Wetlands for the Future

Editors

A.J. McComb and J.A. Davis

Gleneagles Publishing
Adelaide

Assessing the ecological health of estuaries in southwest Australia

D. M. DEELEY and E. I. PALING

Biological and Environmental Sciences, Murdoch University, Perth, Western Australia 6150

Abstract

Southwestern Australia experiences a Mediterranean climate and its estuaries are at risk from eutrophication because of a limited oceanic tidal range, constricted or barred ocean entrances, shallow estuarine basins, extensive agricultural clearing and draining in catchments, fertilized sandy coastal plain soils with limited phosphorus sorption capacity, large inland catchments with little vegetative cover at the onset of winter rains and widespread erosion. Elevated nutrient and sediment loads to southwest estuaries have caused disruption of ecosystem processes in some estuaries including seagrass loss, macroalgal and phytoplankton nuisance, and anoxia. To investigate estuarine health in the southwest, five sites in each of eight estuaries and five sites at a marine reference location (45 sites), were investigated using a range of physical, chemical and biological indicators. Water and sediment nutrient concentrations were determined at each site, together with an assessment of the community structure of phytoplankton, and benthic macro-invertebrates. Historical water quality information was evaluated to provide perspective. Despite some past analytical uncertainty, there appears to have been a significant increase in median total phosphorus concentrations in both the upper Swan (12 to 90 $\mu g\ L^{-1}$), and Peel-Harvey estuaries (15 to 70 $\mu g\ L^{-1}$), from 1945 to 1995. Surface salinity transects in summer 1995 typified classical, lagoonal and reverse estuaries, depending on basin morphology, marine contact and runoff volumes. Summer N, P and chlorophyll *a* concentrations in estuarine waters were correlated and were consistent with the extent of agricultural development particularly for sandy coastal catchments. Phytoplankton species diversity (evenness) was inversely related to total cell densities and estuarine nutrient status. Macro-invertebrate diversity (evenness) was inversely related to abundance. The proportion of cyanophyte and dinoflagellate cells in phytoplankton communities was high for some diverse pristine sites with low levels of chlorophyll *a*, low for all sites with moderate levels of chlorophyll a, and high for some sites with high levels of chlorophyll a. The suite of environmental indicators evaluated here collectively provided a useful indicator of estuarine health.

Introduction

Many estuaries in southwestern Australia have been experiencing symptoms of eutrophication including increased frequency, intensity and duration of phytoplankton blooms (Kinhill, 1988; Hosja and Deeley, 1993; Thompson and Hosja, 1996), nuisance

accumulations of macroalgae (Lukatelich *et al.*, 1987; Lavery *et al.*, 1991; McComb and Humphries, 1992; Hodgkin and Hamilton, 1993), a loss of seagrasses (D'Adamo *et al.*, 1992; Walker and McComb, 1992), and benthic hypoxia and anoxia (Douglas *et al.*, 1996).

Few studies have estimated the mass flux of nutrients to southwest estuaries (Rochford, 1951; Kinhill, 1988; Bott, 1993), and consequently it has been difficult to draw conclusions about the onset and progression of eutrophication.

The Mediterranean climate experienced by southwestern Australia, results in highly seasonal streamflow with considerable inter-annual variability. The coefficient of variability (Cv) of annual runoff for the southwest (Cv > 0.5), is greater than that of European (Cv = 0.28) and North American (Cv = 0.36) rivers (Finlayson and McMahon, 1988; Eyre, 1997), and the variability increases as annual rainfall decreases.

Time delays of years between increased nutrient inputs and the appearance of symptoms of eutrophication (Hodgkin and Hamilton, 1993), together with seasonal variability in runoff inputs and primary production in estuaries, makes it difficult to establish causal links.

The difficulty of defining estuarine response to historical nutrient loadings and in estimating permissible loadings, highlights the need for diagnostic indicators of estuarine health. Environmental indicators of ecosystem disfunction need to be consistent with the common symptoms of degradation (Roux *et al.*, 1993), and to differentiate between the presence, early onset or absence of stress-related impacts (Cairns Jr *et al.*, 1993).

Characteristics ascribed to ecosystems under stress may include; increased circulation of contaminants (Cairns Jr *et al.*, 1993), reduced species richness and/or diversity (Rapport, 1995a), reductions in the biotic size spectrum to favour smaller life forms (Rapport, 1995b), simplification of food webs (Havens, 1994), increased primary productivity (Amir and Hyman, 1993), increased dominance by undesirable exotic species (Havens *et al.*, 1996), increased prevalence of disease (Cairns Jr *et al.*, 1993), or morphological aberrations (Clarke, 1994; Tracy and Hough, 1995), and reduced population stability (Keough and Quinn, 1991; Fairweather, 1993; Rapport, 1995b).

This paper reports on the first year of range-finding investigations into the health of estuaries in southwestern Australia. Environmental indicators investigated included; (i) time series of TP concentrations to quantify changes in the circulation of contaminants, (ii) relationships between nutrients and chlorophyll *a* to assess the level of primary productivity, (iii) assessment of species richness and diversity of key trophic groups in impacted and pristine locations, and (iv) screening for the presence of potentially harmful phytoplankton.

Materials and methods

To investigate estuarine health in the southwest, eight estuaries, and a marine reference location were investigated (Table 1) using a range of physical, chemical and biological indicators.

Estuaries were chosen (Figure 1) from a region extending over 280,000 km^2 with a range of annual rainfall from Broke Inlet exceeding 1,300 mm and areas of the Avon catchment below 400 mm. Average annual evaporation ranged from 1,000 mm at Augusta at the mouth of Hardy Inlet, to over 2,500 mm in areas of the Avon catchment.

Catchment clearing percentage ranged from 4% at Broke Inlet to more than 85% for the large Swan Avon catchment. Estuarine types ranged from barrier estuaries with engineered permanent openings on the west coast, to poorly flushed inlets and coastal barrier lagoons on the south coast which close to the ocean much of the time.

Sampling strategy

Five sites were selected in each estuary (40 sites), from near the mouth, moving upstream toward the upper reaches of the tidal influence. Sites generally followed the mid-line of the estuary and were selected to have broadly consistent depths. In several instances, sites were placed off the mid-line to avoid busy navigation channels or deep holes.

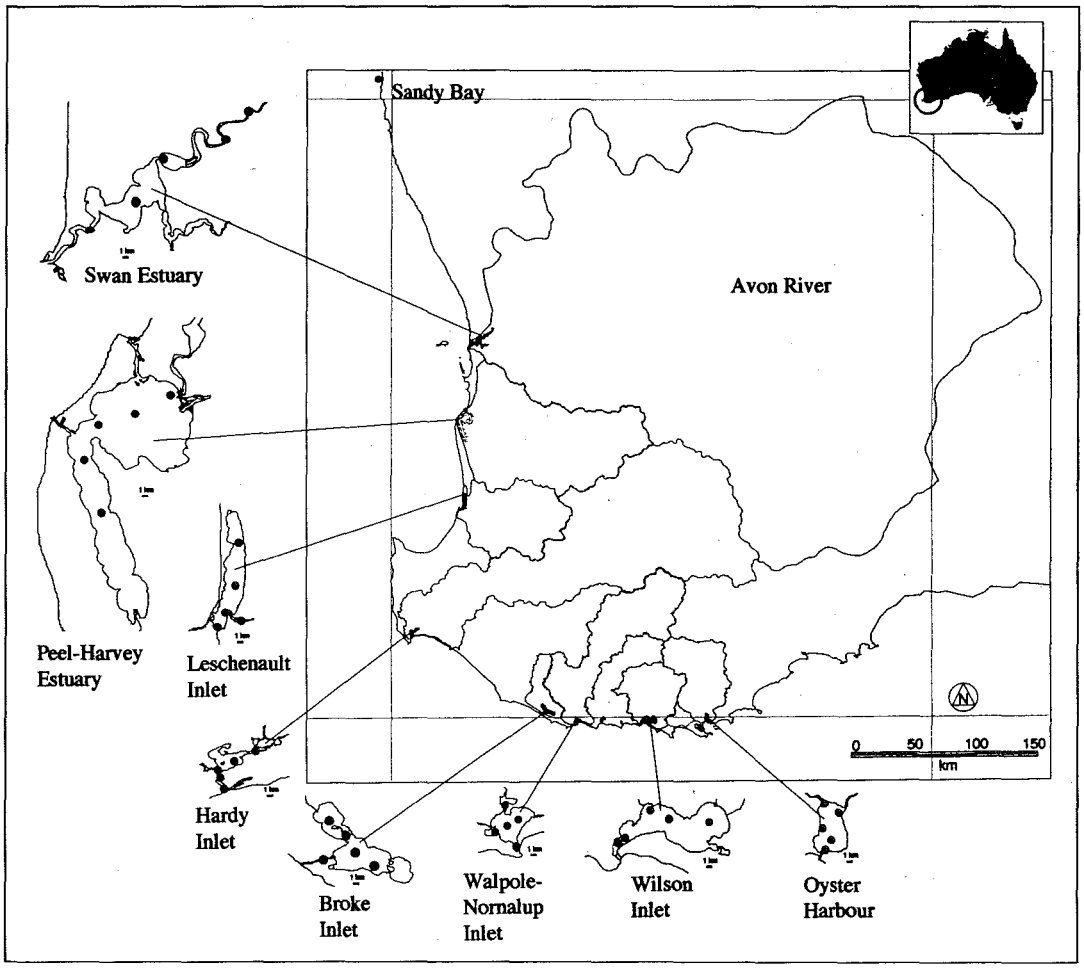


Figure 1. Map showing each of the estuaries where 5 sampling sites were established (n=45).

Five additional sites were selected at the marine reference location, Sandy Bay, which were within a shallow coastal lagoon of <5 m protected from the prevailing swell by an outer barrier reef. All 45 sites for this investigation had fine textured sediments, except for Sandy Bay where soft sediments lay in depressions in the exposed reef pavement.

Marine charts and line drawings showing depth contours (Hodgkin and Clark, 1988; Hodgkin and Clark, 1989), were digitized and the GIS package MAPINFO used to provide catchment areas and estimates of estuarine volume and mean depth. Raw flow data were provided by the State Water and Rivers Commission (WRC). Gap filled data were available from the WRC from Busselton to Walpole. Normal hydrographic procedures (Wanielista *et al.*, 1997), were used to fill gaps in the daily flow record from 1985 to 1995 for rivers draining catchments of the Swan, Peel-Harvey, Leschenault, Wilson, Oyster Harbour and Stokes Inlets and to compute annual totals (Table 1).

Water quality, sediment quality and biota were sampled using standard techniques (Baker and Wolfe, 1987; APHA, 1992; Kramer *et al.*, 1994).

Table 1. Description of each location where 5 sampling sites were established (n=45).

Location	Area open water (km^2)	Mean depth (m)	Mean annual runoff (Mm^3)	Runoff C.v.[a]	Catchment area (km^2)	% cleared
Sandy Bay[b]	–	2.5	–	–	–	–
Swan Estuary[c]	37.2	4.3	624	0.71	122,960	85
Peel-Harvey Estuary[c]	134.6	1.0	557	0.45	11,434	75
Leschenault Inlet[c]	27.7	0.7	495	0.57	4,824	48
Hardy Inlet	12.7	1.1	749	0.70	22,702	75
Broke Inlet[d]	43.6	2.0	166	0.43	918	4
Walpole-Nornalup Inlet	14.8	2.2	373	0.45	4,208	50
Wilson Inlet[d]	50.4	2.2	175	0.47	2,309	60
Oyster Harbour	15.9	1.7	65	0.63	2,991	72

[a]C.v. = coefficient of variation is the standard deviation expressed as a fraction of the mean (Muirden, 1995).
[b]Sandy Bay is a nearshore marine reference location.
[c]Significant volumes of freshwater have been diverted for potable or irrigation supplies.
[d]Seasonally closed estuaries.

Physical and chemical sampling. Salinity (conductivity), temperature and dissolved oxygen were monitored at 0.5 m intervals using a calibrated YEOCAL 601 probe.

Samples analysed for TP from 1974 to the present (Henderson *et al.*, 1983), were digested using excess $HCL0_4$ or persulphate digests followed by colourimetry (Murphy and Riley, 1962). Surface samples from 1945 to 1954 were analyzed for TP using limited $MgNO_3$ which may have extracted only 50% of the particulate bound P (Jack, 1977). This analytical uncertainty precluded compensatory adjustment.

Surface and bottom waters were sampled at 45 sites in 1995 for bio-available P, BAP (Oliver and Douglas, 1994; Robinson *et al.*, 1994), TKN and TP using whirlpaks (NASCO Pty Ltd). Samples were frozen for transport to laboratories. Known volumes of samples were filtered (0.45μm) and filters stored in Alfoil and frozen prior to determination of chlorophyll a and other pigments by acetone extraction and colourimetry (APHA, 1992). Filtered samples were analyzed for oxidised N by Cu-Cd reduction followed by colorimetry (Technicon method 158-71W), and for NH_3-N by forming indophenol blue followed by colorimetry. Dissolved inorganic N, (DIN) which was expressed as the sum of dissolved forms (NH_3-N + NO_2-N + NO_3-N). Total N was expressed as the sum of TKN and oxidized N (NO_3-N + NO_2-N). Analytical precision (2s), was better than ±10% for each of TP, BAP, NO_3-N, NH_3-N and TKN.

Nine replicate samples of sediment from each site were cored (50 mm dia) sectioned (0–2 cm, 2–10 cm) thoroughly mixed and sub-sampled. Particle size distribution was determined following wet sieving. Organic matter content was measured as the percentage loss on ignition (LOI) at 550°C.

Biological sampling. Phytoplankton were collected using integrated trawls from the bottom to the surface with a 20 μm net. Samples were immediately preserved with Lugol's solution. Standard microscope techniques (APHA, 1992), were used to enumerate phytoplankton. This included appropriate dilutions and counting standard numbers of graticule grids. More than 250 species of phytoplankton were identified by experienced taxonomists.

Benthic macro-invertertebrates were sampled by a diver using SCUBA, and bulking sediment from 4 replicate (150 cm dia x 100 cm), cores. Animals were collected after wet sieving, and sorting fresh material on the day of sample collection. Macro-invertebrates were preserved in 2% formalin for storage, prior to identification using standard taxonomic keys.

Data analysis

Chemical and biological data were tested using the Kolmogorov test for normality and skewed data transformed using the ladder of powers prior to statistical treatment.

Measures of biological community structure were derived including, species richness, Shannon-Weiner diversity, and Pielou's evenness.

$$\text{Shannon-Weiner diversity} \quad H' = -\Sigma p_1 \log p_1 \qquad ...1$$

where p_1 is the number of individuals of the ith species (Shannon and Weaver, 1949)

$$\text{Pielous evenness} \quad J' = \frac{H'}{\log s} \qquad ...2$$

where s is the number of species in the sample (Pielou, 1975).

Correlations between physico-chemical variables and biological community structure were derived using the BIOENV module of the PRIMER software package. The method

was based on the construction of a dissimilarity matrix using the principal components analysis (PCA) of physical chemical variables, and by obtaining a similarity matrix from non-metric multi-dimensional scaling (MDS) of the biological variables, followed by simple linear correlation between the two (dis) similarity matrices (Clarke and Ainsworth, 1993).

Transformed (log(n+1)) phytoplankton data were correlated to surface salinity, log transformed DIN and log BAP. Transformed (4th root), benthic macro-invertebrate community data were correlated to bottom salinity and dissolved oxygen saturation, percent of sediments < 250 μm, and % LOI 550°C.

Results

Total phosphorus (TP), concentrations in the upper reaches of the Swan estuary at Sandringham are summarized in Figure 2. Median concentrations around 12 μg L^{-1} from 1945 to 1955 increased to above 100 μg L^{-1} from 1975 to 1985 and decreased in recent years to around 30 μg L^{-1}. A LOWESS smooth (Cleveland, 1985), fitted to the log transformed series (Figure 2), shows the non–linear trend over time.

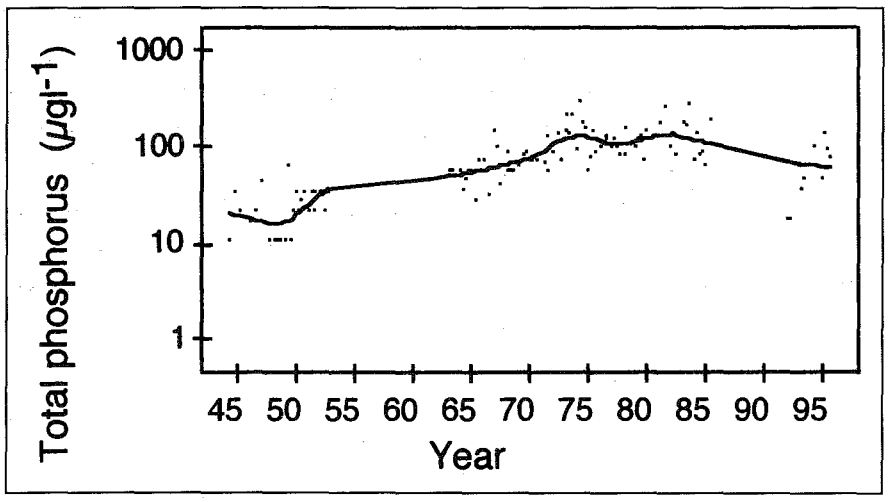


Figure 2. Surface total phosphorus concentrations (μg L^{-1}) sampled from the Swan River at Sandringham from 1945 to 1995 with Lowess smooth. (Note: log scale, gaps in series where no monitoring occurred).

A similar pattern of TP concentrations has been observed from the centre of Peel Inlet (Figure 3). Median TP concentrations of 15 μg L^{-1} from 1945 to 1955, increased to over 80 μg L^{-1} from 1982 to 1992. There appears to have been a slight reduction in median TP concentrations in the last five years.

There appears to have been a real increase in the concentration of TP in both the Swan and Peel-Harvey estuaries since 1945. Gaps in the data record and inconsistent analytical techniques add uncertainty. Adjusting data upwards for the period 1945–55, when particulate P fractions may have been underestimated by up 50% would not greatly

influence the magnitude of the observed trend. Peak concentrations exceeded 500 μg L^{-1} for both estuaries in the 1980's.

Typical summer salinities for estuarine categories are presented in Figure 4. The Swan Estuary, Hardy Inlet and Oyster Harbour showed the normal pattern for estuaries where salinity decreased upstream through residual runoff inputs from large inland rivers (Avon, Blackwood and Kalgan Rivers).

The three lagoonal estuaries were well mixed vertically and horizontally, because of wind-driven circulation patterns. Walpole-Nornalup Inlet has a narrow natural opening and sand bars at the mouth of Wilson and Broke Inlets were closed at the time of observation. The surface salinity in Peel-Harvey and Leschenault inlets showed a reverse pattern with increasing salinity upstream. These two estuaries had limited exchange in their upper reaches, and evapo-concentration produced salinities above marine levels. One site at Leschenault Inlet was in the mouth of a river entering the basin from the side, and had a lower salinity.

There was a non-linear relationship between TN and log chlorophyll a (Figure 5), and TP and log chlorophyll a (Figure 6). Pristine sites at Sandy Bay and Broke Inlet had very low levels of nutrients and chlorophyll a. Highly impacted sites in the Swan and Peel-Harvey estuaries had very high nutrient and chlorophyll a concentrations.

Species richness of phytoplankton and benthic macro-invertebrate communities were averaged for the five sites in each estuary (Figure 7). Patterns of species richness for the two trophic groups were broadly consistent in most estuaries except the coloured south coast estuaries (Broke, Walpole-Nornalup and Wilson Inlet). For phytoplankton, the lowest species richness was in the Swan, Peel-Harvey and Walpole-Nornalup Inlets and in Oyster Harbour. For benthic macro-invertebrates, higher values of species richness were observed for pristine sites at Sandy Bay, and in Leschenault Inlet. Low species

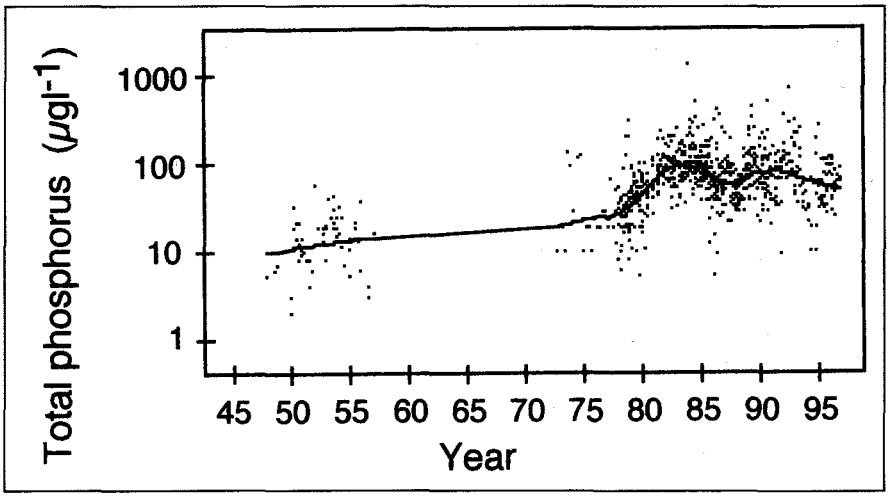


Figure 3. Surface total phosphorus concentrations (μg L^{-1}) sampled from the centre of Peel Inlet from 1945 to 1995 with Lowess smooth. (Note: log scale, gaps in series where no monitoring occurred)

richness for benthic macro-invertebrates was observed for the Peel-Harvey system. It may be expected that when averaging data for a transect of sites along an environmental gradient, such as observed in classical and reverse estuaries (Figure 4), variability between sites may be greater than when combining data from spatially more homogeneous lagoonal estuaries. Standard deviation values (error bars in Figure 7) showed no relationship between site variability and estuarine morphology (Figure 4).

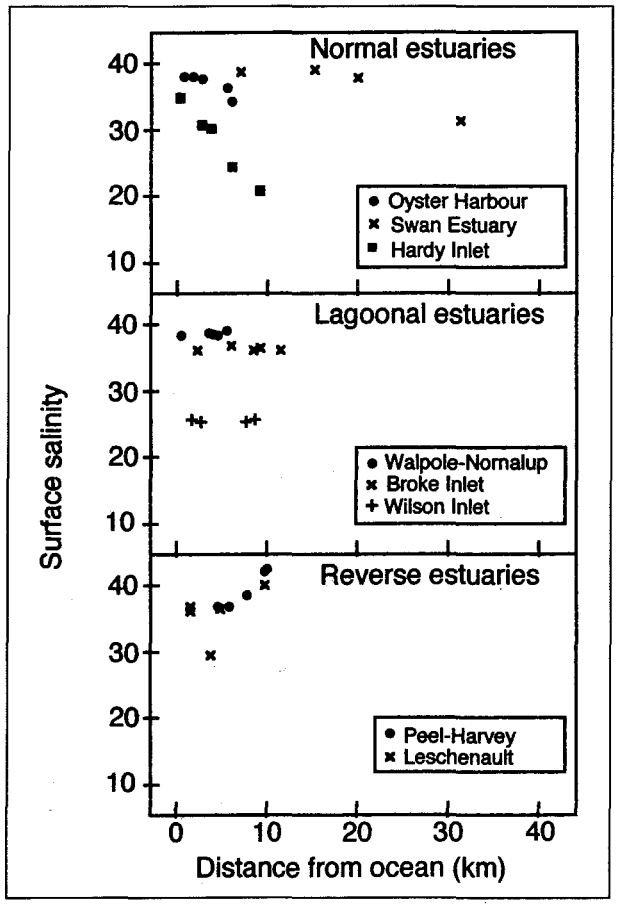


Figure 4. Surface salinity and distance from the ocean for a snap-shot taken in each estuary from January and April 1995 when river flows were minimal or had ceased.

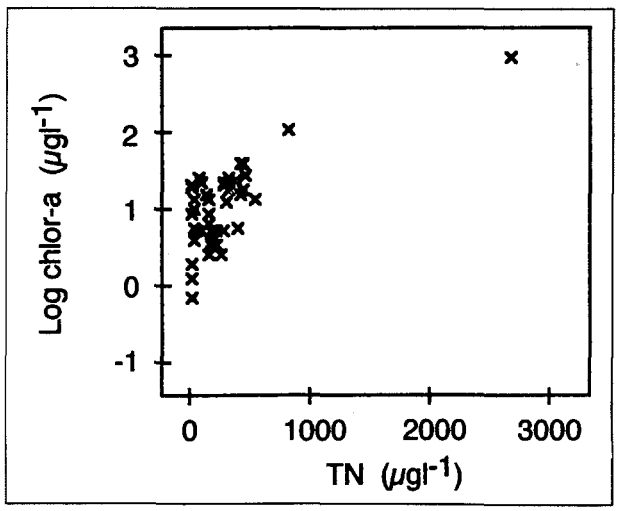


Figure 5. Relationship between log surface chlorophyll a (μgL^{-1}) and total nitrogen (TN) for a sample collected from each site from January and April 1995.

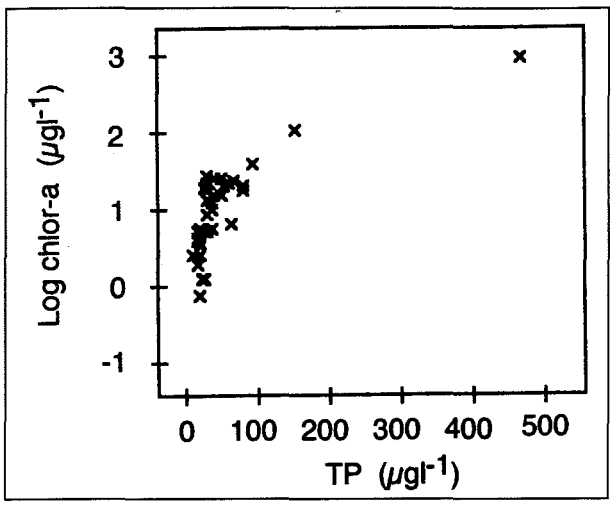
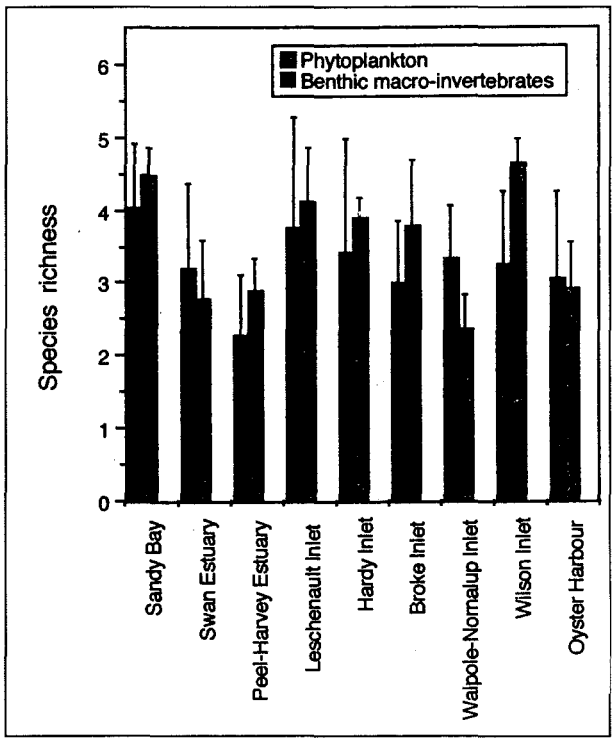


Figure 6. Relationship between log surface chlorophyll a (μgL^{-1}) and total phosphorus (TP), for a sample collected from each site from January and April 1995.

Figure 7. Average species richness for phytoplankton and benthic macro-invertebrates observed for each estuary over summer 1995. Error bars are standard errors.



Diversity (evenness) of phytoplankton and benthic macro-invertebrates plotted against normalized abundance (Figure 8), showed decreasing evenness with increasing total individuals. The highest evenness was for pristine sites at Sandy Bay and in Broke Inlet, and lower evenness was observed in estuaries with extensive agricultural development and cleared coastal sands within their catchments.

Correlations between a similarity matrix for MDS of phytoplankton community data and a dissimilarity matrix from PCA of chemical data for 45 estuarine sites are summarized in Table 2. Chemical variables thought to influence phytoplankton

distribution included surface salinity, log DIN and log BAP. Salinity was the most highly correlated (r = 0.48), of the chemical variables, and DIN (r = –0.03), and BAP (r = –0.05), were poorly correlated.

Table 2. Correlation between phytoplankton community structure and chemical data (Clarke and Ainsworth, 1993).

Variable	r^2
Surface salinity	0.48
Log inorganic N	–0.03
Log BAP	–0.05
Surface salinity & log inorganic N	0.41
Surface salinity & log BAP	0.41
Log BAP & log inorganic N	–0.04
Surface salinity, log BAP & log inorganic N	0.39

a Correlation coefficient harmonic weighted Spearman.

Correlations between benthic macro-invertebrate data and chemical and sediment characteristics thought to influence the benthos were poor (r = <0.15), and are not presented. Physico-chemical data tested included bottom salinity, bottom O_2 saturation, percentage of sediment < 250 μm, LOI 550°C. The best correlation was with bottom O_2 saturation (r = 0.14).

The percentage of cells of potentially harmful cyanophytes, and dinoflagellates, showed a bimodal distribution when plotted against log chlorophyll a (Figure 9). The proportion of cyanophyte and dinoflagellate cells in phytoplankton communities was high for some diverse pristine sites with low levels of chlorophyll *a*, and low for all sites with moderate levels of chlorophyll a. Sites with the highest levels of nutrients and chlorophyll a also had significant agricultural development in the catchment, with a high proportion of cleared sandy soils on coastal plain catchments.

Discussion

The symptoms of eutrophication may appear some time after the onset of unacceptable levels of nutrient input (Harris, 1994). There were large gaps in the TP series (Figures 2 and 3), when both estuaries appear to have undergone a transition in TP status. Without accompanying chlorophyll *a* or other biological data, it is difficult to relate TP concentrations to the onset of symptoms of eutrophication.

The data indicate that time series of TP concentrations can be used to assess changes in contaminant cycling in estuaries which has been identified as a key indicator of ecosystem health (Cairns Jr *et al.*, 1993). A less fragmented series would be required to statistically describe long term non-linear trends (Gilbert, 1987).

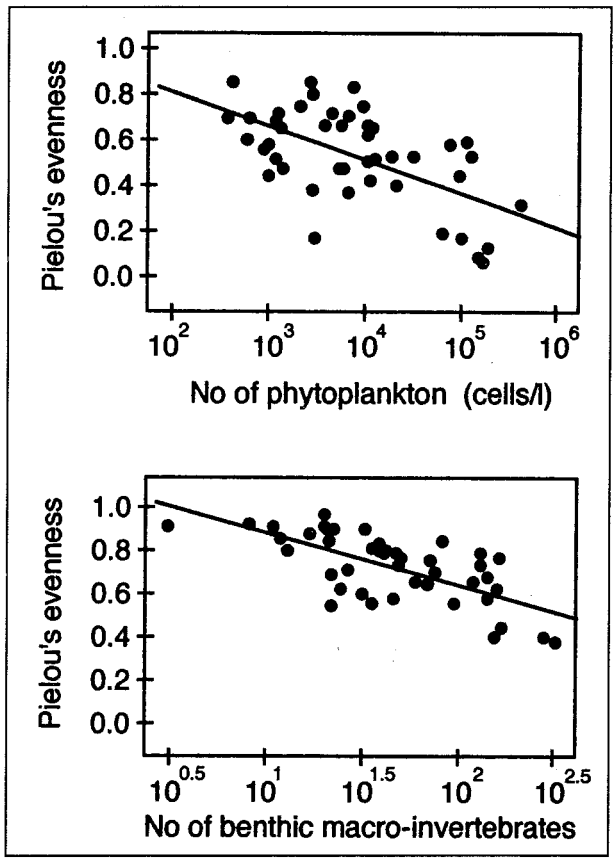


Figure 8. Pielous' evenness for phytoplankton expressed as a function of abundance (Σcells L^{-1}), and Pielous' evenness for benthic macro-invertebrates as a function of abundance for all 45 estuarine sites.

The salinity data (Figure 4), show that estuaries may be grouped on the basis of their basin morphology. Salinity differences in the three types of estuaries described may exert a controlling influence (Table 2), over the distribution of biota (Baker and Wolfe, 1987).

As expected, concentrations of TN, TP and chlorophyll a were highly correlated. During summer, a significant proportion of the total nutrients may occur as phytoplankton biomass (Thompson and Hosja, 1996), which may also control chlorophyll a concentrations. Unimpacted sites had the lowest TN, TP and chlorophyll a concentrations, with the converse for highly impacted sites.

These data suggest that a water quality snap-shot in summer may differentiate between pristine and highly enriched sites (Figures 5 and 6), but clearly an improved understanding of spatial and temporal heterogeneity of phytoplankton biomass would be required (Harris, 1994), before these indicators had appropriate diagnostic precision (Cairns Jr *et al.*, 1993), for tasks other than broadscale regional impressions.

Impacted sites in the Swan and Peel-Harvey estuaries had reduced species richness, and pristine marine sites at Sandy Bay had higher species richness (Figure 7). Patterns of species richness for the two trophic groups were consistent at the two extremes of disturbance, but less clear for the other estuaries. Clearly there is a range of factors including salinity which may influence the distribution of organisms (McComb and Humphries, 1992; Havens *et al.*, 1996).

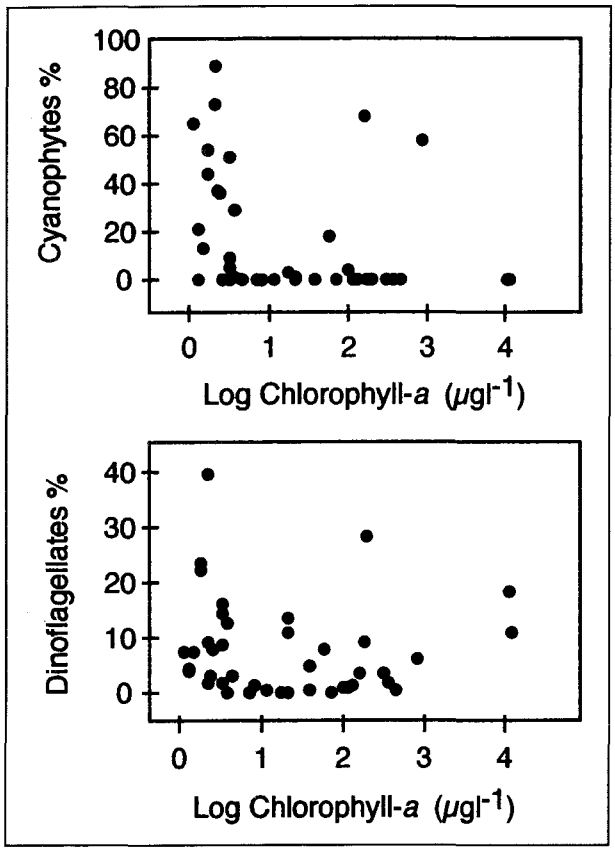


Figure 9. Proportion of cyanophyte (blue-green) and dino-flagellate (red tide) cells in phytoplankton communities for an integrated sample collected from each site between January and April 1995.

The relationship between evenness and abundance (Figure 8), was consistent with ecological theory that defined the evenness measure (Pielou, 1975). Highest evenness was observed for pristine sites and the lowest for highly impacted sites which had greater abundances of phytoplankton and benthic macro-invertebrates.

A data point for a particular site may not be fixed within the data clouds shown in Figure 8, but may cycle around (Bunn, 1995), under the influence of seasonal cycles and sampling bias resulting from limited monitoring of heterogeneous populations. A degrading estuary may require years of observation using this sort of relationship before longer term trends could be separated from short term noise.

Management of estuarine eutrophication may require trade-offs between diversity and abundance of organisms (Rapport, 1995a). Measures of evenness and abundance have a role in an estuarine indicator suite, but additional quantitative sampling would need to confirm early impressions resulting from these range finding investigations.

The proportion of cyanophyte and dinoflagellate cells in phytoplankton communities (Figure 9), may offer some diagnostic ability, but because of potential health implications, and the degraded state of estuaries experiencing blooms, this measure may have limited utility as an early warning indicator (Havens *et al.*, 1996).

Conclusions

There has been no consistent water quality monitoring program for rivers and estuaries in the southwest of Australia needed to identify trends in the health of estuaries over decades.

Historical TP data for Swan and Peel-Harvey estuaries suggest a long term increase in contaminant cycling. Relating contaminant concentration to biological response in estuaries would improve the diagnostic precision of this indicator (Rapport, 1995a).

There were differences in nutrient concentrations and chlorophyll *a* in estuaries, and differences in the evenness and abundance of biotic communities when comparing pristine and impacted sites. More than simply reaffirming ecological norms, these relationships may provide insights to possible longer term trajectories under greater or reduced nutrient loadings.

All indicators investigated were instructive, but additional quantitative sampling is required to better define their utility.

Acknowlegements

We gratefully acknowledge financial support by the National Landcare Program and the Water and Rivers Commission. Thanks to Professor Arthur McComb for his guidance and wisdom during the projects inception, to Wasele Hosja for undertaking phytoplankton identification and enumeration, to Phil Papas for undertaking benthic macroinvertebrate identification and enumeration, and to Brian Chester and staff of the Water and Rivers Commission, for provision of gap filled hydrographs from Busselton to Walpole, digitized catchment boundaries and historical estuarine water quality data.

We would especially like to thank our graduate assistant, Kevin Bancroft, for his tireless efforts in the field and with data processing. The Marine and Freshwater Research Laboratory provided prompt analysis of chemical samples, and Kirrily White digitized marine charts and line drawings of estuaries, and provided estimates of estuarine surface area and mean depth.

We also acknowledge the support of staff from the Swan River Trust, Peel-Inlet Management Authority, Leschenault Inlet Management Authority, Wilson Inlet Management Authority, and Albany Waterways Management Authority for the provision of vessels and other logistical support.

References

Amir, S., and Hyman, J., 1993. Measures of ecosystem health and integrity. Water Science & Technology 27: 481–488.

APHA, 1992. Standard methods for the examination of water and wastewater. 18th Ed. American Public Health Association, Washington DC.

Baker, J.M. and Wolfe, W.J. (Eds), 1987. Biological surveys of estuaries and coasts. Cambridge University Press, Cambridge.

Bott, G., 1993. Relationships between the extent of conventional broadscale agriculture and stream phosphorus concentration and phosphorus export in southwest Western Australia. Rep. No. 2. Office of Catchment Management.

Bunn, S.E., 1995. Biological monitoring of water quality in Australia: Workshop summary and future directions. Australian Journal of Ecology 20: 220–7.

Cairns Jr, J., McCormick, P.V. and Niederlehner, B.R., 1993. A proposed framework for developing indicators of ecosystem health [Review]. Hydrobiologia 263: 1–44.

Clarke, G.M., 1994. Developmental stability analysis: An early-warning system for biological monitoring of water quality. Australian Biologist 7: 94–104.

Clarke, K.R. and Ainsworth, M., 1993. A method of linking multivariate community structure to environmental variables. Marine Ecology Progress Series 92: 205–19.

Cleveland, W.S., 1985. Robust locally weighted regression and smoothing scatterplots. Journal of the American Statistics Association 74: 829–36.

D'Adamo, N., Simpson, C., Mills, D., Imberger, J. and McComb, A., 1992. The influence of stratification on the ecological response of two Western Australian embayments to nutrient enrichment. Science of the Total Environment, Supplement, 829–50.

Douglas, G.B., Hamilton, D.P., Gerritse, R.G. and Adeney, J.A., 1996. An investigation of water quality at two sites in the Swan River Estuary, 1993/4., Rep. No. 96–2. CSIRO.

Eyre, B.D., 1997. Transport, Retention and Transformation of material in Australian Estuaries. Estuaries submitted October 1996.

Fairweather, P.G., 1993. Links between ecology and ecophilosophy, ethics and the requirements of environmental management. Australian Journal of Ecology 18: 3–19.

Finlayson, B.L. and McMahon, T.A., 1988. Australia v the world: a comparative analysis of streamflow characteristics. In: Fluvial geomorphology of Australia., pp. 17–40. Academic Press, Australia.

Gilbert, R.O., 1987. Statistical methods for environmental pollution monitoring, Van Nostrand Reinhold, New York.

Harris, G.P., 1994. Nutrient loadings and algal blooms in Australian waters—a discussion paper, Rep. No. 12/94. LWRRDC.

Havens, K.E., 1994. Experimental perturbation of a freshwater plankton community: A test of hypotheses regarding the effects of stress. Oikos 69: 147–53.

Havens, K.E., Aumen, N.G., James, R.T. and Smith, V.H., 1996. Rapid ecological changes in a large subtropical lake undergoing cultural eutrophication. Ambio 25: 150–155.

Henderson, A., Humphries, R.B. and Hosja, W., 1983. Time series plots of Swan-Canning estuarine system physico-chemical dat, 1944–1989, Rep. No. 1. Waterways Commission of Western Australia.

Hodgkin, E.P. and Clark, R., 1988. An inventory of information on the estuaries and coastal lagoons of South Western Australia. Nornalup and Walpole Inlets, Rep. No. 2. Environmental Protection Authority.

Hodgkin, E.P. and Clark, R., 1989. An inventory of information on the estuaries and coastal lagoons of South Western Australia. Broke Inlet and other estuaries of the Shire of Manjimup., Rep. No. 6. Environmental Protection Authority.

Hodgkin, E.P. and Hamilton, B.H., 1993. Fertilizers and eutrophication in southwestern Australia: Setting the scene. Fertilizer Research 36: 95–103.

Hosja, W., and Deeley, D.M., 1993. Harmful phytoplankton surveillance in Western Australia., Rep. No. 43. Waterways Commission of Western Australia.

Jack, P.N., 1977. Seasonal variations in the water of the Swan River, Rep. No. 14. Government chemical laboratories.

Keough, M.J. and Quinn, G. P., 1991. Causality and the choice of measurements for detecting human impacts in marine environments. Australian Journal of Marine and Freshwater Research 42: 539–54.

Kinhill, 1988. Peel-Harvey Environmental Review and Management Program Stage II. Kinhill Engineers Pty Ltd.

Kramer, K.J.M. Brockman, U.H. and Warwick, R.M., 1994. Tidal estuaries: Manual of sampling and analytical procedures, Balkema, A.A., Rotterdam.

Lavery, P.S., Lukatelich, R. J. and McComb, A.J., 1991. Changes in the biomass and species composition of macroalgae in a eutrophic estuary. Estuarine, Coastal and Shelf Science 33: 1–22.

Lukatelich, R.J., Schofield, N.J., and McComb, A.J., 1987. Nutrient loading and macrophyte growth in Wilson Inlet, a bar-built southwestern Australian estuary. Estuarine, Coastal and Shelf Science 24: 141–65.

McComb, A.J. and Humphries, R., 1992. Loss of nutrients from catchments and the eco-logical impacts in the Peel-Harvey estuarine system, Western Australia. Estuaries 15:529–537.

Muirden, P.D., 1995. Surface water resources of the Busselton-Walpole region., Rep. No. WS 166. Water and Rivers Commission.

Murphy, J. and Riley, J. P., 1962. Anal. Chim. Acta. 27: 31–6.

Oliver, R. L., and Douglas, G. B., 1994. Measurement of bio-available phosphorus in natural waters using the Fe-strip adsorption method,. CSIRO.

Pielou, E.C., 1975. Ecological diversity, Wiley Interscience, New York.

Rapport, D.J., 1995a. Ecosystem health: An emerging integrative science. In: Evaluating and monitoring health of large scale ecosystems. Eds D.J. Rapport, C.L. Gaudet and P. Calow. Vol. I 28, pp. 5–31. Springer-Verlag, Berlin.

Rapport, D.J., 1995b. Ecosystem health: More than a metaphor. Environmental Values 4: 287–309.

Robinson, J.S. Sharpley, A.N. and Smith, S.J., 1994. Development of a method to determine bioavailable phosphorus loss in agricultural runoff. Agriculture Ecosystems & Environment 47: 287–97.

Rochford, D.J., 1951. Studies in Australian estuarine hydrology. I. Introductory and comparative features. Australian Journal of Marine and Freshwater Research 2: 1–116.

Roux, D.J., van Vliet, H.R. and van Veelen, M., 1993. Towards integrated water quality monitoring: Assessment of ecosystem health. Water SA 19: 275–80.

Shannon, C.E. and Weaver, W., 1949. The mathematical theory of communication, University of Illinois Press, Urbana.

Thompson, P.A. and Hosja, W., 1996. Nutrient limitation of phytoplankton in the Upper Swan River Estuary, Western Australia. Marine & Freshwater Research 47: 659–67.

Tracy, M. and Hough, R.A., 1995. Developmental instability as a biomonitor of environmental stress: An illustration using aquatic plants and macroalgae. In: Biomonitors and biomarkers as indicators of environmental change. Ed. F.M. Butterworth. pp. 313–37. Plenum Press, New York.

Walker, D. I., and McComb, A. J., 1992. Seagrass degradation in Australian coastal waters. Marine Pollution Bulletin 25: 191–195.

Wanielista, M., Kersten, R. and Eaglin, R., 1997. Hydrology: Water quantity and quality control. 2nd Ed. John Wiley and Sons, New York.