Gippsland Lakes

Environmental Audit

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Review of Water Quality and Status of the Aquatic Ecosystems of the Gippsland Lakes

prepared for

Gippsland Coastal Board

prepared by

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Executive Summary

The water quality, hydrology and catchment land use of the Gippsland Lakes region of eastern Victoria has been the subject of a number of previous studies and reviews. In this preliminary assessment CSIRO have reviewed much of that information and have attempted to bring it together into a single authoritative document.

Over the years catchment land use has changed with the result that nutrient exports and loads to the Lakes have increased, water quality has deteriorated and other impacts on the aquatic ecosystems of the Lakes has occurred.

Much of the scientific work on the water quality of Gippsland Lakes seems to have been done as a result of major algal bloom outbreaks, which have occurred about every ten years in the lakes. Knowledge about the lakes has therefore advanced in a sporadic manner. In between times Victorian State agencies have carried out regular water quality monitoring.

Other work in the Lakes has been the result of monitoring of the fisheries and because of other aquatic ecosystem impacts. There is also much anecdotal information about changes in aquatic macrophyte populations, losses of fringing reed beds and *Melaleuca* swamps, increasing salinity and other effects. All this needs to be placed in a general framework so that the total impact of land use change, agricultural and urban development on the Lakes may be assessed.

About a third of the river flows and over half the nutrient load to the Lakes comes from the Latrobe River system, which enters into Lake Wellington, with the result being that the impact of these loads is strongly felt in the Lake. The other major impact on Lake Wellington is saltwater intrusion through McLennans Strait.

Lake Victoria and Lake King, the lower two lakes of the system, are much more marine environments than Lake Wellington. Water quality surveys have shown that salinity increases steadily from around a salinity of 5 in Lake Wellington to 25-30 near the Entrance, and a salinity of 35 in Bass Strait. With saltwater incursion greater in the lower lakes, the physical structure is quite complex. Often stratification (layering effect) results in depletion of oxygen from the bottom (saltier) waters, which has a significant impact on the ecosystem and overall water quality.

The picture that emerges is of a series of tidally flushed, coastal embayments, which show marked interannual variability and habitat change due to nutrient enrichment. The overall health of the Lakes ecosystem is heavily influenced by freshwater and nutrient inflows from the catchment during wet periods and there is some recovery during dry periods when the water clears and a more marine environment is restored.

The fisheries data seem to indicate that at present the influence of habitat change and environmental factors (water temperature and salinity) is more severe than the influence of over-fishing.

Algal bloom dynamics in the Lakes are complex. Not only are the nutrient loads to the system from the catchment high enough to stimulate the growth of blooms but also there is a significant internal source of nutrients – phosphorus and ammonia from decomposition in the sediments. It appears that major blooms can be associated with the major climatic patterns such as the southern oscillation index (ENSO) events. Further blooms are linked to minor climatic events (calm periods following storm events).

The usual situation appears to be that nutrients in Lakes Victoria and King are depleted in surface waters as a result of algal growth and that nutrients build up in bottom waters during stratified periods. Vertical mixing will make these nutrients available for subsequent algal growth and a bloom may develop.

Reductions in the rates of nutrient release by the sediments are an important management tool and should be factored in to any nutrient management plan. Marked reduction in catchment loads should control production, decomposition and sediment nutrient fluxes.

The only long-term solution is to markedly reduce the nutrient loads from the Latrobe system by both restoring the catchment to a more sustainable land use and by replacing riparian vegetation and reducing erosion.

The importance of wetlands, in reducing nutrient loads to the lakes should not be underestimated. The capacity of existing wetlands and means for increasing that capacity to reduce the loads should be investigated and acted upon. Simultaneously, it may be worthwhile to consider engineering large-scale artificial or enhanced natural wetlands at critical points within the catchment and at the river mouths to trap loads of suspended solids and nutrients before they enter the Lakes. Wetlands in addition to reducing nutrient loads provide conservation areas, wildlife refuges and areas of increased biodiversity.

Recent work on sampling for toxicants indicates that, with the exception of mercury (and perhaps selenium), toxicants are not a problem in the Gippsland Lakes. Levels of pesticides and PAHs (polyaromatic hydrocarbons) are, in fact, much lower than in Port Phillip Bay.

The mercury problem does require further investigation, as the evidence indicates rising mercury levels in fish and sediment concentrations are approaching alarmingly high levels. Such exceedences imply a likelihood of ecosystem impacts, although it should be pointed out that the presence of high sulfide concentrations would ensure that mercury was present as insoluble mercury sulfide, which is not bioavailable. The chemical form of mercury needs verification.

Future management of the Lakes should be based on the results of ecosystem modelling. A three-dimensional hydrodynamic model of the Lakes needs to be constructed to examine in more detail the precise interactions of freshwater and marine inflows, tidal mixing and Entrance dynamics, and saltwater/freshwater balances. The hydrodynamic model could be integrated with an ecological model to provide a synthesis of scientific understanding of the ecosystem.

The integrated model would be used to predict the impacts of management decisions regarding the Lakes, through modelling various nutrient load scenarios. The scenarios may be related to flow control, catchment practices, farm management, urban development, sewerage systems, and the control of boating and fishing activities.

The model would also provide the basis for measuring improvements in the system. The actual results from future water quality monitoring programs and other monitoring programs could be compared with results predicted by the model.

Overall there is need for the establishment of management plans accompanied by effective performance monitoring. There is a need to be able to determine what the trends are, how well catchment actions are performing and how much progress has been achieved. In a system which shows strong interannual variation this will not be easy, but it can, and must, be done.

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1 Terms of Reference for the Review

This review did not instigate further data collection or commission specific research investigations. The objective was to gain an understanding of the issues involved and then to review the scientific research and data collection that has already been done. From the development of an integrated understanding of the system, the issues and the science, the review team has produced a "state of the environment" report which identifies gaps and data needs as well as potential management actions.

Under the terms of the contract the review team was asked to consider, in particular, the following points:

- Sources of nutrient and toxicant inputs to the system
- Nutrient pools, transformations and fluxes within the system
- Spatial and temporal trends in water and sediment chemistry
- The role of sediments in nutrient and toxicant dynamics in the system
- Potential role of physical and biological processes in sediment resuspension
- Nuisance algal blooms
- Impacts on significant flora and fauna
- Distribution of dominant benthic habitats
- Impact of water extraction
- Impacts of land use changes within the catchments
- Freshwater/salinity balances within sections of the system
- Entrance dynamics
- Environmental impacts on fish stocks

These topics are dealt with in the following sections.

2 Introduction

The water quality, hydrology and catchment land use of the Gippsland Lakes region (Fig. 1) of eastern Victoria has been the subject of a number of previous studies and reviews but the results have not, until now, been brought together into a single authoritative document.

Since the 1860s catchment land use in the major Gippsland catchments has changed with the result that nutrient exports and loads to the Lakes have increased, water quality has deteriorated and there have apparently been other impacts on the aquatic ecosystems of the Lakes.

Major engineering activities that have impacted on the Lakes include the opening of the Entrance in 1889, and control of the rivers through construction of dams and other water harvesting activities. The extraction, processing and use of mineral resources such as gold, coal and oil have and continue to have an impact on the Lakes. Other primary industries and their associated secondary industries such as forestry, farming and fishing also add to the impact. Include tourism, recreational activities, and the pressures from urbanisation into the equation and it becomes quite a complex system to understand and manage appropriately.

There has been an attempt to both understand and manage the Lakes. Resulting in a number of scattered pieces of often, high quality research. However the topics are many and various and the results have generally appeared in the “grey literature”. Numerous management action plans have been produced, but overall no great improvement in water quality and ecosystem health seems to have been achieved (see the chronology by Smith in Young and Sjerp, 1995). However, as we shall show, trend detection in this system is not easy. A slow deterioration in water quality does however seem to be occurring – particularly in Lake Wellington. (Of course it is possible to argue that without all the management activities the water quality of the Lakes would be even worse than it is now!)

The time has come to pull a lot of this scattered information together into a form that might become the basis for more concerted management planning and action in future. In particular the time has come to set some clear goals for improvement and to put in place a sound performance monitoring system to measure performance against clear objectives.

2.1 Past studies

Much of the scientific work on the water quality of the Gippsland Lakes seems to have been done as a result of major algal bloom outbreaks. Large blooms of *Nodularia spumigenia* (a toxic, nitrogen fixing cyanobacterium) have occurred about every ten years in the lakes and major water quality investigations have resulted.

Large amounts of fieldwork were carried out in the 1970s as a result of the 1974 blooms and again in the late 1980s as a result of the large *Nodularia* blooms of 1987-88 (Gippsland Lakes Algal Bloom Seminar, 1988). Knowledge about the lakes has therefore advanced in a sporadic manner.

In between times, Victorian State agencies have carried out regular water quality monitoring (eg Brown, 1993; Chessman and Marwood, 1988; EPA 1974, 1983, 1986, 1990, and 1993; Longmore, 1988 and 1989a, 1989b; 1994a, 1994b; Longmore et al., 1988; LVW&SB, 1978; Robinson et al., 1994; and Robinson, 1995). The data are scattered, sampling frequencies have changed over the years and the data have been summarised in various ways; sometimes as means, sometimes as medians, sometimes in terms of compliance with State Environmental Protection Policies. At present, it is difficult to present good data on long-term trends in water quality and ecosystem impacts.

Other work in the Lakes has been the result of monitoring of the fisheries and because of other aquatic ecosystem impacts (crab plagues, loss of seagrass). There is also much anecdotal information about changes in aquatic macrophyte populations, losses of fringing reed beds and *Melaleuca* swamps, increasing salinity and other effects. All this needs to be placed in a general framework so that the total impact of land use change, agricultural and urban development on the Lakes may be assessed.

From a review of the results of the investigations of the 1970s (Bek and Bruton, 1979), it is clear that the major features of the impact on the Lakes from land use change, agriculture and other developments were, quite clear even then. Bek and Bruton did an excellent job of bringing the "state of the art" of catchment and estuarine ecology (as it was then) to bear on the problem.

Since the 1970s we have learnt much about the ecology of Australian estuaries. With the completion of the Port Phillip Bay Environmental Study in particular (PPBES, Harris et al, 1996), we have a much clearer idea of what the potential impacts and their mechanisms might be and what to look for. In addition to the detailed process studies of the PPBES, we can also compare data for the Lakes to data from a recent set of empirical studies of a number of coastal lakes and lagoons around the world (see Hinga et al. 1995, Monbet 1992) as well as for lakes along the east coast of Australia (Scanes et al., 1998).

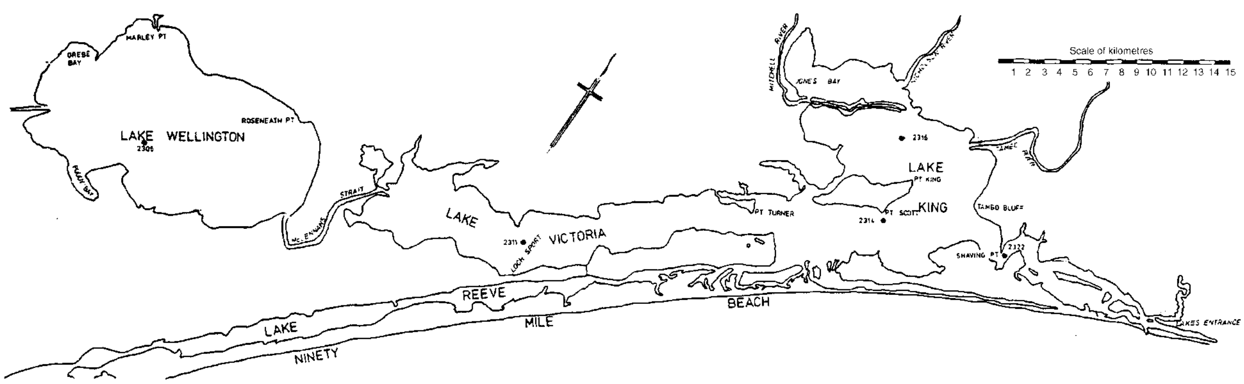

Figure 1: Map of Gippland Lakes, with EPA water quality sampling sites (2306, 2311, 2314, 2316 and 2322) shown.

3 The Historical Setting

It is necessary to point out at the beginning that the Lakes, as we know them today, are a recent phenomenon. Bird (1965, 1978) has thoroughly documented the recent geological history of the Lakes. Before the opening of the Entrance in 1889, the Lakes were a series of coastal lagoons that only opened to the sea after heavy rainfall and run-off from the major catchments. Even then, the marine influences would have been small and the system of lakes and marshes were almost entirely populated by freshwater species.

The level of the lakes would have fluctuated markedly after rainfall, with the level rising and inundating the freshwater marshes around the open water areas before the flow through the opening scoured out enough sand to lower the level again. Periodic inundation of freshwater marshes is important for the survival and health of these systems (see GHD, 1991).

When the Entrance was opened permanently, the level of the Lakes dropped and became more constant, marine species began to invade and the present ecosystems were established. It should be remembered however that a period of just over one hundred years is small in relation to the time scales of many ecosystem processes, so that the Lakes are still changing and responding to the engineering work at the Entrance. Many of the changes that we are presently seeing are part of that ecosystem response.

Further changes in the Lakes system are still being wrought by the invasion of species such as carp. These fish, which stir up the water, increase turbidity and reduce aquatic plant populations (Roberts et al., 1997; Robinson, 1995) have been introduced to the Lakes areas since the 1960s (SOE, 1996) and are now having significant environmental effects in the Lakes (Gunthorpe, 1997).

Undoubtedly, there are other introduced marine species in the Lakes as there are in Port Phillip Bay (Harris et al. 1996). Whether these are having similar impacts on ecosystem function in the more marine parts of the Lakes is not known.

Against the background of changes in the hydrology of the Lakes caused by the opening of the Entrance, other changes have taken place in the catchments feeding the system. Since the advent of human settlement, forests have been cleared, and towns and sewer systems have been built. A lot is known about the land classifications, soil types and sediment delivery potential of the Gippsland catchments (Aldrick et al. 1992; Erskine et al., 1990; Grayson et al., 1994).

The early development of gold mining in the catchments is a particularly important historical event because of the association of these activities with various forms of pollution, particularly mercury pollution. Similarly, the large-scale development of power generation plants adjacent to the extensive coal deposits in the Latrobe Valley has implications for the ecology of the Lakes.

Throughout the southeastern corner of Australia, in areas around the Southern Highlands, clearing of land for agriculture leads to gullying, erosion, and increased exports of nutrients and suspended solids (SOE, 1996). High stock densities and grazing of stock adjacent to unfenced waterways increases this affect. Consequently, in most cases, waters of rivers and marshes that were clear become more turbid. Increasing salinity and sodicity of soils resulting from agriculture leads to dispersive clays that remain in suspension, reduce the underwater light intensity and lead to the loss of aquatic plants.

In many of these deforested systems in NSW and Victoria, there is a close correlation between export of suspended solids and nutrients (Grayson et al., 1994). This is because most of the nutrient exports are in the form of particulate material sourced from stream banks and gullys in the catchments (Erskine et al., 1990).

As we shall see, further impacts on the Lakes arise from the building of dams, extraction of water for irrigation and the regulation of the rivers flowing into the Lakes. Floods are now less frequent, inundation of the marshes is less common (GHD, 1991), saltwater intrusions are more frequent and the vegetation is changing accordingly (Ducker et al., 1977; Clucas and Ladiges, 1980; Birkin, 1991; GHD, 1991).

What were flourishing freshwater marshes are becoming more saline and depauperate as the balance between freshwater and marine influences continues to shift towards the marine. This situation is further exacerbated by the development of saline areas in the lower Latrobe catchment resulting from deforestation, irrigation and consequent ground water rise. This too is a frequent result of agricultural development in southeastern Australia and it contributes to the inexorable drift towards saline conditions in rivers and low land marshes.

The Lakes ecosystems are changing in response to a whole series of human activities over the last hundred years so continued change is inevitable. The Lakes system is not and probably never will be, at steady state. Really, the question is which changes are desirable and which deleterious and what might be done to push the balance one way or the other.

4 Climate Variability

Another factor that leads to long-term change in the Lakes and contributes to the lack of stable conditions is climate variability. The southeastern corner of Australia, from Brisbane to Tasmania, is subject to strong climate variability resulting from El Nino Southern Oscillation (ENSO) events (Harris et al. 1988, Harris and Baxter 1996).

During strong ENSO events when the Peruvian oceanic upwelling is shut off (the so-called "warm" events in the eastern Pacific Ocean), southeast Australia experiences severe droughts. We have just come through a particularly strong event of this kind.

Following ENSO droughts, we frequently see so-called La Nina events ("cold" events in the eastern Pacific) which are characterised by warmer than average ocean temperatures around eastern Australia and above average rainfall across southeastern Australia. During the writing of this report, flooding and heavy rains occurred across central NSW, and parts of Victoria.

4.1 Rainfall

The long-term rainfall record for the Gippsland region (Fig. 2) shows strong evidence of such climate variability.

Indeed, there is good circumstantial evidence that the major *Nodularia* blooms have been associated with strong La Nina events and above average rainfall (Robinson 1995). Such a sequence of events is plausible because *Nodularia* grows in response to lowered salinity and increased nutrient inputs (see papers in Gippsland Lakes Algal Bloom Seminar, 1988). This is precisely what happens in marine coastal waters after heavy rains, catchment erosion and nutrient inputs.

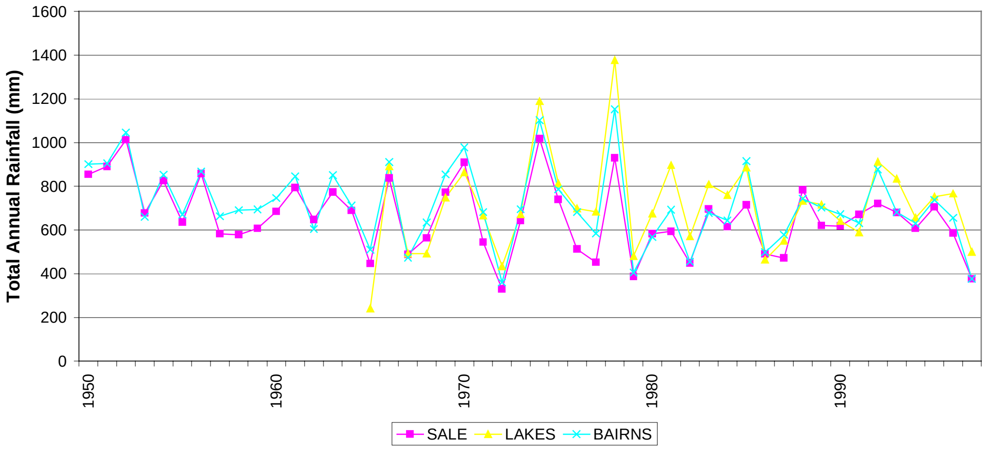

Figure 2: Total annual rainfall from 1950 to 1997, measured at Sale, Lakes Entrance and Bairnsdale. (Supplied by Bureau of Meteorology).

Statistical analysis of the long-term rainfall record shows two very important features:

(a) long-term fluctuation in rainfall over periods of decades, with a particularly noticeable decline in rainfall from over 55 mm per month to less than 45 mm per month since the 1950s (Fig. 3) and

(b) significant fluctuations in rainfall at scales of 4.75 years, 7.75 years and 28 years (Fig. 4). These last temporal fluctuations are characteristic of ENSO events. The long-term decline in rainfall since the 1950s has a significant impact on run-off and the salinity balance of the Lakes.

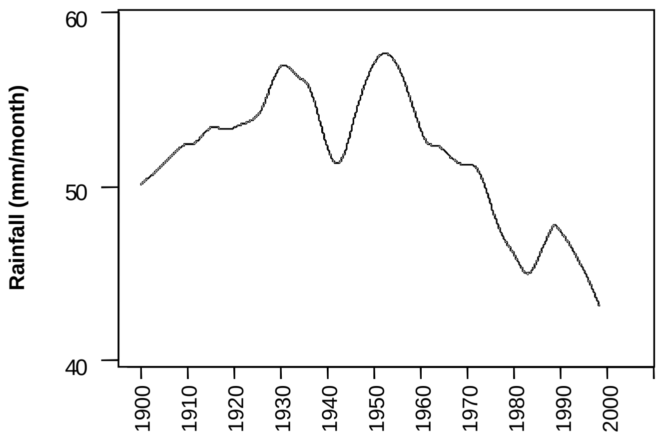

Figure 3: Analysis of Sale rainfall data (1900 to present). Plot shows smoothed monthly rainfall per year.

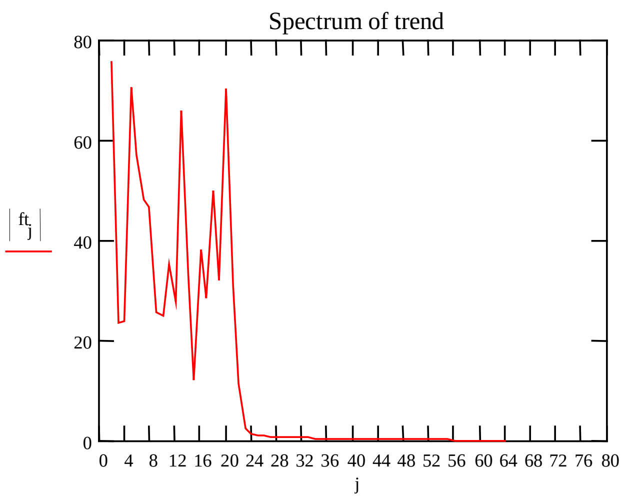

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Figure 4: Analysis of Sale rainfall data (1900 to present). Plot shows spectrum of filtered series, which has three distinct peaks corresponding to 28 years, 7.75 years, and 4.75 years.

4.2 Storm events

Smaller and more frequent algal blooms result from individual storms, river run-off and nutrient inputs into local bays and estuarine areas. The precise causes of each bloom may be quite complex (Chessman in Gippsland Lakes Algal Bloom Seminar, 1988). Clearing of land for agriculture and urban development make such blooms more likely by increasing the overall delivery of nutrients to the Lakes. Only the major bloom events can be associated with the larger ENSO events.

We do know what the response of coastal lakes and estuaries is to increased nutrient loads (Hinga et al. 1995) and the response is common to these systems world wide. Increased nutrient loads increase the overall biomass of plankton in the water and increase the frequency of nuisance algal blooms (Harris 1986). As we shall see, the overall nutrient loading to the Lakes is already at the point where severe environmental degradation and ecosystem impacts can be expected.

Under Australian conditions, much of the nutrient input to the Lakes results from individual storm events. Cullen and O'Loughlin (1982) showed many years ago that in this country the vast majority of the erosion and nutrient movement was associated with individual storm events. Indeed, in one NSW catchment as much as 90% of the nutrient transport and loading occurred in as little as 5% of the time.

So, major rainfall events, such as recently occurred in the Tambo River, account for most of the erosion and nutrient load. To some extent, extreme events are also correlated with ENSO events – being more likely during La Nina events and less likely during drought periods.

All in all, with major loads of nutrient arriving in individual high rainfall events, the Lakes will never be in steady state. Residence times of nutrients and suspended solids in the Lakes are many years in the sediments. Although the residence time of water in the Lakes is about 85-120 days (Lester, 1983), it will take many multiples of this residence time to flush the system out completely. Thus, the Lakes are always responding to the last big nutrient-loading event when the next one arrives.

Analysis of the water quality data from the period 1986 to present indicates major interannual variation in salinity and water temperature (Figs. 5 and 6). Long-term fluctuations (3-5 year) in temperature and salinity are likely to be driven by climate variability and individual storm events. Water temperature was at a minimum in 1986 of around 15 °C and a maximum in 1998 of about 19 °C. The increase in temperature may also be another indication of the long-term warming trend which is occurring in Southeast Australia. Note that other studies have provided a clear indication of this trend in the region (see Harris and Baxter 1996).

Taken together with the long-term responses to marine incursions and catchment land use change described above, no two years in the Lakes will ever be the same. It is against this background that we must begin to assess and explain the events taking place in the Lakes and to suggest remedial action.

The strong interannual variability and lack of steady state response explains the difficulty of measuring and detecting long-term changes in water quality in the Lakes. Nevertheless, the existing data should be brought together and summarised. We believe that the data will show a continuing deterioration in water quality in the Lakes as nutrient loads from the catchment and salinity levels have risen (Longmore et al. 1988).

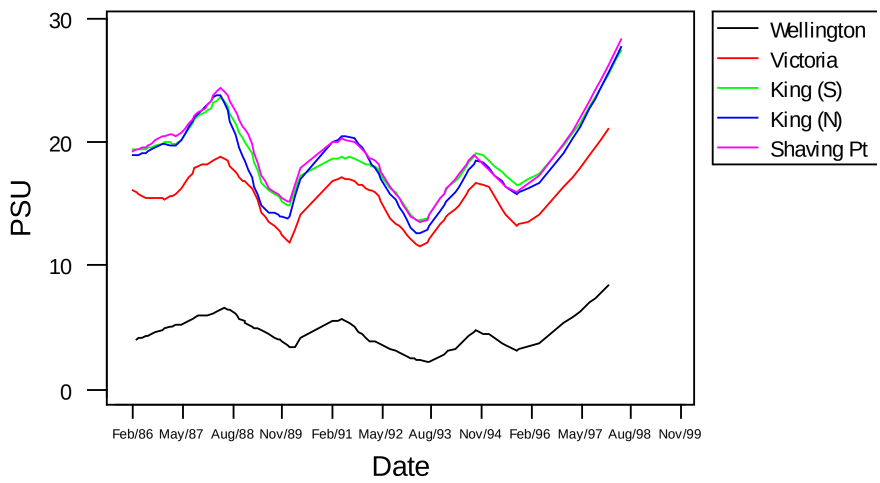

Figure 5: Smoothed salinity data from EPA surface water quality sampling (1986 to 1998)

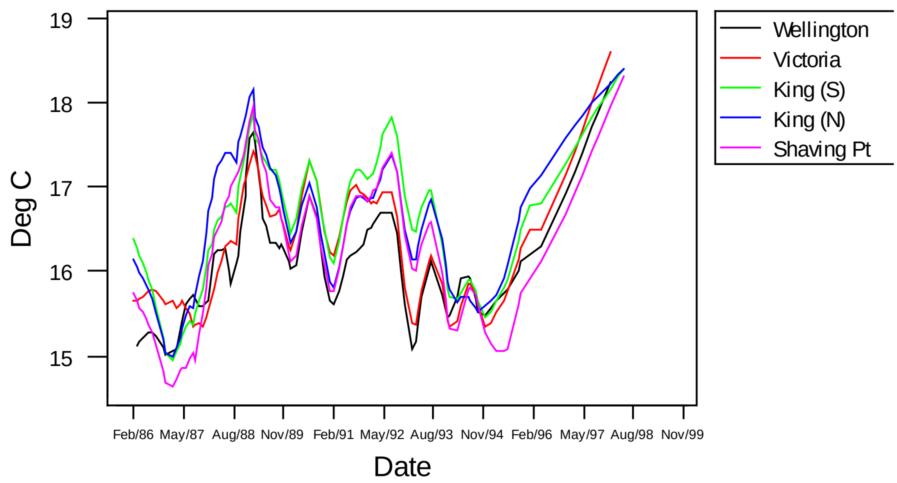

Figure 6: Smoothed water temperature data from EPA surface water quality sampling (1986 to 1998)

Note: The application of a robust smoothing technique was used to identify trends. The degree of smoothing was quite modest (span=0.2; passes=1) in order to preserve some of the finer structure.

4.3 Variability of residence time

The residence time of water in the Lakes has changed considerably since the advent of human interventions. The figure of 85 to 120 days is only an average figure for the modern system with a dredged Entrance. In fact, the residence time of water in various parts of the Lakes varies from a few days during storm flows (when freshwater inflows are sufficient to cover the surface of the lakes and exit from the Entrance) to almost infinity during drought.

Climate and rainfall variability causes the water residence times in the Lakes to vary from year to year and this variation controls the response to nutrient loads. Also, flushing will be much quicker and more effective near the mouth than in regions of the Lakes further away from the Entrance. Part of the variable response to nutrient loads is caused by plankton populations being flushed from the system by high flows, whereas populations of rooted plants and other benthic species are not being similarly affected. Thus the balance of plankton versus rooted aquatic vegetation is partly controlled by flushing of the system.

4.4 Flushing and flow regulation

Increased flushing tends to reduce plankton populations. This has been exploited in situations such as the Peel Harvey system where excessive nutrient loads led to large *Nodularia* blooms (Hodgkin in Gippsland Lakes Algal Bloom Seminar, 1988) and a second entrance was built to increase tidal flushing.

It has been suggested that a similar solution might apply in the Lakes. As we shall see, however, the situation in the Lakes is complicated by strong vertical stratification in the water column and some highly complex hydrodynamics and physics, so the situation in the Lakes precludes a categorical recommendation of a simple solution such as this.

River regulation and water extraction in the major Gippsland catchments has also changed the variability in water residence times in the Lakes by cutting down on the high flows and reducing flows during drier periods (GHD, 1991). The long-term decline in rainfall since the 1950s exacerbates the situation and, together with impoundment, regulation and extraction has led to markedly reduced run-off entering the Lakes.

River regulation tends to push Australian lowland river and estuarine systems towards situations very similar to permanent drought (as has happened in the Hawkesbury Nepean system in NSW, Harris, 1996). It is not possible to quantify this effect on the Lakes at present. Indeed, a full three-dimensional physical computer model of the Lakes system is the only way to fully quantify many of these interactions and this has not yet been built.

Nevertheless, it is possible to say with confidence that reducing the freshwater inflows and reducing the water residence times in the system, thus reducing the rate of flushing, will push the system towards increased salinity and an increased probability of algal blooms and other ecosystem impacts.

It is probably true to say that one of the largest and most insidious impacts on all Australian coastal systems is the alteration of the freshwater inflow regimes and hence the alteration of the impact of the natural climate and rainfall variability. Recent ecosystem modelling (Harris unpublished) has shown some very strong non-linear (and hence surprising) effects of alterations in the water residence times of shallow coastal embayments. Small changes in residence times can have major ramifications in terms of ecosystem impacts ranging from plankton populations, to seagrasses and fish.

4.5 Climate change

While concern has been expressed about longer-term climate change (resulting from increased levels of carbon dioxide in the atmosphere) and there is evidence of a long-term warming signal in southeastern Australia, climate variability is, for the time being, of greater significance. The interannual variability in climate is at least equal to the long-term warming signal and the impacts on the aquatic ecosystems of the Lakes are expected to be more significant.

Nevertheless, the long-term effect of global warming and sea level rise must be considered, although the response to long-term sea level rise is more of an engineering problem than an ecological problem.

Similarly, any subsidence of the land due to groundwater extraction or mining activities will exacerbate the situation. Design of sea walls and dykes must take the possibility of sea level rise and possible subsidence into account.

One essential requirement for future action is the establishment of an adequate monitoring program which, despite interannual variation in climate and rainfall, can detect and monitor improvements in water quality and ecosystem health. Despite a plethora of management plans and actions, it is impossible to tell at present if things are improving or not.

5 The Catchment

There is well-documented evidence of the effects of land use change in the major Gippsland catchment (Aldrick et al., 1992). Sufficient work has been done to begin to quantify not only the magnitude of the loads to the Lakes but also to identify the major source areas (Pooley, 1978; EPA 1983; Erskine et al., 1990; Grayson et al., 1994). However, a note of caution is required: loads have only been quantified for short periods and the longer-term climate and rainfall variation may mean that the data are not totally representative.

It is clear that the major sources of nutrients are erosion and gullying in catchments during storm events and that there is a good correlation between flows and loads of suspended solids and nutrients. As expected, most of the load is associated with large rainfall events and most of the load is transported during brief, extreme events (Grayson et al., 1994).

Although the Lakes represent one of the largest coastal embayments in southeastern Australia (with a surface area of 364 km^2), they also have large catchments feeding them. The total catchment area feeding the Lakes is about 20,600 km^2.

This is the largest of the coastal embayments studied by Scanes et al., (1998). Even though there is a good proportionality between catchment area and the area of the receiving waters (Fig. 7) the catchment area of the Gippsland system is still on the high side of the overall trend.

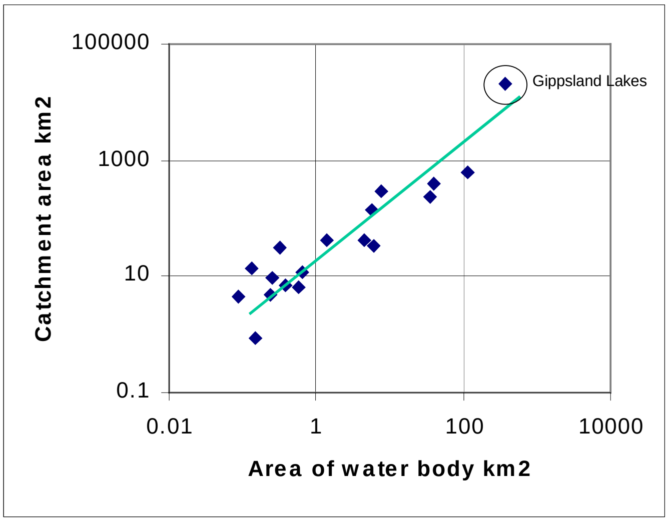

Figure 7: Graph of catchment areas verses area of the receiving water body for coastal embayments in southeastern Australia (Scanes et al., 1998). Note that the axes use log scales.

Even if the nutrient export from the catchments is small on a per unit area basis (as is the case with forested catchments), the overall impact may be nevertheless large, particularly if a small area of the Lakes is disproportionately loaded with nutrients. This is the case for the northern arm of Lake King (Jones Bay) where the full impact of loads from the Mitchell River catchment is felt.

Overall, the nutrient exports from the Latrobe River system (including the Latrobe, Macalister and Thomson Rivers) are twice those of the Mitchell River on the basis of per unit area of catchment (EPA 1983, 1986). This, together with the overall much larger catchment area of the three river systems combined, means that by far the most important nutrient load to the Lakes comes from that system (EPA 1983 and Table 1).

All of the nutrient inputs from the Latrobe River system enter Lake Wellington so the impact of these loads is strongly felt in this lake causing water quality degradation. Overall, it would appear that the nutrient load to the Lake is dominated by high flow events in the river system (either erosion or surface runoff) (Robinson 1995). Even though the inputs from waste water treatment plants are significant during low flow periods, their overall contribution to water quality degradation in the Lake is small. It should also be noted that continuing upgrades to the wastewater treatment plants in the region has led to significant reductions in nutrient flows to the Lakes.

Because the Mitchell River catchment is largely forested, it does not export a large load of suspended solids. However, because of its large size and the restricted area of the receiving waters, it is a significant source of dissolved inorganic nitrogen to the northern arm of Lake King (EPA 1986; Longmore, 1989b). As it appears that the lower Lakes (Victoria and King) are nitrogen limited, the Mitchell River has a significant impact on this portion of the Lakes.

Table 1: Calculated average annual loads (Oct 76 to Jun 78) for selected analytes (tonnes) with percentage contributions from each stream shown in brackets. (Graham et al. 1978).

	Tambo River	Nicholson River	Mitchell River	Avon River	Latrobe River	Thomson River less Macalister	Macalister River	Total to Gippsland Lakes
Total Phosphate	29 (7.2)	14 (3.5)	77 (19.0)	16 (4.0)	122 (30.2)	81 (20.0)	65 (16.1)	400
Ortho Phosphate	12 (13.8)	3 (3.2)	13 (13.8)	3 (3.2)	27 (28.7)	22 (23.4)	13 (13.8)	94
Nitrate/Nitrite	103 (11.6)	46 (5.2)	146 (16.4)	25 (2.8)	447 (50.3)	52 (5.8)	70 (7.9)	889
Total Kjeldahl	770 (23.4)	150 (4.6)	770 (23.4)	150 (4.6)	870 (26.4)	200 (16.1)	380 (11.6)	3290
Ammonia	13 (13.5)	3 (3.1)	6 (6.2)	4 (4.2)	34 (35.4)	-6 (-)	36 (37.5)	96
Silica	2480 (8.2)	2600 (8.6)	9020 (29.7)	1490 (4.9)	7530 (24.8)	4090 (13.5)	3160 (10.4)	30370
Suspended solids	112200 (17.6)	18800 (3.0)	92900 (14.6)	26800 (4.2)	111800 (17.6)	164500 (25.8)	109600 (17.2)	636600
TDS (EC at 25 C x 0.6)	29600 (8.7)	14800 (4.3)	93200 (27.2)	16500 (4.8)	139600 (40.8)	23900 (7.0)	24500 (7.2)	342100
Daily Avg Flow (ML/day)	1133 (10.8)	261 (2.5)	3132 (29.9)	613 (5.9)	2504 (23.9)	1470 (14.0)	1363 (13.0)	10476
Catchment area (km^2)	2876 (13.7)	530 (2.5)	4778 (22.8)	1650 (7.9)	4786 (22.8)	1648 (7.8)	2012 (9.6)	20999

6 The Lakes

6.1 Lake Wellington

The nutrient status of Lake Wellington is eutrophic by OECD standards (Bek and Bruton, 1979; Robinson, 1995) with frequent algal blooms and the Lake appears to be phosphorus limited. This is the usual situation and the response of the lake to nutrient loads can be compared to international guidelines.

The Lake appears to be becoming more eutrophic with time and the frequency of algal blooms has increased since the 1970s as the nutrient load from the rivers has increased. Total phosphorus in lake waters increased markedly between the 1970s and the 1990s (Robinson, 1995). The strong interannual variability (Fig. 8) in the data makes trend analysis difficult, but an upward trend from the 1970s to the 1980s is clearly present followed by and further increase to around 1993-4. Phosphate levels in Lake Wellington appear to have fallen slightly from 1994 to present (Figs. 9 and 10). (Probably because of reduced P inputs during the drought, and reduction in loads from wastewater treatment plants.)

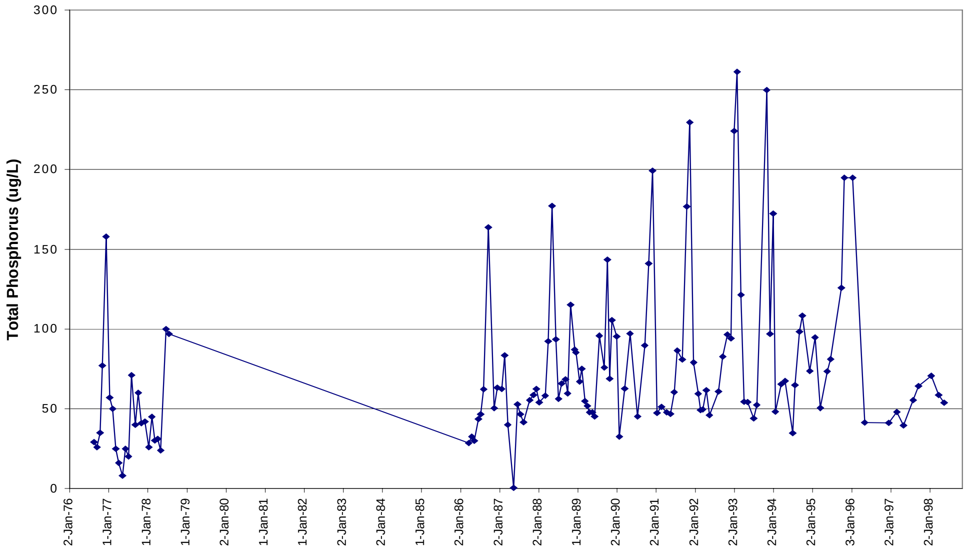

Figure 8: Total Phosphorus (µg/L) as measured monthly by the EPA at a fixed site within Lake Wellington (Stn 2306) from 1976 to 1978 and from 1986 to 1998.

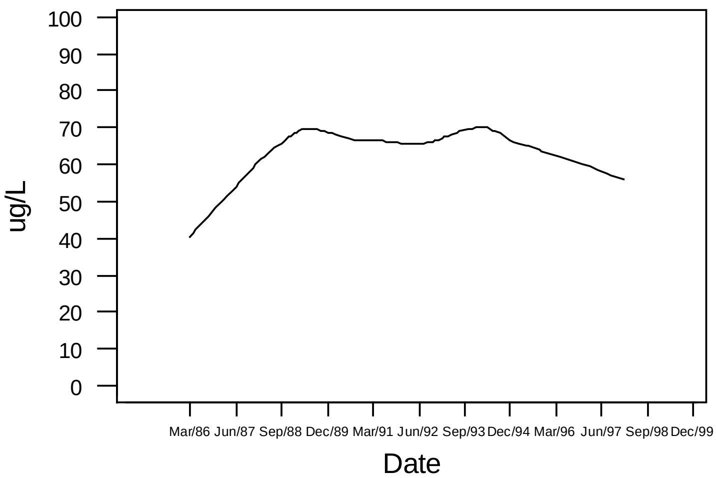

Figure 9: Smoothed Total Phosphorus levels for Lake Wellington data (Stn 2306, 1986-1998). Heavy smoothing was applied to tease out gross trends.

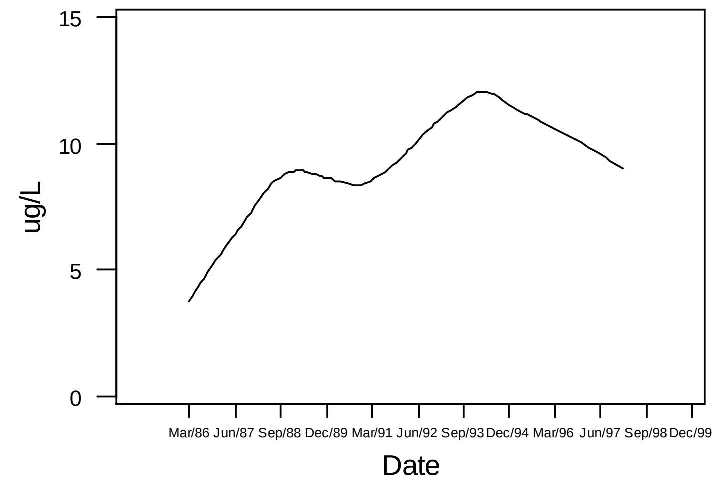

Figure 10: Smoothed Phosphate levels for Lake Wellington data (Stn 2306, 1986-1998). Heavy smoothing was applied to tease out gross trends.

Lake Wellington is also changing because of the longer-term effects of saline incursions from the sea. After the opening of the Entrance seawater flooded into the lower portions of the Lakes and salty water underlies most of the system.

In effect, the Lakes are a typical salt-wedge estuary in which freshwater flows out over the salt water below and mixing processes at the interface ensure that salt is exported in the surface waters. Mass balance considerations necessitate that salt water is drawn into the estuary so that, while the net flow is outwards in surface waters, the net flow in deeper saline waters is inwards towards the riverine end of the estuary. During drought, there will be occasions when the freshwater inflows are less than the evaporation rate thus drawing saline waters into Lake Wellington (Black, 1989; Black and Hatton, 1989; Lake Wellington CSMPCWG, 1993; Hatton et al., 1989; Lester, 1983; Longmore, 1989a; Mobley et al., 1983; Newell, 1991; Pooley and Helman, 1986).

Much of the original freshwater marsh system is slowly being replaced by saline scalds and salt marshes, as part of the long-term response to marine incursions described above.

As catchment flows have been regulated, and river flows have been reduced even more during the more frequent droughts since the 1960s, Lake Wellington has become more saline (See Fig. 2, annual rainfall, and Fig. 3, smoothed annual rainfall).

Enough work has been done to adequately model the relationship between river flows and salinity in the Lake (Longmore, 1989a). Lake Wellington's salinity rises during periods of drought (Fig. 5), rising steeply if the net water balance of the Lake is negative and salt water is drawn in through the channel connecting to Lake Victoria by evaporation. The causes of rising salinity in Lake Wellington, the surrounding wetlands and the Latrobe estuary are well-understood (Newell, 1991; Lake Wellington CSMPCWG, 1993).

Evidently, the Lake was clear and dominated by aquatic macrophytes until the late 1960s. This was, of course, a desirable state. During the 1968 drought, the salinity of the Lake rose to the point where the aquatic plants were killed and the Lake has since become turbid and dominated by plankton (Ducker et al., 1977).

Lakes naturally exist in two stable states – either clear and macrophyte dominated or turbid and dominated by phytoplankton (particularly cyanobacteria or blue-green algae, Blindow et al. 1993) and can be switched from one state to the other by events which kill the macrophytes.

In the case of Lake Mokoan, an off-river storage in northeastern Victoria, the macrophytes were killed by drawing the lake down during drought. When refilled the lake became turbid and dominated by blue-green algae.

The salinity levels in Lake Wellington now sometimes exceed 20 which is more than 50% sea water (Marwood, 1989), This salinity is sufficient to cause the death of fringing aquatic plants and trees. Simple models can easily explain the variability in Lake salinity in relation to inflows (Mobley et al., 1983; Marwood, 1989; Longmore, 1989a).

The freshwater wetlands around Lake Wellington are being replaced by saline marshes, a situation exacerbated by the rising saline ground waters of the surrounding areas of the lower Latrobe catchment. More than 10,000 ha are now affected (DCE, 1991). In addition, bank erosion is endemic and the situation is now made worse by the presence of carp which were introduced in Victoria in the early 1960s (Ducker et al., 1977; Clucas and Ladiges, 1980).

Restoration of the natural wetlands surrounding the Lake is essential. Deliberate flooding of these areas with freshwater inflows would be desirable. A system of weirs and sluice gates would be required. Some studies of this have been carried out (GHD, 1991).

Lake Wellington tends to be highly turbid because of the inflows of sodic dispersed clays from the catchment soils and from the increased loads of eroded material from gully and stream banks after rain. In addition, the Lake is large and shallow enough to enable resuspension of settled material by wind forced mixing. This, together with the presence of carp in the Lake causes the water to be continually turbid.

Even if aquatic plants were replanted artificially, it is doubtful whether they could recolonise in a turbid, dark and unstable environment. This turbid environment favours the growth of phytoplankton such as cyanobacteria, which are able to control their buoyancy in the water column and float to the surface during calm weather. If blooms of potentially toxic cyanobacteria are not more frequent it is probably because the Gippsland Lakes are in an exposed region of coastal eastern Victoria which experiences much windy weather. In this respect, they are like Port Phillip Bay in being characterised by shallow bathymetry, large surface area and a large wind fetch (Harris et al 1996). This wind forced mixing increases turbidity and reduces stratification, which results in reduced algal growth.

The biomass in Lake Wellington is now dominated by phytoplankton because few plants can live in an environment which is partly salt water and partly fresh. The saline incursions are sufficiently frequent and severe that recolonisation by freshwater plants is unlikely for that reason also.

Hatton (1989), Black (1989) and Marwood (1989) measured and modelled the saltwater inflows to the Lake and studied whether or not the placing of a barrier in the channel would reduce the influx of salt water. The significant result was that while a barrier would stop some of the saltwater incursions from Lake Victoria, there were occasions when the barrier would actually stop the saltwater getting out again, thus exacerbating the problem.

One way to stop salt water getting into Lake Wellington would be to place a lock across McLennans Strait. Such a barrier would impact on fish migration and would be an inconvenience for boat operators. It may also lead to ongoing management issues through its operation and maintenance. Even if all saltwater incursions were eliminated, it may not be possible to return the Lake to its pre-1968 condition given the magnitude of the nutrient and suspended solids loads from the Latrobe River and the presence of carp.

Another solution is to take the Latrobe River system flow out of the Lake entirely and divert it to sea, thus allowing Lake Wellington to become more marine. This would exclude the carp but, of course, would do little for the deleterious influences of salt on the surrounding marshes.

6.1.1 Future options

The only long-term solution is to markedly reduce the nutrient loads from the Latrobe system by both restoring the catchment to a more sustainable land use and by replacing riparian vegetation and reducing erosion. A program for elimination of the carp would also be worthwhile. While elimination is still not possible (CSIRO is involved in research aimed at controlling carp, see Roberts and Tilzey 1997), programs to markedly reduce their numbers would be desirable.

A large-scale nutrient management plan for the entire Latrobe, Thomson, Macalister River catchment system is also required. To achieve significant improvements in water quality, a 50% reduction in catchment nutrient loads is required (Robinson, 1995).

The importance of wetlands, in reducing nutrient loads to the lakes should not be underestimated. The capacity of existing wetlands and means for increasing that capacity to reduce the loads should be investigated and acted upon. Simultaneously, it may be worthwhile to consider engineering large-scale artificial or enhanced natural wetlands at critical points within the catchment and at the river mouths to trap loads of suspended solids and nutrients before they enter the Lakes. Wetlands in addition to reducing nutrient loads provide conservation areas, wildlife refuges and areas of increased biodiversity. However, detailed investigation would be required to establish design requirements necessary to manage wetlands during periods of major flooding.

6.2 Lakes Victoria and King

The lower two Lakes in the Gippsland system are much more marine environments than Lake Wellington. Salinities increase steadily from around a salinity of 5 in Lake Wellington to 25 to 30 near the Entrance. (Seawater has a salinity of about 35.) Numerous water quality surveys (see eg Longmore, 1988) have documented this pattern.

The lower lakes are flushed by tidal exchange though the Entrance. Minor modifications to the Entrance will have little effect on the tidal flushing overall, but may have more significance locally. We do not know precisely how the tidal inflows and outflows exchange with the Lakes' waters. This will effectively control the expression of the loads in terms of water quality and algal blooms. This is a crucial piece of information to a more complete understanding of the transport of fluvial inputs through the Lakes and to the sea.

Analysis of EPA water quality data from the Lakes shows no clear trend or marked deterioration in water quality in the lower Lakes since the mid 1980s (Figs. 11 and 12). Marked effects of major freshwater inflows are seen in the 1987-88 La Nina (wet period) event with subsequent fluctuations from year to year. The data does show an improvement in water quality during the drought of the late 1990s – similar to Port Phillip Bay (Harris et al., 1996). The long residence time of water in the lower Lakes (85-100 days) will tend to smooth out the effects of the individual rainfall events and show the observed 3-5 year fluctuations.

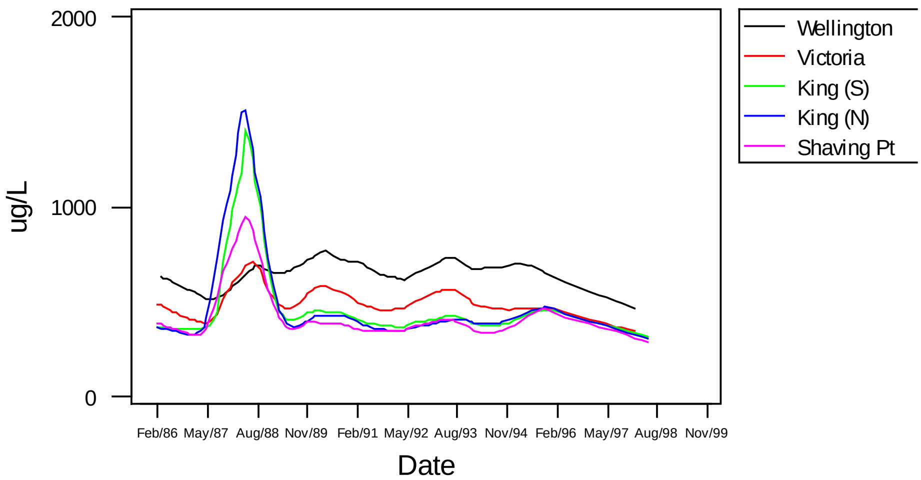

Figure 11: Smoothed Total Nitrogen data from EPA surface water quality sampling (1986 to 1998).

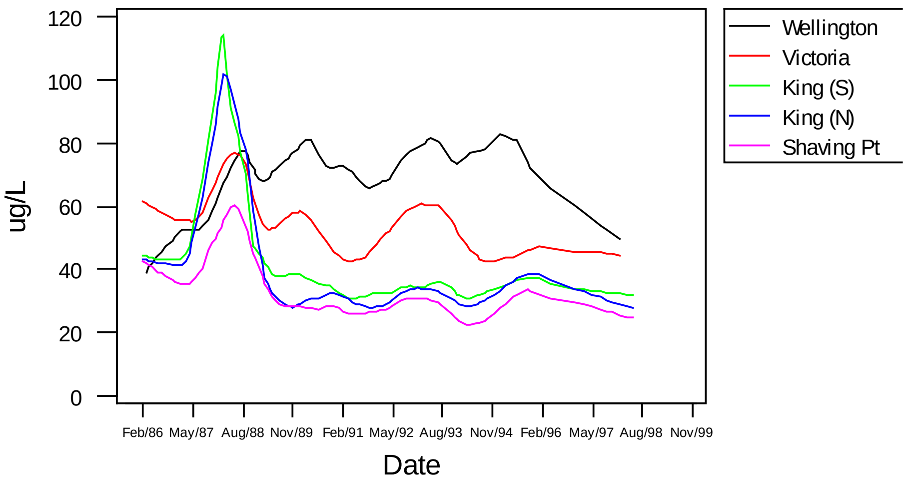

Figure 12: Smoothed Total Phosphorus data from EPA surface water quality sampling (1986 to 1998).

There is a marked difference in the structure of the water columns of the Lakes between summer and winter (Bek and Bruton, 1979; Longmore, 1988). Peak rainfalls usually occur between May and December so that the normal winter pattern is of a salt wedge estuary with freshwater outflows overlying saltier bottom waters. The residence time in surface waters may be quite short with freshwater inflows crossing the Lakes and exiting the Entrance taking a fraction of the suspended solids and nutrient loads with them. In Lakes Victoria and King, surface salinities in winter range from a salinity of less than 10 to a salinity of 25 and bottom waters are closer to fully marine concentrations (about 35). There is also a strong vertical temperature gradient in winter.

In summer, the pattern is different (Bek and Bruton, 1979; Longmore, 1988). Freshwater inflows are markedly reduced and evaporation may dominate the water balance for long periods. Lakes' waters are usually vertically mixed so that the water column is both isothermal and isohaline. In summer, the isohalines (contours of constant salinity) are usually vertical indicating the presence of minimal salinity stratification and strong salinity gradients across the surface of the Lakes (Bek and Bruton, 1979). The northern arm of Lake King (Jones Bay) can be regarded almost as a separate water body with reduced salinities and more marked nutrient impacts because of the inputs of freshwater, nutrients and suspended solids from the Mitchell River. Conditions in this area are less marine and more heavily impacted than the rest of Lake King (Longmore, 1989b).

This complex pattern of tidal exchanges and density stratification is the reason why it is impossible to predict what the effect of the construction of a second Entrance or a series of tunnels under the dunes might be. While a more effective exchange with the sea would produce more effective flushing, it might also lead to a longer period of stratification with full seawater at salinity of 35, underlying the slightly fresher Lakes water even in summer. As we shall discuss, the strong oxygen demand and high rates of ammonia release from bottom sediments in the lower Lakes make such an effect undesirable.

Because Lakes Victoria and King they are more marine than Lake Wellington, they are clearer (Fig. 13) (the higher salinity causes the suspended solids to flocculate and settle out) and are characterised by extensive beds of seagrasses and aquatic macrophytes. There is also a rich marine benthic fauna (Poore, 1982). The species present can be said to be some of the more opportunistic species and it appears that the abundance and spatial extent of the macrophytes and seagrass beds vary from year to year (MacDonald, 1997). The seagrass beds require high light intensities and clear water and are adversely affected by large freshwater inflows and the associated turbidity. Thus, the beds appear to shrink during wet periods and expand again during periods of drought when the environment is more marine and the water clearer.

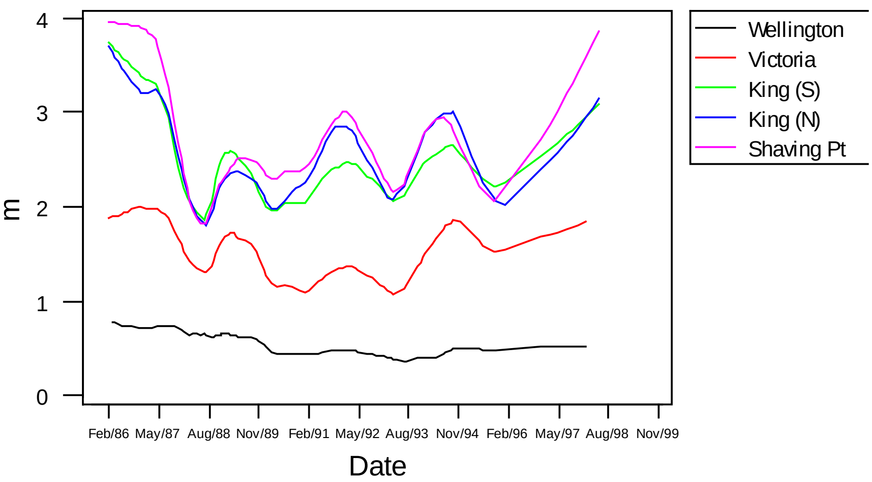

Figure 13: Smoothed secchi disk depths data from EPA water quality sampling (1986 to 1998).

There appears to be only anecdotal information about this important aspect of the ecology of the Lakes. However, a recent analysis of the long-term changes using historical aerial photographs illuminates the subject (Roob and Ball 1997). Although the photographic data are scattered in time, it is possible to show that the seagrass beds were at their maximal extent in the late 1960s and the late 1990s. The beds seem to have decreased in aerial extent during the late 1970s and early 1980s. These long-term fluctuations are to be expected because of the long-term fluctuations in rainfall, which occur in the region. Seagrasses and aquatic macrophytes seem to do better during periods of drought, lower nutrients and clearer water.

Photographic evidence in one of the reports indicates large growths of epiphytic algae on the blades of seagrass and submerged macrophytes on occasions (Gunthorpe, 1997). This is indicative of marked eutrophication in the water column. Excessive growth of epiphytes leads to shading and the death of the seagrasses. Similarly the appearance of large *Nodularia* blooms in wet years (such as 1987-88 – a La Nina year) in the lower lakes is also indicative of severe degradation in water quality. Evidently, the nutrient status of even the lower lakes is sufficient to produce all the signs of degraded water quality resulting from nutrient inputs in wet years.

Oxygen depletion in bottom waters is another sign of widespread eutrophication in the Lakes. Oxygen depletion in bottom waters occurs when the oxygen demand of the sediments is high enough to remove oxygen from the water while the exchange of surface and bottom waters is sufficiently slow to preclude re-oxygenation from the atmosphere.

The physical structure of Lakes Victoria and King is quite complex so the pattern of anoxia is similarly complex. Density stratification is prevalent in winter and even in summer when the water column is more frequently vertically mixed, areas of widespread oxygen depletion are observed. Sediment oxygen and carbon fluxes are high, another sign of high nutrient enrichment, high water column production rates and high decomposition rates in bottom waters. As expected under these circumstances, large ammonia releases from the sediments are observed.

Fluctuation in the extent of the seagrass beds and the prevalence of oxygen depletion in bottom waters appear to have a significant effect on the recruitment of some of the more important marine fisheries in the Lakes. Again the data are sparse and the arguments are based on the timing of events in the Gippsland Lakes and other Victorian coastal embayments (Port Phillip Bay and Western Port). (See arguments presented in MacDonald (1979) and Rigby (1982) and Longmore et al., (1990)).

Indications are that Lakes Victoria and King are important areas for bream reproduction and recruitment. Increased recruitment is dependent on successful reproduction and development of juvenile fish. From spawning to recruitment takes 5 to 6 years. However, successful spawning occurs in waters with a salinity between 18 and 21, and water temperature between 22 and 23°C. Optimal larval survival rates then depend on having large areas of vegetated benthic environment (M. MacDonald, personal communication).

The fisheries data seem to indicate that at present environmental factors (eg. changes in habitat, loss of seagrass, and variation in salinity and water temperature) are more severe than the influence of overfishing (Gunthorpe, 1997). However, this should not overshadow the impacts that overfishing can and has exerted on the Lakes. Recent restrictions on the commercial and recreational fish catch, together with some good recruitment years (Walker et al. 1998) seem to have controlled the situation. Continued monitoring of the fishery should establish whether it is actually the fisheries management practices that are improving the level of fish stocks.

Data in MacDonald (1997) show that bream reproduction is more successful in years when freshwater input is lower than average (ie. drought periods - when the seagrass beds are in recovery). Carp numbers appear to increase after wet periods (periods of lower salinity and higher turbidity). This is a pattern that may be expected in a system that oscillates between freshwater and marine influences depending on climate and rainfall variability (Hobday and Moran 1983). Hobday and Moran did show a link between bream recruitment and climate variability driven by ENSO events.

A more recent and thorough analysis of the recruitment data for Black Bream (Walker et al. 1998) shows that good and bad recruitment years are driven by both rainfall and temperature and only one of their models showed a clear link to ENSO events. Certainly, bream require a combination of water column productivity and extensive beds of benthic vegetation for good recruitment so interannual variability, driven by longer term climatic variability and individual rainfall events can lead to complex patterns from year to year.

The picture that emerges is of a series of tidally flushed coastal embayments that show marked interannual variability and habitat change due to nutrient enrichment. The overall health of the Lakes ecosystem is heavily influenced by freshwater and nutrient inflows from the catchments during wet periods and there is some recovery during dry periods when the water clears and a more marine environment is restored.

6.3 Comparison with other lakes and estuaries

Water quality measured in the early 1980s (EPA 1986) showed that the average algal biomass of the water of the Lakes (as measured by the concentration of the photosynthetic pigment chlorophyll a in $mg.m^{-3}$) was in the eutrophic range as recognised by OECD standard criteria.

Lake Wellington was phosphorus limited and eutrophic (average chlorophyll 16 $mg.m^{-3}$ with values as high as 50 $mg.m^{-3}$). All indications are that the lower Lakes are nitrogen limited and also eutrophic (chlorophyll a averaging 5-6 $mg.m^{-3}$). This average chlorophyll concentrations in the lower Lakes (Victoria and King) is five to six times the average value in Port Phillip Bay, entirely consistent with the nutrient loadings and international comparisons published in Hinga et al. (1995) and Monbet (1993).

It may be unfair to make comparisons with average values for Port Phillip Bay due to the differing physical characteristics of each. However, comparing data from the EPA's fixed site program for Hobsons Bay near the mouth of the Yarra River and chlorophyll a levels from the northern Lake King site for the corresponding period, we see that the difference is quite clear (Fig. 14).

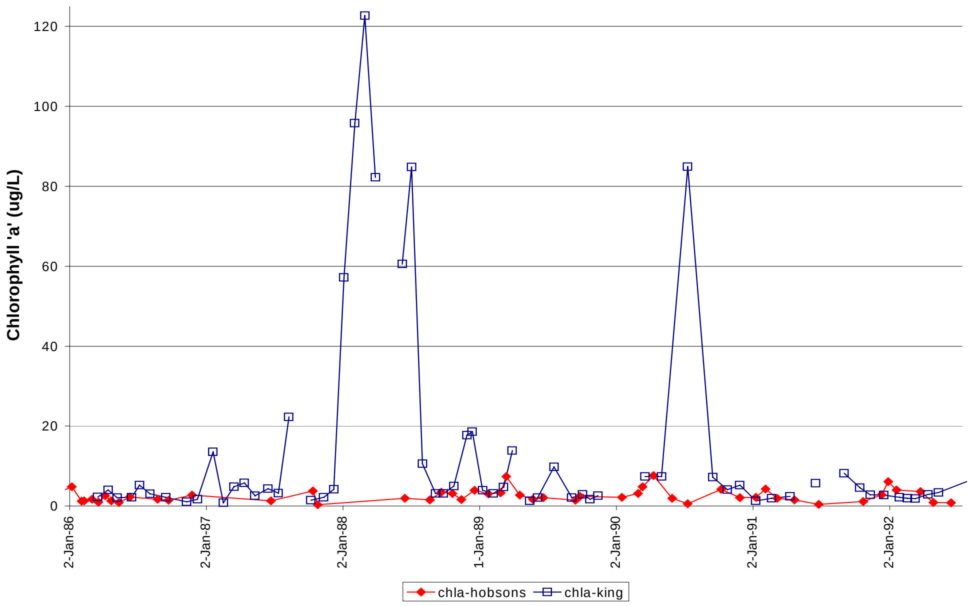

Figure 14: Graph of chlorophyll 'a' from EPA surface water quality sampling programs (1986 – 1992) for Hobsons Bay (Stn 1991) in Port Phillip Bay and for Lake King North (Stn 2316). The lower levels are in Hobsons Bay.

When nutrient loads to the Lakes are calculated on a load per unit surface area basis, then it is possible to compare the loads to other coastal lakes and estuaries. When this is done (Fig. 15) it becomes clear that the loads to the Gippsland Lakes are on the verge of critical for nitrogen and slightly less so for phosphorus.

A comparison of the Lakes with other coastal embayments along the eastern coast (Scanes et al., 1998) reveals that the Gippsland Lakes are similar to other systems. They are showing symptoms of nutrient enrichment or eutrophication viz. frequent algal blooms, loss of seagrasses and some fish populations and oxygen depletion in bottom waters. There is, therefore, nothing unusual about the ecosystem responses of the Lakes system to the increased nutrient loads.

Comparison of the nutrient loading and responses of the Gippsland Lakes to other estuaries and coastal embayments around the world gives a similar result. The Gippsland Lakes nutrient load per unit area of lake surface (8 - 10 g N m^{-2} y^{-1}) is about three times the load to Port Phillip Bay (Harris et al., 1996) and exceeds the load predicted to produce severe water quality degradation in the Bay.

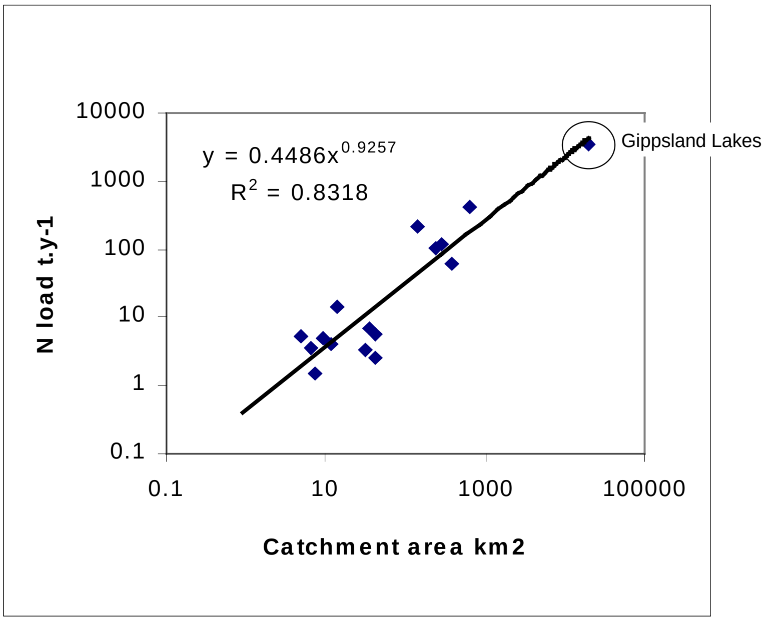

Figure 15: Graph of nitrogen loads per unit surface area for coastal embayments in southeastern Australia. (Scanes et al., 1998). Note that the axes use log scales.

When compared to other estuaries around the world, the nutrient load to the Lakes is similar to (or even greater than) nutrient loads to heavily impacted estuaries such as Tokyo Bay and parts of San Francisco Bay.

The picture that emerges is for a need to cut the nutrient loads to the Lakes significantly in order to make a marked improvement in water quality and restore ecosystem health.

7 Nutrient budgets

Another way to look at the functioning and health of the Lakes is to examine the budgets of the major nutrients entering and leaving the lakes. To explain the significance of this technique, it is sufficient to note that healthy estuaries and lagoons retain and recycle a large fraction of the nutrients entering the system and, in particular, show high denitrification efficiencies.

Denitrification is the process whereby nitrogen loads to the system are converted from nitrate to nitrogen gas – which is then lost to the atmosphere. By this means, the aquatic ecosystem has a strong capability to be self-cleansing. Port Phillip Bay, for example, converts about 80 to 90% of the nitrogen load to nitrogen gas and is thus in a healthy condition (Harris et al. 1996).

Ammonia fluxes from the sediments are a sign of low denitrification efficiencies, so the large fluxes of ammonia from the sediment in the Lakes are another way to measure the health of the system.

Bek and Bruton (1979) constructed nutrient budgets for the major elements entering the Lakes and showed that the Lakes retained between 46 and 56% of the total suspended solids loads and between 35 and 40% of the total phosphorus loads. Both of these figures are in the expected range for eutrophic water bodies. Retention of total organic carbon was about zero (-16% to 2% - a negative figure indicating that the Lakes were a net exporter of organic carbon on occasion). Retention of nitrate depended on flow conditions. Retention of nitrate varied from –32% during floods to 75% during drought. Denitrification efficiency, and hence retention of nitrate, is higher during drought when water retention times are longer, seagrass beds are more extensive and internal nutrient recycling is more efficient. During floods, nitrate is washed from the system and is exported to the ocean. We have developed a budget for Total Nitrogen for a two year period (1988 to 1990) for Lakes Wellington and Victoria. The average retention rates for these two lakes were calculated to be 31% and 24% respectively. These values are low and are consistent with a eutrophic status for these waterbodies.

Ammonia retentions show clear signs of eutrophication and high sedimentary ammonia fluxes. Retention coefficients range from – 482% (flood) to – 314% (drought). In other words, as a result of the decomposition of the high primary production in the system, the Lakes are decomposing organic matter and exporting the ammonia to the ocean. Between three and four times as much ammonia leaves the Lakes as enters them in rivers and streams.

When oxygen concentrations within and close to the sediments are sufficiently low, ammonia is not converted to nitrate (nitrification). The conversion of nitrate to nitrogen gas, which is lost to the system (denitrification), does not occur. Consequently, ammonia builds up in bottom waters where it can be a significant internal source of fertiliser for renewed production in the water column.

Davis et al. (1977), who studied the sediment characteristics, found that in all the Lakes the sediments are coarse at the margins and finer in the centre, with depositional zones in deeper waters where most of the production is decomposed. It is in the central lake basins where most of the ammonia fluxes occur and most of the deoxygenation takes place.

Longmore (personal communication) has recently measured sediment fluxes of the major elements and has shown that the rate of decomposition of organic matter on the sediments is up to twice that of Port Phillip Bay and that denitrification efficiencies are frequently of the order of 30%. Parts of the Lakes (at the margins where sediments are coarser and seagrasses more abundant) have higher efficiencies than this, but an effort must be made to reduce nutrient loads, reduce decomposition rates and increase denitrification efficiencies so that efficiencies closer to 100% are more widespread.

Longmore (1994a) also reported that Lakes Victoria and King have rich growths of microphytobenthos. Microphytobenthos can reach quite a high biomass and may have a profound effect on cycling of nutrients in the Lakes. In Port Phillip Bay microphytobenthos photosynthesize during the day and intercept nutrients released from the sediments, thus reducing nutrient loads to the water column (Harris et al. 1996, Beardall and Light, 1997). It is likely that the microphytobenthos are similarly responsible for increased denitrification efficiencies in the Lakes and hence act to improve water quality. The importance of microphytobenthos needs to be investigated further, in order to enable nutrient budgets to be adequately established.

8 Algal blooms

It is now clear why the algal bloom dynamics in the Lakes are complex. In addition to the major events associated with the major ENSO events, further blooms are to be expected frequently. Not only are the nutrient loads to the system from the catchments high enough to stimulate the growth of blooms, but there is also a significant internal source of nutrients – phosphorus and ammonia from decomposition in the sediments.

Longmore (1994) measured sedimentary releases of nutrients and showed high rates of release of both nutrients. Rates were comparable to other eutrophic estuaries around the world. Thus after periods of vertical stratification, when nutrients build up in bottom waters, wind or tidal mixing can suddenly release large amounts of nutrients into surface waters and stimulate the growth of blooms.

Reductions in the rates of nutrient release by the sediments are an important management tool and should be factored into any nutrient management plan. Marked reduction in catchment loads (by about half) will control production, decomposition and sediment nutrient fluxes. In the case of phosphorus, direct intervention to control phosphorus release rates by means of sediment remediation could be investigated. Unfortunately, the presence of carp in high numbers in places such as Lake Wellington may serve to negate or reduce the efficacy of any such plans.

The usual situation appears to be that nutrients in Lakes Victoria and King are depleted in surface waters as a result of algal growth and that nutrients build up in bottom waters during stratified periods. Vertical mixing will make these nutrients available for subsequent algal growth and a bloom may develop. This is one reason why actions attempting to improve the flushing of the lower Lakes with seawater (and probably increase the density stratification in the system) must be viewed with caution.

Recent computer modelling of shallow estuaries (Harris unpublished) shows that the response to nutrient loads in these systems is particularly sensitive to physical processes. Water residence times and vertical mixing dominate the processes that control flushing and internal nutrient loads from the sediments. It is possible to get a wide range of responses to external loads over time as rainfall, freshwater flows and wind alter the physical structure of the Lakes. The balance of seagrass growth (the desired state) and algal blooms (the undesirable state) is a very sensitive indicator of ecosystem health and is strongly modified by climate variability.

9 Toxicants

Work recently done by MAFRI (Fabris personal communication) indicates that, with the exception of mercury (and perhaps selenium), toxicants are not a problem in the Gippsland Lakes. This is consistent with experience in other Victorian coastal embayments like Port Phillip Bay. Levels of pesticides and PAHs are, in fact, much lower than in Port Phillip Bay. Other heavy metals are also present in low concentrations (Glover, 1981).

Historical data show that selenium was elevated in fish in the 1980s (Glover et al., 1980). The probable source was the output from the Latrobe Valley power stations (Lake Macquarie in NSW has a major problem in this respect), both atmospheric and aqueous discharges, although the latter now goes directly to the sea.

The mercury problem does require further investigation. There is evidence of rising mercury levels in fish and sediment concentrations are alarmingly high. The latter are up to 400 times the recommended lower sediment guideline concentration for mercury and up to 40 times the upper guideline. Such exceedences imply a likelihood of ecosystem impacts, although it should be pointed out that the presence of high sulfide concentrations would ensure that mercury was present as insoluble mercury sulfide, which is not bioavailable. The chemical form of mercury needs verification.

Recent MAFRI data (Fabris personal communication) indicate that mercury concentrations in fish have risen 40% since the 1980's and that bream are close to (but under) the NH&MRC guidelines for human consumption. Other species were previously over the guidelines but only bream have been analysed recently, and it is presumed that similar increases would be expected in mercury in these species.

The selenium concentrations measured in fish in 1980 (Glover et al.) exceeded the NH&MRC guideline value of 1 mg/kg. This is worthy of further examination as the only data were for livers and not muscle tissue. Even so, current research indicates that there are health benefits from increased selenium in the diet. Selenium has been shown to counteract mercury toxicity in many species of fish. It has been suggested, for example, that mercury guidelines in fish be modified on the basis of selenium levels. Further examination of existing selenium concentrations and forms in sediments and fish is warranted. With the diversion of the power station effluents selenium sources may now be absent, unless there remains a flux from contaminated sediments or an input from other sources.

The likely sources of the mercury are historical gold mining operations, which were widespread in the Gippsland catchments and operated until earlier this century. Other sources that might have contributed mercury include a paper mill and the Latrobe Valley power stations mentioned earlier. We have already discussed the high rates of decomposition in the sediments of the Lakes. Under these circumstances the cycling of various readily available forms of organic mercury will be rapid and will contribute to the accumulation of mercury in food chains. Such processes are not well understood and further work is warranted. CSIRO is already working on similar problems in Tasmania and Papua New Guinea. Because of the risk to public health, further work on mercury cycling and food chain accumulation in the Gippsland Lakes is recommended as a matter of some urgency.

10 Actions for the future

In conclusion we recommend the following:

1. In the absence of knowledge that may be gained from proposed modelling (see recommendations below), it is recommended that overall nutrient loads to the Gippsland Lakes need to be reduced significantly. A 50% reduction has been estimated based on the loads published in reports available for this review. In order to achieve this reduction, catchment land use needs to be changed towards more sustainable practices. Reafforestation and the regeneration of riparian vegetation should be encouraged so as to reduce erosion during storm flows. Nutrient reduction targets should be set and adhered to.

2. A thorough analysis of the long-term trends in water quality needs to be carried out to determine the rate of degradation of the Lakes system. This will also involve a thorough analysis of the trends in flows and loads to the system. There is a need to examine the causes of the temporal fluctuations in water quality as well as the sources and transport of nutrients and suspended solids through the system. The existing sources of nutrients need to be quantified in both spatial and temporal terms. The existing water quality monitoring programs should be evaluated, with reference to knowledge gained from this and other tasks.

3. A three-dimensional hydrodynamic model of the Lakes needs to be constructed to examine in more detail the precise interactions of freshwater and marine inflows, tidal mixing and Entrance dynamics, and saltwater/freshwater balances. The hydrodynamic model would be integrated with an ecological model to provide a synthesis of scientific understanding of the ecosystem of the Gippsland Lakes. The integrated model will, in particular, need to examine the interaction of physical processes, sediment nutrient fluxes and algal blooms as well as the influence of flow regulation in the rivers on salt balances. The modelling would be undertaken in conjunction with an extensive field data collection program (eg. water level monitoring, CTD surveys, thermistor chains, local meterological data including solar radiation, nutrient measurements and measurements of input loads including impact of storm events).

4. The integrated model should be used to predict the impacts of management decisions (scenarios) which are designed to alter the nutrient loads to the Lakes. The scenarios may be related to flow control, catchment practices, farm management, urban development, sewage systems, and the control of boating and fishing activities.

5. Measurement of sediment nutrient fluxes in conjunction with research on the influences of external and internal nutrient loads on the dynamics of algal blooms is required. The major factors influencing nutrient dynamics need to be better understood. This should include establishing the importance of microphytobenthos in the Lakes.

6. The existing distribution of benthic habitats and fringing vegetation needs to be recorded (create temporal reference point against which change can be measured) and the value of each habitat established. In conjunction with this study the factors influencing seagrass growth, loss and recovery need to be studied with particular reference to the interactions between habitat change, variability and fish recruitment in major commercial species. A time series of records of seagrass extent needs to be constructed if possible.

7. Further investigation and research on mercury sources, sinks and cycling is required to enable more effective management of what might be a growing public health problem. Possible interactions between mercury and selenium toxicity need to be examined.

8. Further investigation and research on the likely impact to the system from the coarse sediment plugs (sand slugs) within the catchment streams and rivers is required. Work should include investigation of possible remedial actions to mitigate or remove the impact.

9. Overall there is need for the establishment of management plans accompanied by effective performance monitoring. There is a need to be able to determine what the trends are, how well catchment actions are performing and how much progress there has been. In a system which shows strong interannual variation this will not be easy, but it can, and must, be done.

11 References

Aldrick, J.M., Hook, R.A., van de Graff, R.H.M., Nicholson, B.M., O'Beirne, D.A., and Schoknecht, N.R., (1992). A study of land in the catchment of the Gippsland Lakes Volume 1 and 2. Land and Catchment Protection Division, Dept. Conservation and Natural Resources, Victoria.

Beardall, J. and Light, B. (1997). Microphytobenthos in Port Phillip Bay, Victoria: Distribution and primary productivity. Port Phillip Bay Environment Study Technical Report no. 30, Melbourne.

Bek, P.A., and Bruton, G., (1979). The Hydrochemistry of the Gippsland Lakes and Streams. Studies Series No. 238. Ministry of Conservation, Victoria.

Bird E.C.F. (1978). The Geomorphology of the Gippsland Lakes Region. Environmental Studies Series, Publication No. 186. Ministry for Conservation, Victoria.

Bird E.C.F. (1965). A Geomorphological Study of the Gippsland Lakes. Department of Geography Publication G/1, Australian National University, Canberra.

Birkin, E.J. (1991). An Overview Report on the Wetlands of the Central Gippsland Region. Dept of Conservation and Environment, Central Gippsland Region, Victoria.

Black, K.P., (1989). Hydrodynamics and salt transport in an estuarine channel. Part 3, Numerical model simulations. Technical Report No. 10. Victorian Institute of Marine Science.

Black, K.P., and Hatton, D.N., (1989). Hydrodynamics and salt transport in an estuarine channel. part 2, Field data analysis. Technical Report No. 9. Victorian Institute of Marine Science.

Blindow, I., G. Andersson, A. Hargeby and S. Johansson (1993). Long term pattern of alternative stable states in two shallow eutrophic lakes. Freshwater Biology 30: 159-167.

Brown, R., (1993). Marine water quality monitoring 1991 yearly report, Publication No. 386. Environment Protection Authority, Victoria.

Chessman, B.C., (1990). Weather, water quality and algal blooms in the Gippsland Lakes. Paper presented at Algal Blooms Seminar 1988.

Chessman, B.C., and Marwood, P.L., (1988). Estimates of loads of non-filterable residue in streams of the Lake Wellington catchment. EPA SRS 88/004. Environment Protection Authority, Victoria.

Clucas, R.D., and Ladiges, P.Y., (1980). Die-back of Phragmites australis (Common Reed) and increased salinity in the Gippsland Lakes. Environmental studies series No. 292. Ministry of Conservation, Victoria.

Cullen, P and E.M. O'Loughlin (1982) Non point sources of pollution. p.437-453 in "Prediction in water quality" eds E.M. O'Loughlin and P.Cullen, Australian Academy of Science, Canberra.

Davis, R.A, Delporte, B.C., and Marsden, M.A.R., (1977). Morphology, surface facies and sediment distribution, Gippsland Lakes, Victoria. Report to Gippsland Regional

Environmental Study, by University of Melbourne Geology Department for Ministry of Conservation.

Department of Conservation and Environment (DCE) (1991). Gippsland Lakes Management Plan.

Ducker, S.C., Brown, V.B., and Calder, D.M., (1977). An identification of the aquatic vegetation in the Gippsland Lakes. Prepared by University of Melbourne, School of Botany for Ministry of Conservation.

EPA (1974). A report on the Gippsland Lakes algal bloom - Autumn 1974. Water Quality Branch, Environment Protection Authority, Victoria.

EPA (1983). Water quality of Gippsland lakes inputs. Publication No. 161. Environment Protection Authority, Victoria.

EPA (1986). Gippsland Lakes and Rivers. Water quality series data No. WQ7/86. Environment Protection Authority, Victoria.

EPA (1990). Gippsland Lakes 1988 yearly report, (water quality data). WQD 90/014. Environment Protection Authority, Victoria.

EPA (1993). Marine water quality monitoring network 1990 data summary. Publication No. 374. Environment Protection Authority, Victoria.

Erskine, W.D., Rutherfurd, I.D., and Tilleard J.W., (1990). Fluvial Geomorphology of Tributaries to the Gippsland Lakes. Prepared by Ian Drummond and Associates P/L for Dept of Conservation and Environment.

Gippsland Lakes Algal Bloom Seminar (1988). Discussion papers from the seminar.

Glover, J.W. (1981). Gippsland regional environmental study: Heavy metals budget - Latrobe River catchment. Report No. ESS 326. Ministry of Conservation, Victoria, Environmental Studies Division Report, 54 pages.

Glover, J.W., Bacher, G.J., and Pearce, T.S. (1980). Gippsland regional environmental study: Heavy metals in biota and sediments of the Gippsland Lakes. Report No. ESS 279. Ministry for Conservation, Victoria, Environmental Studies Division Report, 97 pages.

Graham, W.A.E., Herbert R.K., and Franklin K.F., (1978). Input streams assessment. Gippsland Environmental Study, Special Projects Group, Water and Materials Science Division, State Rivers and Water Supply Commission, Victoria

Grayson, R., Gippel, C.J., Finlayson, B.L., Hart, B.T., Hawken R., McMahon, T.A., and Tilleard J., (1994). Mitigation of sediment and nutrient movement from sources in the Latrobe River catchment into the Gippsland lakes.

Gunthorpe, L., (Ed), (1997). Gippsland Lakes Fish Habitats. Report No. 16. Fish habitat assessment group, Department of Natural Resources and Environment.

Gutteridge Haskins & Davey P/L (GHD), (1991). Water Supply Options for the Lower Latrobe River Wetlands.

Harris, G.P. (1996). Catchments and aquatic ecosystems: nutrient ratios, flow regulation and ecosystem impacts in rivers like the Hawkesbury-Nepean. Discussion paper, Co-operative Research Centre for Freshwater Ecology, University of Canberra, ACT, 57p.

Harris, G.P. and Baxter, G. (1996). Interannual variability in phytoplankton biomass and species composition in a subtropical reservoir. Freshwater Biol. 35: 545-560

Harris, G., Batley, G., Fox, D., Hall, D., Jernakoff, P., Molloy, R., Murray, A., Newell, B., Parslow, J., Skyring, G., and Walker, S., (1996). The Port Phillip Bay Environmental Study: final report, CSIRO, ACT 239p.

Harris, G.P., Davies, P., Nunez, M. and Meyers, G. (1988). Interannual variability in climate and fisheries in Tasmania. Nature (Lond) 333: 754-757

Hatton, D.N., Black, K.P., Longmore, A.R. and Colman, R.S., (1989). Hydrodynamics and salt transport in an estuarine channel: Part 1. Field data collection program. Working Paper No. 15. Victorian Institute of Marine Science.

Hinga, K R, Jeon, H. and Lewis, N F (1995) Marine eutrophication review. Pt. 1: Quantifying the effects of nitrogen enrichment on phytoplankton in coastal ecosystems. NOAA Coastal Ocean Program, Decision Analysis Series No. 4, NOAA Coastal Ocean Service, Silver Springs, Md USA.

Lake Wellington catchment salinity management plan community working group (Lake Wellington CSMPCWG), 1993. Dropping the Watertable: the Draft Lake Wellington Catchment Salinity Management Plan.

Latrobe Valley Water and Sewerage Board (LVW&SB), (1978). Gippsland Regional Environmental Study: Gippsland Lakes Water Quality Assessment Project, Final Report. Prepared for Ministry for Conservation.

Lester, C.J., (1983). Environmental water management : modelling long-term circulation and pollutant dispersion patterns in the Gippsland Lakes estuary. University of Melbourne, masters degree, research project.

Longmore, A.R., (1994a). Nutrient and chlorophyll concentrations in the Gippsland Lakes 1988-89. EPA Publication No.419. Environment Protection Authority, Victoria.

Longmore, A.R., (1994b). Nutrient release rates for sediment cores from the Gippsland Lakes 1988. Publication No. 420. Environment Protection Authority, Victoria.

Longmore, A.R. (1989a). Saltwater Intrusion into Lake Wellington, Gippsland Lakes, Southeastern Australia: Its Cause, Effects and Control. SRS 89/006. Environment Protection Authority, Victoria.

Longmore, A.R., (1989b). Nutrient Budget for northern Lake King, Gippsland Lakes, 1987-88. SRS 89/004. Environment Protection Authority, Victoria.

Longmore, A.R., (1988). Gippsland Lakes water quality monitoring, 1987-88: MSL Internal Rep. No. 170. Marine Science Laboratories, Queenscliff.

Longmore, A.R., Gibbs, C.F., and Marchant, J.W., (1988). Water quality in the Gippsland Lakes, July 1984- June 1985 : spatial and temporal trends. MSL Tech. Rep. No. 60. Marine Science Laboratories, Queenscliff.

Longmore, A.R., Norman, L., and Strong, J., (1990). Oxygen concentrations, temperatures and salinities in the Gippsland Lakes, Victoria, 1987-88 and survival of black bream, Acanthopagrus butcheri. EPA SRS 89/005. Environment Protection Authority, Victoria.

MacDonald, M (Ed), (1997). Gippsland Lakes fin fisheries and black bream stocks. DNRE Report No. 4. Bay and fisheries and stock assessment group, Fisheries Victoria. Dept of Natural Resources and Environment.

Marwood P., (1989). Impact of Stream Inflow on Lake Wellington Salinity. EPA unpublished report. Environment Protection Authority, Victoria.

Mobley, M.C., Hyduke, E.P., and Collet, L.C., (1983). A regression model of historical salinity data from the Gippsland Lakes. MSL Technical Report No. 28. Marine Science Laboratories, Queenscliff.

Monbet, Y., (1992) 'Control of phytoplankton biomass is estuaries: a comparison of microtidal and macrotidal estuaries', Estuaries 15, 563–71

Newell, B.S., (1991). Hydrodynamics of the Latrobe River Estuary. Department of Conservation and Environment.

Pooley G., and Helman Z., (1986). Review of the impacts of water abstraction from the Thomson River on the Thomson River and the Gippsland Lakes. Gippsland Lakes Assessment Working Party.

Pooley G.J., (1978). The origin and fate of nutrients from natural sources in the Latrobe Valley - Victoria, final report. Gippsland Environmental studies program, report No. 210. Ministry for Conservation.

Poore G.B.C., (1982). Benthic communities of the Gippsland Lakes, Victoria. Museum of Victoria. Journal paper, Aust. J. Mar. Freshwat. Res. 33, 901-915.

Rigby B.A., (1982). An ecological study of the estuarine fish assemblage in the Gippsland Lakes. MSL Internal Report No. 3. Marine Science Laboratories, Queenscliff.

Roberts, J., Ebner, B., and Sainty, G., (1997). Carp in Australia. In: Banens RJ, Lehane R, editors. Proceedings, 1995 Riverine Environment Research Forum, Attwood, Victoria, 4-6 October. Canberra: Murray-Darling Basin Commission, 1997: 47-8.

Roberts, J., and Tilzey, R., (editors), (1997). Controlling Carp – exploring the options for Australia. Proceedings of a workshop 22-24 October 1996, Albury. Published by CSIRO Land and Water, Griffith.

Robinson D.P., (1995). Water quality and nutrient status of Lake Wellington. EPA Publication No. 460. Environment Protection Authority, Victoria.

Robinson D.P., Rooney G.R., and Jirik S., (1994). Temporal trends analysis of data from the water quality monitoring network, 1984-89. EPA Publication No. 430. Environment Protection Authority, Victoria.

Roob, R and Ball, D. (1997) Victorian marine habitat database: seagrass, Gippsland Lakes. A report for Fisheries Victoria, Department of Natural Resources and Environment. Marine and Freshwater Research Institute, Victoria.

Scanes, P., Coade, G., Large, D. and Roach, T., (1998). Developing criteria for acceptable loads of nutrients from catchments. In proceedings from Coastal Nutrients Seminar, October 1997, pp89-99. Published by AWWA, NSW.

State of the Environment Advisory Council, (SOE), (1996). Australia State of the Environment 1996. Department of the Environment, Sport and Territories. CSIRO Publishing, Collingwood, Australia.

Walker, S, Sporcic, M and Coutin P (1998) Development of and environment-recruitment model for black bream: a case study for estuarine fisheries management. Final report FRDC project 96/102, Marine and Freshwater Resources Institute, Victoria

Young L and Sjerp E (compiled and edited), (1995). Gippsland Lakes conference. Papers and proceedings from the conference held at Lakes Entrance April 1995. Gippsland Lakes Management Council.