Freshwater Influences on Hydrology and Seagrass Dynamics of Intermittent Estuaries

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by

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Contents

Abstract

Many temperate estuaries have intermittently open and closed mouths, a feature that is often related to intermittent freshwater input. These systems, often overlooked due to their small size, can have large hydrological and physico-chemical variability over medium-term time scales. This variability presents potential difficulties for estuarine species, particularly where anthropogenic alterations to freshwater flows can cause large deviations from natural patterns of tidal influence and hence inundation regimes of shallow benthic substrates. Patterns of change in physico-chemical conditions, including inundation regimes, are particularly important for sessile benthic organisms. The foci of this study were the multiple causal links between freshwater inflow and the extent and decompositional processes of seagrass beds in these systems.

Influences of hydrological variability on physico-chemical conditions and seagrasses were examined in two central Victorian estuaries with anthropogenically-modified, but naturally-intermittent, freshwater flows and mouth openings. Comparisons focused on differences between the Anglesea River estuary with artificially-augmented freshwater inflow and the adjacent Painkalac Creek estuary, in which the volume and timing of inflows were altered by a reservoir. Eight other estuaries (those of Barham River, Skenes Creek, Kennett River, Wye River, St George River, Erskine River, Spring Creek and Thompsons Creek) were sampled to provide a regional context for the two main estuaries across climatic and topographic gradients.

Hydrological changes during the three-year field component were affected by the ending of a drought and then a major flood a year later, as well as by ongoing anthropogenic flow augmentation and reduction. These changes in hydrology were associated with an initially high seagrass coverage that was substantially reduced and showed signs of recovery only in the system that was affected by flow reduction. Such influences and responses also changed seasonally but to a much lesser extent than the responses to stochastic climatic events.

Natural freshwater flows in both systems were intermittent and varied substantially between years. Flooding flows represented up to 89% of the long-term annual average flow volume. Water quality in streams was broadly typical of the region, with the exception of low pH in some tributaries, especially those of Anglesea estuary. Anthropogenic changes to freshwater inflow were most evident at times of low natural flows and resulted in longer and more frequent periods of zero inflow to Painkalac estuary and a continual base flow to Anglesea. This base flow, from ponds containing coal ash, neutralised acidic waters flowing from upstream and increased conductivity, except at times of high natural flow.

A three-state conceptual model of the magnitude and variability of estuarine water levels, based largely on the degree of tidal influence was identified (with states being defined as either closed, perched or tidal). This model was quantitatively assessed and found to be valid for the two estuaries that were the main focus of the study. These states in turn had a large influence on the area and inundation of benthic habitat. Floods tended to open the mouths of estuaries, which then remained tidal given sufficient flow to overcome sedimentary processes at the mouths. Low and zero inflow was a precondition for closure of the mouths of the estuaries. When closed, differences in inflow resulted in different endpoints in salinity patterns. From an initial pattern similar to a classic ‘salt wedge’, Painkalac estuary, with reduced inflow, quickly destratified and gradually became more saline, at times hypersaline. Anglesea estuary, with augmented flow, tended to remain stratified for longer until becoming completely fresh, given a long enough period of closure.

Episodic changes in the water quality of the estuaries were associated with different components of the freshwater flow regimes. At high flows, fresh waters of low pH with a high metal load entered Anglesea estuary. Except during the largest flood, when the estuary was completely flushed, these waters were neutralised at the halocline, resulting in precipitation of metals. High flows into Painkalac were associated with elevated concentrations of clay-sourced suspended solids. During a closed period, with zero flow, a release of sediment-bound nutrients triggered by anoxia was observed in Painkalac, followed by an algal bloom.

The large decline in seagrass extent that was observed in both estuaries was closely related to floods and the subsequent reductions in potential habitat associated with the tidal states that followed. Analysis of historical patterns of extent against rainfall records suggested that periods of drought and extended mouth closures were related to establishment and expansion of beds. This model was similar to that described for South African estuaries and contrasted with more-seasonal patterns reported for local marine embayments.

Rates of *in situ* decomposition of seagrass detritus showed a mix of seasonal and disturbance-driven patterns of change, depending on the estuary. Variability of these rates at a scale of 10s of metres was typically not significant, but there were intermittent episodes with significant and spatially-consistent differences. A negative correlation between decomposition rate and seagrass extent was also observed. A novel technique for assessing cellulose decomposition potential in sediment, adapted from soil science, proved to be a useful tool for estuarine research. Results from this component of the study highlighted both small-scale variability that was inconsistent through time, and also stable differences in decomposition potential between depths and estuaries that were consistent with differences in hydrological state and salinity.

Given the relative lack of knowledge about processes in intermittent estuaries, particularly those relating to changes in freshwater inflow, results from this study will be of value both locally and for similar systems elsewhere. Locally, it is likely that flow to both Anglesea and Painkalac estuaries will be reduced, following closure of the mine power station at Anglesea and due to increased demand from the reservoir above Painkalac. There is potential to manage flows from each of these sources to minimise downstream effects. Regionally, and globally, there are many intermittent estuaries in areas with

Mediterranean-type climates. It has been predicted that the climates of these regions will become drier but with an increase in intensity of storm events, both of which have ramifications for flow regimes to estuaries. In the context of these likely changes, it is hoped that results of this study will contribute to more informed management of intermittent estuaries. Examples of such actions include management of flow and inundation regimes to avoid or minimise unnatural losses of seagrass or to maintain water quality at times of highest risk.

1. Introduction

1.1. Freshwater flow and estuaries

Inflow of fresh water is an inherent feature of estuaries, which, by any definition, are places in which fresh and marine waters mix. The volume, variability and chemical composition of freshwater flows are a primary influence on physical, chemical and biological environments within estuaries and are a major contributor to their extremely dynamic nature.

The geomorphology of estuaries is influenced by freshwater flow and is an important factor in the nature of flow-related effects in an estuary (Gillanders & Kingsford, 2002). Influences of fresh water can occur over a wide range of time scales from thousands of years (e.g. infilling of estuaries since the Holocene sea level rise) to days (e.g. scouring of channels during large floods). The size and shape of estuaries are determined by their underlying morphology, surrounding lithology and a combination of the dynamic forces of freshwater flow, tides and waves. These factors have been used in various combinations to describe and classify types of estuaries (e.g. Rochford, 1959; Bird, 1967; Ketchum, 1983; Digby *et al.*, 1999; Kench, 1999; Roy *et al.*, 2001).

Relative inputs of fresh versus marine water, combined with morphology, largely determine the vertical and horizontal salinity distribution within an estuary. This distribution is typically considered the most important chemical determinant for estuarine biota (Hodgkin, 1994). Other constituents of fresh water, such as suspended sediments, nutrients, carbon, acids and pollutants, interact with each other and with salinity, as well as physical and biological processes within estuaries to create a range of habitats through space and time.

Changes in the volume and timing of freshwater flow can affect the functioning of estuaries in different ways, through multiple pathways (reviewed in Gillanders & Kingsford, 2002). While most of this variability in flow has historically been due to natural causes, anthropogenic alterations of freshwater flow to estuaries are increasingly common, arising, for example, from water extraction for agriculture, industry and domestic use (Kennish, 2002). Predicted changes in climate are also likely to alter flow regimes (e.g. Chiew & McMahon, 2002).

Knowledge of the effects of changes in flow on ecosystems is an issue of increasing importance in the field of aquatic ecology. While considerable work has been done on flow-related effects in freshwater systems, there has been less examination of the changes in estuarine ecosystems (Estevez, 2002; Pierson *et al.*, 2002). Since 1990 there has been an increasing amount of research on the effects of changes in freshwater flow to estuaries, much of which has yet to be published in the peer-reviewed literature or is being done outside Australia (Estevez, 2002).

1.2. Intermittent estuaries

Coastal water bodies can be described in terms of their degree of connection to the ocean and their relative freshwater inflows along a broad continuum from coastal lakes, with little to no connection to the ocean at one end, via coastal lagoons and estuaries to bays and straits, which are essentially partially-enclosed marine bodies. In this continuum, the systems that are the subject of this study have some characteristics of both coastal lagoons and estuaries.

The term ‘estuary’ has been defined in many ways and has been used to describe bodies of water ranging from open embayments, to lagoons that are isolated from the sea by barriers for most of the time. Other examples of water bodies that have been included in various definitions (reviewed in Perillo, 1995) are; ‘open estuaries’ where large volumes of fresh water mix with salt water in seas or oceans, ‘reverse estuaries’ that have no freshwater input, but are driven by evaporation and mixing of marine and hypersaline waters, glacial valleys (fjords) and drowned river valleys. Depending on sediment supply, and fluvial and coastal energy, estuaries that are semi-enclosed may either have a deep, permanently-open connection to the sea or may be largely isolated by sedimentary barriers (Hodgkin, 1994). For this study, a relatively recent definition of estuaries was adopted:

> "*a semi-enclosed coastal body of water that extends to the effective limit of tidal influence, within which sea water entering from one or more free connections with the open sea, or any other saline coastal body of water, is significantly diluted with fresh water derived from land drainage, and can sustain euryhaline biological species for either part or the whole of their life cycle*"(Perillo, 1995).

The estuaries studied here are in a subclass that have intermittently open sedimentary barriers that connect them to the sea. In the introduction to the definition above, Perillo (1995) makes the point that such water bodies are only estuaries in the strict sense when open to the sea, hence the term 'intermittent estuary'. Often these estuaries are also intermittent in another sense, as used in Cuff and Tomczak (1983), in that freshwater inflow is not continuous and there is not always significant dilution of seawater.

The term 'coastal lagoon' may also be used to describe the intermittent estuaries of this study when they are not open, or have no freshwater input. A coastal lagoon has been defined as '*a shallow water body separated from the ocean by a barrier, connected at least intermittently to the ocean by one or more restricted inlets, and usually oriented shore-parallel*' (Kjerfve, 1994). Implicit in this definition is that 'the entrances are narrow compared with the coastwise extent of the enclosing barriers' (Bird, 1994), and that there may or may not be a component of freshwater flow diluting a coastal lagoon. Often, larger coastal lagoons may have more than one entrance from the sea, and the term 'choked' lagoons has been used to describe the subset of coastal lagoons with only one entrance (Kjerfve, 1994).

The type of systems examined here may then, depending on supply and residence times of fresh water, and opening and closing of the mouth, be considered estuaries or coastal lagoons (Table 1.1). That they share some, but not all, characteristics of both estuaries and coastal lagoons indicates that they are located at the junction of these two types of systems in the continuum from landlocked coastal lakes to open, marine embayments and straits.

Freshwater flow?	Freshwater dilution?	Entrance State Open	Entrance State Closed
Yes	**Yes**	Estuary	Coastal lagoon
No	**Yes**	Estuary	Coastal lagoon
No	**No**	Coastal lagoon (Tidal inlet)	Coastal lagoon

Table 1.1. Categorisation of water bodies in this study based on entrance opening and freshwater flow and residence based on the definitions in Kjerfve (1994) and Perillo (1995).

'Intermittent estuary' is the term used to describe these systems in this study but similar systems are also variously referred to in the literature either as choked coastal lagoons, ICOLLs (intermittently closed and open lakes or lagoons), or as ephemeral, bar-built, temporarily open/closed or blind estuaries.

This type of estuary is common in regions of the world that have Mediterranean-type climates and low tidal ranges (*i.e*. southern and western Australia (Kench, 1999), the Cape region of South Africa (Cooper, 2001), the lower west coast of North America (e.g. Elwany *et al.*, 2003), the mid-latitude coasts of South America (Isla, 1995) and the Mediterranean (e.g. Cardona, 2006)). There is also a tendency for smaller barrier systems to close in microtidal regions with other climate types (e.g. the east coast of Australia: Roy *et al.*, 2001), more arid parts of southern Africa (Cooper, 2001) and the South Island of New Zealand (Kirk & Lauder, 2000)). The prevalence of intermittent estuaries in these parts of the world is reflected in a morphodynamic classification for microtidal estuaries proposed by Cooper (2001). In this dichotomic classification, the first splitting criterion separates 'normally open' and 'normally closed' systems based on a bimodal frequency distribution of mouth opening reported for estuaries in the KwaZulu-Natal region of South Africa and four of the five final categories contain estuaries that may close. Such a distinction is also reflected in the historical tendency to separate the study of 'estuaries' (typically open mouths) from 'choked coastal lagoons' (mostly closed or highly constricted mouths) when, in fact, a continuum of 'connectedness' exists between such systems.

Globally, estuarine research has traditionally focused on large, permanently-open systems resulting, for example, in dominant categorisations of estuaries, and zones within estuaries, based on salinity structures that assume a continually-tidal system (e.g. Rochford, 1951; Odum, 1959). Such a focus may reflect the relative importance of these systems as ports and fishing grounds. This has been the case in Australia, where much early estuarine research was associated with east-coast oyster farming (Rochford, 1951). While recent reviews and classifications of estuarine structure and function have recognised the large number of small, intermittently-open estuaries in temperate Australia (e.g. Digby *et al.*, 1998; Roy *et al.*, 2001; Gillanders & Kingsford, 2002) little is known about them. This has been reflected in either generalities being proposed or the physical and ecological properties of smaller estuaries being ignored in favour of discussion of those of larger, more-often studied estuaries. While this focus is understandable, there are hydrological and ecological aspects of small, intermittent estuaries that are considerably different from those of larger estuaries. These aspects need to be recognised when developing models of ecosystem function for Australian estuaries.

Substantial bodies of work on intermittent estuaries exist for South Africa (where 73% of estuaries are intermittently open: Nozais *et al.* 2001), and to a lesser extent, for south-western Australia where these estuaries are the dominant type (Lenanton, 1974a; Environmental Protection Authority, 1987, 1988a, 1988b, 1988c, 1989). In recent years, there has been an increased research focus on these estuaries on the east coast of Australia (e.g. Pollard, 1994; Dye & Barros, 2005; Jones & West, 2005), resulting in the now-locally-common acronym 'ICOLL'.

Intermittent estuaries may be particularly vulnerable to anthropogenic changes in freshwater flow due to their prevalence in Mediterranean climates, and the strong influence that freshwater flows have on their hydrology. Many regions where intermittent estuaries exist already have altered freshwater flows due to urbanization and irrigated agriculture (Gillanders & Kingsford, 2002). In Mediterranean-type climates, runoff is likely to be affected disproportionately by climatic change compared with other areas, with decreases in overall runoff and an increase in the frequency of extreme events more likely (Bolle, 2003). Increased periods of closure associated with reduced runoff will increase the residence times of water in these estuaries, potentially leading to problems with nutrients and pollutants. Conversely, an increased frequency of large flows may provide greater flushing of the estuaries. Work on the relationships between freshwater flow, mouth openings and associated physical and ecological changes is needed to provide an understanding of current processes to then allow prediction of potential changes (Sherwood, 1988).

1.3. Seagrass beds in intermittent estuaries

Despite a large body of work on estuarine seagrasses globally, including the bays, gulfs and larger estuaries of temperate eastern Australia (e.g. Larkum *et al.*, 1989), very little research has been done on seagrass ecosystems in smaller estuaries. The majority of ecological work relating to seagrass systems in intermittent Australian estuaries has been done in Western Australia and, to a lesser extent, NSW, focussing mainly on fish (as summarised in Jones & West, 2005). In Victoria, almost all of a relatively small body of research has been done in coastal embayments, particularly Western Port and Port Phillip Bays (e.g. Shepherd & Robertson, 1989; West *et al.*, 1989: see Figure 1.1 for locations). Seagrasses in these environments are likely to be subject to much smaller changes in the physico-chemical properties of surrounding water and in sediment dynamics than seagrasses in the small estuaries that are typical of the coast between the Coorong (in South Australia) and Port Phillip Bay. Because their highly variable hydrology, it is likely that seagrass productivity in these smaller water bodies is linked to the hydrological regime more strongly than in the open, marine-influenced bays of Port Phillip and Western Port. This hypothesis is supported by evidence mostly from South Africa (e.g. Hanekom & Baird, 1988; Talbot *et al.*, 1990; Adams *et al.*, 1992; Allanson & Read, 1995) but also from Western Australia (Carruthers *et al.*, 1999), suggesting that changes in freshwater flow dominate the growth patterns of seagrasses in small estuaries, in contrast to the more seasonal patterns predominantly observed in large embayments of Victoria (reviewed in Chapter 6).

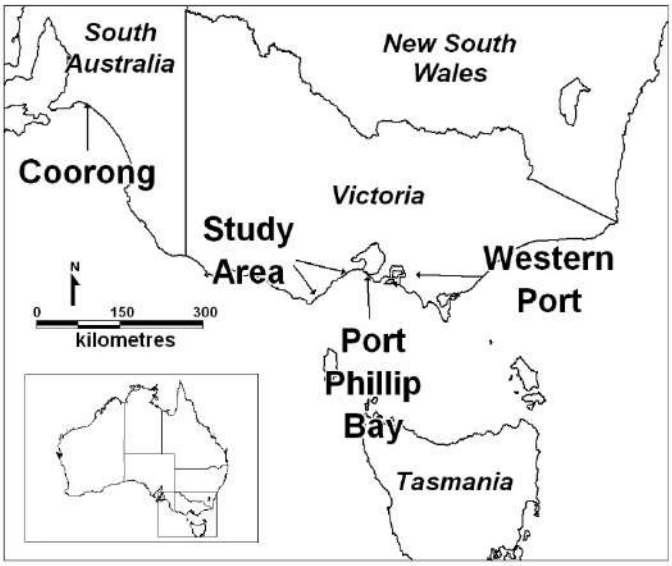


Figure 1.1. Map of south-eastern Australia, showing locations of States, study area and locations mentioned in this section. The rectangle in the inset shows the extent of the main map.

1.4. Estuaries of southern Australia

Estuaries on Australia's south-eastern coast are typically considered to be well-mixed with short periods of stratification, based largely on characterisations of east-coast estuaries (Digby *et al.*, 1999; Kench, 1999; Gillanders & Kingsford, 2002), but less is known about hydrological processes in the intermittent estuaries that are common along southern and western coasts. Part of this lack of knowledge stems from a 'paucity of research examining the timescale of estuary opening and closure and the associated physical and ecological changes that occur in these ephemeral estuaries' (Kench, 1999).

There is little information in the formal literature on inter-annual or seasonal mixing for typical Victorian estuaries, as opposed to the two large marine embayments (see, for example, Sherwood, 1988). A few technical reports exist, at least for estuaries in the west of the State (e.g. Sherwood, 1983, 1985; Sherwood *et al.*, 1998), and the hydrodynamics of some of these systems have been reported in the context of ecological changes in theses and journal articles (e.g. Newton, 1994, 1996; Walsh & Mitchell, 1995; Rouse, 1998; Arundel, 2003; Matthews & Fairweather, 2003, 2004, 2006; Matthews & Constable, 2004; Matthews, 2006).

Longitudinal gradients in tidal range and climate are likely to affect mixing regimes of estuaries along the Victorian coast. Tidal range increases from west to east, with localised increases along the central part of the coast near Port Philip Bay. Climate changes from a Mediterranean-type to a wet, non-seasonal type along the same axis. These west-east gradients are discussed in Chapter 2, but it is likely that the mixing regimes of Victorian estuaries grade from being similar to those of south-western Australia in the west to those of temperate eastern Australia in the east and comparisons with results of studies in systems from both regions are made throughout this thesis.

1.5. Background, aims and thesis structure

This study investigates links between freshwater flows, estuarine hydrology, estuarine water quality and seagrasses. It comprises a multidisciplinary assessment of the effects of freshwater flow on the hydrology and water quality of small, intermittent estuaries and associated patterns of change in the extent of seagrass beds and rates of detrital processes.

1.5.1. Background

This research commenced in late 1998 as a two-year Master of Science project jointly funded by Alcoa of Australia and Deakin University. The initial aim of the project was to increase knowledge of the ecology of the Anglesea River and its estuary with the specific objectives of :

1. Describing their physico-chemical characteristics with particular reference to: sediments and suspended solids; hydrodynamics; bathymetry; temperature, salinity and dissolved oxygen; pH and trace metal concentrations; and nutrients;

2. Quantifying the dominant seagrass, algae, zooplankton and macroinvertebrates in the Anglesea River system;
3. Comparing both the physico-chemical and biological data to those of adjacent or similar river systems;
4. Investigating causal linkages between: trends within and between the physico-chemical and biological data; and trends in the physico-chemical data and/or biological data and human activities – in particular any possible influences attributable to Alcoa's coal mine and power plant; and
5. Recommending appropriate management strategies to protect and, if necessary, enhance the Anglesea River ecosystem.

The study arose from a lack of knowledge about aquatic ecosystems at Anglesea, where Alcoa operates an open-cut coal mine and an associated power station. Particular areas of interest were stream and estuarine hydrology, cycling of trace metals, the trophic status of the system and quantitative ecological information. The only earlier study on water quality and bioaccumulation (Atkins & Bourne, 1983) was conducted in an unusually dry period. Since then the estuary has been substantially physically modified, meaning that results from that study may not be representative of the present system.

At the end of 2000, the scope of the study was expanded to that of a PhD-level project, with a change of focus from an ecological description of the Anglesea River and estuary, to an examination of the effects of changes in volume and composition of fresh water on the physical and ecological characteristics of intermittent estuaries. Anglesea estuary (with augmented flows) and the adjacent Painkalac Creek estuary (with water extraction) were the focus of the bulk of the study. Eight other regional estuaries were examined at a lesser level of detail to provide a broader context for the results from the two main estuaries studied (see Section 2.3).

The two main estuaries studied provided an excellent opportunity to examine flow-related effects on estuarine ecology. While they have many similarities, there is a clear, long-term and quantifiable contrast in freshwater flow regimes due to anthropogenic augmentation of flows in one estuary and reduction of flow in the other. In addition, existing long-term datasets relating to both estuaries provided a temporal context on the scale of decades against which to compare data from the three years of sampling for this study.

An increased understanding of hydrology and seagrass dynamics in this type of estuary, as well as these specific estuaries, has implications for the management of such systems in conditions of reduced or altered freshwater flow. Locally, it may have relevance for the provision of environmental flows for Painkalac estuary, and for management of the reductions in flows to Anglesea estuary that are anticipated to follow eventual closure of the mine and power station.

Results of this study will be relevant for managers of intermittent estuaries world-wide. A number of government bodies in Australia and overseas (e.g. Cooperative Research Centre for Coasts and Estuaries, Commonwealth State of Environment Reporting, National Land and Water Resources Audit, Victorian Environment Protection Authority, South African Department of Environmental Affairs) are engaged in studies to determine management goals for intermittent estuaries. This study will contribute to the protection of these systems, particularly in relation to management of potential effects of upstream water diversions.

1.5.2. Aims and objectives

The broad aim of this study was to investigate the characteristics of, and links between, freshwater flows, estuarine hydrology, estuarine water quality and seagrass extent and decomposition in intermittent estuaries. These links are shown schematically in Figure 1.2. To address this aim, the specific objectives detailed in the following sub-sections were developed.

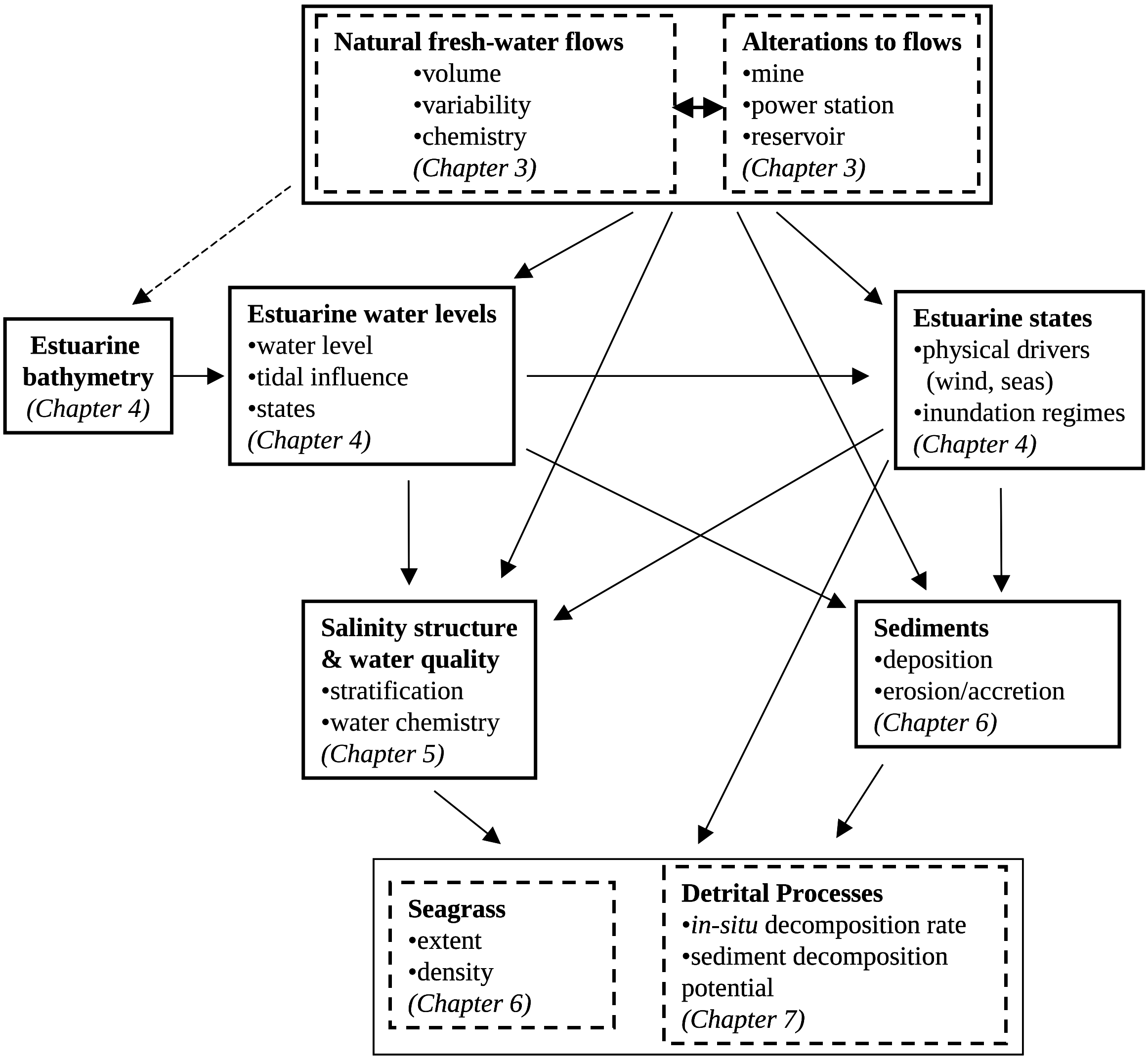


Figure 1.2. Aspects of estuarine functioning and their interrelationships that are addressed in this study. Chapters where data on various components are presented are shown in italics. Lines indicate links between aspects that were examined. The dashed arrow represents a link that is likely to exist but was not the subject of this study and dashed boxes represent closely related aspects.

1.5.2.a. Fresh waters (Chapter 3)

An understanding of the nature of freshwater flows was needed as a basis for examining their effects in the estuaries. This component of the study aimed to:

- assess links between rainfall and stream flow;

- quantify the volume and temporal variability of flows from the catchments;
- determine the physical and chemical characteristics of flows from the catchments, including links with hydrology;
- assess anthropogenic changes to the volume and timing of flow to the estuaries;
- assess anthropogenic changes to water quality; and
- examine temporal and spatial patterns in historical water quality.

1.5.2.b. Estuarine bathymetry and hydrology (Chapter 4)

A detailed examination of Anglesea estuary and a coarser temporal- and spatial-scale assessment of Painkalac estuary were completed. Associated objectives were to:

- quantify the bathymetry of Anglesea estuary;
- develop, and assess the validity of, a conceptual model of relative tidal influences in intermittent estuaries based on changes in water level and constriction of entrances; and
- examine the relative influence of freshwater, marine and coastal processes in relation to the conceptual model.

1.5.2.c. Estuarine salinity structure and water quality (Chapter 5)

The general aim of this component was to develop an understanding of estuarine hydrology and water chemistry in relation to flow-related physical processes, and the chemistry of freshwater inflows. Related specific objectives were to:

- examine relationships between freshwater flow, tidal influence and salinity regime, including stratification; and
- examine links between water quality and freshwater inflow, estuarine hydrology and salinity regime, with reference to the conceptual hydrologic model.

1.5.2.d. Estuarine sedimentation and seagrasses (Chapter 6)

Hydrology and water quality can determine the suitability of estuaries as habitat for seagrasses. Sedimentation and sediment mobility are further physical factors that can influence seagrass extent. Objectives of this section were to:

- describe historical changes in seagrass distribution in Anglesea estuary and examine potential links with flow;
- quantify changes in seagrass distributions and densities in Anglesea and Painkalac estuaries;
- examine sediment dynamics via deposition rates and net erosion/accretion in the lower parts of the estuaries;
- relate changes in seagrass extent and density to hydrological and sedimentary changes, particularly changes that are likely to be associated with flow; and
- assess relative influences of hydrologic events and seasonal changes on seagrass dynamics.

1.5.2.e. Detrital processes (Chapter 7)

Decomposition of detritus is a major process in the transfer of seagrass primary production to other components of estuarine ecosystems. Measurement of these rates, and of changes in these rates in association with physico-chemical variation, provides insight into the functioning of these ecosystems. Specific objectives were to:

- quantify *in situ* decomposition rates of seagrass detritus in litterbags, as a measure of transfer of material between the seagrass and microbial components of the estuarine ecosystem, and examine differences in rates through time and between estuaries and sites;
- assess decomposition potential of the estuarine sediments and their associated microbial communities between different times, water depths, estuaries and among locations at two smaller spatial scales within the estuaries by measuring cellulose decomposition rates with the cotton-strip assay method, a novel technique in estuarine research;

- compare decomposition potential of sediments between seagrass beds and other areas with accumulated seagrass macro-detritus; and
- relate changes in these measures of decomposition to hydrological and other flow-related changes.

1.5.3. Thesis structure

The structure of this thesis reflects various ways that fresh water can influence the hydrological, chemical and ecological structure and function of estuaries (Figure 1.2). The thesis is broadly ordered in two ways; the first geographical, moving from the catchment and fresh waters to the estuary and marine influences, and the second discipline-based, moving from physical to chemical to biological processes. General design and site information is given in Chapter 2, with more detailed methodologies described in appropriate chapters. Chapter 3 relates to the amount, timing and quality of freshwater inputs to the two main estuaries studied, and examines anthropogenic changes to those inputs. Chapter 4 deals with estuarine bathymetry and water levels and develops a “three-state” conceptual model of relative tidal influences and degrees of constriction at the mouths of the estuaries. The hydrologic and meteorologic processes that drive maintenance of, and transitions between, the three states are also addressed in Chapter 4, along with inundation regimes associated with each state. Chapter 5 investigates stratification and water quality through space and time, examined in the context of hydrologic state and freshwater flows. Chapter 6 deals with changes in the extent of seagrass beds, sedimentary processes and seagrass density. Chapter 7 addresses detrital processes of seagrasses. A synthesis of findings is presented in Chapter 8.

2. Design of Study and Location Descriptions

2.1. *Study area: regional context*

The Otway Range runs parallel to the south-east facing coast of the study area in central-west Victoria (temperate south-east Australia; Figure 2.1). In contrast to the south-west facing coast of western Victoria, the south-eastern aspect of this coastline shelters it from much of the energy associated with the predominant westerly wind and swell.

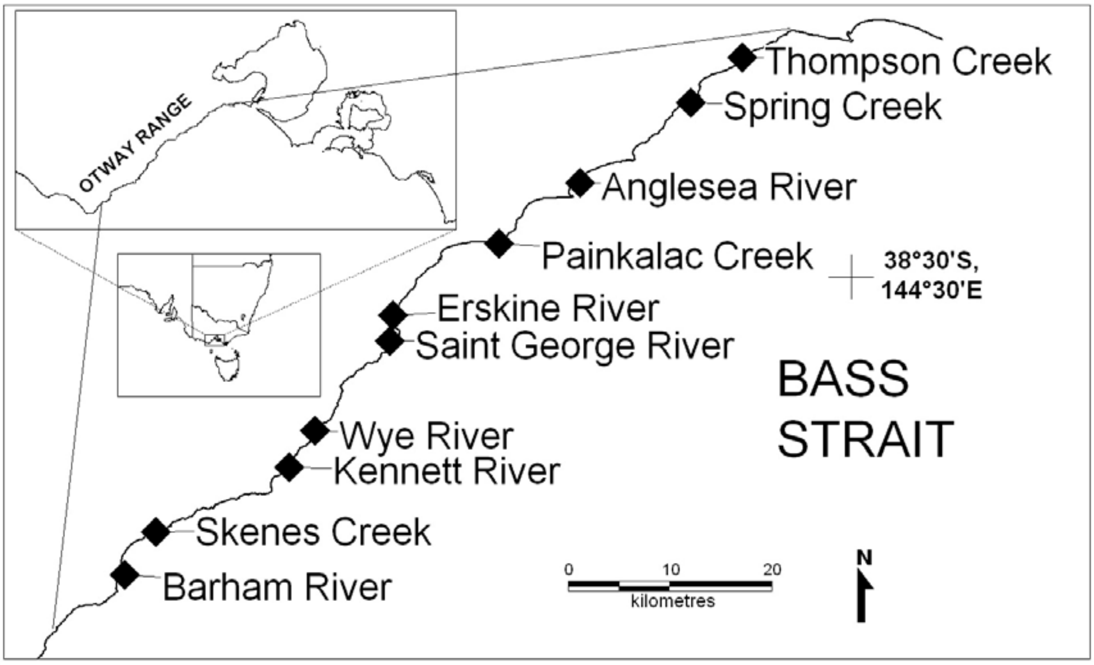


Figure 2.1. Location of study area and estuaries sampled. Towns with different names from their associated rivers are, from west to east, Apollo Bay (Barham River), Lorne (Erskine River), Aireys Inlet (Painkalac Creek), Torquay (Spring Creek) and Breamlea (Thompson Creek). Saint George River is not associated with a town. Port Phillip Bay is the larger of the two embayments shown on the intermediate scale map, the entrance to which is 20km from Thompson Creek. Positions of the western and eastern ends of the coastline shown are 38º51'S 143º34'E and 38º18'S, 144º36'E respectively.

The Otways grade from over 650m elevation inland from Wye River to just under 400m in the highest points of the Anglesea and Painkalac catchments. The spine of the Range is between 10km and 15km inland meaning that the catchments of estuaries in the study area are smaller and steeper than many others in south-eastern Australia. The Range extends westwards beyond the study area while at the eastern end it gradually decreases in elevation such that much of the catchments of Spring and Thompson Creeks are on flatter land that extends towards Port Phillip Bay and Melbourne. Due to their location near the end of the Range, the topographies of the Anglesea and Painkalac catchments differ from those in the rest of the study area. They have lower relief and are larger catchment areas than most Otway streams but are steeper and smaller than other systems outside the Range to the east.

2.1.1. Rainfall and river flow

Mean annual rainfall increases westwards along the Otway Range from a low of 653mm at Anglesea to 924mm near Erskine River and a peak of 1928mm on the ridge of the Range inland from the Barham estuary. To the east of Anglesea, mean annual rainfall further decreases to below 600mm at the coast near Spring and Thompson Creeks (Land Conservation Council (Victoria), 1976).

2.1.1.a. Seasonality

The study area is part of ~8000km of Australian coast that is classified as winter-rain dominated (winter defined as May to October: Commonwealth of Australia: Bureau of Meteorology, 2006b). This area covers most of the southern coastline of mainland Australia, from Shark Bay in Western Australia (26^{o}13’S, 113^{o}11’E) to east of Wilsons Promontory in Victoria (38^{o}37’S, 146^{o}52’E) as well as all but the central east coast of Tasmania.

This rainfall classification is broadly consistent with the division of the study area into two categories of seasonal stream flow regime by McMahon and Finlayson (2003). The flow regime of the eastern portion of the study area is classified as ‘early Spring’ and the western portion as ‘extreme Winter’, based on the global classification scheme in Haines *et al.* (1988). This division between two types of flow regime reflects the gradient in rainfall within in the study area, as discussed above. Other regions of the world fall within the same seasonal rainfall categories (Table 2.1) and many of these have a Mediterranean-type climate.

Flow regime[a]	Coastline	Region	M-type climate?[b]
extreme Winter	*W Otway Range*	Australasia	√
	W Victoria/E South Australia	Australasia	√
	SW West Australia	Australasia	√
	W Cape Region, South Africa	Africa	√
	Israel/Palestine	W Asia	√
	Portugal	Europe	
	California, S Oregon	N America	√
early Spring	*E Otway Range*	Australasia	
	E Victoria	Australasia	
	W & E Cape Region, South Africa	Africa	(√)
	Black Sea & Turkey	Asia/Europe	(√)
	W & S Spain	Europe	√
	S Baltic	Europe	
	central Chile	S America	√
	central Argentina	S America	
	S Brazil	S America	
	E USA	N America	
	S Island, New Zealand	Australasia	

Table 2.1. Coastlines of the world with similar river flow regimes to those of the study area. a –as identified in Haines *et al*. (1988), b – Mediterranean climate type, as delineated in Bolle (2003) and discussed in Chapter 1; brackets indicate that part of the region described has this climate type. Coastlines of the study area are shown in italics.

A link between the seasonal variability of flow and occurrence of intermittent estuaries is highlighted in Table 2.1, showing coastlines of the world with similar flow regimes to the study area. Most coastlines where intermittent estuaries occur are included in these two of the fifteen global types of flow regime defined by *Haines et al.* (1988). Intermittent estuaries also occur in micro-tidal regions within two other classification types. One of these types, 'moderate autumn', includes the southern east coast of Australia; the other, 'moderate winter', includes Tasmania, part of eastern Victoria, the North Island of New Zealand, parts of the Mediterranean and Uruguay.

2.1.1.b. Inter-annual variability

Inter-annual variability in rainfall can be an important driver of estuarine processes, but, compared with seasonal differences, is relatively little studied. This may be due to the long duration such studies require and/or the relatively small inter-annual variation in rainfall and flow (compared to seasonal variations) in many parts of the world where estuaries have been studied.

To compare inter-annual variability of rainfall in the study area with the rest of Australia, the formula:

$$IV = (R_{90}-R_{10})/R_{50}$$

was used, where IV is inter-annual variability and R_n is the n^{th} percentile of the distribution of annual rainfall (Commonwealth of Australia: Bureau of Meteorology, 2006a). By this measure, inter-annual variability in rainfall (through the 20th Century) was low in the study area relative to the rest of Australia (IV<0.5, where for most of coastal Australia, 0.5<IV<1.0). The IV of annual rain in the study area was similar only to the monsoonal areas of far northern Australia and south- and west-facing portions of the southern Australian coastline. Within the study area, inter-annual variation of summer rains was approximately twice as large as for winter rains and overall variability decreased westwards along the coast (Commonwealth of Australia: Bureau of Meteorology, 2006a).

Compared to other systems in the State, Otways estuaries tend to have more seasonal and intermittent freshwater inputs, smaller catchments and smaller water areas. All estuaries in the study area are fed by intermittently flowing watercourses with the exception of the relatively large Barham River in the west, which has flowed constantly since at least 1977. Augmentation of flow in the Anglesea River resulted in a continuous flow to this estuary through the study period and presumably has done so since the Alcoa power station became operational in 1969 (Atkins & Bourne, 1983). There are dams or weirs that retain water and modify flows to varying extents located in the catchments of four other estuaries – those of Painkalac Creek, Erskine River, St George River and Barham River.

2.1.2. Tides

The Otways coast is micro- to meso-tidal with tidal waves moving from west to east, having refracted around Tasmania (Easton, 1970). Along the coast of the study area, tidal range increases from west to east (Short, 1996). The mean spring range of astronomical tides increases from 1.34m at Apollo Bay near the Barham River mouth to 1.59m at the Erskine River in the centre of the study area (Easton, 1970), to 1.6m at Barwon Heads, 11km east of the Thompson Creek mouth (Nelson & Keats, 1978). Qualitative observations for Victorian ports derived from pre-1970 Ports and Harbour Branch tide tables by Easton (1970) show similar changes between these three ports, but with a larger difference from Lorne to Barwon Heads (mean spring rises of 1.5m, 1.8m and 2.2m, respectively).

Tides in this region are semi-diurnal, typically with substantial height differences between the two low tides of the day and smaller height differences between the two high tides (Easton, 1970). Meteorological influences can account for a substantial proportion of tidal movements in the area, for example a meteorological range of 0.8m (+0.5m and –0.3m) was measured by Nelson and Keats (1978). High meteorological tides are associated with low barometric pressures and westerly winds. Conversely, low meteorological tides are associated with high barometric pressure and easterly winds (Nelson & Keats, 1978).

2.2. Study area: waterways and catchments

2.2.1. Anglesea River

2.2.1.a. Catchment

The Anglesea River has two major tributaries, Marshy and Salt Creeks, both of which have intermittent and predominantly seasonal discharge. These tributaries each comprise approximately half of the riverine catchment area and are similar lengths (Figure 2.2; Table 2.2). Each tributary has one major sub-catchment, Edwards Creek and Breakfast Creek in the Marshy and Salt Creek catchments, respectively.

Most of the catchment consists of intact native vegetation, mostly heathland and open, dry-sclerophyll forest. A small proportion of the rim of the catchment is grazed, ~2km^2 in the Marshy Creek subcatchment, and 0.3km^2 in the Breakfast Creek subcatchment of Salt Creek. Active and disused gravel pits as well as dirt tracks used by trail bikes and four wheel drives are scattered through the forested section of the catchment. Small stock and fire-fighting dams are located in the headwaters, primarily in the Marshy Creek subcatchment.

A heavy-vehicle driver-training complex, previously a proving ground for agricultural machinery, is located in the upper part of the Marshy Creek subcatchment (~9.5km^2), while a cleared area of ~0.6km^2 on a river flat in the middle of the catchment has been grazed and has a rifle range.

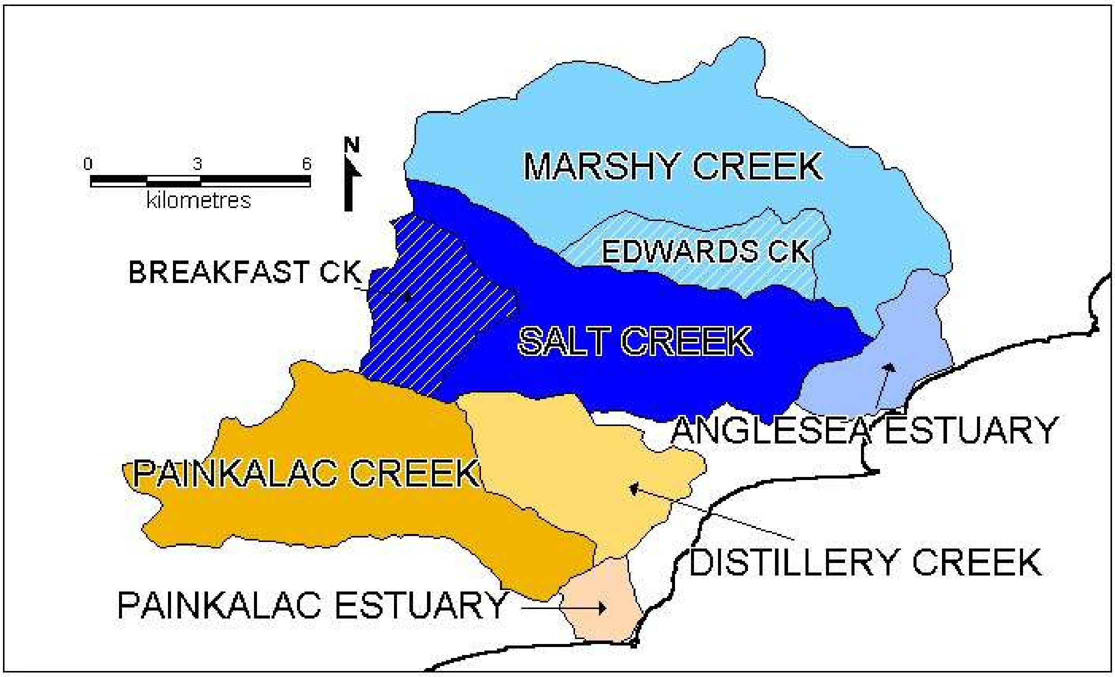


Figure 2.2. Major sub-catchments of the Anglesea River and Painkalac Creek.

Estuary	Sub-catchment	Area (km^2)	Area (% Tot.)	Max Length (km)
Anglesea	**Salt Creek**	51.2	41	15
	Breakfast Creek	(12.4)	(9.9)	5.4
	Marshy Creek	65.0	52	17.5
	Edwards Creek	(11.3)	(9.1)	7.9
	Direct Estuarine	8.74	7.0	2.6
	Total (catchment)	**125**	**100**	**18.7**
Painkalac	**Mainstem**	39.2	65	15
	Distillery Creek	18.0	29	8.9
	Direct Estuarine	3.85	6	1.4
	Total (catchment)	**61.1**	**100**	**15.1**

Table 2.2. Areas and maximum stream lengths of sub-catchments of the estuaries of Anglesea River and Painkalac Creek. Areas for Salt & Marshy Creeks include sub-catchments of Breakfast and Edwards Creeks – dimensions of these are shown in brackets.

Salt and Marshy Creeks meet about one kilometre above the head of the estuary to become the Anglesea River (although on some maps Marshy Creek is known as Anglesea River from its headwaters). A brown-coal mine, operated by Alcoa, is located in the lower reaches of Salt Creek, which has been diverted around the open cut in a concrete channel. The coal is burnt in a power station located near the junction of the tributaries (Figure 2.3). Ash ponds serving as temporary storage for waste from the power station are located next to it.

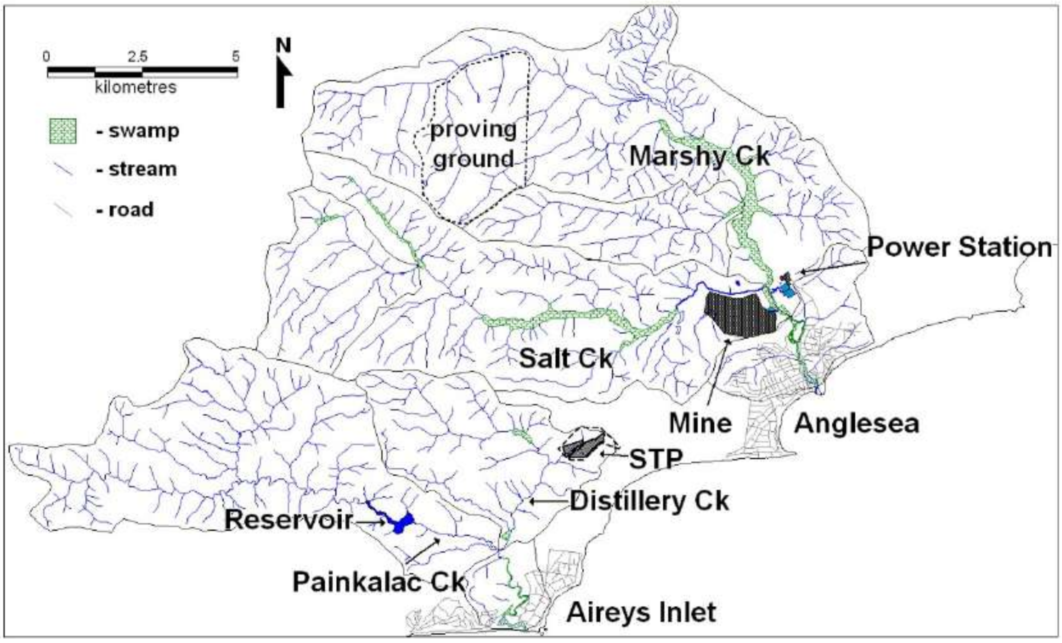


Figure 2.3. Waterways of the Anglesea River and Painkalac Creek catchments. Data from Royal Australian Survey Corps (RASC) 1:25000 maps except for the Painkalac reservoir, STP and the Anglesea estuary, mine and power station. The reservoir was delineated based on post-processed differentially-corrected GPS (dGPS) recorded positions at the dam wall and infilling of contours from the RASC 1:50000 map. The location of the STP (sewage treatment plant) is approximate and was derived from a 1:25,000 Barwon Water map (Barwon Water, 2003a). Additional features at Anglesea were digitised from an aerial photo taken on 29/8/98 and provided by QUASCO and Alcoa of Australia. Georectification of the photo was done using points recorded with dGPS.

The mine and power station operations contribute two sources of water to the estuary: the first source is groundwater, used for a number of processes within the power station, including transportation of ash from both the furnace and the chimney precipitators to the ash ponds. Ash settles in these ponds, and their overflow continually discharges to the Anglesea River. Sediment in the ponds is cleared to the mine pit approximately every five years. On average this water source was about 10 times greater in volume than that of the second industrial source, groundwater seepage and rainwater from the mine pit. This water was pumped to holding ponds which also overflow to the Anglesea River. At times water is transferred between holding ponds of the two sources to manage pH of the discharges. The volumes of flow from both discharges are less variable through the year than natural flows, often comprising the entire flow of the river when the tributary creeks are dry, but only accounting for a small fraction of larger natural flows (Section 3.3.5).

The town of Anglesea straddles the estuary and covers most of the area draining directly to the estuary (Figures 2.2, 2.3; Table 2.2). The town was sewered in 1974, with an ocean outfall north east of the estuary, but most of the town's stormwater, estimated at 1700ML/year (Atkins & Bourne, 1983), drains to the estuary via 16 stormwater outlets (Donaldson *et al.*, 1998).

2.2.1.b. Estuary

The Anglesea estuary is 2.6km long and 110m wide near its mouth, narrowing to 40m one km upstream and ~15m in the upper reaches (Figure 2.4). It has a surface area of 15ha when bank full. Surface area increases as marsh areas in the upper estuary are inundated at higher water levels (see Chapter 4). Estuary outflows run across a beach, landward of which is an area of shallow marine sand (a flood-tide delta) that extends to ~300m upstream. In the lower estuary, to the road bridge one km upstream, a central channel is fringed by sand and mud flats of relatively low relief. At the upper limit of the flats there is a low (~0.5m high) seawall, backed by reclaimed land. In the mid-estuary, the clay banks of the channel become steeper, with no fringing rock walls. Emergent macrophytes grow along most banks of the mid- and upper estuary. The upper estuary (known as Coogoorah Park) contains constructed channels. These were dredged west of the original channel in 1984-85 to isolate a section of peat perceived as a fire risk to the town following bushfires in 1983. A network of channels and low islands with extensive macrophyte beds at lower elevations now covers ~ 17ha in the upper third of the estuary (Figure 2.4).

Water level in the estuary can vary by 1.6m (see Chapter 4). At low levels, the central channel remains submerged because it is between 1.5 and 3m deeper than the banks. At extremely low water levels some flat areas within the channels become exposed in Coogoorah Park; however, the majority of the time water is confined within relatively steep-sided banks while at high water levels sections of the islands are flooded. The channel becomes shallower downstream of the road bridge. Sand- and mud-flats in this part of the estuary become exposed at medium to low water levels while at higher levels the water may reach the top of the sea walls. At the highest water levels, the main road can be flooded, often prompting artificial opening of the estuary (Section 4.3.5.a).

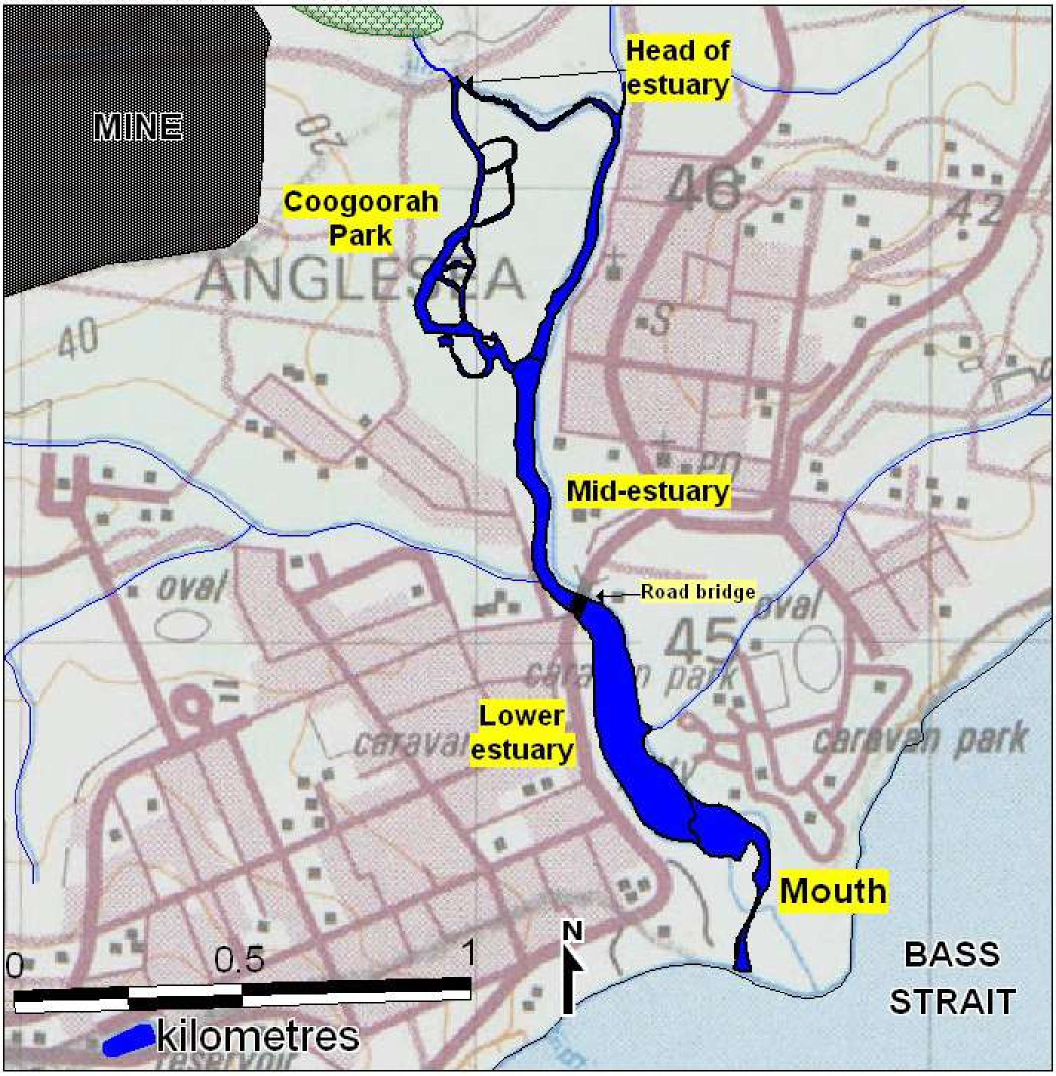


Figure 2.4. Anglesea estuary, its sections and surrounds. Data from Royal Australian Survey Corps 1:50,000 map except for the Anglesea estuary boundary which was digitised from an aerial photo described in the caption of Figure 2.3.

The mouth of the estuary faces south and is located to the west of a point extending to a submerged reef 200m offshore to the south (Figure 2.4). On the eastern side of the mouth are scattered rocks while to the west is Anglesea Beach which curves to face south east at its opposite end (~400m from the mouth). The estuary exits to the beach through an area of low-lying sand, flanked by a low headland to the east and dunes to the west. The sandbar at the mouth often reduces or prevents water exchange between the estuary and the sea and has done so since at least the late 1800s (Cecil & Carr, 1989). Along the beach, there is net eastward longshore transport of sand, which is then trapped in the mouth of the estuary and behind the rocks to the east of the mouth (Nelson, 1981). Waves on the beach average 1m and the beach type varies from a low tide terrace at the west end to a transverse bar and rip near the mouth of the estuary depending on wave conditions (Short, 1996).

Macrophytes along the estuary are dominated by *Phragmites australis* and *Juncus krausii*, with some stands of *Typha domingensis*. Throughout this study, seagrasses were found only below the road bridge (see Chapter 6). Prior to the construction of Coogoorah Park, seagrasses were sampled in the upper estuary by Atkins & Bourne (1983).

Fish species commonly observed in the estuary include black bream (*Acanthopagrus australis*), yellow-eyed mullet (*Aldrichetta forsteri*), sea mullet (*Mugil cephalus*), smooth toadfish (*Tetractenos glaber*), short-finned eel (*Anguilla australis*) and common galaxias (*Galaxias maculatus*). Both mullet species use macrophytes and seagrasses as nursery and growth habitat and black bream spawn in the lower estuary (McCarraher, 1986). Ducks are the most commonly observed birds on the estuary, while wading birds use the flats of the lower estuary when the water level is suitable.

Since the late 1800s, the estuary has been a major tourist attraction for Anglesea (Cecil & Carr, 1989). It is used for fishing, canoeing, and swimming. Walking tracks along the shorelines are well used and a weekend market is held on the western shore. Several businesses directly used the estuary during the study period; paddle boats operated in the lower part of the estuary in summer and holiday times and two adventure companies used the river for canoeing, windsurfing, nature tours and other activities.

2.2.2. Painkalac Creek

2.2.2.a. Catchment

Painkalac's mainstem and major sub-catchment, Distillery Creek, meet 200m above the estuary (Figure 2.2, Figure 2.3). Flow in both creeks was seasonal and intermittent. In terms of catchment area and length, the mainstem is approximately twice the size of Distillery Creek (Table 2.2).

The majority of the Painkalac Creek catchment is intact native vegetation contained within conservation and forestry reserves (Parks Victoria, 1999). Apart from some unsealed roads, the upper catchment has been little disturbed and the forestry areas of the upper catchment have not been logged since at least 1983 (OREN, 2003).

Painkalac Reservoir is located 2.8km upstream of the estuary on the mainstem of Painkalac Creek. The annual volume of water extracted from the reservoir is around 175ML (Barwon Water, 2003b). There are two areas of swamp on Distillery Creek, 0.6 and 4.0km above the estuary (Figure 2.3). A sewage treatment plant (STP), with 100% water reuse (Radcliffe, 2004), and its associated 20ha plantation is located on a tributary which joins Distillery Creek 3.1km above the estuary (Figure 2.3). Stormwater from approximately one half of the towns of Aireys Inlet and Fairhaven drains to the estuary.

2.2.2.b. Estuary

Painkalac estuary is 3.8km long and is 100m wide near the mouth, narrowing to 10m upstream. It has a surface area, when bank full, of ~15.7ha. An area of intermittently-exposed mudflats is located 750m from the entrance and an anabranch-like inlet 200m long is located 1km from the entrance (Figure 2.5). Stormwater outlets are found up to 2km upstream, in association with dense housing, while further upstream there are runoff channels from paddocks and sparser residential areas. Areas of swamp are found in a conservation reserve on the floodplain within the bends of the lower 3km of the estuary.

The floodplain is divided by a causeway and road bridge 900m upstream from the entrance.

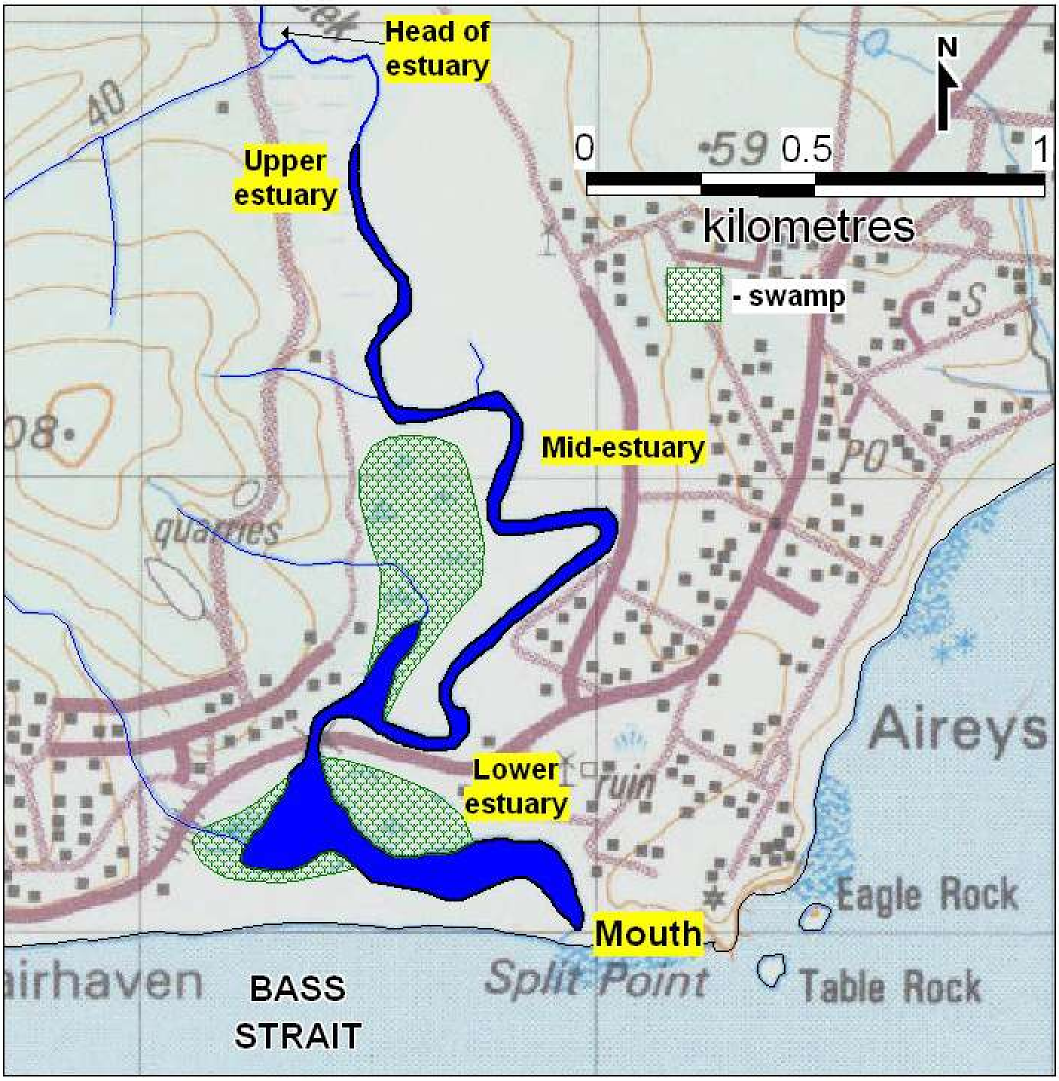


Figure 2.5. Painkalac estuary, its sections and surrounds. Data from Royal Australian Survey Corps 1:50,000 map.

A distinct channel runs almost the entire length of the estuary, becoming shallow with steep banks in the uppermost reaches. In the lowest one kilometre of the estuary, mudflats and salt marsh fringe the channel. The depth of the estuary varies by at least 1.9m over time (Chapter 4); however, the central channel always remains submerged, with a maximum depth 4-4.5m below the banks. At the highest water levels, an area of low-lying houses and shops just upstream of the road bridge can be flooded. A 'trigger' height used for artificial openings of the mouth (2.1m above

Australian Height Datum (AHD)) is marked on the Great Ocean Road bridge (M. Jackman, Surfcoast Shire, pers. comm.).

The mouth of the estuary faces south and is bounded by cliffs and intertidal reefs to the east, and high sand dunes and a beach to the west. The beach runs east-west and its slightly higher wave energy than Anglesea is reflected in the more permanent presence of a transverse bar and rip system (Short, 1996). As at Anglesea, the sandbar at the mouth often restricts flow and intermittently closes (see Section 4.3.4).

Freshwater macrophytes (*Juncus krausii, Isolepis nodosus, Gahnia filum, Phragmites australis* and *Schoenus breviculmis*) fringe the estuary in areas of lower relief. These decrease in extent from the middle to the head of the estuary (Reilly, 1998). Salt marsh (particularly *Sarcocornia quinqueflora*) lines the shallower mudflats in the lower estuary. Seagrasses, while mainly limited to the lower ~1.2km, were found as isolated plants up to ~3.5km upstream (see Chapter 6).

Fish species in the estuary include black bream (*Acanthopagrus australis*), short-finned eel (*Anguilla australis*), yellow-eyed mullet (*Aldrichetta forsteri*), common galaxias (*Galaxias maculatus*), spotted galaxias (*Galaxias truttaceus*), pouched lamprey (*Geotria australis*) estuary perch (*Macquaria colonorum*), sea mullet (*Mugil cephalus*) and smooth toadfish (*Tetractenos glaber*) (McCarraher, 1986; Koehn & O'Connor, 1990). Water birds are generally more abundant and diverse than in the Anglesea estuary. Regularly present waterbirds observed by Reilly (1998) were black swan (*Cygnus atratus*), Pacific black duck (*Anas superciliosa*), little pied- and great cormorants (*Phalacrocorax melanoleucos, P. carbo*), white-faced heron and great egrets (*Egretta novaehollandiae, E. alba*) and masked lapwing (*Vanellus miles*).

The estuary is used less than Anglesea for direct recreational activities but is a focal point of Aireys Inlet, which is smaller town than Anglesea. Fishing, canoeing and swimming are the main uses of the estuary by humans.

2.2.3. Other waterways

Dimensions of other estuaries sampled and their rivers are compared with those of Anglesea and Painkalac in Table 2.3.

Estuary	Catchment Area (km^2)	Length (km)	Estuary Length (km)
Thompson Creek	305	42	6.0
Spring Creek	57.1	19.4	4.0
Anglesea River	125 (125)	17.5 (20.4)	2.6 (3.5)
Painkalac Creek	61.1 (61.0)	15 (18.9)	3.8 (3.25)
Erskine River	30.4	11.9	1.0
Saint George River	33.9	13.6	1.5
Wye River	24.6	10.8	1.0
Kennett River	20.6	12.5	1.2
Skenes Creek	18.2	9.3	0.25
Barham River	79.5	18.7	2.25

Table 2.3. Spatial characteristics of catchments and lengths of estuaries sampled. Data from Mondon *et al.* (2003) except for the Anglesea River and Painkalac Creek estuaries where calculations were made from Royal Australian Survey Corps 1:25000 map data transferred to GIS (with data from Mondon *et al.* (2003) in brackets). Estuaries are listed in order from east to west.

Catchments in the region typically have largely intact natural vegetation with small holiday towns located on the estuaries. Larger estuaries also tend to have some grazing on the floodplain of the estuary. At the east end of the study area, Spring Creek and Thompsons Creek estuaries have catchments that are mostly agricultural and flow eastwards from the relatively flat foothills at the east end of the Otway Range before turning south to the coast. Dams or weirs are present in the catchments of the Erskine River, St George River and Barham River. All estuaries except Saint George estuary are adjacent to towns. Estuaries to the west of Anglesea and Painkalac are predominately confined in steep valleys. To the east, in flatter topography, Spring Creek estuary is largely confined in a shallow valley while Thompson Creek estuary spreads over flat coastal plains and includes a large expanse of mudflats with intertidal seagrass and salt-marsh habitat.

2.3. Study design

Anglesea estuary was the initial focus of this study, which subsequently expanded to include Painkalac estuary and, in less detail, salinity structure and water quality of eight other regional estuaries (Table 2.3). The field component of the study commenced in December 1999 and concluded in February 2002.

There are clear, long-term and quantifiable anthropogenic differences in freshwater flow between the two estuaries, due to Alcoa's augmentation of flow in the Anglesea River and water extraction from Painkalac Creek (Section 2.2). There are also many similarities between the two estuaries including catchment size, estuary size and morphology, rainfall, and tidal regime. This allowed examination of the influence of different flow regimes on intermittent estuaries. Long-term flow and water quality data, particularly from the Anglesea system, provided a decadal context for the three years of this study.

During project planning it was recognised that a comparison of two estuaries limits the study's usefulness for generalisation. A lack of replication at the estuary scale means that results of the study can be confounded by specific differences between the two estuaries, unrelated to differences in hydrology. No other estuaries were considered similar enough to Anglesea and Painkalac estuaries to be included as replicates due to the location of these estuaries and the various geographic, hydrologic and climatic gradients across the region (Section 2.1). Two approaches were used to compensate for this lack of replication. The first was a process-based comparison of the two estuaries that aimed to identify causal links between freshwater flows and estuarine processes in each estuary and examine differences via conceptual models of the linking processes. The second approach was to assess Anglesea and Painkalac estuaries in the context of patterns of differences in all ten regional estuaries across the hydrologic and climatic gradients.

Sampling was designed to examine processes over several temporal and spatial scales. This was reflected in the timing, intensity and breadth of sampling of the various components of the study. The timing of the various components also reflected, to some extent, the change in the nature of the project from late 2000 (see Section 1.5.1), with the addition of seagrass and decomposition-related components at that time (Table 2.4). External datasets with longer timespans were also used in the study: rainfall data (from 1926 onwards); stream flow data (from 1974 onwards); water quality data (from 1972 onwards); and aerial photos of Anglesea estuary (from 1964 onwards) are described fully in the relevant sections. Combined, the elements of the study described in Sections 2.3.1 to 2.3.4 below provided a detailed understanding of drivers and processes on a 'whole-system' basis that can be extrapolated to intermittent estuaries generally but especially those in Mediterranean climates and those with relatively small catchment areas.

Component	Approx. Freq
Freshwater Inputs	
Stream Flow	
A - Tributaries/Mine Sources	daily
A - Tributaries	1 month
P - Tributaries (mainstem) *	continuous
P - Tributaries (inc. Distillery Ck)	1month
A - Flow To Estuary	continuous
A - Flow To Estuary	1 month
P - Flow to Estuary *	continuous
P - Flow to Estuary	1 month
Water Quality	
A - General	1 month
A - Nutrients/TSS	1 month
A - Turbidity	1 month
P - General	1 month
P - Nutrients/TSS	1 month
P - Turbidity	1 month
Alcoa monitoring *	1 month
Estuarine Physical Processes	
Water Height/State	
A - Logged	continuous
A - Spot	1 month
P - Logged	continuous
P - Spot	1 month
Tide at Lorne *	continuous
Sand Bars	
A - General features	1 month
A - GPS trace	1 month
P - General features	1 month
P - GPS trace	1 month
Sediment dynamics	
A&P - Sedimentation rates/OM	integrated[a]
A&P - Net erosion/accretion	integrated[a]
Estuarine Chemical Processes	
Water Quality	
A - General	1 month
A - Nutrients/TSS	1 month
A - Turbidity	1 month
P - General	1 month
P - Nutrients/TSS	1 month
P - Turbidity	1 month
O - General	3 months
O - Turbidity	3 months
Alcoa monitoring *	1 month
Estuarine Biological Processes	
Seagrass extent & density	
A - GPS mapping	6 months
A - transect mapping	3 months
P - transect mapping	3 months
A&P - small-scale bed edges	1 month
A&P - shoot density	1 month
Detrital Processes	
A&P - Seagrass decomposition	1 month
A&P - Cloth strip assay	4 months

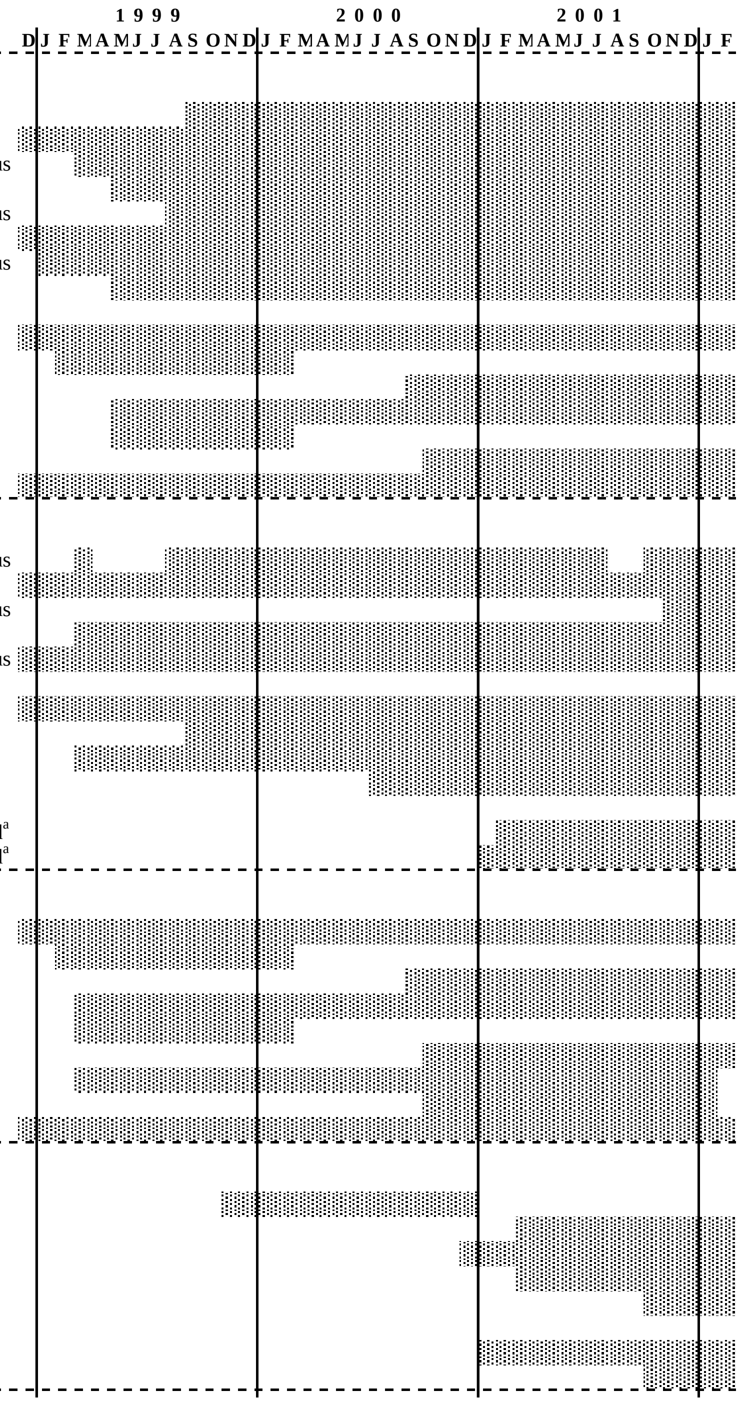


Table 2.4. Duration and frequencies of sampling for each component of this study. A – Anglesea, P – Painkalac, O – Otway (regional) estuaries, * - external dataset, a – integrated over 1 month intervals.

2.3.1. Freshwater inputs

Major anthropogenic influences on freshwater flow and quality (the Anglesea power station and Painkalac Reservoir) are located at single points relatively low in the catchments. This meant that natural waters could be measured relatively easily at locations upstream of those points and that anthropogenic influences could be assessed both indirectly, by comparisons between points above and below the discharge/extraction points and directly, by measurement at discharge points. There were limited data on natural flows prior to this study, so rainfall was used as a surrogate in long-term comparisons, using annual data. In the case of Anglesea, a 30-year dataset from Alcoa's monitoring program was made available to investigate long-term variability in water quality.

Discharge and water quality in streams above and below the reservoir and power station/mine were measured at time scales from half-hourly (logged flow) to monthly (Table 2.4). Acidic flows originating primarily in the Anglesea catchment were also investigated to determine their potential influence in the estuaries.

2.3.2. Estuarine physical processes

The degree of intermittency of the estuaries was first investigated by recording variation in water levels in Anglesea and Painkalac estuaries, coupled with visual assessments of the degree of connection between the estuaries and the sea (Table 2.4). Examination of early results from this component resulted in a conceptual model with three 'states', or degrees, of tidal influence, described as the entrance being either 'closed', 'perched' or 'tidal'. The validity and transferability of this model were examined using the logged data and a classification-tree approach. This resulted in a series of quantitative, dichotomous rules that separated periods when the estuary was in different states based on the height and variability of water level. These rules were successfully applied to logged data from a short period in Painkalac estuary.

The three-state model provided a framework in which physical, chemical and biological processes and changes could be assessed. Influence on the degree of intermittency, or state, was a major pathway through which freshwater flows could affect the estuaries. Similarly, both wind and sea state were potential drivers of sediment dynamics on the coastline and at the mouths of the estuaries. Because of this, the importance of all three potential drivers of transitions between states was compared, with a semi-quantitative assessment of those drivers likely to have influenced each transition through the three years of the study. Differences in timing of transitions and their associated drivers between Anglesea and Painkalac estuaries were then compared.

A direct effect of variation in water levels in the estuaries was to change the extent of inundation of estuarine substrates. Unlike estuaries with permanent connections to the sea, inundation in intermittent estuaries can vary over a wide range of temporal scales and elevations. This was examined in detail in Anglesea estuary, using logged water level and a digital bathymetric model of the estuary to compare the inundation regimes of estuarine substrate between each of the three states.

Sediment dynamics were investigated primarily in the context of their potential influence on seagrass extent and decomposition. As seagrasses were essentially limited to the lower estuaries, deposition rates and net erosion/accretion of sediments were measured at fixed sites, randomly located in the lower parts of Anglesea and Painkalac estuaries, over the final thirteen months of the study (see Section 2.3.4).

2.3.3. Estuarine chemical processes

The model of estuarine states also provided a mechanistic framework in which the role of freshwater inputs in estuarine water chemistry, including salinity stratification, could be assessed. The sampling design was based on depth profiles at sites along the entire lengths of Anglesea and Painkalac estuaries. Seven patterns of vertical and longitudinal salinity stratification were identified in the resulting data (Section 5.3.1.c) which were then used as the basis for comparisons of salinity structure with flow and hydrologic state. These patterns of stratification were also used in examining other aspects of water chemistry.

Comparisons of water quality among single sites in the lower part of each of the ten regional estuaries (Figure 2.1, Table 2.4) provided a context for the more detailed processes examined in Painkalac and Anglesea estuaries. Where relevant, State and national guidelines for water quality were used to assess potentially negative influences of individual water quality characteristics.

2.3.4. Estuarine biological processes

While the hydrologic and chemical components of the estuaries were compared at a system-wide level, aspects of the biological component also explicitly addressed smaller-scale variability between and within sites in the lower estuaries. Changes in the extent and density of seagrass beds and in the rates of detrital processes in the two estuaries were measured in both Anglesea and Painkalac estuaries (Table 2.4). Potential causes of these changes were assessed using existing knowledge and the physical and chemical data described above. The availability of historical aerial photographs of Anglesea estuary also permitted measurement of changes in the distribution of seagrass beds from 1981 to 2002.

Mapping of seagrass extent encompassed all areas of the two estuaries where seagrasses were present. Initially, this was done by using a differentially-corrected GPS to directly trace all bed edges at Anglesea only and, later, by fixed transects run across each estuary. Fine-scale measures including small-scale changes in the deep edges of beds and estimates of shoot density were used at fixed sites in the lower estuaries (Table 2.4).

Detrital processes were also examined at the fixed sites (Table 2.4). Loss of *in situ* seagrass material in litterbags was used as a comparative measure of biomass transfer from seagrasses to other components of the ecosystems. This rate was dependent on the nature of *in situ* seagrass material present, and this varied both temporally and spatially. Microbial decomposition potential between and within estuaries, sites, depths and times, and between buried wrack and seagrass beds, was assessed using a standard method of cotton strip degradation, adapted from soil science (Boulton & Quinn, 2000). The use of this technique has not previously been reported for estuaries.

3. Freshwater Inputs

3.1. Introduction

The small size of their catchments and stochastic seasonal rainfall result in naturally intermittent fresh water flows into estuaries in the study area. Exceptions to this are Anglesea estuary (which has an anthropogenic perennial flow) and the Barham estuary, with its relatively large catchment at the wet, western end of the study area, which had flowed continuously since at least 1977 (Department of Primary Industries (Vic), 2005). These intermittent flows are typical of many Australian streams and are more likely to be found in association with intermittent estuaries than larger, perennial rivers with sufficient energy to overcome coastal sedimentary processes and maintain an open entrance (Kench, 1999).

The aims of this part of the study were to:

- assess links between rainfall and stream flow;
- quantify the volume and temporal variability of flows from the catchments; and
- determine anthropogenic changes to the volume and timing of flow to the estuaries.

These aims were components of a larger study: to compare the flow regimes of the Anglesea and Painkalac systems in a regional context and to quantify the influences of human activities on those regimes.

In conjunction with flow regimes, the physical and chemical properties of fresh water inputs are an important determinant of the aquatic environment of estuaries. To further characterise fresh water inputs, the following aspects of the systems were investigated:

- physical and chemical characteristics of flows from the catchments, including links with hydrology;
- anthropogenic physico-chemical changes to fresh waters;
- temporal and spatial patterns in historical water quality; and;
- sources of acidic inflows.

Where relevant, results were compared with government guidelines for protection of aquatic biota to provide an indication of their potential biological importance. In the course of this study, it became apparent that acidic flows were an important aspect of water chemistry in the catchment of Anglesea in particular (Section 3.4.1). Data from additional sites throughout the catchments of Anglesea and Painkalac estuaries (Figure 3.2) were used in an investigation of sources of these flows, along with sites on the opposite side of the Otway Range (Section 3.4.2, Appendix D).

3.2. Methods

Surface flow (a term considered interchangeable with discharge herein unless otherwise specified) was obtained from gauging stations and field measurements of instantaneous flow. Flows at gauging stations were measured by either electronic depth logger or daily instantaneous readings of water height on gauge plates. Instantaneous channel flows were mostly measured using an electro-magnetic flow meter (30x1second readings, where possible at 0.4 of the total depth and in three sections across the stream). In some cases, when streams were too small for use of the flow meter, an approximation of flow was obtained by diverting flow to a 1 litre container and timing the rate of fill.

Gauging stations with flow loggers were located in three places during the study period, two sites on Painkalac Creek were being monitored for other purposes and a site immediately above the Anglesea estuary was established for this project. Instantaneous flows were measured intermittently at 19 sites on tributaries of Anglesea River and Painkalac Creek over 84 days during the study period. This report focuses on sites draining major sub-catchments and flowing into the estuaries, while data from other sites are only referred to where necessary. Details of these sampling locations and sampling frequency are in Section 3.2.2 and Appendix B.

3.2.1. Stream Gauging

At the lower end of the Anglesea catchment, five gauging stations were established by Theiss Environmental Services for this project (Table 3.1,

Figure 3.1a). Heights at four of these were recorded on most working days by Alcoa employees between 27/9/1999 and 27/2/2002 (Table 3.2). Details of gauge ratings and flow calculations are given in Appendix A.

Location	**Latitude (deg S)**	**Longitude (deg E)**	**Height (m above MSL)**	**Data type**
Anglesea River	38°23'44"	144°10'59"	1.1[a]	Logged (1/2 hour & Daily)
Marshy Creek	38°23'04"	144°10'45"	~17[b]	Daily
Salt Creek	38°23'17"	144°10'26"	~15[b]	Daily
Ash Ponds	38°23'22"	144°10'46"	4.5[a]	Daily
Mine Overflow	38°23'40"	144°10'44"	7.7[a]	Daily

Table 3.1 Gauging stations in the lower Anglesea catchment established for this study. Heights were determined by a - differentially corrected GPS or b – from 1:25,000 topographic map.

Gauge	**1999 (96)**	**2000 (366)**	**2001 (365)**	**2002 (58)**	**Total %**
Marshy Ck	78	219	257	47	67.9
Salt Ck	78	218	248	47	66.8
Ash Ponds	76	218	253	46	67.0
Mine Reclaim	77	217	225	46	63.8

Table 3.2 Number of daily readings of non-logger flow gauges by Alcoa staff per year. (x) = days in study period for which the gauge was installed in that year.

A logger was installed and maintained by Theiss for this project at a site immediately above the head of the estuary from 26/8/99 to 31/3/02. Flow data were recorded half-hourly at this site. In addition, half-hourly data from two existing gauges in Painkalac Creek were obtained from Theiss. One of these gauges was located at the sampling site above the dam on Painkalac Creek; the other was located below the dam, approximately 1.7 km above the downstream site used in this study. Flow data from above and below the dam were obtained from 26/3/99 to 28/2/02 and 20/1/99 to 31/3/02 respectively.

3.2.2. Water quality

3.2.2.a. Sampling

Two sets of data have been analysed; data collected as part of this study, and a longer, unpublished data series that has been collected at Anglesea by Alcoa of Australia and has not been previously analysed. In Anglesea catchment, the three key sites (Figure 3.1)allowed comparisons between Salt and Marshy Creeks above Alcoa's operations and characterisation of changes to water quality before the Anglesea River enters the estuary. Additional sites (Figure 3.2) were used in the investigation of the source of acid flows. Other comparisons at Anglesea were possible using the data from Alcoa's monitoring program; assessments of the water quality of discharges from the mine and ash ponds are primarily based on these data.

The positions of the three key sites in Painkalac catchment allowed comparisons of water quality between locations above and below the reservoir as well as between waters entering the estuary from Painkalac and Distillery Creeks. A limitation of this design was that detection of effects of the reservoir on water quality depended on that effect being expressed a few kilometres downstream. As part of the investigation of acidic flows, five additional sites were sampled along Distillery Creek (Figure 3.2) and six sites were sampled in an area of similar geology on the opposite side of the Otway Ranges (Appendix D). Historical water quality data from Painkalac Creek were also available from a State Government sampling location at the flow gauge below the dam. Direct flows to the estuaries as stormwater were not examined in this study.

The six key sites and thirteen additional sites in Anglesea and Painkalac catchments were sampled between 1 and 37 times on 81 days between 8/12/1998 and 19/2/2002 (Table 3.3, locations in Figure 3.2, Appendix B). Sixteen of these were in the Anglesea catchment and three were in the Painkalac catchment. Key sites in the Anglesea catchment were just above the estuary and near the bottom of each of the two major tributaries but above mine operations (Figure 3.1a). In Painkalac Creek key sites were just above the estuary in the mainstem and Distillery Creek as well as a site on the mainstem above the dam (Figure 3.1b).

a)

b)

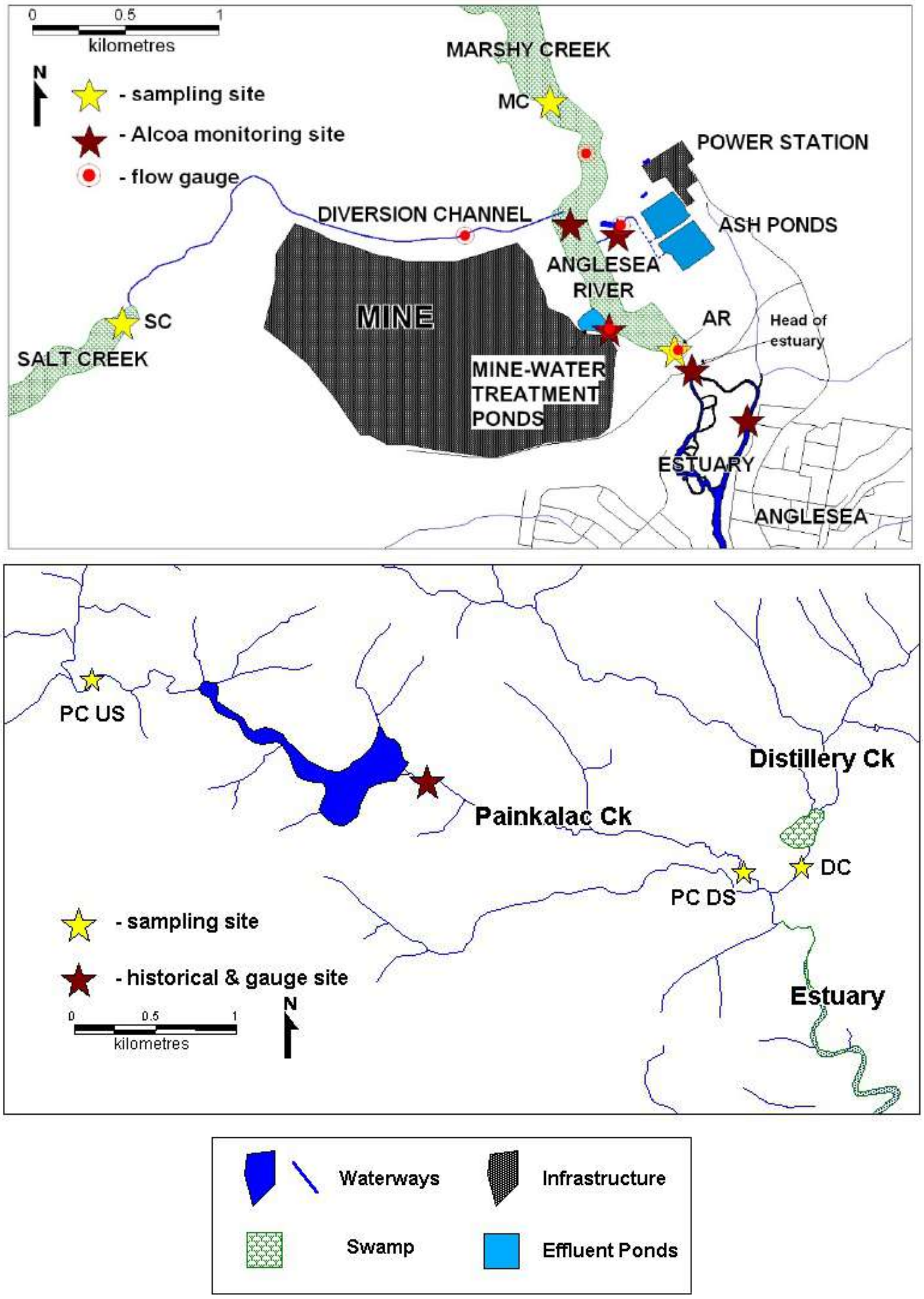


Figure 3.1a). Water quality sampling locations in the lower Anglesea catchment. Data for effluent from the ash ponds and mine are primarily unpublished data from Alcoa's monthly monitoring program. b). Water quality sampling locations in the Painkalac catchment.

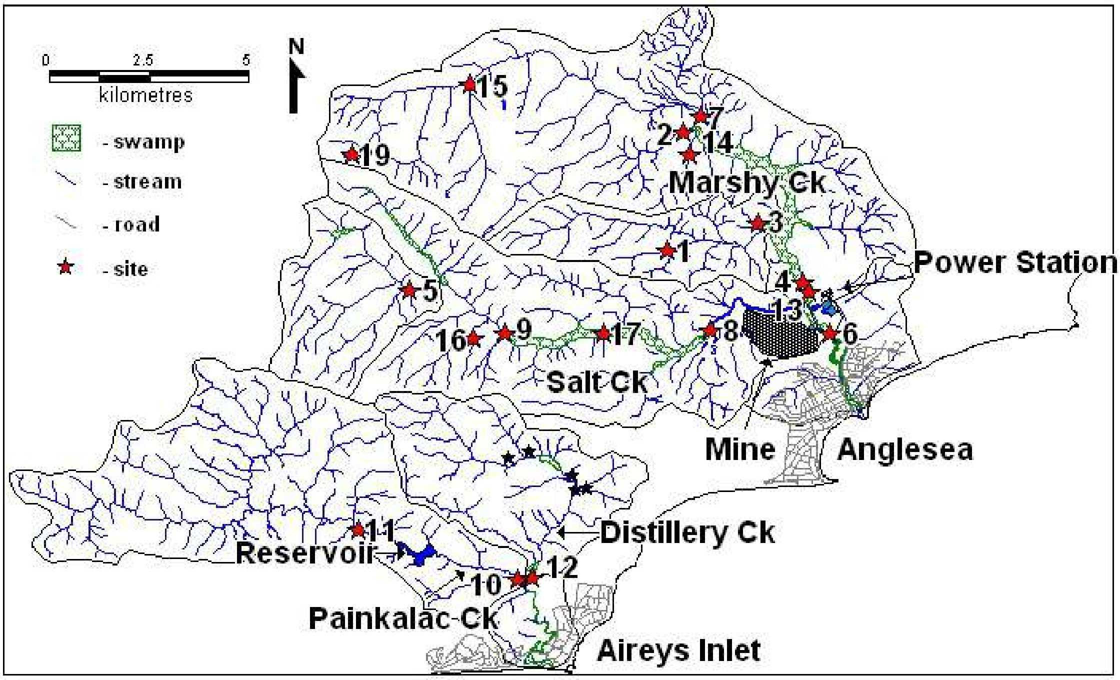


Figure 3.2. Freshwater sampling sites throughout the catchments of Anglesea and Painkalac. One-off sites in the Distillery Creek sub-catchment are shown as black stars.

Some data were collected jointly with Deakin University honours candidates and has been presented in their theses (Meyrick, 1999; Hermon, 2002). Trips and sites where data were shared are shown in Appendix B.

Site no.	Waterway	Location	Times sampled	Times flowing	First date	Last date
4	Marshy Ck	above Alcoa 1	37	28	16/12/98	18/2/02
6	Anglesea R.	above estuary	37	37	8/12/98	18/2/02
8	Salt Ck	above Alcoa	33	12	22/1/99	18/2/02
10	Painkalac Ck	above estuary	39	21	18/5/99	19/2/02
11	Painkalac Ck	above dam	14	13	29/10/99	18/2/02
12	Distillery Ck	above estuary	24	10	18/5/99	18/2/02

Table 3.3. Key freshwater sites sampled as part of this study, with locations and timing of sampling. Sites are also shown in Fig 2.18. Details of additional sites are given in Appendices B and D.

Various combinations of flow, conductivity (reported as conductivity at 25°C), temperature, dissolved oxygen, pH, turbidity, redox potential, suspended solids and nutrients (total P (TP), total N (TN), soluble reactive P (SRP) and the sum of nitrate and nitrite (NOx)) were measured at the above sites. Details of variables measured are given in Appendix B.

3.2.2.b. *In situ* physico-chemical measurements

Two multiprobes were used to measure conductivity, temperature, pH, and dissolved oxygen though the study. A YSI 600XL in the first part of the study (8/12/1998–3/7/2000), following which a Yeokal 611 that also measured turbidity and redox potential was used (29/9/2000-19/2/2002). There were exceptions to this due to meter availability and malfunctions of individual sensors (Table 3.4).

Variable	Meter	Dates
Conductivity/ Temperature	YSI 600XL	24-27/4/2001 16/8/2001
	Yeokal 602 Mk2	3/10/1999 8/4/2000-22/4/2000 11/6/2000 22/8/2000 28-29/3/2001
D.O.	Yeokal O_2 model 57	14/12/1999-19/3/2000 15/9/2000
pH	Hanna Instruments Model H18519	18/2/1999-18/3/1999 18/5/1999-3/10/1999*
	Alcoa lab	16/12/1998 9/3/2000
	YSI 600XL	24-27/4/2001 16/8/2001

Table 3.4. Periods where 'non-standard' instruments were used during the sampling program. * - period ends 20/7/1999 for Painkalac estuary. D.O.=dissolved oxygen

When the Yeokal model 57 meter was used to measure dissolved oxygen, results in mg/L were converted to percentage saturation using equations from Weiss (1970) assuming an atmospheric pressure of 760mm Hg. Meters were calibrated for dissolved oxygen and pH/redox daily when sampling. More stable variables such as conductivity, temperature and turbidity were checked periodically and calibrated if necessary according to manufacturers instructions. Problems with calibration of conductivity were identified for the period of 22/4/1999 to 11/6/2000. Based on water samples collected at these times, correction factors were identified and conductivity, salinity and dissolved oxygen were recalculated using established equations for temperature correction of conductivity to EC 25 and EC15 (YSI Incorporated, no date), calculation of salinity (Fofonoff & Millard Jr, 1983) and calculation of dissolved oxygen (in mg/L and as % saturation: Weiss, 1970; Benson & Krause, 1984).

3.2.2.c. Laboratory analyses

Nutrients

Nutrient analyses (TP, TN, SRP, NOx) were conducted using flow injection spectrophotometry according to accepted procedures documented by the National Association of Testing Authorities (NATA)-accredited Deakin University Water Quality Laboratory. These analytical methods were derived from MDFRC (1994), Hosomi & Sudo (1986) and Benson *et al.* (1996).

Total suspended solids

Total suspended solids were measured by the Deakin University Water Quality Laboratory using the method described in APHA *et al.*(1998) and were reported in mg/L.

3.2.2.d. Alcoa data

Alcoa conduct a NATA-accredited water quality monitoring program in and around the Anglesea site (Figure 3.1a) and have been collecting data monthly for over 30 years. The data were collated by Alcoa and provided for this project. Variables measured and their temporal ranges are provided in Table 3.5.

Parameter	Start date
pH	30/3/1972
Conductivity	13/1/1975
Temperature	14/5/1973
Total Solids	30/3/1972
Suspended Solids	30/3/1972
Turbidity	18/6/1973
Colour	16/12/1974*
Aluminium	24/4/1979
Iron	30/3/1972
Zinc	24/41979
Rainfall	1/1968

Table 3.5. Temporal scope of measurement of variables from Alcoa monthly monitoring. * - some earlier measurements made. Final date of collection for all variables used in this study was 13/2/2002.

Each of the water quality variables shown above was sampled in surface waters at five locations (Figure 3.1a): upstream of the mine at the bridge below the junction of Marshy Creek and the Salt Creek diversion, below the ash pond overflow, below the ponds containing water pumped from the mine, downstream of the site at the Coal Mine Road culvert (upper estuary) and further downstream in the Anglesea River estuary (500m below its head).

The dataset is not continuous and there are omissions for various places, times and parameters. Further details are given in Appendix C.

3.3. Results and discussion: hydrology

3.3.1. Rainfall

There are three rainfall stations currently run by the Bureau of Meteorology in the catchments of the Anglesea River and Painkalac Creek. The station at Anglesea has the longest record (from late 1926). While comparisons of rainfall between Anglesea, the upper Anglesea catchment and Aireys Inlet are possible from 1965 and 1994, respectively, references to historical rainfall in this section use data from the Anglesea station unless otherwise specified.

The period before this study was exceptionally dry. During 1997-99, mean annual rainfall at Anglesea was 140mm below the long-term average of

661mm (1927-2002). This was the driest three-year period since 1967-69 (Figure 3.3). Total annual rainfalls in the three full years of the study increased each year (Figure 3.3); 1999 was dry (with 527mm), rain from May 2000 onwards resulted in a near-average annual total of 689mm, and 2001 was the fifth wettest year in 76 years of records with 838mm. During the study, October 2000 and April 2001 had much higher than average rainfall (220% and 336% of mean respectively: Figure 3.4). These were the 15th and 3rd highest monthly falls on record (n=935).

Rainfalls recorded at Painkalac (Aireys Inlet) and Anglesea were similar during the study (Figure 3.4), both these stations were close to the coast and, with the exception of greater mean rainfall at Anglesea in January, have had very similar monthly rainfalls since 1995.

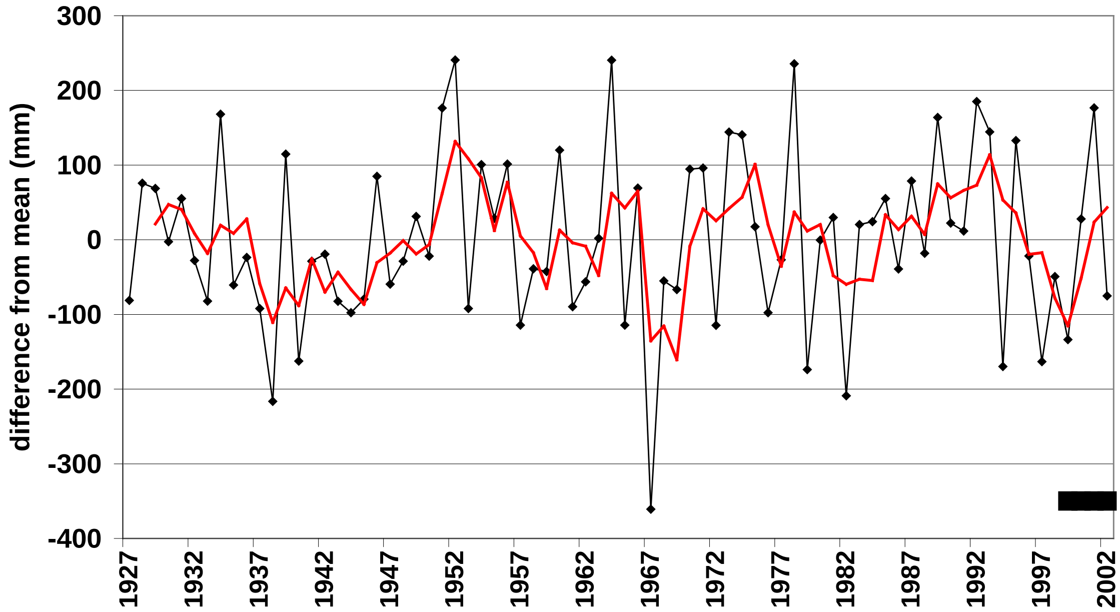


Figure 3.3. Annual residual rainfall (difference from the long-term mean of 661mm) at Anglesea, 1927-2002. The red line links 3 year moving averages. Periods above zero indicate above-average rainfall. Periods below zero indicate below-average rainfall or droughts. Study period is indicated as a bar in the lower right hand corner.

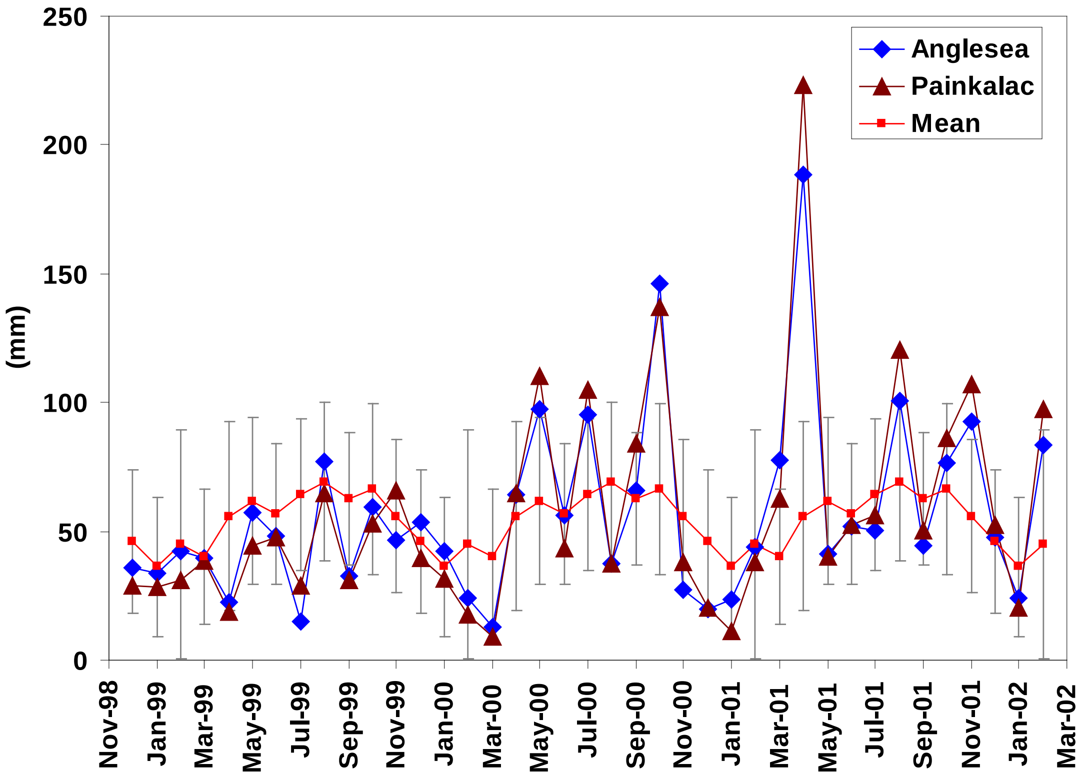


Figure 3.4 Monthly rainfall at Anglesea and Painkalac during the study period. The cyclical 'Mean' series shows the long-term monthly means (± 1 std dev.) at Anglesea between 1927 and 2003.

3.3.2. Flow patterns

Natural stream flows in the catchments of Anglesea and Painkalac estuaries are moderately seasonal, tending to little or no flow from mid-summer to autumn and greater flows through winter and spring. Large flows and periods of no flow, however, have occurred at all times throughout the year and inter-annual variation of these characteristics is pronounced.

During the study flows were typically low or zero in late summer-autumn but varied at other times of the year with less flow in 1999 than in the following two years. For 91% of the time between 26/8/99 and 31/3/02, when both flows were logged, daily flow into the Anglesea estuary was greater than that into Painkalac (Figure 3.5, Figure 3.6, Table 3.6).

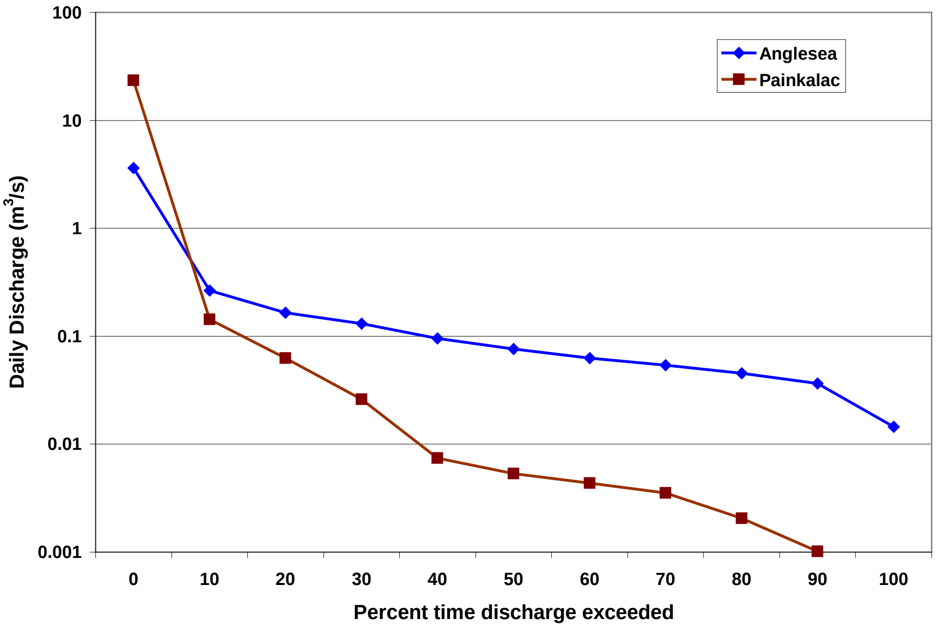


Figure 3.5. Flow duration curves for Anglesea River and Painkalac Creek at gauges above the estuaries during the study. Periods of record are 16/8/1999-31/3/2002 and 22/1/1999-31/3/2002 respectively. Painkalac data are from Theiss Environmental Services.

River	Daily Discharge (m^3/s) Median	10th Percentile	90th Percentile	No flow days
Anglesea	0.077	0.036	0.265	0
Painkalac	0.005	0.001	0.142	58

Table 3.6. Descriptive flow statistics for Anglesea River and Painkalac Creek as measured at gauging locations above the estuaries during the study period. Statistics are based on data recorded between 25/8/1999 to 31/3/2002 at Anglesea (*n*=936) and 21/1/99 to 31/3/02 at Painkalac (*n*=1167).

Variability of flow was greater in Painkalac than Anglesea on several temporal scales (Table 3.7). The mean coefficient of variation for flow in both waterways increased with duration, from days to years. This is due to an increasing number of measurements in each period, serial correlation between half-hourly flow measurements, seasonal changes in rainfall, and intermittent flood and high-flow events.

These differences in variability between estuaries are mostly attributable to intermittent flow in Painkalac Creek and continuous flow in the Anglesea River (see Section 3.3.5). It also appears that flow in Anglesea River was proportionately less variable than that of Painkalac Creek over longer time scales. As the Anglesea gauge had three periods when flow was too high to measure accurately, coefficients of variation were also compared with these dates removed. The ratios between the creeks for this second comparison were similar but smaller in absolute terms.

While partial records exist of historical flows in Painkalac and Anglesea, rainfall records are more consistent and temporally extensive. Given substantial inter-annual variation, the relationship between rainfall and flow was examined and assessed in terms of its usefulness for placing flows during the study period in a longer-term context and for examining the frequency of short-term hydrologic events such as floods.

Despite increasing variability, predictability of flow based on rainfall increased over longer measurement intervals for major subcatchments above and below points of modification (Table 3.8). The exception to this was at Painkalac Creek below the dam, where the annual correlation in the long-term dataset was strongly influenced by proportionally high runoff following the loss of vegetation in the catchment in the large bushfires of 1983.

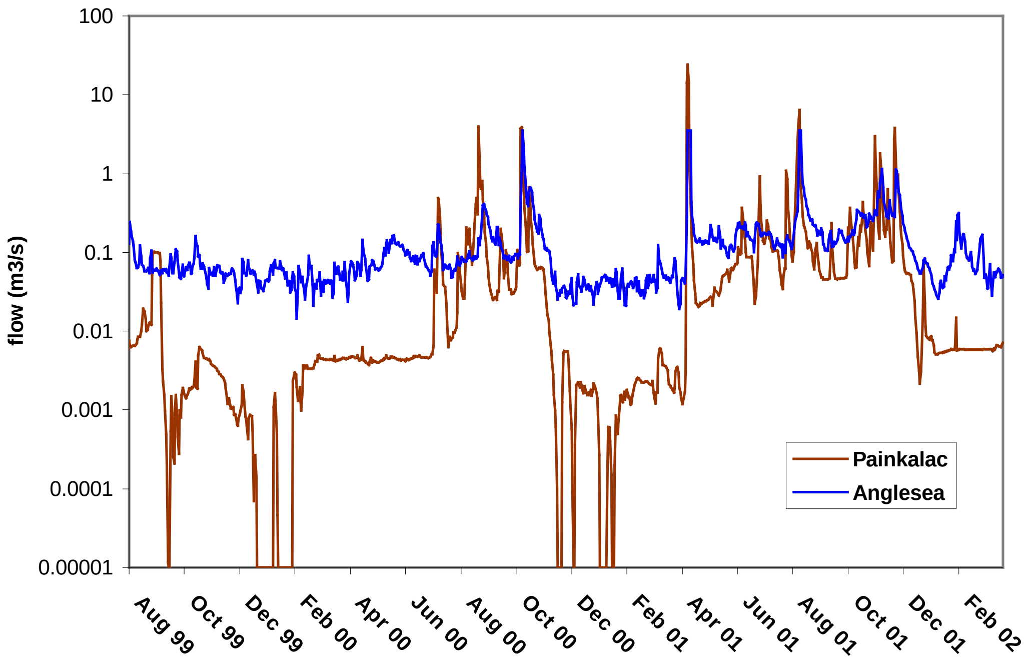


Figure 3.6. Daily flows in the lower sections of Anglesea River and Painkalac Creek, 26/8/1999 to 31/3/2002. Where flow in Painkalac Creek was zero, a flow of $1x10^{-5}$ was substituted to allow flow to be plotted on a log scale.

	Day	Week[a]	Month	Season	Year[b]
Anglesea	8.2%	24.4%	52.3%	90.0%	193%
Painkalac	11.5%	52.1%	114%	243%	596%

Table 3.7. Mean coefficients of variation for half hourly flows at gauging locations above the Anglesea and Painkalac estuaries on five time scales. Anglesea flows > gauge rating were replaced by maximum rated flow. [a] – complete weeks only, [b] -2000 and 2001 only.

Factors including preceding rainfall, absorptivity of soils, groundwater levels and connectivity, slope, aspect, catchment size, temperature, humidity, season, intermittency of streams, vegetation type and anthropogenic changes in flow due to storage, extraction and augmentation of flows may all affect the relationship between rainfall and runoff. At best (using monthly totals for the Painkalac upstream site), 50% of the variation in flow could be accounted for in terms of a general monotonic relationship with rainfall as measured using Spearman's rank correlation coefficient (Table 3.8).

Period	Location	Statistic	Measurement Duration Day	Week	Month	Year
1967-1982	**Salt Ck**	r_s	0.219	0.387	0.540	0.701
		n	5456	818	184	14
1999-2002	**Salt Ck**	r_s	0.129	0.284	0.453	-
		n	570	95	29	(2)
1999-2002	**Marshy Ck**	r_s	0.277	0.398	0.565	-
		n	587	97	29	(2)
1999-2002	**Painkalac Ck US**	r_s	0.323	0.540	0.708	-
		n	1058	153	35	(2)
1980-2002[a]	**Painkalac Ck DS[b]**	r_s	0.231	0.364	0.550	0.248
		n	5528	791	184	14
1999-2002	**Anglesea River**	r_s	0.264	0.392	0.589	-
		n	896	126	30	(2)
1999-2002	**Painkalac Ck DS**	r_s	0.241	0.373	0.567	-
		n	1121	159	37	(2)

Table 3.8. Correlations between rainfall at Anglesea and flow, presented as Spearman's rank correlation coefficient (r_s) for varying periods of measurement. Whole weeks, months and years only were used in correlations, flows greater than rated heights were extrapolated, where flow data were missing for <4 days in a week (Salt and Marshy Creek data), means were used in weekly calculations, means were also used for monthly and annual calculations. Records began and ended at various times in the years shown, [a]: no data from 7/4/1992 to 20/1/1999. [b]: historical comparison for Painkalac creek below the dam (DS) was with rainfall at Anglesea, the comparison during the study period was with rain at Aireys Inlet. Rainfall data provided by Bureau of Meteorology, Commonwealth of Australia, flow data from Painkalac Creek provided by Theiss Environmental Services.

3.3.2.a. Anglesea

Tributaries

Throughout the study, Marshy Creek had consistently greater flow than Salt Creek, which also had longer dry periods and flowed more intermittently than Marshy Creek. This was in direct contrast to the relative flows observed in 1981-2 by Atkins and Bourne (1983). Daily flows measured from October 1999 show that during floods, however, Salt Creek had higher flows than Marshy Creek leading to greater total flows for 2000 and 2001 (Figure 3.7a-c). Median flows (Table 3.9) reflect the pattern described above, with lower values for Salt Creek reflecting the 'flashier' nature of flows in this tributary.

Stream	Oct-Dec, 1999	2000	2001	Jan-Feb, 2002
Marshy Creek	0.010 (74)	0.020 (217)	0.017 (257)	0.002 (47)
Salt Creek	0.000 (74)	0.000 (217)	0.011 (248)	0.000 (47)
Anglesea River	0.056 (92)	0.070 (366)	0.139 (365)	0.068 (58)

Table 3.9. Median flows for Anglesea River and tributaries in m^3/s. Numbers in brackets indicate number of days for which there were data.

Flows ranged from zero to >>0.40m^3/s and >6.32m^3/s (the maximum discharges of each rating table) in Marshy and Salt Creeks, respectively. Marshy Creek had a similar flow pattern in each year of the study while Salt Creek had very little flow until September 2000, after which it had a similar flow pattern to Marshy Creek (Figure 3.7).

Anglesea River into the estuary

Flow to the estuary from Anglesea River was continuous for the study period (Figure 3.6). Instantaneous discharge peaked at a conservatively estimated 24.3m^3/s in a flood on 23/4/2001. There were two other floods later in 2001 where the ratings table of this gauge was exceeded. Unless otherwise stated, flows during these periods have been substituted with maximum reliable extrapolated flow from rating tables provided by Theiss Environmental Services.

During times when there was no flow from either creek upstream of Alcoa, there was a smaller, more consistent flow (mean=0.047 m^3/s, coefficient of variation=34.5%) than during times when at least one of the tributaries was flowing above Alcoa (mean=0.176 m^3/s, coefficient of variation=185.1%). The mean flow for times of no upstream flow was exceeded for 78% of the study period, while the mean flow during times with upstream flow was exceeded for only 18% of the study period (Figure 3.5). This is illustrated in the patterns of flow in Figure 3.6 and Figure 3.7 with greater downstream flows in winter/spring 2000 and autumn to early summer 2001 matching peaks in flow in the tributaries shown in Figure 3.7a)-c).

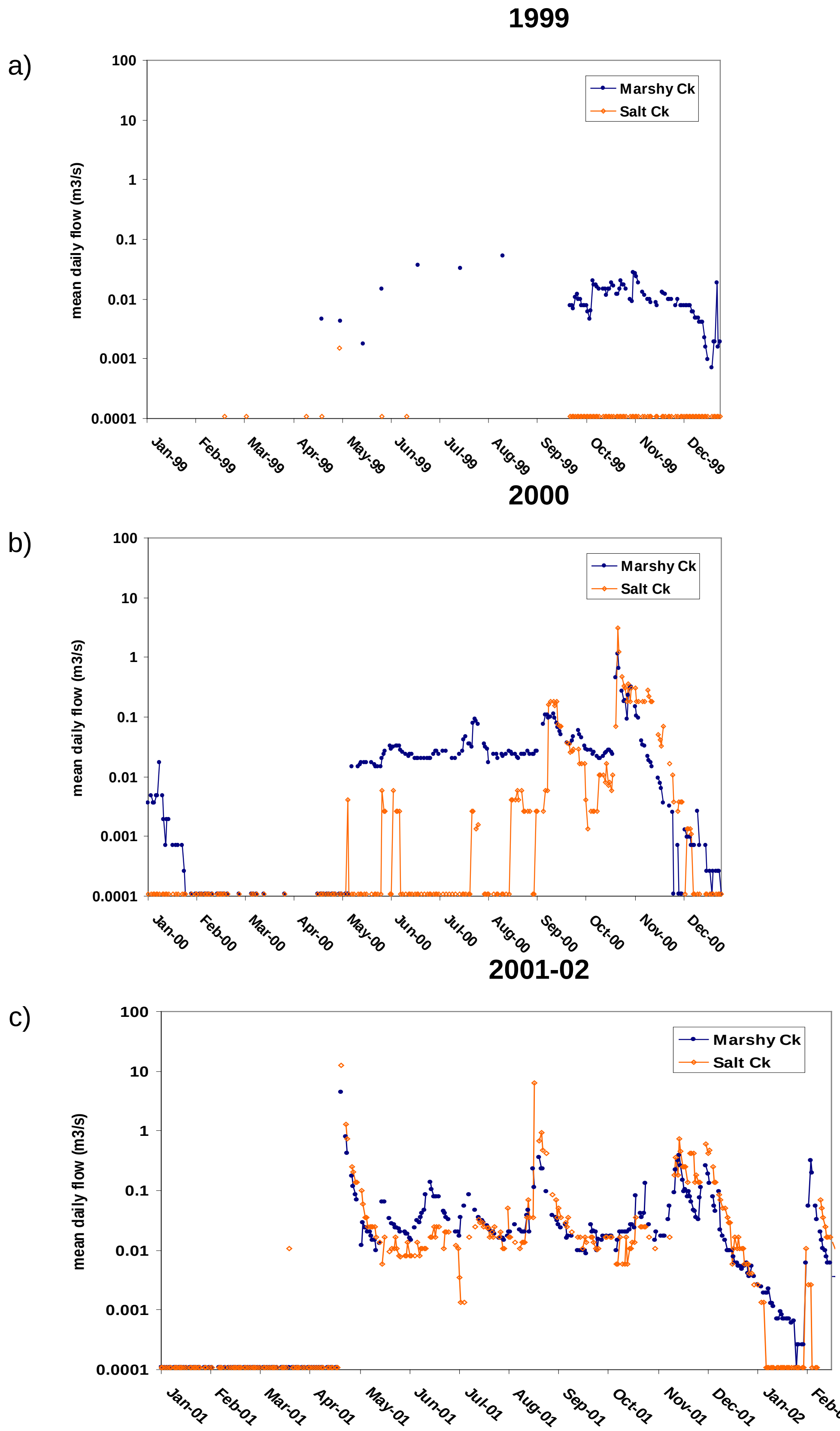


Figure 3.7a-c. Plots of daily flows for Marshy and Salt Creeks , February 1999 to February 2002. Zero values were substituted with 1×10^{-4} m^3/s to allow presentation on a log scale. Data prior to September 1999 were collected at water quality sites immediately upstream of gauging locations. Points connected by lines represent consecutive daily measurements.

Overall, the pattern of discharge into Anglesea estuary was more consistent than that into Painkalac estuary, with variability in flow increasing with volume during times of natural flow and Alcoa flows providing a relatively constant base flow during other times. Natural flows became larger and more frequent towards the end of the study period, including several floods from October 2000 onwards (Section 3.3.3).

3.3.2.b. Painkalac and Distillery Creeks

In contrast to the Anglesea River, both Painkalac Creek and Distillery Creek flowed intermittently into their estuary during the study period and Painkalac Creek flowed more often than Distillery Creek, particularly before the flood of April 2001. While Anglesea River flowed for more days than Painkalac Creek, flow was also greater in Anglesea River 91% of the time when flows were measured at gauging stations in both waterways (Figure 3.5, Table 3.6). This is consistent with the size and nature of the catchments as well as the differing anthropogenic alterations to flow (Section 3.3.5). The catchment of Painkalac Creek is 35% of the size of the Anglesea River catchment with similar rainfall, leading to less runoff. It is also steeper and without the peat swamps typical of the waterways in the Anglesea catchment. These have a dampening effect on Anglesea's hydrograph that, for example, was particularly evident in the first flood period in September 2000 (see Section 3.3.3.b).

While the gauge below the dam was useful as an indicator of flows to the estuary, there were two particular sets of circumstances in which it was expected that flows to the estuary would substantially differ from those recorded at this gauge location. First, at times when there was natural flow, but no release from the dam, the upstream gauge was considered a useful indicator of potential flow from other tributaries (but see Section 3.3.4). Second, at times of very little to zero flow below the dam, flows sometimes did not extend to the estuary. This second case was consistent with data from this study and also with five months of data from 1974 in which periods of flow were recorded at a gauge located at a site now below the dam but not at a gauge immediately above the estuary. From comparisons of flows measured simultaneously at the gauge below the dam and at the site above the estuary (see Appendix A), it appears that in 1999 daily flows below 0.0015m^3/s at the gauging station did not enter the estuary as surface flow. In 2000, 2001 and 2002 this 'threshold' flow at the gauge was 0.004m^3/s. When gauged flows below these thresholds were substituted with zero values, there were 13 periods where there was no freshwater flow to the estuary from Painkalac Creek. These periods ranged from one day (16/5/2000 and 22/5/2000) to 96 days (16/12/2000-21/3/2001) (Table 3.10).

Period	Dates	Duration (days)
1	22-28 Jan 1999	7
2	30 Jan–8 Feb 1999	10
3	12–19 Apr 1999	8
4	3 Oct-3 Nov 1999	32
5	7-8 Nov 1999	2
6	9-26 Dec 1999	18
7	28 Dec 1999-14 Mar 2000	78
8	10-12 May 2000	3
9	16 May 2000	1
10	22 May 2000	1
11	28 Nov-8 Dec 2000	11
12	16 Dec 2000-21 Mar 2001	96
13	27 Mar-21 Apr 2001	26

Table 3.10 Periods of no flow in Painkalac Creek above the estuary based on modified data from the flow logger below Painkalac reservoir (see Appendix A).

Flows from Distillery Creek were observed for a ~two-month period from 1/10/2000 and then again from until the flood that started on 22/4/2001. Apart from a brief period in November 2001, Distillery Creek flowed throughout the remainder of the study although always at a lesser volume than Painkalac Creek.

3.3.3. Floods

During the period of continuous flow measurement (21/1/99-31/3/02 for Painkalac; 26/8/99-31/3/02 for Anglesea), six distinct floods occurred (Figure 3.8). Except for the September 2000 event, all occurred in both Anglesea River and Painkalac Creek. Dates of the floods and their maximum flows (where available) are shown in Table 3.11.

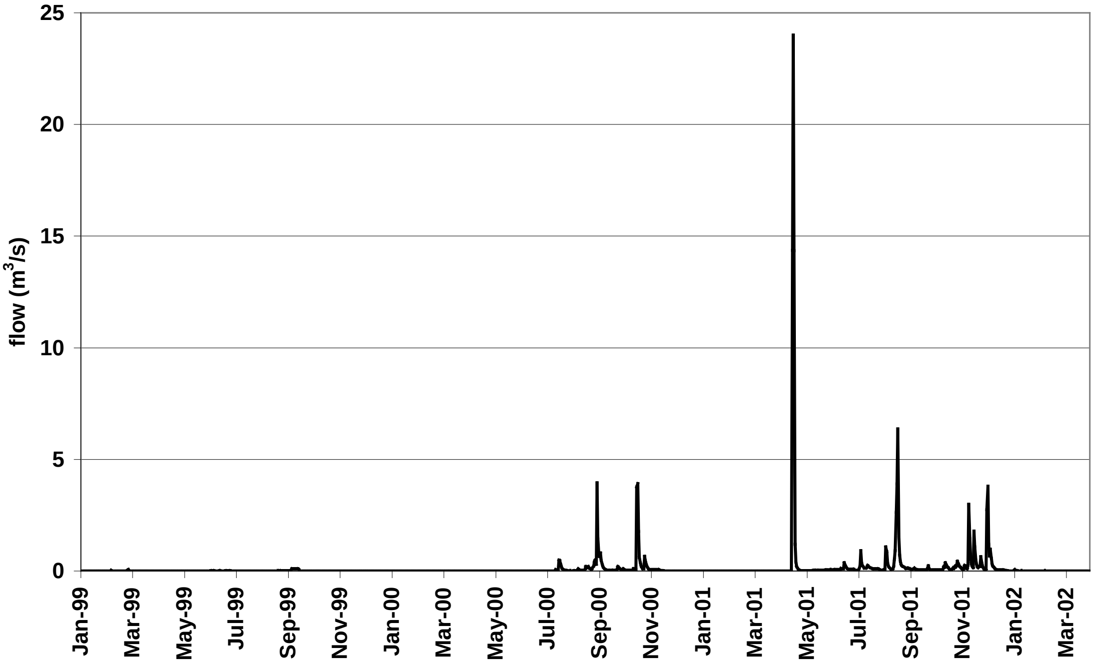


Figure 3.8 Mean daily flow in Painkalac Creek downstream of the reservoir over the study period, illustrating the timing and magnitude of floods. This gauge was the only one of the three in the area to be reliably rated for all measured flows.

Dates	7/9/00-21/9/00	23/10/00-31/10/00	22/4/01-30/4/01	19/8/01-30/8/01	11/11/01-22/11/01	3/12/01-10/12/01
Anglesea	0.425 (0.39m)	>3.61* (0.92m)	>3.61* (2.42m)	>1.62* (1.24m)	1.36 (0.65m)	1.51 (0.68m)
Painkalac below dam	5.51	6.64	45.6	17.5	4.62	9.73
Painkalac above dam	5.79	7.52	> 9.22	>9.22	5.70	>9.22

Table 3.11 Flood events and maximum half hourly flows in m^3/s. * Different ratings were used for the Anglesea gauging station over time (see Appendix A). Maximum rated flow is shown (as >'x') for floods with peaks greater than maximum rated heights and peak flood heights at the Anglesea gauge are shown in brackets.

The floods of April 2001 were the largest of the six and washed away trees and banks in both waterways as well as the gauging structure in Anglesea. Roads were cut and standing waves were observed in the lower estuaries. The sandbars at the entrances of both estuaries were deeply scoured and water levels in the estuaries were subsequently much lower and fully tidal (see Chapter 4). In Painkalac below the dam, where maximum rated flows were not exceeded and long-term data exist, a volume equivalent to 89% of the long-term annual average flow was recorded over the nine days of the flood (Table 3.12), with 39% on 23 April, when the flood peaked.

Proportion of:	Flood dates 7/9/00-21/9/00	23/10/00-31/10/00	22/4/01-30/4/01	19/8/01-30/8/01	11/11/01-22/11/01	3/12/01-10/12/01
Long-term annual mean	0.152	0.180	0.887	0.287	0.161	0.172
Annual total	0.304	0.360	0.461	0.149	0.083	0.090

Table 3.12. Proportion of annual flows at the Painkalac downstream gauge recorded during floods in the study period. Long-term annual mean (5246ML) was calculated from the years 1980-1991, 2000 and 2001. Total annual flows were 2617ML and 10,092ML in 2000 and 2001, respectively.

Combined, the flows during flood periods in Painkalac were equivalent to 67% of the annual flow in 2000 and 78% of the total flow in 2001 over periods of 24 and 41 days, respectively.

3.3.3.a. Historical context

In 20 years of recorded flow in Painkalac Creek (1974-92; 1999-2002), only one flood, in September 1976, exceeded the flow recorded for Painkalac in April 2001 (by 30%). The 1976 flood was also the largest on record for Salt Creek (data from June 1967 to November 1982). It had a peak daily flow of 8.68m^3/s, a peak instantaneous flow of ~60-80m^3/s and overflowed the diversion channel into the open-cut mine. One other recent flood, which also flooded the mine in November 1995, was probably a similar size or larger than the April 2001 flood but no flow data were available for this time.

Large rainfalls were associated with the flood of April 2001 and occurred mainly between 21-23 April, with totals of 163mm, 173mm and 205mm falling at Anglesea, upper Anglesea and Aireys Inlet respectively. Rainfall in this short period was equivalent to the annual difference from long-term average rainfall in that year. Falls between 9am on 21 April and 9am on 23 April were the only two consecutive days (apart from two days in 1933) included in the highest 20 daily falls recorded since 1927 (n=28,269). An exceptionally high daily fall was also recorded in October 2000, the 51.4mm recorded to 9am on 24/10/2000 being ranked 20th of all recorded daily falls.

Based on 3-day totals from large rain events, there have been 10 other times since 1927 when similar or larger floods than those of April 2001 could have occurred, including those of 1976 and 1995 (Table 3.13). Of these eleven events, nine occurred in the warmer half of the year (between November and April) while the other two occurred in August and September.

Max. rainfall (mm) over: 3 days	**1 day**	**Dates of Event**	**Duration (days)**	**Total rain (mm)**
180.6	165.1	15-19 Feb. 1954	5	181.6
163	***62.4***	***19-24 Apr. 2001***	***6***	***169***
132	94.6	1-5 Feb. 1990	5	134.4
131.4	74.7	29 Nov.-6 Dec. 1933	8	150.7
123.7	87.5	19-23 Aug. 1951	5	129.2
117.6	106.7	5-7 Feb 1973	3	117.6
111.5	82.6	16-19 Feb. 1928	4	129.3
108.4	52.3	17-22 Jan. 1946	6*	119.8
107.2	39.4	20-28 Apr.1960	9	134.9
106.2	*58*	*20-24 Sept 1976*	*5*	*111*
96.2	*46*	*3-8 Nov. 1995*	*6*	*98.4*

Table 3.13 Rainfall events at Anglesea with 3 day falls equal to or greater than those during known large floods (shown in italics). The rain event associated with the large flood during the study period is in bold. Duration refers to consecutive days of rain except '*' for which there was one day of no rain in the period.

Of the three historical rainfall events for which there are flow data (Feb 1973, Sept 1976 and Feb 1990), a flood was only recorded for September 1976 (in both Salt Creek and Painkalac Creek before construction of the dam).

A likely explanation for the lack of floods associated with the February 1973 rain event is absorption into dry, permeable soils at the end of summer, particularly the peat beds of Salt Creek. Flow data for 1990 were only available from below the reservoir, where a small peak of $0.053m^3/s$ was recorded at the gauge below the dam. In this case, it is difficult to tell if there was a flood upstream of the dam. Although there are no flow records for the November 1995 rain event, evidence of an associated flood is provided by the record of mine flooding at that time. This inconsistent flow response to rainfall is particularly relevant given the frequency of these large rain events in late summer/autumn, increasing the difficulty of predicting floods and associated effects such as acidic flows based on rainfall (Sections 3.4.2, 5.3.5).

3.3.3.b. Flood hydrographs: 1999-2002

During each flood, flow in Anglesea River peaked later and at a lesser volume than Painkalac Creek and then receded at a slower rate. Lag times ranged between 9 hours (for the second peak of the largest flood) to ~4.5 days in the first flood monitored, when there was a very minor peak in Anglesea River (Table 3.11). The duration of lags tended to be inversely related to the size of the flood, and smaller for second peaks where flood discharges were bimodal. The differences between hydrographs may be related to the relatively flat, swampy floors of the Salt and Marshy Creek valleys; the prevalence of peaty soils in those watercourses; the greater steepness of the Painkalac catchment; lags in rainfall between catchments or a combination of any of these differences.

Peaty soils in the Anglesea catchment, dry from the preceding drought, are thought to have absorbed much of the first flood, in September 2000, which was only seen as a slight increase in flow in the Anglesea River relative to the other floods (Table 3.11). This spate was the first substantial flow from Salt Creek for at least 21 months and probably longer, based on rainfall patterns prior to the study. While small, this initial flow was very important in terms of water quality, was strongly acidic and contained high concentrations of metals (Sections 3.4.1, 3.4.2, 5.3.5).

No estimates of peak flows at the upstream site in Painkalac Creek were possible for the three largest floods but, for periods where data is available, the flood hydrographs are very similar to those from the downstream gauge, indicating that releases during large floods were consistent in timing and magnitude to natural flows. This was also the case at Anglesea, where flows from Alcoa were a very small proportion of flood flows (Section 3.3.5).

3.3.4. Low flow

3.3.4.a. Anglesea and Painkalac

Variation in the duration and timing of no-flow periods between Anglesea River, Painkalac Creek and Distillery Creek appeared to be mainly associated with catchment characteristics, including interactions with groundwater. Of the six main locations where flow was observed, Distillery Creek had the longest periods of no flow followed by Salt Creek. Both these creeks had a similar pattern of flow throughout the study period, with little or no flow until winter/spring 2000 (Figure 3.9). These sub-catchments are adjacent to each other on either side of the eastern boundary of the Painkalac Creek catchment (Figure 2.2).

In the Anglesea catchment, Marshy Creek flowed more consistently than Salt Creek. This may be related to an uncapped artesian bore in the lower Marshy Creek catchment, 3.3 km upstream of junction, in the Edwards Creek sub-catchment (38°22'18"S 144°10'44"E). This bore flowed at a mean rate of 0.0027m^3/s through the study, but was not always realised as surface flow at the Marshy Creek site downstream. Despite the longer periods of no flow at the bottom of Salt Creek, Breakfast Creek flowed into upper Salt Creek with a similar frequency to Marshy Creek downstream flows. Natural flows immediately above Alcoa ceased in the summers of 1999/2000 and 2001/2002, although the section of Anglesea River above the estuary continued to receive discharge water from Alcoa.

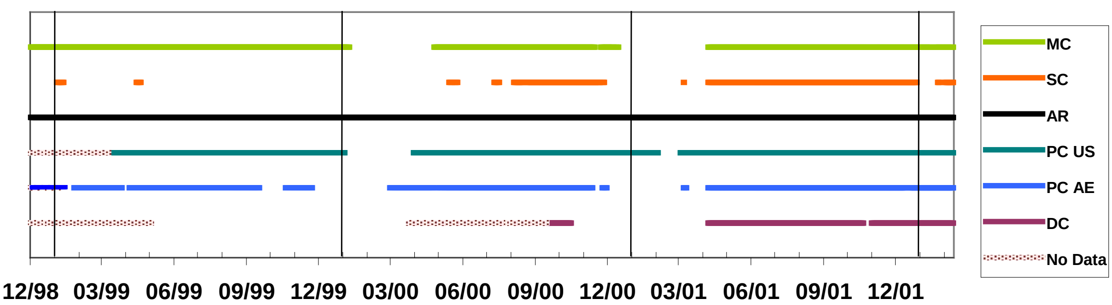


Figure 3.9 Periods of flow in tributaries above and below the dam and power station for the estuaries of Painkalac Creek and Anglesea River. Codes are, MC: Marshy Creek above Alcoa, SC: Salt Creek above Alcoa, AR: Anglesea River below Alcoa, PC US: Painkalac Creek above the reservoir, PC AE: Painkalac Creek above the estuary and DC: Distillery Creek above the estuary. Data for: MC and SC are daily readings;

AR and PC US are logged half-hour flows at gauges; PC AE are logged half-hour flows at the gauging point modified for low flows by observations above the estuary (see Sect 3.3.2.b); and DC are ~monthly observations.

At locations where historical data exist (Painkalac Creek and Salt Creek), it appears that there were more days of no flow during the study period than in previous years (Figure 3.10). This was particularly the case in Salt Creek in the first part of the study even compared to the period of comparable rainfall in 1967-69. The relatively longer dry periods in Salt Creek compared with Marshy Creek are also different to earlier observations. In 1981 Marshy Creek was reported to have had a small discharge compared with Salt Creek which only appeared to flow 'after heavy local rainfall' (Atkins & Bourne, 1983).

During the present study, Painkalac Creek had longer dry periods below the dam than historically (Figure 3.10). This was the case when compared to both the periods before and after construction of the dam (historical data from pre- and post-1979, respectively). The proportion of no-flow days above the estuary that were derived from the below-dam gauge are consistent with, if slightly greater than, pre-dam dry spells measured at a gauge that was located near the current above-estuary site (Figure 3.10).

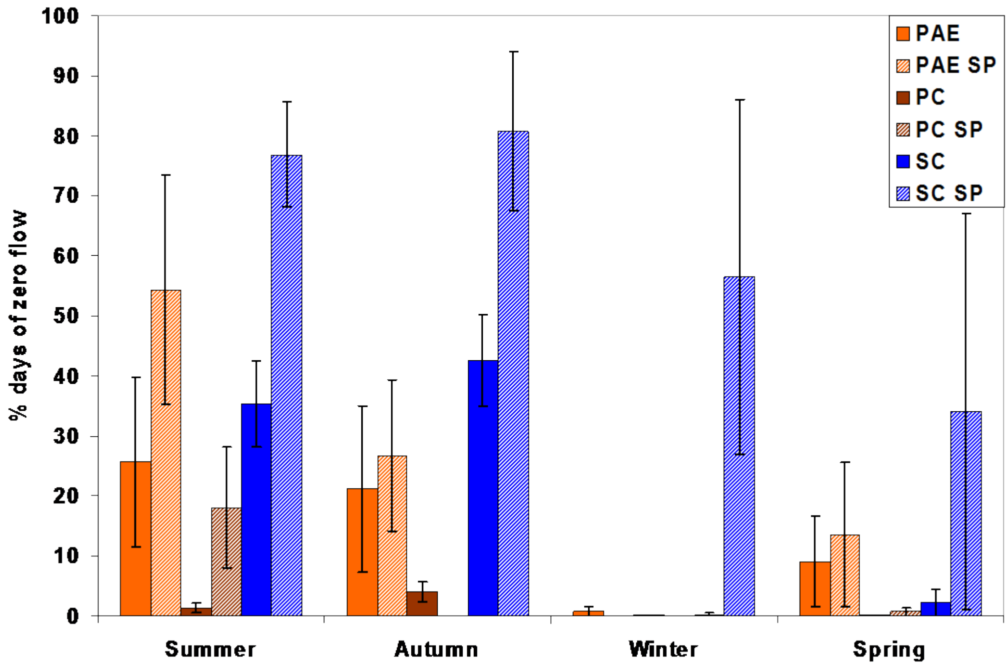


Figure 3.10 Seasonal mean (± s.e.) percentages of no-flow days for Painkalac and Salt Creeks for annual historical data and equivalent annual percentages for the study

period. PAE: Painkalac Creek above Painkalac estuary, measured 11/2/1967-1/10/1974; PAE SP: above Painkalac estuary in the study period, derived from data at gauge below dam and manual measurements, 21/9/1999-31/3/2002; PC: Painkalac Creek below dam 27/3/1974-5/4/1992; PC SP: Painkalac Creek below dam in the study period, 21/1/1999-31/3/2002; SC: Salt Creek, 9/6/1967-28/11/1982; SC SP: Salt Creek in the study period, 16/12/1998-27/2/2002.

3.3.4.b. Regional patterns

Four of the other estuaries studied had gauging stations in their catchments (maintained by Theiss Environmental Services) although only two of these were operational during the study period (Table 3.14). These stations were at either end of the broader study area; in the eastern branch of the Barham River and in Thompson Creek.

Across the region intermittency increased and median discharges decreased from west to east (though over varying periods). Anglesea River was an exception to this pattern, with anthropogenic influences resulting in continuous and supplemented flows. Effects of these properties on estuarine water quality are discussed in Chapter 5.

Estuary	Distance above estuary (km)	Period of data	Median flow (m^3/s)	% dry days	Location
Thompson Ck	11.1	27/7/94-31/3/02	0.0010	23.3	mainstem
Anglesea River	*0.15*	*25/8/99-31/3/02*	*0.077*	*0.0*	*mainstem*
Painkalac Creek	*2.0*	*21/1/99-31/3/02*	*0.005*	*4.96*	*mainstem*
Erskine River	6.3	1/12/89-13/7/97	0.095	2.62	mainstem
St Georges River	2.5	22/4/70-22/2/89	0.13	0.596	mainstem
Barham River	7.2	13/10/77-31/3/02	0.31	0.000	East branch only

Table 3.14 Summary of flow data available for tributaries of other estuaries in the study. Data provided by Theiss Environmental Services.

Despite a higher degree of seasonality in rain in the western part of the study area, patterns of flow in Barham River and Thompson Creek during the study were similar with regard to seasonality of flow, but flows in Thompson Creek were much more variable (Figure 3.11). Higher rainfall in the west generally, combined with the larger catchment area and regulated flows of the Barham River in particular are likely contributors to the differences in the hydrographs. Despite these differences, periods of high and low to zero flow for both Thompson Creek and the Barham River match those of Anglesea River and Painkalac Creek, with the obvious difference of a base flow in Anglesea.

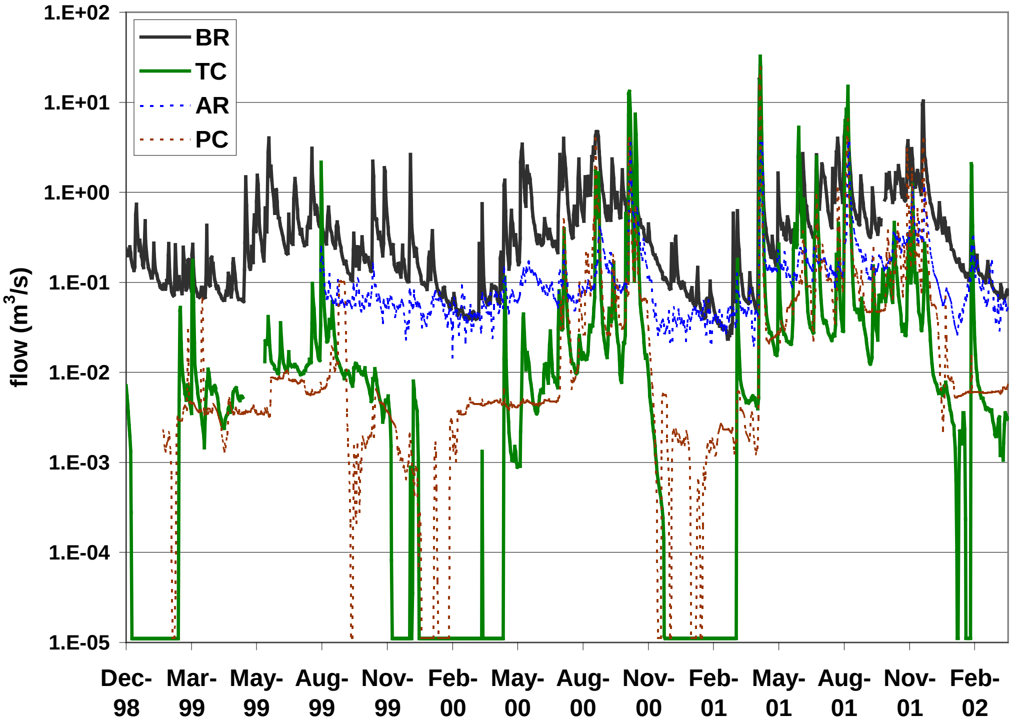


Figure 3.11 Daily flows at gauging stations in Barham River (BR), Thompson Creek (TC), Anglesea River (AR) and Painkalac Creek below the dam (PC) between 1/12/1998 and 31/3/2002. Zero flows have been substituted with 1.1×10^{-5} to allow presentation on a log scale.

3.3.5. Modifications to flows

Freshwater flows reaching Painkalac and Anglesea estuaries are both modified by human infrastructure. The Alcoa operations on the Anglesea River contributed a relatively constant amount of groundwater-sourced process water to the River year-round. In addition, an uncapped artesian bore in the lower Marshy Creek catchment contributed to the flow of that tributary, but not enough to create a perennial flow in the downstream reach above Alcoa (see Section 3.3.4). The dam on Painkalac Creek reduced downstream flow and changed patterns of stream flow, particularly at lower rates. These modifications were most apparent during times of low natural flows and resulted in the greatest relative differences in inflow between the estuaries during times when there was no flow into Painkalac estuary and an entirely anthropogenic flow into Anglesea estuary.

The flows from Alcoa contributed a significant proportion of flows to Anglesea , representing at least 50% of flow to the estuary for 88% of the study period (Figure 3.12). Proportionally, the contribution of Alcoa's discharges to total flow was smallest during floods, when they represented less than 1% of total flows into Anglesea estuary for relatively short periods of time (<4% of all flow records). In contrast to Painkalac, the timing of flows into Anglesea was not modified.

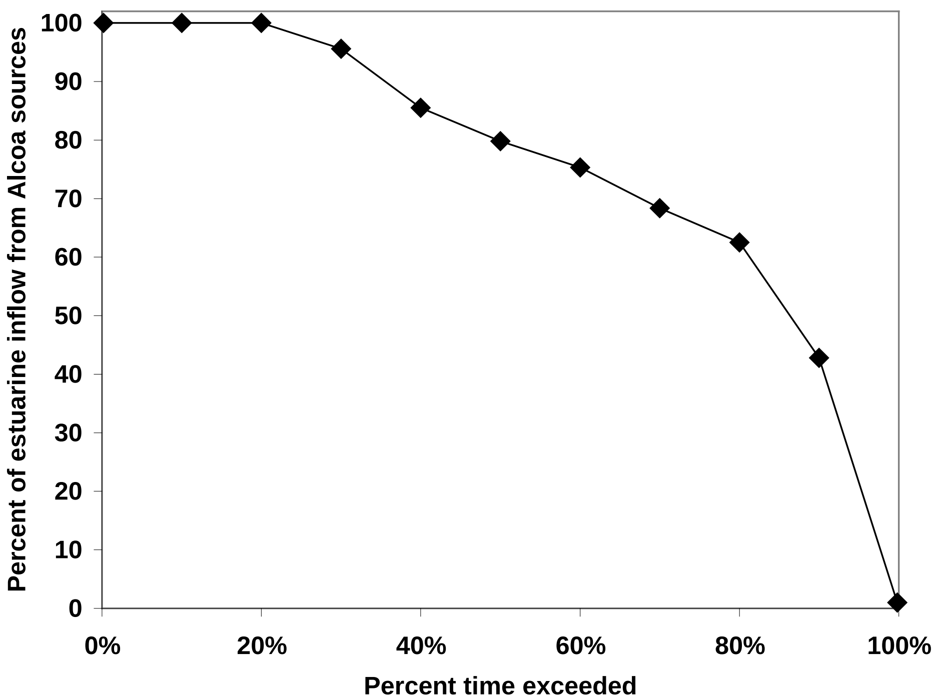


Figure 3.12. Exceedance curve of daily Alcoa flow as a percentage of daily total flow (calculated using the difference between flows at upstream and downstream gauges). Alcoa flows during floods represented as a nominal 1%. Data collected from 27/9/1999 to 27/2/2002, *n*=541.

Around 175,000m^3 of water was extracted from Painkalac Creek each year (Barwon Water, 2003b), while an annual mean of approximately 4,065,000m^3 flowed through the reservoir. Before mid-2000, several peaks in flow upstream of the Painkalac Reservoir were not reflected below the dam, resulting in a considerably reduced variability of flow (Figure 3.13). After that time, flow patterns were quite closely matched for all flows above 0.005m^3/s. Despite this change, periods of no flow downstream were shorter and more frequent than those upstream and tended to be earlier in the summer. For a substantial part of the first three summer/autumns of the study, it appears that the dam also contributed a small artificial flow to Painkalac Creek. The change in the relationship between upstream and downstream flows at this time may have been related to either; changes in management practices, with upstream flow gauging allowing flow patterns to be more closely matched or, to the filling of the dam and subsequent proportional releases.

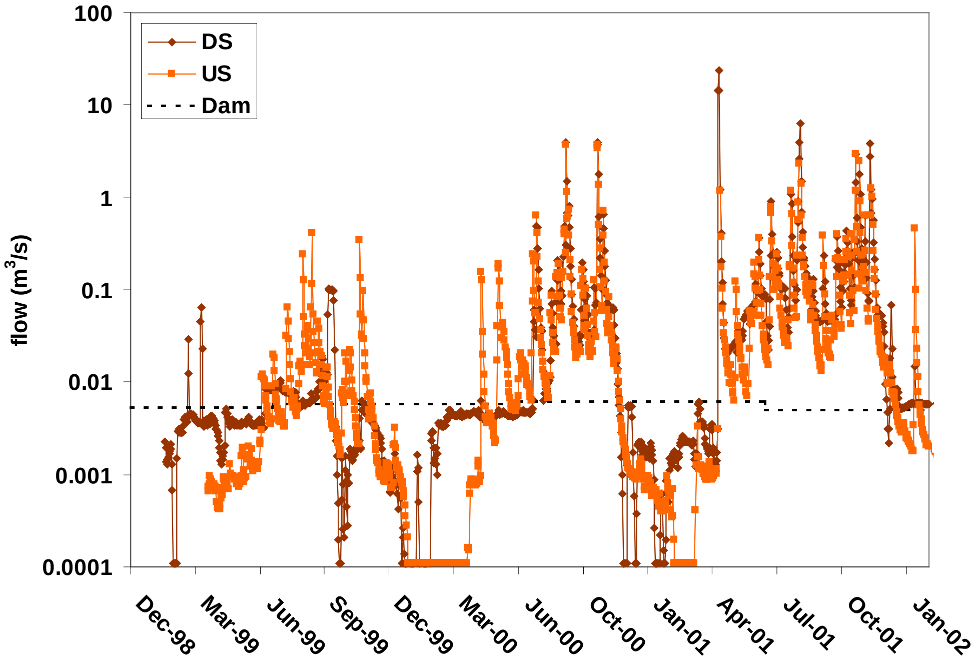


Figure 3.13. Flow above and below the reservoir on Painkalac Creek. US: upstream gauge, DS: downstream gauge, Dam: approximate annual average off-take by financial year (lower in 2001/02 than previous years) (Barwon Water, 2003b).

3.4. Results and discussion: water quality

Differences in geology and vegetation between the catchments of Marshy, Salt and Distillery Creeks and those of the upper reaches of Painkalac Creek were reflected in different patterns of water quality through times of high and low discharge. Overall, conductivity, turbidity and concentrations of dissolved oxygen, suspended solids, total aluminium and total zinc tended to increase with flow. pH and concentrations of nutrients and total iron tended to be greatest at lower flows. Temperature was essentially seasonal. No differences in water quality were evident due to the reservoir on Painkalac Creek but increased conductivity, pH and concentrations of and nitrate/nitrites were related to Alcoa's discharges as were the patterns of supply and transport of metals to the estuary. Increases in temperature and dissolved oxygen at Anglesea were associated with a change in the morphology and vegetation of the waterway through Alcoa's site.

3.4.1. pH

One of the most distinctive features of freshwaters in the Anglesea and Painkalac catchments, with high potential to affect their estuaries, was the low pH in three of their four main sub-catchments. It is also a characteristic that has been substantially altered by Alcoa's discharges into the Anglesea River during the last 30 years.

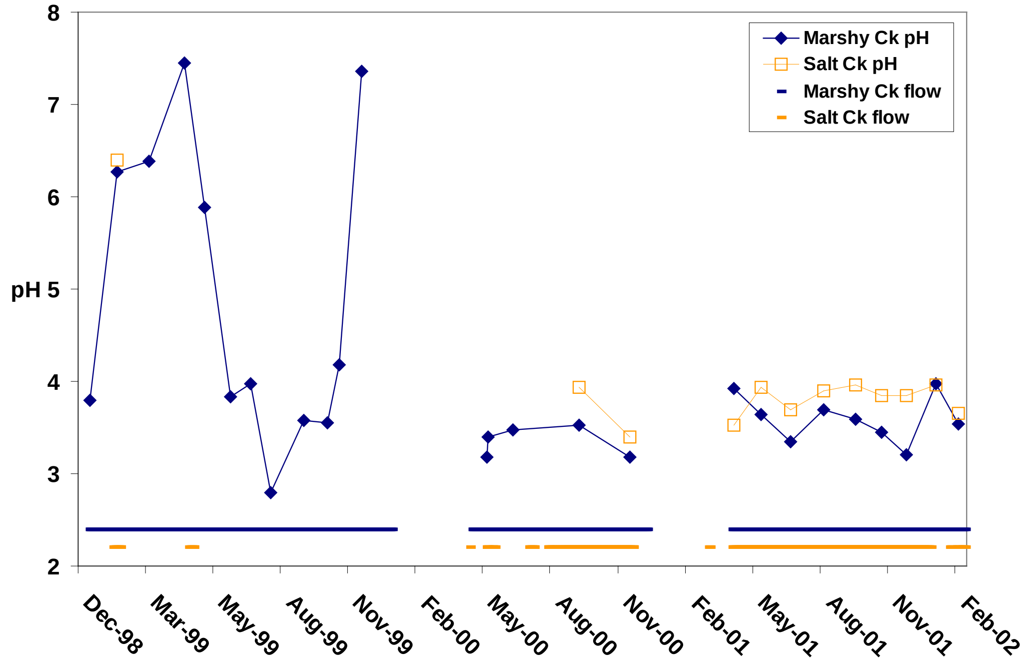
3.4.1.a. Catchment processes

A combination of geology and hydrologic, chemical and biological processes resulted in an overall distribution of pH that was bimodal but varied among sites and times, particularly with flow. Waters in the Anglesea catchment and Distillery Creek were typically low in pH, ranging from 2.8 to 7.5 (Figure 3.14). Neutral pH in these waterways occurred only in association with very low flows in 1999. On these occasions, a rust-coloured floc appeared at the Marshy Creek site, most likely iron precipitates and/or colonies of iron-associated bacteria. At times of low pH, waters were typically very clear. These associations are further examined in Section 3.4.4 and Appendix C.

Waters of the mainstem of Painkalac Creek were mostly neutral and did not differ greatly between the site above the dam and the site just above the estuary (Figure 3.14b). Some variation from this pattern was seen with pH decreasing to 5.2 at the downstream site following the floods of April and

August 2001. Two pH values above 9.0 were recorded in late 1999, one at each Painkalac Creek site. These values coincided with times of very low flow and are most likely associated with increased carbonate concentrations (products of algal respiration) in the slow-moving waters.

a)



b)

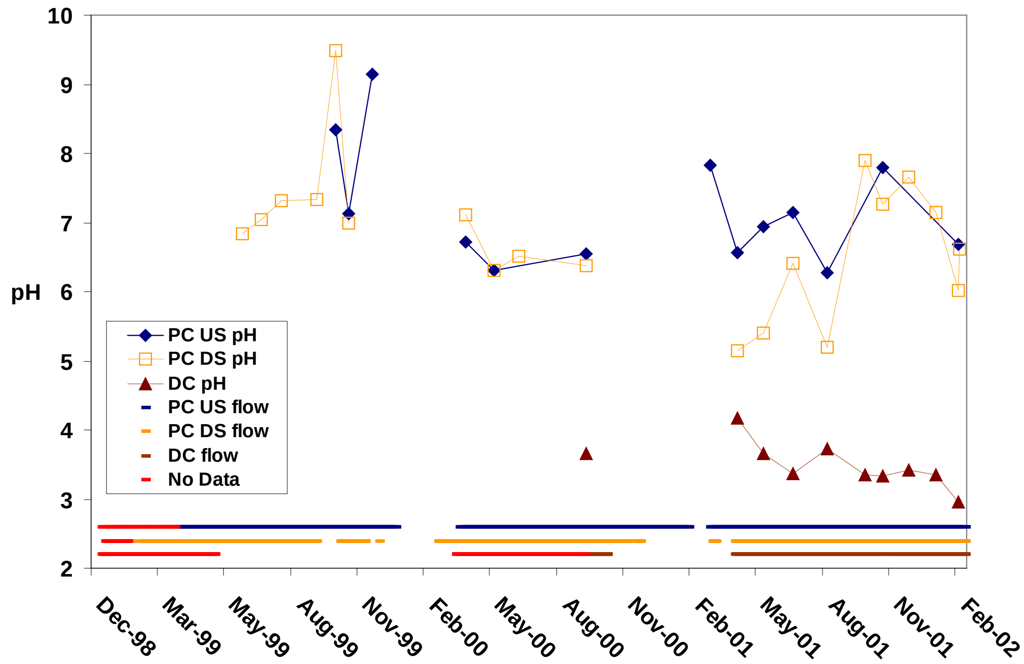


Figure 3.14. pH and periods of flow in: a) tributaries of the Anglesea River above Alcoa; and b) at the three sites in the Painkalac catchment. Bars at the bottom of the figures indicate times when the creeks were running. PC US: Painkalac above dam, PC DS: Painkalac Creek above the estuary, DC: Distillery Creek above the estuary. PC DS flow is estimated periods of flow based on the gauge below Painkalac Reservoir (Sect. 3.3.2.b).

Potential reasons for the difference in pH between Distillery and Painkalac Creek are discussed in detail in Appendix D; however, several attributes separate Distillery Creek from the rest of the Painkalac catchment. A small area of *Melaleuca* swamp, similar to those of the Anglesea catchment, is found in Distillery Creek. Distillery Creek also has the same outcropping geological units (the Eastern View Group) as the majority of Anglesea catchment unlike the catchment of Painkalac Creek (see Appendix D). It is likely in these areas that acid generation is associated with oxidation of sulphur-bearing minerals, e.g. for pyrite:

$$\mathrm{FeS_2} + {}^{15}\!/_{4}\,\mathrm{O_2} + {}^{10}\!/_{4}\,H_2 0 \rightarrow \mathrm{FeOOH} + 2\mathrm{SO_4}^{2-} + 4\mathrm{H}^{+}$$

The lower pH measured at the downstream Painkalac site with floods may also be related to flows from the relatively small proportion of these units that outcrop in the Painkalac Creek sub-catchment. Alcoa's long-term monitoring data from the confluence of Salt and Marshy Creeks above the power station showed a pattern of change in pH with flow consistent with that recorded during the study period. A bimodal distribution of pH was evident, with neutral pH occurring in 12 of 30 summers, particularly in the drier years from 1994-1999 (Figure 3.15). The relocation of the upstream site to a location with more accessible, or ponded, water in times of low flow may also be responsible for the seasonal pattern from the early 1990's becoming obvious, as no summer samples were taken in most of the preceding years, except for a period between the late 1970's and early 1980's. Similarly, historical data from Painkalac Creek suggest that pH during the study was typical of longer-term patterns. Mean monthly pH immediately below the dam from October

1976 to April 1987 was 7.3. The only measurement of substantially acidic (pH<6.0) waters during this period was a pH of 5.2 recorded during a flood (State Government data: Department of Primary Industries (Vic), 2005).

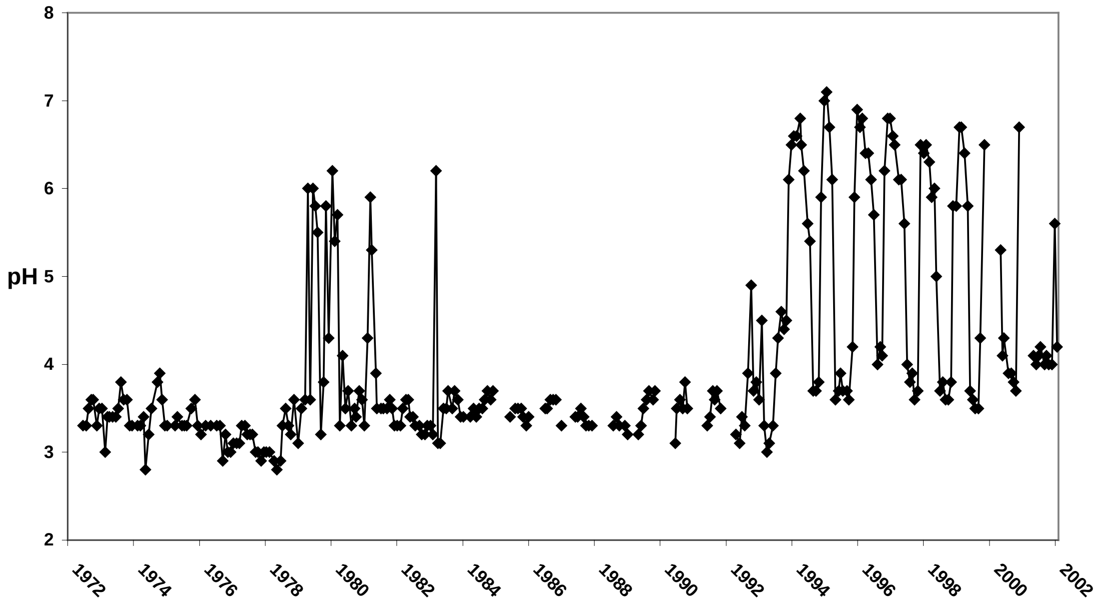


Figure 3.15. Long-term monthly pH of the Anglesea River upstream of Alcoa. Data from Alcoa.

In the 10 years for which flow data were available for Salt Creek (1972-1982), higher pH (>5) was associated only with flows below 0.037m^3/s whereas low pH (<5) was associated with a range of flows up to 1.27m^3/s (pH data from Alcoa of Australia, flow data provided by Theiss Environmental Services). A lack of corresponding flow data for Marshy Creek meant that only a partial comparison was possible.

3.4.1.b. Anthropogenic changes in pH

Sites above and below Painkalac dam had similar pH, except for the high flow period from April to August 2001 (Figure 3.14b). This change was likely due to flows from tributaries entering the creek between the two sites, mainly in reaches between the dam and the site above the estuary (Figure 3.1b).

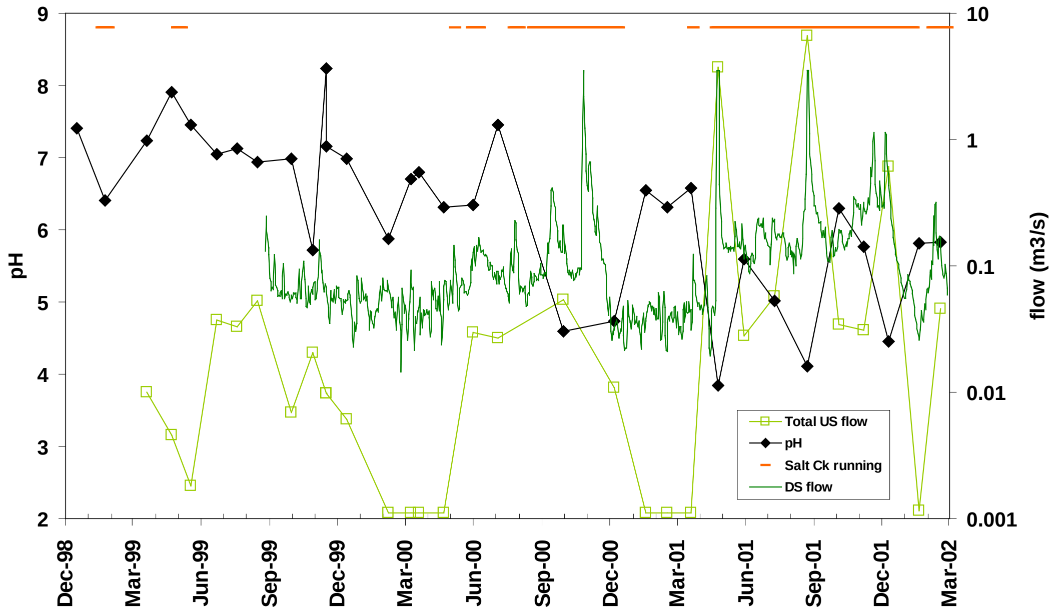
The main effect of Alcoa's inputs on the pH of the Anglesea River is due to waste water discharged through the ash ponds of the power station. Coal ash is typically alkaline and has a large component of soluble material (Mudd

& Kodikara, 2000; Daniels *et al.*, 2002), although pH of discharges from the ponds at Anglesea may be modified by mixing of acidic waters from the mine. Flows from the ponds have been almost entirely alkaline since 1986, following nine years of intermittently acidic pH (unpublished data, Alcoa of Australia). During this study, the pH of discharges from the ash-ponds ranged from 6.7 to 9.3. The flow from the mine reclaim ponds was more intermittent than that from the ashponds. It was also more variable in terms of pH (3.6 to 7.2), most likely due to the practice of treating the acidic effluent from mine de-watering with alkaline ashpond water before release into Anglesea River.

The pH of Anglesea River below Alcoa was generally between 6 and 8 from the start of the study until September 2000 from when, coincidentally with increased flows, it ranged between 3.8 and 6.6 (Figure 3.16). The lowest five pH values were all associated with flood events described in Section 3.3.3.

At times when there was natural flow upstream, pH below the mine was significantly less acidic than the least acidic of Salt and Marshy Creeks (*t*-test, $p=1.95\times10^{-7}$, $n=24$). The difference in pH ranged between 0.5 and 4.1 with two exceptions. During the largest flood (of April 2001), fresh, acidic (pH = 3.9)waters flushed the estuary, and on 22 January 1999, when upstream flows were neutral (pH = 6.4). In both cases, the difference in pH was less than 0.1. This difference depended on the pH, buffering capacity and proportional flow from all sources. Examples of this are discussed in Section 3.4.2.

Figure 3.16. pH of waters entering the Anglesea estuary plotted with upstream (US)



and downstream (DS) flow rates against time. Periods when Salt Creek was running are indicated at the top of the figure. Zero flows have been replaced with 0.0011 to allow presentation on a log scale. Peaks in downstream flow are curtailed at a nominal 3.5 m^3/s where the ratings table of the gauging point was exceeded. Peak flows from upstream are estimates.

Long-term pH data for waters downstream of Alcoa are available for a site at the head of the estuary. While the site was not used in this study due to potential estuarine influences, patterns of pH between 1978 and 1998 were generally consistent with those observed in this study. A period of high pH at the end of the series coincides with a series of dry years while a period of low pH in the 1980s coincided with low pH releases from the ash ponds. For the period in which flow data were available for Salt Creek, there was an inverse relationship between log flow and pH at the head of the estuary. In this period, pH only fell below five at flows greater than 0.006m^3/s and a clearly bimodal distribution of pH, similar to that of upstream waters, was evident (Figure 3.17).

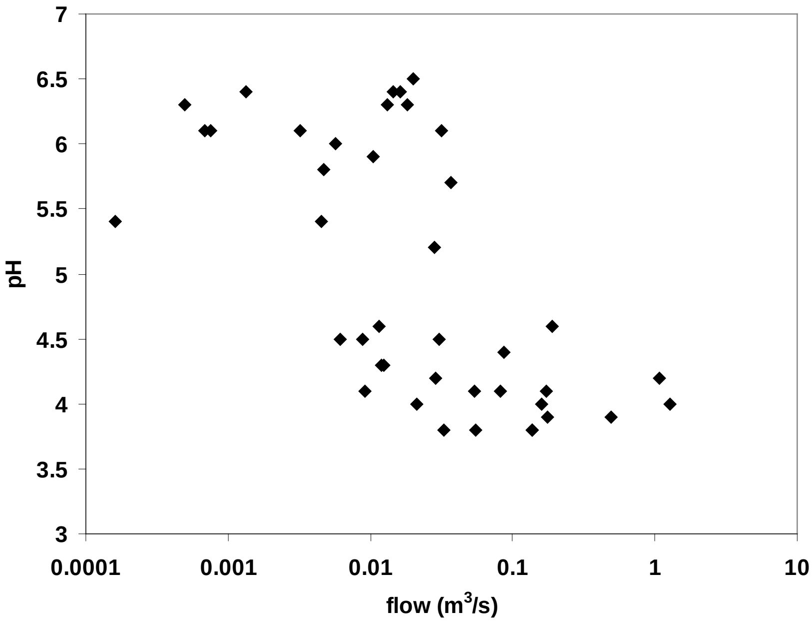


Figure 3.17. Salt Creek flow vs. pH in surface waters at the head of Anglesea estuary, monthly data, 1978-1982. Only flows > 0.0001 m^3/s are shown.

3.4.2. Estuarine acidification events and metals

Examples of acidification of the Anglesea estuary were recorded in late 2000, and at least twice in 2001. During these times the estuary had a pH below five for most of its length. This acidification was not recorded in the Painkalac Creek estuary at any time during the study, although low pH was recorded occasionally at upstream sites.

The events in late 2000 were associated with first substantial flows from Salt Creek for at least 2 years. A fish kill and precipitation of a milky white substance that caused the estuary to have a vivid aqua-blue colour were observed at this time. Both events recorded in 2001 were associated with floods. These events are examined in detail in Appendix E, and a full description of pH in the estuaries is given in Section 5.3.5.

The first estuarine acidification was associated with the first substantial flows from Salt Creek during the study period (during September 2000: Figure 3.7) and, based on rainfall, for some years. Flows on 12 and 13/9/2000 were $0.162 m^3/s$ and $0.185 m^3/s$, respectively, in contrast to the preceding 22 months at least, during which the maximum flow had been $0.006 m^3/s$. These flows continued for five days and then gradually decreased with flow from Marshy Creek becoming dominant from 25/9/2000. Flow from both creeks increased with a flood on 26/10/2000, following which Marshy Creek subsided faster than Salt Creek. Blue colouration of the estuary was again observed on 15/11/2000. The two events in 2001 did not result in such vivid colouration, largely because acidic freshwater flushed all brackish water from the estuary and neutralisation occurred in Bass Strait (Section 5.3.5).

While pH was low in all streams, high concentrations of aluminium (Figure 3.18) and zinc, along with low $Cl^-:SO_4^{2-}$ ratios in Salt and Distillery Creeks (see Appendix E) indicate acid generation due to oxidation of sulphide minerals (van Breemen, 1992). The gradual decreases in metal and sulphate concentrations following the initial flow of Salt Creek were consistent with an initial 'slug' of ions being released in the first substantial flow in this sub-catchment for some time. These high concentrations of metals and acids were most likely produced over time by a wetting and drying cycle of sulphidic sediments that transported dissolved metals in acidic waters to streambeds in the Salt Creek catchment where metal and sulphate compounds remained until re-dissolved in the flood of September 2000. This mechanism is discussed by Hermon (2002) and in Appendix D. It is also similar to observed mechanisms for acid sulphate soils in northern New South Wales (Callinan *et al.*, 1992; Wilson *et al.*, 1999) and the Northern Territory (Hart *et al.*, 1987). It is important to note that this event may not necessarily be caused by a large rain event but is dependent on the positioning of the water table in relation to the cumulative rainfall and previous periods of flushing and oxidation of soils (Wilson *et al.*, 1999).

Alcoa's long-term data (Appendix D, Appendix E) suggest that, while the very high aluminium concentrations recorded in 2000 are unlikely to have occurred since 1979, there have been many lesser events where elevated concentrations of metals have entered the estuary. Figure E.5 in Appendix E illustrates the seasonal pattern of aluminium concentrations above and below Alcoa as well as large inter-annual variability.

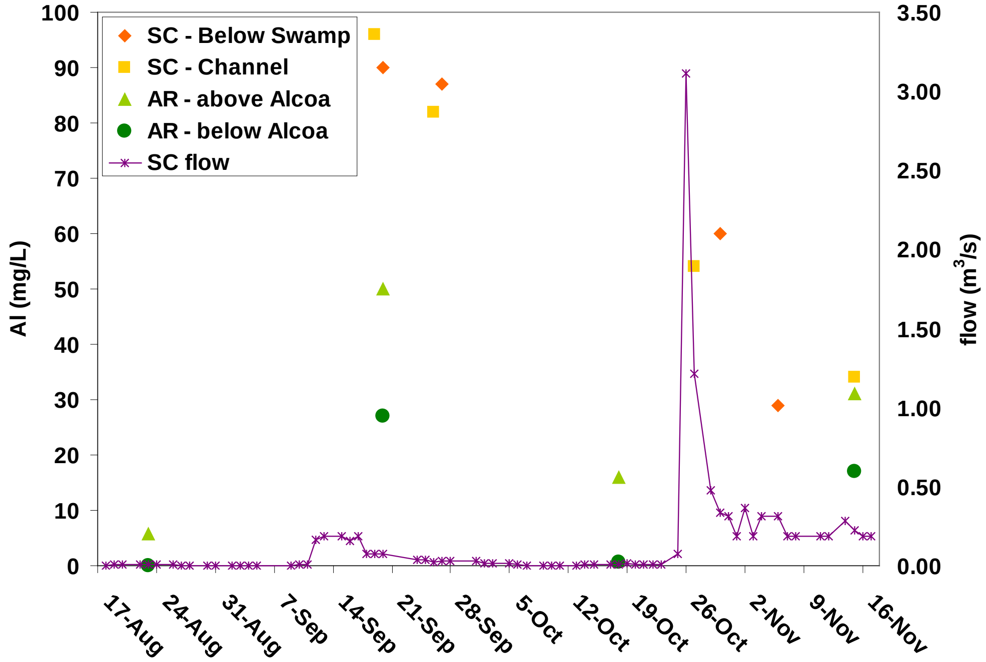


Figure 3.18. Aluminium concentrations during the 2000 acidification event at four locations from the bottom of Salt Creek (SC) through the mine diversion channel to above the Anglesea River (AR) estuary. Data from Alcoa and Environment Protection Authority.

3.4.3. Other variables

3.4.3.a. Conductivity

In the first half of the study period, conductivity in Marshy, Salt and Distillery Creeks ranged between 1100μS/cm and 2100μS/cm when they flowed. In the second half of the study, conductivity decreased with flow to values as low as 400μS/cm in floods. This range was consistent with Alcoa's long-term monitoring data from 1975 to 1998. Conductivities in Painkalac Creek both above and below the reservoir ranged between 20μS/cm and 500μS/cm throughout the study period. The higher conductivities in the early part of the study period were close to or above State guidelines (EPA Victoria, 2003b), but were within the more elevated range typical of lowland rivers in western Victoria (ANZECC & ARMCANZ, 2000).

Although naturally saltier than Painkalac Creek, discharges from the ashponds have further increased the conductivity of waters flowing into Anglesea River since their commissioning. Highest conductivities occur at times when the discharge represents the entire flow of the river, with dilution increasing with upstream flow (as seen in Figure 3.19 from April 2001 onwards). When there were natural flows, mean conductivity at the site below Alcoa was 2500µS/cm greater than that of the saltier upstream tributary.

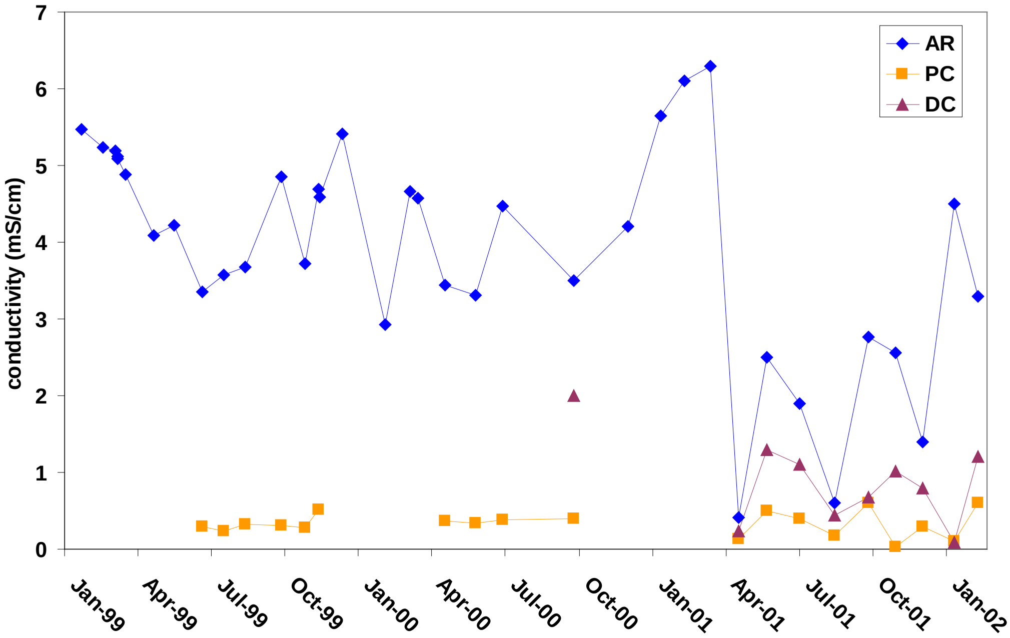


Figure 3.19. Conductivity of waters entering the Anglesea and Painkalac estuaries. AR: Anglesea River, PC: Painkalac Creek and DC: Distillery Creek.

3.4.3.b. Temperature

Temperature of sites in both catchments showed a seasonal trend, ranging between 7 and 21^{o}C. No difference in temperature was measured between upstream and downstream sites in Painkalac, nor is it evident in comparisons of historical data from below the dam before and after construction (data obtained from Victorian State government: Department of Primary Industries (Vic), 2005). Temperature upstream of Alcoa tended to be greater than in the Anglesea River below Alcoa. This was most likely related to reduced shading of the waterways through the Alcoa site as, unlike at some power stations, there is essentially no thermal pollution of waters at Anglesea due to the use of a recirculatory cooling system.

3.4.3.c. Dissolved oxygen

Dissolved oxygen concentrations were periodically low at all sites, none of which complied with the 25th percentile of 85% saturation specified as a trigger level for dissolved oxygen in State water quality objectives for rivers and streams in the region (EPA Victoria, 2003b). Of the six sites, Marshy Creek and Distillery Creek consistently had low concentrations of dissolved oxygen (Figure 3.20a and b). This may be related to the shaded, swampy nature of the streams above the sampling points, both of which had a large amount of detrital matter and thus potential for oxygen demand, in combination with limited light for photosynthesis. Dissolved oxygen concentrations at all sites tended to be lower as flows approached zero; however, the hypoxic conditions seen at the Marshy and Distillery Creek sites suggest that these places may become unsuitable for aquatic fauna (ANZECC & ARMCANZ, 2000), restricting passage at times of low flow. Oxygen concentrations below Alcoa were greater than those above Alcoa on 21 of 24 times measured while no difference was evident between sites above and below the reservoir (Figure 3.20).

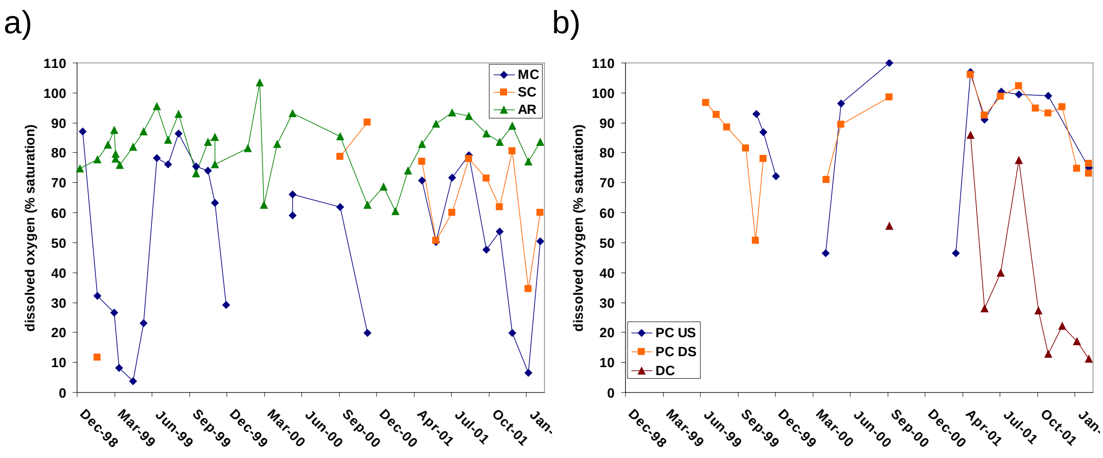


Figure 3.20. Percent saturation of dissolved oxygen at a) Anglesea and b) Painkalac sites during the study period. MC: Marshy Creek, SC: Salt Creek and AR: Anglesea River below Alcoa. PC US: Painkalac Creek upstream of dam, PC DS: Painkalac Creek above the estuary and DC: Distillery Creek.

3.4.3.d. Nutrients

Concentrations of nutrients were measured between February 1999 and February 2000 at the Marshy Creek, Anglesea River and two Painkalac

Creek sites. No samples were taken from Salt or Distillery Creek as there was little to no flow in either during this time. Both total phosphorus and total nitrogen concentrations were highest during low flows. Seventy-fifth percentiles for total nitrogen at all sites were above or close to the State objective for the region of 600μg/L (EPA Victoria, 2003a) except for Anglesea River where it was half that concentration. Despite having the lowest concentrations of total nitrogen, the Anglesea River site had clearly the greatest concentrations of available nitrogen as expressed by the combined concentrations of nitrates and nitrites (NO_x). While there is no State guideline for this parameter, the mean concentration of 65μg/L at this site was well above the mean upstream concentration (16μg/L) as well as the trigger value of 40μg/L for lowland rivers in south-east Australia (ANZECC & ARMCANZ, 2000). The increase in NOx between the Marshy Creek site and the below Alcoa site is strongly suggestive of an anthropogenic input, most likely from groundwater pumped for use as process water in the power station, but potentially also from stormwater inputs between these sites.

All concentrations of soluble reactive phosphorus (SRP) were below the quantifiable limit of 10μg/L although its presence was detected in some samples. Total phosphorus concentrations were also generally low with 15 of 27 samples below the quantifiable limit of 10μg/L. The upstream Painkalac Creek site had a 75th percentile (30μg/L) that was greater than the State water quality objective for the region (25μg/L) (EPA Victoria, 2003a). This dataset, however, was too small to be usefully compared with the objective as it was dominated by the only sample that was above the detection limit, which was taken at a time of low flow.

3.4.3.e. Turbidity and suspended solids

During the study, Marshy Creek had comparable concentrations of suspended solids to Alcoa's long-term data (Table 3.15), with the exception of a concentration of 118mg/L measured in March 1999, in a period of no flow. Concentrations above Anglesea estuary were higher than the mean recorded by Alcoa at that site over the entire sampling period.

Turbidity was measured from September 2000 to the end of the study. During sampling, turbidity was greater at the two Painkalac Creek sites than at Distillery Creek or any of the sites in the Anglesea catchment (Figure 3.21). Reasons for this were not clear, although means were influenced by some very high (~200NTU) values recorded during a minor flood. Turbidity also tended to be slightly higher at the downstream site compared to the upstream site in Painkalac Creek. Neither of these sites met the State water quality objective of a 75th percentile less than 10NTU.

Statistic	MC	SC	AR	PC US	PC DS	DC
Mean	22.9	-	46.5	22.75	20.8	-
75th percentile	9.25	-	55.5	-	24.25	-
Min	4.5	-	19.5	17	14.5	-
Max	118	-	102.5	28.5	25	-
n	7	0	8	2	6	0
Long-term mean	13.2		10			

Table 3.15. Descriptive statistics for suspended solids in mg/L, measured between February 1999 and November 1999. MC: Marshy Creek, SC: Salt Creek and AR: Anglesea River below Alcoa. PC US: Painkalac Creek upstream of dam, PC DS: Painkalac Creek above the estuary and DC: Distillery Creek. No flow was observed in SC and DC during this period. Long-term mean from Alcoa monthly sampling, 1972-2002 at the confluence of MC and SC, 1998-2002 at AR. Before 1998, the detection limit of Alcoa data was 1mg/L, from 1998 it was 10mg/L. All below detects have been replaced with the detection limit appropriate for the time of sampling.

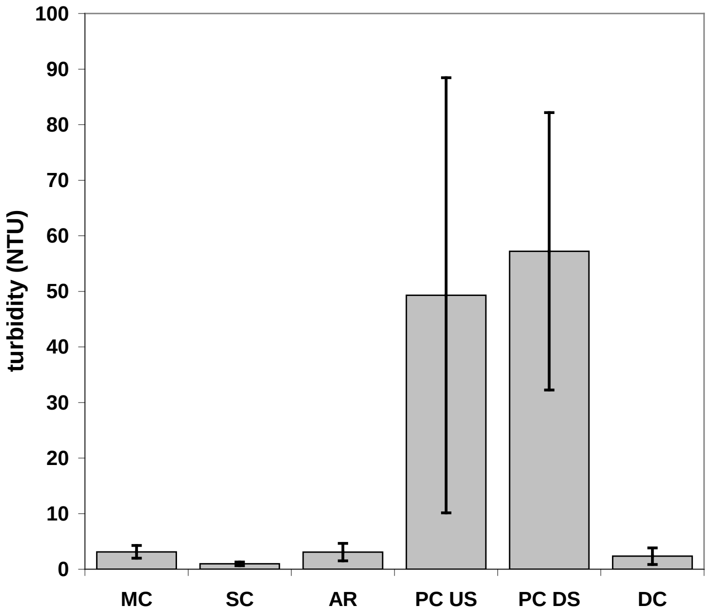


Figure 3.21. Mean turbidity (± s.e.) measured between September 2000 and February 2002. MC: Marshy Creek, SC: Salt Creek and AR: Anglesea River below Alcoa. PC US: Painkalac Creek upstream of dam, PC DS: Painkalac Creek above the estuary and DC: Distillery Creek.

The relative clarity of Distillery, Marshy and Salt Creeks was typical of acidic waters. In the later part of the sampling period Painkalac Creek sites were noticeably less clear than at earlier sampling times, when no measurements of turbidity were made.

3.4.4. Links between water quality variables

Water quality variables measured in Alcoa's monitoring program were analysed using principal components analysis to examine redundancies among the variables in describing temporal and spatial changes in water quality, and to determine components of water quality responsible for most of the overall variability between waters of upstream, discharge, and downstream locations through the three decades of Alcoa's operations.

Seventy percent of the overall variability in the data was explained by three independent components identified as; 'Salinity' (associated with high total solids, conductivity, pH, temperature, suspended solids and low iron and aluminium), 'Precipitates/Iron' (associated with high turbidity, suspended solids, colour, iron and low aluminium) and 'Acid episodes' (associated with high zinc, aluminium and low pH and colour). These components reflected processes observed during the study period:

- the first component related most closely to the influences of Alcoa's ash-pond discharge and of marine waters at the downstream sites (Section 3.4.3.a);
- the second component reflected the naturally occurring phenomena of iron-associated material in upstream waters at times of low flow; and
- the third component related to acid episodes such as those discussed in Sections 3.4.1, 3.4.2 and Appendix E.

Factor scores for each component were calculated and seasonal, inter-annual and between-site differences examined. 'Salinity' showed distinct seasonal influences, with different patterns in timing between the upstream site and the two Alcoa inputs. 'Precipitates/Iron' also showed seasonal patterns, which were less evident for 'Acid episodes'. Inter-annual variability was greatest at the upstream and mine-water sites. Details of these patterns and the principal components analysis are given in Appendix C.

3.5. Summary and conclusions

Marshy Creek and Painkalac Creek flowed more consistently than either Salt or Distillery Creek. This pattern was the opposite of that observed between Marshy and Salt Creeks in 1981 and may reflect changes in the location and/or the extent of reduced groundwater potential as a result of mining. The extent of dry periods in Salt Creek was also greater than those recorded from 1967 to 1982, or the subset of those flows associated with an equivalent period of low rainfall between 1967 and 1969. Six floods were recorded during the study, the first in September 2000. The largest of these was in April 2001 and was comparable in size to only two other floods in the previous 20 years. Flood peaks were lower and receding limbs longer in Anglesea River than Painkalac Creek, especially during the first flood, which also marked the start of substantial flows from Salt and Distillery Creeks.

Anthropogenic changes resulted in marked differences between the naturally intermittent flow regimes of the Anglesea River and Painkalac/Distillery Creeks, particularly during times of low natural flow. During the study period, there were a greater number of days and longer periods where there was no flow into the Painkalac estuary than would be expected, based on flows upstream of the reservoir. The release of waters from the dam was less variable than upstream flow until mid-2000, after which patterns of flow were more closely matched. In contrast, flow from Alcoa sources was continuous, representing the majority of flow to the estuary for around 90% of the study period. Anthropogenic effects on flow were negligible during floods, but smaller peaks in flow in Painkalac were not expressed below the dam in the first part of the study period. Overall, more water flowed into the Anglesea estuary at a less variable rate than waters flowing into the Painkalac estuary during the study.

In addition to differences in flow regimes entering the estuaries, there were also physico-chemical differences in inflowing waters that arose from both natural and anthropogenic causes. Anthropogenic contributions to differences in water quality varied with flow rates. These anthropogenic changes arose solely in the Anglesea system and no patterns were observed that suggested that the reservoir in Painkalac Creek was altering water quality (Table 3.16). The most noteworthy water quality variable in freshwaters was pH, which was below 5 in Marshy, Salt and Distillery Creeks almost all of the time while Painkalac Creek was generally neutral. pH was related to flow, in that waters in the acidic creeks were occasionally neutral at times of low flow and strongly acidic waters in Salt and Distillery Creek were associated with the first substantial flows from September 2000. (see Appendices D and E)

While concentrations of aluminium and iron were generally high in tributaries of the Anglesea River, very high concentrations of aluminium were associated with the initial strongly acidic flows from Salt Creek which, along with waters of Distillery Creek at this time, probably had an origin in oxidation of sulphide minerals. An association of acid streams with the geological strata of Eastern View Group is clear from the distribution of acidic waters between sub-catchments and in the equivalent strata on the opposite side of the Otways (Appendix D).

Variable	Est. diff?	Alcoa effect?	Pattern of differences	Flow related changes? A	P
Conductivity	Y	Y	PCboth<ARUS<ARDS	Y (↓)	Y(↓)
Temperature	Y	Y?	PCboth(=)ARUS<ARDS	N	N
Dissolved O_2	N	Y?	ARUS<ARDS=PCboth	Y (↓)	Y(↓)
Total N	Y	Y?	ARDS<PCboth<ARUS	Y (↓)	?
NOx	Y	Y	ARUS=PCboth<ARDS	?	?
Total P	N	Y[a]	ARUS=PCboth<ARDS	Y (↓)	?
SRP	? [b]	?	none	?	?
TSS	Y	Y	PCboth=ARUS<ARDS	Y (↓)	Y?(↑)
Turbidity[c]	Y	N	ARboth<PCboth	Y (↓)	Y (↑)
pH[d]	Y	Y	ARUS<ARDS<PCDS(<)PCUS	Y(↓)	Y(↓)

Table 3.16. Summary of patterns of difference in water quality variables above and below the Anglesea power station and Painkalac reservoir. 'Est. diff?' refers to whether a substantial difference in waters entering the estuaries was apparent. 'Alcoa. effect' refers to whether an anthropogenic effect was evident at Anglesea (question marks indicating uncertainty). No such effects were observed at Painkalac. Whether changes in a variable were flow related in either system is also indicated, with positive (↑) or negative (↓) relationships indicated by arrows in brackets. Codes: ARUS, ARDS=Anglesea River upstream and downstream, respectively; PCUS, PCDS=Painkalac Creek upstream and downstream, respectively; 'both' refers to upstream and downstream sites. a=an increase in %>detection limit, b=all samples > detection limit, c=but low in Distillery Creek, d=typical pattern shown, but temporally variable (discussed below).

Several changes to the water quality of the Anglesea River were caused by the mine and ash-pond discharges. These changes were dependent on the rate and chemical characteristics of flows from upstream. While the mine discharge was the major contributor of zinc to the system, the ash pond contributed a large amount of alkaline salts to the watercourse, neutralising upstream flows except at times when the volume and/or acidity of upstream flows was too great. This resulted in waters entering the estuary having a pH below 5 in September/October 2000 and then on three occasions in 2001, all associated with floods.

At times when flows were neutralised at the ash-pond discharge, it is likely that there was a settling of precipitated metals in the sediments downstream, as the saturation limit for dissolved metal species decreases at higher pH. It is also likely that some of these metals were then re-mobilised when acidic waters flowed downstream. A similar mechanism is thought to have led to the large concentrations of Al from Salt Creek in September 2000. Relative contributions from all upstream sources meant that the pH in Painkalac estuary was much less affected by acidic flows than the Anglesea estuary, which had a pH of less than 5 throughout much of the estuary on more than one occasion. Prior to the neutralisation of upstream waters by the ash-pond discharge this may have happened more frequently.

The ash-pond discharge also increased the conductivity of Anglesea River, except in times of flood. The effect of this, in terms of salinity, was to extend the low salinity region of the estuary upstream, which may have restricted the range of stenohaline freshwater organisms (although ionic composition alone can also be biologically important: e.g. Radke *et al.*, 2003; Zalizniak *et al.*, 2006).

Phosphorus concentrations in both estuaries were relatively low; however, total nitrogen concentrations in Marshy Creek and entering the estuary of Painkalac Creek were close to water quality guidelines, while substantially higher concentrations of nitrates were measured below Alcoa in 1999 at levels well above national guidelines.

Principal components analysis of long-term water quality data from the Anglesea catchment showed that the ten variables measured could be reduced to three attributes of water quality. These three attributes together accounted for 70% of the total variability in water quality between upstream waters, Alcoa's two discharges and two downstream sites as measured monthly between 1972 and 1998. The first two attributes, 'Salinity' and 'Precipitates/Iron', showed seasonal trends in upstream waters and were more consistent in the mine and ash-pond discharges. The third attribute was primarily influenced by a few instances of high zinc concentrations in the mine discharge. Results from the analysis of historical data were broadly consistent with patterns from the study period with its seasonally acidic flows and more constant input from Alcoa.

Several physico-chemical changes in natural waters were associated with flow rate. This was most pronounced in the Anglesea catchment but changes were also observed in Painkalac. In Anglesea, interactions between flow, water tables, sulphidic sediments, peats and iron bacteria were thought to have been responsible for an essentially dichotomous set of water quality characteristics that were expressed at times of high and low flows. These differences are summarised in Table 3.17.

Variable	Low flow	Mod./High flow
pH	6-8	3-4
Iron	↑	↓
Aluminium	↓	↑
Dissolved oxygen	↓	↑
TN, TP	↑	↓
Conductivity	↑	↓

Table 3.17. pH and associated changes in other water quality variables in upstream waters of the Anglesea River catchment. ↑=increase, ↓=decrease.

The mechanisms responsible for these patterns were thought to stem largely from generation of acid by oxidation of sulphidic sediments in the catchment (see Appendix D), related to the raising and lowering of the water table, possibly enhanced by the intermittency of streams of the area. Such mechanisms have been extensively discussed in the literature relating to acid sulphate soils (e.g. Dent & van Mensvoort, 1992; Sammut & Lines-Kelly, 1996; Wilson *et al.*, 1999).

Increased solubilities of metals at low pH, presumably derived from clays (van Breemen, 1992) were the likely reason for elevated concentrations of aluminium recorded at times of high flow. The extremely-high iron concentrations during low flows (with higher pH), may be the result of concentration of iron as oxides by iron bacteria (regularly observed at times of low flow) and re-dissolution of iron compounds in hypoxic waters associated with low flows through peaty streambeds. Complexation of this iron with humic acids (which were responsible for the high colour of these waters) probably led to the large filterable portions of iron recorded by Meyrick (1999).

4. Estuarine Character and Hydrology

4.1. Introduction

Intermittent estuaries represent an extreme type in the continuum of hydrological variability in estuaries. In a ‘classical’ estuary, with an unconstricted mouth, differences in tidal influence vary with diurnal and spring-neap cycles. Other permanently-open estuaries may have mouths that change in cross-section and so have a variable throttling effect on tidal movements of estuarine waters. Intermittent estuaries have no tidal exchange for extended periods of time.

There have been many models put forward as mechanisms for opening and closure of intermittent estuaries (summarised in Ranasinghe & Pattiaratchi, 2003). Typically these relate openings to short term high-flow and/or storm events and closures to longer-term movement of sand either along the adjacent shoreline or landward during appropriate wave conditions. Once an intermittent estuary is open, maintenance of that state is dependent on water movements through the mouth having enough energy to stop any long- or cross-shore sand movements from re-closing the estuary (Bird, 1967; Ranasinghe & Pattiaratchi, 2003). These water movements can be either tidal exchange through the mouth or freshwater discharge from upstream, or both. Conversely, for an estuary to remain closed, any fresh or marine flows over the sand bar must be insufficient to erode an opening faster than the opposing coastal processes building the bar.

Specific objectives designed to investigate the influence of fresh water flow on estuarine hydrology were to:

- quantify the bathymetry of Anglesea estuary;
- develop and assess the validity of a conceptual model of relative tidal influences in intermittent estuaries based on changes in water level and constriction of entrances; and
- examine the relative influence of freshwater, marine and coastal processes in relation to the conceptual model.

4.2. Methods

4.2.1. Bathymetric model

A bathymetric model of Anglesea estuary was created using SURFER (Golden Software Inc, v6.04) to interpolate spot depth measurements at locations recorded by differentially corrected GPS. In addition to spot depths, boundaries of the estuary at a height of 1.49m AHD were created using a combination of spot depths, 1:25,000 map data and a georectified aerial photo. The position of the centre channel was marked at a finer resolution by interpolation of spot depths. Details of this method are given in Appendix F.

4.2.2. Depth variations

4.2.2.a. Benchmarks

Fixed benchmarks were used to measure water height in Anglesea and Painkalac estuaries. In Anglesea, the mark was set at the lower edge of the bevel on the footrail on the inner corner of the wooden jetty (Figure 4.1). This was surveyed to 1.69m AHD from Permanent Survey Mark 18 on 10/3/1999. In Painkalac, a mark at the bottom edge of a 45^{o} bevel on the southwest corner of the north-eastern bridge foundation was used as a height datum (Figure 4.1). This was surveyed to 1.20m AHD from Angahook Permanent Survey Mark 15 on 29/11/2002.

4.2.2.b. Height measurements

Between 8/12/1998 and 19/2/2002, water height was measured 109 times at the jetty datum point in Anglesea and 58 times at the bridge datum point in Painkalac. These measurements were taken on two time scales, monthly over years and within hours and days during an individual trip. Notes on the extent of inundation of the littoral areas of each estuary were also recorded.

In Anglesea, a depth logger was installed 400 m upstream of the benchmark on 12/3/1999 and removed on 18/2/2002 (see location in Figure 4.1a). While operable, depth and temperature were measured at 10-minute intervals. Due to vandalism, malfunctions and subsequent repair times, there are periods where there are no logger data (Table 4.1). The logger sensor was initially placed at a depth considered sufficient to ensure that it remained submerged at all times. Following the flood of 8 November 2000, water level dropped below the sensor on some low tides. It was not possible to relocate the logger to a suitable secure site.

Period	Start date	Finish date	Interval (days)	Sensor elevation (mAHD)
1	**12/3/1999**	**24/3/1999**	**13**	**-0.369**
2	25/3/1999	15/8/1999	144	N/A
3	**16/8/1999**	**23/10/1999**	**69**	**0.827**
4	24/10/1999	14/11/1999	22	N/A
5	**15/11/1999**	**4/2/2000**	**82**	**0.827**
6	5/2/2000	7/2/2000	3	N/A
7	**8/2/2000**	**15/7/2001**	**524**	**0.827**
8	16/7/2001	30/9/2001	77	N/A
9	**1/10/2001**	**18/2/2002**	**141**	**0.807**

Table 4.1. Depth logger deployment times for Anglesea estuary. Periods during which depth was logged are shown in bold. N/A – not applicable.

A logger was available for short-term use in Painkalac and was deployed 250 m upstream of the benchmark (Figure 4.1b). This logger was programmed identically to the one in Anglesea and operated from 9/11/2001 to 19/2/2002 with a short gap from 13-18/1/2002. The sensor was at an elevation level of 0.320m AHD.

a)

b)

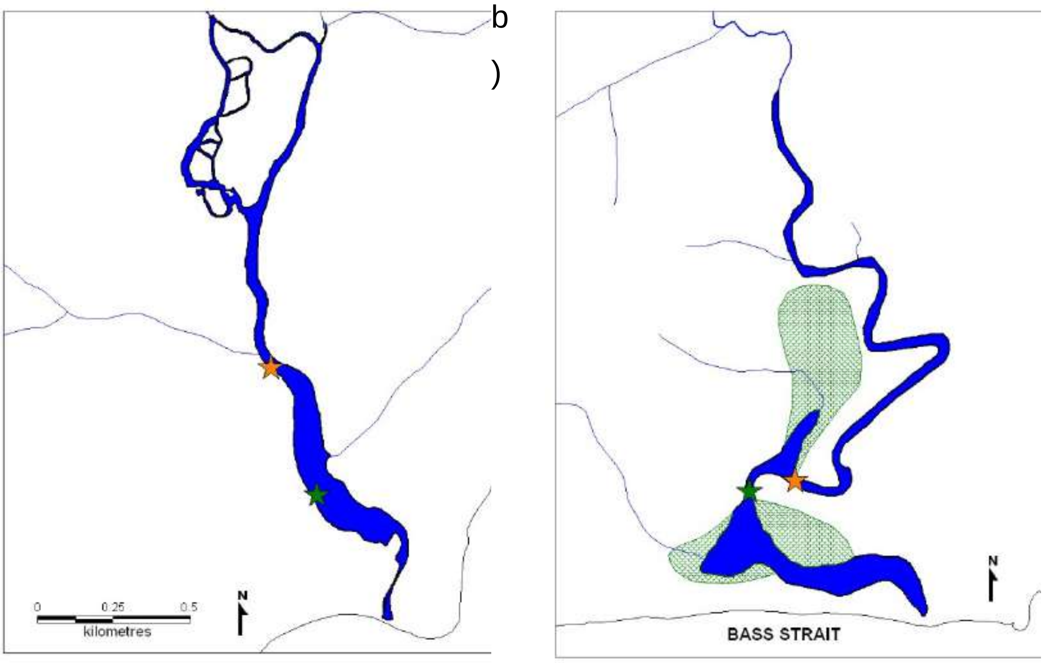


Figure 4.1. Logger (red stars) and height benchmark (green stars) locations in a) Anglesea and b) Painkalac estuaries.

Tidal data from a gauge at Lorne (38°32'S, 143°59'E) were obtained from the Victorian Channels Authority. This gauge is 12.8km southwest of the mouth of Painkalac estuary and 23.1km from the mouth of Anglesea estuary. Daily observations of water height in Painkalac, whether the estuary was tidal and the presence of a connection to the sea from May 1987 onwards were kindly made available by Mrs Pauline Reilly (see Reilly (1998) for methods).

Due to the differences in the frequency and duration for which water heights in each of the two estuaries were logged, comparisons of hydrologic state between estuaries were based on a combination of sources of data (Section 4.3.4).

4.2.3. Bar morphodynamics

From 8/12/1998 to 19/2/2002, approximately monthly, the following features of the entrance sandbars of the Anglesea and Painkalac estuaries were recorded:

- whether there was a channel crossing the bar;
- the width and depth of any channel;
- flow direction and speed of flow;
- evidence of marine water incursion (e.g. wrack deposits, patterns in sand); and
- the apparent state of the estuary (closed, perched (partially open/overflowing) or open/tidal).

For Anglesea and Painkalac estuaries, the perimeter of the channel or, when closed, the seaward boundary of the estuary was traced by GPS and differentially post-processed using Victorian government GPSnet base station data (as for bathymetric positional data: see Appendix F). Anglesea was sampled 26 times between September 1999 and February 2002. Painkalac was measured 13 times between July 2000 and February 2002. Mean 95% precisions of horizontal positions in each trace ranged between

1.80m and 3.20m over all occasions. The worst 95% precision of any individual position in a trace ranged from 1.99m to 5.87m.

On each occasion, a shoreline length of approximately 1 km was traced at the Anglesea mouth and a length of approximately 500 m was traced at the Painkalac mouth due to the narrower beach and less extensive sand deposits inside the mouth of the Painkalac estuary.

4.2.4. Quantitative classifications of hydrologic state

To quantify the nature of the hydrologic states at Anglesea, and to confirm the validity of the visual classifications from plots of water level, a classification tree analysis was done. The method used for this analysis was one of a series of techniques known as classification and regression trees (CART). These tools explain changes in a response variable in terms of one or more predictor variables by repeatedly splitting data into more homogenous groups based on rules associated with a single predictor variable (De'Ath & Fabricius, 2000). In effect, this process creates rectangular categories parallel to axes of the predictor variables. These analyses have very few assumptions and can be used for a wide range of distributions (Brieman *et al.*, 1984). One particular benefit of this type of analysis, when compared to a PCA or similar, is the ability to describe a response in terms of the original predictors.

The dependent variable for the classification tree was state, as qualitatively identified from time series plots of water level. The nine independent variables were, for each period, the number of heights recorded and the mean, standard error, skewness and kurtosis of water heights and the mean magnitude, standard error, skewness and kurtosis of ten-minute changes in height over each period. Further details of the analysis are in Appendix F.

4.3. Results and discussion

4.3.1. Bathymetry of Anglesea estuary

Anglesea estuary can be divided into four sections based on morphology (Figure 4.2). The upper estuary consists of the original channel and

Coogoorah Park, to the west of the original estuary, consisting of a network of shallow dredged channels with steep sides, and fringing flats with reedbeds. The middle section comprises a long straight reach with moderately sloping sides, a partially defined central channel and few fringing mudflats or reedbeds. The lower estuary is wide and shallow and has fringing mudflats at lower elevations than the upper estuary and Coogoorah Park (~0.7-1.0 m AHD vs >~1.3 m AHD). A central channel runs though most of this part of the estuary, becoming less defined towards the mouth. The final section of the estuary is the entrance, which is defined by an area of mobile beach sand/flood tide delta between two fringing sand dunes and a mobile and variable connection between the estuary and Bass Strait.

The elevation range of the estuary represented in the model was –1.70 to 1.49m AHD (Figure 4.2). At a water level of 1.49m AHD, the volume of the estuary was calculated as 157ML. As the upper limit of the model was constrained by the (relatively high) water level at the time of survey, volumes for heights above 1.49m AHD were based on a vertical extension of the surface area of the estuary at that height and are therefore underestimates, given flat areas in the upper estuary that were excluded from the model by necessity.

In general, the modelled volume of the estuary increases more per unit of height at higher water levels (Figure 4.3). In the lower estuary, the increase in volume per unit height becomes consistently greater above the height of 0.8m AHD, which is reflected to a lesser extent in the volume/height relationship for the whole estuary. In contrast, there is a gradual increase in volume/cm height for the rest of the estuary, with a slightly greater increase at the highest level, at which flat fringing areas in the upper estuary begin to be inundated.

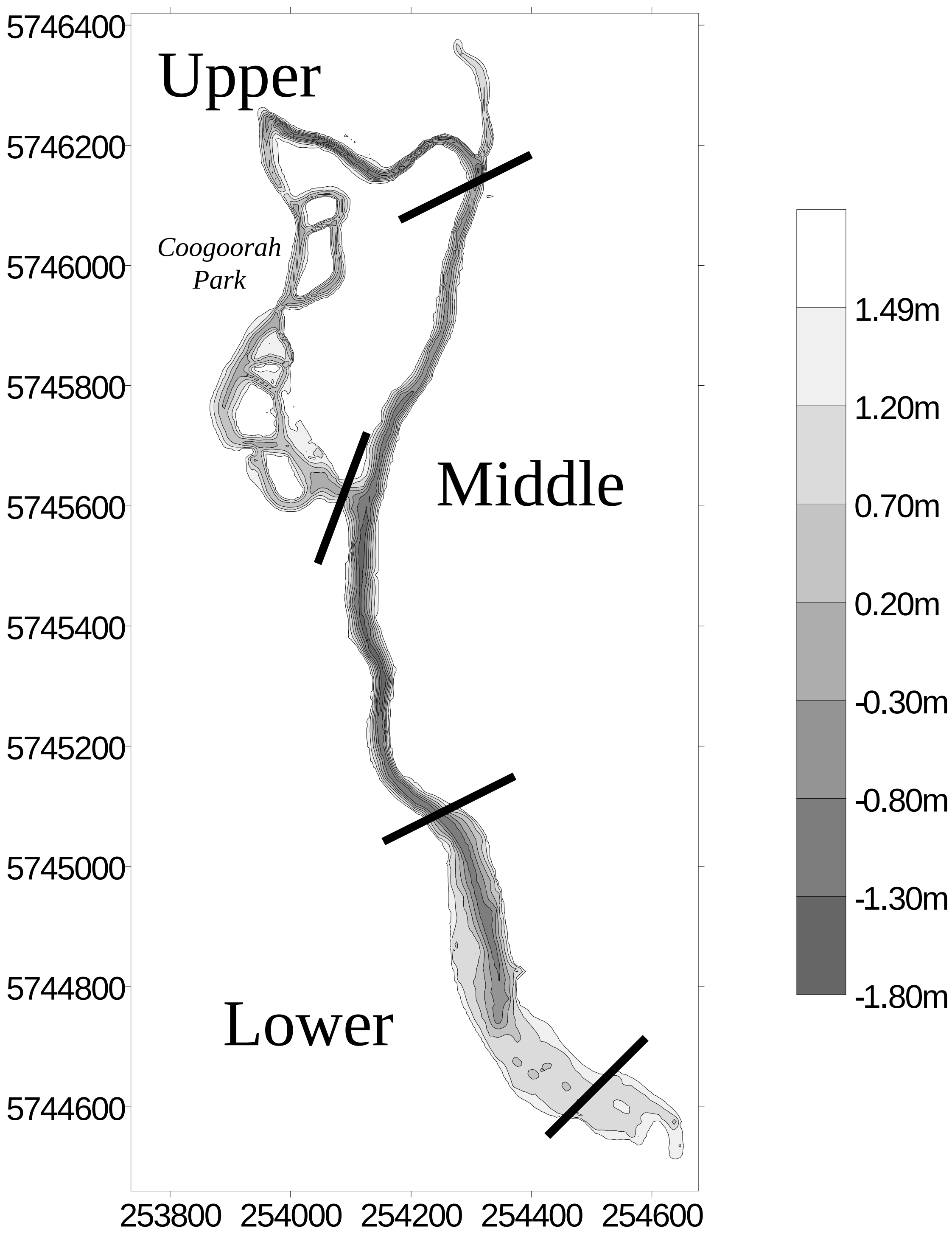


Figure 4.2. Contour plot of the digital bathymetric model of Anglesea estuary with sections of the estuary marked as shown. The entrance area is in the lower right hand part of the figure. Depth is in metres AHD, eastings and northings are from the Australian Map Grid 66, Zone 55, AGD 66.

Water height (m AHD)	Volume (ML) Whole estuary	Lower estuary	Coogoorah Park	'Original' estuary[a]	1981 estuary[b]
1.8 (max. recorded)	210*	71.3*	41.9*	168*	-
1.65	184*	61.7*	35.4*	149*	180
1.49	157	51.5	28.4	129	-
1.05	96.4	28.3	13.8	82.6	110
0.2 (min. recorded)	31.9	8.76	0.568	31.3	-

Table 4.2. Modelled volumes of Anglesea estuary and sections at selected water heights. *volumes above 1.49m AHD estimated based on planar areas at 1.49m. [a] original estuary = estuary less Coogoorah Park [b] volume estimated in 1981 (Atkins & Bourne, 1983).

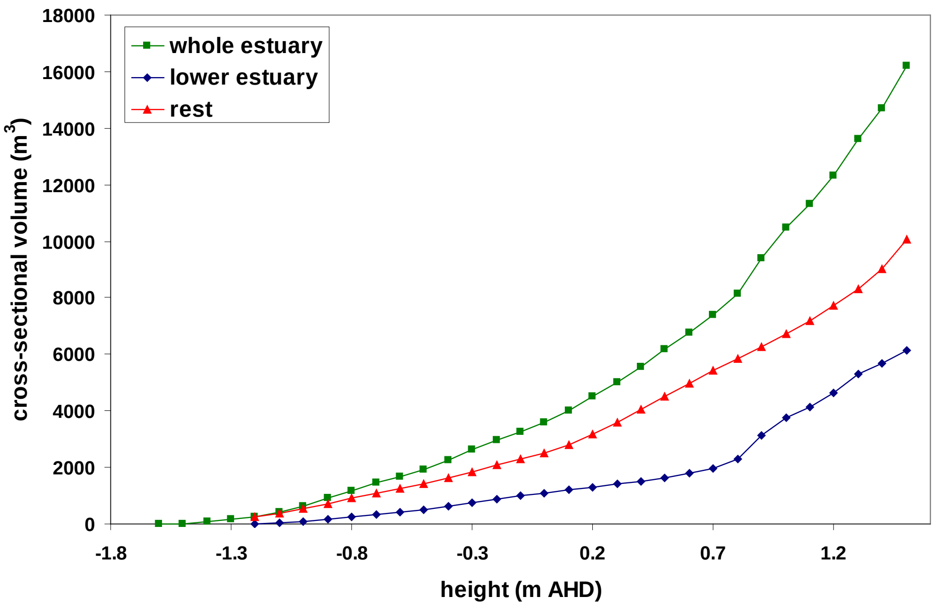


Figure 4.3. Cross-sectional volumes of Anglesea estuary from the bathymetric model. Points represent volumes of 10cm thick horizontal sections of the whole estuary, the lower section and the remaining sections combined.

At 1.05m AHD the estimated volume of the estuary excluding Coogoorah Park was 25% smaller than the estimate for the pre-Coogoorah Park estuary by Atkins and Bourne (1983) using mean cross-sections (Table 4.2). At 1.65m AHD, the highest water level recorded by Atkins and Bourne (1983), the current modelled volume was 17% smaller than the estimate for the estuary in 1981 (Table 4.2). Both of the 1.65m estimates are acknowledged underestimates, in the case of this study, due to increases in volume above 1.49m being based on vertical rise over the surface area at that height, and the 1981 volume 'is a conservative estimate since…water has already inundated a large area of low lying bush and *Phragmites* flats'.

At certain heights there was a disproportionately large area of substrate in the estuary, reflecting relatively flat topography at those heights (Figure 4.4). Distinct peaks in substrate area are seen in five height intervals. Of these, the lower two intervals (-1.0 to -0.9m and -0.5 to -0.4m) were submerged at all times during the study and are found in all parts of the estuary, excepting Coogoorah Park which has a maximum depth of around –0.3m AHD. The largest peak (0.8 to 0.9m) corresponds to flats in the lower estuary, the next peak (1.1 to 1.2m) is related to the sand body at the entrance of the estuary and the highest point (1.4 to 1.49m) is an indication of the flattening topography at these heights in the upper estuary and Coogoorah Park.

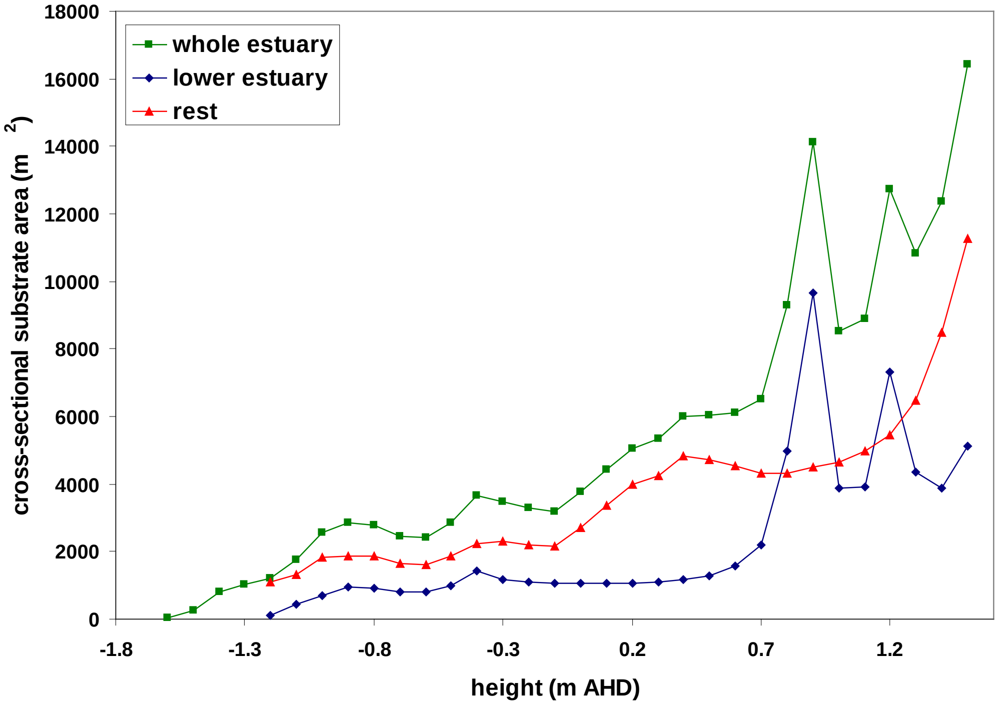


Figure 4.4. Cross-sectional areas of substrate in Anglesea estuary from the bathymetric model. Points represent areas of substrate within 10cm high horizontal sections of the whole estuary, the lower section and the other sections combined.

4.3.2. Hydrologic states: variations in depth

4.3.2.a. Estuarine states

Inundation of both Anglesea and Painkalac estuaries appeared to follow three broad patterns during the time of the study: identified as closed, perched and tidal states. Each of these states was associated with a particular pattern of temporal changes in water level (Figure 4.5). Neither estuary was observed to have a water level below mean sea level (0m AHD).

Closed states

During closed periods, the bars of the estuaries were highest relative to sea level and the estuaries were essentially isolated from tidal influence. Occasionally during these periods, particularly at Anglesea, a small amount of water flowed out and/or large waves entered the estuaries. At these times there was little short-term fluctuation in water depth. When closed, the water level in the Anglesea estuary was typically high (~1.5m AHD) while that of Painkalac varied seasonally with freshwater flow and evaporation (Table 4.3). An example of the potential difference between the two estuaries when both were closed can be seen in photographs taken at the end of the summer of 1999/2000 when Anglesea was full (Figure 4.6a) and fringing flats were permanently exposed in Painkalac (Figure 4.7b).

Perched states

The estuaries were defined as perched when the height of the bar was at an intermediate level, allowing tidal exchange during higher tides but maintaining a water height well above that of sea level. Intermediate water depths and a restricted and attenuated tidal pattern reflecting the pattern of spring tides were typical of these periods. A channel to the sea was always present (Figure 4.5).

Tidal states

In a tidal state, the bars of the estuaries had deep channels and there was a relatively unrestricted exchange with the sea. During these periods, mean water levels were relatively low (<~0.9m AHD) and there were large daily fluctuations in depth.

Although each of these states was relatively distinct there was some blurring of boundaries between the states, particularly between perched and tidal, depending on the depth of the channel through the mouth (see Section 4.3.2.c, Appendix F).

State	Water level	Daily range
Closed	High[a], High/Low[b]	Low
Perched	Intermediate	Intermediate
Tidal	Low	High

Table 4.3. Characteristics of the three hydrological states of the estuaries. [a] Anglesea, [b] Painkalac.

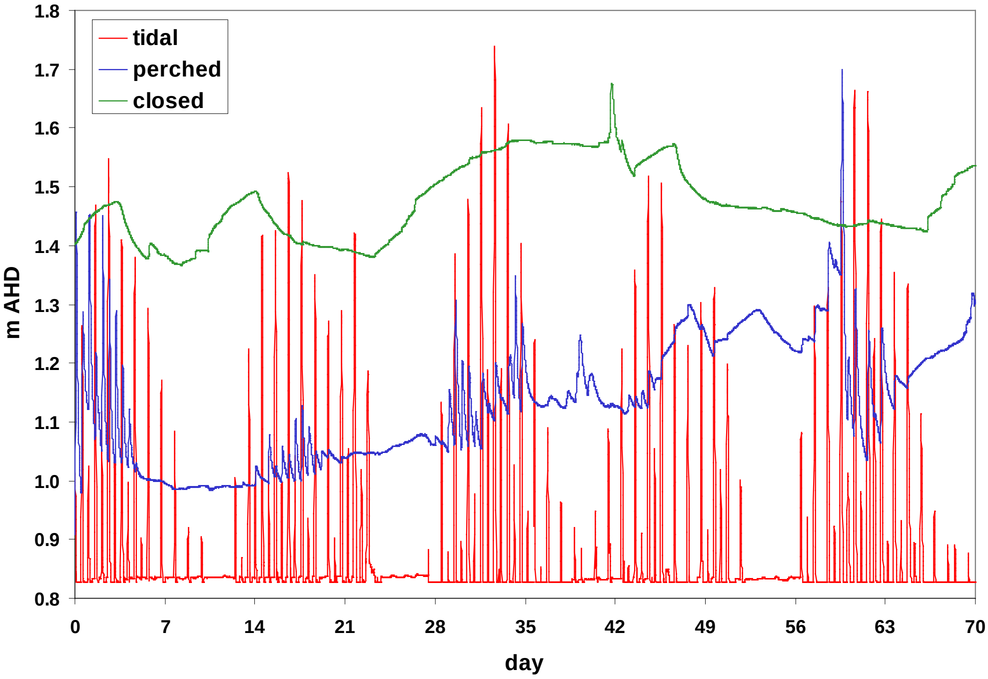


Figure 4.5. Logged water levels from Anglesea during three ten-week periods illustrating patterns of variation in water level during the three hydrological states. Tidal: 25/4/2001-3/7/2001, Perched: 11/2/2001-21/4/2001, Closed: 15/11/1999-24/1/2000.

4.3.2.b. Water levels

The maximum water level recorded manually at the datum point at Anglesea was 1.77mAHD, on both 19/5/1999 and 31/5/2000. The minimum level recorded was 0.17mAHD on 25/4/2001, immediately after the large flood of that month. Photos of high and low water levels in the lower estuary are shown in Figure 4.6.

a)

b)

Figure 4.6. Lower Anglesea estuary at a) high and b) low water levels. a) taken on 8/2/00 at a level of approximately 1.44m AHD. b) taken on 25/4/2001 at a height of approximately 0.8m AHD. The approximate location of Site A3 (see Chapter 6) is shown in b) with a red 'X'.

a)

b)

Figure 4.7. Lower Painkalac estuary at a) high and b) low water levels. a) taken on 2/10/2000 at a level of approximately 1.88m AHD. b) taken on 8/2/2000 at a height of approximately 1.03m AHD. The approximate location of Site P3 (see Chapter 6) is shown in b) with a red 'X'.

In Painkalac, the maximum recorded water level was 1.88mAHD on 2/10/2000. Following the large floods of April 2001, the minimum level recorded was 0.05mAHD at low tide on 26/4/2001. Photos of high and low water levels in the lower estuary are shown in Figure 4.7

From the start of sampling until the floods of September (in Painkalac) and October 2000 (in both estuaries), the water level of both estuaries remained above 0.8mAHD (Figure 4.8). During this time the water level in Anglesea showed a different pattern of variation to that in Painkalac.

In this first period, the water level in Anglesea was relatively constant except for two periods of rapid change in May/June of 1999 and 2000. In Painkalac, the water level showed a seasonal pattern from June 1999 to August 2000 with low water levels in summer/autumn and high water levels in winter/spring. The large drop in water level in Painkalac between late August and early September 2000 was associated with an artificial breaching of the sandbar on 1/9/2000.

The water level of both estuaries was more variable in the second half of the study than the first. In October 2000, decreases in water level were observed in both estuaries. Of particular note are two occasions on which the water level of both estuaries was reduced simultaneously by flooding, in April 2001 and August 2001. A large decrease in water level in Painkalac was also associated with floods of November and December 2001.

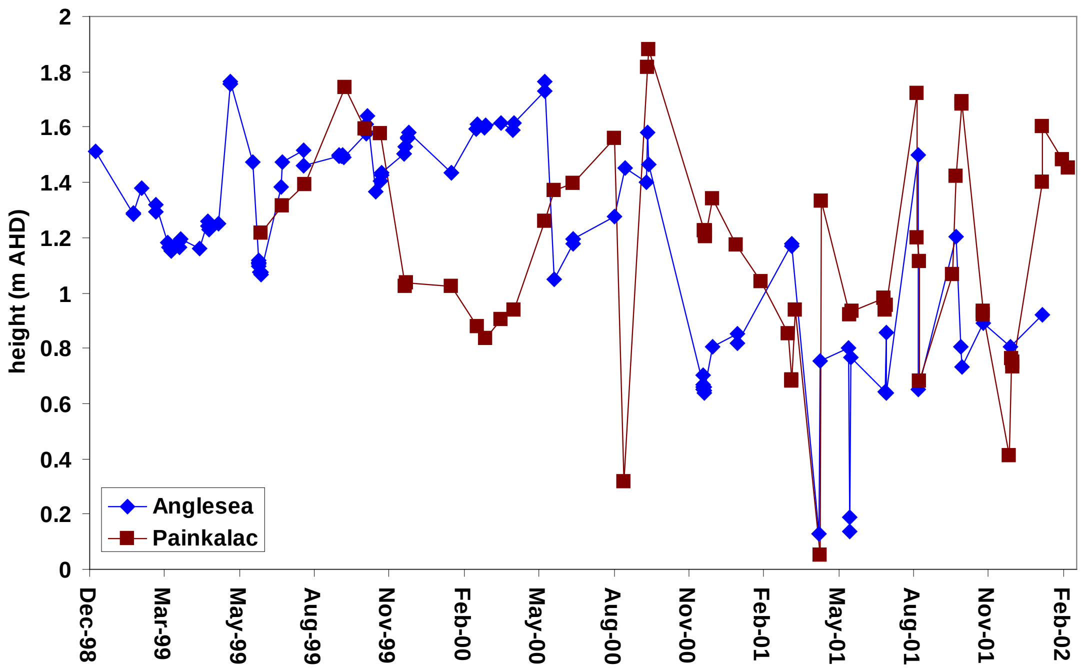


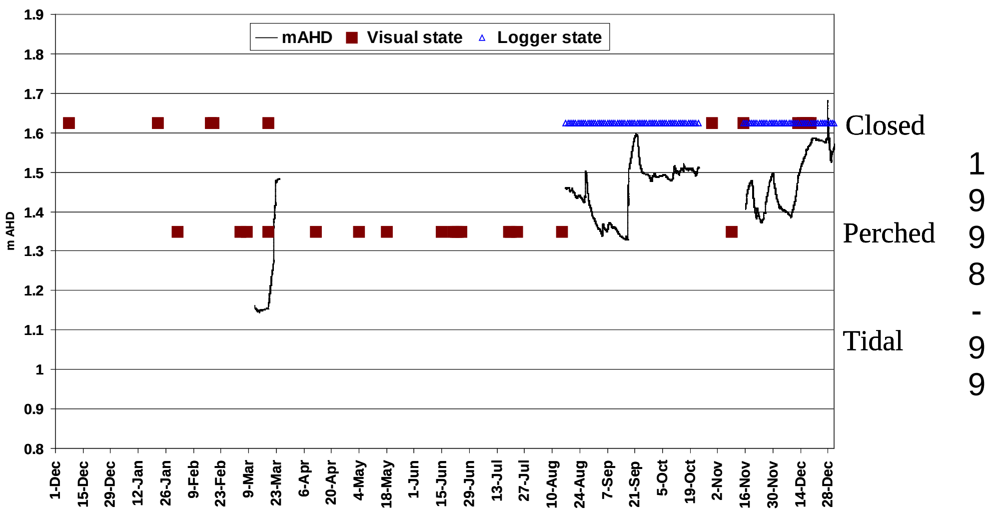
Figure 4.8. Measured water heights at height datum points in Anglesea and Painkalac estuaries during the study period. Note that automatic logging of water levels was not possible below ~0.8m AHD at Anglesea and 0.3mAHD at Painkalac. Deployment times for loggers were not continuous and started on , and encompassed a subset of times from mid-1999 at Anglesea and November 2001 at Painkalac (see Section 4.2.2.b).

4.3.2.c. Qualitative description of states: water level traces

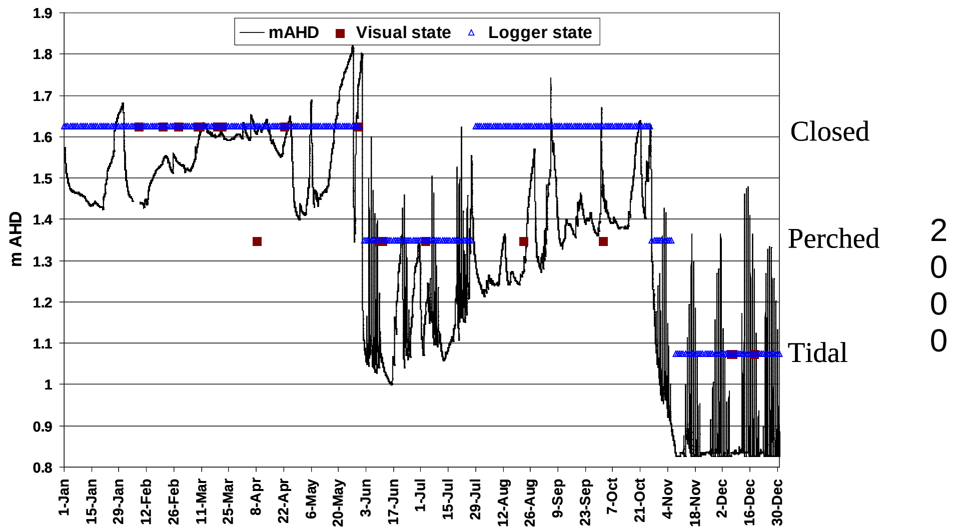
Compared to the manual measurements, logger data provided much greater temporal resolution of changes in water level (10 minute intervals versus ~monthly). Qualitative determinations of the state of each estuary for the times that the loggers were deployed were made based on visual examination of time series plots.

During the time of deployment at Anglesea, the maximum water height recorded was 1.82m AHD on 27/5/2000. Following the breaching of the bar on 8/11/2000 the depth sensor was regularly exposed at low tide during tidal periods (at levels below ~0.8m AHD – see Table 4.1). At the lowest manually recorded water level, following the large floods of April 2001, the water level (0.17m AHD) was about 0.7m below the depth sensor. This placed some restrictions on the use of the height data from tidal periods in analysis.

a)



b)



c)

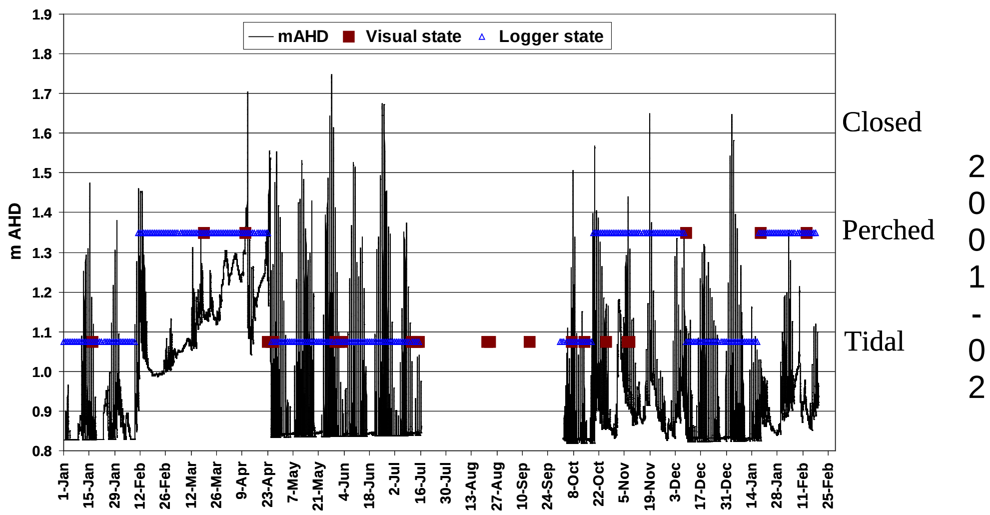


Figure 4.9. Logged water height data with hydrologic states, as determined from mouth observations and logger data, in Anglesea for a) 1998/99, b) 2000 and c) 2001/02. The logger became exposed at <0.827 mAHD during 16/8/99-15/7/01, and <0.807 mAHD during 1/10/01-18/2/02.

The maximum daily range as measured by the logger was 0.911m on 27/5/2001; an underestimate, as the sensor was exposed to the air at a water level of 0.827mAHD on this day. A survey on 1/6/2001 recorded a minimum water level of 0.13mAHD, suggesting a maximum range around 1.6m at that time. The minimum range of water levels recorded was 1mm (on 15/9/1999) and 142 days had a range of water level less than 1cm, all between the start of depth logging on 16/8/1999 and 24/3/2001 (586 days).

Periods of various estuarine states are distinct in the plotted height data (Figure 4.9, Table 4.4). Some discrepancies between 'logger state' and 'visual state' are shown in Figure 4.9b and c. These are discussed in Section 4.3.4.

State	#	Duration (days)	Mean level (m AHD)	Mean daily range (m)	From	Till
Closed	1	68[a]	1.45	0.015	16/8/99	23/10/99
	2	81[a]	1.49	0.019	15/11/99	4/2/00
	3	113[a]	1.57	0.028	8/2/00	31/5/00
	4	90	1.38	0.044	29/7/00	26/10/00
Perched	1	56	1.16	0.16	2/6/00	27/7/00

	2	12	1.02	0.22	27/10/00	7/11/00
	3	71	1.14	0.09	11/2/01	23/4/01
	4	50	0.92	0.25	19/10/01	08/12/01
	5	31[a]	0.91	0.14	18/01/02	18/02/02
Tidal	1	93	0.87[b]	0.21[b]	8/11/00	9/2/01
	2	81[a]	0.88[b]	0.36[b]	25/4/01	15/7/01
	3	17[a]	0.86[b]	0.28[b]	01/10/01	18/10/01
	4	39	0.86[b]	0.31[b]	09/12/01	17/01/02

Table 4.4. Duration, dates, mean height and range of states of Anglesea estuary as determined from logger data. [a] - underestimated as period begins or ends with deployment or removal of logger, [b]- range underestimated and mean overestimated by logger becoming exposed (at <0.827 mAHD 16/8/99-15/7/01, and <0.807 mAHD 1/10/01-18/2/02).

A total depth range of 1.13m (0.38m-1.51m AHD) was recorded while the logger in Painkalac was deployed (9/11/2001 to 19/2/2002). During this period the maximum daily range was 827mm on 7/12/2001 (a tidal cycle) and the minimum daily range was 4 mm on 6/2/2002 (Figure 4.10). Although the deployment was relatively short, all three states were observed during that time (Figure 4.10, Table 4.5). On 8/2/2002, following a brief freshwater pulse, the water level rose relatively rapidly (26 cm over 11 hours) before slowly draining out and/or evaporating until the end of the deployment period. This last event illustrates the large effect a relatively small flow (1.9 ML in 11 hours) can have on water levels in a closed estuary.

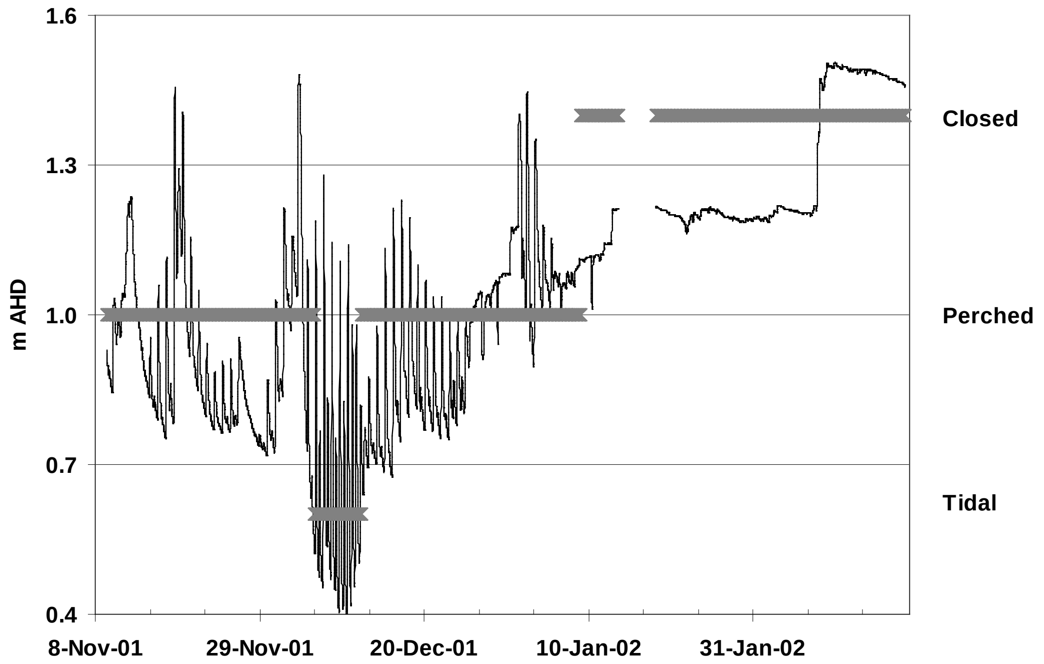


Figure 4.10. Depth variations in Painkalac. Height datum is AHD. Data recorded at 10-minute intervals. State shown as grey line at three levels from closed (top) to tidal (bottom).

State	#	Duration (days)	Mean level (m AHD)	Mean daily range (m)	From	Till
Closed	1[a]	4.8	1.14	0.052	9/1/02	13/1/02
	2[a]	31.9	1.30	0.021	18/1/02	19/2/02
Perched	1[a]	26.5	0.91	0.259	9/11/01	5/12/01
	2	28	0.97	0.246	12/12/01	8/1/02
Tidal	1	6	0.64	0.710	6/12/01	11/12/01

Table 4.5. Duration, dates, mean height and range of states of Painkalac estuary as determined from logger data. [a] - underestimated as period begins or ends with deployment or removal of logger. Mean daily range includes whole days only.

4.3.3. Quantitative description of states

To quantify the nature of the hydrologic states at Anglesea, and to confirm the validity of the visual classifications from plots of water level, a classification tree analysis was done. In this analysis, summary statistics of logged height data for each defined period were used as potential predictor variables to describe the state of each period. The summary statistics used related to water level, the rate of change in water level and the duration of each period (Appendix F, Table F.10).

Seven bivariate rules were derived in the analysis that successfully separated periods in which the Anglesea estuary was in different states (Table 4.6). These rules were based on a subset of 6 of the 9 independent variables used (mean, standard error, skewness and kurtosis of height as well as the mean magnitude and standard error of the 10-minute rate of change in water level). While only one of the seven rules derived from Anglesea data could be successfully applied to the Painkalac data directly, five of the rules showed patterns that were reflected in Painkalac albeit with differing cut-off criteria. A detailed examination of the results of this analysis is given in Appendix F.

Variable	***Rule***	***= State***	***Valid?***	***Pattern?***
Mean height	< 0.91	Tidal	Y	Y
Mean height	> 1.38	Closed	N	Y
Standard error height	$< 1.06 \times 10^{-3}$	Closed [a]	N	Y
Skewness height	> 2.92	Tidal	N	Y
Kurtosis height	> 9.47	Tidal	N	N
Mean magnitude rate	$< 1.64 \times 10^{-3}$	Closed	N	?
Standard error rate	$< 4.22 \times 10^{-5}$	Closed	N	Y

Table 4.6. Rules derived from Anglesea data and applicability for separation of states in Painkalac. 'Valid?' refers to whether the rule derived from Anglesea can be directly applied to the Painkalac data, 'Pattern?' refers to whether or not a similar pattern of separation between states was evident for the Painkalac data. [a] – this rule only distinguishes between perched and closed states; all other rules distinguish between the specified state and both other states.

The successful differentiation of states by quantitative analysis of the magnitude and rate of change in water level supports the original method of classification based on observation of time series plots and visual assessments of the mouths (Figure 4.9, Figure 4.10).

4.3.4. Integrated measures of state

Logged height data did not encompass the entire study period, nor was it available for both estuaries most of the time. To compare states between estuaries, and to assess links with physical and biological measures three sources of data relating to state were used. These were site visits (visual assessments of state), logger data and daily observations made by Reilly at Painkalac (unpublished data – for methods see Reilly (1998)).

Visual assessments of state tended towards a greater degree of marine influence than actually present and so were modified consistently, based on systematic differences between those assessments and logged patterns of fluctuations in water levels (see Appendix F for details).

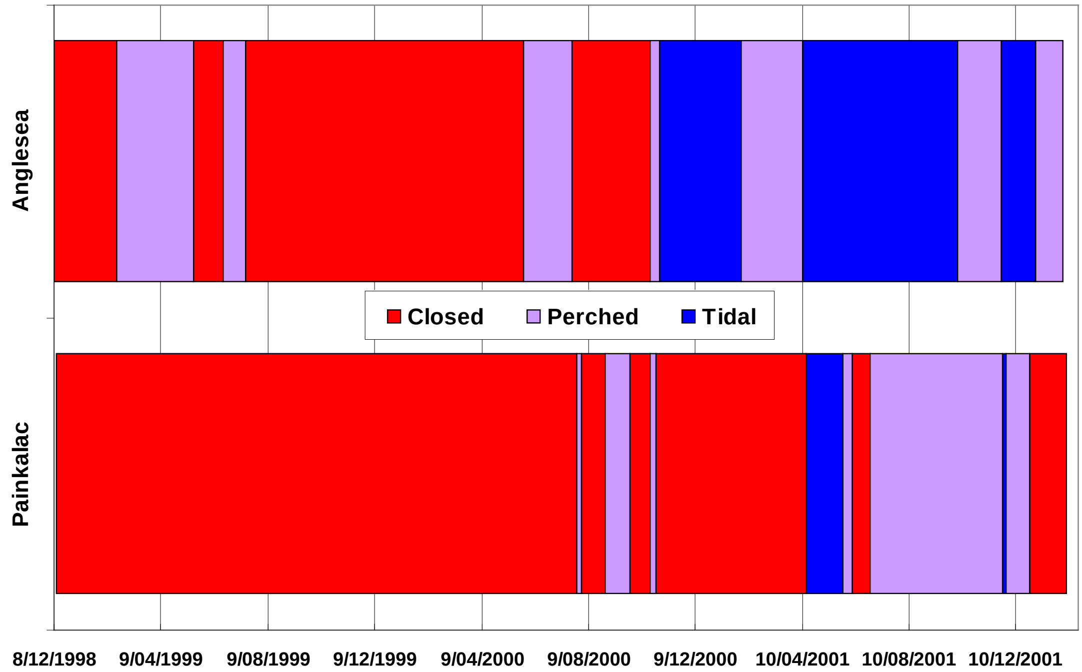


Figure 4.11. States of Anglesea and Painkalac estuaries throughout the sampling period. State of each estuary was identified by logged water height, visual observation of the mouths and daily observations by Reilly (unpublished data).

Overall, proportions of time that the estuaries were in each state were significantly different (chi-square test, $p=1.0 \times 10^{-15}$, 2 d.f.). Anglesea estuary was more often tidal and less frequently closed than Painkalac estuary (Figure 4.11, Table 4.7).

	Closed	Perched	Tidal
Anglesea	44.4	28.8	26.8
Painkalac	76.0	19.9	4.1

Table 4.7. Percentage time that the estuaries were in each state from 12/1998-2/2002.

The proportions of time in each state for both estuaries combined varied significantly with season (chi-square test, p=0.0017, 6 d.f.) with proportionally greater periods of closed states in summer and autumn and a larger proportion of time in perched states in winter and spring. Substantial differences in these patterns were observed between estuaries and years however. Over all seasons, Anglesea was closed less often than Painkalac (Table 4.7). This was the case in each season except spring, in which both estuaries were closed for just over 50% of the time. The duration of individual closed and perched periods were similar in both estuaries, but tidal periods were substantially longer in Anglesea than Painkalac.

Both estuaries changed from being predominantly closed/perched prior to April 2001 to be mostly perched/tidal after this date (Figure 4.11). While Painkalac was closed from March 1999 until April 2001, Anglesea was closed for only 62% of the time and was perched (30%) and tidal (8%) for the rest of this period. Following the floods of April 2001, Anglesea remained tidal for several months, while Painkalac returned quickly to a perched, then closed state. This pattern was repeated on a smaller temporal scale from December 2001.

4.3.5. Physical influences on estuarine states

The influence of freshwater flow on the hydrologic state of Anglesea and Painkalac estuaries was examined from two perspectives: in the context of conditions prevailing during periods of constant state and in the context of conditions during transitions between states. Factors considered in these examinations were freshwater inflow; and, in lieu of direct measurements of sediment transport:

- sea state;
- south-north wind vector; and
- west-east wind vector.

Measures of wind and sea state were obtained from the Bureau of Meteorology.

Mean daily freshwater flow was higher in closed than tidal states in both estuaries. There was little difference in mean daily flow between perched and tidal states in Anglesea, but the difference in flow was much more pronounced in Painkalac (Figure 4.12). For thirty five percent of the time that

Painkalac was closed there was no inflow (Section 3.3.4). This pattern suggests that, for both estuaries, freshwater flow played a role in maintaining perched and tidal states but that, for Painkalac, greater flow volumes were required to maintain tidal states than perched states. An alternative explanation for the difference in mean flow between tidal and perched states in Painkalac, but not Anglesea, was that the duration of high flow events required to maintain a tidal state was longer in Painkalac than in Anglesea. This alternative explanation does not appear to be the case as the durations of high flow periods were similar in both systems. It is more likely that the shorter tidal periods in Painkalac meant the highest flows (that initiate tidal states) represented a greater proportion of all flows to Painkalac during tidal periods and that longer tidal periods in Anglesea meant that lower flows were included in the average for that estuary (see Figure 4.14a). ‘Upwards’ transitions that ended these tidal periods are discussed in Section 4.3.5.a.

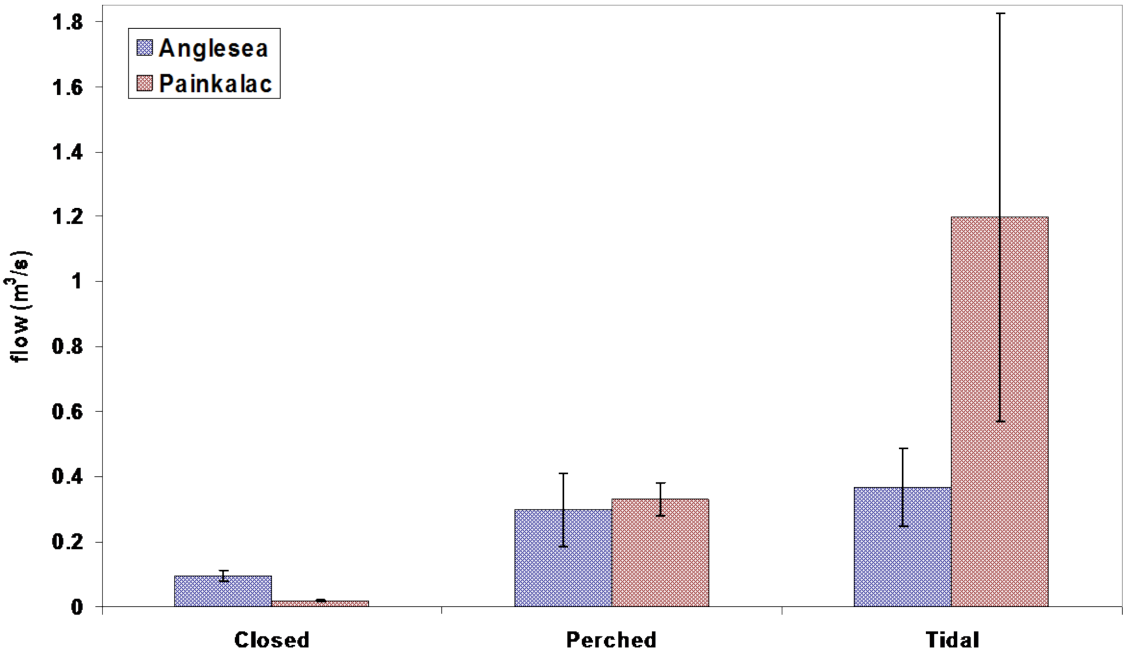
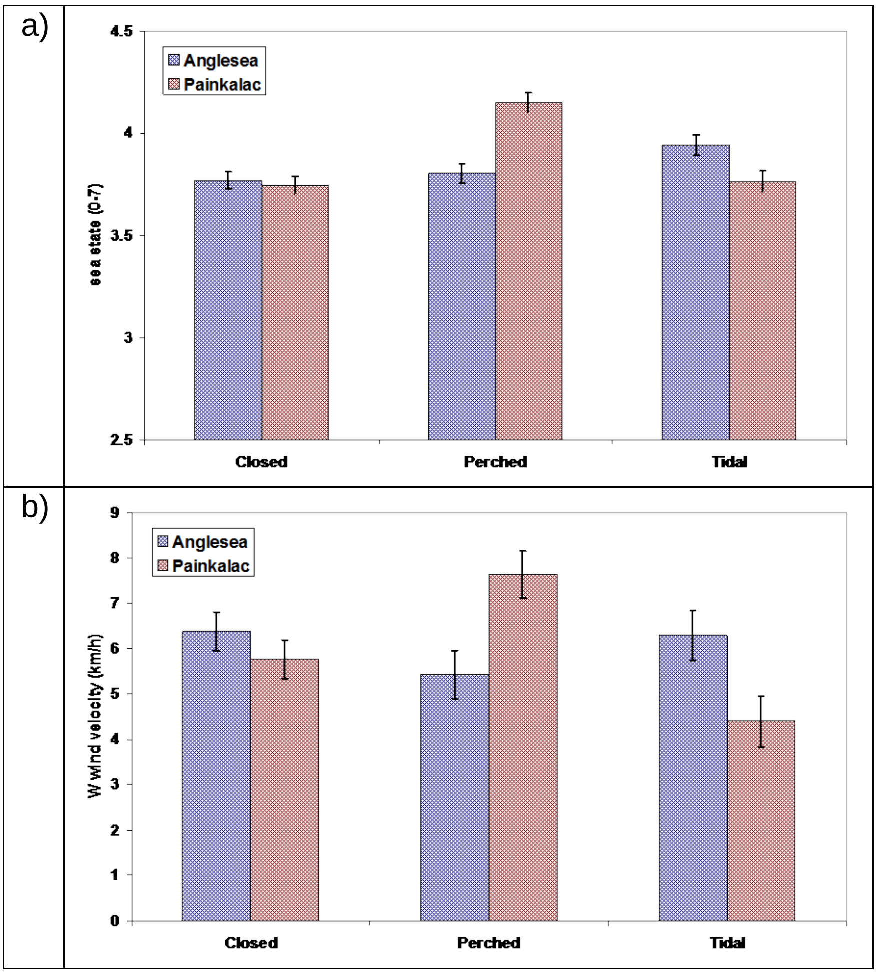


Figure 4.12. Mean daily flow rates (± std error) for closed, perched and tidal states in Anglesea and Painkalac estuaries. Where flow exceeded gauging stations, estimates of peak flows from Section 2.2.3 were used. Period sampled was 16/8/1999 to 18/2/2002.

Patterns of association between hydrologic state and both sea state and prevailing winds differed between estuaries. There were clear differences between the prevalent sea state and winds associated with perched states at

Anglesea and those associated with perched states at Painkalac, with smaller differences in sea state and winds during closed and tidal states. During perched periods at Painkalac, mean sea state and westerly wind component were higher than during perched periods at Anglesea (Figure 4.13a and b). Also, at times when Anglesea was perched there was a mean southerly wind component, while for Painkalac there was a mean northerly wind component (Figure 4.13c).



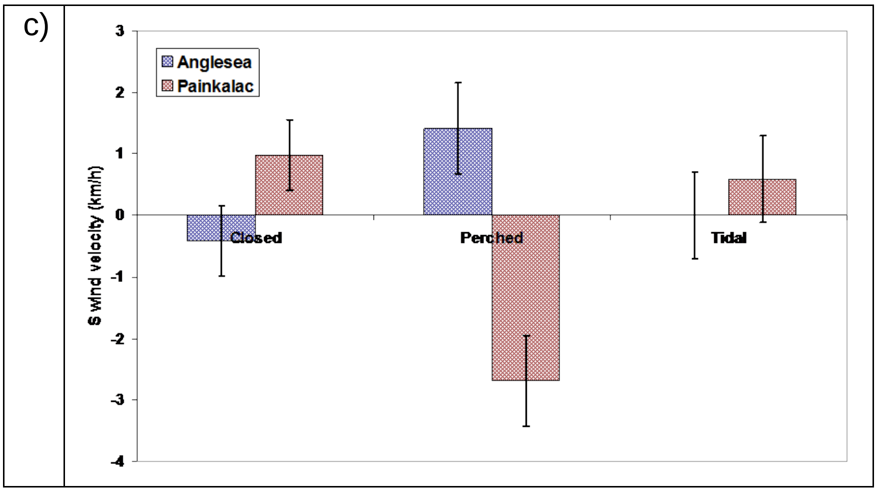


Figure 4.13. Mean (± std error) daily a) sea state as coded from 0 to 7, b) westerly wind vector and c) southerly wind vector for Anglesea and Painkalac estuaries during closed, perched and tidal states during the study period.

Differences during closed and tidal states were not as consistent. There were no clear differences in mean sea state. The westerly wind component was greater during tidal states at Anglesea than at Painkalac and the net north-south wind component was northerly for closed states at Anglesea and southerly for closed states at Painkalac (Figure 4.13).

The extent to which these observed differences either reflect non-causal but coincidental differences in timing of the various states in each estuary or reflect differences in coastal sedimentary processes with causal links to states in the two estuaries is difficult to determine. Given that flows during perched periods were similar between the two estuaries, it may be that the observed differences in wind and sea during perched states were related to coastal aspect, geomorphology and local marine sediment availability. Potential causal links are examined in the next section.

4.3.5.a. Transitions between states

'Downwards' transitions, in the direction from closed to tidal states tended to be more rapid than 'upwards' transitions, when the degree of closure increased in the estuaries.

'Downwards' transitions

Figure 4.14a shows that downward transitions of state were often closely associated with peaks in freshwater flow and that changes from closed to tidal states required larger flows than changes from closed to perched or perched to tidal.

Of the floods listed in Section 3.3.3, the largest was associated with changes from closed to tidal in Painkalac and perched to tidal in Anglesea. The second-largest flood was not associated with any change in state. Anglesea remained tidal, as expected, but Painkalac remained perched. This unexpected result for Painkalac may be due to high longshore sediment transport associated with a strong westerly wind and very high sea state at the time (Figure 4.14b, d).

The third largest flood at Anglesea was associated with a change from a closed to perched state, followed by a transition to a tidal state (Figure 4.14a). The December 2001 flood (third largest at Painkalac and fourth-largest at Anglesea) occurred in conjunction with a change from perched to tidal states in both estuaries. The fourth and fifth-largest floods at Painkalac were both associated with changes from closed to perched states. The fifth-largest flood at Anglesea, and sixth-largest at Painkalac, was not associated with a change in a perched state in either estuary, possibly due to a combination of relatively lower flows high westerly and southerly winds and a peak in sea state (Figure 4.14).

Peaks in freshwater flow appeared to account for three of four downwards changes in state at Anglesea (during the time for which continuous flow records are available) and four of six at Painkalac. The other downward change at Anglesea (closed to perched in June 2000) was associated with a sustained period of slightly increased flow that is similar to several others where no change of state was seen. This change did, however occur in combination with a period of high estuarine water levels, a strong westerly wind component and high seas and was probably a storm-associated event whereby waves overtopping the bar provided sufficient hydraulic head and erosion to precipitate a transition (Table 4.8).

The two downward transitions at Painkalac that did not coincide with flood flows were from closed to perched in July 2000 and July 2001. Both coincided with minor peaks in flow, the first was due to an artificial breaching of the estuary due to concern about flooding, the second was in association with a peak in sea state which may have contributed to an increase in water levels by overtopping of the sand bar (Figure 4.14a), d); Table 4.8).

'Upwards' transitions

The amount of sand movement required to partially or fully close the mouths of the estuaries is likely to be dependent on the energy of water movement through the mouths. Thus upwards transitions are more likely to occur at times of low freshwater flow. The magnitude of tidal flows will also affect the likelihood of upwards

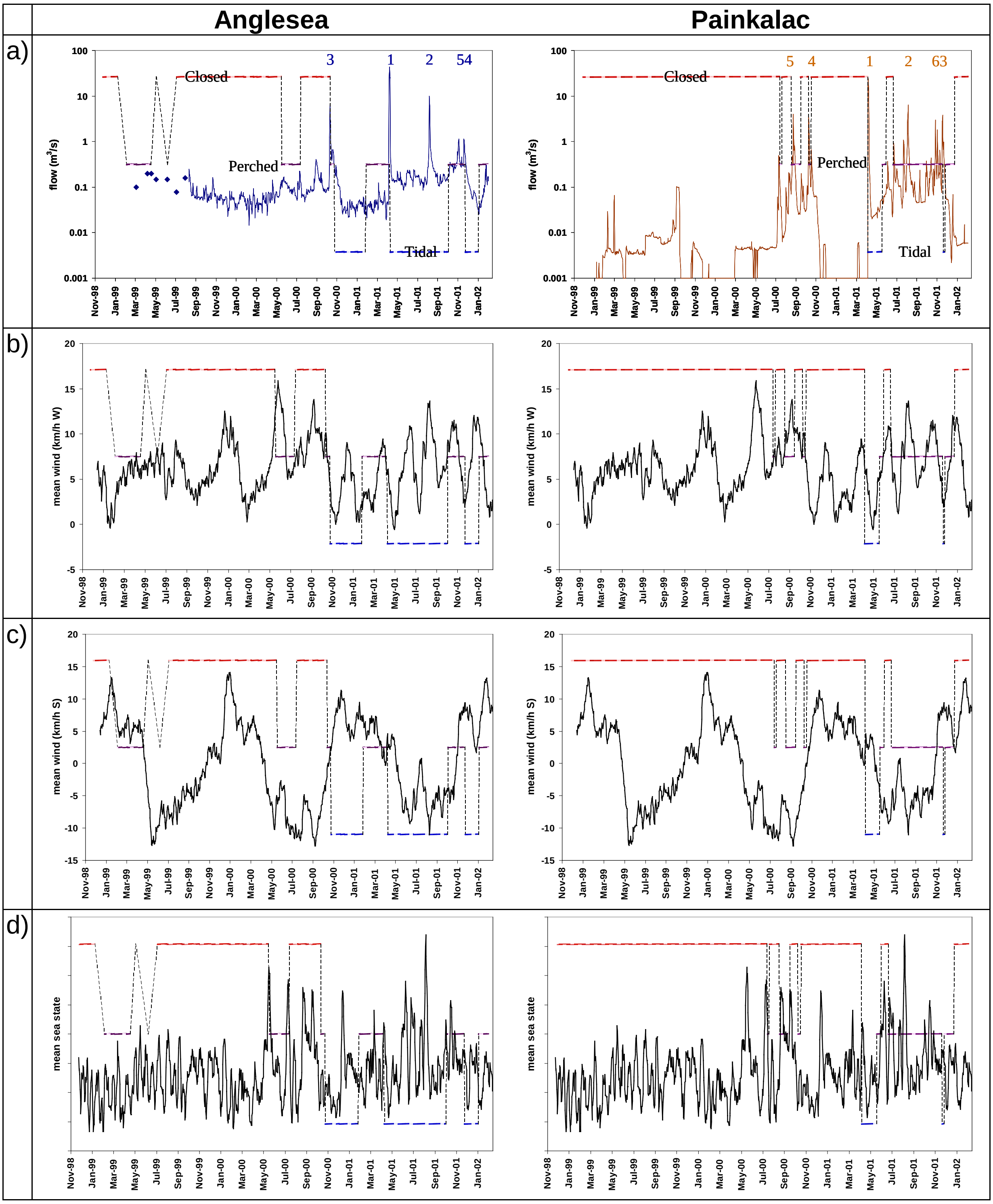


Figure 4.14. Estuarine states and; a) freshwater flow - floods indicated by rank, b) four-week moving average of the westerly component of three-hourly wind at Airey's Inlet, c) four-week moving average of the southerly component of three-hourly wind at Airey's Inlet and d) seven-day moving average of sea state at Cape Otway (Calm to Very High rated from 1-7)

transitions, for example, tidal flows would be smaller in transitions from perched states than from tidal states and smaller during neap tides than at spring tides.

Of the upwards transitions for which there are flow data, three of four at Anglesea and five of seven at Painkalac occurred at times of low flow relative to surrounding times (Table 4.8). Those that did not (perched to closed on 28/7/2000 for Anglesea, flow >0.2m^3/s, and perched to tidal at Painkalac on 1/11/2000 and 12/12/2001, flow >0.1m^3/s) coincided at Anglesea with a localised increase in sea state and an increase in the southerly wind component at Painkalac (Figure 4.14).

Only one occurrence of near-simultaneous upwards transitions of state was observed in the study period, in January 2002, with Anglesea moving from tidal to perched and Painkalac moving from perched to closed. On this occasion, a prolonged period with a high westerly wind component was the apparent cause, clearly supported by changes in channel location at Painkalac but not as obvious at Anglesea. Persistent westerly wind components were more commonly associated with upwards transitions at Painkalac (4 of 7) than at Anglesea (2 of 6) (Table 4.8). Contributing factors to this may be beach alignment (the beach at Painkalac faces more to the east than that at Anglesea) or a larger reserve of sand immediately westward of the mouth of Painkalac, possibly due to the extended closed period prior to and in the first part of the study.

The two upwards transitions at Painkalac associated with southerly wind components were also the only two upwards movements without particularly low freshwater flows suggesting that such conditions are a strong ‘upwards’ driver (Table 4.8). At Anglesea, the only upwards transition that was associated with southerly winds (February 2001) followed a time of a particularly deep channel through the bar.

Higher sea states were associated with stronger westerly wind components at times of upwards transitions, the exception being the transition of

Anglesea from perched to closed in July 2000 at which a high sea state was the only associated potential cause. This transition was potentially affected by the remains of a borrow area in the mouth area that was dug in late December 1999 and was still evident in July 2000.

					Potential cause(s)					
date		from	to	direction	Manual opening	High flow	Low flow	W wind 4-week	S wind 4-week	sea state
~18/02/1999	Anglesea	C	P	Down	?	?	N/A	N/A	N/A	?
~22/06/1999		C	P	Down	?	?	N/A	N/A	N/A	?
1/06/2000		C	P	Down	?	X	N/A	N/A	N/A	√
27/10/2000		C	P	Down	X	√	N/A	N/A	N/A	X
7/11/2000		P	T	Down	X	√	N/A	N/A	N/A	X
24/04/2001		P	T	Down	X	√	N/A	N/A	N/A	√
8/12/2001		P	T	Down	X	√	N/A	N/A	N/A	X
~18/05/1999		P	C	Up	N/A	N/A	?	X	X	?
~19/07/1999		P	C	Up	N/A	N/A	?	X	X	?
28/07/2000		P	C	Up	N/A	N/A	X	X	X	√
10/02/2001		T	P	Up	N/A	N/A	√	X	√	X
18/10/2001		T	P	Up	N/A	N/A	√	√	X	√
17/01/2002		T	P	Up	N/A	N/A	√	√	X	X
29/07/2000	Painkalac	C	P	Down	√	√	N/A	N/A	N/A	√
1/09/2000		C	P	Down	√	X	N/A	N/A	N/A	√
24/10/2000		C	P	Down	X	√	N/A	N/A	N/A	X
22/04/2001		C	T	Down	X	√	N/A	N/A	N/A	√
7/07/2001		C	P	Down	X	√	N/A	N/A	N/A	√
6/12/2001		P	T	Down	X	√	N/A	N/A	N/A	X
4/08/2000		P	C	Up	N/A	N/A	√	X	X	X
30/09/2000		P	C	Up	N/A	N/A	√	√	X	√
1/11/2000		P	C	Up	N/A	N/A	X	X	√	X
~4/06/2001		T	P	Up	N/A	N/A	√	√	X	X
16/06/2001		P	C	Up	N/A	N/A	√	√	X	√
12/12/2001		T	P	Up	N/A	N/A	X	X	√	X
9/01/2002		P	C	Up	N/A	N/A	√	√	X	X

Table 4.8. Changes of hydrologic state in Anglesea and Painkalac estuaries and potential cause(s). Highlighted dates indicate times when both estuaries experienced a similar change of state. Infrequent observations at Anglesea early in the study period did not allow exact dates of state changes to be examined for causes ("?"). N/A is shown where causes do not relate to the direction of the transition. C=closed, P=perched, T=tidal, X=cause of transition unlikely, √=potential cause of transition.

Mouth dynamics

Changes in the location of the entrance channels of the two estuaries provided further information about processes affecting state. While Nelson (1981) identified longshore drift as the cause of mouth closure at Anglesea, westward movement of the channel between 19/12/1999 and 30/5/2000, associated with upwards transitions of state and with prolonged episodes of both southerly and westerly wind, suggest a mixture of long-shore and cross-shore sediment movements as causal agents of mouth closure. The relative importance of onshore sediment movements at Anglesea may also be increased by available reserves trapped by the reef on the point immediately east of the mouth.

Painkalac also appeared to be influenced by both long- and cross-shore sand movements although southerly winds had a greater association with upwards transitions at Painkalac than at Anglesea during times when substantial freshwater flows were present (1/11/2000 and 12/12/2001). At these times, similar eastward movements of the entrance channel occurred. Overall, openings of the mouth at Painkalac tended to move the channel to a more central location, followed by movement of the channel eastwards over time.

Direct anthropogenic influences

In addition to changes in state and movements of the sand bars at the mouths of the estuaries due to natural causes, various direct human influences can play a part in mouth dynamics and maintenance of and transitions between states.

The mouth of Anglesea estuary has been subject to more structural alterations than the mouth of Painkalac. The most dramatic of these was the construction of a weir in 1975, the remains of which are still evident when the mouth of the estuary is deeply scoured. The presence of these remains may reduce the amount of sand removed from the entrance of the estuary during large opening events such as that in April 2001. Such a process would be consistent with historical aerial photographs, anecdotal evidence of a gradual incursion of sand into the estuary and a 1.4m shallowing at a site 320m upstream from the entrance between 1981 (Atkins & Bourne, 1983) and this survey, in 1999.

Artificial openings of intermittent estuaries in western Victoria are relatively common and are a topical management issue in the region due to flooding of infrastructure and farming land creating pressure for such openings but with the risk of negative environmental consequences such as fish kills (Barton & Sherwood, 2004). Two artificial openings occurred during the study period, both in Painkalac in July and September 2000. Both openings caused the estuary to move from a closed to perched state for a period of weeks before the estuary re-closed. In Painkalac, a trigger height used for artificial mouth openings (marked on the bridge support opposite the height benchmark) was at 2.1m AHD. The trigger for opening of Anglesea was near-inundation of the Great Ocean Road near the jetty at a height of approximately 2m AHD. The last manual opening of Anglesea estuary prior to the study was in November 1995 (M. Jackman, Surfcoast Shire, pers. comm..)

4.3.6. Inundation regimes and tidal prisms: Anglesea

The changes in water levels associated with each state lead to markedly different inundation regimes for shallower sections of the estuaries in that large areas that are aquatic (*i.e.* permanently submerged) in closed periods become intertidal or terrestrial in perched and tidal periods. Areas most affected by changes in inundation regime were of low relief and at elevations between the lowest tides and the higher levels at which water is consistently held by the sandbars. Figure 4.15 shows frequencies of inundation for the lower portion of Anglesea estuary during each of the three states. Relative to the estuary as a whole, a greater proportion of the lower estuary was exposed to the air more often during tidal and perched periods with at least 58% of this area exposed at some stage during these periods compared to 12% of the total area during closed periods.

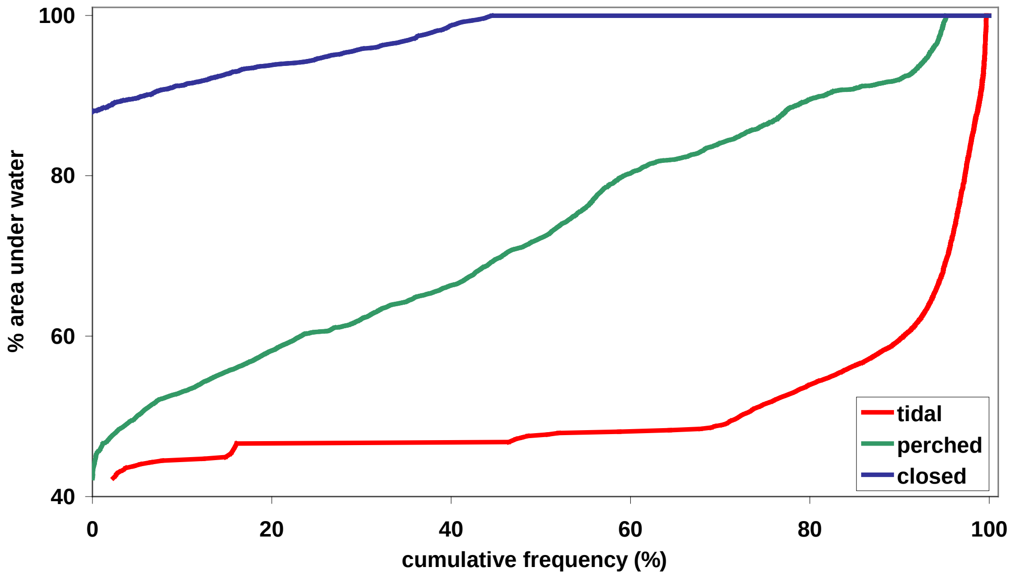


Figure 4.15. Frequency of inundation of substrate for lower Anglesea estuary (below the Great Ocean Road bridge – *i.e.* seagrass habitat) for each state during the study period. Frequencies were derived from logged height data and the bathymetric model with a maximum elevation of 1.49m AHD.

Due to the paucity of data on changes in water level below 0.8m AHD, the daily tidal prism for tidal periods was calculated based on the maximum tidal range manually recorded on one day. Examples of neap and spring prisms are shown in Table 4.9, the tidal prism of Anglesea can vary by 80,000m^3 during tidal periods while there was a greater range in perched periods, with a similar maximum but a negligible minimum. These tidal prisms are a significant fraction of the estuary volume in most cases, indicating that during these states, residence time of water in the estuary is often short (days).

	Date	High tide (mAHD)	Prism (m^3)	Max vol. (m^3)	% of vol.
Tidal (neap)	6/12/2000	0.834	13,900	74,400	19
Tidal (spring)	25/4/2001	1.263	94,000	122,600	77
Tidal (mean, n=10)		1.062	44,500	100,100	44
Perched (min. neap)	5/3/2001	1.049	557	96,100	0.58
Perched (max. spring)	19/10/2001	1.555	91,800	168,800	54
Perched (mean, n=223 days)		1.166	19,200	112,000	17

Table 4.9. Tidal prisms for neap, spring and mean tides in tidal and perched states in Anglesea. The size of the prism is also shown as a percentage of the maximum total estuary volume for each case.

4.4. Summary and conclusions

Based on visual assessment of plotted water level variation in Anglesea, a three state model of tidal influence was identified based on full, partial or zero tidal influence (tidal, perched and closed states respectively). To examine the validity of this model and to quantify the relationship between water level and the hydrologic states, a classification tree analysis was used to construct bivariate rules for classification of time periods as ‘closed’, ‘perched’ or ‘tidal’ based on statistical summaries of water level and the 10-minute rate of change in water level in Anglesea. A high degree of success in the creation of these rules supported the use of a three-state model. When tested with a shorter –term dataset from Painkalac, only one of seven derived rules (relating to kurtosis of the distribution of water levels) was not transferable from Anglesea using slightly modified cut off values.

This three state model was also used as a basis for a separate categorisation using observations of marine influence made at the mouths of the estuaries. When these observational categorisations were compared with logged water level fluctuations at Anglesea, a tendency for overestimation of tidal influence in the mouth observations was found, usually associated with larger seas washing across the bar into the estuary. As estimations of state for some of the study period at Anglesea, and most of the time at Painkalac, were solely based on these observations, further information was used to create 'integrated' measures of state (see Section 4.3.4 and Appendix F).

Periods of low flow were associated with closed periods, during which Anglesea had relatively high and constant water levels and water levels in Painkalac estuary sometimes declined with evaporation. Low inflows were also associated with 8 of 11 'upwards' transitions. Floods (and artificial openings) were closely associated with 'downwards' transitions between states in both estuaries, as for intermittent estuaries elsewhere (e.g. Pollard, 1994; Young & Potter, 2002; Stretch & Zietsman, 2004). Both long-shore and cross-shore wind influences were related to mouth closures, with cross-shore influence strongest at Painkalac. High seas were related to both upwards and downwards transitions. The state of the estuary strongly influenced frequencies of inundation of substrate in the lower estuaries.

5. Estuarine Stratification and Water Quality

5.1. Introduction

Water quality of estuaries is determined by a combination of the mixing of fresh and marine waters, each with their own characteristics, and internal physico-chemical and biological processes. In terms of mixing, salinity is almost always the most important water quality variable and is a standard measure used to assess mixing processes (Hodgkin, 1994). Salinity itself is a dimensionless measure of the salt content of water and ranges from zero in distilled water to ~35 in the open ocean. Areas along coastlines, in estuaries and in inland waters can have salinities much greater than 35 (hypersalinity) due to concentration of salts by evaporation. Because of its salt content, sea water is more dense than fresh water and the two do not readily combine without a source of energy to create mixing, for example by currents created by freshwater flow, wind and tides. Without this mixing energy, fresh water flowing on to salt water will form a separate layer on top, a phenomenon referred to as salinity stratification. The degree of stratification can have major effects on chemical and biological processes within an estuary, and is often used to classify an estuary (e.g. Rochford, 1951; 1959; Eyre, 1998).

To investigate the relationship between freshwater inflow, hydrologic state, stratification and water quality, five variables were measured: temperature, dissolved oxygen, nutrients, pH and water clarity.

5.2. Methods

Physico-chemical variables in estuarine waters were measured in depth profiles at two spatial scales, both longitudinally within the Anglesea and Painkalac estuaries and regionally at single sites in estuaries along the Otway Ranges.

5.2.1. Sites

5.2.1.a. Longitudinal surveys – water quality

Six sites along the length of Anglesea and five sites along the Painkalac were sampled for physico-chemical measures (Figure 5.1). Coordinates and site descriptions are given in Appendix F.

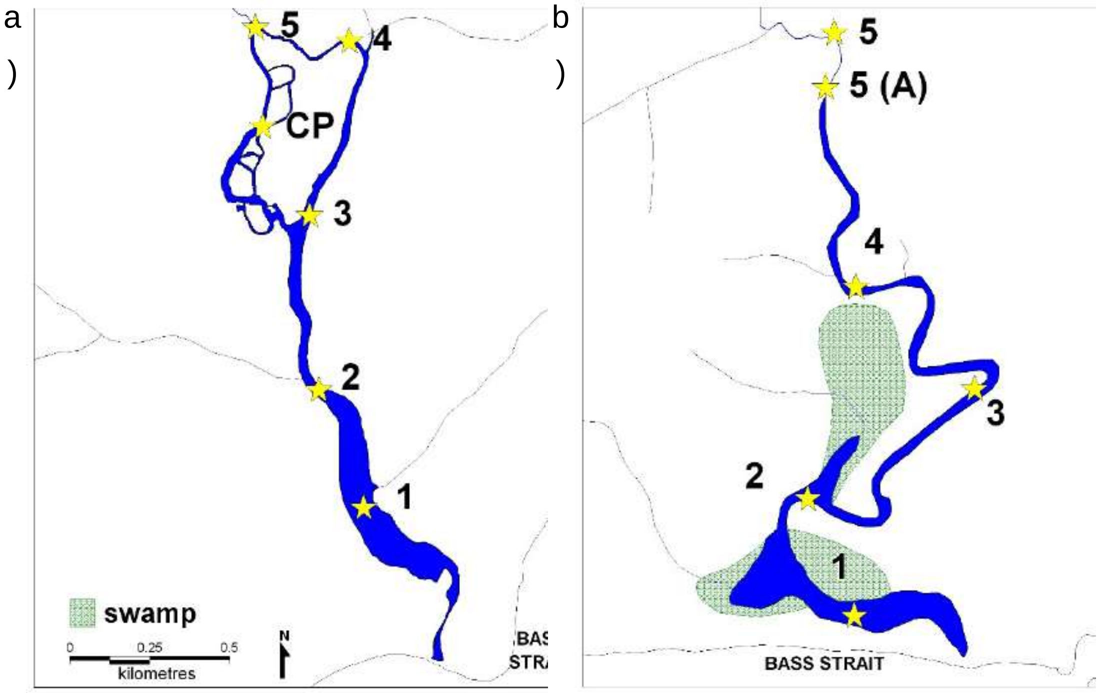


Figure 5.1. Locations of longitudinal survey sites in a) Anglesea and b) Painkalac estuaries. Scales are identical. Area marked as swamp at Painkalac was identified from Royal Australian Survey Corps 1:25000 map and consists largely of emergent macrophytes that were infrequently inundated. CP – Coogoorah Park site, 5(A) in Painkalac was sampled on occasion when Site 5 was inaccessible (see Appendix F).

5.2.1.b. Otways estuaries

Eight other estuaries in the region (Figure 2.1) were sampled at three-month intervals to provide a broader context for results from Anglesea and Painkalac. Despite geographic differences between catchments such as size, steepness and rainfall (see Chapter 2), an examination at this scale was important given existing anthropogenic modifications of the morphology and hydrology of both Anglesea and Painkalac estuaries which may have resulted in these estuaries being atypical of the region.

At least one location in each Otway estuary was sampled, where physico-chemical profiles and/or notes on seagrasses, including shoot density, were recorded (Table 5.1). Details of sampling are given in Section 5.2 (profiles) and Section 6.2 (shoot density). Coordinates for these sites are given in Appendix F.

Estuary	Site	Distance upstream (km)	Depth profile	Seagrass observations
Barham	DS	0.15-0.80	X	√
	GOR	0.82	√	X
	US	1.33	√	X
Skenes	*GOR*	0.16	√	X
Kennett	*GOR*	0.07	√	√
Wye	*GOR*	0.12	√	√
St Georges	*GOR*	0.32	√	√
Erskine	*DS*	0.19	√	√
	GOR	0.73	√	X
Spring	*DS*	0.20	√	√
Thompsons	*DS*	0.13	√	X
	US	1.46	X	√

Table 5.1. Sites within regional estuaries along the Otways and central Victorian coast from west to east. GOR=Great Ocean Road bridge, US=upstream, DS=downstream, near mouth. Sites used for water quality comparisons were located in morphologically equivalent parts of the lower sections of the estuaries and are italicized. Sites for which physico-chemical depth profiles or seagrass observations were made are also noted.

5.2.2. Sampling details

In Anglesea, measurements were made approximately monthly at all sites from December 1998 to February 2002. In Painkalac, measurements were made quarterly at all sites and monthly at Site 2 between March 1999 and February 2002. The eight regional estuaries were sampled on a quarterly basis for all variables except concentrations of nutrients and suspended solids. Details of site locations and the dates and depths sampled are given in Appendix F.

Variables measured were salinity, temperature, dissolved oxygen and nutrient concentrations, pH, Secchi depth, concentrations of suspended solids, turbidity and oxidation-reduction potential (ORP). Secchi depth was measured once per site/time and concentrations of nutrients and suspended solids were measured in near surface waters and in bottom waters when salinity stratification existed. All other variables were measured at the surface and then at 0.5 m depth intervals. When possible, sample depths were corrected to AHD using logger data. At other times spot measurements at surveyed height datum points were used for measurement of water heights at the time of sampling. Periods for which each variable was sampled are detailed in Chapter 3, as are methods for each variable (except Secchi depth, which is discussed in Section 5.3.6). Table 5.2 lists the number of times each variable was sampled at each site.

	Anglesea						**Painkalac**				
	1	**2**	**3**	**4**	**5**	**CP**	**1**	**2**	**3**	**4**	**5**
Temperature (ºC)	40	46	39	36	38	34	21	40	17	16	16
Salinity	40	46	39	36	38	34	21	40	17	16	16
Dissolved Oxygen (%)	37	39	36	34	35	32	21	36	17	16	16
pH	35	40	34	34	33	32	21	34	17	16	16
Secchi depth (m)	36	35	35	34	34	33	19	29	17	16	16
Turbidity (NTU)	10	13	9	9	9	9	11	13	8	7	7
Redox potential	10	13	9	9	9	9	11	13	9	8	8
TSS (mg/L)	8	1	8	1	8	8	3	6	3	2	3
Nutrients	10	1	10	1	10	10	4	8	4	3	4
Total times visited	40	46	39	36	38	34	21	40	17	16	16

Table 5.2. Number of times each variable was sampled per site during the study period. "Nutrients" represents four separate measures; total P, soluble reactive P, total N and NOx and were measured, along with TSS, to 8/2/2000 only. Turbidity and redox potential were measured from 30/9/2000 onwards.

Additional data on estuarine water quality were obtained from Alcoa's long-term monitoring program. These data provided water quality information for surface waters at the head and middle of Anglesea estuary monthly (see Section 3.2.2.d and Figure 3.1a).

5.3. Results and discussion

5.3.1. Salinity

5.3.1.a. Salinity ranges and frequencies

Overall, the range of salinity in Anglesea (0.0 to 35.6) was less than that in Painkalac estuary (0.0 to 51.9). The difference in the maxima was due to hypersalinity developing twice in Painkalac, in the summers of 1998/99 and 2000/01, but not in Anglesea. Hypersalinity was associated with periods of no flow in Painkalac combined with separation of the estuary from the ocean by a sand bar. The continuous fresh water inflow into Anglesea (at a greater rate than evaporation) ensured a maximum salinity around that of sea water. Minima in both estuaries were recorded at times of high freshwater flow.

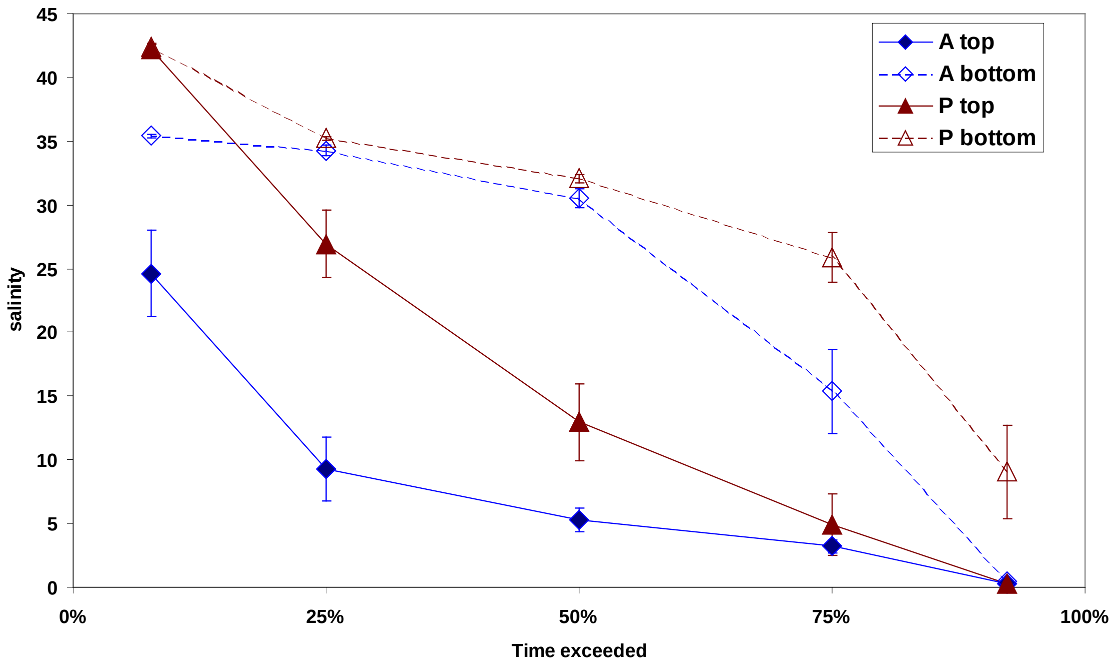


Figure 5.2. Mean (± s.e.) percentile exceedances for salinity for all sites in Anglesea (A) and Painkalac (P) estuaries over the 13 times on which both estuaries were longitudinally sampled.

Over the study period, salinity distributions of top and bottom waters in Painkalac were consistently greater than the equivalents in Anglesea (Figure 5.2). Most of the time, surface waters of Anglesea were relatively fresh with a salinity between ~4 and 10. This reflects the consistent inflow of waters with a salinity of between 3.5 and 1, becoming lower only in flood conditions.

While the salinity of inflowing waters to Painkalac was lower than at Anglesea (always below 0.4 from the mainstem and typically below 1 from Distillery Creek), surface salinities in the estuary were usually greater than Anglesea due to greater mixing and relatively smaller and more intermittent freshwater input (see Section 5.3.1.d). Frequency of near-marine bottom salinities in Painkalac were also greater than those in Anglesea. For at least 75% of the time though, Painkalac was not hypersaline. At no time did Anglesea estuary become hypersaline (max of ~35: Figure 5.2).

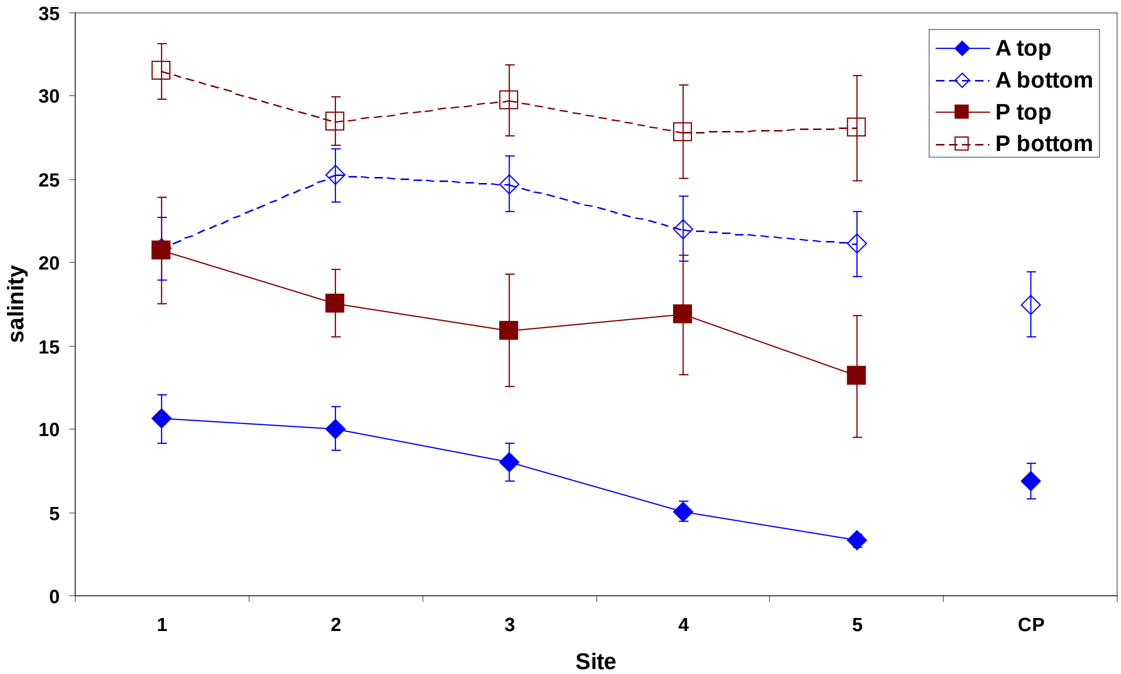


Figure 5.3. Mean salinities (± std error) for equivalent sites along Anglesea (A) and Painkalac (P) estuaries. Sites 1 to 5 were in equivalent positions along the estuary from downstream to upstream. CP = Coogoorah Park, a site in the artificial channel network of the upper Anglesea estuary. *n* was variable between sites (see Section 5.2, Appendix F).

On a site-by-site basis, surface and bottom waters of each Painkalac site had greater salinities than the equivalent longitudinal sites in Anglesea. Surface salinities at sites in both estuaries were consistently smaller than bottom salinities. The salinity of Anglesea surface waters was typically less variable than the salinity of bottom waters in Anglesea and both surface and bottom waters of Painkalac (Figure 5.3).

On average, salinities of surface and bottom waters in both estuaries increased from upstream to downstream consistent with typical horizontal and vertical salinity gradients (Figure 5.3). Vertical differences within sites were equivalent in magnitude to the horizontal differences in mean salinity along the length of the estuaries.

Compared to other estuaries in the region during the study, both Anglesea and Painkalac estuaries had higher surface-water salinities than most estuaries to the west and lower surface salinities than estuaries to the east. Salinities of bottom waters were comparable to those of Erskine, St George and Kennett estuaries to the west and Thompson to the east (Figure 5.4, Table 5.3).

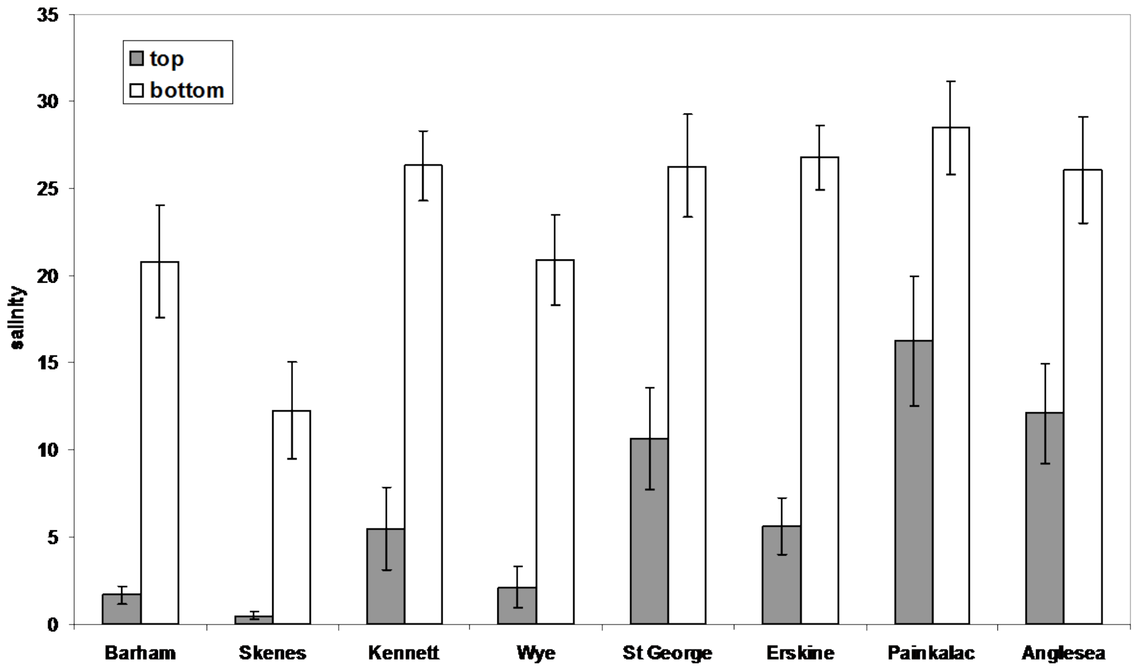


Figure 5.4. Mean surface and bottom salinities (± std error) at sites in estuaries sampled along the Otway Range from west to east from March 1999 to January 2002. Painkalac and Anglesea are represented by Sites 2, at approximately the same location in the estuaries as the other sites. *n*=13 for all sites with the following exceptions. Skenes (*n*=11: missing 6/00, 6/01); Kennett & Wye (*n*=12: missing 6/01). Spring & Thompsons Ck are not shown as they were included late in the study, and for those estuaries *n*=4.

Estuary	Painkalac	Anglesea	Spring	Thompson
top	15.1 (9.1)	11.4 (5.2)	26.8 (4.6)	36.4 (1.3)
bottom	34.9 (0.9)	33.8 (1.2)	27.2 (5.7)	36.5 (1.3)

Table 5.3. Means (and std errors) of salinities in surface and bottom waters of estuaries for times at which Spring and Thomson were sampled (*n*=4), (3,6,11/01, 1/02).

5.3.1.b. Temporal patterns: Site 2 Anglesea & Painkalac

As a result of the sampling design (Section 5.2), salinity was measured more frequently at Site 2 than at other sites in Painkalac. Comparisons between this site, and the equivalent site in Anglesea (Site 2) provides the highest level of temporal resolution for comparison between the two estuaries (Figure 5.5).

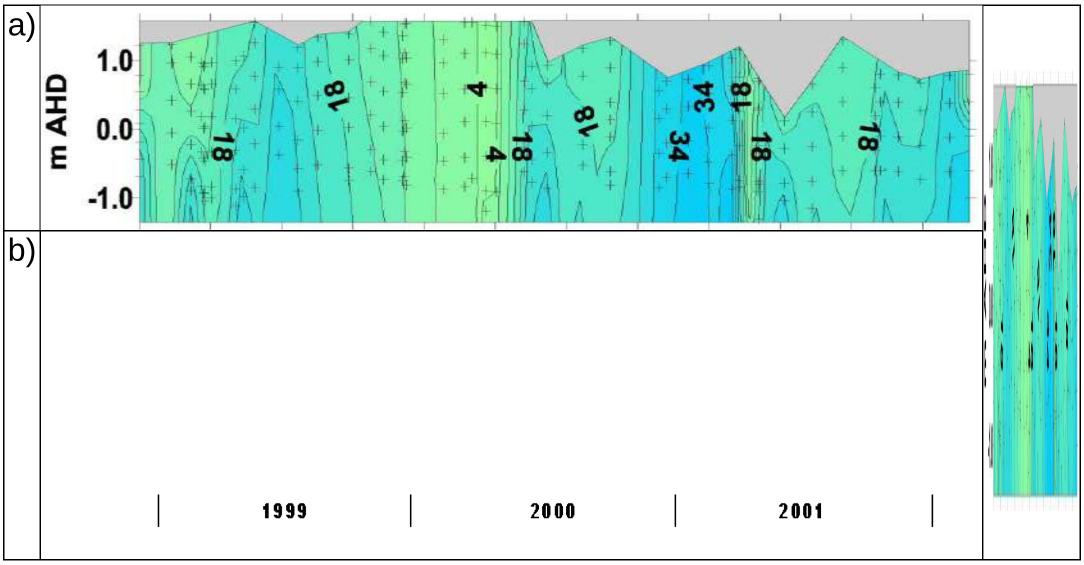
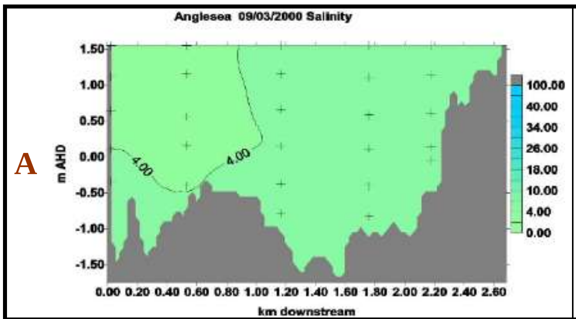


Figure 5.5. Vertical salinity structure of site 2 at a) Anglesea and b) Painkalac though the study period (8/12/1998 (Day 0) to 19/2/2002 (Day 1169)). Shaded areas at the top of each figure reflect changing water levels at the sites, the shaded area at the left of b) represents time from the start of the study until Painkalac was sampled. Crosses represent sampling points across depth and time.

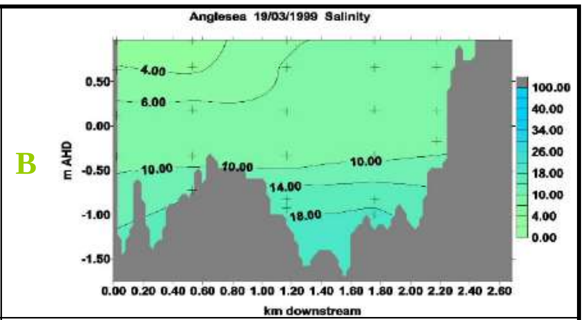
Anglesea was less saline than Painkalac for most of the study period. Initially, Site 2 at Anglesea was stratified with surface salinities less than 18. During this period, Site 2 at Painkalac was well mixed and initially hypersaline, reducing to salinities around 30 by early May 1999 (~day 150). Following this, Anglesea became more saline, as Painkalac became fresher and mildly stratified (up to mid-August (day 250)). Both estuaries then entered a relatively long phase of vertical homogeneity, Anglesea becoming more saline until mid-May 2000 (~day 525), Painkalac with salinity increasing to a maximum of 19 in late March (~day 465) and then decreasing until the end of July (day 600).

An influx of marine waters in Anglesea led to a second period of stratification at Site 2 over winter and early spring (mid-May to mid-October (~days 525 to 675)), following which Anglesea was less stratified and relatively marine until April 2001. From the end of July to early November (~days 600 to 700), Painkalac had moderate salinities and was stratified. Following this, the estuary was vertically mixed and salinity increased over summer to be hypersaline by April 2001. At this time both estuaries experienced a major change, coinciding with the largest flood observed in the study (on day 866). Both estuaries then remained stratified to varying degrees with moderate salinities until the end of the study period (day 1169, 19/2/2002).

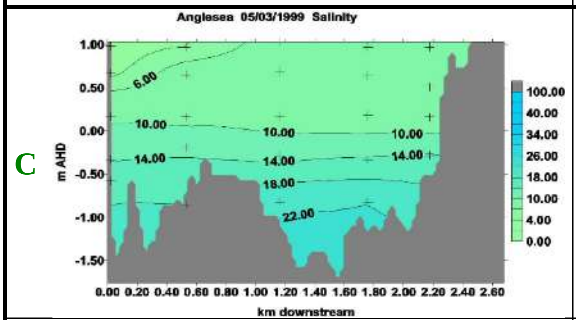
5.3.1.c. Temporal patterns: Longitudinal, Anglesea & Painkalac



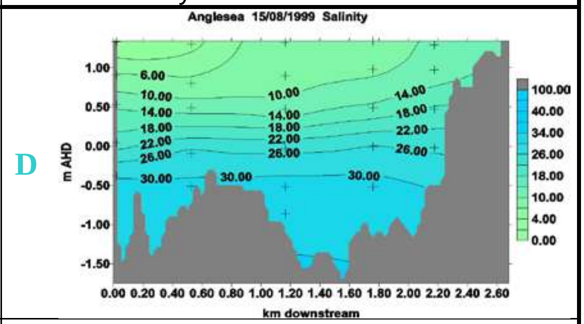
Estuary all fresh water: salinity ≤ 6, no stratification



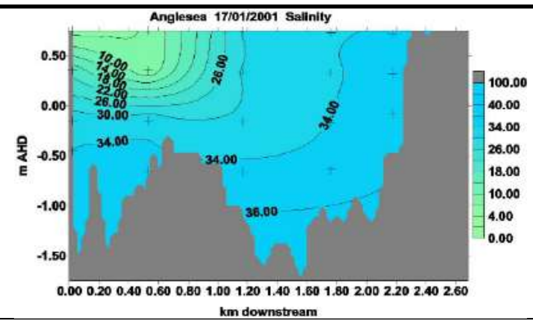
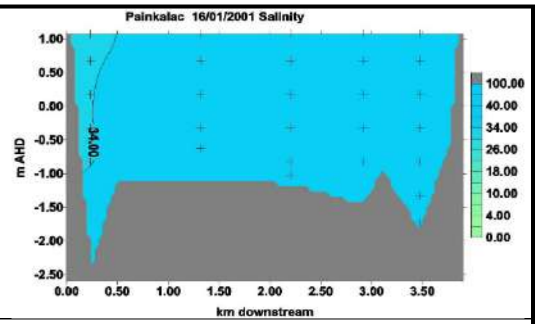
Fresh water throughout the surface of the estuary (<10 to >1m); vertical stratification, salinity 10-20 in bottom waters



Fresh water on surface of estuary (<10 typically >1m deep), often with a fresh water lens upstream. Vertical stratification, bottom waters with salinities between 20 and 30.



Surface <10 (typically <1m deep); stratified, often with a fresh water lens upstream. Bottom >30.

<table>
<tr>
<td>E </td>
<td>F </td>
</tr>
<tr>
<td>Classic ‘salt-wedge’ stratification, marked horizontal and vertical salinity differences. Surface salinity <10, bottom waters >30.</td>
<td>Very little to no freshwater influence or stratification, bottom salinities >30. Only seen in Painkalac. Includes periods of hypersalinity.</td>
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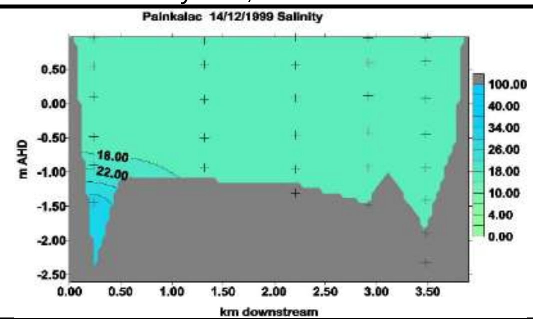
<td>G </td>
<td></td>
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<tr>
<td>Well mixed, salinity 10-25 (possible pockets of bottom water >30). Only seen in Painkalac.</td>
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Table 5.4. Examples of longitudinal patterns of salinity stratification observed in Anglesea and Painkalac estuaries in order of increasing marine influence (except “G” – well-mixed at intermediate salinity).

The patterns observed in detail at Site 2 were reflected in temporal changes of the longitudinal salinity profiles of the estuaries. Seven patterns of salinity structure were identified in Anglesea and Painkalac estuaries during the study, described, with graphical examples, in Table 5.4. Four of these patterns (B to E) represented different patterns of stratification and three (A, F and G) represented patterns of vertical mixing. Of the patterns of stratification, A and B were only observed in Anglesea, while F and G were only observed in Painkalac (Figure 5.6). Patterns D and E were, at times, reflections of the phase of tide in which the estuaries were sampled, with the halocline moving upstream on incoming tides.

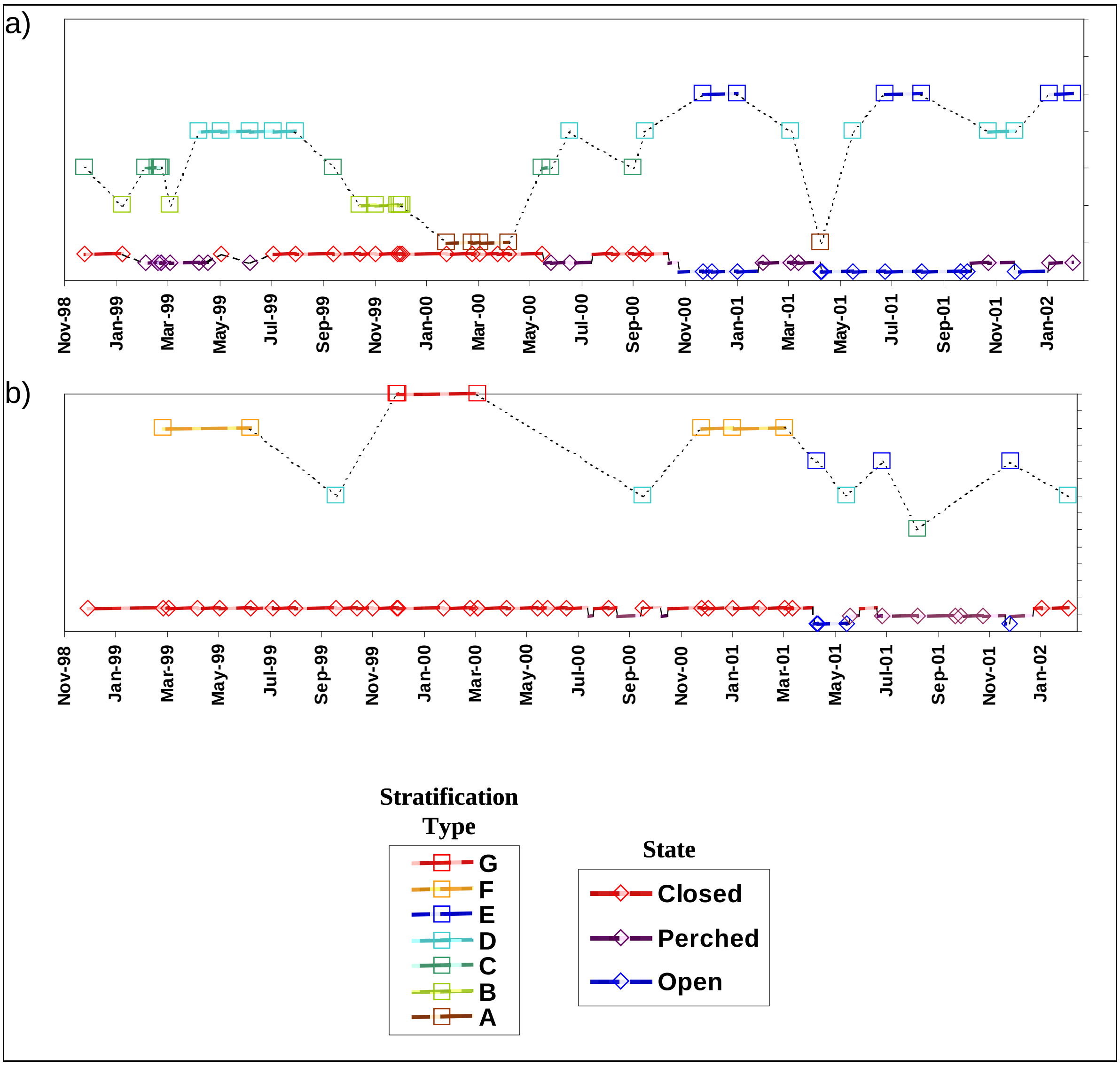


Figure 5.6. Longitudinal stratification patterns and hydrologic states in a) Anglesea and b) Painkalac estuaries though time. Salinity of stratification types increases vertically from 'A' to 'F', pattern 'G' was well mixed, at an intermediate salinity. Square and diamond shaped markers indicate sampling times, state changes without associated markers were recorded using loggers or unpublished data (P. Reilly).

The Anglesea estuary was stratified from December 1998 to February 2000 with varying degrees of marine influence (stratification patterns B, C and D). Between February 2000 and May 2000 it destratified and was predominantly fresh (pattern A). From May 2000 onwards, the estuary was stratified (patterns C, D and E), with a relatively greater marine influence following the floods of April 2001, during which the estuary was flushed with fresh water (pattern A) (Figure 5.6a).

The Painkalac estuary was well mixed during most of the first two and a half years of the study period (patterns F and G), but showed stratification between July and October 1999 and in October 2000 (pattern D). During this time, salinity increased to become hypersaline during summer and decreased during winter. In March, May and June 1999 there was a very thin (<50cm) layer of fresh water overlying a well-mixed body of water with salinities in the mid-thirties (pattern F). Following the April 2001 floods and deep breach of the mouth, the pattern of stratification changed to highly stratified (patterns D and E), with salinities greater 30 in bottom waters, except for August 2001 (pattern C) (Figure 5.6b).

5.3.1.d. Influences of state and freshwater flow on stratification

Longitudinal stratification patterns observed in Anglesea and Painkalac estuaries were dependent on a combination of estuary, state and freshwater inflow (Figure 5.7). Stratification (patterns B to E) was typical in both estuaries during tidal and perched states. When the estuaries were closed, they tended to mix and destratify but with differing patterns of salinity. In association with low freshwater inflows, Anglesea gradually became fresher and better mixed when closed, eventually reaching pattern A. In similar circumstances, but with zero inflow, Painkalac destratified more quickly to a uniform brackish salinity (pattern G) and then, with no further inflow, became uniformly hypersaline (pattern F).

During tidal states in both estuaries, only patterns D and E were observed, with the exception of short-term complete freshwater flushing (pattern A) observed during a flood in Anglesea (Figure 5.7). Whether pattern D or E was observed depended on the state of the tide; pattern D was associated with an outgoing tide, pattern E was associated with an incoming tide pushing the halocline upstream.

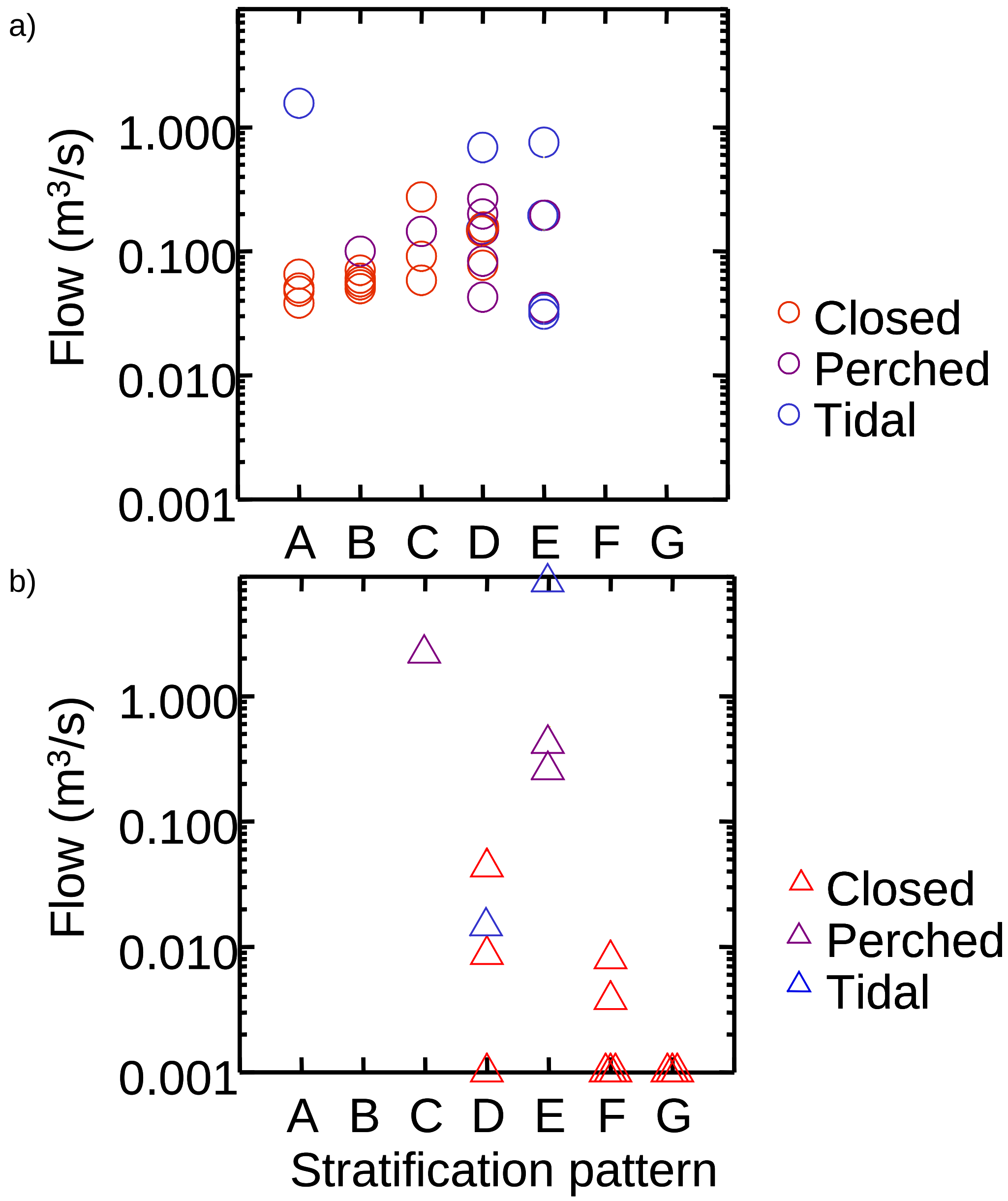


Figure 5.7. Freshwater inflows associated with times that each stratification pattern was observed in a) Anglesea and b) Painkalac estuaries (A=Anglesea, P=Painkalac). States are identified by colour. Flow was measured as mean flow for the week preceding salinity measurement except for the first six flow measurements for Anglesea, which were instantaneous flow on the day of sampling. Where flow was zero in Painkalac it was substituted with 0.001m^3/s to allow presentation on a log scale.

In perched states, salinity patterns in Anglesea ranged between B and E on the thirteen sampling occasions, with patterns C and D being most common (Figure 5.7a). During times when Painkalac was perched, patterns C and E were observed once each (Figure 5.7b).

In Anglesea, the patterns of stratification observed during perched periods reflected the previous state and stratification pattern, the amount of freshwater flow and the degree of incursion of marine waters. Overall, salinity tended to move towards intermediate levels during perched periods (Figure 5.6a). The first perched period in Anglesea was associated with an influx of marine waters that increased salinity of bottom waters to >20 (pattern B to C) followed by a reduction in salinity (pattern B) towards the end of the period as the estuary re-closed and fresh water inputs became the major influence on salinity. The second perched period in which there were multiple measurements (June-July 2000) became more saline between the start and middle of the period, moving from pattern C to D. The final perched period with more than one observation (Jan-Feb 2001) showed ‘salt wedge’ stratification (pattern E) on both occasions.

In Painkalac, the patterns observed while the estuary was perched reflected relatively high freshwater flows (in July and August 2001) flushing saltwater downstream and out of the estuary. No longitudinal surveys were done in Painkalac in other perched periods.

Patterns of stratification were most different between estuaries when the estuaries were closed (Figure 5.7). The only stratification pattern not observed in either estuary during these periods was E (‘salt wedge’) and the only pattern in common between the estuaries was D, observed early in closed periods.

In Anglesea, the estuary was stratified at all times when closed, except for a period in which salt-water had been effectively flushed from the system (pattern A, Feb-April 2000). During the long closed period in Anglesea (July 1999 to May 2000) a steady decrease in salinity (from pattern D to A) was observed, consistent with the continuous freshwater input and no marine-water incursions (Figure 5.6a).

In Painkalac, closed periods tended to be associated with little to no freshwater flow (Section 4.3.5). While closed, Painkalac was only stratified on three of eleven sampling occasions (Figure 5.6b, Figure 5.7b). At the start of the longest closed period, Painkalac was well-mixed and highly saline (pattern F). Sustained flow from mid-September 1999 led to a reduction in salinity and stratification in October 1999 (pattern D) (with overtopping of the bar by marine waters possibly contributing to stratification). This was followed by destratification at lower salinity (pattern G) from December 1999. In the next prolonged closed period (Nov 2000 to Apr 2001) the estuary was also not stratified and was saline (pattern F), with salinity increasing from ~30 to ~40 (hypersaline) due to a lack of freshwater input and evaporation.

5.3.2. Temperature

Temperature varies both seasonally and diurnally and tends to vary more in fresh waters and shallow areas of estuaries than in deep estuarine and marine waters. It can affect estuarine organisms both directly and indirectly. Each species has an optimal temperature range associated with their physiology so changes in ambient temperatures can affect organisms directly or through interactions with other organisms with different temperature ranges (e.g. Kerr & Strother, 1985; Bulthuis, 1987; Bintz *et al.*, 2003; Caffrey, 2004). Temperature also affects other aspects of water quality such as the amount of dissolved oxygen that a given body of water can hold at saturation and the rate of chemical processes (e.g. Weiss, 1970; Cosovic & Kozarac, 1993).

5.3.2.a. Temperature ranges and frequencies

Overall temperatures in Anglesea estuary were more often higher than those in Painkalac, although ranges were similar (Table 5.5). In Anglesea, the temperature distribution of bottom waters was consistently warmer than that of surface waters. In Painkalac, surface waters had a greater mean range and so were both warmer and cooler than bottom waters through the study period.

Compared to the eight other estuaries sampled, mean temperatures at Site 2 in both Anglesea and Painkalac were within the range of mean temperatures of similar locations in other estuaries (Table 5.5). Maxima and minima were slightly higher than those of most other estuaries while ranges were similar to all those except Spring and Thompson, which were only sampled on a limited number of occasions.

Patterns of mean temperature at sites along Anglesea and Painkalac estuaries differed (Figure 5.8). In Anglesea, mean bottom temperatures were greater than mean surface temperatures at all sites; the difference between the two increasing upstream. In Painkalac, mean temperature was greatest in surface waters at downstream sites but higher in bottom waters upstream.

Estuary	**Mean**	**Minimum**	**Maximum**	**Range**	***n*** [a]
Barham	15.9	9.3	23.8	14.5	26
Skenes	16.4	9.4	24.6	15.2	22
Kennett	17.0	8.4	26.1	17.7	24
Wye	17.8	9.2	29.6	20.3	24
St George	15.8	8.1	24.1	16.1	26
Erskine	17.3	8.8	26.0	17.3	26
Painkalac	17.3	10.0	26.6	16.7	26
Anglesea	18.1	10.1	25.5	15.4	26
Spring[b]	18.5	14.8	25.2	10.4	6
Thompson[b]	17.5	14.7	22.4	7.8	8

Table 5.5. Summary statistics for estuarine water temperature (°C) on occasions where several estuaries were sampled during the study. [a]: *n* includes top and bottom measurements from single sampling occasions, [b]: Spring and Thompson Creeks were sampled on limited occasions in the latter part of the study period (see Section 5.2, Appendix F).

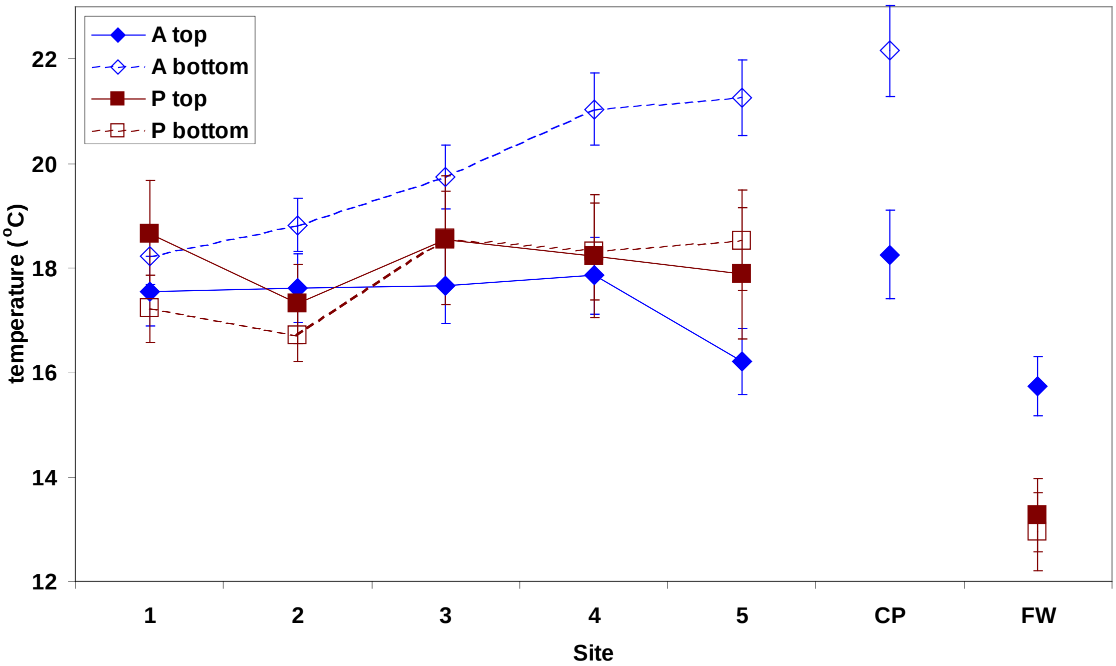


Figure 5.8. Mean temperatures (± std error) for sites along Anglesea (A) and Painkalac (P) estuaries. *n* was variable between sites (see Section 5.2, Appendix F). Sites 1 to 5 were in equivalent positions along the estuary from downstream to upstream. CP = Coogoorah Park, a site in the artificial channel network of the upper Anglesea estuary. FW = freshwater downstream sites, solid Painkalac symbol = Painkalac Creek, open Painkalac symbol = Distillery Creek.

Site 5 in Anglesea had a markedly lower mean surface-water temperature than the other sites. This was associated with a commonly occurring freshwater lens at the head of the estuary. The large differences between mean temperature at the head of Painkalac estuary and mean freshwater temperatures are most likely related to intermittent flows (mainly in the cooler months) and shallow, unshaded reaches between the freshwater sites and the head of the estuary.

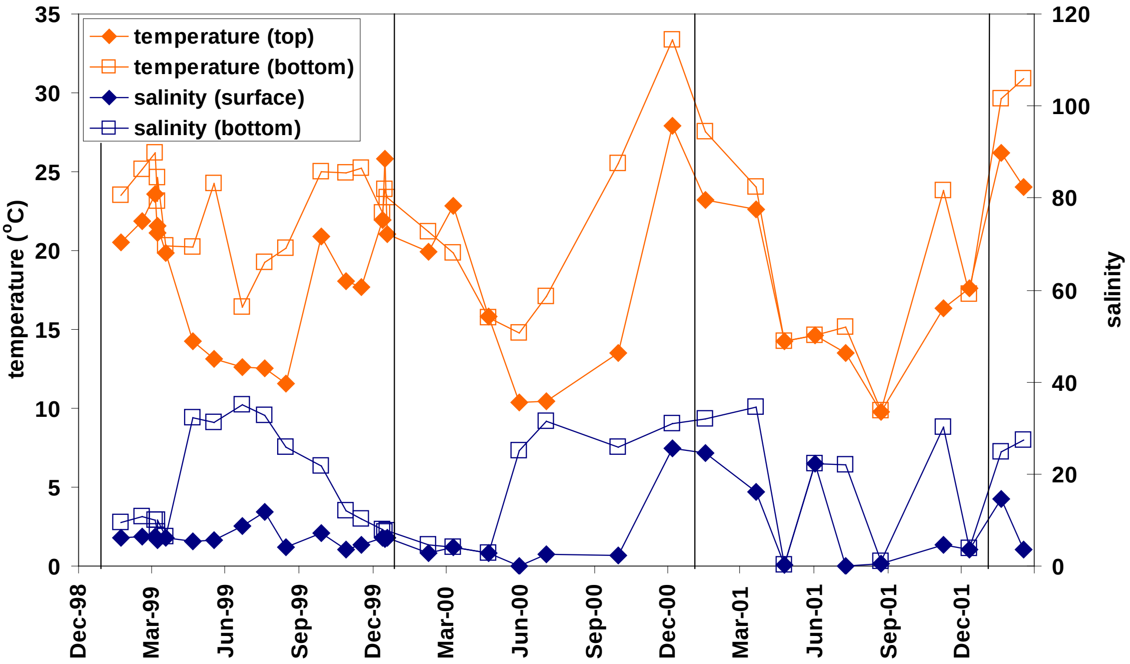


Figure 5.9. Temperature and salinity for top and bottom waters of Coogoorah Park site in Anglesea estuary illustrating warming of bottom waters with salinity stratification.

Maximum water temperatures in Anglesea were often observed in bottom waters at the upper sites (4, 5 and Coogoorah Park), typically changing at the halocline. This was the reverse of what is often found in larger stratified estuaries, in which the bottom waters are cooler than surface waters during summer. Warming of bottom waters in the upper part of Anglesea was most likely due to solar heating in shallow channels with minimal mixing with the upper, freshwater layer. The effect was also observed downstream, but less often and to a lesser degree. A clear link with the existence of salinity stratification at the Coogoorah Park site can be seen in Figure 5.9 in which a difference in temperature between surface and bottom waters was present at all times where there were also differences in salinity. This pattern was also reflected at Site 1 during the first half of the study period, but not in the latter part of the study when the estuary was largely tidal.

5.3.2.b. Temporal patterns: Sites 2

A seasonal pattern is obvious in Figure 5.10, showing temperatures at Site 2 in both Anglesea and Painkalac. The pattern of slightly greater mean temperatures in Anglesea seen in Figure 5.8 was reflected on multiple occasions for depth-averaged data. In particular, the mean temperature at Site 2 in Anglesea was between two and four degrees warmer than the equivalent in Painkalac from April to September 1999 inclusive and between one and three degrees warmer from May 2000 to March 2001, with the exception of December 2000 (Figure 5.10).

These two periods (in which Site 2 at Anglesea had a greater mean temperature) coincided with times when Anglesea had a relatively greater marine influence than Painkalac as indicated by estuarine state (perched or tidal vs closed: Figure 4.11), salinity measurements at the sites (Figure 5.5), and salinity stratification patterns (C, D or E vs mostly F: Figure 5.6). In December 2000, when Painkalac was warmer than Anglesea, Painkalac was closed and well mixed, but with high water levels extending across shallow mud flats where waters were easily heated. Later that summer, water levels in Painkalac had fallen into the channel, reducing the amount of solar heating of the water body.

While Anglesea was closed and stratified, maximum summer temperatures were in the bottom waters, consistent with heating of bottom waters as described above. In winter, minimum temperatures were in surface waters, a more typical estuarine pattern. In Painkalac, there was little temperature stratification in the first half of the study while the estuary was closed and well-mixed but in the latter part of the study, extremes of temperatures were seen in surface waters.

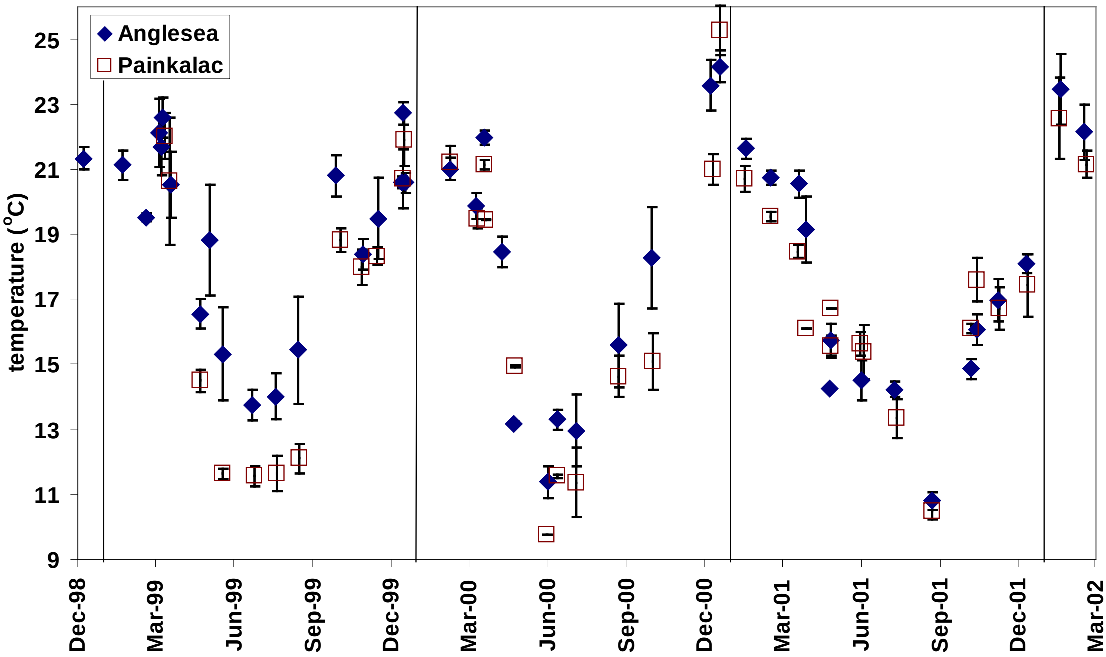


Figure 5.10. Mean (± s.e.) temperature for all depths at Site 2 in both Anglesea and Painkalac estuaries for all dates sampled during the study period.

5.3.3. Dissolved oxygen

Oxygen is a primary determinant of the habitability of estuarine waters for animals. The main net source of dissolved oxygen in estuaries is the atmosphere (ANZECC & ARMCANZ, 2000). While oxygen is also generated *in situ* by photosynthesis during daylight, much of this is consumed by plant and algal respiration during the night. It is also consumed by animal and microbial respiration, particularly when there is a large amount of decaying organic matter present, as well as by abiotic oxidative chemical reactions. All measurements reported in this section were taken in the daytime and so do not address diurnal variation.

In stratified estuaries with little salt-water exchange, depletion of dissolved oxygen is a relatively common phenomenon in bottom waters that are effectively isolated from the atmosphere (Rochford, 1951; Roy *et al.*, 2001). It has potential to affect animals directly and via trophic interactions (e.g. Carter, 1994; Breitburg *et al.*, 1997; Eby & Crowder, 2004) and to influence exchange of nutrients, metals and other pollutants between sediments, organisms and the water column (e.g. Carter, 1994; Kristiansen *et al.*, 2002;

Griscom & Fisher, 2004). Two main factors influence deoxygenation of bottom waters: time of isolation of bottom waters and oxygen demand within those waters (Rochford, 1974).

5.3.3.a. Dissolved oxygen ranges and frequencies

The greater range of individual measurements of dissolved oxygen in Painkalac compared to Anglesea (Table 5.6) is reflected in mean minima and maxima in Figure 5.11. Apart from the greater mean maximum for sites in Painkalac, surface waters of both estuaries had similar distributions of mean dissolved oxygen. There was a much larger difference between the distributions of mean saturation of bottom waters. Painkalac and Anglesea had similar mean maxima but bottom waters of Anglesea sites were more oxygenated, with a considerably greater mean minimum.

Estuary	Min.	Site	Depth	Date	Max.	Site	Depth	Date	*n*
Anglesea	20.4	4	2.4	16/11/1999	187.9	CP	0.7	17/1/2001	997
Painkalac	0.0	5	2.3	10/12/2000	197.0	2	1.5	14/12/1999	592

Table 5.6. Minimum and maximum percent saturation of dissolved oxygen recorded in Anglesea and Painkalac estuaries during the study period.

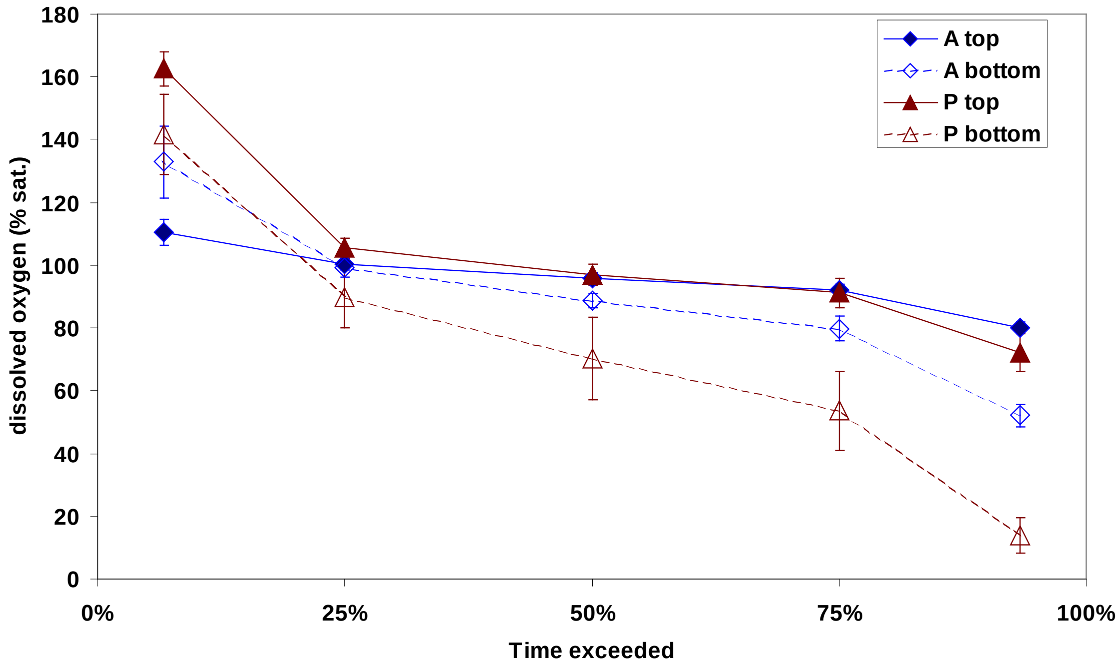


Figure 5.11. Mean (± s.e.) percent exceedances for saturation of dissolved oxygen of sites in Anglesea (A) and Painkalac (P) estuaries for the 15 times on which both estuaries were longitudinally sampled for dissolved oxygen.

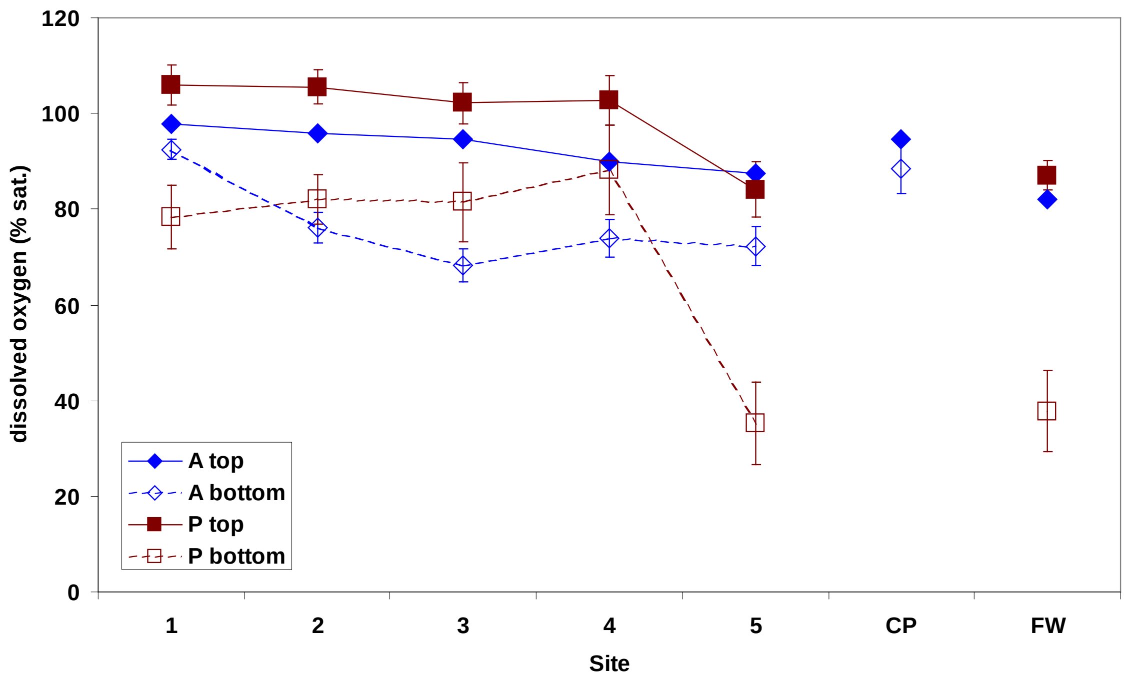


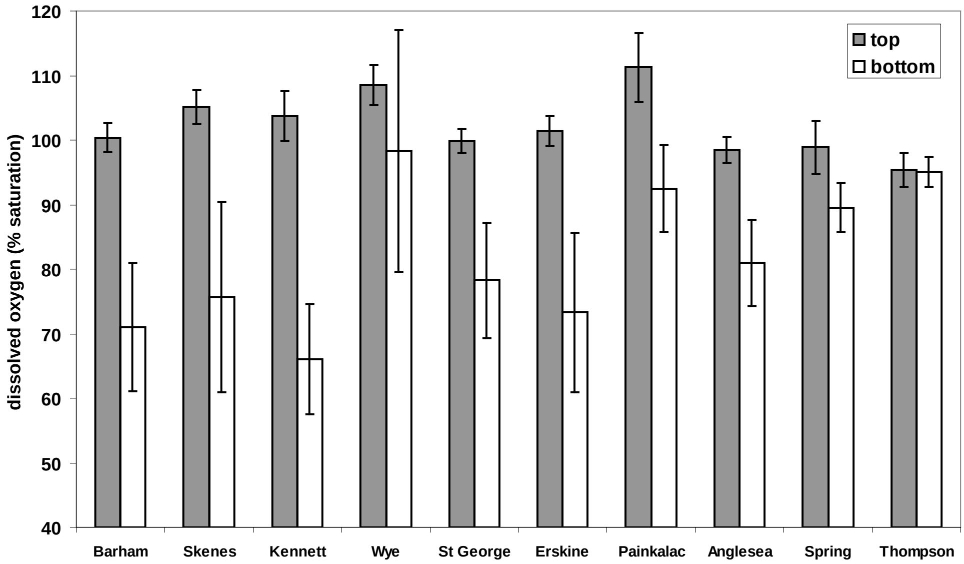
Figure 5.12. Mean percent saturation of dissolved oxygen (± std error) for sites along Anglesea (A) and Painkalac (P) estuaries. *n* was variable between sites (see Section 5.2, Appendix F). Sites 1 to 5 were in equivalent positions along the estuary from downstream to upstream. CP = Coogoorah Park, a site in the artificial channel network of the upper Anglesea estuary. FW = freshwater downstream sites, solid Painkalac symbol = Painkalac Creek, open Painkalac symbol = Distillery Creek.

In terms of mean saturation by site (Figure 5.12), bottom waters in both Anglesea and Painkalac were lower than surface waters. In both estuaries, mean saturation of surface waters was typically close to 100% and decreased slightly upstream. Exceptions to this were Site 5 in Painkalac which, at 84%, was substantially lower than other sites in Painkalac and Coogoorah Park in Anglesea, which did not follow the general trend and was equivalent to the mid-estuary site (Site 3) at 95%. Slight super-saturation in Painkalac was indicative of high net primary production in most of the estuary.

Longitudinal patterns of mean dissolved oxygen in bottom waters also differed between estuaries (Figure 5.12). In Anglesea, levels were highest (~90%) at Sites 1 and Coogoorah Park, the shallowest sites in the estuary, with similar levels (~70%) at the other sites. Low mean dissolved oxygen (35%) in bottom waters of Site 5 in Painkalac may reflect high net respiration and little mixing, consistent with the nature of the site (one of several deep holes in the upper estuary) and the regular occurrence of pockets of trapped saline water (illustrated in pattern G, Table 5.4).

Differences in mean percent saturation between surface and bottom waters were greatest at Site 5 in Painkalac and smallest at Sites 1 and Coogoorah Park in Anglesea. This may reflect the morphology of the estuaries at these sites, with the Painkalac site being in a deep and narrow section of the estuary and both Anglesea sites being shallow and more readily vertically mixed. The mean concentration of dissolved oxygen at the head of the Anglesea estuary (Site 5) was consistent with the mean concentration of inflowing waters (Figure 5.12). Site 5 in Painkalac was also consistent with dissolved oxygen in inflowing waters from Painkalac Creek but was substantially greater than the mean concentration in Distillery Creek.

Reduced, but variable, oxygen concentrations in bottom waters were common among regional estuaries (Figure 5.13). In general, mean saturation of surface waters was close to 100% among these estuaries, with Painkalac having the greatest mean concentration of 111%. Mean saturation of bottom waters ranged from 66% in Kennett to 98% in Wye, where variation was high, largely due to a bloom of benthic algae in December 1999. No oxygen differential between top and bottom waters was observed in the Thompson, although this estuary was added in the later part of the study and



so sampled for a shorter time (*n*=3).

Figure 5.13. Mean (± std error) percent saturation of dissolved oxygen of surface and bottom waters at times when regional estuaries were sampled. *n* = 10, with the following exceptions: Spring and Thompson *n*=3 (6/01, 11/01, 1/02), Skenes, Kennett and Wye *n* = 9 (not sampled 6/01). Anglesea and Painkalac are represented by Site 2 only.

5.3.3.b. Temporal patterns

As would be expected from the data represented in exceedance curves and site means above, dissolved oxygen concentrations in Anglesea were typically below those in Painkalac and below saturation while concentrations in Painkalac were supersaturated for much of the early part of the study period (Figure 5.14).

Comparison of Site 2 in each of the estuaries across time (as per previous sections) shows that Painkalac was very highly (>110%) saturated only in the warmer months of the first year of sampling (Figure 5.14). At the same time, some of the lowest mean concentrations recorded in Anglesea were near the start of the study in association with hypoxic bottom waters (Figure 5.15a).

With the exception of Painkalac in April 2000 (~day 490), low mean dissolved oxygen concentrations occurred only in bottom waters, usually in association with salinity stratification (Figure 5.14, Figure 5.15). Extremely high dissolved oxygen concentrations in Painkalac in December 1999, followed by a drop in oxygen concentration, were very likely to have been associated with an algal bloom.

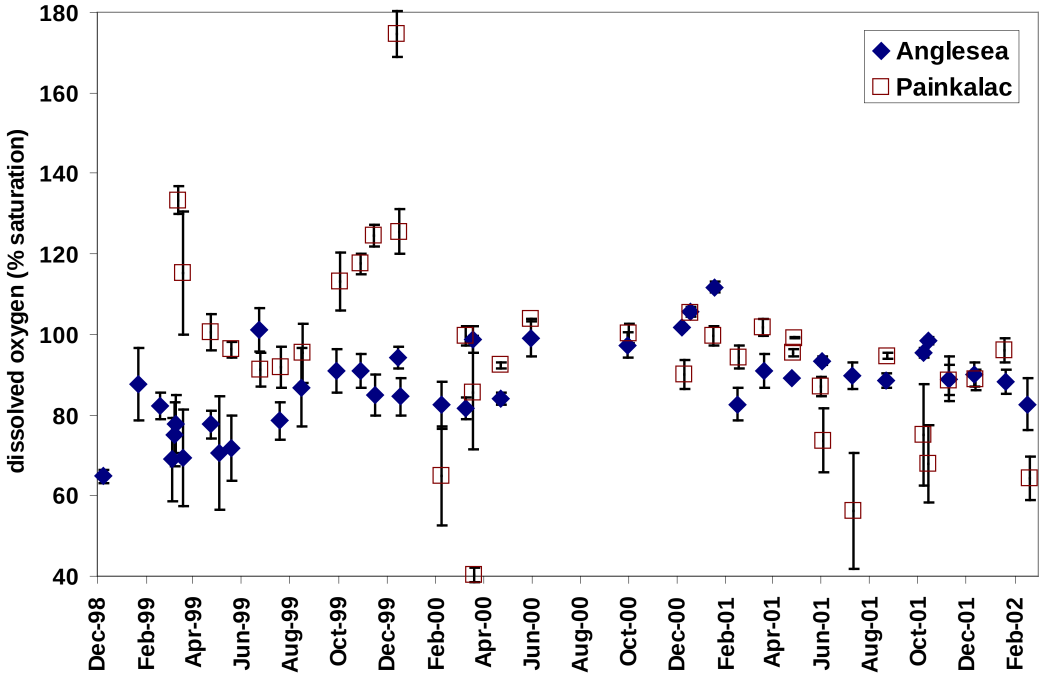


Figure 5.14. Mean (± s.e.) percent saturation of dissolved oxygen for all depths at Sites 2 in Anglesea and Painkalac estuaries for all dates sampled during the study period

a)

b)

Figure 5.15. Dissolved oxygen across depth at Site 2 in a) Anglesea and b) Painkalac though the study period (8/12/1998 (Day 0) to 19/2/2002 (Day 1169)). Shaded areas at the top of each figure reflect changing water levels at the sites, the shaded area at the left of b) represents time from the start of the study until Painkalac was sampled. Crosses represent sampling points across depth and time.

5.3.4. Nutrients

Four species of nutrients were measured. Total nitrogen (TN) and total phosphorus (TP) represented the potential pool of nutrients available in the system, while nitrate plus nitrite (NOx) and soluble reactive phosphate (SRP) are measures of nutrients that are immediately available to plants and algae.

Globally, estuaries are typically nutrient rich, with the most commonly found management-related issue being eutrophication: excessive primary production that is associated with anthropogenic nutrient enrichment (Kennish, 2002). External sources of nutrients to Anglesea and Painkalac estuaries are likely to be mainly of terrestrial and fluvial origin, as nutrients in the waters of western Bass Strait, as for most Australian waters, are low (e.g. 1980 concentrations: NOx<0.014mg/L, SRP<0.009mg/L, TP<0.012mg/L: Gibbs *et al.*, 1986).

5.3.4.a. Ranges and spatial distribution

A large proportion of samples from both Anglesea and Painkalac had concentrations of bio-available nutrients (NOx, SRP) below the detection limit (d.l.) of 0.010mg/L (Table 5.7). Total phosphorus concentrations in samples from Anglesea, but not Painkalac, were also often below the detection limit

(0.010mg/L, as for TN; Table 5.7). Because of this, minima could not be specified for these nutrient species. As a conservative measure (*i.e.* overestimating nutrient concentrations), samples with concentrations of one or more nutrient species below the detection limit were treated as if they had 0.010mg/L of those nutrient species in calculations of summary statistics and analyses (Helsel, 1990).

Locations and dates of maxima for each species in Anglesea were all in the mid- to upper estuary: nitrogen species during winter/spring in surface waters; phosphorus species in bottom waters during summer/autumn. These results suggest a freshwater (possibly storm water) source of high concentrations of nitrogen species and benthic release of phosphorous possibly associated with lowered dissolved oxygen concentrations.

Maxima for all nutrient species in Painkalac were recorded in December 1999. NOx was greatest in surface waters near the mouth of the estuary while all other species were highest in bottom waters at the head of the estuary. These values (except for NOx) were one to two orders of magnitude greater than other values measured in Painkalac waters. This was associated with symptoms of an algal bloom (see Section 5.4). A pulse of SRP at all five sites in Painkalac was also observed in March 1999, with concentrations decreasing upstream but with no bloom detected.

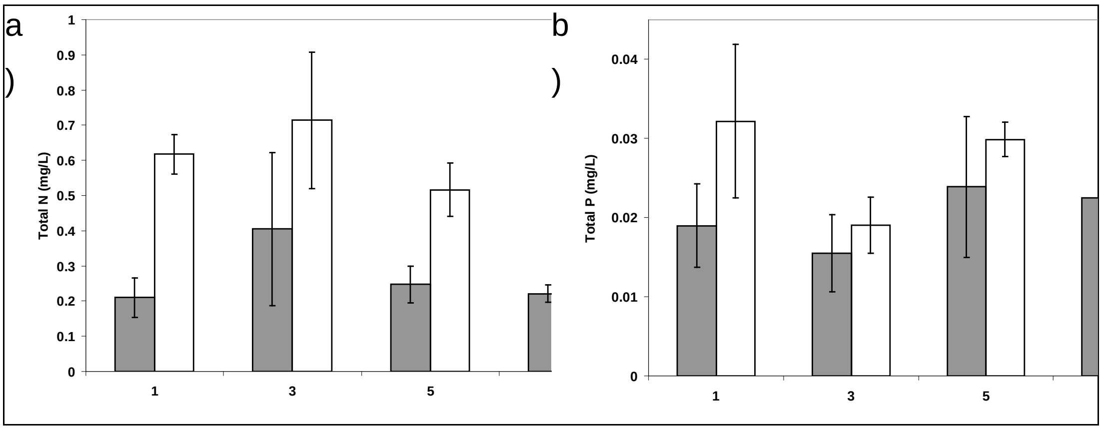
	Total N			**NOx**			**Total P**			**SRP**		
	min	<dl	max	min	<dl	max	min	<dl	max	min	<dl	max
Anglesea	0.035	0%	1.0	d.l.	34%	0.13	d.l.	33%	0.080	d.l.	92%	0.015
Painkalac	0.37	0%	8.4	d.l.	71%	0.090	0.012	0%	1.7	d.l.	69%	0.80

Table 5.7. Ranges (in mg/L) and percentages of observations below the detection limit (0.010mg/L) for species of nutrients in Anglesea and Painkalac estuaries. *n*=76 for Anglesea and *n*=35 for Painkalac.

The vertical distribution of nitrogen in Anglesea was variable for total nitrogen but NOx tended to be greater in surface waters and was greater in the upper estuary than the lower estuary. Relatively small top to bottom differences at

Sites 1 and Coogoorah Park may have reflected their shallower depth and greater mixing, as observed for salinity, temperature (at Site 1 only) and dissolved oxygen. No clear vertical or longitudinal patterns in the concentration of phosphorus were seen; results for SRP were below the detection limit, except for bottom waters in the mid- and upper-estuary. There were two measurements of SRP at the detection limit (0.010mg/L) at Site 5 and one measurement of SRP at 0.013mg/L at Site 3. In Painkalac there was only one occasion (3/10/1999) on which all sites were stratified and hence surface and bottom water samples were taken throughout the estuary.

For the subset of sampling occasions where longitudinal comparisons between surface waters of the two estuaries were possible, Painkalac tended to have greater concentrations of all nutrient species, except for NOx, which was greater in Anglesea (Figure 5.16). Mean TN was greater (though associated with more variability) in the middle of both estuaries, where mean TP was the lowest. Bioavailable components tended to decrease upstream in Painkalac while NOx increased upstream in Anglesea but SRP was below detection limits, consistent with the chemical composition of inflows (Section 3.4.3.d).



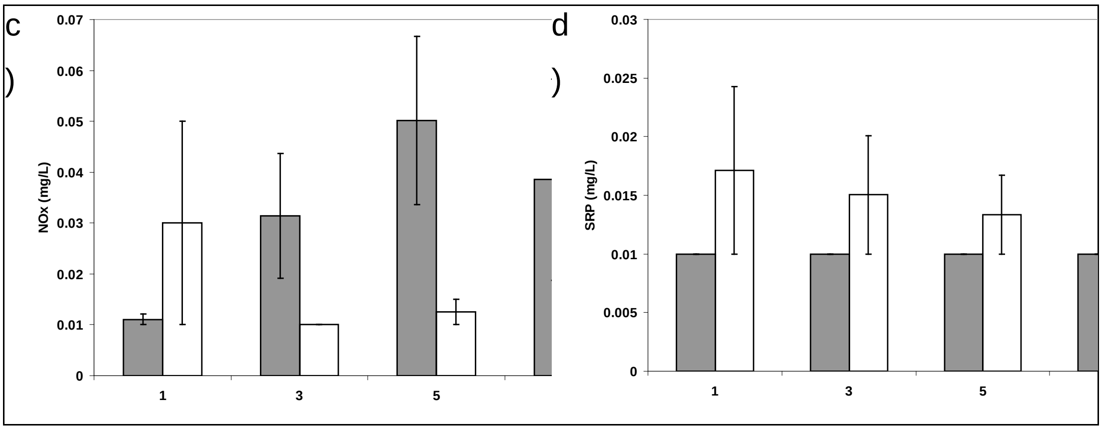


Figure 5.16. Mean (± std. error) nutrient concentrations in surface waters of sites at Anglesea and Painkalac at times when all sites were sampled (*n*=4). a) total nitrogen, b) total phosphorus, c) nitrates and nitrites (NOx), d) soluble reactive phosphorus (SRP). All samples from Anglesea were below the detection limit of the analysis for SRP.

5.3.4.b. Comparison with water quality objectives

Nutrient objectives for Victorian estuaries and inlets (modified from national objectives) are given as 75th percentiles, based on at least 12 monthly samples over a year (ANZECC & ARMCANZ, 2000; Environment Protection Authority, 2001). On this basis, not enough samples were taken to assess any individual site against the guidelines (max of 10 at any site). Pooling of samples from each estuary allows the comparison shown in Table 5.8.

75th percentile	Total N	NOx[a]	Total P	SRP[b]
Guideline	0.300	0.030	0.030	0.005
Anglesea	*0.359*	*0.035*	0.016	0.010*
Painkalac	*0.680*	0.010*	0.030	*0.014*

Table 5.8. Comparison of Victorian nutrient guidelines with samples taken from Anglesea and Painkalac estuaries. Values greater than the objectives are italicized. Concentrations are in mg/L. *n* =76 for Anglesea and 35 for Painkalac. SRP = soluble reactive phosphorus. [a] guideline includes ammonia, [b] guideline is half the detection limit used in analyses for this study. Values below detection limits are indicated by '*'.

Total nitrogen was greater than the objective in both estuaries and more than twice the objective in Painkalac. In contrast to this pattern, concentrations of NOx were only slightly greater than the objective in Anglesea but well below the objective in Painkalac. Both estuaries met the objective for total phosphorus and while a comparison for soluble reactive phosphorus was not possible in Anglesea, due to the detection limit of the analysis being greater than the objective, the 75th percentile of SRP concentrations in Painkalac was almost three times that of the objective.

5.3.5. pH

The pH of an estuary is influenced by acidity, alkalinity and volume of marine and freshwater inputs, mixing within the estuary and biological processes (mainly primary production) within the estuary. The pH of fresh water is more variable than that of marine waters (typically 6.5 to 8 vs ~8.2: ANZECC & ARMCANZ, 2000). pH is often not considered to be a variable of concern for estuarine organisms, as marine waters have a high buffering capacity and the small associated range of slightly alkaline pH is not large enough to have significant effects on bioavailability of toxicants (Knezovitch, 1994). This is reflected in, for example, pH being omitted in textbook discussions that include salinity, dissolved oxygen and nutrients as controlling factors of the composition and distribution of estuarine biota (e.g. Hodgkin, 1994). Fresh water mixing that is associated with large depressions in pH is, however, likely to increase bioavailability of heavy metals (Knezovitch, 1994).

In parts of Australia, acidic waters generated by a range of processes flow into estuaries (ANZECC & ARMCANZ, 2000). The impacts of acidic fresh water flows from acid sulphate soils on Australian estuaries have only been studied in detail in relatively recent years (e.g. Sammut *et al.*, 1993; Sammut *et al.*, 1995; Roach, 1997; Johnston *et al.*, 2004).

5.3.5.a. Ranges and spatial distribution

A larger range of pH was measured in Painkalac (3.8 to 9.6) than in Anglesea (3.6 to 8.4) although the occurrence of low pH in Painkalac was less common and more spatially restricted than in Anglesea. Overall distributions of pH in Anglesea and Painkalac were similar, with surface waters more acidic than bottom waters and all waters in both estuaries having a pH between six and eight for most of the time (Figure 5.17). The main differences in pH distributions between the estuaries were larger minima and maxima in Painkalac and greater differences between surface and bottom waters in Anglesea, except in extreme conditions. These patterns reflect high pH in Painkalac at times of little stratification, occasional flushing of Anglesea estuary with low pH water and a pattern of surface waters being less acidic and bottom waters being more acidic in Painkalac than in Anglesea (see Section 5.3.5.c).

When comparing mean pH for surface and bottom waters for sites along each estuary, a gradient of increasing pH was evident in all waters except for Anglesea bottom waters, probably due to increasing marine influence (Figure 5.18). Lower mean pH at the heads of the estuaries was consistent with the mean pH of inflowing waters from Anglesea River and Painkalac Creek but not with those of Distillery Creek, probably reflecting the smaller frequency and volume of flow from this tributary compared to the mainstem of Painkalac Creek.

On average, surface waters were more acidic than bottom waters at all sites except Site 5 at the head of Painkalac. This difference was smallest in the lower part of both estuaries, and at the Coogoorah Park site in Anglesea.

Means of pH of surface waters in Anglesea were consistently lower than the equivalent sites in Painkalac (Figure 5.18).

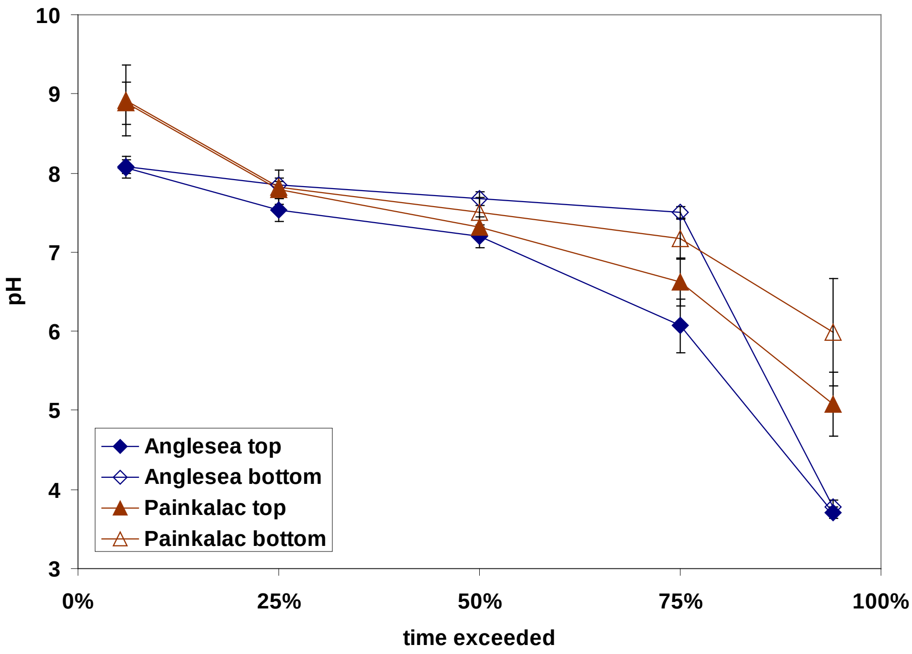


Figure 5.17. Mean (± s.e.) percent exceedances of pH for Sites 1-5 combined in Anglesea and Painkalac estuaries for the 16 times on which both estuaries were longitudinally sampled.

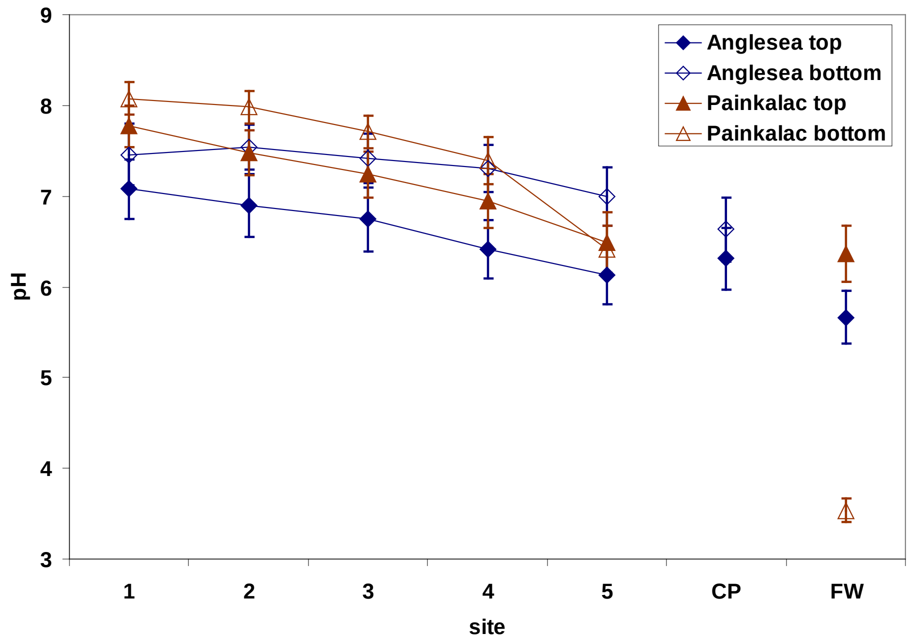


Figure 5.18. Mean pH (± std error) for sites along Anglesea (A) and Painkalac (P) estuaries at times of longitudinal sampling. *n* was 14 to 16, except for Painkalac freshwater sites, which were only sampled during times of flow (see Section 5.2, Appendix F, Appendix B). Sites 1 to 5 were in equivalent positions along the estuary from downstream to upstream. CP = Coogoorah Park, a site in the artificial channel network of the upper Anglesea estuary. FW = freshwater downstream sites, solid Painkalac symbol = Painkalac Creek, open Painkalac symbol = Distillery Creek.

5.3.5.b. Regional comparison with water quality objectives

There are no specified trigger levels for pH in Victorian estuarine water quality objectives, based on the premise that altered pH is likely to be only a local issue for discharges to marine waters with a high buffering capacity (Environment Protection Authority, 2001). This line of reasoning is not applicable to intermittent estuaries such as those of the Otway coast, in which there is no continuous marine influence. National guidelines for estuaries in south-east Australia recommend a minimum of 7.0 and maximum of 8.5, based on a comparison with the "median pH range measured in the test system" (ANZECC & ARMCANZ, 2000).

As the trigger levels in the guidelines were derived from 80th and 20th percentiles of reference data, box plots of pH for each estuary are presented in Figure 5.19. In all cases, the interquartile range was within the upper and lower trigger levels. Comparisons for Spring and Thompson Creek should be treated with caution due to the low number of samples in those estuaries, but for the times sampled both estuaries were well within the specified range. Although *n* differs by an order of magnitude between the Anglesea, Painkalac and the rest of the estuaries, and so a wider range of values would be expected, on at least one occasion all estuaries exceeded at least one of the trigger levels and six of the eight estuaries exceeded both, though not by as much as Anglesea and Painkalac.

On average, Painkalac had the most alkaline bottom waters and Anglesea had the most acidic surface waters of all the estuaries. There was a general trend for bottom waters to become more alkaline from west to east, from 7.3 in the Barham to 8.1 in Painkalac. Compared to the other estuaries, Spring and Thompson were unusual, in that mean pH of top and bottom waters were very similar. This pattern was consistent with top-bottom similarities for other water quality parameters in that both these estuaries were usually well-mixed and strongly marine influenced over the limited number of times they were sampled (2001/2, *n*=3).

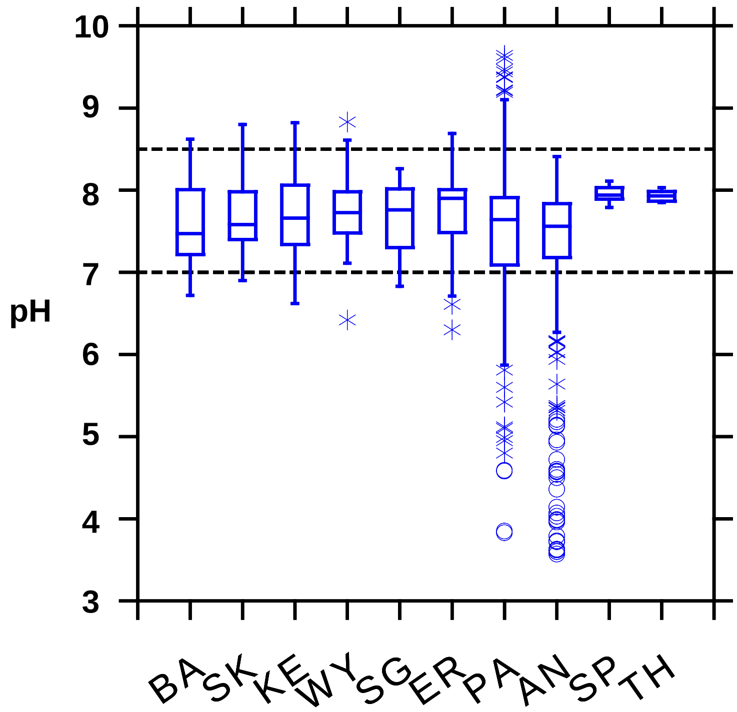


Figure 5.19. pH by estuary through the sampling period compared with ANZECC & ARMCANZ trigger levels for south-east Australia (shown as dashed lines). Estuaries are presented from west to east with codes are as follows and *n* in brackets: BA – Barham (59), SK – Skenes (22), KE – Kennett (41), WY – Wye (28), SG – St George (40), ER – Erskine (47), PA – Painkalac (197), AN – Anglesea (408), SP – Spring (7) and TH – Thompson (8).

5.3.5.c. Temporal changes

Figure 5.20 highlights the wide range of pH measured in Anglesea and Painkalac through the study period. A qualitative difference in the ranges of pH between the first and second halves of the study period is the most obvious characteristic of the figure, with intermittent periods of low pH from late September 2000 onwards. Inflowing waters were more acidic than estuarine waters, a notable exception being a pH of 9.5 in Painkalac Creek in late October 1999 that preceded a high pH event in the estuary. During the first half of the study, pH in Painkalac was almost always alkaline, while Anglesea was occasionally acidic. In the second half of the period, acidic events in Anglesea were reflected at a lesser magnitude in Painkalac, most likely reflecting flows from the previously dry Distillery Creek (Section 3.3.4).

As for other water quality variables, pH was sampled more frequently at Site 2 in Painkalac than at other sites in that estuary. A comparison of this site and its equivalent in Anglesea provides the best temporal resolution when comparing full depth profiles between estuaries.

Most of the time, the entire water column at Site 2 in both estuaries had a pH between 7 and 8. There was little overall difference in pH with depth at Site 2 in either estuary although there was a tendency for bottom waters to be more alkaline than surface waters (Figure 5.21).

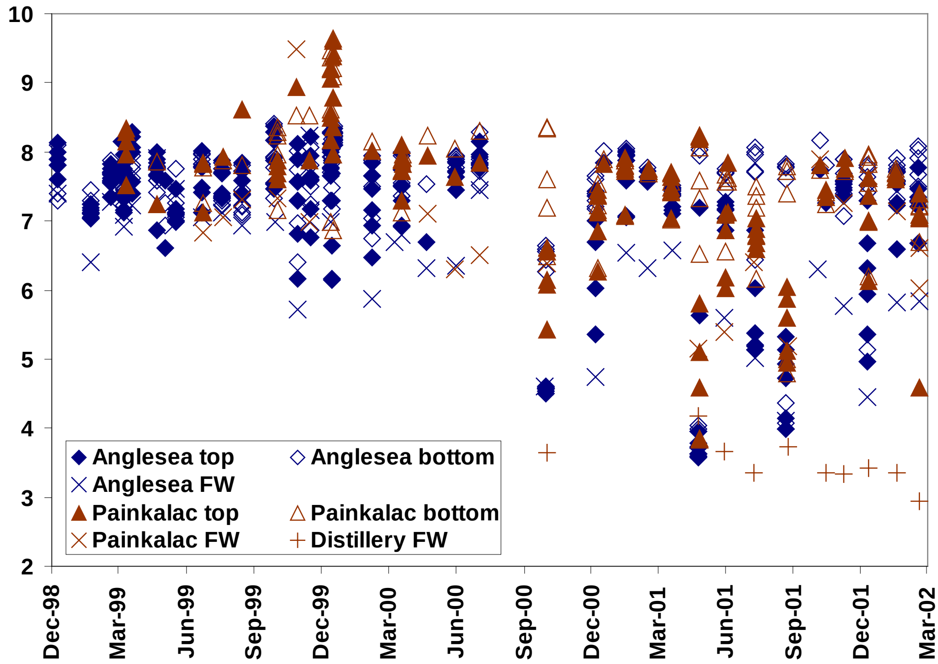


Figure 5.20. pH in surface and bottom waters from all sites in Anglesea and Painkalac estuaries and inflowing fresh water (FW).

Acid events where pH was below 7 at Site 2 in Anglesea are clearly evident at days 662 (30/9/2000), 869-870 (25-26/4/2001) and 945/988 (10/7 and 22/8/2001). Although these events were reflected in upstream sites in Painkalac (Figure 5.20), the only depression of pH seen at Site 2 in that estuary was in July-August 2001 (days 947 and 989). These occasions all coincided with periods of high flow (Section 3.4.2).

a)

b)

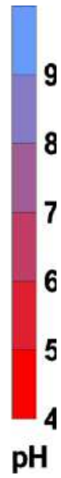


Figure 5.21. Vertical pH structure of Site 2 at a) Anglesea and b) Painkalac though the study period (8/12/1998 (Day 0) to 19/2/2002 (Day 1169)). Shaded areas at the top of each figure reflect changing water levels at the sites, the shaded area at the left of b) represents time from the start of the study until Painkalac was sampled. Crosses represent sampling points across depth and time.

In Anglesea, there were two short periods in September and December 1999 (days 294 and 370) where pH was above 8. In contrast, there was a period of eight months between August 1999 and April 2000 in which the pH at Site 2 in Painkalac was above 8. Within this period at Painkalac, on 14 and 16 December 1999 (days 371 and 373) there was a peak in pH, during which pH was above 9 at all depths. On 27 April 2001 (day 871), the pH at all depths was also above 8, in contrast to the previous day on which the pH was below 8. This instance reflected the incursion of a high tide in the newly tidal Painkalac estuary.

Since June 1972 there have been 22 times when the pH at Alcoa's mid-estuary surface water site was below five, with an average frequency of 0.7 times a year and mean duration of 1.8 months (Alcoa of Australia, unpublished data). On three of these occasions, pH was measured below four, twice in the mid-1970s and once in 1983, for a single month on each occasion (the April 2001 event reported above was not detected in Alcoa's program as sampling that month took place four days before the flood that lowered estuarine pH) (Figure 5.22).

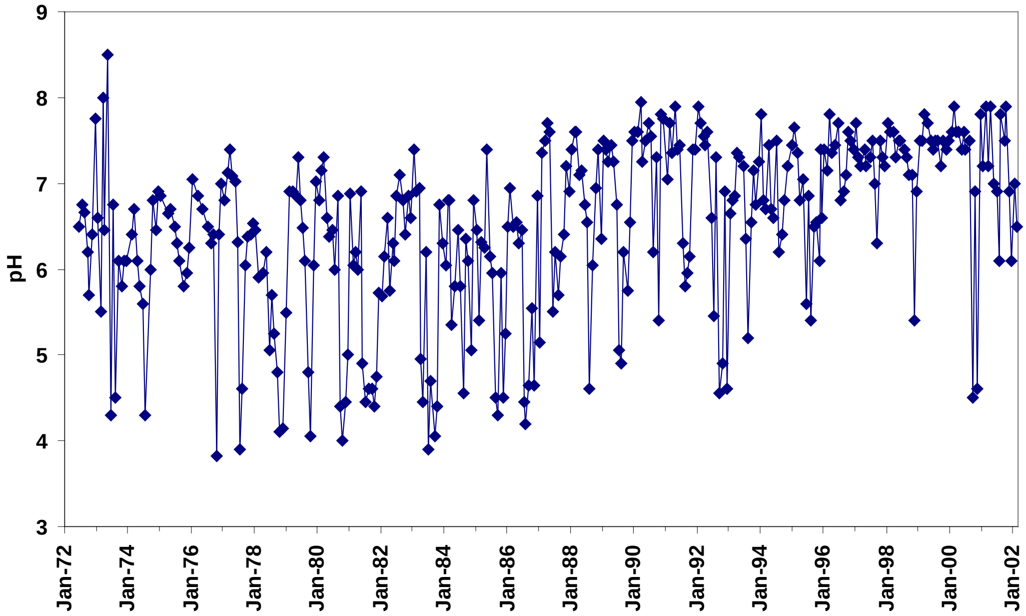


Figure 5.22 Monthly pH of surface waters at Alcoa's mid-estuary site in Anglesea, July 1972 to March 2002.

Between December 1995 and August 2000 there was a period with only two short depressions in pH. This period corresponds to four consecutive years of below average rainfall discussed in Section 3.1, including the second-driest three year period since the start of records in 1927. The second half of the study period was similar to the period from 1987 to 1995, before which maximum pH values were around 0.5 units more acidic.

5.3.6. Secchi depth, turbidity and TSS

Light is an essential requirement for photosynthesis and the light climate in the Anglesea estuary is potentially critical in determining depth limits of the aquatic flora of the estuary, including seagrasses, benthic macroalgae and phytoplankton. Although the incidence and attenuation of photosynthetically active radiation (PAR) was not measured directly as part of the study, three surrogate measures of water clarity were recorded: Secchi depth, concentrations of suspended solids and turbidity.

Secchi depth is the depth at which a disc with alternating black and white quadrants cannot be seen from the surface. This variable was measured throughout the study period. Suspended solids were measured as the concentration in mg/L of filterable solids in water samples (which were also analysed for nutrients). Turbidity is a measure of scattering of light passing through the water. It is measured in nephelometric turbidity units (NTU). Measurement of turbidity commenced in September 2000, when a multi-probe with this capability became available. Unlike Secchi depth, both suspended solids and turbidity were measured at multiple depths within a site.

A range of biotic and abiotic factors can affect these measures of water clarity, including seasonal and other temporal changes in the abundance and composition of phytoplankton and zooplankton. Sediments washed into the estuary from waterways and stormwater drains or re-suspended from the estuary bed will reduce clarity (ANZECC & ARMCANZ, 2000). The relative influence of marine and fresh waters will also affect clarity, as will other chemical changes (Kirk, 1986), including flocculation of dissolved compounds associated with changes in pH.

5.3.6.a. Secchi depth

Secchi depth was the only measure of water clarity that was measured throughout the study period. A total of 207 readings were taken at Anglesea, on 36 of the 46 dates the estuary was sampled (Figure 5.23a). At Painkalac, 97 readings were taken, on 29 of the 40 dates the estuary was sampled (Figure 5.23b).

Measurement of Secchi depth is dependent on the total water depth being greater than the Secchi depth at the time of sampling. When the Secchi disc can be seen on the bottom of a sampling location, Secchi depth may only be reported as ">*x*", where *x* was the water depth. A larger proportion of readings at Anglesea (27%) were greater than the available depth than those at Painkalac (5%). This was largely due to the greater depth of Painkalac including for the site most frequently sampled (Site 2).

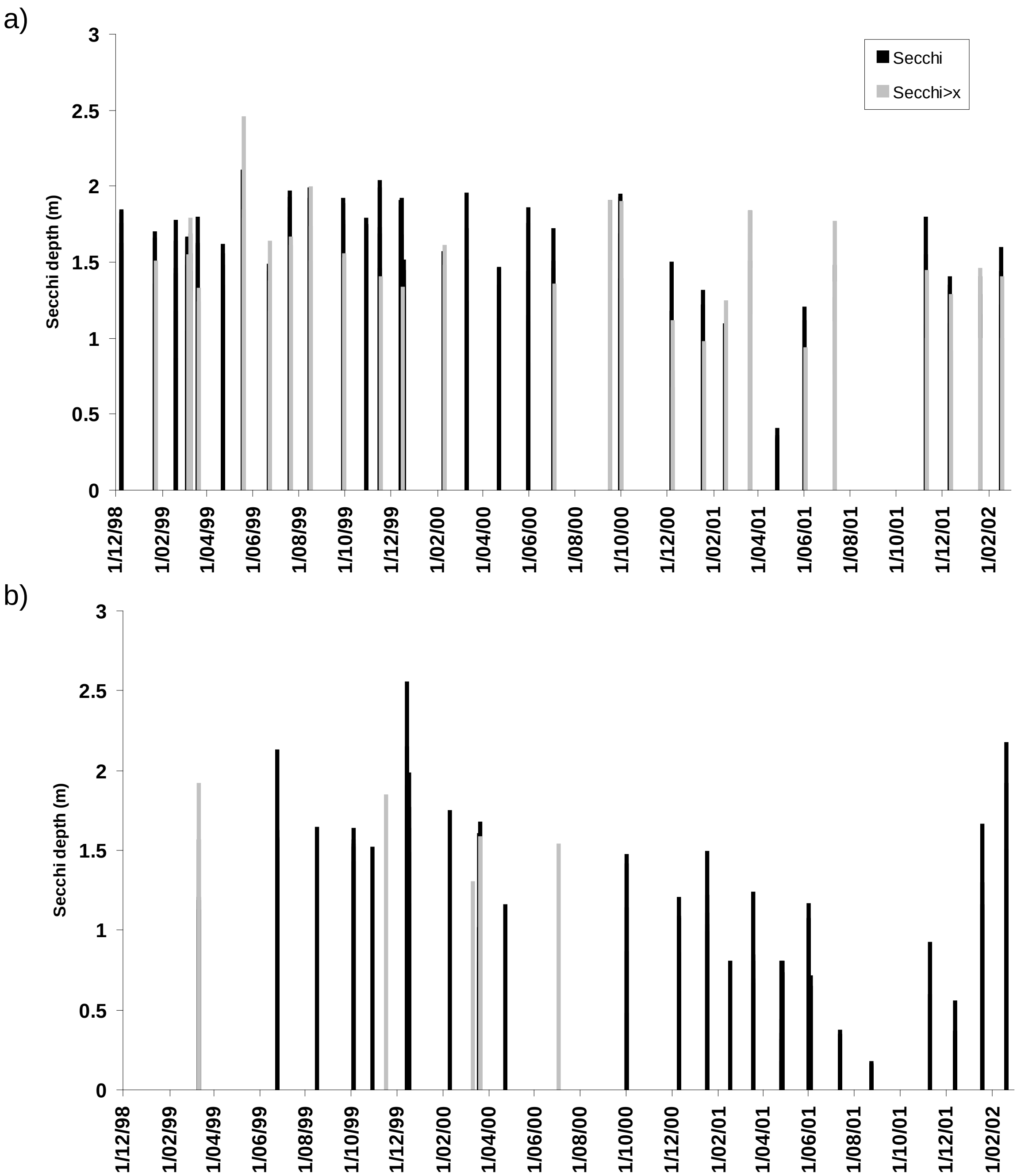


Figure 5.23. Maximum Secchi depths from all sites sampled in a) Anglesea and b) Painkalac estuaries. Grey lines indicate depths for times when the Secchi disc was still visible on the bottom of the estuary.

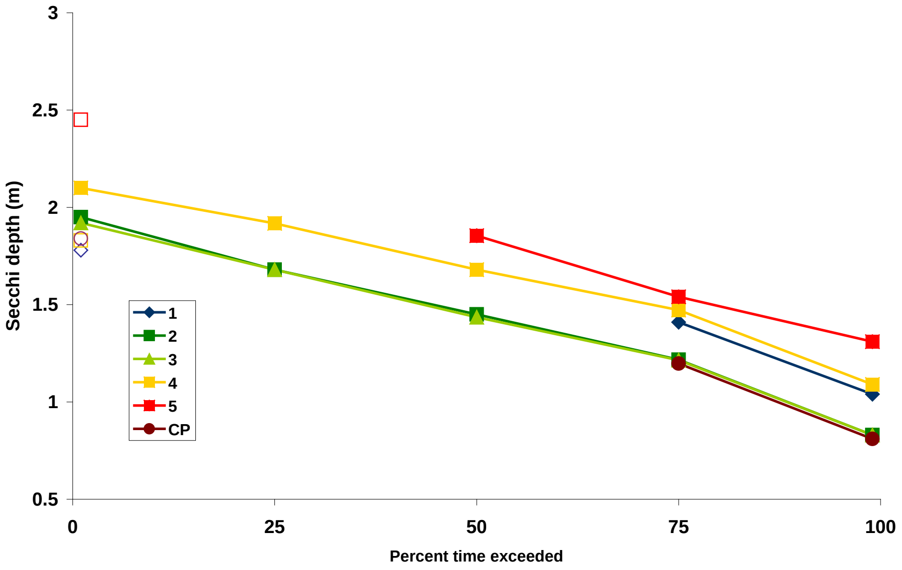
Secchi depths in Anglesea and Painkalac ranged from 0.32m to >2.45m and 0.14m to 2.55m respectively. Minima for each estuary were recorded in flood conditions, during April 2001 at Anglesea (Site 1) and during August 2001 at Painkalac (Sites 2 and 4). Maxima were recorded at times of low/zero fresh water flow, during May 1999 at Anglesea Site 5 and during December 1999 at Painkalac Site 2. It should be noted that these maxima may not reflect times with the clearest waters as greater Secchi depths could potentially have occurred at times and sites limited by water depth. Because of this, medians and percentiles were used to describe the results, as opposed to means, with substituted values of *x* where Secchi depth was ">*x*".
Overall, Secchi depth was greater at Anglesea than at Painkalac with medians of 1.47m and 1.08m respectively. This was largely related to a proportionally greater decrease in water clarity of Painkalac compared to Anglesea in the second half of the study period (Figure 5.23, Figure 5.25).

During the study, sites within Anglesea estuary had relatively similarly shaped, but offset distributions of visibility as measured by Secchi depth (illustrated by parallel lines in Figure 5.24a). In contrast, there was no consistent longitudinal separation of sites in Painkalac estuary based on Secchi depth (Figure 5.24b). Minima were consistent between all sites while Site 5 had the largest maximum and therefore range. In common with Anglesea, Site 1 had the lowest range of depths, possibly because of the increased potential for wind-driven currents to reduce clarity in the relatively shallow and open waters of the lower estuaries and, in the case of Painkalac, a reduced influence of creek-borne sediments near the mouth.

As a result of the sampling design (Section 5.2), Secchi depth was measured more frequently at Site 2 than other sites in Painkalac. Secchi depths from these sites are compared in Figure 5.25. A similar pattern to that shown in Figure 5.23a & b can be seen in that, for the first part of the study, Secchi depths were broadly similar between the two estuaries. A decrease in Secchi depth was evident in 2001, with the greater decrease in Painkalac. Around this time, an increase of sediment-associated colour was observed in

Painkalac Creek above the estuary and a series of high turbidities were recorded in waters flowing into the estuary.

a)



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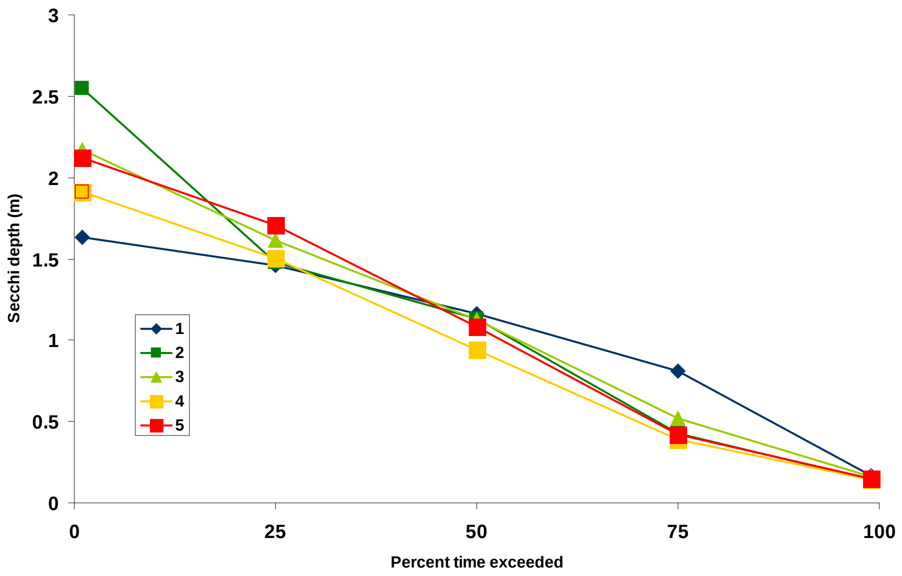

Figure 5.24. Percent exceedance of Secchi depths at sites in a) Anglesea and b) Painkalac estuaries for times at which all sites were sampled (*n*=32 for Anglesea, *n*=14 for Painkalac). Secchi depths greater than water depth were assumed to be larger than measured depths from other times. Except for maxima (represented by open shapes), these cases are not shown.

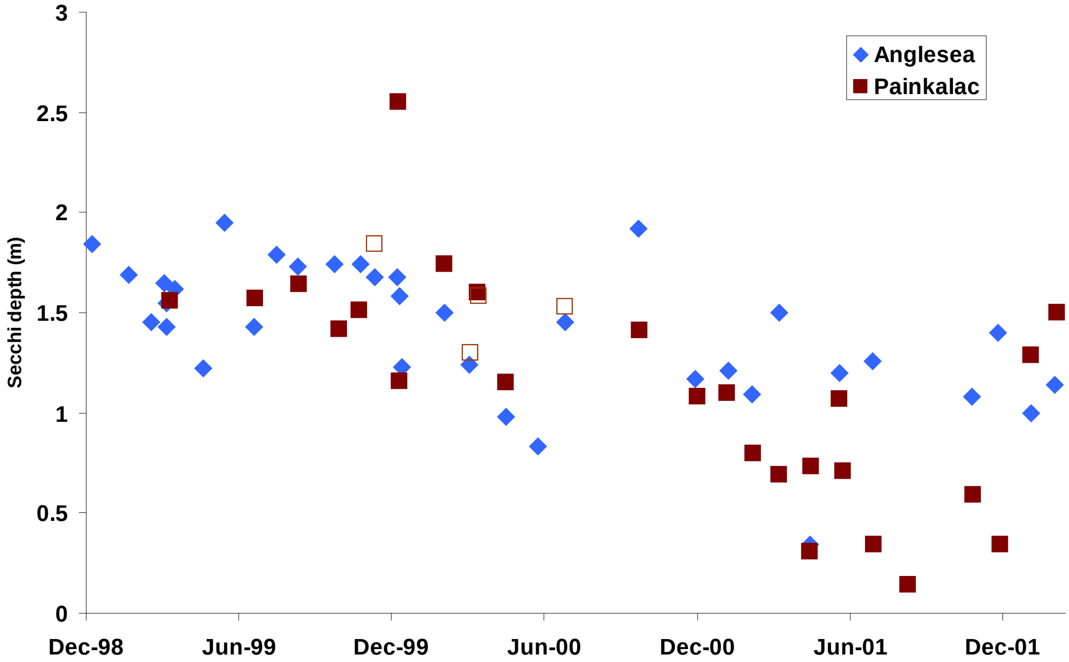

Figure 5.25. Secchi depths from Site 2 in Anglesea (*n*= 35) and Painkalac (*n*=29). Where Secchi depth was greater than water depth, water depth is shown as an open symbol.

5.3.6.b. Secchi depth vs other measures of clarity

As Secchi depths could not be transformed to approximate a normal distribution, the nonparametric Spearman's rank correlation coefficient was used to assess the strength of relationships between Secchi depth and other measures of clarity.

Secchi depths showed moderate to weak negative and approximately linear relationships with suspended solids from surface and bottom waters of each estuary (Table 5.9). Of the four comparisons, bottom waters had the largest inverse correlation with Secchi depth but only the correlation for Anglesea was significant.

	Anglesea TSS		Painkalac TSS	
	Surfac e	botto m	surfac e	botto m
***r**s*	-0.122	-0.545	-0.409	-0.568
p	0.599	0.011	0.166	0.043
n	21	21	13	13

Table 5.9. Correlations between Secchi depth and TSS. r_s represents Spearman's rank correlation coefficient, critical p values were adjusted to 0.017 using Bonferroni corrections. Where there was no stratification and only one TSS sample was taken, this value was used for both surface and bottom datasets (1/21 for Anglesea, 5/13 for Painkalac). Only times with actual (not ">*x*") Secchi depths were used for comparisons.

Turbidity of some components of the water column showed better correlations with Secchi depth (Table 5.10). At Anglesea, only surface waters showed a significant correlation, where in Painkalac, surface, bottom and mean turbidities were significantly correlated with ln Secchi depth. The higher correlations between surface turbidity and Secchi depth can be explained because Secchi readings by their nature are an integrated measure of water clarity downwards from the surface. As such, some caution must be used in their interpretation when measured in systems that are highly stratified in a similar vertical range to the Secchi depth.

	Anglesea turbidity			Painkalac turbidity		
	Surface	botto m	mean	Surface	botto m	mea n
***r**s*	-0.670	-0.016	-0.368	-0.963	-0.419	-0.854
p	<0.001	0.934	0.041	<0.001	0.006	<0.00 1
n	31	31	31	41	41	41

Table 5.10. Correlations between ln Secchi depth and turbidity. r_s represents Spearman's rank correlation coefficient, critical p values were adjusted to 0.017 using Bonferroni corrections. Only times with actual (not ">*x*") Secchi depths were used for comparisons.

5.3.6.c. Total suspended solids

With the exception of a concentration of 193mg/L recorded in a one-off sampling of a plume from a stormwater drain above Site 2 at Anglesea, the highest concentrations and variability of suspended solids recorded were in the surface waters of Anglesea (Figure 5.26). Concentrations in Painkalac were similar to, or lower than, those of Anglesea bottom waters. In contrast to Anglesea, concentrations were often greater in bottom waters of Painkalac than top waters, particularly later in the year when surface waters had very low concentrations of suspended solids

Concentrations of suspended solids in Anglesea estuary ranged from 3.5mg/L to 30mg/L. Peaks at sites in the surface waters of the upper estuary appeared to be related to increases in upstream concentrations. Concentrations in Painkalac tended to be lower than in Anglesea, ranging from 1.5mg/L to 21.5mg/L. No close link between upper sites and TSS in freshwater inflow was apparent.

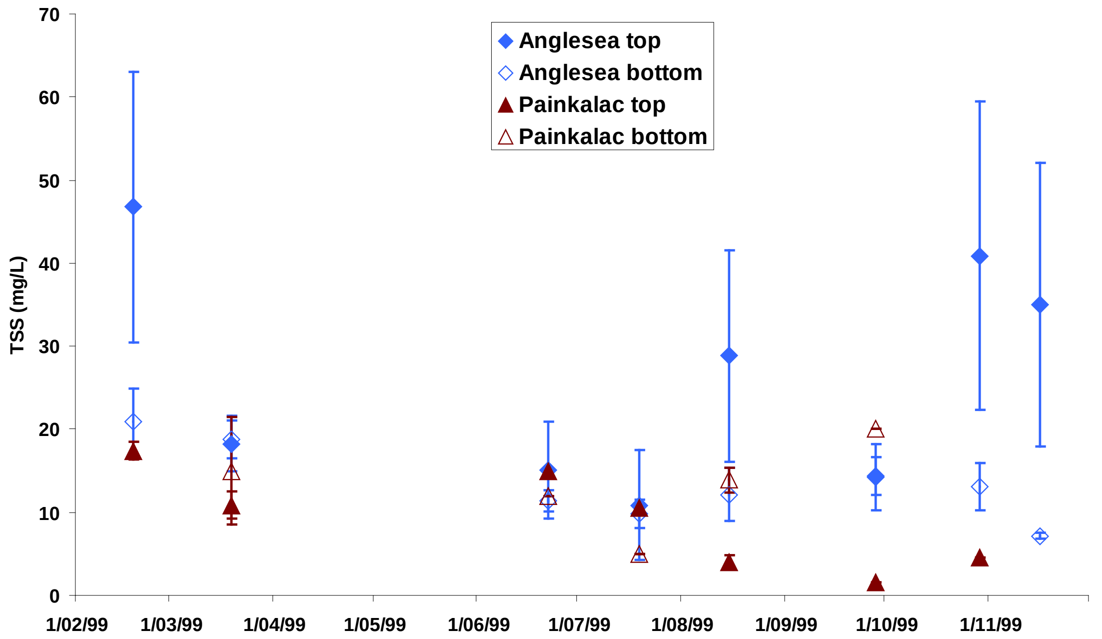

Figure 5.26. Mean total suspended solids (± standard error) for top and bottom waters of Anglesea and Painkalac estuaries, 1999. *n*=1 for Painkalac July, August and November sampling times. No error bars are included for these dates.

Using data from 12/2/1978 to 13/2/2002, a moderately strong (Pearson's r=0.56) and highly significant ($p=1.0\times10^{-15}$) correlation was seen between concentrations of suspended solids in waters entering the head of the estuary at Coal Mine road and those in estuarine surface waters at Alcoa's downstream site. This suggests that in the longer-term, fluvial sources have been a substantial contributor to suspended solids in the estuary.

5.3.6.d. Turbidity

Study period

Overall, turbidities in Painkalac were equivalent to, or higher than those in Anglesea, with the exception of December 2000 and January 2001 (Figure 5.27). Mean turbidity (from trips where all sites were sampled in both estuaries) was higher in Painkalac (23.0 NTU) than in Anglesea (9.8 NTU). Maxima in Anglesea and Painkalac were 47 NTU (7/12/2000, Site 2, bottom) and 122.4 NTU (23/8/2001, Site 4, bottom) respectively. These, and other high turbidities were associated with flood conditions. Respective minima for the estuaries were 0.0 (16/2/2002, Sites 4 and 5, top) and 0.7 (19/2/2002, Site 3, top). Despite some large values in Painkalac, turbidities in both estuaries were generally lower in spring/summer 2001/02 than in the equivalent period in 2000/01 (Figure 5.27).

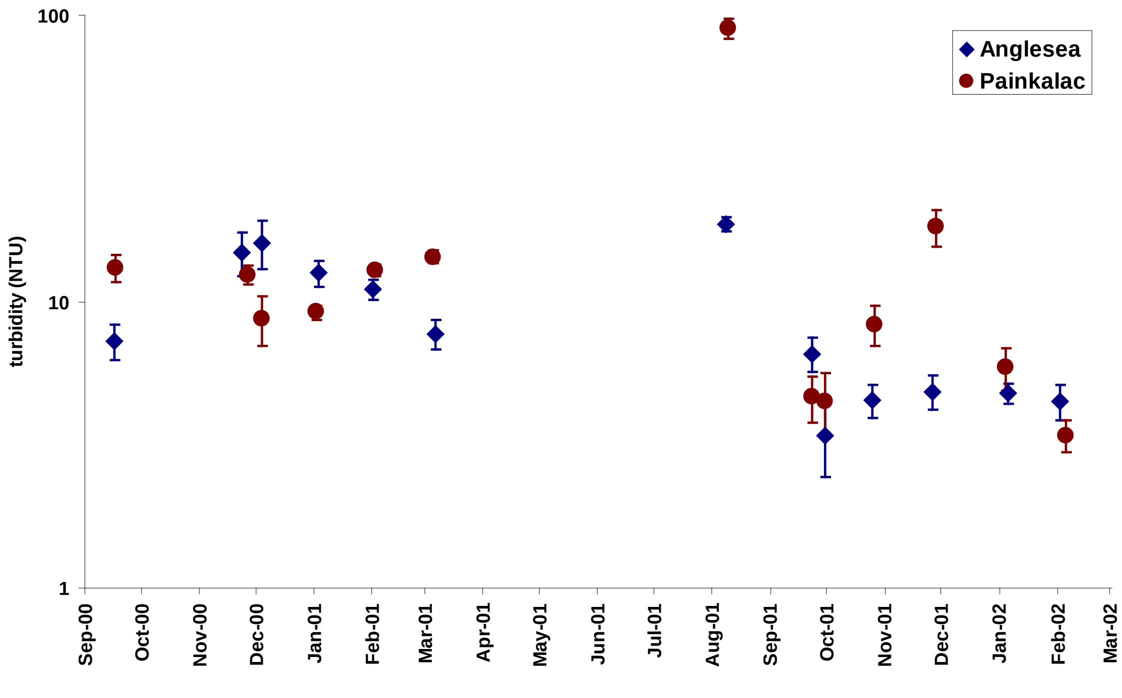

Figure 5.27. Mean turbidities (± std error) for all sites and depths in each estuary by sampling date. Note the log scale of the y-axis.

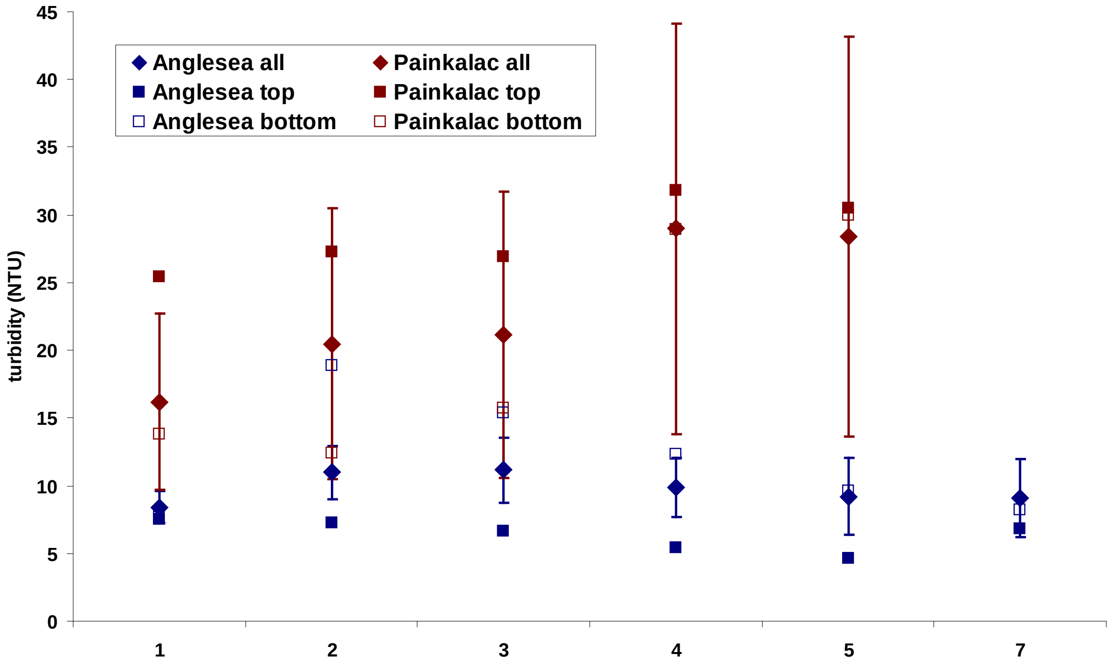

Figure 5.28. Mean depth-integrated turbidities (± std error) and mean top and bottom turbidities by site for Anglesea and Painkalac estuaries for times at which all sites were sampled in both estuaries (*n*=7, 30/9/2000-19/2/2002, see Appendix E for dates). "all" refers to averages for all times and depths at each site, "top" and "bottom" represent the shallowest and deepest sampling points for each site and time.

A longitudinal comparison of turbidity in both estuaries is shown in Figure 5.28. Over the sampled period, mean turbidities were consistent along Anglesea estuary but increased upstream in Painkalac estuary. Turbidity was consistently more variable at Painkalac sites than at Anglesea. Mean turbidities were higher in bottom waters than top waters in Anglesea while the opposite was the case in Painkalac.

The estuaries differed significantly in that, during the time of sampling, the turbidities of inflowing waters from Anglesea River and Distillery Creek were lower than those in the estuaries while the turbidity of Painkalac Creek was usually greater than that of the estuary. This suggests that Painkalac Creek was a substantial contributor of sediments to its estuary during the latter period of the study, a pattern that was consistent with the results for suspended solids from Painkalac in 1999, but not with the results from Anglesea in that year (Section 5.3.6.c). Compared to national guidelines, turbidity in both estuaries in the latter part of the study period was potentially an issue of concern, a conclusion supported by comparison with regional estuaries (Figure 5.29).

There were five times when regional estuaries were also sampled for turbidity, allowing a regional comparison of Anglesea and Painkalac estuaries for this parameter (Figure 5.29). Anglesea and Painkalac estuaries had similar or greater mean turbidities than other estuaries sampled, except in October 2000, when turbidities in these estuaries were the lowest of the eight sampled. This may be related to the fact that atypically, flows in the eastern end of the Otways were greater than those in the west at that time (see Figure 3.11). Painkalac had the maximum value, in August 2001, while the minimum value was measured in Kennett River in November 2001. Minima in Anglesea and Painkalac were greater than those for all other estuaries.

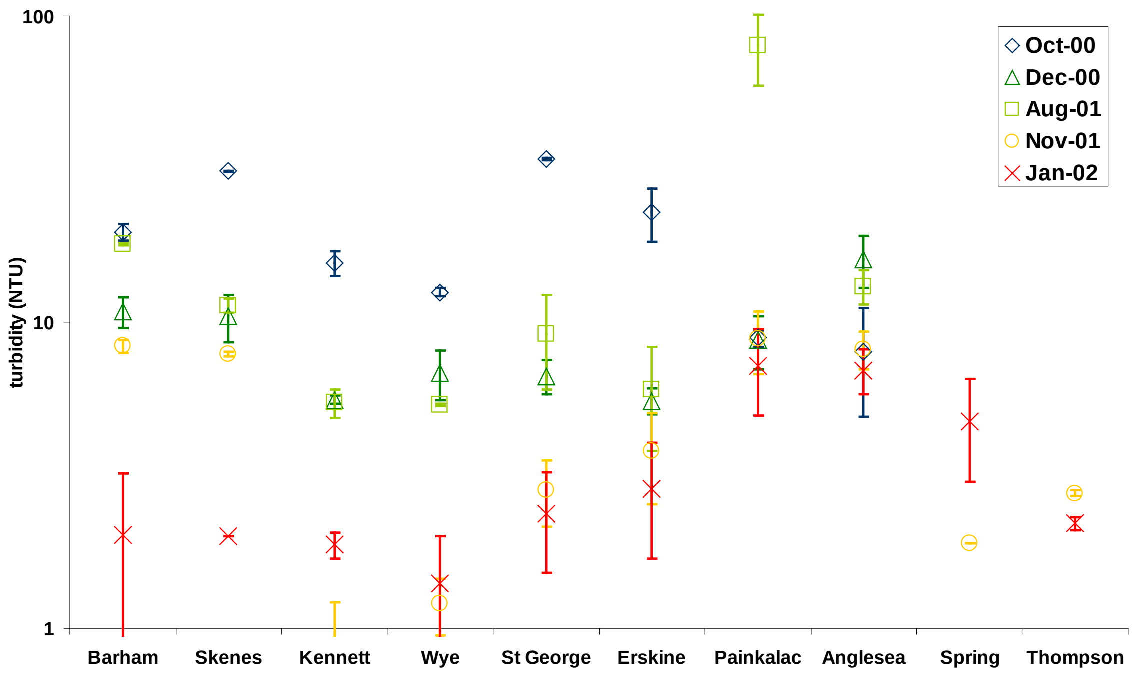

Figure 5.29. Mean turbidity (± std error) for all depths at sites in the lower parts of all Otways estuaries from west to east. Painkalac and Anglesea are each represented by Site 2. Spring and Thompson Creeks were only sampled in Nov 2001 and Jan 2002. Mean turbidity of Kennett River in Nov 2001 was 0.96 NTU.

Long-term patterns

Long-term turbidity was measured by Alcoa in surface waters at a site in the mid-estuary (between Sites 3 and 4 of this study). No seasonal pattern was evident in these data.

Inter-annual variability of turbidity at this site was greater than between-month variability (Figure 5.30) with cycles in the order of ten years apparent in the data. In most years, turbidity at this site was within national guidelines for turbidity but years with relatively high turbidity (1973, 1974, 1993, 1994) fell outside the guidelines. In the context of the long-term dataset, turbidity during the study period was low, and had been decreasing in the five years prior to the study.

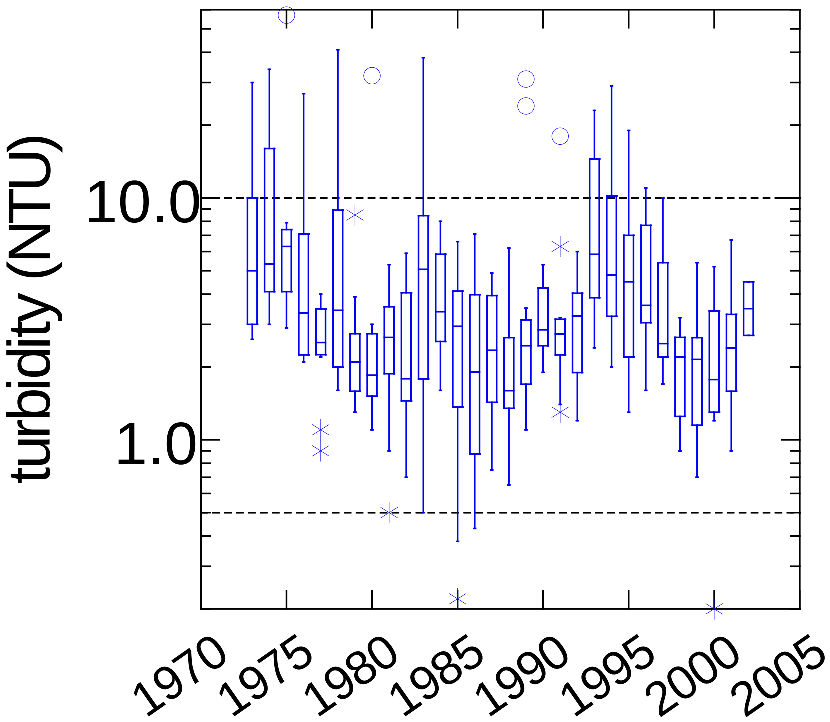

Figure 5.30. Annual turbidity in surface waters of Anglesea estuary at Alcoa's monitoring site, 18/6/1973 to 13/2/2002. Upper and lower dashed lines are 80th and 20th percentiles specified as default trigger levels in national guidelines (ANZECC & ARMCANZ, 2000).

5.4. Summary and conclusions

During the study there were clear differences in salinities and stratification between estuaries. These differences were related to flow directly and indirectly via hydrological state. During tidal and perched periods, similar patterns of salinity structure were observed in both estuaries while during closed periods, differences that were clearly flow-related were obvious. During tidal periods, salinity structure alternated between a salt wedge on incoming tides and full vertical stratification on ebb tides as freshwater flowed out of the estuaries. While perched, a reduced marine influence resulted in the same two patterns seen during tidal periods as well as less saline versions of vertical stratification. When the estuaries were closed, the continual flow in Anglesea tended to reduce salinity but maintain a vertical salinity gradient until, over a period of several months, the estuary became completely fresh. In Painkalac, the relative lack of flow led to the estuary being well mixed during closed periods, with occasional stratification at times of flow. Evaporation resulted in hypersaline conditions in this estuary over the summers of 1998/99 and 2000/01. A large portion of the differences in the salinity structure of the estuaries over time could be related to Painkalac estuary being closed for much greater proportion of the time than Anglesea, and tidal for less of the time. In the context of a regional gradient that increased from west to east, salinities were higher than would be expected in Painkalac and lower than would be expected in Anglesea, consistent with the reduction and augmentation of freshwater flow to these estuaries.

Four aspects of the water quality of Anglesea and Painkalac estuaries were considered particularly likely to cause substantial biological effects:

- completely fresh and hypersaline conditions in Anglesea and Painkalac respectively;
- hypoxia and nutrient release resulting in an algal bloom in Painkalac;
- occasional acidification of Anglesea estuary and;
- periods of low water clarity in Painkalac.

Conditions associated with each of these phenomena are summarised in Table 5.11.

Phenomenon	Estuary affected	Flow	State	Strat. Pattern
Freshwater flushing (fast)	A, P[a]	**flood**	T	any → **A**
Freshwater flushing (slow)	A	**low**	**C**	D → **A**
Hypersalinity	P	nil	C	G → **F**
Eutrophication	P	**nil**[b]	**C**	**G**
Acidification	A, (P)	**high**	any	A,C,D,E

Light reduction	P	**high**	P, T	A,D,E

Table 5.11. Conditions associated with phenomena of potential biological importance. Conditions likely to be prerequisites for each phenomenon are shown in bold. Codes are as follows: A – Anglesea, P – Painkalac; T – tidal, P – perched, C – closed. Arrows between stratification patterns indicate a sequential transition to reach the indicated pattern. a = assumed for Painkalac (not sampled in early part of large flood). b=nil to low flow most likely a prerequisite condition.

5.4.1. Salinity regime

The two extremes of salinity regimes, freshwater flushing and hypersalinity, both had potential for large biological effects in the estuaries. In Anglesea there was a prolonged period in which waters of the estuary were essentially fresh (salinity ~4), associated with an extended closure of the mouth in which estuarine waters were diluted by discharges from Alcoa (Chapter 3). The relatively high salinity of these discharges compared to most fresh water resulted in a mean salinity of ~2 in inflowing waters from Anglesea River, compared to <1 for most Australian Rivers and a global average of 0.1 (Hart & McKelvie, 1986). This may have resulted in less of an effect than would have occurred with fresher waters although differences in the ionic composition of these waters may have had a separate effect (e.g. Gardner *et al.*, 1991; Radke *et al.*, 2003; Zalizniak *et al.*, 2006). In Painkalac estuary there was only a short period of freshwater flushing, associated with high flow volumes during a flood, closely followed by incursion of tidal flows. Of more potential relevance to biota of Painkalac estuary were the two extended periods of hypersalinity, both of which were a result of evaporation in an isolated system with a closed mouth and zero freshwater inflow.

5.4.2. Nutrients

In the single year that nutrients were measured, both estuaries were predominantly closed, although there were periods when Anglesea was perched. Although concentrations of NOx were elevated in inflows to Anglesea, concentrations in the estuary were close to State guidelines and no evidence of planktonic algal blooms was observed. That this occurred in a period with little to no tidal flushing during which there was a relatively high cover of seagrasses (see Chapter 6) suggests that these elevated inputs of

NOx are not great enough to create eutrophication in the system (*sensu* Webster & Harris, 2004). In contrast, bloom conditions were observed in Painkalac, in association with a release of phosphorus from anoxic sediments in the upper estuary in late 1999, following a model described by Davis & Koop (2006). Prior to this, a pocket of high salinity water in a deep hole in the upper estuary had been isolated by a layer of fresher water for some months and had become increasingly hypoxic. Extremely high concentrations of nutrients recorded at this location were very likely to have been released from the sediments and causing an algal bloom, as indicated by high pH (>9) and supersaturated dissolved oxygen concentrations. Following this episode, it is likely that decreased concentrations of dissolved oxygen throughout the estuary were associated with decomposition of phytoplankton from the bloom.

5.4.3. Acidification

Episodes of lowered pH occurred in both estuaries and were clearly flow related. While this aspect of water quality is not typical of all intermittent estuaries, such events have been recorded in a substantial number of estuaries, most often associated with drainage of acid sulphate soils (e.g. Sammut *et al.*, 1993; Sammut *et al.*, 1995; Roach, 1997; Johnston *et al.*, 2004). Results from this study indicate that for the subset of smaller estuaries affected by acid drainage, effects are more likely to be estuary-wide than in the larger estuaries that have been the focus of studies of such events. Due to apparent differences in the proportion of each catchment that generated these acids, and hence the volume and strength of acids flowing into each estuary, conditions of lowered pH in Painkalac were less extensive than in Anglesea. During times when acidic inflows do not completely flush the estuary, but rather are buffered at the halocline, large and potentially localised deposition of metals such as Al are likely (see Section 5.3.5, Appendix E).

5.4.4. Light availability

The final aspect of estuarine water quality with likely implications for biota was the reduction in water clarity by suspended sediments in Painkalac estuary. This was not related to differences in flow between the estuaries, but rather highly turbid episodes in the mainstem of Painkalac Creek. These episodes were only observed at times of higher flow in the second half of the study (see Section 3.4.3.e). The degree of this reduction in clarity was such that light levels in the deeper parts of Painkalac were potentially limiting for plant growth at times. An ameliorating factor, in terms of seagrasses was that the times of decreased clarity often coincided with perched or tidal states. Two favourable aspects of these states for seagrasses were that water level was lowered, decreasing the depth and the proportional reduction in light to the seagrass beds, and that clear, inflowing marine waters were often present beneath the turbid freshwater layer.

6. Seagrass and Sedimentary Processes

6.1. Introduction

Seagrass beds are a potentially important component of the ecosystems of smaller estuaries of southern Australia, given their relatively high productivity and habitat value in larger systems throughout temperate Australia (Larkum *et al.*, 1989). Despite this, little has been done to investigate the relationships between estuarine hydrology and seagrass habitats. Of the two taxonomic groups of seagrasses studied in Anglesea and Painkalac estuaries, the Zosteraceae (comprising *Heterozostera tasmanica* and *Zostera muelleri* – see Section 6.2.1.b for discussion of taxonomy) have been the most studied locally, primarily in Port Phillip and Westernport Bays, two large marine embayments to the east of the study area. *Zostera* has been well studied as a genus both along the east coast of Australia and in the northern hemisphere but little of this research has been done in intermittent estuaries (e.g. Moore & Short, 2006). The other group, the genus *Ruppia,* has not been the subject of detailed study along the Victorian coast and, while often found in saline environments, are not considered by many authors to be ‘true’ seagrasses (e.g. Larkum *et al.*, 1989).

Two models for variation in seagrass extent provided a context for interpretation of the results of this chapter. The first, based on seagrass dynamics of local embayments, suggests a perennial cover of both *Zostera* and *Ruppia*, with seasonal peaks in warmer months. The second, based largely on South African work in similar estuaries (e.g. Talbot *et al.*, 1990; Adams & Talbot, 1992), suggests that freshwater inputs can cause a regime of disturbance that over-rides seasonal growth patterns. Due to inherent differences in hydrological conditions, sediment dynamics and physico-chemical variability between bays, permanently open estuaries and intermittent estuaries the second model was expected to be more applicable to the dynamics of seagrass beds in these southern Australian systems.

Talbot *et al.* (1990) found that flooding in two small South African estuaries (the Nahoon and the Kwelera) removed almost all seagrass and other macrophytes. Following the floods, *Zostera capensis* recovered over a period of years while *Halophila ovalis*, previously not seen in these estuaries, rapidly colonised their middle and lower sections. Sedimentation processes associated with mild flooding in these estuaries were thought to have been an important factor in the temporal dynamics of seagrasses. In contrast to complete scouring associated with a major flood, minor floods were seen to either reduce growth or smother seagrass beds. A reduction in these small floods by river regulation was also thought to have been partially responsible for a long-term increase in cover and biomass of *Z. capensis* in the Kromme estuary in South Africa (Adams & Talbot, 1992).

A decline in salinity gradients associated with a decrease in freshwater flow can be important in structuring macrophyte communities. An example of this is the lack of zonation or separation of macrophyte species in an estuary (the Kromme) that receives little flow (Adams *et al.*, 1992). The greater fluctuation in salinity seen in two intermittently closed estuaries was thought responsible for a dominance of polyhaline species such as *Ruppia cirrhosa*. This is supported by the work of Baird and Heymans (1996) who used a network analysis of carbon flows to conclude that the relatively homogenous and high salinity of the Kromme estuary following freshwater reduction led to a movement from a phytoplankton-dominated system to one dominated by benthic macrophytes and macroinvertebrates.

Light attenuation from riverine suspended material is another potential limiting factor in the distribution of seagrasses. For example, consistently high turbidity associated with river inflow has been postulated to restrict submerged macrophytes to the lower reaches of the Gamtoos estuary (Adams *et al.*, 1992).

This component of the study used analysis of aerial photographs (over 16 years), direct mapping using dGPS and fixed transects (over 27 months) and measurements of small-scale variables (bed-edge position, shoot density, sediment deposition rate, organic matter content and net erosion/accretion over 13 months) to examine variability in seagrass extent and density in the Anglesea and Painkalac estuaries. Specific aims of this component of the study were to:

- describe historical changes in seagrass distribution in Anglesea estuary and examine potential links with flow;
- quantify changes in seagrass distributions and densities in Anglesea and Painkalac estuaries on a seasonal timescale;
- examine sediment dynamics via deposition rates and net erosion/accretion in the lower parts of the estuaries;
- examine changes in density of seagrasses in these estuaries;
- relate changes in seagrass extent and density to hydrological changes, particularly changes that were likely to have been associated with flow; and
- assess relative influences of hydrologic events and seasonal changes on seagrass dynamics.

6.2. Methods

There were two parts to the seagrass component of the study; one examining overall extent of seagrass beds in the two estuaries and one focusing on fixed sites in the lower estuaries. The fixed sites were also the focus of decomposition studies reported in Chapter 7.

6.2.1. Extent

Three methods were used to estimate seagrass extent on a whole-estuary scale: aerial photo interpretation was used to assess the historical extent of seagrass in Anglesea estuary, seagrass beds in Anglesea were physically traced with a GPS in the first half of the study, and a fixed-transect method was used in both estuaries later in the study.

6.2.1.a. Aerial photo interpretation

Six 1:20,000 scale colour aerial photos taken between 1982 and 1998 (provided by Alcoa of Australia and QUASCO) were scanned at 1200 dpi and georectified from prominent landmarks in the Anglesea area that were located to an average 95% precision of 2.3m using post-processed dGPS (Table H.1, Appendix H). A GIS was then used to trace and calculate areas of apparent seagrass coverage at a scale of 1:2,000, with contrast adjusted for optimum definition of seagrass beds. While the depth of water, coverage and scales of the photos varied (Table H.1, Appendix H), deep edges of beds could be identified in photos from all years. To assess potential links between annual rain and seagrass area, a time-series analysis of correlations between seagrass area and annual rainfall of the 10 preceding years was done. To assess potential links between longer-term rain/flow conditions and seagrass area a series of regressions of seagrass area and cumulative rainfall over time intervals from one to ten years prior to the measurement of seagrass area were done. Recognising the non-independence, patterns of R^2 with increasing time intervals were assessed against potential models of flow effects on seagrasses.

Seagrass beds in a 1:5,400 scale black-and-white photo from 1964 were also delineated for qualitative comparison with the later photos (Appendix H).

6.2.1.b. GPS traces

By either wading or towing a GPS on snorkel, traces of seagrass beds were made with a similar precision to that of the bathymetric survey (Appendix F, Table H.3). Three of these surveys were done in the Anglesea estuary, in November 1999, April 2000 and December 2000. No surveys of seagrasses in Painkalac were done using this method (see Section 6.2.1.c).

Seagrasses were identified as either '*Zostera*', *Ruppia* or mixed beds and classified as either patchy or dense during the mapping. Patchy beds were defined as having gaps between stands of greater than ~50cm. Areas of bleached/senescent seagrass were also mapped separately when present.

Beds delineated as *'Zostera'* comprised mainly *Zostera muelleri*, with some isolated stands of *Heterozostera tasmanica*, identified during surveys primarily on the basis of leaf-tip morphology. Since completion of the surveys, there has been a revision of the *Heterozostera* genus, with the description of a new species, *H. nigricaulis*, that is sympatric with *H. tasmanica* in Victoria (Kuo, 2005). *H. nigricaulis* often has a similar leaf tip to *Z. muelleri*, but can be usually distinguished in the field by black, wiry, erect stems. While the presence of such stems was not noted in seagrasses in this study, by the time of the mapping exercises most individual stems were small (~10-20cm long), and such a feature would not have been easily noticed.

A molecular phylogenetic study of Zosteraceae (Les *et al.*, 2002) recommended that *Z. muelleri* be included with *Z. capricorni, Z. mucronata* and *Z. novazaelandica* as one species, *Z. capricorni*, on the basis of genetic similarity between, and large phenotypic plasticity within, these species (although den Hartog and Kuo (2006) point out that the name *Z. muelleri* should be given preference as it was described before *Z. capricorni*). Of the three previously defined *Zostera* species, *Z. muelleri* is the only species that has been recognised historically with a distribution that includes the study area (Kuo & McComb, 1989) and as such, comparisons with other studies focus on those associated with *Z. muelleri*.

Les *et al*. (2002) also recommended that *Heterozostera* should become a subgenus of *Zostera* on the basis of molecular similarities. Kuo however, following his division of the previously monotypic *Heterozostera* genus into four species, recommended that such a revision be delayed pending further investigation of the taxonomic and phylogenetic relationships between the newly-identified species and the rest of the Zosteraceae (Kuo, 2005). It may then be that *'Zostera'* beds reported in this study are composed of up to three species of Zosteraceae: *H. nigricaulis*, *H.*(or *Z.*) *tasmanica* and *Z. muelleri* (or *capricorni*), but current taxonomy is unclear.

Beds delineated as *Ruppia* were identified only to genus. Although three species of *Ruppia* have been recorded in southern Australian estuaries, *R. megacarpa* is considered to be the most common estuarine species. Occasional examination of leaf–tip morphology (*sensu* Robertson, 1984), when there were more mature specimens available, suggested that at least *R. megacarpa* was present, although *R. polycarpa* and *R. tuberosa* have also been reported from Victorian estuaries and bays (Jacobs & Brock, 1982).

6.2.1.c. Fixed-transect method

When seagrass mapping was extended to both estuaries in December 2000, it became apparent that it was logistically unfeasible to continue mapping each bed individually and so a fixed-transect method was used in Painkalac. This method had the combined advantages of improving sampling efficiency and allowing sampling in turbid conditions. From the following survey, in March 2001, this method was also used in Anglesea. Seagrasses were then mapped 5 times in Anglesea using this method and 6 times in Painkalac. To allow comparisons with times when seagrasses were mapped using GPS, transect data were derived from the maps at half-metre intervals along appropriate lines through the Anglesea GPS survey results.

Transects were marked with permanent pegs or paint markings at either end. A rope marked at half-metre intervals was laid between the ends on each occasion. In Anglesea 10 transects were used, at distances of around 50m from each other, between the Great Ocean Road bridge and the mouth of the estuary, corresponding to the area downstream of Longitudinal Site 2 and incorporating the lower-estuarine sites (Figure 6.1a). Occasional inspections showed no seagrass growth upstream of this area. In Painkalac, 16 transects were established at ~100m intervals , corresponding to the area downstream of Longitudinal Site 3 and incorporating the lower-estuarine sites (Figure 6.1b). The two upstream transects were not formally surveyed, but inspected for regrowth after the first three sampling occasions, following the loss of beds at these sites early in the study.

a
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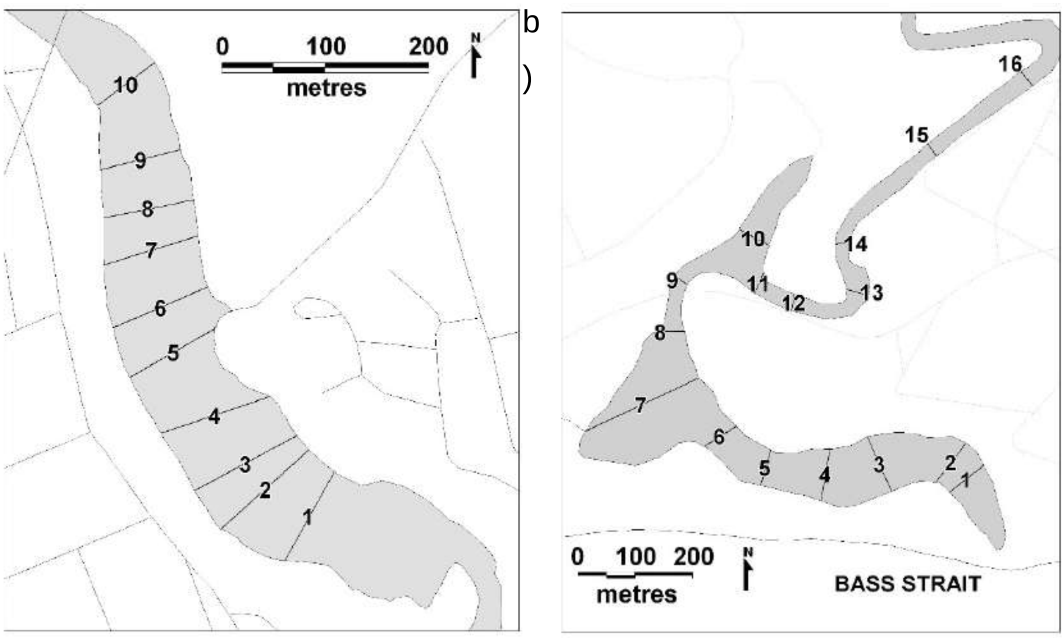


Figure 6.1. Locations of transects used to measure seagrass extent in a) Anglesea and b) Painkalac estuaries.

The same categories of density and composition of benthic cover used in the GPS mapping were used along the transects. Start and finish points of each type of benthic cover were recorded to the nearest half-metre. Profiles of transects were derived from the bathymetric model for Anglesea. Painkalac profiles were limited to measurement of three depths with maximum depth interpolated from soundings at longitudinal sites, incorporating knowledge of the bathymetry of the estuary gained from snorkelling the transects (see Appendix H for details).

Two possible approaches for representing seagrass extent were considered: percent cover and linear distance along transects. The first measure provides equivalency between transects along the estuary regardless of relative and absolute lengths, while the second is a more accurate measure of total cover of seagrasses in the estuaries. Due to uneven transect lengths between estuaries (total length for Anglesea=940m, Painkalac=830m), the influence of one disproportionately long transect in Painkalac (T7: see Figure 6.1), and differences in the density of transects along the estuaries, temporal patterns represented by changes in linear metres of seagrass may differ from those represented by percent cover *per se*. Of the two measures, linear metres was a more representative unit for the total amount of seagrass within an estuary, but did not allow comparisons between transects within estuaries, nor for comparisons between estuaries. On balance, it was decided to represent the extent of seagrasses as percentage cover in this section to examine processes across the whole estuaries with even spatial representation and to allow direct comparisons between estuaries.

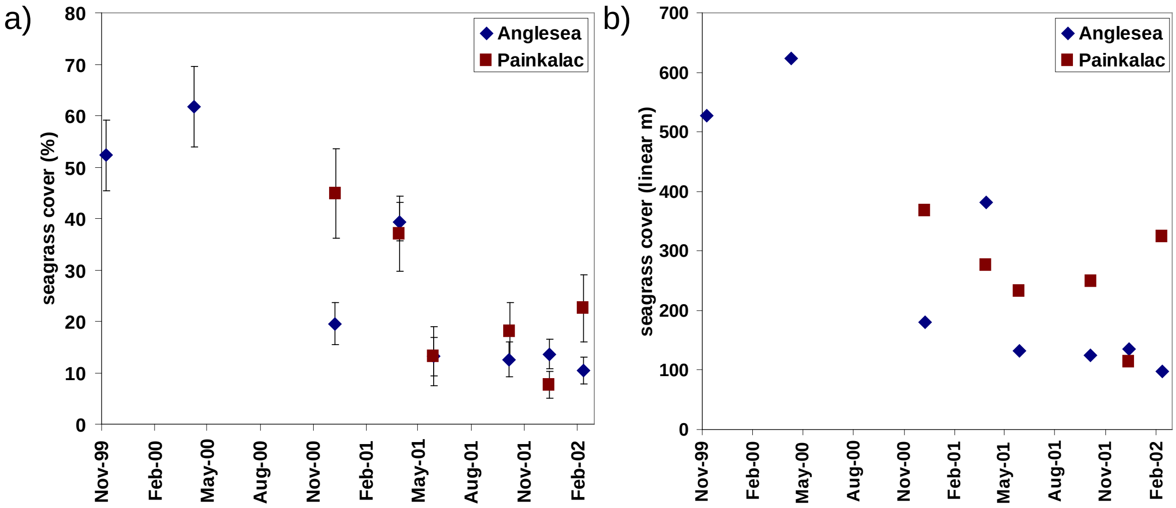


Figure 6.2. Seagrass coverage in Anglesea and Painkalac estuaries through time, measured as a) mean percent cover along transects (± s.e., *n*=10 for Anglesea and 16 for Painkalac) and b) total linear metres along all transects.

While there were some differences in temporal patterns shown by each measure, most notably in the extent of seagrass in Painkalac in May 2001, caused by expansion of seagrass along the proportionally-long transect 7 in Painkalac, the overall pattern of change in both estuaries was similar regardless of the measure used (Figure 6.2).

6.2.2. Lower estuarine sites

6.2.2.a. Site descriptions

Three sites were located haphazardly within the lower part of each estuary, in the general vicinity of Site 1 from the longitudinal surveys. The areas in which the sites were located were broadly identified by observations of densely growing seagrass where the form of the estuary comprised a central channel with mud and sand flats on the fringes. In the text, these sites are distinguished from longitudinal sites by use of the initial of each estuary as a prefix. The sediment dynamics, seagrass shoot density and decomposition components of the study were focused on these sites (Sections 6.3.4, 7.3 and 7.3.2). Sites were 40-70m from each other in Anglesea estuary and 200-500m from each other in Painkalac estuary reflecting the relative sizes of this part of each estuary (Figure 6.3).

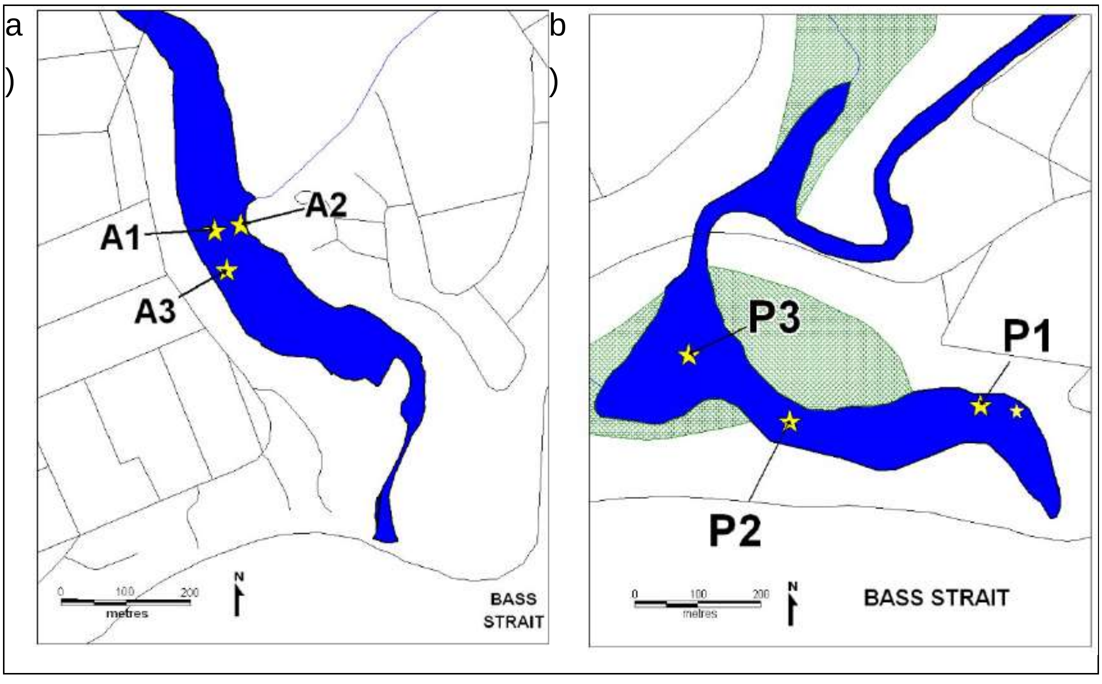


Figure 6.3. Lower estuarine sites in Anglesea and Painkalac estuaries. Initial Site P1 is shown as a smaller yellow star 59 metres to the east of P1.

Sites were ten metres long by six metres wide, with their long axes aligned parallel to the shore. The sites were located within the deeper portions of seagrass beds and at similar elevations (-0.71m to 0.86m AHD). At this depth, the sites encompassed the seaward edge of the flats and the upper part of the slope to the central channel. At the time of establishment it was thought that the sites would remain submerged; however, following changes in the hydrological regime after the April 2001 floods, all sites proved to be intertidal when the estuaries were fully open.

Following direct human disturbance, and to minimise future disturbance, Site 1 in Painkalac (P1) was moved 59m upstream along the shore in March 2001, following the first two deployments of sediment traps and seagrass detritus (Figure 6.3b).

Site elevations

Water depths at nine points in each of the lower estuarine sites were measured to the nearest centimetre on 13/10/01 (Anglesea) and 14/10/01 (Painkalac). On these dates, Anglesea estuary was tidal and Painkalac estuary was perched. Within each site water depth was recorded at the four corners and the centre of the site as well as at points halfway along each boundary. Times were noted for each depth recording to allow correction for fluctuations in water height during the sampling time as recorded at the nearby depth logger in Anglesea and the height benchmarks in both estuaries (see Section 4.3.2).

Over the first half hour of measurements at Anglesea (during survey of Site A1) linear interpolations between water levels logged at 10-minute intervals were used to correct for water level. Subsequent to this, the water level fell below the elevation of the depth sensor so interpolations between the last logged depth and measured depth from the jetty datum 3 hours later (a 7.2cm fall) were used to correct depth measurements from Sites A2 and A3 to AHD. At Painkalac, a fall of only 5mm in water level was recorded over the three hours and 15 minutes of sampling and no corrections for fluctuations in water depth were made.

Anglesea Sites A1 and A3 tended to have lower relief than Site A2 and the Painkalac sites. The centres of the sites at Anglesea were also slightly higher than those at Painkalac (Table 6.1).

Site	Depth (m AHD)			Depth
	Centre	Maximum	Minimum	Range (m)
A1	0.36	-0.13	0.54	0.67
A2	0.19	-0.31	0.71	1.02
A3	0.48	-0.13	0.55	0.68
P1	0.11	-0.71	0.71	1.42
P2	0.32	-0.63	0.86	1.43
P3	0.27	-0.71	0.78	1.49

Table 6.1 Centre point, maximum and minimum depths and depth range of lower estuarine sites (m AHD) as measured in October 2001.

6.2.2.b. Bed-edge changes

Although sites in the lower estuaries were initially located completely within seagrass beds, deep edges of the beds contracted and crossed into the sites early in the final year of the study. Between March 2001 and February 2002, on a ~five-weekly basis, the location of the edge of any seagrass bed within the three lower estuary sites in each of Anglesea and Painkalac estuaries was recorded at one-metre intervals along the sites parallel to the shore. At each interval, the distance of the bed edge from the fixed longitudinal centre line of the site was recorded to the nearest half metre (Figure 6.4).

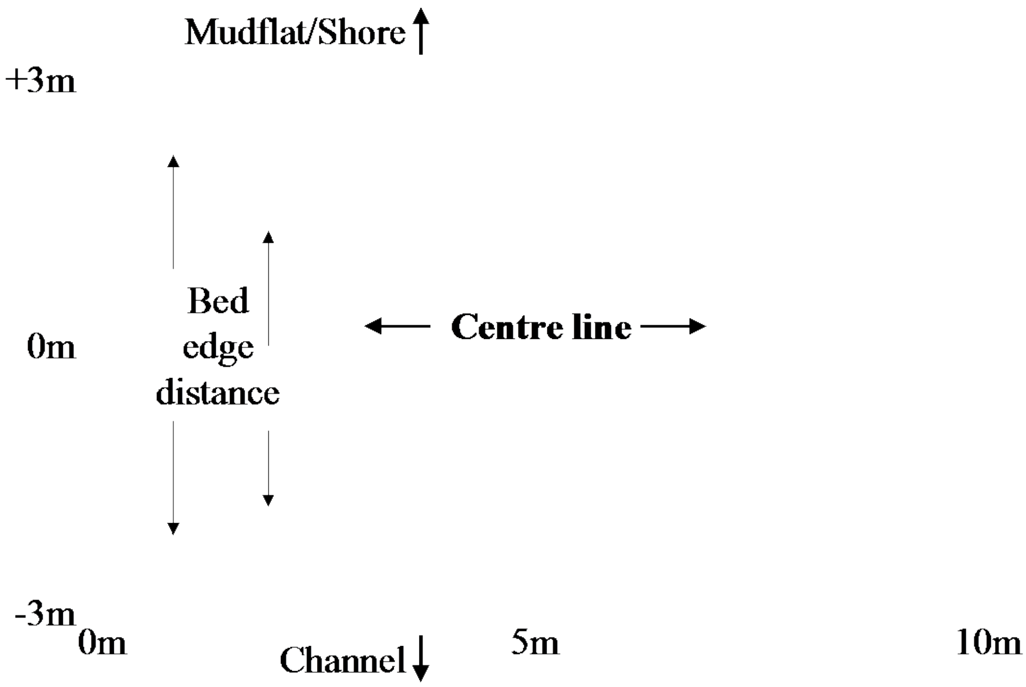


Figure 6.4. Diagram of a lower-estuarine site, illustrating the method of measurement of deep edges of seagrass beds. Measurements of distance from the centre line (to the nearest half-metre) were made at 1m intervals along the shore from 0m to 10m.

6.2.2.c. Shoot density

In response to qualitatively-observed changes in density during the bed-edge mapping in 2001, estimates of seagrass shoot density were made whilst either wading or on snorkel by counting shoots within a 0.25 m^2 quadrat. Five replicate measurements of shoot density were made in each lower-estuary site in Anglesea and Painkalac five times between 1 October 2001 and 14 February 2002, in association with other sampling at the lower-estuarine sites. Quadrats were located haphazardly within the vegetated area of each site (see Section 6.3.4). When no definable bed edge existed, but isolated shoots were present (at densities < ~1 per 0.25m^2), quadrats were located haphazardly within the general area of the site where such shoots had been observed. Throughout sampling, there was only one leaf cluster observed per shoot.

6.2.2.d. Sediment dynamics and organic matter content

Deposition rates and net erosion/accretion were measured at three sites in the lower reaches of both Anglesea and Painkalac estuaries through 2001 and into 2002. Sediment traps were used to measure deposition rate and organic matter content of settling particles while fixed markers were used to measure changes in height of the substrate. When compared with changes in bed height, deposition rates can also give an indication of sediment resuspension rates near the substrate.

Traps

Cylindrical tubes with one closed end and an aspect ratio (height:mouth diameter) of > 5 were used as sediment traps, taking into account the relatively low-energy environment and following the advice of Hargrave and Burns(1979) and Blomqvist and Håkanson (1981). Following a trial of three trap sizes (Appendix G), polypropylene tubes with a diameter of 15.6 mm and a depth of 96 mm were used.

Deployment

Each deployment over the main study involved placing 3 or 4 replicate paving blocks at randomly determined places within the three 10mx6m sites in the lower part of Anglesea and Painkalac estuaries.

Traps were deployed in groups of three held in grommets in plywood boards that were attached to the blocks. Tubes were placed in 3 of 4 holes that were ~15 cm from each other in a rectangular pattern. Paving blocks placed on the soft substrate were slightly embedded and openings of the tubes were approximately 7 cm above the surface of the sediment to avoid capturing bedload sediments in the traps.

Tubes were deployed for eleven consecutive periods of about 5 weeks each, ranging from 25 to 41 days. For the first and second deployments at Anglesea sites, three blocks were used. Following losses and disturbance of units, four blocks were deployed at all other times and sites. On retrieval, tubes were capped and kept on ice then frozen within 12 hours and returned to the lab. Due to various disturbances during deployment, not all blocks were successfully retrieved and 100% retrieval was only possible for the July

2001, August 2001 and January 2002 deployments (see Table 6.3 for details).

Prior to processing, tubes were thawed and their contents emptied and washed on pre-ignited Whatman GC 50 filter papers in a Buchner funnel. In approximately 30% of samples, organisms (crustaceans, gastropods and small fish) were removed at this point. Sediments were then dried in open, pre-ignited, pre-weighed aluminium trays at 70^{o}C for at least 24 hours. Dry sediment and trays were weighed to the nearest 0.001g on a top loading balance prior to ashing at 550^{o}C for 1 hour in a muffle furnace. Following ashing samples were reweighed. Deposition rates used in analyses were time averaged over deployments, calculated as g(dry wt)/m^2/day. Organic matter content was analysed as the dry weight percentage loss on ignition. Thirteen procedural controls for the filtering, drying and ashing process (using pre-ignited filter papers and trays) returned differences between initial dry weight and ashed weight of –1 to +3 mg, negligible in comparison to the mean loss of 151mg for sediment samples.

Data analysis

Mixed-model, nested ANOVAs were used to determine significant variability in deposition rates and organic matter content in deposits between estuaries, sites within estuaries and blocks within sites for each deployment period. Because of the loss of units in some deployments, randomly selected blocks and/or sites, were omitted from the analyses to allow a balanced design for each period (Table 6.3, Table 6.2). Assumptions of homoscedasticity and normality for each ANOVA were checked by visual inspection of plots of residuals versus expected values and probability plots of residuals. No transformations were required.

Retrieval month	Feb 01	Mar 01	Apr 01	May/Jun 01	Jul 01	Aug 01	Oct 01	Nov 01	Dec 01	Jan 02	Feb 02
sites	3	3	2	2	3	3	3	3	3	3	3
blocks	2	2	2	2	4	4	3	4	3	3	4
n	3	3	3	3	3	3	3	3	3	3	3
days (A,P)	27,26	32,33	24,29	37,35	39,39	38,38	38,41	27,28	33,33	32,32	27,25

Table 6.2. Levels of site and block factors and number of replicate sediment tubes in the ANOVA design for each deployment time and lengths of deployments for each estuary. Estuary was a fixed factor, site and block were both random factors.

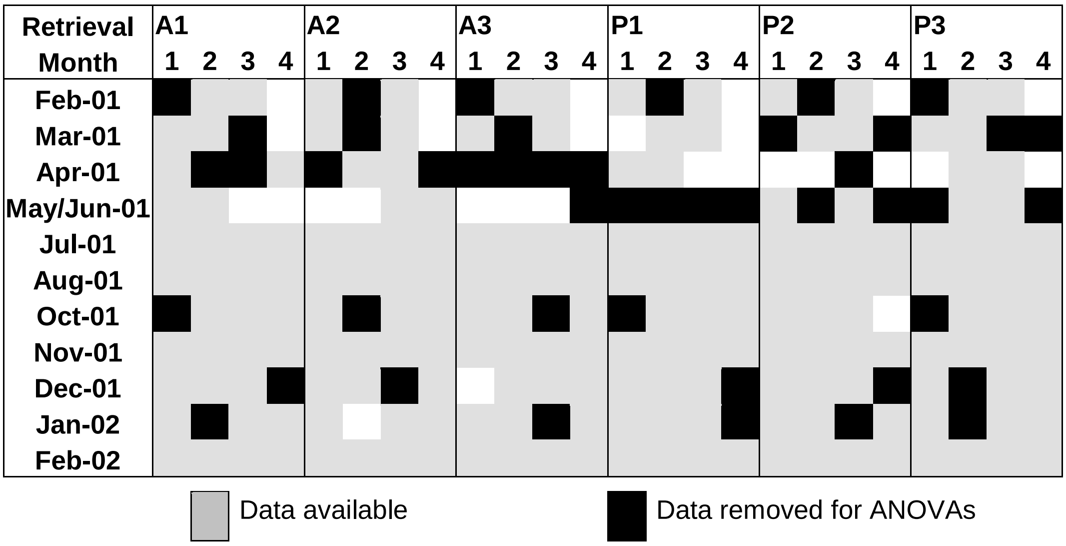


Table 6.3. Summary of successful deployments of sediment traps by site and block. Black shaded cells indicate data that were removed for purposes of balancing nested ANOVAs for each deployment period.

Net erosion/accretion

At each lower estuarine site, three plastic rulers attached to large tent pegs were embedded at random locations within each site. To avoid errors from scouring around the rulers, bed height was measured at a point 22.5cm from the face of the ruler using a spirit level. Bed heights were recorded 12 times between 18/1/2001 and 14/2/2002 on an approximately monthly basis (intervals between sampling occasions ranged from 25 to 45 days). Changes in bed height were measured eleven times, however not all sites or rulers were checked on each trip due to near-zero visibility at times. Table G.2 in Appendix G includes detailed results.

Data are expressed as either daily rate of change in bed height or absolute change, depending on the analysis.

6.3. Results and discussion

6.3.1. Anglesea: aerial photos 1964-1998

Areas of seagrass derived from colour aerial photographs from 1982 to 1998 varied considerably between years and showed an inverse relationship to the amount of rainfall in the previous three years (Figure 6.5, Figure 6.6, Section 3.1).

The spatial patterns of changes in the beds from the colour photographs (Figure 6.5) were similar to those observed in on-ground mapping (1999 and 2000: Section 6.3.2), in that the majority of the reduction in extent of seagrasses was in bed edges moving upstream from the mouth and inwards from the sides of the estuary.

All photos used for direct quantitative comparison were taken at identical scale and interpreted by the same, experienced operator. In the absence of ground-truthing however, caution is needed in interpreting apparent areas from photographs as the actual extent of seagrass beds, as other features of the estuary (e.g. algal wrack, terrestrial detritus, emergent macrophytes) may have been included in estimates of seagrass area while sparser beds and smaller plants would not be visible in the photos. In addition, the photos were taken in different months of each year, between May and September (see Appendix H).

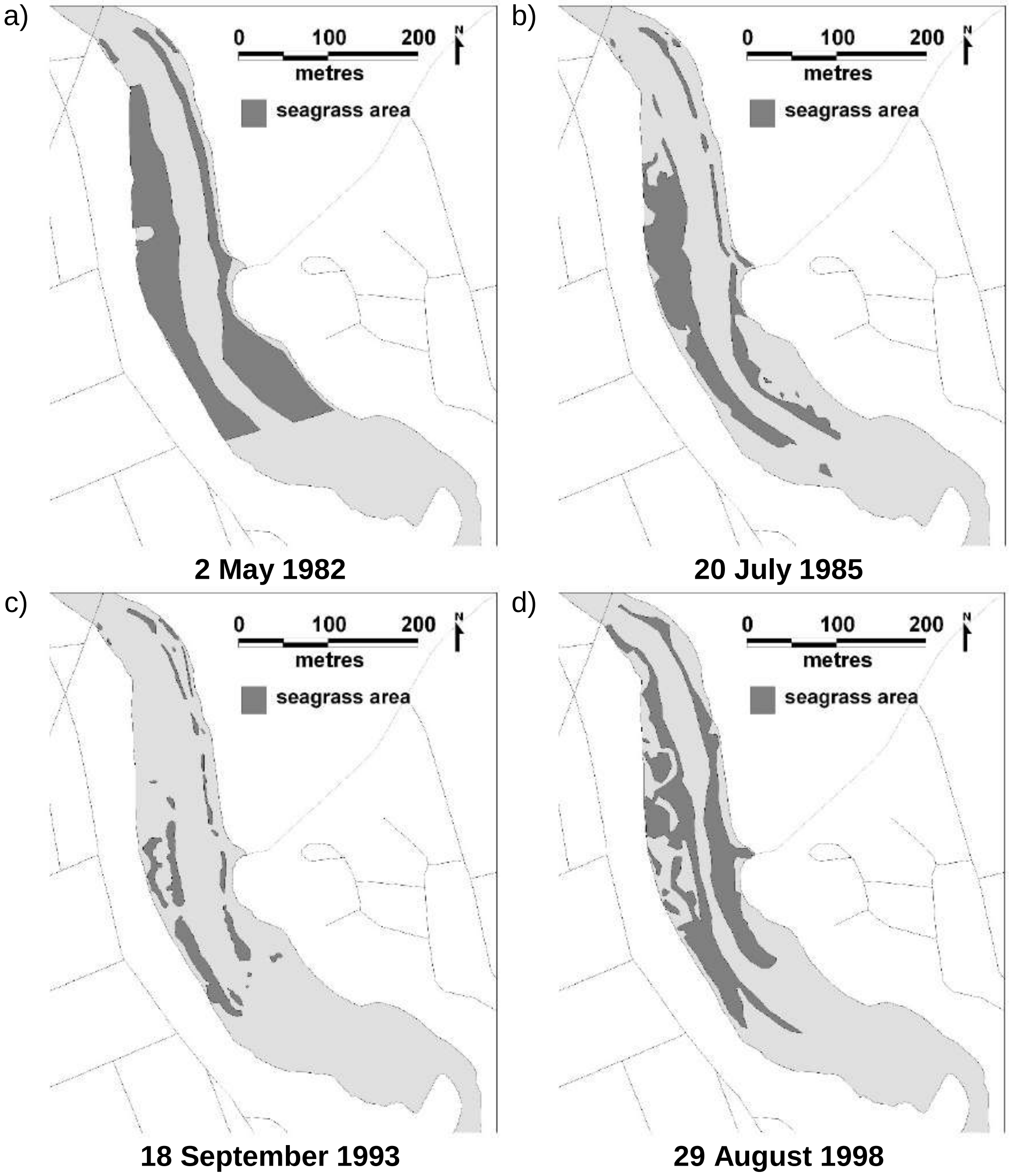


Figure 6.5. Seagrass areas as delineated in selected aerial photos prior to the study illustrating a) 1982, the earliest distribution (but with limited coverage) – note the circular scar from a drain in the western bed, b) 1985, the second earliest distribution, c) 1993, the distribution with the minimum area and d) 1998, the distribution in the year prior to this study. In a) (the earliest colour photo available), the photo did not include the area of the estuary towards the mouth, as indicated by the straight southern boundary of the beds shown. Based on the shape of the beds, and examination of subsequent photos, it was estimated that an area of ~ 5000m^2 of seagrass was not included in the frame of the photo (see Appendix H).

Despite such limitations, these data illustrate that large changes in the areal extent of seagrass beds are likely have occurred in the Anglesea estuary and that, on this basis, over sixteen years there was a large reduction in extent followed by an increase in the years immediately preceding this study. Support for the validity of the overall pattern of extent is provided at the beginning and end of the photo series. The large extent of seagrass apparent in 1982 is supported by anecdotal references in Atkins and Bourne (1983), while the areas of beds identified in the 1998 photo are consistent with an increase in area in the beds identified by on-ground mapping in late 1999 (see Section 6.3.2).

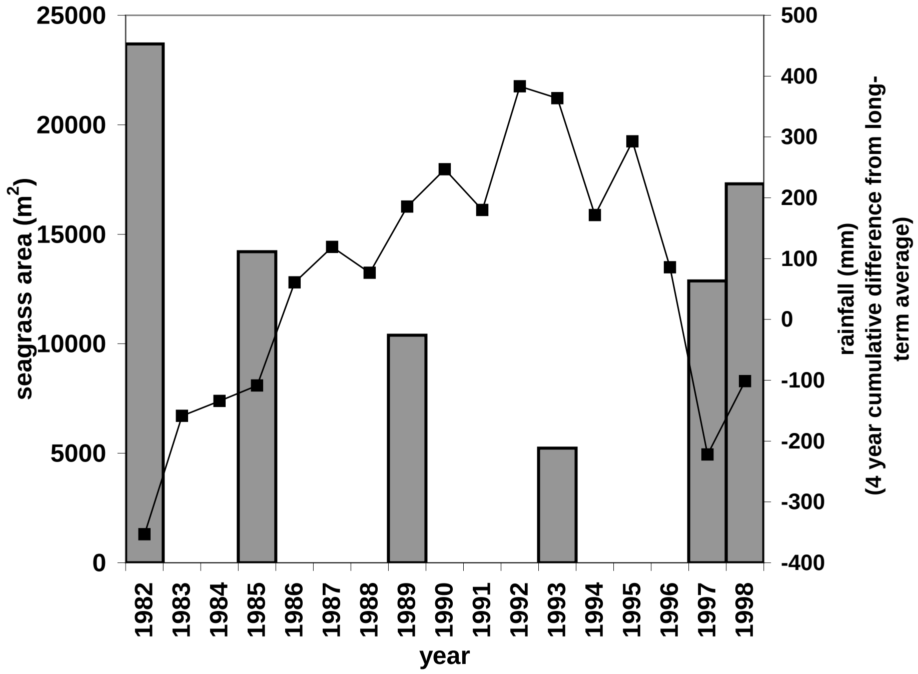


Figure 6.6. Areas of seagrass in the lower Anglesea estuary derived from aerial photos, 1982-1998 (bars) and differences in four-year rainfall totals from the long-term average. Photos from 1982 and 1997 did not cover the entire estuary, so associated areas were very likely to have been underestimated (by an estimated ~5000m^2 and ~600m^2, respectively: see Appendix H).

Based on the observed declines in seagrass extent in Anglesea estuary following an extended dry period and the patterns reported from a South African estuary by Adams and Talbot (1992), the effects of cumulative inter-annual rainfall on seagrass extent were examined using historical rainfall data and seagrass areas derived from the aerial photographs. While in a time series analysis seagrass extent was not significantly cross-correlated with total rainfall for any single preceding year, the R^2 of the linear relationship between seagrass extent and cumulative rainfall increased from

0.62 to 0.79 as the period of integration of rainfall total was expanded from 1 to 4 years (Figure 6.7). As the period of integration was further increased, R^2 was variable but remained at ~0.75 until it declined to 0.34 for 10-year rainfall totals and remained low, corresponding with negative (though non-significant) cross correlations in the time series analysis for lag times greater than 10 years. While the regressions used to derive R^2 values were not independent of each other, these results support a model of seagrass dynamics in which floods cause large declines in seagrasses, which may then recover over a period of several years of stable conditions.

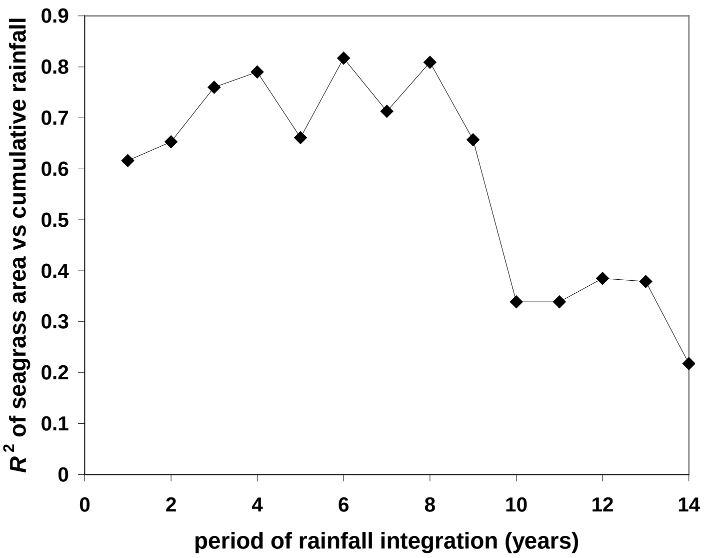


Figure 6.7. R^2 values of relationships between seagrass area in Anglesea estuary and cumulative rainfall over different measurement periods from one to fourteen years. Analyses were done using areas derived from aerial photos, 1982-1998, as per Figure 6.6 and historical rainfall data from Anglesea.

In the context of the pattern observed between 1982 and 1998, the relatively small 6200 m^2 of seagrass visible in the 1964 photo (Figure 6.8) suggests that fringing beds have been a feature in Anglesea for some time but that the area covered has been very variable. The area of seagrass observed was similar to that delineated in the 1993 photo (~5,500m^2), but the beds were closer to the mouth, extending ~150m into an area now included in the flood-tide delta at the mouth (Figure 6.8). The area was smaller than that expected from the relationship with rainfall shown above (four-year cumulative rainfall for that year was only 97mm greater than the long-term average). This may have been due to a flood related to a large rainfall of 64mm in 2 days on 11-12/2/1964 (see Section 3.3.3).

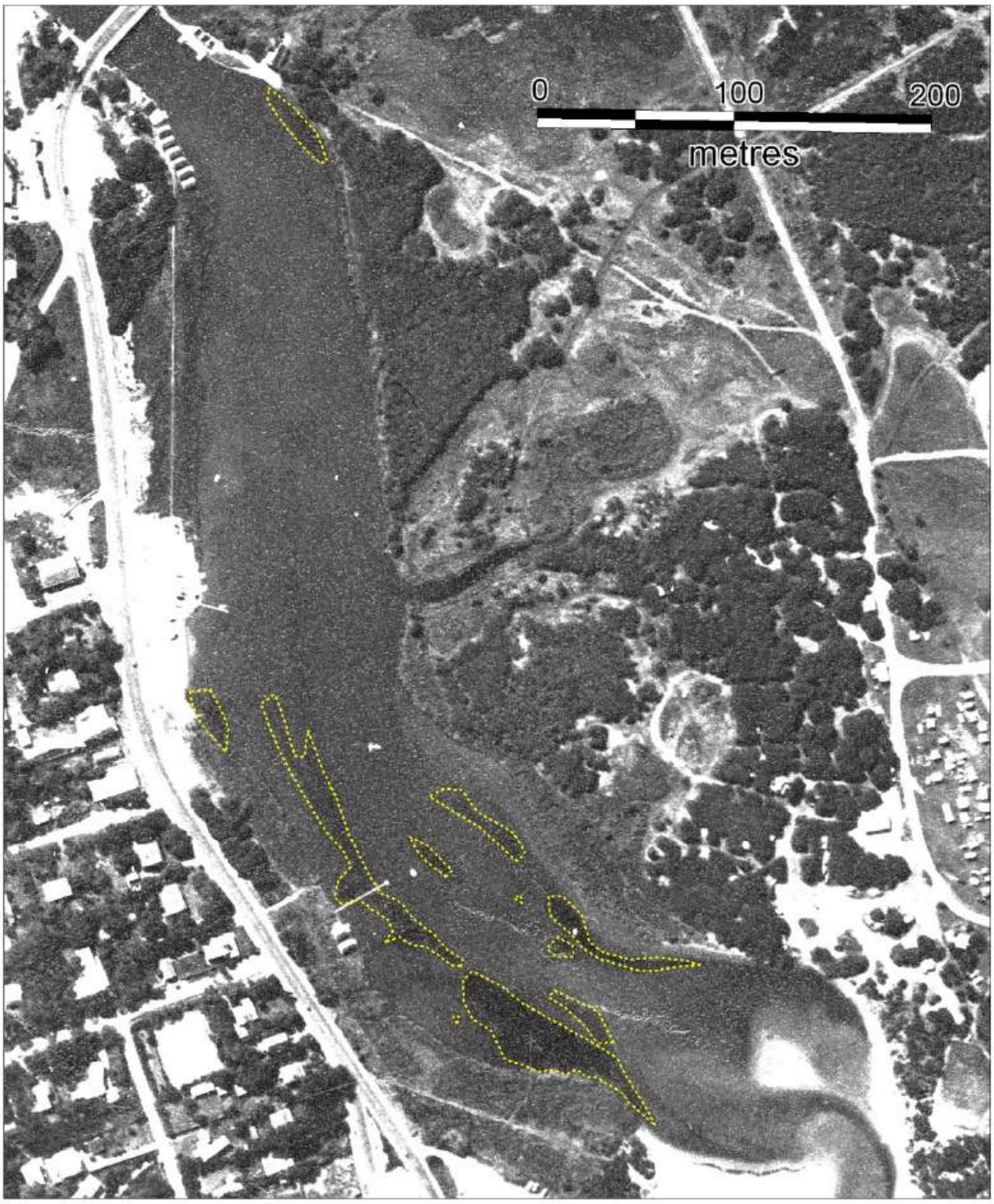


Figure 6.8. Aerial photo of the lower Anglesea estuary, 28 March 1964. Note the dense seagrass beds were confined primarily to the area near the mouth, with some fringing beds further upstream (outlined in yellow). The dark area on the western shore south of the boathouses was most likely macrophytes.

6.3.2. Anglesea: GPS mapping (Nov 1999-Dec 2000)

Between the three survey times during which all seagrass beds were completely mapped in Anglesea estuary, there were changes in both the composition and area of seagrass beds (Figure 6.9, Figure 6.10). The most common type of bed over all surveyed times was a mix of species. Pure *Ruppia* beds were extensive in April 2000 only, while little or no pure *Zostera* bed was present in any survey (0, 208 and 291m^2 respectively). Details of measured seagrass extent by composition and density are in Appendix H.

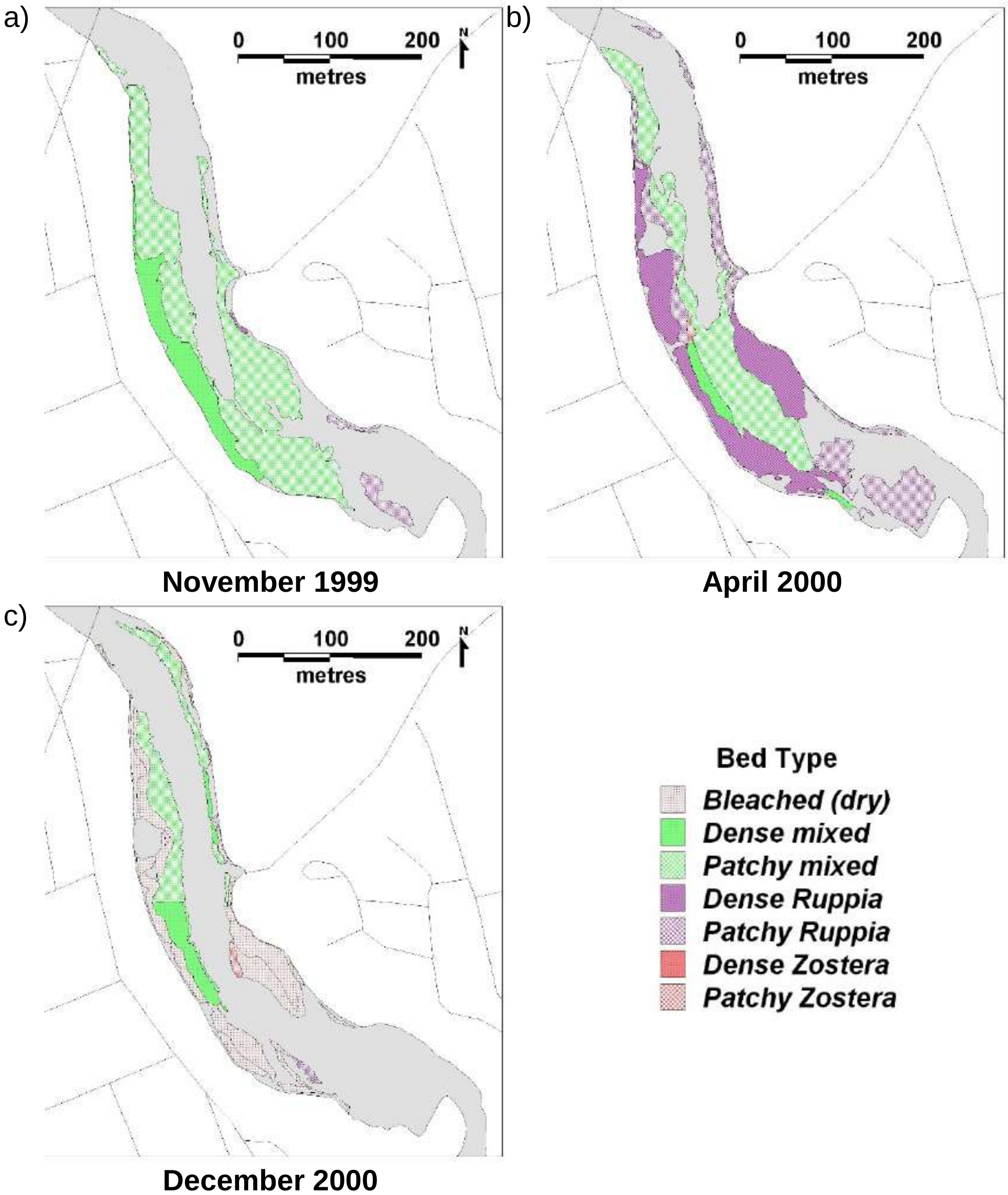


Figure 6.9. Extent of seagrass beds in Anglesea estuary through time as measured by tracing with dGPS.

The greatest cover of *Zostera* (as both mixed and pure beds) was in November 1999 (Figure 6.10), at which time it extended to the edges of the estuary, and across the channel in the seaward part of the estuary, while small beds of pure *Ruppia* were present towards the mouth (Figure 6.9a).

By April 2000, the extent of *Zostera* had decreased, while the extent of *Ruppia* had increased, resulting in large areas of pure *Ruppia* beds, particularly in the shallower parts of the estuary (Figure 6.9b). This change was most likely associated with a reduction in the salinity of the estuary over this time (see Section 0).

By December 2000, there had been a large reduction in the area of seagrass beds, which were now present almost entirely as mixed beds (Figure 6.9c, Figure 6.10). Much of this change was due to the estuary becoming tidal and either permanently or intermittently exposing seagrasses, especially *Ruppia* beds, depending on elevation (Section 6.3.3, Appendix H). These beds were still present, but bleached and dead or dying in the December survey. Some reductions in extent at this time were also evident in deeper areas, possibly associated with acidic flows (Section 6.3.3).

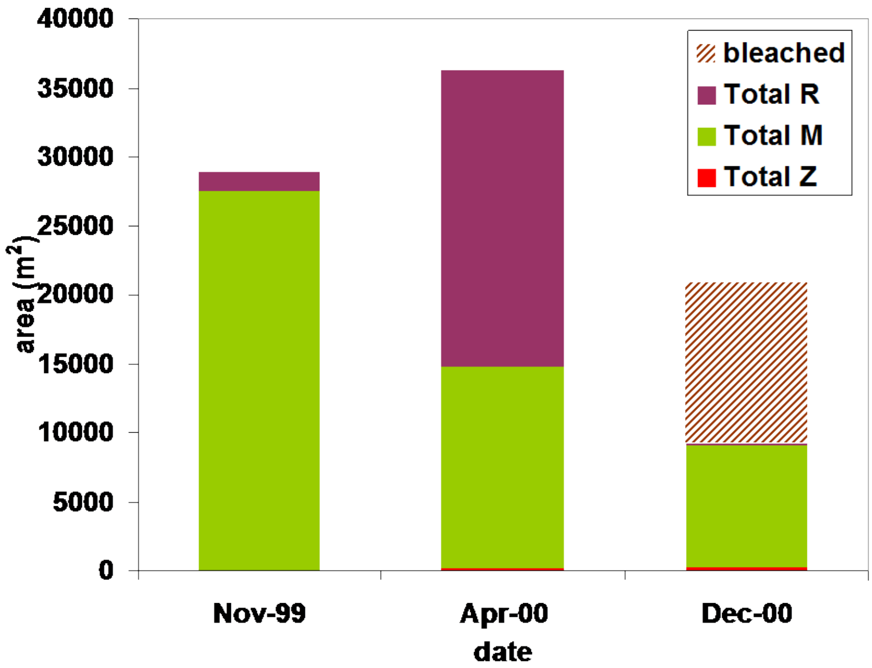


Figure 6.10. Extent of seagrass beds in Anglesea estuary through time. R – *Ruppia* beds, M – mixed beds, Z – *Zostera* beds. Detailed areas are given in Table H.4 and Table H.5 in Appendix H.

6.3.3. Fixed transects (Dec 2000-Feb 2002)

Prior to comparative measurements between estuaries, seagrass beds were more extensive in Anglesea than throughout the period from December 2000 to February 2001. Within the period where both estuaries were measured, there were large changes in the extent of seagrass beds in both estuaries (Figure 6.2a). In Anglesea, there was substantial expansion of the beds over the summer of 1999/2000, followed by a decline in area to a level which remained relatively constant from May 2001 to February 2002. In Painkalac, the area of seagrasses declined from December 2000 to May 2001, from when the area of beds remained relatively low but variable, until an increase over the summer of 2001/02.

Possible causes of the reduction in the area of seagrass beds prior to December 2000 are discussed in Section 6.3.2 above. While there were no quantitative measurements of seagrass area in Painkalac prior to this time, an apparent decrease in the size of both plants and beds during this time was noted while conducting water quality sampling:

- in the lower part of Painkalac estuary, continuous beds, with individual shoot lengths of ~1m were present in October 1999;
- fringing seagrasses upstream to between Longitudinal Sites 4 and 5 (~3km from the mouth, Figure 5.1b) were noted in March, August and October 1999;
- a loss of beds in shallow areas was observed in association with reduction of water levels from evaporation during the summer of 1999/2000 (see Section 4.3.2.b);
- regrowth in these shallow areas was observed by July 2000; and
- a second loss of seagrasses was observed in association with exposure of shallower areas of mudflat by artificial breachings of the sandbar and three short perched periods in the time before the first transect survey in December 2000.

Seasonal growth of *Ruppia* had a large effect on the total area of seagrass beds in both Anglesea and Painkalac estuaries after December 2000. This effect was seen in Anglesea estuary at the end of the summers of 1999/2000 and 2000/01 (Figure 6.11a) and was also seen in Painkalac early in the summer of 2000/01 (Figure 6.11b).

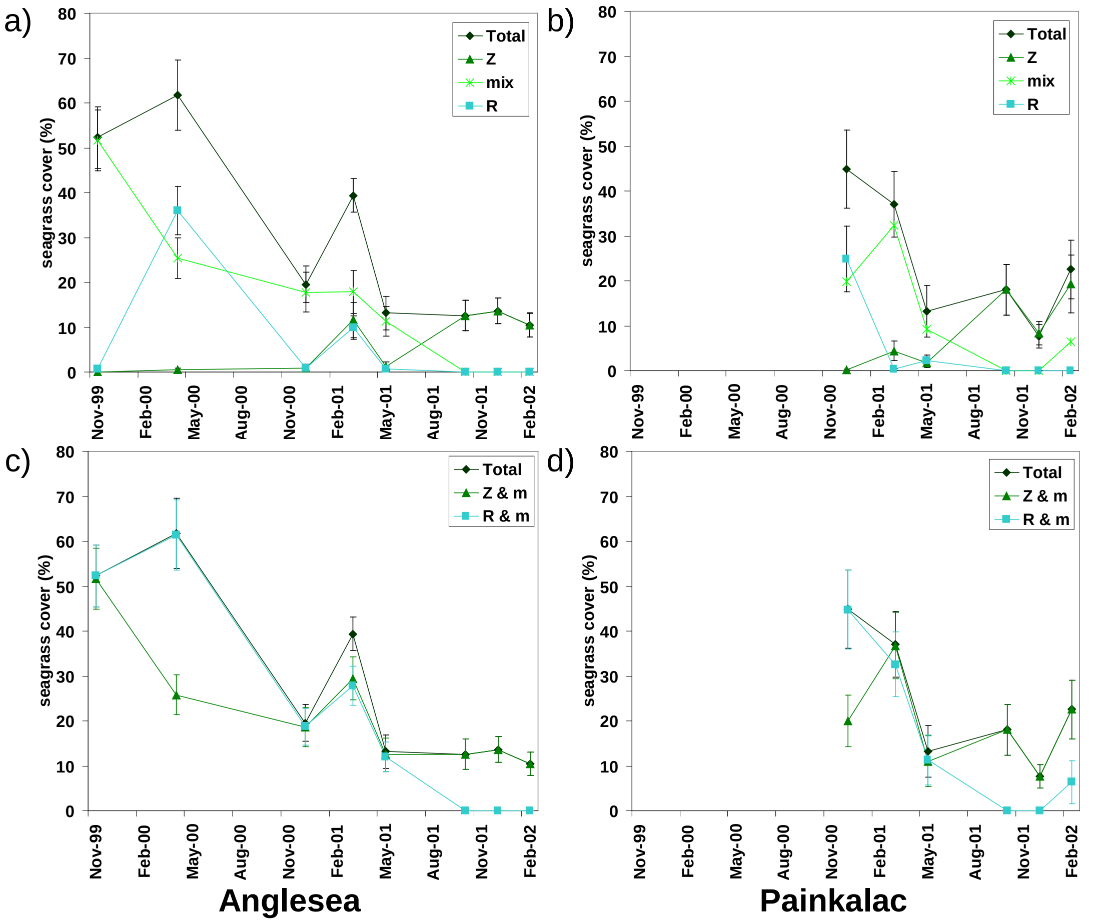


Figure 6.11. Mean (± s.e.) percent cover of seagrasses along transects in: a) and c), Anglesea; and b) and d), Painkalac; estuaries. Figures a) and b) illustrate the proportional composition of seagrasses by bed type: Z – *Zostera*, mix – mixed beds and R – *Ruppia* beds. Figures c) and d) illustrate coverage of each species, in both pure and mixed beds: Z & m = pure *Zostera* plus mixed beds, R & m – pure *Ruppia* plus mixed beds. *n*=10 for Anglesea and 16 for Painkalac.

6.3.3.a. *Ruppia* beds

Ruppia spp. frequently undergo large, often seasonal, changes in coverage and biomass and are capable of living in environments where inundation and salinity vary dramatically. For example, in Australia, *Ruppia* species are known to exist in environments that are inundated for as little as two months a year and across salinity ranges between 0 and 220‰ total dissolved solutes (Brock, 1985). Of the *Ruppia* species recorded in southern Australian estuaries, *R. polycarpa* and *R. tuberosa* typically have ‘annual’ life cycles and are found primarily in temporary salt lakes, while *R. megacarpa* has a nominally more ‘perennial’ life history, although it has been consistently observed to have an ‘annual’ habit in ephemeral salt-lake systems (Brock, 1982b; Jacobs & Brock, 1982; Brock & Lane, 1983). In Western Australia, biomass and percent cover of *R. megacarpa* are known to vary by at least an order of magnitude and to peak and then decline at differing times during warmer months of the year. Within both permanently-open and intermittent estuaries, this process varied among locations and in response to changes in conductivity, turbidity and inundation (Congdon & McComb, 1979; Carruthers *et al.*, 1999). Salinity affects germination of seeds of two of the species in different ways; *R. megacarpa* is more likely to germinate in fresh water and *R. tuberosa* is more likely to germinate in hypersaline water (Brock, 1982a).

Through this study, *Ruppia* cover in both estuaries ranged between 0 and 61 percent (Figure 6.11c,d) with a maximum of 36 percent cover as pure *Ruppia* beds (Figure 6.11a, b). Within the context of a decline in coverage in both estuaries, seasonal peaks were observed twice each in Anglesea and Painkalac estuaries. In Anglesea, *Ruppia* stands (pure and mixed) represented 61 and 28 percent cover in April 2000 and March 2001 respectively (Figure 6.11a), while in Painkalac *Ruppia* stands accounted for an average 45 and 6 percent of total cover in December 2000 and February 2002, respectively (Figure 6.11b).

Anglesea *<u>Ruppia</u>*

Fluctuations in area of *Ruppia* beds in Anglesea and Painkalac can be related to effects of salinity and inundation regimes overlaid on a seasonal growth pattern. In Anglesea, *Ruppia* was only present as mixed beds in November 1999 and by April 2000 had increased to its greatest coverage recorded in the study, mostly as pure stands (Figure 6.11). During this period, salinity had decreased in the estuary (from pattern B to A: Table 5.4, Figure 5.6) as freshwater gradually diluted remnant seawater in the closed estuary, temperature was consistently warm (>19^{o}C) and water levels were high (>1.4m AHD) (Sections 4.3.2, 5.3.2). These conditions are known to be conducive to the growth of *Ruppia*, and are likely to have favoured the growth of *Ruppia* over *Zostera* (Brock, 1982a; Kerr & Strother, 1985; Vollebergh & Congdon, 1986; Carruthers *et al.*, 1999).

Between April 2000 and December 2000 there were several environmental factors that may have contributed to the decline of *Ruppia* coverage aside from the typical winter dieback. The most obvious of these factors was a change in inundation regime resulting in the exposure of a large portion of the mudflats on which it was growing (see Figure 4.9b, Figure 4.15a). In the closed period prior to June 2000 water levels in the estuary were greater than 1.4 m AHD. Following this, reductions in water level to as low as 1.0m AHD were associated with a perched period from June 2000 until late July 2000. Levels then fell below 0.8m AHD with tidal periods from the end of October 2000 through the December 2000 seagrass survey, exposing much of the *Ruppia* on the upper shore (see 'bleached' beds in Figure 6.9c). During this period of decline in *Ruppia* cover, there was also an acid flow event in September/October 2000 in which seagrasses to a depth of ~0.9m AHD were exposed to water with a pH as low as 4.6 and precipitated metals would have covered deeper areas (see Section 3.4.2). Salinity (becoming more saline) and light availability (no change/increased with a reduction in water depth) were not likely to have been major stressors of *Ruppia* beds at this time.

The second peak in *Ruppia* extent in Anglesea (in March 2001) was also at a typical time of year for this genus but in distinctly different environmental conditions than the first peak. Water levels increased in the interval from December 2000, becoming perched in mid-February, three weeks before the sampling date. Salinity was much higher, moving between patterns D and E (Table 5.4), with a distinct marine influence (Section 5.3.1.c). pH remained neutral and temperatures were slightly greater than in the previous summer (Section 5).

By the end of May 2001, *Ruppia* extent, although it was still present in mixed beds, had declined from 28 to 12 % mean cover. By October 2001, there were no beds containing *Ruppia* along any of the transects in Anglesea and there was no regrowth during the following summer (Figure 6.11a,c). In this time, aside from seasonal changes in temperature and day length, potential disturbances included:

- a change in state from perched to tidal, associated with the large floods of April 2001; and
- periods of acidic waters (pH < 5.5) in the lower estuary in April and August 2001 (Section 5.3.5.c).

Despite an increase in water levels in two perched periods towards the end of the period, no regrowth of *Ruppia* was observed by February 2002 (Section 5.3.1.c). This may be related to earlier episodes of lower pH, the exposure to air of much of its potential habitat and/or a lack of sufficiently fresh water to trigger germination of seeds as in the previous two summer blooms (and as has been observed in Western Australia: Carruthers *et al.*, 1999).

Painkalac <u>Ruppia</u>

The greatest extent of *Ruppia* in Painkalac, as expressed by mean percent cover, was at the first (December 2000) and last (February 2002) sampling times. Much of the cover in Painkalac in December 2000 was as pure beds in shallower parts of transects (Figure 6.11). Some dry and bleached *Ruppia* beds were also observed at this time. Water levels in the four previous months had been high (>1m AHD), alternating between closed and perched hydrologic states (Section 5.3.1.c), providing a large area of shallow habitat covered with brackish water that was slowly increasing in salinity (Figure 5.5).

Expansion of mixed beds combined with die-off of higher pure *Ruppia* beds resulted in a net decline in coverage of *Ruppia* by March 2001. It is likely that this was the result of a reduction in water level and hence available habitat through evaporation (Section 4.3.2). From this date onwards, negligible (zero to two percent) areas of *Ruppia* were present as pure beds.

By May 2001, the estuary had become fully tidal and overall seagrass extent declined markedly with the exposure of large areas of habitat and scouring associated with the large flood. No *Ruppia* was observed in the estuary in October or December 2001 but by February 2002 there was six percent mean cover of *Ruppia* as mixed beds. The growth of *Ruppia* in Painkalac at that time, when none was seen in Anglesea, was most likely related to the higher water levels associated with closure of Painkalac but not Anglesea.

6.3.3.b. *Zostera* beds

As for many temperate seagrasses (Duarte *et al.*, 2006), *Z. muelleri* and *H. tasmanica* in Victorian bays have been observed to have highly seasonal patterns of growth rates and both above- and below-ground standing crop, with smaller differences between seasons in *H. tasmanica* than *Z. muelleri* (Bulthuis & Woelkerling, 1983b; Kerr & Strother, 1989; 1990; Edgar *et al.*, 1994). In Australia and elsewhere, these seasonal patterns have sometimes been overshadowed by large-scale decline and, less often, recovery over periods of years (reviewed in Clarke & Kirkman, 1989; Duarte *et al.*, 2006). A local example of this is the modification of seasonal patterns at some sites in Westernport Bay related to smothering of seagrass beds by mobile sediments (Edgar *et al.*, 1994).

Leaf length of *H. tasmanica* has been shown to be inversely proportional to light availability while shoot density declined with persistent reductions in light (Bulthuis, 1983a). In laboratory conditions photosynthetic rates and light compensation points for *H. tasmanica* increased from 5^{o}C to maxima occurring at 30 and 35^{o}C, respectively, indicating an optimal temperature, if not light limited, of 30^{o}C (Bulthuis, 1983b). Similar results have been found for *Z. muelleri* (Kerr & Strother, 1985). *Z. muelleri* has also been shown to be strongly photosynthetically inhibited as pH decreases below 7.8 (Millhouse & Strother, 1986) and as salinity increases or decreases from that of seawater (Kerr & Strother, 1985). Despite this, apparent photosynthesis in both species has been maintained over a wide range of temperatures and, at least for *Z. muelleri*, to salinities as low as ~6 (Bulthuis, 1983b; Kerr & Strother, 1985). Low salinities have stimulated germination of several *Zostera* species in laboratory studies while photosynthesis and production are generally greatest at intermediate (10-20) to high, but not hypersaline, salinities (reviewed in Moore & Short, 2006).

Unlike *Ruppia*, *Zostera* beds were present in both estuaries throughout this study, primarily in the form of mixed beds until coverage of *Ruppia* declined to near-zero in the last five months of the study period (from October 2001 onwards). Seasonal peaks in extent were observed in the summers of 1999/2000 and 2000/01 (at Anglesea), and 2000/01 and 2001/2 (at Painkalac), but not at Anglesea in the final summer of the study (Figure 6.11).

Between November 1999 and December 2000, mean cover of *Zostera* in Anglesea declined from 52% to 19%, most of the reduction in extent occurring by April 2000. In this period, *Zostera* disappeared from large areas of previously mixed-species beds in shallower parts of the estuary, becoming pure stands of *Ruppia* (Figure 6.9a,b). As discussed above, waters of the

Anglesea River became progressively fresher during this time and it is likely that this was a major cause of the decline in Zosteraceae in this period, primarily through reductions in photosynthesis related to salinity (see Kerr & Strother, 1985) but also potentially due to interactions with expanding *Ruppia* beds. In October 2000 there was a flood and an associated transition from a closed to a tidal state. This may have been responsible for reductions in extent in shallow areas due to exposure, as well as being a possible cause of the recession of deep edges, such that beds were divided across the central channel (Figure 6.9).

Between December 2000 and March 2001, there was an expansion of *Zostera* beds in both estuaries, as would be expected from seasonal growth dynamics. This expansion took place in the context of a tidal-to-perched Anglesea estuary with stratified but saline waters and a closed, well-mixed and saline-to-hypersaline Painkalac estuary (Section 5.3).

By May 2001 there had been a substantial (~50%) decrease in the mean cover of *Zostera* in both estuaries. While the timing of this reduction was consistent with the beginning of a seasonal decline, the scale and speed of the decline was most likely related to the flood of April 2001. This flood increased sediment deposition rates (in Painkalac) and erosion, filled the estuaries with fresh, acidic waters and resulted in tidal states in both estuaries, with water levels persistently lowered by ~1m in both estuaries thus exposing large portions of beds (see Section 4.3.2).

Despite low-pH events in Anglesea, no further decrease in seagrass extent in this estuary was observed during the study (*i.e.* in October, December 2001 and February 2002) but nor was an increase seen in the summer of 2001/02. In contrast, there were fluctuations but an overall increase in percent cover of *Zostera* beds in Painkalac in this time. The decreased percent cover recorded in December 2001 was primarily linked to a short tidal period and associated drop in water level in early December. Some of this loss of cover was observed as a 7 percent mean cover of senescent seagrasses on 14

December. After this survey, water levels increased through a perched and a tidal period until the final survey in February 2002.

6.3.4. Small-scale processes (2001-2002)

At the lower-estuarine sites in both estuaries, changes in the deep edges of seagrass beds were measured from March 2001 to February 2002 and shoot density in the vegetated portions of these sites was measured between October 2001 and February 2002. These measures were closely tied to each other as shoot density was only measured in the vegetated portion of any site and the vegetated portion of the site was defined based on a sufficient density to provide a definable bed edge on a half- to one-metre scale.

6.3.4.a. Deep bed edges – small scale changes

Deep edges of beds were measured for the first time in the month before the large flood of April 2001. At this time, as measured in the large-scale surveys, there were peaks in percent cover of Zosteraceae and a peak of *Ruppia* cover in Anglesea but a decline from earlier coverage in Painkalac (Figure 6.11). In both estuaries there was substantially less seagrass coverage than in the preceding summer although this was only a qualitative observation for Painkalac.

Sites A1 and A3 were the only two sites for either estuary in which a definable bed edge could be measured on every occasion (Figure 6.12). With the exception of two sampling occasions in Painkalac, all times and sites where no bed edge was recorded were a result of insufficient densities to define a boundary (<~1 shoots per $0.25m^2$). In August 2001 in Painkalac, there was insufficient visibility to observe any boundary and in July 2001, in the same estuary, a combination of low densities and poor visibility made sampling impossible.

In Anglesea, the direction and size of changes in the locations of bed edges were initially variable between sites and times, but edges receded at all sites between July and October, then became relatively stable and consistent between sites until the end of the sampling period. A net recession of the mean location of bed edges at Sites A2 and A3 was evident over the duration of sampling, with only a slight recession at A1. Between March 2001 and January 2002 there were recessions in mean locations of bed edges of between 3.3 and 5.7 metres at Painkalac sites, but substantial expansions at Sites P1 and P3 between the January and February measurements resulting in net recessions of 1.6 and 4.2 metres, respectively, during the sampling period (Figure 6.12). The existence of defined bed edges in early November but not in December suggests regrowth that was then reduced by floods in mid-November and December.

Compared to the large-scale pattern of changes in bed edges in Section 6.3.3, small-scale changes were consistent in Painkalac for all times sampled. In Anglesea the reduction in seagrass extent following the large flood was evident on a small scale at Sites A1 and A2 by early July, but continued to decrease to October, in contrast with mean large-scale percent cover, which had reached its minimum by May (Figure 6.11).

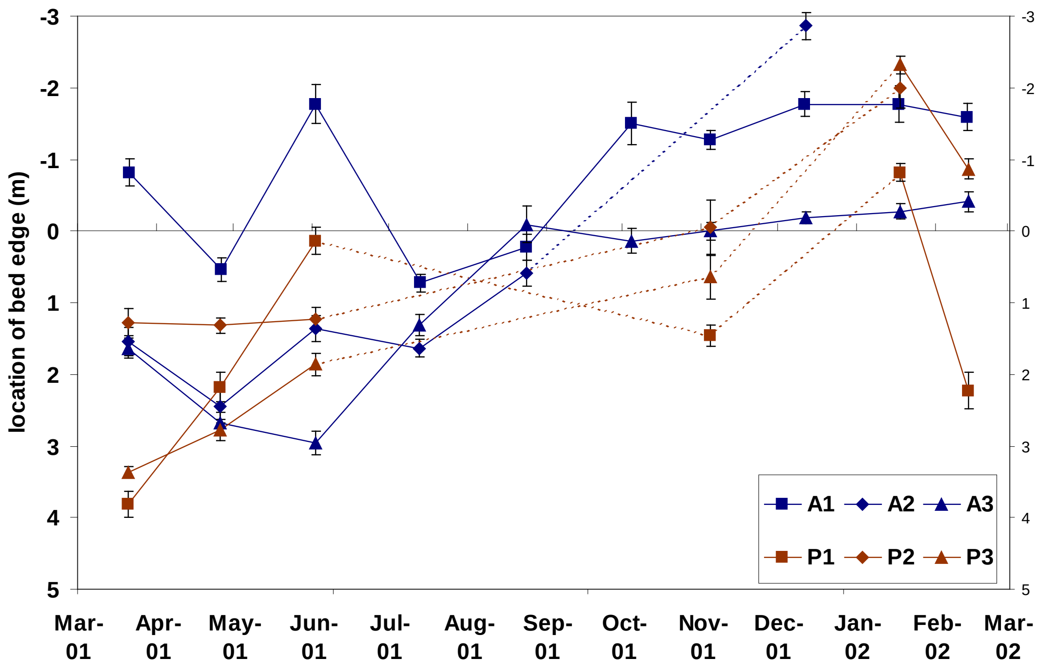


Figure 6.12. Mean (± s.e., *n*=11) locations of deep bed edges in relation to the longitudinal centre line of lower estuarine sites between March 2001 and February 2002 (as illustrated in Figure 6.4). Times where no value is shown reflect a

combination of bad visibility or lack of a definable bed edge in surveys at the relevant site. Solid lines connecting means indicate sampling on consecutive occasions.

6.3.4.b. Shoot density

Shoot density was highly variable over the five months that it was measured. At many times and sites, mean density was below the density at which a bed edge could be defined (~1 shoot per 0.25m^2 – see Section 6.3.4.a). Maximum densities were recorded at Sites P2 and P3 in January 2002, at which time bed edges had receded to a 'core' part of the beds, in which densities were higher than those in the area of recession and expansion (Figure 6.12, Figure 6.13). The subsequent reduction in density at these sites was most likely related to lower densities in the expanding portion of the beds.

For most of the period in Anglesea, densities were higher at Sites A1 and A3 than at A2, a fact reflected in the lack of a defined bed edge at A2 for most of this period. A slight increase in density was apparent between January and February 2002 at this site. The decrease in mean density at A1 cannot be attributed to an expansion of the bed, nor to any flow or state-related cause but at this time it was also noted that 80 percent of leaves had become brown.

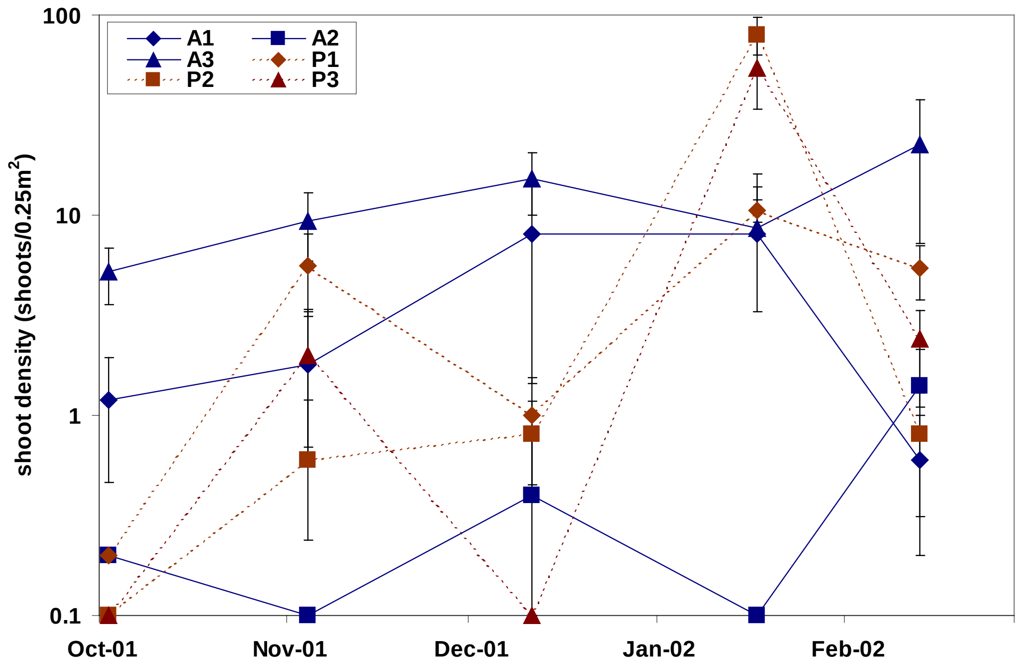


Figure 6.13. Mean (± s.e., *n*=5) shoot density of seagrasses in lower estuarine sites of Anglesea and Painkalac estuaries. Zero values are represented as 0.1 shoots/0.25m^2 to allow presentation on a log scale.

In Painkalac, two peaks in density are apparent, in early November and mid-January (Figure 6.13). These peaks may reflect the beginnings of summer growth, which were then reversed with two floods in mid-November and early December, followed by a continuation of growth, resulting in increased density by January and expansion of beds in February.

Compared with densities observed in more marine environments in the region, shoot densities were low in Anglesea and Painkalac, although similar to those recorded in most regional estuaries (Table 6.4). Of the estuaries in which seagrasses were observed, Breamlea had the most perennial and extensive beds. This was most likely related to the estuary having a permanently open mouth, large areas of shallow and intertidal substrate, and comparatively low and intermittent freshwater flows.

Location	Date	Shoot density (per m²)	Source
Anglesea	10/01–2/02	0–67	this study
Painkalac	10/01–2/02	0–320	this study
Erskine	11/01	4.0	this study
Spring	11/01	26	this study
Breamlea	11/01, 1/02	1320, 1780	this study
Westernport Bay	3/78-5/79	544-3440	Bulthuis & Woelkerling (1983a)
Westernport Bay	4/98-1/99	450-2500	Campbell & Miller (2002)
Port Phillip Bay	9/82-2/83	640-1000	Bulthuis *et al.*(1992)

Table 6.4. Comparative shoot densities of Zosteraceae in regional estuaries and marine embayments.

The highest quantitatively measured shoot densities in Anglesea and Painkalac estuaries were qualitatively sparser than densities observed in 1999 and 2000. Although impressions of high density were likely to have been influenced by the much larger shoots in these years, densities appeared similar to those that have been measured in more marine environments (see Table 6.4).

6.3.4.c. Sediment deposition rates: Lower estuary

In total, 708 samples were collected from the lower estuarine sites in Anglesea and Painkalac. Deposition rates recorded in individual tubes ranged from 29.4 to 2220 g/m^2/day (dry weight) with an overall mean of 232 g/m^2/day and median of 178 g/m^2/day.

On average, deposition rates were greater in Anglesea (179±8.0g/m^2/day) than Painkalac (110±7.8g/m^2/day) but had a greater range in Painkalac than in Anglesea. Differences in overall mean deposition rates were evident between sites within Anglesea estuary, but not between sites in Painkalac (Figure 6.14).

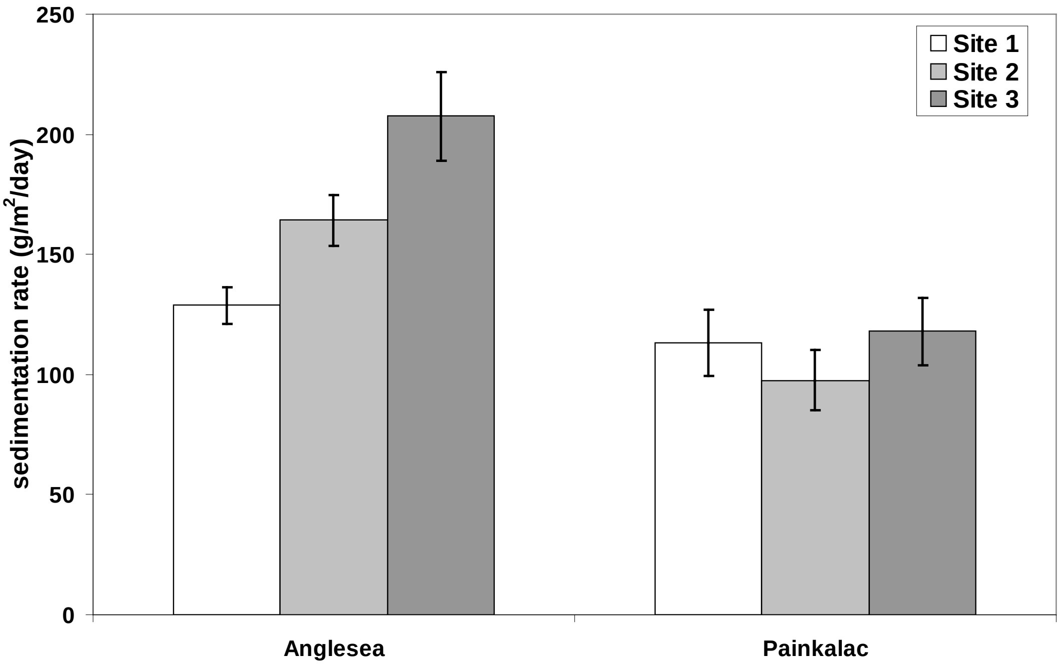


Figure 6.14. Mean (± s.e.) deposition rate by site for all times (*n*=102, 99, 99 and 97, 102, 105 for Anglesea and Painkalac Sites A1, A2, A3 and P1, P2, P3 respectively).

The fully nested experimental design allowed differences in mean rates at each of the three spatial scales to be examined in isolation from each other for each deployment time (Table 6.5). At the block scale, significant differences were seen for eight of the eleven deployment periods while rates of deposition were significantly different at site and estuary scales on five and two times respectively. Due to differing replication at the site and block scales, substantial differences in power between deployments were likely.

Scale\Time	1/1[a]	3/1[a]	4/1[a,b]	5/1[a,b]	6/1	8/1	9/1[c]	10/1	11/1[c]	12/1[c]	1/2
Estuary	NS	NS	NS	NS	*	NS	NS	*	NS	NS	NS
Site	NS	NS	***	**	*	NS	**	NS	NS	*	NS
Block	**	NS	NS	NS	***	***	***	***	***	***	***

Table 6.5. Results of eleven separate nested ANOVAs of deposition rates during each deployment (shown as month/year in the top row). Shading indicates a significant result at *: $\alpha=0.05$, **: $\alpha=0.01$, *: $\alpha=0.001$. 3 sites per estuary and four blocks per site (*n*=3) were used except for a – 2 blocks per site, b- 2 sites per estuary and c – 3 blocks per site.**

At times when differences between sites existed, there was no consistent pattern between sites in Anglesea or Painkalac despite the overall difference in rates seen in Figure 6.14. This suggests that the relative rates of deposition between sites were variable, but that cumulative differences in the magnitude of deposition resulted in the differing overall means between the sites in Anglesea.

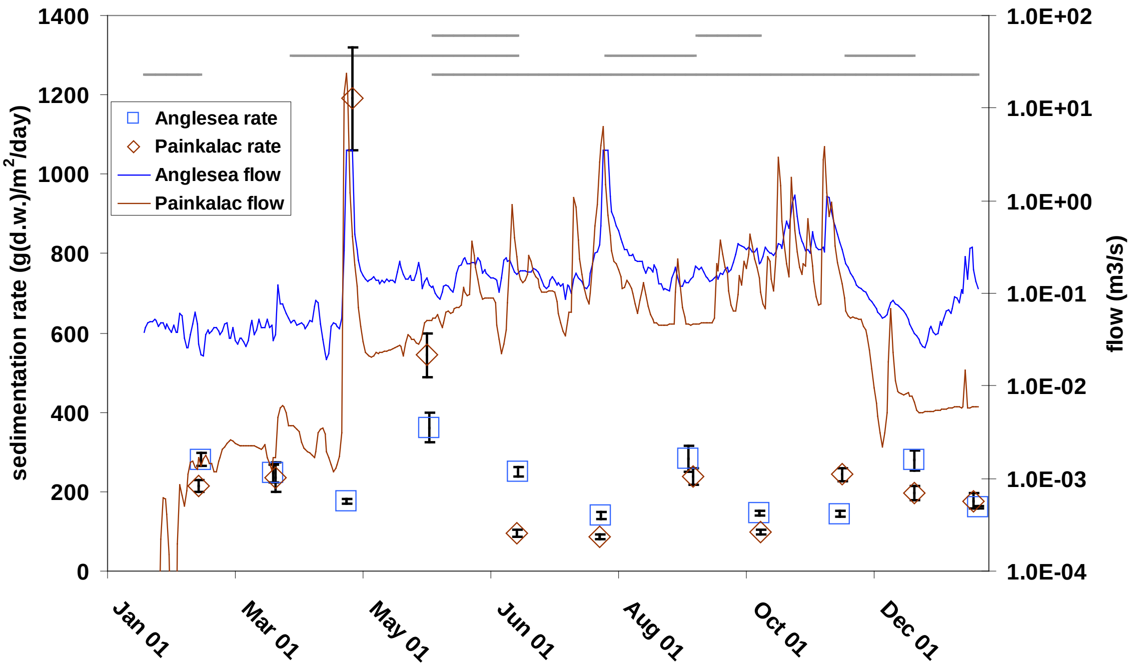


Figure 6.15. Mean (± s.e.) deposition rates and freshwater flow in Anglesea and Painkalac estuaries, with periods in which there were significant differences on three spatial scales indicated by grey lines at top. Mean rates are shown on the end date of deployment periods. The upper grey line represents inter-estuarine differences; the middle, inter-site differences (within estuaries and the lower, inter-block (within site) differences. The large difference in rates for the April deployment is probably due to collection of the Anglesea tubes in the early stages of the flood and the Painkalac tubes two days later. *n* was variable, between 15 and 36 tubes per estuary (see Table 6.3).

Deposition rates and variability in both estuaries increased in periods following high flows, most obviously that of April 2001 (Figure 6.15). These effects in Anglesea tended to last longer than in Painkalac, which may account for both times at which there were significant inter-estuarine differences being during the second deployment post-flood.

Significant between-site differences occurred only in deployments immediately after or incorporating the three largest high-flow events, except the large flood of April, which was followed by two deployment periods in which there were inter-site differences in mean deposition rate. Significant within-site (between-block) differences were common (8/11 deployments), reflecting small-scale heterogeneity in deposition rate. Significant differences were observed for all periods for which >2 blocks were retrieved per site, but for only one of the four deployments where only two blocks were included in the analyses. It is likely that at least part of the three non-significant results is related to the reduction of power in the four analyses with lower replication.

In addition to influences from freshwater flow, degree of tidal influence (estuarine state) was considered likely to influence deposition rates through mechanisms including periodic exposure of substrate, alteration of wind resuspension via changes in fetch and depth, and resuspension via tidal flows. Deposition rates were higher in closed and tidal states than in perched states and, greater in Painkalac than Anglesea during tidal states (Figure 6.16).

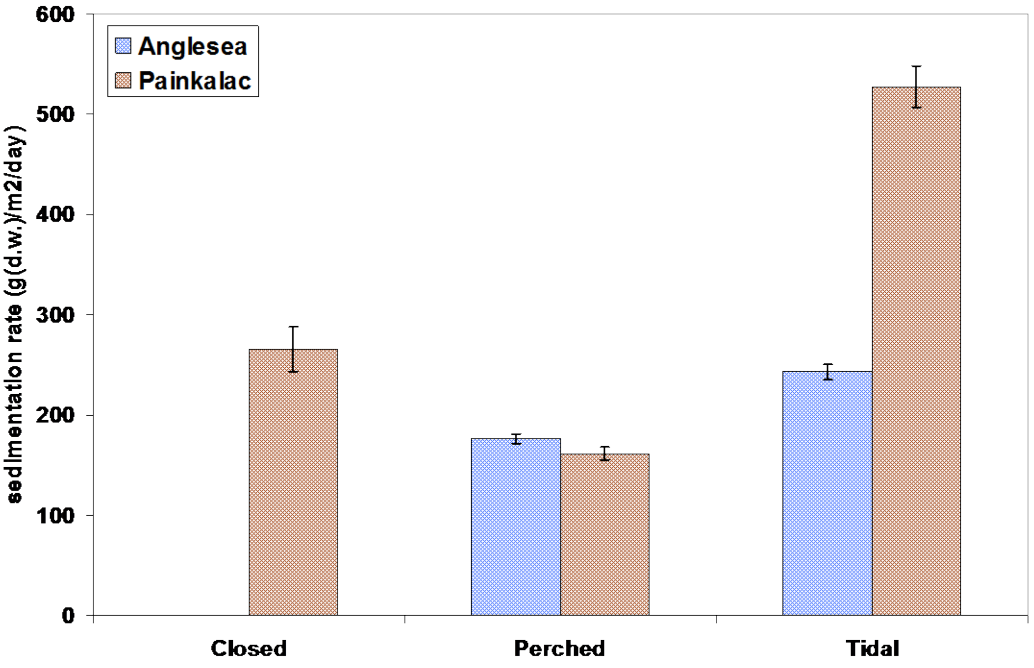


Figure 6.16. Mean (±s.e.) deposition rates during the three states for Anglesea and Painkalac estuaries. Where a deployment period encompassed more than one state, values were weighted according to the proportion of the period for each state. *n*=0, 195, 252 and 178, 213, 125 for closed perched and tidal states in Anglesea and Painkalac respectively. Anglesea estuary was not closed in the sampling period.

The high mean deposition rates during tidal periods may reflect allochthonous sediments associated with high flows at the beginning of most tidal periods and/or increased resuspension rates through the mechanisms listed above. The high mean rate for closed periods in Painkalac was influenced by very high deposition rates associated with a flood at the end of the April deployment period in which the estuary was mostly closed (Figure 6.15). When the data for this deployment were excluded, the mean rate for closed periods in Painkalac was similar to that for perched periods.

The higher mean deposition rate in Painkalac during tidal periods probably relates to the timing and duration of tidal states in Painkalac compared to Anglesea. Most of the time that Painkalac was tidal was in periods immediately after floods, while Anglesea was tidal for longer periods that were not so closely associated with large floods. One alternative explanation, that a greater proportion of fine particles in Painkalac led to greater resuspension and net deposition rates does not seem to be the case, given the similar deposition rates between the estuaries during perched states. Similarly, increased wind resuspension at times when Painkalac was tidal compared to times when Anglesea was tidal is not likely given similar wind velocities during these periods (Figure 4.13b,c).

6.3.4.d. Organic matter content: Deposited sediments

Of the 708 samples of settled sediment from the main sites, the mean organic matter content (as measured by loss on ignition) was 11.7% (±0.13%). The minimum, maximum and median values for sediments from individual tubes were 2.8, 21.4 and 11.2 % respectively.

Overall, mean percent organic matter content of settled sediments at Anglesea was greater than that at Painkalac consistent with the results from the longitudinal sites. Within Anglesea, Sites A1 and A2 had greater proportions of organic matter than Site A3 which was similar to all three Painkalac sites (Figure 6.17).

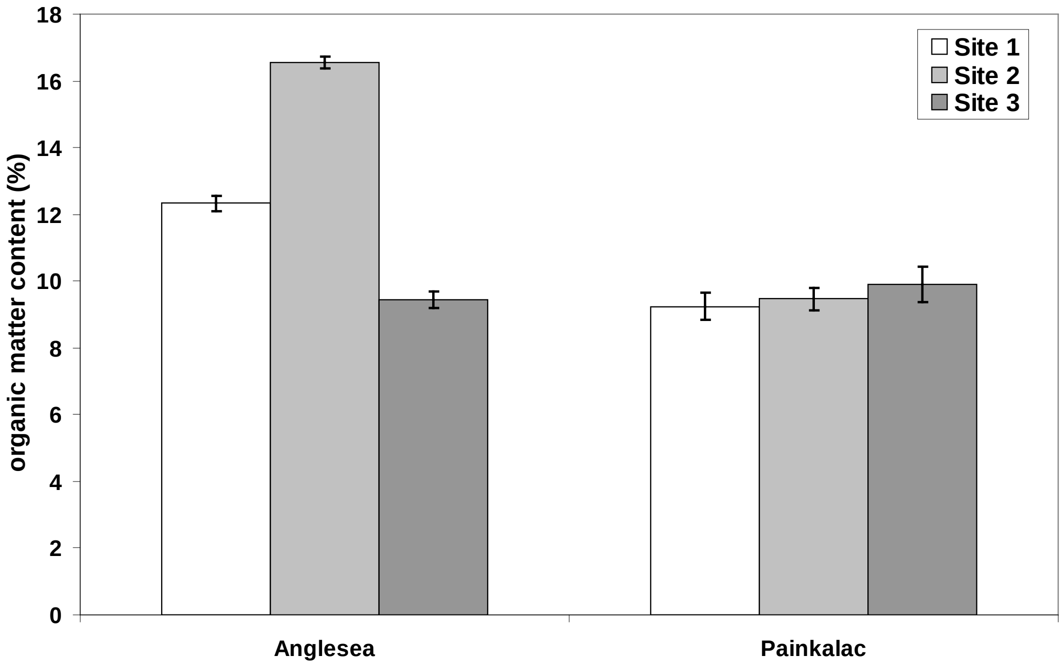


Figure 6.17. Mean (± s.e.) percent organic matter of settled sediments by site for all times (*n*=102, 99, 99 and 97, 102, 105 for Anglesea and Painkalac Sites A1, A2, A3 and P1, P2, P3 respectively).

The fully nested experimental design allowed patterns of difference in mean organic matter content at each of the three spatial scales to be examined in isolation from each other for each deployment time (Table 6.6). Most of the time there were no significant differences in the organic matter content of settling sediments between estuaries, with exceptions in the April and November 2001 deployment times. The typical pattern of spatial differences was at both site and block scale (8 of 11 deployment times).

Scale\Time	1/1[a]	3/1[a]	4/1[a,b]	5/1[a,b]	6/1	8/1	9/1[c]	10/1	11/1[c]	12/1[c]	1/2
Estuary	NS	NS	*	NS	NS	NS	NS	NS	*	NS	NS
Site	*	NS	***	**	***	***	***	***	*	***	**
Block	**	NS	NS	*	***	***	***	***	***	*	***

Table 6.6. Results of eleven separate nested ANOVA's of organic matter content of sediments collected during each deployment (shown as month/year in the top row). Shading indicates a significant result at *: α=0.05, **: α=0.01, *: α=0.001. 3 sites per estuary and four blocks per site (*n*=3) were used except for a - 2 blocks per site, b - 2 sites per estuary and c - 3 blocks per site.**

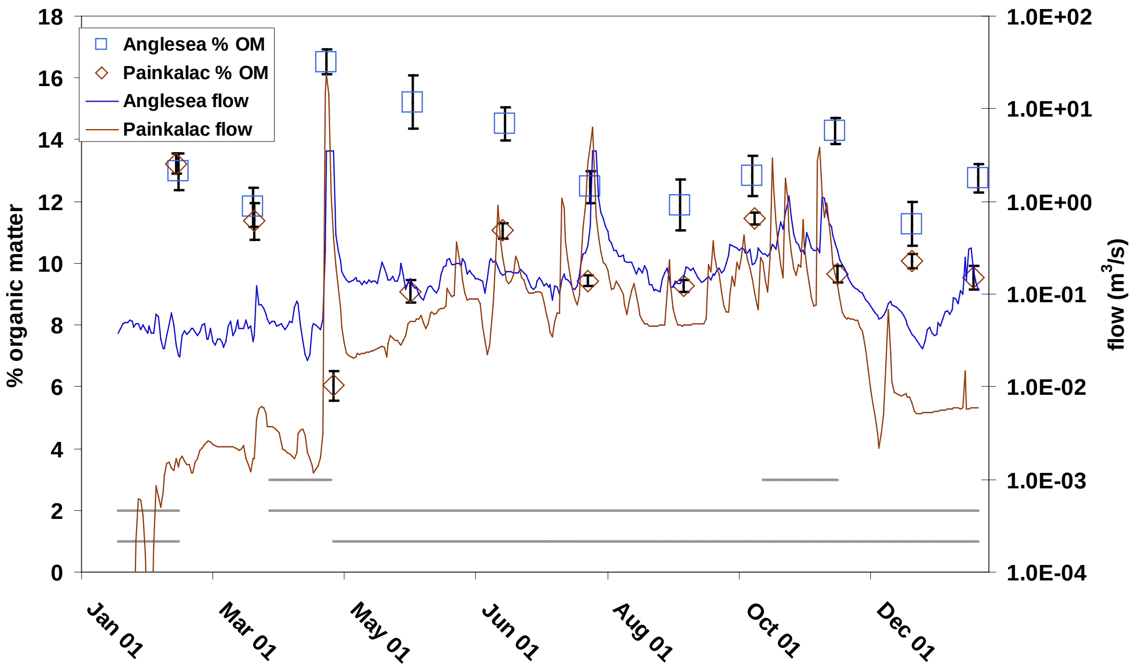


Figure 6.18. Mean (± s.e.) percent organic matter and freshwater flow in Anglesea and Painkalac estuaries, with periods in which there were significant differences on three spatial scales indicated by grey lines at bottom. Mean percent OM contents are shown on the end date of deployment periods. The upper grey line represents inter-estuarine differences; the middle, inter-site differences (within estuaries and the lower, inter-block (within site) differences.

At both times where a significant difference between estuaries was observed, the OM content was greater in Anglesea sediments (Figure 6.18). This was consistent with the greater overall organic matter concentrations in sediments at Anglesea but each occasion also occurred in association with high flows, which appeared to have differing effects on the OM content of depositional sediments. Overall, there was a significant positive relationship between flow and mean organic matter content in Anglesea (R^2=0.57, p=0.0070) but a significant negative relationship in Painkalac (R^2=0.57, p=0.0072). These opposing results were potentially due to erosion of clay soils in the Painkalac catchment and the mobilisation of organic rich sediments in the streambeds and/or stormwater system of the Anglesea catchment.

Differences between sites, for the 10 of 11 periods where this factor was significant, were consistent in Anglesea but not in Painkalac. In Anglesea, Site A2 always had the largest percentage of organic matter, followed by Site

A1 and then A3. This pattern matches the relative distances of each site away from major storm water drains. In particular, Site A2 was located near a drain that was observed to deposit large quantities of organic material into the estuary and of the upper 10% of values recorded in both estuaries (>16.75% organic matter), 85% were from Site A2. Conversely, of the lowest 10% of values (<7.7% organic matter), 75% were from sites closest to the mouths of the two estuaries (A3 and P1).

Significant between-block (within site) differences were observed on nine of eleven deployments. The only two deployments following which no differences were observed were also deployments in which only two blocks were included in the analyses. As for results from analyses of deposition rate, it may have been that variability at the within-site spatial scale was typical of these estuaries, but was not detected on these occasions due to lower replication at the block level and hence lower power in the statistical analyses.

Both Anglesea and Painkalac had slightly larger organic matter content in settled sediments during perched states compared to tidal (Figure 6.19) while the mean during the times when Painkalac was closed was higher than during the two other states. This may be due to the increased marine influence (and hence reduced organic matter content) associated with tidal and, to a lesser degree, perched states.

In terms of patterns of scales at which significant spatial differences were observed, heterogeneity at the block (metre) and site (tens of metres) scales was typical throughout the sampling period. The two periods in which there were significant differences on the between-estuary scale also coincided with simultaneous peaks in flow and changes of state. There were positive and negative correlations between flow and organic matter content in Anglesea and Painkalac estuaries respectively. It is likely that the significant differences between estuaries were related more to the largest inter-estuarine differences in organic matter content occurring at times of high flow than to states or transitions in state.

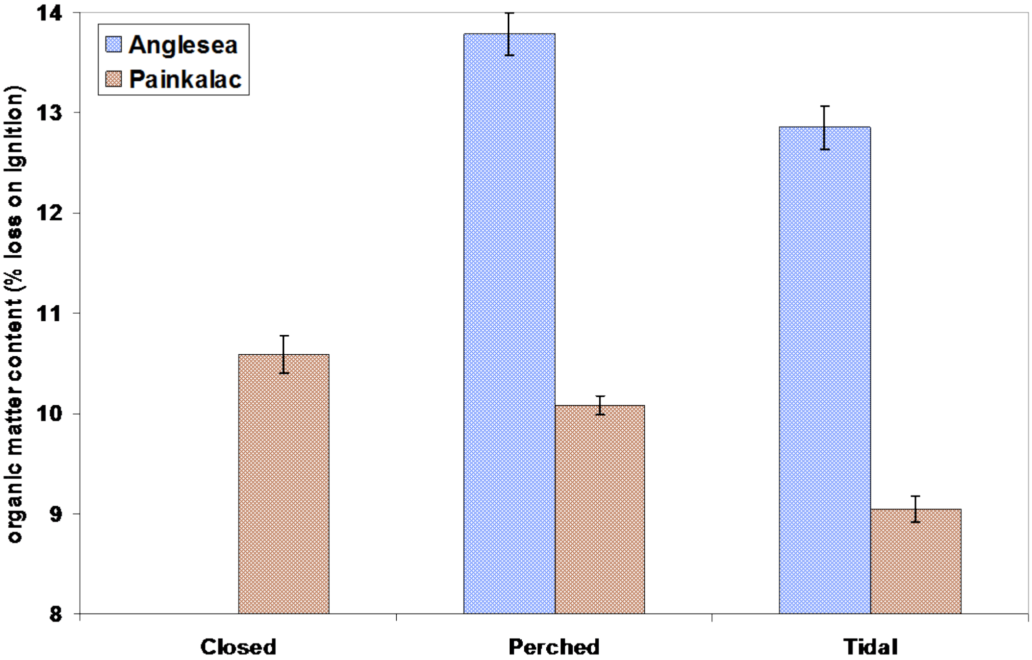


Figure 6.19. Mean (±s.e.) percent organic matter content in sediment collected during the three states for Anglesea and Painkalac estuaries. Where a deployment period encompassed more than one state, values were weighted according to the proportion of the period for each state. *n*=0, 195, 252 and 178, 213, 125 for closed perched and tidal states in Anglesea and Painkalac respectively. Anglesea estuary was not closed in the sampling period.

6.3.4.e. Rate and organic matter comparisons

Only 15 tubes collected sediment at a rate greater than 1kg/m^2/day, and except for one (from March 2001), these were all from Painkalac, following the major flood. A comparison of deposition rate and organic matter content indicates relatively low organic matter content at these highest deposition rates (Figure 6.20). These values (mainly from Sites P1 and P2 in April and May 2001; Figure 6.20b,d) indicate mineral particles (e.g. clay, sand) from either freshwater flows or locally resuspended marine sands as likely sources of sediment associated with the flood flows at that time. This second mechanism was the most likely cause of the low organic content of sediments collected at Site A3 in September (Figure 6.20a,c).

The maximum deposition rate recorded was almost certainly an underestimate as it is based on tubes that were nearly filled. Because of the increase in the ratio of mouth diameter to effective tube depth with filling and associated increases in turbulent mixing at the surface of the collected sediment (*sensu* Hargrave & Burns, 1979), it is likely that resuspension of sediment from within the tube occurred in the fuller tubes. Despite this, individual rates were extremely high (Table 6.7), comparable to those reported near hydraulic dredging and shipping operations (Adriano *et al.*, 2005).

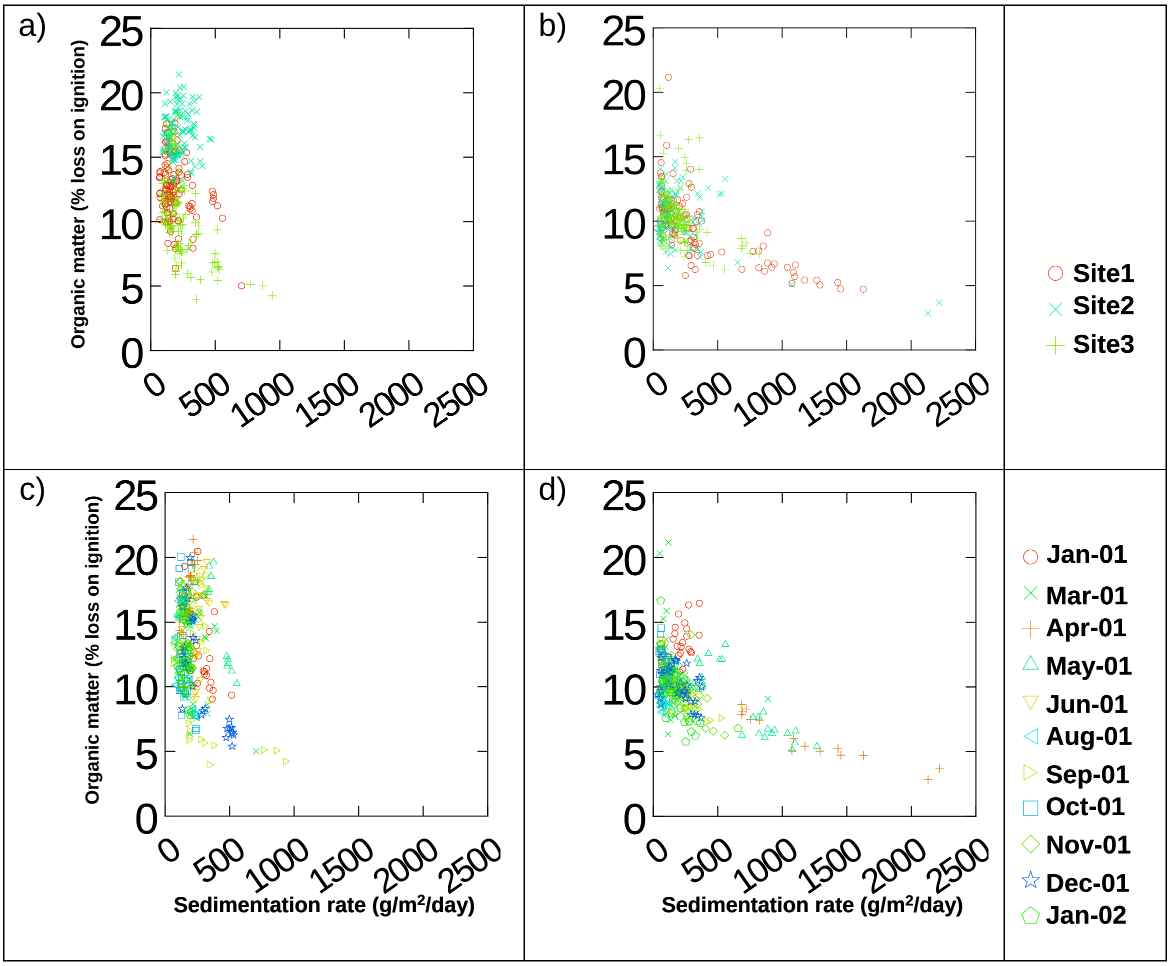


Figure 6.20. Deposition rate vs organic matter content for individual tubes by site within estuary (a: Anglesea, b: Painkalac), and by deployment for each estuary (c: Anglesea, d: Painkalac).

Organic matter content of settled material tended to be greater than that of *in situ* sediments at the longitudinal sites most of the time, as would be expected, given a greater proportion of fine material. Results were within the range of those found in studies elsewhere (Table 6.8).

Location	Type	Range (g/m²/day)	Method	Study
Anglesea, Vic	Intermittent estuary	68-940	Tube traps	this study
Painkalac, Vic	Intermittent estuary	29-2220	Tube traps	this study
Peel-Harvey, WA	Large estuary	0-~240	Funnel traps	(Gabrielson & Lukatelich, 1985)
Port Phillip Bay, Vic	Marine embayment	4.4(±0.8)[a]	Radionuclide abundances	(Hancock & Hunter, 1999)
Wellington Harbour N.Z.	Marine embayment	30.7[b]	Tube traps	(Nodder & Alexander, 1999)
Venice Lagoon	Coastal lagoon	113-2609[c]	Custom traps	(Adriano *et al.*, 2005)
W Mediterranean	Marine	5.0-35	Tube traps	(Gacia *et al.*, 1999)
Narragansett Bay NE USA	Large Estuary	20(±8.8)-51(±16.7)	Tube traps	(Oviatt & Nixon, 1975)

Table 6.7. Deposition rates from selected studies. Where standard errors are shown, means are given. [a] calculated from mean residence time in water column, [b] mean, se not available [c] site means.

Location	Type	Range (%)	Method	Study
Anglesea, Victoria	Intermittent estuary	10.3(±0.3)-16.9(±0.2)	LOI	this study
"	"	3.4(±0.2)-14.0(±0.5)	"	(*in situ*)
Painkalac, Victoria	Intermittent estuary	9.6(±0.2)-10.4(±0.2)	LOI	this study
"	"	2.6-8.2	"	(*in situ*)
Peel-Harvey estuary, WA	Large estuary	16.6(±1.1)-30.2(±0.5)	LOI	(Gabrielson & Lukatelich, 1985)
Narragansett Bay NE USA	Estuary	10(±1)-14(±1)	LOI	(Oviatt & Nixon, 1975)

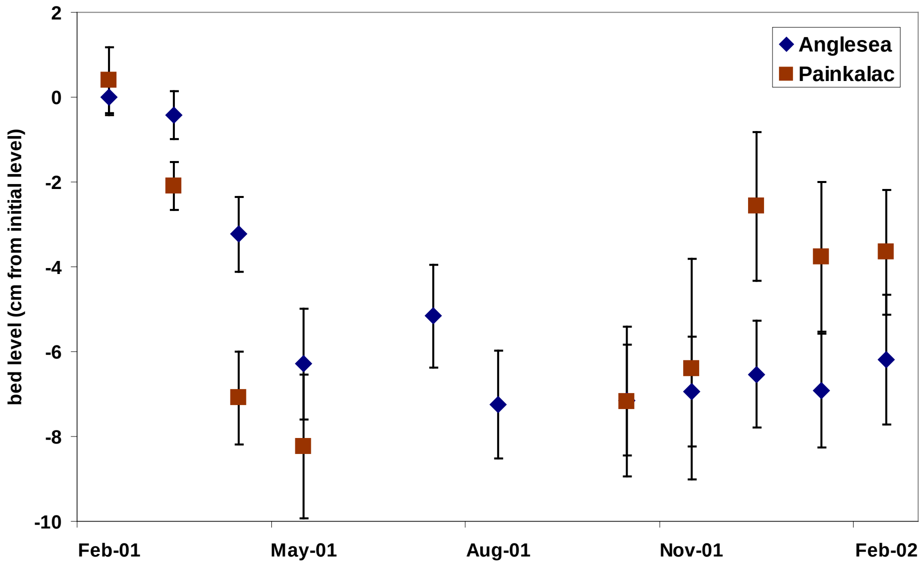
Table 6.8. Organic matter content of depositional material from selected sources. Where standard errors are shown, site means are given.

6.3.4.f. Net erosion/accretion

On average, there was net erosion at all sites in the lower part of both estuaries between 18/1/2001 and 14/2/2002. The majority of this erosion occurred in the first four months of measurement, particularly in the two sampling periods from late March to late May 2001 (Figure 6.21). Both of those sampling periods encompassed high flows from the largest flood in the entire study period, followed by the largest tides experienced in the estuaries.

Differences in erosion/accretion between sites became pronounced with time in both estuaries (Figure 6.22). Net erosion at Anglesea was greatest at Site A2, with a mean reduction in bed level of 11.6cm by the final measurement while the least erosion over this time was at Site A1, with a mean of 2.7cm (Figure 6.22a). By the end of the sampling period in Painkalac, Site P2 showed a slight net accretion (0.8cm) while Sites P1 and P3 had similar degrees of erosion (5.6 and 6.1 cm).

All Anglesea sites appeared to be eroded to differing degrees by the April flood (Figure 6.22a). Site A2 then experienced some accretion but was re-eroded by the August flood as were Sites A1 and A3 to a lesser degree. Sites A1 and A3 then accumulated sediment towards the end of the study period. These differences may have been due to slightly different elevations (and hence inundation during tidal periods), more easily disturbed sediments at Site 2, differing proximities to storm-water drains, or a combination of



these.

Figure 6.21. Mean (± s.e.) bed levels in Anglesea and Painkalac estuaries as measured in lower estuarine sites relative to the bed levels measured on 18/1/2001. Missing Painkalac data for July and August were due to zero visibility conditions.

There was substantial variation between Painkalac sites over time, but no patterns as consistent as those between sites in Anglesea (Figure 6.22b). All sites experienced considerable erosion with the April flood but later in the period appeared to experience large and inconsistent changes in bed level.

There was no consistent relationship between mean flow and erosion/accretion rate over the ~monthly time-scale in either estuary (Figure 6.23). In Anglesea, the fastest rates of erosion (>1cm/day) only occurred in the periods with the highest flows but high flows were not always associated with high rates of erosion (Figure 6.23a).

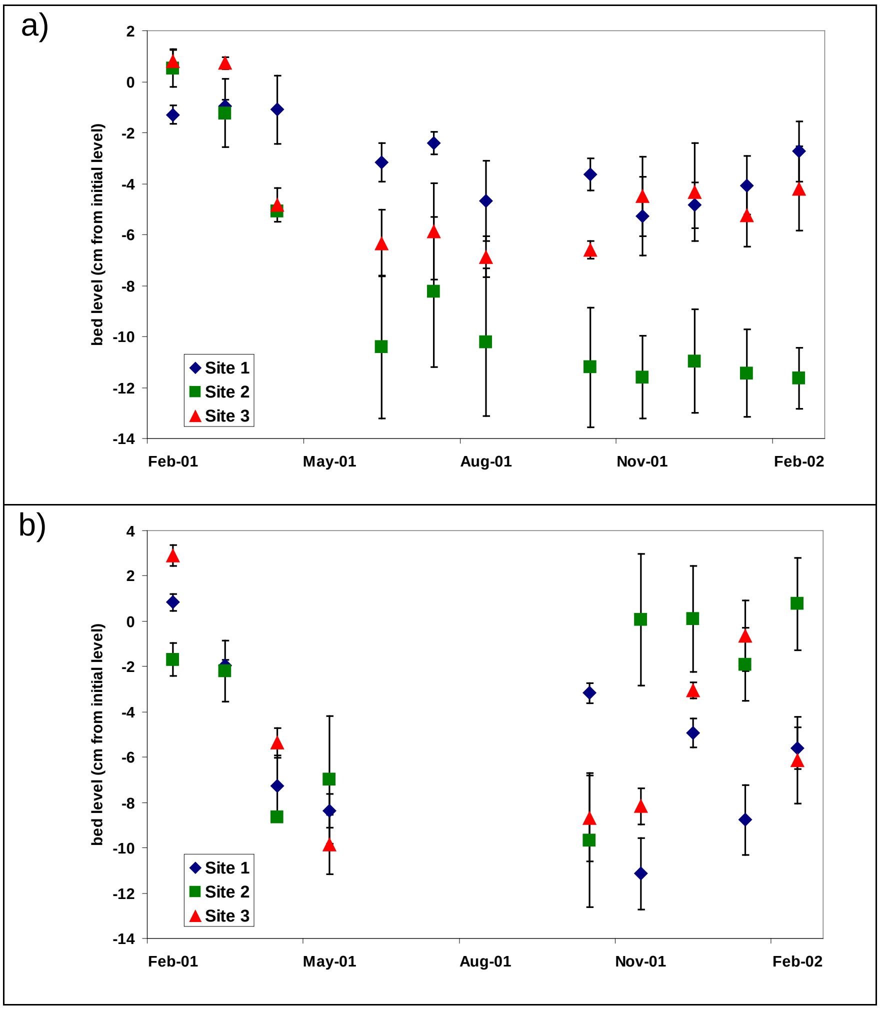


Figure 6.22. Mean (± s.e.) bed levels at each site in a) Anglesea and b) Painkalac estuaries relative to the bed levels measured on 18/1/2001. Missing Painkalac data for July and August were due to zero visibility conditions

In contrast to Anglesea, the greatest rates of erosion in Painkalac occurred over a range of flows, including the highest flows (Figure 6.23b). These results suggest that processes in addition to freshwater flow were altering bed levels on a local scale in Painkalac.

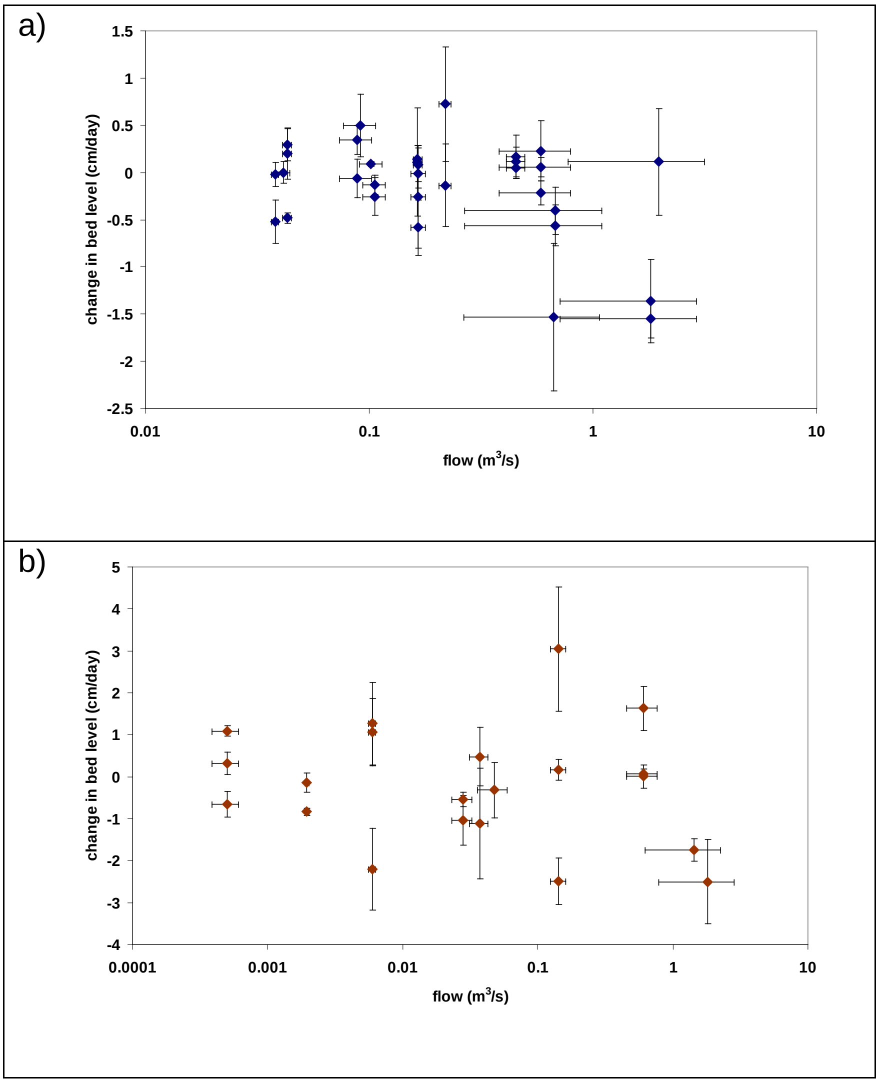


Figure 6.23. Mean (± s.e.) flow vs mean daily change in bed level (± s.e.) by time and site for a) Anglesea and b) Painkalac. Means for standard length deployments only were used. Note the log scale for flow and the greater range on the x and y-axes for b).

No clear relationship existed between erosion/accretion and deposition rate in either estuary (Figure 6.24). Two features of these comparisons are noteworthy. First, the four extremely large depositions (all >600g(d.w.)/m^2/day and in Painkalac) occurred during periods of moderate to large erosion of sites. This suggests that the amount of resuspension of sediments is not closely related to erosion or accretion on this temporal scale, but that when very high deposition rates occurred, a large component of depositional material was likely to have been resuspended autochthonous sediments. Second, on the relatively few occasions where there was accretion, deposition rates were moderate.

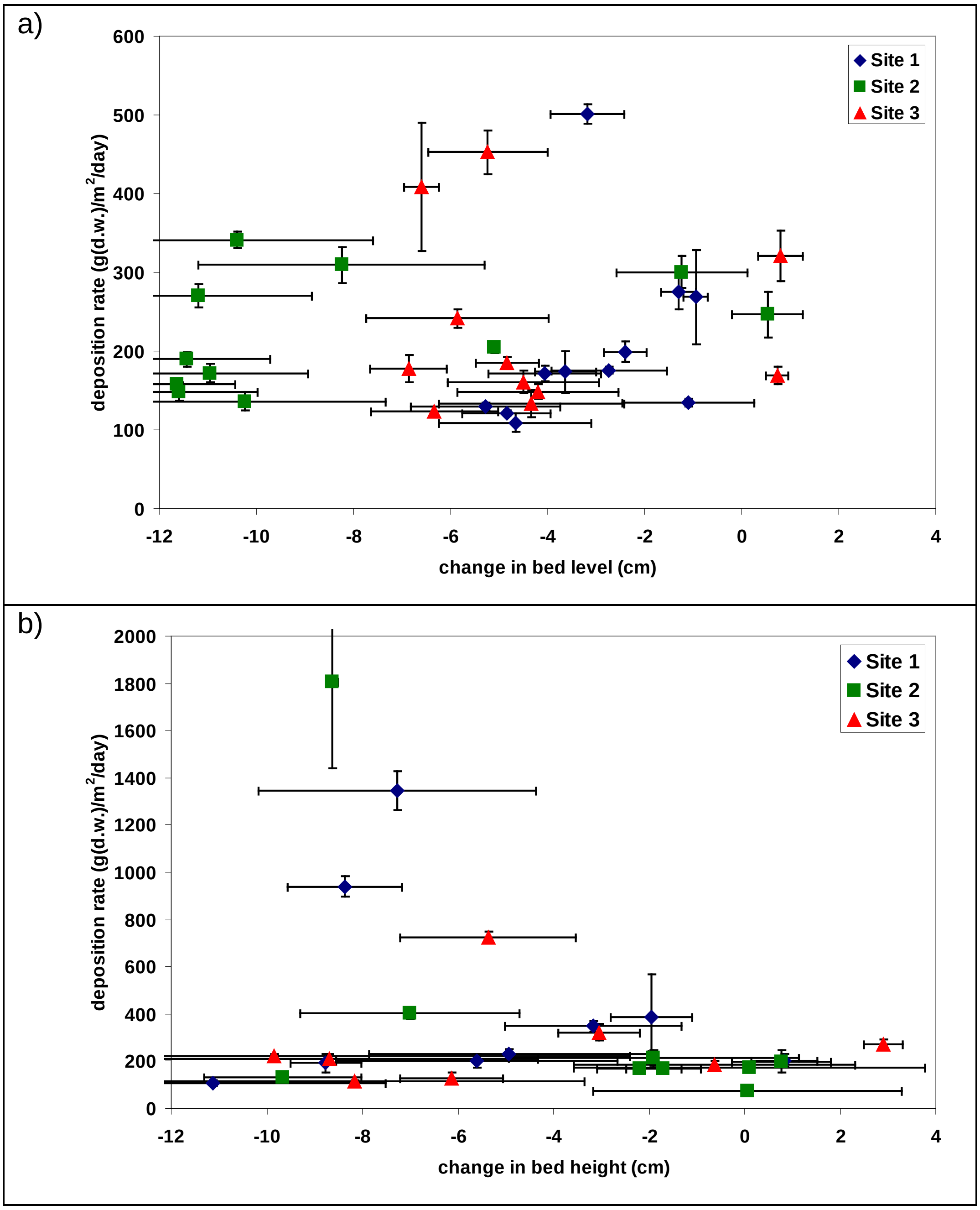


Figure 6.24. Change in bed height versus deposition rate (means ± s.e.) for sites in a) Anglesea and b) Painkalac by deployment periods.

6.4. Summary and conclusions

Historically, the area of seagrass in Anglesea estuary appears to have fluctuated dramatically, with seagrass areas derived from photos ranging from 5200m^2 to 23,700m^2. An inverse relationship with rainfall for several years prior was apparent. This may have been causal, with seagrass extent showing decline in floods and re-establishment in drier, more stable conditions. Alterations in the hydrologic regime of the estuary potentially provided a large and consistently available area of potential habitat through these drier years. Other major events in the estuary during the period of photos, notably the Ash Wednesday bushfires of 1983, the subsequent construction of Coogoorah Park, and the flooding and pumping of turbid waters from the open-cut mine in 1995, may also have influenced this pattern of coverage. These events may also have been related to the distribution of seagrasses in Anglesea being limited to the lower estuary, in contrast to the fringing seagrasses present in the middle and upper parts of the estuary in both Painkalac (during the first half of this study), and historically in Anglesea (in 1981: Atkins & Bourne, 1983).

Within the study period, despite seasonal variability, the effects of a major flood and associated transitions of the estuaries to tidal states were clearly evident in the recession of seagrass beds and a change in species composition. Smaller floods were also associated with a measured decrease in seagrass extent in Anglesea and the apparent disruption (no peak in Anglesea) or interruption (reduced peak in Painkalac) of recolonisation and/or seasonal increases in the spring-summer following the major flood of April 2001. Reductions in shoot densities were also associated with floods in both estuaries.

A substantial departure from the cyclic, seasonal patterns of growth observed in *H. tasmanica* and *Z. muelleri* in Western Port and Port Phillip Bays (Bulthuis & Woelkerling, 1983b; Kerr & Strother, 1989; 1990) was evident in Anglesea and Painkalac estuaries, in association with variability in freshwater flow and estuarine hydrology. Apparent causes of these changes were low salinity (with positive effects on *Ruppia* and negative effects on *Zostera*) and estuarine state, which was closely linked, via inundation regimes, to available areas of subtidal and intertidal habitat (Table 6.9, Table 6.10). While measured regrowth did not return beds to the condition and extent observed at the beginning of the study, the results of these early qualitative observations and the aerial photo analysis suggest that extended periods of low flow with relatively-high water levels can allow periodic growth of extensive and dense seagrasses throughout the lower parts of Anglesea and Painkalac estuaries.

State	Tidal	Perched	Closed
Salinity	↑	↑	↕
Water level	↓	-	↕
Seagrass coverage	low	med/high	variable
Genus favoured	*Zostera*	-	variable

Table 6.9. Links between state and seagrass extent and composition. Variable responses during closed periods were related to flow and salt-water overtopping of the bars (see Table 6.10). Arrows indicate increases and/or decreases in salinity and water level associated with each state.

Hydrologic influences	+ flow		- flow	
	- overtopping	+ overtopping	+ overtopping	-overtopping
Salinity	fresh	brackish	marine	hypersaline
Water level	high	high	variable	low
Seagrass extent	↑	↑	↑	↓
Genus favoured	*Ruppia*	neither	*Zostera*	?

Table 6.10. Conceptual model of relationships between hydrologic influences (*i.e.* freshwater flow and seawater overtopping of sandbars), salinity and water level during closed periods with implications for seagrass extent and composition. Arrows indicate increases and decreases in seagrass area associated with each combination of flow and overtopping. '+' indicates presence of a given factor, '-' indicates absence. '?' indicates uncertainty.

In terms of the relative degree of influence on seagrasses, climatic factors appeared to be more important that anthropogenic modifications to flow. The largest losses of seagrasses were associated with a large flood, with a discharge that was orders of magnitude greater than any anthropogenic changes. Seasonal changes in extent were also evident, although these were probably influenced by alterations to flow regimes, via the prolonged tidal period in Anglesea estuary when compared to Painkalac in the summer of 2001/02.

Similarly, the establishment of substantial beds occurred in association with three-to-four year periods of low rainfall. It is in the context of these naturally-dry situations that anthropogenic changes to flows are likely to have the most influence, as outlined in Table 6.10. Potential mechanisms of influence include additional flows reducing salinity in an estuary and favouring polyhaline species or conversely, increased periods of no flow leading to lowered water levels through evaporation, with hypersalinity and exposure of potential habitat resulting.

7. Detrital Processes

7.1. Introduction

Breakdown of dead material from seagrasses via detrital processes constitutes an important trophic pathway for seagrass meadows, particularly because the rate of the alternative process of herbivory in these systems is thought to be relatively low, especially in temperate regions (Mateo *et al.*, 2006; but see Valentine & Duffy, 2006). There are many inherent and environmental factors that influence the decomposition rate of seagrass detritus, including:

- the volume, physical location, stage of decay and composition of detrital material;
- physico-chemical properties of sediment and water (e.g. temperature, salinity, pH, redox, oxygenation and nutrient concentrations, especially nitrogen); and
- the composition and relative abundances of microbial, meiofaunal and macrofaunal invertebrate communities (reviews by Klug, 1980; Harrison, 1989; Klumpp *et al.*, 1989; Enriquez *et al.*, 1993; Mateo *et al.*, 2006).

Many of these factors are likely to be influenced by changes associated with freshwater inflow and may result in variability in decomposition rates across different spatial and temporal scales.

This component of the research measured *in situ* decomposition rates in Anglesea and Painkalac estuaries over time, from January 2001 to February 2002. It also measured decomposition potential of sediments using a technique that was new in estuarine research. Aims of the investigation were to:

- quantify *in situ* decomposition rates of seagrass detritus, as a measure of transfer of material between the seagrass and microbial components of the estuarine ecosystem, and examine differences in rates through time and between estuaries and sites;
- assess decomposition potential of the estuarine sediments and their associated microbial communities between different times, water

depths, estuaries and among locations at two smaller spatial scales within the estuaries by measuring cellulose decomposition rates with the cotton-strip assay method;

- compare decomposition potential of sediments between seagrass beds and other areas with accumulated seagrass macro-detritus; and
- relate changes in these measures of decomposition to hydrological and other flow-related changes.

Two complementary measures of detrital processes were used to address these aims: the first was the rate of *in situ* decomposition of freshly-collected, locally-sourced seagrass detritus in litterbags; and the second was the rate of decomposition of standard cellulose test strips. The seagrass decomposition component was intended as a relative measure of the actual rates of matter transfer from seagrass detritus in any given place or time while the cotton strip assay (CSA) technique measured potential decomposition rate using a standard material across times and places. The low per-unit time cost of CSA testing also meant that additional spatial factors, which could not be included in the litterbag component, could easily be included to examine variability in decomposition potential between depths and at a scale of metres.

Measurement of leaf decomposition in litterbags is a long-established technique in both terrestrial and aquatic ecology, and its uses and limitations have been discussed in detail (e.g. Boulton & Boon, 1991). The CSA originated as an assay of cellulose decomposition potential in soils (Latter & Howson, 1977) and, although primarily used in terrestrial settings, its use in aquatic environments includes the hyporheic zone of running fresh waters (Boulton & Quinn, 2000), bogs and marshes (reviewed in Maltby, 1988), a dyked, partially-reclaimed coastal wetland (Mendelssohn *et al.*, 1999) and, in one known instance, estuaries (A. Boulton, unpublished data). The assay uses the loss of tensile strength in strips of standardised cotton fabric as an indicator of microbial degradation of cellulose in the surrounding media and was developed for ecological and textile science soil-burial tests in the late 1970s (Latter & Walton, 1988b, 1988a; Sagar, 1988).

7.2. Methods

7.2.1. Seagrass decomposition

Seagrass decomposition rates at lower estuarine sites in Anglesea and Painkalac estuaries were measured by mass loss of leaves and detritus in litter bags over 11 continuous deployment periods between January 2001 and February 2002 (Table I.1, Appendix I).

Prior to each deployment, detritus and living leaves from the vicinity of each site were collected by hand and kept on ice. Detritus or, when no detritus was present, the oldest attached leaves available (which would be the next to be lost) were sorted from this material and cleaned of epiphytes and animals to make up four replicates, each of ~1 gram wet weight, per site. On two occasions, it was not possible to locate enough material from Painkalac estuary to use in litterbags due to a combination of poor visibility and a large reduction in seagrass extent (described in Chapters 5 and 6). Thus, for deployments 7 and 8, in September and October 2001, material from Anglesea was substituted. On two other occasions, a lack of material near individual sites required that seagrass detritus from other sites within an estuary was used. For deployment 7, some seagrass from Site A3 was deployed at Site A2 and for deployment 11, material from Site P3 was deployed at Site P2. On deployment 5, bags containing material from Sites P1 and P3 were accidentally swapped. Times and sites where non-local material was used are detailed in Table I.1, Appendix I.

Preweighed detritus was transferred to plastic mesh bags (mesh size 1mm) attached to bricks by short (~15cm) tethers. These bricks were then placed in random locations within each lower estuarine site in Anglesea and Painkalac (Figure 6.3), such that the bags lay on the surface of the sediment. On retrieval, mesh bags were immediately placed in plastic bags with seawater and put on ice. Large epiphytes and any animals associated with the detritus and bag were removed prior to reweighing the samples. Mass for each replicate was determined as wet weight of material that had been patted dry between two sheets of paper towelling and weighed to the nearest

10mg. While use of dry weight may have been useful in terms of reducing variability, restrictions in the amount of material available meant that this was not possible in this experiment. Deployment times averaged 33 days and ranged from 25 to 41 days (Table I.1, Appendix I). While it is recognised that decay rates of seagrass detritus are not linear (Klumpp & Van der Valk, 1984; Mateo *et al.*, 2006), neither are they necessarily exponential, but are complex and dependent on environmental conditions and the composition of material (Boulton & Boon, 1991; Moore & Fairweather, 2006). For this reason, given the relatively short times of deployment (Harrison, 1989), lack of exponential changes in rate on these time-scales in Westernport Bay (Klumpp & Van der Valk, 1984), and the use of material at varying stages of decomposition across locations and times, decomposition rate was recorded as % mass loss per day, based on a linear decomposition rate.

For the first three deployment periods, three bags were randomly placed within each site. From the fourth deployment onwards, four bags were used, in an attempt to provide some more redundancy for losses, breakages and vandalism during deployments. Overall, there was a mean recovery rate of 83%, with means for individual deployments ranging from 72% to 96%, except for a 56% recovery rate for deployment 3, during which there was a large flood. Details of successfully recovered and intact replicates are given in Table I.2, Appendix I.

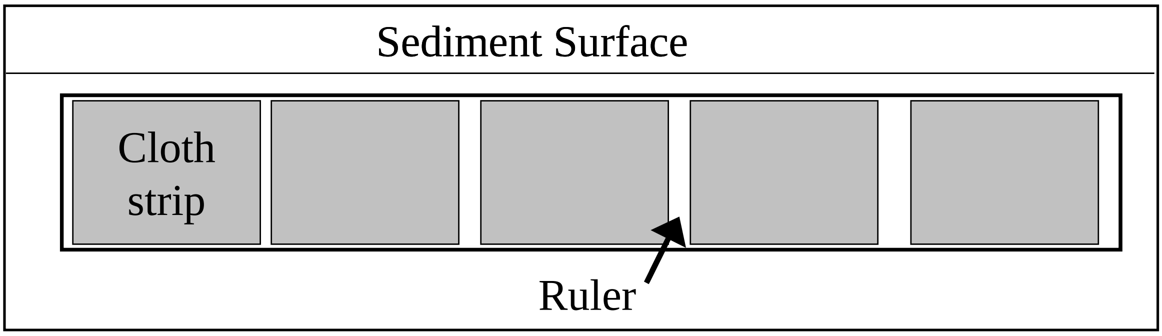
Due to variable replication and a lack of independence between consecutive deployments, results from each time were analysed separately, using a nested design similar to that used for sediment deposition rates (Section 6.2.2.d). This allowed comparisons between estuaries and sites within estuaries for each time. Incomplete recovery and human interference with some litter bags meant that comparison using only ANOVA was not possible for deployment 3, while deployments 1, 2 and 4 were analysed using only two sites per estuary. Valid replicates within each site varied with time and were randomly removed as required to balance the ANOVA designs (Table I.2, Appendix I).

7.2.2. Cotton strip assay

The methodology of the sample testing and field deployments broadly followed that of Howson (1988) and Boulton and Quinn (2000). Strips of standard Shirley Soil Burial Test Fabric (Shirley Dyeing and Finishing Ltd, Hyde, Cheshire) were cut to a size of 5cm (warp) by 4 cm (weft). Groups of five strips, chosen so that no two strips comprised the same threads in the warp or weft from the original piece of cloth, were wrapped in aluminium foil and autoclaved at 121ºC for 15 min. On site, haphazardly-allocated groups of strips were attached lengthwise along the flat sides of 30cm rulers that had been soaked in 70% ethanol, with rubber bands holding the ends of each strip. A small gap (~1cm) separated each strip to avoid spread of bacterial or fungal colonies between strips. Aluminium tags with identification codes were attached to a hole at the end of each ruler with a plastic cable tie.

Strips were deployed for 21 days at each of two times: first from 13/10/2001 to 3/11/2001 at Anglesea, 14/10/2001 to 4/11/2001 at Painkalac, and then from 25/1/2002 to 15/2/2002 in both estuaries, coinciding with deployments 8 and 11 of the seagrass litterbags, respectively. At each of the lower estuarine sites in Anglesea and Painkalac estuaries, five rulers were placed randomly within each of the shallow and deep halves of the site. Rulers were placed edgeways into the sediment with the long edges parallel to the surface, so that the strips were in contact with sediments on one side, across the depth range of ~1 to 5 cm (Figure 7.1).

Figure 7.1. Diagram of deployed 30cm ruler with strips attached.



To explore the influence of buried detritus on cellulose decomposition potential, two haphazardly chosen sites outside seagrass beds, but with substantial amounts of large, buried and slightly emergent seagrass detritus were sampled in the Anglesea estuary (OM1 and OM2). Both these sites were 10m long by 3m wide, running parallel to the main axis of the estuary and at an equivalent elevation to the shallow halves of Sites A1, A2 and A3. Five rulers were placed in each of these sites on the first deployment, with two procedural controls placed at random locations and processed as for controls from the main sites.

At the start of deployment at each estuary, a group of five strips (air controls) was exposed to the air and placed in a plastic bag on ice as a control. At each site, rulers with 5 control strips were placed in the shallow and deep halves of the site and left for ~20 minutes to control for any handling effects or site differences during deployment (especially in abrasion Harrison *et al.*, 1988; Boulton & Quinn, 2000). These procedural controls were then retrieved, rinsed in estuarine water and placed on ice in plastic bags. At the end of each day when strips were retrieved, they were gently washed in tap water and air dried. On return to the lab, both air and abrasion control samples from the first deployment were placed in a desiccator pending tensile testing (13-14/10/2001 – 14/11/2001). Control samples from the second deployment were not kept in a desiccator, but stored in bags immediately following air-drying until tensile testing (from 25/1/2002 to 19-20/2/2002).

A spare ruler was placed against strips being deployed to minimise abrasion while they were gently placed in the sediment. A similar procedure was used on removal. Strips were removed from the ruler in the water and gently rinsed before being placed in sealed plastic bags on ice. At the end of each day all strips were gently rinsed in cold tap water and air dried. Once dry, the edges of the strips were frayed so that all strips were 100 threads wide (~3cm x 5cm long) allowing an equal section of cloth to be tested and reducing the likelihood of tears from the edge of the material (Standards Association of Australia, 1988).

Within two days of each retrieval, test strips were dried in a fan-forced cabinet at 50^{o}C for 4 hours before conditioning for 24 hours at 20^{o}C and 65%

humidity (Standards Australia, 1995). After conditioning, the tensile strength of the strips was tested using a tensometer (Lloyd LR30K Material Testing Machine, speed ±0.5%, force ± 0.5 kg or 1%). Separation speed of the jaws was set at 100mm/minute and maximum load before breakage was recorded as kgf (1kgf=9.807N). The length of strip between the jaws was set at 3 cm. Underestimation of tensile strength, caused by tearing of material where it is held in the jaws of the tensometer, has been identified as a problem with this method (Howson, 1988); such tearing was minimised by covering the steel jaws with masking tape and experimenting with spare strips to find a suitable jaw tension.

Tensile strength of cotton strips has a non-linear relationship with rotting rate, which has been approximated by a cubic function (Hill *et al.*, 1988). For this study, decomposition rates were calculated using the equation recommended by Correll *et al*. (1997):

$$R = \frac{1}{t}\left(\frac{y_o}{y} - 1\right)^{\frac{1}{3}}$$

where R is decomposition rate, t is time elapsed, y_o is initial tensile strength (represented here by the mean of procedural controls for a site – see Appendix I) and y is tensile strength of the sample. This relationship has been shown to be stable over a wide range of proportional tensile strengths with an optimal remaining tensile strength of 2/3 that of the original material (Correll *et al.*, 1997).

On testing, it was found that reductions in tensile strength were considerable and unpredictable for the controls from the second deployment compared to those from the first deployment and also from other estuarine results (A. Boulton, pers. comm.). It is likely that this was due to residual moisture in the strips allowing some decomposition in storage during the interval between the start of the deployment and testing of tensile strength. As these values were considered unreliable, the initial tensile strength (y_o) for samples from the second deployment was derived from the respective site means of procedural controls from the first deployment time (see Appendix I for details).

The main experiment using CSA was analysed using a five-factor, mixed-model ANOVA that simultaneously tested for differences in mean decomposition rate between times, estuaries, depths, sites (nested within estuaries) and rulers (nested within depths and sites (within estuaries)) (Table 7.1) as well as between combinations of these factors (see Section 7.3.2). Following both deployments, some rulers were not recovered, and during testing, some strips tore along the jaws of the tensometer (resulting in underestimations of tensile strength; Table 7.2). Because of this, replication of rulers for each depth/site combination was randomly reduced from 5 to 3, and the number of replicates for each ruler was reduced from 5 to 4 for the ANOVA, to allow a balanced design, without which analysis using this design would have been less robust and 'almost intractable' in terms of calculating appropriate *F* ratios (Quinn & Keough, 2002). Pseudo- or quasi- *F* ratios were used to assess H_o for several of the terms (shown in Section 7.3.2, Table 7.6), for which no appropriate denominator terms for actual *F* ratios existed. While these measures are only an approximation of *F*, *p* values in this case were either clearly significant or not at the α=0.05 level, increasing confidence in the results. A subsequent analysis, pooling non-significant terms with p>0.5, did not change the results (see Table I.7, Appendix I).

Factor	**Time**	**Estuary**	**Depth**	**Site(E)**	**Ruler(DxS(E))**
Random/Fixed	R	F	F	R	R
Levels	2	2	2	3	3

Table 7.1. Details of factors in five-factor ANOVA comparing mean decomposition rates. *n*=4, as described above, Site and Ruler factors are nested. Details of interaction terms are shown in Table 7.6.

	Time 1 no.	Time 1 %	Time 2 no.	Time 2 %	Total %
strips on lost rulers	15	5.0	30	10.0	7.5
jaw breaks	17	5.7	5	1.7	3.7
valid measurements	268	89.3	265	88.3	88.8
used in ANOVA	144	48.0	144	48.0	48.0

Table 7.2. Retrieval rates, numbers of valid samples and proportions of samples used in ANOVA. Percentages are of the total number of strips deployed.

The sites with high amounts of buried detritus were compared with shallow halves of sites A1 and A3 in a balanced, mixed-model design, in which there were three factors (Table 7.3). Although all rulers from these depths and sites were retrieved, some tearing of strips at the jaws of the tensometer meant that *n* was reduced from 5 to 4 for some rulers and so, to balance the ANOVA design, one replicate was discarded at random from each ruler with 5 valid replicate strips.

Factor	**Detritus**	**Site**	**Ruler (Site)**
Random/Fixed	F	R	R
Levels	2	2	5

Table 7.3 Details of factors in ANOVA comparing mean decomposition rates at sites with and without buried seagrass detritus. *n*=4

Where interaction terms in ANOVAs were highly non-significant ($p>0.5$), they were pooled with the residual to increase the power of the *F*-tests for remaining terms and reduce the probability of a Type II error (Quinn & Keough, 2002).

As the CSA is known to be temperature sensitive (Ineson *et al.*, 1988), temperature loggers (Tinytalk II, Gemini Data Loggers (UK) Ltd) were deployed in the centre of each of the shallow and deep halves of Sites A1 and P1 and buried at a similar depth to that of the rulers. Temperature was recorded every 10 minutes for part of deployment one (from 26/10/2001 onwards) and every 20 minutes for all of deployment two.

7.3. Results

7.3.1. Seagrass decomposition

Decomposition rates measured by individual litterbags ranged from –1.8 to 3.9 % of wet weight/day. Overall mean rates (± s.e.) were 1.1% (±0.07) and 1.5% (±0.11) for the 110 and 97 samples from Anglesea and Painkalac estuaries, respectively. No clear seasonal signal was evident in decomposition rates of either estuary (Figure 7.2). Peaks in Autumn and late-Spring/Summer were more apparent in Painkalac than Anglesea, although decomposition rates in January 2002 were greater than those in January 2001. Most of the time decomposition rates were similar between sites in each estuary (Figure 7.3), more so in Painkalac than Anglesea.

When decomposition rates were compared between estuaries and among sites nested within estuaries for each deployment, significant differences between means were observed on only two and three of the ten testable deployments, respectively. When differences were evident however, they tended to be highly significant (Table 7.4). Due to losses of replicates, no analysis of Deployment 3 was possible, although mean rates for the two estuaries were very similar at this time (Figure 7.2), with a large difference between sites in Painkalac, while samples were only retrieved from one site in Anglesea (Figure 7.3). Of the remaining ten deployments, half showed no significant differences for either factor. There were no deployment periods where decomposition rates varied significantly between both estuaries and sites.

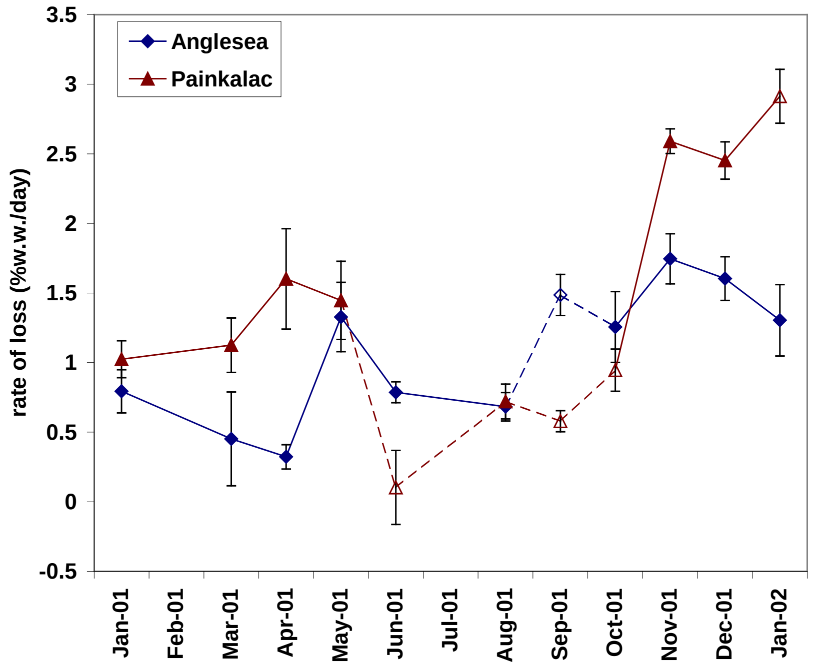


Figure 7.2. Mean (±s.e.) decomposition rate of seagrass material in litter bags in Anglesea and Painkalac estuaries. Times where material was not sourced from the vicinity of some or all sites measured are shown as open symbols associated with dashed lines. Means are located at middle dates of each deployment.

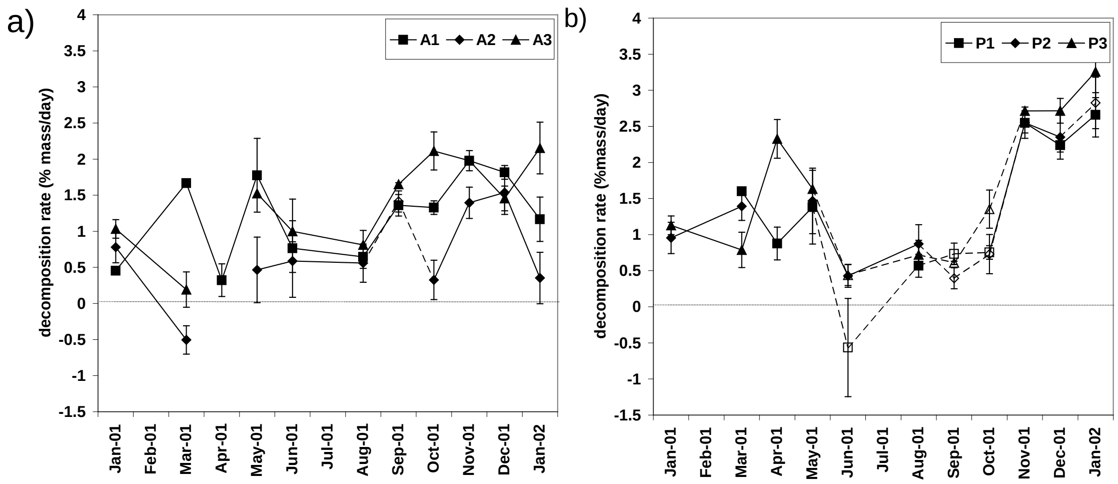


Figure 7.3. Mean decomposition rates (± s.e.) of seagrasses in litterbags deployed at lower estuarine sites in a) Anglesea and b) Painkalac estuaries between January 2001 and February 2002. Means are located at middle dates of each deployment. Times where material was not sourced from the vicinity of a site are shown as open symbols associated with dashed lines.

7.3.1.a. Differences between estuaries

On the two occasions where between-estuary differences were significant, the location of greater mean decomposition rates changed from Anglesea to Painkalac in association with an increase in decomposition rates in Painkalac through Spring 2001 (Figure 7.4). The first of these differences, however, reflects a difference in rate for similar materials, rather than that of *in situ* material as, for this period, material deployed in Painkalac was sourced from the area of Site A3 due to a lack of detrital material and difficulties in collection in near-zero visibility at that time (Table I.1, Appendix I). Large, non-significant differences in mean decomposition rates between estuaries were observed during three other deployments. In the cases of deployments 5 and 9 the power of the tests was reduced relative to other times due to missing litterbags (n=2). During deployment 11 the largest observed between-estuary difference was indistinguishable from a chance result due to relatively large variation at the site scale.

Deployment Month	$1^{2,2}$ 1/01	$2^{2,2}$ 3/01	3 4/01	$4^{2,4}$ 5/01	$5^{3,2}$ 6/01	$6^{3,2}$ 8/01	$7^{3,3}$ 9/01	$8^{3,3}$ 10/01	$9^{3,2}$ 11/01	$10^{3,3}$ 12/01	$11^{3,3}$ 1/02
Estuary	0.319	0.818	-	0.229	0.080	0.975	0.009	0.571	0.102	0.004	0.107
Site(Estuary)	0.912	0.009	-	0.880	0.671	0.448	0.604	0.001	0.366	0.874	8×10^{-5}

Table 7.4. Results (*p*-values) of nested ANOVAs of decomposition rates for each deployment. Shading indicates a significant result at α=0.05. Number of sites per estuary and replicate bags per site are shown in superscript after the deployment dates. Data from the last deployment were ln-transformed to achieve homoscedasticity. ANOVA tables are given in Table I.3 in Appendix I.

7.3.1.b. Site-scale differences

Significant variability at the site (10s of metres) scale was observed on three deployments, spanning almost the entire range of decomposition rates (Figure 7.4). On all these occasions, there were larger differences between sites in Anglesea, which was perched, than in Painkalac, which was either perched or tidal (Figure 7.3, Figure 7.4). On the latter two of these deployments, material had been transferred between sites, which could be expected to decrease variability between sites. For Deployment 8, the same mix of detritus from Anglesea sites was used in Painkalac, while for Deployment 11, detritus from Site P2 was used at Site P3, the decomposition rates of which had the greatest difference of the three pairs of sites in Painkalac. Across the three times where decomposition rates were different between sites, the relationships between sites were not consistent, suggesting that variation in the processes associated with differences in decomposition at this scale was associated with both time and location, although in Anglesea rates were lowest at Site A2 on all these occasions.

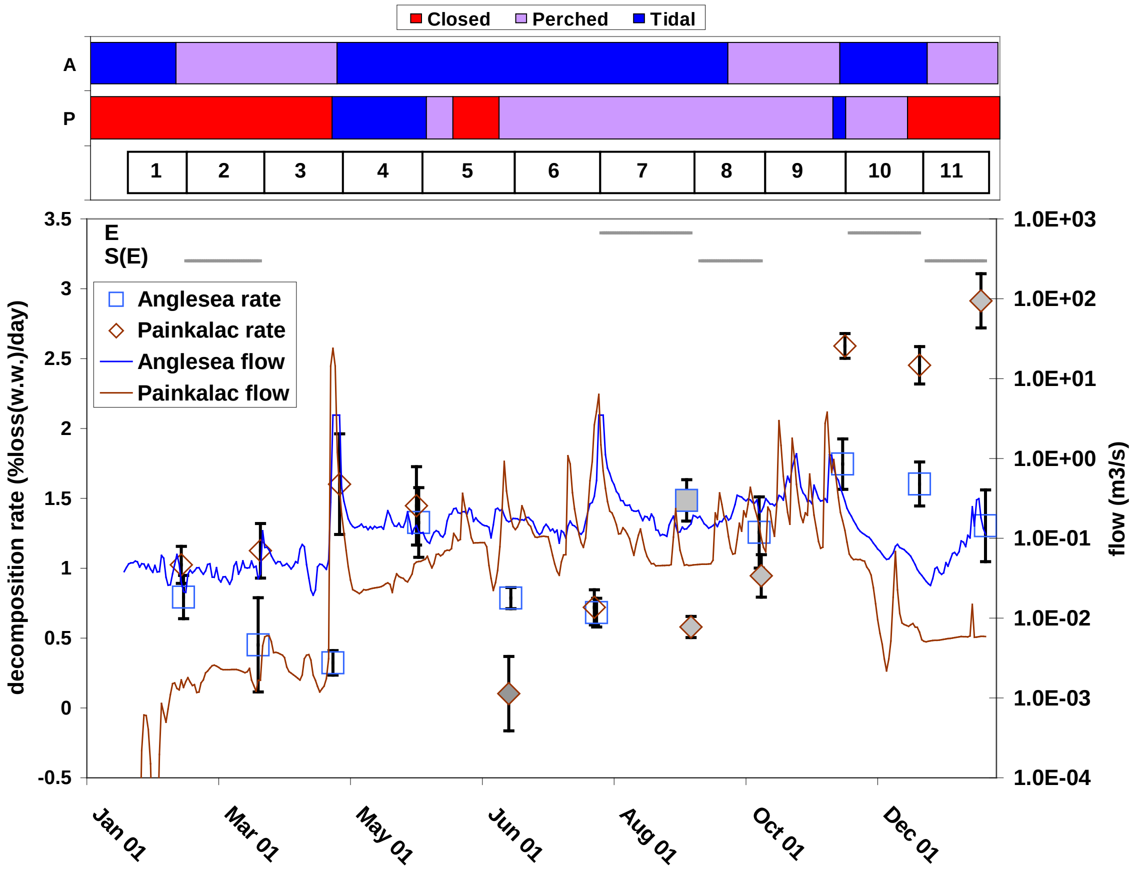


Figure 7.4. Mean (± s.e.) decomposition rates and freshwater flow in Anglesea and Painkalac estuaries, with periods in which there were significant differences on two spatial scales indicated by grey lines at top. The upper grey line represents inter-estuarine differences; the lower, inter-site differences (within estuaries). Mean rates are shown on the end date of deployment periods, which are shown above the main chart, along with hydrologic states of each estuary. Shaded symbols represent times when material was not sourced from the vicinity of some or all sites measured.

Decomposition rates appeared to respond differently to flow in each estuary (Figure 7.4). In Anglesea decomposition rates peaked in each of the three deployments following floods, where Painkalac showed no consistent pattern. Despite similar temporal patterns of peaks and declines following floods (see Section 6.3.4), there was no significant correlation between mean decomposition rates and mean deposition rate nor between mean decomposition rate and mean percent organic-matter content of either estuary (Table 7.5). There was a significant negative correlation between decomposition rate and seagrass cover (within sites) for Anglesea using both site and estuary means (Table 7.5).

		Anglesea			Painkalac		
	Estuary means	***p***	***r***	***n***	***p***	***r***	***n***
a)	**Sediment deposition rate**	0.758	0.105	11	0.662	0.149	11
	OM of deposited sediments	0.412	-0.276	11	0.411	-0.276	11
	Site seagrass cover	0.002	-0.849	10	0.252	-0.556	6
	Site means						
b)	**Sediment deposition rate**	0.983	0.004	31	0.668	0.080	31
	OM of deposited sediments	0.018	-0.423	31	0.306	-0.190	31
	Site seagrass cover	0.006	-0.541	24	0.083	-0.446	16

Table 7.5. Results of correlation analyses between decomposition rates and other variables measured in lower estuarine sites using: a) estuary means and b) site means. Site seagrass cover refers to percent cover of lower estuarine sites based on locations of bed edges for Deployments 2-11. Times where the edge was unmeasurable were omitted from the analysis.

Mean decomposition rates in Painkalac were greatest when the estuary was closed (Figure 7.5), which also corresponded with higher water levels and warmer months of the year. Compared to Anglesea, decomposition rates were similar in perched periods but greater when the estuaries were tidal.

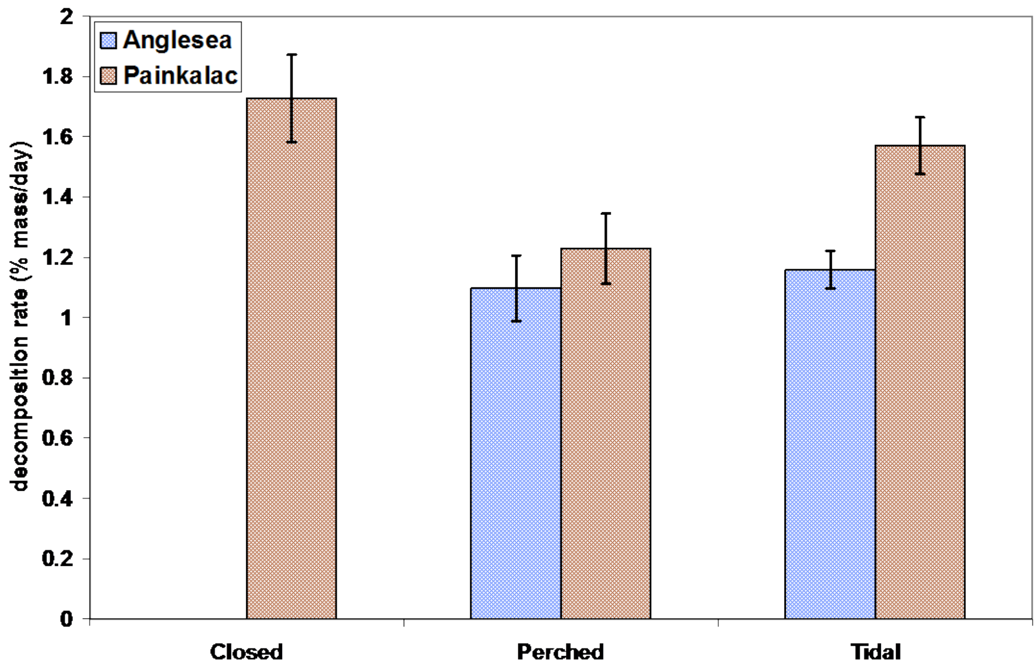


Figure 7.5. Mean (±s.e.) decomposition rates of seagrass in litterbags by estuary and state. Weightings were used for deployment periods which spanned more than one state. *n*=49, 54, 59, 86 and 32 for bars from left to right.

7.3.2. Cotton strip assay

7.3.2.a. Temperature

Sediment temperatures ranged from 14.7°C to 21.9°C over the first deployment period and from 15.8 °C to 27.7 °C over the second deployment (Figure 7.6, Table I.4 (Appendix I)). Mean temperatures were 3.5 degrees warmer during the second deployment compared to the first. At Anglesea, both shallow and deep loggers recorded marked diurnal variations, consistent with water temperature at the logger location (Section 5.3.2). In Painkalac, temperatures also showed diurnal fluctuations, except at the deep location during the first deployment where sediment temperature was constant around 16.7°C except for a 1 degree drop in the final 10 hours of the deployment. This consistency in temperature was potentially related to greater submersion during a period in which water levels had increased in association with periods of high seas on 18/10/2001 and 31/10/2001, and/or the location of the deep logger well below the halocline compared to the shallow logger which was located at the bottom edge of the halocline.

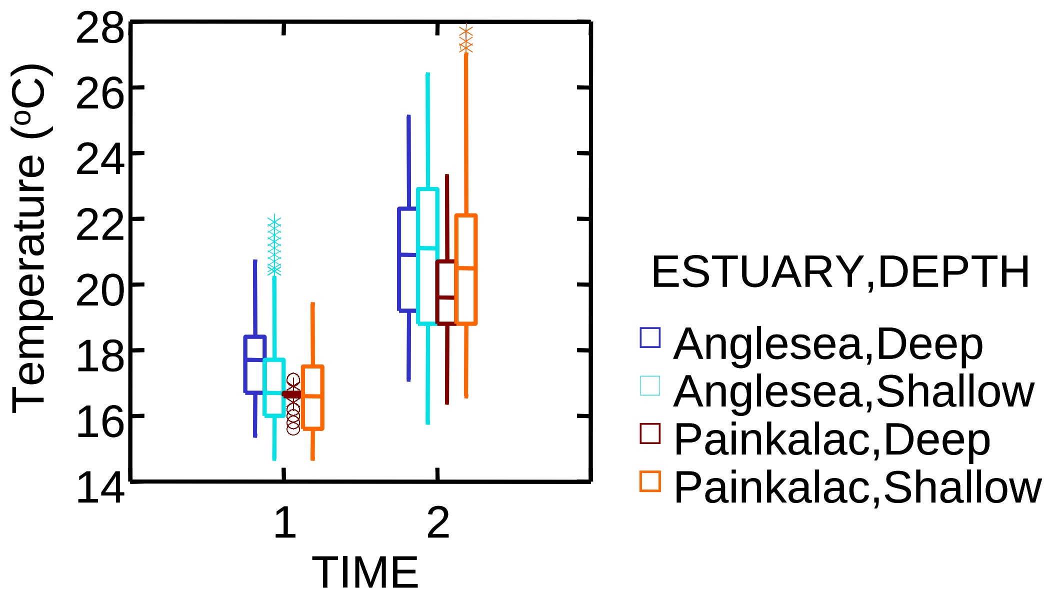


Figure 7.6. Sediment temperature by time, estuary and depth as recorded by loggers at Sites A1 and P1, from left to right *n*=1120, 1120, 1243, 1243, 1529,1529, 1496 and 1496.

7.3.2.b. Main comparison

In total, 583 valid measurements of changes in tensile strength were made. Tensile strengths of cotton strips ranged from 11% to 93% of original strength (y_o), with a mean of 41% and standard deviation of 17%, within the range for which *R* is robust (Correll *et al.*, 1997). The decomposition rate of the strips (*R*) varied from 0.020/day to 0.095/day, with a mean of 0.056/day and a standard deviation of 0.013/day.

Results of the ANOVA showed significant differences in mean decomposition potential (as measured by *R*) for two main effects and two interactions (Table 7.6). Temporal and spatial variability of decomposition potential resulted in significant differences between combinations of time and ruler positions (within deep and shallow halves of sites) and between combinations of time and site (within estuary). That these interactions, but not the time by estuary interaction, were significant indicates that, on smaller spatial scales (metres to 10s of metres), there is substantial variability in decomposition potential across periods of months. Despite this variability, Painkalac had a significantly greater decomposition potential than Anglesea (Figure 7.7a), as did deep locations compared to shallow (Figure 7.7b). Overall, significant sources contributed 66% of the total variance as measured by sums of squares, differences between estuaries contributing 47%, differences between depths contributing 12% and the remaining 7% accounted for by the two significant time interactions.

A comparison between seagrass decomposition rates for the two deployments (8 and 11) that coincided with those of the CSA (Table 7.7) showed a different pattern of significant results from that of the analysis of decomposition potential. Although the design of the seagrass study did not include multiple depths nor a spatial scale equivalent to that between rulers in the cotton-strip study, comparisons across times, estuaries and sites within estuaries were possible. While there were significant differences in decomposition potential among combinations of time and site (within estuary), there were no such differences in actual seagrass decomposition rates, which varied significantly among sites but consistently so by site across both times. On the other hand, there was significantly greater decomposition potential in Painkalac but no interaction between time and estuary (Figure 7.8b) whereas actual decomposition rates varied significantly depending on the interaction between both time and estuary (Figure 7.8a).

Source	S.S.	d.f.	M.S.	F-ratio	*p*
Time	0.000573	1	0.000573	1.524	0.2846
Estuary	0.025739	1	0.025739	35.302[a]	**0.0040**
Depth	0.006740	1	0.006740	30.665[a]	**0.0052**
Site(Estuary)	0.002799	4	0.000700	1.403[a]	0.3101
Ruler(DepthxSite(Estuary))	0.004777	24	0.000199	1.865	0.0670
TimexEstuary	0.000040	1	0.000040	0.106	0.7607
TimexDepth	0.000004	1	0.000004	0.021	0.8917
TimexSite(Estuary)	0.001504	4	0.000376	3.522	**0.0213**
TimexRuler(DepthxSite(Estuary))	0.002562	24	0.000107	2.542	**0.0002**
EstuaryxDepth	0.000396	1	0.000396	2.37[a]	0.1885
DepthxSite(Estuary)	0.000888	4	0.000222	0.844[a]	0.5807
TimexEstuaryxDepth	0.000025	1	0.000025	0.131	0.7353
TimexDepthxSite(Estuary)	0.000761	4	0.000190	1.782	0.1654
Residual	0.009144	216	0.000042		

Table 7.6. Results of ANOVA for main comparisons of decomposition rate of cotton strips. Significant terms are indicated by bold *p*-values. a=pseudo-*F* ratio with d.f. of 1,4; 1,4; 5,9; 2,5 and 8,14 for marked terms from top to bottom, *n*=4.

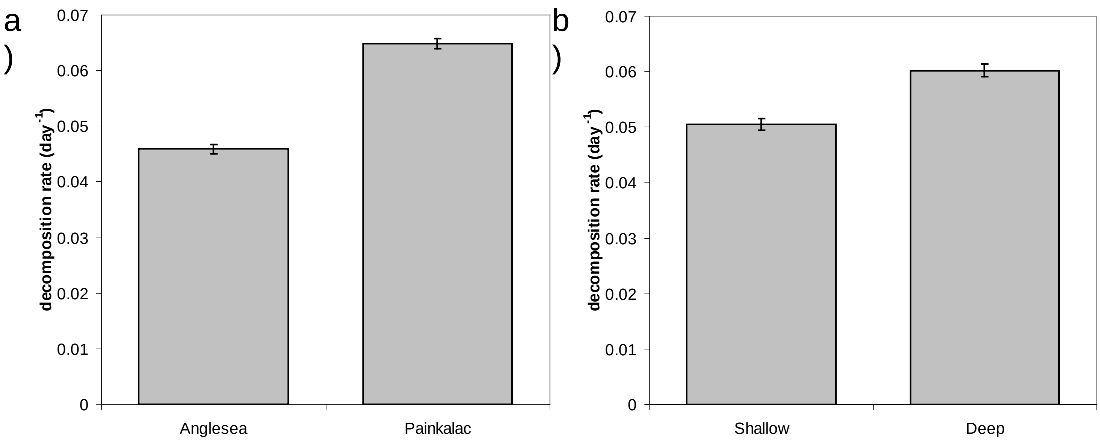


Figure 7.7. Mean (± s.e., *n*=144) decomposition rates of cotton strips between a) estuaries and b) depths during three-week periods in October-November 2001 and January-February 2002.

Source	S.S.	d.f.	M.S.	F-ratio	*p*
Time	9.771	1	9.771	42.373	**4.70×10^{-7}**
Estuary	4.187	1	4.187	0.300	0.639
Site(Estuary)	12.311	4	3.078	13.347	**3.31×10^{-6}**
TimexEstuary	10.860	1	10.860	47.098	**1.86×10^{-7}**
Residual (pooled)	6.456	28	0.2306		

Table 7.7. Results of ANOVA for comparisons of decomposition rate in litter bags for the two deployments that coincided with cotton-strip assays, *n*=3. Significant terms are indicated by bold *p*-values. The timexsite(estuary) interaction in the full model was not significant (*p*=0.972) and so was pooled with the residual. Results for the full model are presented in Table I.8, Appendix I.

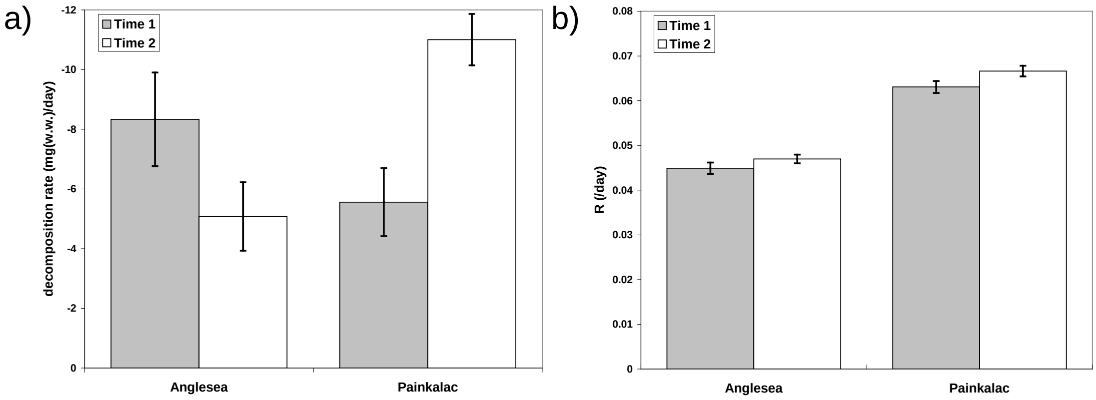


Figure 7.8. Mean (± s.e.) a) decomposition rate of seagrass in litter bags for the times coinciding with the CSA experiment (significant timexestuary differences) and b) decomposition potential, by estuary and deployment period (significant between estuary difference). *n*=9 and 72 for a and b, respectively.

7.3.2.c. Organic matter experiment

Results of the analysis of the experiment comparing areas with and without buried seagrass detritus are shown in Table 7.8. Significant differences in mean decomposition rate were observed for the interaction between the detritus and ruler (within site) factors, suggesting that mean decomposition rates were the result of combined influences of small-scale variability (metres) and the presence or absence of buried material. These differences are shown in Figure 7.9, where a tendency for higher decomposition rates at sites with buried detritus is seen, along with substantial differences among rulers within each site. There were also significant spatial differences in mean decomposition rate between sites, at a 10s-of-metres scale.

Source	S.S.	d.f.	M.S.	F-ratio	*p*
Detritus	0.005380	1	0.005380	10.51	0.1905
Site	0.002028	1	0.002028	13.35	**0.0064**
Ruler(Site)	0.001215	8	0.000152	4.75	**0.0002**
DetritusxSite	0.000512	1	0.000512	4.28	0.0722
DetritusxRuler(Site)	0.000956	8	0.000120	3.73	**0.0013**
Residual	0.001896	60	0.000032		

Table 7.8. Results of ANOVA for comparison of decomposition rate of cotton strips in and out of areas with buried seagrass detritus. Significant terms are indicated by bold *p*-values.

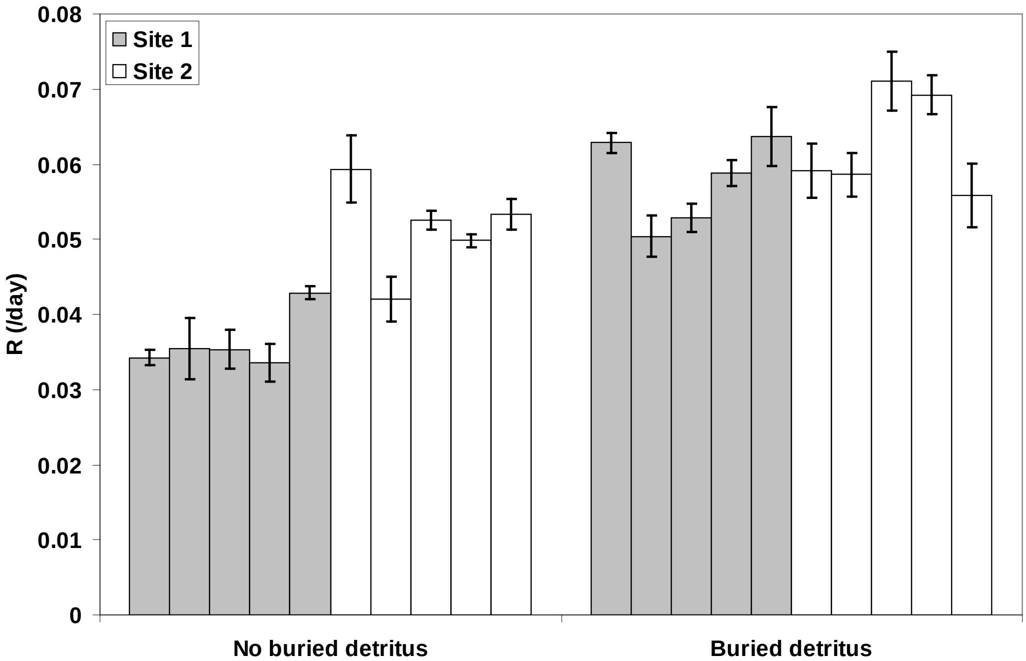


Figure 7.9. Mean decomposition rate of cotton strips (± s.e., *n*=4) by ruler within site for locations with and without buried seagrass detritus.

7.4.   Discussion

7.4.1. Context of experiments and limitations of techniques

The relative importance of decomposition of seagrass detritus as an ecological process is dependent on factors like the relative amount of seagrass production in a given system, rates of herbivory and the proportion of detrital material that is either retained in the seagrass beds, transferred to other compartments or exported from the system. In the case of Anglesea and Painkalac estuaries, this sampling program coincided with a period in which seagrass extent (and biomass) declined dramatically, so that bed edges retreated within lower-estuarine sites along with reductions in shoot density (see Chapter 6). By the middle of the sampling period there were few detrital deposits of large seagrass pieces in either estuary, reflected in difficulties with sourcing enough material to place in litterbags (Table I.1, Appendix I). In this sense, it is likely that the natural contributions of substantial amounts of new seagrass detritus to the systems were low compared to other years, especially during Winter and Spring.

During large declines in seagrass extent and production, the dynamics of decomposition in these systems may also be of particular importance as the ratio of primary to secondary production within the beds decreases. There is evidence suggesting that such large-scale changes in seagrass beds are common in intermittent estuaries and are followed by recolonisation over periods of several years (Talbot *et al.*, 1990, see Chapter 6).

In drawing conclusions from both the litterbag and CSA experiments, the nature and limitations of each technique must be considered. Both methods measure only a component of a complex process. Effectively, the mass losses recorded from the litterbags are a sum measure of several processes including leaching and absorption of material from the detritus, mechanical and biological fragmentation into pieces smaller than the mesh of the bags, consumption of the detritus by microbes and, potentially, invertebrates (Harrison, 1989; Boulton & Boon, 1991). The losses in tensile strength of the strips in the CSA are a combined measure of a reduction in the degree of polymerisation of the cellulose fibres and of the breakdown of the cellulose itself (Howard, 1988). As such, the CSA is a measure of the potential rate of only one part of the processes being measured in the litterbags. Some limitations are shared by the two methods (see Harrison, 1989; Boulton &

Boon, 1991; Correll *et al.*, 1997; Boulton & Quinn, 2000), in that both methods:

- introduce artificial structures (litterbags, bricks, rulers and ruler-shaped holes in the sediment) that are not easy to control for;
- are potentially sensitive to the duration of deployment, as rates of decomposition are non-linear through time;
- may be affected by small changes in deployment elevation relative to the sometimes-steep oxygen gradient across the water-sediment boundary; and
- may represent ‘islands’ of resources for microbial communities that differ in spatial arrangement to naturally-occurring substrates.

Other limitations were particular to each method used. Differences in the size and composition of detrital material varied over time and space, and while some aspects of this were intended to be included as part of the design, such as natural variability in surface area:volume and C:N ratios, artefacts such as variation in the degree of fragmentation required for material to escape from the litterbags were also introduced. A bias in the material collected towards more refractory components of detritus was also likely, due to longer residence times leading to accumulation and hence increased availability. The ‘representativeness’ of the rate of cellulose degradation as a measure of the decomposition potential of plant material also must be qualified by the fact that the strips represent a processed and uniform source of cellulose, whereas plant material, although composed primarily of cellulose, contains other nutrients, particularly nitrogen compounds, that are necessary for growth of microbes (Howard, 1988). In this respect, CSA is likely to be more applicable to comparisons with seagrass detritus than with other vascular plants as seagrasses have lower proportions of N than most plants (although fresher material may contain relatively large amounts of proteins and amino acids: Klumpp *et al.*, 1989), and the process of seagrass decomposition is often limited by N availability (Menéndez *et al.*, 2003; Mateo *et al.*, 2006).

7.4.2. Patterns in decomposition rate and potential

Although direct comparisons of seagrass decomposition with other studies are not possible, due to the use of wet weight measurements, mean measured rates of decomposition were consistent with the faster dry-weight rates reported for other species of *Zostera* from lab and litterbag studies (Harrison, 1989). Mean rates in Anglesea (1.1%(w.w.)/day) and Painkalac (1.5%(w.w.)/day) estuaries were also slightly faster than that of ~0.7%(d.w.)/day of *H. tasmanica* measured over 46 days in Autumn in Corner Inlet (a marine embayment in eastern Victoria: Klumpp & Van der Valk, 1984). Mass gains measured in seven of the 207 valid litterbag samples were possibly due to trapped silt, growth of microbes (Boulton & Boon, 1991), additional fine detritus entering the bags or increased absorptive capacity of the detritus.

Compared to measurements of decomposition potential using CSA in other environments (reported as percent loss in tensile strength per day – CTSL), overall mean decomposition potential in Anglesea and Painkalac was moderate to high compared to that of various freshwater wetlands (Table 7.9).

Location	CTSL (%/day, ±s.e.)	Treatment/notes
Anglesea estuary	2.36 (±0.04)	overall mean
Painkalac estuary	3.34 (±0.03)	overall mean
Florida Everglades [a]	1.05 (±0.09)	none
Florida Everglades [a]	7.04 (±0.08)	nutrients added
Amazon basin	~0.7 (±~0.1)	anoxic swamp
Amazon basin	~3.3(±~0.05)	aerobic creek
New Zealand stream[b]	~0.5	smallest mean
New Zealand stream[b]	~1.7	largest mean

Table 7.9. Comparative rates of tensile strength loss (CTSL) in immersed sediments at similar depths to this study. Examples selected from those a: reviewed in Maltby (1988) and b: reported in (Boulton & Quinn, 2000).

Significant differences in decomposition rate among sites within estuaries were observed on only three of the ten occasions where they were effectively examined. At these times, and in general, Anglesea had greater among-site differences in decomposition rate than did Painkalac. When there were significant differences, Site A2 consistently had the lowest decomposition rate of the three sites in Anglesea, suggesting spatially consistent influences driving differences between sites.

Site-scale differences in seagrass decomposition rate were consistent with patterns in decomposition potential, as measured during two of the periods when there were significant site-scale differences in seagrass decomposition (8 and 11). On both of these deployments, significant differences in mean *R* were observed among combinations of site and time. During the first of these, Sites A1 and A2 had substantially lower decomposition potential than Site A3. During the second deployment, there were only small differences in decomposition potential between sites, although mean *R* at Site A2 was still lower than the other sites. As for seagrass decomposition rates, there were no substantial among-site differences in decomposition potential in Painkalac at these times.

Mechanisms responsible for the intermittent variability in decomposition rate at the site scale were not obvious and there were no clear links between differences in decomposition rates at sites and values for flow, state/inundation, season/temperature, salinity, pH, or sediment deposition rates. In Anglesea, the combination of the negatively correlated mean percent organic matter in deposited sediments and mean decomposition rate for sites (see Table 7.5) effectively separated the three sites (Figure 7.10a), whereas no grouping of sites or correlation was evident for Painkalac. A likely cause of this difference was the proximity of Anglesea sites to stormwater drains; Site A2 in particular was close to a drain that was the source of a large amount of fine organic material while A1 was closer to outlets than A3. The resulting among-site variability in decomposition rates was, however, not as consistent as the among-site differences in organic matter of deposited sediments.

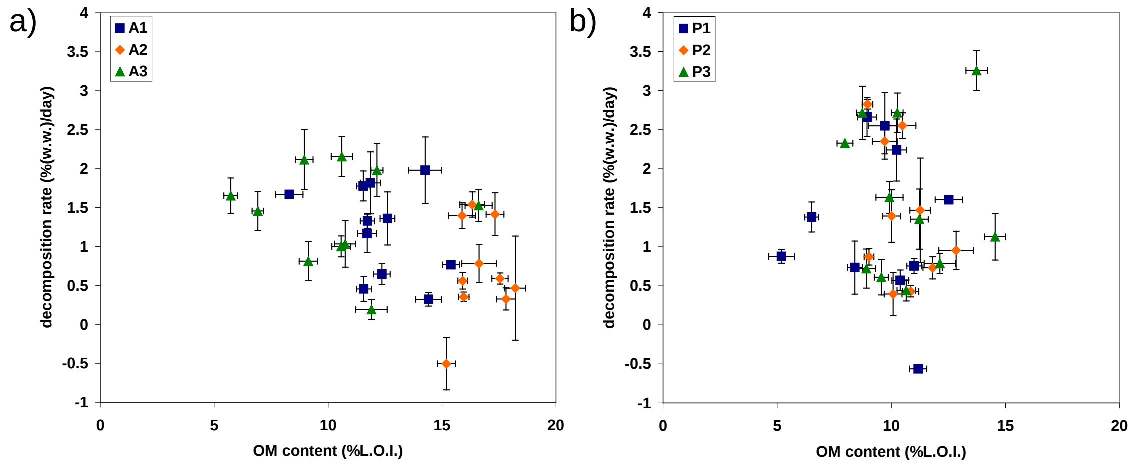


Figure 7.10. Mean decomposition rate vs mean percent organic matter of deposited sediments at sites in a) Anglesea and b) Painkalac estuaries.

The two times when seagrass decomposition rates were significantly different between estuaries apparently resulted from different processes affecting the rate of decomposition. The first significant difference occurred when a flow-associated peak in decomposition was occurring in Anglesea while the rate in Painkalac was low, as it had also been for the previous two Winter deployments (see Figure 7.4). The second significant difference also occurred when the decomposition rate at Anglesea was also relatively high, following a flood, but in this case the rate in Painkalac had increased such that it was greater than that in Anglesea.

Reasons for this large increase in decomposition rate in Painkalac compared to Anglesea during this period may be linked to the higher water levels and substantial increase in extent of seagrasses in Painkalac (see Chapters 4, 6). Temperature and salinity were similar in both estuaries but, apart from a short tidal period, Painkalac was perched then closed, when Anglesea was tidal then perched. Decomposition potential was consistently greater in Painkalac than Anglesea before and after the increase in actual decomposition rate in Painkalac but was also consistently greater in the deeper halves of sites (Section 7.3.2.b). These results support the hypothesis that the increase was due either to the faster decomposition of newer seagrass material present in Painkalac at that time, or to the more consistent inundation of the litterbags.

Several factors associated with inundation have been reported to change the decomposition rate of seagrasses. These may relate to differences in decomposition rates between hydrologic states and the differences in decomposition potential between depths and estuaries. Greater *in situ* decomposition rates of seagrass in shallow subtidal areas compared to intertidal locations have been reported (Gallagher *et al.*, 1984; Machás *et al.*, 2006), despite alternate wetting and drying and exposure to sunlight increasing rates of decomposition (Vähätalo *et al.*, 1998; Vähätalo & Søndergaard, 2002) and a probable increase in mechanical fragmentation of detritus due to wave action (Mateo & Romero, 1996).

Increased *R* at depth may have been related to exposure to waters of greater salinity (~18+, see Figure 5.5). In a study in Denmark, however, decomposition rates decreased with increasing salinity except for the most saline site (salinity = ~15), where a marked increase in decomposition rate was observed (Mendelssohn *et al.*, 1999). In the same study, and that of Boulton and Quinn (2000), the availability of nutrients was the largest influence on decomposition of the strips, a factor that may have been influential in the increased rates of seagrass decomposition in Anglesea following floods.

The inverse relationship between seagrass coverage within sites and decomposition rate (see Table 7.5) is counter-intuitive, given the typical elevation of nutrients within seagrass beds (e.g. Boon, 1986). It is possible that this relationship may have been due to increased water movements, the provision of ‘islands’ of resources where none existed (and hence increased concentrations of microbes in the litterbags) or to releases of nutrients from newly unvegetated sediments. Changes in shoot density and the location of bed edges (see Section 6.3.4) meant that the seagrass beds at the sites, in terms of above-ground material, were close to non-existent in the middle and later parts of the sampling period. This was especially evident when compared to earlier periods when individual shoots were larger and present in denser and more extensive beds prior to, and to a lesser degree during, the early deployments of the decomposition study. Thus, the mechanisms responsible for the inverse relationship between cover and decomposition rates were also potential causes of the greater decomposition rates in Painkalac in January 2002, compared to January 2001, given similar hydrologic conditions in those periods (see Section 4.3).

This study is believed to be the first to report:

i. decomposition rates of seagrass from intermittent estuaries in Australia and;
ii. *in situ* decomposition rates of *Zostera muelleri/capricorni*,

although other *Zostera* species have been studied, most frequently *Z. marina* (reviewed in Harrison, 1989). The approach of this study, however, differed from most investigations of *in situ* seagrass decomposition in that rates of locally-available material were measured over space and time. Previous research has most often followed the trajectory of a single source of material, sometimes at different locations and, less frequently, over more than one time. In this study, different processes appeared to affect decomposition rates in the two estuaries. In Anglesea, there was an increase in mean decomposition rates following flood events, while, in Painkalac, the most obvious relationship was the increase in decomposition rates associated with the closure of the estuary in the final months of the study.

The cloth strip assay proved a useful technique for estuarine use. Ease of use meant that a stratified sampling design that examined effects at multiple spatial scales was possible and that a separate examination of the effects of buried detritus outside seagrass beds could be made. Results showed that there was variability across combinations of time and sites at scales of metres to 10s of metres but that there were also significant differences in decomposition potential between estuaries and depths. Results from this component of the study also provided useful information for interpreting the decomposition rates of *in situ* seagrass detritus. In particular, they provided a comparative measure of the microbial component of the integrated rate of leaching, fragmentation and microbial consumption that was measured in the litterbag component of the study. Further studies examining the effects of a range of natural and anthropogenic environmental modifications on decomposition potential would increase the value of the CSA as an estuarine research tool for use as a process-based indicator of health *sensu* Fairweather (1999). The technique also holds great potential as part of a suite of measurements examining detrital processes alongside techniques such as decomposition of both *in situ* and uniformly sourced detritus, microbial assays and stable isotopes.

8. General Discussion

8.1. Main findings

8.1.1. Fresh waters

Discharge patterns of fresh water in the catchments of Anglesea and Painkalac estuaries were intermittent and fitted a pattern typical of flow along an east-west gradient of the Otway Ranges. Flow was more closely related to rainfall over longer time periods, in Painkalac compared to Anglesea catchment, and for naturally-flowing tributaries rather than for anthropogenically modified flows. R^2 values ranged from 0.065 for long-term daily comparisons for Salt Creek to 0.67 for monthly totals at the Painkalac Creek site above the dam (see Section 3.3.2). In most aspects, water quality was also typical of the region, an important exception being the generation of acids in the catchments of the Anglesea River and Distillery Creek (Section 3.4). This resulted in low pH for all but the smallest flows from these catchments. Outcroppings of the Eastern View Group (and oxidation of sulphides in brown-coal-bearing strata) were associated with this phenomenon and also resulted in acidic flows on the opposite side of the Otway Ranges.

Intermittent flows, which varied substantially between years, had major implications for water quality in the Anglesea catchment in particular. Interactions between groundwater and sulphidic sediments over a prolonged drought led to a 'first flush' effect, in which initial flows from Salt Creek had high titratable acidity and concentrations of aluminium orders of magnitude greater than relevant water quality guidelines (see Section 3.4.2, ANZECC & ARMCANZ, 2000).

A dam on Painkalac Creek reduced flow into Painkalac estuary, and the mine and power station on the Anglesea River increased flow in Anglesea River. Effects of these anthropogenic changes were greatest during times of low natural flows. Periods of zero flow into Painkalac were longer and more frequent, but groundwater-sourced process water was continually discharged into Anglesea estuary, augmenting any natural flows (Section 3.3). These changes had no apparent effects on water quality in Painkalac Creek but, depending on the volume and composition of upstream flows, there were major increases in pH, conductivity, temperature, and dissolved oxygen in Anglesea River (Section 3.4).

8.1.2. Estuarine waters

The volume and timing of freshwater inflow was a major determinant of both hydrologic state and salinity structure of the estuaries. Low or zero flows were a precondition for closures of the mouths of the estuaries, which, without artificial opening, tended to remain in that state for up to 17 months until higher flows opened the mouths. Fully tidal states were only observed following flood events, while smaller peaks in flow, and high seas were associated with transitions to perched states (Section 4.3).

Seven patterns of salinity stratification were observed in Anglesea and Painkalac estuaries. Of these, four were exclusive to only one of the estuaries (A and B in Anglesea, and F and G in Painkalac: see Section 5.3.1.d). When tidal, salinity structure of both estuaries was similar to that of a classical ‘salt-wedge’ estuary. During perched and closed states a lesser degree of stratification was typical of the estuaries. Differences were most pronounced during extended closed periods, due to differences in freshwater flow regimes. At these times Anglesea tended to remain stratified for longer than Painkalac while gradually becoming fresh, Painkalac became well mixed and brackish and then, with evaporation, hypersaline.

Aside from salinity and inundation regimes associated with hydrologic states, three aspects of estuarine water quality had high potential for causing biological effects (Sections 5.3, 5.4). These were:

- hypoxia and nutrient release resulting in an algal bloom in Painkalac. This was associated with a period of zero flow and a small pool of isolated saline water in a deeper part of the upper estuary when the rest of the estuary was well mixed. Despite longer periods of stratification, anoxia of bottom water was not observed in Anglesea;

- acidification events in Anglesea estuary, associated with higher flows that were sufficient to overcome the buffering capacity of Alcoa's discharges which have resulted in at least one fish kill, and had potential to affect most estuarine biota and;
- periods of low water clarity in Painkalac associated with higher flows that caused erosion in the catchment of Painkalac Creek with potential to reduce photosynthetic rates and to clog and/or smother benthic fauna.

8.1.3. Seagrass dynamics and detrital processes

Within the three-year study period a large decrease in seagrass extent was observed in both estuaries, followed by some recovery in Painkalac estuary (Chapter 6). The decline in seagrass cover (from 60% to 10%: Figure 6.11) was associated with floods, erosion and mobilisation of sediments and exposure of habitat during tidal states. The recovery of seagrasses in Painkalac estuary was associated with higher water levels, due to the return of Painkalac to perched and closed states. Anglesea remained tidal over the same time and seagrasses did not expand in area. While some seasonality in seagrass extent was measured, particularly in the more *r*-selected *Ruppia*, longer-term and stochastic hydrologic events appeared to be the main drivers of seagrass extent.

Changes in the historical extent of seagrasses in Anglesea derived from aerial photos were consistent with the results from ground surveys, in that there were negative correlations between long-term (>3 year) rainfalls and seagrass extent (Section 6.2.1.a). This supported the hypothesis that long periods with low flow and no large floods (which, in these systems, represent a large proportion of annual flows) create stable conditions with relatively high water levels that allow establishment of beds.

Decomposition rates of *in situ* seagrass detritus varied inconsistently between the two estuaries through time, with estuary means ranging from 0.10 to 2.9%(w.w.)/day. A weak seasonal signal was apparent in Painkalac estuary but rates were twice as fast in January 2002 (at the end of the sampling period) than in January 2001. In Anglesea, decomposition rates appeared to peak immediately following floods and then slowly decline over a period of months (Section 7.3.1).

During seven of ten litterbag deployments, there were no significant differences in decomposition rate among sites within each estuary, although on the three times when there were differences they were highly significant. These differences between sites were primarily found in Anglesea and were most likely related to consistent differences in character among sites, including distance from storm-water drains.

Use of a standard cotton strip assay to measure cellulose decomposition potential of sediments proved to be a useful technique for estuarine use. Its relative ease of replication allowed simultaneous assessment of differences across several spatial scales as well as between depths and sites with and without buried detritus (Section 7.3.2). Decomposition potential varied with time on spatial scales from metres to tens of metres but also was significantly greater in Painkalac than Anglesea and in deeper halves of the lower estuarine sites (below ~0.3mAHD). The presence of buried detritus alone did not result in differences in mean decomposition potential, but did interact with metre-scale variability to produce significant differences, with a tendency for greater rates in areas with buried seagrass detritus.

8.2. Implications

In the context of naturally intermittent inflows, the increased periods of zero flow at Painkalac represented a comparable regime to what could be expected from water extraction elsewhere, or a drying climate, as has been predicted for this, and other regions with Mediterranean-type climates (Bolle, 2003). The augmentation of flows at Anglesea was not so much comparable with potential climatic changes than with other human-influenced flow regimes, for example the artificially-low (versus naturally-zero) flows common in regulated rivers (McMahon & Finlayson, 2003).

In terms of water quality, Painkalac was typical of undisturbed catchments in the region, except for elevated turbidity/sediment loads during times of high flow. Anglesea was unusual, both in terms of the delayed runoff characteristics due to its peaty streambeds, and in relation to the low pH of flowing waters. Neutralisation of these waters by discharges from Alcoa's ash ponds is likely to have changed not only the timing of acidic inflows to Anglesea estuary but also the temporal and spatial patterns of metal inputs to the estuary in that the precipitation of dissolved species of metals that would have occurred in the estuary prior to the establishment of the power station now occur in reaches just above the estuary. These metals, unless otherwise sequestered upstream, are now likely to move to the estuary in pulses during floods at higher concentrations than would have occurred naturally.

These changes to freshwater inflows at Anglesea and Painkalac estuaries caused substantial changes in hydrology and salinity structure. Continual low flow when Anglesea was closed resulted in very different conditions to those in Painkalac estuary, where there were extended periods of zero flow. The contrasts in both estuary volume and salinity substantially modified the extent and character of available habitat between the estuaries (although estuarine organisms are often well adapted to such variable conditions e.g. Brock, 1982b; Kanandjembo *et al.*, 2001; Mackay & Cyrus, 2001). Extended closures and hypersalinity in Painkalac did not apparently affect growth of seagrasses despite the associated period of eutrophication but limited connectivity with the sea would have affected organisms such as fish that normally spend only part of their life cycle in estuaries.

Hydrologically, the estuaries were similar to other intermittent estuaries, both in Australia and globally (e.g. Whitfield, 1992; Pollard, 1994; Young & Potter, 2002; Stretch & Zietsman, 2004). In terms of inundation regime, Painkalac tended be more isolated from the sea and 'lagoonal' while Anglesea estuary had a greater marine influence and was more often similar to permanently-open barrier estuaries. The timing and duration of salinity stratification was intermediate between estuaries on the east coast of Australia, which are stratified for ‘brief periods following floods’ (Digby *et al.*, 1999) and those of Western Australia which stratify regularly on a seasonal basis. In these respects they are similar to some South African estuaries (e.g. Largier & Slinger, 1991; Whitfield, 1992).

Changes in the hydrologic state and stratification patterns in each estuary appeared to be the largest contributor to changes in seagrass extent, creating changes in both the area and quality of habitat that was present. While changes in water quality, especially low pH, were also likely to affect seagrasses (e.g. Millhouse & Strother, 1986), these changes were of a temporary nature and have not restricted the ability of seagrasses in these systems to recover. Although effects on the large areas of emergent macrophytes and salt marsh at higher elevations were not specifically examined, both hydrological and chemical factors were likely to have also influenced these components of the estuarine flora. Links to decomposition rate of *in situ* seagrass detritus were less clear, although increased cellulose decomposition potential with depth, and in Painkalac, suggested that inundation regimes were likely to have an effect on this process.

Flow-related changes to seagrass extent and density observed in the two intermittent estuaries that were the primary focus of this study were similar to those observed, most often anecdotally, in other intermittent estuaries (e.g. Bally, 1987; Jones & West, 2005). A critical link between flows and estuarine conditions was the state of the sand bars, the building and breaching of which have been shown to affect the physical, chemical and biological nature of these estuaries (e.g. Lenanton, 1974b; Harvey, 1988; Young *et al.*, 1997; Kirk & Lauder, 2000; Young & Potter, 2002; Ranasinghe & Pattiaratchi, 2003; Froneman, 2004; Saintilan, 2004; Stretch & Zietsman, 2004). This study has demonstrated that relatively small alterations in flow at times of naturally-low or zero flow, while not representing a large proportion of total inflows, can have large effects (with differences in water levels, water quality and seagrass extent during closed states: Chapters 4, 5, 6). In contrast to the seasonal patterns of more stable, marine-influenced systems, floods in these small systems represent a substantial portion of total annual flows and create major disturbances at irregular intervals that can effectively ‘reset’ the systems. Rates of recovery of seagrasses also appeared to be closely linked to flow regimes subsequent to these disturbances.

It is likely that when the mine is closed, discharges from Alcoa to the Anglesea estuary will cease and the estuary will revert to one with intermittent inflows. In Painkalac, demand for water from the reservoir is increasing and it is unlikely that natural flows will be restored, although management of volume and timing of environmental flows from the reservoir is likely to be improved. Overall, it is likely that flows to each estuary will be reduced, but that Anglesea will continue to have the greater inflows. In addition to these potential changes, these systems are also likely to be sensitive to climate change. For regions with Mediterranean-type climates, including the study area, reduced rainfall is likely to lead to increased periods of closure in intermittent estuaries, a potentially positive outcome for seagrasses. In contrast, predicted increases in the intensity of storms would be likely to lead to more substantial opening events that would result in fully tidal states and thus a reduction in the proportion of time that estuaries are perched. This type of change would increase the frequency of estuary-wide disturbances such as those observed in this study.

Each of the 20 specific objectives listed in Section 1.5 were addressed in detail in the body of the thesis. Accomplishment of these aims has led to an understanding of the nature and effects of freshwater flow into two intermittent estuaries in which all the links shown schematically in Figure 1.2 have been explored. In these numerous, but little-studied systems, floods have a powerful and lasting influence on the nature and extent of available habitat. Despite this, relatively small modifications of flow regimes can also have substantial effects, especially in systems with small catchments and extended periods with zero or low natural flow.

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Appendix A. Flow Gauge Ratings and Calculations

Information on the calculation of flows from recorded water heights at four manually-read gauging stations and details of stage-discharge ratings of the automatic station in Anglesea River are given below. Data from the manual stations were provided as direct recordings of height. Data from the automatic station were processed by Theiss and provided as flow.

There are many uncertainties associated with the use of stage-discharge relationships, particularly with non-purpose-built structures (Gordon *et al.*, 1992). Despite this, data from these stations provided important quantitative information on relative flow rates and times of no flow in the Anglesea system.

Marshy Creek

The Marshy Creek gauge was located at a culvert above Alcoa and rated by Theiss Environmental Services for heights of 0.1m to 0.58m at 1cm intervals (corresponding to flows of 0 to 32.6 ML/day, respectively). Some readings by Alcoa staff were made to the nearest mm and, because stage-discharge equations were not available from Theiss, polynomial equations were used to interpolate between readings and to convert the height readings to flow rates. In the case of the Marshy Creek gauge it was necessary to fit separate curves to the upper and lower sections of the ratings tables.

Equations for the Marshy Creek gauge were:
For water heights of:

0 to 0.100m: $Q=0$;

0.101 to 0.299m: $Q=612510h^6-678840h^5+304960h^4-70744h^3+8991.3h^2-592.77h+15.791$ ($R^2=0.9999$); and

0.30 to 0.58m: $Q=9057.2h^5-21515h^4+20312h^3-9294.0h^2+2120.1h-191.64$ ($R^2=1.000$), where Q=discharge (ML/day) and h = gauge height in metres. The latter equation was also used to extrapolate to 0.70m, above which there were only three readings, all associated with floods.

Salt Creek

The Salt Creek gauge was located in the diversion channel around the mine. This gauge was originally established in 1967 and recorded flow data from 1967 to 1982. A graph of water levels from a flood in November 1992 at this station was also made available by Alcoa.

This station was repaired and the rating table checked before data were collected for this study. The station was rated for the greatest range of flows of the Anglesea gauges, from 0 to 0.73m, corresponding to zero to 546ML/day.

Two equations were required to interpolate readings from this gauge:
For heights of:

0.000m to 0.119m: $Q=-50613000h^6+14287000h^5-1355100h^4+60241h^3-741.36h^2+25.087h+0.00155$ (R^2=1.0000); and

0.120m to 0.249m: $Q=2130500h^5-2016300h^4+747170h^3-134900h^2+12351h-443.82$, ($R^2$=1.0000).

From 0.250m to 0.730m interpolation was not necessary because the five height readings in this range were only measured to the nearest centimetre. For one reading, at 0.74m, flow was extrapolated using the height-flow relationship of three rating points at heights from 0.70m to 0.73m.

For flood flows, large extrapolations have been made for qualitative comparisons. These flows were calculated based on an equation fitted to the upper portion of the ratings table (0.25m to 0.73m). This equation is:

$Q=531.3344h^2+433.4708h-51.7396$ (R^2 = 1.0000)

Ash Pond

The ash pond gauge, at a pre-existing square concrete channel was rated to 0.30m although 46% of the readings were above this height. Further gauging was difficult due to overtopping of the gauging structure into adjacent reedbeds. A single equation was used for interpolation of readings within the rating table:

$Q=-42069h^5+44406h^4-18431h^3+3899.1h^2-380.96h+13.528$ (R^2=1.0000).

Mine Reclaim Discharge

The mine reclaim gauge was located at a weir over which mine seepage water was discharged. This gauge was rated for all recorded flows and two equations were used to calculate flow for heights of:

0.095m to 0.160m: $Q=-23875000h^5+15784000h^4-4145600h^3+541290h^2-35089h+902.12$ (R^2=1.0000); and

0.160m to 0.210m: $Q=25000000h^5-23208000h^4+8600000h^3-1589000h^2+146480h-5390.3$ (R^2=1.0000).

The weir was not overtopped until a water height of greater than 0.095m was reached.

Anglesea River

The Anglesea River gauge downstream of Alcoa was established as a purpose-built structure and included a logger that recorded height at half-hour intervals. These data were processed by Theiss and provided as flow rates from 25/8/1999 to 31/3/2002. Heights were provided for times when the rating of the station was exceeded. The rating table for this gauge was revised several times as detailed below. Notably, the installed section was washed downstream in the flood of April 2001, following which the banks were rated at their post-flood shape and later reshaped and re-rated (Table A.1).

Table	Effective from	Max rating (ML/day)	Max. extrap. (ML/day)	Notes
1.00	1/7/1999	37.1	-	First rating
2.00	1/9/1999	36.9	-	Revision – low flow
2.01	1/9/1999	312	411	Revision – above 13ML/day
1.01	26/10/2000	312	411	Revision - below 66 ML/day
3.00	24/4/2001	89.8	131	New rating – post flood-damage
4.00	22/8/2001	94.3	137	New rating - repairs
5.00	8/2/2002	100	143	Revision – slight downward all flow

Table A.1. Details of rating tables for the flow gauging station in Anglesea River downstream of Alcoa, above Anglesea estuary. Ratings calculated by Theiss Environmental Services.

Flow into Painkalac estuary: Corrections of gauge data

As mentioned in Section 3.3.2, periods where zero flow was recorded where Painkalac Creek entered its estuary did not match records from the flow gauge located below the reservoir. To correct for this discrepancy, ~five-weekly observations of flow in the section of Painkalac Creek immediately above the estuary were compared with flow records from below the dam for times of low and zero flow. Based on field observations, there were two periods of no flow in Painkalac Creek below the dam, from 17/12/1999 to 10/3/00 and from 8/12/00 to 18/3/01 (84 and 100 days, respectively, but with long gaps between data). At the gauge below the dam, there were eight periods of no flow between 20/1/1999 and 31/3/2002. These periods ranged in duration from 5.5 hours to 18 days and 14 hours with an average dry period of 7 days and 19 hours (Table A.2).

From comparisons of flows measured simultaneously at the gauge below the dam and at the site above the estuary, it appears that in 1999 daily flows below 0.0015m^3/s at the gauging station did not enter the estuary as surface flow. In 2000, 2001 and 2002 this ‘threshold’ flow at the gauge was 0.004m^3/s. When gauged flows below these thresholds were substituted with zero values, there were 13 periods where there was no freshwater flow to the estuary from Painkalac Creek. These periods ranged from one day

(16/5/2000 and 22/5/2000) to 97 days (16/12/2000-21/3/2001) (Table 3.10). These periods are largely consistent with field observations and provide additional insight into shorter periods of no flow into the estuary that were not possible to observe with monthly visits.

Period	Logged flow below dam (1.7km upstream)	Modified logged flow (estuarine input)	Interpolated field observations (estuarine input)
1	<a	22/1/99-28/1/99	no obs.
2	2/2/99-7/2/99	30/1/99-8/2/99	no obs.
3	<a	12/4/99-19/4/99	no obs.
4	7/10/99-10/10/99	3/10/99-3/11/99	3/10/99: $0.00214 m^3/s$, 28/10/99: trace flow
5	<a	7/11/99-8/11/99	no obs.
6	<a	9/12/99-26/12/99	17/12/99-10/3/00
7	8/1/00	28/12/99-14/3/00	17/12/99-10/3/00
7	11/1/00-30/1/00	28/12/99-14/3/00	17/12/99-10/3/00
7	3/2/00-20/2/00	28/12/99-14/3/00	17/12/99-10/3/00
8	<b	10/5/00-12/5/00	no obs.
9	<b	16/5/00	no obs.
10	<b	22/5/00	no obs.
11	2/12/00-8/12/00	28/11/00-8/12/00	8/12/00-18/3/01
12	19/12/00-22/12/00	16/12/00-21/3/01	8/12/00-18/3/01
12	17/1/01-26/1/01	16/12/00-21/3/01	8/12/00-18/3/01
13	<b	27/3/01-21/4/01	no obs.

Table A.2 Comparison of periods of no flow in Painkalac Creek above the estuary based on different measurement techniques. Logged flow was recorded half-hourly while field observations were made approximately every five weeks. Periods were determined from times when logged flow data were below the thresholds required for surface flow to the estuary (<a=<$0.0015 m^3/s$, <b=<$0.004 m^3/s$). no obs. – no field observations for this period.

Appendix B. Freshwater Sampling Details

Code	Parameters	Total
A	F	103
B	C,T,O,P,Tb,Rp,F	70
C	C,T,O,P,F	45
D	C,T,O,P,N,Ss,F	43
E	C,T,O,P,Rp,F	15
F	C,T,O,P,N,F	5
G	C,T,P,F	4
H	C,T,O,P,Tb,F	3
I	C,T,O,N,Ss,F	2
J	C,T,O,P	2
K	C,P,N,Ss	2
L	C,T,F	1
M	C,T,O,P,N,Ss	1
N	C,T,P,N	1
O	T,O,P,F	1
P	C,T,P,N,F	1
Q	C,T,O	1

Table B.1. Combinations of parameters measured in fresh waters for this study. Codes A to Q are used in Table B.2. Total refers to the number of times a combination was sampled over all dates and sites. F: flow, C: conductivity, T: temperature, O: dissolved oxygen, P: pH, Tb: turbidity, Rp: redox potential, N: nutrients (total N&P, NOx, soluble reactive P), Ss: suspended solids. Times when only flow was measured were typically records of no flow.

Date	Site																		
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19
				MC		AR		SC		PC DS	PC US	DC							
8/12/1998						O													
16/12/1998				C									C						
22/01/1999				J		C		C											
18/02/1999						D	A												
19/02/1999					D			A											
5/03/1999				C	C	C	A	A											
8/03/1999						C													
9/03/1999						C													
18/03/1999				I	I														
19/03/1999						D													
12/04/1999							A	A											
22/04/1999				C	C	C	A	A											
4/05/1999				A		A			Q										
18/05/1999				C	C	C	A			A		A							
15/06/1999					A	A	A	A		A		A							
22/06/1999			D	D	D	D													
23/06/1999		D																	
24/06/1999										D		A							
25/06/1999	D																		
19/07/1999		D	D	D	D	D													
20/07/1999	D									D									
15/08/1999				D	D	D													
16/08/1999		K	K				A			D									
26/09/1999									D										
27/09/1999				A	D	A		A											
29/09/1999	A			D		D		A					D						

2/10/1999		D	D											A					
3/10/1999										**D**									
28/10/1999	D	D	D					**A**		**D**				A					
29/10/1999				**D**	D	**D**	A				**D**								
9/11/1999										**A**									
15/11/1999	C	D	D	**D**	D	**M**	A	**A**		**D**	**D**	**A**							
16/11/1999						**J**													
13/12/1999	A				P	**N**		**A**											
14/12/1999										**A**									
15/12/1999		A	F	**F**										A					
17/12/1999					A				A	**A**	**F**	**A**							
7/02/2000	A	A	F	**A**	A	**F**	A	**A**		**A**	**A**	**A**							
9/03/2000				**A**		**C**				**A**			A						
10/03/2000								**A**		**A**									
19/03/2000						**C**													
20/03/2000		A	A	**A**			A	**A**											
8/04/2000										**A**									
22/04/2000	A	A	C	**A**	A	**C**		**A**	A		**C**								
23/04/2000										**C**									
30/05/2000	C	A	C	**C**	C		A	**A**		**C**	**C**								
31/05/2000				**C**		**C**													
11/06/2000								**A**		**L**									
3/07/2000				**G**		**G**		**A**		**G**									
22/08/2000										**A**									
3/09/2000										**A**									
29/09/2000	B	B	B	**B**	B	**B**		**B**	B										
1/10/2000										**B**	**B**	**B**							
6/11/2000							A												
7/12/2000	A	B	B	**B**	B	**B**	A	**B**	B					A					
8/12/2000										**A**		**A**							
17/01/2001				**A**		**B**		**A**		**A**		**A**							
15/02/2001				**A**		**B**		**A**		**A**		**A**							
18/03/2001				**A**				**A**		**A**	**B**	**A**							
20/03/2001						**B**													
30/03/2001				**A**				**A**				**A**							
24/04/2001								**C**		**C**	**C**	**C**							
25/04/2001				**C**		**C**													
30/05/2001				**C**		**C**		**C**		**C**	**C**	**C**							
9/07/2001				**H**				**H**		**H**		**C**							
10/07/2001						**C**													
11/07/2001	E	E			E		E		E		**E**				E	E			
16/08/2001				**G**			C												
22/08/2001				**B**		**B**		**B**											
24/08/2001										**B**	**B**	**B**							
4/10/2001	E	B	B	**B**	B	**B**	B	**B**	B						E		E	B	E
7/10/2001										**B**		**A**							
14/10/2001										**A**		**B**							
7/11/2001	B	B	B	**B**	B	**B**	B	**B**	B	**B**	**B**	**B**		E	E		B	B	E
9/12/2001										**A**		**A**							
10/12/2001				**B**		**B**		**B**											
12/12/2001										**B**		**B**							
19/01/2002				**B**		**B**		**B**		**B**		**B**							
13/02/2002										**A**		**A**							
18/02/2002				**B**		**B**		**B**		**B**	**B**	**B**							
19/02/2002										**B**									
Total	14	16	15	**38**	22	**40**	18	**33**	9	**39**	**14**	**24**	3	5	3	1	2	2	2

Table B.2. Dates, sites and water quality parameters measured during the study. Codes A through Q refer to combinations of parameters (see Table B.1). Columns in bold are sites for which results are examined in detail in Section 2.3. Collaborative data that were also used in Meyrick (1999) and Hermon (2002) is shown with shaded cells.

Catchment	No.	Site	Latitude	Longitude
Anglesea	**6**	**Anglesea River**	**38º23’44” S**	**144º10’59” E**
(Marshy Creek)	**4**	**Above Alcoa 1 (MC)**	**38º23’03” S**	**144º10’29” E**
	1	Bald Hills (MC)	38º22’33” S	144º08’13” E
	2	E trib1 (MC)	38º20’57” S	144º08’35” E
	3	Allardyce S (MC Bore)	38º22’12” S	144º09’49” E
	7	Gum Flat Rd (MC)	38º20’45” S	144º08’54” E
	13	Above Alcoa 2 (MC)	38º23’03” S	144º10’29” E
	14	E trib 2 (MC)	38º21’16” S	144º08’39” E
	15	Test Ground(MC)	38º20’14” S	144º04’56” E
	19	Headwaters (MC)	38º21’07” S	144º02’54” E
(Salt Creek)	**8**	**Salt Creek above Alcoa**	**38º23’38” S**	**144º08’56” E**
	5	Breakfast Creek (SC)	38º22’58 S	144º03’40” E
	9	Hughies Track (SC)	38º23’35” S	144º05’24” E
	16	Denham Trib 1 (SC)	38º23’40” S	144º04’51” E
	17	Denham Track (SC)	38º23’37” S	144º07’06” E
	18	Denham Trib 2 (SC)	38º23’39” S	144º05’23” E
Painkalac	**10**	**Mainstem DS**	**38º26’54” S**	**144º05’26” E**
	11	**Mainstem above Dam**	**38º26’10” S**	**144º02’47” E**
	12	**Distillery Creek DS**	**38º26’53” S**	**144º05’45” E**

Table B.3 Locations of freshwater sites. Major sites are in bold. Datum is WGS 1984.

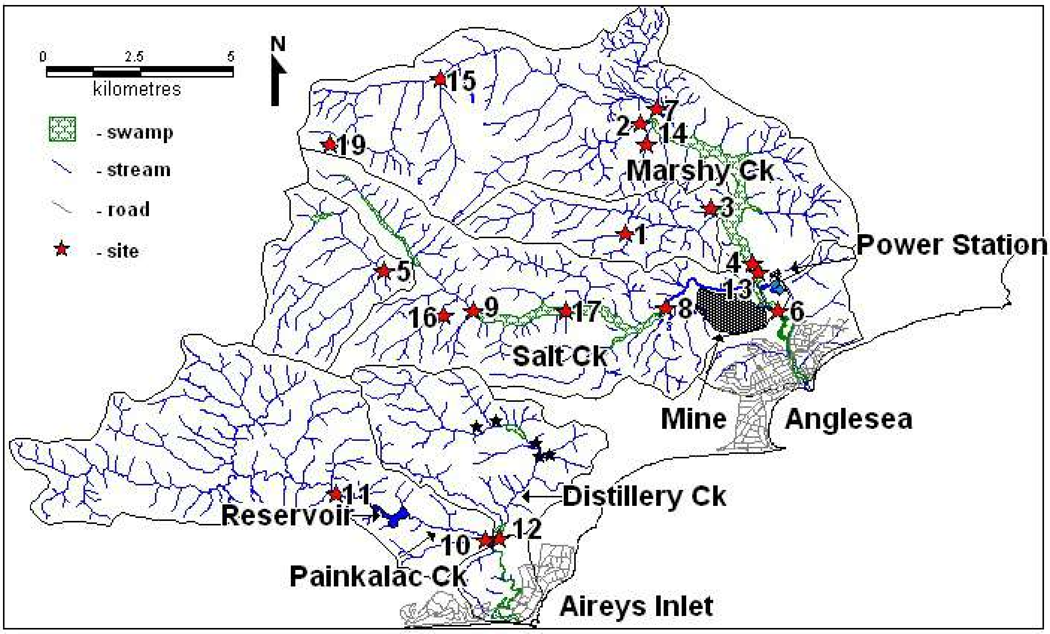

Figure B.1. Locations of freshwater sites as listed in Table B.3 (red stars). Black stars along Distillery Creek are locations of additional sites sampled to investigate sources of acidic flows (Appendix D).

Appendix C. Alcoa Monitoring Data – Methods and PCA

Methods

Since 1972, shortly after commissioning of the power plant, Alcoa have been monitoring water quality monthly at five sites in the lower Anglesea catchment; upstream, at mine and ash-pond discharges and surface waters at the extreme upstream end and in the middle of the estuary (Figure 3.1).

Section 3.2.2.d describes the site locations and temporal range of the Alcoa water quality dataset. There are omissions in the dataset, some presumably from a lack of water (at the upstream and mine-reclaim sites) and some for unknown reasons. Numbers of gaps in the data are given in Table C.1 below.

Variable	US	Ash ponds	Mine water	DS	mid-estuary	Total possible
pH	293	350	248	228	348	350
Cond.	264	319	217	288	319	319
Temp.	293	350	248	228	348	337
Total solids	293	350	248	288	348	350
Suspended solids	293	350	248	288	348	350
Turbidity	276	330	227	285	331	336
Colour	268	323	221	288	323	336
Al	221	276	176	276	265	276
Fe	249	306	206	276	293	350
Zn	221	276	179	276	265	276

Table C.1 Number of measurements by variable and site from Alcoa's water quality monitoring program, 1968-2002. US=Upstream, DS=Downstream. Total possible refers to the number of months in which each variable was sampled, dates of initial measurement varied between 1972 and 1979.

Site locations are shown in Table C.2. In about 1994, the Marshy Creek sampling site was moved slightly downstream to a section of creek more similar in appearance to other sites (M. Thorp, *pers. comm.*). The location of the current site is given in Table C.2.

Site	Latitude	Longitude	Elevation (m AHD)
Upstream	38°23’29”	144°10’35”	~5[a]
Ash Ponds	38°23’24”	144°10’46”	4.5[b]
Mine Water	38°23’40”	144°10’44”	7.7[b]
Downstream	38°23’44”	144°11’02”	1.35*
mid-estuary	38°23’56”	144°11’14”	1.35*

Table C.2. Locations of Alcoa water quality monitoring sites. a – estimation from topographic map, b – by differentially corrected GPS, *- estuarine sites

A NATA audit of the monitoring program led to an increase in the detection limits reported for some parameters in 1986. Old (pre-1986) and new detection limits are shown in Table C.3.

	S.S.	Col.	Al	Fe	Zn
Old d.l.	1	5	0.1	0.1	0.03
New d.l.	10	5	0.1	0.1	0.1

Table C.3. Old and new detection limits for variables where ‘below detect’ results were reported. All units for included values are mg/L, except colour (measured in platinum-cobalt colour units). Detection limits were not exceeded for pH, conductivity, temperature, total solids and turbidity (although one ‘out of range’ reading was recorded at the mine reclaim site on 22/7/74 - possibly >1000, this reading was deleted for analyses).

The occurrence of many readings under the detection limits of the analytical techniques needed to be addressed for the purposes of analysis. This aspect of the dataset is typical of much environmental pollution data, that are often censored at lesser concentrations by the detection limits of analytical methods and usually have ‘outliers’ at greater concentrations (Porter *et al.*, 1988). These characteristics must be considered in attempts to describe the data statistically and balanced with the assumptions of normal distribution of data that are associated with many statistical methods.

Analyses for this study were done using values lower than the NATA-revised detection limits because of the information (albeit less exact) contained in those values (Porter *et al.*, 1988). Unfortunately this was only possible for a subset of the data and a large amount was still presented as ‘<x’. This latter data were revised to the pre-NATA detection limit as a compromise approach

to dealing with the censored data. The approach was partially conservative, in that the ‘worst case’ scenario (i.e. values below the detection limit are close to the detection limit) was assessed but still attempted to retain as much information as possible by using the lower, pre-1986, detection limits despite potentially reduced accuracy at these smaller values. It must be noted, however, that use of substituted data for values below the detection limit can make a large difference when making parametric comparisons between times or sites when there is a large proportion of data below the detection limit (Helsel, 1990).

Links Between Water Quality Variables

The large amount of data from Alcoa was used to investigate overall variability in water quality at their five sites (Table C.2, Figure 3.1) and to identify links between the measures that are responsible for that variability. This was done using principal components analysis (PCA), based on linear correlations between data from all sites and times between March 1972 to October 1998 using the SYSTAT program (v9, SPSS Inc, 1998). Factor scores from the PCA were used to examine differences between sites and seasons.

In effect, the components resulting from the PCA represent the attributes of water quality (as measured by the ten recorded variables) that changed most between sites and dates. Examination of the relative contribution of each variable to the components provided information about correlations between variables in terms of their contribution to overall differences in water quality between natural water upstream of Alcoa, effluents from the mine and power station and surface water at two locations downstream of Alcoa with differing estuarine influences.

Not all variables were measured at each site and time, due to measurement of different variables starting between 1972 and 1979 as well as intermittent dry periods at the mine and upstream sites when sampling was impossible. As these missing values were unlikely to be randomly distributed through space and time, a pairwise, rather than listwise method was used so that, for cases where one or more variables was missing, the remaining data were used to calculate correlations for the PCA. Much of the data had a strong positive skew and so all variables were log(x+1) transformed before analysis to increase the likelihood of linear correlations between variables and of multivariate normality (Quinn & Keough, 2002).

Three principal components with an eigenvalue greater than one were identified in the PCA. These components were then orthogonally rotated using the varimax method resulting in components explaining 29.3%, 25.4% and 15.3% of the total variance, respectively. The associated factor loading matrix is shown in Table C.4.

	'Salinity': Component 1	**'Precipitates/Iron': Component 2**	**'Acid episodes': Component 3**
Total solids	**0.922**	-0.103	-0.082
Conductivity	**0.922**	-0.112	-0.072
pH *	**0.768**	0.095	**-0.351**
Temperature	**0.551**	0.100	0.059
Turbidity	0.104	**0.857**	0.013
Suspended solids *	**0.166**	**0.777**	0.068
Colour *	-0.091	**0.775**	**-0.270**
Iron *	**-0.479**	**0.723**	0.031
Zinc	0.024	0.083	**0.853**
Aluminium *	**-0.235**	**-0.178**	**0.763**

Table C.4. Rotated factor loading matrix from PCA of Alcoa data, 1972-1998. Bold values are component loadings that show significant correlation between the variable and the respective Component at the α=0.01 level (> 0.162, <-0.162) (Norman & Streiner, 1994). '*' denotes factorially complex variables.

These components represent three combinations of variables that are orthogonal (independent) and together account for 70% of the total variability in the data. Half of the variables are factorially complex (are significantly associated with more than one component) despite rotation to minimise this. Aluminium is associated with all components; however, along with zinc, it is a major contributor to Component three only. Concentration of suspended solids is positively associated with Components one and two while pH, colour and iron concentration are positively associated with one component and negatively associated with another although the pairs of components differ for each variable.

The first component was associated with total solids, conductivity, pH and temperature and, to a lesser degree, positively associated with suspended solids and negatively associated with iron and aluminium. Changes in scores for this component are consistent with increasing salinity, which is inherently associated with total solids and conductivity and also tends to increase pH and therefore reduce proportions of metals in solution and also potentially increase suspended solids. The association with higher temperatures may be due to the ash pond and mid-estuary sites, both of which were relatively warm and saline.

The second component was associated with colour, total iron concentrations, turbidity and suspended solids and, to a lesser degree, negatively associated with aluminium. The association between colour and iron was particularly strong at the upstream site, with high values of both during summer and autumn and with lower values in winter and spring.

The third component was associated with total aluminium and zinc concentrations and negatively associated, to a lesser degree, with pH and colour. The presence of clear, acidic waters with high dissolved metal concentrations would account for this association, which was thought to represent acidic events, similar to that of late 2000.

Scores for each of the three components were calculated for each site/time combination between 1979 and 1998 on which all variables were measured using factor coefficients and the log (x+1) transformed data. Results were plotted by date for each site (Figure C.1a-e) and as seasonal averages of each component by site (Figure C.2a-c). Inter-annual variability is most easily seen in Figure C.1a-e while general patterns by season and/or site are most clearly seen in Figure C.2a-c.

Scores for Component one show a clear seasonal influence at the upstream site with higher scores in the summer and autumn, times of typically low flow (Figure C.2a). Higher pH, temperature suspended solids and iron and lower aluminium are typical of this site at these times of year although there was a large amount of inter-annual variation in this seasonal peak, possibly related to times with no flow (Figure C.1a). The mine and ash pond showed a different, and less obvious seasonal pattern, with higher values in winter and spring (Figure C.2a,b). Of these two sites, scores for the ash pond were consistently higher, probably reflecting the higher pH and salt content of this water. An increasing trend in scores for this component was evident at the mine site (Figure C.1c). The downstream and mid-estuary sites both show a pattern between those of the discharges and the upstream waters with little seasonal change aside from a slight decrease in spring (Figure C.1d,e; Figure C.2a). Like the ash-pond scores, the consistently higher scores for the mid-estuary site, when compared to the site at the head of the estuary (‘downstream’), are most likely related to the influence of salt water, rather than the pH/suspended solids influence on the upstream site.

Scores for Component two also showed a strong seasonal influence at the upstream site that was moderated downstream with seasonal and inter-annual patterns similar to that of scores for Component one (Figure C.1a; Figure C.2b). Likely causes of these patterns are the high iron and suspended solids concentrations along with increased colour at times of low flow, as well as increased aluminium in winter and spring. Scores for the mine and ash-pond discharges again were relatively consistent throughout the year with the ash-pond scores consistently greater than those of the mine discharge (Figure C.2b). Reasons for the difference in scores between the discharges are unclear, as this pattern is not seen in any of the variables with significant factor loadings. Inter-annual variability of these factor scores appears greater at the ash pond site than at the mine site, a difference potentially associated with the composition of ash from different sections of coal (Alcoa, unpublished data). The downstream and mid-estuary sites show a similar, though increasingly attenuated, seasonal pattern to that of the upstream site (Figure C.2b). Of the variables associated with this component, this pattern is most obvious for suspended solids and, in mirror-image, for aluminium at the lower two sites. This pattern is consistent with increased phytoplankton abundance and decreased aluminium inputs in the warmer parts of the year.

From the factor loadings of Component three it would be expected that a seasonal pattern in factor scores for the upstream site would be obvious along with peaks of aluminium and zinc with acidic winter and spring flows. This is not the case, and in fact very little seasonal variation in scores of Component three is evident (Figure C.2c). This may be due to irregular and non-seasonal flows containing high concentrations of these metals to estuarine sites (Section 3.4.2) or may be related to outliers associated with high zinc concentrations in the mine discharge in the mid-1990s that affect scores for this component (Figure C.1c and caption). The influence of the outlying records from the mine discharge may explain the unexpectedly high standardised factor score coefficient for total solids.

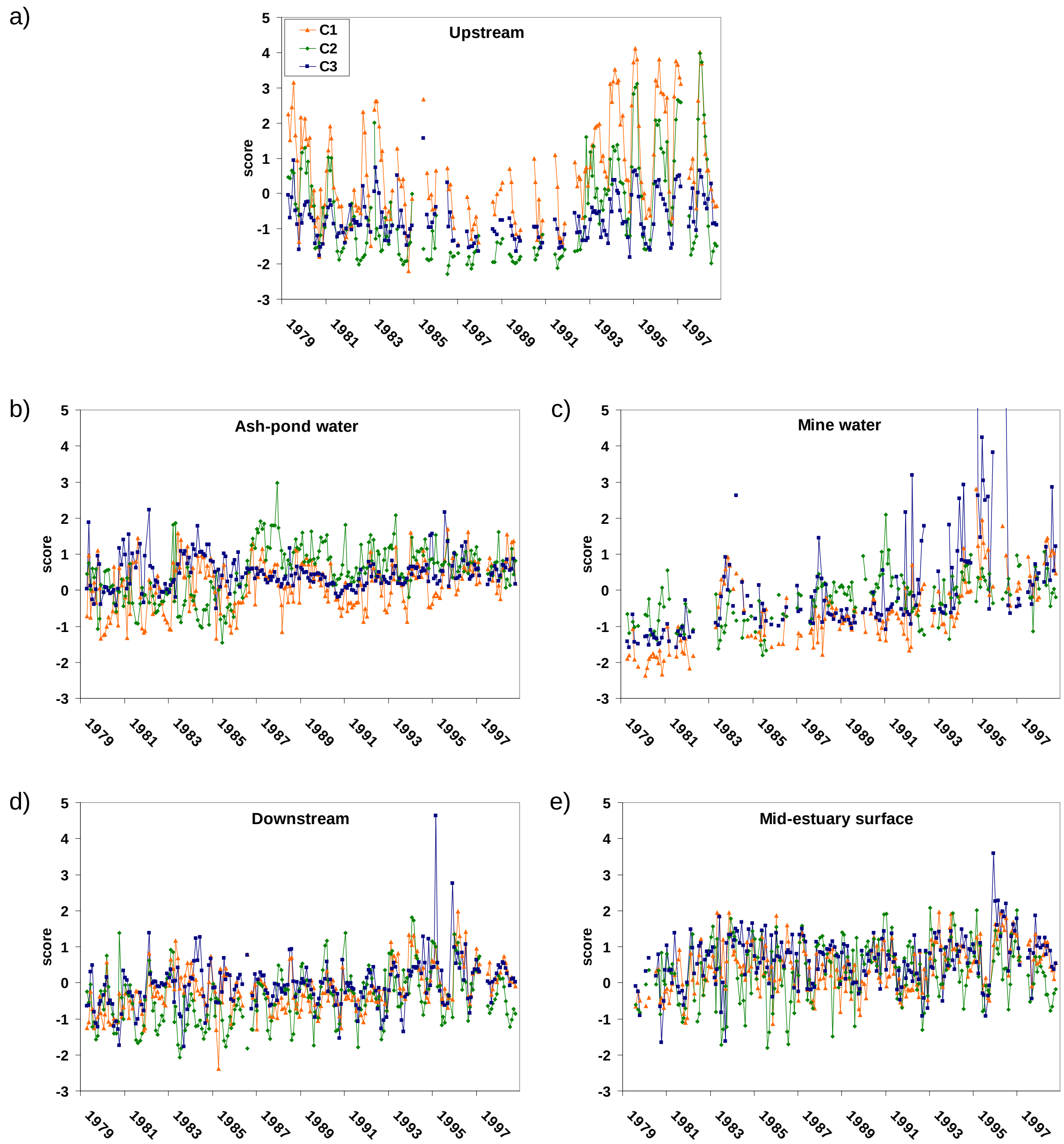

Figure C.1. Standardised scores for each of the three Components at each site in Alcoa's monitoring program, 1979-1998. Gaps indicate times when either no sampling was done or not all variables were sampled. Component 3 scores that are not shown in c) are 13.5, 5.8 and 5.2 on 9/3/95, 16/5/96 and 17/7/96 respectively.

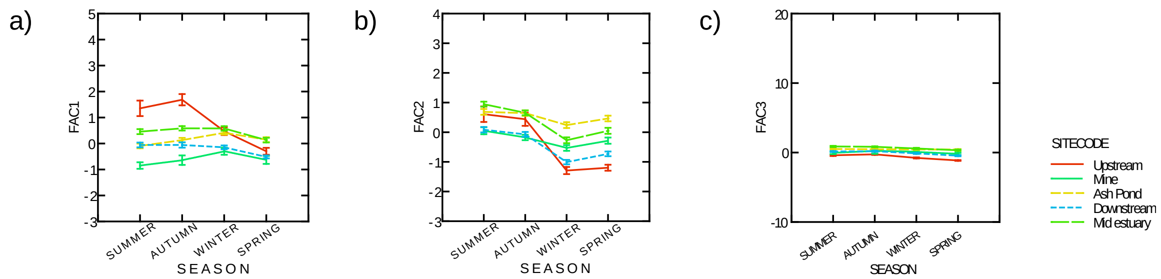

Figure C.2. Seasonal means (± s.e.) of standardised factor scores by site for a) Component1, b) Component 2 and c) Component 3. Data from 1979 to 1998 were used.

Despite a non-significant factor loading, the coefficient for total solids was 0.612 compared with 0.497 for zinc, the second highest for this component. The influence of salts associated with total solids on calculated values of this component may explain the higher scores that in summer and autumn as well as for the ash-pond, mid-estuary and downstream sites when compared to the mine and upstream sites. The prevalence of zinc concentrations below the detection limit may have also influenced calculation of this component. Interpretations of factor scores for Component 3 should therefore be treated with caution.

As can be seen in Figure C.1 a-e, Components one and two appear to be responsible for most of the within-site variability at the upstream and ash pond sites, while Component three is dominant at the mine site. A more equal influence between components is evident at the downstream and estuarine sites. Seasonal influences are strongest at the upstream site, weakest at the mine and ash pond sites and intermediate at the downstream and mid-estuary sites.

Use of the PCA to reduce the ten water quality variables to three components allowed an examination of links between variables and identification of potential processes that drive variability of water quality between sites and through time in the Anglesea system. Comparisons of scores for each component between seasons, years and sites further developed an understanding of the nature of the changes of each component in space (from natural and Alcoa sources upstream to estuarine waters). The results of the analysis, however, are specific to the data used and comparisons of water quality at Anglesea against published guidelines and other studies must be done using original variables as presented in Chapters 3 and 5.

Appendix D. Sources of Acidity

The acidic flows observed in the Anglesea and Distillery Creek catchments were unusual for the region and were a subject of much local interest following the estuarine acidification of 2000. At this time, a review of existing data was done by Alcoa (Gower, 2000) to assess the likelihood of groundwater extraction at Alcoa contributing to the generation of acidic waters in the catchment. This report concluded that, based on historic pH of the streams and the limited movement of waters between aquifers within the Eastern View Group (see section on geology below), the groundwater extraction did not impact on the water table at or near the surface to cause acidification. Hermon (2002) then investigated sources of acidity in the Anglesea catchment in late 2001 and concluded that there was potential for generation of acidity both in the peaty Quaternary sediments of the valley floors and from sulphidic sediments in the strata underlying the stream beds and outcropping in the rest of the catchment. In addition to the above, the wider spatial and temporal extent of sampling for this study provide further data of use in determining the sources of acid generation.

As has been discussed (Section 3.4.1), the natural waters in Marshy, Salt and Distillery Creeks (but not Painkalac Creek) were always acidic at times of high flows and mostly acidic at times of lower flows. The low $Cl^-:SO_4^{2-}$ values and high aluminium concentrations associated with the 2000 flow event from Salt Creek (Section 3.4.2) are strongly suggestive of an acid sulphate, rather than organic, origin for the strongly acidic slug of water. In addition, hydrologic conditions that existed in the previous years would lead to a build up of metals and acids in an area of suitable sediments. Following this event the buffering capacity of Salt Creek decreased and the ratio of $Cl^-:SO_4^{2-}$ increased; however, this ratio in Salt and Distillery Creeks remained well below that of Marshy Creek.

Additional sampling for this study was done in two areas outside of those sampled by Hermon or Alcoa – the Distillery Creek catchment and the Carlisle State Forest on the northern slopes of the Otway Range. The main reason for this sampling was for a comparison of water quality in other areas known to have similar peats and riparian vegetation to those of the waterways of the Anglesea catchment and that therefore had potential for similar pH characteristics. A short section on the geology of the area is also given as background information.

Geology

The Anglesea and Painkalac catchments are located on the western edge of the Torquay Basin and eastern edge of the Otway Ranges High. In the west the Painkalac runs through Lower Cretaceous sediments of the Otway Group while further east both waterways cut through various Tertiary sedimentary strata. Pleistocene and recent alluvial formations are associated with the creek and estuary beds (Abele, 1979) (Figure D.1, Table D.1).

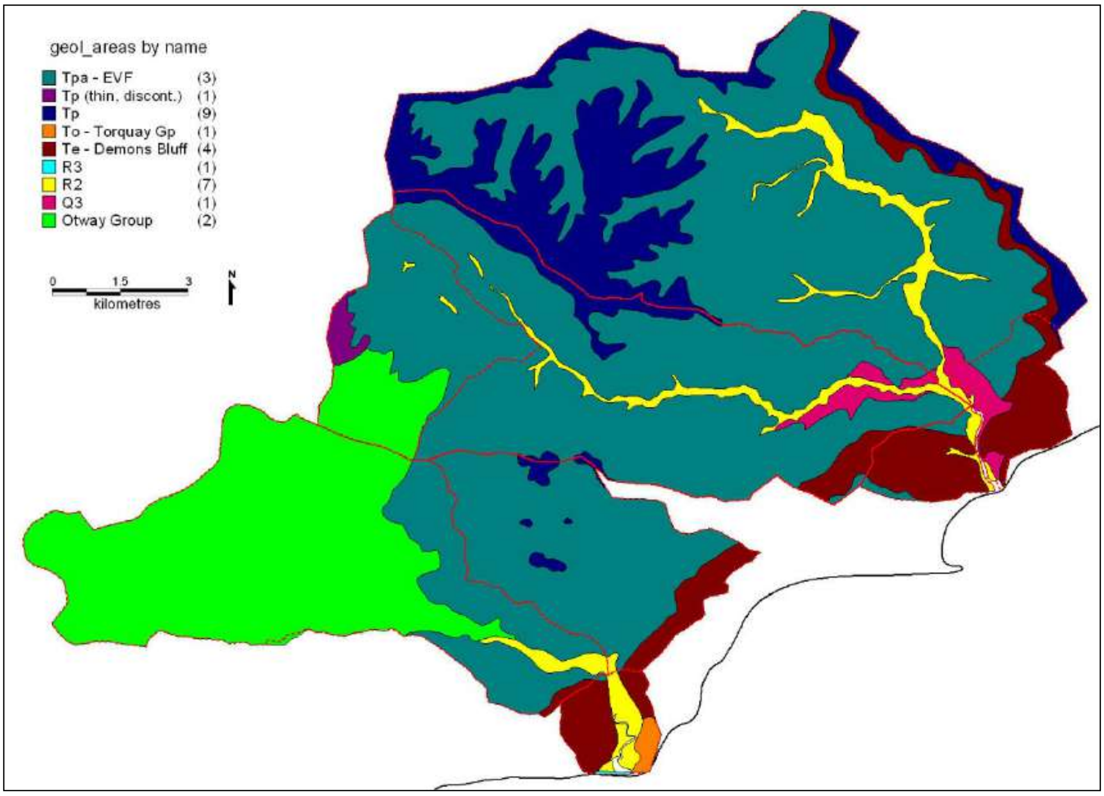

Figure D.1. Outcropping geological strata in the Anglesea and Painkalac catchments. Data from Abele (1979) and Douglas & Ferguson (1988). Catchment and sub-catchments are shown as red dashed lines.

Period	Epoch	Unit	Description
Quaternary	Recent	R3	Beach and dune sand. Estuarine sand, silt, clay
		R2	River alluvium. Lake and swamp deposits
	Pleistocene	Q3	High level alluvium and outwash deposits
Tertiary	Pliocene	Pliocene Sands (Tp)	Ferruginous, clayey quartz sand, gravel, sandy clay
	Late Oligocene/Miocene	Torquay Group (To)	Limestone, sandy and clayey limestone, marl, calcareous and clayey silt
	Oligocene/Eocene	Demons Bluff Group (Te)	Carbonaceous clayey silt to very fine sand with thin quartz sand and gravel beds
	Eocene/Palaeocene	Eastern View Group (Tpa)	Quartz sand and gravel, clayey silt, carbonaceous clay, brown coal
Lower Cretaceous/ Jurassic		Otway Group	Felspathic sandstone, mudstone, shale, with thin conglomerate and coal beds

Table D.1. Geologic units in the Anglesea and Painkalac catchments. Adapted from Abele (1979) with modifications from Holdgate *et al.* (2001).

Anglesea

The Anglesea River network cuts through geologic units from the uplifted Lower Cretaceous Otway Group in the far west to the Pliocene sediments capping the plateau and Pleistocene alluvial and outwash sediments near the town of Anglesea. The most extensive outcropping stratum is the Eastern View Group (EVG) comprising most of the slopes of the tributaries. This group is also coal bearing and is the stratum that is mined at Anglesea. The EVG is overlain by the Demons Bluff Group (DBG) and younger, unnamed Pliocene sands and gravels. The Pliocene sands are predominantly found in the northern part of the catchment and on top of the Bald Hills while the DBG outcrops around the town of Anglesea and in a strip along the eastern slope of the Marshy Creek valley. Units of the Otway Group only outcrop in the headwaters of Breakfast Creek (Abele, 1979; Holdgate *et al.*, 2001).

Quaternary alluvium and outwash deposits are situated on the eastern side of the estuary and along both creeks near the mine. Much of these deposits near the pre-diversion location of Salt Creek have been removed as overburden from the open cut. These sediments are located at a distinctly higher elevation than present river levels (Abele, 1979).

Fifty-six and 79 percent respectively of the lengths of Marshy Creek and Salt Creek above the mine comprise recent river alluvium and peat (Figure D.1). These deposits also extend up the lower reaches of smaller tributaries and are associated with a distinct floristic community, as described below.

Painkalac

The Painkalac catchment includes a much greater proportion of Otway Group outcrop than the Anglesea catchment. The edge of this outcrop runs approximately north-south on the eastern slopes of the valley of the mainstem of Painkalac Creek. The DBG outcrops on hills either side of the estuary and a small area of Torquay group overlies the DBG to the east of the creek mouth.

The surface units of the Distillery Creek sub-catchment are mostly of the EVG with a strip of the DBG outcropping in the south east and patches of Pliocene sands on ridge tops. The mainstem sub-catchment cuts through the Otway group with the EVG outcropping on eastern slopes and on either side of the floodplain upstream of the DBG outcrop. The floodplain is covered with recent alluvial sediments and is ~5km long and up to 600m wide. Table D.2 details proportions of outcrop in each sub-catchment.

Unit	%* Anglesea	% Marshy	% Salt + Breakfast	% Salt – Breakfast	% Breakfast	%* Painkalac	% mainstem	% Distillery
Otway Group	3.2	0.0	7.9	0.01	31	50	77	0.0
EVG	66	64	77	82	62	38	20	86
DBG	7.3	2.9	2.3	3.1	0.0	7.1	0.58	11
Torquay Group	0.0	0.0	0.0	0.0	0.0	0.76	0.0	0.0
Pliocene sands	17.	28	6.3	6.5	5.7	0.94	0.0	3.2
Quat. Alluv.	1.9	0.39	2.4	3.3	0.0	0.0	0.0	0.0
Recent alluv. & marine	4.3	4.7	3.9	5.0	0.58	3.0	2.0	0.44

Table D.2 Percentage outcrop of various geological units in Anglesea and Painkalac catchments and subcatchments. * Anglesea and Painkalac figures include all sub-catchments as well as the portion of the catchment draining directly to the estuary. Areas derived from 1:63360 map data in Abele (1979) and sub-catchment areas delineated using 1:25000 topographic maps with reference to (Douglas & Ferguson, 1988) for the far western portion of the Painkalac Creek catchment. Portions of the stratigraphy of the Torquay Group have recently been moved to the DBG (Holdgate *et al.*, 2001), this is not reflected in the figures above as the resolution of stratigraphy in the map was not sufficient to revise the areas for those Groups.

Distillery Creek - Upstream

On 1-2/10/2000 (the time of the first acidification of Anglesea estuary), six sites were sampled along Distillery Creek. These were the main site above the estuary, four sites along the creek between 2.3km and 4.9km upstream, and a small tributary at 2.4 km upstream (Table D.3, Figure B.1).

The boundaries of the lower, middle and upper Eastern View group all outcrop in the vicinity of the sites, although available information does not provide enough resolution to place the sites exactly within any of the strata. The small tributary that was sampled flows from a sub-catchment in which both the upper Eastern View Group and the Demons Bluff formation outcrop. A swamp, similar to those in the Anglesea catchment was located between the upper two sites and the lower three on Distillery Creek.

Site/Distance Upstream (km)	pH	Flow (m^3/s)	Comment
0.0 (DC)	3.65	?-little apparent	
2.3	4.07	0.0066	
2.6*	6.57	?-some	Tributary
3.0	3.25	0.0012	Below marsh
4.3	-	0.0	Above marsh/Below waterfall
4.9	5.36	0.0029	Above waterfall
PC US	6.54	0.172	
PC DS	6.38	0.0261	

Table D.3. Flow and pH of sites in the Painkalac and Distillery Creek catchments on 1,2/10/2000. DC: Distillery Creek above the estuary, PC US: Painkalac Creek above the dam and PC DS: Painkalac Creek above the estuary. '?' – flow observed but not measured. '*' – on small southern tributary of Distillery Creek. Locations are shown in Figure B.1.

The flows measured at these sites clearly show that, in this area, a large component of the overall flow was subsurface. This was particularly evident at the sites above and below the waterfall (no surface flow at downstream site). No overall pattern of pH was evident; however, the substantial decrease in pH at the below marsh site is consistent with both the possibility of acidification via organic processes in the marsh and/or by pyritic sediments in the hyporheic zone.

Carlisle State Forest

Six sites were sampled in the catchments of the Carlisle and Gellibrand Rivers in the Carlisle State Forest (38°40'S, 143°25'E) on 16/11/1999. The outcropping geological stratum in this area is the Wangerrip Group, an area of groundwater recharge, consisting of Palaeocene to early Oligocene marine sediments, including sandstone, conglomerate clay, mudstone and brown coal (Land Conservation Council (Victoria), 1976). Holdgate *et al*. (2001) have concluded that this group is essentially the same as the Eastern View group, and that the uplift of the older Otway group sediments occurred subsequently.

The mean pH at these sites was 4.1 and ranged from 3.7 to 4.6. These acidic flows associated with the corresponding strata on the opposite side of the Otway Range provide further support for the link between the Eastern View Group and naturally occurring acidic flows.

Likely Sources of Acidity

While the pH of Marshy, Salt and Distillery Creeks was consistent with acids generated in sulphidic sediments, humic/fulvic acid generation in the peaty sediments of the swamps may also be contributing to the acidity of the creeks. The higher titratable acidity, low chloride-sulphate ratios and Al concentrations in waters from Salt and Distillery Creeks suggest a greater contribution from sulphidic sediments in these sub-catchments.

The Eastern View Group outcrops least in the Painkalac Creek mainstem and Breakfast Creek sub-catchments, both of which include the portions of the EVG that directly overly the Otway Group (Table D.2, Figure D.1). Discharges from these sub-catchments had neutral pH during most the study period, indicating less acid generation in these substrates than those of other sub-catchments.

Two possible reasons for the stronger acidity of flows in Distillery and Salt Creeks than that of Marshy Creek relate to the finer scale geology of the area. Both the Distillery and Salt Creek sub-catchments are in an area in which the lower Eastern View Group outcrops and which is also the recharge zone for the aquifer associated with this stratum. There are therefore possibilities of a greater acid generating potential due to the nature of this stratum, a greater frequency of oxidation and reduction of pyrites due to increased fluctuation of groundwater levels in a recharge zone, or both. The unusually low rainfall in the three years prior to this study may have exacerbated the latter phenomenon.

Despite the above possibilities, the swamp deposits in the Anglesea catchment and Distillery Creek sub-catchment are of a similar age and composition to others in central Victoria that are known to produce sulphuric acid (Abele, 1979; Rampant *et al.*, 2003). Neither should production of acids via bacterial humic/fulvic pathways be discounted, given the apparently suitable conditions in the swamps and the highly acidic peats found at other locations on the coastal plans of western Victoria (Land Conservation Council (Victoria), 1976). It may be that these pathways account for the relatively constant acidity in the creeks which is supplemented intermittently by acid sulphate sources.

Further research in the form of another Alcoa-sponsored Deakin University PhD project is currently underway to identify sources of acidification and potential effects of groundwater extraction in the Anglesea catchment. In addition, detailed analysis of data collected at all freshwater sites for this study, along with existing geological data would provide a more spatially and temporally extensive assessment of the dynamics of the period leading up to, during and following a large flow of acidic water with large concentrations of metals.

Appendix E. Acid/Metal Events: Anglesea Estuary

The events in late 2000 were associated with first substantial flows from Salt Creek for at least 2 years. A fish kill and precipitation of a milky white substance that caused the estuary to have a vivid aqua-blue colour were observed at this time (Figure E.1). Both events recorded in 2001 were associated with floods.

Figure E.1. Anglesea River mid-estuary, 30/9/00, showing blue colouration during an acid-flow event.

The first estuarine acidification was associated with the first substantial flows from Salt Creek during the study period (Figure 3.7) and, based on rainfall, for some years. Flows on 12 and 13/9/2000 were 0.162m^3/s and 0.185m^3/s, respectively, in contrast to the preceding 22 months at least, during which the maximum flow had been 0.006m^3/s. These flows continued for five days and then gradually decreased with flow from Marshy Creek becoming dominant from 25/9/2000. Flow from both creeks increased with a flood on 26/10/2000, following which Marshy Creek subsided faster than Salt Creek. Colouration of the estuary was again observed on 15/11/2000. The two events in 2001 did not result in such vivid colouration, largely because the low pH, fresh water flushed all salt water from the estuary and neutralisation occurred in Bass Strait (Section 5.3.5).

In 2000, due to public concerns about the colour of the estuary, and the prohibition of swimming due to low pH, a period of intense sampling on behalf of Alcoa, the Environment Protection Authority and this study followed the first estuarine acidification event and included three measurements of Cl^- and SO_4^{2-} concentrations. Results from this period are shown in Table E.1.

Measure	Marshy Ck			Salt Ck			Distillery Ck		
	mean	s.e.	*n*	mean	s.e.	*n*	mean	s.e.	*n*
pH	3.8	0.092	5	3.9	0.019	6	3.7	0.005	2
Al	0.94	0.33	3	89	2.9	4	58	N/A	1
Zn	<0.1	N/A	3	1.6	0.067	3	2.4	N/A	1
SO_4^{2-}	61	N/A	1	1000	N/A	1	580	N/A	1
Cl^-	310	N/A	1	94	N/A	1	95	N/A	1
$Cl^-:SO_4^{2}$	5.1	N/A	1	0.094	N/A	1	0.16	N/A	1

Table E.1. Freshwater results from 13/9/2000 to 1/10/2000. All units except for pH are mg/L. Data from Alcoa and Environment Protection Authority (Gower, 2000) except for some of the pH measurements. N/A – not applicable. ANZECC and ARMCANZ (2000) guidelines are: 6.5<pH<8.5 – default values to be used where site specific information is lacking, Al < 0.055mg/L – a moderate reliability value for freshwaters with pH<6.5 and Zn < 0.0080mg/L –for slightly to moderately disturbed freshwater systems.

While pH was low in all streams, the high concentrations of aluminium and zinc, along with the low $Cl^-:SO_4^{2-}$ ratios in Salt and Distillery Creeks are very suggestive of acid generation due to oxidation of sulphidic soils (van Breemen, 1992). These characteristics were still evident in Salt Creek on 6/11/2000, when Alcoa results show a $Cl^-:SO_4^{2-}$ ratio of 0.20. This ratio had increased to 0.26 by July 2001 and to 0.30 by December 2001 (Hermon, 2002), very much lower than the ratio of 21.5 found in sea water and ratios >1 typical of inland waters (Hart & McKelvie, 1986; Carter, 1994).

Along with this increase in $Cl^-:SO_4^{2-}$, concentrations of aluminium and zinc decreased from the time of the first flow in Salt Creek (e.g. Figure E.2 for aluminium). It appears that there was a consistent decrease in concentration in the Salt Creek sites with time. Dilution and neutralisation by proportionally increased Marshy Creek and ash-pond inputs appear to have reduced concentrations in the Anglesea River sites prior to the flood of 26/10/2000.

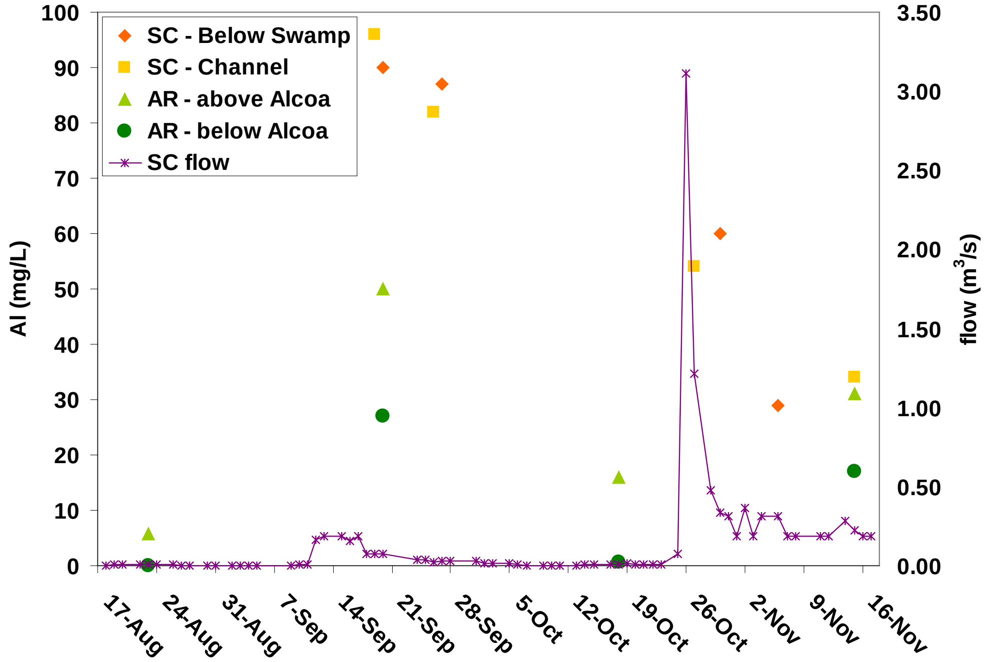

Figure E.2. Aluminium concentrations during the 2000 acidification event at four locations from the bottom of Salt Creek (SC) through the mine diversion channel to above the Anglesea River (AR) estuary. Data from Alcoa and Environment Protection Authority.

Relative 'strength' of (or titratable) acidity of waters from Marshy Creek and Salt Creek was determined on two dates by measuring its pH when mixed with increasing proportions of seawater (Figure E.3, Figure E.4). On 7 December 2000, the waters of Salt Creek had a clearly greater buffering capacity than those of Marshy Creek, despite having a higher pH. By 25 April 2001 the difference in buffering capacity was much reduced, but Salt Creek waters still had a greater buffering capacity. This is of particular relevance in the context of estuarine acidification events that occurred in that year (Section 5.3.5).

Figure E.3 also shows that waters from the ash pond have a high buffering capacity, close to that of seawater. This is consistent with the large increase in pH that is typical in the Anglesea River with the addition of ash-pond water and points to the potential for precipitation of metals downstream of this input. In the estuary, the results of titrations of Marshy Creek water and Salt Creek waters from April 2001 suggest that a pH of 7 would have been reached at a salinity of between 15 and 25, in contrast with a salinity of ~30 for the first flows from Salt Creek (Figure E.3, Figure E.4).

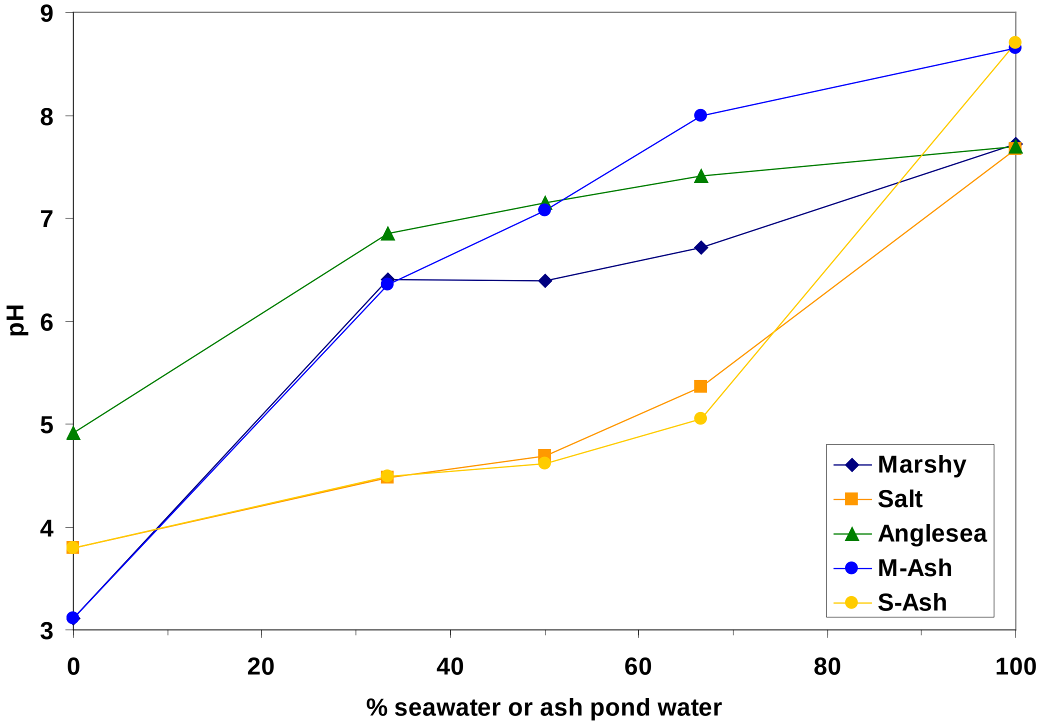

Figure E.3. pH titrations of freshwater with seawater and ash-pond effluent. M-Ash: Marshy Creek with ash pond, S-Ash: Salt Creek with ash pond. Samples collected 7/12/2000.

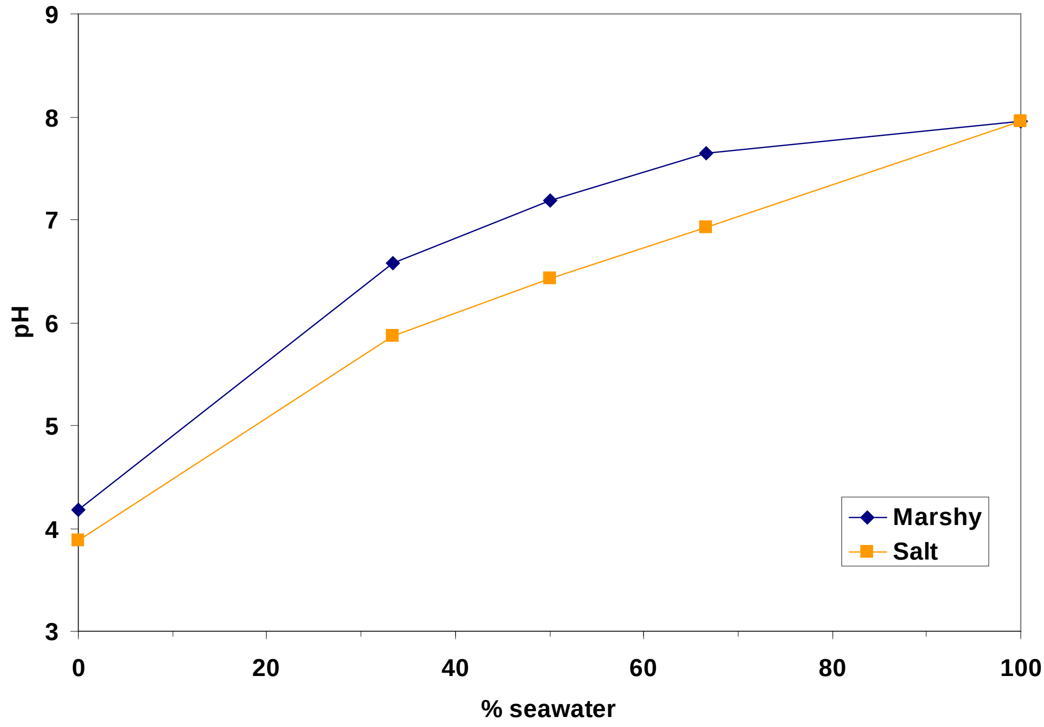

Figure E.4. pH titrations of freshwater with seawater. Samples collected 25/4/2001, in the tail of the largest flood during the study period.

The gradual decreases in metal and sulphate concentrations following the initial flow of Salt Creek are consistent with a 'slug' of ions released over time by a wetting and drying cycle of sulphidic sediments in the Salt Creek catchment that was not released until the first substantial flow in this sub-catchment. This mechanism is discussed by Hermon (2002) and is similar to observed mechanisms for acid sulphate soils in northern New South Wales (Callinan *et al.*, 1992; Wilson *et al.*, 1999) and the Northern Territory (Hart *et al.*, 1987). It is important to note that this event may not necessarily be caused by a large rain event but is dependent on the positioning of the water table in relation to the cumulative rainfall and previous periods of flushing and oxidation of soils (Wilson *et al.*, 1999).

Metals

Alcoa's long-term data (Appendix D) suggest that, while the very high aluminium concentrations recorded in 2000 are unlikely to have occurred since 1979, there have been many lesser events where significant concentrations of metals have entered the estuary. Figure E.5 illustrates the seasonal pattern of aluminium concentrations above and below Alcoa as well as large inter-annual variability. The large concentrations of aluminium in 1982 and 1983 coincide with the last sustained period of below average rainfall prior to the period preceding this study (Section 3.1, Figure 3.3), adding support to the proposed mechanism of ion generation outlined above. The peak in 1982 also coincided with the onset of substantial (>0.01m3/s) flows, which began in Salt Creek in the last week of August, two weeks prior to the peak concentration for that year. Flow data were not available for subsequent years.

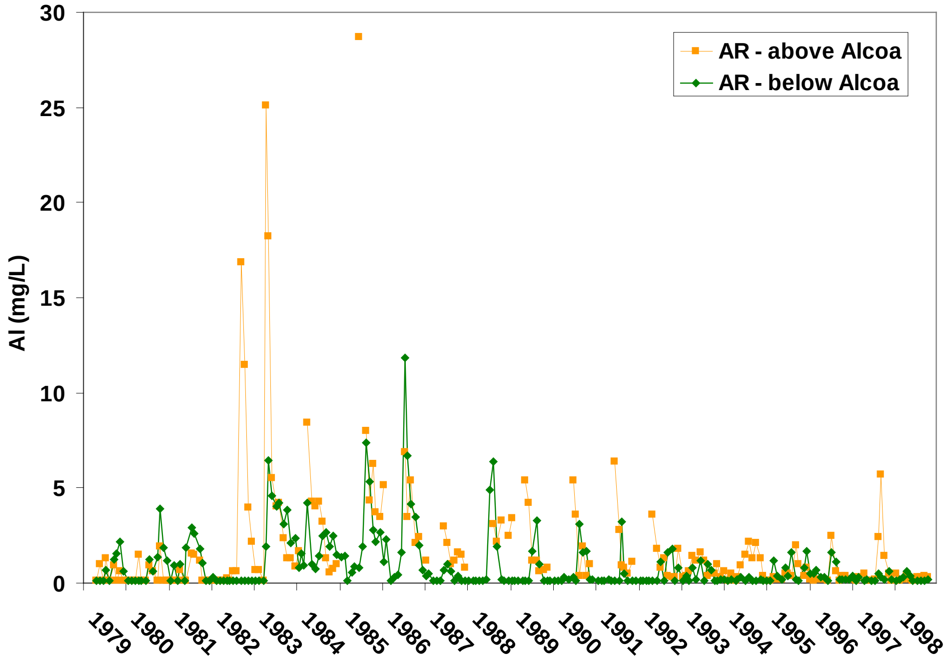

Figure E.5. Historic aluminium concentrations in the Anglesea River (AR) above and below Alcoa's operations, 1972-1998. Note: ANZECC guideline for Al concentrations in freshwaters below a pH of 6.5 is 0.055mg/L (ANZECC & ARMCANZ, 2000). Data from Alcoa. Connected points indicate sampling in successive months.

Given the greatly increased mobility of metals in acidic waters, the interaction of flow, buffering capacity and pH between natural sources and Alcoa inputs is critical to the movement of metals from the catchment into either the wetlands downstream of the Ash-pond inflow if neutralised at that point, or into the estuary. Similarly, these components are critical to the fate of those metals that enter the estuary.

The work of Meyrick (1999) demonstrated that at a time of low flow (during which Salt Creek was not flowing at all) the neutralisation of natural waters by Alcoa inputs resulted in decreased total concentrations of aluminium, iron and manganese. At this time, however, a small increase in zinc was seen, attributable to the release of mine waters (Table E.2). The proportion of particulate to dissolved components of all these metals was increased through Alcoa while bioavailability of the dissolved component, as measured by lability in diffusive gels, remained 100% for aluminium and increased from 30%, 12% and 90% to 100% for iron, manganese and zinc respectively.

Site/	Al		Fe		Mn		Zn	
Measure	**Total**	**DGT**	**Total**	**DGT**	**Total**	**DGT**	**Total**	**DGT**
MC	0.20	0.126	88.5	1.740	0.45	0.051	0.0060	0.0054
AR	0.020	0.0019	1.3	0.011	0.056	0.016	0.010	0.0060
MC:AR	*10*	*66.3*	*65.4*	*158*	*8.04*	*3.19*	*0.6*	*0.9*
ANZECC	0.055[A]	-	0.30[B]	-	1.90[C]	-	0.0080[C]	-

Table E.2. Metal concentrations (mg/L) for total and diffusive gel (DGT) labile species above and below Alcoa's operations on 15-16/8/1999 (Meyrick, 1999). MC: Marshy Creek above Alcoa and AR: Anglesea River above estuary. The ANZECC row refers to ANZECC and ARMCANZ (2000) guidelines. [A] – moderate reliability value for freshwaters with pH<6.5, [B]- Canadian guideline, indicative only, [C] – guidelines for slightly to moderately disturbed freshwater systems.

Table E.2 shows that despite the increase in bioavailable species as a proportion of dissolved metals, absolute concentrations of labile Al and Fe decreased proportionally more than total metals did at the downstream site. Labile Mn increased as a proportion of total Mn, while the proportion of bioavailable Zn species was essentially unchanged between sites. When compared to the ANZECC guidelines it appears that Al and possibly Fe were of concern biologically.

Historic metal concentrations

Long-term data from Alcoa's monitoring sites suggest that concentrations of metals, particularly aluminium, are very likely to be influencing the biota of the system, with concentrations usually many times higher than the ANZECC guideline (Figure E.6) and, at the peak of the September 2000 event, up to 1750 times the guideline. A seasonal pattern is evident, with lower concentrations of aluminium at the upstream site in summer than in other seasons and greatest variability in autumn. This pattern is very likely to have been influenced by flow and periods of neutral pH in upstream waters during summer. The pattern is reflected downstream of Alcoa and into estuarine waters but at lower concentrations than at the upstream site.

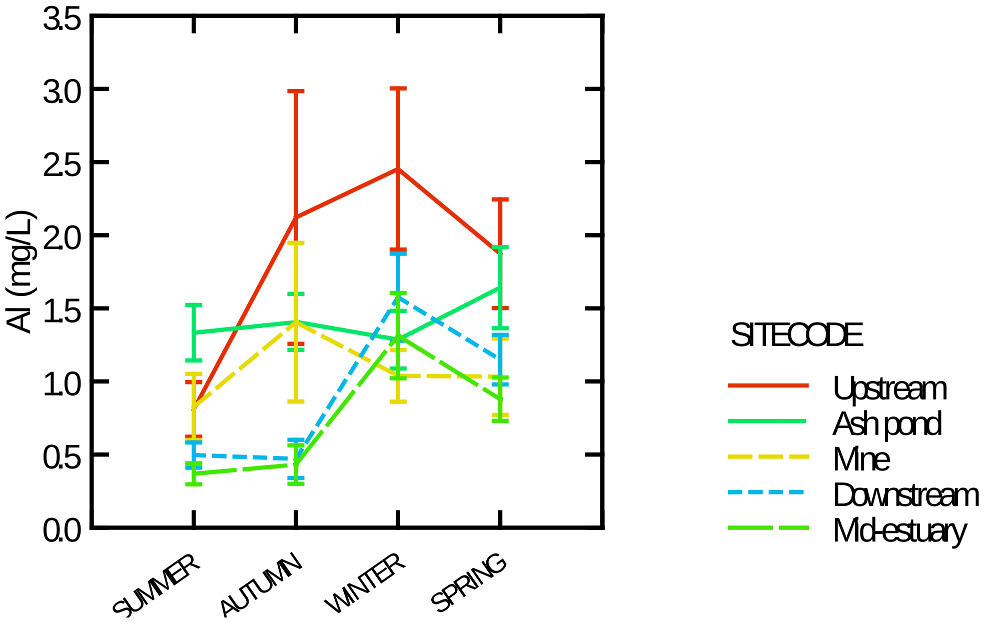

Figure E.6. Mean (± s.e.) total aluminium concentrations from Alcoa's monitoring program by season, 1979 to 1998. Values below detection limits were replaced by 0.1mg/L.

The mine and ash-pond inputs contained substantial, but generally lower concentrations of aluminium than the upstream site. That the Alcoa inputs are not reflected in downstream concentrations in summer may be related to the volume of flow or a greater tendency for the species of aluminium from the discharges to be removed from the water column downstream.

It is more difficult to assess the other two metals measured in Alcoa's monitoring against water quality guidelines. The guideline value for iron is indicative only as iron is an essential component of biological systems and values of concern are likely to vary between systems. Despite this, it is likely that concentrations of iron in the Anglesea system have affected biota upstream of Alcoa, as only the lowest concentrations from 1972 onwards were below the Canadian guideline and 25% of the time concentrations were greater than 70 times that guideline. Iron concentrations at long-term sites are shown in Figure E.7.

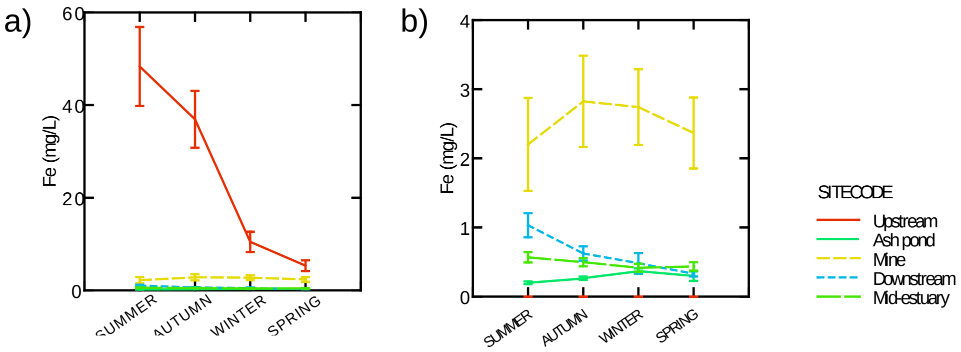

Figure E.7. Mean (± s.e.) total iron concentrations from Alcoa's monitoring program by season, 1972 to 1998. Values below detection limits were replaced by 0.1mg/L. a) includes the upstream site, b) is the same data with a smaller range on the y-axis and the upstream site excluded.

The upstream site had considerably greater concentrations of iron than any of the other Alcoa sites (Figure E.7a). This was particularly true in summer and autumn, a pattern suggesting that high concentrations of this metal are not directly associated with acidic flows. This is further illustrated in Figure E.8, showing that lower concentrations of iron at the upstream site only occurred at low pH and peak concentrations occurred only at more neutral pH (note log scale on y-axis). More recent data from Meyrick (1999) and Hermon (2002) showed no clear pattern in proportion of dissolved iron with pH.

The mine discharge also contributed iron to the system (Figure E.7b) while concentrations of iron from the ash-pond discharge were consistently low. Seasonal patterns of iron concentrations downstream broadly followed those of the upstream site but at much lesser concentrations.

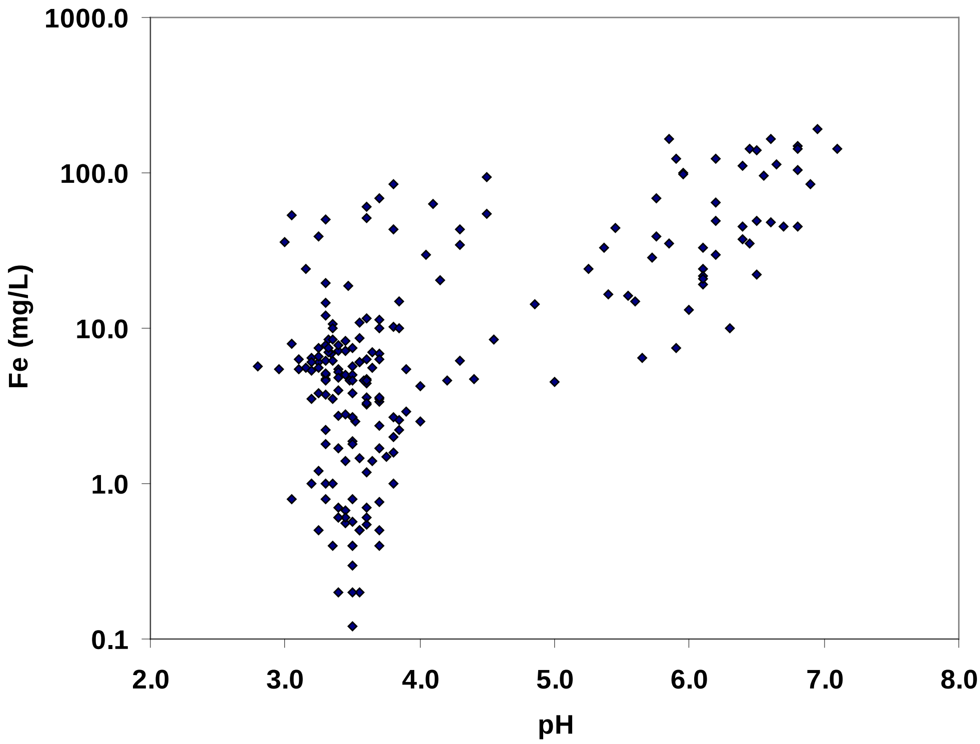

Figure E.8. Total iron concentrations vs. pH at Alcoa's upstream monitoring site 1982 to 1998.

The detection limit used for analysis of zinc in Alcoa's monitoring was initially 3.75, then 12.5 times the current guideline and over 80% of measurements at the upstream Alcoa site were below that limit. It is therefore difficult to make comparisons between the guideline and environmental concentrations but it is clear that, unlike aluminium and iron, natural concentrations of zinc in the system were usually no more than an order of magnitude greater than the ANZECC guideline of 0.008mg/L.

Figure E.9 shows that the mine discharge has had relatively high zinc concentrations during times of flow, while concentrations of zinc in the ash pond discharge were only slightly greater than those upstream. Temporal patterns of zinc concentration in downstream waters were similar to those seen for aluminium. During this study, concentrations of zinc were elevated in natural waters in late 1999 (Table E.2) and were slightly higher than historical levels at concentrations between 0.1mg/L and 0.3mg/L during winter and spring of 2001.

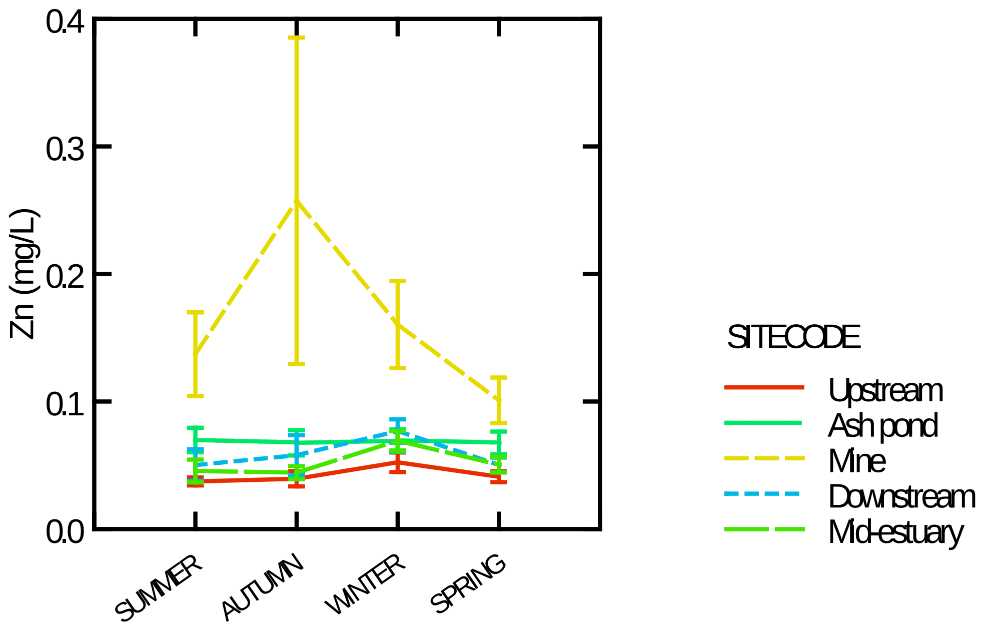

Figure E.9. Mean (± s.e.) total zinc concentrations from Alcoa's monitoring program by season, 1979 to 1998. Values below detection limits were replaced by 0.03mg/L, the lower of the two detection limits for the data.

In 1981, Atkins & Bourne (1983) also sampled fresh waters, estuarine sediments and estuarine biota for metals. Fresh waters were sampled for Al, Fe and Zn at Salt and Marshy Creeks, the ashponds and mine-reclaim ponds and the swamp below Alcoa in February, March and April. Their results were largely consistent with the results from Alcoa's monitoring program at this time of year, with concentrations of aluminium and zinc at all sites and times being less than their respective detection limits of 1.0 and 0.1 mg/L. High concentrations of iron at the ash pond were the only unusual results for this metal (Figure E.10), all three concentrations being equivalent to the upper 1% from Alcoa's monitoring program and considerably greater than Alcoa's data from the licence point in those months (1.5, 1.2, 1.2mg/L vs. 0.3, 0.3, <0.1mg/L). This difference was unexpected, given the short distance between sampling sites and it may have been that dilution occurred between the number 2 ashpond overflow where Atkins and Bourne sampled and the licensing point 450m downstream. Markedly greater concentrations at Marshy Creek than at Salt Creek were consistent with observations during this study, where iron flocs and/or bacterial colonies during summer/autumn were observed primarily in Marshy Creek.

That the presence of high concentrations of iron, possibly other metals and a low pH in waters of the Anglesea catchment are likely to be a long-term, natural phenomenon is also reflected in 19th century newspaper reports of iron-rich, 'chealate' mineral waters at Anglesea (Cecil & Carr, 1989).

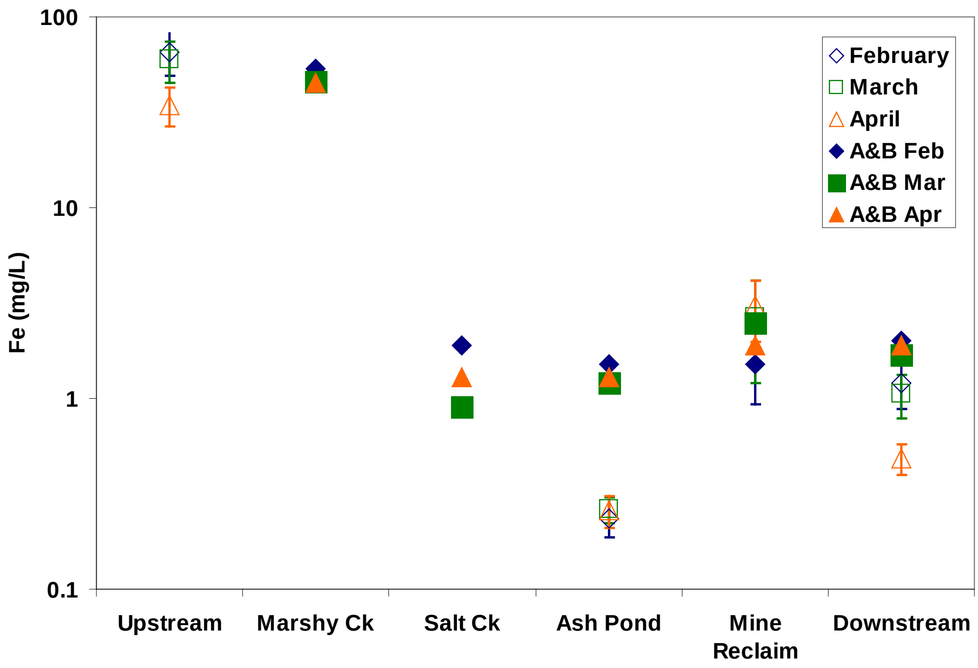

Figure E.10. Long-term monthly mean (±s.e.) concentrations of iron from Alcoa monitoring and concentrations recorded by Atkins & Bourne (1983) at fresh water sites.

Appendix F. Estuarine Physico-chemical Sampling

Sites

Locations of sites used for water quality sampling in Anglesea and Painkalac estuaries are given in Table F.1. Site 5 in Painkalac estuary was not always accessible and on seven of sixteen occasions it was necessary to sample ~200m downstream at Site 5(A). Dates on which this occurred are shown in Table F.4.

Estuary	Site	Description	Latitude	Longitude
Anglesea	1	Lower-estuary opposite jetty	38°24’36” S	144°11’16” E
	2	North of road bridge	38°24’24” S	144°11’10” E
	3	Mid estuary (N of foot bridge)	38°24’07” S	144°11’09” E
	4	Upper estuary, major bend	38°23’49” S	144°11’14” E
	5	Head of estuary	38°23’48” S	144°11’02” E
	7	Back channels (Coogoorah Park)	38°23’58” S	144°11’03” E
Painkalac	1	Lower estuary	38°28’03” S	144°05’49” E
	2	Upstream of road bridge	38°27’52” S	144°05’43” E
	3	Mid-estuary	38°27’41” S	144°06’05” E
	4	Bend on W floodplain	38°27’30” S	144°05’50” E
	5	Near head of estuary	38°27’05” S	144°05’47” E
	5A	185m downstream Site 5	38°27’10” S	144°05’46” E

Table F.1 Locations of longitudinal estuary sites. Datum is WGS 1984.

Estuary	Site	Estuarine Section	Distance upstream from mouth km	Proportion
Anglesea	1	Lower	0.67	0.26
	2		1.08	0.42
	3	Mid	1.66	0.64

	4		2.26	0.87
	5	Upper	2.60	1.00
	CP	CP	2.10	0.81
Painkalac	1	Lower	0.47	0.12
	2		1.03	0.27
	3	Mid	1.79	0.47
	4		2.63	0.69
	5	Upper	3.52	0.93
	5A	Upper	3.33	0.88

Table F.2. Locations of sites along Anglesea and Painkalac estuaries.

Water Quality Sampling Dates

Date	1	2	3	4	5	7	Date Total
8/12/1998	5	8	9	7	8		37
22/01/1999	4	5	6	5	4	3	27
18/02/1999	4	6	6	5	5	4	30
5/03/1999	4	5	5	5	5	3	27
8/03/1999	4	5	6	4	5	4	28
9/03/1999	4	6	5	5	5	4	29
19/03/1999	4	6	6	5	4	4	29
22/04/1999	4	6	5	5	5	4	29
3/05/1999		4	6				10
18/05/1999	5	6	6	6	6	5	34
22/06/1999	5	4	5	6	5	4	29
19/07/1999	4	6	6	5	5	4	30
15/08/1999	4	5	6	5	5	4	29
28/09/1999	4	6	6	6	5	4	31
29/10/1999	5	6	6	6	5	4	32
16/11/1999	5	6	6	6	5	8	36
13/12/1999	5	6	6	6	5	4	32
15/12/1999	4	6	6	5	5	4	30
18/12/1999	4	6	6	6	5	4	31
8/02/2000	4	7	6	6	5	4	32
9/03/2000	5	6	6	5	5	4	31
19/03/2000	2	2	2	2	2		10
8/04/2000		7					7
22/04/2000	5	6	6	6	5	4	32
31/05/2000	5	6	7	6	6	5	35
11/06/2000		5	5		3		13
3/07/2000	4	5	6	5	5	4	29
22/08/2000		5					5
15/09/2000	4		6		5		15
30/09/2000	4	7	7	6	6	5	35
7/12/2000	2	4	4	4	4	3	21
18/12/2000		4					4
17/01/2001	3	4	4	4	4	3	22
15/02/2001	4	5					9
20/03/2001	4	5	5	5	4	3	26

29/03/2001	4	6					10
25/04/2001	3	5	5	4	4	3	24
26/04/2001	3	5					8
1/06/2001	3	4	4	3	3	1	18
10/07/2001	3	5	4	4	3	3	22
22/08/2001	4	5	6	5	5	4	29
7/10/2001		4					4
14/10/2001		4					4
8/11/2001	3	4	5	5	4	3	24
10/12/2001	3	4	4	4	3	2	20
19/01/2002	4	4	4	6	4	3	25
16/02/2002	4	5	4	5	3	3	24
Site Total	**158**	**241**	**213**	**183**	**175**	**128**	**1098**

Table F.3. Number of depths sampled at sites in Anglesea estuary by date.

Date	1	2	3	4	5	Date Total
12/03/1999	6	6	6	5	4	27
18/03/1999		2				2
22/04/1999		6				6
18/05/1999		5				5
24/06/1999	7	7	7	6	7	34
20/07/1999		5				5
16/08/1999		6				6
3/10/1999	6	6	6	6	7	31
28/10/1999		6				6
15/11/1999		6				6
14/12/1999	8	6	6	5	6	31
16/12/1999	6	6	6	5	6	29
8/02/2000		6				6
10/03/2000		5				5
19/03/2000		4	5	5	*4*	18
20/03/2000	6	4				10
23/04/2000		5				5
30/05/2000		5				5
11/06/2000		5				5
3/07/2000		6				6
22/08/2000		6				6
1/10/2000	8	7	7	6	6	34
10/12/2000	7	6	6	5	6	30
18/12/2000		5				5
16/01/2001	7	5	6	5	5	28
16/02/2001	4	4				8
19/03/2001	5	5	5	4	*4*	23
28/03/2001		4				4
26/04/2001	7	4	4	4	*3*	22
27/04/2001	7	7				14
31/05/2001		5	5	5	*7*	22
4/06/2001	6	5				11
12/07/2001	6	6	5	4	*8*	29
23/08/2001	6	6	5	4	*7*	28
7/10/2001		6				6
14/10/2001	7	6				13
9/11/2001	6	5				11
12/12/2001	5	5	4	4	*7*	25

18/01/2002	5	5	5			15
19/02/2002	7	6	6	6	6	31
Site Total	**132**	**215**	**94**	**79**	**93**	**613**

Table F.4. Number of depths sampled at sites in Painkalac estuary by date. Italicised text for Site 5 indicates that samples were taken at Site 5(A) as Site 5 was inaccessible.

The sites were spread approximately equally between the mouths and heads of the estuaries. Distances upstream in km and proportion of the length of the estuary are shown in Table F.2.

Table F.5 details locations of lower estuarine sites that were used in sedimentation, seagrass and decomposition studies towards the end of the study period.

Estuary	Site	Description	Latitude	Longitude
Anglesea	1	N of jetty, W bank	38°24’40.9” S	144°11’10.6” E
	2	Opp jetty, E bank	38°24’40.7” S	144°11’12.3” E
	3	S of jetty, W bank	38°24’43.0” S	144°11’11.4” E
Painkalac	1	Near ramp, E bank	38°28’02” S	144°05’59” E
	2	Below saltmarsh, E bank	38°28’26” S	144°05’46” E
	3	Below mudflat, W bank	38°27’59” S	144°05’39” E
	1(old)	Opp. ramp, E bank	38°28’02” S	144°06’01” E

Table F.5. Locations of lower estuarine sites in Anglesea and Painkalac. Datum is WGS84. Painkalac Site 1 was moved 60m upstream following the first two deployments due to site disturbance.

Table F.6 details locations of sampling locations in regional estuaries used in water quality and seagrass components of the study.

Estuary	Site	Latitude	Longitude	Max depth
Barham	Lower	38°45’54” S	143°40’04” E	-
	GOR bridge	38°45’50” S	143°40’14” E	2.6
	US (upstream)	38°45’41” S	143°39’53” E	3.0
Skenes	GOR bridge	38°43’27” S	143°42’41” E	1.3
Kennett	GOR bridge	38°39’59” S	143°51’44” E	3.0

Wye	GOR bridge	38º38’03” S	143º53’28” E	1.6
St Georges	GOR bridge	38º33’16” S	143º58’34” E	2.6
Erskine	Footbridge (near mouth)	38º32’00” S	143º58’41” E	2.0
	GOR bridge	38º32’09” S	143º58’25” E	1.1
Spring	DS (near mouth)	38º20’31” S	144º19’03” E	0.9
Thompsons	Mouth	38º18’06” S	144º22’30” E	1.5
	US (intertidal flats)	38º17’48” S	144º23’05” E	-

Table F.6. Locations, coordinates and maximum depths of regional estuarine sites. Datum is WGS84. Depths were only measured at profile sites.

Bathymetric Model: Anglesea

Measured depths

The main data used for the model of Anglesea estuary were depths measured to the nearest 10 cm at positions recorded with differentially corrected GPS (dGPS). Pathfinder Office software (Trimble Navigation Ltd, v2.70) was used for post-processing differential correction of logged field data (Trimble GeoExplorer) with base station files supplied by GPSnet (Land Victoria). When surveyed, the water level in the estuary was consistent throughout working hours (Table F.7) as recorded by a logging depth sensor located above the road bridge (Figure 4.1)

In the lower estuary, the waterline was traced by dGPS when the water level was relatively low (0.836m AHD) and used as a contour to create additional points for the model in Mapinfo (MapInfo Corp. v6.0).

Date	Time	Area covered	Water height (m AHD)	No. points
7/12/2000	14:45-15:30	Lower estuary	0.836	559*
2/10/1999	16:04-16:54	Head/ partCoogoorah	1.485	19
1/10/1999	14:50-16:02	Upper	1.485	63
30/9/1999	14:12-19:26	All	1.492	321

Table F.7. Summary of dGPS-located bathymetric dataset for Anglesea estuary. * - points created over a waterline traced by dGPS and recorded as six line and area features.

Based on precisions of individual recorded positions calculated by the GPS software, the mean 95% precision of individual points collected in 1999 was 2.35m, with a range of 1.70m-3.45m. Of the six sections of waterline mapped as lines and areas in the lower estuary in December 2000, the average 95% precisions of points comprising each line or area ranged from 2.03m-2.50m and the worst precisions of any point in each line or area ranged from 2.68m-4.30m.

These data were imported into Excel as Cartesian coordinates (AGD 66, AMG Zone 55) and depths were converted to Australian Height Datum using water height measurements taken at the jetty benchmark and recorded by a depth logger (see below). These three dimensional data were then imported into SURFER (Golden Software Inc, v6.04).

Training data

The methods of interpolation available made it necessary to create three types of points in addition to the GPS data in order to obtain a realistic bathymetric representation of the estuary. These were the edge of the estuary, areas of shallow swamp and a centre line of the estuary. All points were created in Mapinfo using a Cartesian coordinate system (AGD66, AMG Zone 55), combined with the points from the GPS data in Excel and imported into SURFER.

As most of the depth data were collected at a relatively high water level of ~1.49m AHD, a series of data points representing the edge of the estuary was created at 1.49m AHD. The location of the perimeter was determined firstly by the survey points marking the edge which were then interpolated with guidance from 1:25000 map data (Royal Australian Survey Corps, 1977) based on photos from 1969 and 1974, ground-truthed in 1975, and a georectified aerial photo (QUASCO) taken on 18/9/1993. As Coogoorah Park was created after the most recent topographic map, all guidance in this area was based on the aerial photo.

In the upper estuary, particularly Coogoorah Park, there are sections of swampland that are inundated when water levels are high and have a defined bank edge separating them from the main body of the estuary. At the time of sampling this bank was 0.3m deep and additional training points at 1.19m AHD were added between dGPS obtained points to better define this feature. As for the perimeter, a map and aerial photo were used to aid in accurate location of these points.

The main channel of the estuary was further defined for the model by adding points by linear interpolation between the deepest dGPS recorded points. This was done to avoid 'bullseyes' or 'holes' around the dGPS points that would otherwise have occurred given the method of contouring and the large number, and relative proximity to each other, of points used to define the edge of the estuary

Model creation

SURFER was used to calculate depths for a grid of points at 5m intervals from the x,y,z data described above. Kriging used for the calculations using a linear variogram model with no anisotropy. The grid extended from 253735E, 5744460N to 254677E, 5746420N. Volume, planar area and surface areas of the estuary were then calculated at 0.1m intervals over the range of z values. The grid was also truncated to calculate volume and areas for the estuary below the road bridge and for Coogoorah Park.

To allow volumes to be calculated from logged height data a second order polynomial was fitted to the height-volume curve between 0.8 and 1.49m AHD using Excel. This allows interpolation between the 0.1m height intervals to the more precise (mm) measurements of height that were logged. The relationship is:

$$V = 54752h^2 - 703.66h + 36592$$

where V = volume in m^3 and h = height in metres AHD ($R^2 = 0.99998$).

For heights above 1.49m AHD, in the absence of topographic information above this height, a linear relationship was used to estimate volume based on vertical rise of the water area of the estuary at 1.49m. This equation is:

$$V = 169913(h-1.49) + 157265.$$

Estuarine States

Comparison of visual assessments with logged water levels

Of the 70 visual observations of state made at the mouth of Anglesea, 45 were made at times for which there were logger data. Of these, 37 (82%) agreed with determinations based on logger data (Figure 4.9). Of the occasions on which the sources of information differed, two were within a day of a transition between states. The six remaining discrepancies all overestimate the amount of exchange and tidal fluctuation in the estuary. All of these times were near spring tides and an increase in sea state or swell (Table F.8) when high marine waters and temporary exchanges across the bar were taken as indications of a volume of exchange greater than actually occurring.

Date	Vis. state	Log. State	Notes
8/4/00	P	C	Low→Mod. sea
22/8/00	P	C	Spring tides overtopping bar, inc fw from 20/8/00
2/10/00	P	C	Spring tides & high seas
23/4/01	T	P	***Transition*** P→T – large floods
26/10/01	T	P	Large tides, mod.-rough seas
7/11/01	T	P	Large tides, mod.-rough seas
8/11/01	T	P	Large tides, mod.-rough seas
9/12/01	P	T	***Transition*** P→T – large seas on 10th

Table F.8. Dates of different assessments of state based on visual measures and logged water height. Swell & sea data from Cape Otway provided by the Bureau of Meteorology. Tide data from the Victorian Ports Corporation.

The majority of visual determinations of state without corroborating logger data from Anglesea are before mid-August 1999 (Figure 4.9a). These data indicate a general pattern of closure in summer, with short, perched periods followed by a perched state throughout winter. These results must be interpreted in context of the errors in estimation as identified above. Based on these data, assessments of the estuary as closed seem to be accurate as there were no visual misclassifications of tidal or perched state as closed. Incorrect assessments of closed states as perched based on the observation of a short term overflow of the bar were common as were classifications of perched states as tidal based on water movement associated with large tides and seas (Table F.8).

Ten of 16 times made prior to installation of the logger were assessed as perched at times of large tides (>1m). Based on measured heights, tide data, and field notes there is evidence to suggest that some of these assessments were made when the estuary was essentially closed despite short-term movements of water across the bar (Table F.9). This was not the case with Painkalac, where there was often no flow and hence low estuarine water levels during closed periods, reducing the incidence and ambiguity of these situations.

For Anglesea, some visual assessments of a perched state were revised to closed as shown in Table F.9. One misclassification was identified at Painkalac based on observations by Reilly (unpublished data), when an assessment of the estuary as closed (on 14/10/2001) was made at a neap tide during a tidal period. Unless otherwise specified, these re-interpreted states are used in comparisons between state and other variables from Section 4.3.4 onwards.

Date	Revised to:	Peak tides >1m?	Comments
1/2/99	-	y	height suggests perched
5/3/99	-	y	mod. seas, height suggests perched
8/3/99	-	n	
19/3/99	-	n	
12/4/99	-	n	
4/5/99	-	y	
18/5/99	C	y	big seas prior to observation
15/6/99	-	y	
21/6/99	-	y	
22/6/99	-	y	
23/6/99	-	n	
25/6/99	-	n	
19/7/99	C	y	too high for substantial tidal influence
20/7/99	C	y	too high for substantial tidal influence
23/7/99	C	n	too high for substantial tidal influence
15/8/99	C	y	Closed on 18/8/99 (logger)

Table F.9. Observations of a perched state at Anglesea that are not verifiable by logger data with revised assessments based on tidal data, manual height measurements, sea state/ swell and field notes.

In addition to the above modifications, there were some short-term changes in state that were recorded by Reilly (unpublished data). A comparison of changes in state based on visual assessments alone and the record that integrates logger data, changes in visual assessments and Reilly's observations is seen in Figure F.1.

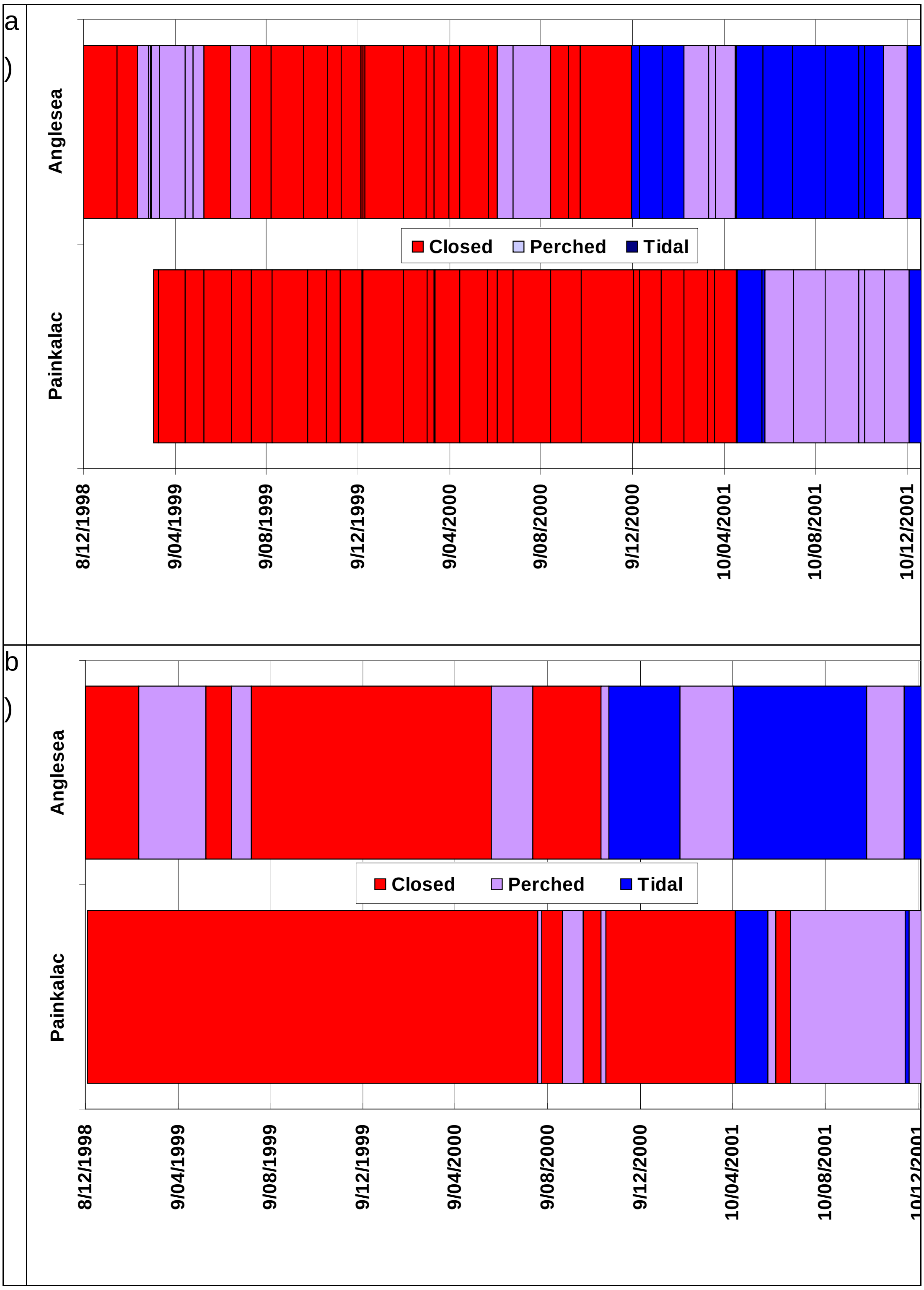

Figure F.1. Comparison of hydrologic states in Anglesea and Painkalac estuaries from a) visual assessments of mouth conditions only (dates of assessment indicated by vertical bars) and b) integrated visual assessments, logger data and daily observations by Reilly (unpublished data).

Description of states: classification tree analysis

In order to quantify the nature of the hydrologic states at Anglesea, and to confirm the validity of the visual classifications from plots of water level, a classification tree analysis was done. The method used for this analysis was one of a series of techniques known as classification and regression trees (CART). These tools explain changes in a response variable in terms of one or more predictor variables by repeatedly splitting data into more homogenous groups based on rules associated with a single predictor variable (De'Ath & Fabricius, 2000). In effect, this process creates rectangular categories parallel to axes of the predictor variables. These analyses have very few assumptions and can be used for a wide range of distributions (Brieman *et al.*, 1984). One particular benefit of this type of analysis, when compared to a PCA or similar, is the ability to describe a response in terms of the original predictors.

The dependent variable for the classification tree was state, as qualitatively identified from time series plots of water level. The nine independent variables were, for each period, the number of heights recorded and the mean, standard error, skewness and kurtosis of water heights and the mean magnitude (|rate|), standard error, skewness and kurtosis of ten-minute changes in height over each period (Table F.10).

		Water height (m AHD)				**Rate of change (m/10min)**			
Period	***n***	**mean**	**standard error**	**skew**	**kurt**	**mean |rate|**	**standard error**	**skew**	**kurt**
Closed1	9743	1.45	6.88×10^{-4}	-0.342	-0.843	3.05×10^{-04}	7.76×10^{-06}	7.00	122.5
Closed2	11708	1.49	6.91×10^{-4}	0.582	-0.419	3.82×10^{-04}	6.80×10^{-06}	2.53	42.96
Closed3	16330	1.57	6.99×10^{-4}	0.318	0.204	5.17×10^{-04}	8.68×10^{-06}	0.792	42.60
Closed4	12813	1.38	9.10×10^{-4}	0.518	-0.115	5.50×10^{-04}	1.14×10^{-05}	6.39	92.47
Perched1	8063	1.16	1.28×10^{-3}	0.965	0.555	2.13×10^{-03}	6.05×10^{-05}	3.86	27.72
Perched2	1728	1.02	2.99×10^{-3}	0.741	-0.0825	3.14×10^{-03}	1.81×10^{-04}	3.00	20.29
Perched3	10367	1.14	1.06×10^{-3}	0.545	0.285	1.64×10^{-03}	4.22×10^{-05}	5.00	51.85
Perched4	7343	0.924	1.20×10^{-3}	2.128	6.327	3.76×10^{-03}	9.89×10^{-05}	3.21	22.24

Perched5	4552	0.907	$1.08\text{x}10^{-3}$	1.589	3.683	$2.39\text{x}10^{-03}$	$7.71\text{x}10^{-05}$	3.92	26.90
Tidal1	13533	0.869	$7.44\text{x}10^{-4}$	3.354	12.87	$3.13\text{x}10^{-03}$	$8.03\text{x}10^{-05}$	3.91	27.63
Tidal2	11794	0.878	$1.19\text{x}10^{-3}$	3.311	11.33	$6.04\text{x}10^{-03}$	$1.51\text{x}10^{-04}$	1.63	14.04
Tidal3	2481	0.864	$2.07\text{x}10^{-3}$	2.922	9.47	$5.65\text{x}10^{-03}$	$2.52\text{x}10^{-04}$	2.66	12.32
Tidal4	5760	0.861	$1.36\text{x}10^{-3}$	3.357	13.84	$4.69\text{x}10^{-03}$	$1.49\text{x}10^{-04}$	3.05	17.03

Table F.10. Summary statistics for water height and rate of change during each of the periods identified as particular states by inspection of time plots. skew=skewness, kurt=kurtosis.

In addition to the identification of the best predictor variable(s), estimates of the fit of the classification (equivalent to R^2) were determined for each model, with the aim of the procedure being to minimise misclassification. A proportional reduction in error (PRE) of close to 1 means that the categorical variable of 'state' can be accurately predicted by simple rules based on values of the predictor variables.

Classification trees make use of stopping rules, whereby no further splitting of groups is done once a minimum level of improvement in homogenisation of groups is reached. In this case there were four values that this was dependent on: the total number of splits in the tree (set to the maximum possible); the minimum reduction in PRE allowable for any split (0.01); the minimum value of the splitting criteria (Gini index) for a node to be split (0.01); and the minimum number of periods permitted in any final leaf of the tree (1). With such small values as stopping criteria, a classification tree will typically become overlarge, despite reducing within-group variability of response variables (Brieman *et al.*, 1984). In this case, despite a set of very small values governing stopping rules, the 13 periods were perfectly classified by two nodes defined by rules associated with mean height (Figure F.2).

As mean water height was sufficient to classify the periods of time into states perfectly, the classification procedure stopped and variables following mean height in the list above were not used in the initial classification tree. In order to examine the explanatory power of the other variables, the analysis was re-done with the dependent variables re-ordered using 30 random permutations. As expected, each of these analyses resulted in a perfect correct classification of the states, however this was achieved with sets of rules based on different variables, all of which successfully isolated either closed or tidal states from both others, except for the standard error of water height which only distinguished between closed and perched states (Table F.11).

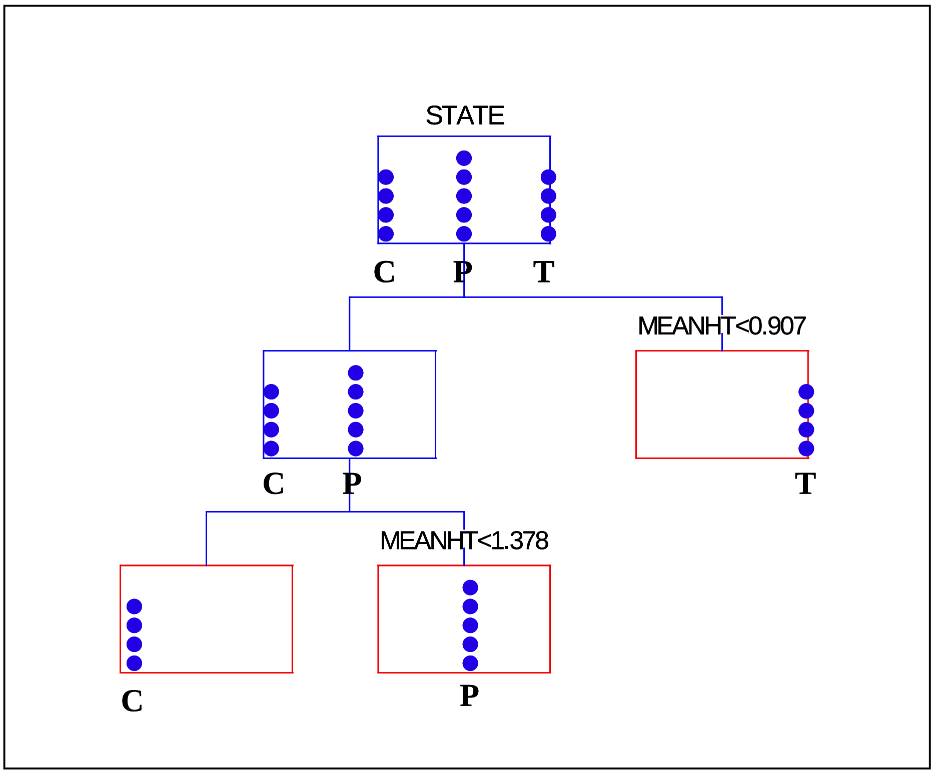

Figure F.2. Results from initial classification tree analysis of hydrologic states of Anglesea estuary. Proportional reduction of error (PRE) is 1.000. The Gini index was used as the measure of heterogeneity. Stopping rules were: minimum split index=0.01; minimum improvement in PRE=0.01; maximum number of nodes=22; and minimum count in node=1.

Variable	*Rule*	*= State*
Mean height	< 0.91	Tidal
Mean height	> 1.38	Closed
Standard error height	< $1.06x10^{-3}$	Closed [a]
Skewness height	> 2.92	Tidal
Kurtosis height	> 9.47	Tidal
Mean magnitude rate	< $1.64x10^{-3}$	Closed
Standard error rate	< $4.22x10^{-5}$	Closed

Table F.11. Rules from classification trees defining states in Anglesea estuary. [a] – this rule only distinguishes between perched and closed states; all other rules distinguish between the specified state and both other states.

The usefulness of these variables in identifying states is clear in scatterplots, in which the three states are clearly identified by many combinations of two variables in Figure F.3.

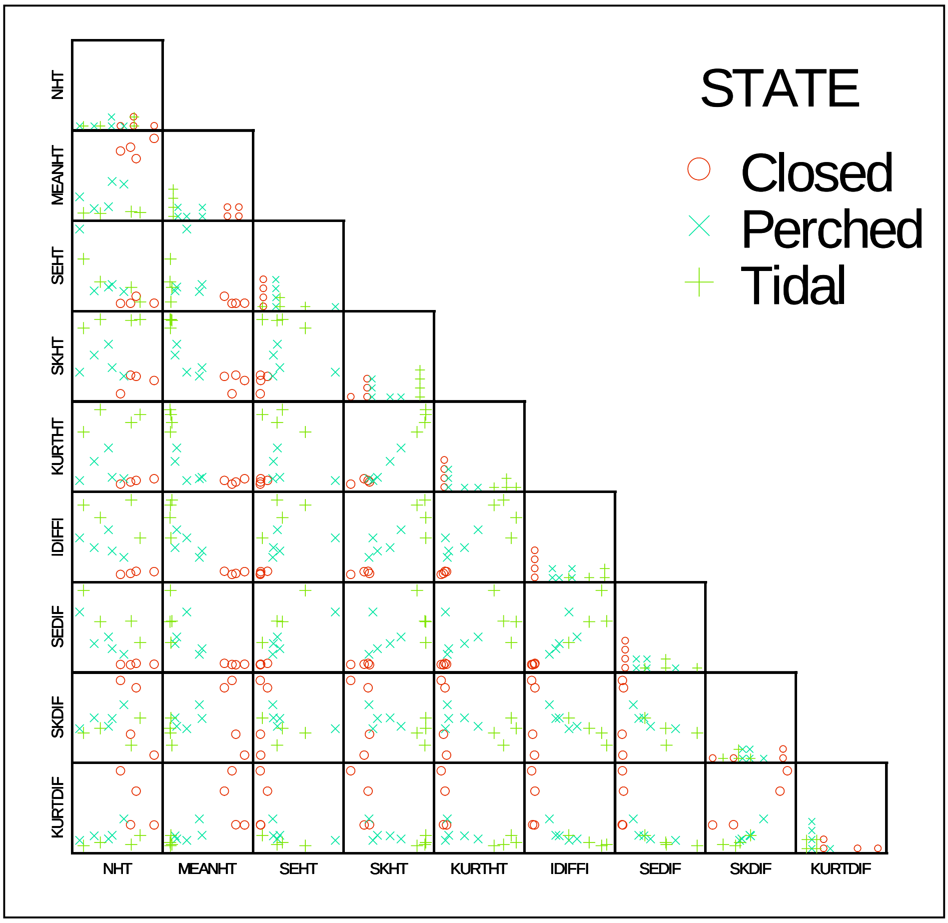

Figure F.3. Scatterplot matrix of variables associated with classification tree rules identifying hydrologic states in Anglesea estuary. Plots of single variables along the diagonal are dot histograms.

While the results of the classification tree analysis may seem obvious given the separation of states in much of the scatterplot matrix, they do provide confirmation of the somewhat arbitrary classifications based on visual analysis of time plots of water height. The results of the classification tree also provide a set of rules by which periods from other times or estuaries may be assessed.

As seen in Section 4.3.2.c, during the period of logged height in Painkalac five periods were identified as one of the three states. While these periods are short compared to those at Anglesea, the rules derived from the Anglesea data can be applied to those from Painkalac. When this was done, the rule based on mean height successfully distinguished the tidal period from the others but the remaining rules did not apply to the data from Painkalac. A possible reason for this inability to distinguish closed and perched states was that in the creation of splitting rules, the higher of the distinguishing values was used by the SYSTAT program. In fact, even with modification, rules that apply to both estuaries could not be defined as there were overlaps in values at all other splitting points identified in the Anglesea analysis (Table F.12).

Despite this, patterns of differences between the states were similar for most of the statistics identified with the exception of kurtosis of water height (Table F.13, Figure F.4). Both estuaries moved from closed to perched states at a mean height of about 1.15mAHD and from perched to tidal at around 0.9mAHD. The standard error of height increased from closed through perched states, further increased in Painkalac during the tidal period, and would very likely have done in Anglesea without the curtailing of lower water levels.

Skewness of height generally increased from closed to tidal states, with some overlap between closed and perched states. Skewness of the Painkalac perched and tidal periods was less than that for those states in Anglesea. This was also most likely due to the censoring of lower values in the Anglesea data. Kurtosis increased from closed to tidal states in Anglesea but did not show a clear pattern in Painkalac. It is likely that this is an artifact of the censoring of the Anglesea data and the short duration of some of the Painkalac periods.

As would be expected, the mean magnitude of the rate of change in height was lowest during closed periods and greatest during tidal periods. This applied to both periods however there were overlaps in ranges between states and between estuaries. Standard error of the rate of change in height showed a similar pattern.

Given the differences in the lower part of the distribution (where water height was below the logger on low tides in Anglesea) some of these differences are not surprising, particularly skewness and kurtosis of height during tidal periods at Anglesea. Additionally, some of the periods in Painkalac were shorter than the two-week spring-neap tidal cycle and would not have included the full range of tidal variation. Of the statistics examined, all but kurtosis of height appeared to be useful descriptors of the state of the estuaries.

State	*n*		mean ht		s.e. ht		sk. ht		kurt. ht		|rate|		se rate	
	A	P	A	P	A	P	A	P	A	P	A	P	A	P
C	4	2	1.57	1.30	6.88×10^{-4}	1.50×10^{-3}	-0.342	0.505	-0.843	-1.56	3.05×10^{-4}	3.79×10^{-4}	6.80×10^{-6}	8.23×10^{-5}
			1.38	1.14	9.10×10^{-4}	1.99×10^{-3}	0.582	0.637	0.204	1.04	5.50×10^{-4}	3.50×10^{-3}	1.14×10^{-5}	1.35×10^{-5}
P	5	2	1.16	0.967	1.06×10^{-3}	2.51×10^{-3}	0.545	0.0963	-0.083	-0.474	1.64×10^{-3}	3.17×10^{-3}	4.22×10^{-5}	1.07×10^{-4}
			0.907	0.914	2.99×10^{-3}	2.66×10^{-3}	1.589	0.989	6.327	0.875	3.76×10^{-3}	3.50×10^{-3}	1.81×10^{-4}	9.84×10^{-5}
T	4	1	0.878	0.640	7.44×10^{-4}	6.92×10^{-3}	2.922	1.12	9.47	0.448	3.13×10^{-3}	0.0145	8.03×10^{-5}	7.03×10^{-4}
			0.861		2.07×10^{-3}		3.357		13.8		6.04×10^{-3}		2.52×10^{-4}	

Table F.12. Ranges of summary statistics used in classification tree rules created using Anglesea data. Shaded figures show overlaps in the ranges of values between states in the two estuaries. C=closed, P=perched, T=tidal, A=Anglesea, P=Painkalac, s.e.=standard error, sk.=skewness, kurt=kurtosis, ht=height.

Variable	*Rule*	*= State*	*Valid?*	*Pattern?*
Mean height	< 0.91	Tidal	Y	Y
Mean height	> 1.38	Closed	N	Y
Standard error height	$< 1.06 \times 10^{-3}$	Closed [a]	N	Y
Skewness height	> 2.92	Tidal	N	Y
Kurtosis height	> 9.47	Tidal	N	N
Mean magnitude rate	$< 1.64 \times 10^{-3}$	Closed	N	?
Standard error rate	$< 4.22 \times 10^{-5}$	Closed	N	Y

Table F.13. Rules derived from Anglesea data and applicability for separation of states in Painkalac. 'Valid?' refers to whether the rule derived from Anglesea can be directly applied to the Painkalac data, 'Pattern?' refers to whether or not a similar pattern of separation between states was evident for the Painkalac data. [a] – this rule

only distinguishes between perched and closed states; all other rules distinguish between the specified state and both other states.

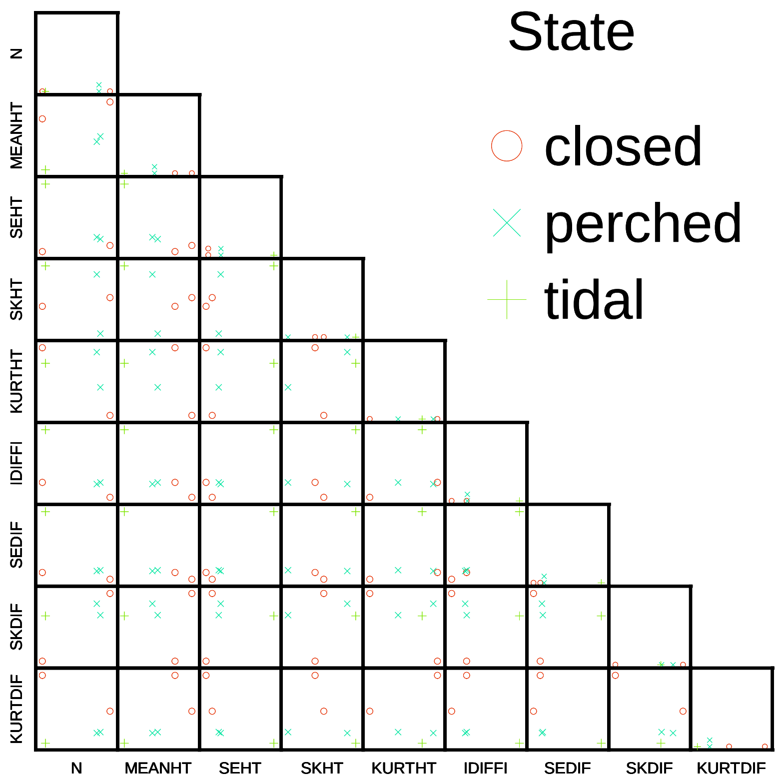

Figure F.4. Scatterplot matrix of variables associated with classification tree rules identifying hydrologic states in Painkalac estuary. Plots of single variables along the diagonal are dot histograms.

Appendix G. Sediments

Pilot study – sediment traps

Three trap sizes were trialled during the first deployment from 18/1/2001 to 14/2/2001 (Table G.1).

Size	Height (mm)	Mouth diameter (mm)	Aspect ratio (height:diam.)	Mouth area (mm^2)	*n*
Small	73	10.5	6.95	86.6	5
Medium	96	15.6	6.15	191	18
Large	131	27.2	4.82	581	12

Table G.1. Dimensions and numbers of tubes used in pilot study.

For the pilot study, solid house bricks with notched sides were used to attach tubes to with cable ties around the circumference of bricks and tubes. Tubes were attached so that the mouths protruded approximately 2 cm above the surface of the bricks.

Along with medium tubes that were deployed as per the main design, eight haphazardly selected bricks had either small or large tubes attached in addition to the medium tubes. Of these, two bricks, both with medium and small tubes were not successfully retrieved.

Mean deposition rates and the percentage organic matter by weight were compared between sizes on individual bricks with t-tests using a Bonferroni-modified α for each of the four groups of comparisons.

No significant difference in deposition rate between small and medium tubes was observed but on one brick (in Anglesea) a significantly greater percentage ($p=0.016$, $\alpha=0.025$) of organic matter content was observed in small tubes than in medium tubes (Figure G.1b)).

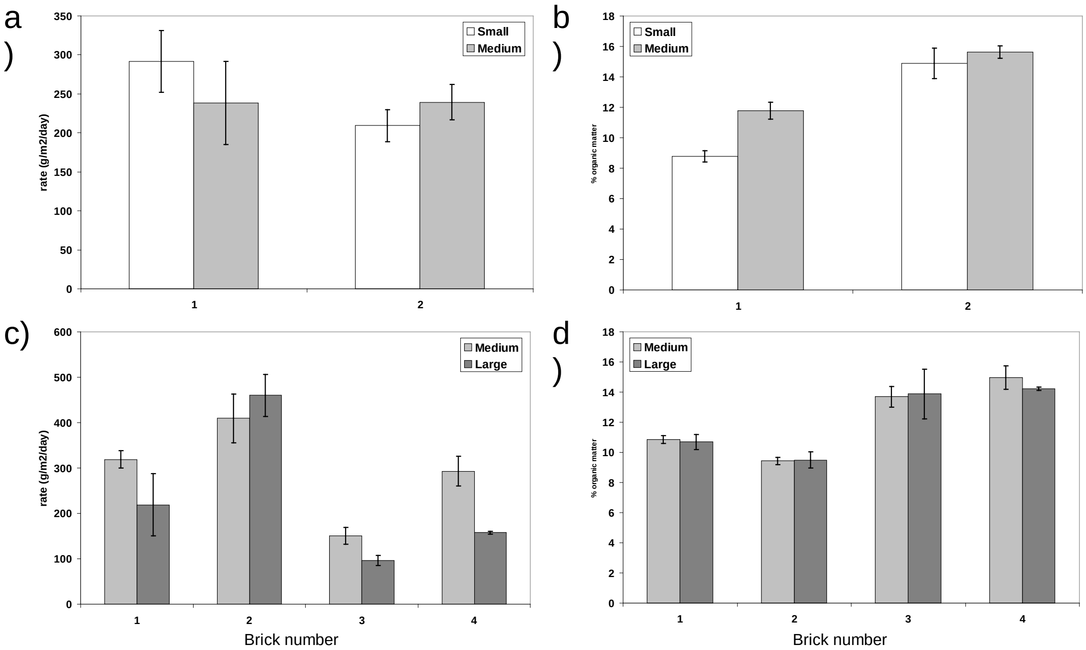

Figure G.1. Mean (± s.e.) deposition rates (a),c)) and organic matter content (b),d)), as measured by small, medium and large sediment traps on 2 (small/medium; a) and b)) or 4 (medium/large; c) and d)) bricks.

No significant differences between medium and large tubes were seen in terms of deposition rate or percent organic matter. Two bricks showed non-significant differences in that large tubes collected less sediment than medium tubes (Figure G.1c; p= 0.084, 0.053, α=0.0125).

Although the aspect ratios of the tubes were similar, potential problems with small and large tubes were identified. The aspect ratio of the large tubes was just below the value of 5 suggested by Hargrave and Burns (1979) to create a turbulence-free boundary layer at the bottom of the tubes. Causes of the lower proportion of organic content collected in small tubes are not clear but the relatively small mass collected in these tubes (mean = 1.112g) could have led to greater proportional errors from handling and measurement of collected sediments. For these reasons, medium tubes were used for this component of the study.

Changes in bed height

	Date	Date	A1				A2			A3			P1			P2			P3		
	Anglesea	Painkalac	A	B	C	C[1]	A	B	C	A	B	C	A	B	C	A	B	C	A	B	C
0	18/01/01	18/01/01	in	in		in	in	in	in	in	in	in	in	in	in	in	in	in	in	in	in
1	14/02/01	13/02/01	-0.8	-1.1		-2.0	1.0	1.5	-0.9	1.1	-0.1	1.4	-0.1	2.2	0.4	-1.3	-3.2	-0.6		3.3[a]	2.5
2	20/03/01	21/03/01	-0.4	0.4	in		-2.7	-0.2	-2.4	0.1	0.6	-0.9	-2.7	-3.3		0.8	-0.3	-2.0		no vis.	
3	25/04/01	27/04/01[b]	0.2	-3.0	2.9	-0.3	-3.3	-6.3		-6.3	-4.1	-6.3	-3.6	-13.0	-5.2	-8.3	-4.9	-6.1	-9.0	-6.4	-6.5
4	01/06/01	31/05/01	-2.0	-1.1	-3.8	-1.4	0.1	-8.9	-8.9[c]	-3.3	-0.3	-0.9	-5.2	2.8	-0.9	6.2	0.3	-1.6	-8.3	0.7	
5	13/07/01	no vis.	0.7	1.6	-0.5		-0.4		1.0	5.2	-1.0	-2.8									
6	21/08/01	no vis.	-1.9	-4.4	-0.5		0.8	0.3	-1.2	-4.9	-0.5	2.4									
7	05/10/01	03/10/01	-0.5	3.9	-0.3		-2.1	-0.5	-0.3	1.1	-0.5	0.2	2.0	9.4	4.2	-6.0	0.6	-2.6	2.2	-1.3	-1.0
8	03/11/01	04/11/01	0.0	-0.4	-0.1	-6.0	-2.6	-0.4	1.8	5.6	0.1	0.6	-11.4	-5.4	-7.1	9.5	1.6	18.1	-0.4	-0.1	2.1
9	11/12/01	12/12/01	-1.8	0.1	-0.1	3.5	2.1	0.7	-0.9	0.8	0.3	-0.6	7.8	2.2	8.6	2.1	-1.3	-0.7		-0.1	0.7
10	15/01/02	18/01/02	0.2[d]	0.5[d]	0.3		-1.0	0.2	-0.6	-2.2	-0.7	0.2	0.2	-4.4	-7.3	-2.6	-0.7	-2.7	12.0	3.4	6.6
11	14/02/02	12/02/02	1.5	2.8	-0.3		-1.4	0.2	0.6	1.8	1.1	0.2	1.6	-0.1	8.0	1.4	6.6	0.0	-2.3	-3.9	-10.3

Table G.2. Changes in bed height measured in 2001 and 2002. Intervals for individual rulers were typically around one month, longer intervals are indicated by missing values. Site P1 was moved in March 2003, values shown after this are from the relocated site. Location C at Anglesea Site 1 was thought missing and a new location was established on 30/3/1, the original location is shown as C[1]. Small variations in measurement dates (largely due to locating or reading rulers in low visibility conditions) are indicated by super script letters: [a] measured on 14/2/2001, [b] Painkalac Site 1 measured on 26/4/2001, [c] measured on 4/6/2001, [d] measured on 18/1/2001.

Appendix H. Seagrass Extent

Aerial Photographs

The areas of seagrass delineated from aerial photographs are shown in Table H.1, along with a qualitative assessment of water height and clarity at each sampling time based on the extent of inundation and which parts of the bottom of the estuary were visible. Even in the worst viewing conditions (e.g. 1982 and 1993), deep edges of beds were apparent on photographs.

Date	Measured area (m^2)	Water depth	Water clarity
28/3/1964	5546[a]	High	Intermediate
2/05/1982[b]	23695	High	Intermediate
20/7/1985	14201	Medium	High
28/6/1989	10388	Low	High
18/9/1993	5234	Low	Low
29/8/1997[b]	12871	High	High
29/8/1998	17304	High	High

Table H.1. Measured areas of seagrass, water depth and water clarity in aerial photographs. [a]=another 2370m^2 of vegetation that was thought to be emergent macrophytes, rather than seagrasses, was identified high on the shore. [b]= part of estuary not included in photo: 2/5/82 at 500m upstream from the mouth; 29/8/97 at 570m upstream of the mouth. The 1964 photo was provided by the Surf Coast Shire, all other photos were from QUASCO with permission from Alcoa of Australia.

Sections of the lower estuary were beyond the bounds of photos taken in 1982 and 1997. In these years, areas of the estuary from the mouth to 500m and 560m upstream, respectively, were not included in the photos. The areas of seagrass at that were recorded in these excluded areas (from photos taken in other years) are shown in Table H.2.

Date	Area excluded from 2/05/1982 coverage (m^2)	Area excluded from 29/8/1997 coverage (m^2)
2/05/1982	n/a	6006
20/7/1985	927.67	2407
28/6/1989	158.7	837
18/9/1993	0	443
29/8/1997	0	n/a
29/8/1998	217.66	1352

Table H.2. Mapped areas outside the coverage of photos that did not include the whole estuary.

A comparison of seagrass areas measured in each year with seagrass areas recorded within the bounds of the 1982 and 1997 photos is shown in Figure H.1. Despite some seagrass areas in the lower estuary being excluded from estimates, temporal patterns in the results remained similar, suggesting that conclusions reached on the basis of the results presented in Section 6.3.1 were not compromised by the incomplete coverage of the estuary in two of the six years.

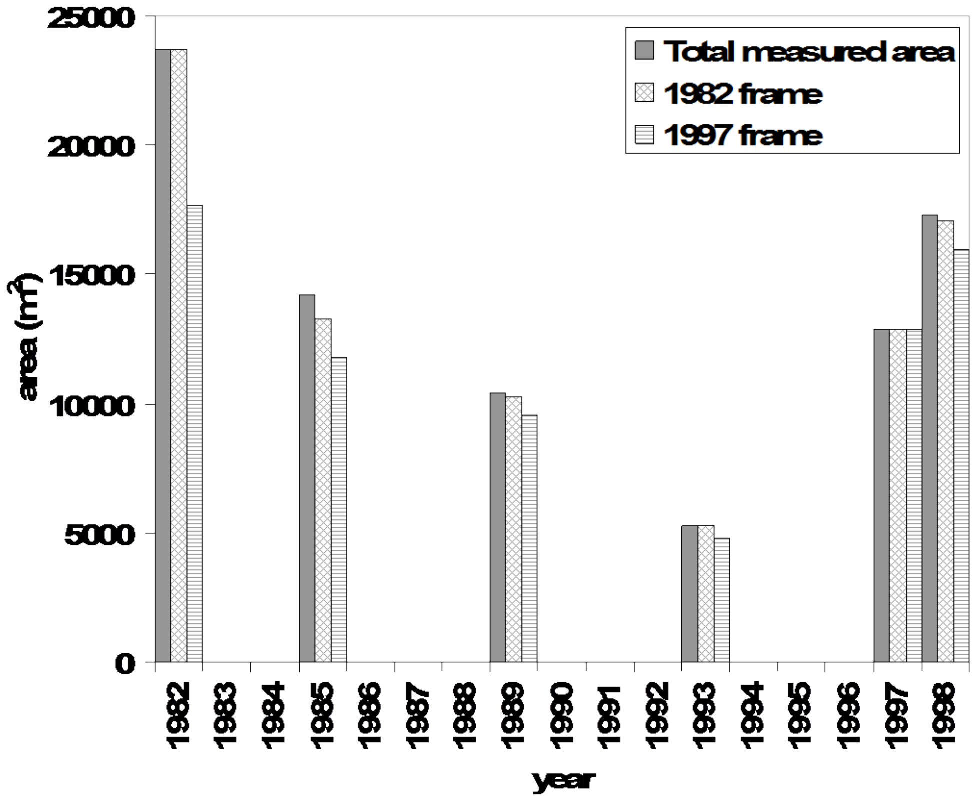

Figure H.1. Comparison of total areas of seagrass measured and areas of seagrass measured within the frames of 1982 and 1997 aerial photographs.

Estimates of seagrass area outside the coverage of the aerial photos were made based on bathymetry, extent of seagrass within the photos and extent of seagrass in 1985 (for the 1982 estimate) and 1998 (for the 1997 estimate). These areas were 5000m^2 and 600m^2 for 1982 and 1997, respectively.

GPS Mapping

Date	No. features (lines & areas)	No. points (within features)	Mean 95% precision (m)	Worst 95% precision (m) feature mean	Worst 95% precision (m) overall
8-9/11/1999	8*	914	2.46	3.27	4.49
6-9/4/2000	21	1874	3.04	4.58	8.47
8/12/2000	37	1965	2.31	3.02	4.76

Table H.3. Sampling details of GPS-mapped seagrass beds in Anglesea estuary. 'Features' includes lines and areas that were traced, each of which comprises multiple point locations. Precision refers to the precision of the positional measurement of the individual points (as calculated using the software *Trimble Pathfinder Office 2.51*). Feature mean refers to the average of the worst precision of points in each feature. * - mapped locations of seawalls were used as boundaries for some beds in the November 1999 survey.

Composition	Bed type	Date Nov-99 (m^2)	Apr-00 (m^2)	Dec-00 (m^2)[a]
Mixed	**Dense**	5,625	1,330	2,586
Mixed	**Patchy**	21,861	13,233	6,164
Ruppia	**Dense**	85	12,154	230
Ruppia	**Patchy**	1,293	9,422	0
Zostera	**Dense**	0	8	0
Zostera	**Patchy**	0	200	292
Mixed	**Both (subtotals)**	27,486	14,563	8,749
Ruppia		1,378	21,576	230
Zostera		0	208	292
All species (summed)	**Both**	28,863	36,348	9,271

Table H.4. Areas of seagrass beds by species composition and density category for the three survey dates. [a]=excludes bleached/dying beds

Composition	Bed type	Area bleached/dying absolute (m^2)	Percent of total
Mixed	**Dense**	4,678	64
Mixed	**Patchy**	5,060	45
Ruppia	**Dense**	1,087	83
Ruppia	**Patchy**	739	100
Zostera	**Dense**	10	100
Zostera	**Patchy**	25	8
Mixed	**Both (subtotals)**	9,738	53
Ruppia		1827	89
Zostera		34	11
All species (summed)	**Both**	11,599	56

Table H.5. Composition and bed density of bleached and dying seagrasses recorded in December 2000 by absolute area and percentage of that category of bed in Anglesea estuary.

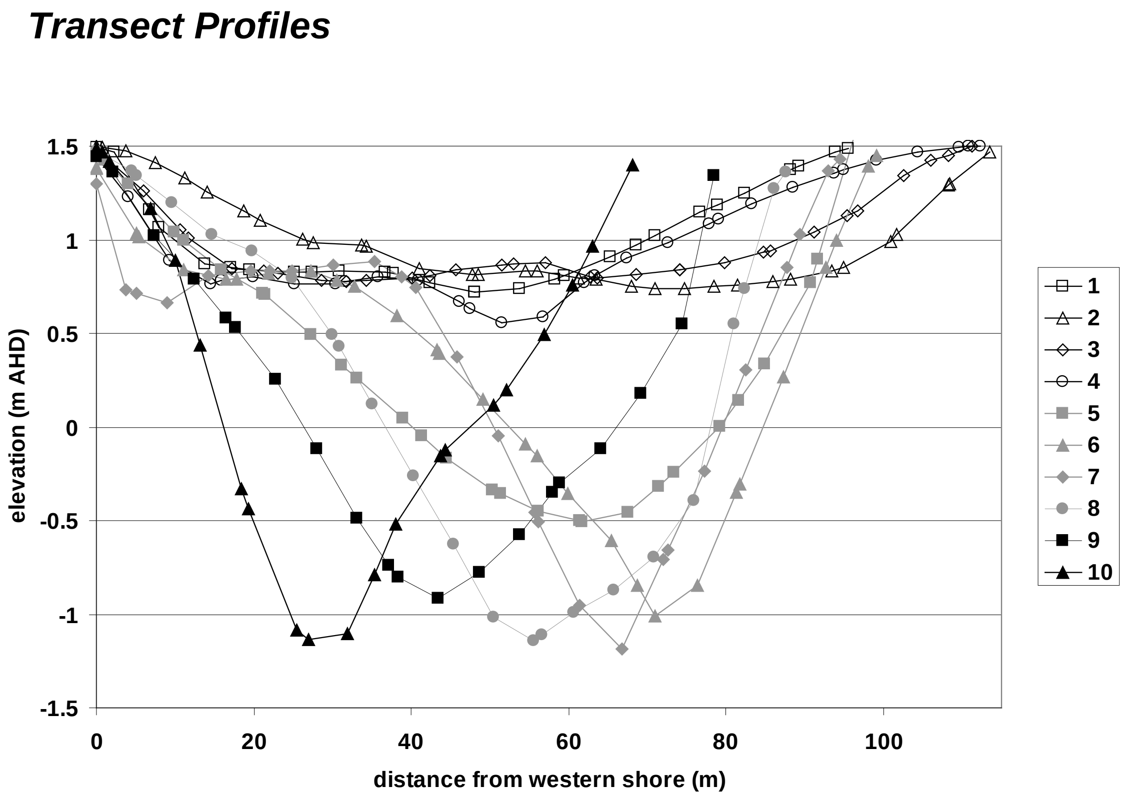

Figure H.2. Depth profiles of seagrass transects in Anglesea estuary. Transect numbers shown in the legend relate to transects from near the mouth of the estuary upstream as shown in Figure 6.1a. Profiles derived from bathymetric model.

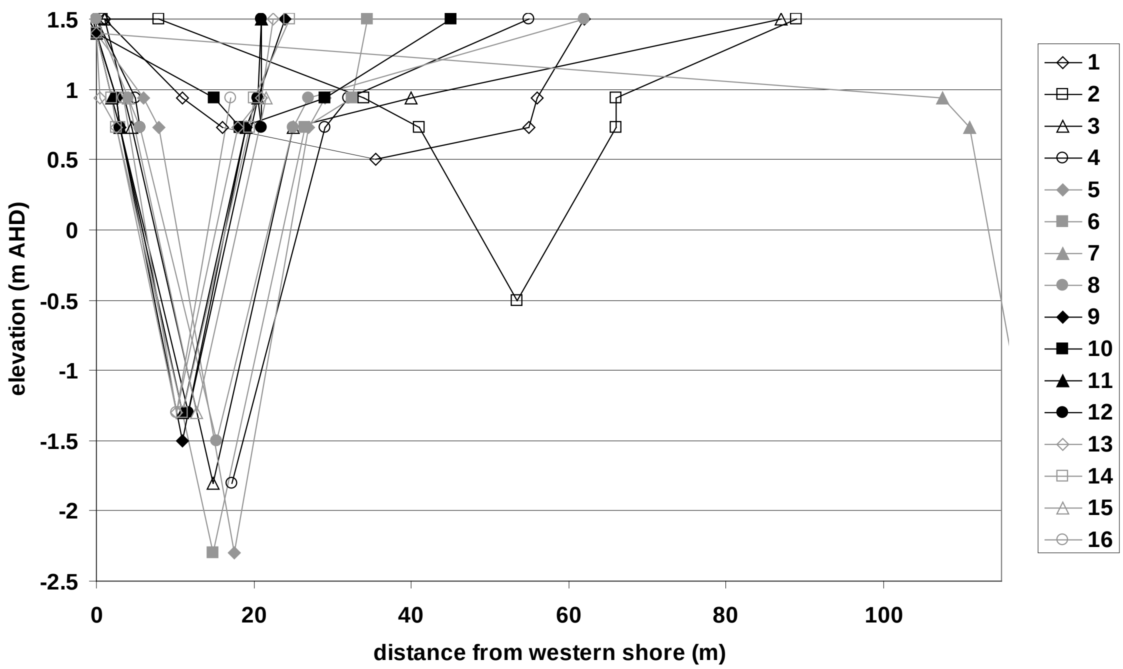

Figure H.3. Depth profiles of seagrass transects in Painkalac estuary. Note the different scale of the y-axis compared to Figure H.2. Transect numbers shown in the legend relate to transects from near the mouth of the estuary upstream as shown in Figure 6.1b. Maximum depths are estimates derived from depths measured at longitudinal sites. Transect seven (off-scale on the x-axis) had a maximum depth of –1.9m AHD and a length of 297m, comprised mainly of shallow flats, with a deep channel width of ~25m at a distance of ~120m from the western shore .

Appendix I. Detrital Processes

Litterbags

Time	Estuary	Date: deployed	Date: retrieved	Duration (days)	Replicates 1	2	3	Total	Source of material[a]
1	A	18/01/2001	14/02/2001	27	2	3	3	8	
	P	18/01/2001	13/02/2001	26	0	3	2	5	
2	A	16/02/2001	20/03/2001	32	3	3	3	9	
	P	15/02/2001	21/03/2001	34	1	2	3	6	
3	A	22/03/2001	23/04/2001	32	4	0	0	4	
	P	22/03/2001	26/04/2001	35	3	0	3	6	
4	A	26/04/2001	1/06/2001	36	4	3	4	11	
	P	27/04/2001	31/05/2001	34	4	4	1	9	
5	A	4/06/2001	13/07/2001	39	3	4	4	11	
	P	4/06/2001	12/07/2001	38	3	2	4	9	P1 and P3 swapped
6	A	14/07/2001	21/08/2001	38	4	3	4	11	
	P	13/07/2001	20/08/2001	38	4	4	2	10	
7	A	24/08/2001	1/10/2001	38	3	4	4	11	A2: 1/4 from A2, 3/4 from A3
	P	23/08/2001	3/10/2001	41	4	4	4	12	All sites: from A3
8	A	7/10/2001	3/11/2001	27	4	4	4	12	
	P	7/10/2001	4/11/2001	28	3	3	3	9	All sites: 1/2 A2, 1/A1, 1/4 A3
9	A	8/11/2001	11/12/2001	33	3	4	3	10	
	P	8/11/2001	11/12/2001	33	3	3	2	8	
10	A	14/12/2001	15/01/2002	32	4	4	4	12	
	P	14/12/2001	15/01/2002	32	3	4	4	11	
11	A	18/01/2002	14/02/2002	27	4	3	4	11	
	P	18/01/2002	12/02/2002	25	4	4	4	12	P2: from P3

Table I.1. Details of deployment periods, replicates successfully retrieved from each site and variations in location of source materials. [a]: only specified if not from the vicinity of the site of deployment.

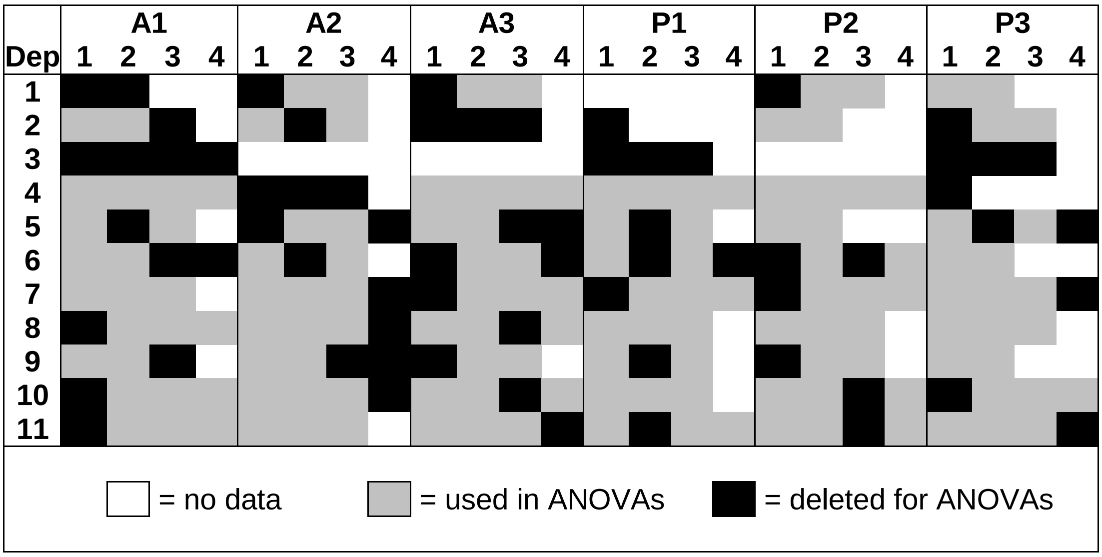

Table I.2. Litter bags retrieved and randomly deleted for use in ANOVAs by site and deployment period (Dep). Numbers below site labels are replicate identification numbers.

Deployment	Factor	S.S.	d.f.	M.S	*F*-ratio	*p*
1	Estuary	0.033	1	0.033	1.728	0.3192
	Site(Estuary)	0.038	2	0.019	0.094	0.9121
	Residual	0.805	4	0.201		
2	Estuary	0.153	1	0.153	0.068	0.8182
	Site(Estuary)	4.475	2	2.237	18.630	**0.0094**
	Residual	0.480	4	0.120		
4	Estuary	0.206	1	0.206	2.926	0.2293
	Site(Estuary)	0.141	2	0.071	0.129	0.8801
	Residual	6.557	12	0.546		
5	Estuary	1.598	1	1.598	5.445	0.0799
	Site(Estuary)	1.174	4	0.293	0.610	0.6710
	Residual	2.887	6	0.481		
6	Estuary	7.5×10^{-5}	1	7.5×10^{-5}	0.001	0.9746
	Site(Estuary)	0.261	4	0.065	1.069	0.4476
	Residual	0.366	6	0.061		
7	Estuary	3.108	1	3.108	22.828	**0.0088**
	Site(Estuary)	0.545	4	0.136	0.704	0.6043
	Residual	2.321	12	0.193		
8	Estuary	0.682	1	0.682	0.380	0.5711
	Site(Estuary)	7.182	4	1.796	10.327	**0.0007**
	Residual	2.087	12	0.174		
9	Estuary	1.601	1	1.601	4.457	0.1024
	Site(Estuary)	1.437	4	0.359	1.305	0.3663
	Residual	1.651	6	0.275		
10	Estuary	3.036	1	3.036	33.568	**0.0044**
	Site(Estuary)	0.362	4	0.090	0.297	0.8740
	Residual	3.649	12	0.304		
11[a]	Estuary	5.850	1	5.850	4.300	0.1068
	Site(Estuary)	5.443	4	1.361	16.665	$\mathbf{7.67 \times 10^{-5}}$
	Residual	0.980	12	0.082		

Table I.3. Results of ANOVAs of decomposition rate (% wet weight/day) for each deployment period (except 3, with no replication of sites for Anglesea estuary). Significant *p*-values are shown in bold. a: data were transformed to natural logarithm before analysis to meet the assumption of homoscedasticity.

Cotton Strip Assay

Deployment	Location	Min	Max	Range	Mean	Std Dev	*n*
1	**A1 (shallow)**	14.7	21.9	7.2	16.9	1.36	1120
	A1 (deep)	15.4	20.7	5.3	17.7	1.15	1120
	P1 (shallow)	14.7	19.4	4.7	16.6	1.08	1243
	P1 (deep)	15.6	17.1	1.5	16.7	0.26	1243
2	**A1 (shallow)**	15.8	26.4	10.6	20.9	2.52	1529
	A1 (deep)	17.1	25.1	8.0	20.9	1.96	1529
	P1 (shallow)	16.6	27.7	11.1	20.7	2.22	1496
	P1 (deep)	16.4	23.3	6.9	19.6	1.45	1496

Table I.4. Descriptive statistics of sediment temperatures recorded during deployments of cotton strips. All values are degrees Celsius.

Controls

Breaking strains of control strips for deployment 1 (air and procedural) ranged from 59.06 kgf to 68.67 kgf with an overall mean of 64.03 kgf and standard deviation of 2.36 kgf (see Section 7.2.2 for methodological details of deployment and tensile testing). Means and standard deviations of breaking strain for strips on any particular control ruler ranged from 60.78kgf to 66.27 kgf and 0.819kgf to 2.72 kgf respectively (Figure I.1). Twenty five of the 85 control strips used tore at the edge of the jaws where they were held in the tensometer. These results were discarded, creating uneven numbers of replicate strips per ruler and resulting in no valid controls for the deep half of Site A2 (Table I.5).

Site	Depth	Anglesea Realised N	Painkalac Realised N
1	**S**	5	4
	D	4	3
2	**S**	3	5
	D	0	4
3	**S**	4	4
	D	3	3
OM 1	**1**	2	-
	2	4	-
OM 2	**1**	5	-
	2	5	-
Air control		2	5

Table I.5. Replicates with valid breaking strain data from deployment 1 control rulers (initial *n*=5). OM (organic matter) sites were located at Anglesea at a shallow depth only. Controls were placed randomly within these sites.

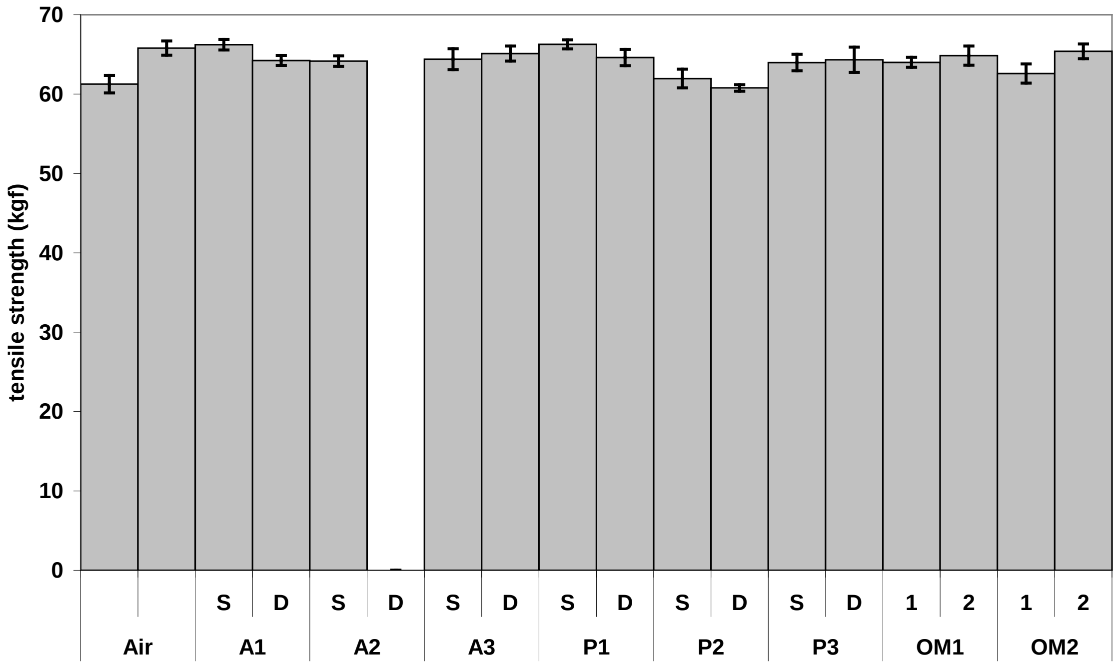

Figure I.1. Mean (±s.e.) of breaking strengths of air and procedural control strips, *n* was variable (see Table I.5).

Due to the unbalanced nature of the dataset, analysis of controls could not be done using nested ANOVA without removal of replicates representing whole sites. As there was no reason to expect differences at any particular scale, but rather an interest in differences between all physical locations, a one-way ANOVA was used to test for differences in breaking strength between all control rulers, including both air and procedural controls (Table I.8). A significant difference in mean tensile strength was found among control rulers (at α=0.05). Post-hoc multiple comparisons using Tukey's HSD test showed a significant differences between one of the air controls and the procedural control with the lowest mean tensile strength (air no. 2>P2 deep, p=0.036). Significant differences were also observed between Site P2 deep and the two procedural controls with the highest means (Site P1 shallow: p=0.024 and Site A1 shallow: p=0.014). No significant differences between pairs of controls from shallow and deep parts of each site were observed (minimum p=0.983), nor were any differences between randomly placed rulers in the OM sites (p=1.000, 0.694). To maximise the ability for generalisation of control procedural control tensile strengths within sites, combined means from strips on shallow and deep rulers were used as y_o on a site basis.

Source	**SS**	**d.f.**	**MS**	***F***	***p***
Controls	165.4	16	10.34	2.634	0.005
Error	188.4	48	3.924		

Table I.6. Results from one-way analysis of variance of control strips from deployment 1. Visual examination of residuals and expected values showed the assumptions of normality and homoscedasticity to be valid.

Compared with the controls from the first deployment, procedural controls from the second deployment were much more variable between individual rulers (Figure I.2). It is thought that these results were primarily an artifact of more humid storage conditions during the deployment (unlike the procedural controls from deployment 1, the strips from each control ruler were air-dried and then stored in a sealed plastic bag). Results for the air controls were similar to those from deployment 1, suggesting that differences among procedural controls from the second deployment were due to variable decomposition of the strips that had been inoculated with estuarine microbes while in inappropriate storage conditions. For this reason, it was necessary to use site means of procedural controls from deployment 1 as y_o in calculations for the second deployment.

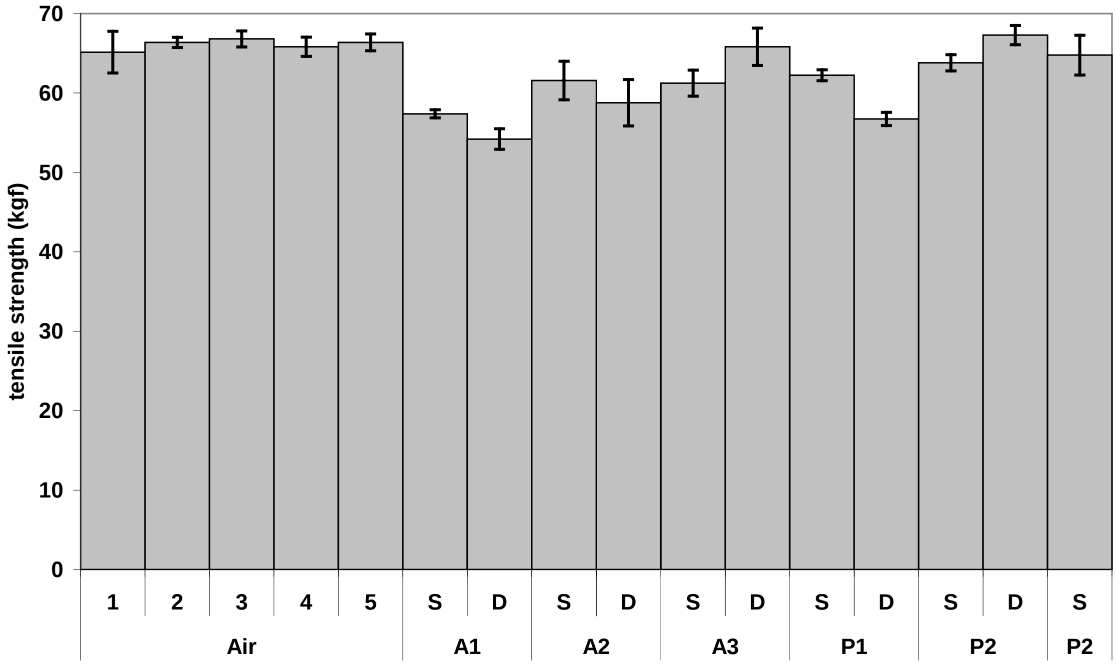

Figure I.2. Mean (±s.e.) of breaking strengths of air and procedural control strips for deployment 2, *n* was variable, between 3 and 5.

Main comparison

Source	S.S.	d.f.	M.S.	*F*-ratio	*p*
Time	0.000573	1	0.000573	1.524	0.2846
Estuary	0.025739	1	0.025739	36.783	**0.0037**
Depth	0.006740	1	0.006740	18.123[a]	**0.0009**
Site(Estuary)	0.002799	4	0.000700	1.431[a]	0.3012
Ruler(DepthxSite(Estuary))	0.004126	22	0.000188	1.757	0.0904
TimexSite(Estuary)	0.001504	4	0.000376	3.522	**0.0213**
TimexRuler(DepthxSite(Estuary))	0.002562	24	0.000107	2.357	**0.0006**
EstuaryxDepth	0.000396	1	0.000396	1.331[a]	0.2980
TimexDepthxSite(Estuary)	0.000761	4	0.000190	1.782	0.1654
Residual	0.010101	223	4.53×10^{-5}		

Table I.7. Results of ANOVA for main comparisons of decomposition rate of cotton strips with the interaction terms: TimexEstuary, TimexDepth, DepthxSite(Estuary) and TimexEstuaryxDepth pooled with the Residual term. Significant terms are indicated by bold *p*-values. a=pseudo-F ratio with d.f. of 1,13; 5,9 and 2,13 for marked terms from top to bottom.

Source	S.S.	d.f.	M.S.	*F*-ratio	*p*
Time	9.771	1	9.771	298.499	**6.59×10^{-5}**
Estuary	4.187	1	4.187	0.302	0.637
Site(Estuary)	12.311	4	3.078	94.025	**3.30×10^{-4}**
TimexEstuary	10.860	1	10.860	331.783	**5.34×10^{-5}**
TimexSite(Estuary)	0.130933	4	0.0327	0.124	0.972
Residual (pooled)	6.325556	24	0.264		

Table I.8. Results of full ANOVA for comparisons of decomposition rate in litter bags for the two deployments that coincided with cotton-strip assays. Significant terms are indicated by bold p-values. Results of the same analysis where the timexsite(estuary) interaction is pooled is presented in Table 7.7.