The impact of extreme low flows on the water quality of the Lower Murray River and Lakes (South Australia)

Luke M. Mosley[A,*], Benjamin Zammit[A], Emily Leyden[A], Theresa M. Heneker[B], Matthew R. Hipsey[C], Dominic Skinner[D], Kane T. Aldridge[D]

[A] *Environment Protection Authority, GPO Box 2607, Adelaide, SA 5001, Australia*

[B] *Department for Water, GPO Box 2834, Adelaide, SA 5001, Australia*

[C] *School of Earth and Environment, University of Western Australia, 35 Stirling Highway, Crawley WA 6009, Australia*

[D] *School of Earth and Environmental Sciences, University of Adelaide, Adelaide, SA 5005, Australia*

*Corresponding author:

Email: luke.mosley@epa.sa.gov.au

Phone: (61 8) 8463 7808

Fax: (61 8) 8124 4673

Abstract

The impact of extreme low flows on the water quality of the Lower Murray River and Lower Lakes (Alexandrina and Albert) in South Australia was assessed by comparing water quality from five sites during an extreme low flow period (March 2007–November 2009) and a preceding reference period (March 2003–November 2005). Significant increases in salinity, total nitrogen, total phosphorus, chlorophyll *a* and turbidity were observed in the Lower Lakes during the low flow period. Consequently, water quality guidelines for the protection of aquatic ecosystems were greatly exceeded. Principal Component Analysis, empirical and mass balance model calculations suggested these changes could be attributed primarily to the lack of flushing resulting in concentration of dissolved and suspended material in the lakes, and increased sediment resuspension as the lakes became shallower. The river sites also showed significant but more minor salinity increases during the extreme low flow period, but nutrient and turbidity concentrations decreased. The most plausible reasons for these changes were decreased catchment inputs and increased influence of saline groundwater inputs. The results highlight the vulnerability of arid and semi-arid lake systems to reduced flow conditions as a result of climatic changes and/or water management decisions.

Keywords: hydrological drought, biogeochemistry, climate change, salinisation, eutrophication

Introduction

Droughts are predicted to become more frequent and severe in many regions of the world due to climate change (IPCC 2008). Different types of droughts can be classified, typically based on meteorological, hydrological, or agricultural indices (Tallaksen and van Lanen 2004; Tsakiris et al. 2007). Increased hydrological droughts (extreme low flows) are likely in many mid-latitude arid and semi-arid regions , and large environmental, social and economic impacts could result (Meyer et al. 1999, Mishra and Singh 2010).

One recognised effect of extreme low flow conditions is that large changes in water quality can occur. Such changes have been documented in some lake and river systems in the temperate climate zone (Schindler et al. 1996; van Vliet and Zwolsman 2008) but there is a paucity of information available on arid and semi-arid rivers and lakes (Schindler 1997), apart from a few studies in reservoirs (Naselli-Flores 2003; Baldwin et al. 2008). From the studies in temperate zones, complex and variable changes to water quality have been observed during extreme low flow periods. For example, reductions in non-point source pollution, derived from runoff from agricultural catchments, can improve receiving water quality (Caruso 2002), while the lack of in stream dilution from point source pollution can deteriorate it (Chessman and Robinson 1987). Low flows lead to longer water residence times and reduced water volumes, which often results in higher water temperatures (due to reduced thermal capacity and higher sensitivity to atmospheric and other heat inputs); salinity increases (due to increased influence of saline groundwater inputs and evaporation); lowered dissolved oxygen concentrations; and more favourable conditions for algal blooms (Donnelly et al. 1997; Caruso 2002; van Vliet and Zwolsman 2008; van Vliet et al. 2011). Deteriorating water quality during extreme low flow conditions can negatively impact socio-economic outcomes (Tallaksen and van Lanen 2004) and decrease aquatic biodiversity (Caruso 2001; 2002; Bond et al. 2008). Empirical, statistical and mass balance approaches have proved useful in predicting river and lake water quality changes under changing hydrological and catchment loading conditions (Dillion 1975; Chapra 1975; Brett and Benjamin 2008; van Vliet and Zwolsman 2008; Koklu et al. 2010). However the applicability of these types of models to extreme low flow situations and/or arid to semi-arid systems remains unclear.

Given predictions of increased frequency of low flow drought conditions in arid and semi-arid water systems it is essential that the impacts and drivers of water quality under these conditions are better understood. Recently Australia's largest arid to semi-arid river catchment, the Murray-Darling Basin, experienced a severe drought with catchment flows in the Southern part of the basin the lowest in over 100 years of records (Murphy and Timbal 2008; LeBlanc et al. 2009). This outcome was driven by reductions in autumn rainfall (Murphy and Timbal 2008**;** Ummenhofer et al. 2011) and management decisions in a river system with a very high amount of water resource development (CSIRO 2008; Kingsford et al. 2011). Using data from five river and lake sites in the lower reaches of this system, we tested hypotheses that: (1) water quality was significantly altered during the extreme low flow period compared to a hydrological reference period; and (2) the changes were driven primarily by concentration of dissolved and suspended material due to water volume reductions.

Methods

Description of study area

The Lower Murray region study area comprised the lower reaches of the Murray River below Lock 1 and the two lakes, Albert and Alexandrina, collectively known as the Lower Lakes (Figure 1). This area comprises the lowest portion of the Murray-Darling Basin which has a total catchment area of 1,061,469 km^2 (equivalent to 14% of Australia's total area) and is a highly regulated river system with a series of locks, weirs and storages. The Lower Murray River varies from 5–20 m deep with high turbidity, nutrient and algal levels (Mackay et al. 1988; Maier et al. 2001; Mosley and Fleming 2010).The river channel discharges into the large (821.7 km^2 total surface area) and shallow Lower Lakes, which are freshwater, eutrophic, and highly turbid (Geddes 1984; Fluin et al. 2007; Cook et al. 2009). Under sufficient flows, water exits from the Lower Lakes through a series of barrages separating the lakes from the Coorong (a coastal lagoon), out the river mouth and into the coastal ocean (Figure 1). The barrages are gated structures completed in the late 1940s to prevent seawater intrusion into the lakes as water resource development in the Murray-Darling Basin began to exacerbate this effect. The barrage design typically maintains the Lowers Lakes at near full capacity at a water level of approximately +0.75 m AHD (Australian Height Datum).

The Lower Murray River and Lakes region is very important for environmental and socio-economic reasons. The Lower Lakes and Coorong are collectively recognised as one of Australia's most significant ecological assets and have been designated a wetland of international importance under the Ramsar convention (Paton et al. 2009). The region contains major drinking water supply off takes for the city of Adelaide (population 1.2 million) and several regional townships, several large irrigated agricultural areas, a high amount of tourism and recreational activity, and high cultural values (Bell 1998).

Sample sites, sampling and analytical methods

Water quality was measured at five sites (Figure 1); two in the Lower Murray River (Tailem Bend and Murray Bridge), two in Lake Alexandrina (Milang and Goolwa) and one in Lake Albert (Meningie) from 2003–2010. The water quality parameters analysed were salinity/total dissolved solids (TDS), temperature, pH, turbidity, nutrients (total nitrogen, TN; oxidised nitrogen, NO_x; total phosphorus, TP; filterable reactive phosphorus, FRP) and chlorophyll *a*.

Samples were taken approximately monthly by grab sampling at the lake sites and weekly at the river sites located at the drinking water treatment stations. Samples were collected in polyethylene plastic bottles in accordance with standard methods (APHA 2005). Subsamples were filtered through 0.45 µm membrane filters immediately following collection and stored in the dark at 4ºC for determination of dissolved nutrients (NO_x, FRP). All analyses were undertaken at the Australian Water Quality Centre, a National Association of Testing Authorities (NATA) accredited laboratory. TDS was calculated by measuring specific electrical conductivity (EC, µS/cm at 25ºC) using a calibrated conductivity meter and multiplying by a factor of 0.55 (local factor obtained by comparison of EC with TDS data measured via evaporation). Turbidity was measured by a nephelometer and where required (in absence of gravimetric measures) converted to total suspended solids (TSS) using a conversion equation (TSS = turbidity/0.5 + 0.2, r^2 = 0.91) determined over a range of wind speeds (Skinner 2011). Alkalinity was measured by titration with a pH 4.5 end-point and chlorophyll *a* by acetone extraction (APHA 2005). Temperature and pH were measured at the time of sample collection (12pm ± 4 hrs) using calibrated field instruments (YSI multiparameter probes). Nutrients were measured by standard colorimetric methods (APHA 2005) and TN:TP ratios were calculated to indicate which nutrients may be limiting or in oversupply (Redfield 1958; Harris 2001). There was inadequate ammonia data to report but it is notable that where data was available concentrations were very low compared to TN.

All water quality data was stored in, and extracted from, the South Australian Environment Protection Authority's database.

Hydrology and meteorology

Water level, flow and volume data for the River Murray (measured downstream of Lock 1), Lake Alexandrina (average from five stations), and Lake Albert (average from two stations) was provided by the Department for Water (South Australia). Additional calibrated hydrological model outputs ('BIGMOD', MDBC 2002) were also obtained for inflows to, and outflows from, the lakes, and net losses (= evaporation + seepage – rainfall) for the river and lakes. The net loss indicates the amount of inflow required to prevent water levels declining. Lake volume and surface area was determined using ARC GIS 3D analyst^{TM} with the measured water levels and surveyed bathymetry. Local meteorological data (wind speed, air temperature, and rainfall) used to assess relations with water quality was obtained from the Bureau of Meteorology for the river (Murray Bridge station ID 24521) and lake (Hindmarsh Island station ID 23894) sites. Whole-of-basin rainfall was obtained from the Murray-Darling Basin Authority.

Data analysis

Non-parametric summary statistics were used with median values summarising the centre of the dataset and the interquartile range (IQR, 75th percentile minus the 25th percentile) used to represent the data spread. For the Lower Lakes sites there was a large percentage (40–80%) of dissolved nutrient data below the analytical detection limit. This made it impossible to calculate summary statistics by normal methods. In these cases, the summary statistics were calculated by the Maximum Likelihood Estimation technique (Helsel and Hirsch 2002). In this method a log-normal distribution was fit to the data above the detection limit and the fitted distribution was used to extrapolate a collection of values below the detection limit. These extrapolated values were not considered as estimates for specific samples, but only used collectively with the data above the detection limit to estimate summary statistics.

Water quality data from an extreme low flow period (March 2007– November 2009) were compared to those in a preceding reference period (March 2003–November 2005) (Hypothesis 1). The extreme low flow period had monthly average flows less than the 10th percentile (commonly used to define extreme low flow conditions, Tallaksen and van Lanen 2004) of average monthly flows from 1970–2010, whereas the reference period had monthly flows within the interquartile range in this period. There were a few other occurrences of monthly flows less than the 10th percentile in the 40 year period but these were only of very short duration (1–3 months). The long term (1970–2010) average whole-of-basin rainfall of 483 mm compares to 466 mm and 448 mm in the reference and extreme low flow periods respectively. The water quality in the reference period was similar to longer term averages (Mackay et al 1988). Statistical differences in water quality parameters between these two periods were determined using the non-parametric Mann-Whitney U test (Helsel and Hirsch 2002). These tests were performed in the Microsoft Excel™ add-in program XLSTAT™ with statistical significance ascribed with α of 0.05. As this test is based on ranks, any of the dissolved nutrient data below the detection limit was able to be treated as the value of the detection limit. Some of the sites had insufficient datasets to enable statistical comparisons for all parameters.

A combination of mass balance calculations, empirical fits and statistical analyses were used to compare water quality with hydrological factors (Hypothesis 2). The mass of salt (TDS), TN, TP and TSS present in the water column were calculated separately for the River Murray, Lake Alexandrina and Lake Albert using data from Murray Bridge, Milang and Meningie, respectively. The mass was calculated by dividing the measured concentration by the river or lake volume (V_{river} or V_{lake} respectively) at the time of sampling. If sources are in balance with the sinks, the mass remains at a constant level and the system is said to be at steady state or dynamic equilibrium (Chapra 1997). A mass balance model (Chapra 1975; 1997) for Lake Alexandrina was also used to determine if mass changes over time ($d\mathrm{M}/\mathrm{d}t$) could be explained by:

$$d\mathrm{M}/\mathrm{d}t = Q_{in(t)}c_{in(t)} - Q_{out(t)}c_{out(t)} - v_s A_s m \qquad (1)$$

where Q_{in} and Q_{out} were the monthly inflow and outflow volumes respectively in a particular time step (t) as provided by the BIGMOD model outputs, c_{in} and c_{out} are the mean monthly concentrations in the inflow and outflow respectively, and the net sedimentation term ($v_s A_s m$, used for TN, TP, and TSS but not for TDS) comprises the net settling velocity (v_s, m per day) of this material (used 0.025 m/day as best fit to data), the area of the sediment water interface (A_s, m^2) estimated using ARC GIS 3D analyst™, and the suspended solids concentration in the water column (m, weight per m^3). A second type of model that predicts TP levels well in temperate lake systems (see review by Brett and Benjamin 2008) was also assessed for its applicability to our arid lake system under extreme low flow conditions:

$$TP_{lake} = TP_{in} / (1 + \sigma T_w) \quad (2)$$

where TP_{lake} is the concentration predicted in the lake, TP_{in} is the flow weighted concentration of TP in the river inflow (mg/m^3) calculated using measured TP data at Tailem Bend and modelled flow from BIGMOD, T_w is the mean hydraulic residence time (V_{lake}/Q_{in}), σ is the first-order rate coefficient for TP loss from the lake ($year^{-1}$) where $\sigma = k\ T_w^X$ (where k = 0.47 and x = -0.53, Brett and Benjamin 2008). Equation 2 predicts that lower TP concentrations in the river inflow, and increasing hydraulic residence time, will result in lower lake TP concentrations**.** Fits of the water quality data to river flow and water level using simple regression methods (power curves) were also undertaken as these have proved useful for explaining low flow water quality changes in other locations (van Vliet and Zwolsman 2008).

To assess the potential changes in sediment resuspension with declining water levels, the shear stress at the sediment water interface was estimated using the equations provided by Kang et al. (1982) and Chapra (1997). The shear stress was calculated using data on mean water depth, wind speed and fetch for Lake Alexandrina from the sources noted above. Shear stresses greater than a critical value of 1 $dyne/cm^2$ are considered to appreciably resuspend bottom sediments, while stresses of approximately 1 to 2 $dyne/cm^2$ only resuspend fine particles (Kang et al. 1982).

A Principal Components Analysis (PCA) was performed on all the parameters in the hydrological and water quality dataset except for dissolved nutrients in the lakes (insufficient data above the detection limit). PCA reduces a large number of variables to a few independent, composite variables (principal components) that explain much of the variance of the original data (Puckett and Bricker 1992; Koklu et al. 2010). Variables that are highly correlated with a principal component can indicate underlying processes influencing the data (Vega et al. 1998). The water quality data was normalised using Kendall Tau rank procedure (Ma et al. 2010), the principal axis method was used to extract the components and this was followed by a varimax (orthogonal) rotation. The data was suitable for PCA analysis based on results of tests of the Measure of Sampling Adequacy (> 0.5), and Bartlett's sphericity ($p<0.05$). All data analysis procedures were performed using XLSTAT™. Factors were considered significant and were retained for rotation when eigenvalues were greater than 1 and they occurred prior to the change in slope of the associated scree plots (Vega et al. 1998). The factor loadings were classified as ‘strong’, ‘moderate’ and ‘weak, corresponding to values of >0.75, 0.75–0.50 and 0.50–0.30, respectively (Koklu et al. 2010).

Results

Water quality

There were significant ($p<0.0001$) increases in salinity at all river and lake sites during the extreme low flow period (Table 1 and Figure 2). The salinity increases in the lake sites were very large with median levels three to ten times that of the reference period. The highest increase was at Goolwa where the median salinity of 1195 mg/L during the reference period rose to 12045 mg/L during the extreme low flow period. The river salinities increased by a smaller amount but there was a slightly larger increase at Tailem Bend (closest to the Lower Lakes entrance).

Turbidity levels at both river sites during the extreme low flow period (median of 17 and 21 NTU) were approximately half those found during the reference period (median of 35 and 40 NTU) (Table 1 and Figure 2). In contrast to the river patterns, turbidity levels were significantly higher in the Lower Lakes during the extreme low flow, apart from at Goolwa (Table 1 and Figure 2). The increase was particularly large in Lake Albert, where median turbidity levels at Meningie during the extreme low flow period were over eight times those in the reference period.

The median TN levels at Murray Bridge and Tailem Bend were not significantly different in the extreme low flow and reference periods (Table 1 and Figure 2). In contrast, TN levels were higher ($p<0.0001$) at all the Lower Lakes sites during the extreme low flow period with median levels approximately double that of the reference period. NO_x levels at Murray Bridge and Tailem Bend lowered ($p<0.0001$) during the extreme low flow period to approximately 60% of the level during the reference period. NO_x levels in the lakes were very low compared to the river sites. While apparent increases in NO_x levels were observed during the extreme low flow period at Milang and Goolwa, the median values at all lake sites were very low (at or near the detection limit of 0.005 mg/L). The very low levels of dissolved nitrogen despite high TN levels in the lakes indicate that a large amount of the nitrogen is in organic form.

TP levels at the river sites during the extreme low flow period were approximately 40% lower ($p < 0.0001$) than those found in the reference period (Table 1 and Figure 2). Large and significant ($p<0.0001$) decreases in FRP concentrations also occurred in the river during the extreme low flow period, particularly at Tailem Bend, where the median FRP concentration during the extreme low flow period was less than half that of the reference period level. However, TP were higher ($p<0.0001$) at all sites in the Lower Lakes during the extreme low flow period with median levels nearly double that of the reference period (Table 1 and Figure 2). In the Lower Lakes, FRP levels did not significantly change apart from at Goolwa, where concentrations were higher during the extreme low flow period that the reference period. At the other sites in the Lower Lakes (Milang and Meningie), median FRP values in the lakes during the extreme low flow and reference periods were below the detection limit (0.005 mg/L). On average TN:TP ratios were close to Redfield stoichiometry (16:1) in the river at Murray Bridge although they were significantly higher during the extreme low flow period (Table 1). All lake sites had TN:TP ratios above Redfield stoichiometry during both the extreme low flow and reference periods and the ratio increased during the low flow period at Meningie (Table 1).

Chlorophyll *a* levels in the Lower Lakes increased significantly during the extreme low flow period, apart from at Goolwa (Table 1). Chlorophyll increases were particularly large in Lake Albert where levels at Meningie more than doubled. Insufficient chlorophyll *a* data was available at the river sites (only available at Tailem Bend during the extreme low flow period) to use in our assessment.

pH levels decreased slightly during the extreme low flow period at Murray Bridge, Tailem Bend and Goolwa, but there were no significant changes at Milang and Meningie (Table 1). No significant change in water temperature was found at any site during the extreme low flow period (Table 1).

Median concentrations of many parameters exceeded the ANZECC (2000) water quality guideline values (bold values in Table 1). In some cases the guidelines were exceeded during the reference period but the extreme low period generally resulted in much greater exceedances (e.g. salinity/TDS, TN, TP, chlorophyll *a* at the lake sites; NO_x at the river sites). Median TP at

Tailem Bend during the low flow period fell below the guideline level but salinity slightly exceeded the recommended (taste-based) salinity threshold for drinking water in Australia (ADWG 2004).

Relationship of water quality with hydrology

The median flow over Lock 1 in the extreme low flow period was approximately one third of that in the reference period (Table 2 and Figure 3a). The median water levels in Lakes Alexandrina and Albert during the extreme low flow period were approximately 1 m lower than those during the reference period and represented a 35% and 55% reduction in water volume respectively (Table 2 and Figure 3c). The lowest water levels during the extreme low flow period were reached in April 2009 and represented a 64% and 73% reduction in the volume of Lakes Alexandrina and Albert respectively. These are the lowest water levels in over 90 years of records (see inset to Figure 3b). The low water levels and inflows meant there was no outflow from the lake system during the extreme low flow period (Figure 3a). During this period the lake levels fell below mean sea level (approximately +0.2m AHD) downstream of the barrages, reversing the usual positive hydraulic gradient from the lake to the sea. In contrast to the changes in the lakes, the river level and volume reduction during the low flow period, while still statistically significant, was comparably much less (only 16% reduction in volume, Table 2). Net losses from the Lower Lakes (720 GL/year for both lakes) during the low flow period were greater than the average river inflows (588 GL/year) during the same period. Due to their smaller surface area to volume ratio, the net losses from the river were lower than the lakes but losses reduced proportionally more during the low flow period (Table 2). This is likely because most of the shallow wetlands (high evaporative losses) along the river became disconnected by the time the water level fell below -0.2 m AHD in early 2008. Local annual rainfall was higher during the low flow period (Table 2), highlighting the whole-of-basin drivers of the hydrological drought.

Figure 4 shows the mass of salt, nitrogen, phosphorus and suspended solids in the River Murray and Lower Lakes from 2003–2010. For salinity, the mass showed a large (2-3 times) increase over the extreme low flow period in both lakes but the mass was relatively constant in the river. TN mass was relatively constant in the river and lakes during the extreme low flow period. Some increases in TP and TSS mass were observed in the Lake Alexandrina (Milang) during the extreme low flow period, particularly in early-mid 2008, but mass in Lake Albert (Meningie) was relatively constant. It is noted that some of the shorter term fluctuations in TSS, TP and TN mass are not represented well by the model. Where TP monitoring data was available (2005–2010) for comparison, the mass balance model for Lake Alexandrina predicted the changes in lake TP concentration better than the Brett and Benjamin (2008) model (Figure 5). TP and TSS in the river showed a declining trend during the low flow period.

There was a relatively poor fit (r^2 values < 0.35) of Murray Bridge and Milang water quality data with river flow using power equations (Figures 6a and c respectively). Other types of empirical fits, including up to 3rd order polynomial equations, were trialled and yielded similarly poor results (not shown). Visual examination of the data shows a peak in water quality concentrations in the low-mid flow range which limits a simple empirical approach. For example, conductivity at Murray Bridge peaks at about 1500 ML/day flow and turbidity, TN, and TP concentrations peak at about 4000 ML/day flow with lower water quality concentrations either side of this flow. The power equation fits of water quality with mean river and lake depth were more successful, particularly in the Lower Lakes where they explained up to 90% of the data variance (Figures 6b and d), indicating that water level (and hence volume) changes are better predictors of water quality change than flow.

The Principal Component Analysis results also supported hydrological influences as a key driver of water quality change (Table 3). River/lake volume, surface area, and mean depth were found to load strongly on the first component while inflow had a moderate loading. There was also a moderate to strong inverse relationship with salinity/conductivity on the first component, apart from Tailem Bend where the relationship was weak. This first component was subsequently labelled the “hydrological” component and accounted for 35–50% of the total variance in the entire water quality dataset. TP, TN, chlorophyll *a* and turbidity loaded moderately to strongly on the second component. Dissolved nutrients (FRP, NO_x) also loaded moderately on the second component for the river sites, as did temperature at Milang and Goolwa. The second component was labelled the “nutrient and suspended solids” component. Together with the hydrological component, the nutrient and sediment component accounted for 55–64% of the data variance.

The potential for increased resuspension of the lake bed sediments as the lake water level declined was assessed as an additional potential influencing factors on the “nutrient and sediment” component. The critical shear stress for appreciable fine sediment resuspension was calculated to be exceeded on the median wind speed once water depth decreases below 2 m (Figure 7). For Lake Alexandrina, this mean lake depth occurred in approximately February 2007 at a water level of approximately 0.1 m AHD. For Lake Albert, this mean lake depth was less than 2 m for the duration of the study period, but became less than 1 m from December 2007. Hence based on the shear stress calculations it is plausible that there was an increased background (on median wind speeds) level of sediment resuspension during the low flow/water level period that did not occur during the higher water levels in the reference period. The 75th percentile wind speed exceeded the estimated critical shear stress at all water depths (Figure 7).

Discussion

Lake water quality

As hypothesised the lake water quality changed significantly during the extreme low flow period with large salinity, TN, TP, turbidity and chlorophyll *a* increases. The water quality changes in the Lower Lakes appear more extreme than many previous studies and exceeded guidelines for protection of aquatic ecosystems (ANZECC 2000; Nielsen et al. 2003). The salinity increases were particularly large and resulted in a large scale loss of aquatic ecosystem diversity and socio-economic impacts (Kingsford et al. 2011). Salinity increases observed at Goolwa were much larger than the other sites due to leakage of seawater back into the lake through the barrages (Fig. 1, Aldridge et al. 2011). This situation was created by the lake level being lower than the sea for substantial periods of time and the regulation structures not being designed to cope with this reverse hydraulic gradient. Water temperature did not increase at any sites during the low flow period (Table 1), unlike findings in other locations (Murdoch et al. 2000, van Vliet et al. 2011). This likely is due to the air temperature not changing significantly in our study area (Table 2) despite a general regional warming across the river basin (Murphy and Timbal 2008).

The water level in the Lower Lakes reduced substantially from 2007-2009 as a consequence of the extremely low river inflows being insufficient to meet the net losses. There was a greater reduction in water volume in the lakes compared to the river due to their much shallower depth and larger surface area exposed to evaporation. As the lake water levels declined there was a correspondingly large increase in salinity (TDS), TN, TP, TSS, and chlorophyll *a*). The lack of outflows during the low flow period meant there was no flushing of mass from the system. The PCA, mass calculations, and empirical equations (with water level as dependent variable) were consistent in supporting hypothesis (2) that the water quality changes were driven primarily by concentration of dissolved and suspended material due to water volume reductions. In contrast to the concentration of TP and TN, dissolved nutrient concentrations remained very low, which is likely due to rapid assimilation by algae (Cook et al. 2009). The high TN:TP ratios throughout the study period in the Lower Lakes also indicated P-limitation as found in many other lakes (Schindler et al. 2008). A lack of flushing has previously been noted to concentrate nutrients, algae and salt in lakes in North America (Dillion 1975; Schindler et al. 1996) but in some cases to decrease turbidity (Schindler 1997).

The concentration of turbidity in the Lower Lakes during the lake volume decline is likely due to the dominance of the colloidal fraction in the Murray system which stays in suspension (Douglas et al. 1993). The low value (0.025m/day) of the net sedimentation parameter used in our mass balance model to fit the observed TN, TP, and TSS data also indicates colloidal sized and/or low density organic matter is present (see Chapra 1997). Our results suggested there was also potential for increased resuspension of this fine material under median wind conditions as the lakes became progressively shallower during the extreme low flow period. Hence it is plausible that part of the observed increase in turbidity as water levels lowered was driven by resuspension, in addition to concentration of mass. In addition, finer clay sediments are predominantly found towards the middle of the lakes (Barnett 1993), so there was likely more mobilisation of finer sediment materials as water levels declined. Further research during water level decline is required to separate out the relative importance and dynamics of these sediment processes. The lower turbidity levels at Goolwa compared with the other sites are likely due to dilution from the seawater intrusion and salt-induced flocculation of clay colloids (Mosley et al. 2003).

The mass balance model, despite its simplicity and limitations (e.g. only considering inflow, outflows and settling as a driver of water quality change, based on data from one site and assuming mass was uniformly distributed across the lake), represented the general TDS, TN, TP and TSS trends well. The advantage of the process-based mass balance model approach is that can be readily applied to pre-impact predictions in other lakes whereas the empirical approaches (e.g. power equations used with moderate success in our study to predict lake water quality with mean depth) require an extreme low flow situation to occur to develop them for a particular water system. However, the advantage of the power equations is that they are easier to apply for management purposes. The TP model that works well in temperate lake systems (Brett and Benjamin 2008, equation 2) did not represent the Lower Lakes trends well (predicted decreasing TP in the lake due to lower TP concentrations in the River Murray and increased hydraulic residence time in Lake Alexandrina). This suggests this type of model is not applicable under extreme low flow conditions, particularly where no outflow occurs for an extended period.

River water quality

The Lower Murray River water quality was much more resilient to the extreme low flow conditions than the Lower Lakes, but significant changes were observed (supporting Hypothesis 1). Only minor salinity increases were observed in the river (110-165 mg/L TDS increase) during the low flow period compared to the Lower Lakes (1,100-11,000 mg/L TDS increase). Nutrient (TP, NO_x, and FRP but not TN) and turbidity levels decreased significantly at the river sites in comparison to the Lower Lakes, where levels increased. Similar patterns of higher salinity, but lower nutrient and sediment levels have been observed during low flow conditions in other rivers (Caruso 2002; Schindler 1997; van Vliet and Zwolsman 2008).

The drivers of water quality in the Lower Murray River under extreme low flow conditions appear somewhat different from those in the Lower Lakes. Unlike the Lower Lakes the volume changes in the river were relatively minor and flushing/river flow continued to occur throughout the extreme low flow period that was sufficient to export salt and other constituents through to the Lower Lakes. The low river flows were a consequence of decreased Basin catchment runoff, which is also the main source of nutrients and sediment to the Murray River (Mackay et al. 1988). Our empirical plots of water quality versus river flow revealed salinities, turbidity and nutrient levels peaked at intermediate flows with low concentrations during the extreme low flow period. It is likely that these complex water quality-hydrology patterns relate to changes in catchment export (Costelloe et al. 2005). (1) During low flows (<1000 ML/day) there is likely to be very little catchment runoff and hence salt, sediment and nutrients accumulate on the landscape. However in the case of salinity, there was likely reduced dilution of saline groundwater inputs (Caruso 2002) and much longer water residence times that would have promoted salt accumulation. The larger salinity increase observed at Tailem Bend (that exceeded drinking water guidelines) in our low flow period is likely due to more saline lake water being periodically wind-driven up the river (SA Water, unpublished data). (2) Moderate flows (2000-5000 ML/day) appear to mobilise large amounts of salt, turbidity and nutrients off the catchment and floodplain which move down the river. The amount of mobilisation may also be a function of the length of the antecedent dry period where salt and other constituents have built up in the arid landscape. (3) Under high flows (>7000 ML/day), the salt and other constituents are likely to have already been flushed and there is large amount of dilution water.

There are also local sources of agricultural pollution (in particular TP and FRP) that likely decreased over the extreme low flow period due to severe restrictions in water allocations and inability of irrigators to access water and discharge nutrient-laden drainage water back to the river (Mosley and Fleming 2010). Correspondingly, the TN:TP ratios increased in the river during the low flow period and suggested P-limitations on primary productivity as the TN level did not change significantly. TN levels in this reach of the river have been shown to be relatively insensitive to external loadings (Mosley and Fleming 2010) with denitrification increasing with water residence time in this reach of the river (Harris 2001). The very low-flow conditions may also have increased sedimentation of particulate inorganic and organic material in the river channel which may have contributed to the observed reductions in TP and turbidity. Further process-specific research is required during low flow conditions to assess potential changes in internal biogeochemical processes in the river.

Management implications

Regulated rivers have previously been noted to be resilient to drought as their control structures enable less variability in flow and water levels than for unregulated systems (Meyer et al. 1999). While this was true to an extent for the lower Murray River sites, in the Lower Lakes there was insufficient water availability to maintain water levels and flushing, resulting in severe water quality impacts. Further assessment of water quality during future low flow events is recommended in our study area, as well as the time period for recovery from the recent event. Along with many other arid and semi-arid river systems, median river flows in the southern Murray-Darling basin are predicted to decline further (13% decrease by 2030) over the next 20 years due to climate change (CSIRO 2008). Hence extreme low flow periods will likely become more frequent and intense in these vulnerable systems. Careful water resource planning and management will be required in arid systems to prevent water quality deteriorating to the point where socio-economic and environmental values are threatened. Our findings support the recommendations of the recent draft Murray-Darling Basin plan that a substantial increase in environmental water availability is required to maintain system flushing, water levels and quality in the lower reaches of the system (MDBA 2010).

Conclusion

The results of this study demonstrate that extreme low flow conditions significantly impacted water quality at the downstream end of Australia’s largest river system. The impacts were very different in the river compared to the lakes and highlighted there can be dynamic and complex interactions between hydrology, catchment inputs and biogeochemical processes during extreme low flow periods. In the Lower Lakes there was a shift to a much more saline and nutrient and algal-enriched system during the low flow period. These water quality changes were attributed primarily to insufficient river flows resulting in lake volume reductions and concentration of mass (dissolved and suspended) in the system. Increased fine sediment resuspension as the lakes became shallower was also a plausible contributor to the observed increases in turbidity. Simple mass balance calculations and models were useful in predicting the general direction and magnitude of lake water quality changes under low flow conditions. In contrast to the lake results, nutrients and turbidity decreased at the river sites, but there were relatively minor increases in salinity. These changes were attributed primarily to reduced catchment inputs and increased influence of saline groundwater inputs. Droughts and low flow conditions are projected to intensify in the future in southern Australia and other semi-arid regions of the world. The water quality in terminal lakes and storages could be particularly sensitive to declining flows, as exemplified in the present study.

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Table 1 – Summary statistics (number of samples, n; median; interquartile range, IQR) and p-values (for significant differences based on Mann-Whitney U test) for (a) general water quality parameters and (b) nutrients and Chlorophyll *a* between the extreme low flow (March 2007 – November 2009) and preceding reference (March 2003 – November 2005) periods at Murray Bridge, Tailem Bend, Milang, Meningie, and Goolwa. Exceedances of water quality guidelines for 95% ecosystem protection from ANZECC (2000) are shown in bold. Where available, the guideline values used were specific one provided for lowland rivers (Murray Bridge, Tailem Bend sites) and lakes (Milang, Goolwa, Meningie sites) in South-Central Australia. The chlorophyll *a* data was assessed against the hyper-eutrophic guideline value. The guideline value for salinity in the river is for drinking water values from ADWG (2004).

(a)

		River Sites				**Lake Sites**					
Parameter		**Murray Bridge**		**Tailem Bend**		**Milang**		**Goolwa**		**Meningie**	
		Low flow	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**
TDS (mg/L)	n	133	137	132	143	97	31	45	31	67	30
	Median	355	263	**519**	358	**2112**	**711**	**12045**	**1191**	**4048**	**1104**
	IQR	79	46	170	132	1551	69	6082	766	2844	156
	p-value	< 0.0001		< 0.0001		< 0.0001		< 0.0001		< 0.0001	
Water Temp. (°C)	n	142	34	143	81	101	9	41	10	26	10
	Median	17.0	18.0	16.0	18.0	16.0	15.0	17.0	15.5	14.1	16.5
	IQR	8.0	6.8	8.0	7.0	7.0	8.0	7.0	5.8	6.5	4.3
	p-value	NS		NS		NS		NS		NS	
pH	n	144	136	141	143	108	10	44	10	69	10
	Median	7.6	7.4	7.8	7.6	8.5	8.6	8.4	8.6	8.5	8.5
	IQR	0.2	0.2	0.3	0.2	0.3	0.3	0.3	0.3	0.2	0.1
	p-value	< 0.0001		< 0.0001		NS		0.0015		NS	
Turbidity (NTU)	n	143	134	142	143	28	31	25	31	69	30
	Median	21	40	17	35	56	36	8	15	89	11
	IQR	6	20	6	16	88	23	9	7	56	17
	p-value	< 0.0001		< 0.0001		0.002		0.0002		< 0.0001	

(b)

		River Sites				Lake Sites					
Parameter		**Murray Bridge**		**Tailem Bend**		**Milang**		**Goolwa**		**Meningie**	
		Low flow	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**
TN (mg/L)	n	143	136	142	143	29	31	29	31	66	30
	Median	0.63	0.63	0.71	0.74	**2.75**	**1.15**	**1.93**	**1.13**	**3.15**	**1.51**
	IQR	0.25	0.28	0.25	0.32	2.18	0.39	0.98	0.30	1.45	0.57
	p-value	NS		NS		< 0.0001		< 0.0001		< 0.0001	
NO_x (mg/L)	n	143	136	142	143	29	31	29	31	66	30
	Median	0.073	**0.114**	**0.122**	**0.204**	0.005	0.002	0.009	0.001	0.006	0.001
	IQR	0.107	0.072	0.192	0.154	0.006	0.002	0.021	0.001	0.004	0.003
	p-value	< 0.0001		0.0001		< 0.0001		< 0.0001		NS	
TP (mg/L)	n	143	134	142	143	29	10	29	10	66	10
	Median	0.055	0.095	0.058	**0.103**	**0.154**	**0.088**	**0.104**	**0.061**	**0.188**	**0.107**
	IQR	0.015	0.043	0.192	0.055	0.157	0.028	0.095	0.062	0.068	0.046
	p-value	< 0.0001		< 0.0001		0.0079		0.004		0.001	
FRP (mg/L)	n	143	134	142	143	26	31	29	31	60	30
	Median	0.0120	0.0165	0.0130	0.0310	0.0025	0.0007	0.0036	0.0008	0.0047	0.0008
	IQR	0.0090	0.0178	0.0140	0.0285	0.0040	0.0033	0.0051	0.0016	0.0024	0.0025
	p-value	< 0.0001		< 0.0001		NS		0.0089		NS	
TN:TP ratio (molar units)	n	143	134	142	143	26	10	29	10	63	10
	Median	24.4	15.0	27.0	15.9	29.9	27.8	41.2	42.1	39.1	29.7
	IQR	7.8	4.3	9.9	5.7	5.0	9.9	13.4	5.3	44.4	7.7
	p-value	< 0.0001		< 0.0001		NS		NS		0.002	
Chl. a (µg/L)	n	n/a	n/a	142	n/a	12	32	26	32	53	31
	Median	n/a	n/a	12.3	n/a	**34.9**	**25.0**	**28.2**	**25.0**	**63.5**	**31.0**
	IQR	n/a	n/a	7.2	n/a	8.3	10.4	35.6	7.0	44.8	21.4
	p-value	n/a		n/a		0.0236		NS		< 0.0001	

NS = not significant at 5% level (p>0.05), n/a = data not available and/or statistical test not applied

Table 2 – Summary statistics (number of samples, n; median; interquartile range, IQR) and p-values (for significant differences based on Mann-Whitney U test) for hydrological and meteorological parameters in the extreme low flow (March 2007 – November 2009) and preceding reference flow (March 2003 – November 2005) periods at the river (Murray Bridge, Tailem Bend) and lake (Milang, Meningie, and Goolwa) sites. Note the annual river flow, rainfall and net loss statistics are averages for the full calendar years (2003–2005 reference, 2007–2009 low flow).

Parameter		**River**		**Lake Alexandrina**		**Lake Albert**	
		Low flow	**Ref.**	**Low flow**	**Ref.**	**Low flow**	**Ref.**
Daily inflow (ML/day)	n	1005	1004	n/a	n/a	n/a	n/a
	Median	1300	3830	n/a	n/a	n/a	n/a
	IQR	620	1305	n/a	n/a	n/a	n/a
	p-value	< 0.0001		n/a		n/a	
Annual inflow (GL)	n	3	3	n/a	n/a	n/a	n/a
	Average	537	1588	n/a	n/a	n/a	n/a
	IQR	n/a	n/a	n/a	n/a	n/a	n/a
	p-value	n/a		n/a		n/a	
Level (m AHD)	n	1005	1004	1006	1006	1006	1006
	Median	-0.31	0.74	-0.42	0.71	-0.23	0.71
	IQR	0.65	0.31	0.86	0.26	0.50	0.26
	p-value	< 0.0001		< 0.0001		< 0.0001	
Volume (m^3)	n	1005	1004	1006	1006	1006	1006
	Median	290026099	346639299	899411484	1584896060	113381003	264188343
	IQR	35046266	16849166	487095643	169753730	71366552	46323059
	p-value	< 0.0001		< 0.0001		< 0.0001	
Surface area (m^2)	n	1005	1004	1006	1006	1006	1006
	Median	53375308	54719509	559937517	649976235	143584614	170215383
	IQR	1295030	181469	83914253	15313995	23917854	1108589
	p-value	< 0.0001		< 0.0001		< 0.0001	
Mean depth (m)	n	1005	1004	1006	1006	1006	1006
	Median	5.45	6.36	1.60	2.43	0.78	1.54
	IQR	0.56	0.27	0.61	0.20	0.37	0.22

	p-value	< 0.0001		< 0.0001		< 0.0001	
Daily air temperature (°C)	n	1006	1005	982	994	982	994
	Median	15.9	15.6	15.0	15.0	15.0	15.0
	IQR	7.3	6.6	5.6	5.3	5.6	5.3
	p-value	NS		NS		NS	
***Daily wind speed* (km/h)**	n	1006	1006	1006	995	1006	995
	Median	8.0	7.5	22.0	22.5	22.0	22.5
	IQR	10.5	12.0	11.5	10.8	11.5	10.8
	p-value	0.004		NS		NS	
***Annual rainfall* (mm)**	n	3	3	3	3	3	3
	Average	426	315	423	374	423	374
	IQR	n/a	n/a	n/a	n/a	n/a	n/a
	p-value	n/a		n/a		n/a	
***Annual net loss* (GL)**	n	3	3	3	3	3	3
	Average	55.3	70.3	569.4	588.9	144.3	146.8
	IQR	n/a	n/a	n/a	n/a	n/a	n/a
	p-value	n/a		n/a		n/a	

NS = not significant at 5% level (p>0.05), n/a = data not available and/or statistical test not applied

Table 3 – Principal component analysis results for the river and lake sites showing varifactors (VF) with the amount of variability (%) in the dataset each one explains. VF1 was subsequently labelled the "hydrological" component and VF2 the "nutrient and suspended sediment" component.

	River sites				**Lake Sites**					
Parameter	Murray Bridge		Tailem Bend		Milang		Goolwa		Meningie	
	D1	D2	D1	D2	D1	D2	D1	D2	D1	D2
Inflow	**0.58**	0.32	**0.60**	0.13	**0.67**	0.14	**0.69**	0.19	**0.68**	-0.13
Level	**0.97**	0.06	**0.97**	-0.06	**0.98**	-0.09	**0.97**	-0.02	**0.97**	-0.06
Volume	**0.97**	0.06	**0.97**	-0.06	**0.98**	-0.09	**0.97**	-0.02	**0.97**	-0.06
Surface area	**0.97**	0.06	**0.97**	-0.06	**0.98**	-0.09	**0.97**	-0.02	**0.74**	-0.19
Mean depth	**0.97**	0.06	**0.97**	-0.06	**0.98**	-0.09	**0.97**	-0.02	**0.97**	-0.06
TDS	**-0.67**	-0.23	**-0.30**	0.10	**-0.69**	0.04	**-0.82**	-0.11	**-0.72**	0.05
TN	-0.29	**0.51**	-0.24	**0.64**	-0.47	**0.57**	**-0.71**	0.21	-0.17	**0.80**
TP	0.20	**0.77**	-0.13	**0.78**	-0.15	**0.77**	-0.44	**0.57**	-0.04	**0.89**
NOx	0.08	**0.38**	-0.35	**0.44**	n/a	n/a	n/a	n/a	n/a	n/a
FRP	-0.04	**0.61**	-0.31	**0.56**	n/a	n/a	n/a	n/a	n/a	n/a
Chlorophyll *a*	n/a	n/a	n/a	n/a	-0.07	**0.77**	-0.25	**0.66**	-0.13	**0.83**
Turbidity	0.46	**0.61**	0.03	**0.69**	0.06	**0.77**	0.45	**0.61**	-0.03	**0.88**
pH	**-0.52**	-0.23	-0.22	-0.28	-0.09	0.02	0.30	**0.54**	0.12	-0.03
Water Temp.	0.11	0.28	0.11	0.71	-0.26	**0.56**	-0.06	**0.41**	0.04	-0.11
Variability (%)	*40*	*16*	*35*	*20*	*42*	*21*	*50*	*14*	*37*	*25*
Cumulative %	*40*	*56*	*35*	*55*	*42*	*63*	*50*	*64*	*37*	*61*

Fig. 1. Sampling sites (Murray Bridge, Tailem Bend, Milang, Meningie and Goolwa) within the River Murray and Lower Lakes study area and the Murray-Darling basin. Also shown on the left inset map is the flow and water level monitoring station at Lock 1.

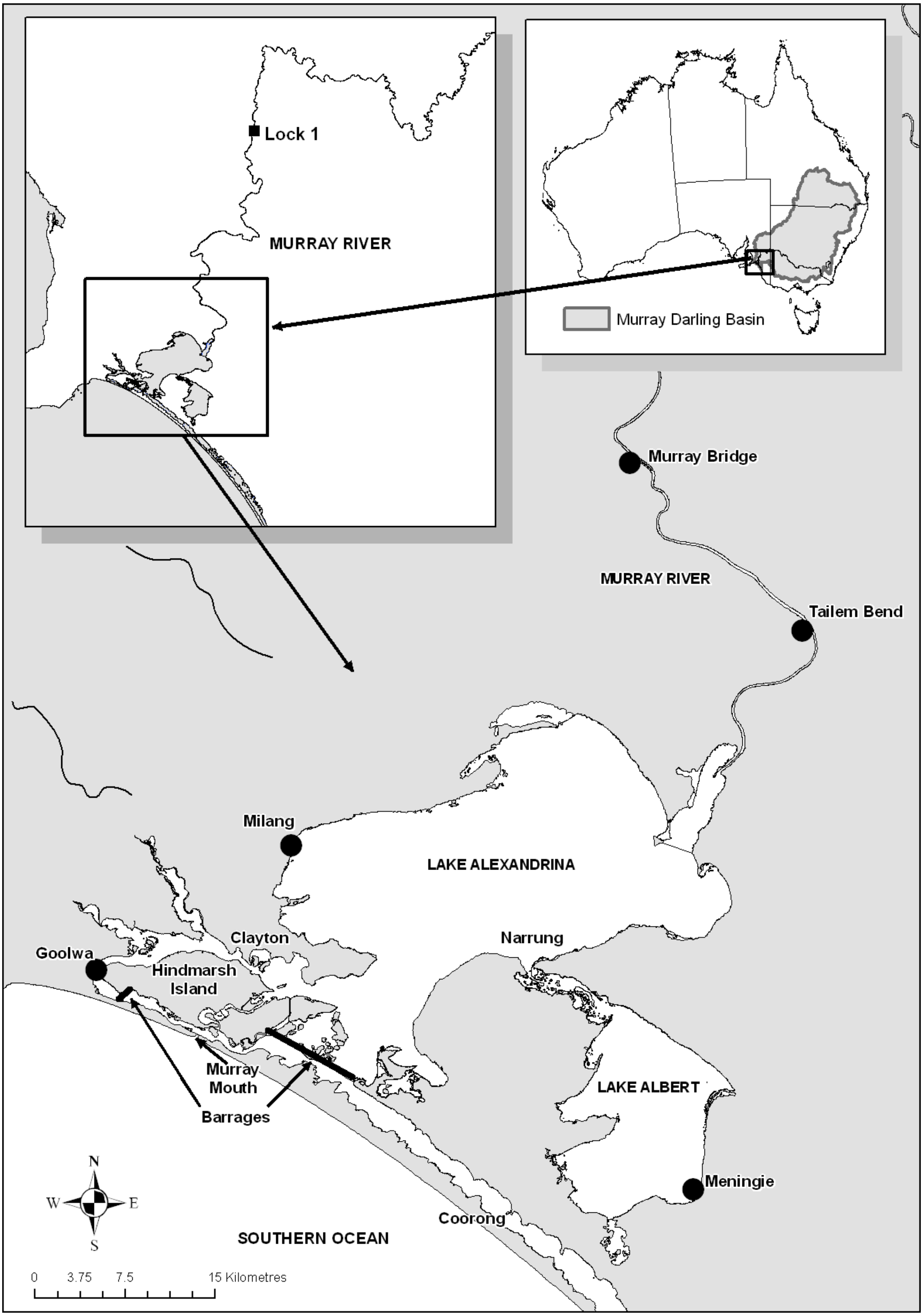


Fig. 2. Salinity (TDS), turbidity, TN, and TP at Tailem Bend (Lower Murray River) and Milang (Lower Lakes) from 2003-2010.

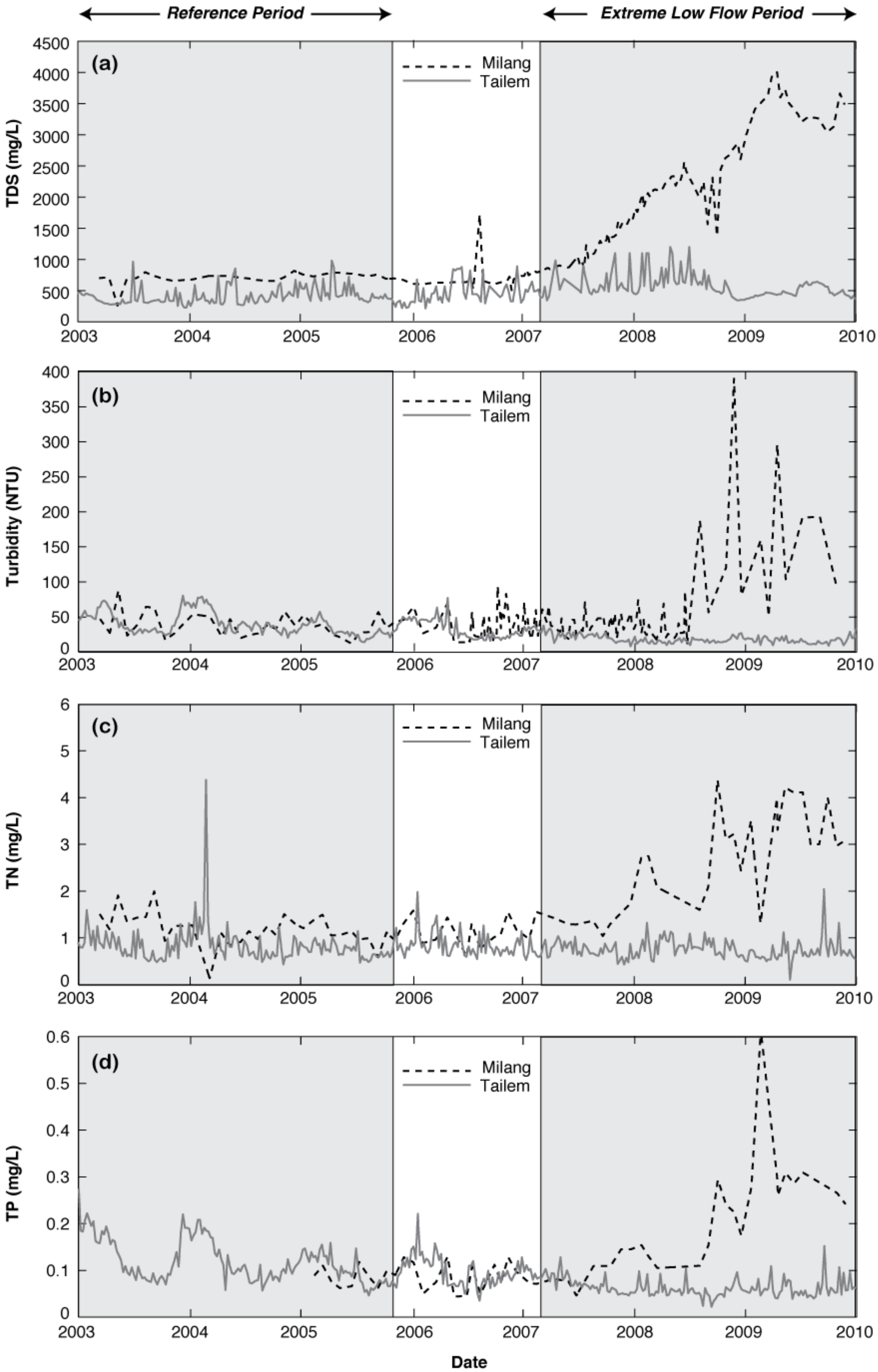


Fig. 3 (a) Murray River flow over Lock 1 and lake system/barrage outflows from 2003-2010, (b) Water levels in the Lower Murray River (downstream from Lock 1) and Lakes from 2003–2010 (with inset from 1920–2010), and (c) Water volumes in the Lower Murray River and Lakes from 2003–2010.

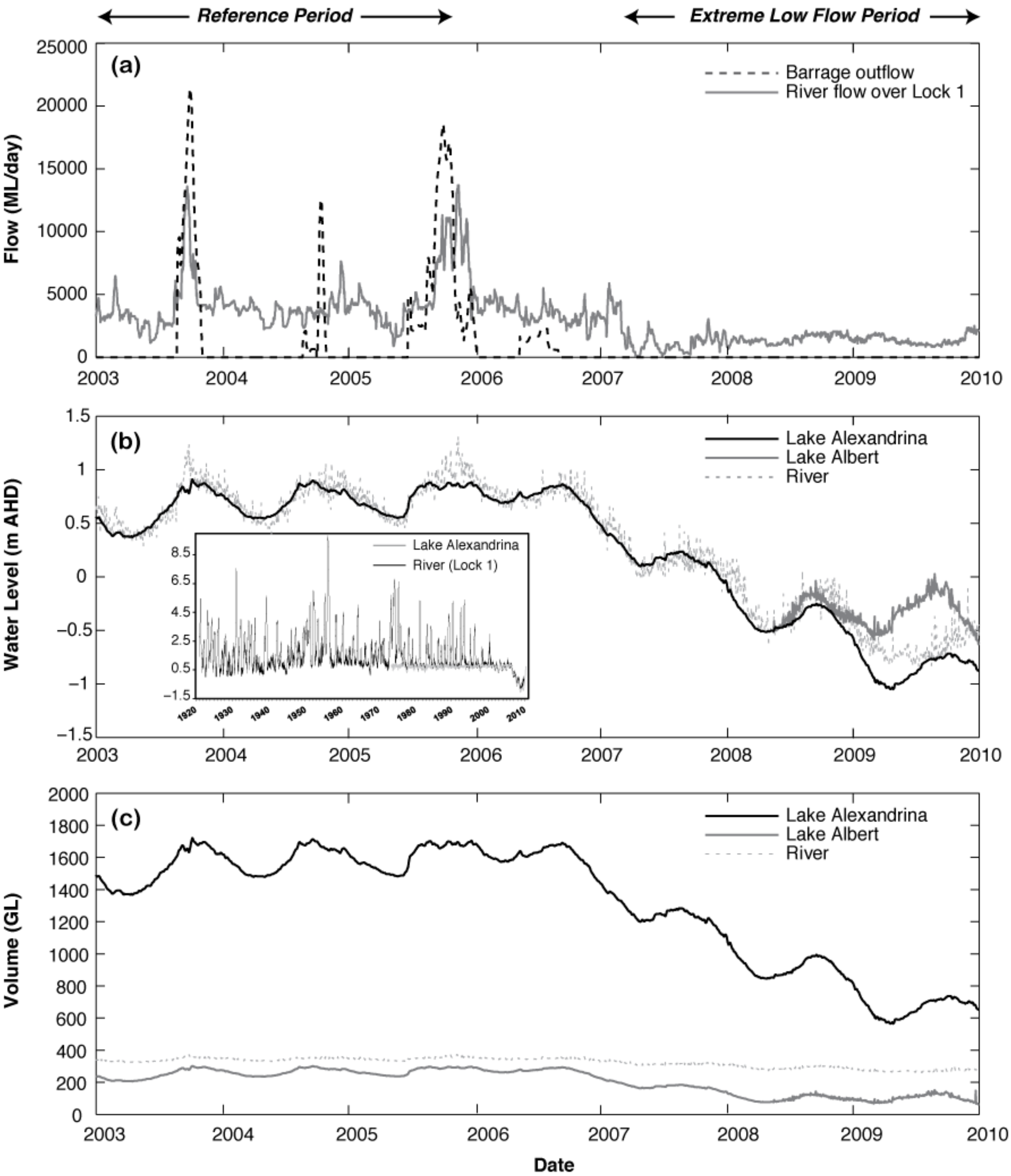


Fig. 4 – Mass of salt (TDS), TN, TP and TSS (calculated) in the river and lakes from 2003-2010. Also shown is the monthly mass balance model results for Lake Alexandrina. Parameters used in the mass balance model (see equation 1) were: $c_{in}= c_{out}$ using Tailem Bend input data, v_s = 0.025m/day, and *m* using measured Milang data.

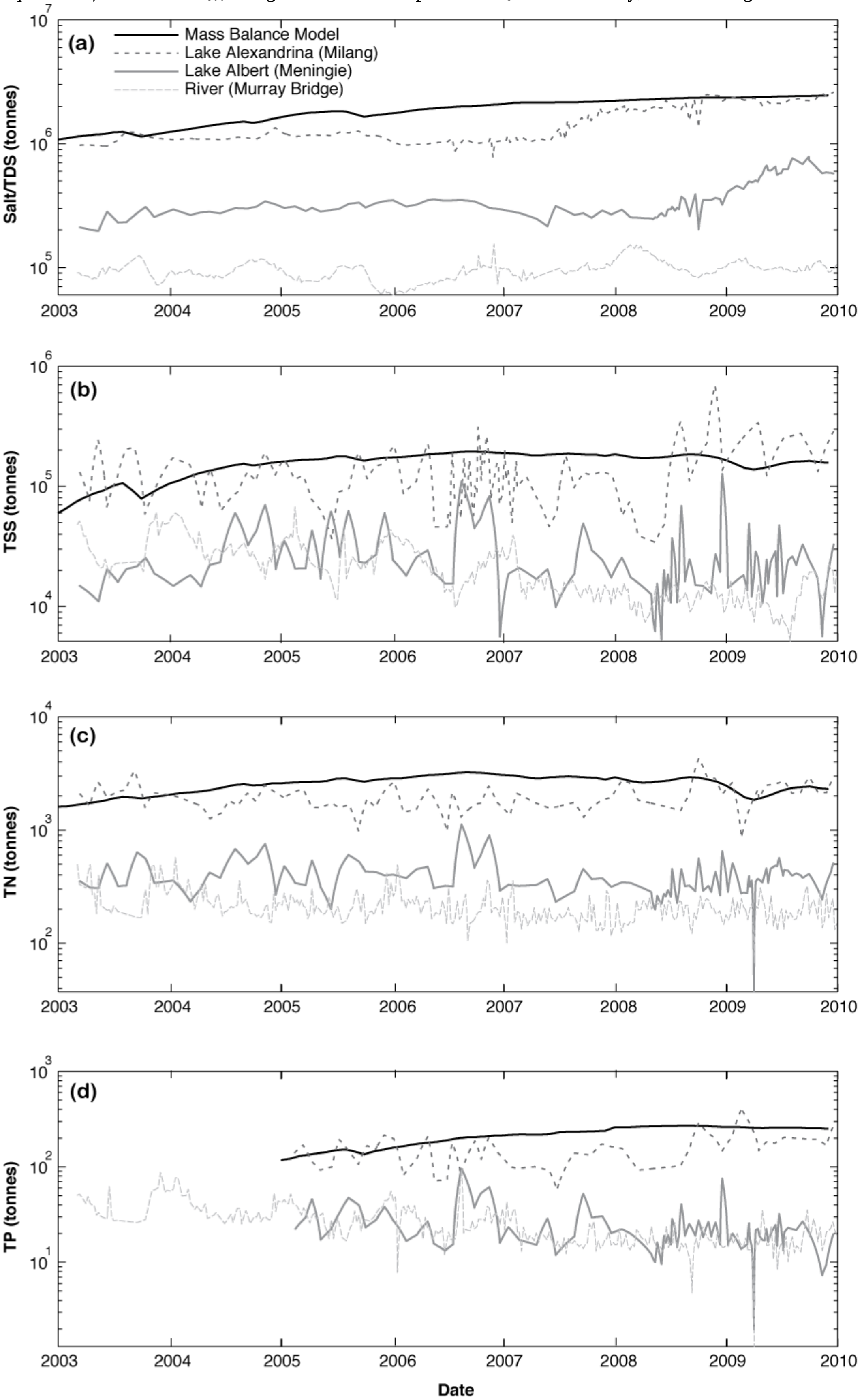


Fig. 5. Comparison of TP model predictions (Chapra 1975, 1997 and Brett and Benjamin 2008) with measured Milang data from 2005-2010. The estimated lake inflow (Tailem Bend data) TP concentration (TPin) is also shown.

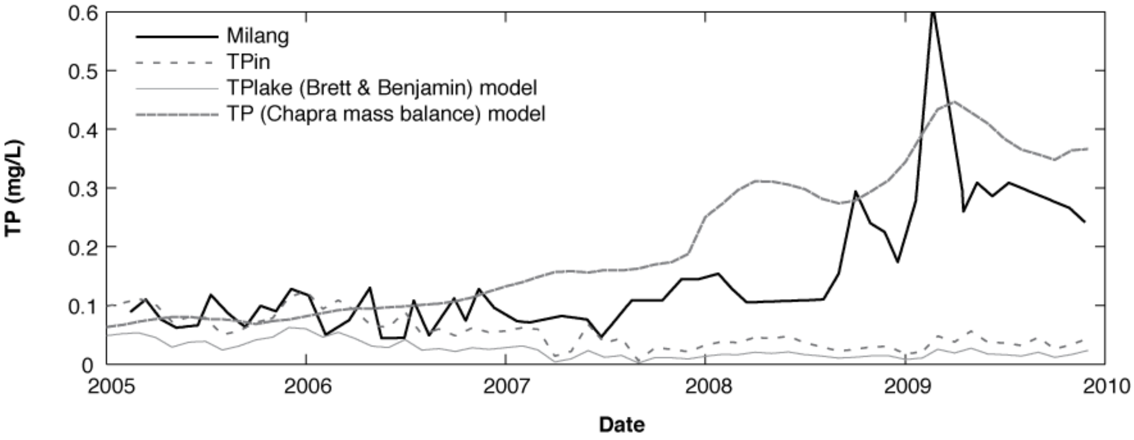
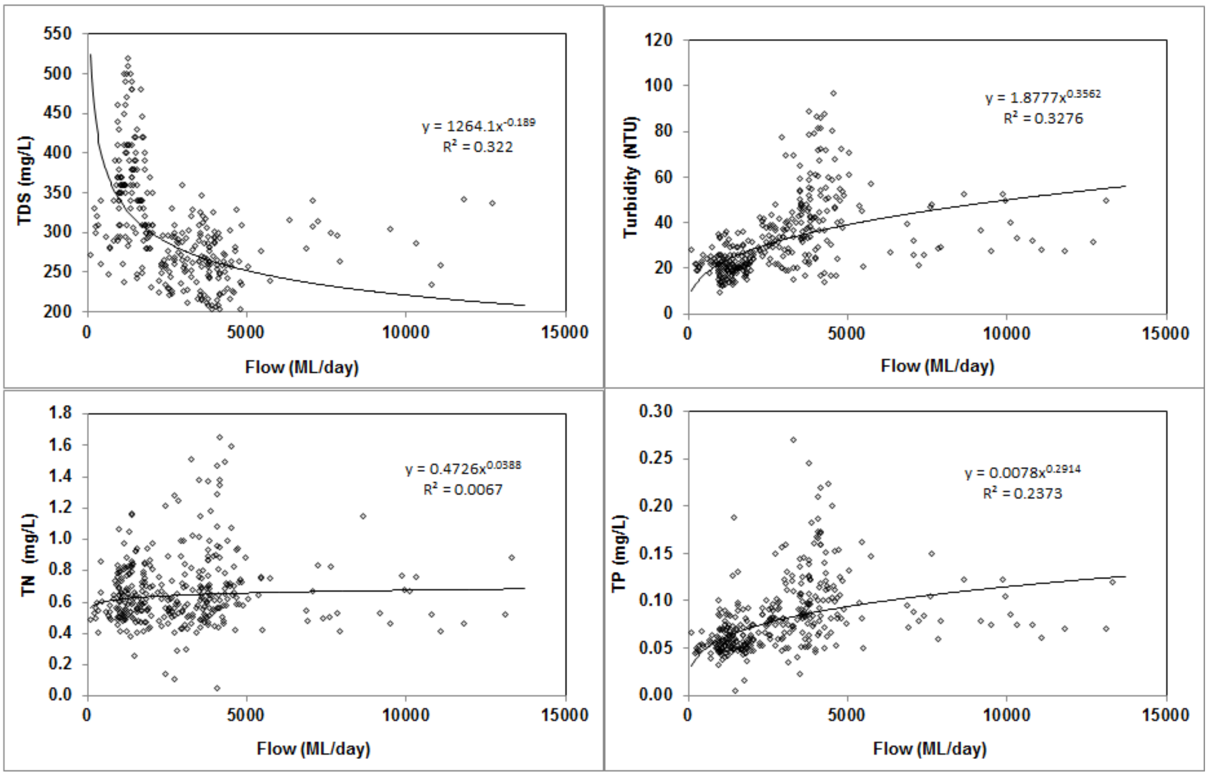
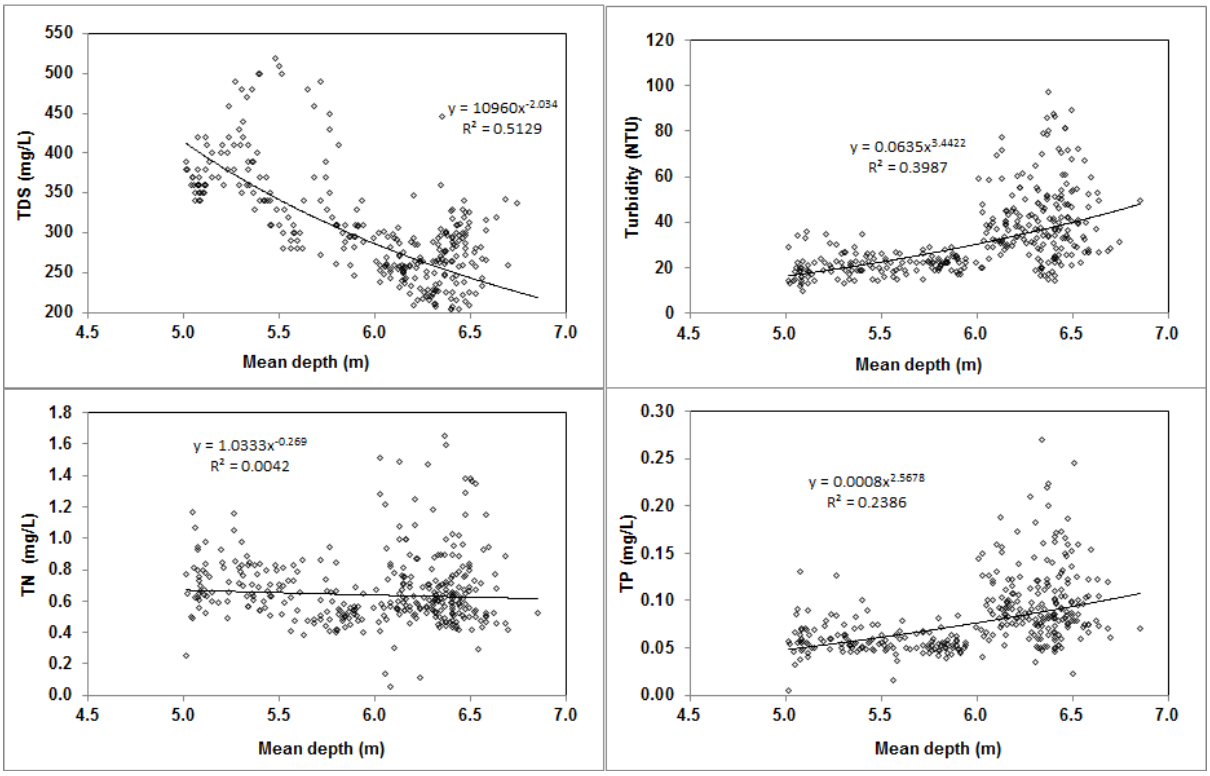


Fig. 6. Relationships between flow over Lock 1 and mean water depth with salinity (TDS), turbidity, TN and TP at Murray Bridge (a-b) and Milang (c-d)

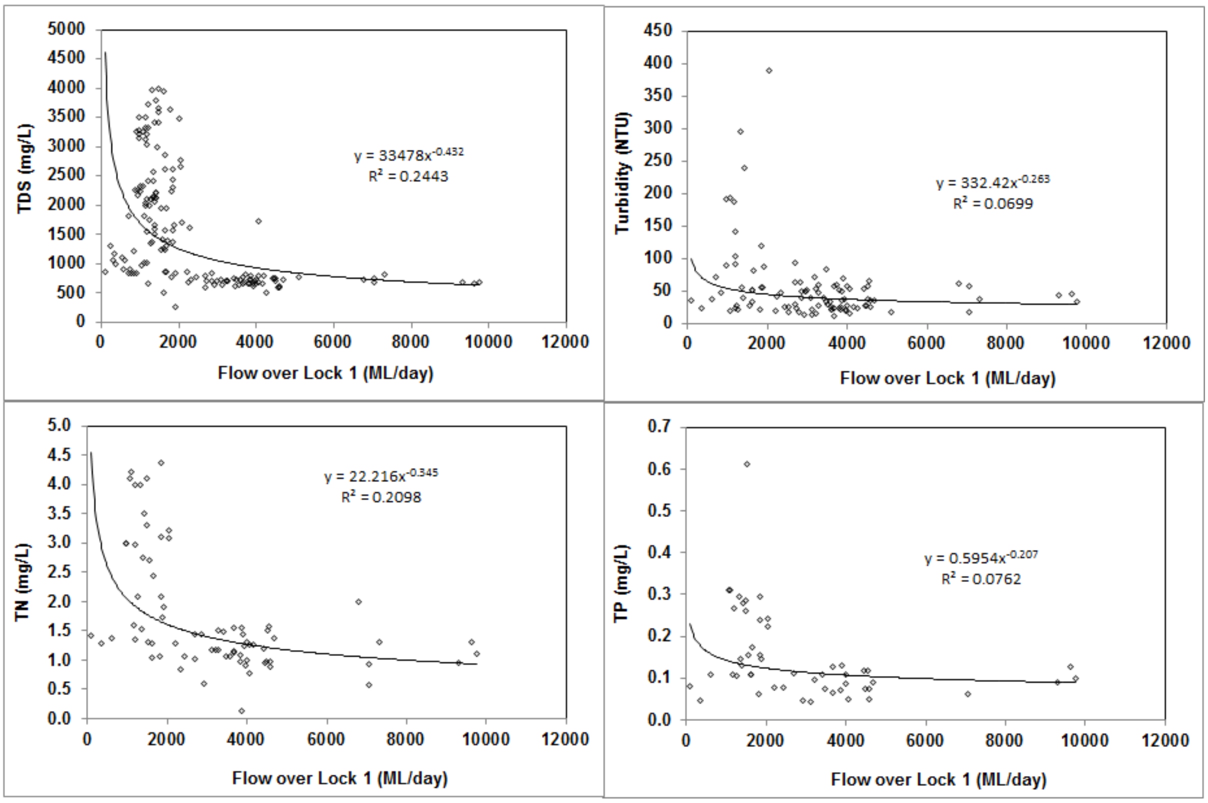
(a)



(b)



(c)


TDS (mg/L)
Flow over Lock 1 (ML/day)
y = 33478x$^{-0.432}$
R^2 = 0.2443
Turbidity (NTU)
Flow over Lock 1 (ML/day)
y = 332.42x$^{-0.263}$
R^2 = 0.0699
TN (mg/L)
Flow over Lock 1 (ML/day)
y = 22.216x$^{-0.345}$
R^2 = 0.2098
TP (mg/L)
Flow over Lock 1 (ML/day)
y = 0.5954x$^{-0.207}$
R^2 = 0.0762

(d)

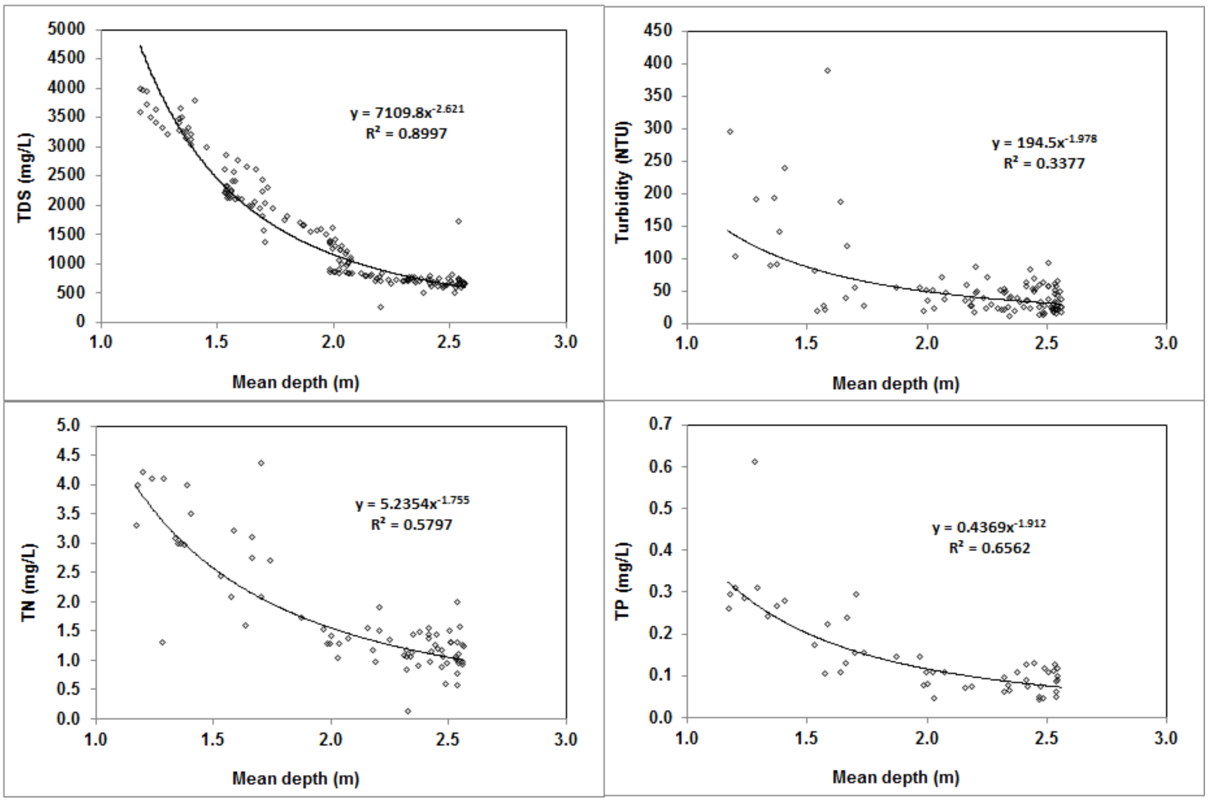
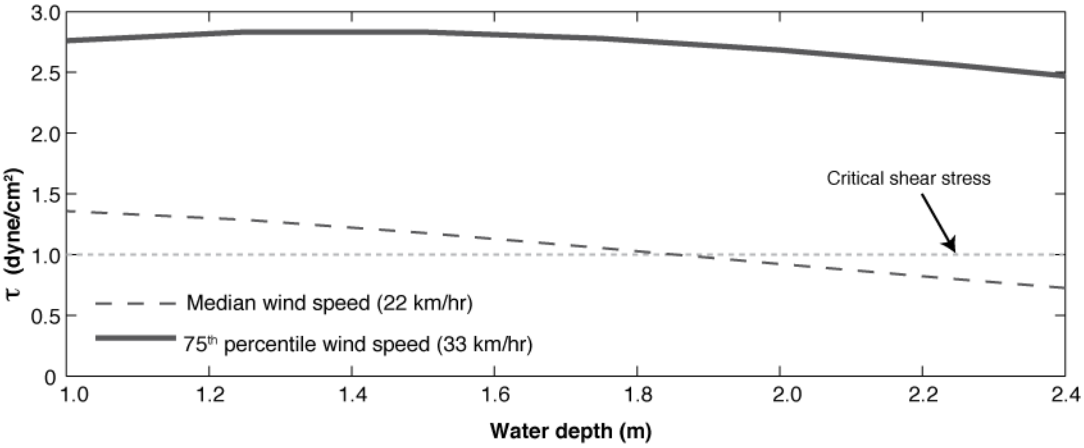
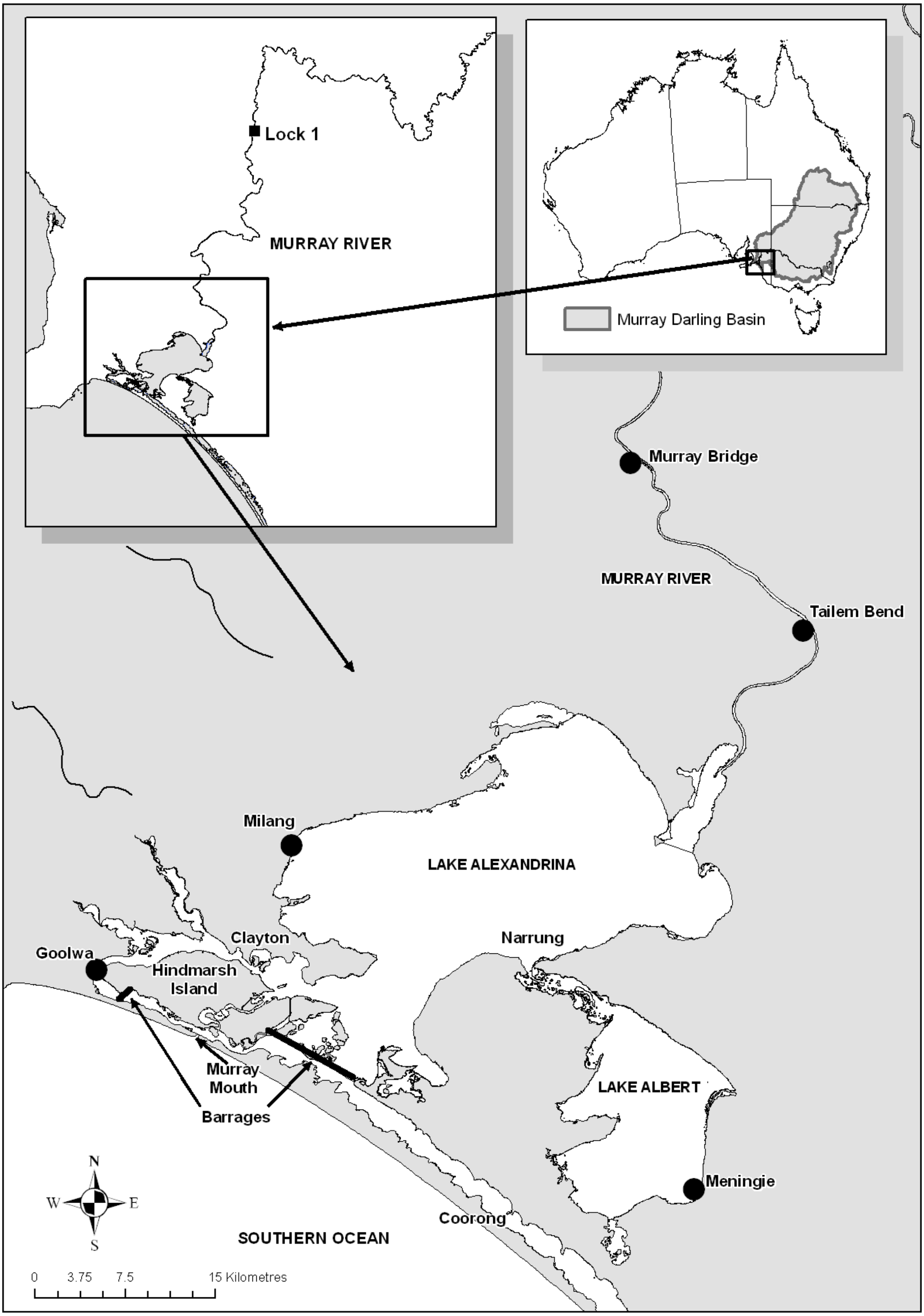
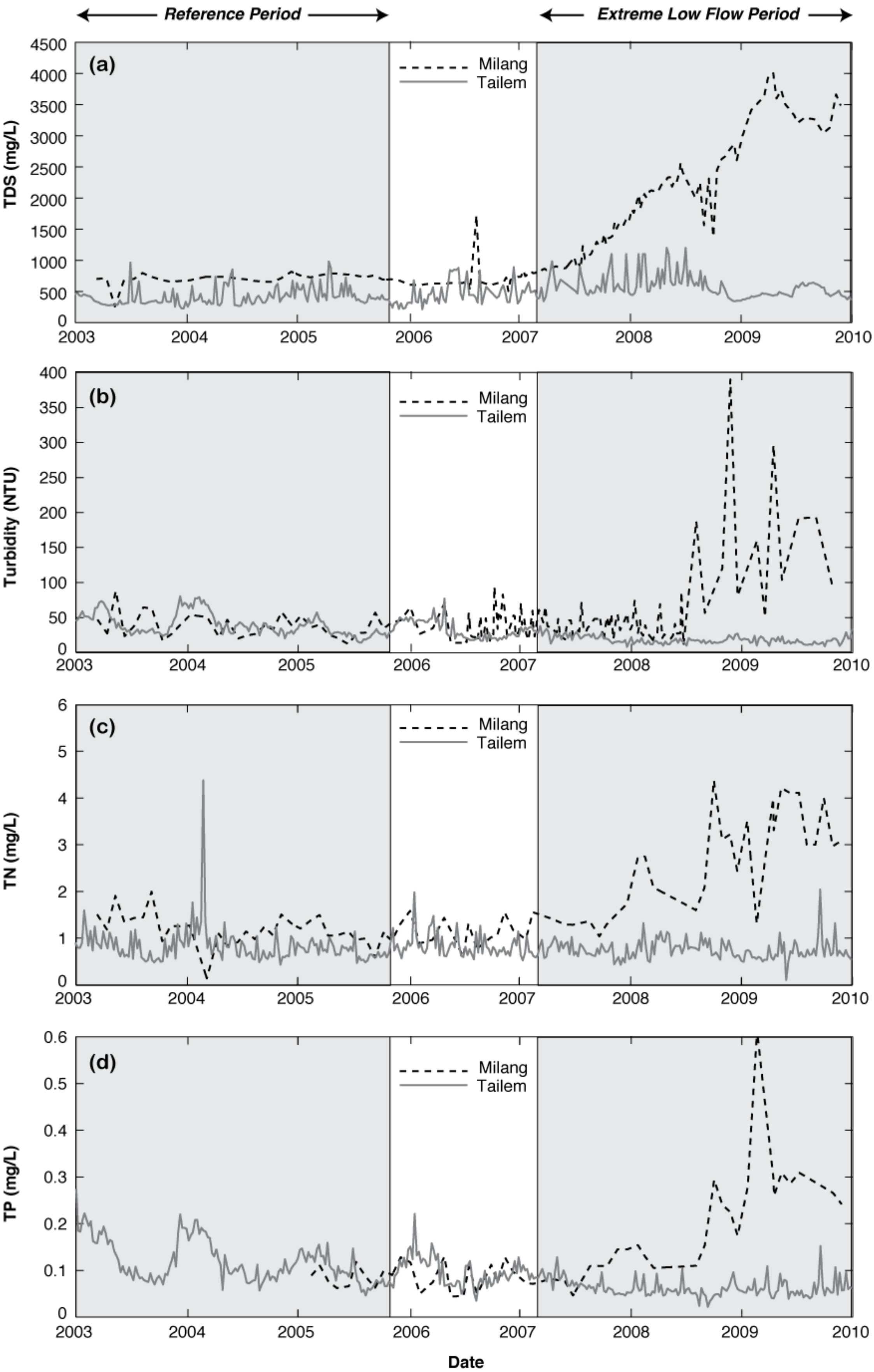

TDS (mg/L)
Mean depth (m)
y = 7109.8x$^{-2.621}$
R^2 = 0.8997
Turbidity (NTU)
Mean depth (m)
y = 194.5x$^{-1.978}$
R^2 = 0.3377
TN (mg/L)
Mean depth (m)
y = 5.2354x$^{-1.755}$
R^2 = 0.5797
TP (mg/L)
Mean depth (m)
y = 0.4369x$^{-1.912}$
R^2 = 0.6562

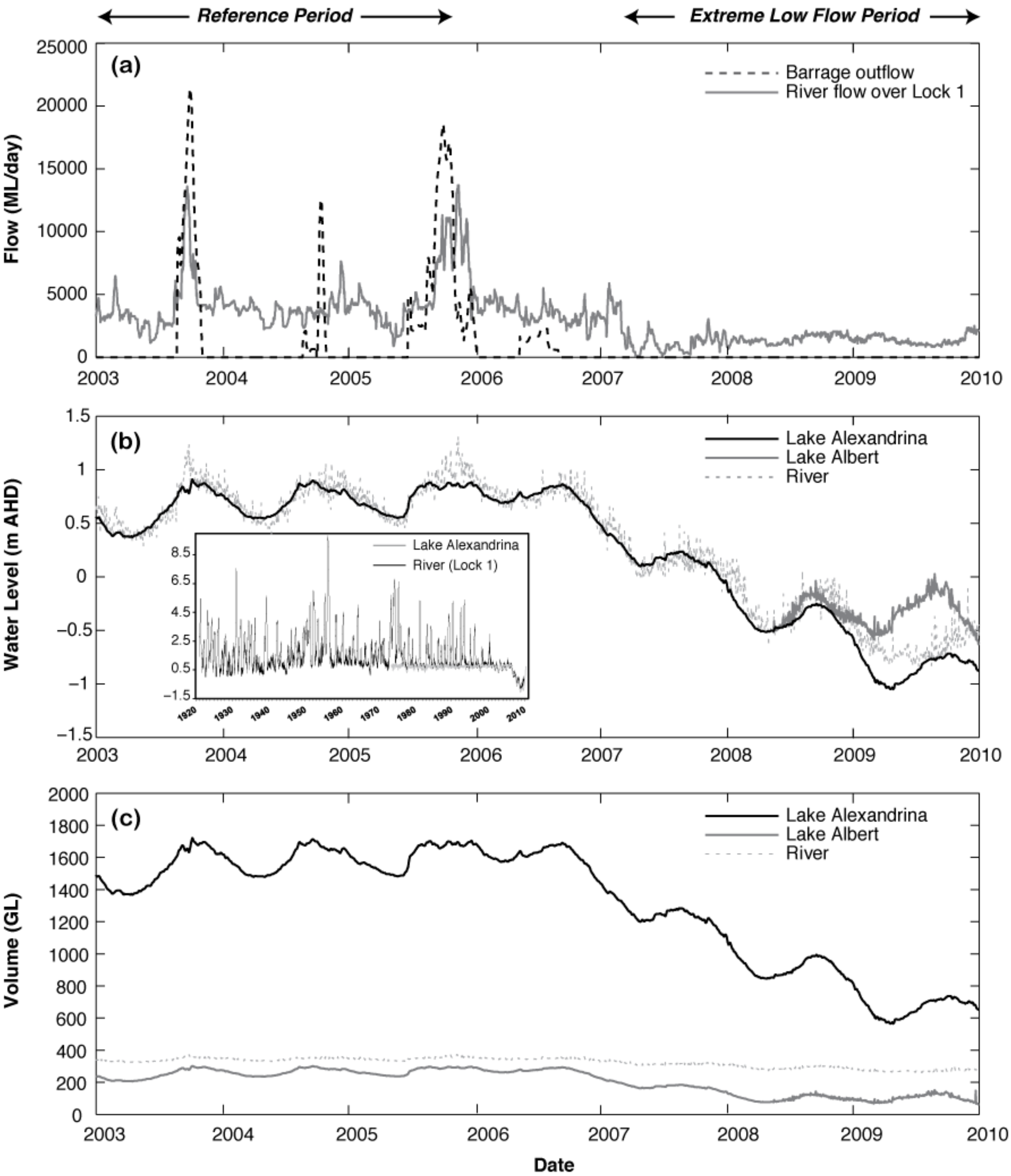
Fig.7. Shear stress (τ) estimated at the sediment water interface versus water depth and wind speed (median and 75th percentile, Table 2) using the equations provided by Kang et al. (1982) and Chapra (1997). The critical shear stress value of 1 dyne/cm^2 is when appreciable fine sediment resuspension begins to occur. A fetch of 30 km for Lake Alexandrina was used in the calculations.



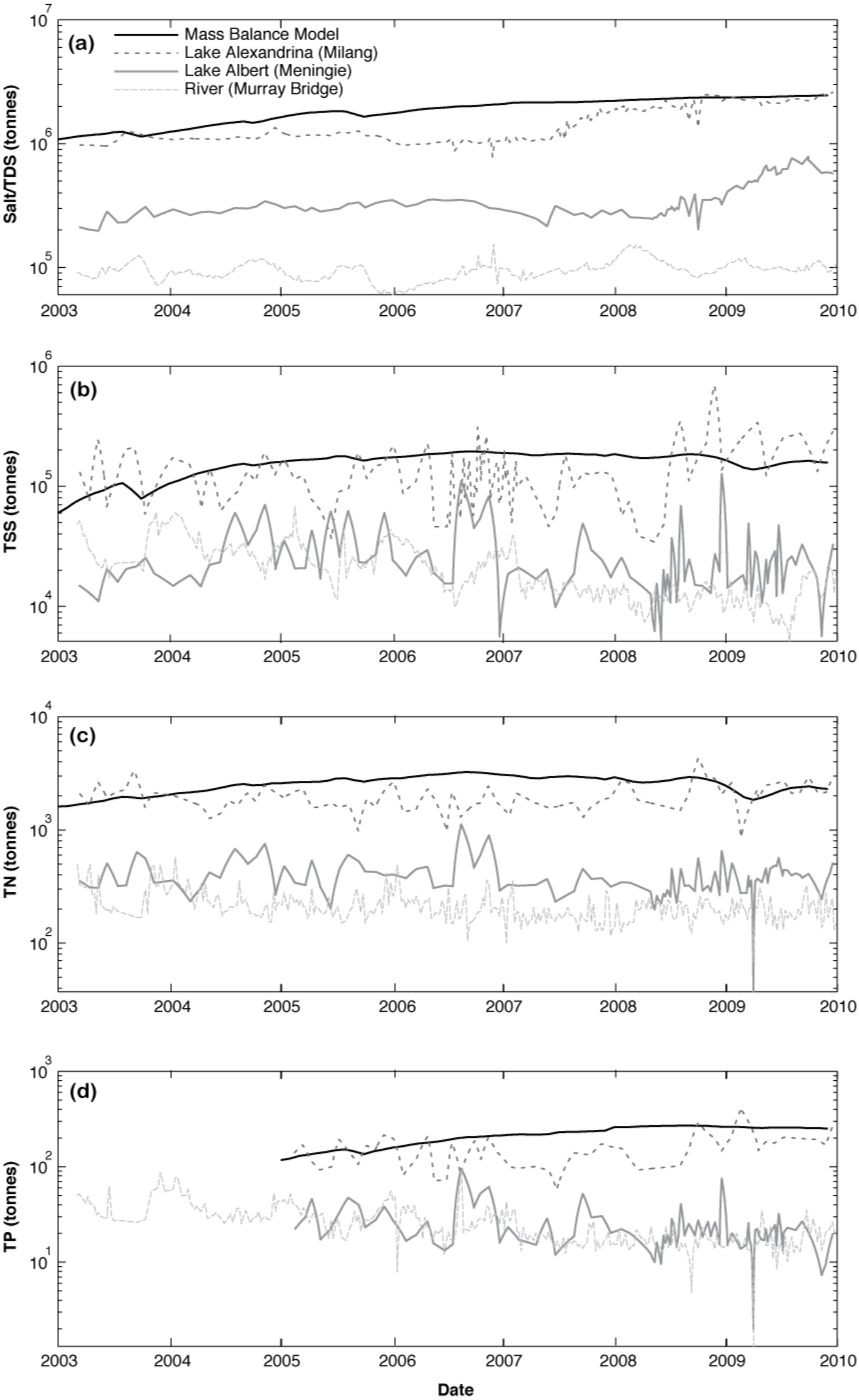

Lock 1
MURRAY RIVER
Murray Darling Basin
Murray Bridge
MURRAY RIVER
Tailem Bend
Milang
LAKE ALEXANDRINA
Clayton
Narrung
Goolwa
Hindmarsh Island
Murray Mouth
Barrages
LAKE ALBERT
Meningie
Coorong
SOUTHERN OCEAN
N
W
E
S
0
3.75
7.5
15 Kilometres



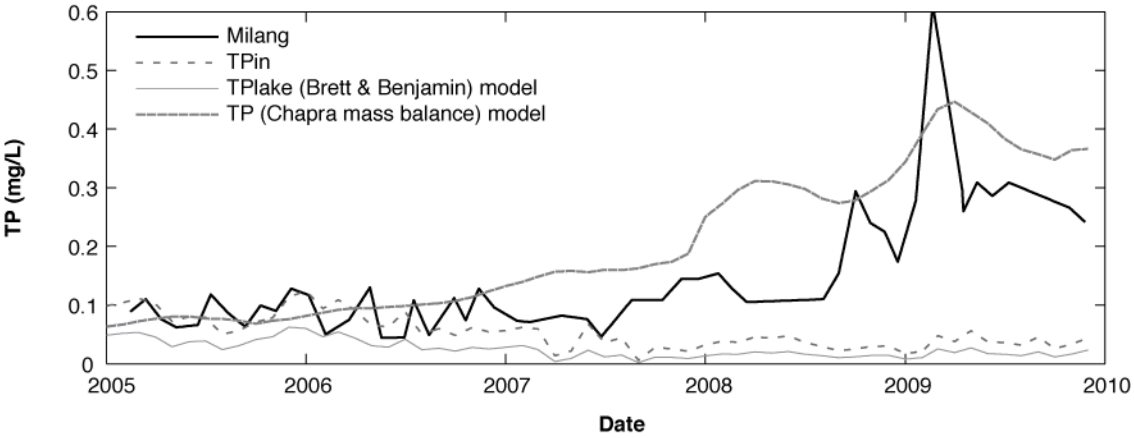
Reference Period
Extreme Low Flow Period
(a)
(b)
(c)
(d)
Milang
Tailem
TDS (mg/L)
Turbidity (NTU)
TN (mg/L)
TP (mg/L)
Date



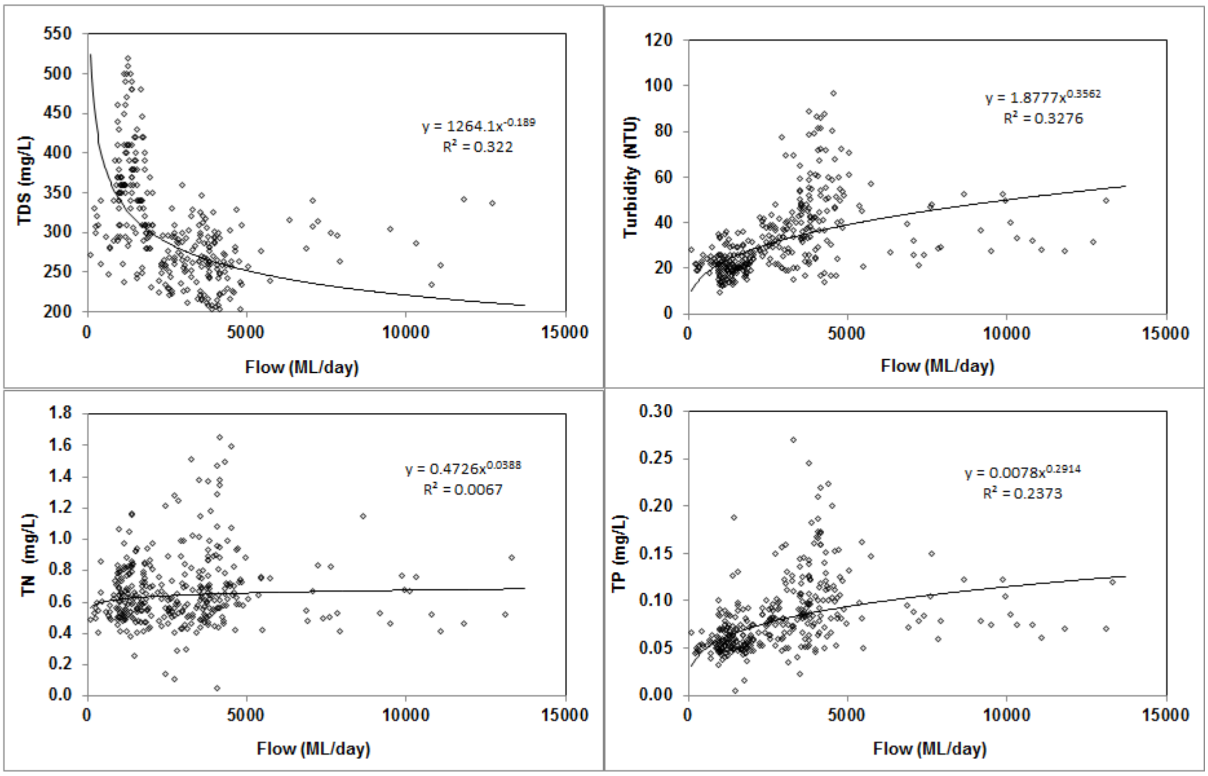
Reference Period
Extreme Low Flow Period
(a)
Barrage outflow
River flow over Lock 1
Flow (ML/day)
(b)
Lake Alexandrina
Lake Albert
River
Water Level (m AHD)
Lake Alexandrina
River (Lock 1)
(c)
Lake Alexandrina
Lake Albert
River
Volume (GL)
Date



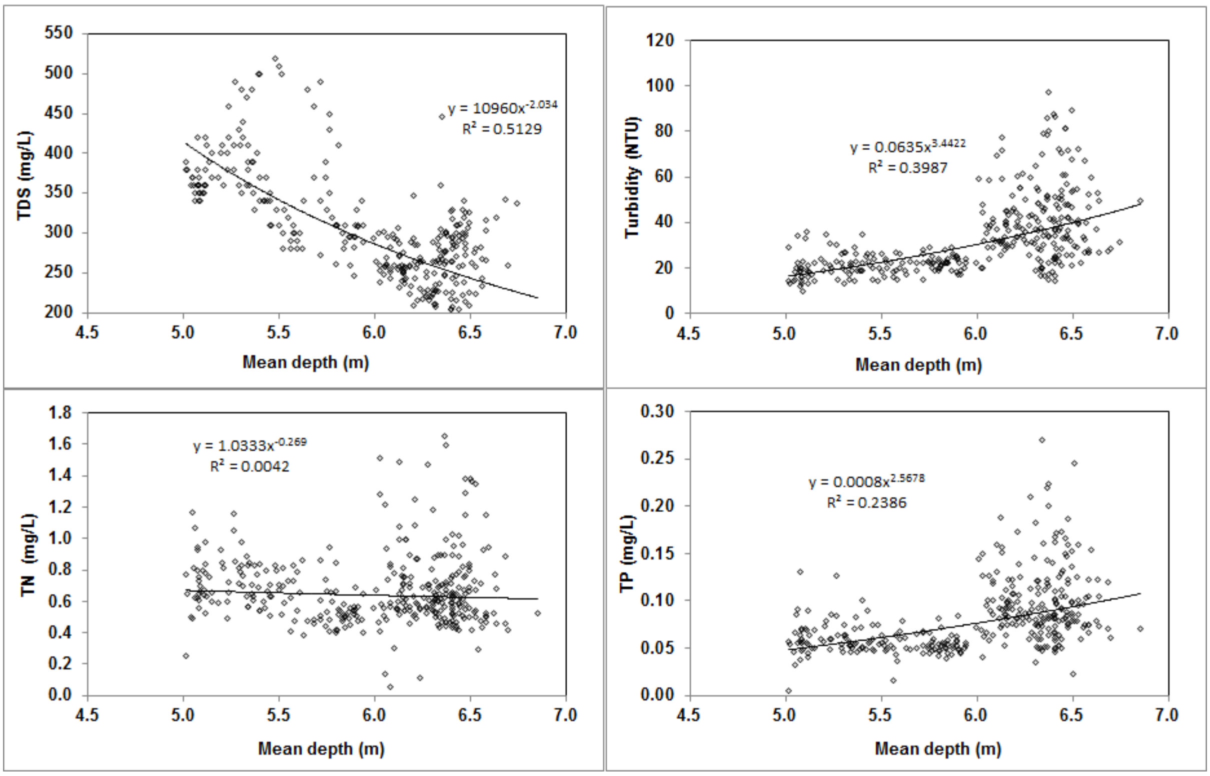
(a)
Mass Balance Model
Lake Alexandrina (Milang)
Lake Albert (Meningie)
River (Murray Bridge)
Salt/TDS (tonnes)
(b)
TSS (tonnes)
(c)
TN (tonnes)
(d)
TP (tonnes)
2003
2004
2005
2006
2007
2008
2009
2010
Date

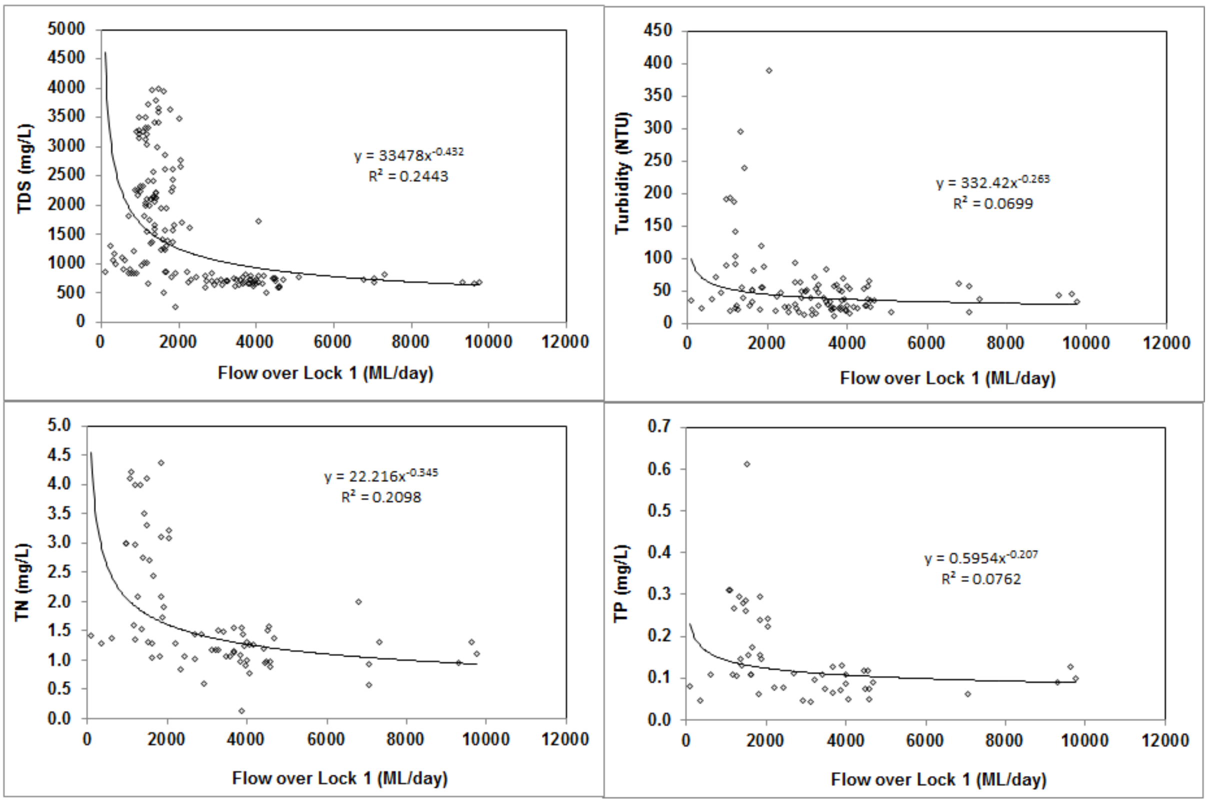


Milang
TPin
TPlake (Brett & Benjamin) model
TP (Chapra mass balance) model
TP (mg/L)
0.6
0.5
0.4
0.3
0.2
0.1
0
2005
2006
2007
2008
2009
2010
Date



TDS (mg/L)
Flow (ML/day)
y = 1264.1x^-0.189
R² = 0.322
Turbidity (NTU)
Flow (ML/day)
y = 1.8777x^0.3562
R² = 0.3276
TN (mg/L)
Flow (ML/day)
y = 0.4726x^0.0388
R² = 0.0067
TP (mg/L)
Flow (ML/day)
y = 0.0078x^0.2914
R² = 0.2373


TDS (mg/L)
y = 10960x-2.034
R² = 0.5129
Mean depth (m)
Turbidity (NTU)
y = 0.0635x3.4422
R² = 0.3987
Mean depth (m)
TN (mg/L)
y = 1.0333x-0.269
R² = 0.0042
Mean depth (m)
TP (mg/L)
y = 0.0008x2.5678
R² = 0.2386
Mean depth (m)


TDS (mg/L)
Flow over Lock 1 (ML/day)
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R² = 0.2443
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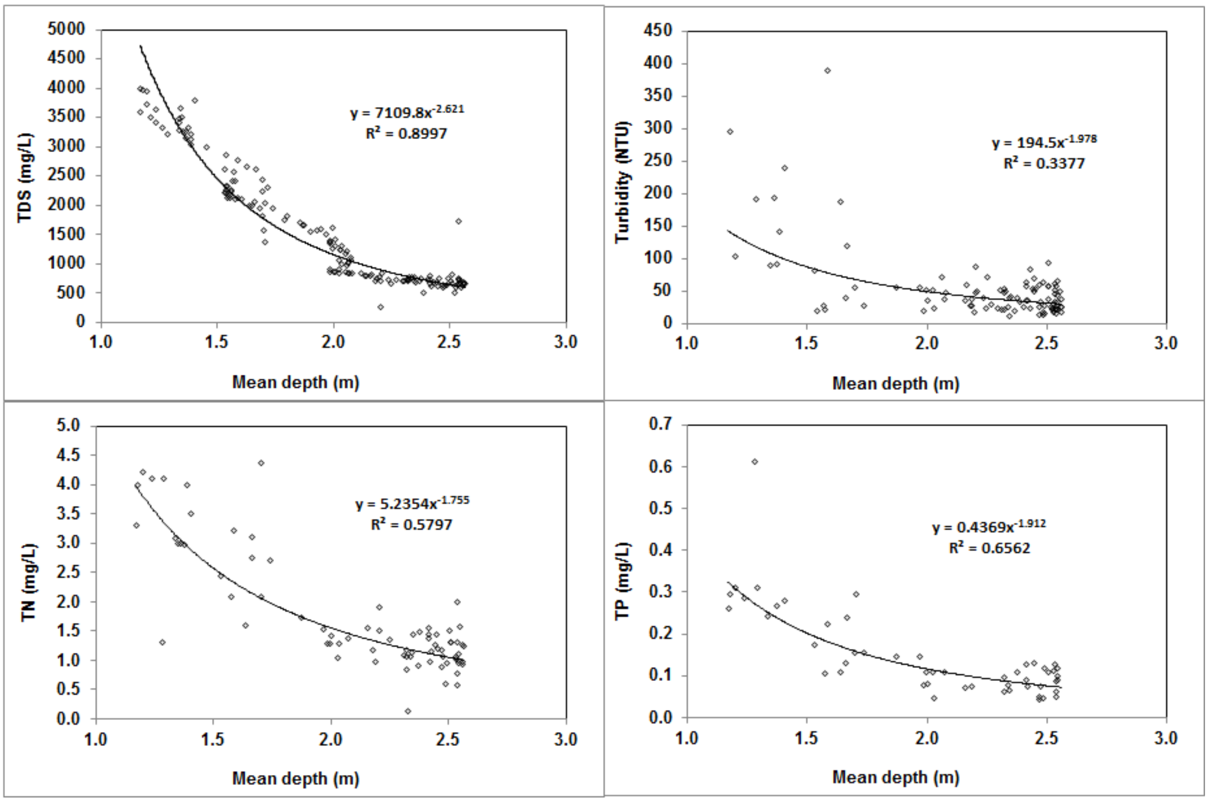
R² = 0.2098
TP (mg/L)
Flow over Lock 1 (ML/day)
y = 0.5954x-0.207
R² = 0.0762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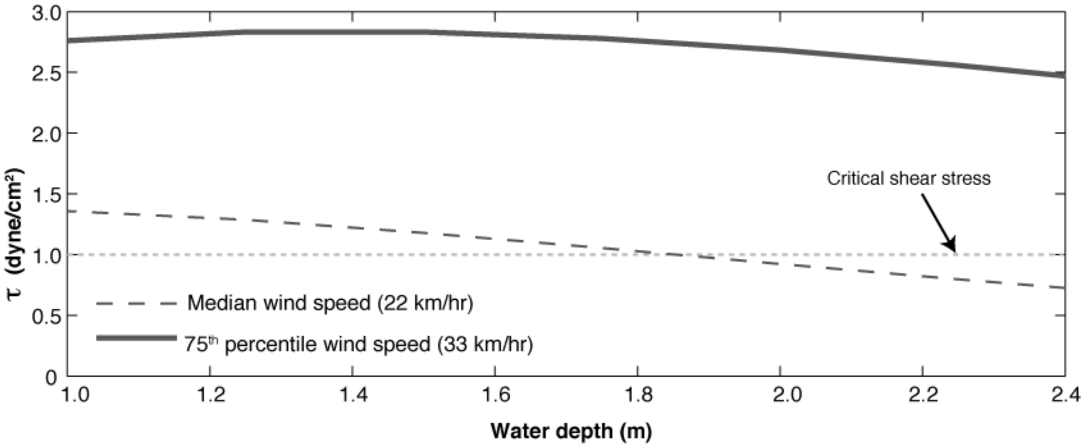
R² = 0.8997
Mean depth (m)
Turbidity (NTU)
y = 194.5x^-1.978
R² = 0.3377
Mean depth (m)
TN (mg/L)
y = 5.2354x^-1.755
R² = 0.5797
Mean depth (m)
TP (mg/L)
y = 0.4369x^-1.912
R² = 0.6562
Mean depth (m)



3.0
2.5
2.0
1.5
1.0
0.5
0
τ (dyne/cm²)
Critical shear stress
Median wind speed (22 km/hr)
75th percentile wind speed (33 km/hr)
1.0
1.2
1.4
1.6
1.8
2.0
2.2
2.4
Water depth (m)