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Derwent Estuary Biogeochemical Model: Scenario Report

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1. EXECUTIVE SUMMARY

The calibrated 2003 biogeochemical model (Wild-Allen et al. 2009) was used to address three management scenarios. The scenarios were: 1) a near-pristine scenario excluding anthropogenic nutrient loads; 2) a 2015 active management scenario assuming improved treatment of industrial effluent, STP reuse and marine nutrients constrained to 2003 concentrations; and 3) a 2015 business-as-usual scenario including reduced Derwent River flow, improved treatment of industrial effluent, increased STP loads and marine nutrients increased to 2008 concentrations.

All model simulations demonstrated broad similarities in seasonal nutrient characteristics and phytoplankton succession with highest biological productivity and nutrients simulated in the middle reaches of the estuary. There appears to be natural accumulation of nitrogen in the upper and mid estuary in winter and persistent elevated chlorophyll concentrations in the middle reaches associated with the dynamics of the salt wedge front. There was also lower dissolved oxygen saturation in bottom waters and surface sediments (seasonal mean saturation 40-60%) in the deeper parts of the mid to lower estuary, particularly in autumn, but also in spring for all scenarios.

Modelled annual mean near surface chlorophyll concentrations, show that the estuary under the current flow management guidelines and without any anthropogenic loads (i.e. near-pristine scenario) would be predominantly mesotrophic (54%) and partially eutrophic (46%) [although it is very likely that for a pristine scenario where river flow was unmanaged the results of this classification would be different]. In 2003 eutrophic conditions occurred over 82% of the region and this increase to 87% in the 2015 business-as-usual scenario. In the 2015 active management scenario the eutrophic area of the estuary was reduced to 72% of the region with the remaining area classified as mesotrophic.

The active management scenario simulation had lower dissolved inorganic nitrogen (DIN), dissolved inorganic phosphate (DIP) and chlorophyll concentrations and higher dissolved oxygen (DO) percent saturation in bottom water and surface sediment than the 2003 calibrated model. The active management scenario simulation demonstrated the greatest water quality improvement in the middle reaches of the estuary compared to the 2015 business-as-usual scenario and the 2003 calibrated model.

The business-as-usual scenario had higher DIN, DIP and chlorophyll concentrations and lower DO percent saturation in bottom water and sediment than the 2003 calibrated model. The lower river flow in the 2015 business-as-usual scenario allowed excursion of the marine salt wedge upstream into the estuary and there was an enhanced influx of nutrients across the marine boundary. In the business-as-usual scenario the model favoured seagrass and macroalgae growth in shallow parts of the upper and middle reaches and Ralphs Bay due to a combination of low attenuation (and increased propagation of light) and elevated sediment nutrient concentrations. The near-pristine scenario favoured less seagrass and macroalgae growth possibly due to nutrient limitation.

Nitrogen budgets for all scenarios showed contrasting nitrogen inputs from marine, river and point source loads were very nearly balanced by denitrification and marine export. Modelled denitrification was found to be a key process in maintaining the health of the estuary and whilst this component of the model is consistent with sparse data, improved observation and validation of the modelled algorithms is a priority for future work. The modelled budgets suggest that a decline in denitrification efficiency could result in a rapid accumulation of nitrogen and an associated decline in water quality, in the estuary.

This study has shown that interactions between river flow, nutrient sources and water quality are complex but well simulated by the biogeochemical model. Low sediment dissolved oxygen saturation was found to vary with total nitrogen load into the estuary, provisionally by an exponential relationship. To achieve sediment DO oxygen concentrations in excess of 40% saturation over 95% of the region for 98% of the year then under average flow conditions nutrient loads to the estuary should be constrained to levels proposed in the 2015 active management scenario. Under low Derwent flow nutrient loads to the estuary would need to be reduced further to avoid extension of low sediment DO. This analysis could be improved by excluding the large refractory DON component of total nitrogen and repeating each scenario simulation for a range of river flows.

2. SCENARIO INTRODUCTION

The 2003 calibrated Derwent Estuary biogeochemical model (Wild-Allen et al 2009) was implemented with modified forcing and point source nutrient loads to simulate 3 hypothetical management scenarios for the Derwent Estuary. The Derwent Estuary Program, a regional partnership between state and local governments, industries and conservation groups, agreed on three proposed management scenarios: a near-pristine case, which omitted anthropogenic loads; a 2015 active management case; and a business-as-usual projection of point source loads and river flow into the estuary in 2015. All scenarios were completed with 2003 weather data. Greater detail on the changes made to the calibrated model with respect to the scenarios can be found in Section 3 ‘Scenario Forcing’.

Model results from scenario simulations were compared against the calibrated 2003 biogeochemical model to identify regions of change. The 2003 biogeochemical model was validated against observations collected throughout the estuary and captures the essential biogeochemical dynamics of carbon, nitrogen, phosphorus and dissolved oxygen cycling through phytoplankton, detritus and dissolved phases in the estuary. For a clear understanding of the model performance and uncertainties and a full description of the simulated biogeochemical dynamics of the estuary the technical report should be read in detail (Wild-Allen et al., 2009). These scenario simulations are hypothetical projections of plausible conditions in the estuary (in 2003) given alternative point source loads and river forcing.

2.1 Near-Pristine Scenario

This scenario simulation was designed to simulate plausible conditions in the estuary in 2003 in the absence of anthropogenic loads. Compared to the 2003 calibrated model run (Wild-Allen et al 2009) the following changes to forcing and point source loads into the estuary were made:

- All (10) sewage treatment plant (STP) inputs omitted.

- All (3) industry inputs (nutrients, DOC, POC and effluent colour) omitted.

- No change to Derwent River flow (uses 2003 flow and current management regime).

- 96 stormwater inputs reduced to 12 forested catchments containing the major drainage rivers and rivulets using 2003 rainfall data.

- No change to 2003 marine boundary conditions derived from observations.

This near-pristine scenario is indicative of pre-European conditions in the estuary but does not fully represent these conditions due to the inclusion of modified Derwent River flow and the derivation of boundary conditions, at New Norfolk and Iron Pot Lighthouse, from observations made in 2003.

2.2 Active Management Scenario (~ 2015)

This scenario was designed to simulate conditions in the estuary under projected ‘active management’ practices achievable in about 2015, assuming levels of Derwent River flow similar to

2003. Compared to the 2003 calibrated model run (Wild-Allen et al 2009) the following changes to forcing and point source loads into the estuary were made:

• 8 Sewage treatment plant (STP) inputs reduced and 2 increased, based on present (2008) and proposed Council improvements, effluent re-use schemes and projected increased load from increased urbanisation.

• 2 Industry (Nyrstar zinc refinery; Impact fertiliser) inputs remain at 2003 levels; Norske Skog effluent colour removed, carbon load decreased and nutrient input increased to Best Available Techniques (BAT) guidelines associated with effluent processing plant upgrades in 2007-10.

• No change to Derwent River flow (uses 2003 flow and current management regime).

• No change to stormwater inputs (uses 2003 flow and catchment loads).

• No change to 2003 marine boundary conditions derived from observations [which implies active reduction of nutrient concentrations observed at Iron Pot in 2008 to 2003 levels, potentially by reduction of aquaculture waste in adjacent waters].

2.3 Business-as-Usual Scenario (~2015)

This scenario was designed to simulate conditions in the estuary given 'business-as-usual' management practices for the estuary and a reduction in Derwent river flow. Compared to the 2003 calibrated model run (Wild-Allen et al 2009) the following changes were made to forcing and point source loads into the estuary:

• 8 Sewage treatment plant (STP) inputs increased and 2 decrease, based on present (2008) Council loads and projected increased load from increased urbanisation.

• Industry inputs: same as for 2015 active management scenario [2 Industry inputs remain at 2003 levels; Norske Skog effluent colour removed, carbon load decreased and nutrient input increased to BAT guidelines].

• Derwent River flow reduced to low flow year (uses 2007 flow and current management regime).

• Stormwater inputs use 2003 flow and catchment loads except for 19 catchments in greater Hobart which have increased catchment urbanisation.

• 2003 Marine boundary condition for ammonium increased to levels observed in 2008 to account for increased aquaculture adjacent to the estuary.

This business-as-usual scenario is indicative of current management practices and projections for the estuary, although conditions in the estuary could get much worse should the influx of waste aquaculture nutrients from adjacent waters increase further and/or Derwent River flow decline below 2007 levels, due to drought and/or water extraction.

3. SCENARIO FORCING

The biogeochemical model is coupled with a hydrodynamic model and a sediment model implemented on a curvilinear model grid (Figure 3.1). Figure 3.2 shows a cross section along the axis of the estuary and division of the estuary into upper, middle and outer reaches used to describe simulation results in this report. A full description of the model, compilation of parameter values, initial conditions and standard meteorological and boundary forcing is provided in Wild-Allen et al., (2009). Scenario variations to the 2003 simulation forcing are detailed below.

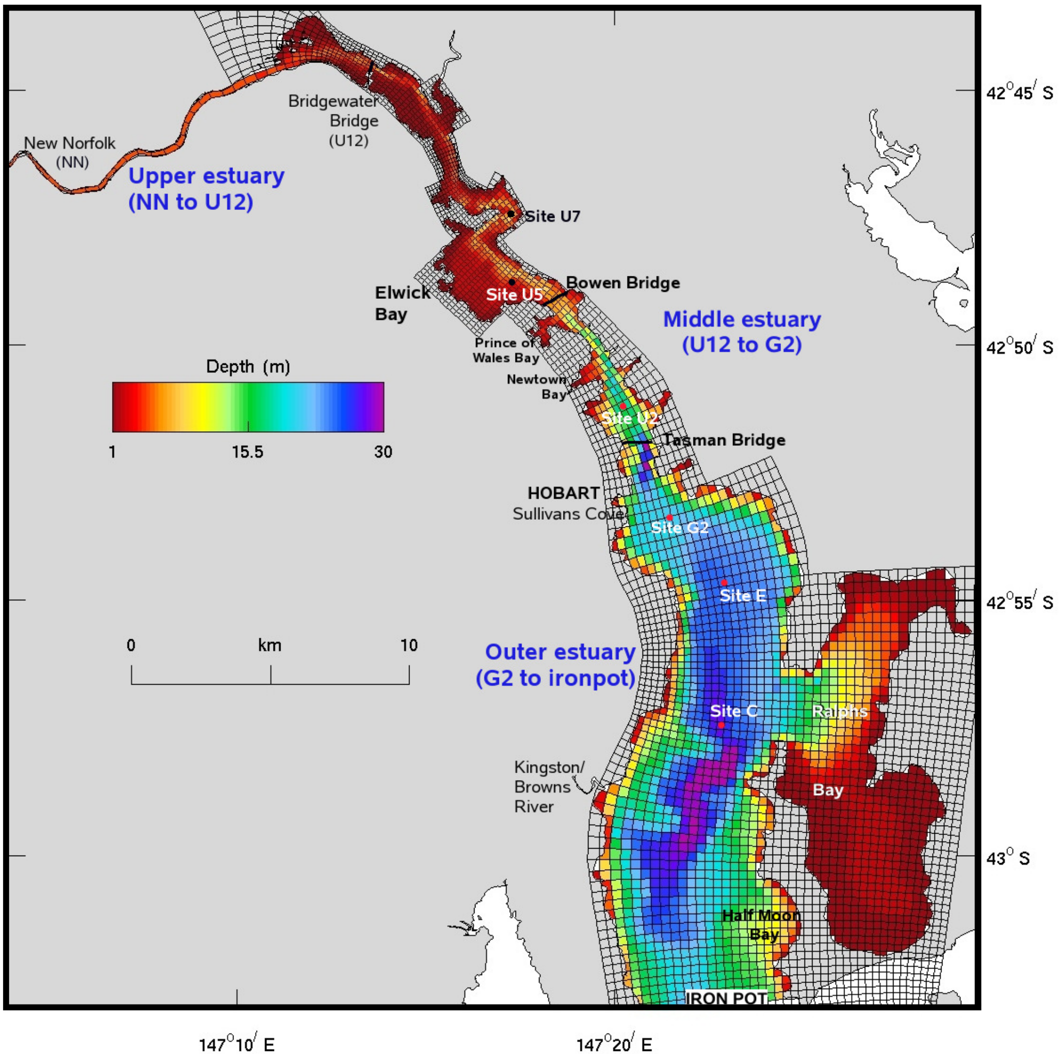


Figure 3.1 Map of the Derwent Estuary bathymetry showing the model grid and geographic locations.

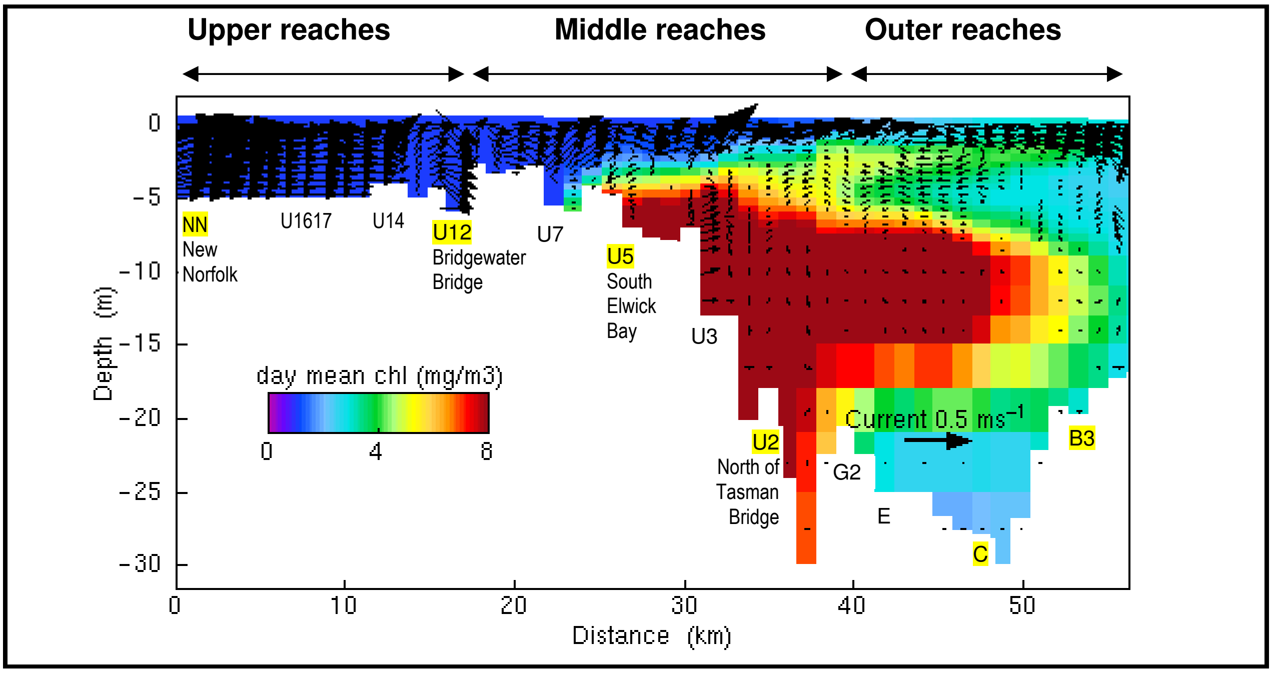


Figure 3.2 Cross section of chlorophyll concentration and current flow along the axis of the Derwent Estuary showing approximate site locations with respect to map view (Figure 3.1) and division of estuary into upper middle and lower reaches [G2 is across from Sullivans Cove and B3 is across from Half Moon Bay].

3.1 River Loads

For the near-pristine and 2015 active management scenarios Derwent river flow into the model domain was based on observations in 2003 at Meadowbank augmented with Tyenna River flow equivalent to the 2003 calibrated simulation (Wild-Allen et al., 2009). Nutrient concentrations were provided as an upstream boundary condition derived from observations by DEP at New Norfolk (Wild-Allen et al., 2009).

For the 2015 business-as-usual scenario Derwent river flow was based on similar observations made in 2007 when the river had a significantly reduced flow. In comparison to 2003, 2007 featured lower mean flows throughout the year with a single major flood event in August (Figure 3.3); in 2003 flood events occurred in late August and September. Whilst 2007 was a dry year with low flow it is possible that an even drier year could occur and, augmented by water extraction for irrigation, Derwent river flow could be less than recorded in 2007, [it should be noted that river flow is regulated to remain above 25 m^3s^{-1} at all times by dam release]. Nutrient concentrations for the business-as-usual scenario were provided as an upstream boundary condition derived from observations by DEP at New Norfolk in 2003 (Wild-Allen et al., 2009). In comparison with the other simulations the reduced Derwent river flow for the 2015 business-as-usual scenario, resulted in a net reduction in nutrient load to the upper estuary.

The timing of storm water loads in the business-as-usual scenario was unchanged from the 2003 simulation, and flow events may not coincide with periods of elevated Derwent river flow. In contrast to the 2003 simulation, the near pristine scenario and the 2015 active management scenario, the hydrodynamics simulated in the business-as-usual scenario, whilst consistent with our understanding of the estuary under reduced flow conditions, have not been validated against observations.

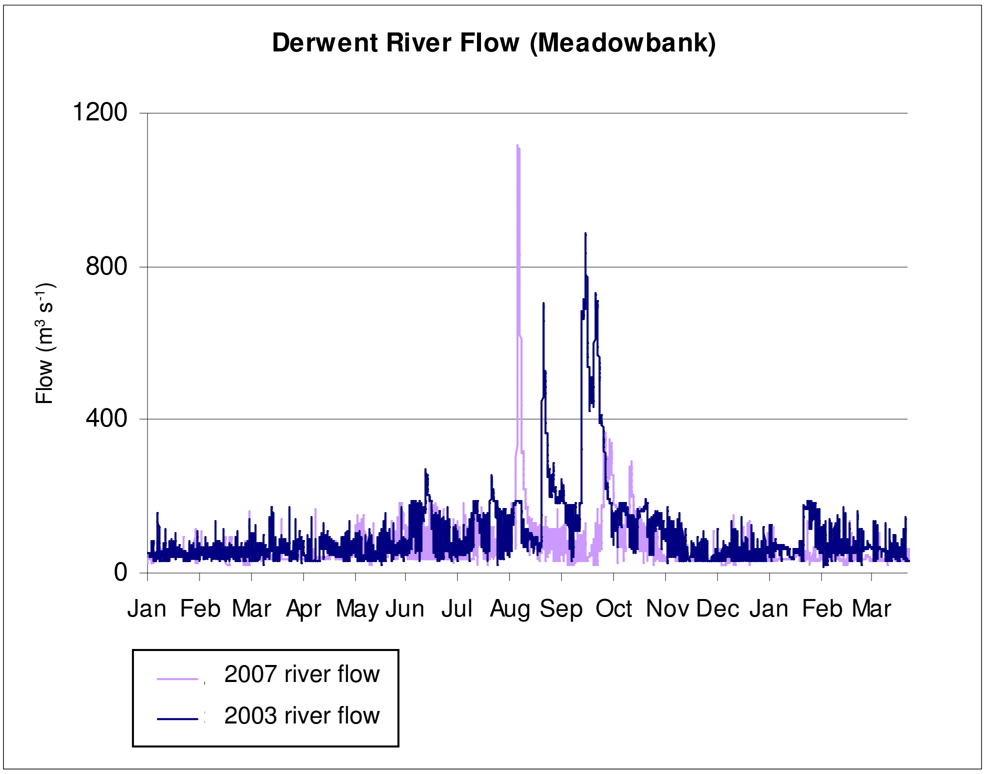


Figure 3.3 Comparison of 2003 and 2007 river flow used to force the northern model boundary. The 2003 flow is used in the 2003 calibrated model and for the near-pristine and 2015 active management scenarios. River flow from 2007 (example of a low flow year) was used in the 2015 business-as-usual scenario (Flow data from Hydro Tasmania Consulting).

3.2 Marine Boundary Conditions

The 2015 active management scenario assumed identical nutrient concentrations at the marine boundary as derived from observations for the 2003 model simulation. As observations in 2008 show an increase in nutrient concentrations in surface waters off Iron Pot lighthouse, the active management scenario is assuming the source of this increase will be mitigated, possibly by a reduction in aquaculture waste discharged into adjacent waters.

For the near-pristine scenario marine boundary there was concern that observations collected in 2003 adjacent to the marine boundary may include waste nutrients and elevated plankton concentrations associated with aquaculture loads in the D'Entrecasteaux Channel. A review of limited observations pre-2003 was inconclusive as there were insufficient observations to identify any temporal trend in loads. In the absence of historical data, marine boundary concentrations for the near-pristine simulation were derived from observations in 2003.

Marine boundary conditions for the 2015 business-as-usual scenario were updated to reflect increased concentrations of ammonia observed at the mouth of the estuary (sites B1, B3 and B5) since 2003. Ammonia was increased from 2003 to 2008 observed values, which were approximately double (Appendix 9-1). Nitrate and phosphate remained at 2003 concentrations as they had not increased significantly in the observations. Between 2003 and 2008 salmonoid aquaculture has increased in the adjacent D'Entrecasteaux Channel and Huon Estuary which could contribute to the observed changes in nutrient concentration observed at the mouth of the estuary. Although the trajectory of the aquaculture industry to 2015 is unknown, using the 2008 nutrient observations was considered to be a conservative estimate for the business-as-usual future management scenario. Further work is required to understand the connectivity between the D'Entrecasteaux Channel and the Derwent Estuary and quantify the flux of nutrients and plankton between these systems.

3.3 Sewage Treatment Plant Loads

For the 2015 active management and business-as-usual scenario simulations the sewerage treatment plant point source locations were identical to the 2003 simulation (Figure 3.4). The near-pristine scenario had no sewage treatment plant loads.

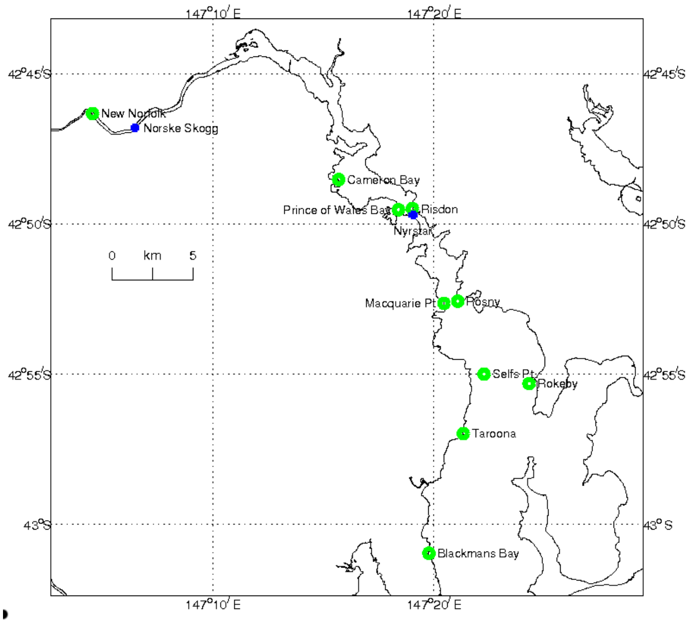


Figure 3.4 Map of Derwent Estuary indicating positions of sewerage treatment plants (large green circles) and industry source loads (small blue points).

Sewage treatment plant nutrient loads used in the two 2015 scenarios are shown in Figure 3.5. These forecasts were derived from data supplied by the EPA and the Derwent Estuary Program. Total suspended solids were converted to labile detrital particulate nitrogen and phosphorus at a fixed Redfield ratio. Refractory particulate detrital nitrogen and phosphorus were gained from subtracting the labile components from the total nitrogen and total phosphorus respectively (as detailed in Wild-Allen et al. 2009).

For the 2015 active management scenario STP loads were significantly reduced compared to the 2003 simulation; the converse was true for the 2015 business-as-usual scenario (section 3.6).

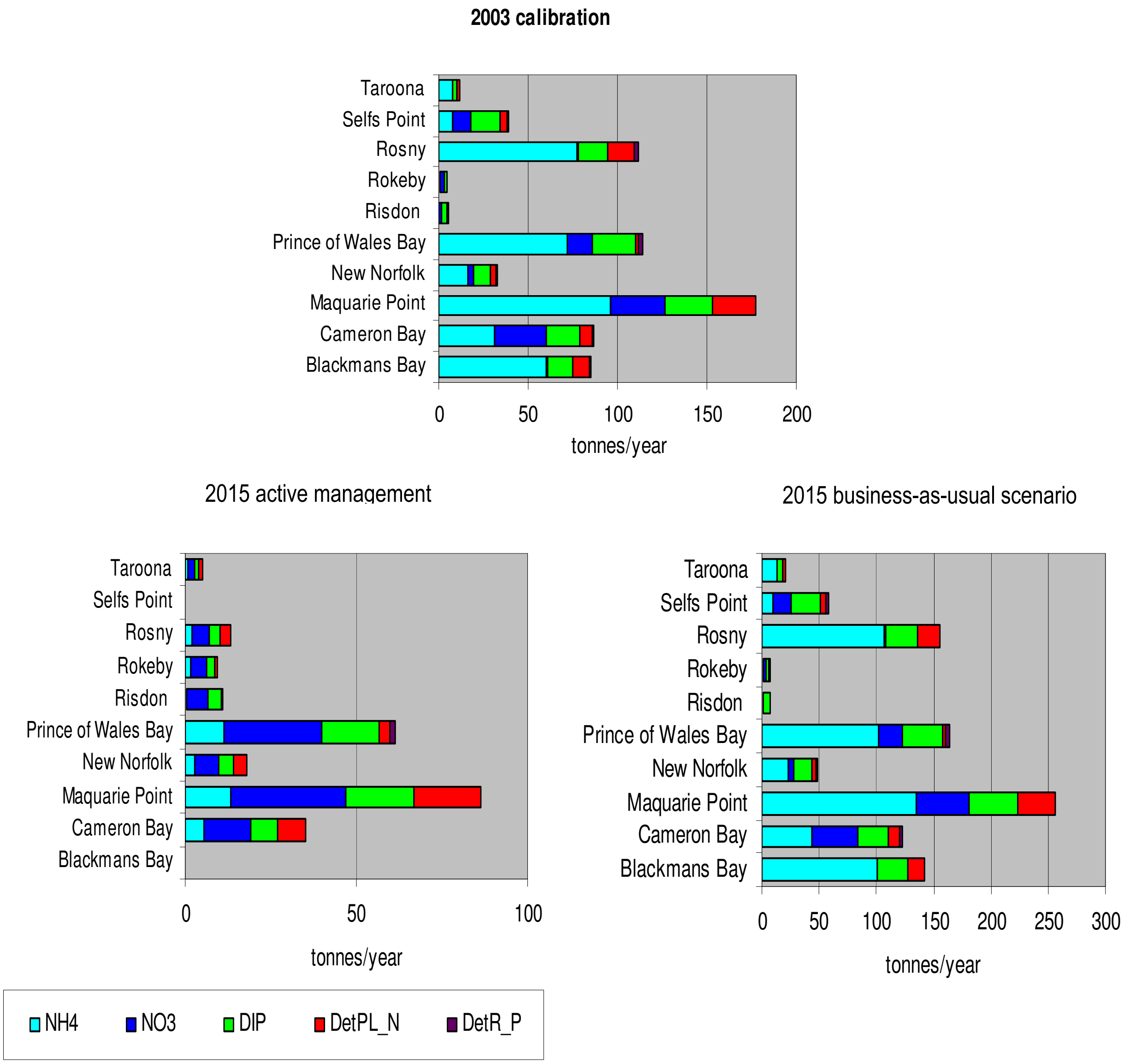


Figure 3.5 Sewage treatment plant nutrient loads (12 months) derived from EPA and DEP forecasts used in the model scenarios. [Note scale change between plots; DetPL_N is labile detrital nitrogen (with associated carbon and phosphorus at Redfield ratio); DetR_P is refractory detrital phosphorus].

3.4 Stormwater Loads

Point source locations for stormwater discharge into the estuary for the scenario simulations were identical to the 2003 model simulation (Figure 3.6).

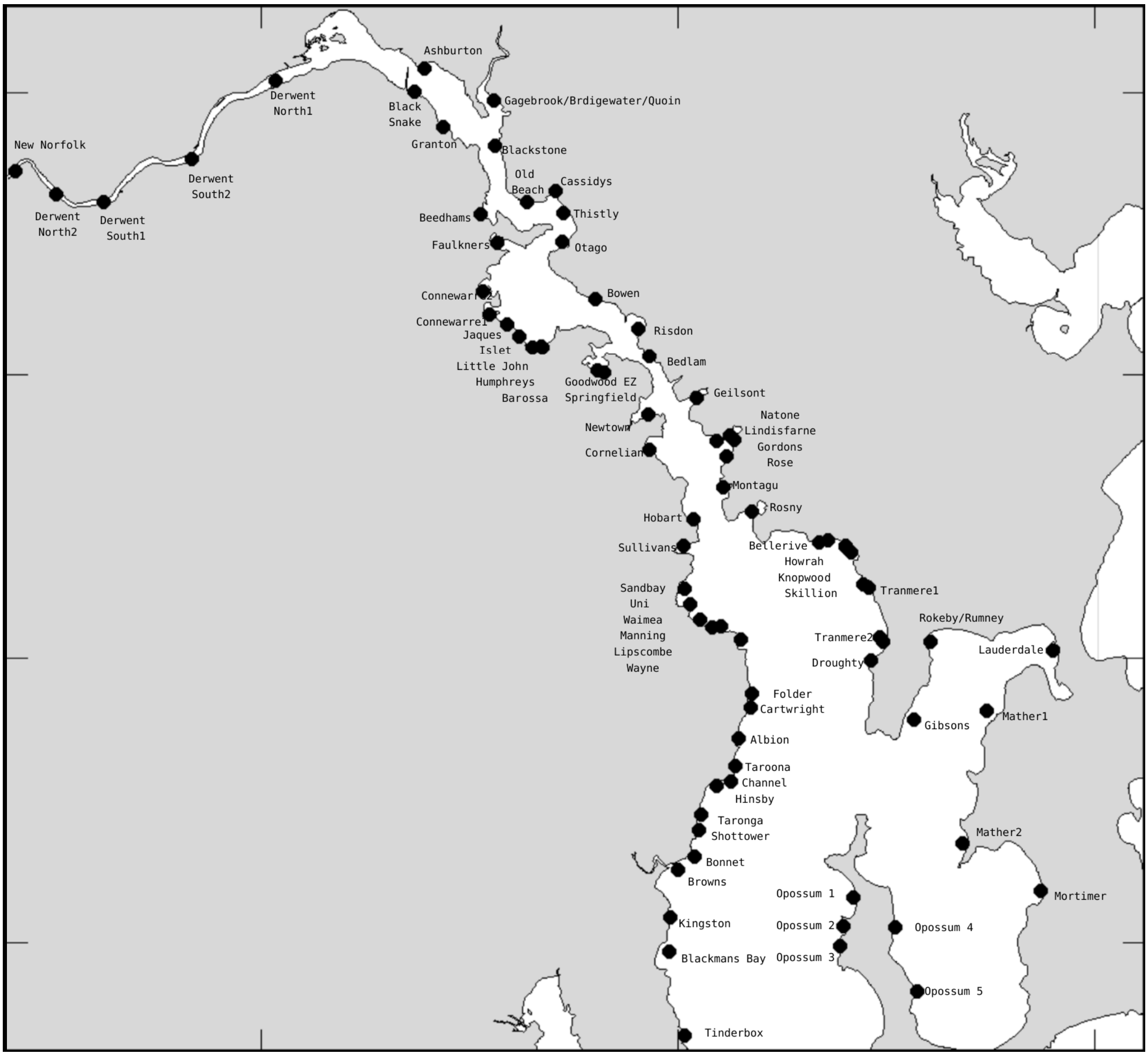


Figure 3.6 Stormwater and rivulet point sources used for the model. Entry points were placed with respect to the associated stormwater drains, rivulets or land contouring.

Stormwater sources for the near-pristine scenario were reduced from 96 to 12 from the major drainage sites (rivulets and rivers): Browns, Hobart, Newtown, Faulkner, Ashburton, Risdon, Rokeby, Rosny, Sandy Bay, Humphreys, upper Derwent north and upper Derwent south. These 12 catchments were changed in the catchment model, from urban to forested catchments and stormwater loads were computed using 2003 rainfall data (Jason Whitehead, DEP). Conversions and multipliers were used based on observations made in forested catchments similar to those used in the 2003 calibrated model [TN: 20% DIN. DIN: 95% nitrate 5% ammonia TP 17.4% DIP. TSS multiplier 0.05, TN multiplier 0.14 TP multiplier 0.16] (for stormwater catchments and analysis details see Wild-Allen et al 2009).

Stormwater nutrient loads for the active management 2015 scenario were assumed to be unchanged from the 2003 model simulation (see Wild-Allen et al. 2009).

For the 2015 business-as-usual scenario stormwater nutrient inputs from greater Hobart were changed to allow for increased urbanisation to the following 19 catchments: Sandy Bay, Quoin, New

Norfolk, Lauderdale, Granton, Browns River, Rokeby, Cassidys, Droughty, Gagebrook, Gibsons, Rokeby, Stanfields Blackstone, Bridgewater Blacksnake, Ashburton, Faulkner and Rusts. For these catchments modelled stormwater loads (Jason Whitehead, DEP) were adjusted based on conversions and multipliers computed from observations in similar urban catchments [Total nitrogen (TN): 36% dissolved inorganic nitrogen (DIN); DIN: 65% nitrate 35% ammonia; Total phosphorus (TP): 17% dissolved inorganic phosphate (DIP); Total suspended solids (TSS) multiplier 0.05, Total nitrogen (TN) multiplier 0.14, Total Phosphorus (TP) multiplier 0.16] (for stormwater catchments and analysis details see Wild-Allen et al 2009). All other stormwater catchment nutrient inputs remained the same as for 2003.

Total nutrient loads from stormwater are compared by scenarios in section 3.6. Between the scenarios the increase in urbanisation and associated decrease in agriculture and forest catchment loads resulted in less nutrient runoff from stormwater for the 2015 business-as-usual scenario than for the active management scenario (Figure 3.7). These projections are based on assumptions made in the MUSIC catchment model (operated by Jason Whitehead, DEP) of typical nutrient loads derived from urban and forested areas in the Derwent catchment.

3.5 Industry Loads

All industry loads were removed from the near pristine scenario.

Three industry point sources were included in the 2015 active management and business-as-usual scenarios: Norkse Skog (paper mill), Nyrstar (zinc refinery) and Impact Fertiliser (fertiliser plant). For both scenarios future projections of nutrient loads from Nyrstar and Impact Fertiliser remained at 2003 levels, however following effluent treatment improvements at Norske Skog significant changes in carbon loads and effluent colour were anticipated. Model loads were provided by Norske Skog in accordance with the Best Available Technology (BAT) guidelines that they have agreed to achieve before 2015. These correspond to a decrease in carbon, an increase in DIN load, and removal of effluent colour as the mill changes to softwood processing in 2010 (Figure 3.7). There was no difference between the active management and business-as-usual scenarios with respect to carbon and nutrient loads from Norske Skog.

3.6 Nutrient and Carbon Load Summary

Nutrient and carbon loads for the pristine scenario from stormwater, sewage treatment plant and industry were lower than for the 2003 simulation and the 2015 active management and business-as-usual scenarios (Figure 3.7, 3.8). The 2015 active management scenario had lower nutrients than 2003 and business-as-usual scenario. The sole source of carbon included in the model nutrient loads was from Norske Skog. There is a marked decrease in carbon in the 2015 active management and business-as-usual scenarios when compared to 2003 due to the treatment facilities coming online at Norske Skog (in 2008) (Figure 3.7, 3.8).

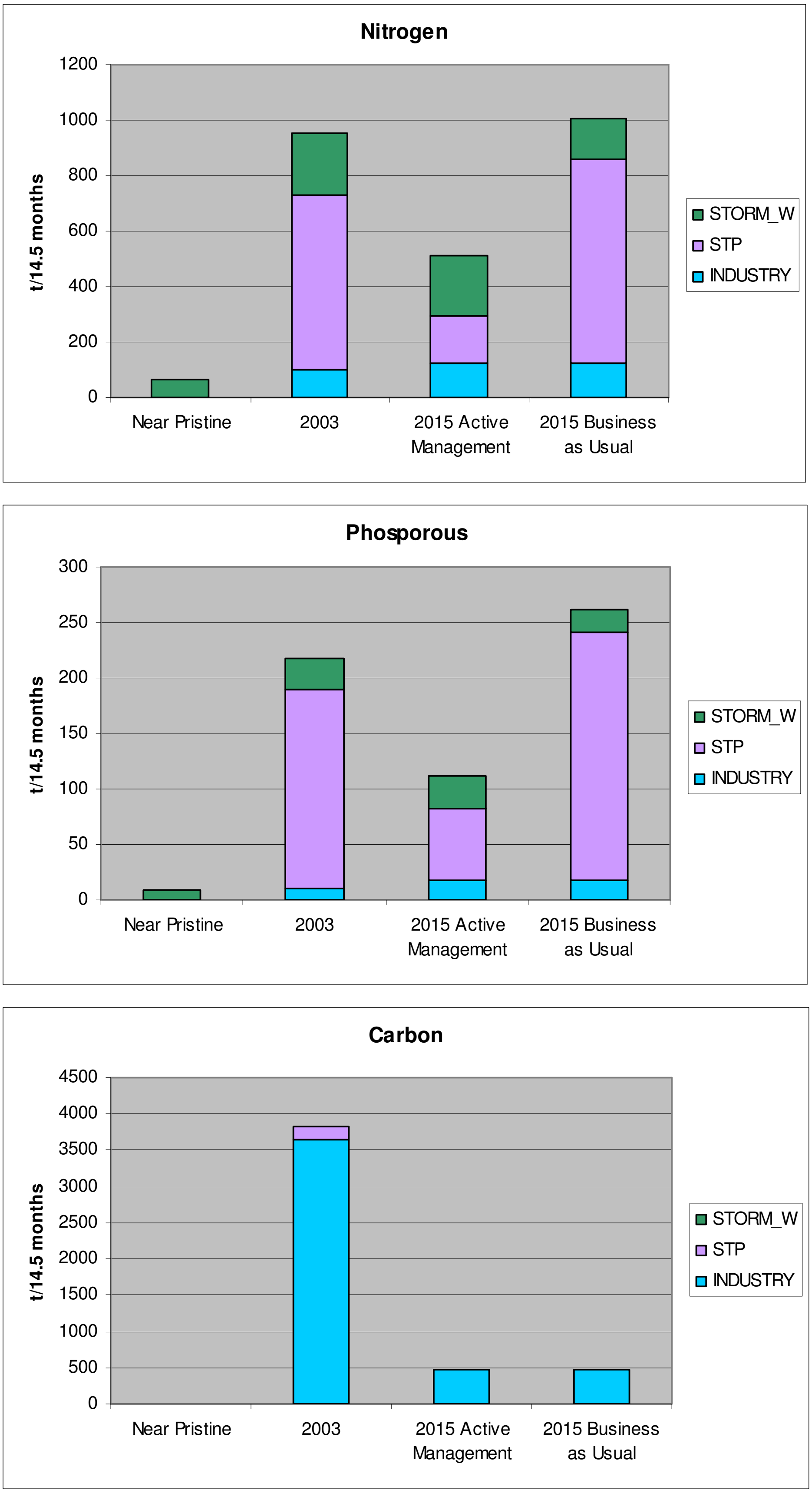


Figure 3.7 Nutrient inputs (nitrogen phosphorus and carbon tonnes per 14.5 months) for the three scenarios and the 2003 calibrated model.

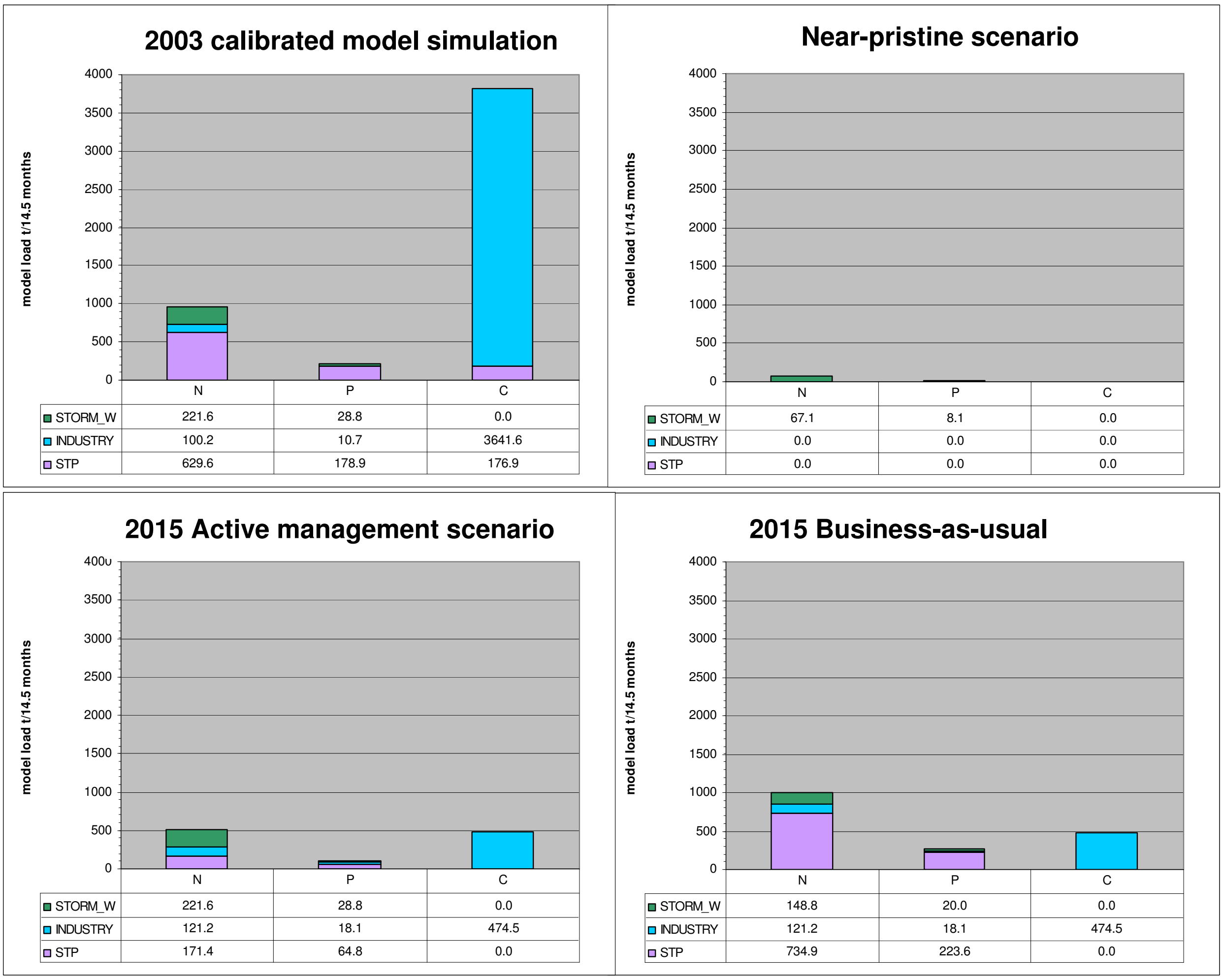


Figure 3.8 Summary of nutrient input (nitrogen, phosphorus and carbon loads) comparing stormwater, sewage treatment plant and industry loads for the 2003 calibrated model and the three model scenarios.

4. SCENARIO RESULTS

4.1 Salinity

The salinity structure and hydrodynamics of the estuary in the 2003 simulation, the near pristine scenario and the 2015 active management scenario are identical and fully described in Herzfeld et al., 2006. For the 2015 business-as-usual scenario the reduction in Derwent River flow altered the salinity structure and hydrodynamics of the estuary compared to the 2003 simulation.

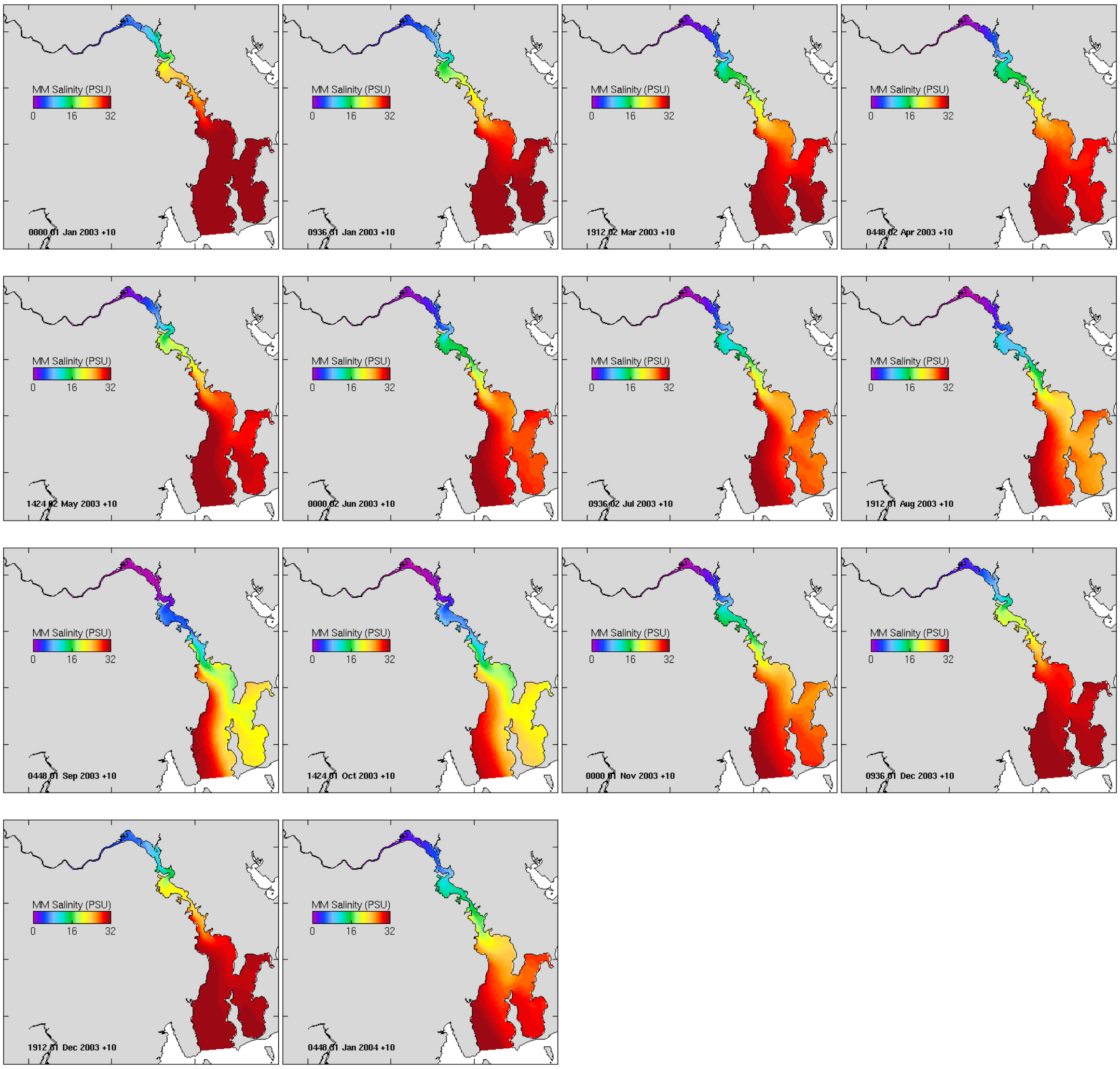


Figure 4.1 Monthly mean surface salinity for the 2003 simulation, the near pristine scenario and the 2015 active management scenario simulation from Jan'03-Feb'04.

In general differences in the salinity field between the two model runs were small as river flow is regulated to remain above 25 m^3s^{-1} throughout the year, however in the business-as-usual scenario elevated salinities extended further upstream than in the 2003 simulation. During August conditions in the estuary were fresher in the business-as-usual scenario due to a major flood event (Figure 4.2). In the 2003 simulation fresher conditions also occurred in September and October (Figure 4.1).

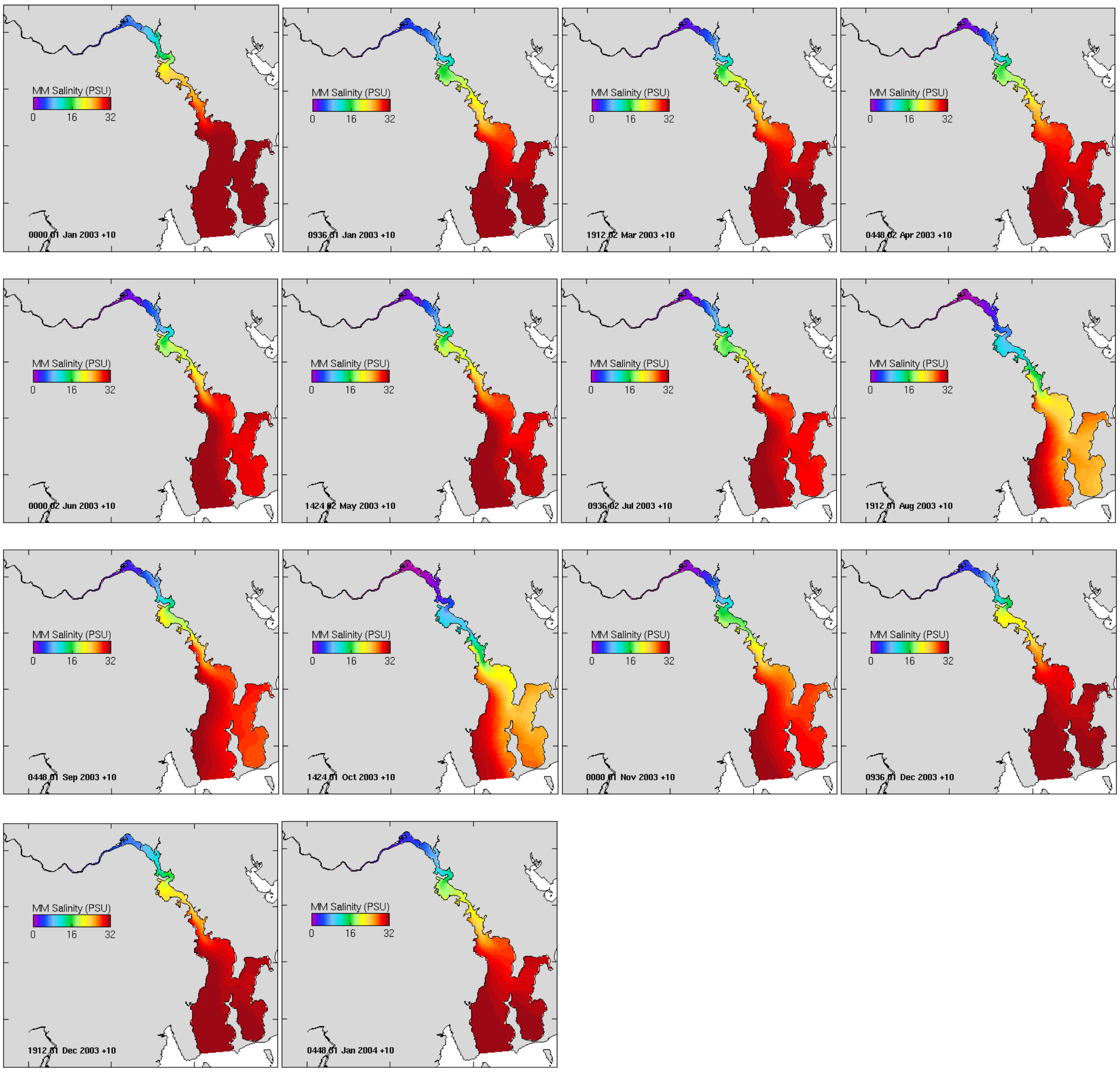


Figure 4.2 Monthly mean surface salinity for the 2015 business-as-usual scenario simulation (with reduced Derwent River flow) from Jan'03-Feb'04.

Variation in the salinity profile of the estuary during September [flood event in 2003; low flow in 2007] is shown in Figure 4.3. In the business-as-usual scenario less river water enters the estuary and the salt wedge extends further upstream (Figure 4.3). This shift in the salt wedge also carries bottom waters from the middle reaches, which can be elevated in nutrients and reduced in dissolved oxygen, further upstream. The net transport of marine water into the estuary across the marine boundary under low flow is greater, and the retention time of water in the estuary is increased. In the business-as-usual scenario the greater flushing period provides a longer opportunity for biogeochemical cycling in the mid estuary.

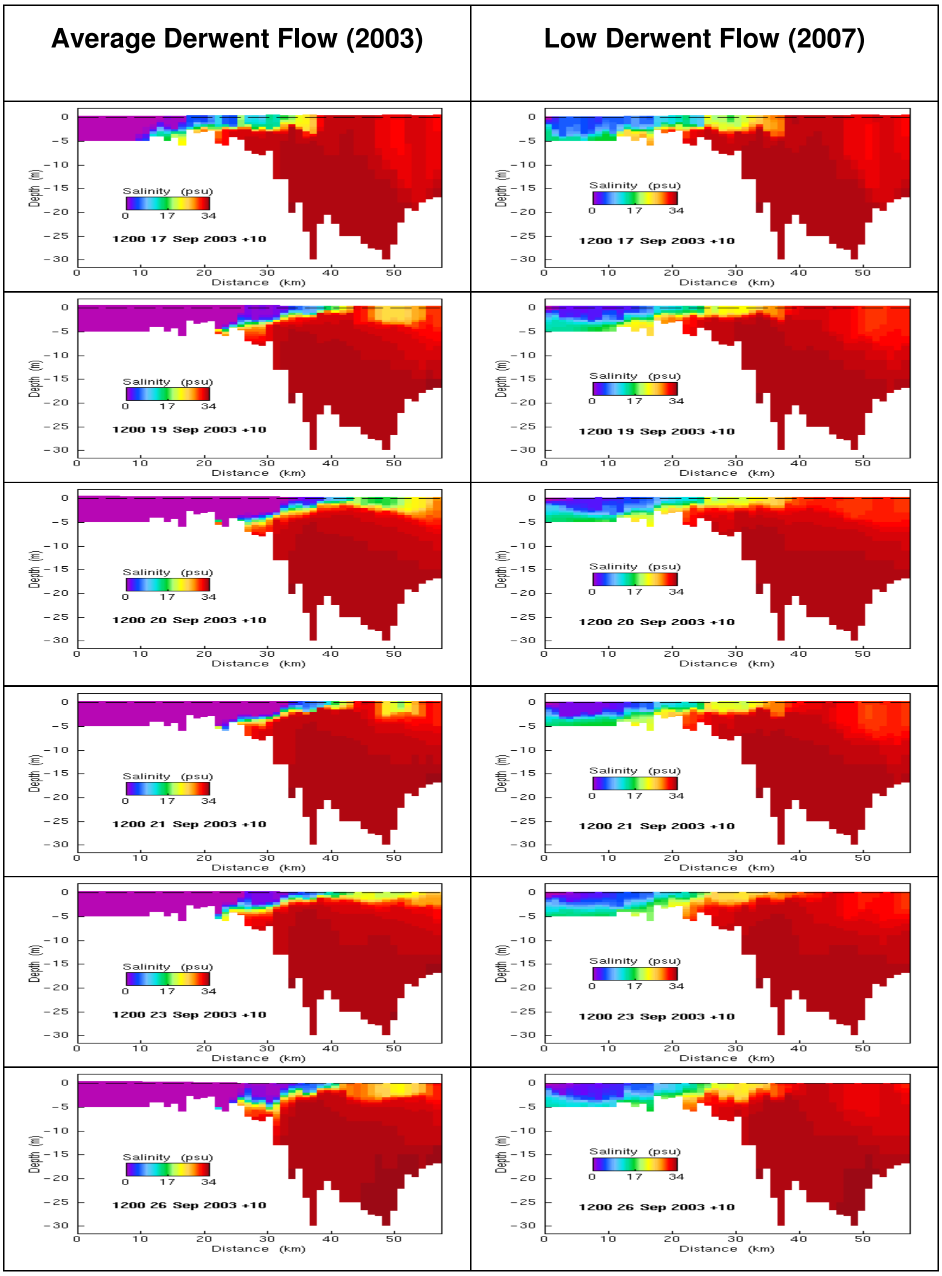


Figure 4.3 September salinity cross section along the axis of the estuary showing the 2003, near pristine and 2015 active management scenario (left) and the 2015 business-as-usual scenario (right) during September 2003.

4.2 Water Quality

4.2.1 Dissolved Inorganic Nitrogen

The seasonal mean near surface concentration of dissolved inorganic nitrogen (DIN) for all scenarios was highest in the upper and middle reaches of the estuary in winter with lowest DIN concentrations in the outer reaches and outer bays in summer (Figure 4.4).

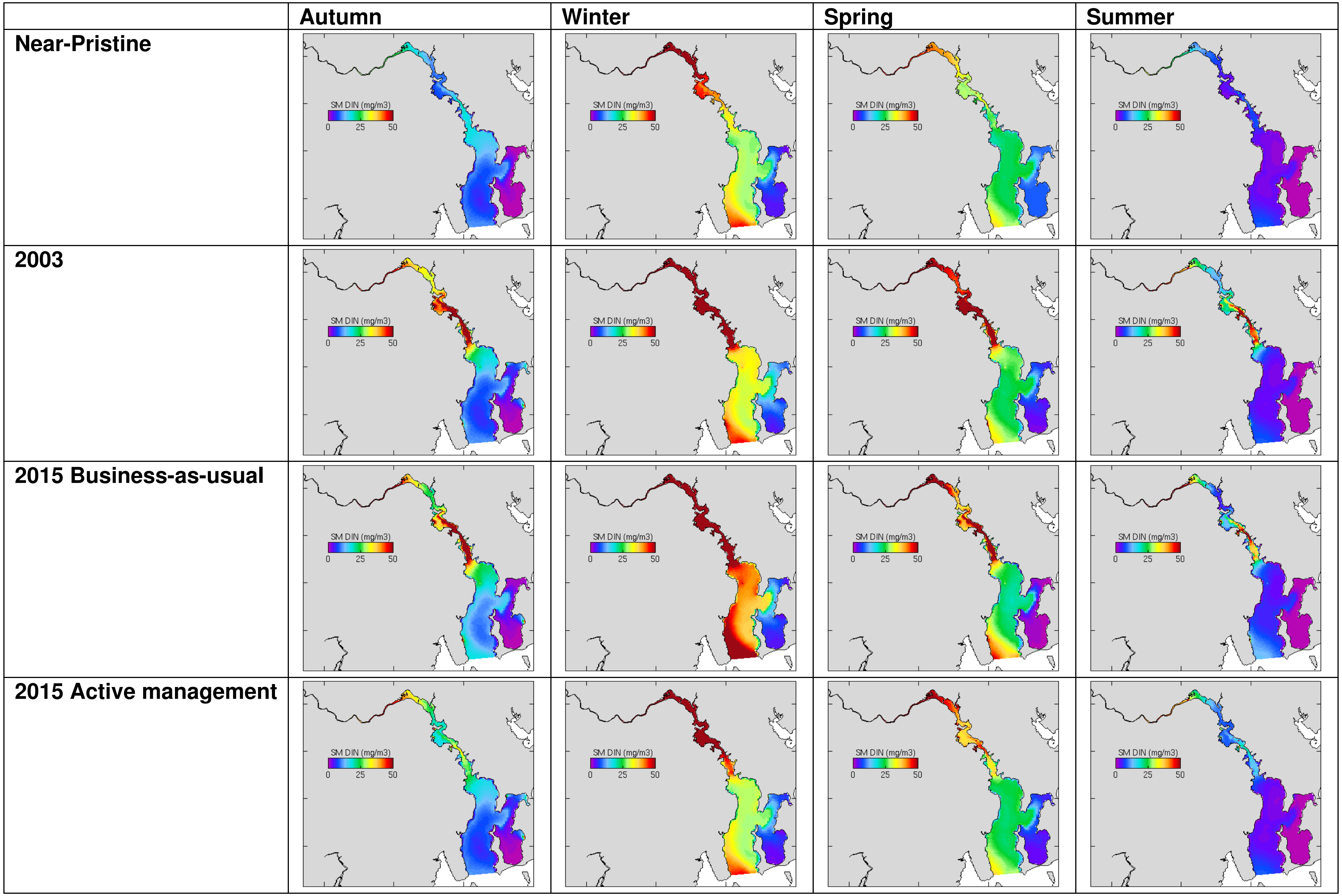


Figure 4.4 Seasonal mean near surface concentration (0-11m) of dissolved inorganic nitrogen for three scenarios and the 2003 Derwent Estuary calibrated model simulation

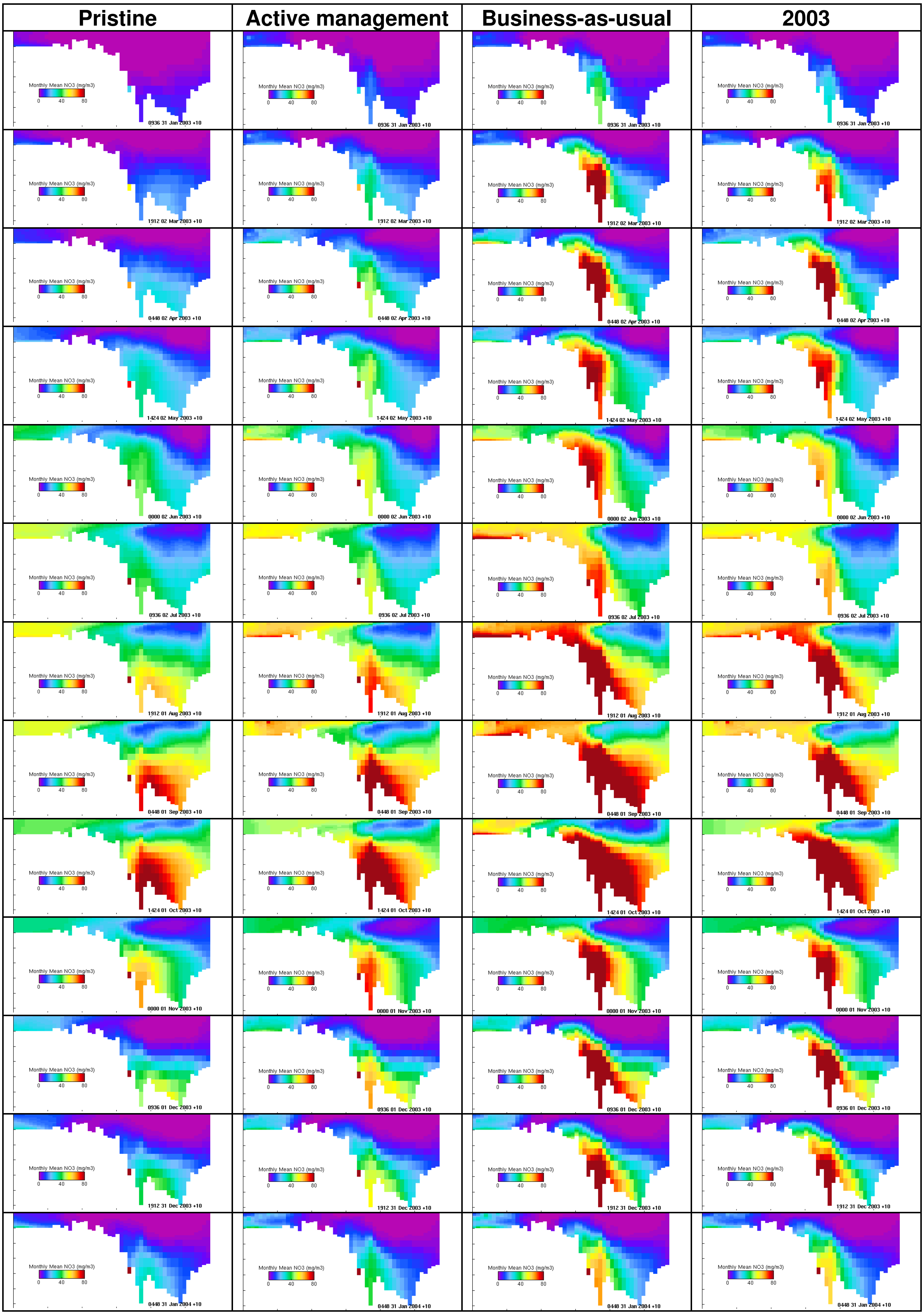


Figure 4.5 Cross sections of monthly mean DIN along the axis of the estuary from New Norfolk to Iron Pot from Feb '03 – Jan '04 for the three scenarios and the 2003 model simulation.

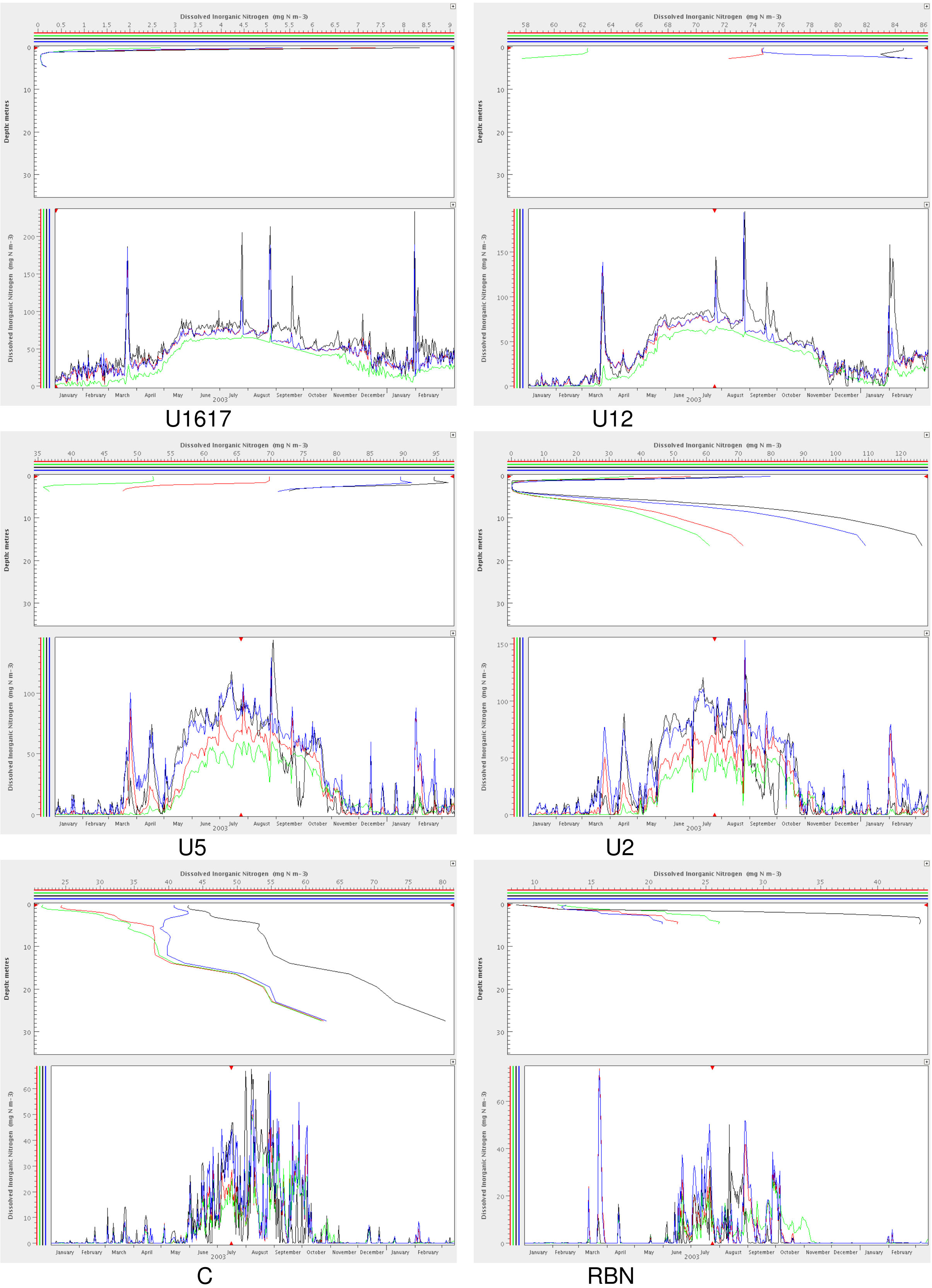


Figure 4.6 Annual time series (bottom) and winter depth profiles (top) of dissolved inorganic nitrogen at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

The near-pristine scenario simulation in general had the lowest DIN concentration of all the model runs for all seasons. The active management scenario had higher DIN concentrations in the upper and mid estuary than the near-pristine scenario but mean concentrations were lower than in the 2003 simulation.

The business-as-usual scenario had greater surface DIN concentrations throughout the year than the other scenarios, including elevated concentrations in the lower reaches in winter and spring. This is likely due to an increase in ammonia flux across the marine boundary (due to the reduced Derwent flow) and the business-as-usual elevated nutrient boundary condition. In Ralphs Bay the business-as-usual scenario had the lowest DIN concentrations in spring compared to all other runs, likely due to a change in nutrient supply associated with the low Derwent river flow hydrodynamics and/or phytoplankton uptake. The business-as-usual scenario simulation also had lower near surface DIN concentrations in parts of the middle reaches and inner bays compared with 2003 but concentrations were still higher than the active management and near-pristine scenarios.

Cross-sections along the axis of the estuary show elevated DIN concentrations at depth in winter in all simulations (Figure 4.5). In 2003 elevated DIN concentrations persisted throughout the year, due to continuous supply of anthropogenic nutrient to the estuary. The active management scenario shows a reduction in DIN concentration in spring, summer and autumn compared to 2003; the business-as-usual scenario simulates higher concentrations of DIN in bottom waters throughout the year.

At six different sites throughout the estuary (Figure 4.6) the lowest surface DIN concentrations were simulated by the near-pristine scenario followed by 2015 active management scenario; time series of annual surface DIN concentration for the business-as-usual scenario and the 2003 simulation were more similar. In the middle reaches (e.g. U5 and U2 Figure 4.6) the active management scenario had lower surface DIN concentrations than the business-as-usual scenario and the 2003 calibration.

Scenario Comparisons DIN

Figures quantifying the spatial and temporal differences between simulations are presented as spatial plots of number of days a relative threshold is exceeded (Figure 4.7). Both 2015 scenarios are compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario is compared with the 2003 simulation using negative thresholds to quantify the reduction in DIN concentration with the removal of anthropogenic inputs.

The business-as-usual scenario for the outer reaches up to Droughty Point and into Ralphs Bay north had over six months of the year where DIN was 25 to 50% higher than 2003 (Figure 4.7). The increase in DIN in this area was primarily due to the elevated ammonia flux across the marine boundary (see section 3.2). In addition the projected increase in population contributed additional STP DIN from the Blackmans Bay outfall, assuming no upgrade of the STP to tertiary treatment.

The active management scenario showed a dramatic reduction in DIN with respect to the business-as-usual scenario and a status quo or reduction in DIN in the middle and outer reaches compared with 2003 (Figure 4.7). The only area where the proportion of DIN was higher than in 2003 was in Ralphs Bay where for two months of the year DIN exceeded the 2003 simulation by 10%.

Near pristine conditions showed a reduction of over 50% in 2003 DIN for over six months of the year in the middle reaches and for three month of the year in the upper reaches (Figure 4.7). In the middle reaches for over three months of the year near-pristine DIN was 75% less than concentrations simulated in 2003. DIN at the southern boundary of near-pristine had not changed when compared with 2003 except near sewage treatment plants outfalls.

The increase in DIN loads from the Norske Skog paper mill in both the active management and business-as-usual scenario does not appear to have a great impact on the near surface DIN concentration of the upper estuary possibly due to rapid assimilation by phytoplankton. The change in hydrodynamics associated with the reduction in Derwent river flow in the business-as-usual scenario has a major impact on near surface DIN concentration in the lower reaches due to increased marine influx and in the upper reaches due to the upstream excursion of the salt wedge and nutrient rich mid-estuary deep water.

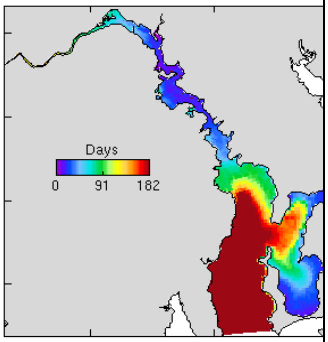
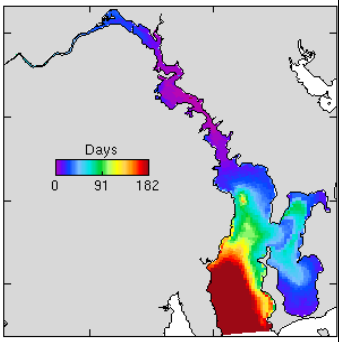
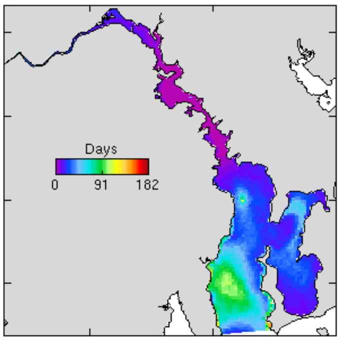
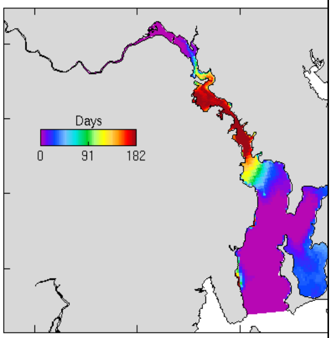
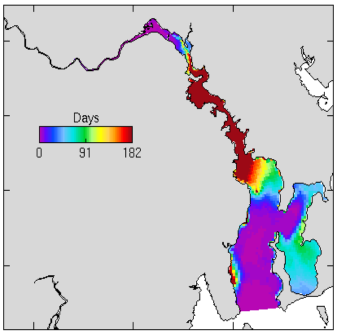
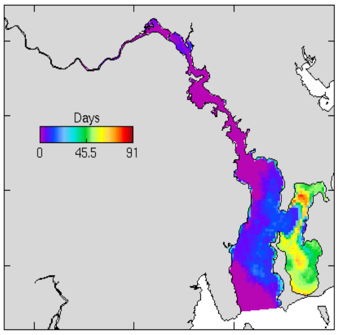
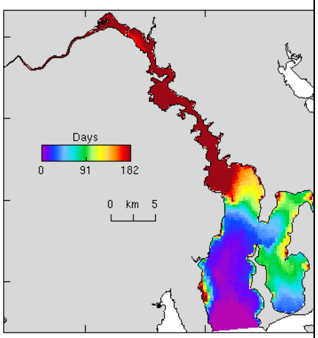
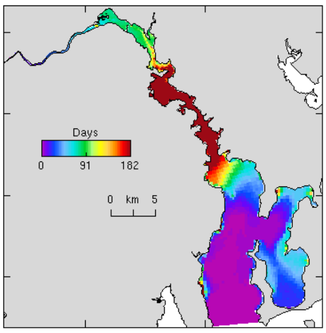
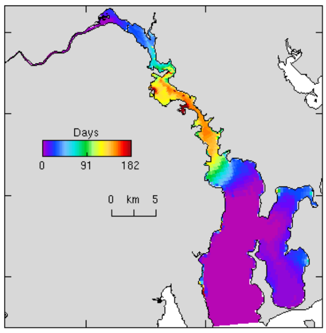
DIN			
	2003 as baseline		
2015 Business-as-usual scenario compared with 2003	 **2015 DIN > 25% 2003**	 **2015 DIN >50% 2003**	 **2015 DIN > 75% 2003**
2015 Active management scenario compared with 2003	 **2015 DIN < 50% 2003**	 **2015 DIN <25% 2003**	 **2015 DIN > 10% 2003**
Near-pristine scenario compared with 2003	 **near-pristine < 25% 2003 DIN**	 **near-pristine <50% 2003 DIN**	 **near-pristine <75% 2003 DIN**

Figure 4.7: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 near surface (0-11m) dissolved inorganic nitrogen thresholds exceeds 2003. Near-pristine scenario: Number of days in year when near pristine dissolved inorganic nitrogen less than 2003 thresholds. Note change in thresholds between plots.

4.2.2 Dissolved Inorganic Phosphate

Near surface seasonal mean dissolved inorganic phosphate (DIP) concentrations were lowest in the near-pristine scenario, followed by the 2015 active management scenario, and the 2003 simulation. Greatest near surface DIP concentrations were simulated by the 2015 business-as-usual scenario (Figure 4.8).

The near-pristine scenario simulation had very low concentrations of DIP in the upper estuary and in Ralphs Bay with only slightly elevated concentrations along the marine boundary in winter.

The business-as-usual scenario had the highest concentrations of near surface DIP compared with active management, near-pristine and 2003, particularly in the inner reaches and bays where concentrations were 15-20 mg/m^3. In contrast to the dynamics of DIN in the business-as-usual scenario concentrations of DIP were elevated in Ralphs Bay, indicating supply of phosphorus in excess of Redfield ratio. In comparison to DIN in the outer reaches of the estuary there was only a slight increase in DIP compared with 2003 associated with the slightly greater influx of water across the marine boundary.

The active management scenario had lower near surface DIP concentrations when compared with 2003 and the business-as-usual scenario. The greatest differentiation between scenarios for DIP occurred in the middle reaches of the estuary associated with contrasting point source loads and estuarine recirculation.

Cross-sections along the axis of the estuary show elevated DIP concentration at depth (Figure 4.9). In the near pristine scenario elevated concentrations were simulated in autumn and winter in the mid to outer reaches of the estuary. In 2003 anthropogenic supply of DIP to the estuary resulted in a considerable elevation in concentration particularly in the mid-estuary with high concentrations persisting throughout the year. The business-as-usual scenario showed further elevation of DIP concentrations above 2003 levels in all seasons, with increased deep water concentrations throughout the estuary outcropping into surface waters of the mid and upper reaches. In contrast, the active management scenario simulated a decline in deep DIP concentrations from 2003 levels including seasonal depletion of deep water DIP in spring.

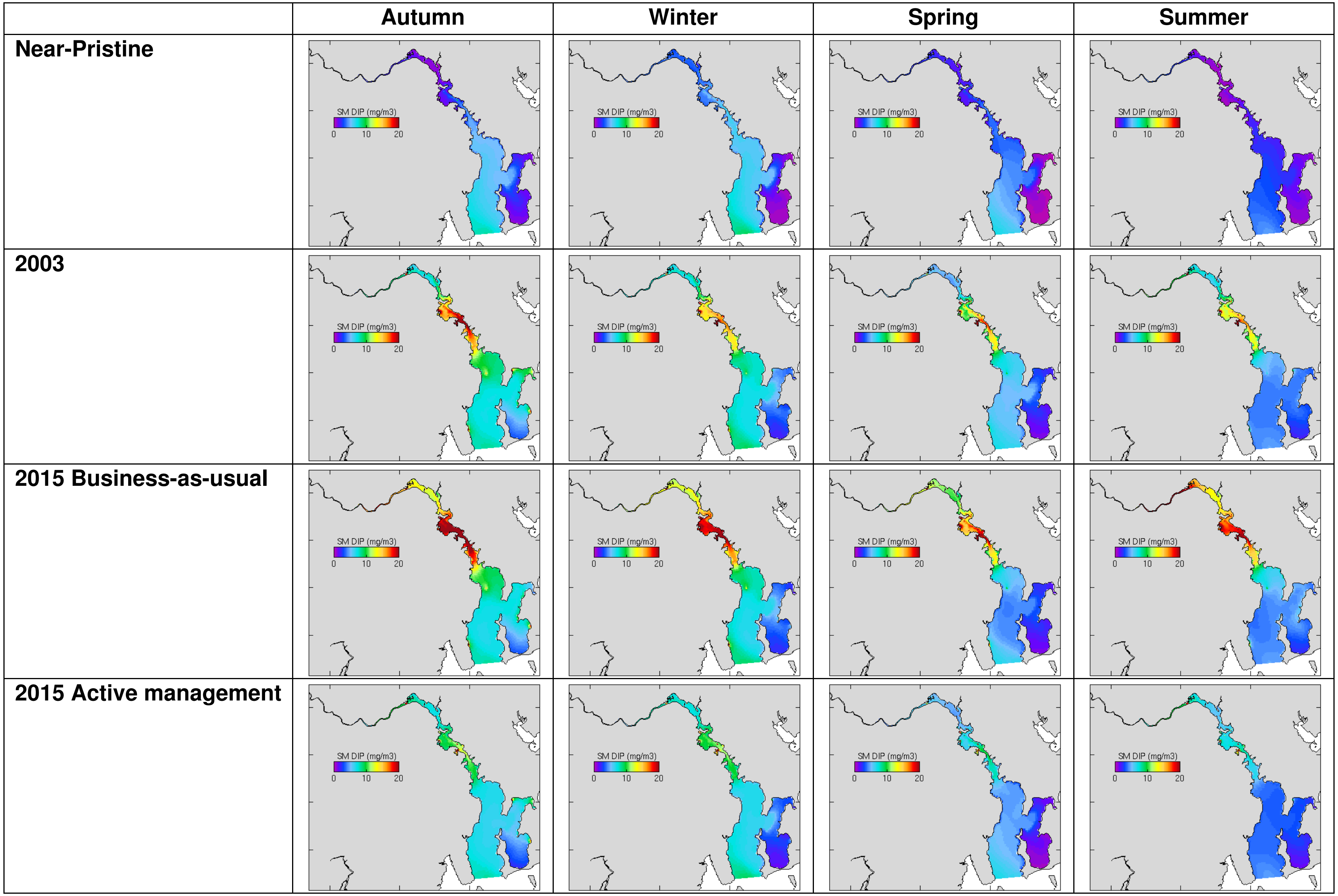


Figure 4.8 Seasonal mean near surface concentration (0-11m) dissolved inorganic phosphate for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

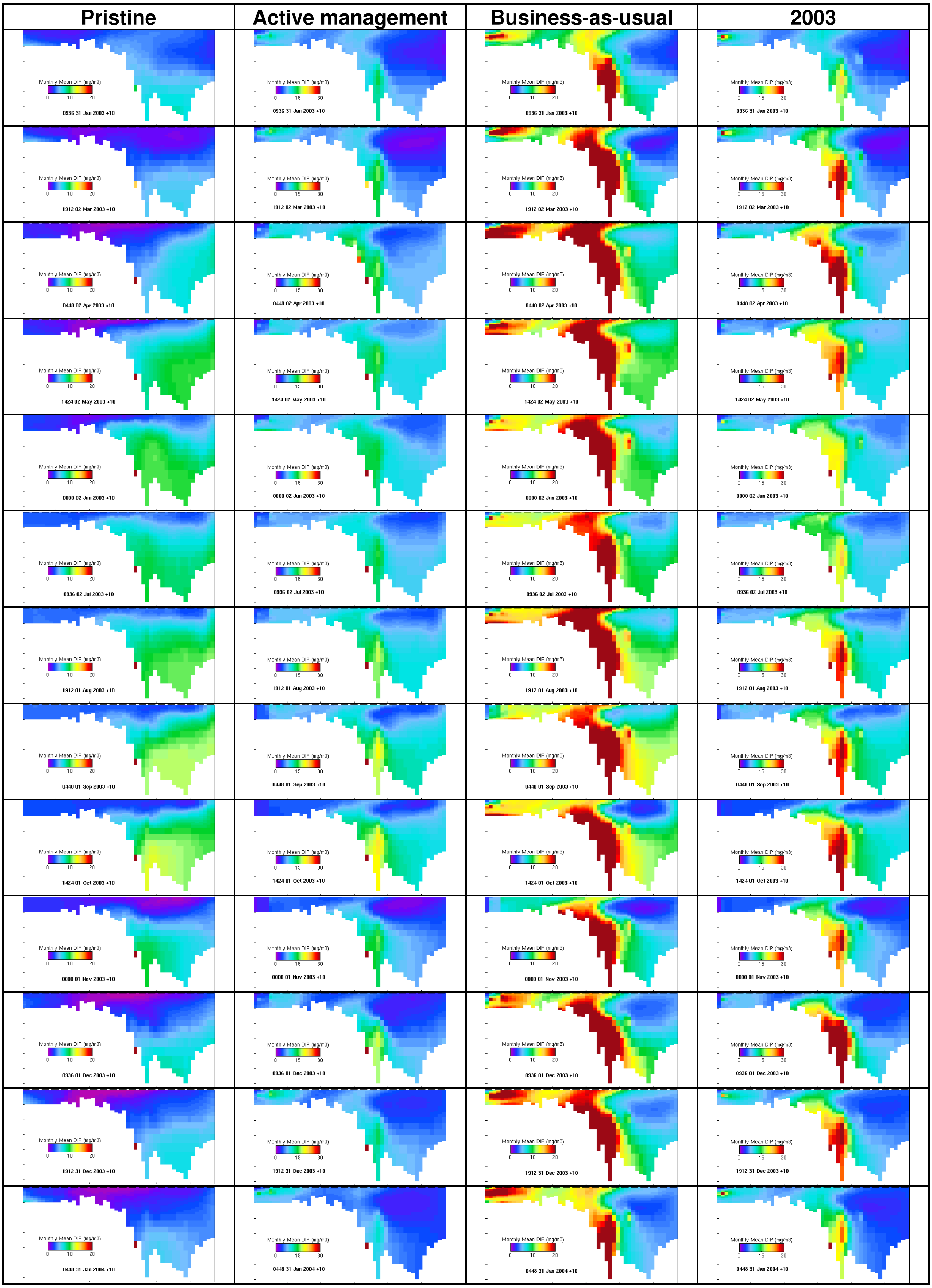


Figure 4.9 Cross sections of monthly mean DIP along the axis of the estuary from New Norfolk to Iron Pot from Feb ’03 - Jan ’04 for the three scenarios and 2003 model simulation.

Scenario Comparisons DIP

Similar to the analysis of near surface DIN Figure 4.10 quantifies the spatial and temporal differences between model simulations as spatial plots of number of days a relative threshold is exceeded. Both 2015 scenarios are compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario is compared with the 2003 simulation using negative thresholds to quantify the reduction in DIP concentration with the removal of anthropogenic loads.

The business-as-usual scenario had six months or more where near surface DIP exceeded 25% of 2003 concentrations in the upper reaches, middle reaches and inner embayments. In the upper reaches for six months of the year DIP exceeded 75% of 2003 concentrations whilst in southern Ralphs Bay DIP levels were more than 25% of 2003 concentrations for over three months (Figure 4.10).

In comparison the active management scenario simulations showed a dramatic improvement with respect to the business-as-usual scenario with a status quo or reduction in concentration in the middle and outer reaches and bays. There was an increase in DIP in the upper reaches (DIP was ~10% higher for one month a year when compared to 2003) due possibly to treated effluent from Norske Skog (Figure 4.10).

The near-pristine scenario had six months or more where DIP concentrations were reduced by 75% compared to 2003 in the middle and upper reaches and inner bays and for most of Ralphs Bay north and south (Figure 4.10). The outer reaches of the estuary had similar DIP concentrations in the near pristine scenario and the 2003 simulation due to the equivalence of boundary conditions for both simulations.

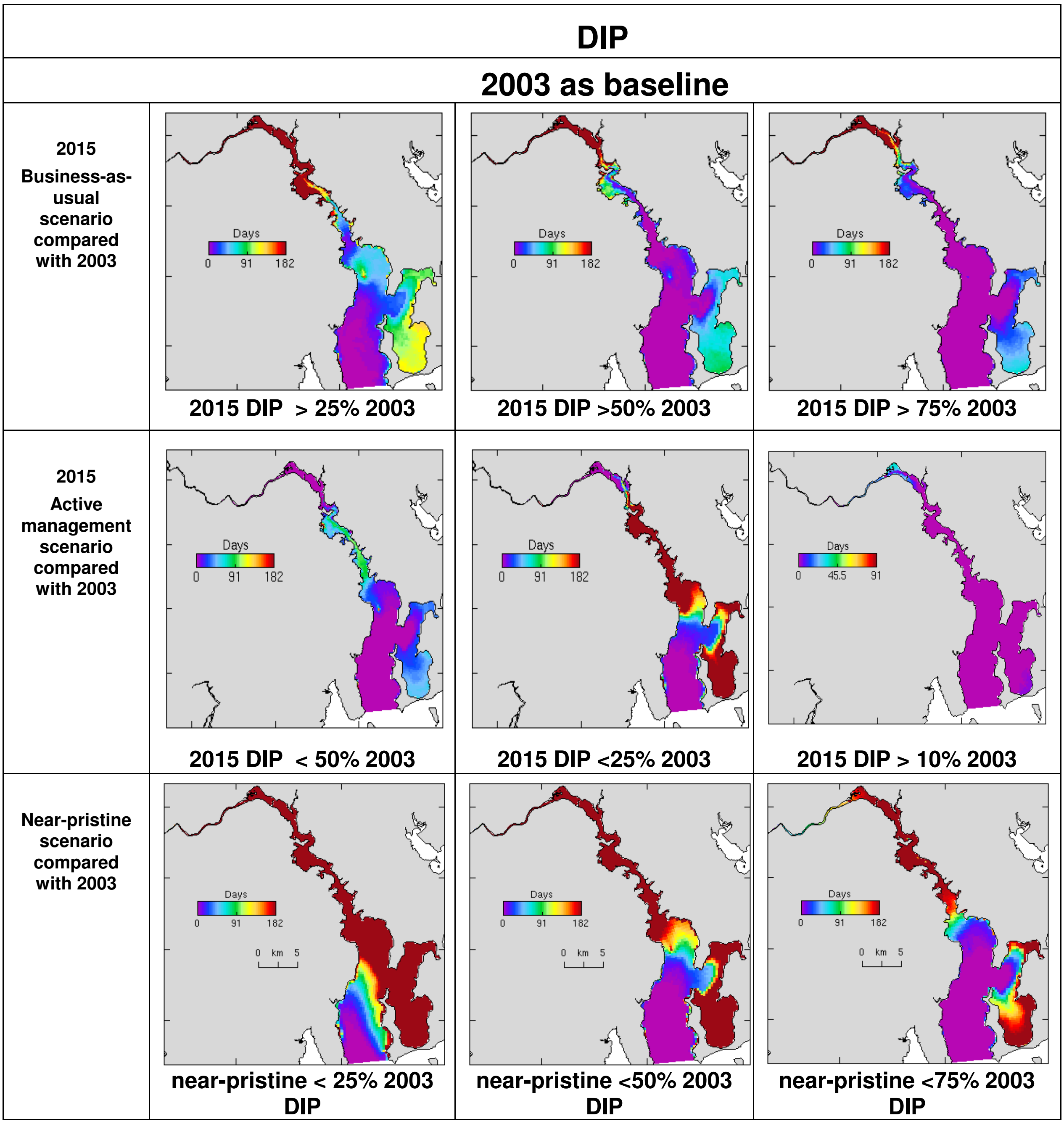


Figure 4.10: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 near surface (0-11m) dissolved inorganic phosphate thresholds exceeds 2003. Near-pristine scenario: Number of days in year when near pristine dissolved inorganic phosphate less than 2003 thresholds. Note change in thresholds between plots.

4.2.3 Chlorophyll and Phytoplankton Biomass

In the 2003 model validation exercise (Wild-Allen et al., 2009) chlorophyll concentrations were well simulated in most parts of the estuary although in the upper estuary modelled concentrations exceeded observations in autumn and in the middle reaches and inner bays the spring and autumn blooms persisted longer than observed. In these places and seasons model (including scenario) results should be treated with more caution.

Modelled seasonal mean near surface concentrations of chlorophyll were lowest in the near pristine scenario, followed by the 2015 active management scenario and the 2003 simulation; the highest levels of chlorophyll were simulated in the 2015 business-as-usual scenario. In all simulations the chlorophyll concentrations were highest in the middle reaches of the estuary (Figure 4.11) and ranged from 4 $mg.m^{-3}$ to >8 $mg.m^{-3}$.

For the business-as-usual scenario mean near surface chlorophyll concentrations were very high (> 8 $mg.m^{-3}$) in the middle reaches of the estuary in autumn, spring and summer. In winter the business-as-usual scenario chlorophyll concentrations remained higher than in the other model runs (> 6 $mg.m^{-3}$). The business-as-usual scenario had the highest chlorophyll compared with the other scenarios in winter in Ralphs Bay consistent with the lower concentrations of DIN that were simulated for the same time period. In the upper reaches of the estuary elevated chlorophyll concentrations were found extending above the Bridgewater causeway in summer and autumn, likely supported by the upstream movement of the salt wedge and entrainment of nutrient rich mid estuary water into the upper reaches. Availability of photosynthetically active radiation (PAR) in the upper estuary was also greater due to the smaller contribution of coloured dissolved organic material (CDOM) associated with the reduced Derwent river flow.

Figure 4.12 display the difference in chlorophyll concentrations between the scenarios at specific sites throughout the estuary. The near-pristine scenario had the lowest chlorophyll concentrations for all sites followed by the active management scenario. Chlorophyll concentrations for the business-as-usual scenario and 2003 were similar for the sites chosen. Similar to the dynamics of DIN and DIP, sites in the middle reaches of the estuary display the greatest differentiation between the scenarios (Figure 4.12 U2 and U5). Here the active management scenario chlorophyll concentrations were clearly lower than the business-as-usual scenario and the 2003 simulation. In Ralphs Bay North chlorophyll concentrations were variable and for some periods in the year the active management scenario concentrations were slightly higher than in the other simulations.

Figure 4.13, Figure 4.14 and Figure 4.15 show the modelled contribution of dinoflagellates and large phytoplankton to the phytoplankton biomass of each model run at six different sites throughout the estuary. There were no significant changes in small phytoplankton biomass between model runs so this group is not shown. [In 2003 there were no observations of phytoplankton species, so this aspect of the model has not been validated against observations but is consistent with our understanding of phytoplankton succession in the estuary (see Wild-Allen et al., 2009).]

All sites and scenarios, including the near-pristine scenario, show an autumn and early summer increase in dinoflagellate concentration. Overall the business-as-usual scenario had the highest levels of dinoflagellate biomass. The active management scenario and 2003 simulation had similar levels of dinoflagellate biomass and the near-pristine scenario had the lowest dinoflagellate concentrations. Again the middle reaches were where the greatest difference between scenarios can be observed with near-pristine lower than the 2003 simulation and the active management scenario. In Ralphs Bay, the

2003 simulation and the active management scenario had the highest concentrations of dinoflagellates when compared with business-as-usual and near-pristine scenarios.

Large phytoplankton had the lowest biomass in the near-pristine scenario (Figure 4.14). In summer, the 2003 simulation generated high concentrations of large phytoplankton in the middle reaches although the maximal concentrations were simulated in late winter in the business-as-usual scenario.

The succession and depth profiles of the dinoflagellates compared with large phytoplankton over the year can be seen in Figure 4.15. This succession was similar to limited observations taken within the estuary. The location, succession and timing of phytoplankton blooms were similar for all model simulations.

Monthly mean chlorophyll concentrations along the axis of the estuary are shown in (Figure 4.16) for the different model simulations. The transect plots show high concentrations of chlorophyll in the middle reaches of the estuary even in the near-pristine scenario. The active management scenario shows a decrease in chlorophyll when compared to 2003 and the business-as-usual shows chlorophyll levels remain high and persist throughout winter in the middle reaches. It should be noted that the 2003 calibrated model slightly overestimated chlorophyll in the upper reaches in autumn, and the middle reaches and inner bays in late spring and late autumn (Wild-Allen et al., 2009). Results in these locations and periods should be interpreted with more caution.

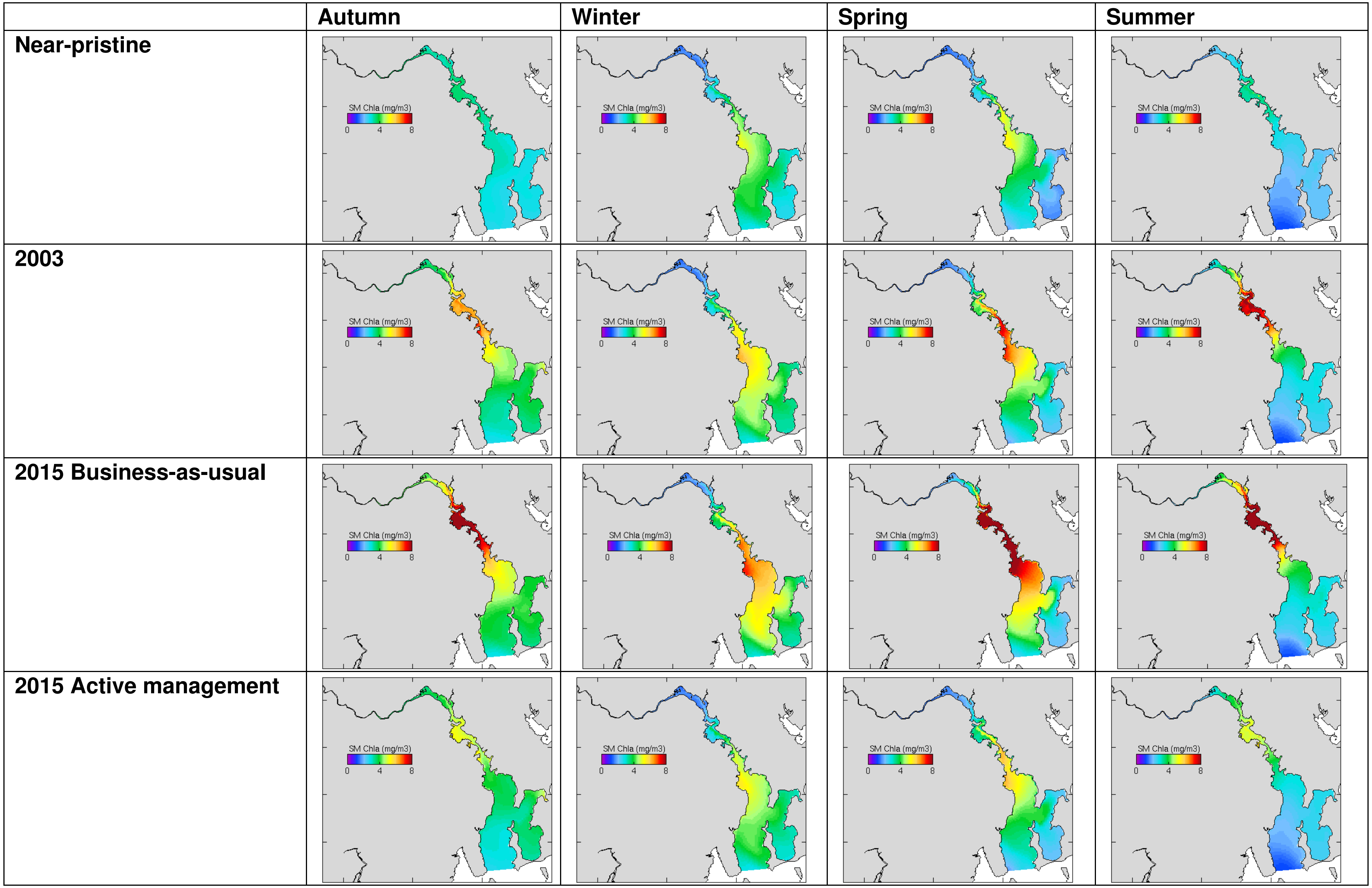


Figure 4.11 Seasonal mean near surface concentration (0-11m) chlorophyll for three scenarios and the 2003 Derwent Estuary calibrated model simulation

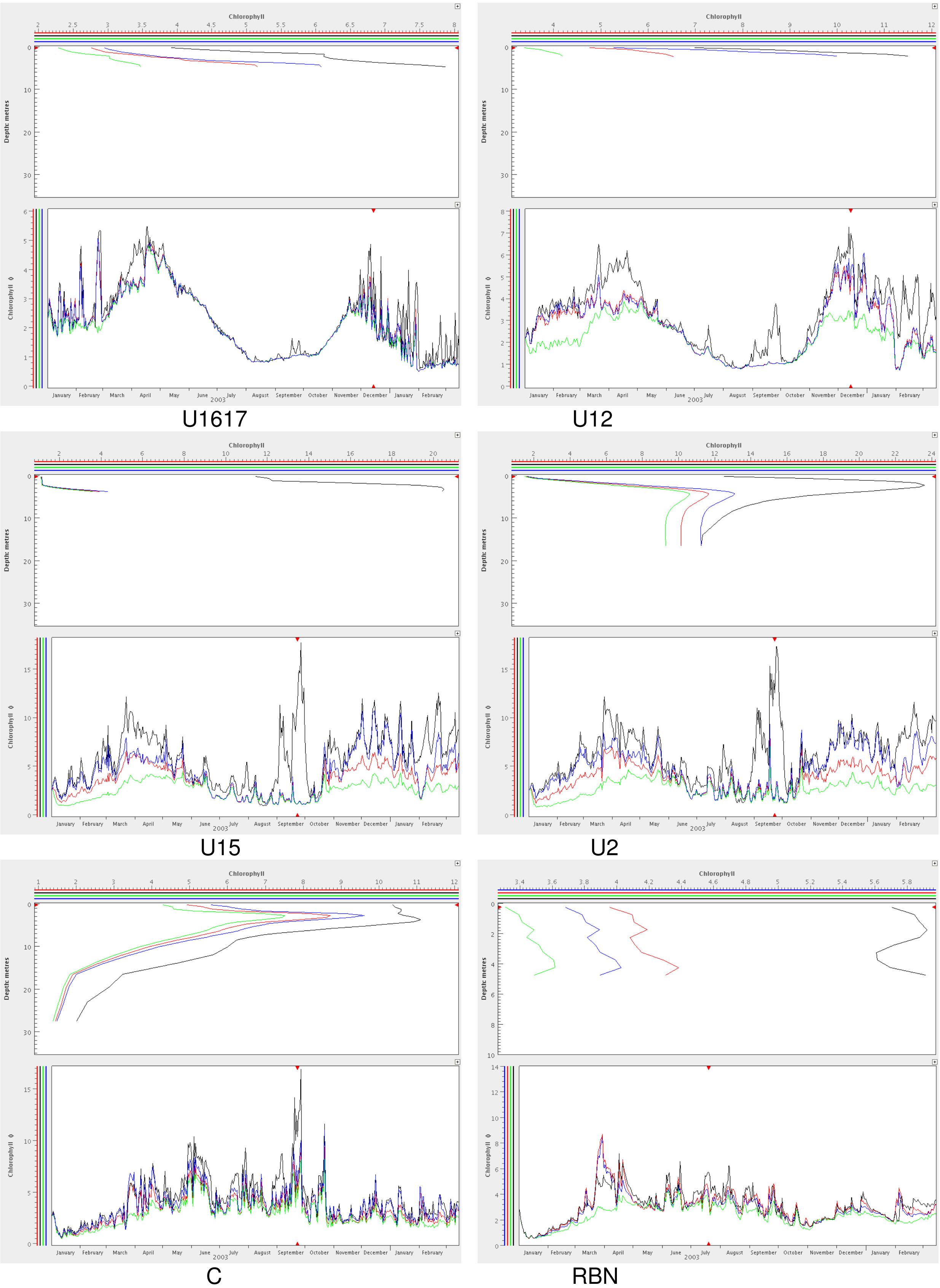


Figure 4.12 Annual time series (bottom) and winter or spring depth profiles (top) of chlorophyll at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

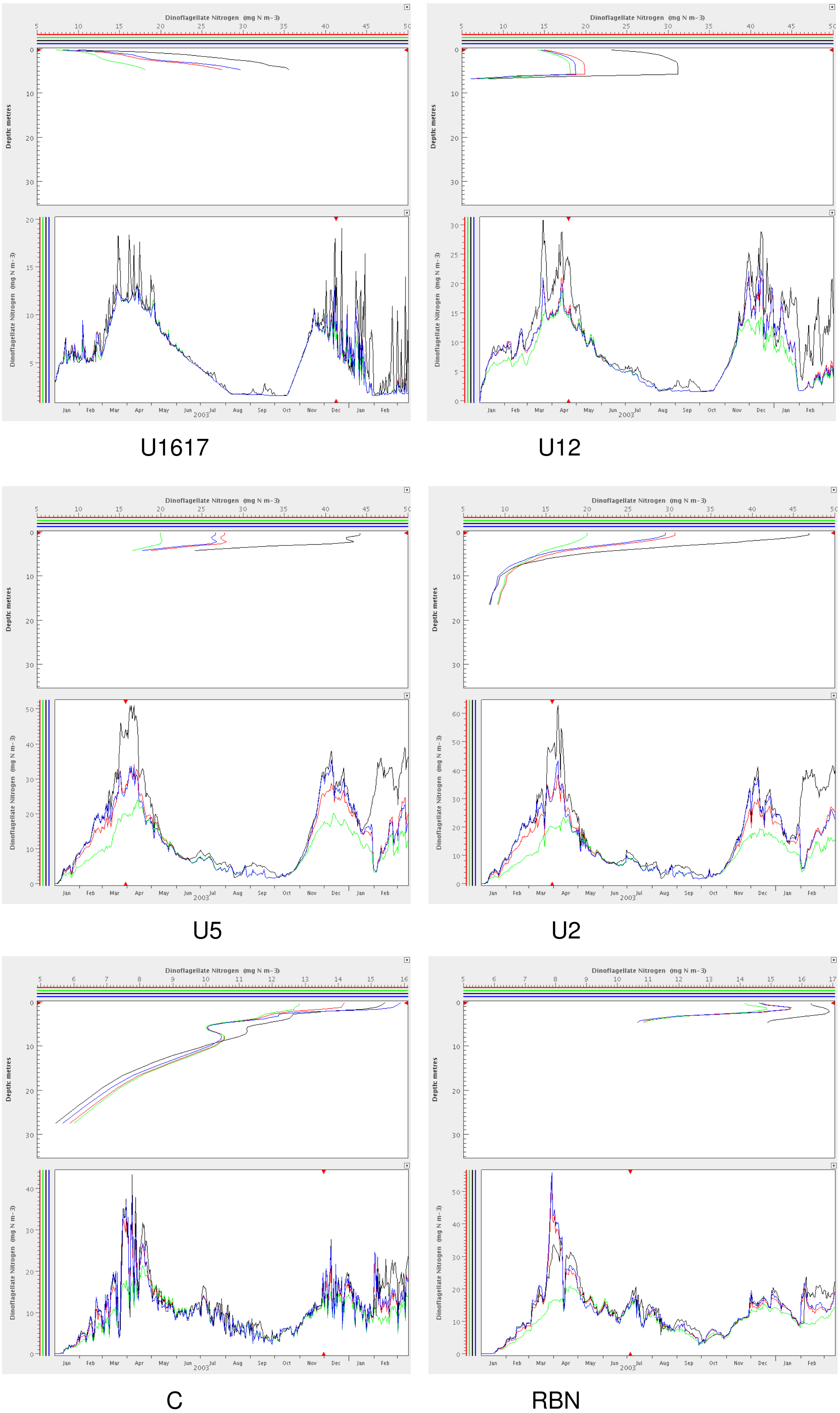


Figure 4.13 Annual time series (bottom) and autumn or spring depth profiles (top) of dinoflagellate nitrogen biomass at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

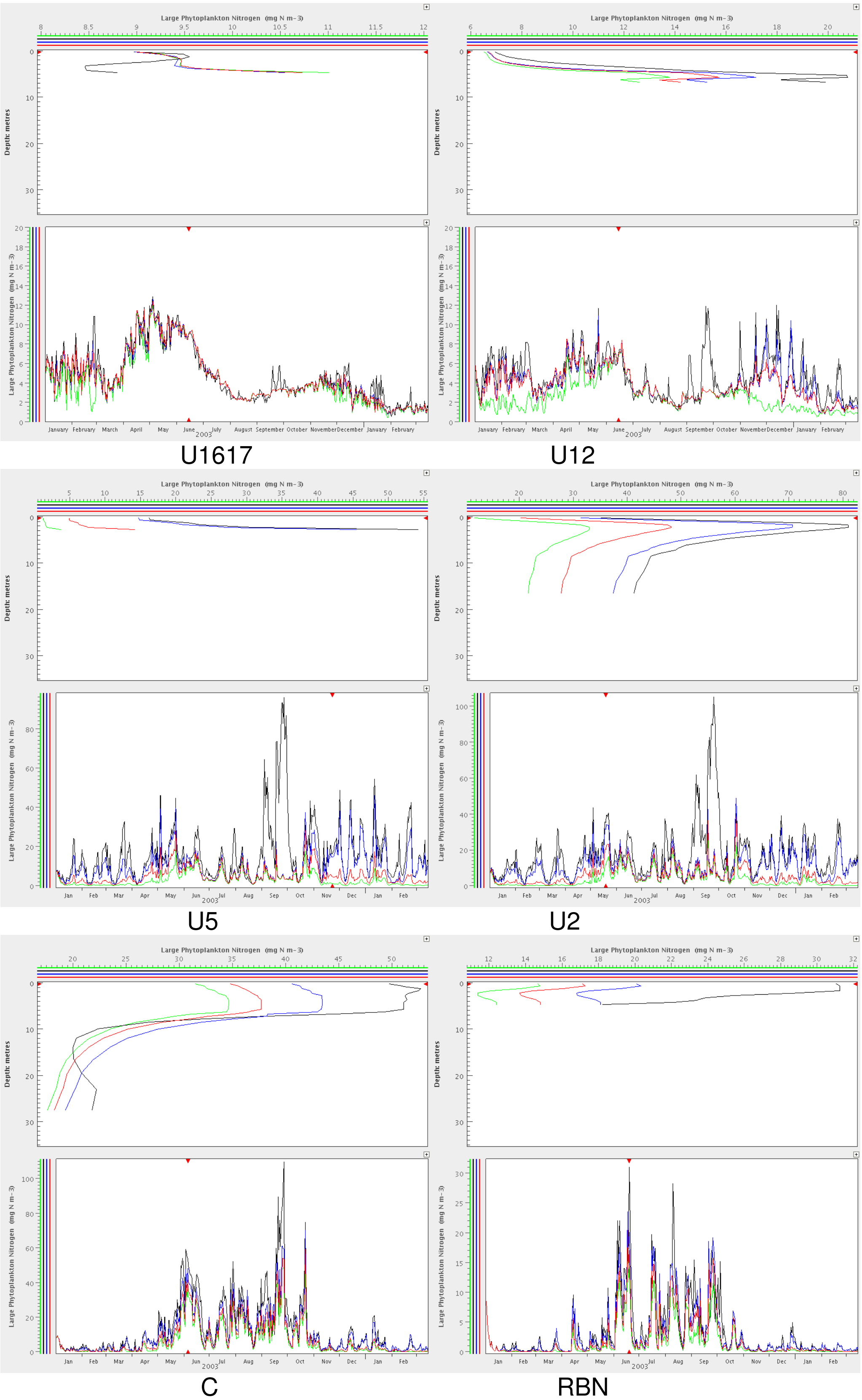


Figure 4.14 Annual time series (bottom) and autumn or spring depth profiles (top) of large phytoplankton nitrogen biomass at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

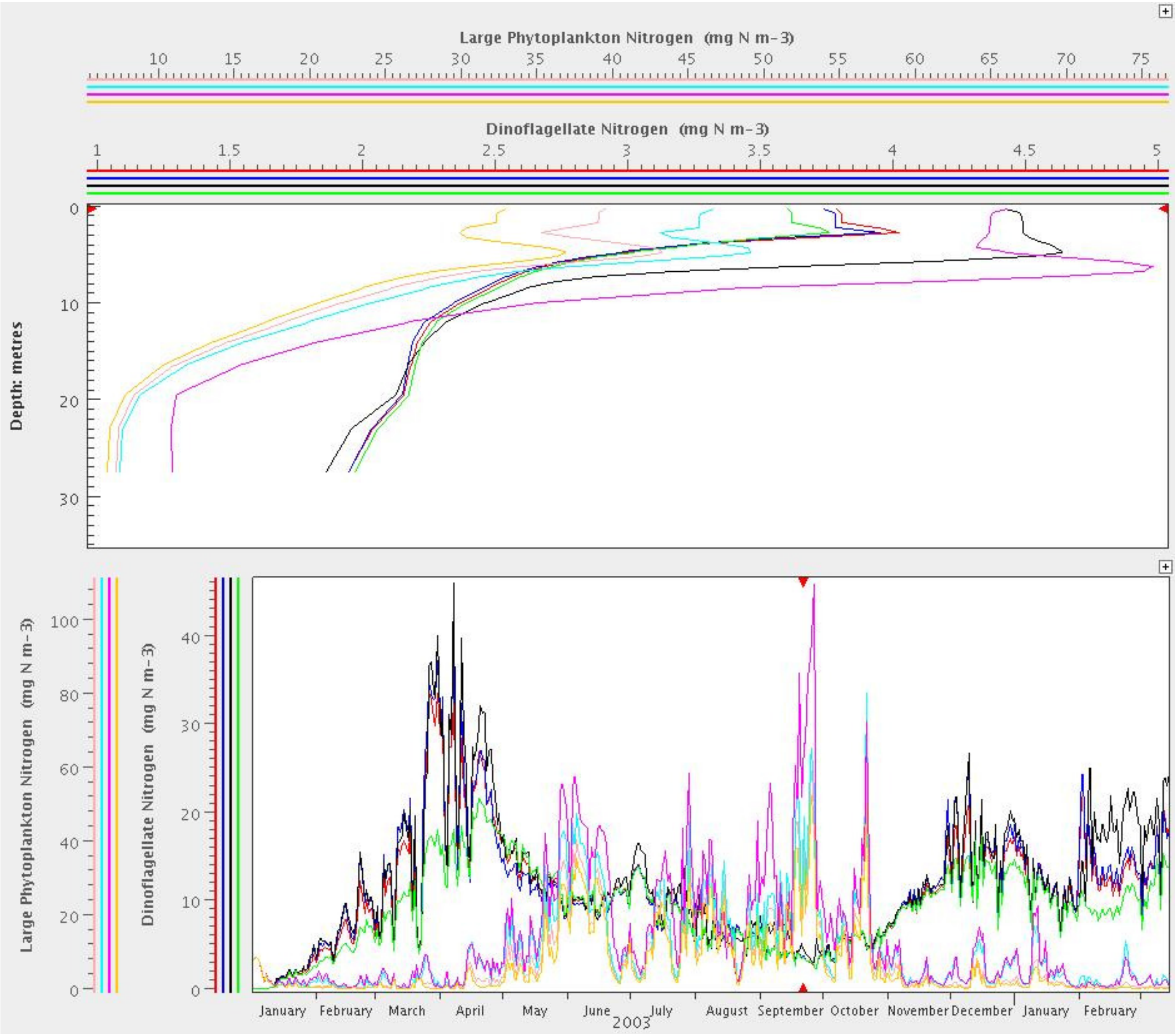


Figure 4.15 Compilation of Figure 4.13 and Figure 4.14 showing annual succession and depth profile of dinoflagellates (autumn and summer blooms) and large phytoplankton (diatoms: late autumn spring blooms) at site C in the Derwent Estuary (see Figure 3.1 for location). Figure demonstrates simulations from different scenarios dark and light blue are calibrated 2003 model run, red and light pink are 2015 active management scenario, black and dark pink are business-as-usual 2015 management scenario green and yellow are near-pristine.

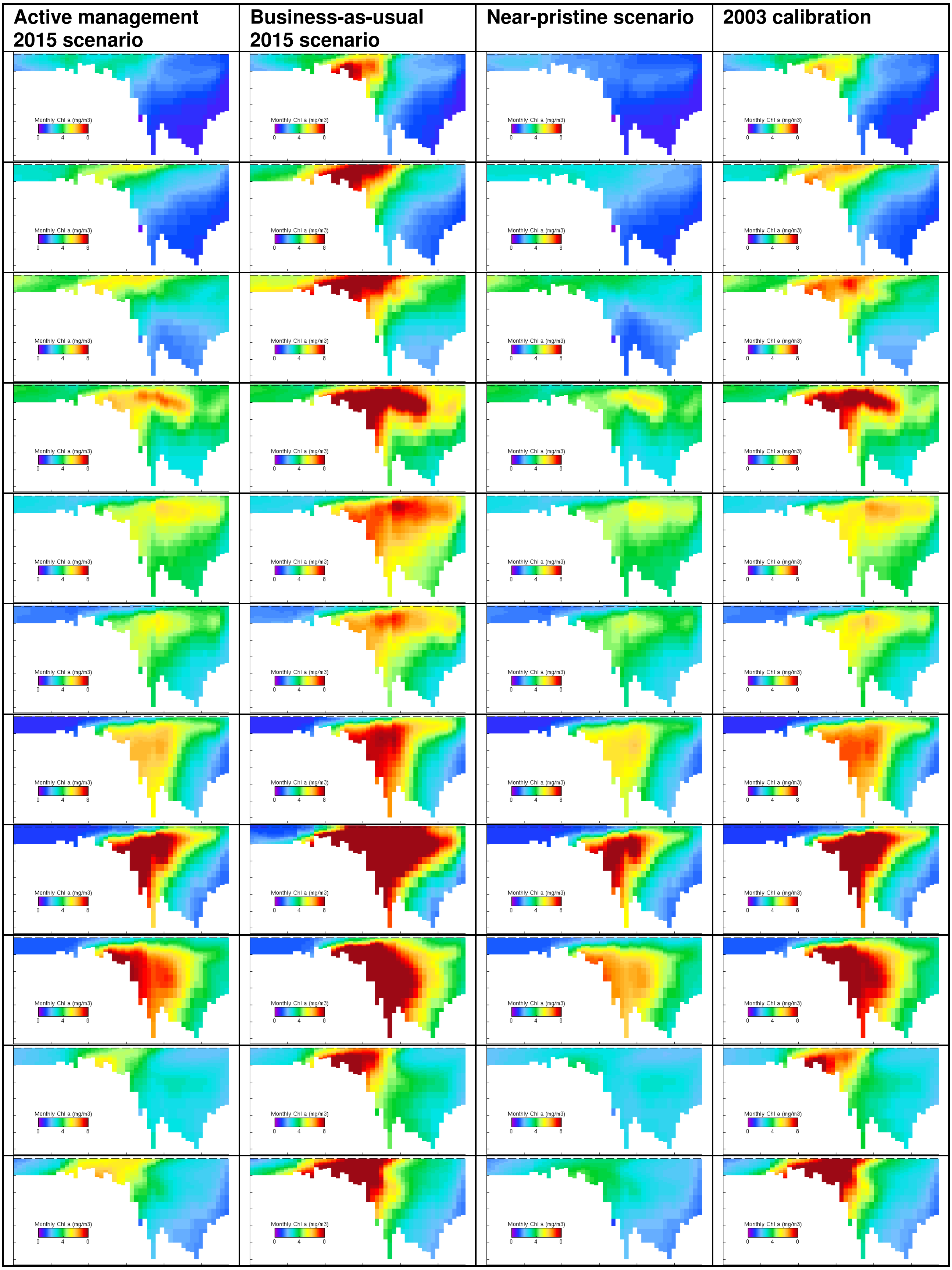


Figure 4.16 Cross sections of monthly mean chlorophyll along the axis of the estuary from New Norfolk to Iron Pot from Feb '03 - Dec '03 for the three scenarios and 2003 model simulation.

Scenario Comparisons Chlorophyll

Figures quantifying the spatial and temporal differences between simulations are presented as spatial plots of number of days a relative threshold is exceeded (Figure 4.17). Both 2015 scenarios are compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario is compared with the 2003 simulation using negative thresholds to quantify the reduction in chlorophyll associated with the removal of anthropogenic loads.

Chlorophyll in the 2015 business-as-usual scenario exceeded 2003 values by over 25% for six months of the year for the upper and upper middle reaches and for approximately one month in the year in the outer reaches (Figure 4.17). For more than two months of the year chlorophyll exceeded 75% of 2003 levels in the middle and upper reaches of the estuary.

The active management scenario simulation show a dramatic improvement with respect to the business-as-usual scenario with a status quo or reduction in chlorophyll concentration throughout the estuary when compared with 2003 (Figure 4.17). In the middle reaches of the estuary chlorophyll concentrations were reduced by more than 25% of 2003 concentrations for up to 5 months of the year.

Near-pristine chlorophyll concentrations were 25% lower than 2003 levels for six months of the year in the middle reaches and for three months of the year in Ralphs Bay (Figure 4.17).

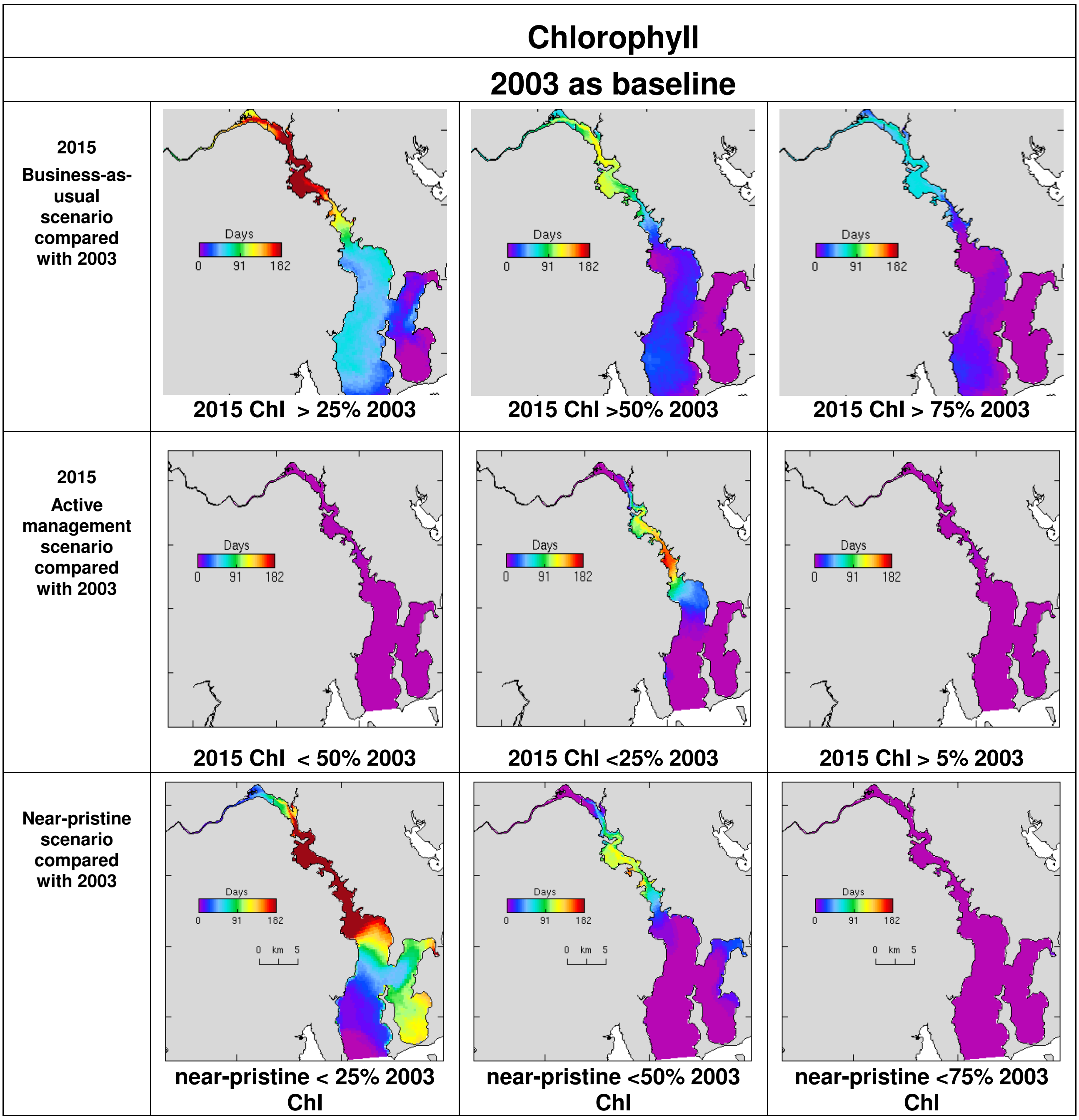


Figure 4.17: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 near surface (0-11m) chlorophyll thresholds exceeds 2003. Near-pristine scenario: Number of days in year when near pristine chlorophyll less than 2003 thresholds. Note change in thresholds between plots.

4.2.4 Zooplankton Grazing

There were no observations of zooplankton or grazing in 2003 and this aspect of the model, whilst consistent with our understanding of zooplankton dynamics in the estuary, is not validated against observations (Wild-Allen et al., 2009). Zooplankton model (including scenario) results should therefore be treated only as a plausible hypothesis of conditions prevailing in the estuary. The plots of temporal evolution and depth profile of modelled small (Figure 4.18) and large (Figure 4.19) zooplankton grazing suggest that secondary production in the estuary is dominated by small zooplankton grazing for all scenarios and the 2003 simulation. The middle reaches had the highest levels of zooplankton grazing for all scenarios and the 2003 simulation. The general succession of zooplankton grazing for all three scenarios did not appear to change relative to the 2003 simulation.

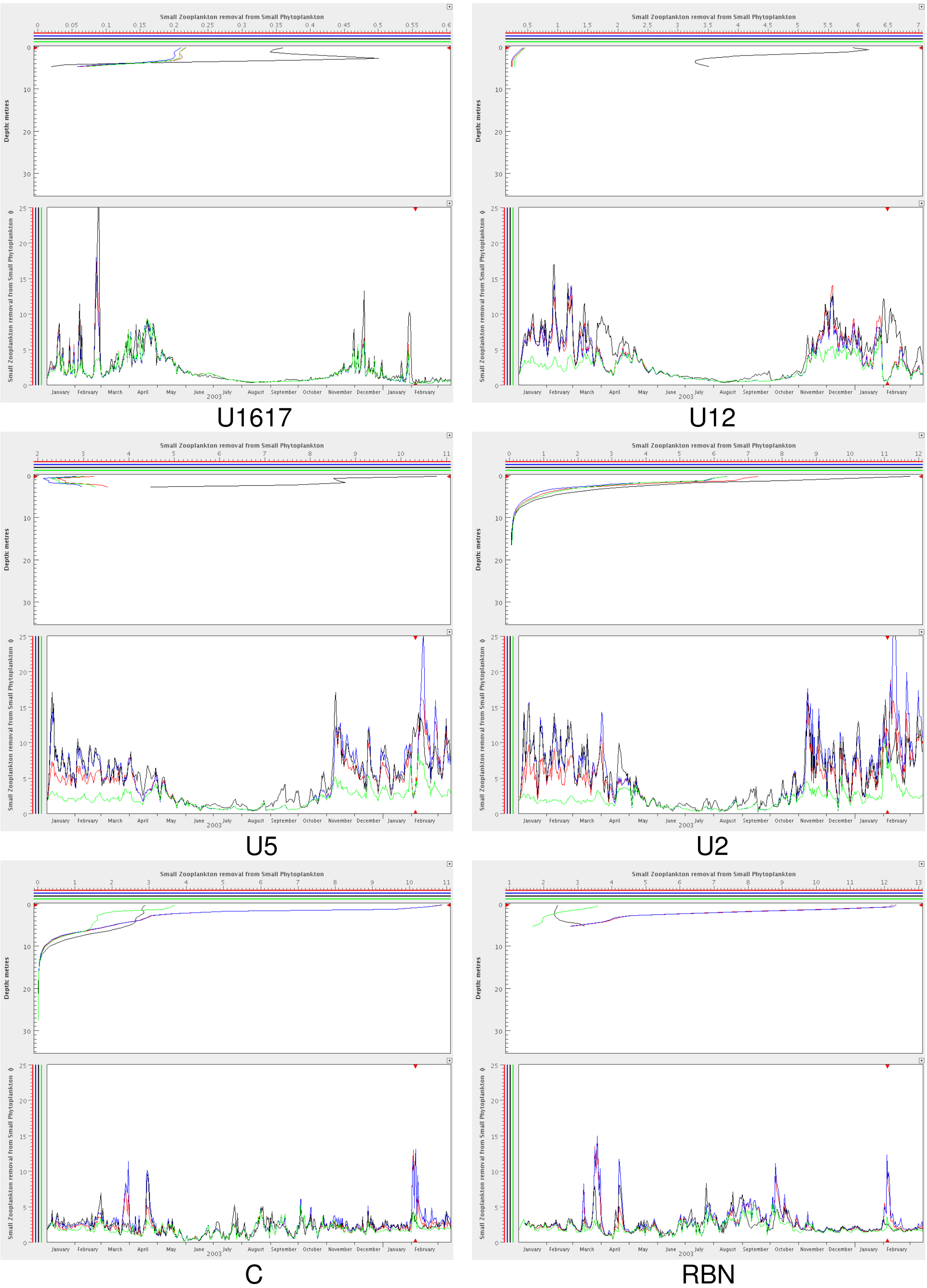


Figure 4.18: Annual time series (bottom) and summer depth profiles (top) of small zooplankton grazing at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

Modelled small and large zooplankton grazing occurred throughout the year proportional to zooplankton biomass and prey concentration. In general simulated grazing levels were lower over winter and elevated in spring and autumn associated with the seasonal increase in phytoplankton production. In the outer reaches of the estuary the seasonality in modelled grazing was reduced with occasional spikes in activity possibly due to patchiness in plankton populations. Modelled grazing occurred at similar intensities throughout all regions of the estuary although small zooplankton grazing was maximal in the upper layers of the water column and large zooplankton grazing greater at depth (Figure 4.18 and Figure 4.19). This depth stratification of modelled grazing intensity results from the contrasting depth distributions of small neutrally buoyant phytoplankton and large phytoplankton that sink.

Differences in grazing activity between the model simulations were generally small except in the middle reaches of the estuary. The business-as-usual scenario and the 2003 simulation had elevated grazing by both large and small zooplankton in the middle reaches, particularly during spring and autumn, compared with the active management and near-pristine scenarios. This was due to greater prey abundance in the middle reaches of the business-as-usual scenario and the 2003 simulation compared to the other model runs.

The near-pristine scenario had the lowest zooplankton grazing levels compared with the other simulations. Grazing was five times lower for the near-pristine scenario in the middle reaches in spring than for the business-as-usual scenario, due to the contrast in prey abundance between the two model scenarios.

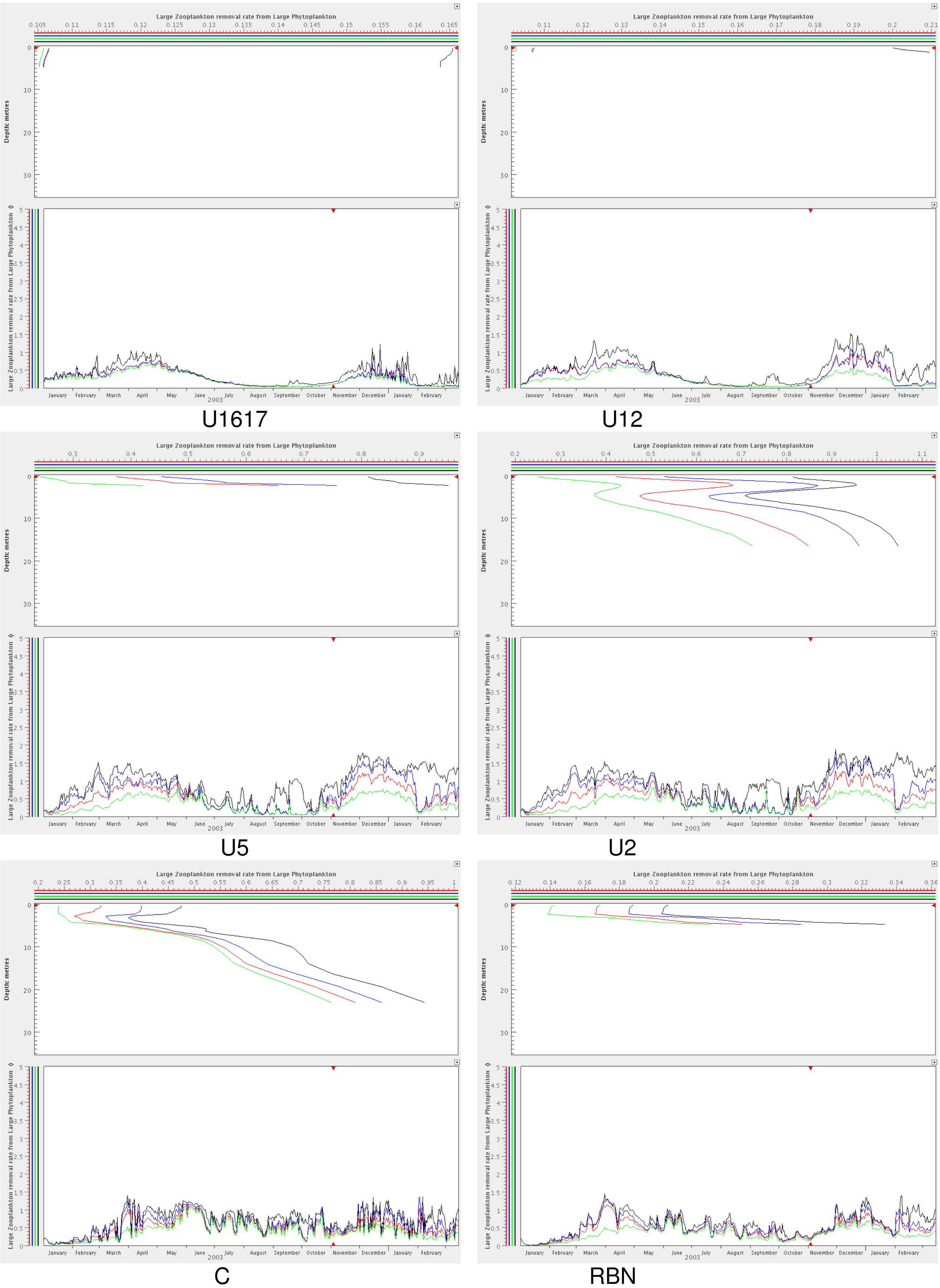


Figure 4.19 Annual time series (bottom) and spring depth profiles (top) of large zooplankton grazing at 6 sites in the Derwent Estuary: upper estuary (U1617, U12); middle estuary (U5, U2); outer estuary and Ralphs Bay (C and RBN). Blue is calibrated 2003 model, red is 2015 active management scenario, black is business-as-usual 2015 management scenario, green is near-pristine.

4.2.5 Bottom Water Dissolved Oxygen Saturation

Modelled seasonal mean bottom water dissolved oxygen (DO) saturation was highest for the near-pristine scenario (Figure 4.20 and Figure 4.21) compared with the other scenarios and the 2003 simulation. Bottom water DO saturation in the active management scenario simulation was slightly lower in the middle reaches of the estuary than the near-pristine scenario. The business-as-usual scenario and 2003 were very similar with lower DO bottom water saturation than near-pristine and the active management scenario. The near-pristine scenario had highest DO saturation during summer 2004 in the middle to outer reaches when compared with 2003 and the other scenarios.

Seasonal mean bottom water DO saturation was lowest in the deeper section of the middle reaches from the Bowen Bridge and out towards the entrance of Ralphs Bay for all scenarios and the 2003 simulation (60-75% Figure 4.20).

Cross-sections of monthly mean DO saturation along the axis of the estuary show that in the business-as-usual scenario the deeper bottom waters are less oxygenated in the autumn and spring (Figure 4.21). The reduced Derwent river flow used in the business-as-usual scenario also results in the excursion of mid-estuary bottom water up-stream into the upper reaches of the estuary during summer and autumn reducing the oxygen content of the upper estuary. The cross-sections also show a higher degree of DO saturation in the near-pristine scenario throughout the year and along the axis of the estuary.

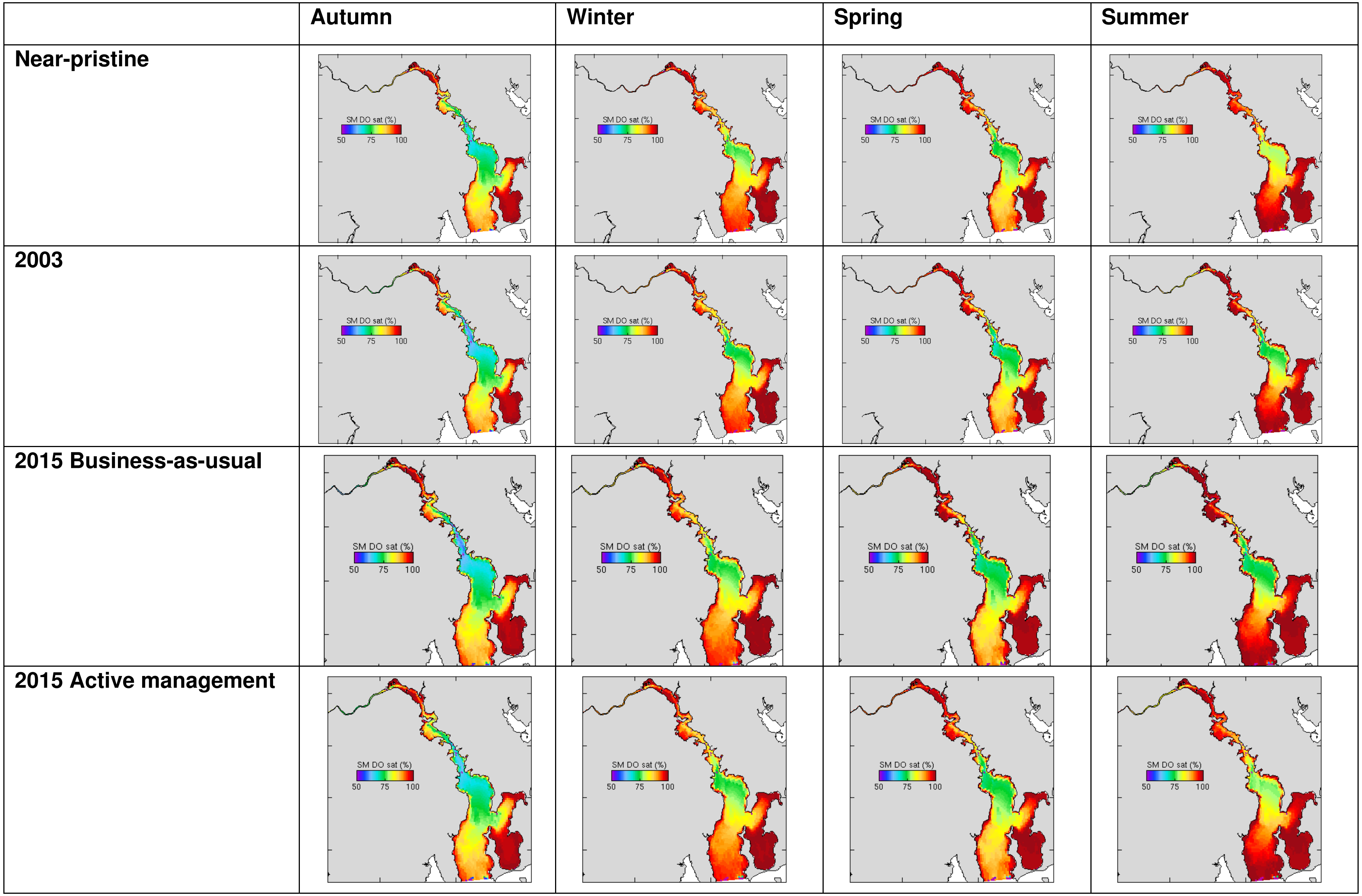


Figure 4.20 Seasonal bottom water dissolved oxygen saturation (%) for three scenarios and the 2003 Derwent Estuary calibrated model simulations

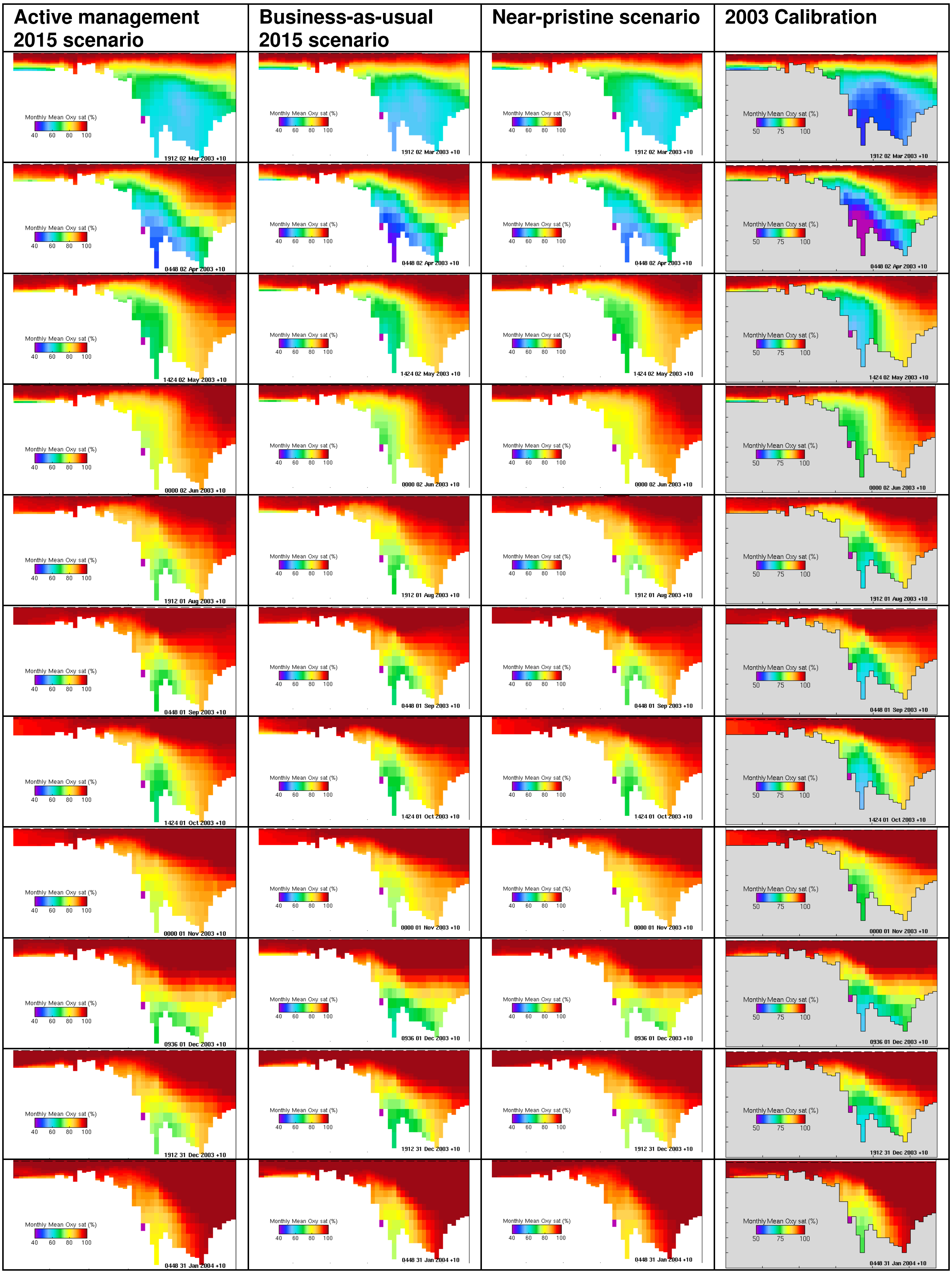


Figure 4.21 Cross sections of monthly mean dissolved oxygen saturation along the axis of the estuary from New Norfolk to Iron Pot from Feb '03 - Jan '04 for the three model scenarios and the 2003 simulation.

Scenario Comparisons Bottom Water DO Saturation

Figures quantifying the spatial and temporal differences between simulations are presented as spatial plots of number of days that bottom water oxygen saturation falls below a relative threshold (Figure 4.22). Both 2015 scenarios are compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario is compared with the 2003 simulation to show increases in oxygen levels and quantify anthropogenic impact.

The 2015 business-as-usual scenario had lower bottom water DO saturation (1% less than 2003 values) for around three months of the year in the upper, lower-middle and outer reaches of the estuary (Figure 4.22). In the upper middle reaches, western shoreline and areas of Ralphs Bay bottom water DO saturation was also lower than 2003 values (1% less than 2003 values) for one month of the year.

For the 2015 active management scenario bottom water DO saturation was 1% lower than 2003 values in the upper and middle reaches for one month of the year and 5% lower in the middle reaches for ~2 weeks (Figure 4.22). Bottom waters of the Estuary in the active management scenario are better oxygenated overall than the business-as-usual scenario.

The greatest difference between bottom water DO saturation in the model runs was between the 2003 simulation and the near-pristine scenario (scale difference in Figure 4.22). Near pristine bottom water DO saturation was 1-5% greater than simulated in the 2003 model run in the upper estuary and in the middle reaches between Newtown Bay and past Sullivans Cove (Figure 4.22). For 3 months of the year near pristine bottom water DO saturation in the mid estuary was up to 10% higher than simulated in 2003.

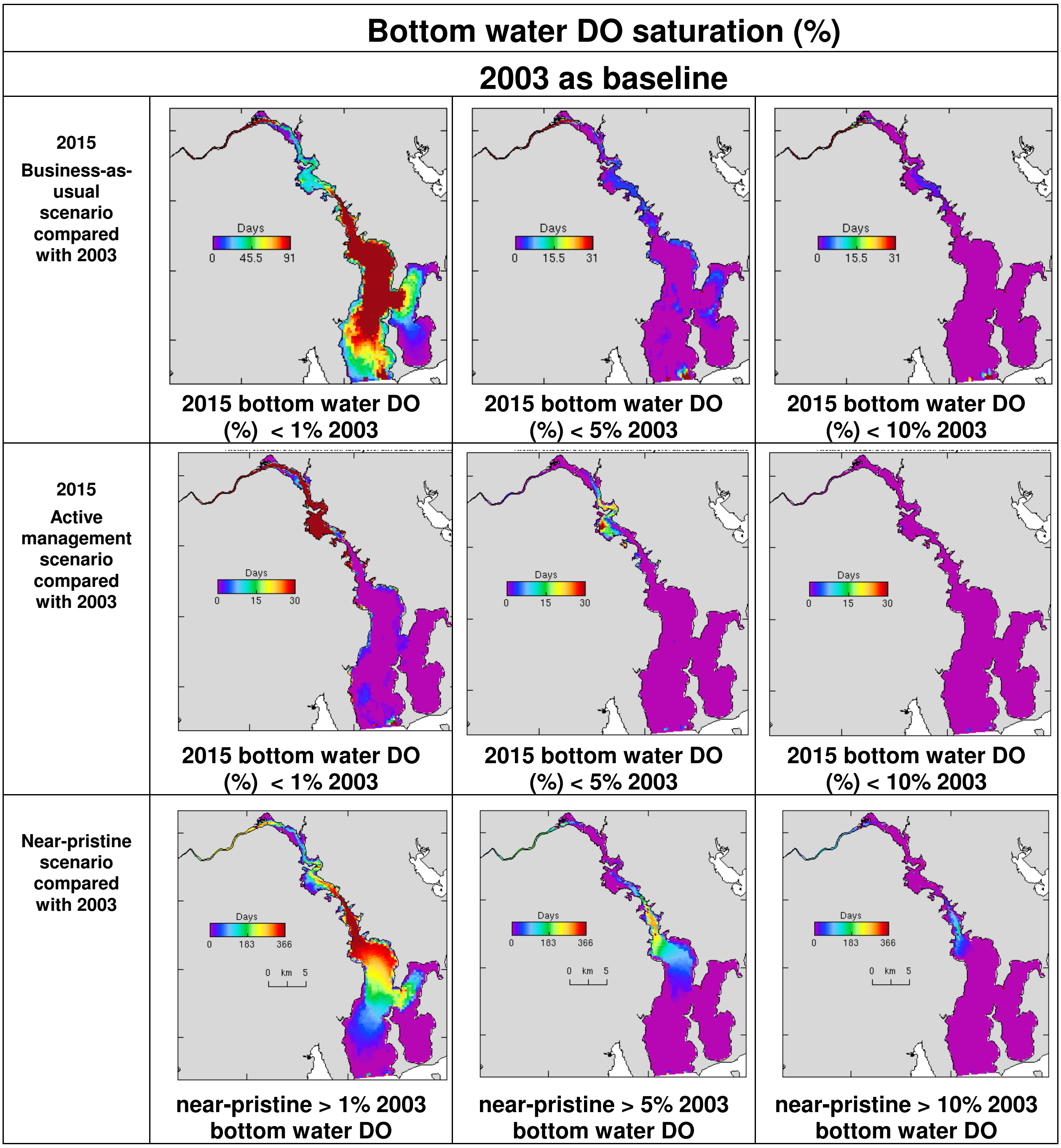


Figure 4.22: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 bottom water dissolved oxygen saturation falls below 2003 threshold. Near-pristine scenario: Number of days in year when near pristine bottom water dissolved oxygen saturation exceeds 2003 thresholds. Note changes in threshold and timescale between plots.

4.3 Benthos

There were no observations of benthic properties in the estuary in 2003. Benthic model (including scenario) results should therefore be treated as a possible hypothesis of conditions in the estuary and interpreted with caution. It should also be noted that the model does not include spatial gradients in infauna bio-irrigation or bio-turbation which are known to have considerable influence on sediment biogeochemical processes. This is recognised as a priority area for future observational studies and model refinement.

4.3.1 Sediment Dissolved Oxygen Saturation

Surface sediment DO saturation was calculated for the top 5 mm of the sediment layer. In deep water sediment DO saturation was often less than bottom water DO saturation where sediment utilisation of DO exceeded exchange rates. Modelled sediment processes that utilise DO include remineralisation of detritus and nitrification; DO can be generated in the sediment by denitrification and in the euphotic layer, by photosynthesis of microphytobenthos.

Seasonal mean surface sediment dissolved oxygen (DO) saturation was highest in the near pristine scenario, followed by the 2015 active management scenario and the 2015 business-as-usual scenario; the 2003 simulation had the lowest surface sediment dissolved oxygen concentration (Figure 4.23). Autumn and spring were the seasons with lowest sediment DO for all simulations, whilst summer was generally most oxygenated. In the 2003 simulation and business-as-usual scenario seasonal mean surface sediment DO saturation was less than 40% in a small area of the middle reaches in autumn and spring. By comparison the active management scenario was ~50% and the near-pristine scenario ~60% for the same period and area.

In the upper reaches, modelled sediment DO saturations were reduced in summer and autumn and throughout the year in a number of relatively coarsely resolved deep holes in the bathymetry above Bridgewater Bridge (Figure 4.24). By comparison of the 2003 and near pristine scenarios, anthropogenic inputs are shown to reduce modelled sediment DO content in the upper estuary all year around. The business-as-usual scenario simulated the lowest sediment DO content likely associated with the reduced Derwent River flow, poorer flushing of the estuary and general upstream excursion of the salt wedge under low flow conditions. The active management also showed a small decline in sediment DO levels compared to 2003 conditions, probably associated with the changes in nutrient load and biogeochemical cycling off Norske Skog associated with their effluent processing plant upgrade. It should be recalled that the model grid in the upper estuary cannot resolve the complex channel bathymetry and resulting hydrodynamics very accurately, so results, should only be treated as indicative of conditions in the upper reaches. Sub-grid scale undulations in the bathymetry which accumulate organic material could experience considerably lower sediment DO conditions than simulated.

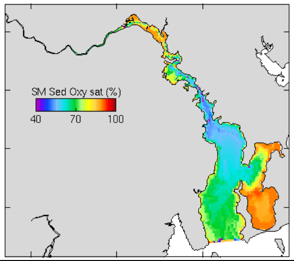
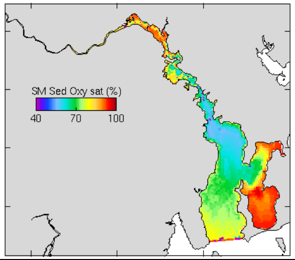
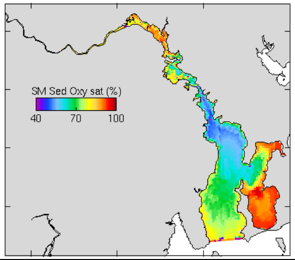
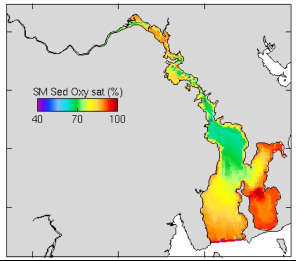
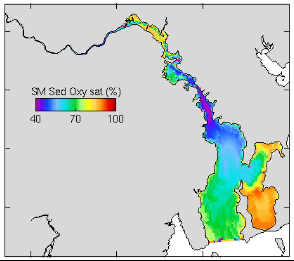
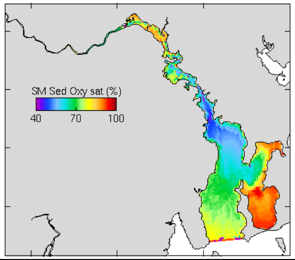
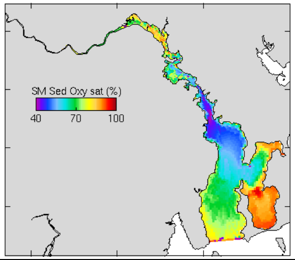
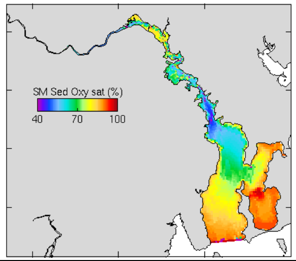
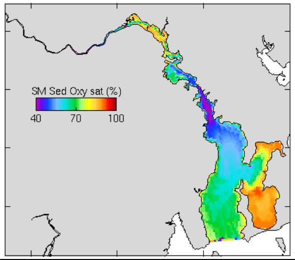
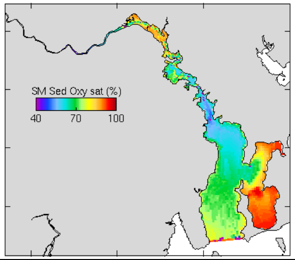
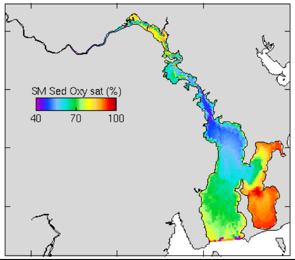
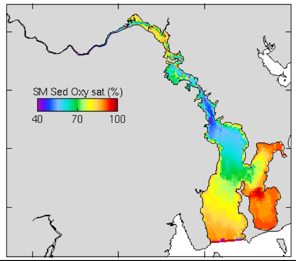
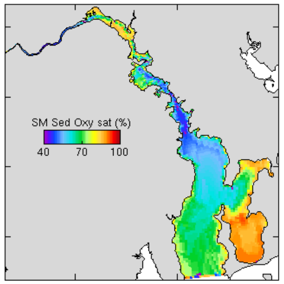
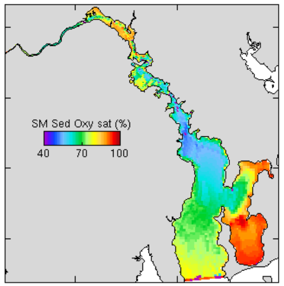
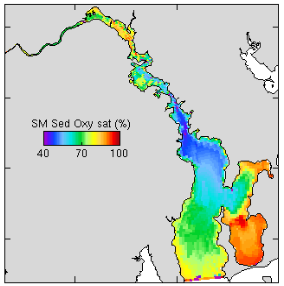
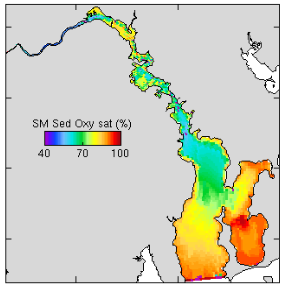
	Autumn	**Winter**	**Spring**	**Summer**
Near-pristine				
2003				
2015 Business-as-usual				
2015 Active management				

Figure 4.23 Seasonal mean surface sediment dissolved oxygen saturation (%) for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

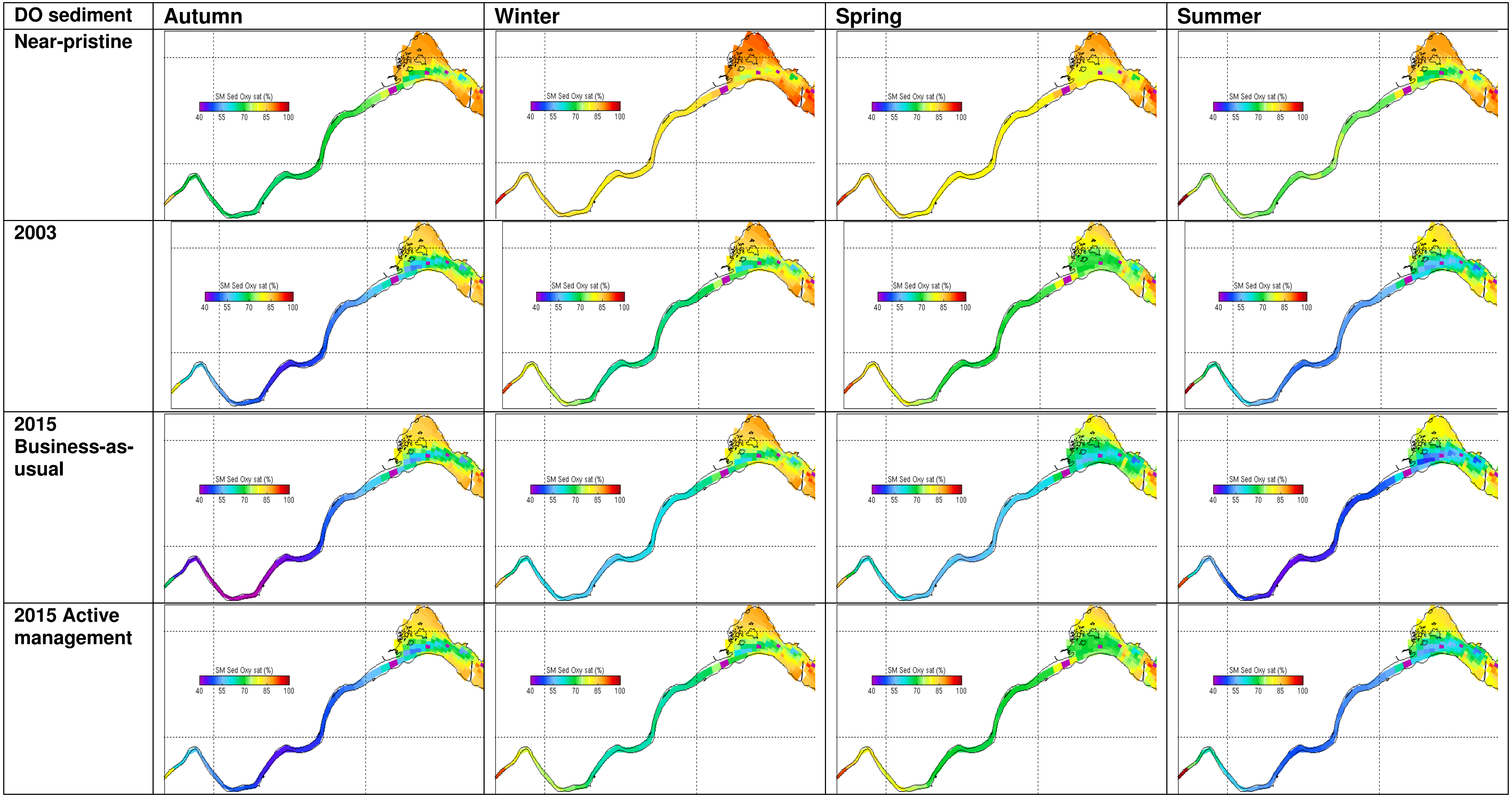


Figure 4.24 Seasonal mean surface sediment dissolved oxygen saturation (%) in the upper estuary for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

Scenario Comparisons Sediment DO Saturation

Figures quantifying the spatial and temporal differences between simulations are presented as spatial plots of number of days that surface sediment dissolved oxygen saturation changed by a relative threshold (Figure 4.25). Both 2015 scenarios are compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario is compared with the 2003 simulation to show differences between the model runs and quantify the anthropogenic impact on surface sediment dissolved oxygen saturation. For pelagic variables (e.g. DIN, DIP) most contrast between simulations occurred in the central middle reaches. For sediment oxygen the model showed high sensitivity in the middle to outer reaches of the estuary where deeper water limited ventilation of the sediment.

Business-as-usual scenario surface sediment DO saturation was 1% less than 2003 values for more than 6 months of the year in all parts of the estuary and 5% less than 2003 values for 3 months of the year in the upper estuary, Elwick Bay and part of the lower reaches (Figure 4.25).

The active management scenario simulated only slight depletion of surface sediment DO saturation in the upper estuary (1% less than 2003 values) whilst large areas of the mid estuary featured elevated sediment DO compared to the 2003 simulation. In the mid estuary active management scenario surface sediment DO increased by 5% of 2003 values in a small area of the middle reaches for the whole year.

Surface sediment DO saturation in the near pristine scenario was more than 1% higher than 2003 levels for most of the estuary and the whole year. In the mid and upper estuary surface sediment oxygen saturation was elevated by more than 10%, relative to 2003, for much of the year (Figure 4.25).

Sediment DO concentration is of interest to managers as at low DO levels heavy metals bound to sediments may alter their redox state and become bio-available for uptake and assimilation into biota (Jo Banks pers.comm.). In the Derwent Estuary there are large areas of sediment with elevated heavy metal content from historical industrial activity. Spatial plots of the number of days of low sediment DO saturation for each simulation are shown in (Figure 4.26).

Periods of low sediment DO occurred in the deeper waters of the mid to lower reaches of the estuary, the upper estuary and isolated deep holes in the bathymetry throughout the estuary. Deeper holes in the model bathymetry were generally poorly flushed and had a tendency to accumulate organic material and drawdown dissolved oxygen via remineralisation and nitrification. In all simulations sediment DO saturations of <10% were simulated for more than 6 months of the year in isolated deep holes throughout the estuary. The net sum of these areas was small (~1%) relative to the whole estuary.

In the near pristine scenario low sediment DO (<40% saturation) occurred for less than a week in the deeper water of the mid to lower reaches. In the 2003 simulation the area of sediment with <40% DO saturation was greater, due to anthropogenic inputs, and included the upper reaches and a larger portion of the mid estuary. In the mid estuary in 2003 simulated bottom water DO was <40% saturation for ~2 months near the Tasman Bridge. Results from the business-as-usual scenario are similar to the 2003 simulation, but under active management the simulated area and duration of low sediment DO saturation in the upper and mid estuary was smaller.

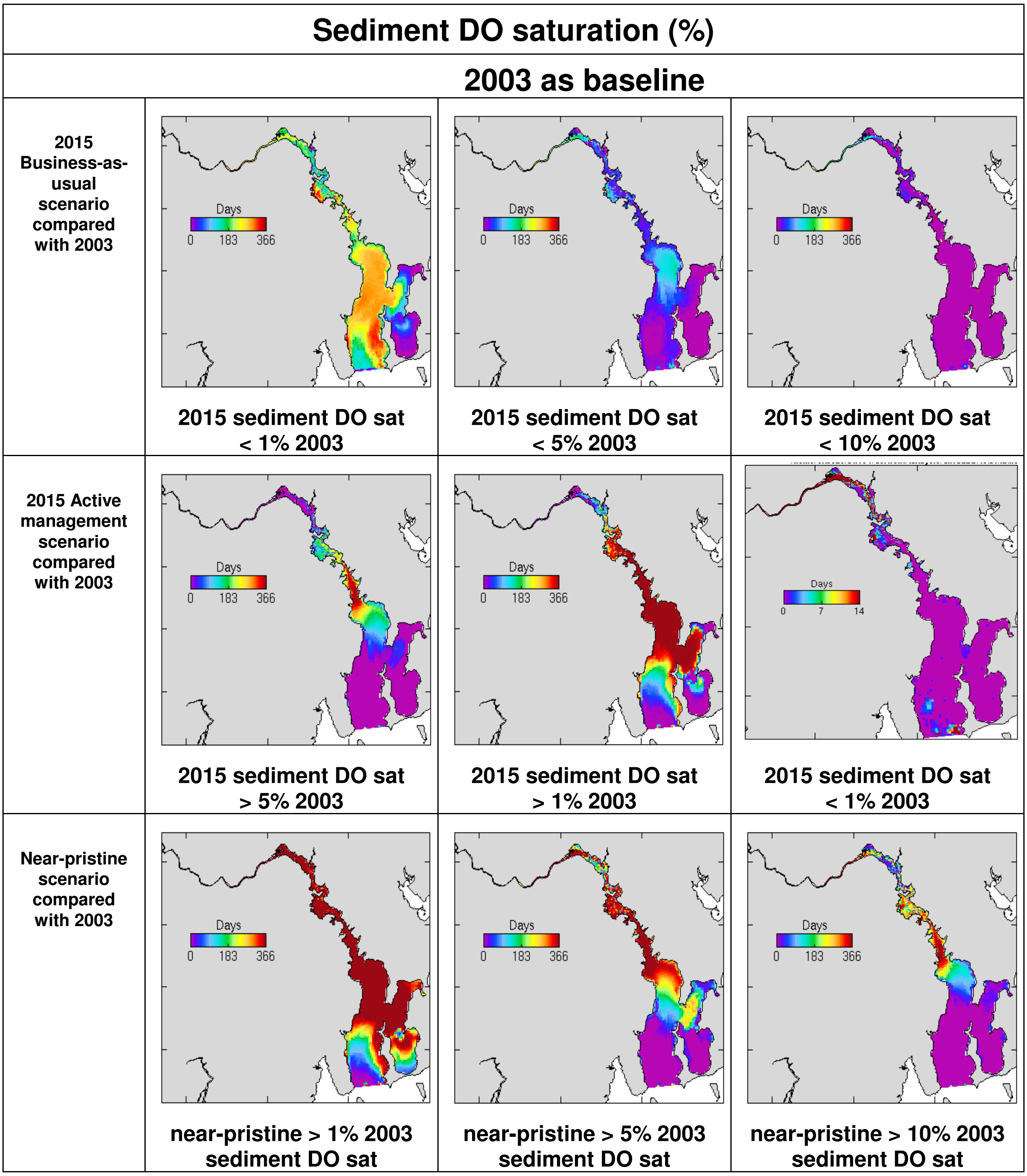


Figure 4.25: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 surface sediment dissolved oxygen saturation fell below 2003 threshold. Near-pristine scenario: Number of days in year when near pristine surface sediment dissolved oxygen saturation exceeded 2003 thresholds. Note changes in thresholds and timescales between plots.

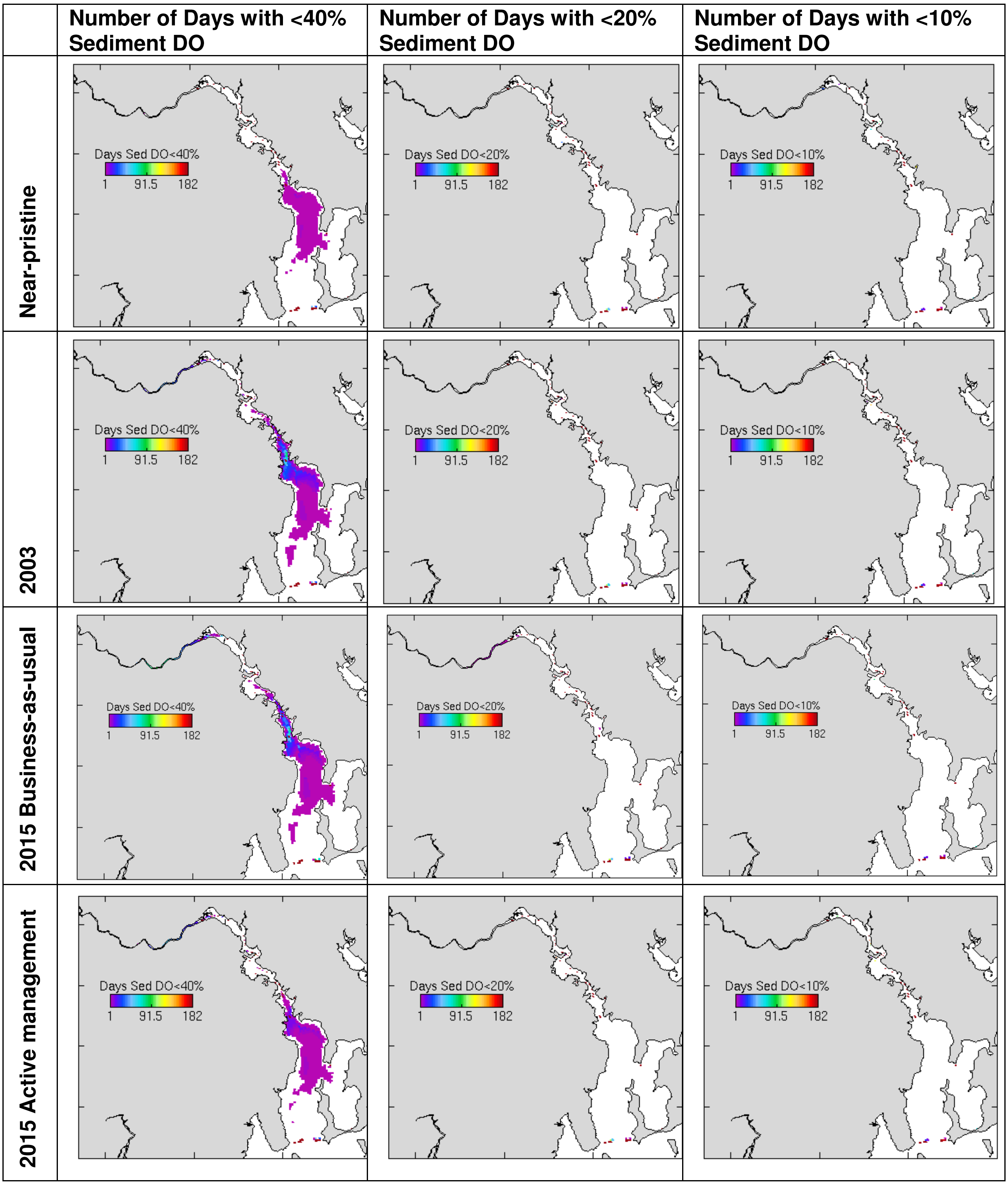


Figure 4.26 Number of days in year when surface sediment dissolved oxygen saturation fell below thresholds of <40% (left), <20% (middle) and <10% (right) saturation.

4.3.2 Light Attenuation and Intensity for Epibenthos

Seasonal mean near surface attenuation coefficient (Figure 4.27 and Figure 4.28) and 24 hour mean light reaching the epibenthos (Figure 4.29) showed small variation between model simulations. The 2003 simulation had the highest attenuation in the upper estuary. In Ralphs Bay a persistent area of elevated attenuation occurred so the south likely associated with resuspension in the shallow water. In spring the business-as-usual scenario had least attenuation in Ralphs Bay possibly due to the increased influence of marine waters relative to the other simulations which included flood events in spring.

In the upper estuary the seasonal contrast in attenuation is primarily due to the influence of coloured dissolved organic matter (CDOM) associated with the timing of the major Derwent river floods. In the business-as-usual scenario a large flood event occurred in winter followed by a relatively dry spring. In the 2003 Derwent flow data, used in all other simulations, major flood events occurred in spring. In other seasons the generally reduced Derwent flow and influx of CDOM in the business-as-usual scenario resulted in less attenuation in the upper reaches of the estuary. This allowed a slight increased in light penetration to the epibenthos in the shallow waters above Bridgewater causeway and in Elwick bay.

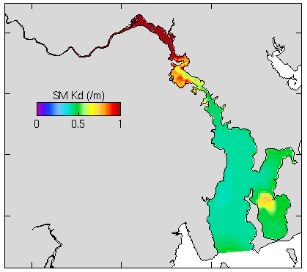
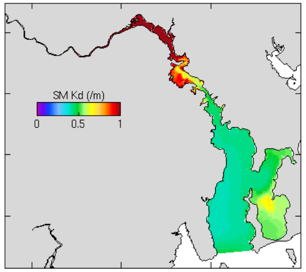
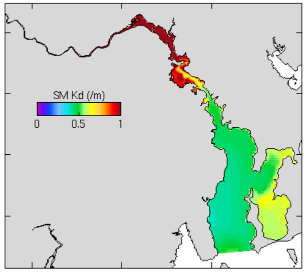
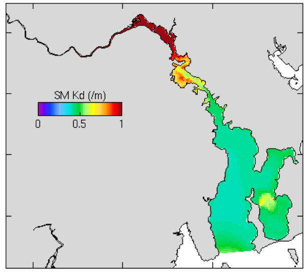
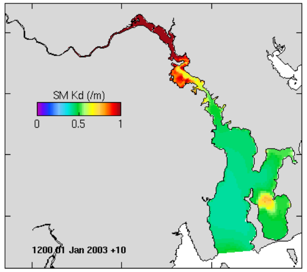
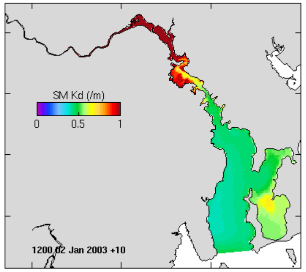
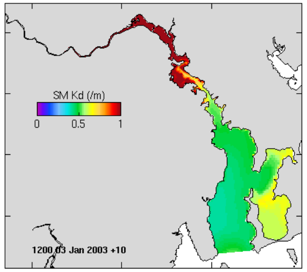
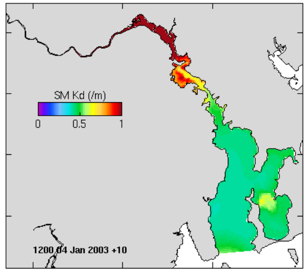
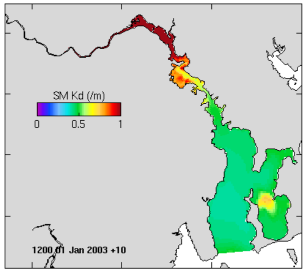
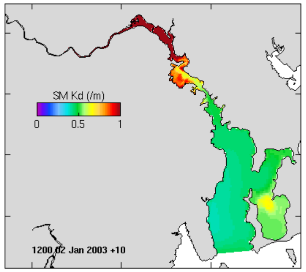
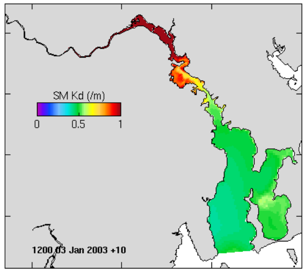
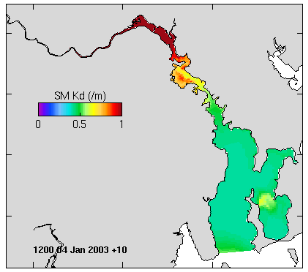
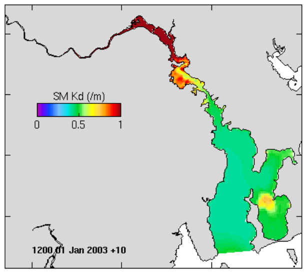
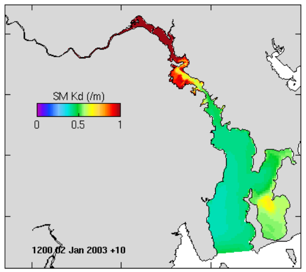
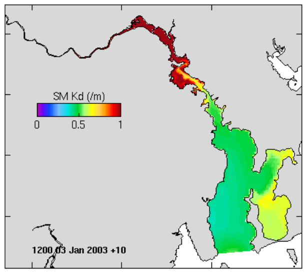
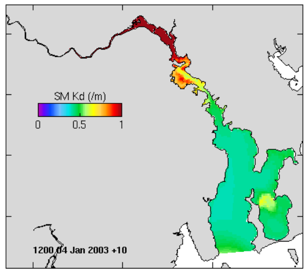
	Autumn	**Winter**	**Spring**	**Summer**
Near-pristine				
2003				
2015 Business-as-usual				
2015 Active management				

Figure 4.27 Seasonal mean near surface attenuation coefficient (0-11m) for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

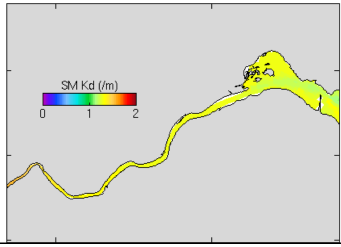
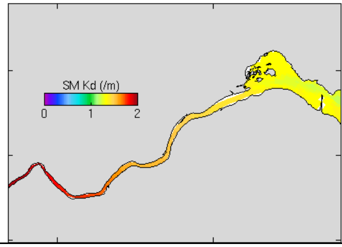
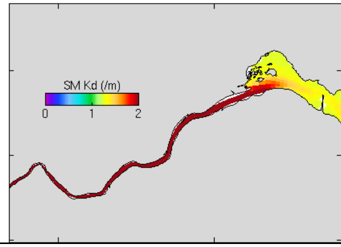
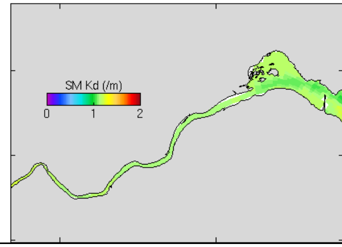
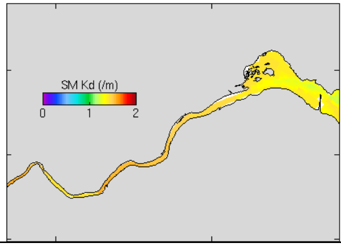
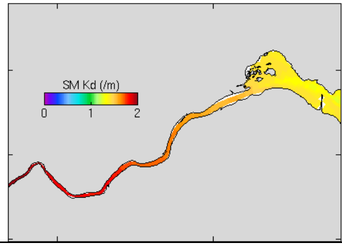
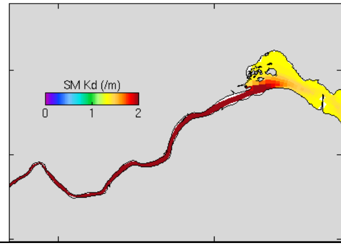
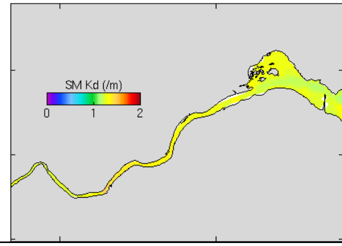
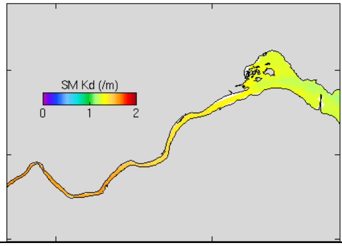
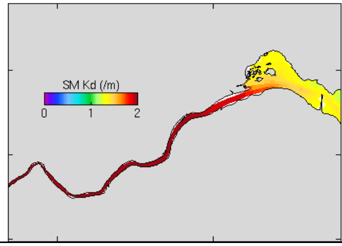
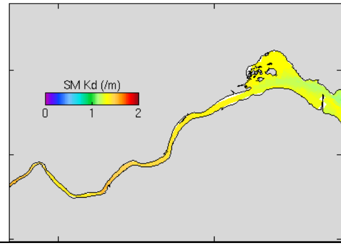
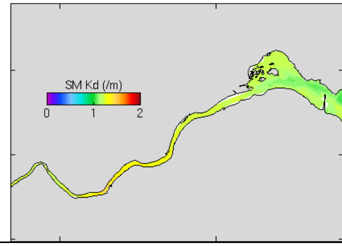
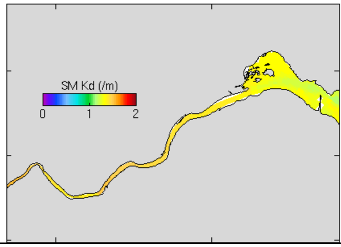
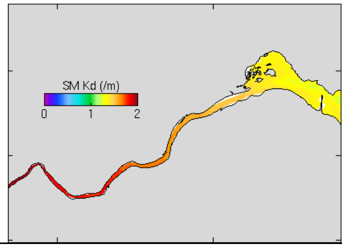
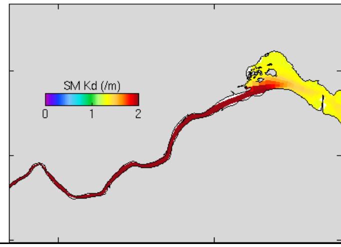
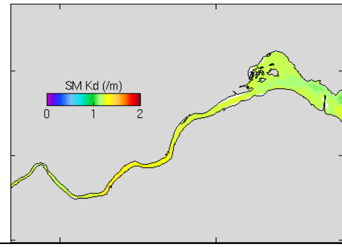
	Autumn	**Winter**	**Spring**	**Summer**
Near-pristine				
2003				
2015 Business-as-usual				
2015 Active management				

Figure 4.28 Seasonal mean near surface attenuation coefficient (0-11m) in the upper estuary for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

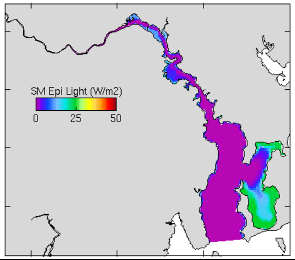
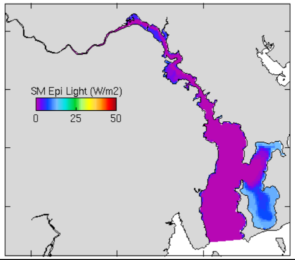
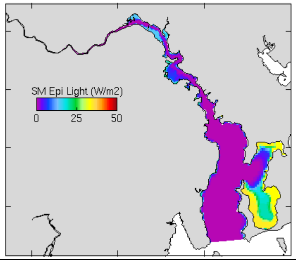
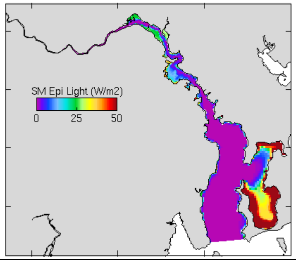
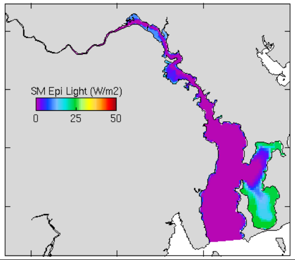
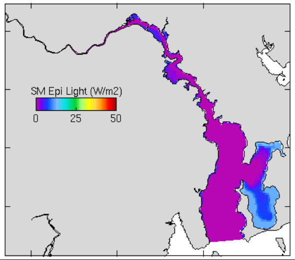
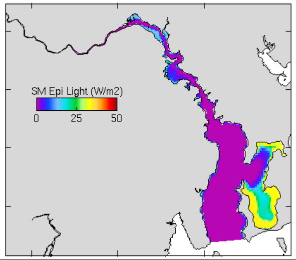
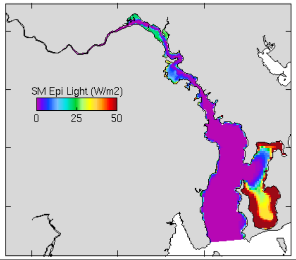
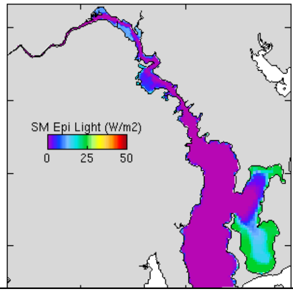
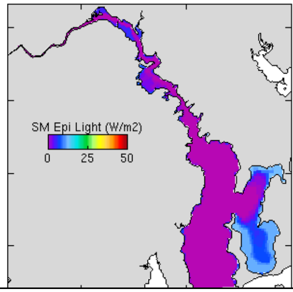
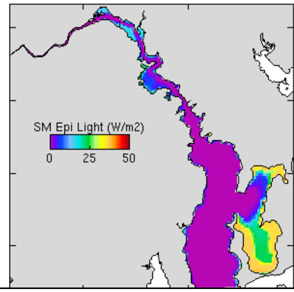
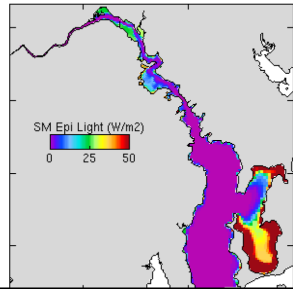
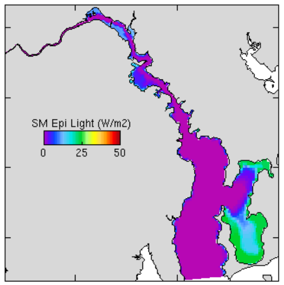
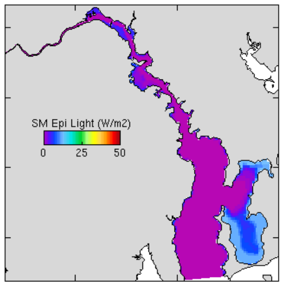
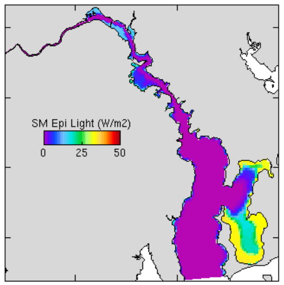
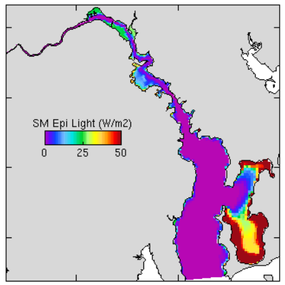
	Autumn	**Winter**	**Spring**	**Summer**
Near-pristine				
2003				
2015 Business-as-usual				
2015 Active management				

Figure 4.29 Seasonal mean 24 hour light reaching Epibenthos for three scenarios and the 2003 Derwent Estuary calibrated model simulation.

Scenario Comparisons Light Attenuation (Kd)

To identify any small spatial and temporal difference between simulations spatial plots of number of days that the near surface attenuation coefficient changed by a relative threshold were prepared (Figure 4.30). Both 2015 scenarios were compared with the 2003 simulation to show the likely evolution of the estuary given contrasting management. The near pristine scenario was compared with the 2003 simulation to show differences between the model runs and quantify the anthropogenic impact on near surface attenuation coefficient.

In the 2015 business-as-usual scenario simulation light attenuation was up to 10% higher than the 2003 calibrated model in the middle reaches for one month of the year, likely due to elevated suspended particulate material associated with the plankton. In the upper reaches of the estuary light attenuation was reduced by 5-10% for 4-6 months of the year compared to the 2003 simulation likely due to a decrease in the influx of CDOM and the cessation of coloured effluent discharge by Norske Skog. Similar reductions in attenuation were simulated in the shallow waters of Ralphs Bay in the business-as-usual simulation. These may result from improved ventilation of the bay with clearer marine waters associated with the reduction in Derwent river flow or from a decrease in particulate point source loads into the bay.

In the 2015 active management scenario there was trivial increase in near surface attenuation throughout the simulation. Conversely attenuation in the mid estuary and upper estuary was reduced by ~5% of 2003 values for up to 6 months of the year (Figure 4.30). This reduction in attenuation is due in part to the reduction in suspended particulate material in the mid estuary and in part to the cessation of coloured effluent discharge by Norske Skog. As Derwent river flows were identical for the active management scenario and 2003 simulation there would have been no reduction in CDOM concentration.

The near-pristine scenario showed reductions in attenuation coefficient of ~10% of 2003 values for more than 4 months of the year in the mid estuary and the upper reaches. Similar to the active management scenario these changes would have been due to reductions in the quantity of optically active substances in the water column including mill effluent.

These contrasting model runs suggest that variation in Derwent river flow has the most impact on light attenuation in the upper estuary with Norske Skog paper mill effluent playing a lesser role. In a low flow year CDOM influx to the estuary is reduced, attenuation is reduced and increased levels of light would penetrate shallow water to the epibenthos.

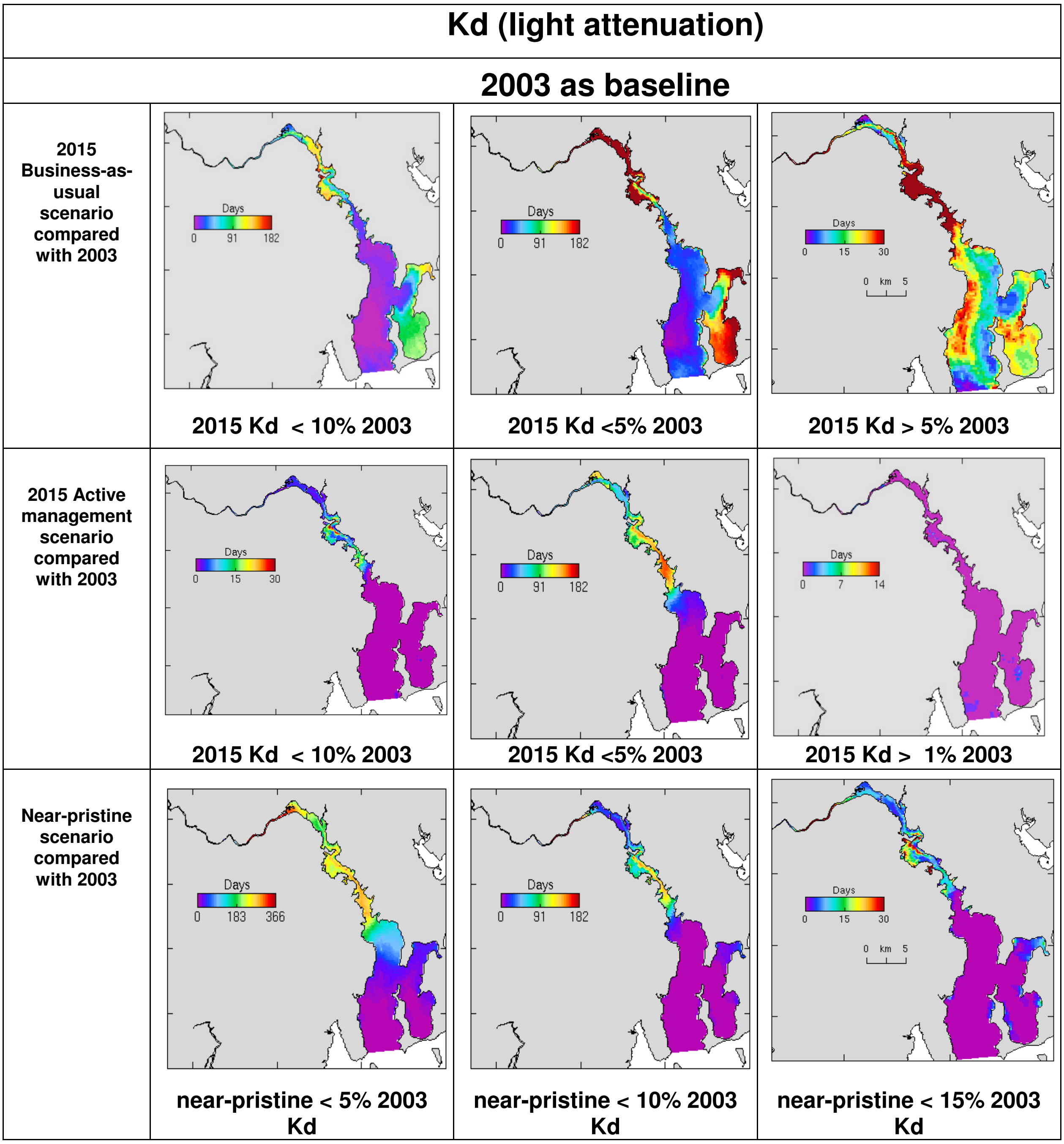


Figure 4.30: Comparison of active management, business-as-usual and near-pristine scenarios with 2003 Derwent Estuary calibrated model simulation. Business-as-usual and active management scenarios: Number of days in year when 2015 near surface (0-11m) attenuation coefficient varied from 2003 levels. Near-pristine scenario: Number of days in year when near pristine near surface (0-11m) attenuation coefficient exceeded 2003 thresholds. Note changes in thresholds and timescales between plots.

4.3.3 Seagrass and Macroalgae

Modelled epiphytic macroalgae take up nutrients from the water column and in the euphotic zone grow in the epibenthic model layer. They may shade modelled seagrass, which also grows in the epibenthos but take up nutrients from the sediment. In the Derwent model macrophytes were initialised with low uniform biomass throughout the model domain and model results shown in Figure 4.31 should be interpreted as areas with potential for seagrass and/or epiphytic macroalgae growth following cumulative growth and/or death of plants over the course of a year (see also Wild-Allen et al., 2009). Modelled macrophyte biomass, whilst consistent with our understanding of plant growth is unvalidated in the model and the model does not resolve gradients in substrate, disturbance or recruitment which can have significant impacts on resulting distributions. Macrophytes have been identified as a priority area for improved observation and model parameterisation.

In all model simulations seagrass biomass was simulated in the shallow waters of Ralphs Bay, above and below the Bridgewater bridge, in Elwick Bay and in small areas of several other bays including Kingston beach. Macroalgae biomass dominated and covered more extensive areas in the upper estuary and Elwick Bay, in the deeper waters of Ralphs Bay and along the coast of the mid and lower estuary where there was access to both pelagic nutrients and light (Figure 4.31).

The model favoured seagrass growth most successfully in the 2015 business-as-usual scenario likely due to the simulated reduction in attenuation in the upper estuary and Ralphs Bay. Similar distributions and magnitudes of biomass also were simulated in 2003 but under active management seagrass biomass was reduced in the shallow waters in the south of Ralphs Bay and even more so in the near pristine scenario despite reductions in attenuation in the upper estuary. These results appear counter-intuitive but likely result from contrasting accumulation of organic material and sediment nutrient concentration between the scenarios. Under pristine conditions sediment nutrient levels are lower potentially restricting seagrass uptake and growth, whilst under greater nutrient loads sediments throughout the estuary accumulate more nutrients that allow elevated seagrass growth rate. The model parameterisation of seagrass growth is derived from literature values and these results could be better constrained by better local observation of seagrass growth dynamics.

Epiphytic macroalgae biomass was high in the two future scenarios and the 2003 simulation, but much lower in the near pristine scenario. This is due to the spatial and temporal reduction in pelagic nutrient concentrations in the near pristine scenario which would limit macroalgae growth particularly in the mid to upper reaches.

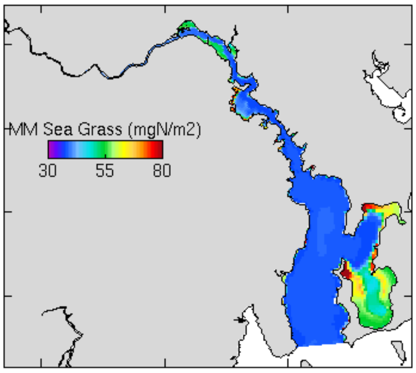
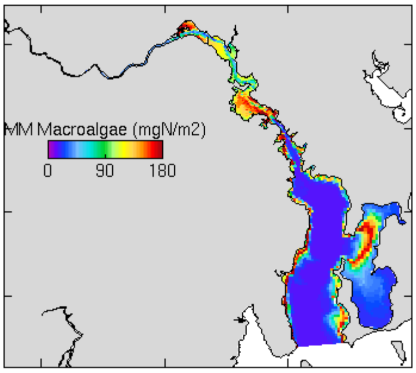
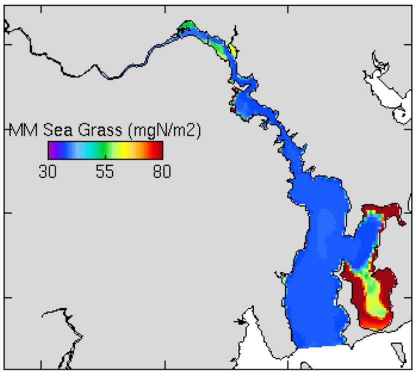
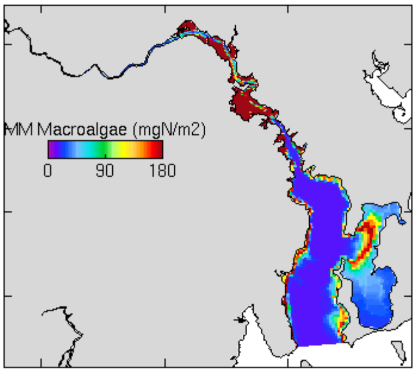
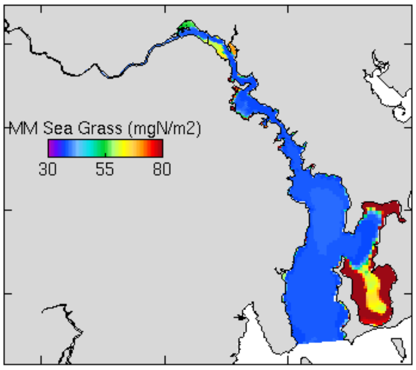
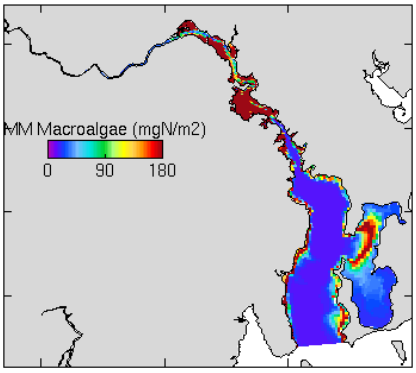
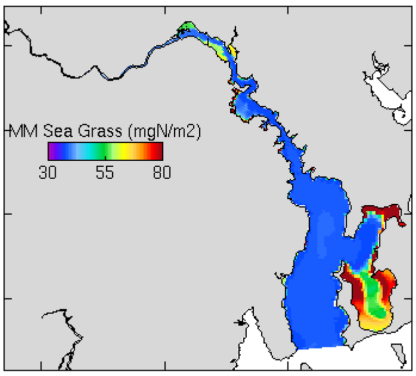
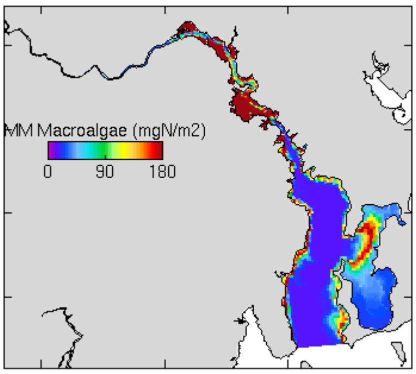
February 2004	**Seagrass**	**Macro-algae**
Near-pristine scenario		
2003		
2015 Business-as-usual scenario		
2015 Active management scenario		

Figure 4.31 Monthly mean modelled seagrass and macroalgae biomass in Feb'04 (following 13 months of simulation) for three model scenarios and the 2003 Derwent Estuary calibrated model simulation

4.3.4 Chlorophyll Classification

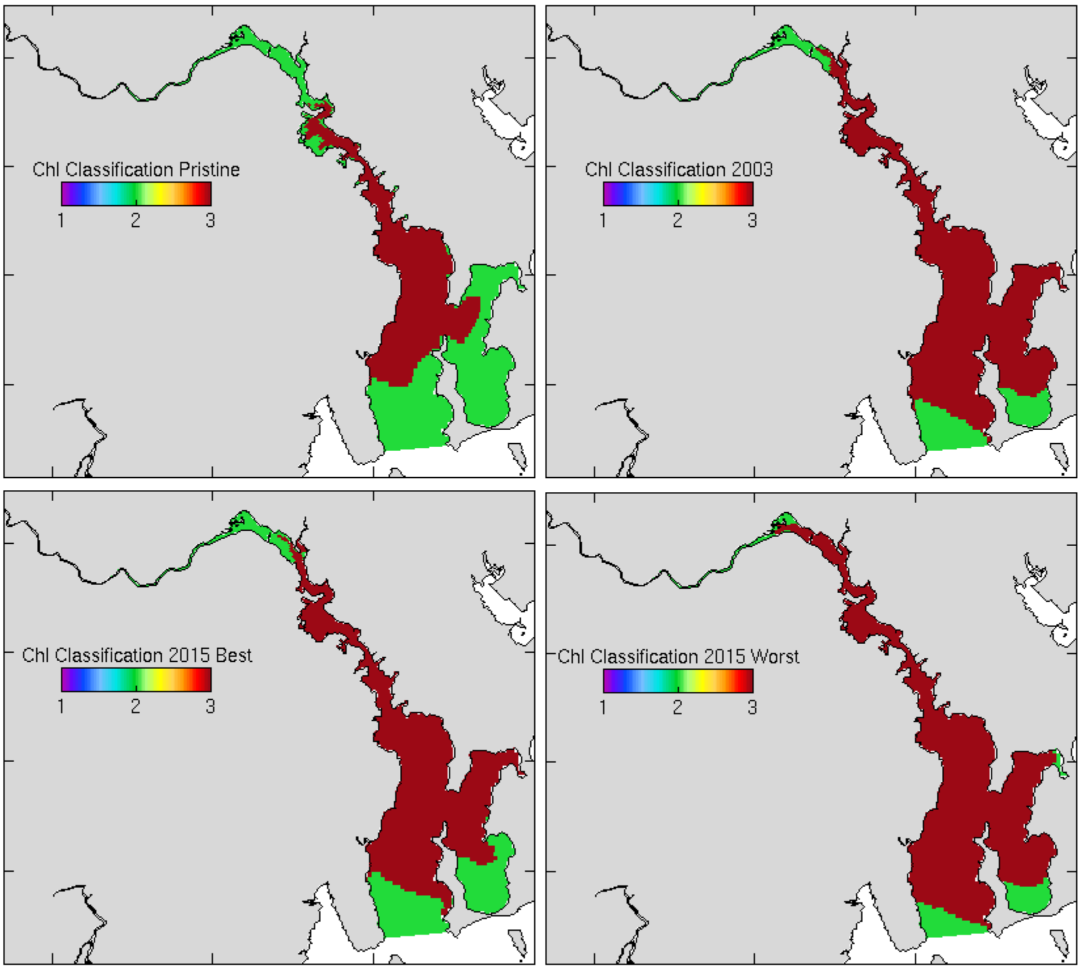
Annual mean near surface (0-11m) chlorophyll concentrations can be used to classify regions into oligotrophic, mesotrophic and eutrophic (Smith et al 1999). The Derwent Estuary was classified into these three categories based on the chlorophyll concentrations from the three scenarios and the 2003 simulation (Figure 4.32). [In the upper reaches in autumn and the mid estuary and inner bays in spring and autumn the 2003 model simulated greater concentrations of chlorophyll than observed and model results for these areas should be treated with caution (see Wild-Allen et al., 2009)].

The modelled estuary was predominantly eutrophic for the 2003 calibration simulation and for both future scenarios. More of the estuary was classed as eutrophic (87%) in the business-as-usual scenario when compared to the other scenarios and the 2003 simulation. The business-as-usual scenario shows an increase in eutrophic classification extending both upstream into the upper estuary and downstream towards the marine boundary. Interestingly, in the business-as-usual scenario the very northern end of Ralphs Bay was classified as mesotrophic. This was likely due to a reduction in catchment and stormwater nutrient loads (and resulting phytoplankton biomass), associated with increased local urbanisation in this scenario [in the MUSIC catchment model natural and forested areas deliver greater loads of nutrient in stormwater compared to urban areas (computed by Jason Whitehead, DEP)].

The active management scenario simulations shows a slight decrease in eutrophication compared with the 2003 simulation with reduction in eutrophic area in the outer reaches of the estuary, in southern Ralphs Bay, and in the upper estuary downstream to the Jordan River.

The near-pristine scenario simulation shows a decrease in eutrophic classification compared with 2003 and other scenarios. The near-pristine scenario simulations show that the estuary contains naturally eutrophic areas in the middle and lower reaches and at the entrance of Ralphs Bay (46%). In this scenario the majority of the estuary was mesotrophic (54%).

Annual mean near surface chlorophyll concentrations used to classify the region are shown in Figure 4.33. From this figure it is clear that in the near pristine scenario much of Ralphs Bay had chlorophyll concentrations approaching 3 mg Chl m^{-3} and was close to eutrophic classification. Similarly in the upper reaches near Bridgewater bridge in the active management scenario and the 2003 simulation concentrations were approaching eutrophic classification. These figures also show that for all simulations the lowest annual mean near surface chlorophyll concentration are found in the upper reaches of the estuary where high attenuation limits light penetration and phytoplankton growth.



	Near pristine	2003 simulation	2015 active management	2015 business-as-usual
Oligotrophic (<1mg Chl /m3)	0.0	0.0	0.0	0.0
Mesotrophic (1-3mg Chl /m3)	54.1	18.3	27.9	12.7
Eutrophic (>3mg Chl /m3)	45.9	81.7	72.1	87.3

Figure 4.32 Regional chlorophyll derived classification for three scenarios and the 2003 Derwent Estuary calibrated model simulation (summarized in table as % area) based on annual mean chlorophyll in near surface (0-11m) layer after Smith (1998). In the figure legend 1 is oligotrophic (purple) 2 is mesotrophic (green) 3 is eutrophic (dark red).

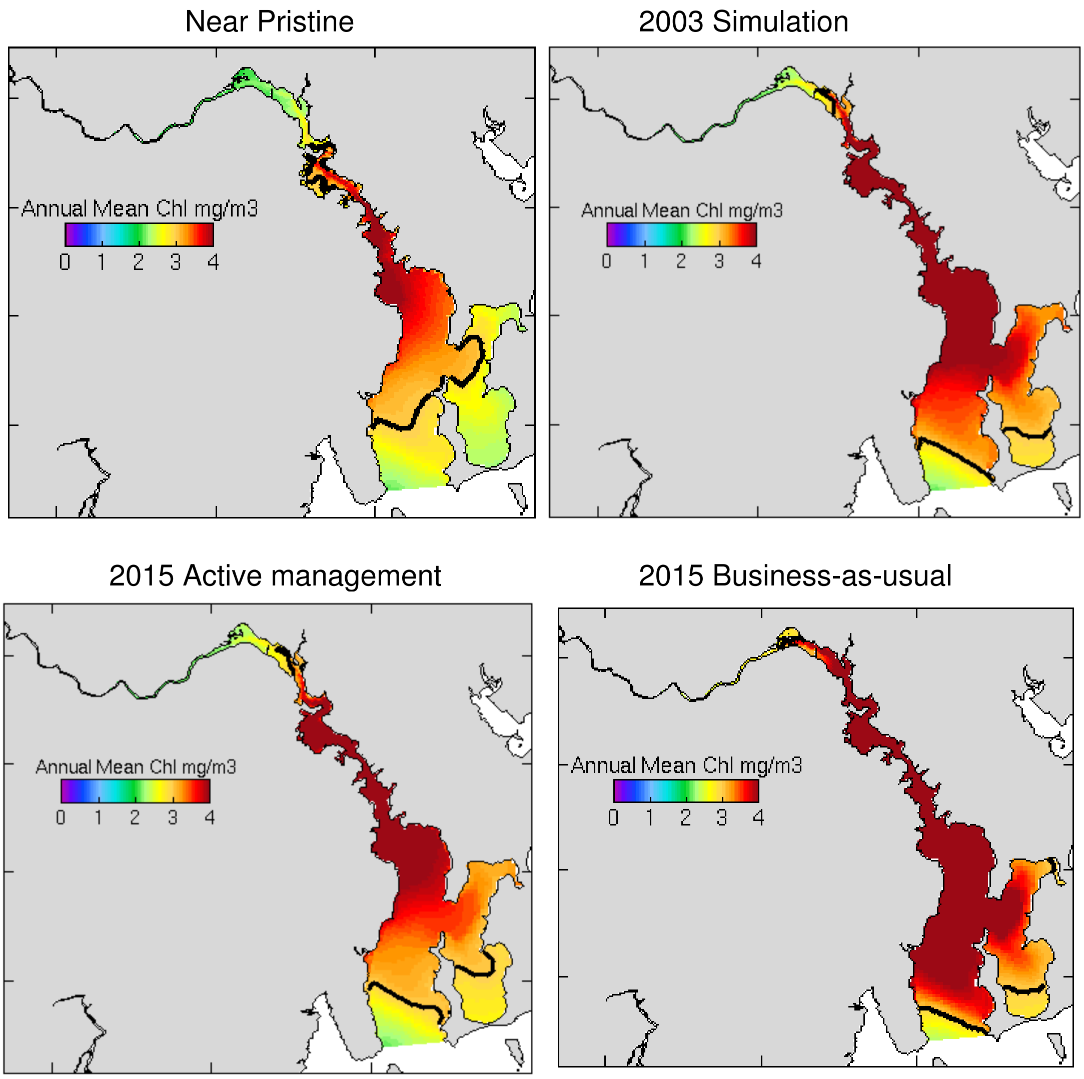


Figure 4.33 Annual mean near surface chlorophyll concentration for three scenarios and the 2003 Derwent Estuary calibrated model simulation [contour line distinguishes mesotrophic and eutrophic classified area (after Smith 1998)].

4.4 Nutrient Budgets

Figure 4.34 shows the amount and relative proportion of nitrogen influx and export from the estuary. In all simulations the greatest influx of nitrogen to the estuary was across the marine boundary, followed by the Derwent River. Point source loads of nitrogen from STPs were high in the 2003 simulation and the business-as-usual scenario; stormwater and industry loads accounted for smaller fractions. The greatest nitrogen export term in all simulations was via denitrification, which exceeded fluxes across the marine boundary. This aspect of the model has not been rigorously validated against observations due to the lack of denitrification observation in 2003, however modelled values are in the range recently observed in 2008 (see Wild-Allen et al., 2009 for further discussion). Improved observation and validation of modelled denitrification is a priority for future work.

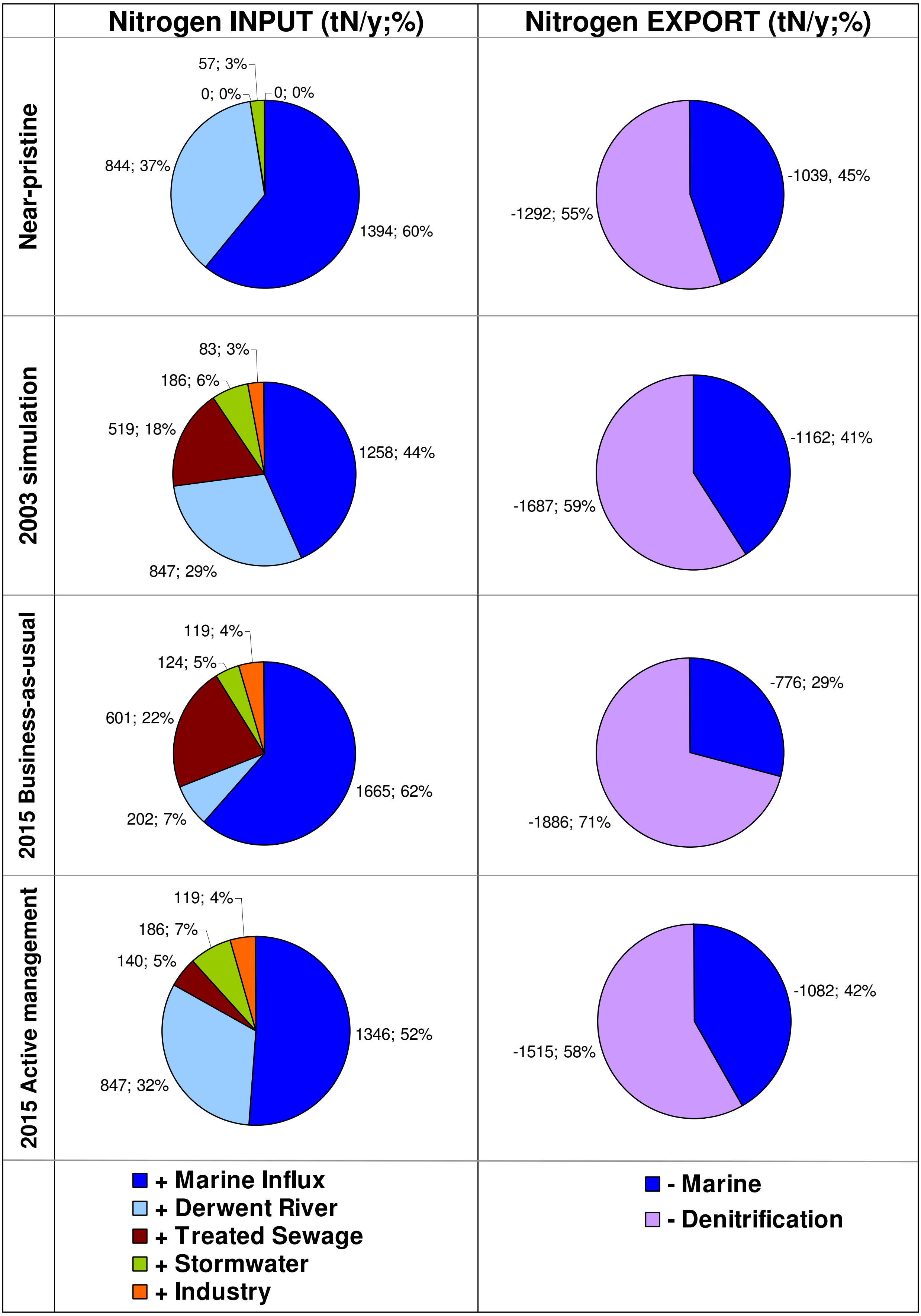
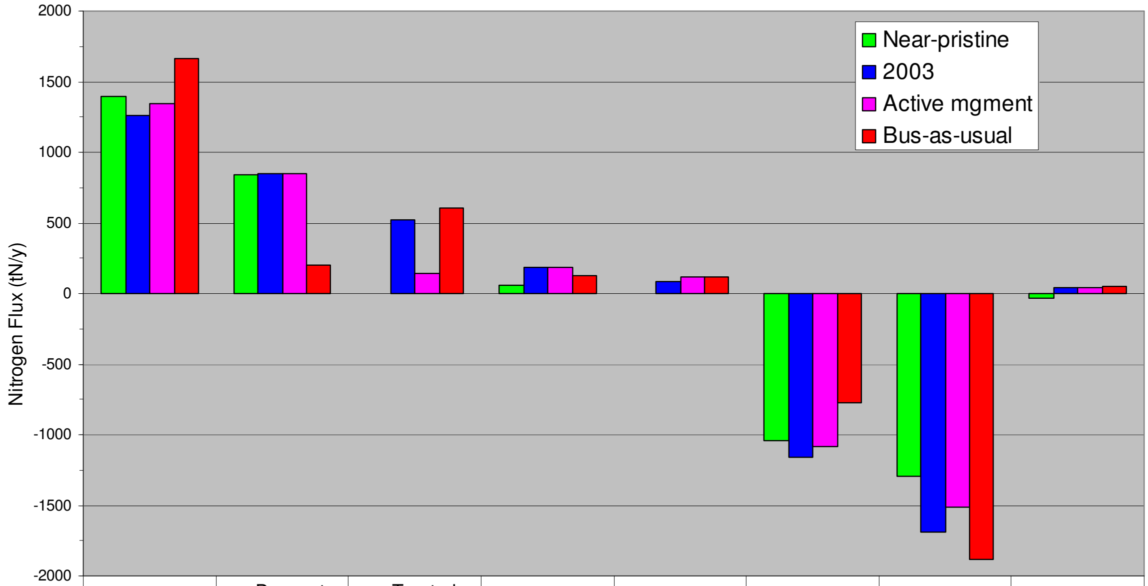


Figure 4.34 Nitrogen input and export (tN/y) for three scenarios and the 2003 Derwent Estuary calibrated model simulation

In the business-as-usual scenario the reduced Derwent River flow resulted in a reduced river load to the estuary, a decrease in export across the marine boundary and an increased marine flux of nitrogen into the model, from a combination of elevated transport and elevated ammonium concentration of inflowing marine water. The net sum of nitrogen input to the estuary was greatest in 2003, slightly less in the business-as-usual scenario (due primarily to the reduced river flow), and lowest in the near-pristine scenario. These simulations confirm the critical role that the Derwent River flow has in regulating water exchange throughout the estuary.

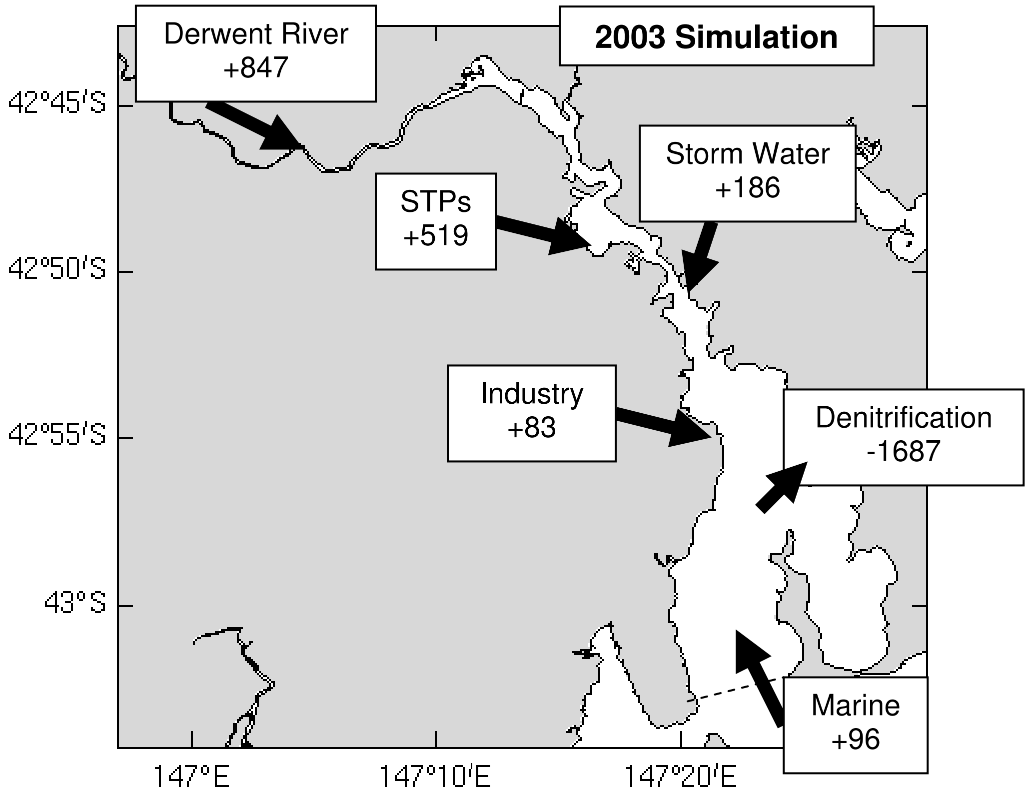
Nutrient budgets of the estuary for the three scenarios and the 2003 calibration show that in comparison to the magnitude of fluxes into and out of the estuary the net accumulation of material within the estuary is negligible (Figure 4.35 and Figure 4.36). However it should be noted that the model has limited capacity to accumulate and/or bury nutrients in the sediment as even the most refractory pools of nutrient are remineralised over a timescale of ~3 months (Wild-Allen et al., 2009).

In the business-as-usual scenario there was a greater nitrogen influx from marine sources due to low river flow and the subsequent intrusion of marine waters, with elevated ammonium concentration, into the estuary. This resulted in elevated influx of nitrogen across the marine boundary compared to the 2003 calibrated model run and the 2015 active management scenario (Figure 4.35). Variation in net nitrogen flux across the marine boundary between the near-pristine, 2003 and active management scenarios is due to variations in nitrogen content of water leaving the estuary integrated over the daily tidal cycle. Accordingly the near-pristine scenario with lowest estuarine nutrient concentrations has slightly elevated influx and reduced outflow compared to the active management scenario and the 2003 simulation (which had slightly less influx and greater loss).

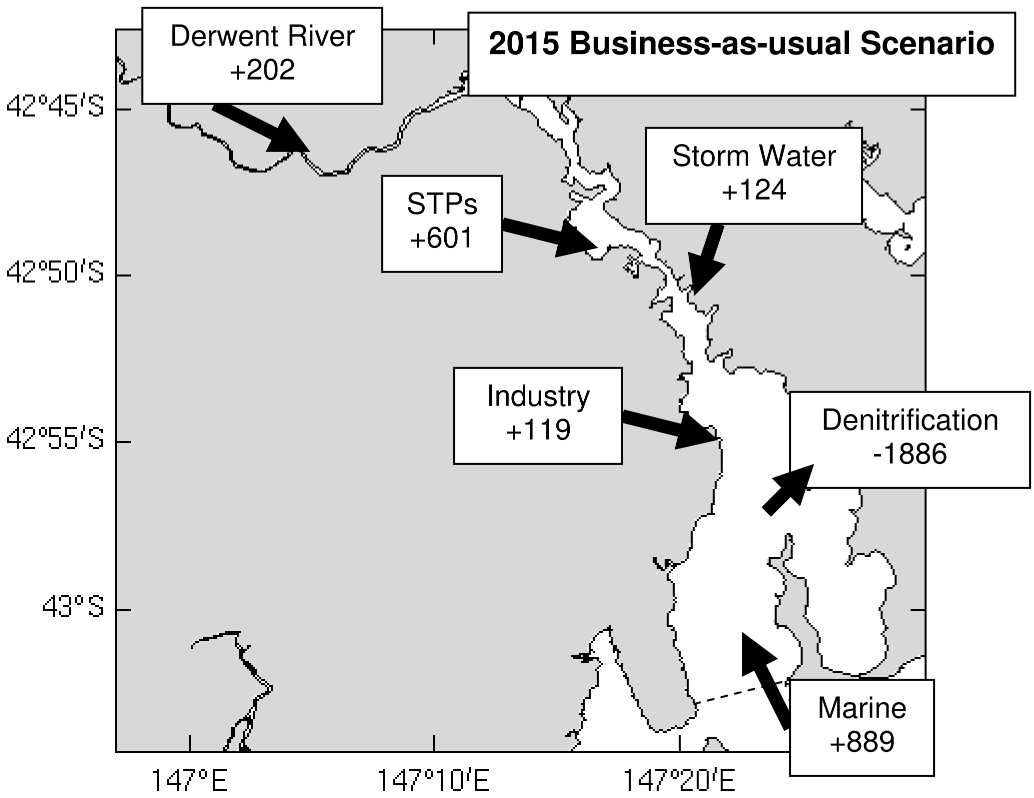


	+ Marine	+ Derwent River	+ Treated Sewage	+ Stormwater	+ Industry	- Marine	- Denitrification	Net Flux
Near-pristine	1394	844	0	57	0	-1039	-1292	-35.4
2003	1258	847	519	186	83	-1162	-1687	44.9
Active mgment	1346	847	140	186	119	-1082	-1515	40.3
Bus-as-usual	1665	202	601	124	119	-776	-1886	48.3

Figure 4.35 Annual nitrogen flux into and out of the estuary, including total denitrification and net flux, for the three model scenarios and the 2003 Derwent Estuary calibrated model simulation


Derwent River
+847
2003 Simulation
Storm Water
+186
STPs
+519
Industry
+83
Denitrification
-1687
Marine
+96
42°45'S
42°50'S
42°55'S
43°S
147°E
147°10'E
147°20'E

.


Derwent River
+202
2015 Business-as-usual Scenario
Storm Water
+124
STPs
+601
Industry
+119
Denitrification
-1886
Marine
+889
42°45'S
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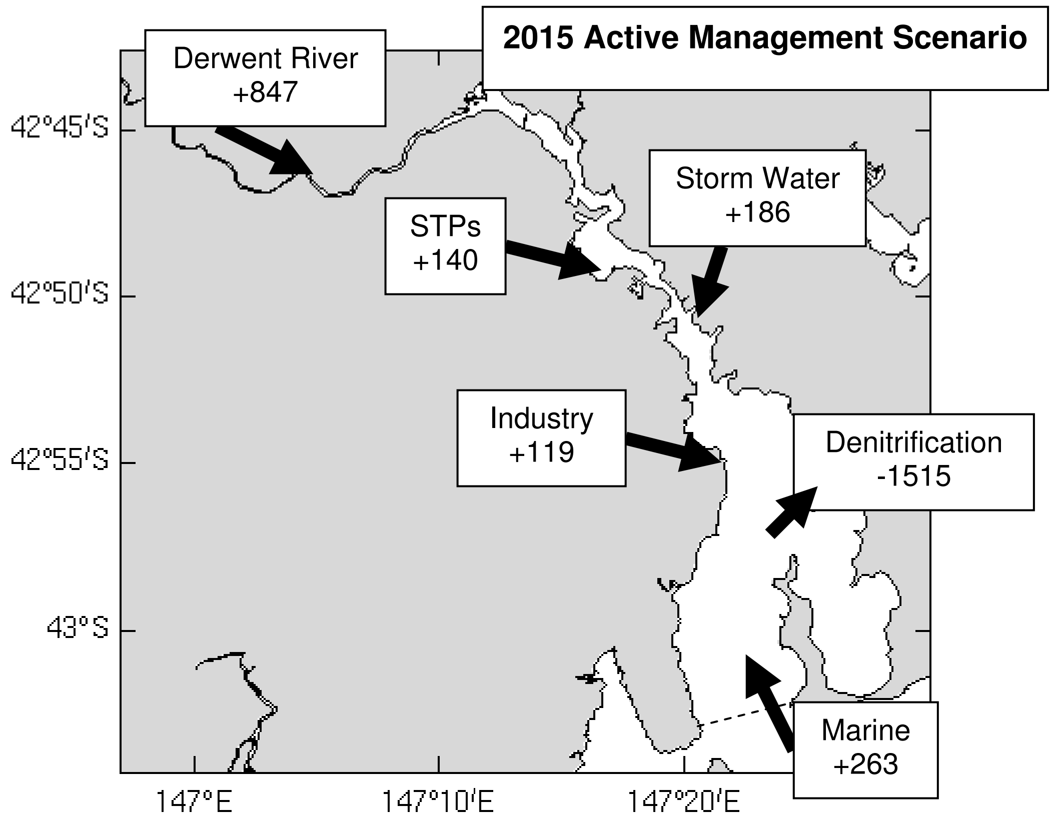
43°S
147°E
147°10'E
147°20'E



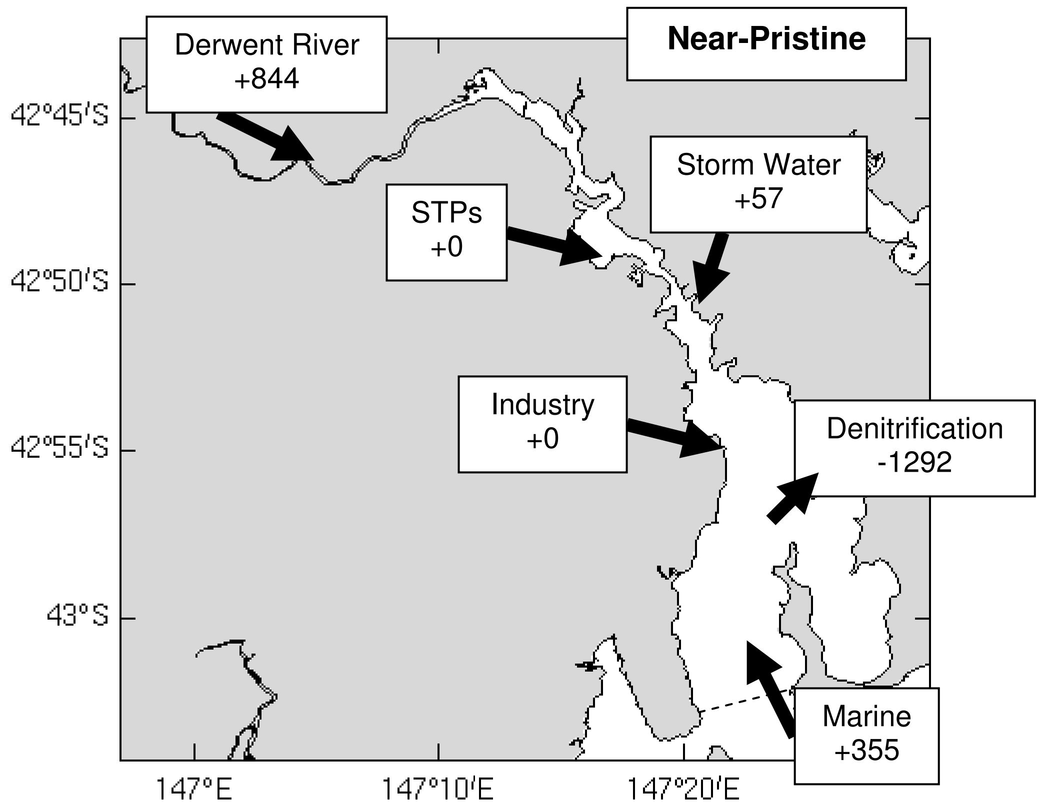


Figure 4.36 Estuarine nutrient budgets for the 2003 Derwent Estuary calibrated model simulation and the three model scenarios.

Sewage treatment plants also contributed a higher nitrogen flux in the business-as-usual scenario, but were much reduced under active management. Stormwater nitrogen was slightly less in the business-as-usual scenario compared with the active management scenario. This occurred because in the MUSIC catchment model loss of agricultural and forestry area and increase in urban area corresponded to a decrease in stormwater nitrogen load (Jason Whitehead, DEP). In both 2015 management scenarios industry nitrogen loads were higher than for the other simulations due to the new effluent treatment plant at the Norske Skog paper mill.

Simulated net annual denitrification flux increased with nitrogen load to the estuary. Accordingly denitrification was lowest for the near pristine scenario and greatest for the 2015 business-as-usual scenario. The 2003 simulation and the active management scenario were intermediate in denitrification flux. Overall the estuary nitrogen net flux was trivial and the model suggests that the nitrogen budget in the estuary is balanced due to effective denitrification of excess loads. This result whilst consistent with observations made by Jeff Ross, et al., (pers. comms.) is not well validated due to sparse observations confounded by small scale spatial and temporal variability in sediment biogeochemistry. The model suggests that denitrification is a critical process controlling the nitrogen balance of the estuary. Should denitrification fluxes be reduced by, for example, a less favourable dissolved oxygen environment then the estuary may begin to accumulate nitrogen.

5. MANAGEMENT IMPLICATIONS

These scenario simulations have explored a range of estuarine nutrient loads and river flow conditions and shown plausible variation in the estuarine biogeochemistry. The strength of these scenario simulations lies in the initial well calibrated model (Wild-Allen et al., 2009). The scenarios were sensitive to changes in nutrient loads from sewage treatment plants, storm water, industry, river flows and marine fluxes and the model integrated the estuarine circulation, light environment, sediment dynamics and biogeochemical cycling to demonstrate likely management impacts on water quality values. Results from the range of scenario simulations illustrate the strong relationship between river flow, nutrient loads and water quality.

5.1 River Flow, Nutrient Loads and Water Quality

In a salt wedge estuary like the Derwent, the river flow sets up an overturning 2-layer circulation, in which marine water enters at the mouth in the bottom layer and is advected upstream, is entrained into the surface layer and then exits with the river water as part of an out-flowing lower salinity surface layer. Changing river flow has two effects on this circulation: the strength of the circulation increases with river flow; and the salt wedge is pushed further downstream. One extreme in this relationship is at zero river flow when there would be no stratification and no salt wedge circulation. In this circumstance the estuary would only be flushed by tidal mixing and in a long estuary with low tidal amplitude, this would lead to very long flushing times in the middle and upper reaches, of order hundreds of days. Conversely, at very high river flow under flood conditions, the salt wedge could be pushed right out of the estuary, which would also remove the salt wedge circulation, leaving the estuary fresh to the mouth. In this case, the flushing time is likely very short, of order hours.

The dominant effect of changes in river flows is on flushing rates, but the interaction of flow with nutrient loads and eutrophication is more complex, and depends on the source of nutrients. Under natural variation river loads are generally proportional to river flows, but if long-term changes to base flows through water extraction allow more intensive agriculture in the catchment, river loads might increase whilst flow declines. At the mouth of the estuary elevated nutrients in deep water can be brought into the estuary by the upstream flow in the salt wedge. Variability in river flow increases the exchange of deep water at the boundary and in winter when nutrients are elevated across the shelf, the inflow of nutrients into the estuary increases. Other sources of nutrients for example from aquaculture in the D'Entrecasteaux Channel might also enter Derwent Estuary depending on the interaction of the Derwent circulation with the circulation in Storm Bay and D'Entrecasteaux. Point source loads from STPs and industry within the estuary are independent of river flow, although stormwater loads are often associated with increased river flow depending on the proximity of local rainfall.

In general, the impacts of a given nutrient load decrease as flushing rates increase. Flushing removes total nutrients from the water body via export, and the amount of nitrogen (as DIN, plant biomass or organic matter) that accumulates in the water body in response to a given load will generally increase as flushing rates decrease. So for point source loads, reduced river flows result in greater water quality impacts including higher chlorophyll and lower bottom water DO. For river loads, where nutrient load and estuary flushing drop together the impact on water quality is less clear but at very low flows, nutrients in the river water will be strongly diluted by seawater in the estuary, so in the absence of other loads, average chlorophyll concentrations in the estuary should be low (although localised impacts in the upper estuary could be high). Marine loads vary seasonally depending on the variability in river flow and the nutrient concentration at the mouth. High natural concentrations of marine nitrate occur in Storm Bay in winter, when the whole system tends to be nutrient replete. In summer and autumn elevated nutrient loads, for example from aquaculture waste, could fuel additional phytoplankton growth in otherwise depleted waters. When other loads are absent, nutrient concentrations in the inflowing salt wedge will tend to establish mean concentrations for the estuary.

At long flushing times, internal sinks can be as or more important than export in determining water quality in the estuary. This study suggests denitrification is an important sink in the Derwent. If the denitrification flux was independent of load and flushing, then it would be more important relatively at low loads and low flushing. If the denitrification flux scaled with biomass in the estuary, it could be considered crudely as another export process equivalent to a background flushing rate. In practice, the denitrification sink is typically low under oligotrophic conditions, maximum under intermediate (mesotrophic) conditions, and declines under eutrophic conditions. Denitrification therefore offsets water quality impacts due to reductions in river flow up to a certain point, but then exacerbates them.

Under stratified conditions environmental impacts can also increase. Stratification allows oxygen drawdown in bottom waters, which tends to reduce denitrification rates and release ammonia and DIP into bottom waters, amplifying into a positive feedback loop. Under the range of base flows considered for the Derwent, the water column is mostly stratified throughout the estuary. If flows were reduced to the point where parts of the lower estuary became well-mixed, this might have localised beneficial effects, but likely very severe effects on the upper estuary. Intermittent high flow events are important at flushing the salt wedge out of the upper estuary, along with isolated pockets of hypoxic water. Once the water column is stratified, the salt wedge circulation is the only mechanism to ventilate the salt wedge with fresh marine water with higher DO content. For given oxygen consumption rates, hypoxia is expected to be worse under low river flow.

The interactions between river flow, nutrient sources and water quality are complex but well simulated by the biogeochemical model. Within this study the business-as-usual scenario shows the net effect of higher marine and point source loads concurrent with low Derwent River flow. Bottom water DO was reduced due to a combination of elevated nutrient load and associated biomass, and inhibition of ventilation by stratification and a longer flushing time.

5.2 Oxygen Drawdown and Nutrient Load

A key management recommendation, based on this modelling study, is the maximum nutrient load which the estuary can absorb, and still meet the environmental water quality criteria. The methodology that DEWHA is using is based on a US-EPA approach, which requires loads to be set as "Total Maximum Daily Loads" or TMDL. In the Derwent Estuary a critical environmental criteria is the drawdown of oxygen, because it has direct ecological effects, is a direct symptom of eutrophication, and results in the transformation of sediment bound heavy metals to bio-available forms. The model results can be used to set a TMDL to achieve given spatial and temporal bottom oxygen targets of e.g. 40% or 20% saturation over % time x % area or percentiles of monthly concentration. In this study, however, only a very limited set of scenario runs were resourced and to choose a load that meets a given target requires interpolation of results between these runs. Interpolation of results is complicated by the variation in river flows and loads between simulations because river flows interact in a different way with each load source (river, point source, marine).

From the scenario analysis of sediment DO saturation a sensitive indicator is the % region with DO less than 40% saturation for 7 days or more, or for 14 days or more (Table 1). Low sediment DO is known to have negative impacts on sediment in-fauna and biodiversity, denitrification flux and sediment bound metal chemistry. Impacts vary with temporal exposure to low DO and are generally more severe under persistent, rather than intermittent exposure. In Table 1, the number of days in the year with low sediment DO is summed; however the days were not necessarily sequential. From spatial plots (Figure 4.23 and Figure 4.26) DO drawdown is shown to occur in the deeper channels of the mid estuary, which are known to contain elevated concentrations of sediment bound heavy metals. Reductions in sediment DO saturation in this part of the estuary could therefore have a significant impact on metal release.

Model Simulation	1 day	7 days	14 days	30 days
Near-Pristine	23	1	1	1
2003	32	10	7	3
Active management	26	5	3	2
Business-as-usual	33	9	6	3

Table 1: Area of Derwent Estuary (%) with sediment DO <40% saturation for 1,7,14 and 30 days.

In Table 1 results show the 2015 business-as-usual scenario has lower areal impacts (6 and 9%) than the 2003 scenario (7 and 10%) which corresponds to the magnitude of total nitrogen input to the estuary (greatest in the 2003 simulation - Figure 4.35). Plotting impacted area against total nitrogen load to the estuary (Figure 5.1) show a provisional exponential relationship between load and modelled area of reduced sediment DO saturation. The impact of reduced flow in the business-as-usual scenario is shown to increase the impacted area in excess of the impact expected for that total nitrogen load. Included in ‘total nitrogen’ are significant quantities of dissolved organic nitrogen which are refractory and very slowly remineralised to bio-available forms. The target load would therefore be better based on labile nitrogen and exclude the refractory DON component.

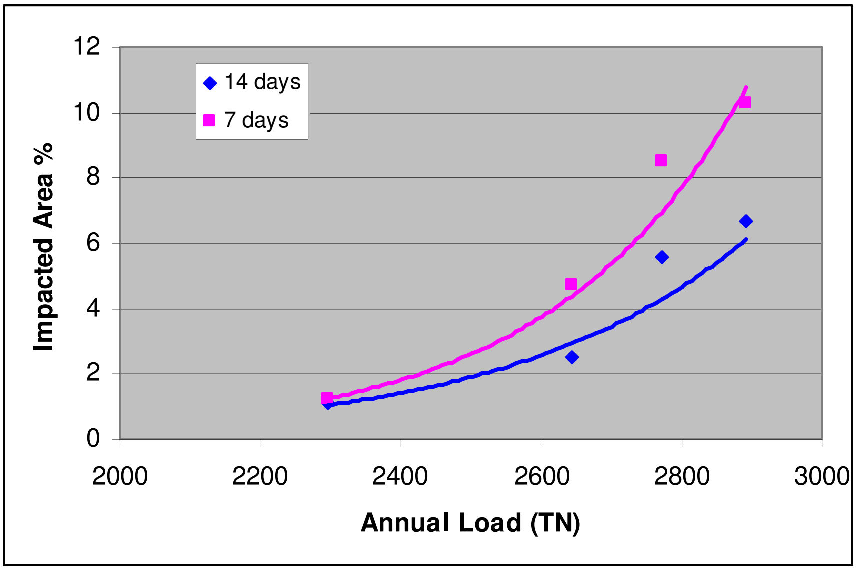


Figure 5.1 Annual total nitrogen input to the estuary and area of estuary with sediment DO saturation less than 40% for 7 and 14 days from the near-pristine, 2003, active management and business-as-usual model simulations.

From the model analysis completed to date it appears that to achieve sediment DO oxygen concentrations in excess of 40% saturation over 95% of the region for 98% of the year then under average flow conditions nutrient loads to the estuary should be reduced to levels proposed in the 2015 active management scenario. Under low flow conditions loads would need to be further reduced to avoid additional spatial and temporal impact. This analysis is based on a very limited set of load and flow scenario simulations and should be interpreted with caution. For improved confidence in this analysis each scenario should be repeated for a range of Derwent River flows.

6. CONCLUSIONS

The modelled annual net nitrogen flux into the estuary was small for all scenarios and the 2003 simulation indicating that for all scenarios nutrient input to the estuary was balanced by near-equivalent export. Overall the estuary water quality was better and the DO saturation higher under the active management scenario than the business-as-usual and 2003 model runs.

The low flow year used in the 2015 business-as-usual scenario resulted in extension of the salt wedge upstream into the estuary due to lower river input. Higher levels of denitrification in the business-as-usual scenario helped to offset the increased nutrient fluxes across the marine boundary and from sewage treatment plant and industry loads. Subsequently the net nitrogen flux was the same for business-as-usual as for the other scenarios. However the business-as-usual scenario simulation suggested that the estuary would have higher proportions and concentrations of nutrients and phytoplankton and lower relative levels of DO in the bottom waters and sediment due primarily to the increase in nutrients from marine sources and sewage treatment plants compared to the 2003 model.

Comparing the near-pristine scenario with the 2003 calibrated model, results suggest that in the absence of anthropogenic loads the estuary would have experienced reduced levels of near surface DIN, DIP and chlorophyll concentrations and elevated bottom water and surface sediment dissolved oxygen saturation. Near pristine nutrient concentrations in the mid estuary were considerably lower than simulated in 2003 which suggests that anthropogenic loads entering the estuary are retained and recycled in the estuarine circulation. The near-pristine simulation shows that with no anthropogenic input, eutrophic areas remained in the estuary under the present river flow management scheme.

This study has shown that a biogeochemical model used to test hypothetical scenarios can be an extremely powerful tool in aiding managers in their decision process as well as looking at underlying environmental impacts that may occur in the estuary. The strength of this study lies in the initial well calibrated model simulation. The model was sensitive to changes in nutrient loads from marine sources, sewage treatment plants, storm water, industry and river flows. Results from the range of scenario simulations illustrate the strong relationship between river flow, nutrient loads and water quality.

Analysis of modelled sediment dissolved oxygen saturation showed spatial and temporal occurrence of low DO (<40% saturation), could be related to total nitrogen load into the estuary, provisionally by an exponential relationship. To achieve sediment DO oxygen concentrations in excess of 40% saturation over 95% of the region for 98% of the year then under average flow conditions nutrient loads to the estuary should be constrained to levels proposed in the 2015 active management scenario. Under low Derwent flow the spatial and temporal occurrence of low sediment dissolved oxygen was greater than expected for equivalent total nitrogen loads and average river flow. Nutrient loads to the estuary would therefore need to be reduced further to avoid additional spatial and temporal occurrence of low sediment DO. This analysis could be improved by excluding the large refractory DON component of total nitrogen and repeating each scenario simulation for a range of river flows.

7. RECOMMENDATIONS FOR FUTURE WORK

This modelling study has shown that water quality in the estuary relates to the complex interaction of nutrient loads with river flow, stratification and flushing time. These relationships have been explored for a very limited set of load and flow scenarios and a priority for future work would be to extend this set of scenarios to include a greater range of flow and load combinations. Of particular interest would be the simulation of the active management and business-as-usual scenarios under contrasting flow regimes.

In this study stormwater loads to the estuary were calculated from rainfall using the MUSIC catchment model (by Jason Whitehead, DEP), and there was considerable uncertainty in their reliability and the model parameterisation of forested and urban catchments. Whilst stormwater loads are understood to be a minor component of nutrient input to the estuary, it would still be good to include some high resolution observations of contrasting catchments and constrain the parameterisations in the model with local data.

Model results suggest denitrification is a key process maintaining the health of the estuary, and this should be confirmed with observations and a full validation of the model algorithms and parameters. Other priority improvements to the model are noted in Wild-Allen et al., 2009. For future studies it would be good to improve the model grid resolution in the upper estuary to better resolve the complex channel bathymetry and deep holes that accumulate organic material. In addition it would be worthwhile to extend the model grid into Storm Bay and the D'Entrecasteaux Channel to accurately quantify the circulation and nutrient fluxes from these regions. Finally the scenario simulations could be usefully extended to multiple years to evaluate cumulative impacts on water quality over larger timescales.

8. REFERENCES

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9. APPENDIX

Appendix 9-1 Observations from B1, B3 and B5 at surface and bottom of nutrients (nitrate ammonia and DIP) in 2008 used to derive the marine boundary condition for the business-as-usual scenario.