Impacts of Hydrological Changes on Phytoplankton Succession in the Swan River, Western Australia

TERENCE U. CHAN[1],*, DAVID P. HAMILTON[1],†, BARBARA J. ROBSON[1], BEN R. HODGES[2], and CHRIS DALLIMORE[1]

[1] *Centre for Water Research, University of Western Australia, Crawley WA 6009, Australia*
[2] *Department of Civil Engineering, University of Texas at Austin, Austin, Texas 78712*

ABSTRACT: **The Swan River estuary, Western Australia, has undergone substantial hydrological modifications since pre-European settlement. Land clearing has increased discharge from some major tributaries roughly 5-fold, while weirs and reservoirs for water supply have mitigated this increase and reduced the duration of discharge to the estuary. Nutrient loads have increased disproportionately with flow and are now approximately 20-times higher than pre-European levels. We explore the individual and collective impacts of these hydrological changes on the Swan River estuary using a coupled hydrodynamic-ecological numerical model. The simulation results indicate that despite increased hydraulic flushing and reduced residence times, increases in nutrient loads are the dominant perturbation, producing increases in the incidence and peak biomass of blooms of both estuarine and freshwater phytoplankton. Changes in salinity associated with altered seasonal freshwater discharge have a limited impact on phytoplankton dynamics.**

Introduction

The ecology and biodiversity of estuarine and coastal waters in many parts of the world are under threat from increasing anthropogenic inputs of nutrients (Nixon 1995; Cloern 2001). Many of these threats can be attributed directly to expansion of human populations along riparian zones and coastal catchments (Cooper and Brush 1993). The threats to coastal ecosystems are especially exacerbated in Australia where 80% of the population lives within 50 km of the coast and the major land drainage basins have undergone large-scale land clearing and hydrological modification since European settlement (Harris 2001). Declining water quality and high rates of sedimentation are the most obvious manifestations of nutrient enrichment and land clearing (Zann 1995).

Knowledge of the nutrient assimilative capacity of coastal and estuarine ecosystems is essential for management and rehabilitation. Globally, current large-scale efforts to control eutrophication are based largely on the premise that improvements in biodiversity and water quality will be linked directly to reductions in nutrient loads. Such assessments give only rudimentary consideration to response times, hysteresis effects, and hydrological controls, thereby neglecting possible non-linear responses to changes in nutrient loading (Harris 1999).

While the major focus of eutrophication man-agement has been on nutrient control strategies, it is also important to consider hydrological modifications that may have an impact on the eutrophication response. On the Australian continent, weirs and dams have contributed directly to algal blooms by increasing residence times and stratification of the impounded waters (Sherman et al. 1998) and decreasing flushing of downstream estuaries (Davies and Kalish 1994), even though some dredging or estuary opening strategies have improved water quality through increasing flushing with marine water (Hearn and Robson 2000; Ranasinghe and Pattiaratchi 2000). The complexities of the interactions among freshwater flow and composition, estuary topography and hydrodynamics, and human alterations of these features indicate that numerical models may be important in quantifying the hydrological responses of estuaries and the resultant changes in water quality.

The objective of this study was to develop a quantitative understanding of the way in which the hydrology and water quality of a Western Australian estuary, the Swan River, have been altered by changes in watershed land use patterns and tributary regulation associated with European settlement and development. We use a coupled hydrodynamic-ecological model to make assessments for pre-modification and post-modification cases, with the major focus placed on the likely changes to phytoplankton biomass and species composition.

Study Site

GENERAL DESCRIPTION

The watershed of the Swan River is large (121,000 km²) and dominated by the Avon River

---

* Corresponding author; tele: (61 8) 9380 1683; fax: (61 8) 9380 1015; e-mail: chan@cwr.uwa.edu.au.

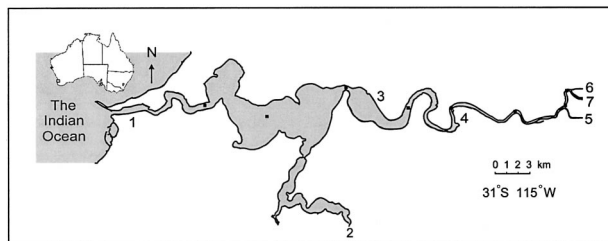† Current address: Department of Biological Sciences, University of Waikato, Hamilton, New Zealand.

Fig. 1. Map of the Swan River showing the nine field monitoring sites (■) and the locations of some of the major changes that have had an impact on the estuary hydrology (1–7 described below). Note the narrow constriction (the Narrows) between 2 and 3, which delineates the lower basin towards the ocean from the upper reaches. 1) Fremantle Channel: dredged from ~2 to 14 m (occurred in 1892); 2) Canning River: Kent St Weir (1920s) and Canning Dam (1940); 3) Perth City: urbanization; 4) Ascot Waters: boat harbors and marinas (1990s); 5) Helena River: Mundaring Weir (1902); 6) Ellen Brook: agriculture (1950s) and urbanization (1990s); and 7) Avon River: clearing and salinization (1900s) and river training (1958–1971).

watershed (120,500 km²). Rainfall varies over the watershed from ~900 mm yr^{-1} in coastal regions to ~300 mm yr^{-1} in eastern regions. The climate can be considered to be Mediterranean with around 70% of rainfall confined to the winter and spring months of June through September. Tributary runoff is highly seasonal with little or no flow occurring in the first 4–5 mo of each year in the Avon River. Runoff from smaller tributaries is also highly seasonal, but may vary from negligible in summer (e.g., Ellen Brook) to continuous in the case of some urban drains (Donohue et al. 2001). Groundwater inflows, which occur mostly through the sandy soils of the Swan Coastal Plain, vary little seasonally, and may contribute up to 10% of freshwater inputs to the estuary in summer and fall months when flows from surface-fed tributaries are small (Linderfelt and Turner 2001).

In summer and fall, water of marine origin intrudes up the Swan River, along the Swan Coastal Plain, to approximately 50 km upstream of the estuary mouth at Fremantle (Fig. 1). In winter, rainfall and associated streamflow drives the salt wedge seaward, occasionally close to the ocean entrance at Fremantle in very wet seasons (Stephens and Imberger 1996). Tidal excursions of the salt wedge are typically on the order of 1–3 km although synoptic forcing may displace the salt wedge by around 10 km, corresponding to the duration of passage of low-pressure and high-pressure systems (Hamilton et al. 2001).

The highly seasonal hydrology of the Swan River estuary is reflected in a well-documented succession of phytoplankton taxa (John 1994; Thompson and Hosja 1996; Chan and Hamilton 2001). The phytoplankton dynamics are of particular interest

in the upper estuary reaches (~20 to 40 km from the mouth), from the constriction at the Narrows up to the confluence with Helena River (Fig. 1), as problematic algal blooms occur frequently in this region. The high-flow period of winter and early spring is usually dominated by freshwater diatoms, which are typically succeeded by a short-lived bloom of freshwater chlorophytes. In summer and fall, estuarine and marine species are dominant and typically show transitions between dinoflagellates (e.g., *Gymnodinium* spp. and *Prorocentrum* spp.) and the cosmopolitan coastal diatom *Skeletonema costatum* (Chan and Hamilton 2001). Blooms of dinoflagellates (Hamilton et al. 1999) and more recently (February 2000) the blue-green alga *Microcystis aeruginosa* (Hamilton 2000) are of particular concern in terms of biodiversity, amenity, and long-term impacts on the estuary ecosystem.

POST-EUROPEAN MODIFICATIONS

The hydrology of the Swan River has undergone substantial modifications in the past century, and it is likely that these changes have also affected phytoplankton succession. Several dams, notably Canning Dam (Fig. 1, location 2) and Mundaring Weir (Fig. 1, location 5), were constructed for water supply through the 1900s, restricting freshwater discharges to the estuary. In their original state, these tributaries (i.e., Canning River and Helena River) were unlikely to have exerted a major influence on winter flows, which are dominated by the Avon River. Their relative contribution would have been greater in drier months due to the proximity of the tributaries to the high rainfall zone near the coast and the extended period of little or no flow in the Avon River.

In contrast to flow reductions from reservoir construction, clearing of native vegetation is estimated to have increased flows in the Avon River by 4–5 times over the past 100 years, and has increased groundwater recharge and nutrient and sediment discharges from the catchment (Viney and Sivapalan 2001). Clearing was particularly widespread between 1940 and 1970. The subsequent increases in runoff prompted adoption of a river training scheme, in which large sections of the Avon River were cleared of vegetation (ripping of the river bank), then straightened and deepened by bulldozer (Riggert 1978). It is now generally accepted that these modifications had a severe impact upon the ecology of the Avon River and led to major problems with sediment erosion and riverbank stability along many parts of the river (Harris 1996). Perhaps of even more concern is the progressive increase in salinization, waterlogging, and land degradation in the Avon River catchment, which has resulted from clearing of

remnant vegetation and reduced water loss via evapotranspiration (Harris 1996).

Modeling

A three-dimensional hydrodynamic model (Estuarine and Lake Computer Model; ELCOM) coupled with an ecological model (Computational Aquatic Ecosystem Dynamics Model; CAEDYM) was used to simulate physical and ecological processes in the Swan River estuary.

ELCOM has been developed to simulate hydrodynamics and transport in stratified water bodies with spatially-varying wind stress, surface heat exchange, tidal boundaries, and multiple inflows (including groundwater sources). The simulation method solves the three-dimensional Reynolds-averaged, unsteady, hydrostatic, Boussinesq, Navier-Stokes, and scalar transport equations on a Cartesian mesh. The hydrodynamic algorithms are a semi-implicit, finite-difference approach based on a second-order Euler-Lagrange advection of momentum with an implicit solution of the free surface evolution. Scalar transport uses a conservative discretization of a flux-limiting third-order method. Turbulence modeling uses a mixed-layer approach in the vertical with constant eddy viscosities for the horizontal. Detailed descriptions of the hydrodynamic model can be found in Hodges et al. (2000) and Hodges (2000).

CAEDYM consists of a set of subroutines containing a series of equations that describe the major biogeochemical processes influencing water quality. These include primary and secondary production, nutrient and metal cycling, oxygen dynamics, and the movement of sediment. The equations relevant to the phytoplankton model are described in detail by Griffin et al. (2001), with the exception that no grazing by zooplankton is included in this application. Zooplankton grazing was considered in this study to be of secondary importance relative to the effects of advection and transitions between freshwater and brackish conditions (Chan and Hamilton 2001). The biota were represented in the model simulations by four taxa of either freshwater and estuarine phytoplankton.

ELCOM and CAEDYM are coupled such that ELCOM simulates salinity and temperature, passing values for these parameters to CAEDYM for modification of ecological state variables, while CAEDYM passes the water quality variables to ELCOM for advective and dispersive processes (Fig. 2).

In this study, the coupled model is applied to a 40 km length of the Swan River estuary, from the mouth at Fremantle to the confluence with Helena River (Fig. 1). The simulation grid uses an along-channel and cross-stream coordinate system that
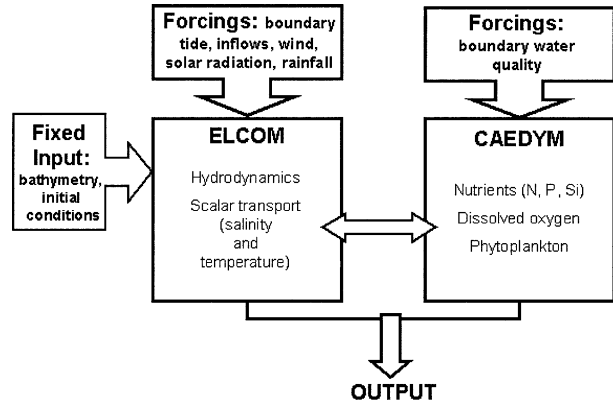

Fig. 2.   Schematic of the coupling between the hydrodynamic model ELCOM and the ecological model CAEDYM.

effectively straightens the estuary. This approach neglects effects of curvature in the river, which can be shown to be a second-order effect (Hodges and Imberger 2001). Neglecting the river curvature significantly simplifies the model computations at the expense of cross-channel processes. While cumulative effects of cross-channel processes are important in sediment transport and erosion studies, the overall impact should be negligible for the residence time and flushing rate observed in the upper Swan River. The grid cells have a longitudinal aspect ratio of 10:1, using 1000 m in the along-river direction and 100 m across-river. In the vertical direction, a grid spacing of 0.5 m is used in the upper 7 m of the domain, increasing incrementally to 2 m in the bottom-most layer. This paper focuses on results for the Swan River upstream of the Narrows, where the depth is less than 6 m and is resolved with 0.5 m vertical spacing.

The bathymetry used in the model was obtained from an intensive bathymetric survey (20 by 20 m resolution) commissioned by the Water and Rivers Commission over the entire estuary in 1997, and averaged to the required model grid resolution. Meteorological forcing inputs included solar radiation, wind, air temperature, humidity, and cloud cover, which were entered into the model based on 15-min readings taken at Perth Airport, 5 km to the south of the most upstream estuary sampling station.

Model boundaries were defined at the ocean entrance, where tidal elevations were prescribed at 15-min intervals, and at the confluence with Helena River in the upper estuary (Fig. 1, location 5), where daily discharge was entered as a total for the six major, gauged tributaries; Avon River, Ellen Brook, Susannah Brook, Jane Brook, Henley Brook, and Helena River, and for smaller ungauged tributaries. The latter estimate was made by applying a rainfall runoff coefficient to each

catchment area based on the coefficient derived for the nearest gauged tributary. Other inputs included estimates of daily groundwater discharge and recharge on the south and north shores of the estuary, based on model simulations by Linderfelt and Turner (2001), and daily discharge from three gauged urban drains in the upper estuary and from the Canning River in the lower reaches. Localized surface runoff adjacent to the estuary was assumed to be 50% of the daily rainfall on the catchment of the lower estuary (Peters and Donohue 2001). This catchment consisted of a band of land around the estuary perimeter, varying from 0.5 to 1.5 km in width. Daily rainfall was also entered directly onto the water surface of the estuary. Water quality composition at the upper domain boundary and for the Canning River was derived from weekly sampling at these stations. Drain composition was derived from fortnightly sampling of one of the drains and was assumed to be identical for the other two drains, and for diffuse runoff from the catchment. Composition measurements included salinity, temperature, dissolved oxygen, phosphate, ammonium, nitrate, total phosphorus, total nitrogen, silica, biochemical oxygen demand, and suspended solids. Composition of groundwater inflows was based on average values from bore tests located in two transects across the estuary (Linderfelt and Turner 2001). Field measurement of nitrate in rainfall indicate that peak concentrations coinciding with the peak rainfall volume would result in a nitrate load of less than 60 kg yr^{-1}. The groundwater nitrate load has been estimated at 30–60 t yr^{-1} or about 10% of the nitrogen load in the upper reaches (Linderfelt and Turner 2001). Our calculated rainfall nitrate contribution is, at most, 0.2% of this. As nitrate was measured at higher concentrations than other nutrients (ammonium, nitrite, phosphate), direct rainfall was approximated as having negligible solutes. Data from the model simulations were compared with measured vertical profiles or surface, mid-depth, and bottom samples at 9 stations along the estuary (Fig. 1). The major focus of this study is the upper reaches, where 6 of the 9 stations are located and where the majority of algal blooms are reported (Thompson and Hosja 1996). Measured data at the estuary stations included the same parameters as those measured for the tributaries, as well as chlorophyll *a* (chl *a*) and Secchi depth. Surface (0–5 m) integrated cell counts were also taken at each station and differentiated to taxon level. A complete description of the methods and additional measurements taken for the estuary samples is given in Chan and Hamilton (2001). For use in the model, cell counts for each taxon were converted

to chl *a* as a measure of biomass according to chl *a* per cell values given in Griffin et al. (2001).

The model configuration for this study included phytoplankton parameters for their responses to light, salinity, temperature, nitrogen, phosphorus, silica, and carbon, as well as migration and settling velocities. Additional parameters were required for oxygen exchanges and nutrient cycling. Parameters were calibrated within the literature ranges observed for similar phytoplankton species or in other estuarine studies. These parameters included maximum growth and respiration rates, half saturation constants for nitrogen and phosphorus, light saturation, response to temperature, settling rates, and salinity tolerances for each of the four phytoplankton groups. The model calibration runs involved successive runs over one year with the aim to iteratively reduce differences between measured and simulated variables. The primary focus of the model calibration was to reproduce the observed changes in phytoplankton biomass and succession over a 1-yr simulation, but matching concentrations of nutrients and dissolved oxygen was also an integral part of the calibration.

Four different scenarios were developed to run as separate simulations, based on past conditions in the estuary. The effect of removing the Mundaring and Canning Weirs was simulated by adjusting Helena River and Canning River inflows according to gauged monthly inflows to these impoundments. A pre-European settlement scenario was simulated based on results from a watershed model that specifically examined inflow volume and composition prior to European settlement (Viney and Sivapalan 2001). The catchment model factors for reduction of flow (1/5th), phosphate (1/10th), total phosphorus (1/16th), ammonium (1/4th), and nitrate and total nitrogen (1/16th) in the Avon River, Ellen Brook, and Helena River, were applied to the inflow file input for the present-day case with all other inputs remaining the same as present. In the third scenario, flow was kept at present-day levels while incoming nutrients were reduced to pre-European levels as described above. In the final scenario, nutrients were kept at present-day levels, while flow was reduced to pre-European levels as above.

Results

Water quality comparisons between field, model, and scenario results, integrated over the volume of the upper estuary, are presented for salinity, inorganic nitrogen, and inorganic phosphorus (Fig. 3). The measured data are based on water column means from the six monitoring stations in the upper reaches. Model salinity results compare well with field measurements for most of the year, but
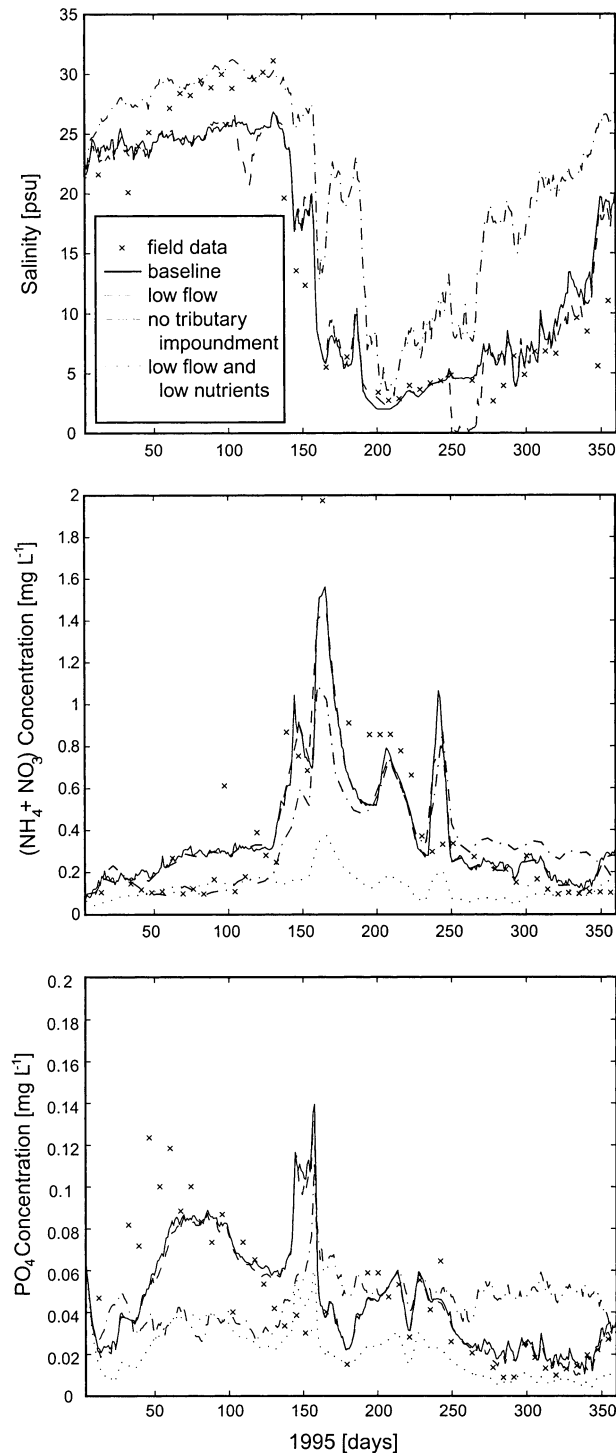
Fig. 3. A) Salinity integrated over the upper estuary. B) Dissolved inorganic nitrogen ($NO_3 + NH_4$) comparison. C) Filterable inorganic phosphorus (PO_4) comparison. Comparison of baseline (1995) case against a scenario with Canning Dam and Mundaring Weir removed and a scenario with reduced inflow corresponding to a pre-European catchment. Solid line (—) is the baseline case, $-\cdot-\cdot$ is the low flow case, $\_\cdot\_\cdot\_$ is the case without tributary impoundment, . . . is the low nutrient and low flow case, and x is the field data.
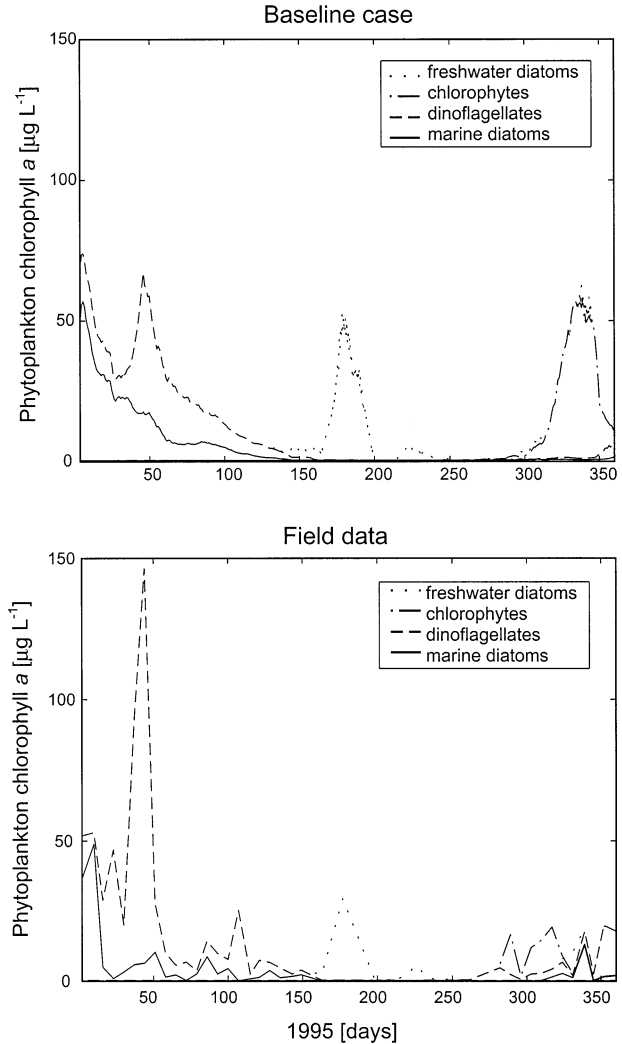

Fig. 4. Relative phytoplankton biomass (chlorophyll *a*) integrated over the upper estuary for the a) baseline (1995) case and b) field (1995) data.

there is some discrepancy during fall (days 60–150), with the upper reaches somewhat fresher than observed values. Simulations of inorganic nutrients match field measurements except during this same fall period, when they are lower than observed values.

Our simulation of biomass for the four primary phytoplankton groups integrated over the volume of the upper estuary with the corresponding field data are presented in Fig. 4. This simulation was the outcome of repeated model calibration runs that were designed to minimize errors between measured and simulated biomass of phytoplankton groups as well as nutrients and dissolved oxygen. The primary difference between the measurements and simulations occurs during the chlorophyte bloom (days 290–350), when the simulated

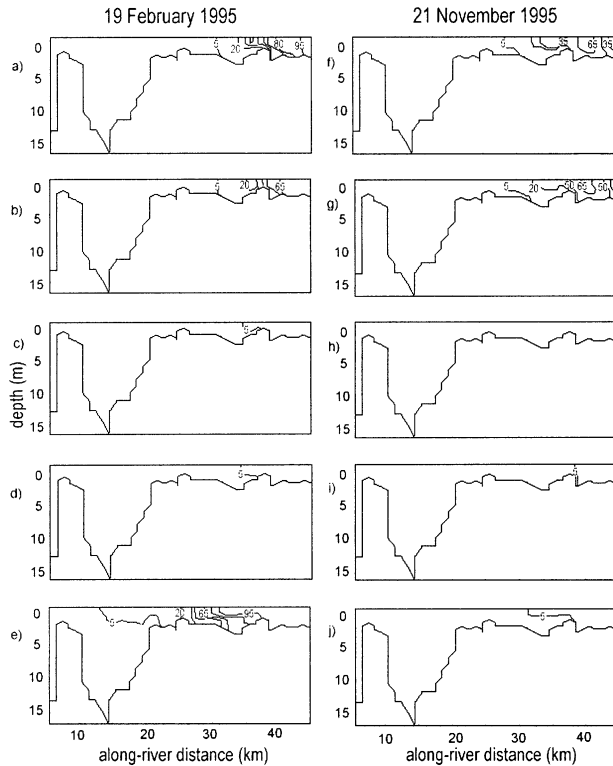19 February 1995   21 November 1995

Fig. 5. Along-river biomass transects from the estuary mouth at left (at the inner edge of the 5 km ocean buffer zone) up to the confluence with Helena River on the right. Dinoflagellates dominated on day 50 (February 19) for (a) the present-day (base) case, (b) the case without tributary impoundment, (c) the pre-European case, (d) the pre-European case without flow reduction, and (e) the pre-European case without nutrient reduction. Chlorophyte dominated on day 325 (November 21) for (f) the present-day (base) case, (g) the case without tributary impoundment, (h) the pre-European case, (i) the pre-European case without flow reduction, and (j) the pre-European case without nutrient reduction.
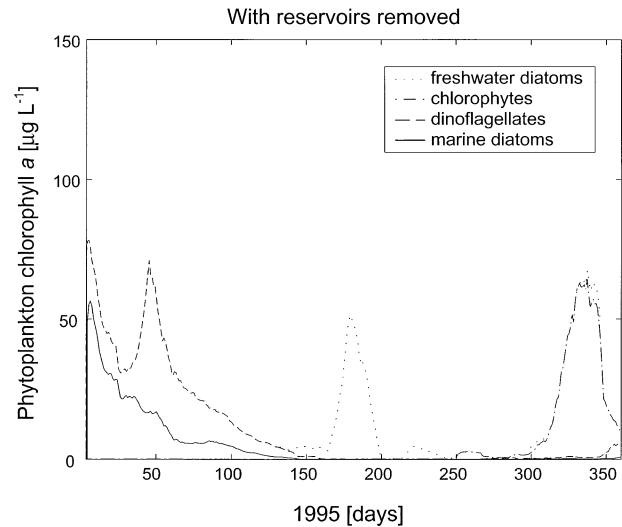
With reservoirs removed

Fig. 6. Relative phytoplankton biomass (chlorophyll *a*) integrated over the upper estuary for the case with reservoirs (Mundaring Weir and Canning Dam) removed.

bloom persists longer and the biomass is higher than observed in the field. A similar, but less pronounced effect is evident in the comparisons of dinoflagellate biomass. For both phytoplankton groups (i.e., chlorophytes and dinoflagellates) the simulated decline of post-bloom biomass could not be reproduced without adjusting parameters outside of literature ranges. The peak biomass and seasonal succession of phytoplankton groups provides a good predictor of what has been observed in the Swan River. An example of the spatial distribution over the estuary can be seen for two selected days; day 50 in Fig. 5a and day 325 in f.

Increased Flow in the Absence of Tributary Impoundments

The model results show that removal of impoundments and the resultant increases predicted to occur in streamflow had a relatively small impact

on the dynamics of the Swan River estuary. In comparison to the present-day (base) case, salinity was reduced slightly around days 110, 250, and 330 (Fig. 3), but there was little difference in nutrient concentrations at any time. The main difference in the phytoplankton community was an increase in the duration and peak of chlorophyte biomass during the spring bloom (beginning ~day 310, Figs. 5g and 6). Chlorophyte simulations were particularly sensitive to changes in salinity. This was evident around days 330–340 in the scenario without tributary impoundments, with reduced salinity allowing a greater window of opportunity for chlorophyte populations to increase rapidly. It is evident that salinity is the critical influence on chlorophytes at times of high biomass. Residence time in the upper estuary was reduced slightly with this scenario, which may also have affected the time for chlorophyte growth potential to be realized (Chan and Hamilton 2001).

Pre-European Watershed

Under a reduced flow (1/5th) and nutrient (1/4th to 1/16th) regime, as estimated for pre-European settlement (Viney and Sivapalan 2001), the winter freshwater period is of shorter duration and salinity is elevated over the base case (Fig. 3). Inorganic nitrogen and phosphorus concentrations are lower throughout the year. The divergence from the base case is most pronounced in winter, when nutrient concentrations reach maximal levels with the commencement of substantial seasonal freshwater flows. The initial concentrations of phytoplankton and nutrients in the water column for
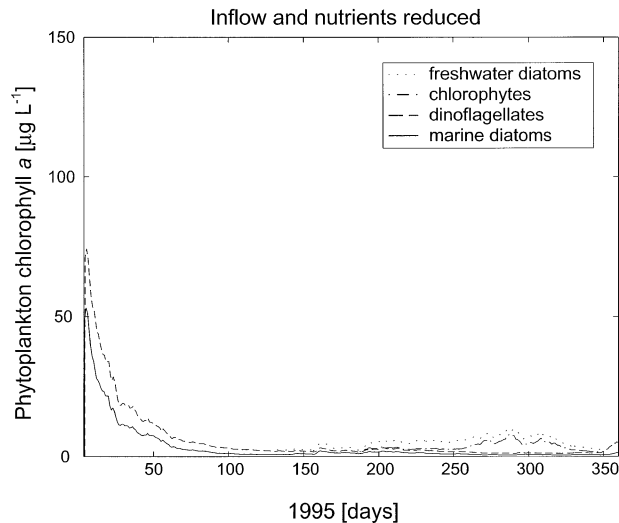
Fig. 7. Relative phytoplankton biomass (chlorophyll *a*) integrated over the upper estuary for the case with both inflows and nutrients reduced.
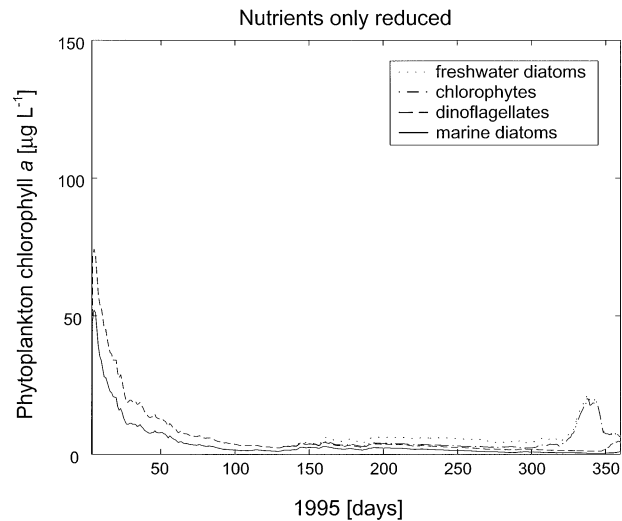

Fig. 8. Relative phytoplankton biomass (chlorophyll *a*) integrated over the upper estuary for the case with nutrients only reduced, while inflow remains at baseline levels.

the beginning of this scenario were identical to those for the base case, but declined steadily through the early phases of the simulation. The elevated level of marine diatoms near the start of this simulation was an artifact of the relatively high initial levels of this group. The general effect of reduced levels of nutrients was to reduce the biomass of all phytoplankton groups (Figs. 5c,h and 7).

Dinoflagellates, in particular, remained at substantially lower levels throughout the year than in the base case. The difference in the upper reaches can also be seen in comparing the day 50 base case biomass transect in Fig. 5a with that of the scenario shown in Fig. 5c. This pre-European scenario would have increased residence times in the upper estuary, provided greater opportunity for species adapted to higher salinities to grow, and increased the likelihood of phytoplankton growth potential being realized (Chan and Hamilton 2001). These effects appear to be outweighed by reduced levels of nutrients to support phytoplankton growth.

Freshwater diatoms are particularly disadvantaged in this scenario. When winter inflow begins (~day 150), simulated salinity through the water column in the upper reaches decreases to ~13 psu but then immediately increases back to 20 psu and remains at this level until day 200, while in the base case the upper estuary was much fresher (<10 psu) during this period. Freshwater diatoms are mostly outside of their usual salinity tolerance, and their biomass is reduced to 1/9th of peak values in the base case. Chlorophytes were reduced to around 1/10th of levels simulated in the base case (Fig. 5f,h).

Pre-European Watershed Without Flow Reduction

A simulation was run with nutrient concentrations reduced as for the pre-European simulation, but with inflows unchanged from the present-day (base) case. Phytoplankton succession and biomass were largely unchanged from the pre-European simulation which had both tributary flow and nutrient levels altered (Figs. 5f and 8), indicating that reductions in nutrients were largely responsible for the decrease in biomass over the base case. For chlorophytes, while peak biomass reached only around 1/3rd of levels in the base case, it still exceeded levels for the pre-European scenario that had both flow and nutrients reduced (Figs. 5i and 8).

Pre-European Watershed Without Nutrient Reduction

In this scenario, the flow regime was set to the predicted low pre-European levels but nutrients were set to present-day (base) levels. Salinity is unchanged for this scenario from the other pre-European scenario (Fig. 3), but dissolved inorganic nutrients in the upper reaches are elevated over the base case at times of low flow. Under the low flow case, an increase in occurrence and duration of stratified conditions produces anoxia that enhances sediment nutrient release. Douglas et al. (1997) observed elevated levels of inorganic nutrients in bottom waters when anoxia occurred under prolonged stratification. Calibration of nutrient release rates on the basis of these observations and of sediment oxygen uptake rates on the basis of
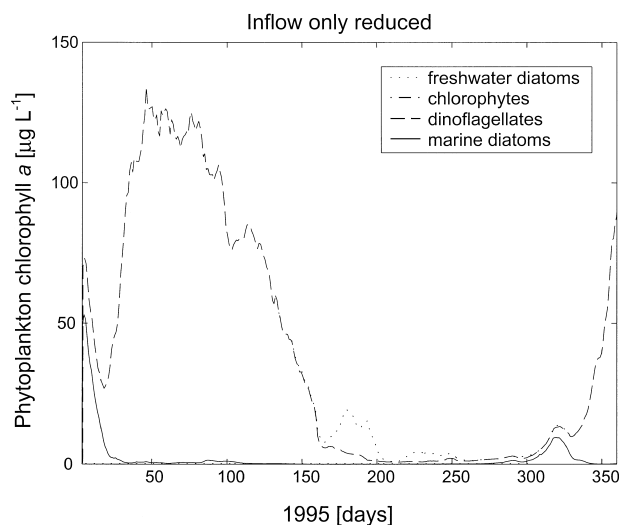
Fig. 9. Relative phytoplankton biomass (chlorophyll *a*) integrated over the upper estuary for the case with inflow only reduced, while nutrients remain at baseline levels.

benthic chamber deployments (Herzfeld et al. 2001) provide confidence that the interactions of stratification, anoxia, and sediment nutrient release may be simulated with some certainty.

Despite the combination of increased nutrient levels and reduced flushing, chlorophytes do not reach high concentrations in this scenario (Figs. 5j and 9), as high salinity imposes a major constraint on biomass development. Dinoflagellates become the dominant group, benefiting from both higher salinities and higher nutrient concentrations over the summer period. While nutrient levels are conducive to blooms at any time of the year, the pre-European levels of flow in winter-spring (days 200–300) are still sufficient for flushing to prevent high levels of biomass.

Discussion

Although the phytoplankton seasonal succession and peak biomass is well represented, the dinoflagellate and chlorophyte groups were of longer duration than observed in the field. This difference may be attributable to zooplankton grazing, as Griffin et al. (2001) found previously that grazing hastened post-bloom decreases of dinoflagellate biomass. The increased duration of simulated dinoflagellate biomass in fall also partially explains the decreased inorganic nutrients exhibited in the model at this time. The duration of blooms modeled in the scenarios may thus also be overestimated. Due to the ephemeral nature of the phytoplankton blooms in most of the scenarios, this would only be a factor in the final, low-flow, high-nutrient, scenario (Fig. 9).

Model simulations indicate that flow, salinity,

and nutrients are the main factors influencing phytoplankton biomass and succession, but the influence of other factors should also be considered. The temperature regime is unlikely to change substantially under the different scenarios. Reduced temperatures in winter are likely to hinder the attainment of phytoplankton growth potential, especially when peak winter flows reduce residence times in the upper estuary to fractions of a day (Chan and Hamilton 2001). Stratification and mixing in the water column are, however, altered by the changing flow regime between scenarios. Water column stability has implications for the light climate experienced by the phytoplankton community (Monbet 1992) as well as for nutrient release from bottom sediments (Douglas et al. 1997). In the Swan River, the influence of mixing on light regime is mitigated by the relatively shallow mixed layer depths, and the potential for light limitation is considered to be low given the relatively high water clarities that are experienced over the periods of highest phytoplankton biomass (Chan and Hamilton 2001).

Nutrients in streamflow, and from benthic regeneration under stratification-generated anoxia, appear to be the most important factors influencing phytoplankton productivity in the presence of the hydrological changes that have taken place in the Swan River watershed since European settlement. These observations are consistent with others on the Swan River (Thompson 1998) and other microtidal estuaries (Malone et al. 1988; Mallin et al. 1993) although the latter two studies were in systems with less seasonality of rainfall and more limited salinity ranges than we observe in the Swan River estuary.

Although the simulations indicate that the greatest effects on phytoplankton biomass are associated with European settlement and nutrient enrichment, salinity plays an important role in phytoplankton succession. For example, Marshall and Alden (1990) found that the oligohaline-mesohaline gradients in estuaries were even more important than variations in nutrients in determining the composition of phytoplankton communities.

The emphasis of this study was to examine possible changes in phytoplankton succession due to the impact of major anthropogenic activities on the hydrology of the Swan River. It should be noted that in addition to the watershed hydrological changes examined here, there are hydraulic changes that may also have had a significant impact on the phytoplankton succession and biomass. The dredging of a sandbar across the mouth of the estuary for navigation purposes (Fig. 1, location 1) is likely to have had an important effect in increasing exchange of estuarine water with the ocean

(Riggert 1978). Removing the sill may have moderated the effects of increasing nutrient levels by increasing flushing, although there may be confounding effects related to the tolerance of the various phytoplankton groups to changes in salinity and stratification.

More recently, rapid growth of the city of Perth (Fig. 1, location 3) has led to the transformation of traditionally rural or natural catchments to urban catchments (e.g., Ellen Brook, Fig. 1, location 6). Catchment models (Sivapalan personal communication) indicate corresponding increases in stormflow and more rapid response of tributary inflows to rainfall, due to the increased fraction of impermeable surfaces in urban areas. Development of marinas and boat harbors in the upper estuary (Fig. 1, location 4), while not unduly influencing the hydraulic residence time of the entire estuary, may lead to localized variations in water residence time at the scale of the enclosure. Both types of developments are likely to adversely affect water quality in parts of the estuary, although on what scale remains uncertain. Modeling of such developments would be useful for identifying their impacts, and if performed prior to the inception of development, may assist in planning for the mitigation of any negative consequences (Hamilton and Turner 2001).

Conclusions

The coupled hydrodynamic-ecological model ELCOM-CAEDYM has been used to simulate the effects of post-European development of catchment conditions and tributaries on the ecology of the Swan River estuary. Phytoplankton succession and biomass in the estuary are likely to have been affected only slightly by the changes in hydrology due to impoundment of water in Mundaring Weir and Canning Dam. A far greater impact is attributable to changing land use of the catchment. Increased discharge and the associated decrease in salinity have allowed chlorophyte biomass to increase. Increased nutrient inputs from clearing of native vegetation and expansion of agriculture have allowed an increase in biomass of all four of the main groups of phytoplankton and, in particular, diatoms and dinoflagellates. Model results suggest that the pre-European phytoplankton community was very low in biomass and dominated by chlorophytes. The dominant impact of the hydrological changes examined in this study is the increased availability of nutrients.

Monitoring and prediction of the impacts of ongoing changes to the catchment of the Swan River, such as the conversion of rural to urban catchments, is essential if the impact of such changes on the ecology is to be properly managed.

Literature Cited

Chan, T. U. and D. P. Hamilton. 2001. The effect of freshwater flow on the succession and biomass of phytoplankton in a seasonal estuary. *Marine and Freshwater Research* 52:869–884.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210: 223–253.

Cooper, S. R. and G. S. Brush. 1993. A 2,500-year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries* 16: 617–626.

Davies, P. E. and S. R. Kalish. 1994. Influence of river hydrology on the dynamics and water quality of the upper Derwent estuary, Tasmania. *Australian Journal of Marine and Freshwater Research* 45:109–130.

Donohue, R., W. A. Davidson, N. E. Peters, S. Nelson, and B. Jakowyna. 2001. Trends in total phosphorus and total nitrogen concentrations of tributaries to the Swan-Canning estuary, 1987 to 1998. *Hydrological Processes* 15:2411–2434.

Douglas, G. B., D. P. Hamilton, R. G. Gerritse, and J. A. Adeney. 1997. An investigation of water quality at two sites in the Swan River estuary, 1993–1994. Division of Water Resources Report No. 96-2. Commonwealth Scientific and Industrial Research Organisation, Canberra, Australian Capital Territory.

Griffin, S. L., M. Herzfeld, and D. P. Hamilton. 2001. Modelling the impact of zooplankton grazing on the phytoplankton biomass during a dinoflagellate bloom in the Swan River estuary, Western Australia. *Ecological Engineering* 16:373–394.

Hamilton, D. P. 2000. Record summer rainfall induced first recorded major cyanobacterial bloom in the Swan River. *Environmental Engineer* 1:25.

Hamilton, D. P., T. U. Chan, M. S. Robb, C. B. Pattiaratchi, and M. Herzfeld. 2001. The hydrology of the upper Swan River estuary with focus on an artificial destratification trial. *Hydrological Processes* 15:2465–2480.

Hamilton, D. P., P. A. Thompson, R. Kurup, and J. Horner-Rosser. 1999. Dynamics of dinoflagellate blooms in the Swan River estuary, p. 273–286. *In* A. J. McComb and J. A. Davis (eds.), Proceedings of the 5th International Wetlands Conference. Gleneagles Press, Adelaide, South Australia.

Hamilton, D. P. and J. Turner. 2001. Integrating Research and Management of an urban estuarine system, the Swan River, Western Australia. *Hydrological Processes* 15:2383–2385.

Harris, G. P. 1996. Catchments and aquatic ecosystems: Nutrient ratios, flow regulation and ecosystem impacts in rivers like the Hawkesbury-Nepean. Discussion paper, Co-operative Research Centre for Freshwater Ecology, University of Canberra, Australian Capital Territory.

Harris, G. P. 1999. Comparison of the biogeochemistry of lakes and estuaries: Ecosystem processes, functional groups, hysteresis effects and interactions between macro- and microbiology. *Marine and Freshwater Research* 50:791–811.

Harris, G. P. 2001. Biogeochemistry of nutrient and phosphorus in Australian catchments, rivers and estuaries: Effects of land use and flow regulation and comparisons with global patterns. *Marine and Freshwater Research* 52:139–150.

Hearn, C. J. and B. J. Robson. 2000. Modelling a bottom bound-

ary layer and its control of massive alga blooms in an estuary. *Applied Mathematical Modelling* 24:843–859.

HERZFELD, M., D. P. HAMILTON, AND G. B. DOUGLAS. 2001. Comparison of a mechanistic sediment model and a water column model for hindcasting oxygen decay in benthic chambers. *Ecological Modelling* 136:255–267.

HODGES, B. R. 2000. Numerical Techniques in CWR ELCOM. Centre for Water Research, Technical Report WP1422BH. The University of Western Australia, Perth, Western Australia.

HODGES, B. R. AND J. IMBERGER. 2001. A simple curvilinear method for numerical models in open channels. *Journal of Hydraulic Engineering* 127:949–958.

HODGES, B. R., J. IMBERGER, A. SAGGIO, AND K. WINTERS. 2000. Modeling basin-scale internal waves in a stratified lake. *Limnology and Oceanography* 45:1603–1620.

JOHN, J. 1994. Eutrophication of the Swan River estuary, Western Australia and the management strategy, p. 749–757. *In* W. J. Mitsch (ed.), Global Wetlands: Old World and New. Elsevier, Amsterdam, The Netherlands.

LINDERFELT, W. R. AND J. V. TURNER. 2001. Interaction between shallow groundwater, saline surface water and nutrient discharge in as seasonal estuary: The Swan-Canning system. *Hydrological Processes* 15:2631–2653.

MALLIN, M. A., H. W. PAERL, J. RUDEK, AND P. W. BATES. 1993. Regulation of estuarine primary production by watershed rainfall and river flow. *Marine Ecology Progress Series* 93:199–203.

MALONE, T. C., L. H. CROCKER, S. E. PIKE, AND B. W. WENDLER. 1988. Influences of river flow on the dynamics of phytoplankton production in a partially stratified estuary. *Marine Ecology Progress Series* 48:235–249.

MARSHALL, H. G. AND R. W. ALDEN. 1990. A comparison of phytoplankton assemblages and environmental relationships in the three estuarine rivers of the Lower Chesapeake Bay. *Estuaries* 13:287–300.

MONBET, Y. 1992. Control of phytoplankton biomass in estuaries: A comparative analysis of microtidal and macrotidal estuaries. *Estuaries* 15:563–571.

NIXON, S. W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41:119–219.

PETERS, N. E. AND R. DONOHUE. 2001. Integrating research and management of an urban estuarine system. *Hydrological Processes* 15:2671–2686.

RIGGERT, T. L. 1978. The Swan River estuary: Development, Management and Preservation. Swan River Conservation Board, Perth, Western Australia.

RANASINGHE, R. AND C. PATTIARATCHI. 2000. Tidal inlet velocity asymmetry in diurnal regimes. *Continental Shelf Research* 20:2347–2366.

SHERMAN, B. S., I. T. WEBSTER, G. J. JONES, AND R. L. OLIVER. 1998. Transitions between *Aulacoseira* and *Anabaena* dominance in a turbid river weir pool. *Limnology and Oceanography* 43:1902–1915.

STEPHENS, R. AND J. IMBERGER. 1996. Dynamics of the Swan River estuary: The seasonal variability. *Marine and Freshwater Research* 47:517–529.

THOMPSON, P. A. 1998. Spatial and temporal patterns of factors influencing phytoplankton in a salt wedge estuary, the Swan River, Western Australia. *Estuaries* 21:801–817.

THOMPSON, P. A. AND W. HOSJA. 1996. Nutrient limitation of phytoplankton in the Upper Swan River estuary, Western Australia. *Marine and Freshwater Research* 47:659–667.

VINEY, N. R. AND M. SIVAPALAN. 2001. Modelling catchment processes in the Swan-Avon River basin. *Hydrological Processes* 15:2671–2686.

ZANN, L. 1995. Our Sea, Our Future. Major Findings of the State of the Marine Environment Report. Department of Environment, Sport and Territories, Canberra, Australian Capital Territory.

SOURCE OF UNPUBLISHED MATERIALS

SIVAPALAN, M. personal communication. Centre for Water Research, University of Western Australia, Stirling Highway, Nedlands, Western Australia, 6009.