CHAPTER 1: Phytoplankton Ecology and Eutrophication: An Introduction

1.1 Phytoplankton and algal blooms

Phytoplankton are microscopic autotrophic or mixotrophic organisms that occur in water ranging from fresh to hypersaline. These organisms often exist in a state of quasi-equilibrium, where biomass production and loss processes are in balance (Evans and Parslow, 1985) resulting in no net population growth and a relatively static biomass. Phytoplankton ecology and an understanding of the factors regulating bloom development and decline are a pivotal component in the overall health of any aquatic system. Phytoplankton biomass, growth rate and species composition can be influenced by a number of environmental factors including light, turbidity, nutrient availability, temperature and grazing. Periodically there may be transient departures from the *status quo* when population densities and biomass may increase markedly resulting in what has historically been classified as a bloom (Paerl, 1988; Hallegraaf, 1993). This basic characterisation of what constitutes a bloom, historically based in terms of biomass, has been re-evaluated over recent decades in the light of the increasing scientific and public awareness of phytoplankton populations, or harmful algal blooms (HAB) that have deleterious effects, either directly or indirectly, on human socio-economic conditions (Smayda, 1997).

Algal blooms may cause non-toxic effects on humans, such as skin rashes and irritation or may cause the death of fish through induced anoxia in the water column in conditions of high biological oxygen demand overnight or, alternatively or additionally, they may cause species specific hepatotoxic or neurotoxic effects depending on the type of bloom. HABs and non-toxic blooms may have deleterious effects on the commercial and recreational use of rivers, requiring management or intervention to minimise economic or social impacts. Ability to predict their occurrence or positively influence or reduce their effects will be important in the maintenance of socially, recreationally and economically viable waterways.

1.2 The Issue of Eutrophication

The level of eutrophication of a system is assessed on the biomass and persistence of high concentrations of phytoplankton chlorophyll *a* (Chl*a)*. Globally there has been an increasing concern over anthropogenic eutrophic conditions that lead to higher primary production (Epstein *et al*. 1993) and consequently more productive heterotrophic communities. An increasing frequency of HABs has been linked to increasing levels of eutrophication (McComb, 1981; Smayda, 1990; Epstein *et al*., 1993; Hallegraeff, 1993; John, 1994; Justić *et al*., 1995; Paerl *et al*., 1999). There are three major anthropogenic influences affecting phytoplankton biomass and bloom development. Pollution from external sources of nutrients such as sewage and fertilisers, soil erosion and acid rain can add nitrogen and phosphorus to a system. Atmospheric deposition, land run-off, and river flow play an important role in regulating estuarine primary production and biomass accumulation (Malone *et al.,* 1988; Paerl *et al*., 1990; Gallegos *et al*., 1992; Mallin *et al*., 1993). Indirect effects influencing phytoplankton biomass include changes in trophic interactions in populations. This may be a result of phytoplankton removal by pelagic zooplankton, benthic filter feeders and/or higher predators such as planktivorous fish species, with the consequent reduction of controlling effect that these higher orders have on the biomass of lower trophic levels (Shapiro and Wright, 1984; Post and McQueen, 1987; Buskey *et al*., 1997). Habitat loss, particularly of riparian vegetation and wetlands that sequester nutrients (Chambers, 1987; Epstein *et al*., 1993), results in an increase in the amounts of nitrogen and phosphorus entering aquatic systems. Wetlands are capable of removing up to 98% of the nitrogen and phosphorus from the water passing through them (Chambers, 1984 & 1987).

Nutrient loadings to waterways around the world have increased with increases in human activity (Heathwaite *et al*., 1996) but there have also been changes in nutrient quality and stoichiometry (Justić *et al*., 1995), with a concurrent increase in both organic and inorganic nitrogen concentrations (Butler *et al.,* 1979; Smayda, 1989 & 1990). The paradoxical situation has been noted (Paasche *et al*., 1984), of algal blooms occurring in conditions of very low nutrient, especially nitrogen. Increased capacity for waterways to support algal growth has lead, in many cases, to blooms of such intensity as to cause catastrophic deoxygenation and death of higher consumers. These cultural or accelerated eutrophic conditions are primarily a result of changes in land use through agricultural practices and industrial and population-based waste production (Reynolds, 1997). This global phenomenon has created the need for management strategies that enable predictive ability for phytoplankton succession (Roelke, 1998; Roelke *et al.*, 1999). This will require an understanding of the complex interaction of environmental and biological factors that control phytoplankton growth and community composition.

1.3 Eutrophication in the Swan River - causes for concern - the need for action.

Extreme climatic variability coupled with clearing of remnant vegetation for agricultural purposes have led to modifications to the hydrology and ecology of many river and estuary systems in Australia (Harris, 1995). Relatively dense human settlement combined with changes in land-use of catchment areas, such as clearing of remnant vegetation for agricultural purposes, have had major impacts on water quality of estuarine ecosystems both in Australia and worldwide (Harris, 1995). Major changes in land-use of the Swan coastal plain, Western Australia, since early European settlement (Riggert, 1978) plus modifications to the hydrology of the Swan River have lead to a decline in the health of the Swan-Canning Estuary system. Warning signs have been periodic events symptomatic of eutrophication, such as red tides (Hamilton *et al.,* 1999), fish kills, cyanobacterial blooms (Hamilton, 2000) and the accumulation of organic matter in the bottom sediments of deep holes (Douglas *et al.,* 1996).

1.3.1 Location and description

The Swan River Estuary (31°57”S Latitude and 116°04”E Longitude), fed by a catchment area of approximately 121,000 km^2 (Peters and Donohue, 2001), is the second largest estuary in south-western Western Australia. It enters the Indian Ocean at the port of Fremantle and extends approximately 60km upstream from its mouth to Ellen Brook, one of its subsidiary streams. The Swan River Estuary is usually considered to include a lower estuary which has a large tidally driven marine influence, between Perth Water and Fremantle, and an upper estuary, the 40km of water between the causeway and Guildford (see Figure 1.1). This section of the river is narrow and shallow (generally < 4 m depth) with deeper pockets (6 m) and exhibits strong seasonal differences influenced by tidal and climatic conditions (Hodgkin, 1987). Its hydrodynamic properties are regulated primarily by climate. Cool wet winters (maximum mean 17.4°C during July), with approx. 630 mm rain between April and November, create seasonal river flow. This flow diminishes during the hot, dry summer (maximum mean 29.9°C during February) to create a true tidally-driven estuary situation with reduced river flow (Spencer, 1956; Hodgkin and Lenanton, 1981; Hodgkin, 1987). A salt wedge penetrates some 50 km upstream as the winter rainfall run-off declines. Average upper estuary depths are 2-3 m. A series of deeper pockets (6 m depth) occur along the upper reaches of the river and these are reported to have higher concentrations of NH_4^+ and PO_4^{3-} (Douglas *et al.*, 1996; Jack, 1987). A recognised pattern of bloom succession (John, 1987; Thompson and Hosje, 1996; Twomey and John, 2001) has been established within the system (see Figure 3.2). Chlorophyte-dominated blooms occur in the upper estuary during early spring, giving way to dinophyte- and cryptophyte-dominated blooms through the summer and autumn periods. Seasonal variation of rainfall, and its subsequent effect on the spatial distribution of salinity, coupled with nutrients, have been shown to influence the distribution and succession of phytoplankton species (John, 1984 & 1987; Thompson, 2001; Twomey and John, 2001).

Hodgkin (1987) describes the Swan River as a seasonal estuary. The Swan River Estuary dynamics were classified more recently by Stephens and Imberger (1996) as varying between a winter rain-driven gravitational overflow and a salt wedge condition governed by both discharge and topographic constraints, with the degree of flushing influenced by tidal dynamics. This system alternates between three distinct phases. It is a poorly flushed water course during the summer drought period, or a river during periods of high rainfall where the subsequent high discharge flushes away the salt water. Between these two phases are periods of high stratification where the salt-wedge regime is established during periods of decreasing (late spring) or increasing (autumn-early winter) river discharge (Douglas *et al.*, 1996). This salt wedge penetrates some 50km upstream as the winter rainfall run-off declines. Like other estuaries of south-western Australia, the Swan is unusual for its highly variable biological and hydrological characteristics. These are a reflection of the extreme seasonality of river flow, a direct consequence of rainfall patterns (Spencer, 1956; Stephens and Imburger, 1996).

Eutrophication is a recognised and often severe problem in many rivers and estuaries world wide, including rivers on the eastern seaboard and in the south-west of Australia.

Since European settlement there has been a 16 fold increase in total nutrient yields in the Swan-Avon river catchment (Viney and Sivapalan, 2001). The trend towards larger

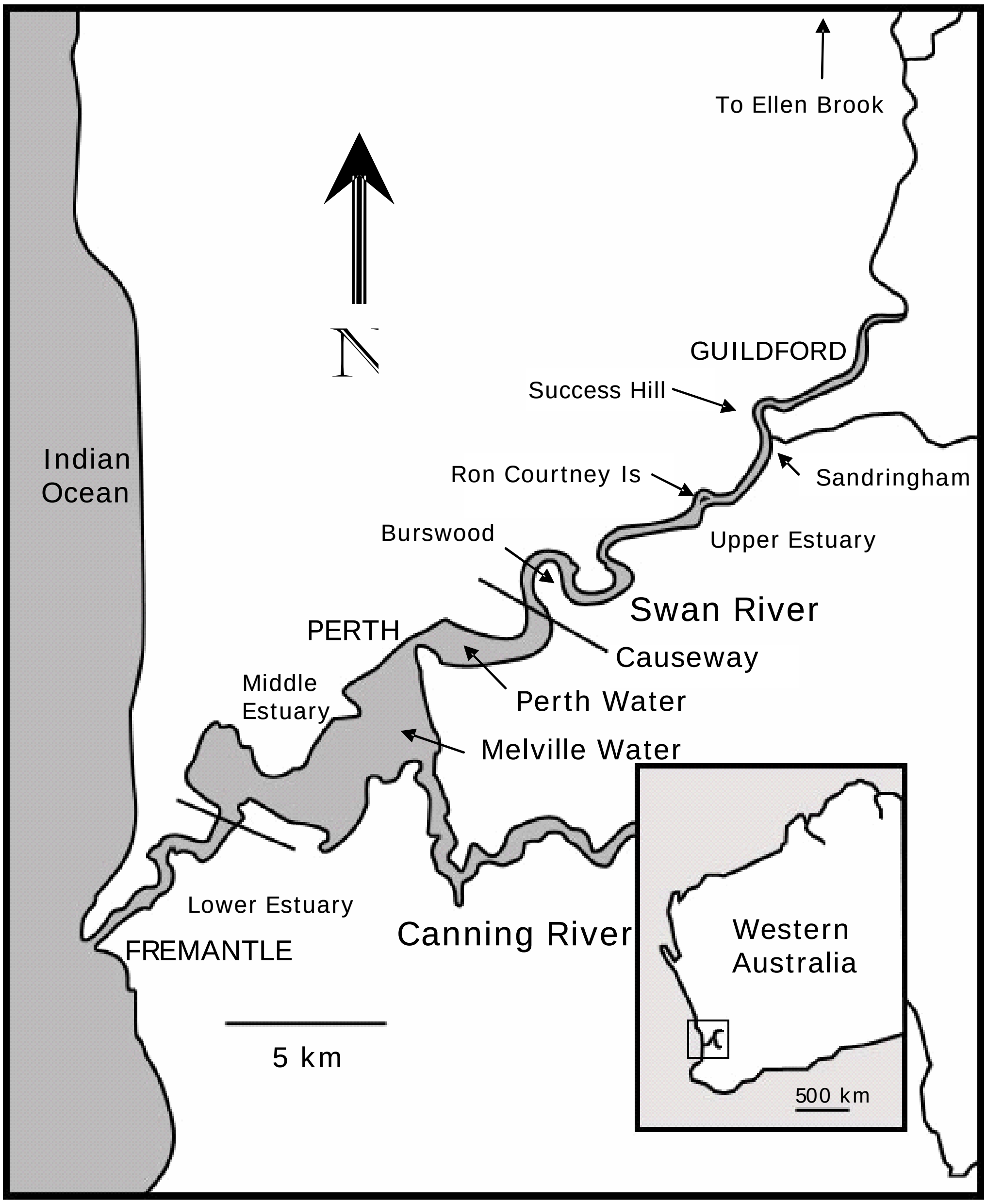

Figure 1.1 Location map for the Swan-Canning Estuary, Perth, Western Australia showing lower, middle and upper sections of the Swan Estuary and the major landmarks of Perth city, the port of Fremantle and Guildford in the upper reaches.
Inset: Location of Swan-Canning river systems in south-west of Western Australia.

and more persistent bloom events has been noted over the past few decades as being cause for concern (John, 1987 & 1994; Thompson and Hosja, 1996; Thompson *et al.*, 1996; Thompson, 2001). Several catastrophic events due to algal blooms in the Swan River have already occurred. Of note was the January 1992 deoxygenation of the entire length of the Swan River between the causeway and Midland (see Figure 1.1) following the collapse of a large dinoflagellate bloom. This resulted in mass mortality of fish and benthic invertebrates (Deeley *et al.*, 1993). Although toxic cyanobacterial (blue-green algae) blooms have occurred sporadically in the Swan River, giving rise to increased management efforts (Thompson *et al.*, 1997), concern that conditions conducive to development of harmful algal blooms in the Swan River may occur were realised early in 2000 (January 2000). A large-scale bloom of the fresh-water hepatotoxic blue-green alga *Microcystis aeruginosa* form *flos-aquae* (Wittrock) Kircher 1898 (Komarek and Anagnostidis, 1999) (Jacob John, pers.comm.; Hamilton, 2000) during late January and early February of this year followed unseasonal rainfall and caused thick green scums along shorelines. This HAB, the largest recorded for the Swan River to date, necessitated the closure of the whole length of the Swan River (approx 50 km) for a period of 12 days (John, 2000). Extensive fish kills (>250 000 mortalities recorded) occurred again in 2003 (April – June) caused by the ichthyotoxic dinophyte *Karlodinium micrum* (density $>10^6$ cells ml^{-1}) and were attributed to elevated nitrate levels from runoff in the Swan-Canning catchment area following high rainfall and a subsequent warm spell (http://www.wrc.wa.gov.au/srt/algalalert/FishKillAlgae.pdf, access date 29/12/2003). The extent of these HAB, combined with its drastic effect on the socio-economic aspects of the river, reinforced the need for waterways health management and a predictive ability for the future reduction or elimination of nuisance and harmful algal blooms.

The main theme that emerged from a Scientific Committee on Oceanic Research (SCOR) working group on mathematical models in biological oceanography was the need "to have at least as much information on the fluxes as on the biomasses" (Platt *et al.*, 1981) for an understanding of biological oceanographic systems. The most common fluxes considered are through, firstly, trophic transfers such as grazing, predation, egestion and, secondly, detritus formation and also elemental cycling which includes nutrient uptake, excretion and advective processes. Fluxes are a product of population density and organism physiology. Top-down and bottom-up controls operate simultaneously. The relationship between top-down and bottom-up control varies depending on the scale of interest, and has important consequences for how we model phytoplankton biomass control in a natural food web. Grazing and nitrogen recycling are intricately connected: i.e. the presence of large zooplankton simultaneously provides top-down control of biomass and bottom-up nutrient supply (Glibert, 1998). It has become increasingly evident that an evaluation of the relative strength of each, predators and resources, is needed to determine how the development and sustainability of populations or communities is regulated.

The issue of “nuisance” blooms in the Swan River estuary has primarily focused on bottom-up control of phytoplankton dynamics relating to investigations into the physical and chemical conditions of the estuary (John, 1987; Gerritze, 1992; Hosja and Deely, 1994; John, 1994; Douglas *et al*., 1996; Thompson and Hosja, 1996; Hamilton *et al*., 1999; Horner Rosser, 1998; Horner Rosser and Thompson, 2001). However, little is known about the relationship between phytoplankton biomass and zooplankton grazers in the Swan-Canning Estuary (Rose, 1998). Studies have demonstrated that copepod grazing, by locally occurring *Gladioferens imparepes* and *Sulcanus inflictus*, can account for loss of up to 45% of standing stock (Griffin *et al*., 2001; Griffin and Rippingale, 2001), but little is known about the impact of other locally occurring zooplankton grazers. It has been recognised that grazing pressure can both enhance, through nutrient release and recycling, and reduce phytoplankton biomass (Stone, 1990; Svensson and Stenson, 1991; Ferrier-Pagès and Rassouzadegan, 1994; Koepfler and Lewitus, 1995). The importance of nano- and micro-zooplankton in foodwebs and planktonic community dynamics (Ferrier-Pagès and Rassouzadegan, 1994; Burkhill *et al*., 1995), particularly those tending towards eutrophy, has been emphasised. No studies detailing the microheterotroph community or its role in top-down control of phytoplankton biomass and/or species composition have been undertaken for the Swan-Canning Estuary which, like other southern Western Australian estuaries, is subject to seasonally highly varying environmental conditions.

1.4 General Objectives of this Research

This study has been one component of a collaborative project with the aim of enabling predictive management strategies for the minimisation of eutrophication and control or prevention of harmful algal blooms (HAB) in the Swan and Canning Rivers. Based on a two dimensional model developed by the Centre for Water Research, University of

Western Australia under the guidance of Dr David Hamilton, this project was designed to provide ecological information on the phytoplankton assemblages of the upper Swan River.

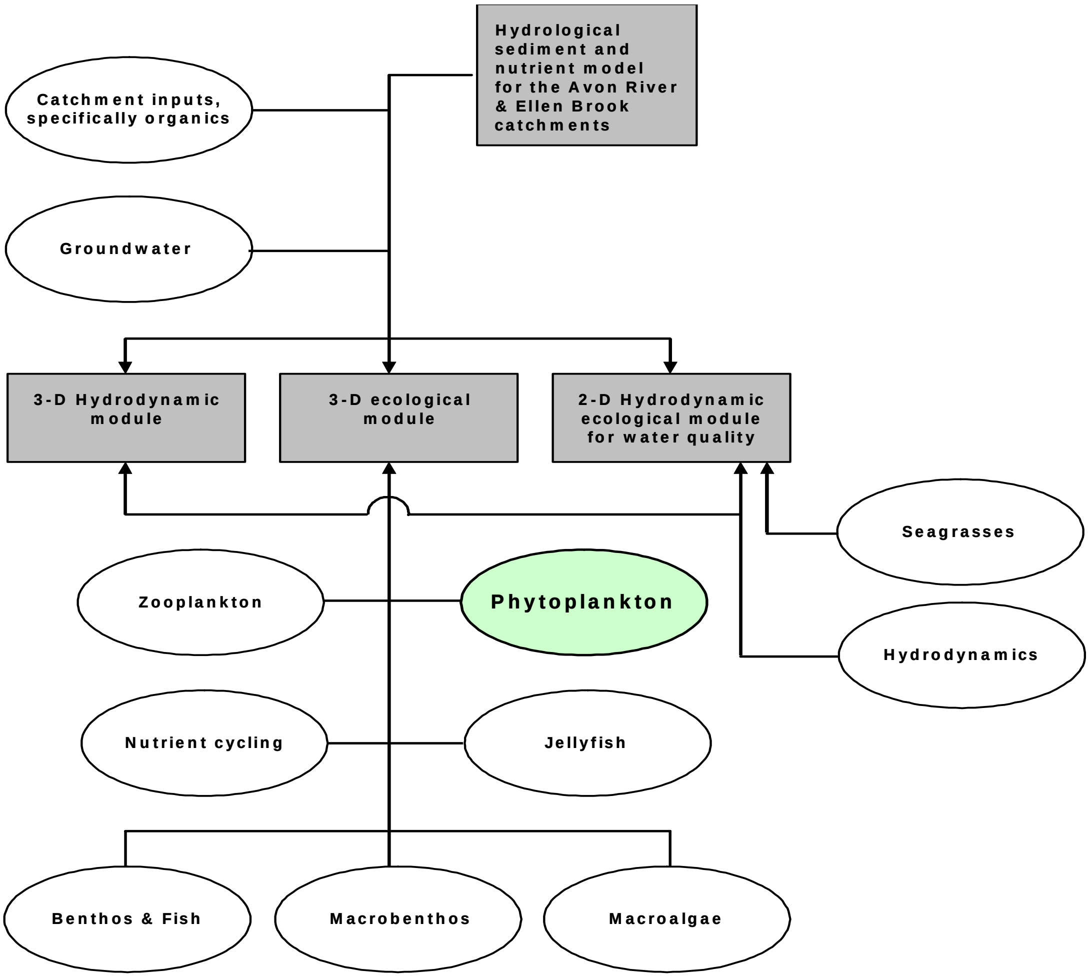

Figure 1.2 Outline of the Estuarine Research Foundation of WA funded interdisciplinary project to model the Swan-Canning Estuary, showing the relationship of this phytoplankton project to the whole.
(Modified from Hamilton, 1996)

The aim of this study was to determine the extent to which phytoplankton populations in this region are controlled by physico-chemical parameters (bottom-up) or biological (top-down) factors. That is, to determine which factors, physical, biological or physiological, have the greatest influence on controlling phytoplankton biomass under various ambient conditions for this system. It provides the first information on nitrogen fluxes influencing the phytoplankton biomass occurring in the upper Swan River Estuary. This region of the estuary was chosen as representative of the region most influenced by allochthonous input and which historically has been the region of most frequent and intense algal bloom activity over the past decade. Flux rates and biomass information from this research have already been used to validate a model of the Swan River developed as a tool to aid predictive modelling and management of the Swan River environment (Hamilton *et al.*, 1999).

The objectives of this thesis, presented in the form of separate chapters, are:

1. To establish short- (within bloom) and medium-term (annual) diurnal variation in the physico-chemical environment of the upper Swan River Estuary (Chapter 2).
2. To determine short-term (within bloom) and medium-term (annual) diurnal variation in phytoplankton species composition of the upper Swan River Estuary (Chapter 3).
3. Following previous studies that indicate nitrogen to be the limiting nutrient in this system, to determine flux rates of nitrogen species within the system. Seasonal uptake kinetics for the inorganic nitrogen sources (NO_3^- and NH_4^+) and the organic nitrogen source (urea) were determined (Chapter 4). The nitrogen source(s) of preference and the rates of uptake of different nitrogen sources (nitrate, ammonium and urea) on a short-term (within bloom) and medium-term (annual) time scale (Chapters 5 and 6) are determined and compared with ambient levels during the study and related to long-term trends in ambient nutrient levels.
4. To determine the microheterotroph species composition and grazing pressure of micro-plankton (< 300μm) and nano-plankton (< 20μm) size classes in the upper Swan River Estuary (Chapter 7). The influence of mesotrophs, specifically Copepod grazing, on phytoplankton biomass was investigated by S. Griffin (2003) as part of a different sub-programme of the larger collaborative study.
5. To combine the results from the different research sections in this study to determine the balance between bottom-up and top-down control for the upper Swan River Estuary (Chapter 8). Ultimately the data collected will be used to validate a predictive model developed for the Swan River, in collaboration with The Centre for Water Research at the University of Western Australia (with Dr David Hamilton). Due to time constraints, this component of the analysis and

interpretation will be beyond the scope of this dissertation. The aim will be to enhance or facilitate predictive management capabilities for this economically and recreationally important river system

As each Chapter deals with a different aspect of phytoplankton ecology (ie. environmental or physiological conditions), each chapter is presented with its own relevant literature review, techniques section and discussion. Therefore a degree of repetition has been unavoidable. Published papers relevant to this research project, plus seminars presented at National or International scientific meetings and based on the results of this research, are listed in Appendix III.

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