra</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Staurastrum</td>
<td>+++(+++++)</td>
<td>73-3</td>
<td>2-9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>CRYPTOPHYTA</strong></td>
<td></td>
<td></td>
<td></td>
<td><strong>DINOPHYCEAE</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nephrocytium</td>
<td>++</td>
<td>—</td>
<td>3-9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Staurastrum</td>
<td>+++(+++++)</td>
<td>20-6</td>
<td>78-3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Staurastrum</td>
<td>+++(+++++)</td>
<td>20-6</td>
<td>78-3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>CHRYSOPHYCEAE</strong></td>
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<td></td>
<td></td>
<td><strong>EUGLENOPHYTA</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Anabaena</td>
<td>++</td>
<td>3-0</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Stigeoclonium</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Triposceras</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Volvox</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Zygnema</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>EUGLENOPHYTA</strong></td>
<td></td>
<td></td>
<td></td>
<td><strong>OCHROMONAS</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Euglena</td>
<td>+</td>
<td>&lt;0-1</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Trachelomonas</td>
<td>+</td>
<td>0-7</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>OCHROMONAS</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dinobryon</td>
<td>+++(++++)</td>
<td>20-6</td>
<td>78-3</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mallomonas</td>
<td>++</td>
<td>&lt;0-1</td>
<td>2-1</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pseudokephyrion</td>
<td>++</td>
<td>—</td>
<td>0-8</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ochromonas</td>
<td>+++(++++)</td>
<td>—</td>
<td>—</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Key: ++++, dominant or subdominant; ++++, common; ++, occasional; +, rare.*
fully be considered in this chapter. They relate to Wingecarribee and Fitzroy Falls reservoirs, both of which were operated as independent systems before December 1977.

Some morphological data for the two reservoirs are given in Table 28, and Tables 34 and 35 summarise information on nutrient levels and phytoplankton during the filling period. It may be noted that the catchments of both reservoirs have considerable areas of farmed land, and that neither reservoir has displayed any but the most transient thermal stratification.

The changes which have taken place in both reservoirs may be considered in three phases. In the first phase, the filling phase, there was an immediate release of nutrients from inundated soil and vegetation. Thus the period 1973-74 was marked by high nutrient concentrations, high colour, and high turbidity. Algal populations were most likely limited by

Table 34 Some nutrient and phytoplankton population data for Wingecarribee and Fitzroy Falls Reservoirs during initial filling phases. Data relate to interval October-October.

<table>
<thead>
<tr>
<th>Year</th>
<th>Total PO₄-P (mg m⁻³)</th>
<th>Silica (mg 1⁻¹)</th>
<th>Chlorophyll-a (mg m⁻³)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>mean</td>
<td>max.</td>
<td>min.</td>
</tr>
<tr>
<td>WINGECARRIBE</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1973-74</td>
<td>41.1</td>
<td>61</td>
<td>26</td>
</tr>
<tr>
<td>1974-75</td>
<td>31.5</td>
<td>39</td>
<td>22</td>
</tr>
<tr>
<td>1975-76</td>
<td>34.4</td>
<td>51</td>
<td>25</td>
</tr>
<tr>
<td>1976-77</td>
<td>25.9</td>
<td>36</td>
<td>21</td>
</tr>
<tr>
<td>FITZROY FALLS</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>1973-74</td>
<td>35.4</td>
<td>92</td>
<td>18</td>
</tr>
<tr>
<td>1974-75</td>
<td>30.1</td>
<td>47</td>
<td>20</td>
</tr>
<tr>
<td>1975-76</td>
<td>31.1</td>
<td>50</td>
<td>21</td>
</tr>
<tr>
<td>1976-77</td>
<td>28.3</td>
<td>44</td>
<td>20</td>
</tr>
</tbody>
</table>

* Not detected.
### Table 35

Algal counts in Wingecarribee and Fitzroy Falls Reservoirs during initial filling phases. Data are peak levels as cells ml\(^{-1}\).

<table>
<thead>
<tr>
<th>Date</th>
<th>Total algal count</th>
<th>Dominant alga</th>
<th>Dominant algal count</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>WINGECARRIBE</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>19. 9.74</td>
<td>13,870</td>
<td>Anacystis</td>
<td>7,406</td>
</tr>
<tr>
<td>13.11.75</td>
<td>14,986</td>
<td>Botryococcus</td>
<td>13,425</td>
</tr>
<tr>
<td>19. 8.76</td>
<td>7,815</td>
<td>Cyclotella</td>
<td>5,655</td>
</tr>
<tr>
<td>16. 9.76</td>
<td>6,389</td>
<td>Cyclotella</td>
<td>1,867</td>
</tr>
<tr>
<td>16.12.76</td>
<td>19,397</td>
<td>Sphaerocystis</td>
<td>13,192</td>
</tr>
<tr>
<td>21. 4.77</td>
<td>5,893</td>
<td>Asterionella</td>
<td>3,691</td>
</tr>
<tr>
<td>11. 8.77</td>
<td>12,992</td>
<td>Asterionella</td>
<td>11,176</td>
</tr>
<tr>
<td><strong>FITZROY FALLS</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>12. 2.76</td>
<td>4,342</td>
<td>Sphaerocystis</td>
<td>3,987</td>
</tr>
<tr>
<td>24. 6.76</td>
<td>4,077</td>
<td>Cyclotella</td>
<td>3,479</td>
</tr>
<tr>
<td>11.11.76</td>
<td>7,885</td>
<td>Actinotaenium</td>
<td>4,860</td>
</tr>
<tr>
<td>13. 1.77</td>
<td>11,934</td>
<td>Actinotaenium</td>
<td>6,761</td>
</tr>
<tr>
<td>10. 2.77</td>
<td>11,901</td>
<td>Actinotaenium</td>
<td>10,870</td>
</tr>
<tr>
<td>15. 9.77</td>
<td>24,381</td>
<td>Cyclotella</td>
<td>14,388</td>
</tr>
</tbody>
</table>

Inadequate light penetration. In a second phase, a quiescent phase up to October 1975, nutrient concentrations were fairly stable at a lower level than recorded initially but still high. Again light was a likely limiting factor to algal populations. In the third phase (October 1975 to October 1977), there was an increase in light penetration following lower turbidity, and, although nutrient concentrations were at their lowest recorded values, sufficient were present to promote large algal populations. In Wingecarribee algal growths have usually been of a green algae and diatom mixture, while the Fitzroy Falls reservoir, because of silica limitation, has produced fewer diatoms and more green algae. In view of the importance of agricultural activity on the catchments of both reservoirs it seems likely that nutrient levels and algal crops will continue to change and will need careful management.

### Acknowledgments

The permission of the Metropolitan Water, Sewerage and Drainage Board, Sydney, to publish this chapter is acknowledged. The views expressed, however, are those of the authors and not necessarily those of the Board.

### Reference

31 Limnological features of some Victorian reservoirs

I. J. Powling

Introduction
The State Rivers and Water Supply Commission of Victoria operates thirty major water storages. Mostly these store water for irrigation, but some are used for urban supplies. Three (Eildon, Eppalock, Tarago) have been studied in some detail over the past ten years; a fourth (Dartmouth) has been studied since November 1977 when impoundment first began (see Chapter 17). All four studies are considered in this chapter since each storage serves a different management objective. The geographical location of the reservoirs is indicated in Fig. 42.

Lake Eildon

Background. Lake Eildon, formed by a dam on the Goulburn River just downstream of its confluence with the Delatite River (Fig. 42), is primarily used to store irrigation water. It is also used to generate hydroelectric power, to assist in flood control when necessary, to supply water to perimeter settlements, and, above all, for recreation. All forms of water-based recreation are enjoyed on and around the lake particularly in summer when some hundreds of boats with

Fig. 42 Location of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam.
overnight accommodation, and thousands of smaller boats are to be seen. Around the shore are boat marinas, camping grounds, holiday houses and picnic areas.

Studies began in August 1968. Primarily they were a response to the need for information on the effect on water quality of the various forms of recreation—particularly as there were no sanitation facilities on the lake for cruiser boats with overnight accommodation. A full-scale bacteriological monitoring program formed part of the studies. Early results of this program pointed to several sensitive areas around the lake where *Escherischia coli* counts were inclined to be high especially after heavy rain. As a result, the first 'Sanitation Barge' for the collection and disposal of sewage from the cruiser boats was installed on the lake in 1972 (there are now three such stations, with more proposed). Since 1972 there has been a marked improvement in the quality of those surface waters which received the bulk of the contamination detected earlier.

In 1969, following the development of a massive bloom of the blue-green alga *Microcystis* sp., a further study began in which limnological events received attention. It was intended to monitor the physical, chemical and biological changes during each year in the hope that predictions could be made concerning the onset of future algal blooms. However, blooms have not recurred, and so the exercise has been simply one of long-term data collection. Nevertheless, several interesting phenomena have been observed and much useful information obtained. The stratification records, in particular, have assisted engineers in design calculations for other storages and related management activities such as destratification for water quality improvement.

The northern part of the reservoir, two-thirds of the total surface area, receives only 10 per cent of the total inflow, and this from predominantly cleared, arable land. The southern section, which has a steeper catchment of predominantly eucalypt forest, receives therefore the bulk of the flow.

**Physico-chemical features.** Important physical features are indicated in Table 36. Thermal patterns for two years (1971-72, 1975-76) are indicated in Fig. 43. From this it can be seen that Lake Eildon is a warm monomictic lake, with a winter holomixis (overturn). The hypolimnion does not normally become anoxic because of bottom release of water for irrigation and power generation, but has tended towards this state following irrigation shutdown in April each year.

Chemical features are indicated in Table 37. Noteworthy is the low concentration of total dissolved solids (mean 42 mg l⁻¹); this is lower than in any other Commission storage. Nutrient levels (phosphorus, nitrogen and silicon), turbidity, colour and chlorophyll-α levels are all higher and transparency lower on average in the northern section of the storage.

**Biological features.** Mean *E. coli* counts are generally higher in the northern section of the storage. The sections differ also in the species composition of their phytoplankton. Overall, the phytoplankton is very diverse and typical of an oligotrophic water, with green algae, particularly desmids, mostly dominant (about 150 species of phytoplankton have been collected, 120 being green
Table 36 Major morphometric and other physical features of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam

<table>
<thead>
<tr>
<th></th>
<th>Eildon</th>
<th>Eppalock</th>
<th>Tarago</th>
<th>Dartmouth</th>
</tr>
</thead>
<tbody>
<tr>
<td>Latitude and longitude</td>
<td>145°54' E</td>
<td>144°30' E</td>
<td>145°54' E</td>
<td>147°31' E</td>
</tr>
<tr>
<td>Height above sea level (crest of dam) (m)</td>
<td>291.65</td>
<td>199.10</td>
<td>161.09</td>
<td>494.00</td>
</tr>
<tr>
<td>Surface area (ha)</td>
<td>13,750</td>
<td>3,230</td>
<td>360</td>
<td>6,800*</td>
</tr>
<tr>
<td>Catchment area (km²)</td>
<td>3,885</td>
<td>2,124</td>
<td>117</td>
<td>3,613</td>
</tr>
<tr>
<td>Ratio catchment area: surface area</td>
<td>28:1</td>
<td>66:1</td>
<td>33:1</td>
<td>53:1</td>
</tr>
<tr>
<td>Shoreline (km)</td>
<td>466</td>
<td>160</td>
<td>21.2</td>
<td>275*</td>
</tr>
<tr>
<td>Capacity (volume stored at FSL) (ML)</td>
<td>3,392,073</td>
<td>311,900</td>
<td>37,600</td>
<td>3,700,000</td>
</tr>
<tr>
<td>Maximum depth (m)</td>
<td>23.9</td>
<td>9.66</td>
<td>10.5</td>
<td>58.82*</td>
</tr>
<tr>
<td>Time and duration of stratification</td>
<td>Summer, 7 months</td>
<td>Summer, 6 months</td>
<td>Summer, 4-5 months</td>
<td>—</td>
</tr>
<tr>
<td>Time of overturn</td>
<td>July</td>
<td>May</td>
<td>early May</td>
<td>—</td>
</tr>
<tr>
<td>Average annual inflow (ML)</td>
<td>1,645,000</td>
<td>162,490</td>
<td>31,450</td>
<td>925,000</td>
</tr>
<tr>
<td>Retention time: volume/average inflow</td>
<td>2 years</td>
<td>2 years</td>
<td>1-2 years</td>
<td>4-32 years*</td>
</tr>
<tr>
<td>Annual rainfall on catchment (mm)</td>
<td>720-1,159</td>
<td>508-1,016</td>
<td>1,092</td>
<td>650-1,500</td>
</tr>
<tr>
<td>Number of tributaries</td>
<td>5 major</td>
<td>2 major</td>
<td>1 major</td>
<td>6 major</td>
</tr>
<tr>
<td>Mean depth (m)</td>
<td>23.9</td>
<td>9.66</td>
<td>10.5</td>
<td>58.82*</td>
</tr>
<tr>
<td>Main use</td>
<td>irrigation 92% irrigation 8% urban</td>
<td>urban irrigation</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Date of construction (completed)</td>
<td>1956</td>
<td>1962</td>
<td>1968</td>
<td>1978</td>
</tr>
<tr>
<td>Commencement of study</td>
<td>1968</td>
<td>1971</td>
<td>1968</td>
<td>1977</td>
</tr>
</tbody>
</table>

* When filled.

algae including 70 desmid species). Silicon levels do not fluctuate more than about 1 mg l⁻¹ in a year, and diatoms rarely achieve dominance in the phytoplankton. The commonest diatoms are Melosira granulata var. angustissima, Synedra nana, Rhizosolenia eriensis and Attheya zachariais; the most common green algae are the small desmids Staurastrum, Cosmarium and Staurodesmus and the small colonial forms Sphaerocystis and Dictyosphaerium. Certain of the flagellates are dominant at various times of the year e.g. Dinobryon divergens and D. sertularia in winter, the dinoflagellates Ceratium hirundella and Peridinium volzii in late summer, and cryptomonads in autumn and winter. Except for one occasion following a drought, blue-green algae have not been significant in the phytoplankton.

The zooplankton is diverse (Table 38), but never abundant except for a purple ciliate, cf. Climacostomum sp. (R. Croome, personal communication) which becomes abundant in summer, particularly in northern bays.

**The future.** Clearly the effect on water quality of the continued development of recreation on and around the lake must be assessed. In December 1977, consultants were engaged to make this assessment so that a planning policy for the lake and its environs could be developed. Water quality studies will continue on completion of this study, but
Table 37: Major chemical characteristics of Lake Eildon, Lake Eppalock, Tarago Reservoir, Mitta Mitta River upstream of Dartmouth, and Dartmouth Dam

<table>
<thead>
<tr>
<th>General information</th>
<th>Eildon</th>
<th>Eppalock</th>
<th>Tarago</th>
<th>Mitta</th>
<th>Dartmouth</th>
</tr>
</thead>
<tbody>
<tr>
<td>Number of samples</td>
<td>52</td>
<td>46</td>
<td>75</td>
<td>28</td>
<td>7</td>
</tr>
<tr>
<td>Cation dominance</td>
<td>Na&gt;Mg&gt;Ca</td>
<td>Na&gt;Mg&gt;Ca</td>
<td>Na&gt;Mg&gt;Ca</td>
<td>Na&gt;Mg&gt;Ca</td>
<td>Na&gt;Mg&gt;Ca&gt;K</td>
</tr>
<tr>
<td>Anion dominance</td>
<td>HCO₃&gt;Cl&gt;SO₄Cl&gt;</td>
<td>HCO₃&gt;SO₄Cl&gt;</td>
<td>HCO₃&gt;SO₄Cl&gt;</td>
<td>HCO₃&gt;SO₄Cl&gt;</td>
<td>HCO₃&gt;Cl&gt;SO₄</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Chemical characteristic (unit)</th>
<th>mean or (max.)</th>
<th>range or (max.)</th>
<th>mean or (max.)</th>
<th>range or (max.)</th>
<th>range or (max.)</th>
<th>mean or (max.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>6.9</td>
<td>6.6-7.5</td>
<td>7.9</td>
<td>7.2-8.1</td>
<td>7.3</td>
<td>6.9-7.5</td>
</tr>
<tr>
<td>Total dissolved solids* (mg l⁻¹)</td>
<td>42</td>
<td>34-54</td>
<td>276</td>
<td>174-405</td>
<td>78</td>
<td>67-83</td>
</tr>
<tr>
<td>Colour (Pt-Co)</td>
<td>15</td>
<td>5-50</td>
<td>20-120</td>
<td>55</td>
<td>30-80</td>
<td>5-80</td>
</tr>
<tr>
<td>Turbidity (JTU)</td>
<td>4</td>
<td>1-12</td>
<td>2-50</td>
<td>3</td>
<td>1-8</td>
<td>1-22</td>
</tr>
<tr>
<td>Hardness (CaCO₃) (mg l⁻¹)</td>
<td>16</td>
<td>13-21</td>
<td>102</td>
<td>66-148</td>
<td>24</td>
<td>20-32</td>
</tr>
<tr>
<td>Alkalinity CaCO₃ (mg l⁻¹)</td>
<td>18</td>
<td>16-22</td>
<td>67</td>
<td>44-96</td>
<td>21</td>
<td>16-30</td>
</tr>
<tr>
<td>Chloride (mg l⁻¹)</td>
<td>6.2</td>
<td>4-12</td>
<td>100</td>
<td>48-140</td>
<td>26</td>
<td>21-33</td>
</tr>
<tr>
<td>Chlorophyll-a (µg l⁻¹)</td>
<td>3.2</td>
<td>(8.2)</td>
<td>2.5</td>
<td>(11.1)</td>
<td>5.2</td>
<td>(23)</td>
</tr>
<tr>
<td>Particulate carbon (mg l⁻¹)</td>
<td>—</td>
<td>—</td>
<td>0.56</td>
<td>(2.3)</td>
<td>0.7</td>
<td>(1.7)</td>
</tr>
<tr>
<td>Silicon (Si) (mg l⁻¹)</td>
<td>—</td>
<td>1.6-3.9</td>
<td>2.3-3.8</td>
<td>—</td>
<td>3.3-6.0</td>
<td>5.5-11.6</td>
</tr>
<tr>
<td>Phosphate P (mg l⁻¹)</td>
<td>0.025 (0.09)</td>
<td>0.042</td>
<td>—</td>
<td>—</td>
<td>0.01-0.12</td>
<td>0.06</td>
</tr>
<tr>
<td>total (mg l⁻¹)</td>
<td>—</td>
<td>0.027</td>
<td>0.024</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>particulate (mg l⁻¹)</td>
<td>—</td>
<td>0.016</td>
<td>0.008</td>
<td>—</td>
<td>—</td>
<td>—</td>
</tr>
<tr>
<td>Nitrate (N) (mg l⁻¹)</td>
<td>0.03 (0.10)</td>
<td>0.26</td>
<td>0.17</td>
<td>—</td>
<td>&lt;0.002-</td>
<td>0.04</td>
</tr>
<tr>
<td>Filterable iron† (mg l⁻¹)</td>
<td>0.042</td>
<td>0.03</td>
<td>0.10</td>
<td>0.6</td>
<td>(1.6)</td>
<td>0.1-4.2</td>
</tr>
<tr>
<td>surface (mg l⁻¹)</td>
<td>—</td>
<td>—</td>
<td>(5.0)</td>
<td>(12.0)</td>
<td>—</td>
<td>(4.30)</td>
</tr>
<tr>
<td>bottom (mg l⁻¹)</td>
<td>—</td>
<td>—</td>
<td>&lt;0.002</td>
<td>(0.19)</td>
<td>&lt;0.002-0.08</td>
<td>&lt;0.002</td>
</tr>
<tr>
<td>Filterable manganese† (mg l⁻¹)</td>
<td>—</td>
<td>—</td>
<td>(2.1)</td>
<td>(2.14)</td>
<td>—</td>
<td>(0.52)</td>
</tr>
</tbody>
</table>

* Electrical conductivity x 0.6.
† Dissolved colloidal iron or manganese.
‡ Daily readings (N = 75).

will be extended to include more detailed examination of particular problems.

Lake Eppalock

**Background.** Lake Eppalock (Fig. 42), situated in central Victoria, impounds the waters of the Campaspe and Coliban Rivers and two minor inflows. Constructed mainly for irrigation purposes, its outflow enters the irrigation system via either the Waranga Channel or the Murray River below its confluence with the Campaspe at Echuca. A small percentage of the outflow is used to supplement urban supplies to the city of Bendigo, 25 km west. Hydraulic turbines operated by hypolimnion discharge provide power to pump the water to the city via 4.5 km of 825 mm and 18.5 km of 900 mm concrete-lined mild steel pipe-
Table 38 Zooplankton of Lake Eildon, Lake Eppalock, Tarago Reservoir and Dartmouth Dam.
Index: +, present; —, not recorded.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Eildon¹</th>
<th>Eppalock¹</th>
<th>Tarago²</th>
<th>Dartmouth¹</th>
<th>Taxon</th>
<th>Eildon¹</th>
<th>Eppalock¹</th>
<th>Tarago²</th>
<th>Dartmouth¹</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>PROTOZOA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td><strong>CRUSTACEA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Chaos diffluens (=Amoeba proteus)</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>Diaphanosoma excisum</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Arcella sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>D. unguiculatum</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Centropyxis sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>Daphnia carinata sensulato</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Diffugia sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>D. lumholtzi</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Euglypha sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>D. thomsoni</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Coleps sp.</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Cladocera</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Actinosphaerium sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>Chenodorus sphaericus</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>purple ciliate cf.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>Moina tenuicornis</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Climacostomum sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>M. micrura</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Vorticella sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>B. longirostris</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td><strong>CNIDARIA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Craspedacusta sowerbyi</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Ceriodaphnia cornuta</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>C. quadrangula</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>ROTIFERA</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brachionus calyciflorus</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>C. lucasi</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>B. quadridentatus</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>C. fluvialis</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>B. diversicornis</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Boeckella fluvialis</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Keratella quadrata</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>C. triarticulata</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>australis</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>B. minuta</td>
<td>+</td>
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<tr>
<td>K. valga</td>
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<td>—</td>
<td>—</td>
<td>—</td>
<td>B. symmertica</td>
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<tr>
<td>K. tropica</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>Calamoecia ampulla</td>
<td>+</td>
<td>+</td>
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</tr>
<tr>
<td>K. cochlearis</td>
<td>+</td>
<td>—</td>
<td>—</td>
<td>—</td>
<td>C. lucasi</td>
<td>+</td>
<td>+</td>
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<tr>
<td>Euchlanis sp.</td>
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<td></td>
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<tr>
<td>Asplanchna brightwelli</td>
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<td>+</td>
<td>+</td>
<td>+</td>
<td>Cyclopoida</td>
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<tr>
<td>Polyarthra sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td></td>
<td></td>
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<tr>
<td>Hexarthra sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Mesocyclops leuckarti</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Conochilus dossuaris</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Mesocyclops sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>C. unicicornis</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>M. decipiens</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Synchaeta sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Microcyclops varicans</td>
<td>+</td>
<td>—</td>
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<tr>
<td>Rotaria cf. citrinus</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Microcyclops sp. A</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Philodina sp.</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Microcyclops sp. B</td>
<td>+</td>
<td>—</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Filinia sp.</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>Notocyclops sp.</td>
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<td>+</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Lecane sp.</td>
<td>—</td>
<td>—</td>
<td>+</td>
<td>+</td>
<td>Tropocyclops sp.</td>
<td>+</td>
<td>+</td>
<td>+</td>
<td>+</td>
</tr>
</tbody>
</table>

¹ Identification by R. J. Shiel (personal communication).
² Identification by D. W. Morton (personal communication).
The friction head along the line increased by 50 per cent in the first twelve months, by 80 per cent in four years, and now, after twelve years, has reached 215 per cent of the original value. To overcome the resulting loss of water to Bendigo, auxiliary pumps have become necessary. Visual inspection inside the pipe showed that a thin, rough layer of iron slime had been deposited on the pipeline walls by iron bacteria, *Sphaerotilus* sp., which were later successfully isolated from the slime deposit (P. A. Tyler, personal communication). Certain areas of the water distribution system in Bendigo also suffered from iron bacterial slime, and consumer complaints of 'dirty water' were common in summer.

It was because of the use of the reservoir for supplementary supplies to Bendigo that in 1971 it was decided to monitor the bacteriological quality of its surface waters.

**Physico-chemical features.** Some important morphometric and other physical statistics are given in Table 36, and Table 37 gives some chemical characteristics. Like Lake Eildon, Lake Eppalock is a
warm monomictic lake. Holomixis usually occurs in May; stratification is pronounced and lasts for seven months. Colour and transparency undergo wide variation. This reflects the nature of catchment soils which are mostly solodic and podsolic with highly dispersive clay subsoils. Thus, the onset of heavy rains in 1973 brought about a marked increase in the colour and turbidity of the lake water; the mean transparency as measured by Secchi disc decreased from 2.4 to 0.9 m, whilst mean turbidity increased from 3.5 to 15 JTU. It was not until the dry winter of 1977, following an equally dry year in 1976, that these levels returned to pre-1973 values.

Comprehensive chemical monitoring of nutrients, turbidity and suspended solids began in late 1975 and will continue in conjunction with bacteriological monitoring. This monitoring began because of the need to be continuously aware of the overall quality of surface waters, for there is increasing pressure from recreational use on both the reservoir itself and perimeter land. Moreover, applications to subdivide land are regularly made, and litigation often follows if opposed. Opposition on the grounds of potential deterioration to water quality must be fully substantiated. As it is, nutrient levels are slightly higher than in other storages (cf. Table 37), especially of nitrate-nitrogen (mean 0.3 mg l\(^{-1}\)).

Also in 1975, detailed investigation began of the effects of thermal stratification and discharge regime on hypolimnion iron and manganese concentrations. Samples were collected adjacent to the outlet tower which, with multiple-level offtakes, operates mainly with all ports below water level open at any one time. Bottom ports are always open. Results showed that dissolved iron and manganese reached high levels in the hypolimnion in summer; destratification of Eppalock was therefore carried out on an experimental basis in March 1977.

**Biological features.** As in Lake Eildon, bacteriological monitoring began before biological monitoring in the wide sense, but transparency readings made during bacteriological sampling enable some interesting comparisons of surface water quality before and after the flood years of 1973, 1974 and 1975. The early bacteriological results demonstrated the potential hazard to drinking water quality of a picnic area adjacent to the outlet tower, and this was consequently closed to the public.

Such biological monitoring as has taken place indicates that phytoplankton standing crops, measured by chlorophyll-\(a\) concentrations, were very low (mean 2.5 \(\mu g\) l\(^{-1}\)) during the period of high turbidity and low transparency but, since November 1977, have increased to a mean of 14.2 \(\mu g\) l\(^{-1}\). Examination of phytoplankton trawls usually revealed species which, by their size and shape, for example Microasterias hardyi and Closterium aciculare, when grazed by microcrustaceans would be rather like eating pineapples without first peeling them. The minor inflowing streams to the east are more saline (range 750 to 2,000 mg l\(^{-1}\) total dissolved solids), and, as a result, a diatom more typical of brackish water, *Campylodiscus* sp., has appeared in trawls from adjacent open water. However, in general the phytoplankton is not diverse and displays no unusual features. *Botryococcus braunii* often becomes dominant in late summer during periods of calm water when it tends to float to the surface. *Cyclotella* species including *C. stelligera* are common, also
Melosira distans and occasionally Microcystis sp. (the same white variant as in Eildon).

The zooplankton (Table 38) is not as diverse as in Lake Eildon, but all major groups are represented. The zooplankton was particularly abundant during the period of high turbidity.

The future. Monitoring of all aspects of water quality in Lake Eppalock will continue for several reasons. The water will always be needed by Bendigo for urban supply and there is every possibility of increased demand. Thus the lake must be protected from faecal contamination, unnecessary inputs of turbidity and suspended solids, and high nutrient loads. Destratification has been shown to reduce effectively hypolimnion iron and manganese levels: time will tell whether this measure will not only increase the capacity of the Bendigo pipeline but also reduce the amount of organic material entering the urban distribution system.

Tarago Reservoir

Background. Tarago Reservoir, on the Tarago River 115 km east of Melbourne (Fig. 42), is a major source of domestic water supplies to the Mornington Peninsula and Westernport regions. These receive no direct treatment, and consequently no recreation, save fishing, is permitted on or around the storage. There are two outlets, a pump 3 km upstream of the dam delivering into a system of open aqueducts and siphons, and an outlet tower at the dam wall with multiple offtake levels. Two of these offtakes were used first early in 1978 when the Westernport pipeline was commissioned. Chemical (including chlorophyll-a) and bacteriological monitoring began soon after the storage filled (1968); detailed studies commenced somewhat later.

Physico-chemical features. Some of the more important physico-chemical features of the reservoir are indicated in Tables 36 and 37.

Early complaints of hydrogen sulphide in supplies from the bottom intake indicated the onset of stratification in the reservoir, but it was not until December 1974 that regular monitoring of temperature and dissolved oxygen began. The monitoring was part of a project, funded by the Australian Water Resources Council, in which biologists and engineers combined to study the effects on water quality of artificial aeration and destratification of Tarago Reservoir. The project was completed in June 1977. Leaving aside induced changes, results have shown that Tarago Reservoir stratifies in summer, with natural holomixis commencing in early May. Under normal conditions it has been found that dissolved iron, manganese and silicon are recycled from the bottom muds when the hypolimnion is deoxygenated. The dissolved iron and manganese concentrations are linearly related to dissolved oxygen concentration at levels below 4 per cent saturation. Attempts are now being made to measure the changes to light attenuation and primary production induced by artificial destratification and also to determine the origin of the dissolved organic colour in the stored waters and its relationship to catchment vegetation and soils.

Biological features. The standing crop of phytoplankton is low (mean chlorophyll-a, 5.2 μg l⁻¹), and may be limited by zooplankton grazing. The phytoplankton is not particularly diverse compared to, for example, Lake Eildon,
but diversity may increase as the reservoir ages. In summary: *Botryococcus braunii* and *Microcystis aeruginosa* often appear dominant in the surface waters during periods of calm; small desmids such as certain species of *Staurastrum*, *Staurodesmus* and *Cosmarium* are becoming more common in the summer; *Synura*, *Dinobryon* and occasionally *Peridinium* are the most common flagellates; and the only diatom of abundance in the phytoplankton is *Melosira distans*. *Sphaerocysts* sp. is the most commonly recorded plant in trawls.

The zooplankton (Table 38) comprises at least three cladocerans (*Daphnia* sp., *Ceriodaphnia* sp. and *Bosmina* sp.), several copepods (*Calamoecia lucasi*, *Boeckella symmetrica*, *Mesocyclops decipiens*), occasional ostracods, several rotifers (mainly *Keratella cochlearis*, *Polyarthra* sp.), ciliated protozoans, rhi-zopods, and *Chaoborus* larvae.

**The future.** It is possible that artificial destratification may become a permanent water quality management technique in Tarago now that the Western-port pipeline is in use. Iron and manganese levels in the hypolimnion can be reduced by this technique and so reduce optimum conditions for growth of bacterial slime deposits in pipelines. However, more work is needed concerning the effect of destratification on the organic matter responsible for water
colour. The resulting alteration to optical properties, and hence light attenuation and primary production, may be highly significant when water quality is evaluated for future domestic use.

Dartmouth Dam

**Background.** Dartmouth Dam (Figs. 21 and 42), constructed in a V-shaped gorge on the Mitta Mitta River (Plate XV), a tributary of the River Murray, when completed will retain the deepest water storage in Australia (170 m). The 4 million megalitre reservoir will increase the seasonal and long-term carry-over capacity of the River Murray system and improve its regulation. The project includes the construction of a 150 MW power station and regulating pondage below the dam. Two outlets have been constructed each with an intake tower. The sill of the lower is 121 metres below full supply level, and that of the upper, 62 metres; both will be well below the thermocline. Provision has been made in the design of the intake structure for the high-level outlet to allow construction of an extension for selected draw-off of epilimnion water in the summer months should this be deemed necessary on environmental grounds.

Intensive monitoring of the Mitta Mitta River and its major tributaries was carried out before and during construction. It began in 1973 as part of a broader environmental survey of the whole area to be affected; further details are given in Chapter 17. The actual impoundment of water in Dartmouth Dam began in November 1977, and monitoring of stored water began two weeks later (when water depth had already reached 21 m). Monitoring was undertaken in order to:

1. determine the depth to which turbid water caused by closing-off activities had settled. This would allow estimate of how long it would take before the river downstream cleared sufficiently for extraction of water for domestic purposes if required;
2. follow thermocline development, and thus be able to predict the onset of hydrogen sulphide production in the hypolimnion of a storage with a high organic load; and,
3. provide a comprehensive record of events taking place early in the life of a water storage. No such record was available for any Victorian storage.

Monitoring will continue at regular intervals as the storage fills, with emphasis on the measurement of temperature, dissolved oxygen, sulphide, iron, manganese and turbidity. The zooplankton present in deep hauls and particulate carbon over the water column will also be studied.

**Physico-chemical features.** Some actual or predicted morphometric and other features of Dartmouth Dam are indicated in Table 36. Table 37 gives chemical data on the Mitta Mitta River above the storage for the period 1974-77, and on water in the storage for the period November 1977 to April 1978.

Hydrogen sulphide appeared in the outlet to the river only three weeks after dam closure and, as some of the water was to augment dwindling water supplies to Dartmouth township, it was decided to aerate the water column artificially in
the vicinity of the dam wall. Before artificial aeration, measurements at depth in the storage showed filterable iron and manganese concentrations of 4.3 mg l⁻¹ and 0.5 mg l⁻¹, respectively. The sulphide concentration was 0.7 mg l⁻¹ at a depth of 15 m when first detected. In the river below the dam outlet, the sulphide, manganese and iron dissipated progressively—in that order. Aeration was only partially successful: temperature destratification was achieved almost immediately, but hydrogen sulphide production was affected only to the extent of lessening its impact near the outlet. The remainder of the storage was anoxic below 3.5 m by the end of February. However, aeration probably prevented the formation of a monimolimnion (a permanently stagnant layer with a greater concentration of solutes than found in overlying layers), a phenomenon predicted by Tyler (1974) as likely to occur because of the decomposition of organic matter and the nature of Dartmouth's basin morphometry. A slight rise in temperature (0.3 °C) in the bottom 3 m after six weeks may have been related to the onset of meromixis.

Provision has been made for the future use of destratification by aeration as the reservoir fills. Using this, it may be possible to prevent corrosion of the stainless steel bulkhead on the low level outlet, and, near the high level outlet, to increase the temperature of water discharged and so mitigate the effect of an altered temperature regime on the stream fauna below the dam. Biological features. The first representative of lake phytoplankton to appear in trawls was the colonial flagellate Synura sp., but this appearance was short-lived and was immediately succeeded by a minor bloom of Anabaena flos-aquae. Volvox sp. achieved numerical dominance after twelve weeks. Zooplankton began to appear in abundance eight weeks after filling. It mainly comprised Ceriodaphnia sp., the colonial rotifer Conochilus sp., and very large (3.5 mm) individuals of Daphnia carinata with large helmets. It is interesting to note that several specimens of Daphnia, immediately after having been hauled from the anoxic zone, had air bubbles trapped under their carapace.

Acknowledgments
The Chairman of the State Rivers and Water Supply Commission and the Commissioners of the River Murray Commission are thanked for permission to use the material contained in this chapter. Mr R. J. Shiel, Department of Zoology, University of Adelaide, and Mr D. W. Morton, Department of Zoology, Monash University, are thanked for identifying zooplankton.

Reference
PART VII

Some special aquatic environments
Introduction
Farm dams are ubiquitous throughout settled areas, yet they are amongst the least studied of Australia's water resources. This is surprising, for besides being academically interesting, they offer great potential for commercial and recreational aquaculture. These two aspects are interwoven in that farm dams are relatively simple ecosystems: a few physical factors dominate, biological diversity may be low, and food chains are short. Despite ecological simplicity, however, biological productivity may be high.

An early study was by Weatherley (1958a,b) who assessed the potential for fish culture of some Tasmanian dams. Later work has been on fish and particularly crayfish culture (e.g. Morrissy, 1970, 1974), on basic studies of physico-chemical features and zooplankton, and more recently, on these aspects plus studies on nutrients and littoral invertebrates. Thus, while data are available on some aspects, mainly the important and distinctive physico-chemical milieu, little is known of many biological features. Hence it is difficult to present a balanced summary; furthermore, there is much variation between dams so that it is not easy to generalise.

Dam types and distribution
There may be over 400,000 farm dams in Australia. Many modes of construction occur and terms used for different types vary between states. Nevertheless, there are four basic, though intergrading types: gully dams, hillside dams, excavated tanks, and raised-edge tanks. Typology is partly expressed in geomorphology, distribution and some physico-chemical features.

(a) Gully dams are formed by a wall constructed across a depression, gully or watercourse (Fig. 44a), and hence are typically in hilly terrain. With storage/excavation ratios up to 10, they provide economical water storage for irrigation. Thus some are large (1 x 10⁶ m³), but most are two or three magnitudes smaller and used directly or indirectly (via pipes and troughs) for watering stock. Most are roughly triangular in shape with the deepest part near the wall and extensive shallows up gully. Water turbidity is not as high as in other dam types. This type is particularly common in eastern N.S.W. (Fig. 45), in the Adelaide Hills, in the hills between Perth and Albany, and in many other places.

(b) Hillside dams (excavated tanks in W.A.) have two or three walls on a hill-slope with no defined watercourse (Fig. 44b). Storage/excavation ratios can be up to 5, and water enters by seepage, or more usually by surface runoff. Underwater slopes along the walls are steep, but the slope along the hillside edge is usually gentle, so that a trough of greatest depth occurs near the far wall. Hillside dams tend to be blunt-crescentic, or rectangular in shape. They are the
Some special aquatic environments

Fig. 44 Farm dam types.

(a) GULLY DAM

(b) HILLSIDE DAM

LEGEND:

--- Contour

--- Water level

■ Dam wall

Excavation

(c) EXCAVATED TANK (or soakage dam)

(d) RAISED-EDGE DAM

major dam type in the most intensively used pastoral and agricultural areas of coastal and inland Australia. At least in W.A., the average volume is $1.5 \times 10^9$ m$^3$ with an increase inland and northwards to offset losses by evaporation (W.A. Department of Agriculture, personal communication). Most are used for intensive direct watering of stock, so that edges are much disturbed and the water is very turbid. In humid areas, e.g. in north-eastern N.S.W., some are much bigger and used for limited irrigation.

(c) Excavated tanks are constructed by digging a square to rectangular shaped hole in flat ground. Storage is below ground level so that storage/excavation ratios $< 1$. Unlike other types, there may be considerable input of groundwater, so much so that in Western Australia they are called soakage dams. All sides shelve steeply to a
Farm dams

flat floor (Fig. 44c). They are restricted to areas where the water table (at least seasonally) is very close to the surface.

(d) Raised-edge tanks have a continuous wall so that water must be pumped from beyond the dam. Because of large surface area: volume ratios, there is very high potential evaporative loss, and thus they are restricted to special situations such as flat ground near a creek in wet coastal areas, or near a bore inland. Shape varies and storage may be either below or above ground level. In all subtypes, the littoral slopes steeply to a flat bottom. They are often called ring tanks or turkey's nests.

Physico-chemical features

Light penetration. Most farm dams are highly turbid; Secchi disc readings of < 100 cm, and often < 20 cm are typical (Table 39). Such low values largely reflect interactions between soil type, water chemistry, and stock usage (Table 40). Thus, Morrissy (in press) has shown that clay turbidity negatively correlates with salinity, and positively with the ratio of monovalent to divalent cations. Further, montmorillonite gives higher concentrations of smaller particles in suspension than kaolinite clay, other factors being equal. The amount of suspended clay is considerably increased by stock trampling near edges, and by wave action (Morrissy, 1974).

High concentrations of dissolved organic matter, derived from direct organic pollution, have a marked effect on reducing turbidity. However, dissolved organic matter associated with abundant plant growth in and near the dam (see Tables 39 and 40) may render the water humic and reduce Secchi disc readings.

New dams and those with bare littorals (Table 39) usually have higher turbidities. This is because of the exposure of underlying and surrounding substrata (Tait, 1976) or, in recently excavated dams, because concentrations of salts and organic matter are low. Dams supplied by surface runoff rather than underground water are also likely to have higher turbidities.

Whatever the average light regime, seasonal changes occur because of phytoplankton blooms, seasonal changes in use by stock (Morrissy, 1974; Walker, 1974), rainfall variability and associated flood inflows (Weatherley, 1958a; Timms, 1970a; see Fig. 46), and changes in salinity due to evaporation (Morrissy, in press).

The highly turbid conditions of farm dams have important biological reper-
Table 39  Secchi disc (S.D.) readings in some farm dams and reservoirs

<table>
<thead>
<tr>
<th>Area</th>
<th>Number of dams</th>
<th>Mean S.D. reading (cms)</th>
<th>Range (cms)</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>South-west W.A.</td>
<td>13</td>
<td>11</td>
<td>2-30</td>
<td>Morrissy, 1970</td>
</tr>
<tr>
<td>South-central Vic.</td>
<td>28</td>
<td>21</td>
<td>1-120</td>
<td>Tait, 1976</td>
</tr>
<tr>
<td>North-east N.S.W.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>all types of farm dams</td>
<td>39</td>
<td>66</td>
<td>1-191</td>
<td>Timms, 1970b (reanalysed)</td>
</tr>
<tr>
<td>new with bare littorals</td>
<td>15</td>
<td>36</td>
<td>1-96</td>
<td></td>
</tr>
<tr>
<td>scanty vegetation</td>
<td>18</td>
<td>95</td>
<td>12-191</td>
<td></td>
</tr>
<tr>
<td>well vegetated</td>
<td>6</td>
<td>66</td>
<td>26-127</td>
<td></td>
</tr>
<tr>
<td>small reservoirs</td>
<td>23</td>
<td>117</td>
<td>9-283</td>
<td></td>
</tr>
<tr>
<td>Sydney area N.S.W.</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>large reservoirs</td>
<td>6</td>
<td>392</td>
<td>40-900</td>
<td>Jolly, 1966</td>
</tr>
</tbody>
</table>

cussions (Bayly and Williams, 1973). For instance, marron (*Cherax tenuimanus*) will feed by day and are protected from predatory birds by high turbidity. Further, turbid waters restrict light penetration so that primary production may be suppressed and heating is restricted to the uppermost strata.

**Temperature.** Surface temperatures tend to reflect prevailing air temperatures, so that seasonal range is largely determined by the climatic regime expressed by such factors as latitude, altitude and coastal proximity (Table 41). In the same area, turbid dams have higher surface temperatures than clear ones (Walker, 1974). Daily heating cycles are damped versions of air temperature variations, but with a lag of 1-3 hours. However, the lag is sometimes longer so that surface temperatures may be higher than air temperatures for many hours or days (Bayly and Williams, 1973).

Studies on temperature profiles show that some dams are always isothermal (e.g. dams in southern Victoria; Tait,

Table 40  Some physico-chemical features of six dams at Copping, Tasmania*

<table>
<thead>
<tr>
<th>Dam</th>
<th>Soil type</th>
<th>Relative stock usage</th>
<th>Relative amount of vegetation</th>
<th>Mean turbidity of JTUs</th>
<th>Nitrate in ppm</th>
<th>Phosphate in ppm</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>clay</td>
<td>++++</td>
<td>+</td>
<td>226</td>
<td>6.65</td>
<td>8.3</td>
</tr>
<tr>
<td>B</td>
<td>clay</td>
<td>+</td>
<td>+++</td>
<td>134</td>
<td>0.66</td>
<td>0.6</td>
</tr>
<tr>
<td>C</td>
<td>basalt</td>
<td>+</td>
<td>+</td>
<td>5</td>
<td>trace</td>
<td>trace</td>
</tr>
<tr>
<td>D</td>
<td>clay</td>
<td>++</td>
<td>+++</td>
<td>139</td>
<td>4.43</td>
<td>2.5</td>
</tr>
<tr>
<td>E</td>
<td>clay</td>
<td>++</td>
<td>++</td>
<td>61</td>
<td>0.89</td>
<td>1.9</td>
</tr>
<tr>
<td>F</td>
<td>clay</td>
<td>++</td>
<td>++</td>
<td>54</td>
<td>0.66</td>
<td>2.1</td>
</tr>
</tbody>
</table>

Table 41  Surface temperature ranges in farm dams in southern Australia

<table>
<thead>
<tr>
<th>Area</th>
<th>Number of localities</th>
<th>Mean range (°C)</th>
<th>Latitude (°S)</th>
<th>Comment</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>South-west W.A. near Boscabel</td>
<td>2</td>
<td>8-30</td>
<td>34</td>
<td>160 km inland</td>
<td>Morrissy, 1974</td>
</tr>
<tr>
<td>Southern Tasmania near Copley</td>
<td>6</td>
<td>7-18</td>
<td>43</td>
<td>near sea</td>
<td>Walker, 1974</td>
</tr>
<tr>
<td>Near Melbourne, Vic.</td>
<td>28</td>
<td>9-3-23-7</td>
<td>38</td>
<td>near sea</td>
<td>Tait, 1976</td>
</tr>
<tr>
<td>N.S.W. Central Coast</td>
<td>6</td>
<td>12-7-26-4</td>
<td>33</td>
<td>near sea</td>
<td>Timms, 1970b</td>
</tr>
<tr>
<td>N.S.W. Far North Coast</td>
<td>9</td>
<td>15-5-29-7</td>
<td>29</td>
<td>near sea</td>
<td>Timms, 1970b</td>
</tr>
<tr>
<td>N.S.W. Northern Highlands</td>
<td>14</td>
<td>9-0-23-0</td>
<td>29-31</td>
<td>c.1,000 m asl.</td>
<td>Timms, 1970b</td>
</tr>
</tbody>
</table>

1976), but many stratify daily, even in winter, or for weeks or months at a time but only in summer. In south-western Western Australia, daily stratification in summer can be very strong (e.g. 10°C difference in 1 m; see Fig. 47). Daily bottom temperatures are determined by air temperatures just before insolation recommences so that profiles vary daily and seasonally. The deeper, well-protected dams in N.S.W. studied by Timms (1970a), behave like reservoirs with holomixis in winter and persistent stratification in summer (Fig. 46).

Possibly between these three extreme types occur intermediates which sometimes stratify daily or which intermittently stratify for days or weeks. For instance, from long-term continuous records for many farm dams in south-western Western Australia, Morrissy (1976, in press) noted that at the height of summer some stratification in deeper waters persists overnight due to warmer minimum air temperatures; complete isothermy is initiated only by ‘cool changes’. Early or late in summer, although mid-afternoon heating is still

Fig. 46  Seasonal changes in water level, temperature, surface and bottom dissolved oxygen content, pH, Secchi disc reading and rainfall for a small reservoir (from Timms, 1970a).
strong, lower minimum air temperatures and early morning easterly winds bring about daily isothermy by sunrise.

Clearly, the stratification regime varies between dams and seasonally. Stratification is greatly enhanced by high values for turbidity, but protection from wind and increased depth also assist. Stratification has important limnological consequences in that it influences some chemical parameters and many biological features.

**Oxygen.** Though oxygen concentrations tend to be variable, even between adjacent dams (Walker, 1974), relative concentrations tend to be higher in summer than in winter, in the afternoon than in the morning, and at the surface than on the bottom (Bayly and Williams, 1973). Summer hypolimnetic anoxia associated with strong daily or seasonal stratification (Fig. 46) is characteristic of many dams (Morrissy, 1974; Walker, 1974; Parker and Wallis, 1977; Timms, 1970a).

**Fluctuations in area and volume.** As might be expected from the use made of dams, areas and volumes fluctuate considerably, even in the equable climate of Tasmania (see Weatherley, 1958a). In regions of very high evaporation in South Australia and Western Australia, dams may need to be 4 m deep in order to retain some water all year round (Carder, 1970). In such regions, minima for dam area and volume are likely to be < 10 per cent of maxima even in normal years. In the Western Australian wheat belt where evaporation is lower (80-160 cm per annum), there is 50 per cent loss from full level over summer (Morrissy, in press). These fluctuations,
besides making the establishment of littoral vegetation difficult or impossible, undoubtedly influence water chemistry.

**Total dissolved solids.** The concentration of total dissolved solids is typically <3,000 ppm, and usually <500 ppm. Total solids may be much higher (x 4) owing to wind action freshly suspending fine particles (Morrissy, 1974). Salt supply is mainly by leaching from the catchment (Morrissy, 1974), but dams near the sea probably receive considerable supplies of salts via the atmosphere (Tait, 1976). Dissolved organic matter from lush pastures may add significantly to concentrations of dissolved solids (Timms, unpublished).

Whatever the origin of total dissolved solids, concentrations fluctuate considerably because of volume variations (e.g. by 96 per cent of minimum value in southern Tasmania: Walker, 1974), and are influenced by evaporation regimes (e.g. Tait, 1976) and water renewal rates (Timms, 1970c). In some areas, such as in south-western Western Australia, overflow is particularly important in preventing year to year increase in salinity (Morrissy, in press).

**Ionic composition.** Analytical data for some 150 farm dams in southern Australia (e.g. Morrissy, 1974; Walker, 1974; Tait, 1976; Timms, 1970b) indicate that ionic composition is somewhat variable even in a given district, but that typically Na⁺ and Cl⁻ dominate, as do these ions in Australian inland waters as a whole (Williams, 1967). Hence it is not surprising that no general relationship between ionic composition and soil parent material exists (Currey, 1976), though in isolated cases some correlations may be apparent as at Copping, Tasmania, where one dam on basalt had much more Ca⁺⁺ and Mg⁺⁺ and extra SO₄⁻⁻ than dams on clay (Walker, 1974). In eastern Australia, HCO₃⁻ and sometimes Ca⁺⁺ or Mg⁺⁺ are likely to be important only away from the coast (Tait, 1976; Timms, 1970b), but in south-western Western Australia, HCO₃⁻ is important in all dams studied by N. M. Morrissy (personal communication).

**pH.** Most farm dams are neutral to slightly alkaline, but when much dissolved organic matter is present acid conditions are likely. Variations are largely seasonal with a winter minimum and a summer maximum, or irregular and related to incidence of flood inflows (see Fig. 46). The former regime applies in southern Australia (Walker, 1974), the latter in eastern (Timms, 1970a) and presumably northern areas. Within a dam considerable spatial heterogeneity can occur, mainly during spring-summer when rates of photosynthesis may vary from one part of a dam to another (Table 42).

Although values for total dissolved solid concentrations and pH are of some intrinsic interest, there is no evidence that the range of values reported affects the biological features of farm dams in any way. However, ionic composition may be important; for example, concentrations of Ca⁺⁺ positively correlated with marron biomass in dams studied by Morrissy (1970).

**Nutrients.** Most farm dams have very high concentrations of nitrates and phosphates (Table 40); these are associated with varying intensities of agriculture on catchments (Moore, 1974) and more specifically to fertiliser use (Currey, 1976) or stock watering (Walker, 1974). However, few are obviously polluted, probably because high turbidity
Table 42 Spatial variation in pH in St Lucia Pond, Queensland*

<table>
<thead>
<tr>
<th>Date</th>
<th>Open water</th>
<th>Weedbeds</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bottom</td>
<td>Surface</td>
</tr>
<tr>
<td>30 July</td>
<td>8.0</td>
<td>8.1</td>
</tr>
<tr>
<td>2 Oct.</td>
<td>8.4</td>
<td>9.1</td>
</tr>
<tr>
<td>5 Nov.</td>
<td>8.0</td>
<td>8.0</td>
</tr>
</tbody>
</table>

* From Timms, 1965.
† H = Hydrilla verticillata.
‡ L = Limnanthemum indicum.

combined with limited macrophyte growth due to varying water levels and stock trampling limits nutrient exploitation. It might be added that many Australian lakes and reservoirs also have elevated nutrient levels, but to a lesser degree (Williams and Wan, 1972); in these nuisance macrophyte growth is also limited by astatic water levels, among other factors.

**Biological features**

**Phytoplankton.** There has been no systematic study of phytoplankton in farm dams; only blooms (Weatherley, 1958a; Timms, 1970a) or the presence of toxic algae (e.g. Aplin, 1967) have been noted. Although, *a priori*, high turbidity might be expected to restrict numbers of individuals and perhaps diversity, this may not happen, for Weatherley (1958a) reported that large populations can occur despite high turbidity. Data on primary production are also lacking, but apparently in turbid dams (e.g. those studied by Morrissy, 1974) food chains are detritus-based on material from allochthonous sources.

**Zooplankton.** Zooplankton communities are little different from those of lakes and reservoirs and consist of a species of *Boeckella*, rarely a species of *Calamoecia*, one or two cyclopoids (usually including *Mesocyclops leuckarti*), and one or two cladocerans from the genera *Daphnia*, *Ceriodaphnia*, *Bosmina* and *Moina* (Jolly, 1966; Tait, 1976; Timms, 1965, 1970a,b). In these studies, the total number of crustacean species recorded per dam might total 10 or more, but the average momentary species composition ranged from 2.2 to 2.7 for copepods and from 1.1 to 1.6 for cladocerans. No definitive data are available for rotifers, but they can be diverse and numerous (Timms, unpublished).

Small dams may have a less diverse fauna (Tait, 1976; Timms, 1970b; Walker, 1974), but many interacting factors obscure the issue. One factor is the increasing chance of finding littoral strays in the zooplankton *sensu stricto* in small dams. Another is the presence of fish, which may preclude cladocerans, unless the water is turbid (e.g. in Tait's (1976) dams *Daphnia carinata sensu lato* was typically restricted to turbid dams).

Seasonal changes in abundance (Fig. 48) are frequently due to variation in the dominance of two to three species (Timms, 1970a) and are either regular or irregular; if the latter then they are often controlled by the passage of floods (Timms, 1970a; Weatherley, 1958a). It

seems that high turbidity itself has no adverse effect on numbers (Walker, 1974).

**Benthos.** Diversity is typically low; there were, for example, only three species in the six dams studied by Walker (1974). Often there are three to four chironomid species involved (J. Martin, personal communication). Dominant species include chironomids and oligochaetes (Walker, 1974; Weatherley, 1958a,b) and chaoborids (Timms, unpublished). Species composition and numbers of chironomids are highly variable in time and space depending on substrate type, recent changes in water level and season, among other factors (J. Martin, personal communication). Common forms include *Chironomus* spp., *Polypedilum nubifer* and *Procladius villosimanus*. Gastropods may also be important but their numbers are adversely affected by fish predation (Weatherley, 1958a). In the turbid dams in Western Australia investigated by Morrissy (1970, 1974, 1975), the marron, *Cherax tenuimanus*, is the dominant species.

Biomass may be high (60 g m$^{-2}$), but is more usually low to very low (< 5 g m$^{-2}$, Weatherley, 1958a; < 1 g m$^{-2}$, Timms unpublished). Walker (1974) attributed low values to the adverse influence of clay substrates. An additional factor, at least in stratified dams, may be hypolimnetic deoxygenation (Morrissy, 1974). In dams that fluctuate greatly in area, much benthos is killed during drawdowns (Weatherley, 1958a).

**Littoral plants.** Aston (1973) and Sainty (1973) reported on the biology of most species of aquatic macrophytes, but information on plant communities is limited. At Gloucester, N.S.W., fifteen species commonly occur in dams with *Eleocharis sphacelata* and *Ottelia ovalifolia* dominant; other recurring species are *Cyperus* spp., *Triglochin procera*, *Marsilea* sp., and *Potamogeton ochraceus* (Timms, unpublished). Weedbed growth is influenced by dam age (Tait, 1976), stock influence (Walker, 1974), and probably substrate type, water level
fluctuations and climate. Zonation is often obvious but its ecological interactions or the influence of plant location and structure on animals are largely unknown. However, work on farm dams (Tait, 1976), billabongs (Shiel, 1976), and lakes (Timms, 1973) indicates that some species associate with particular weed-bed types.

Littoral invertebrates. The edge of a farm dam may be bare earth, rocky, or sparsely or densely covered with vegetation which may be submerged, floating or emergent. These possibilities added to fluctuations in water level, and differences in water chemistry, dam age and size, and the element of chance and other factors, summate to a complex situation difficult to study. Hence little more than species lists are available and these are often difficult to interpret.

Bare littorals and open water are often dominated by notonectids, but not usually if fish are present (Tait, 1976). In turbid dams without vegetation in south-western Western Australia, about ten species (Table 43) including ostracods, corixids, chironomids, notonectids and coleopterans are present, relative numbers depending on dam age. In dams with littoral weedbeds, communities are more complex with ~25-60 species momentarily present (Timms, unpublished; Walker, 1974). Species composition is similar to that in weeded ponds and lake littorals with ostracods, cladocerans, copepods, odonatans, hemipterans, trichopterans, coleopterans and snails dominating. In Walker’s (1974) study there were seasonal changes as temperature and the extent of inundation of littoral vegetation changed. More important, though, was the deleterious effect of stock usage on diversity (via the adverse influence on macrophyte abundance) (Table 40). Walker (1974) also presented data showing that fish may affect diversity and influence community structure.

Fish. Many dams lack fish because of limited opportunities to colonise isolated waters. For instance only 40 per cent of the dams near Melbourne studied by Tait (1976) and 50 per cent of those at Gloucester, N.S.W., being studied by Timms (unpublished) have fish. The latter study suggests that large gully dams are the most likely to have fish, probably

<table>
<thead>
<tr>
<th>Dam</th>
<th>Ooze depth (cm)</th>
<th>Number of species of invertebrates</th>
<th>Total numbers of invertebrates</th>
<th>Numbers of ostracods</th>
<th>Numbers of tubificids</th>
<th>Numbers of marron</th>
</tr>
</thead>
<tbody>
<tr>
<td>4</td>
<td>7</td>
<td>8</td>
<td>58</td>
<td>1</td>
<td>5</td>
<td>59</td>
</tr>
<tr>
<td>1</td>
<td>18</td>
<td>12</td>
<td>163</td>
<td>53</td>
<td>0</td>
<td>14</td>
</tr>
<tr>
<td>2</td>
<td>20</td>
<td>7</td>
<td>841</td>
<td>219</td>
<td>196</td>
<td>5</td>
</tr>
<tr>
<td>3</td>
<td>15</td>
<td>13</td>
<td>1,089</td>
<td>137</td>
<td>265</td>
<td>1</td>
</tr>
<tr>
<td>5</td>
<td>25</td>
<td>10</td>
<td>4,025</td>
<td>3,254</td>
<td>483</td>
<td>0</td>
</tr>
</tbody>
</table>

* From Morrissy (1974) and personal communication.
† The greater the depth of allochthonous ooze, the ‘older’ the dam.
‡ Mainly Tubificidae, Ostracoda, Amphipoda, Corixidae, Notonectidae, Coleoptera, Chironomidae.
+++ These numbers are relative and result from the same sampling intensity in each dam.
because of their location on waterways. The most common species in south-eastern Australia are the mosquito fish (Gambusia affinis), pigmy perch (Nannoperca australis australis), carp gudgeon (Hypseleotris compressus), and short-finned eels (Anguilla australis occidentalis). Trout, tench, red fin, carp or Australian native species are sometimes cultured in farm dams (see Chapter 21). Clear gully dams are most suitable for the sight feeding of fish on invertebrates, whereas turbid dams encourage tactile-chemical appendage feeding of crayfish on detritus (N. M. Morrissy, personal communication).

The influence of fish on dam communities is little understood. Some, particularly the European carp, have obvious deleterious effects (Anon., 1971), but the influence of other species is less apparent. Thus Tait (1976) reported reduced mean lengths of zooplankters in dams with fish, and Walker (1974) considered the reduced diversity and relative rarity of corixids and beetles and the absence of amphipods in one of the dams studied by him was due to predation pressure by tench.

**Succession.** Farm dams change with time (e.g. Reid, 1977). The most obvious developments are growth of littoral vegetation and increased siltation. New dams in north-eastern N.S.W. take three to fifteen years to develop littoral weedbeds, provided stock are largely excluded. Then there is little apparent change for decades, but eventually, depending on dam size and catchment erodibility, littoral vegetation extends inwards as siltation and organic accumulation (largely autochthonous) occurs so that the dam tends towards swamp-land. The farmer usually prevents this by dredging so that succession is halted and recommences. In some areas, notably the wheat-sheep belt of Western Australia, siltation by an allochthonous organic matter-clay complex is a major determinant of biological succession (Morrissy, 1974).

Establishment of a plankton community can occur rapidly but populations may be unstable for a few years. Thus Weatherley (1958a) for one dam recorded an initial planktonic bloom which was probably maintained by decaying organic matter (the dam site was heavily grassed before filling), and it was then two and a half years before another bloom, presumably maintained on phytoplankton, was recorded. By contrast, excavated dams initially have low concentrations of nutrients, so that populations are more stable. In a new dam near Newcastle studied by Timms (unpublished) there was no initial bloom, and early succession was by species additions according to known dispersal powers: the first species (Boeckella minuta) arrived within weeks, and after two years most species likely to live in the dam were apparently there.

Macroinvertebrates respond to dam ageing by changes in species composition and by variations in abundance. In the relatively simple ecosystems in the Western Australian dams studied by Morrissy (1974), the latter process is far more important. In these dams the increase of allochthonous ooze with age provides additional food, but the ooze also has a large oxygen demand and hence causes increasing deoxygenation in the bottom layers of water. Over a time there is an increase in the numbers of detritivores with special breathing adaptations for stagnant water (e.g. tubificids) and a concomitant decrease in
those (e.g. marron) without such adaptations (Table 43). By contrast, in dams which develop vegetation the ecosystem becomes more complex so that species are added and very few subtracted. For instance, corixids and notonectids are pre-eminent in a recently constructed dam near Newcastle, but in an old one nearby, although they are present, other groups such as odonatans, trichopterans, dipterans, coleopterans and snails are more abundant (Timms, unpublished).

Overall, from the little data available, it seems that the major successional features of farm dams which develop vegetated littorals are similar to those in reservoirs (Rzoska, 1966) but occur in a shorter time span. However, for farm dams which remain unvegetated, food chains are often detritus-based and succession may be very rapid.

Multipurpose use
There is not much incentive to use farm dams for purposes other than water storage. Nevertheless, thoughtful and resourceful landholders can use farm dams to enhance the visual attractiveness of their land, provide home based recreation (the author knows a grazier who water-skis on his dam!) and serve as a source of gourmet food. The latter is a subject of a separate chapter (Chapter 21), but other possible uses will be considered here.

For a dam to be attractive to waterfowl, it should have extensive shallows, an irregular edge, and islands, and be partially isolated by trees and shrubs, and protected from stock (Cowling, 1967). Few dams naturally fulfil all or most of these requirements. However, it is possible in many cases, particularly with gully dams and in high rainfall areas, to design dams with these attributes and yet detract little from their primary function of storing water. State wildlife authorities encourage the modification of farm dams to make them attractive to wildlife; a showpiece is a dam on the Swallowfield Wildlife Refuge, near Armidale, N.S.W. (Boyd and Breckwoldt, 1977).

Farm dams can be made to enhance rural beauty by strategic planting of trees and shrubs. Such dams can also be especially valuable for aquatic-based recreation in hot and otherwise waterless areas. In eroded areas, small dams may be used for water diversion and silt catching.

Unfortunately not all these uses are compatible. Dams designed to be attractive to waterfowl are little suited for aquaculture, and vice versa. Those used extensively for irrigation may be emptied during dry seasons and so are of little value for waterfowl when needed most, and no use at all for fish farming. Active water sports like skiing are inimical to most other uses, particularly use as wildlife refuges. Nevertheless the plea is made that farmers give thought to the possibility that their dams become at least bifunctional.

Concluding remarks
All farm dams are of interest as potential sites for aquaculture (Weatherley, 1967). Some, particularly excavated clay dams, are more favourable for productivity studies than salt lakes (Morrissy, 1974). Moreover, others provide useful sites for solving basic ecological problems such as the nature of the influence of fish on zooplankton and littoral invertebrate communities (Tait, 1976) and niche separation in *Daphnia* (Hebert, 1977).

Farm dams range in fundamental nature from the relatively simple eco-
systems of excavated dams on clay to the complex ecosystems of large gully dams. The former are largely dependent on a few extrinsic biological factors (catchment vegetation and stock usage) and dominated by few physico-chemical factors (particularly turbidity induced temperature stratification and deoxygenation) (Morrissy, 1974), while the latter are similar to reservoirs or even lakes in which ecosystem control is vested in many factors, often difficult to delineate.

Notwithstanding the extent of variation in farm dams, most share four distinctive limnological features. These are:

1. High turbidity which influences major physico-chemical factors such as temperature stratification and hypolimnetic deoxygenation and which also may have some indirect influence on zooplankton and littoral invertebrate populations.

2. Extensive ecosystem control by extrinsic factors such as insolation, detritus supply, water balance and degree of stock use.

3. High nutrient levels, but not necessarily high primary production. In many dams food chains are detritus-based on allochthonous matter.

4. Large fluctuations in water level and often considerable variation in total dissolved solid concentrations. Only the former appears to have biological repercussions in that benthos and littoral plants and dependent invertebrates are adversely affected.

These generalisations and others mentioned earlier hide much ignorance. Nevertheless, it seems that the limnological features of farm dams are determined by a limited number of factors divisible into two groups: those of major importance—water balance, substrate type and stock usage; and several of secondary importance—nature of catchment vegetation, presence of fish, dam size, depth and age, and level of nutrient supply. The interaction between all these factors is complex and much remains to be learned.

Acknowledgments
I thank Dr N. M. Morrissy for helpful comments on a draft manuscript, and Messrs R. Tait and W. Walker for the loan of theses. Access to the work of these three colleagues has been invaluable in preparing this review. Dr J. Martin is thanked for information.

References


Hebert, P. D. N. 1977. Niche overlap

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33 Waste stabilisation ponds

B. D. Mitchell

Introduction
A confusing collection of terms has been used to describe shallow man-made basins utilising natural biological processes for the reduction of organic matter and the destruction of pathogenic organisms in waste-water (Marais, 1971; Gloyna, 1971; Missouri Basin Engineering Health Council, 1971). It is simplest to adopt the convention of Gloyna (1971) and use the term ‘waste stabilisation pond’ (W.S.P.) to describe any pond or lagoon used for biological waste treatment. A W.S.P. may be anaerobic, facultative or one of three types of aerobic pond. High-rate aerobic ponds are mechanically mixed and have short retention times; aerated ponds have virtually all oxygen supplied mechanically and have low algal growth; maturation or polishing ponds receive waste-water treated to a secondary degree and serve to provide high quality effluents (Ramani, 1976). By combining these pond types in series W.S.P. systems may be designed to meet a variety of treatment objectives.

The advantages of W.S.P.s as a means of waste-water treatment are linked to their simplicity—ponds are operated and maintained simply, they are easily constructed and have low energy consumption and operating costs. These features, together with an ability to effect marked reductions in waste-water biochemical oxygen demand (B.O.D.), suspended solid (S.S.) concentrations, and bacteria and virus numbers, and the ability to smooth peak hydraulic loads and deal adequately with seasonal industrial and agricultural wastes (Shelef, 1976), have made W.S.P.s attractive to less-developed countries, to small communities and at certain stages of urban development. However, the advantages have been accompanied by a lack of theoretical operational data upon which to base pond management. The approach to pond design and operation has been largely empirical. In many cases this has resulted in overloading and mismanagement of ponds and led to criticism of the usefulness of ponds for waste-water treatment (Barsom and Ryckman, 1970). The failure of ponds to provide effluents of sufficient quality to meet current requirements is not the result of limits of the pond method but rather of limits to our understanding of pond function.

Waste stabilisation ponds evolved from the concept of natural purification, that is, the observation that organic contaminants could be stabilised by naturally occurring organisms when wastewater was discharged to rivers and streams. Although W.S.P.s are, above all else, biological communities, the basic biology of the organisms dwelling in ponds has been neglected. In particular, the interaction of micro-organisms with higher trophic levels has been ignored in the design and operation of ponds. Yet if ponds are to be operated
optimally, then authorities must understand the fundamental structure and function of the community they are attempting to manage. It is as a contribution to aid this understanding that the present chapter has been prepared.

Although anaerobic ponds are used widely throughout Australia as pretreatment prior to aerobic ponding (Parker, Jones and Greene, 1959; Parker, Jones and Taylor, 1950), this chapter will deal with aerobic ponds. Particular emphasis will be placed on secondary and maturation ponds (although certain biological principles will apply to the aerobic portions of facultative ponds), as it is from these that effluents are discharged to the environment. If pond effluents are to be regarded as a resource for water and nutrient reclamation, these ponds provide the sites for recycling schemes.

Pond usage in Australia
Waste stabilisation ponds are used extensively by small communities throughout country areas of Australia.

Table 44 Waste stabilisation pond use throughout Australia. Compiled from information supplied by sewage treatment authorities and Gutteridge, Haskins and Davey (1976).

<table>
<thead>
<tr>
<th>State</th>
<th>Number of towns using ponds and population served</th>
<th>Surface area of ponds (ha)</th>
<th>Annual flow (ML)</th>
<th>Type of treatment</th>
<th>Disposal of pond effluents</th>
</tr>
</thead>
<tbody>
<tr>
<td>South Australia</td>
<td>Adelaide + 15(1400-33000)</td>
<td>340</td>
<td>31755</td>
<td>Secondary and tertiary</td>
<td>Irrigation and ocean discharge. Several irrigation, some river discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td>85</td>
<td>7206</td>
<td>Most tertiary, some secondary, a few primary</td>
<td></td>
</tr>
<tr>
<td>New South Wales</td>
<td>Sydney + 106</td>
<td>19</td>
<td>4501</td>
<td>Tertiary</td>
<td>Ocean discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td>312</td>
<td>56035</td>
<td>Most tertiary, secondary in 2, primary in 1</td>
<td>Land disposal or river discharge</td>
</tr>
<tr>
<td>Victoria</td>
<td>Melbourne + 68(1050-5700)</td>
<td>1450</td>
<td>70000</td>
<td>Primary and secondary</td>
<td>Ocean discharge</td>
</tr>
<tr>
<td></td>
<td>Dutson Downs</td>
<td>283</td>
<td>12100</td>
<td>Primary and secondary</td>
<td>Ocean discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td>200</td>
<td>35920</td>
<td>Many tertiary, several primary</td>
<td>Most irrigation, several stream and ocean discharge</td>
</tr>
<tr>
<td>Northern Territory</td>
<td>5(220-25000)</td>
<td>52</td>
<td>3190</td>
<td>Primary and secondary</td>
<td>Ocean, river or creek discharge</td>
</tr>
<tr>
<td>Tasmania</td>
<td>25(500-8830)</td>
<td>44</td>
<td>5980</td>
<td>Most primary, some secondary</td>
<td>River or ocean discharge</td>
</tr>
<tr>
<td>Queensland</td>
<td>Brisbane + 54(120-25000)</td>
<td>3</td>
<td>3285</td>
<td>Most secondary, several tertiary, some primary</td>
<td>Ocean discharge</td>
</tr>
<tr>
<td></td>
<td></td>
<td>40</td>
<td>9655</td>
<td>Secondary (after septic tanks)</td>
<td>Several land, several river and creek, some seepage</td>
</tr>
<tr>
<td>Western Australia</td>
<td>48(100-20865)</td>
<td>64</td>
<td>6282</td>
<td>Most secondary, some tertiary, a few primary</td>
<td>Most soakage, some irrigation, several rivers, few ocean discharge</td>
</tr>
</tbody>
</table>
In addition lagoons are used at Werribee, Victoria, in the treatment of daily peak and wet weather flows from approximately 75 per cent of the population of Melbourne. At Bolivar, South Australia, domestic waste-water from a population of 600,000 in Adelaide and trade waste equivalent to a population of 700,000 is treated in ponds. A large pond system at Dutson Downs in the Latrobe Valley, Victoria, treats largely industrial waste, and smaller plants serve some metropolitan areas in Sydney. A summary of current ponding practices is provided in Table 44.

Ponds vary in size from approximately 0.2-8 ha, and are characteristically 1.0-1.5 m deep although they may be up to 3 m deep. Large cities using ponds for tertiary treatment usually employ aerated-type ponds, while those used for secondary treatment are usually facultative. Country towns use fully aerobic maturation ponds for tertiary treatment, facultative or aerobic ponds for secondary treatment, while primary ponds may be anaerobic. Anaerobic ponds are used extensively in Australia for the treatment of farm wastes, especially from piggeries (Environmental Control Council and Water Quality Control Council of Queensland, 1975; Bryson, 1972) and cannery waste (Parker, 1966).

Algae in waste stabilisation ponds

The composition of the algal flora. Four phyla of algae occur: the Chlorophyta, Euglenophyta, Chrysophyta and Cyanophyta. However, relatively few genera are found in W.S.P.s irrespective of geographical location, climate, chemical composition of sewage, or degree and type of pretreatment (Ward and King, 1976; Fitzgerald and Rohlich, 1958). No local or regional floras develop and attempts to influence species composition via inoculations are not effective (Fitzgerald and Rohlich, 1964; Bartsch, 1961). The algae occurring in W.S.P.s are those capable of adaptation to high nutrient conditions and a wide range of organic compounds. Characteristically, species are those usually associated with polluted bodies of water. The most common genera are Anabaena, Anacystis, Ankistrodesmus, Chlamydomonas, Chlorella, Euglena and Scenedesmus. This pattern holds true for North America (Goulden, 1976; Palmer, 1969; DeNoyelles, 1967), Europe (White, 1975; Malchow-Möller et al., 1955), India (Patil et al., 1975; Singh and Saxena, 1969), New Zealand (Haughey, 1965), and Australia (S. Hussainy, personal communication; Mitchell, unpublished). Extensive species lists of the algae occurring in W.S.P.s are to be found in Goulden (1976), Benson-Evans and Williams (1975), Singh and Saxena (1969) and Malchow-Möller et al. (1955). The diversity of the algal flora depends upon the rate of organic breakdown in ponds, that is the amount of unoxidised organic material. Ponds with a lot of unoxidised material have few species and species number increases as enrichment declines (Benson-Evans and Williams, 1975; DeNoyelles, 1967).

As in natural water bodies, the algae of W.S.P.s undergo a characteristic succession of dominance during the warmer months when standing crops are high. The usual pattern consists of an initial mixed population of flagellated greens, followed by Euglena, then Scenedesmus, giving way to Chlorella and finally mixed blue-greens (Patil et al., 1975; King, 1972; Haughey, 1965; Singh and Saxena, 1969; DeNoyelles, 1967; Neel et al.,
Waste stabilisation ponds

1961). This succession is related to detention time and the attendant change in organic matter, free \( \text{CO}_2 \), and pH. Succession favours species less resistant to high organic concentrations and those able to use low concentration of free \( \text{CO}_2 \) (King, 1972). Seasonal variation of algal numbers in W.S.P.s also occurs and the factors responsible are those operating for natural water bodies, that is, light, temperature, grazing and organic pollution (White, 1975; Rodgi and Patil, 1971; DeNoyelles, 1967).

**Algae-bacteria interactions.** The most intensely studied aspect of waste stabilisation pond biology concerns the interaction of algae and bacteria and their role in the stabilisation of organic matter. The bacteria aerobically metabolise organics while algae utilise the stable end products of bacterial metabolism (McKinney, 1976; Ganapati, 1975; Gloyna, 1971; Oswald, 1964). Aerobic bacteria (heterotrophs) metabolise organic matter producing bacterial cells, \( \text{CO}_2 \), \( \text{H}_2\text{O} \), \( \text{NH}_3 \) and phosphates. The \( \text{NH}_3 \) produced reacts with \( \text{CO}_2 \) and \( \text{H}_2\text{O} \) to produce ammonium bicarbonate \((\text{NH}_4\text{HCO}_3)\), and the nitrogen is thus available for further metabolism. Nitrification reactions, limited by the amount of \( \text{NH}_3-\text{N} \), do not occur to a major extent in W.S.P.s (McKinney, 1976). When dissolved organic material is low bacteria derive energy from the metabolism of their own storage products (Ganapati, 1975). This endogenous respiration results in a continuous cycling of organic matter. The high standing crops of algae (autotrophs) in W.S.P.s reduce the stable oxidised end products of bacterial metabolism to produce new algal cells and dissolved oxygen during photosynthesis. The oxygen released is essential for the aerobic bacteria. That is, the process of organic stabilisation is essentially reversed as organic matter is synthesised from previously stabilised inorganic compounds. Algae also undergo endogenous respiration, and approximately 20 per cent of the bacterial and algal protoplasm is not metabolised and remains as inert, non-biodegradable organics, or suspended solid. Materials are continually recycled producing non-biodegradable organics. That is, nutrients are continuously converted to a stable form, this conversion being a prime means of nutrient removal in W.S.P.s (McKinney, 1976). The intensity of the algal-bacterial relationship is reflected in diurnal fluctuations of 25-50 mg l\(^{-1}\) of dissolved oxygen and up to two full pH units (King, 1972, 1976).

Although the biochemistry of this process is well understood (Ganapati, 1975), several fundamental questions remain to be answered. The emphasis upon public health aspects of W.S.P. bacteriology has resulted in little interest in the basic microbiology of waste stabilisation. Consequently, although bacterial succession and predominance in a W.S.P. is a complex and diverse process (McKinney, 1976; Pike, 1975; Gann *et al.*, 1968) the types of bacteria concerned in the stabilisation of organic substrates are still largely unknown, as are the complexities of the pathways of degradation. Similarly, the agencies of removal of pathogenic bacteria in W.S.P.s are still unknown. While it has been suggested that removal may be due to antibacterial substances produced by algae (Ward and King, 1976) or the effect of increased pH due to \( \text{CO}_2 \) removal during photosynthesis (Parhad and Rao, 1974; Pike, 1975) it is more likely to be the result of competition with the stabi-
lising bacteria which are better able to utilise organic substrates (McKinney, 1976).

The relationship between algae and bacteria in W.S.P.s has been termed 'symbiotic' (Oswald et al., 1953a,b). However, the complexity of algal-bacterial interactions would suggest that the exact nature of the relationships is far from clear. The literature on W.S.P.s has implicitly assumed that the relationship is mutualistic while several experiments seriously question the whole basis of the biological process of organic stabilisation in W.S.P.s. Ganapati and Amin (1972) have shown that while the relationship is mutualistic for species of Scenedesmus, Nostoc and Aulosira, it is definitely antagonistic for some species of Chlorella and Oscillatoria. This effect is probably the result of the release of extracellular products by the algae. Fogg (1962) reported that few contaminant bacteria will develop in vigorously growing algal cultures. Antibacterial substances have been found in filtrates of Chlorella, Scenedesmus, Chlamydomonas, Oscillatoria and Stichococcus (Ward and King, 1976). Algae may indirectly inhibit bacterial activity by increasing pH during photosynthesis while bacteria may inhibit the growth of algae via competition for phosphates (Ward and King, 1976).

Algae excrete extracellular products, e.g. glycollate and other organic acids, polysaccharides, amino acids and peptides, lipids and vitamins, the turnover rate of which may be sufficient for these compounds to be ecologically important (Fogg, 1962). These compounds may provide an appreciable proportion of the dissolved organic substances that bacteria use as a carbon source, while bacterial oxidation of complex carbon compounds may provide simple substrates for the heterotrophic growth of algae (Ward and King, 1976). The nature of the interaction of algae and bacteria is apparently variable and differences may exist between ponds and in the same pond over time. Further research is needed to clarify this area of W.S.P. biology.

**Algal production.** Current design practices for waste stabilisation ponds take account of algal growth factors. The general practice has been to assume that nitrogen and phosphorus are abundant and in a readily usable state (e.g. Gloyna, 1971) and that the main factors controlling algal activity are light, alkalinity, mixing and retention time (King, 1976; Gloyna, 1968, 1971). In the absence of other nutrient limitation, the algae in a W.S.P. can become carbon limited when bacterial metabolism of organic compounds cannot produce sufficient CO$_2$ to meet the photosynthetic demand (King, 1972). Most of the CO$_2$ used to produce excess oxygen (and thus meet night-time respiratory demand) comes from the carbonate-bicarbonate alkalinity system. This system acts as a CO$_2$ 'battery' being discharged during the day and replenished by respiratory CO$_2$ production at night (King, 1972). Photosynthesis at mid-morning can be up to 50 per cent greater than at midday as a result of rapid depletion of the CO$_2$ accumulated during the night (Bartsch, 1961). Algal succession in W.S.P.s may be related to CO$_2$ uptake from the carbonate-bicarbonate system and the ability of certain groups of algae to use lower concentrations of CO$_2$. This culminates in the dominance of blue-greens (King, 1972).

Fogg and Watt (1965) have shown that
the extracellular products of algal metabolism may constitute a considerable proportion of the total carbon fixed in photosynthesis. Furthermore, the proportion increases as population density decreases and when photosynthesis is inhibited by high light intensity. These two conditions may prevail at just those times when CO$_2$ is limiting, i.e. during summer when self-shading has resulted in population decline.

It has been assumed that carbon is the first nutrient to become limiting in a W.S.P. (Fitzgerald and Rohlich, 1958), and scant attention has been given to nitrogen and phosphorus. But Parker et al. (1959) have noted that algae seldom develop in ponds treating trickling filter effluents. This has also been observed by the author, despite algal blooms in successive ponds. Nitrification reactions in W.S.P.s are limited by the amount of NH$_3$-N (McKinney, 1976) and if this is depleted during passage through biological filters then nitrogen may also be involved in algal limitation in W.S.P.s. This is suggested by the observation that standing crops of algae may be higher in the first pond of a series without pretreatment (DeNoyelles, 1967). Moreover, algae are unable to use the complex polyphosphates in detergents and must rely on bacterial phosphatase to produce orthophosphate. The conversion is not normally completed during biological treatment and a considerable portion of the total phosphorus may be in a form not available for algal assimilation (Hemens and Mason, 1968).

These phenomena highlight one important area of W.S.P. biology that has been almost totally neglected, that is, the fate of nutrients upon entering the pond. Little information is available concerning the forms of nitrogen and phosphorus in W.S.P.s, how they are altered by passage through the ponds, the extent of nutrient cycling or the degree to which nutrients are channelled to the sediments. In the absence of such knowledge of pond function, proposals for nutrient removal and reclamation schemes appear somewhat premature.

Algal problems in waste stabilisation pond effluents. The major operating problem of waste stabilisation ponds has been the presence of algae in effluents, their contribution to B.O.D. and S.S. and, hence, the effect of effluent quality on receiving water bodies. The cause is that fundamental processes in W.S.P.s convert unstable organics into algal cells. During summer, effluent B.O.D. and S.S. may exceed influent values due to the presence of algae, while filtered effluent B.O.D. and S.S. can be negligible (Missouri Basin Engineering Health Council, 1971). High standing crops of algae can contribute significantly to the total oxygen demand of the pond (Bartsch and Allum, 1957) and senescent or dead algal cells in effluents might be expected to increase the B.O.D. of receiving waters (Ward and King, 1976). That is, in the past, W.S.P.s have essentially exported their B.O.D. and nutrient load, albeit in a changed form, to rivers, lakes and the ocean; the unfounded idea was that the algae in effluents could survive in receiving waters, thereby causing no harm. The most common cause of dead algal cells in pond effluents has been improper chlorination (Oswald, 1976; Horn, 1970). The effects of W.S.P. effluents upon receiving waters are complex and dependent upon effluent quality, dilution factor, and the ability of algae to survive. While the effect may be small in some cases (Patrick, 1976), it may result

The detrimental effects of algal-laden effluents and the value of algae as a byproduct have spawned a host of chemical and mechanical alternatives for algal removal (John et al., 1976; Middlebrooks et al., 1974; Oswald and Golueke, 1968). However, most techniques involve considerable capital investment and require expert supervision, and thus detract from the advantages of W.S.P.s as a means of waste treatment. Furthermore, techniques such as centrifugation and flocculation may concentrate pathogenic bacteria and viruses (Cooper, 1962). The problem of algal 'harvest', where product quality is important, is not the same as algal removal. It is difficult to maintain constant standing crops of a desired species (Palmer, 1967) and some algae may be unacceptable as stock feeds. In-pond techniques such as biological harvesting with zooplankton or fish have been largely discounted before adequate research as to their potential has been conducted (Middlebrooks et al., 1974).

The fauna of waste stabilisation ponds and its role in the purification process
In contrast to the amount of work on algae, the study of other organisms occurring in W.S.P.s has been neglected. This appears to reflect the idea that organisms other than algae and bacteria play no useful role in the stabilisation process and are therefore unimportant or even detrimental. However, since it is improbable that algal population dynamics ever operate independently of organisms at higher trophic levels, the importance to pond operation of the ecology of these organisms is self-evident. Additionally, a more direct involvement in waste stabilisation processes is implicated for many animal groups.

Protozoa. The Protozoa of waste stabilisation ponds are largely unstudied yet their abundance (Curds, 1975; DeNoyelles, 1967) suggests they serve an important role in the stabilisation of organic matter. Five classes of Protozoa are encountered in waste-treatment (Curds, 1975): the Phyto- and Zoomastigophora, Rhizopoda, Actinopoda, and Ciliata. These classes represent four distinct nutritional types each of which may be seen as contributing to the stabilisation of organic matter. The Phyтомastigophora are autotrophic (bearing chloroplasts) and absorb light to fix CO₂ thus storing organic products. The Zoomastigophora are saprozoic and rely on dissolved organic matter for nutrition. Their role is similar to that of the bacteria. The phagotrophic ciliates filter bacteria to meet their nutritional requirements while the predatory ciliates consume other protozoans.

Pillai and Subrahmanian (1944) have suggested that Protozoa play a major role in aerobic purification of sewage. They have shown that isolated Protozoa can bring about nearly all the changes associated with purification. Protozoa may serve other functions in W.S.P.s. Ciliates which graze upon bacteria may prevent self-limitation of bacterial numbers and maintain a high rate of assimilation of organic material by bacteria (Curds, 1975). Protozoa may also help in the clarification of effluents by reducing bacterial numbers via predation or flocculation.

Rotifera. While the abundance of rotifers in waste stabilisation ponds has received some attention, the role of this
Waste stabilisation ponds

The group in purification has not. Two classes of rotifer occur in W.S.P.s. The Monogononta are the dominant group while the Bdelloidea are poorly represented (Doohan, 1975). The most common genera of Monogonontans are Brachionus, Keratella, Asplanchna and Filinia (Goulden, 1976; White, 1975; DeNoyelles, 1967; S. Hussainy, personal communication). The filter-feeding rotifers appear to play a secondary role in the removal of organic wastes (Doohan, 1975) but their influence upon algal populations in W.S.P.s can be significant. Algal numbers can be severely reduced as a result of heavy grazing by Brachionus which is, in turn, influenced by the predatory Asplanchna (DeNoyelles, 1967).

Table 45 Species of microcrustacea occurring in waste stabilisation ponds

<table>
<thead>
<tr>
<th>Species</th>
<th>North America</th>
<th>England</th>
<th>Europe</th>
<th>South Africa</th>
<th>Australia</th>
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<tr>
<td>Daphnia magna</td>
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<td>D. pulex</td>
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<td>D. catawba</td>
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<td>D. longispina</td>
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<td>D. carinata</td>
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<td>Bosmina longirostris</td>
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<td>Moina dubia</td>
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<td>M. macropa</td>
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<td>M. brachiata</td>
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<td>M. micrura</td>
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<td>M. rectirostris</td>
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<td>Diaphanosoma sp.</td>
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<td>Chydorus sphaericus</td>
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<td>Alona diaphona</td>
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<td>Ceriodaphnia rigaudi</td>
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<td>Simocephalus exspinosus</td>
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<td>Macrothrix hirsuticornis</td>
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<td>Biapertura affinis</td>
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<td>Leydigia ciliata</td>
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<td>L. leydigi</td>
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<td>L. australis</td>
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<td>Candonocypris assimilis</td>
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<td>Potamocypris sp.</td>
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<td>Mesocyclops leuckarti</td>
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<td>Microcyclops minutus</td>
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<td>Acanthocyclops gigas</td>
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<td>Eurytemora affinis</td>
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<td>Cyclops vernalis</td>
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<td>C. vicius</td>
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<td>Eucyclops speratus</td>
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<td>Boeckella triarticulata</td>
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<td>Diaptomis gracilis</td>
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**Crustacea.** A wide variety of planktonic Crustacea has been recorded from waste stabilisation ponds. Cladocera and Copepoda are the dominant members of the zooplankton, the cladoceran genus *Daphnia* usually being most abundant. A list of the species of microcrustacea found in W.S.P.s from a variety of geographic areas is presented in Table 45.

Filter-feeding planktonic Crustacea can consume significant amounts of algae (e.g. Wetzel, 1975; Hutchinson, 1967) and this aspect of the biology of Crustacea has interested workers in the field of W.S.P.s. However, experimental data are scant and the presence of zooplankton in a W.S.P. is still considered by some to be detrimental to pond function (Dinges, 1973). The observation that algal blooms in W.S.P.s are often followed by peak numbers of zooplankton and clear effluents is widespread. The demand for an increased quality of pond effluents has led to the idea of culturing zooplankton in polishing ponds to remove algae from effluents. Ehrlich (1966) has shown that *Daphnia* can clear effluents of *Chlorella* and bacteria with feasible detention times, and Dinges (1973) has concluded that *Daphnia* culture represents a feasible means of improving W.S.P. effluents. Goulden (1976) has calculated that zooplankton standing crop and respiration in a W.S.P. can amount to 15 per cent of the organic load.

Reid (1976) and Kryutchkova (1968) have shown that when zooplankton standing crops are maximal they can result in 80-100 per cent B.O.D. removal. Zooplankton can remove S.S. via filtration and flocculation and will remove bacteria during feeding (Schroeder and Hepher, 1976; Loedolff, 1965). Elster (1965) reported no loss of pond purification capacity when *Daphnia* produce a ‘clear-water phase’ with low B.O.D., S.S., algae and bacteria. *Daphnia* feed on algae when retention times are long but usually will not significantly remove algae in primary ponds, i.e. those ponds where algal eradication would be detrimental to performance (Missouri Basin Engineering Health Council, 1971). Even in these ponds *Daphnia* may serve a useful purpose. Grazing upon the algae will reduce algal self-shading effects and may keep the algae in a vigorous growth phase. This can increase the efficiency of photosynthesis and the intensity of decomposition of organic matter (Gliwicz, 1975). Despite the potential advantages of zooplankton in W.S.P.s, no concerted attempt has been made fully to incorporate the above ideas into a complete treatment system.

**Fish.** The idea of using sewage for fish culture is not new, nor is the observation that fish grow exceptionally rapidly in sewage fertilised ponds. The practice of fish culture in ponds connected with latrines and animal pens is widespread and longstanding in Asian countries (Hickling, 1968). It is only recently in western countries that the concept of fish culture in waste stabilisation ponds has been tested on a preliminary basis. The only notable exception is the culture of *Cyprinus carpio* in large ponds as an integral part of the waste treatment system at Munich (Hickling, 1968). Several species of fish have been recorded as occurring naturally in W.S.P.s. These include *Cyprinus carpio*, *Mugil cephalus*, *Gambusia affinis*, *Perca fluviatilis*, *Leuciscus cephalus*, *Rutilus*
Waste stabilisation ponds

...smaller filter-feeders may therefore increase (Hillbricht-Ilkowska and Weglenska, 1973) and the metabolism of the whole zooplankton community thereby increase (Hrbáček et al., 1961). This effect can also increase phytoplankton density by restricting zooplankton grazing upon algae as demonstrated in W.S.P.s by Williams et al. (1974). In this context, the use of larger fish may be better than using small planktivorous species and fry.

A further effect of fish in W.S.P.s is that they can increase suspended organic material by feeding and excretion, thus increasing mineralisation rates and the amount of bacteria in suspension (Spodniewska, 1962). This can increase the abundance of organisms on which fish do not feed, e.g. rotifers (Hillbricht-Ilkowska, 1962). Fish may also act as an important nutrient store and so be valuable for nutrient removal (Kitchell et al., 1975; Hrbáček et al., 1961).

Other fauna. A variety of other faunal groups occurs in or is associated with W.S.P.s. None can be fully discussed here though it is appropriate to make brief mention of two, insects and birds. Groups recorded from W.S.P.s but not further mentioned include nematodes, annelids, arachnids and molluscs (see Curds and Hawkes (1975) for further details).

Of insects, the most significant group is the Chironomidae, though the composition of the insect community appears to be largely a function of pond organic loading and detention periods. In general, little attention has been accorded the functional role of these communities in W.S.P.s, though the work of Kimerle and Anderson (1971)
has demonstrated that midge larvae are important in organic matter decompositional processes. These authors also demonstrated that the energy removed from a W.S.P. by emergence and respiration of chironomids may be up to 7 per cent of primary production. Additionally, their work indicated that as well as removing energy from ponds, midge larvae may contribute to the overall efficiency of waste stabilisation by extending the aerobic zone of decomposition further into the substratum. To date, the work of Kimerle and Anderson (1971) remains the most careful assessment of energy pathways within W.S.P.s. It may be noted that the occurrence of chironomids in W.S.P.s is not entirely without disadvantage; at Bolivar near Adelaide, for example, the mass emergence of adults proved a nuisance to nearby residential properties and traffic, and the problem was solved only when larval populations were controlled (Peters, 1975; Glover, 1969; Johnson, 1968).

Apart from a number of reports which simply record the occurrence of particular bird species at W.S.P.s, little has been published on the avifauna of W.S.P.s. The exact role of this group in W.S.P. functioning remains indeterminate. It may be remarked, however, that it is perhaps unwise to dismiss this role as of little likely consequence. It is known, for example, that faecal material from waterbirds may contribute significantly to nutrient budgets in natural inland waters (giving rise to the phenomenon of so-called guanotrophy). The occupation by birds of the topmost trophic level in the W.S.P. biological community should also be brought to attention. On the reverse side of the coin, W.S.P.s in Australia undoubtedly provide significant feeding and refuge areas for waterbirds, whatever their role as sewage treatment plants.

Conclusions
Waste stabilisation ponds have a vital role to play in pollution prevention and water conservation and the present empirically based nature of pond design should not be allowed to preclude ponds as a future method of waste-water treatment. In any event, W.S.P.s appear to offer greater potential for nutrient removal than activated sludge (Brar and Tollefson, 1975) or biological filtration methods (Zanoni, 1976). For full realisation of this potential, however, a good deal more research is needed, as indicated in the body of this chapter. An additional avenue for research and offering much potential as a nutrient removal technique, but not mentioned above, is that involving the use of macrophytes (Dinges, 1976; Boyd, 1970).

Acknowledgments
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References


Some special aquatic environments


Waste stabilisation ponds


Introduction
The swamps and billabongs of the Murray-Darling River system provide food, breeding areas, and refuge for a diverse association of birds, mammals, reptiles and invertebrates in a region otherwise characterised by aridity and the absence of natural bodies of permanent standing water. Their importance as an inland aquatic resource from the viewpoint of wildlife conservation, if from no other view, can scarcely be overestimated. Nevertheless, many swamps and billabongs have been and are subject to considerable and unfavourable changes imposed by man. These have taken place largely in ignorance of effects because these waters have been so poorly investigated ecologically. The changes include complete disappearance resulting from river impoundment and flow regulation, use as recipients of effluents, drainage and filling for urban or agricultural purposes, conversion to sewage stabilisation ponds, utilisation as reservoirs from which water may be pumped for stock, other agricultural and domestic needs, and—the final insult!—use as garbage dumps. Thus, for the area near Albury and Wodonga townships, an area naturally characterised by many hundreds of billabongs and swamps marginal to the River Murray, it has recently been said that ‘the character of local billabongs already has been markedly changed, and few, if any, could be considered pristine environments’ (Walker and Hillman, 1977).

This chapter reviews our present knowledge of billabongs associated with the major river system of south-eastern Australia, the Murray-Darling. Thereby, it is hoped, attention will be focused upon the biological interest of billabongs in this region, the extent of our lack of knowledge will become more evident, and further work will be stimulated. It need hardly be added that sensible management of this most valuable aquatic resource requires a rather more solid data-base than it presently has.

Early work on billabongs associated with the Murray-Darling system is meagre. Russell (1886) and Fletcher (1893) noted the dependence of frogs on seasonal inundation of the Darling floodplain. Further references on the association of flooding and breeding seasonality, particularly of waterfowl, appeared sporadically in the early naturalist literature. Among the first intensive studies was one by Frith (1959) on the ecology of ducks in the Riverina.

The first synecological studies of billabongs began in 1973. These were on billabongs of the River Murray in the Albury-Wodonga area and formed part of a limnological survey, 1973-76, undertaken for the Cities Commission (later, the Albury-Wodonga Development Corporation) (Walker and Hillman, 1977). Microcrustacean communities associated with weedbeds in a Goulburn
River billabong have been studied by Shiel (1974, 1976, 1978), and the microfauna of billabongs between Wodonga and Yarrawonga have been studied by the same author (Shiel, unpublished). Since they are the only billabongs to have been comprehensively investigated, those studied by Walker, Hillman and Shiel provide the basis for most of the discussion in the present chapter. Finally, brief note may be accorded a comprehensive and joint study between CSIRO Division of Wildlife Research and the N.S.W. National Parks and

Plate XVI Aerial photograph of a 30 km section of the Goulburn River near Alexandra, Victoria. Photograph by courtesy of the Victorian Department of Crown Lands and Surveys.
Wildlife Service; the study is taking place in red-gum swamps at Booligal on the Lachlan River, and the aim is to gain knowledge useful for habitat management and waterbird conservation (F. Crome, personal communication).

**Billabong morphology and distribution**

Most billabongs are cut-off meanders or 'ox-bow lakes' formed by lateral displacement (meandering), differential abrasion, and downstream migration of meander loops in the plains tract of a stream or river (Fig. 49). Thus they are usually crescentic and elongated, up to several kilometres long, but generally less than 100 m wide. They are also shallow (rarely > 5 m, average 1-3 m); siltation during successive floods and a seasonal vegetation die-off aid filling. Ultimately, swamplands form, as along the Lachlan River and the eastern coastal plain (Goodrich, 1970), and billabongs may fill completely, as along the Goulburn and Murray. Both extant and filled billabongs can be seen in Plate XVI, which shows a 30 km section of the Goulburn floodplain near Alexandra.

A few billabongs are not ox-bow lakes. These may be natural or man-made depressions in the floodplain (e.g. sand or gravel pits) which have filled with water and in which a characteristic vegetation has developed. Billabongs between Thornton and Eildon on the Goulburn River provide examples. Others, such as those on the Paroo, are deepened holes in the channel of an ephemeral stream and persist when stream flow ceases.

Virtually all Australian rivers flowing through a plains tract meander and develop billabongs. The greatest development of billabongs occurs on the gently sloping (1 to 2 cm km⁻¹) plains of the Murray-Darling system, least on the floodplains of rivers flowing east from the Great Divide. The Australian total probably exceeds 100,000. Along the 30 km section of the Goulburn River shown in Plate XVI, more than 100 billabongs occur, and at least 190 are asso-

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**Fig. 49** Stages in billabong formation. 1. initially straight channel is differentially abraded; 2. curves become loops as the wavelength increases; 3. meanders catch up, narrowing the 'necks' between them; 4. breakthrough isolates loop, forming ox-bow lake or billabong. Modified after Sale and Sale (1967).
associated with the River Murray between Lake Hume and Lake Mulwala, a distance of ~200 river km (from aerial photographs in Loder and Bayly, 1977).

With dam construction and subsequent river flow regulation, the seasonal flooding necessary to replenish billabongs has become less frequent, particularly along the Goulburn and Murray. As a result, many have dried. In areas where billabongs are below the surface of the water table, however, or on the margin of the floodplain below steep slopes, as along the Goulburn between Yea and Seymour, seepage and runoff maintain permanent water. The floodplain of the Darling, moreover, is still subject to frequent flooding, and its billabongs and swamps are therefore frequently replenished with water. River impoundment has not only altered the seasonal flood cycle. Other alterations include those to hydraulic behaviour, to load carried, to erosive capacity and to the nature of meandering. All things considered, it is unlikely under modern conditions that further formation of billabongs will take place (Walker and Hillman, 1977). To preserve some of the existing ones, therefore, is a conservation issue of some priority and urgency.

**Light penetration.** Unlike farm dams (Tait, 1976; Timms, 1977; Chapter 32), most billabongs are not highly turbid; surrounding soils are generally bound by vegetation, substrates are not exposed as they are around many dams, and particulate matter inflow is reduced. Light often penetrates to the bottom, permitting growth of macrophytes across the entire billabong. Seasonal variability in transparency, however, may be great, particularly if stock use the billabong, or in time of flood. Thus, floodwaters carrying suspended clay and organic particles reduced Secchi transparencies from 1-6 m to 60 cm in a Goulburn billabong (Fig. 50). Summer blooms of the green algae, *Volvox*, reduced transparency to less than 20 cm in the same billabong (Shiel, 1974).

**Temperature.** Water temperatures tend to follow air temperatures as in most small bodies of water (Bayly and Williams, 1973), but vegetation may prevent complete mixing and cause small horizontal temperature differences. Where present, floating mats of aquatic plants (*Azolla, Lemna, Riccia*) act as insulators, moderating both winter and summer temperatures. An annual temperature range of 8-0 to 28-5°C has been recorded for a Goulburn River billabong (Shiel, 1976), and one of 7-6 to 22-4°C for a Murray River billabong (Walker and Hillman, 1977). Thermal stratification has not actually been detected in any billabong, but on the evidence given by Timms (1977) probably occurs in deep
billabongs. All billabongs studied thus far have been isothermal at all times.

**Oxygen.** Concentrations of dissolved oxygen are variable, even between adjacent billabongs. Occasionally anoxic conditions may prevail in one billabong, and supersaturation in a nearby one—though such a situation reflects a rather extreme set of local conditions. As a general rule, oxygen concentrations follow the patterns noted by Bayly and Williams (1973), namely higher values in summer than in winter, in the afternoon than in the morning, and at the surface than at the bottom. Nevertheless, some billabongs are prone to oxygen depletion in summer as water levels fall. Others, those with permanent water, may have supersaturated concentrations in summer due to algal blooms (cf. Fig. 50).

Fig. 50 Some physicochemical features of a billabong near Alexandra (Goulburn River). Note that there are no data for 1975. (a) temperature of air (---), and of water at station 1 (Chara) (-----) and 3 (Juncus) (-----); (b) oxygen saturation; (c) maximum depth (------) and concentration of total dissolved solids (TDS) (-----); (d) Secchi disc transparency. From Shiel (1976).
Concentrations of total dissolved solids. Concentrations of total dissolved solids fluctuate markedly with rainfall and flooding. This is illustrated in Fig. 50 which shows that an annual range of 40 to 1,960 ppm (mean, 227) occurred in the billabong it refers to. The single peak immediately followed the inflow of floodwater. In 27 billabongs of the Albury-Wodonga area, concentrations during the period 1976-77 were between 60 and 250 ppm (Shiel, unpublished). Much higher concentrations occurred in a billabong near Renmark (400 to 1,200 ppm), reflecting the effect of irrigation practices on local salinities (see Chapter 10).

pH, major ions and nutrients. With a few exceptions, pH falls within the range 7.0 to 8.0. The order of dominance for the major cations in all billabongs sampled in the Upper Murray and Goulburn catchments was Na > Mg > Ca > K, although several in the Riverland area had more K than Ca. The anionic order of dominance was Cl > SO₄ > HCO₃, or, exceptionally, Cl > HCO₃ > SO₄. These orders of ionic dominance are typical for Australian waters (Williams, 1967). Data on nutrient (phosphate and nitrate) concentrations are available for three billabongs near Wodonga sampled 1975-77 (Shiel, unpublished). Concentrations were found to be very variable; nitrate concentration varied from 2 to 685 mgN m⁻³, phosphate concentrations from 11 to 624 mgP m⁻³. Average concentrations were high, indicating eutrophy. There was some correlation between increased nutrient concentrations and runoff from rain, but order-of-magnitude increases between successive fortnightly samples, without preceding rainfall, were not uncommon (Albury-Wodonga Development Corporation, personal communication).

Biological features

Phytoplankton. There has been no comprehensive study of phytoplankton communities. Blooms of Anacystis (blue-green) have been noted in billabongs near Albury-Wodonga (Walker and Hillman, 1978), and from billabongs of the Goulburn (Shiel, 1974; unpublished) has recorded Volvox, Ulothrix, Chlamydomonas, Schizogonium, Closteridium, Debarya, Spirogyra, Zygnema, Zygnemopsis, Netrium, Closterium, Cosmarium, Euglena, Trachelomonas, Tribonema, Lyngbya, Oscillatoria and Micrasterias.

Hydrophytes. A characteristic feature of billabongs and billabong swamps is the development of a diverse assemblage of emergent, submerged (rooted and non-rooted), and free-floating hydrophytes. Often, although superficially similar in hydrophyte vegetation, billabongs may have different species dominants, community composition and seasonal succession. This may occur even in the same geographical area. Thus, only three of the 35 species occurring in two billabong areas indicated in Table 46 are common to both areas. Additional species occupying similar niches on floodplains or coastal wetlands are listed by Goodrich (1970) and Aston (1973). Genera commonly present in the major hydrophyte communities include Eleocharis, Juncus or Typha (tall, emergent, macrophyte community), Callitriche, Potamogeton, Myriophyllum, Ludwigia or Vallisneria (submerged, rooted community), Ceratophyllum, Utricularia (submerged, non-rooted community), and Lemna,
Table 46  Macrophytes associated with billabongs at Wodonga and Alexandra. + present.

<table>
<thead>
<tr>
<th>Species</th>
<th>Common name</th>
<th>Wodonga</th>
<th>Alexandra</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agrostis avenaceae</td>
<td>bent grass</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Alternanthera sp.</td>
<td>alligator weed</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Azolla filiculoides</td>
<td>Pacific azolla</td>
<td></td>
<td>+</td>
</tr>
<tr>
<td>A. pinnata</td>
<td>ferny azolla</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Brasenia schreberi</td>
<td>water-shield</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Callitrichia umbonata</td>
<td>starwort</td>
<td></td>
<td>+</td>
</tr>
<tr>
<td>C. stagnalis</td>
<td>common starwort</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Ceratophyllum demersum</td>
<td>hornwort</td>
<td></td>
<td>+</td>
</tr>
<tr>
<td>Chara australis</td>
<td>stonewort</td>
<td></td>
<td>+</td>
</tr>
<tr>
<td>Crassula helmsii</td>
<td>swamp Crassula</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Cynodon dactylon</td>
<td>Bermuda grass</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Eleocharis sphacelata</td>
<td>tall spike-rush</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Elodea canadensis</td>
<td>Canadian pondweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Gahnia clarkei</td>
<td>saw-sedge</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Juncus ingens</td>
<td>rush</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>J. subsecundus</td>
<td>rush</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Lemna minor</td>
<td>common duckweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Lottilum rigidium</td>
<td>rye grass</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Ludwigia pepliodes</td>
<td>water primrose</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Myriophyllum propinquum</td>
<td>common water milfoil</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>M. elatinoides</td>
<td>coarse water milfoil</td>
<td>+</td>
<td>+</td>
</tr>
<tr>
<td>Nitella sp.</td>
<td>muskgrass</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Polygonum minus</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>P. lapathifolium</td>
<td>willow smartweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>P. strigosum</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Potamogeton ochreatus</td>
<td>blunt pondweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>P. tricarinatus</td>
<td>floating pondweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Ranunculus rivularis</td>
<td>small river buttercup</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Riccia fluitans</td>
<td>liverwort</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Ricciocarpus natans</td>
<td>—</td>
<td>—</td>
<td>+</td>
</tr>
<tr>
<td>Triglochin procera</td>
<td>water ribbons</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Senecio aquaticus</td>
<td>marsh ragwort</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Spirodela oligorrhiza</td>
<td>thin duckweed</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Utricularia sp.</td>
<td>bladderwort</td>
<td>+</td>
<td></td>
</tr>
<tr>
<td>Vallisneria gigantea</td>
<td>eel grass or ribbon weed</td>
<td>+</td>
<td></td>
</tr>
</tbody>
</table>

Azolla, Riccia and Ricciocarpus (free-floating community). One or more of these genera may be a member of the community. In such communities, regardless of species, the array of spatially discrete microhabitats available for colonisation by aquatic animals is immense. Predictably, therefore, billabongs support a diverse, if not unique, assemblage of micro- and macro-invertebrates.

Zooplankton. Collections of zooplankters from the open water of billabongs invariably contain littoral species, due to the occurrence of floating and submerged vegetation. Planktonic assemblages usually comprise one or two species of Boeckella, a Calamoecia species, two or more cyclopoid species, including *Mesocyclops* and *Microcyclops*, two or three cladocerans from the genera *Diaphanosoma*, *Daphnia*,...
Ceriodaphnia, Chydorus, Bosmina and Moina, and one to four rotifers from the genera Asplanchna, Brachionus, Filinia and Keratella. Average momentary species composition in fifteen billabongs sampled, regularly during 1976-77 was 2.6 for copepods, 2.4 for cladocerans and 3.1 for rotifers, a slightly more diverse community assemblage than that recorded for farm dams, reservoirs or lakes (Tait, 1976; Timms, 1970a,b, Chapter 32). This greater diversity is probably a reflection of the occurrence in billabong zooplankton of species which are facultatively planktonic, e.g. Chydorus sphaericus (Fryer, 1968).

Littoral invertebrates.

(i) Rotifera and microcrustacea. The diversity of the fauna in the littoral area (usually the entire billabong) is very large. To date, for example, 117 taxa of rotifers and microcrustaceans have been recorded from Goulburn and Murray River billabongs (Table 47). Eighty-five species have been recorded from a single billabong at Alexandra, the greatest microfaunal diversity recorded from any freshwater habitat (Shiel, 1976).

In contrast to the spatial heterogeneity of billabong hydrophytes, there is a remarkably constant ‘core’ group of microfaunal species in widely disparate billabongs. A plankton net of mesh size < 60 μm towed through any billabong weedbed usually obtains 10 to 40 species in the following groups: 1 to 6 rotifers from the genera Asplanchna, Keratella, Brachionus, Conochilus and Euchlanis; 5 to 20 Cladocera from the genera Simocephalus, Ceriodaphnia, Macrothrix, Bosmina, Chydorus, Alona and Biapertura; 1 to 4 ostracods; and 3 to 10 copepods from the genera Boeckella, Calamoecia, Mesocyclops, Macro-
cyclops, Microcyclops and Eucyclops. Seasonal occurrences of several other genera in each group are common in a collection, as is the occurrence of several congeneric species (Shiel, 1976). Seasonal variations in the composition of the microcrustacean community associated with Juncus are indicated in Fig. 51.

As well as a ‘core’ group of microfaunal species, groups are also associated with particular microhabitats; these microhabitats largely result from habitat subdivision by the ‘architecturally’ discrete hydrophytes. Within micro-communities are found detritivores, herbivores, carnivores and omnivores—constituting a complex trophic web within the larger trophic web of the billabong as a whole. Some knowledge of these micro-communities is slowly emerging. The micro-community on the finely-divided leaf surfaces of Myriophyllum appears very similar to that on Ceratophyllum, a structurally similar plant; that on Juncus appears similar to that on Typha (both are emergent hydrophytes). Additionally, Walker and Hillman (1977) have found that the number of microcrustacean species in a series of billabongs was directly correlated with the size of the standing macrophyte crop.

(ii) Macro-invertebrates. The diversity of macro-invertebrates is greater in permanent billabongs, that is those with community stability, than in billabongs which dry seasonally. Some groups commonly collected from reedbeds are listed in Table 48. Of these, the most frequently present are odonatan nymphs, corixids, notonectids, and dytiscid beetles. Occasionally, collections may be numerically dominated by hydroid coelenterates, notably the green
### Table 47: Rotifera and microcrustacea recorded from billabongs of the Murray and Goulburn Rivers during 1974-77

<table>
<thead>
<tr>
<th>ROTIFERA</th>
<th>41 G. testudinaris</th>
<th>82 C. leana</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Brachionus angularis</td>
<td>occidentalis</td>
<td>83 Potamocypris sp.</td>
</tr>
<tr>
<td>2 B. budapestinensis</td>
<td>42 Kurzia latissima</td>
<td>84 Herpetocypris sp.</td>
</tr>
<tr>
<td>3 B. calyciflorus</td>
<td>43 Camptocercus australis</td>
<td>85 Diacypris sp.</td>
</tr>
<tr>
<td>4 B. calyciflorus var.</td>
<td>44 Leydigia leidyi</td>
<td>86 Ilyocypris sp.</td>
</tr>
<tr>
<td>5 B. plicatilis</td>
<td>45 L. australis</td>
<td>87 Cypretta spp.</td>
</tr>
<tr>
<td>6 B. urceolaris urceolaris</td>
<td>46 Biapertura affinis</td>
<td>88 Paracypria minuta</td>
</tr>
<tr>
<td>7 B. urceolaris rubens</td>
<td>47 B. intermedia</td>
<td>89 Cypris bennelong</td>
</tr>
<tr>
<td>8 Brachionus quadridentatus melheni</td>
<td>50 B. karua</td>
<td></td>
</tr>
<tr>
<td>9 B. novae-zealandia</td>
<td>51 B. setigera</td>
<td></td>
</tr>
<tr>
<td>10 Brachionus spp.</td>
<td>52 Scapholeberis sp.</td>
<td></td>
</tr>
<tr>
<td>11 Keratella slacki</td>
<td>53 S. kingi</td>
<td></td>
</tr>
<tr>
<td>12 K. tropica</td>
<td>54 Daphnia lumholtzi</td>
<td></td>
</tr>
<tr>
<td>13 K. australis</td>
<td>55 Simocephalus expinosis</td>
<td></td>
</tr>
<tr>
<td>14 K. cochlearis</td>
<td>56 S. setulatus elisabethae</td>
<td></td>
</tr>
<tr>
<td>15 K. sp.</td>
<td>57 S. setulatus gibbosus</td>
<td></td>
</tr>
<tr>
<td>16 Nototheca sp.</td>
<td>58 S. acutirostris</td>
<td></td>
</tr>
<tr>
<td>17 Euchlanis incisa</td>
<td>59 Ceriodaphnia dubia</td>
<td></td>
</tr>
<tr>
<td>18 Lecane sp.</td>
<td>60 C. laticaudata</td>
<td></td>
</tr>
<tr>
<td>19 Epiphanes clavulata</td>
<td>61 C. quadranangula</td>
<td></td>
</tr>
<tr>
<td>20 Asplanchna brightwelli</td>
<td>62 C. cornuta</td>
<td></td>
</tr>
<tr>
<td>21 Polyarthra renata</td>
<td>63 Moina micrura</td>
<td></td>
</tr>
<tr>
<td>22 Filinia longiseta</td>
<td>64 Bosmina meridionalis</td>
<td></td>
</tr>
<tr>
<td>23 F. terminals</td>
<td>65 Neothrix armata</td>
<td></td>
</tr>
<tr>
<td>24 Conochilus dossuarius</td>
<td>66 Pseudoomina lemnae</td>
<td></td>
</tr>
<tr>
<td>25 Cupelopagis vorax</td>
<td>67 Echinisca spp.</td>
<td></td>
</tr>
<tr>
<td>CLADOCERA</td>
<td>68 Ilyocypris sordidus</td>
<td></td>
</tr>
<tr>
<td>26 Diaphanosoma unguiculatum</td>
<td>69 I. spinifer</td>
<td></td>
</tr>
<tr>
<td>27 Latonopsis australis</td>
<td>70 Macrothrix spinosa</td>
<td></td>
</tr>
<tr>
<td>28 Pleuroxus aduncus</td>
<td>71 Macrothrix sp.</td>
<td></td>
</tr>
<tr>
<td>29 Alonella excisa</td>
<td>OSTRACODA</td>
<td></td>
</tr>
<tr>
<td>30 Chydrorid n.gen.sp.</td>
<td>72 Cypridopsis funebris</td>
<td></td>
</tr>
<tr>
<td>31 Chydrorus sphaericus</td>
<td>73 Platycypris sp.</td>
<td></td>
</tr>
<tr>
<td>32 C. eurynotus</td>
<td>74 Ilyodromus ellipticus</td>
<td></td>
</tr>
<tr>
<td>33 Dunhevedia crassa</td>
<td>75 I. smaragdinus</td>
<td></td>
</tr>
<tr>
<td>34 Pseudochydrorus globosus</td>
<td>76 Stenocypris sp.</td>
<td></td>
</tr>
<tr>
<td>35 Alona rectangula rectangula</td>
<td>77 Candona sp.</td>
<td></td>
</tr>
<tr>
<td>36 A. rectangula richardi</td>
<td>78 Candocyonpcypris sp.</td>
<td></td>
</tr>
<tr>
<td>37 A. davidi davidi</td>
<td>79 Strandesia sp.</td>
<td></td>
</tr>
<tr>
<td>38 A. davidi theringi</td>
<td>80 Nevanhamia fuscata</td>
<td></td>
</tr>
<tr>
<td>39 A. cambouei</td>
<td>81 N. fenestrata</td>
<td></td>
</tr>
<tr>
<td>40 A. guttata</td>
<td>82 Cyprinus leana</td>
<td></td>
</tr>
<tr>
<td></td>
<td>83 Potamocypris sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>84 Herpetocypris sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>85 Diacypris sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>86 Ilyocypris sp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>87 Cypretta spp.</td>
<td></td>
</tr>
<tr>
<td></td>
<td>88 Paracypria minuta</td>
<td></td>
</tr>
<tr>
<td></td>
<td>89 Cypris bennelong</td>
<td></td>
</tr>
<tr>
<td></td>
<td>COPEPODA</td>
<td></td>
</tr>
<tr>
<td>90 Attheyella incerta</td>
<td>91 A. australica</td>
<td></td>
</tr>
<tr>
<td>92 Attheyella sp.</td>
<td>93 Elaphoidella sp.</td>
<td></td>
</tr>
<tr>
<td>94 D'Arcithompsonioid n.gen.sp.</td>
<td>95 Ectocyprida medius</td>
<td></td>
</tr>
<tr>
<td>96 Microcyclops varicans</td>
<td>97 M. javanus</td>
<td></td>
</tr>
<tr>
<td>98 Microcyclops spp.</td>
<td>99 Eucyclops agilis</td>
<td></td>
</tr>
<tr>
<td>100 E. euacanthus</td>
<td>101 E. nov. sp.</td>
<td></td>
</tr>
<tr>
<td>102 Paracyclops chilonti</td>
<td>103 Macrocylops albidus</td>
<td></td>
</tr>
<tr>
<td>104 Acanthocyclops vernalis</td>
<td>105 A. sp.</td>
<td></td>
</tr>
<tr>
<td>106 Mesocyclops leuckarti</td>
<td>107 M. decipiens</td>
<td></td>
</tr>
<tr>
<td>108 M. sp.</td>
<td>109 Tropocyclops nov. sp.</td>
<td></td>
</tr>
<tr>
<td>110 T. confinis</td>
<td>111 Tropocyclops spp.</td>
<td></td>
</tr>
<tr>
<td>112 Calamoecia lucasi</td>
<td>113 C. australis</td>
<td></td>
</tr>
<tr>
<td>114 Hemiboeckella scarli</td>
<td>115 Boeckella minuta</td>
<td></td>
</tr>
<tr>
<td>116 B. fluvialis</td>
<td>117 B. triarticulata</td>
<td></td>
</tr>
</tbody>
</table>
Fig. 5: Seasonal changes in the composition and abundance of microcrustacea in a billabong of the Goulburn River. See Table 47 for index to species. After Shiel (1976).
Table 48 Macroinvertebrates frequently collected from the reedbeds of billabongs

<table>
<thead>
<tr>
<th>Taxon</th>
<th>Predominant group(s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Coelenterata</td>
<td>Hydrozoa (<em>Chlorohydra, Hydra, Craspedacusta sowerbyi</em>)</td>
</tr>
<tr>
<td>Platyhelminthes</td>
<td>Turbellaria</td>
</tr>
<tr>
<td>Aschelminthes</td>
<td>Gastrotricha, Nematoda</td>
</tr>
<tr>
<td>Mollusca</td>
<td>Bivalvia (<em>Sphaerium, Corbiculina</em>)</td>
</tr>
<tr>
<td></td>
<td>Gastropoda (<em>Physastra, Lymnaea, Glacidorbis, Pettancylus</em>)</td>
</tr>
<tr>
<td>Annelida</td>
<td>Oligochaeta (<em>Naidium, Chaetogaster, Tubifex</em>)</td>
</tr>
<tr>
<td></td>
<td>Hirudinea (<em>Limnobbella</em>)</td>
</tr>
<tr>
<td>Arthropoda</td>
<td>Tardigrada</td>
</tr>
<tr>
<td>Arachnida</td>
<td>Hydracarina</td>
</tr>
<tr>
<td></td>
<td>Porohalacaridae</td>
</tr>
<tr>
<td>Crustacea</td>
<td>Atyidae (<em>Paraty</em>)</td>
</tr>
<tr>
<td></td>
<td>Palaemonidae (<em>Macrobrachium</em>)</td>
</tr>
<tr>
<td></td>
<td>Parastacidae (<em>Cherax</em>)</td>
</tr>
<tr>
<td>Insecta</td>
<td>Collembola</td>
</tr>
<tr>
<td>Ephemeroptera</td>
<td><em>Atalophlebia, Baetis, Cloeon</em></td>
</tr>
<tr>
<td>Odonata</td>
<td><em>Hemianax, Hemicordulia</em></td>
</tr>
<tr>
<td>Hemiptera</td>
<td><em>Mesoveliidae</em></td>
</tr>
<tr>
<td></td>
<td>Gerridae</td>
</tr>
<tr>
<td></td>
<td>Veliidae</td>
</tr>
<tr>
<td></td>
<td>Corixidae</td>
</tr>
<tr>
<td></td>
<td>Naucoridai</td>
</tr>
<tr>
<td></td>
<td>Belastomatidae</td>
</tr>
<tr>
<td></td>
<td>Notonecida</td>
</tr>
<tr>
<td>Coleoptera</td>
<td>Dytiscidae</td>
</tr>
<tr>
<td></td>
<td>Gyrinidae</td>
</tr>
<tr>
<td></td>
<td>Hydrophilidae</td>
</tr>
<tr>
<td>Trichoptera</td>
<td>Calamoceratidae</td>
</tr>
<tr>
<td></td>
<td>Hydroptilidae</td>
</tr>
<tr>
<td></td>
<td>Hydropsychida</td>
</tr>
<tr>
<td>Diptera</td>
<td>Culicidae</td>
</tr>
<tr>
<td></td>
<td>Ceratopogonidae</td>
</tr>
<tr>
<td></td>
<td>Chironomidae</td>
</tr>
</tbody>
</table>

*Chlorohydra*, which preys on zooplankton and littoral microcrustacea. Also frequently collected are shrimps (*Paraty, Macrobrachium*) and yabbies (*Cherax*).

**Vertebrates.**

(i) Fish. A survey of the fish in several billabongs of the Wodonga area was made during 1976 (Walker and Hillman, 1978). Table 49 lists the species recorded, and is given because it refers to species actually present in a specific set of habitats (here, billabongs) and over a given period and not to a synthesis of previous records for the Murray-Darling system (cf. Lake, 1967). A further ten species have been reported by local fishermen, but were not collected in the survey. Most fish in billabongs are carnivores.

(ii) Amphibia. Tadpoles are an important link in billabong trophic webs. They are detritivore-herbivores, grazing on periphyton or consuming organic material from detritus and mud. In turn, they are preyed upon by yabbies, fish
and birds. Adult frogs feed on macroinvertebrates, and are, likewise, important as a food source for higher-order consumers. Species recorded from Goulburn billabongs include *Litoria aurea* (golden bell frog), *L. ewingi* (Ewing’s tree frog), *L. verreauxi* (Verreaux’s tree frog), *Limnodynastes tasmaniensis* (spotted grass frog) and *L. dumerilli* (eastern banjo frog) (Shiel, unpublished). These and a few others also occur in the Albury-Wodonga area.

(iii) Reptiles. Two species of tortoise, *Chelodina longicollis* (long-necked tortoise) and *Emydura macquarii* (Murray tortoise), are common in billabongs of the Albury-Wodonga area (Walker and Hillman, 1977). Lizards and snakes also frequent the fringing vegetation of these billabongs, particularly *Pseudechis porphyriacus* (black snakes) and *Notechis scutatus* (tiger snake). Both prey on frogs.

(iv) Birds. The avifauna of billabongs and swamps has received greater attention than any other faunal group. Frith (1967) and Slater (1970), for example, have described the distributions of birds associated with floodplains. However, thus far no species list referring to specific billabongs has been published. Table 50, therefore, lists those species of waterbirds recorded from billabongs near the Hume Dam (Hawksview Wildlife Refuge). Altogether, 52 species of waterbird have been recorded, and 140 species of non-aquatic bird (M. R. Webb, personal communication). Many of the latter, particularly the insectivores, whilst not closely associated with water, nevertheless are dependent upon billabongs as a food source.

(v) Mammals. Although water-rats and platypus have been seen along the Murray and Goulburn Rivers in the vicinity of Albury-Wodonga and Alexandra, none has been recorded from billabongs in those areas.

**Concluding remarks**

This chapter has indicated how little is known about the ecology of billabongs. Comprehensive and intensive studies do not exist, and those studies which do have been concerned with only parts of the total ecosystem or have formed part

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Table 49 Fish species recorded from Albury-Wodonga billabongs during 1976. After Walker and Hillman (1978).

<table>
<thead>
<tr>
<th>Species</th>
<th>Common name</th>
</tr>
</thead>
<tbody>
<tr>
<td><em>Carassius auratus</em></td>
<td>goldfish, golden carp</td>
</tr>
<tr>
<td><em>Craterocephalus fluviatilis</em></td>
<td>Mitchellian hardyhead</td>
</tr>
<tr>
<td><em>Cyprinus carpio</em></td>
<td>European carp</td>
</tr>
<tr>
<td><em>Gadopsis marmoratus</em></td>
<td>river blackfish</td>
</tr>
<tr>
<td><em>Galaxias planiceps</em></td>
<td>flat-headed galaxias</td>
</tr>
<tr>
<td><em>Gambusia affinis</em></td>
<td>mosquito fish</td>
</tr>
<tr>
<td><em>Hypseleotris klunzingeri</em></td>
<td>western carp gudgeon</td>
</tr>
<tr>
<td><em>Maccullochella peeli</em></td>
<td>Murray cod</td>
</tr>
<tr>
<td><em>Nannoperca australis</em></td>
<td>southern pygmy perch</td>
</tr>
<tr>
<td><em>Perca fluviatilis</em></td>
<td>redfin, English perch</td>
</tr>
<tr>
<td><em>Retropinna semoni</em></td>
<td>Australian smelt</td>
</tr>
<tr>
<td><em>Salmo trutta</em></td>
<td>brown trout</td>
</tr>
<tr>
<td><em>Tinca tinca</em></td>
<td>tench</td>
</tr>
</tbody>
</table>
Some special aquatic environments

Table 50  Waterbirds recorded from Hawksview Wildlife Refuge, Albury. List supplied by M. R. Webb (personal communication).

<table>
<thead>
<tr>
<th>Species</th>
<th>Common name</th>
</tr>
</thead>
<tbody>
<tr>
<td>Acanthorynchus tenuirostris</td>
<td>eastern spinebill</td>
</tr>
<tr>
<td>Anas castanea</td>
<td>chestnut teal</td>
</tr>
<tr>
<td>A. gibberifrons</td>
<td>grey teal</td>
</tr>
<tr>
<td>A. rhynchotis</td>
<td>blue-winged shoveller</td>
</tr>
<tr>
<td>A. superciliosa</td>
<td>black duck</td>
</tr>
<tr>
<td>Ardea novaehollandia</td>
<td>white-faced heron</td>
</tr>
<tr>
<td>A. pacifica</td>
<td>white-necked heron</td>
</tr>
<tr>
<td>Aythya australis</td>
<td>white-eyed duck</td>
</tr>
<tr>
<td>Bizura lobata</td>
<td>musk duck</td>
</tr>
<tr>
<td>Botaurus poiciloptilus</td>
<td>brown bittern</td>
</tr>
<tr>
<td>Burhinus magnirostris</td>
<td>bush curlew</td>
</tr>
<tr>
<td>Casarca tadornoides</td>
<td>chestnut-breasted shelduck</td>
</tr>
<tr>
<td>Charadrius bicinctus</td>
<td>double-banded dotterel</td>
</tr>
<tr>
<td>C. melanops</td>
<td>black-fronted dotterel</td>
</tr>
<tr>
<td>Chenonetta jubata</td>
<td>maned goose</td>
</tr>
<tr>
<td>Chlidonias hybrida</td>
<td>marsh tern</td>
</tr>
<tr>
<td>Circus approximans</td>
<td>swamp harrier</td>
</tr>
<tr>
<td>Cygnus atratus</td>
<td>black swan</td>
</tr>
<tr>
<td>Dendrocygna eytoni</td>
<td>plumed tree-duck</td>
</tr>
<tr>
<td>Egretta alba</td>
<td>large egret</td>
</tr>
<tr>
<td>E. garzetta</td>
<td>little egret</td>
</tr>
<tr>
<td>E. intermedia</td>
<td>plumed egret</td>
</tr>
<tr>
<td>Fulica atra</td>
<td>coot</td>
</tr>
<tr>
<td>Gallinago hardwickii</td>
<td>Australian snipe</td>
</tr>
<tr>
<td>Gallinula tenebrosa</td>
<td>dusky moorhen</td>
</tr>
<tr>
<td>Halcyon azurea</td>
<td>azure kingfisher</td>
</tr>
<tr>
<td>H. pyrrhopygia</td>
<td>red-backed kingfisher</td>
</tr>
<tr>
<td>H. sancta</td>
<td>sacred kingfisher</td>
</tr>
<tr>
<td>Himantopus leucocephalus</td>
<td>white-headed stilt</td>
</tr>
<tr>
<td>Isobrychus minutus</td>
<td>little bittern</td>
</tr>
<tr>
<td>Malacorhynchus membranaceus</td>
<td>pink-eared duck</td>
</tr>
<tr>
<td>Megaleurus gramineus</td>
<td>little marshbird</td>
</tr>
<tr>
<td>Numenius madagascariensis</td>
<td>eastern curlew</td>
</tr>
<tr>
<td>Nycticorax caledonicus</td>
<td>nankeen night heron</td>
</tr>
<tr>
<td>Oxyura australis</td>
<td>blue-billed duck</td>
</tr>
<tr>
<td>Pelecanus conspicillatus</td>
<td>pelican</td>
</tr>
<tr>
<td>Phalacrocorax carbo</td>
<td>black cormorant</td>
</tr>
<tr>
<td>P. sulcirostris</td>
<td>little black cormorant</td>
</tr>
<tr>
<td>P. melanoleucus</td>
<td>little pied cormorant</td>
</tr>
<tr>
<td>P. varius</td>
<td>yellow-faced cormorant</td>
</tr>
<tr>
<td>Platalea flavipes</td>
<td>yellow-billed spoonbill</td>
</tr>
<tr>
<td>P. regia</td>
<td>royal spoonbill</td>
</tr>
<tr>
<td>Plegadis falcinellus</td>
<td>glossy ibis</td>
</tr>
<tr>
<td>Podiceps cristatus</td>
<td>crested grebe</td>
</tr>
<tr>
<td>P. poliocephalus</td>
<td>hoary-headed grebe</td>
</tr>
<tr>
<td>P. ruficollis</td>
<td>little grebe</td>
</tr>
<tr>
<td>Porphyrio melanotus</td>
<td>eastern swamp hen</td>
</tr>
<tr>
<td>Strixtonetta naevosa</td>
<td>freckled duck</td>
</tr>
<tr>
<td>Threskiornis molucca</td>
<td>white ibis</td>
</tr>
</tbody>
</table>
of a wide survey program. Nevertheless, sufficient is now known to indicate that billabongs are complex, diverse ecosystems sensitive to external impact and of great biological interest and importance. They are the repositories of a very rich and diverse assemblage of plant and animal life. As such, they are also a source from which man-made, permanent water storages and less permanent natural waterbodies may be colonised. Their conservation and study is strongly supported.

References


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Estuaries
Australian estuaries were reviewed only recently (Bayly, 1975), and, in the interests of brevity, references cited in that work are omitted in this account in the context of estuaries *sensu stricto*. However, some of the earlier references reappear here in relation to coastal lakes. Two recent papers on New Zealand estuaries (McLay, 1976; McDowall, 1976) are also of at least some relevance to the Australian scene.

An estuary may be defined as a coastal body of water, partially enclosed by land, but with a free aquatic connection with the open sea, and having salinities that are outside the range encountered in oceanic water and which are inherently more variable. Typically, estuaries contain brackish water (salinity range c. 3-32°/oo), but in certain regions, including particularly Australia, it is not uncommon for them to have hypersaline water (salinity > c. 38°/oo). In either case the environment may be described as *poikilosaline*—a term which emphasises the ecological importance of salinity change rather than its absolute level.

Detritus and plankton. The basis for estuarine food chains in Australian estuaries, as elsewhere, is not only phytoplankton, but also organic detritus. Estuarine waters are often of limited transparency and consequently rates of primary production by phytoplankton may be low. Despite this, overall production in estuaries is usually very high because of an abundance of detritus from the decay of riparian vegetation in the form of both benthic algae and higher plants. This organic detritus with its associated bacteria is probably a major food for planktonic calanoid copepods which, in turn, are an important food for small estuarine fish and the younger stages of larger ones. Detritus is also of major significance for filter-feeding benthic animals such as bivalve molluscs and even higher crustaceans such as penaeid prawns. It is important to bear in mind that bacteria can be consumed directly by, and constitute a major food for, quite large estuarine animals; sea mullet, *Mugil cephalus*, are nourished mainly by consuming bottom mud and digesting and assimilating the bacteria contained therein. With respect to zooplankton, there are a number of truly estuarine calanoid copepods which often occur in great abundance. In the southern half of Australia, *Gladioferens* spp., *Gippslandia estuarina* and *Sulcanus conflictus* are important, while in northern parts species of *Pseudodiaptomus* take over. Certain species of *Acartia* such as *A. baylyi* may also form part of the true estuarine assemblage. Additionally there are quite a large number of marine euryhaline copepods part of whose populations penetrate to at least some extent into estuaries from the sea. In this group, *Acartia tranteri* (previously referred to
Some special aquatic environments

as *A. clausii*, but see Bradford, 1976), *Calanopia* spp., *Oithona nana*, and *Pseudodiaptomus* spp. all commonly occur in considerable abundance. The cladoceran, *Podon intermedius*, may also be added to this group.

*Gladioferens* is probably mainly herbivorous and detritivorous, whereas *Sulcanus conflictus* and *Acartia tranteri* are at least in part predaceous and capable of feeding on the larvae of *Gladioferens*.

Timms (1976) investigated the Myall Lakes, New South Wales, which basically constitute part of an open estuarine system in which the freshwater zone is well developed, and which includes a lake with an unusually stable salinity regime in the range 1-2°/oo. The zooplankton consisted mainly of *Sulcanus conflictus*, *Gladioferens pectinatus*, *G. spinosus*, the ostracod *Paracypria tenuis*, and decapod larvae. However, small numbers of freshwater forms such as *Boeckella*, *Mesocyclops*, *Chydorus*, *Ceriodaphnia* and *Diaphanosoma* were recorded.

Recently some attention has been paid to Australian estuarine rotifers. Sudzuki and Timms (1977) described a new species of *Brachionus* from Myall Lake. It occurred in a salinity range of 1.4-1.9°/oo, and was invariably associated with the copepods *Gladioferens spinosus* and *Sulcanus conflictus*. Other rotifers recorded from this system were *Asplanchna (Asplanchnnea)* sp., *Rus selletia (?parrotii)*, *Monostyla (?robusta)* and *M. stenroosi*.

**Nekton.** Estuaries serve as breeding and nursery grounds for many powerfully swimming aquatic animals including higher crustaceans and fish.

In the warmer regions of Australia (but not in Tasmania, or to any significant extent in Victoria), several species of *Penaeidae* (the commercial or primitive prawns) enter estuaries as post-larvae and grow to maturity before returning to the sea to spawn. In Western Australia the commercially important estuarine species include *Penaeus latisculatus*, *P. merguiensis* and *Metapeneaus dalli*. In New South Wales and Queensland, *Metapeneaus macleayi* and *M. bennettiae* are both commercially significant. The blue swimming crab, *Portonus pelagicus*, also enters warmer estuaries.

Most fish occurring in estuaries are essentially marine species possessing at least some degree of euryhalinity. Bayly (1975) listed some 70 such species known to enter Australian estuaries. The degree of dependency of these fish on estuaries varies, but, following Lenanton (1974), three main categories may be recognised:

(a) a group of predominantly marine fish such as *Arrapis trutta* (Australian salmon) and *Mustelus antarcticus* (gummy shark) which use estuaries as a source of food when salinity levels are high;

(b) an assemblage, including *Chryso phrys* spp. (snapper), *Achoerodus gouldii* (blue groper) and *Sillaginodes punctatus* (King George whiting), which exploit estuaries as nursery habitats for the young stages; and

(c) a group, including such highly euryhaline species as *Mugil cephalus* (sea mullet) and *Aldrichetta forsteri* (yellow-eye mullet), which depend on estuaries for both food and nursery purposes.

Few Australian fish species are exclusive estuarine residents, spending their
entire life cycle within estuaries and not extending into open sea water. The black bream, *Acanthopagrus butcheri*, however, is one such fish. The estuarine catfish or cobbler, *Cnidoglanis macrocephalus*, may be truly estuarine on a facultative basis, but some populations use the sea as a permanent habitat.

In the southern parts of Australia at least, estuaries play a very significant role in the life cycles of migratory or diadromous fish. Catadromous species—those that migrate downstream to spawn in the sea—include the eels, *Anguilla australis* and *A. dieffenbachii*. *Galaxias maculatus* may be described as marginally catadromous; it migrates downstream as an adult, but the migration ends in the estuary where spawning occurs. Anadromous species—those that migrate upstream to spawn in fresh water—include the lampreys, *Geotria australis* and *Mordacia mordax*, and the common smelt *Retropinna retropinna*. Several other migratory freshwater fish, including several species of *Galaxias* and *Gobiomorphus*, migrate through estuaries in the course of their life history, but since these migrations are not related to spawning they are neither catadromous nor anadromous. Migratory or 'sea run' populations of introduced brown trout, *Salmo trutta*, pass through Tasmanian estuaries.

In tropical Australia, *Lates calcarifer* (barramundi) attains maturity in the freshwater sections of a river but migrates downstream at the time of summer floods to spawn in brackish estuarine waters or coastal bays. After spawning the adults stay in the lower reaches of the river but the young migrate upstream into fresh water where they grow to maturity.

**Benthos.** One of the very few studies of phytobenthos in Australian estuaries is that of Allender (see Bayly, 1975) on the macroflora of the Swan River. Here the most important euryhaline forms associated with hard substrata were *Calothrix parietina* (supralittoral), *Enteromorpha* (littoral, lower estuary), *Cladosiphora* (littoral, upper estuary) and *Polysiphonia macrocarpa* (littoral-sublittoral junction). *Gracilaria verrucosa* was another important euryhaline species which occurred in the sublittoral zone on both soft and hard substrata. Influx of fresh water with concomitant halocline formation in winter resulted in the fragmentation or reduction in size of many species. In comparison with fully marine assemblages there was a disproportionately large representation of chlorophyta species with under-representation of the Phaeophyta and Rhodophyta.

Turning to the zoobenthos, a detailed study of the distribution of foraminifera has been carried out in the Gippsland Lakes by Apthorpe. At the marine end of the system calcareous species dominate, but in low salinities (< 12‰) arenaceous species become important. Apart from salinity, the pH of bottom sediments is believed to be important. Where the pH is less than 6.5 arenaceous species only are found, with values exceeding 8.5 calcareous species only are present, and at intermediate values mixtures of both types may be found.

Crabs associated with estuaries in southern Queensland and New South Wales have been the subject of a number of studies. The following species, all of which are euryhaline, have been reported: *Australoplax tridentata*, *Cleistostoma macneilli*, *Halicarcinus australis*, *H. paralacustris*, *Helice haswel-
lianus, Heloecius cordiformis, Macrophthalmus setosus, Sesarma erythroductyla, Uca longidigitum. It is interesting that H. paralacustris is closely similar morphologically to H. lacustris, a freshwater species, but has a larger number of smaller less yolky eggs and retains indirect development.

A detailed study of the ecology of Xenostrobus securis, a mussel widely distributed in Australian estuaries, has been carried out in the Swan River by Wilson (see Bayly, 1975). It survives drastic decreases in salinity and subject- tion to long periods of virtually fresh conditions by tightly closing its valves and retarding physiological activity. Activity is resumed when the salinity rises above 3\%\text{ Several. It is a true estuarine species (no permanent populations occur in fully marine conditions), with an upper limit of salinity tolerance of 32\%\text{ Several.} Pyrazus ebeninus, a gastropod occurring in east Australian estuaries, behaves in a somewhat similar manner to that of X. securis; it is active over the salinity range 24-34\%\text{ Several.} But at lower salinities it retracts into its shell, and during flood conditions buries itself in the sub-stratum.

The benthos of a shallow estuarine system on the south coast of Western Australia has recently been investigated by Lenanton et al. The bottom is almost completely covered with the filamentous alga Chaetomorpha. The major benthic animals are Haploscoloplos kerguelensis (Polychaeta), Melita sp. (Amphipoda), Palaemonetes australis (Decapoda) and Diala lauta (Gastropoda).

Coastal lakes

The word 'coastal' may be applied to the aquatic environment to refer collectively to all bodies of water, irrespective of salinity, that lie near to the coastline. Bayly (1967) considered that, although some homoiosaline (or stable fresh-water) environments are then lumped with poikilosaline waters, there is a certain, perhaps restricted, biological unity associated with the grouping. In other words, 'coastal' has a validity and is meaningful in aquatic biology as well as geography. This arises because of the existence of certain forms, mainly or wholly derived from euryhaline estuarine ancestors, which inhabit even low conductivity fresh waters, but only those close to the coast. Thus, in Australia, Gladioferens (Copepoda: Calanoida) may usefully be referred to as a coastal water genus; most of the species (and, at any one time, most of the individuals in the taxon) live in estuaries, but G. spinosus is capable of living permanently in coastal fresh waters including closed lakes (this is now known to be a facultative ability, not an obligate requirement). Recent work by Timms (1973a,b, 1977a) provides further examples; the polychaete Boccardia limnicola (Blake and Woodwick, 1976), the calanoid copepod Sulcanus conflictus, and an anthurid and sphaeromid isopod are known from closed coastal freshwater lakes in Victoria. Some factor other than intolerance of low salinity prevents these species from spreading inland. This could be historical and reflect the comparatively recent origin from a euryhaline ancestor of species whose powers of dispersal are not great (see Timms's (1973a) comments on G. spinosus). Alternatively, some ecological factor, other than salinity or degree of salinity variation, that is dependent on close proximity to the sea, determines distributions of this kind. That such a factor could exist is pointed out by Bayly (1967: 93-4).
Coastal lakes that are open to the sea, either permanently or intermittently, and which always contain at least some water of either a brackish or hypersaline nature, are effectively estuarine environments which have been discussed above. Here, consideration will be restricted to permanently closed coastal lakes.

**Closed coastal saline lakes.** Bayly (1972) stressed that the historical aspect of saline aquatic environments cannot be ignored in the assessment of their primary biological status. A permanently closed coastal saline lake which has suffered drying could, on restoration as an aquatic environment, temporarily have a salinity of 35°/00 and major ions present in the same proportions as those of seawater. It would be quite erroneous, however, to think of such an environment as a marine one. If it were originally marine, historical events have transformed it into an athalassic habitat. In coastal regions such a change does not preclude the occurrence of organisms of direct marine affinity. It does, however, alter the long-term ecological significance of such organisms.

Bayly (1970) concluded that in coastal situations the biological status of closed salt lakes should be judged by the *predominating affinity of forms that are persistently present*. If the predominant affinity of all forms ever collected from such habitats were the criterion then they may well have to be designated as marine. This results from the sea functioning as a large reservoir of euryhaline species, some of which may be fortuitously transported and established in a basically foreign environment if it happens to have, at the time, the right order of salinity. A marine designation, however, would be misleading. Because of a general absence of resistant stages, the existence of marine forms is usually short-lived, and over a long period of time their ecological significance in such environments would not be as great as that of athalassic saline species.

Application of these general principles to the Australian scene may be illustrated by reference to lower crustaceans. The most important species are those which also occur in athalassic saline waters remote from the coast (and which are much more closely related to freshwater species than marine ones). They include the following: *Calamoecia salina*, *C. citellata*, *Microcyclops* spp. (Copepoda); *Australocypris* spp., *Diacypris* spp., *Mytilocypris* spp., *Platycypris baueri* (Ostracoda): *Daphniopsis pusilla* (Cladocera). However, the following taxa of basic marine or estuarine affinity may, on occasion, also be encountered: *Acartia tranteri*, *Onychocamptus bengalensis*, *Heterolaophonte wellsi*, *Robertsonia propinqua* (Copepoda); *Cytheroma sudaustralis*, *Microcytherura difficultis* (Ostracoda) (see McKenzie, 1977).

The marine polychaetes *Capitella capitata* and *Ceratonereis erythraeensis* may also occur in closed saline lakes.

The fish *Atherinosoma microstoma*, which occurs in the sea along southern Australian coastlines, also occurs in coastal salt lakes permanently cut off from the sea. This is presumably explicable on the basis of repeated introductions by man.

**Closed coastal freshwater lakes.** A series of dune lakes on Fraser Island was studied by Bayly (1964) and Bayly, Ebsworth and Wan (1975). These were very dilute (mean salinity ~40 mg 1⁻¹),
Some special aquatic environments

...strongly acidic (pH 4.0-6.0), and often humified waters in which the relative proportions of major ions were closely similar to those of sea water. The occurrence in the open water of the calanoid *Calamoecia tasmanica* and a rich assemblage of desmids was almost invariable. Molluscs were absent. Fish present included *Melanotaenia* sp., *Hypseleotris klunzingeri* and *Rhadinocentrus ornatus*. Fish and nektonic hemipterans had mutually exclusive distributions, presumably because notonectids and corixids were eliminated by direct predation by fish, or they were outcompeted by fish for limited food. Terrestrial insects and pollen constituted major foods for *Rhadinocentrus ornatus*.

Timms (1969) studied two lakes located near Grafton, New South Wales. These were less acidic, and had a higher concentration of bicarbonate and total ions than those on Fraser Island, but cationic proportions were very similar. The zooplankton was again dominated by *Calamoecia tasmanica*, but *Mesocyclops leuckarti* and *Bosmina meridionalis* were also present. As with the Fraser Island lakes, molluscs were absent.

A series of coastal lakes in eastern Victoria were investigated by Timms (1973a). They fell into two rather distinct groups: those on the floodplain of the lower Snowy River and those associated with dunes. The latter had very similar characteristics to those on Fraser Island. The floodplain lakes, however, had a greater biotic diversity with usually two species of *Boeckella*, *Daphnia lumholtzi* and other cladocerans, and a variety of cyclopoids and rotifers in the plankton. Timms (1973a, 1973b) showed that Lake Barracoota, in the far east of Victoria, has four species of marine or estuarine affinity despite the fact that it is well cut off from the sea and contains quite fresh water.

Bensink and Burton (1975) made a detailed study of two lakes on North Stradbroke Island, Queensland. Their major ion chemistry was closely similar to that described for Fraser Island lakes, but they differed markedly from each other with respect to pH, degree of humification, and transmission of light. In one the depth to which 10 per cent of incident white light penetrated exceeded 5 m, but in the other it was only 1.2 m. In both lakes, as on Fraser Island, *Calamoecia tasmanica* was the dominant zooplankter. The dark lake apparently lacked fish but contained an abundance of *Chaoborus* larvae and *Anisops*. The clear lake contained two fish, *Rhadinocentrus ornatus* and *Hypseleotris klunzingeri*, and, probably as a consequence, had only small numbers of *Chaoborus*, and no notonectids in the open water.

Timms (1977a) investigated a series of seven coastal lakes near Portland, Victoria, all of which contained fresh water (the maximum concentration of total dissolved solids was 728 mg l\(^{-1}\)). The fauna included three species, *Gladioferens spinosus*, *Sulcanus conflictus* and *Boccardia limnicola*, of definite marine or estuarine affinity to which the crab *Halicarcinus lacustris*, which has close estuarine relatives, should perhaps be added. Predominantly, however, the fauna was a freshwater one.

A 'popular' account of man's influence on some twenty coastal dune lakes in northern New South Wales is given by Timms (1977b).
References


INDEX

Ablabesmyia, 30
  A. notabilis, 37
Acanthocyclops, 384
  A. gigas, 367
  A. vernalis, 384
Acanhopagrus butcheri, 393
Acanthonyapus tenuirostris, 388
Acartia, 391
  A. baylyi, 391
  A. clausii, 392
  A. tranteri, 391, 392, 395
Achnanthes, 74, 75, 329
Achoerodus gouldii, 392
Actinomyces, 77, 78
Actinopoda, 366
Actinosphaerium, 336
Actinotaenium, 329, 331
acute toxicity, 119
adelkräftor, 219
adenovirus(es), 128, 133
Aedes, 140
  A. (Macleaya) tremulus, 140
  A. (Ochlerotatus) normanensis, 139, 144
  A. (Ochlerotatus) tremulus, 139
  A. (Ochlerotatus) vigilax, 139, 144
  A. (Stegomyia) aegypti, 139, 140
Aedomyia catasticta, 140
aerobic ponds, 361
Aeromonas hydrophila, 128
Aegopotaenia, 391
Agapetus, 178
Agrostis avenaceae, 382
Ambassidae, 231
Ambassis castelnau, 241, 258
Ambystoma mexicanum, 280
Amniataba percoides, 258
Amoeba proteus, 336
amoebiasis, 132
amoebic meningo-encephalitis, 128, 133
amphibia(ns), 280, 387
Amphipnous cuchia, 273
ampipod(s), 8, 23, 25, 28, 35, 196, 199, 200, 235, 354, 355
Amphora, 329
Anabaena, 74, 76, 79, 307, 362
  A. circinalis, 329
  A. flos-aquae, 342
  A. solitaria, 329
Anacydatis, 331, 362, 381
  A. cyaneae, 308, 319
anaerobic ponds, 361
Anas castanea, 388
  A. gibberifrons, 388
  A. rhynchotis, 388
  A. superciliosa, 388
Anaspidaeae, 8, 28, 33
Anaspidaeae, 59, 274
Angle Vale, 151
Anguilla, 240
  A. australis, 231, 233, 234, 259, 260, 393
  A. australis occidentalis, 355
  A. bicolor, 231, 233
  A. dieffenbachii, 393

Alligator Rivers, 233
alligator weed, 87, 382
Alonella excisa, 384
A. philoxeroides, 87
Amphipnous cuchia, 273
anaerobic ponds, 361
Ananas castanea, 388
  A. gibberifrons, 388
  A. rhynchotis, 388
  A. superciliosa, 388
Anaspidaeae, 8, 28, 33
Anaspidaeae, 59, 274
Angle Vale, 151
Anguilla, 240
  A. australis, 231, 233, 234, 259, 260, 393
  A. australis occidentalis, 355
  A. bicolor, 231, 233
  A. dieffenbachii, 393
<table>
<thead>
<tr>
<th><strong>Index</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>A. obscura, 231, 233</td>
</tr>
<tr>
<td>A. reihardti, 231, 233, 234, 259</td>
</tr>
<tr>
<td>Anisops, 396</td>
</tr>
<tr>
<td>Ankistrodesmus, 74, 329, 362</td>
</tr>
<tr>
<td>Annelida, 7, 196, 197, 198, 369, 386</td>
</tr>
<tr>
<td>Anodonta dahli, 231</td>
</tr>
<tr>
<td>Anopheles, 144</td>
</tr>
<tr>
<td>A. (Anopheles) bancroftii, 141, 145</td>
</tr>
<tr>
<td>A. (Cellia) amictus, 139, 141, 145</td>
</tr>
<tr>
<td>A. (Cellia) annulipes, 140, 144</td>
</tr>
<tr>
<td>A. (Cellia) farauti, 140, 141, 145</td>
</tr>
<tr>
<td>A. (Cellia) punctulatus group, 140</td>
</tr>
<tr>
<td>A. (Cellia) hyrcanus, 145</td>
</tr>
<tr>
<td>Anostraca, 8</td>
</tr>
<tr>
<td>Anoxic mud, 53</td>
</tr>
<tr>
<td>Anthurid, 30, 394</td>
</tr>
<tr>
<td>Antipodrilus davidis, 25, 29, 33</td>
</tr>
<tr>
<td>Antocha, 178</td>
</tr>
<tr>
<td>Anura(n), 229</td>
</tr>
<tr>
<td>Aquaculture, 216-26; intensive, 217</td>
</tr>
<tr>
<td>Aquatic: vectors, 130; weeds, 81</td>
</tr>
<tr>
<td>Arachnida, 197, 369, 386</td>
</tr>
<tr>
<td>Arachnida, 197, 369, 386</td>
</tr>
<tr>
<td>Arcella, 336</td>
</tr>
<tr>
<td>Archerfish, 232</td>
</tr>
<tr>
<td>Ardea novae-hollandia, 388</td>
</tr>
<tr>
<td>A. pacifica, 388</td>
</tr>
<tr>
<td>Ariid, 229</td>
</tr>
<tr>
<td>Arripis trutta, 392</td>
</tr>
<tr>
<td>Artemia salina, 8</td>
</tr>
<tr>
<td>Artuhs Lake, 28, 46, 49</td>
</tr>
<tr>
<td>Artificial destratification, 52</td>
</tr>
<tr>
<td>Ascaris lumbricoides, 129</td>
</tr>
<tr>
<td>Ascellminthes, 386</td>
</tr>
<tr>
<td>Asellidae, 8</td>
</tr>
<tr>
<td>Asellota, 8</td>
</tr>
<tr>
<td>Aseraggodes klunzingeri, 232</td>
</tr>
<tr>
<td>Asplanchna, 367, 383</td>
</tr>
<tr>
<td>A. (Asplanchnella), 392</td>
</tr>
<tr>
<td>A. brightwelli, 336, 384</td>
</tr>
<tr>
<td>A. priodonta, 313</td>
</tr>
<tr>
<td>Assiminea, 143</td>
</tr>
<tr>
<td>Astacidae, 8</td>
</tr>
<tr>
<td>Astacopsis gouldi, 168, 238</td>
</tr>
<tr>
<td>Astacus astacus, 219</td>
</tr>
<tr>
<td>Asterionella, 76, 329, 331</td>
</tr>
<tr>
<td>Atalophlebia, 386</td>
</tr>
<tr>
<td>Atalophlebioides, 197, 198</td>
</tr>
<tr>
<td>Athalassic lakes, 19, 298, 299, 395</td>
</tr>
<tr>
<td>Atherinidae, 216</td>
</tr>
<tr>
<td>Atherinosoma microstoma, 395</td>
</tr>
<tr>
<td>Atherton Tableland, 30</td>
</tr>
<tr>
<td>Attheya, 329</td>
</tr>
<tr>
<td>A. zachariasii, 334</td>
</tr>
<tr>
<td>Attheyella, 384</td>
</tr>
<tr>
<td>A. australica, 384</td>
</tr>
<tr>
<td>A. incerta, 384</td>
</tr>
<tr>
<td>Atyidae, 386</td>
</tr>
<tr>
<td>Aulosira, 364</td>
</tr>
<tr>
<td>Australian Fisheries Council, 220</td>
</tr>
<tr>
<td>Australian Heritage Commission, 169</td>
</tr>
<tr>
<td>Australian Water Quality Criteria, 117</td>
</tr>
<tr>
<td>Australian Water Resources Council, 3</td>
</tr>
<tr>
<td>Australocypris, 296, 297, 299, 300, 395</td>
</tr>
<tr>
<td>A. robusta, 33</td>
</tr>
<tr>
<td>Australoplax tridentata, 393</td>
</tr>
<tr>
<td>Austrobiharzia terrigalensis, 141</td>
</tr>
<tr>
<td>Austrochiltonia, 25</td>
</tr>
<tr>
<td>A. australis, 200, 201</td>
</tr>
<tr>
<td>A. subtenus, 23, 26, 200, 201</td>
</tr>
<tr>
<td>Austroperlidiae, 8</td>
</tr>
<tr>
<td>Avifauna, 9, 370</td>
</tr>
<tr>
<td>Avon Reservoir, 324, 325, 327</td>
</tr>
<tr>
<td>Avon River, 167</td>
</tr>
<tr>
<td>Azolla, 379, 382</td>
</tr>
<tr>
<td>A. filiculoides, 382</td>
</tr>
<tr>
<td>A. pinnata, 382</td>
</tr>
<tr>
<td>Azolla: ferny, 382; Pacific, 382</td>
</tr>
<tr>
<td>Bacillariophyceae, 74, 329</td>
</tr>
<tr>
<td>Bacterial infections, 133</td>
</tr>
<tr>
<td>Barbeis, 386</td>
</tr>
<tr>
<td>Barmah Lakes, 250</td>
</tr>
<tr>
<td>Barramundi, 39, 219, 220, 221, 232, 233, 236, 393; northern spotted, 231; silver, 220, 227, 228, 230, 231, 232, 239, 265; spotted, 216, 231, 265</td>
</tr>
<tr>
<td>Barrine, Lake, 24, 31, 35, 36</td>
</tr>
<tr>
<td>Bass, 234, 235, 236, 369; Australian, 216, 228, 231, 233, 239, 244, 265</td>
</tr>
<tr>
<td>Bass Strait Islands, 14</td>
</tr>
<tr>
<td>Bather's itch, 133</td>
</tr>
<tr>
<td>Bathurst Harbour, 298</td>
</tr>
<tr>
<td>Baumea rubiginosa, 60</td>
</tr>
<tr>
<td>Bdelloidea, 367</td>
</tr>
<tr>
<td>Beardy River, 252</td>
</tr>
<tr>
<td>Beenyup, 151</td>
</tr>
<tr>
<td>Beer Lambert Law, 70</td>
</tr>
<tr>
<td>Beetle, 199, 252, 355</td>
</tr>
<tr>
<td>Index</td>
</tr>
<tr>
<td>----------------------------------------------------------------------</td>
</tr>
<tr>
<td>Beijerinck’s dictum, 47</td>
</tr>
<tr>
<td>Belastomatidae, 386</td>
</tr>
<tr>
<td>belut, 273</td>
</tr>
<tr>
<td>beneficial uses for water, 118</td>
</tr>
<tr>
<td>bent grass, 382</td>
</tr>
<tr>
<td>benthic invertebrates, 174, 195</td>
</tr>
<tr>
<td>benthos, 23, 353, 393, 394</td>
</tr>
<tr>
<td>Bermuda grass, 382</td>
</tr>
<tr>
<td>Biapertura, 383</td>
</tr>
<tr>
<td>B. affinis, 367, 384</td>
</tr>
<tr>
<td>B. intermedia, 384</td>
</tr>
<tr>
<td>B. karua, 384</td>
</tr>
<tr>
<td>B. kendallensis, 384</td>
</tr>
<tr>
<td>B. rigidicaudis, 384</td>
</tr>
<tr>
<td>B. setigera, 384</td>
</tr>
<tr>
<td>bidyan, 247</td>
</tr>
<tr>
<td>Bidyanus bidyanus, 247</td>
</tr>
<tr>
<td>Big Jim, Lake, 46, 63</td>
</tr>
<tr>
<td>bilharzia, 128</td>
</tr>
<tr>
<td>bioaccumulation, 120, 194</td>
</tr>
<tr>
<td>bioassays, 120, 193, 195, 197, 199, 200, 201</td>
</tr>
<tr>
<td>biogenic deposits, 44</td>
</tr>
<tr>
<td>biogeochemical cycles, 155, 156, 157</td>
</tr>
<tr>
<td>biological: control, 87; indicators, 174; monitoring, 192, 193, 195, 196, 197, 198, (values of), 192, 199, 200, 201; succession, 355</td>
</tr>
<tr>
<td>Biophialaria, 141</td>
</tr>
<tr>
<td>Birch report, 4</td>
</tr>
<tr>
<td>bittern: brown, 388; little, 388</td>
</tr>
<tr>
<td>bivalves (molluscs), 7, 24, 28, 32, 33, 142, 196, 197, 198, 199, 200, 201</td>
</tr>
<tr>
<td>Bizauro lobata, 388</td>
</tr>
<tr>
<td>blackberries, 168</td>
</tr>
<tr>
<td>blackfish, 59, 251, 252, 274; river, 241, 244, 251, 252, 259, 260, 265, 387; Tasmanian, 265</td>
</tr>
<tr>
<td>bladderwort, 382</td>
</tr>
<tr>
<td>Blanchards perchlet, 231</td>
</tr>
<tr>
<td>bloom(s), 75, 206, 315, 333, 342, 347, 381</td>
</tr>
<tr>
<td>blue eyes, 231</td>
</tr>
<tr>
<td>bobby cod, 251</td>
</tr>
<tr>
<td>Boccardia limnicola, 30, 394, 396</td>
</tr>
<tr>
<td>Boeckella, 8, 352, 382, 392, 396</td>
</tr>
<tr>
<td>B. fluvialis, 325, 336, 353, 384</td>
</tr>
<tr>
<td>B. minuta, 313, 325, 336, 355, 384</td>
</tr>
<tr>
<td>B. propinquia, 313</td>
</tr>
<tr>
<td>B. symmetrica, 336, 340</td>
</tr>
<tr>
<td>B. triarticulata, 19, 308, 319, 336, 367, 384</td>
</tr>
<tr>
<td>Bolivar, 151</td>
</tr>
<tr>
<td>Borumba Dam, 313</td>
</tr>
<tr>
<td>Bosmina, 340, 352, 383</td>
</tr>
<tr>
<td>B. longirostris, 336, 367</td>
</tr>
<tr>
<td>B. meridionalis, 308, 319, 325, 336, 353, 384, 396</td>
</tr>
<tr>
<td>Bosminopsis deitersi, 325</td>
</tr>
<tr>
<td>Bostockia porosa, 231</td>
</tr>
<tr>
<td>Botaurus poicilloptilus, 388</td>
</tr>
<tr>
<td>Botryococcus, 329, 331</td>
</tr>
<tr>
<td>B. braunii, 69, 319, 338, 340</td>
</tr>
<tr>
<td>Branchionus, 367, 383, 384, 392</td>
</tr>
<tr>
<td>B. angularis, 384</td>
</tr>
<tr>
<td>B. budapestinensis, 384</td>
</tr>
<tr>
<td>B. calyciflorus, 336</td>
</tr>
<tr>
<td>B. calyciflorus calyciflorus, 384</td>
</tr>
<tr>
<td>B. calyciflorus var. amphiceros, 384</td>
</tr>
<tr>
<td>B. diversicornis, 336</td>
</tr>
<tr>
<td>B. novae-zealandia, 384</td>
</tr>
<tr>
<td>B. plicatilis, 384</td>
</tr>
<tr>
<td>B. quadridentatus, 336</td>
</tr>
<tr>
<td>B. quadridentatus melheni, 384</td>
</tr>
<tr>
<td>B. urceolaris rubens, 384</td>
</tr>
<tr>
<td>B. urceolaris urceolaris, 384</td>
</tr>
<tr>
<td>Brachirus salinarum, 232</td>
</tr>
<tr>
<td>B. selheimi, 232</td>
</tr>
<tr>
<td>Branchiodrilus hortensis, 34</td>
</tr>
<tr>
<td>Brachiura sowerbyi, 23</td>
</tr>
<tr>
<td>Brasenia schreberi, 382</td>
</tr>
<tr>
<td>bream, 247; black, 247, 393; bony, 228, 240, 243, 249, 254, 255, 258</td>
</tr>
<tr>
<td>Bridgewater Lakes, 30</td>
</tr>
<tr>
<td>Broken Hill, 152</td>
</tr>
<tr>
<td>Brown, Lake, 24, 29</td>
</tr>
<tr>
<td>Bufo, 280, 283</td>
</tr>
<tr>
<td>B. marinus, 9, 280, 281, 282, 284</td>
</tr>
<tr>
<td>Bulinus, 141, 197</td>
</tr>
<tr>
<td>Bulenmerri, Lake, 24, 28</td>
</tr>
<tr>
<td>bumblebee (fish), 232</td>
</tr>
<tr>
<td>Burdekin River, 30</td>
</tr>
<tr>
<td>Burhinus magnirostris, 388</td>
</tr>
<tr>
<td>Burragerang Reservoir, 325, 327, 328</td>
</tr>
<tr>
<td>Burrinjuck Reservoir, 253</td>
</tr>
<tr>
<td>Burrumbeet, Lake, 23</td>
</tr>
<tr>
<td>buttercup, small river, 382</td>
</tr>
<tr>
<td>button grass, 47</td>
</tr>
<tr>
<td>cacao, 284</td>
</tr>
<tr>
<td>caddis: flies, 178, 179, 198, 199, 252; larvae, 196</td>
</tr>
<tr>
<td>cadmium, 16, 154, 197, 198, 201</td>
</tr>
<tr>
<td>caecelians, 9</td>
</tr>
<tr>
<td>Caenidae, 178</td>
</tr>
<tr>
<td>Calamoceratidae, 386</td>
</tr>
<tr>
<td>Calamoecia, 8, 352, 382</td>
</tr>
<tr>
<td>C. ampulla, 308, 325, 336</td>
</tr>
<tr>
<td>C. australis, 384</td>
</tr>
<tr>
<td>C. citellata, 395</td>
</tr>
<tr>
<td>C. lucasi, 319, 325, 336, 340, 384</td>
</tr>
<tr>
<td>C. salina, 395</td>
</tr>
<tr>
<td>C. tasmanica, 396</td>
</tr>
</tbody>
</table>
Calanoida, 8, 336, 391, 394, 396
Calanopia, 392
Callistemon viridiflorus, 60
Callistocythere, 298
Callitriches, 381
C. stagnalis, 382
C. umbonata, 382
callop, 220, 244
Calothrix parietina, 393
Campaspe River, 335
Camptocercus australis, 384
Campylodiscus, 338
Candona, 296, 384
C. assimilis, 367
Candonopsis, 296, 297
cane toad, 3, 280, 282, 283, 284, 285
capitala, 395
Captains Flat, 320
Carassius, 219, 272
C. auratus, 189, 240, 253, 271, 272, 369, 387
C. carassius, 240, 271, 272
Carcoar Reservoir, 77
Carex appressa, 60, 61
carp, 3, 97, 219, 223, 237, 253, 254, 256, 271, 275, 276; Asian, 253; crucian, 240, 244, 271; European, 59, 97, 189, 223, 227, 228, 240, 243, 244, 246, 249, 253-9 passim, 271, 272, 274, 276, 308, 320, 355, 387; German, 253; golden, 189, 387; grass, 97, 276, 369
carp family, 9
carp gudgeon, 232, 355
Carrum aquifer, 152
Casarca tadornoides, 388
Cataract Reservoir, 324, 325, 327
Cataract River, 324
catchment: control, 315; management, 100; usage, 100, 101, 167
catfish, 189, 223, 229, 239, 244, 248, 249, 265; estuarine, 393; fork-tailed, 231; freshwater, 228, 240, 248; Obbes, 231; toothless, 231
Central Australian goby, 258
Central Plateau, 43, 44
Centralres, 74
Centropagidae, 8
Centropillum, 178, 179
Centropomus undecimalis, 230
Centropyxis, 336
Cerarium, 68, 74, 75, 329
C. hirundella, 71, 334
Ceratodidae, 229
Ceratoncles erythraeensis, 395
Ceratophyllum, 381, 383
C. demersum, 382
Ceratopogonidae, 27, 29, 30, 32, 178, 386
cercariae, 141, 143
Ceriodaphnia, 308, 340, 342, 352, 383, 392
C. cornuta, 336, 353, 384
C. dubia, 384
C. laticaudata, 384
C. quadrangula, 319, 325, 336, 384
C. rigaudi, 367
Chaetogaster, 386
Chaetomorph, 394
Chaetophrora, 74
Chao borinae, 35, 36, 59
Chao borus, 24, 25, 29, 30, 31, 33, 340, 396
Chao sof, 386
Charadrius bicinctus, 388
C. melanops, 388
Charales, 74
Chelodina, 9
C. longicollis, 387
chemical variability, 17, 18, 19
chemically induced diseases, 130
chemistry of Australian waters, 12
chemocline, 53, 55, 56
Chenonetta jubata, 388
Cherax, 386, 387
C. destructor, 256
C. tenuimanus, 348, 353
C. wasselli, 34
Chironomidae, 23, 24, 25, 26, 27, 28, 29, 30, 32, 33, 34, 35, 36, 37, 178, 196, 197, 249, 353, 354, 369, 370, 386
Chironomus, 30, 37, 353
C. bathophilus, 37
C. benthophilus, 37
C. duplex, 23, 25, 26, 27, 31, 33, 37
C. nepeanensis, 30, 37
C. oppositus, 28, 29, 33, 37
C. plumosus, 37
Chlamydia, 133
C. trachomatis, 128
Chlamydotheciales, 49
Chlamydogobius eremius, 258
Chlamydomonas, 74, 329, 362, 364, 381
Chlamydotheca, 296, 298, 300
Chlildionias hybrida, 388
Chlorella, 362, 364, 368
chlorination, 58
chlorine, 77
Chlorococcales, 74
Chlorohydra, 386
Chlorophyceae, 19, 74
Index

Chlorophyta, 74, 329, 362
cholera, 129
Chorisandra cymbaria, 60
Chroococcales, 74
Chrysonomadales, 74
Chrysum, 392
Chrysumphyceae, 74
chydorids, 384
Chydrorus, 383, 392
C. eurynotus, 353, 384
C. sphaericus, 336, 367, 383, 384
Ciliata, 308, 334, 336, 366
Circus approximans, 388
cirrhosis, 142
Cladopelma curtivalva, 25
Cladophora, 393
Clarence River, 244, 246
Cladocera, 23, 34, 142, 237, 240, 253, 325, 340, 336, 352, 354, 368, 383, 384, 395
Cladophora, 393
Clarence River, 244, 246
Cleistostoma macneilli, 393
Climacostomum, 308, 334, 336
Clinotanypus crus, 34
Cloeon, 386
Clonorchis, 143
C. sinensis, 143
Closteridium, 381
Closteriopsis, 329
Closterium, 74, 329, 381
C. aciculare, 338
Club, Lake, 29
Clupeidae, 249
Coliban River, 335
Collembola, 386
Cox River, 196, 296
Coxiella striata, 28, 34
Coxs River, 196, 296
Crab Tree Rivers, 170
cobbler, 239, 393; freshwater, 231
cod: marbled river, 251; Murray, 166, 216, 220, 223, 228, 241, 243, 244, 246, 247, 253, 255, 259, 265, 275, 387; slimy river, 251; slippery river, 251; Tailor river, 251
Coelastrum, 329
Coelenterata, 7, 386
Coelopynia, 30
C. pruinosa, 23, 25, 28, 37
Coleoptera, 178, 199, 354, 356, 386
Coleps, 336
Coliban River, 335
Collembola, 386
Colo River, 170
Colo Wilderness, 170
Coloburiscoides, 178
colour (of water), 15, 18, 58, 75
commercial catches of Australian freshwater fishes, 228
Commonwealth Advisory Committee on Imports of Live Food and Sport Fish, 220
Commonwealth Fishing Industry Research Trust Account, 216
Conchostraca, 30, 33
congolli, 232, 241, 260
Conjugatophyceae, 74
conjunctivitis, 128, 133
Conochilus, 308, 342, 383
C. dossumauris, 336, 384
C. unicornis, 336
Conochironomus, 30
conservation, 60, 62, 163, 169, 378
Cooks Lagoon, 29
Cooper Creek, 289
Coorong, 299
coot, 388
Cootapatamba, Lake, 29
Copepoda, 8, 23, 253, 308, 325, 336, 340, 352, 354, 368, 384, 391, 394, 395
copper, 120, 197, 198
Coraliculidae, 7
Corethra, 142
Corixidae, 354, 355, 356, 384, 396
cormorant, 256; black, 388; little black, 388; little pied, 388; yellow-faced, 388
Cosmarium, 74, 328, 329, 334, 340, 381
Coxs River, 196, 296
Coxiella striata, 28, 34
crabs, 143, 235, 393; blue swimming, 392; freshwater, 143
Craspedacusta sowerbyi, 308, 336, 386
Crasella helmsii, 382
Cretorinecephalus, 275
C. dalhousiensis, 258
C. eyresii, 241, 258
C. fluviatilis, 241, 258, 387
crayfish, 8, 164, 216, 218, 219, 222, 228, 229, 239, 244, 249, 253, 255, 256, 345, 355; Murray, 256; Tasmanian freshwater, 168; Tasmanian giant, 238, 266
criteria, water quality, 117
critical organisms, 120
crocodile, 9, 168
Crocodylus johnstoni, 9, 168
C. porosus, 168
Crucigenia, 329
Cryptochironomus griseidorsum, 25, 33
Cryptocladopelema, 30
Cryptomonadales, 74
Cryptomonads, 74, 307, 329
Cryptomonas, 74, 307, 329
Cryptophyceae, 74
Cryptophyta, 74, 329
CSIRO, 3
Ctenopharyngodon idella, 97, 276
cuchia, 273
Culex (Culex) annulirostris, 139, 140, 145
  C. (Culex) bitaeniorhynchus, 139, 145
  C. (Culex) fatigans, 139, 140, 141
  C. (Culex) squamosus, 139
  C. (Culex) tritaeniorhynchus, 145
Culicidae, 386
cultural eutrophication, 91, 158
cumbungi, 82, 84
Cupelopagis vorax, 384
curlew: bush, 388; eastern, 388
Cyanarcus, 329
Cyanophyceae, 74
Cyanophyta, 74, 329, 362
Cyathura, 30
Cyclestheria hislopi, 30, 34
cyclic salts, 13, 18
Cyclocypris, 296
Cyclopoida, 8, 336, 383, 396
Cyclops vernalis, 367
  C. vicinus, 367
Cyclorana australis, 281
  C. novae-hollandiae, 281
Cyclotella, 74, 76, 328, 329, 331, 338
  C. stelligera, 338
Cygnus atratus, 388
Cymbella, 329
Cyphodema gambosa, 30
Cyprinodon dactyclon, 382
Cyprus, 353
Cypretta, 296, 297, 298, 300, 336, 384
Cyprideis, 296, 298, 300
Cypridopsis, 296, 298
  C. australis, 353
  C. funebris, 384
Cyprinidae, 9, 253, 255, 369
Cypriniformes, 229
  C. leana, 384
Cyprinus carpio, 59, 189, 240, 253, 254, 271, 272, 308, 320, 368, 369, 387
Cypris, 296
  C. bennelong, 384
Cythera, 296
  C. sudaussralis, 395
d'Arcithompsoniuid, 384
Dacelo gigas, 284
Dactylococcopsis, 329
damselflies, 199
Dandenong Creek, 197
Daphnia, 120, 340, 342, 352, 356, 368, 382
  D. carinata, 308, 319, 325, 342, 352, 367
  D. carinata sensu lato, 336
  D. catawba, 367
  D. longispina, 367
  D. lumholtzi, 325, 336, 353, 384, 396
  D. magna, 367
  D. pulex, 142, 367
  D. thomsoni, 336
Daphniopsis pusilla, 395
Darling River, 140, 239, 242, 256, 376, 379
Dartmouth Dam (or Reservoir), 55, 56, 174, 178, 179, 189, 309, 332, 334, 335, 336, 340, 341
Darwinula, 296, 298
Dasyurus, 284
  D. geoffroii, 284
Dawson River, 244, 246
DDE, 196
DDT, 196
Debarya, 381
Decapoda, 7, 196, 249, 392, 394
deforestation, 101
Delatite River, 332
Denarius bandata, 258
Dendrocygna eytoni, 388
dengue fever, 138, 139, 140
Derwent River, 58
Derwent estuary, 58, 121
desertification, 102
Desmidales, 74
Desmidium, 329
desmids, 328, 333, 334, 340, 396
destratification, 57, 338, 339, 340, 342
detergents, 94
Diacypris, 296, 297, 298, 299, 300, 384, 395
Diala lauta, 394
Diamantina River, 289
Diaphanosoma, 367, 382, 392
  D. excisum, 325, 336, 353
  D. unguiculatum, 336, 384
Diaptomidae, 8
Diaptomus gracilis, 367
diatom(s), 19, 74, 76, 196, 307, 308, 313, 319, 328, 334, 338, 340
Dicrotendipes, 30
Dictyosphaerium, 329, 334
Diffugia, 336
digested sludge as a resource from sewage, 153
dimictic, 7
Index

Dinobryon, 74, 328, 329, 340
   D. divergens, 334
   D. sertularia, 334
dinoflagellates, 68, 334
Dinophyceae, 74, 329
Dinotoperla, 178, 196
Diptera, 30, 33, 142, 178, 198, 199, 249, 356, 386
discharge of sewage to estuaries, 159
diseases with aquatic vectors and hosts, 137
disposal at sea, 160
dissolved organic material, 15
distinctive biological features, 7
distinctive physico-chemical features, 6
distribution system, 77, 131
doctor fish, 255
domestic reticulation, 132
dotterel: black fronted, 388; double-banded, 388
Dove Lake, 28, 35, 36
Dracunculus medinensis, 129
dragonflies, 9, 199
drainage systems, 229, 230
dual municipal water-supply systems, 152
duck-billed platypus, 9
ducks: black, 290, 388; blue-billed, 388; freckled, 388; musk, 388; pink-eared, 388; plumed tree, 290, 388; white-eyed, 388; wood, 290; see also waterfowl
duckweed: common, 382; thin, 382
dumping wastes at sea, 159
dune lakes, 32, 34, 35, 170, 395
Dunhevedia crassa, 353, 384
Dutson Downs, 151
dysentery, 142
dystrophic waters, 59
dytiscidae, 386

Eacham, Lake, 30, 31, 36
East Basin Lake, 27
Echinisca, 384
Echinostoma ilocanum, 143
ecological: stability, 109; systems, 155
economic value of waters, 164
ecosphere, 155
ecosystems, 155
Ectocyclops medius, 384
educational value of waters, 165
Edward, Lake, 23, 26
eel grass, 382
eels, 216, 220, 227, 233, 234, 244, 259, 260, 393; glass, 234; long-finned, 231, 233, 234; northern, 231; short-finned, 216, 228, 231, 233, 234, 238, 239, 260, 355; silver, 234; south-Pacific, 231
eeltail, 248
effluent (or discharge) standards, 122
effluent reuse, 150
effluents, 59
egret: large, 388; little, 388; plumed, 388
Egretta alba, 388
   E. garzetta, 388
   E. intermedia, 388
Eichhornia crassipes, 83, 145, 168, 313
Eildon Reservoir (Lake), 163, 168, 332-9 passim
Elakatothrix, 329
Elaphoidella, 384
Eleocharis, 381
   E. phacelata, 353, 382
   E. tuberosa, 143
Eleotridae, 232
elephantiasis, 141
Elodea, 84
   E. canadensis, 84, 319, 382
Elseya, 9
elvers, 216, 220, 234
Empididae, 178
Emydura, 9
   E. macquarii, 387
endangered fish species, 264
endrin, 201
energy as a resource from sewage, 153
Engineering and Water Supply Department, 63
Entamoeba histolytica, 129, 132
Enteromorpha, 393
environmental: impact, 174; impact statement (EIS), 123, 163, 169; indicators, 209; monitoring, 164; planning, 123; survey, 174, 205, 207, 209
Ephemeropertera, 8, 24, 32, 178, 196, 197, 198, 201, 386
Epiphanes clavulata, 384
Eppalock Reservoir, 102, 332-9 passim
Escherichia coli, 129, 333
estuaries, 159, 298, 391, 392, 393
estuarine food chains, 391
Etosha Pan, 299
Eustacius, 256
   E. armatus, 256
Eucalyptus, 60
   E. camaldulensis, 305
   E. maculata, 311
   E. pilularis, 311
   E. signata, 311
Euchlanis, 336, 383
   E. incisa, 384
Eucyclops, 383, 384
Index

E. agilis, 384
E. euacanthus, 384
E. speratus, 367
Eucypris, 298, 300
Eudorina, 329
Euglena, 74, 329, 362, 381
Euglenales, 74
Euglenophyceae, 74
Euglenophyta, 74, 329, 362
Euglypha, 336
Eurytemora affinis, 367
Eustheniidae, 8
eutrophic, 34, 35, 36, 37, 43
eutrophication, 6, 16, 43, 91, 100, 103, 158, 159, 167, 197, 313, 315
excessive weed growth, 90, 206
exotic biota, 168, 266
extensive aquaculture, 217

facultative ponds, 361
farm dams, 18, 31, 32, 43, 345
Farmers Creek, 196, 296
Fasciola hepatica, 138, 142, 143, 144
fascioliasis, 143
Fasciolopsis, 143
F. buski, 143
filariosis, 141, 144
Filinia, 336, 367, 383
F. longiseta, 384
F. terminalis, 384
fish: diseases, 277; farms, 218; ladders, 220, 221, 236, 239, 265, 266
fish-farming, 215
fisheries management, 227, 263
Fitzroy Falls Reservoir, 325, 330, 331
Fitzroy River, 244, 248, 255, 298
flagellates, 334, 340, 342
flamingos, 9
flatworm, 29
floods, 166
fluke, 141, 143
Fluta alba, 273
Fluvialosa, 249
forage fish, 256
foraminiferans, 393
forested catchments, 101
Forth River, 52
Franklin River, 62, 170
Fraser Island, 167, 395, 396
frog, 9, 376, 387; African clawed, 280; eastern banjo, 387; Ewing’s tree, 387; golden bell, 387; spotted grass, 387; Verreaux’s tree, 387
Frontipoda, 178
Fulica atra, 388

Gadopsidae, 251
Gadopsis marmoratus, 241, 251, 259, 274, 387
G. tasmanicus, 59
Gahnia clarkei, 382
Galaxias, 231, 238, 393
G. findlayi, 29
G. maculatus, 200, 201, 238, 240, 256, 393
G. olidus, 240
G. pedderensis, 274
G. planiceps, 240, 387
galaxies: common, 240; flat-headed, 387
Galaxiidae, 9, 231, 238, 259, 260, 264, 265, 274
Gallinago hardwickii, 388
Gallinula tenebrosa, 388
Gambusia, 275
G. affinis, 240, 257, 258, 271, 272, 273, 355, 368, 387
G. dominicensis, 257, 258, 273
gammariad, 28, 29
gastroenteritis, 131, 134
Gastropoda, 7, 23, 24, 27, 30, 32, 196, 197, 198, 199, 353, 386, 394
Gastrotricha, 7, 23, 386
Gaviformes, 9
Geehi River, 183
Georgina River, 289
goosmin, 76
Geotria, 231
G. australis, 393
Gerridae, 386
Giardia lambia, 129, 131
giardiasis, 131
Gippsland Lakes, 205, 207, 208, 393
Gippslandia estuarina, 391
glacial lakes, 12, 32, 33
Glacidorbis, 386
G. hedleyi, 29
Gladioferens, 391, 392, 394
G. pectinatus, 392
G. spinosus, 392, 394, 396
Gladyes, Lake, 24
glassfishes, 231
Glenbawn Dam, 247
Glenelg sewage plant, 152
Gloeocystis, 329
Glossosogobius giurus, 258
Glossosomatidae, 178
Gnotuk, Lake, 24, 27, 28
Gobiidae, 232, 238
Gobiomorphus, 393
goby: Central Australian, 258; desert, 257; flathead, 258
gold dredging, 167
goldfish, 219, 221, 228; 240, 244, 253, 255, 271, 272, 277, 387
Gomphidae, 178
Gomphocythere, 295, 296, 298, 300
Gomphonema, 329
goose, maned, 388
Gordon River, 62, 170
Goulburn River, 163, 247, 256, 332, 376-80
G. pascim, 384, 385, 387
Gracemere Lagoon, 298
Gracilaria verrucosa, 393
Graptoleberis testudinaria occidentalis, 384
grassland catchments, 102
grayling, 233-8 passim; Australian, 231, 236, 239, 265; New Zealand, 236
Great Lake, 24, 28, 31, 32, 45, 46, 47, 49
Great Slave Lake, 31
grebe: crested, 388; little, 388; hoary-headed, 388
Green Lake, 96
Gripopterygidae, 8, 178
groper, blue, 392
grunter, 231, 247; Barcoo, 258; black striped, 258; Gilberts, 231; northern, 247; spangled, 251; Welch’s, 241, 244, 258
guanotrophy, 370
gudgeon, 232, 256; firetail, 241; flat-headed, 241; purple spotted, 241; purple striped, 258; snakehead, 232; western carp, 166, 241, 387
gully dams, 345
guppy, 121, 273
Gymnodinium, 329
Gymnoschoenus sphaerocephalos, 47
Gyrinidae, 386
Gyrosigma, 329
Halcyon azurea, 388
H. pyrrhopygia, 388
H. sancta, 388
Halicaricus australis, 393
H. lacustris, 394, 396
H. paralacustris, 393, 394
Halimiscus searlei, 28, 33
Haploscoloplos kerguelensis, 394
hardyhead; Dalhouse’s, 258, Mitchellian freshwater, 241, 257, 258, 387
Hartz Lake, 28
Hawksview Wildlife Refuge, 389
heart disease, 130
heavy metals, 59, 119, 120, 121, 130, 154, 197, 198, 199, 201
Helicothrix cordiformis, 394
Helice haswellianus, 393
helminth eggs, 153
Helmintidae, 178, 179
Hemianax, 386
Hemiboeckella searlei, 384
Hemicordulia, 386
H. tau, 25
Hemicypris, 296, 297, 298
Hemiplebia mirabilis, 9
Hemiplebiidae, 9
Hemiptera, 196, 197, 198, 199, 354, 386, 396
hepatitis, 129, 131, 134; infectious, 134
hepatosplenomegaly, 142
Hephaestus, 257, 258
H. welchi, 241, 258
herbicides, 81, 86
H. pyrrhopygia, 388
Himantopus leucocephalus, 388
Hippocampus crassus, 23
herring: freshwater, 231, 265, hair back, 249
Heterocypris, 296, 297, 298
Heterolaophonte wellsi, 395
Heterophyes, 143
heterophyid myocarditis, 143
heterotrichs, 308
Hexanematosoma, 231
Hexarthra, 336
Hiawatha, Lake, 24, 36
highland lakes, 29
hillside dams, 345
Himantopus leucocephalus, 388
Hippocampus crassus, 236
Hirudinea, 8, 33, 386
Histriobdellidae, 7
Hornwort, 382
Hubbard Brook Experimental Forest, 101
Hume Dam (Reservoir, Lake), 182-8 passim, 253, 305, 306, 307, 308, 309, 379, 387
humic: acids, 47; lakes, 35
Hunter River, 247
hunters, hunting, 290, 291, 292, 293
Hyalothece, 329
Hydra, 386
Hydrilliformes, 24, 33, 386
Hydrilla verticillata, 313, 352
Hydro-Electric Commission, 44
hydrogen sulphide, 52, 53, 55, 58
Hymenobdellidae, 386
hydrophytes, 381, 382, 383, 386
Hydropsychidae, 178, 197, 386
Hydroptilidae, 178, 197, 386
Hydrozoa, 386
Hylidae, 9
Hyphomicrobium, 50
Hypseleotris, 232, 241, 243, 249, 257, 258
H. compressus, 355
H. galii, 241
H. klunzingeri, 241, 387, 396
Hyridella australis, 201
Hyriidae, 7
ibis, 256; glossy, 388; straw-necked, 389; white, 388
Ilyocryptus sordidus, 384
I. spinifer, 384
Ilyocypris, 296, 297, 298, 300, 384
Ilyodromus, 296, 297, 298, 300
I. ellipticus, 384
I. smaragdinus, 384
indicator species, 120, 192, 194, 196, 197, 198, 208
infective agents, 128
Inland Fisheries Commission, 44
insecticides, 194, 195
intestinal bacteria, 130
introduced: amphibians, 280; fish, 10, 252, 271, 274; pests, 278
iron, 14, 15, 16, 19, 47, 52, 53, 58, 338
iron bacteria, 47, 49, 51, 77, 337
irrigation, 104, 150, 184
Isocypris, 296, 297, 298, 300
Isopoda, 8, 27, 30, 32, 33, 196, 199, 394
Ixobrychus minutus, 388
Janiridae, 8
Jewfish, freshwater, 248
Joe Page Bay, 298
Juddacuttup River, 297
Juncus, 381, 383
J. ingens, 382
J. subsecundus, 382
Kalgan River, 298
kenaru, 248
Keratella, 308, 367, 383, 384
K. australis, 384
K. cochlearis, 336, 340, 384
K. quadrata australis, 336
K. slacki, 384
K. tropica, 336, 384
K. valga, 336
K. valga form tropica, 313
Keratococcus, 74, 329
Kiewa River, 184, 187
King George whiting, 392
King River, 59, 62, 197, 298
King William, Lake, 47
Kings Creek, 297
Kirchneriella, 329
Koo-Wee-Rup aquifer, 152
kookaburra, 284
Kosciusko, 274
Kuhlia rupestris, 231
Kuhliiidae, 231
kunjin virus, 138
Kurandapogon blanchardi, 231
Kurtus gulliveri, 232
Kurzia latissima, 384
Lachlan River, 250, 255, 297, 378
Lagerheimia, 329
Lagoon of Islands, 60, 61, 62, 64, 169
Lakes, see also individual lakes
Lake Ainsworth, 29
Lake Alexandrina, 71
Lake Argyle, 137, 218
Lake Barber, 63
Lake Barracoota, 30, 396
Lake Barrine, 30, 31
Lake Barrington, 52, 53, 55, 56, 57, 64
Lake Bong Bong, 30
Lake Bullenmerri, 27
Lake Burley Griffin, 90-5 passim, 253, 317, 318, 319, 321
Lake Burrumbeet, 26
Lake Canobolas, 253
Lake Charles, 216, 223, 246
Lake Chrisie, 299
Lake Coolongup, 298
Lake Coragulac, 26
Lake Crescent, 26, 28
Lake Dobson, 28
Lake Echo, 45
Lake Edgar, 169
Lake Edward, 25, 101
Lake Eildon, 250, 272, 332, 333
Lake Eliza, 14
Index

Lake Elusive, 29
Lake Eppalock, 332, 335
Lake Eucumbene, 262, 263, 278
Lake Eulamoo, 30
Lake Eyre, 246, 249, 256, 257, 258
Lake Eyre hardyhead, 241, 258
Lake Fenton, 58
Lake Frome, 257, 299
Lake Gidgi, 299
Lake Gordon, 55, 56
Lake Hawthorn, 253
Lake Hiawatha, 30
Lake Huron, 31, 32
Lake Jindabyne, 262
Lake Kariba, 83
Lake King William, 45
Lake Leake, 25, 31, 102
Lake Modewarre, 26
Lake Monger, 30
Lake Mulwala, 16, 184, 187, 189, 197, 210, 211, 314
Lake Pedder, 62, 163, 165, 166, 169, 274
Lake Preston, 298
Lake Purrumbete, 262
Lake Rhona, 28
Lake Rowallan, 52
Lake Sailie, 96
Lake Sambell, 247
Lake Sorrell, 28
Lake St Clair, 28, 32, 45, 47
Lake Tali Karn, 29
Lake Trummen, 96
Lake Washington, 32, 94
Lake Wendouree, 90
Lake Werowrap, 23, 26, 27, 31, 34
lamprey, 231, 393; short-headed, 240, 244
land treatment of sewage, 156
Lakes calcifer, 230, 231, 393
L. niloticus, 220, 230, 275
Laduronopsis australis, 384
lead plumbing, 132
Leake, Lake, 23, 24
Lecane, 336, 384
leeches, 8
Leiopotherapon bidyanus, 241, 247, 258, 259
L. unicolor, 241, 250, 258
Lemna, 379, 381
L. minor, 382
leptocephalus larvae, 234
Leptoceridae, 9, 197
Leptodactylidae, 9
Leptophlebiidae, 8, 178
Leptospermum lanigerum, 60, 61
Leptospiro ma interroga, 128
lethal levels, 120
Leuciscus cephalus, 368
Leydigia australis, 367, 384
L. ciliata, 367
L. leydi, 367, 384
limiting nutrients, 16
Limnanthemum indicum, 352
Limnobdella, 386
Limnocythere, 296, 297
Limnodrilus hoffmeisteri, 23, 28, 33
Limnodynastes dumerilli, 387
L. tasmaniensis, 387
limpets, 196
Lindemanella iota, 232
Lithoglyphopsis aperta, 142
Litoria aurea, 387
L. ewingi, 387
L. verreauxi, 387
littoral invertebrates, 354, 356, 383
liver fluke, 138, 143
liverwort, 382
lizards, 9, 387
Llyn Tegid, 31, 32
lobster, spiny, 256
Lolium rigidium, 382
loons, 9
Lovettia seali, 168, 231, 238
Lower Gordon River, 170
Lower Molonglo Water Quality Centre, 94
Ludwigia, 381
L. pepliodes, 382
Lumbriculus variegatus, 9
lungfish, 9, 265, 266; Queensland, 229, 231
Lymnaea, 143, 386
L. tomentosa, 143, 178
L. truncatula, 142
Lyngbya, 381
maar lakes, 25, 32
maars, 30, 32, 34, 36
Maccullochella macquariensis, 241, 246, 259
M. peeli, 241, 246, 259, 275, 387
Macquaria ambigua, 189
M. australasica, 189, 216, 241, 250, 259
Macquarie River, 247
Macquarie Rivulet, 236
Macrobrachium, 248, 249, 386, 387
Macrocylops, 383
M. albidus, 384
macroinvertebrates, 178, 195, 196, 197, 386, 387
Index

_Macrophthalmus setosus_, 394
macrophyte ponds, 153
macrophytes, 10, 19, 91, 93, 166, 313, 352, 382
_Macrothrix_, 383, 384
  _M. hirsuticornis_, 367
  _M. spinosa_, 384
_Madigania unicolor_, 250
magnification, 192
major ions, 12
_Malacorhynchus membranaceus_, 388
malaria, 137, 140, 141, 144, 145
_Maliomonas_, 329
Malpas Dam, 69
mammals, 9, 320, 376, 387
management of waterfowl, 292
manganese, 14, 16, 19, 44-53 passim, 58, 338
_Mansonia_, 145
  _M. (M.) uniformis_, 139
_Margaritiferidae_, 7
marine hypersaline waters, 14
marron, 216, 218, 222, 239, 348, 353, 354, 356
marsh ragwort, 382
marshbird, little, 388
_Marsilea_, 353
Mary River, 246
maturation ponds, 361
mayflies, 8, 29, 30, 179, 195, 196, 199
McDonnell Ranges rainbow fish, 258
_Mecoptera_, 9, 24, 28, 33
medusa, 308
_Megaleurus gramineus_, 388
_Megalopa_, 24, 32, 196
Mekong schistosome, 142
_Melaleuca_, 289
_Melanotaenia_, 275, 396
_Melanotaeniidae_, 231
_Melita_, 394
melon fish, 249
_Melosira_, 74, 76, 307, 313, 319, 328
  _M. distans_, 329, 339, 340
  _M. granulata_, 328, 329
  _M. granulata var. angustissima_, 334
mengo-encephalitis, 128
mercury, 121
_Merismopedia_, 329
meromictic lakes, 52, 53, 55
Mersey Valley, 52, 54
_meso-eutrophic lakes_, 35
_Mesocyclops_, 308, 336, 382, 384, 392
  _M. decipiens_, 336, 340, 384
  _M. leuckarti_, 313, 325, 336, 352, 353, 367, 384, 396
_Mesocyclops_, 295, 296
mesotrophic, 36, 37
Mesoveliidae, 386
_metacercariae_, 143
_Metacypris_, 296, 298
_Metallogenium symbioticum_, 49
_Metapenaeus bennettii_, 392
  _M. dalli_, 392
  _M. macleayi_, 392
_Metaphreatoicus_, 28, 31, 33
  _M. australis_, 29
methaemoglobinaemia, 130
methane, 52, 153, 154
methanol, 153
methyl mercury, 120
Metropolitan Water Board, 44, 58
_Microsterias_, 381
  _M. hardyi_, 10, 338
_Microcyclops_, 308, 336, 382, 384, 395
  _M. javanus_, 384
  _M. minutus_, 367
  _M. varicans_, 336
_Microcystis_, 68, 74, 313, 329, 333, 339
  _M. aeruginosa_, 340
_Microcytherura_, 296, 298
  _M. difficilis_, 395
midge larvae, 370
milfoil: coarse water, 382; common water, 382
mine effluents, 16
mining, 108, 167
minnow, flat-head, 240
Mitchell River, 237
Mitta Mitta River, 174-9 passim, 183, 185, 187, 189, 305, 309, 335, 341
_Mogurnda mogurnda_, 258
  _M. striata_, 241
_Moina_, 352, 383
  _M. brachiata_, 367
  _M. dubia_, 367
  _M. macropa_, 367
  _M. micrura_, 336, 367, 384
  _M. rectirostris_, 367
  _M. tenuicornis_, 325, 336
_Molonglo River_, 108, 196, 318, 319, 320
Mongarlowe River, 250
monimolimnion, 53, 55, 56
monitoring, 174, 177, 192, 193, 195, 197, 198, 199, 201
_Monogononta_, 367
_Monostyla (?robusta), 392
  _M. stenroosi_, 392
moorhen, dusky, 388
_Mordacia_, 231
  _M. mordax_, 240, 393
_mosquito fish_, 240, 257, 258, 271, 275, 355, 387
_mosquitoes_, 130, 137, 138, 139, 141, 145
Mougeotia, 329
Mount Bold Reservoir, 68, 71
Mountain River, 196
Mozambique mouth-brooder, 276
Mt Kosciusko, 183
*Mugil cephalus*, 241, 243, 368, 391, 392
mullet: bully, 241, 244; sea, 391, 392; yellow-eye, 241, 244, 392
multi-level outlets, 58
Mulwala, Lake, 379
Mumblin, Lake, 23, 25
Muridae, 10
Murray River, 6, 16, 184, 185, 186, 187, 188, 189, 211, 235, 239, 242, 244, 247, 249, 250, 252, 255, 256, 297, 300, 335, 378, 379, 384, 387
Murray-Darling River, 12, 164, 166, 168, 170, 188, 216, 217, 223, 227, 239, 240, 242, 244-9 *passim*, 252, 253, 256, 257, 259, 272, 275, 305, 376, 378, 386
Murray Valley encephalitis, 137, 138, 139
Murrumbidgee River, 239, 243, 244, 247, 248, 249, 250, 255, 256, 297
muskgrass, 382
mussel, 201, 394
Mus*te*lus antarcticus, 392
MVE, 138, 139, 140, 145
Myall Lakes, 29, 167, 392
*Mycticorax caledonicus*, 388
*Myriophyllum*, 381, 383
*M. elatinoides*, 382
*M. propinquum*, 319, 382
*M. verrucosum*, 319
Mysidacea, 33
*Mylitocypris*, 296, 297, 299, 300, 395
*M. splendida*, 26

*Naegleria aerobia* (*N. fowleri*), 128
Naididae, 178
*Naidium*, 386
Nannochorista, 31
Nannochori*stidae*, 9
*Nannoperca australis*, 241, 387
*N. australis australis*, 355
Narrandera Inland Fisheries, Research Station, 216
Narrows Basin, 299
National Museum of Victoria, 174
National Parks, 164
native hen, black-tailed, 389
Naucorididae, 386
Navicula, 329
*Necterosoma pellii*latus, 26, 27

*Nematalosa erebi*, 240, 249, 258
*Nematocentris fluviatilis*, 240
*N. maculata*, 240, 258
*N. tatei*, 258
Nematoda, 7, 23, 24, 30, 32, 33, 129, 141, 178, 369, 386
Nematomorpha, 7, 24, 28, 31
Nemertea, 7, 24, 28, 33
Neoceratodus, 9
*N. forsteri*, 229, 231
Neoniphargus, 28, 33
neosilurid catfishes, 248
Neosilurus, 231, 257, 258
*N. argenteus*, 240, 258
*N. hystrix*, 258
*Neothyridia armata*, 384
Nepean Reservoir, 324, 325, 327
Nepean River, 251, 324
*Nephrocytium*, 329
*Nematoceridae*, 388
Neuroptera, 24, 32, 34
*Newnhia*, 296, 297, 298, 299, 300
*N. fenestrata*, 336, 384
*N. fuscata*, 384
newts, 280
nicky, 251
nightfish, 231
*Nilodorum* biro*, 34
*Nitella*, 382
nitrogen, 12, 14, 16, 19, 90, 314
*Nitzschia*, 329
Noosa River, 235
North Pine Dam, 311, 312, 313, 315
North Pine River, 311
North Stradbroke Island, 396
*Nostoc*, 364
*Nostocinales*, 74
*Notechis scutatus*, 387
*Notochalca*, 384
*Notocyclops*, 336
Notonectidae, 354, 356, 386, 396
Notonemouridae, 8
Notostraca, 8
*Numenius madagascariensis*, 388
nursery fish, 232
nutrients, 12, 16, 17, 18, 20, 157, 208, 209, 321, 351, 365, 370
*Ochromonas*, 74, 329
Odonata, 24, 25, 31, 34, 178, 196, 354, 356, 383
*Oecophylla smaragdina*, 284
Oedogoniales, 74
*Oedogonium*, 74
Index

Oithona nana, 392
Old Cork Beds, 300
Oldfield River, 297
Olga River, 62
Oligochaeta, 7, 23-9 passim, 32, 33, 34, 35, 36, 178, 195, 196, 198, 199, 353, 386
oligotrophic, 34, 35, 36, 37, 43
Oncomelania, 141
O. formosana, 141
O. hupensis, 141, 142
O. lindoensis, 141
O. quadrasii, 141
Oncorhynchus tschawytscha, 240, 260, 271, 272
Onkaparinga River, 14, 16, 17
Onondaga Lake, 94
Onychocamptus bengalensis, 395
Oocystis, 74, 329
Ophiocara aporos, 232
Ord irrigation area, 219, 220
Ord reservoir, 137; see also Lake Argyle
Ord River, 140, 142, 143, 144, 145
ornamental fish, 277
Ornithorhynchus anatinus, 9
Orthocladius, 37
Oscillatoria, 329, 364, 381
osmoregulation, 19
ostariophysian carps, 229
Osteoglossidae, 229
Ostracoda, 23, 26, 27, 32, 295-300 passim, 336, 340, 354, 383, 384, 392, 395
Ottelia ovalifolia, 353
Ovens River, 184, 187, 211, 274
overfishing, 168
ox-bow lakes, 378
Oxyeleotris lineolatus, 232
oxygen, 52, 53, 55, 56, 187, 188, 350
Oxyura australis, 388
oysters, 121

Palaemonetes australis, 394
Palaemonidae, 386
paperbark, coastal, 289
Paracyclops chiltoni, 384
Paracypria, 297
P. minutus, 384
P. tenuis, 392
Paragonimus, 143
P. westermani, 143
Parartemia, 8
Parastacidae, 8, 164, 238, 256, 386
Paratya, 248, 386, 387
P. australiensis, 200, 201
P. tasmaniensis, 201
Paroo, River, 378
pathogenic: amoebae, 133; bacteria, 153
pathogenicity of wastewater, 150
Pediastrum, 329
Pedocellulosa, 50, 51
Pelecanus conspicillatus, 388
pelican, 388
Penaeidae, 391, 392
P. latisulcatus, 392
P. merguensis, 392
penny fish, 258
Perca flavescens, 252
P. fluviatilis, 189, 241, 252, 271, 272, 308, 368, 387
Percalates colonorum, 216, 231, 235, 241, 243
P. novaemaculatus, 231, 235
perch, 244, 252; English, 228, 241, 246, 252, 253, 308, 387; estuary, 231, 233, 235, 241, 244, 265; European, 252, 271; giant, 216, 219, 220, 227, 230; golden, 166, 189, 216, 223, 228, 241-50 passim, 253, 255, 257, 258, 265, 275, 319; jewel, 298; jungle, 231, 265; Macquarie, 168, 189, 216, 228, 241, 244, 250, 259, 265; mountain, 250; Murray, 244, 250; Nile, 220, 230, 233, 275, 276; pigmy, 231, 355; silver, 166, 220, 223, 228, 241, 243, 244, 247, 248, 250, 257, 258, 259; southern pigmy, 241, 387; spangled, 241, 247, 250, 251, 257, 258; western chanda, 241, 258
perched lakes, 14
Percichthyidae, 227, 235, 244, 246, 250
Percidae, 9, 252
Peridiniales, 74
Peridinium, 74, 340
P. volzii, 334
Periinium, 329
pesticides, 130, 192, 193, 196-201 passim
Pettancylus, 386
pH, 15, 18, 75
Phaenocora, 23, 34
Phaeophyta, 393
Phalacrocorax carbo, 388
P. melanocephalus, 388
P. sulphurostris, 388
P. varius, 388
Phillip River, 297
Philodina, 336
Philopod, 241, 243
P. grandiceps, 241
Phoenicopteridae, 9
Phormidium, 329
phosphorus, 6, 12, 14, 16, 17, 19, 90, 92, 211, 314
Phreatoicidae, 8, 28, 29, 33, 35
<table>
<thead>
<tr>
<th>Index</th>
</tr>
</thead>
<tbody>
<tr>
<td>Phreodrilus, 28</td>
</tr>
<tr>
<td>Phryganeidae, 9</td>
</tr>
<tr>
<td>Physastra, 386</td>
</tr>
<tr>
<td>Physignathus lesueurii, 320</td>
</tr>
<tr>
<td>Phytomastigophora, 366</td>
</tr>
<tr>
<td>P. regia, 388</td>
</tr>
<tr>
<td>Phylum, 273</td>
</tr>
<tr>
<td>Platycypris, 26, 296, 297, 299, 300, 384</td>
</tr>
<tr>
<td>P. baueri, 395</td>
</tr>
<tr>
<td>platypus, 389</td>
</tr>
<tr>
<td>Plecoptera, 8, 24, 28, 31, 32, 178, 196, 198</td>
</tr>
<tr>
<td>Plectroplites ambiguus, 241, 244, 258, 275, 319</td>
</tr>
<tr>
<td>Plegadis falcinellus, 388</td>
</tr>
<tr>
<td>Pleistocene glaciation, 32, 43</td>
</tr>
<tr>
<td>P. intermedia, 392</td>
</tr>
<tr>
<td>Poecilia latipinna, 273</td>
</tr>
<tr>
<td>P. reticulata, 121, 273</td>
</tr>
<tr>
<td>P. nigra, 23</td>
</tr>
<tr>
<td>Psephenid larvae, 196</td>
</tr>
<tr>
<td>Pseudechis porphyriacus, 387</td>
</tr>
<tr>
<td>Pseudemysdura, 9</td>
</tr>
<tr>
<td>Pseudochydorus globosus, 384</td>
</tr>
<tr>
<td>Pseudokephyrion, 329</td>
</tr>
<tr>
<td>Pseudomonas aeruginosa, 343</td>
</tr>
<tr>
<td>Pseudomugil, 231, 275</td>
</tr>
<tr>
<td>Psychodidae, 197</td>
</tr>
<tr>
<td>Puntius conchonius, 273</td>
</tr>
<tr>
<td>Pybbery, 249</td>
</tr>
<tr>
<td>Pyrazus ebeninus, 394</td>
</tr>
<tr>
<td>Pyrophyta, 74</td>
</tr>
<tr>
<td>Quadrigula, 329</td>
</tr>
<tr>
<td>quality criteria, 117</td>
</tr>
<tr>
<td>rainbow fish, 231, 240, 257; checkered, 240, 258</td>
</tr>
<tr>
<td>Ranidae, 9</td>
</tr>
<tr>
<td>Ranunculus rivularis, 382</td>
</tr>
<tr>
<td>receiving water quality standards, 122</td>
</tr>
<tr>
<td>reclaimed water, 150, 152</td>
</tr>
</tbody>
</table>
Reclaimed Water Committee, 150
recreational use of water, 133
red gums, 164, 170, 305, 378
redfin, 189, 252, 271, 355, 387
redox potential, 52, 57
relict sea salt, 13
reptiles, 9, 376, 387
resource husbandry, 163
resources from wastewater, 149
Restio tetraphyllus, 61
Retropinna, 231, 275
R. retropinna, 369, 393
R. semoni, 240, 258, 387
Retropinnidae, 236, 238
rhabdocoel, 34
Rhadinocentrus ornatus, 396
Rhizopoda, 340, 366
Rhizosolenia, 328, 329
R. eriensis, 334
Rhiophyta, 393
ribbon weed, 382
Riccia, 379, 382
R. fluitans, 382
Riccio carpus, 382
R. natans, 382
Richardson's Number, 69
Richmond River, 235, 244, 246
Rietha, 28, 31, 37
Risdon Brook Reservoir, 58
River Jumna, 131
river: improvement, 167, 170, 265; management, 170
River Murray, 71, 105, 110, 123, 124, 139, 182, 188, 189, 197, 198, 201, 242, 249, 255, 305, 306, 309, 341, 376, 379
River Murray Commission, 105, 184, 189, 305
river red-gum, see red gums
Rivers and Water Supply Commission, 44
roach, 271
Robertsonia propinqua, 395
Ross River, 84, 139
Ross River virus, 137, 138, 145
rosy barb, 273
Rotaria cf. citrinus, 336
rotaviruses, 129, 131, 134
Rotifer, 7, 308, 313, 336, 340, 342, 352, 366, 367, 369, 383, 384, 392, 396
Rottnest Island, 299
RRV, 138, 139, 145
Russell River, 196
Russelletia (?parrotti), 392
Rutilis rutilis, 271, 272, 368
rye grass, 382
Saginaw Bay, 31, 32
sailfin molly, 273
salamanders, 9, 280
saline, 16
saline: drainage water, 105; environments, 298; lakes, 14, 15, 23, 26, 27, 33, 37, 395
salinity, 6, 12, 15, 18, 19, 27, 29, 31, 101, 104; tolerance, 15, 19
Salmo, 260
S. gairdneri, 240, 260, 261, 271, 272, 319, 369
S. salar, 260, 272
S. trutta, 189, 240, 260, 271, 272, 308, 319, 369, 387, 393
salmon, 260; Atlantic, 260, 262, 272; Australian, 392; quinnat, 240, 244, 260, 262, 271
Salmonella, 129
S. bredeney, 133
S. typhi, 129, 130, 131
salmonellosis, 131, 133
Salmonidae, 9, 59, 227, 260, 262
salt cedar, 101
salt lagoons, 43
salt lakes, 8, 12, 14, 18, 20, 23, 26, 28, 242, 289, 395
Salvelinus fontinalis, 240, 260, 273
Salvinia, 3, 83, 87
S. auriculata, 168
S. molesta, 83, 84, 145
Samuel, Lake, 46
sand mining, 167, 170
sandfish, 260
sandpiper, terek, 389
Saprobiensystem, 194
Sarotheradon mossambica, 276
Savage, Lake, 59
saw-sedge, 382
Scapholeberis, 384
S. kingi, 384
Scenedesmus, 329, 362, 364
scenic rivers, 170
Schistosoma, 128, 137, 144
S. haematobium, 137, 141, 142
S. japonicum, 137, 141, 142, 143
S. mansoni, 137, 141, 142
schistosome dermatitis, 133, 141
schistosomes, 128, 133, 143
schistosomiasis, 133, 141, 142, 143, 144
schistosomiasis japonica, 142
Schizogonium, 381
scientific value of waters, 164
Sclerocypri s tuberculata, 299
Scleropages, 216, 229
S. jardini, 9, 231
S. leichhardtii, 9, 231
Index

Scortum barcoo, 258
Scrivener Dam, 318
sedge, 10, 60
Segmentina hemisphaerula, 143
S. trochoidea, 143
Selenastrum, 329
Semisulcospira, 143
Senecio aquaticus, 382
sequential comparison index, 194
Sergentia, 37
Serpentine dam, 166, 169
Sesarma erythrodactyla, 394
Seven Creeks, 247
sewage, 59, 155, 156, 159; discharge to lakes, 158; discharge to rivers, 158; disposal, 155; fungus, 196; poisoning, 134; treatment, 157
sewerage, 110
shark, gummy, 392
sheathed bacteria, 49
shelduck, chestnut-breasted, 388
Shigella, 129
shigellosis, 131, 133
Shoalhaven River, 237, 324
Shoalhaven scheme, 324
shoveller, blue-winged, 388
shrimp, brine, 8; Tasmanian mountain, 274
silicon, 14, 15, 19
Sillaginodes punctatus, 392
Siluriformes, 229
silveryeye, 250
silversides, 231
Simoccephalus, 383
S. acutirostratus, 353, 384
S. exspinosus, 367
S. exspinosus australiensis, 384
S. vetulus elisabethae, 384
S. vetulus gibbosus, 384
sindbis virus, 138, 139, 145
Sisyr, 34
Skeletaloma costatum, 68
sleeper, 232
smelt, 236; Australian, 231, 240, 257, 258, 387; common, 393; southern, 238
snails, 29, 138, 141, 142, 143, 196, 354, 356; planorbid, 143
snakes, 9, 387; black, 387; tiger, 387
snapper, 392
snipe, Australian, 388
Snobs Creek Freshwater Fisheries Research Station and Hatchery, 216
snow, 230
Snowy Creek, 178
Snowy Mountains Hydroelectric Scheme, 44, 183
snowy River, 396
soakage dams, 346
sole: freshwater, 232; salt-pan, 232; tailed, 232
South Esk, 45, 59, 197
South West Tasmanian Resources Survey, 63
Spathula triculenta, 29
Sphaeriidae, 7, 28, 33
Sphaerium, 28, 29, 33, 386
Sphaeroctys, 307, 328, 329, 331, 334, 340
sphaeromats (= sphaemoids), 30, 394
Sphaerotilus, 337
spike-rush, tall, 382
spinebill, eastern, 388
Spirodeloida oligorrhiza, 382
Spirogyra, 74, 329, 381
spoonbill: royal, 388; yellow-billed, 388
St Clair, Lake, 24, 31
St Lucia Pond, 352
standard composition (of fresh water), 6
standards (for water quality), 117
starwort, 382
State Rivers and Water Supply Commission of Victoria, 332
Staurastrum, 74, 328, 329, 334, 340
Staurodesmus, 329, 334, 340
Stenocypris, 296, 297, 298, 384
Stichococcus, 364
Stictochironomus, 37
Stigeoclonium, 74, 196, 329
stilt, white-headed, 388
stocking of fish, 261, 262, 266
stoichiometry, 12, 13, 17, 18
Stokes Law, 68
stoneflies, 8, 33, 177, 179, 195, 196, 199
stoneworts, 74, 382
Strundesia, 384
Stratiolirilus, 7
Strictonetta naevosa, 388
Strongylura kreffti, 231
Sulcanus conflictus, 391, 392, 394, 396
sullage, 197
sulphur bacteria, 77
Surirella, 329
Surprise, Lake, 23, 25
suspension (of algae), 67
swamp Crassula, 382
swamp harrier, 388
swamp hen, eastern, 388
swamps, 43, 376, 379, 381
Swampy Plain River, 183
swan, black, 388
Swan River, 393, 394
Swan coastal plain, 170
swordtail, 273, 275
Index

Sydney water supply system, 324
Syncarida, 8, 33, 43
Synchaeta, 336
Synedra, 76, 328, 329
S. nana, 334
synergism, 121
Synura, 74, 340, 342

Tabbita Hatchery, 223
Tabellaria, 329
tadpoles, 235, 282, 387
Tali Karn, Lake, 36
Tallowa Dam or Reservoir, 237, 325
Tamar estuary, 58
tamarisk shrub, 101
Tamarix pentandra, 101
tandan, 231, 248; Hyrtls, 258; silver, 240, 244, 258
Tandanus bostocki, 231, 239
T. tandanus, 189, 240, 248
Tanytarsus, 37
T. barbitarsis, 26, 27, 28, 31, 33, 37
Tarago Reservoir, 71, 332, 334, 335, 336, 339
Tarago River, 339
Tardigrada, 386
Tarebia, 143
tcheri, 247
teal, chestnut, 388; grey, 290, 292, 388
trematodes, 138, 143, 144
Tetramotrephes, 7
tenches, 189, 216, 228, 240, 244, 249, 254, 255, 256, 271, 355, 387
tern, marsh, 388
Terrors Creek, 312, 315
Thalassiosira, 329
Thames Water Authority, 78
Theraponidae, 231, 244, 247, 250, 251
thereimictic, 7
Thredbo River, 272
threshold concentrations, 120
Threskiornis molucca, 388
T. spinicollis, 389
Tilapia, 276, 369
Tin Can Bay, 235
Tinbeerwhah Hatchery, 223
Tinca tinca, 189, 240, 255, 271, 272, 387
toads, 281, 283, 284
Tooma River, 183
Tooms Lake, 31
tortoise, 387; long-necked, 387; Murray, 387
toxic chemicals, 134
toxicity, 194, 199, 200, 201
Toxotes chatareus, 232
trace metals, 16
Trachelomonas, 307, 329, 381
'tragedy of the commons', 158
Trapa natans, 143
trematodes, 138, 143, 144
Trichonema, 71, 381
Trichonyx ventralis, 389
Trichoptera, 9, 24, 27, 30, 32, 178, 196, 197, 198, 201, 354, 356, 386
Triglochin procera, 319, 353, 382
Trigonocypria, 296, 298, 299, 300
Trinitotylus nivalis, 178, 179
Triops, 8
Triploceras, 329
Tristania conferta, 311
Triturus, 280
Tropocyclops, 336, 384
T. confinis, 384
T. prasinus, 325
trotta cod, 241, 244, 246, 247, 259, 265
Tubifex, 386
T. tubifex, 23, 29
Tubificidae, 197, 354, 355
Tuggeranong Reservoir, 90
Tukari, 249
Turbellaria, 7, 32, 35, 195, 196, 386
turbidity, 7, 17, 58, 59, 75, 321
Tweed River, 297
Tynpanotomus micropterus, 143
Typha, 82, 381, 383
typhoid, 129, 130, 131, 133
Uca longidigitum, 394
Ulothrix, 381
Unionidae, 7
Upper Nepean River, 324
uranium mining, 233
urban: catchments, 105; hydrology, 105, 110; lakes, 90; runoff, 90, 91, 95
Utricularia, 381

Valley, Lake, 23, 24, 25, 31, 35
Vallisneria, 381
V. gigantea, 382
V. spiralis, 90, 319
Velacumantus australis, 141
Velesunio ambiguus, 200, 201
Index

Veliidae, 386
vertical migrations, 68
Vibrio cholerae, 129
vine scrub, 311
viral diseases, 145
viruses, 128, 129, 134, 137, 138, 153, 360, 366
Viviparus javanicus, 143
volcanic lakes, 34, 35
Volvocales, 74
Volvox, 74, 319, 329, 342, 380, 381
Vorticella, 336

Wagin Lake, 299
Waranga Channel, 335
Warm-Water Fisheries Research Station, 223
Warragamba Dam, 319, 324, 325
Warragamba River, 324
waste: disposal, 158; stabilisation ponds, 360, 361, 363, 364, 367, 370; transport, 155
wastewater, 149, 360
water as a resource from wastewater, 149
water caltrop, 143
water chestnut, 143
water dragon, 320
water fern, 3, 168
water hyacinth, 83, 84, 85, 87, 145, 168
water mites, 23, 29, 30
water primrose, 382
water quality, 3, 51, 57, 59, 177, 192; criteria, 117; management strategies, 123; uses of, 122
water renewal ratio, 18
water ribbons, 382
water-borne disease, 128
water-buffalo, 289
water-cress, 144
water-rats, 9, 387
water-shield, 382
waterbirds, 249, 370, 388, 387
waterfowl, 163, 165, 166, 170, 255, 287, 288, 289, 290, 292, 293, 356, 376; distribution, 288; habitats, 287; management, 287; wetland habitats of, 289
weed management, 82, 85
weeds, 81
Werowrap, Lake, 28
Werribee Sewage Farm, 151
West Basin Lake, 27
West Wallabi Island, 299
western native cat, 284
wetland drainage, 108
wetlands, 166, 170, 289, 290, 298
White Nile, 145
white eye, 250
whitewater, 236, 238; Derwent, 238; New Zealand, 238; Tasmanian, 168, 228, 231, 238, 239, 265
Whyte, 59
wildlife: conservation, 376; refuges, 356
Willaer River, 208
willow smartweed, 382
willows, 168
Wingecarribee Reservoir, 325, 330, 331
Wingecarribee River, 324
Wodonga Creek, 184, 187
Wolfram, 59
Wolgan River, 170
Wollondilly River, 324
world average freshwater (composition of), 14
Woronora Dam, 324, 325, 327
Wuchereria, 137
W. bancrofti, 141, 145
Wyangala Reservoir, 253

Xenopus laevis, 280
Xenostrobus securis, 394
Xenus cinereus, 389
Xeromyx, 9
Xiphophorus helleri, 273
X. maculatus, 273
yabbies, 228, 252, 255, 256, 386
Yan Yean Reservoir, 74
Yarrawonga Weir, 184
yellowbelly, 244
zinc, 59, 196, 197, 198, 201, 319
zinc pollution, 318, 319, 320, 322
Zoomastigophora, 366
zooneses, 131
Zygnema, 74, 329, 381
Zygnumales, 74
Zygnemopsis, 381
Zygoptera, 201
This book, about biological matters determining the quality and usefulness of Australian fresh waters, was written for several reasons. It was written because Australian fresh waters have many distinctive biological and other features, and therefore overseas work and management of water resources is often not relevant and must be used with caution in Australia. A main aim of the book, therefore, has been to aid Australian water management authorities by presenting useful ecological and other biological knowledge on Australian fresh waters. Another reason was to bring together in one volume such knowledge since hitherto this has been scattered, often somewhat inaccessible and sometimes out of date. Since freshwater resources are not abundant in Australia, it is important that they should be well managed. This book aims to help all those concerned with water management to achieve the best management possible. It will be of interest also to other authorities and all concerned with preserving Australia's water resources.

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