Report for 2005TX196B: Watershed Development and Climate Change Effects on Environmental Flows and Estuarine Function

Publications

- Other Publications:
    - Russell, M. J., P. A. Montagna, and R. D. Kalke. In Press 2006. The Effect of Freshwater Inflow on Net Ecosystem Metabolism in Lavaca Bay, Texas. Estuarine Coastal and Shelf Science.
    - Russell, M. J., and P. A. Montagna. Spatial and Temporal Variability and Drivers of Net Ecosystem Metabolism in Western Gulf of Mexico Estuaries. Submitted 2006. Estuaries and Coasts.

Report Follows

Watershed Development and Climate Change Effects on Environmental Flows and Estuarine Function

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Introduction

Coastal environments, as the receiving ecosystems of freshwater inflow from watersheds, have the potential to be greatly influenced by anthropogenic and climatic watershed modifications (Jørgensen 1980, Officer et al. 1984, Rosenberg 1985, Andersen and Rydberg 1988, D'Avanzo et al 1996, Montagna et al. 2002a). Ecosystem ecologists are searching to find suitable indicators of estuarine ecosystem response to changing environmental conditions so that watershed management can, with some degree of certainty, maintain estuaries within an acceptable range of conditions. The dynamic nature of watershed landscape modification, land use/ land cover, and the uncertainties of regional meteorological changes due to climate change require scientists to find an indicator of ecosystem response that integrates at the watershed level scale. The inherent variability in coastal ecosystems such as estuaries, however, requires scientists to find an indicator of estuarine response that is sensitive enough to respond to watershed changes on various spatial and temporal scales, but is simple and efficient enough to make spatial and temporal assessment under highly variable conditions possible (Russell 2005). Here it is proposed that net ecosystem metabolism (NEM) is a good indicator for assessment of the influence of anthropogenic and climatic watershed modifications on coastal environments because NEM has been shown to respond to highly variable spatial and temporal environmental conditions in estuaries (Russell 2005).

Climatic changes and anthropogenic watershed modifications during the next one hundred years will change freshwater inflow and environmental conditions in coastal areas. The Union of Concerned Scientists (Twilley et al. 2001), in their regional summary of the Intergovernmental Panel on Climate Change findings for the Gulf of Mexico (Houghton et al. 2001), reports that precipitation over Texas coastal watersheds will increase from 5-25%. This precipitation will be delivered in higher magnitude freshwater inflow pulses. The incidence of heavy precipitation events and high river flows are estimated to increase by 7.5% and 21% respectively. Anthropogenic watershed development may reinforce this increase in higher magnitude freshwater inflow pulses, because impervious surfaces reduce the infiltration capacity of the soil (U.S. Soil Conservation Service 1986). This decreased infiltration capacity results in a higher percentage of precipitation becoming runoff, which leaves less precipitation to become groundwater. Population in Texas increased by 17.8 million people during the 20th century (US Census 2000). Counties in coastal watersheds, such as San Patricio and Bee counties, have had 14.3% and 28.7% population growth respectively between the years 1990 and 2000, and are predicted to triple over the next one hundred years. The combination of increased magnitude precipitation events and a reduced infiltration capacity due to urbanization will result in freshwater inflow delivered in higher magnitude but shorter duration pulses. Decreased groundwater may result in smaller river flows during the interval between precipitation events (Houghton et al. 2001, Twilley et al. 2001). Thus, even with more predicted precipitation, average conditions may be closer to drought conditions. Thus, precipitation is estimated to be

more variable with higher magnitude pulses interspersed with longer dry periods (Houghton et al. 2001, Twilley et al. 2001). Soil moisture is predicted to increase by up to 25% but this is dependant on estimates of evaporation. The predicted climatic changes and watershed development and the resulting modifications to freshwater inflow and environmental conditions could affect coastal estuarine ecosystem metabolic processes.

The purpose of the present study was to create models to assess how predicted climate and watershed changes might affect freshwater inflow and ecosystem function as indicated by NEM in Texas estuaries. First, freshwater inflow, salinity, and temperature's influence on NEM was modeled in two separate bay systems. Then, a hydrological model was developed to estimate freshwater inflow under the wide range of observed and predicted precipitation conditions. Finally the NEM and hydrological models were linked to provide the means to estimate NEM under various land cover/ land use development and climate change scenarios. Estimates of NEM during inflow events under predicted future climate and watershed development conditions were compared to present day NEM. Comparisons between present and future NEM were used to assess whether predicted changes will result in a more heterotrophic or autotrophic estuarine ecosystem.

Methods and Materials

Study site.

The Texas lagoonal estuarine system contains seven distinct bay systems (Longley 1994). Most of these systems incorporate a primary bay with either a direct or indirect opening to the Gulf of Mexico, and a smaller secondary bay that receives most of the freshwater inflow from rivers. The primary and secondary bays are partially separated by a land constriction at the mouth of the secondary bay. Freshwater inflow thus has a greater influence on secondary bays. One of these secondary bays, Copano Bay, Texas (28° 6.9' N, 97° 1.5' W) and its watershed have unique characteristics that made it particularly attractive for the current research goals. The Aransas River, which drains into Copano Bay, drains a relatively small (2,172 km^2) watershed. The watershed can be broken down into three sub-basins with different land use/ land cover and soil type characteristics (Fig. 1). The Aransas River watershed is one of only a few south Texas river systems that are currently unimpeded by dams. Copano Bay is a shallow, micro-tidal, meso-haline, lagoonal coastal plain estuary with wind dominated mixing (Longley 1994). Mean daily freshwater inflow over the past 40 years from the Aransas River equals 28 m^3 s^{-1} (NOAA 1997). Freshwater is mainly delivered during spring and early summer freshets. Lavaca Bay, located about 100 km to the northeast of Copano Bay (28° 38.4' N, 96° 36.6' W), receives most of its freshwater inflow from the Lavaca River watershed (2,110 km^2). Lavaca Bay receives about 4-6 times the amount of freshwater inflow that Copano Bay does. Lavaca Bay also encountered a wider range of freshwater inflows in 2004 than Copano Bay, and thus provided a model system for addressing salinity effects on ecosystem metabolism.

Sampling design.

Water quality and environmental condition data were collected from field studies and public websites. Water quality parameters were also sampled at one station in upper Copano Bay for one week each month in 2004, and at three synoptically sampled stations out from the river

mouth for one week during five different months in 2004 (Russell 2005). Samples were taken at both surface (0.5 m from surface) and bottom depths (0.25 m from bottom). Sampling in upper Lavaca Bay took place between 2002 and 2004 (Russell 2005, Russell et al In Press). Daily freshwater inflow from the Aransas River was downloaded from the United States Geological Survey (USGS) (USGS station 08189700 Aransas River near Skidmore, TX). Daily freshwater inflow into Lavaca Bay was determined from flow gauges in the Lavaca River, Placedo Creek, and Garcitas Creek which are numbered USGS 08164000, 08188800, and 08164600 respectively. Wind speed was downloaded from the Texas Coastal Ocean Observation Network for stations in Port Aransas, and Sea Breeze, Texas.

Water quality measurements.

During field sampling, dissolved oxygen and other water quality parameters were measured every 15 minutes at surface and bottom depths, or at mid-depth, using YSI series 6 multiparameter data sondes. Models 6920-S and 600XLM data sondes with 610-DM and 650 MDS display loggers were used. The series 6 parameters have the following accuracy and units: temperature (± 0.15^{o}C), pH (± 0.2 units), dissolved oxygen (mg l^{-1} ± 0.2), dissolved oxygen saturation (% ± 2%), specific conductivity (± 0.5% of reading depending on range), depth (± 0.2 m), and salinity (± 1% of reading or 0.1 ppt, whichever is greater). Salinity was automatically corrected to 25°C.

NEM Model.

The goal was to produce NEM models for Copano and Lavaca Bay that could be used to predict NEM changes due to currently available climate change and watershed development predictions. A suite of environmental conditions, driven mainly by temperature, salinity, and freshwater inflow, have already been estimated to explain up to 70% of the total environmental variability at four Texas estuarine sites (Russell 2005). Dissolved oxygen and dissolved oxygen percent saturation could not be included in the NEM model as they are used in the calculation of NEM and would lead to circular reasoning. Other parameters, such as pH and chlorophyll-a had to be avoided because they can be affected by biological processes linked to NEM. This left freshwater inflow, salinity, and temperature as the main candidates for inclusion in the NEM model. The relationships between temperature, salinity, freshwater inflow, and NEM in Copano and Lavaca Bay have been previously analyzed using linear regression analysis (Russell 2005). In Copano Bay, temperature, salinity, and freshwater inflow were found to not have a linear relationship with NEM, but a non-linear trend was evident for freshwater inflow. NEM tended to increase to a peak and then decrease as freshwater inflow increased. Thus, the relationship between freshwater inflow and NEM was examined with a non-linear model based on a model used to explain biological structure characteristics at different cumulative freshwater inflows (Montagna et al. 2002b). The assumption behind the model is that NEM values peak at some maximum value with small increases from base freshwater inflows and NEM values decline prior to and after this peak. The shape of this curve can be predicted with a three-parameter, log normal model:

$$Y = a\,(\exp(-0.5\,(\ln(X/c))/b)^2)),$$

where Y is NEM, X is freshwater inflow, and as explained in Montagna et al. (2002b) the three parameters characterize different attributes of the curve, where "a" is the maximum value, "b" is the skewness or rate of change of the response as a function of freshwater inflow, and "c" is the location of the peak response value on the freshwater inflow axis.

The model was fit to data using the Regression Wizard in SigmaPlot, which uses the Marquardt-Levenberg algorithm to find coefficients (parameters) of the independent variables that give the best fit between the equation and the data (SigmaPlot 2000).

Lavaca Bay results implied a multiple-linear relationship between temperature, salinity, freshwater inflow, and NEM (Russell 2005). The linear response in Lavaca Bay, not found in Copano Bay, may be due to the wider range of environmental conditions influencing Lavaca Bay during 2004. For example, the range of salinity during 2004 sampling in Lavaca Bay was 0-23 ppt while Copano Bay was only 1-12 ppt. Thus, a multiple linear regression analysis was used to create an empirical NEM model for Lavaca Bay that would respond to temperature, salinity, and freshwater inflow.

The fit of the Copano NEM model curve to observed NEM values was assessed using a range of cumulative daily freshwater inflow. An analysis of R^2's using one to twenty-one days of cumulative freshwater inflow into Copano Bay was run and the best fit used for NEM model calibration. This analysis allows for assessment of how many previous days inflow influence NEM in Copano Bay, and provides justification for adjusting the previously used ten-day cumulative freshwater inflow calculation (Russell 2005, Russell et al In Press).

NEM results from Copano Bay (Russell 2005) were fitted to the conceptual non-linear model. NEM results were previously calculated using the dissolved oxygen diurnal curve method as detailed in Chapter 2. The entire set of NEM results from Copano Bay was used for calibration, except those results from April when an abnormally large freshwater inflow event occurred (> 30 million m^3 in ten days). April results were analyzed separately for temporal trends in NEM occurring as this freshwater inflow event subsided (Russell 2005).

The NEM model is comprised of three parameters; a, b, and c. Sensitivity of the model to each of the parameters was assessed by varying one while holding the other two constant. Validation of the both NEM models was completed by comparing observed to simulated NEM results for 2002-2004 and model estimation error was quantified.

Hydrological model.

One goal was to assess the influence of precipitation and watershed land use/ land cover characteristics on NEM. To facilitate this analysis a simple hydrological model for the Aransas River watershed was produced based on the soil conservation service (SCS) curve number method (US Soil Conservation Service 1986) and NEXRAD precipitation data. This hydrological model converted spatially and temporally dynamic precipitation inputs into runoff. The model also adjusted runoff by factoring in watershed soil types and land use/ land cover of the receiving terrestrial landscape by applying the appropriate curve numbers (McCuen 1998). The model served two purposes. It was used to assess the relationship between precipitation and

runoff under present day land use/ land cover conditions. It was also used to estimate runoff from simulated precipitation events using predicted land use/ land cover and climate change conditions. Freshwater inflow into Lavaca Bay was adjusted, using the Aransas River watershed as a reference, to reflect the estimated effects of climate change and watershed development.

Precipitation – Nexrad.
Daily Nexrad precipitation shape files for the year 2004 were downloaded from the National Weather Service website in an undefined geographic projection. Each file was then unzipped four separate times to yield a daily shapefile of precipitation points on a 4 km by 4 km grid for the entire Southeastern United States. The projection for each precipitation shapefile was defined according to the Hydrologic Rainfall Analysis Project (HRAP) coordinate system using ArcInfo 9.0. The point shapefiles were then cropped to the extent of the Texas state border and subsequently spatially joined to a HRAP coordinate grid using their HRAP coordinates. This resulted in a gridded polygon shapefile with each cell's precipitation value taken from the point located at its center. The precipitation polygon file was then cropped to the extent of the Aransas River watershed which had been previously delineated from a digital elevation map. The precipitation polygon file was finally converted to a Raster grid file on a 30 m by 30 m scale which was bounded by the Aransas River watershed polygon.

Watershed characteristics.
Two components of the watershed were needed to convert precipitation to runoff. The first component was the land use/ land cover characteristics of the Aransas River watershed. Land use/ land cover data was downloaded as a Raster grid file from the United States Geological Survey's (USGS) 1992 National Land Cover Dataset. The USGS land use/ land cover data has 30 m by 30 m grid cells. The land use/ land cover data was converted to a polygon shapefile on a scale of 300 m by 300 m. The second component was the soil types in the Aransas River watershed. This data was retrieved from the National Resource Conservation Service's State Soil Geographic (STATSGO) database website as a polygon shapefile. The soil polygon shapefile was cropped to the extent of the Aransas River watershed and split into one shapefile per soil type (Fig. 4.1). The Aransas River watershed soils are characterized into three types: soil type B (silt loam or loam), soil type C (sandy clay loam) and soil type D (clay loam, silty clay loam, sandy clay, silty clay or clay). The land use/ land cover polygon data set attribute table was then edited so that curve number (CN) estimates (McCuen 1998) for each land use/ land cover type were added for each soil type. A polygon shapefile of CN's for each soil type was created and cropped to the extent of each soil type within the watershed. These CN by soil type shapefiles were converted to Raster grid files on a 30 m by 30 m scale and then the mosaic function of ArcInfo 9.0 combined them into one CN Raster grid file for the entire Aransas River watershed (Fig. 4.2).

Runoff.
Calculations of runoff take into account both precipitation and the CN of the receiving land surface. Runoff depth was calculated in every 30 m by 30 m grid cell for each day of 2004 using the following equations:

$Q = (P - 0.2\ S)^2 / (P + 0.8\ S)$
$S = (1000/CN) - 10$
$Q = 0$ at $P < I_a$

Where Q is runoff depth in inches d^{-1}, P is precipitation in inches d^{-1}, I_a is the initial abstraction number, and S represents the proportion of precipitation that will runoff from a particular land use/ land cover located on a particular soil type, with CN being the curve number.

Runoff depths in inches d^{-1} were converted to runoff in $m^3\ d^{-1}$ by multiplying them by a conversion factor of inch to meter of 0.0254 and by 900 m^2 (the area of each 30 m by 30 m grid cell). Each cells daily runoff was then summed to calculate total daily runoff from the entire Aransas River watershed and also from just the portion of the watershed that drains to the point where USGS has a river flow gauge.

Hydrological model verification.
The hydrological model runoff results were compared to observed river flow rates in the Aransas River (USGS gauge 08189700 near Skidmore, TX). Daily averaged river flow rates in $ft^3\ s^{-1}$ were converted to an average river flow in $m^3\ d^{-1}$ by multiplying by 2445.12. Daily modeled runoff was assumed to be delivered to the river gauge location with no lag interval. Each daily runoff sum above the gauge was compared to that same day's average river flow using linear regression analysis. This analysis allowed a comparison of runoff estimates to observed river flows. A close fit between simulated runoff and observed river flow allows prediction of freshwater inflow in the Aransas River under various climatic and watershed development scenarios.

Climate change scenarios.
Climate change predictions estimate a 5-25% increase in precipitation over the next hundred years (Houghton et al. 2001, Twilley et al. 2001). This precipitation is predicted to arrive in more intense pulses increasing the incidence of high river flows by up to 21%. Temperature is predicted to increase between 2.5 ^oC and 5 ^oC with a larger change in summer months than in winter (Houghton et al. 2001, Twilley et al. 2001). To assess the affect of these climate change predictions, NEM was simulated during average 2004 daily precipitation event conditions (1.63 cm) over the Aransas River watershed. Freshwater runoff magnitudes were then adjusted to reflect potential climate change effects on precipitation (average daily precipitation event increased to 2.45 cm) during the next one hundred years and NEM simulated under the modified conditions. These simulations will allow assessment of the effects of climate change on NEM during inflow events.

Watershed development scenarios.
Watershed development over the next one hundred years could result in more land becoming impervious as land use/ land cover is modified to accommodate the growing housing and agricultural needs of an increasing human population. Population in Texas increased by 17.8 million people between the year 1900 and 2000 (US Census 2000). This constitutes a 100-200% increase per 50 years. Texas population densities are now between 75 and 200 people $mile^{-2}$. San Patricio and Bee counties, which are located in the Aransas River watershed, had 14.3% and

28.7% population growth respectively between 1990 and 2000. Population numbers in San Patricio County, which is located in the lower portion of the Aransas River watershed, was predicted to double from around 60000 to 120000 people between 1990 and 2050. Actual population numbers in San Patricio County increased as predicted from 58749 to 67138 people between the years 1990 and 2000. Population densities in the Aransas River watershed portion of San Patricio and Bee counties are approximately 40 people mile^{-2} (US Census 2000). The national average is 80 people mile^{-2}, and urbanized watersheds such as those in Maryland have population densities as high as 542 people mile^{-2}. Thus the Aransas River watershed is considered rural and has < 2% urbanized land cover. Urbanization with increased population numbers produces more impervious surfaces and results in more water running off into the river and stream systems of a watershed during precipitation events (US Soil Conservation Service 1986). During large precipitation events in unrestricted coastal watersheds without dams most of this runoff will flow downstream to become freshwater inflow to estuaries. The influence of watershed development on NEM was assessed by simulating a present day precipitation event (1.63 cm) under current land use/ land cover characteristics in the upper Aransas River watershed. Precipitation was related to runoff using the hydrological model for the Aransas River watershed. Land use/ land cover was then adjusted to simulate 100% and 200% increases in urbanization in the upper watershed (Fig. 3) yielding approximately 3 % and 4.5 % urbanized land cover. This percent urbanization can still be considered rural as most urbanized watersheds have >10 % urbanized land cover. NEM response to urbanization was then simulated under present day precipitation event conditions and compared to present day simulated results.

Combined effects of potential climate change and watershed development.
The effects on NEM from predicted modifications of freshwater inflow due to the combination of climate change and watershed development were assessed. Predicted freshwater inflows during precipitation events may be up to 25 % higher than today, and could increase by greater than 25 % as freshwater inflow arrives in larger pulses due to urbanization (US Soil Conservation Service 1986, Houghton et al. 2001, Twilley et al. 2001). NEM during predicted average freshwater inflow event conditions into Copano Bay, due to the combined effects from potential climate change and urbanization, were simulated and compared to present day NEM results. The ratio of present day to predicted freshwater inflow into Copano Bay was then used to assess potential changes in Lavaca Bay NEM due to similar ratios of change in freshwater inflow to that bay. Estimates of the predicted change in Lavaca Bay NEM also incorporate the potential 5 ^{o}C increase in temperature predicted by climate change models for Texas coastal areas (Houghton et al. 2001).

Results

Environmental conditions.
Temperatures followed a seasonal cycle with lows during winter months and highs during summer (Table 1). Daily average water temperature ranged from a low of 16 ^{o}C in Copano Bay and 13 ^{o}C in Lavaca Bay during January to a high of 30 ^{o}C in both bays during July. Water temperatures during deployments tended to remain stable, but some weeks had daily temperature changes of about 1 ^{o}C, and on a few occasions temperatures decreased by as much as 7 ^{o}C in 24-hours. Rapid daily temperature changes were mostly associated with decreasing salinity (-2 ppt

d^{-1}) during April storm events. Salinity was highest during January-March 2004. Large freshwater inflows (up to 44×10^6 m^3 d^{-1}) beginning in April and continuing through May in Copano Bay and June in Lavaca Bay resulted in large decreases in salinity (down by as much as 22 ppt) in Copano and Lavaca Bays. Salinity was at or near zero by May. Copano Bay remained relatively fresh throughout the year with an annual average salinity of 6 ppt. Freshwater inflow into Copano Bay decreased back to base flow levels (~2×10^5 m^3 d^{-1}) in June and was low throughout most of the remaining year. Salinity did not recover to pre-spring levels until September due to the long residence time in Copano Bay. High freshwater inflow into Lavaca Bay started in April and continued through the beginning of July, reaching an average inflow of 86×10^5 m^3 d^{-1} in June.

NEM response to environmental conditions.

Copano Bay had fairly stable NEM values (mean = -1.12 mg O_2 l^{-1} d^{-1}, SE = 0.10 mg O_2 l^{-1} d^{-1}) over a range of temperatures (16-30 ^oC) (Fig. 4a). Copano Bay NEM results were not significantly related to temperature ($p = 0.4869$, $R^2 <0.01$), and showed little response over a range of salinity (0-14 ppt) ($p = 0.1131$, $R^2 = 0.01$) (Fig. 4b). NEM in Copano Bay increased from a basal value of around -1 mg O_2 l^{-1} d^{-1} during the lowest inflows to around 0 mg O_2 l^{-1} d^{-1} at ten-day cumulative inflows less than 2×10^6 m^3 (Fig. 4c). Less data exists for evaluating NEM under ten-day cumulative inflows above 2×10^6 m^3, but the results show a decreasing trend in NEM down to values around -4 mg O_2 l^{-1} d^{-1} as cumulative inflow increased to 7×10^6 m^3. The limited samples collected at ten-day cumulative freshwater inflows above 7×10^6 m^3 in Copano Bay occurred during one particularly large inflow event in April (Fig. 4c). NEM was initially autotrophic during the first day of the freshwater pulse and then dropped to around -4.00 mg O_2 l^{-1} d^{-1} after a peak one-day inflow on April 12th around 600×10^3 m^3 d^{-1} and then became autotrophic again (1.00 mg O_2 l^{-1} d^{-1}) for a day or two as one-day freshwater inflow subsided to around 150×10^3 m^3 d^{-1} (Fig. 5). NEM values then returned to more normal values (-1 to -2 mg O_2 l^{-1} d^{-1}) as freshwater inflow slowed to 80×10^3 m^3 d^{-1}.

Lavaca Bay NEM results from 2002-2004, when combined, had significant linear relationships with temperature, salinity, and cumulative ten-day freshwater inflow (Fig. 6a-c). Salinity, however, was strongly influenced by ten-day cumulative freshwater inflow with an exponential drop from around 22 ppt to 5 ppt as ten-day cumulative freshwater inflow increased from base flows of 1×10^6 m^3 to around 20×10^6 m^3 ($p < 0.001$, $R^2 = 0.76$) (Fig. 7a). Also, salinity above 20 ppt mainly occurred during winter months when water temperatures were below 15 ^oC (Fig. 7b). Since these three factors are interrelated, a multiple linear-regression analysis was used to assess each factors influence on NEM while accounting for the influences from the other factors. Salinity alone accounted for 40% ($p < 0.001$, $R^2 = 0.40$) of the variance in NEM explained by the multiple linear-regression model ($R^2 = 0.45$). Temperature ($p = 0.073$, additional $R^2 = 0.03$) and cumulative 10-day freshwater inflow ($p = 0.071$, additional $R^2 = 0.02$) together accounted for another 5% of the variance in NEM. The combination of these three factors, thus, explained 45% of the variability in Lavaca Bay NEM and estimates of NEM using the complete three-parameter model had a standard error of 1.64 mg O_2 l^{-1} d^{-1} (Fig. 8).

Copano Bay NEM model calibration.
The fit of the non-linear conceptual model to observed Copano Bay NEM values was assessed under various days of cumulative inflow (Fig. 9). It was determined that a 17-day cumulative inflow provided the best model fit ($R^2 = 0.15$) to observed NEM values in Copano Bay. Adjustment of the previously used 10-day cumulative inflow value in chapter 1 and 2 did not change the pattern of response of NEM to increased freshwater inflow, but changing to a 17-day cumulative flow helps to reduce the error of estimates made from the NEM model. Even with this adjustment, the predictive power of the Copano Bay NEM model remained very low and large changes in NEM would be needed to find significant differences in NEM due to the effects of different environmental conditions.

NEM results from Copano Bay were used for parameter calibration.

NEM (mg O_2 l^{-1} d^{-1}) = (f_x) $\sum$ 17-day cumulative freshwater inflow (m^3)
NEM (mg O_2 l^{-1} d^{-1}) = a (exp (-0.5 (ln (X/c)) /b^2))
a = 9.3465 (mg O_2 l^{-1} d^{-1})
b = 2.512 (mg O_2 l^{-1} d^{-1})
c = 960396 (m^3)
X = cumulative 17-day freshwater inflow (m^3)

Parameters were set to values derived from the fit between the non-linear model and observed NEM over the observed freshwater inflows range. Using these parameters, simulated NEM increases from -1.37 mg O_2 l^{-1} d^{-1} at cumulative 17-day freshwater base flow conditions of 352586 m^3 to a peak of -0.65 mg O_2 l^{-1} d^{-1} at 955003 m^3 (Fig. 10). As cumulative 17-day freshwater inflow continues to increase, NEM steadily decreases to an estimated low of -5.50 mg O_2 l^{-1} d^{-1} at cumulative 17-day inflows close to 20×10^6 m^3. This agrees well with the most heterotrophic values observed during the largest Copano Bay freshwater inflow event that occurred in April 2004 (Fig. 5) as well as the heterotrophic values observed using alternative methods for quantifying respiration and NEM (Russell 2005).

Copano Bay NEM model sensitivity.
The sensitivity of the NEM model was assessed by comparing the shapes of the curves relating NEM over a range of freshwater inflow conditions (17-day cumulative inflow between 100,000 – 15 million m^3) while modifying each parameter separately. The shape of the NEM response curve was calculated at the following range of parameter values: a = 5, 10, and 15 (mg O_2 l^{-1} d^{-1}); b = 1.5, 2.5, and 3.5 (mg O_2 l^{-1} d^{-1}); and c = 500000, 1000000, and 1500000 (m^3). Parameter "a" controls the peak value of NEM with higher values yielding more autotrophic conditions (Fig. 11a). Parameter "b" controls the skewness of the curve and how fast NEM declines from its peak value with increased inflow (Fig. 11b). Parameter "c" controls which freshwater inflow value peak NEM will be located at (Fig. 11c).

Copano Bay NEM model estimation error.
Simulated NEM results were compared to those observed at Copano Bay during 2004 (Russell 2005). The overall fit of the non-linear model to observed NEM results was relatively low ($R^2 = 0.15$) but the relationship was significant ($p < 0.0001$). The standard error of estimating NEM

using the model was 1.46 mg O_2 l^{-1} d^{-1}. This standard error requires a minimum change of ± 2.39 mg O_2 l^{-1} d^{-1} to produce a significantly different NEM result with a 95% confidence interval.

Precipitation.
Precipitation over the Aransas River watershed varied widely during 2004 (Fig. 12). Daily precipitation, falling over the sub-basin above the USGS river gauge 08189700, ranged from zero measurable precipitation up to as much as 11 cm d^{-1} (4.33 inches). Large daily precipitation values occurred mostly during April and May storm events. Precipitation was notably low during the first three months of 2004. Precipitation events (>1.65 cm d^{-1}) occurred seven times between June and December.

Runoff potential.
The interaction of soil types with land use/ land cover characteristics within the Aransas River watershed yielded distinct areas of potential precipitation infiltration. The three soil types influenced infiltration potential within each of their respective areas. This is especially evident in the upper Aransas River watershed (Fig. 3). Infiltration potential was assessed using initial abstraction (I_a) values for each 30 m by 30 m grid cell. Values for I_a ranged from 0 to 1.4, with 1.4 being the highest infiltration potential. Soil type B (silt loam or loam) yielded an I_a value (mean ± standard error) of $0.84 \pm 0.66\times10^{-3}$. Soil type C (sandy clay loam) yielded an I_a value of $0.59 \pm 0.525\times10^{-3}$. Soil type D (clay loam, silty clay loam, sandy clay, silty clay or clay) yielded an I_a value of $0.23 \pm 0.584\times10^{-4}$. Infiltration potential of different land use/ land cover types also influenced runoff potential. Values for I_a ranged between 0 in urbanized areas to 1.4 in forested or shrub land areas.

Simulated runoff and observed river flow.
Simulated runoff and observed river flow from the Aransas River watershed area draining to the USGS gauge location had a wide range during 2004. Simulated runoff during precipitation events ranged between 4×10^3 m^3 d^{-1} and 4.4×10^6 m^3 d^{-1}. Observed river flows during the same precipitation events ranged from 20×10^3 m^3 d^{-1} to 3.3×10^6 m^3 d^{-1}. Simulated monthly runoff matched up well with monthly observed river flow ($p < 0.0001$, $R^2 = 0.99$) (Fig. 13). The river flow response to increased runoff was more scattered at lower flows ($p = 0.0035$, $R^2 = 0.68$). This may be due to differences in the effects of evaporation during smaller precipitation events. The average present day runoff event was calculated as 1,378,730 m^3 d^{-1}. Assuming events last for 3 days with one event during every 17 days; these runoff events would yield a total 17-day cumulative simulated river flow of around 4×10^6 m^3 passing the USGS gauge in the Aransas River watershed before it flows into Copano Bay.

Simulated runoff from the area draining towards the USGS river flow gauge also compared well with simulated total watershed runoff ($p < 0.0001$, $R^2 = 0.91$) (Fig. 14a). Outliers influenced this relationship and so the analysis was re-run without them. The relationship between simulated gauged runoff and simulated total watershed runoff suggests that actual river inflow into Copano Bay is approximately 3-4 times larger than observed USGS river flow. This is not surprising, considering the USGS gauge only measures river flow from about a third of the total area of the Aransas River watershed. Estimated freshwater inflows from the entire Aransas River watershed are, thus, roughly equivalent to those measured from the Lavaca River watershed, which has its

gauge much closer to the point of river discharge than the one in the Aransas River. A slightly less strong, but still significant relationship existed with the outliers removed ($p < 0.0001$, $R^2 = 0.54$) (Fig. 14b). More scatter existed as runoffs increased, implying that as a larger proportion of the watershed receives precipitation, differences between sub-basin land use/land cover, soil types, and their influence on runoff may become magnified. Thus, spatially variable precipitation patterns, such as provided by Nexrad, may be even more important for basin scale runoff estimates than those from smaller sub-basins.

Climate change scenarios.

NEM was simulated under upper Aransas River watershed present day cumulative 17-day freshwater event conditions of 4,136,190 m^3 and predicted future event conditions 91% higher at 7,915,263 m^3 (Table 2). This assumes an average three days of precipitation of 1.65 cm d^{-1} and an average three days of predicted precipitation of 2.46 cm d^{-1} occurring in a 17-day period. The resulting simulated present day and predicted NEM values were not significantly different at -2.11 and -3.43 mg O_2 l^{-1} d^{-1} respectively (t-test, $p > 0.1$). The model, however, suggests the potential of 60% more heterotrophic conditions with increased precipitation. To produce significantly different present day and predicted NEM results with a 95% confidence interval, 17-day cumulative freshwater inflow conditions would have to reach 10×10^6 m^3. This would require another 21% increase in cumulative 17-day freshwater inflow above those predicted in this study.

Watershed development scenarios.

Urbanized areas in the Aransas River watershed currently equal 1.5% (1975 of 134399, 900 m^2 cells) of the total area. These urbanized areas are mostly surrounded by land use/ land cover areas with I_a values of 0.2 to 0.4. The city of Beeville, Texas, which is located in the upper Aransas River watershed, is surrounded by land with higher infiltration potential (I_a from 0.8 to 1.4) (Fig. 3). Increased urbanization would result in a 0 - 71% decrease in infiltration potential in converted lands. The percent decrease depends on which soil type is developed. A present day average precipitation runoff event of 1.63 cm (0.64 inches) homogeneously applied over the upper Aransas River watershed would result in runoffs of 1378730, 1406269, and 1642889 m^3 d^{-1} under present day, low urbanization, and high urbanization conditions respectively (Table 3). These daily flows would yield, again assuming a three-day precipitation event with no other events occurring in the previous 17 days, a cumulative 17-day freshwater inflow of 4136190, 4218807, and 4928667 m^3 respectively (increases of 2% and 19%). NEM is estimated to change from a present day value of -2.11 mg O_2 l^{-1} d^{-1} to -2.14 and -2.44 mg O_2 l^{-1} d^{-1} (1% and 16% more heterotrophic) during low and high urbanization scenarios respectively in the upper Aransas River watershed. This represents a non-significant change in NEM with urbanization (t-test, $p > 0.1$)

Combined effects of climate change and watershed development.

Urbanization and climate change have similar effects on NEM in that they both tend to increase freshwater inflow. The combined influence of increased precipitation and increased impervious surfaces due to urbanization in the Aransas River watershed could potentially increase freshwater inflow to Copano Bay. NEM was simulated under future cumulative 17-day freshwater inflow conditions, due to the combination of climate change and watershed urbanization, of 9431796 m^3

(an 128% increase) (Table 2). The simulated NEM under these predicted conditions is 81% more heterotrophic at -3.82 mg O_2 l^{-1} d^{-1}. The combination of urbanization and climate change, thus, almost causes enough freshwater inflow to produce a significantly more heterotrophic ecosystem. An NEM 8% more heterotrophic than predicted (-3.98 mg O_2 l^{-1} d^{-1}) would be significantly different from a present day average event NEM of -2.11 mg O_2 l^{-1} d^{-1} (t-test, $p < 0.1$).

Predicted NEM in Lavaca Bay.

The Lavaca Bay NEM model is influenced by not only freshwater inflow, but also salinity, and temperature. A similar analysis of how many days cumulative freshwater inflow produce the best model fit showed little improvement (Data not shown) from changing from the previously used 10-day period (Russell 2005). The calibrated NEM model equation for Lavaca Bay is as follows:

$$\text{NEM (mg } O_2\ l^{-1}\ d^{-1}) = -1.82\ (\text{mg } O_2\ l^{-1}\ d^{-1}) + 2.62\times10^{-8}\ (\text{mg } O_2\ l^{-1}\ d^{-1}\ m^{-3})\ FW - 0.07\ (\text{mg } O_2\ l^{-1}\ d^{-1}\ {}^{o}C^{-1})\ T + 0.21\ (\text{mg } O_2\ l^{-1}\ d^{-1}\ ppt^{-1})\ S$$

Where FW is cumulative 10-day freshwater inflow (m^3), T is temperature (^oC), and S is salinity (ppt).

Average cumulative 10-day freshwater inflow conditions in Lavaca Bay are around 10×10^6 m^3 (Fig. 7). Salinity at this average present day inflow is approximately 15 ppt (Fig. 7). Application of a proportional increase in freshwater inflow to Lavaca Bay inflow, as calculated from predicted climate change and watershed development conditions in Copano Bay, results in a predicted future event inflow into Lavaca Bay of around 20-30$\times10^6$ m^3. With more precipitation over the Lavaca Bay watersheds, however, the infiltration capacity of soil there will become saturated more often. This could potential increase the runoff and subsequent inflow into Lavaca Bay during the predicted increased precipitation events. A predicted future freshwater inflow event was, thus, estimated to produce around 40×10^6 m^3 over a 10 day period, which is within the range of those observed during 2004 (Fig. 7). This magnitude of freshwater inflow can drop salinity to around 0-2 ppt (Fig. 7). A 5 ^oC temperature increase as predicted for this region by climate change model predictions (Houghton et al. 2001, Twilley et al. 2001), was also applied to the model. The multiple linear-regression Lavaca Bay NEM model predicts an NEM of -0.16 mg O_2 l^{-1} d^{-1} during present day environmental conditions and an NEM 1563% more heterotrophic at -2.66 mg O_2 l^{-1} d^{-1} during predicted future environmental conditions due to climate change and watershed development (Table 3). The more heterotrophic simulated NEM value during future conditions, however, is not significantly different than simulated present day NEM (t-test, $p > 0.1$). Similar to what was found in Copano Bay, a 7% more heterotrophic NEM value than predicted by the Lavaca Bay NEM model (-2.85 mg O_2 l^{-1} d^{-1}) is required to produce a significantly different NEM than present day. All that would be required to produce a significantly more heterotrophic NEM would be to model the predicted salinity as 0 ppt instead of the 1 ppt used in this study. Thus, with the large amount of variability observed in these systems, the potential for more heterotrophic conditions in Lavaca Bay exists.

Discussion

NEM models.

Observed NEM values from Copano Bay followed the shape of the conceptual NEM model (Fig. 10). NEM data, however, fit the conceptual model better when they where compared to cumulative freshwater inflow from multiple previous days (Fig. 9). A 10-day period was previously identified as the time interval needed to capture the response of estuarine benthic community structure to freshwater inflow events (Kalke and Montagna 1991; Montagna and Kalke 1992). The current research identified the 17-day period as the best time interval to capture the response of estuarine metabolism to freshwater inflow events in Copano Bay.

Copano Bay had an initially increasingly autotrophic NEM with increased freshwater inflow (Fig. 10). This type of ecosystem response to increased inflow is likely indicative of increased nutrient loading. Photosynthetic organisms, which produce oxygen as they produce energy during photosynthesis, can combine this energy with nutrient loads to produce more biomass. Increased loading of nutrients was concluded to result in more autotrophic NEM's in marine mesocosms (Oviatt et al. 1986) and in natural systems with high nutrient inputs (D'Avanzo et al. 1996). NEM was significantly more autotrophic under conditions of higher nitrogen loading in a study of 42 sites in 22 National Estuarine Research Reserves (Caffrey 2004).

The initially increasingly autotrophic NEM in Copano Bay, with increased freshwater inflow, was followed by a peak and then a subsequently decreasing NEM trend as freshwater inflow continued to increase (Fig. 10). Heterotrophic ecosystem responses have been found in systems dominated by organic carbon loading (Smith and Hollibaugh 1993, 1997, Cai et al. 1999, and Raymond et al. 2000). NEM was heterotrophic in these organic matter dominated systems. Overall, Copano Bay remained relatively heterotrophic throughout 2004. This result was also found at most NERR sites (Caffrey 2004). Copano Bay, thus, follows the general trend of shallow water estuaries being net heterotrophic.

NEM model results were significantly related to observed NEM values in Copano Bay but the non-linear relationship had a lot of scatter ($p < 0.0001$ $R^2 = 0.15$) (Fig. 10). The large amount of dispersion in observed NEM values, especially during low freshwater inflow conditions, may be due to factors not included in the model. One factor that may influence NEM during low freshwater inflow periods is turbidity. The balance between primary production and respiration could be influences by reduced light availability due to high turbidity in Copano Bay. High turbidity conditions exist for the majority of the year in Texas bays (NOAA 1997). Higher turbidities and subsequent lower available photosynthetically active radiation (PAR) with depth usually exist in water close to river discharge as compared to water closer to an estuary mouth (Kennish 1986). Secchi depths in Copano Bay ranged between 0.5 m during winter to 0.2 m during summer (personal observation). Surface irradiance, and thus PAR, also varied with season, and on shorter time scales associated with clouds (Russell 2005). Surface irradiance levels dropped markedly during the April and May storm events (Russell 2005), but little effect was observed on NEM. Turbidity and in-situ irradiance measurements need to be compared to simultaneously calculated NEM values before these relationships can be added to the model.

Another factor that may influence NEM could be the balance between nutrients and organic matter in freshwater inflow. A determining factor of the balance between autotrophy and heterotrophy is the balance between organic carbon and nutrient loading (Kemp et al. 1997, Eyre and McKee 2002). Nutrient and organic matter loads in the micro-tidal estuaries of Texas are mainly delivered through freshwater inflow (Whitledge 1989a, 1989b, Longley 1994). The pulsing nature of freshwater inflow sporadically delivers nutrients and organic matter at high magnitudes but for short durations. As freshwater inflow increases, the ratio of nutrients to organic matter within the inflow will also change, decreasing as more terrestrial organic particulates are eroded from the watershed (Jones et al. 1986, Parker et al. 1989). The NEM model predicts an increasing NEM with small increases from base freshwater inflow conditions (Fig. 10). The model then predicts more heterotrophic conditions with increasing flow. It is possible that a shift from high to low nutrient to organic matter ratios occurs as freshwater inflow increases from base flow levels. Quantification of nutrient and organic matter loading during a range of freshwater inflow conditions would be an important addition to the NEM model.

NEM event dynamics during a very high magnitude freshwater inflow in Copano Bay resulted in a similar response, overall, to that of NEM during increased cumulative flow (Fig. 5). Days with net heterotrophic rates as large as -5 mg O_2 l^{-1} d^{-1} (Fig. 5) dominated the ecosystem response when all daily NEM were averaged (average NEM = -1.37 mg O_2 l^{-1} d^{-1}). Copano Bay exhibited an initial autotrophic response the day the freshwater pulse began and then became relatively heterotrophic as freshwater inflow rates continued to increase. This response may be due to an initial phytoplankton response to dissolved nutrient inputs followed by large magnitude benthic respiration rates as organic matter loads become the main constituent in freshwater inflow. Within a few days as the freshwater pulse subsided, however, the system became relatively autotrophic again, even becoming net autotrophic for a few days. This response suggests that the more marine planktonic community may have been flushed downstream during the large freshwater pulse and an autotrophic response by phytoplankton may not have occurred until the planktonic community reestablished itself as inflow subsided. These event dynamics may be responsible for some of the observed scatter in NEM values from Copano Bay as enough days of sampling (n = 69) were completed there that various events must be included in the results.

Copano Bay receives freshwater inflow from three point sources. Error in the NEM model may relate to freshwater influences from other point sources than just the Aransas River. This study focused on the influence of the Aransas River, however, and so stations were located close to its discharge point. Copano Creek discharges into the far northeast section of Copano Bay. Freshwater inflow from Copano Creek would have to travel approximately 16 km directly across the bay in order to influence the results of the present study. The Mission River discharges into the tertiary Mission Bay before freshwater inflow can have any influence on the northwestern area of Copano Bay. Water from the Mission River would have to travel approximately 8 km before it could have any influence on the closest station in the present study. By comparison, the Aransas River discharges directly into southeastern Copano Bay, and this discharge is less than one km from the closest station in the present study and is less than 3 km from the farthest station. The Aransas (8.10 cm) and Mission River (8.47 cm) watersheds receive similar amounts of annual precipitation (Quenzer 1998). The Aransas and Mission River watersheds, however, have rather different agriculture (58% and 25% respectively) and rangeland land use (38% and

73% respectively). Even with these differences in land use, though, nutrient loads from the two rivers are remarkably similar. Total nitrogen loads for Aransas and Mission River are 213,314 and 239,843 kg yr^{-1} respectively, and phosphorus loads are 60,900 and 57,801 kg yr^{-1} (NOAA 1997). Tidal range in Copano Bay is severely dampened by the distance from the Gulf of Mexico, and is usually less than one-half meter (Powell et al. 1997). Residence time in Copano Bay has been calculated to be as high as 3 years (Longley 1994) suggesting that water circulation and replacement by river inflows is very slow. Thus, the influence of the Mission River freshwater inflow on the area of station locations in the present study is significantly reduced by the combination of retention in Mission Bay, little tidal circulation, and the long distance between discharge and station locations. The influence that Mission River freshwater inflow could have on the results of the present study could potentially increase during precipitation events producing increased inflows. Any increase in potential influence from the Mission River, however, would coincide with a similar increased inflow from the Aransas River which would push water away from the present study station locations. Thus, the influence of Copano Creek and the Mission River on NEM results in the present study is assumed to be insignificant when compared to that from the Aransas River.

Lavaca Bay NEM results were similar to Copano Bay results in that they generally became more heterotrophic at higher freshwater inflows. The dominant factor explaining changes in NEM was salinity ($R^2 = 0.40$), with more heterotrophic conditions during low salinities (Fig. 6a). Temperature and freshwater inflow, which were somewhat interrelated to salinity (Fig. 7), added a small ($R^2 = 0.05$), but significant, amount of explanatory power to the overall NEM model (Fig. 6b-c). With 45% of the variability in NEM explained by these three factors, the Lavaca Bay NEM model provides much more confidence in estimates of NEM during changing environmental conditions than the Copano Bay NEM model. It is interesting to note, however, that the two models generally agree about the consequences of increases in freshwater inflow due to climate change and watershed development. The Lavaca Bay NEM model was calibrated with NEM values measured over a much wide range of environmental conditions, especially salinity, than those in Copano Bay. This suggests that the relationship between NEM and salinity in Copano Bay may not have been observed due to lack if sampling over a wider range of conditions. A comparison of the shape of the Lavaca Bay and Copano Bay model curves illustrates the similarities and differences of the two ecosystems. Copano Bay, although overall it is more heterotrophic than Lavaca Bay, shows an autotrophic response at moderate flows. A combination of nutrient ratios, organic carbon loads, and residence times may explain this difference. Lavaca and Copano Bays have very similar phosphorus inputs, but nitrogen loads are higher in Lavaca than Copano Bay. Thus, the nitrogen to phosphorus ratio of nutrients loads into Lavaca Bay is higher (6.63) than that in Copano Bay (4.88) (Longley 1994). The low residence time of Lavaca Bay (77 days) as opposed to Copano Bay (1102 days), however, means that nitrogen is much more available in Copano Bay (residence time weighted nitrogen = 5.83 g m^{-3} yr^{-1}) than Lavaca Bay (residence time weighted nitrogen = 0.66 g m^{-3} yr^{-1}) (Longley 1994). The ratio of organic carbon to nitrogen loads, which are identical in Lavaca and Copano Bays (C:N = 6.16), become much lower in Copano Bay (2.06) than Lavaca Bay (29.70) when nitrogen load availability is adjusted to reflect the differences in residence time between the two bays. The slight autotrophic response at moderate flows in Copano Bay may be due to this higher

availability of nutrients which may dominate metabolic processes until organic carbon loads increase during higher freshwater inflows (Jones et al. 1986, Parker et al. 1989).

Hydrological model.
The hydrological model links spatially and temporally dynamic precipitation data with spatially variable landscape characteristics to produce runoff and freshwater inflow to the downstream estuary. This model was created for the Aransas River watershed. The thrust of this modeling effort was to estimate freshwater inflow into Copano Bay under predicted future climate change and watershed development conditions.

The SCS curve number method (US Soil Conservation Service 1986) for transforming precipitation to runoff was originally designed to assess flooding events due to large magnitude precipitation events in urban watersheds. The original SCS curve number method is stated to have less accuracy when runoff is less than 0.5 inch. The method also assumes zero runoff at precipitation less than the initial abstraction value (I_a), which here is assumed to be S multiplied by 0.2. The SCS curve number method does not account for changing I_a values, which are dependent on soil moisture content (US Soil Conservation Service 1986). Soil moisture content may have significantly changed in the Aransas River watershed during 2004, especially considering the large spatial and temporal variability in precipitation that was observed. Even with the potential short comings of using the SCS curve number method for this type of modeling, the rainfall-runoff model performed quite well. Simulated monthly runoff was strongly related to monthly observed river flows ($p < 0.0001$, $R^2 = 0.99$) (Fig. 13). The model showed more error when simulated monthly runoff and observed river flow were compared while excluding the very high inflow months of April and May ($p = 0.0035$, $R^2 = 0.68$). This increased error could have arisen due to the lack of an evaporation term in the rainfall-runoff model. The significant fit between precipitation, runoff, and observed river flow indicates that the combination of Nexrad precipitation data and the SCS curve number method can be used to model freshwater inflow under various climate change and watershed development scenarios.

Climate change and watershed development.
NEM in Copano Bay was more heterotrophic (mean = -3.43 mg O_2 l^{-1} d^{-1}), but not statistically significantly so, during predicted future precipitation conditions than during present day conditions (mean = -2.11 mg O_2 l^{-1} d^{-1}) (Table 3). NEM becomes approximately 63% more heterotrophic with a 91% increase in cumulative 17-day freshwater inflow. Watershed development had less of an effect on net ecosystem metabolism than climate change (Table 3). It is important, however, to realize that the model only accounts for changes in water quantity arising from urbanization. It does not take into account changes in constituent concentrations in that inflow or water quality. It also does not take into account future water diversions or ground water flows.

The level of urbanization in the Aransas River watershed is very low (currently 1.5%). This is one of the many reasons Copano Bay provides a good study site for assessing the consequences of potential watershed development. The NEM model predicts that Copano Bay NEM, after an average precipitation event, is currently around -2.11 mg O_2 l^{-1} d^{-1} and could change to -2.44 mg O_2 l^{-1} d^{-1} with a 200% increase in urbanization in the upper Aransas River watershed. This is not

a statistically significant change but these estimates of NEM in Copano Bay provide a baseline for comparison to other more urbanized watersheds. It is also important to realize that the present study does not attempt to integrate changes in constituent loads into estimates of NEM. Increased agricultural runoff or sewage treatment discharge both effect the response of an estuary to freshwater inflow. For example, the trend of increased extreme heterotrophic or autotrophic conditions with increased urbanization is supported by findings from urbanized watersheds such as those flowing into the Nueces River Estuary (Borsuk et al. 2001), Chesapeake Bay estuaries (Kemp et al. 1997, Hale et al. 2004), the Husdon River Estuary (Howarth et al. 1992 and 1996), San Francisco Bay (Jassby et al. 1993), and Tomales Bay (Smith et al. 1991). Many of these estuaries and bays have greater than 20% urbanized land in the corresponding watersheds, and eutrophication was concluded as the most influential factor on metabolic rates in these estuaries. Dissolved inorganic nitrogen to total organic carbon ratios in heterotrophic estuaries that act as carbon sinks are typically < 0.5 (Kemp et al. 1997). Texas estuaries fall well within this range at 0.16-0.33 (Longley 1994, Russell 2005). Extremely autotrophic estuaries tend to be carbon sources and usually have dissolved inorganic nitrogen to total organic carbon rations > 0.5. Dissolved oxygen conditions, in the receiving water bodies of heavily urbanized watersheds, often become hypoxic (Kemp et al. 1992). Summertime anoxia/hypoxia in the bottom waters of Chesapeake Bay has occurred since the mid-1930's (Newcombe and Horne 1938), and has spatially and temporally increased in recent decades (Heinle et al. 1980, Officer et al. 1984). Historical increases in spatial and temporal occurrences of hypoxic events are attributed to anthropogenic inputs in many coastal regions (Jørgensen 1980, Rosenberg 1985, Andersen and Rydberg 1988). Years with higher freshwater inflows support elevated respiration as a result of nutrient loading, phytoplankton assimilation, and subsequent organic matter remineralization, which can lead to rapid depletion of dissolved oxygen (Boynton et al. 1982). The present study provides evidence that, as relatively undeveloped watershed become urbanized; NEM becomes more heterotrophic as a consequence of increased freshwater inflow. Increased constituent loading of either nutrients or organic matter, would, undoubtedly modify this response. Future research efforts should explore the relationships between climate change, watershed development, and constituent loading to Texas estuaries.

The degree of change in freshwater inflow from both climate change and watershed development combined to determine the magnitude of heterotrophy in Copano Bay (Table 2). Copano Bay exhibited more heterotrophic conditions (mean = -3.82 mg O_2 l^{-1} d^{-1}), with the combination of relatively modest urbanization of the Aransas River watershed and predicted precipitation increases due to climate change, than from each individually. This estimate of heterotrophy, especially when the standard error of 1.46 mg O_2 l^{-1} d^{-1}, could push Copano Bay dissolved oxygen consumption past the assimilation capacity afforded by diffusion (Russell 2005). This large standard error means that NEM values as heterotrophic as -6 mg O_2 l^{-1} d^{-1} are within the 95% prediction confidence interval. On calm days with average wind speeds less than 2 m s^{-1} daily diffusion from the atmosphere in Texas bays reach a maximum rate of approximately 5 mg O_2 l^{-1} d^{-1}. Thus, an NEM value of -6 mg O_2 l^{-1} d^{-1} would result in a decrease of dissolved oxygen concentrations of around 1 mg O_2 l^{-1} which could quickly lead to dystrophic conditions and hypoxia in the warm, salty Texas bays where dissolved oxygen concentrations at 100%

saturation can already be as low as 6 mg O_2 l^{-1}. It is fortunate that this combination of conditions (i.e. high freshwater inflow and calm winds) is currently rare in Texas.

Conclusions

The model predicts non-significant changes to NEM in Copano and Lavaca Bays over the next one hundred years of climate change and watershed development. However, if variability in model estimates is accounted for, Copano Bay may become so heterotrophic after increased precipitation events that it might suffer from increased occurrences of hypoxia. Increased urbanization, above the 200% increase modeled in this study, could also push NEM in Copano and Lavaca Bay to significantly more heterotrophic values after precipitation events. It is difficult to model NEM during intervals between precipitation events because of the large variability at low freshwater inflows (Fig. 10). Also, assessment of significant changes in NEM on the time scales of climate change and watershed development is hampered by the large variability in daily freshwater inflows. Daily freshwater inflow into Copano Bay, for example, changed as much as 30 million m^3 in 2004, which is much greater than the predicted 1-2 million m^3 d^{-1} increase in inflow due to climate change and watershed development. With uncertainties that exist during attempts to model physical and biological parameters on watershed level scales, it is important to view the simulated NEM results as possible trends arising from possible future conditions. The simulated results suggest a trend towards more heterotrophic conditions in Copano and Lavaca Bay metabolic rates with predicted climate change and watershed development.

Recommendations for Future Research

NEM model results were significantly related to observed NEM values in Copano Bay but the non-linear relationship had a lot of scatter. The large amount of dispersion in observed NEM values, especially during low freshwater inflow conditions, may be due to factors not included in the model. One factor that may influence NEM during low freshwater inflow periods is turbidity. The balance between primary production and respiration could be influences by reduced light availability due to high turbidity in Copano Bay. High turbidity conditions exist for the majority of the year in Texas bays (NOAA 1997). Higher turbidities and subsequent lower available photosynthetically active radiation (PAR) with depth usually exist in water close to river discharge as compared to water closer to an estuary mouth (Kennish 1986). Secchi depths in Copano Bay ranged between 0.5 m during winter to 0.2 m during summer (personal observation). Surface irradiance, and thus PAR, also varied with season, and on shorter time scales associated with clouds (Russell 2005). Surface irradiance levels dropped markedly during the April and May storm events (Russell 2005), but little effect was observed on NEM. Turbidity and in-situ irradiance measurements need to be compared to simultaneously calculated NEM values before these relationships can be added to the model.

Another factor that may influence NEM could be the balance between nutrients and organic matter in freshwater inflow. A determining factor of the balance between autotrophy and heterotrophy is the balance between organic carbon and nutrient loading (Kemp et al. 1997, Eyre and McKee 2002). Nutrient and organic matter loads in the micro-tidal estuaries of Texas are

mainly delivered through freshwater inflow (Whitledge 1989a, 1989b, Longley 1994). The pulsing nature of freshwater inflow sporadically delivers nutrients and organic matter at high magnitudes but for short durations. As freshwater inflow increases, the ratio of nutrients to organic matter within the inflow will also change, decreasing as more terrestrial organic particulates are eroded from the watershed (Jones et al. 1986, Parker et al. 1989). The NEM model predicts an increasing NEM with small increases from base freshwater inflow conditions. The model then predicts more heterotrophic conditions with increasing flow. It is possible that a shift from high to low nutrient to organic matter ratios occurs as freshwater inflow increases from base flow levels. Quantification of nutrient and organic matter loading during a range of freshwater inflow conditions would be an important addition to the NEM model.

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Table 1 Copano Bay (CB) and Lavaca Bay (LB) average daily water temperatures (Temp) (^{o}C), and salinity (ppt) during deployments, and United States Geological Survey average gauged daily freshwater inflow (FW) (10^5 m^3) during 2004.

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Average
CB Temp	16	17	20	21	24	29	30	29	28	28	22	18	24
LB Temp	13			21			30			27			23
CB Salinity	9	11	12	4	1	1	1	2	6	7	7	7	6
LB Salinity	22			8			0			8			10
CB FW	3	3	2	13	16	1	4	3	4	2	7	2	5
LB FW	12	15	15	26	71	86	21	3	2	8	106	5	31

Table 2 Present day and predicted daily precipitation (Precip), total precipitation depths, and freshwater inflows during events due to potential scenarios of climate change, urbanization, and their combination in the upper Aransas River watershed.

Scenario	Precip (cm d^{-1})	Precip Depth (cm d^{-1})	Freshwater Inflow (m^3 d^{-1})	Cumulative Event Inflow (m^3)	NEM ± SE (mg O_2 l^{-1} d^{-1})
Present Climate	1.63	91	1378730	4136190	-2.11 ± 1.46
Predicted Climate	2.46	137	2638421	7915263	-3.43 ± 1.46
Present Urbanization	1.63	91	1378730	4136190	-2.11 ± 1.46
100% Increase in Urbanization	1.63	91	1406269	4218807	-2.14 ± 1.46
200% Increase in Urbanization	1.63	91	1642889	4928667	-2.44 ± 1.46
200% Increase in Urbanization and Predicted Climate	2.46	137	3143932	9431796	-3.82 ± 1.46

Table 3 Present day and predicted environmental conditions due to climate change and watershed development. NEM is a function of cumulative ten-day freshwater inflow (FW), salinity (S), and temperature (T).

$$NEM\ (mg\ O_2\ l^{-1}\ d^{-1}) = -1.82 + 2.62\times10^{-8}\ FW - 0.07\ T + 0.21\ S$$

Scenario	Cumulative Event Inflow (m^3)	Salinity (ppt)	Temperature (^oC)	NEM ± SE (mg O_2 l^{-1} d^{-1})
Present Day	10×10^6	15	25	-0.16 ± 1.64
Predicted Climate and 200% Urbanization	40×10^6	1	30	-2.66 ± 1.64

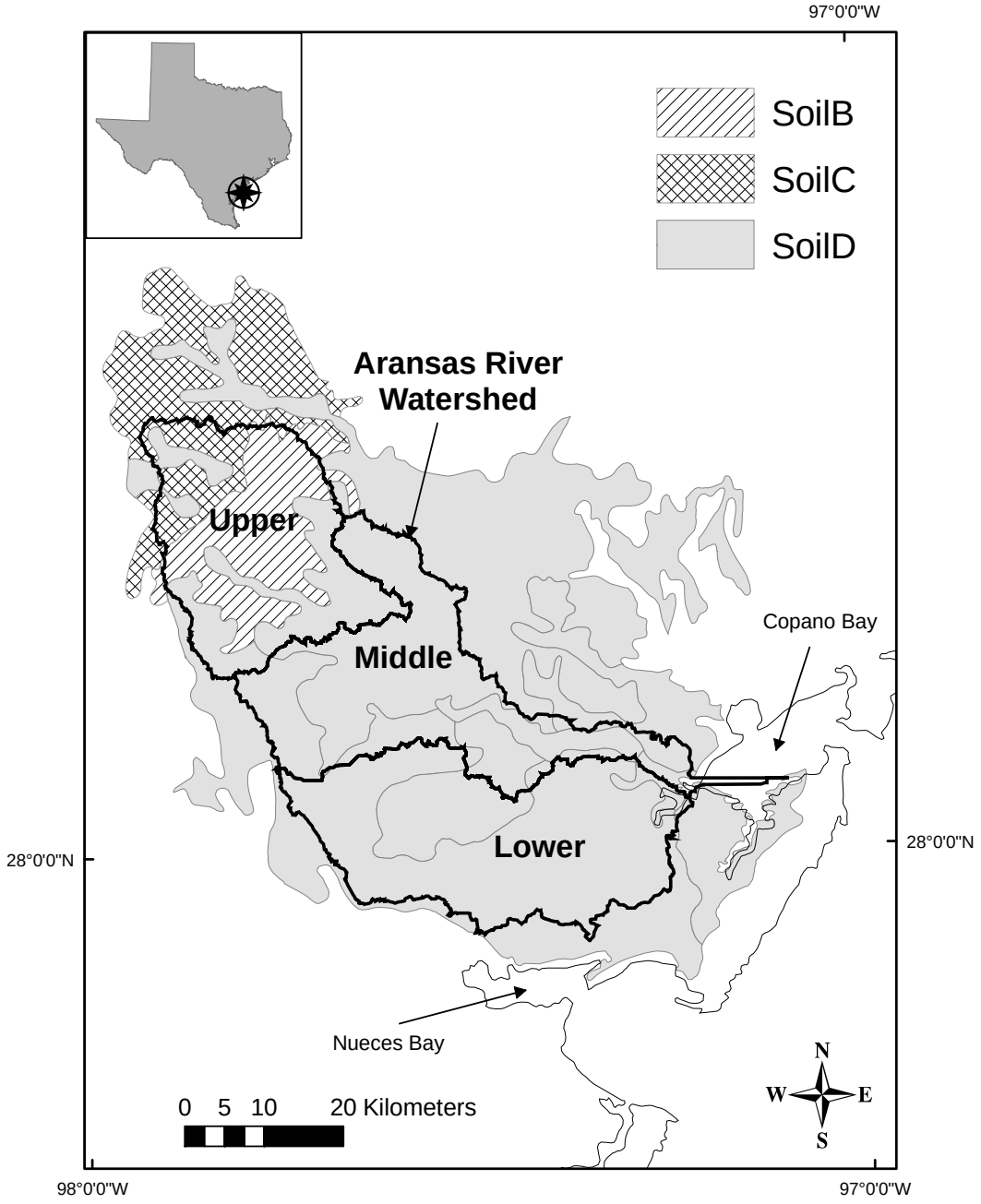

Figure 1 Soil groups in the three regions of the Aransas River watershed draining into Copano Bay. The Aransas River watershed soils are grouped into three types: soil type B (silt loam or loam), soil type C (sandy clay loam) and soil type D (clay loam, silty clay loam, sandy clay, silty clay or clay).

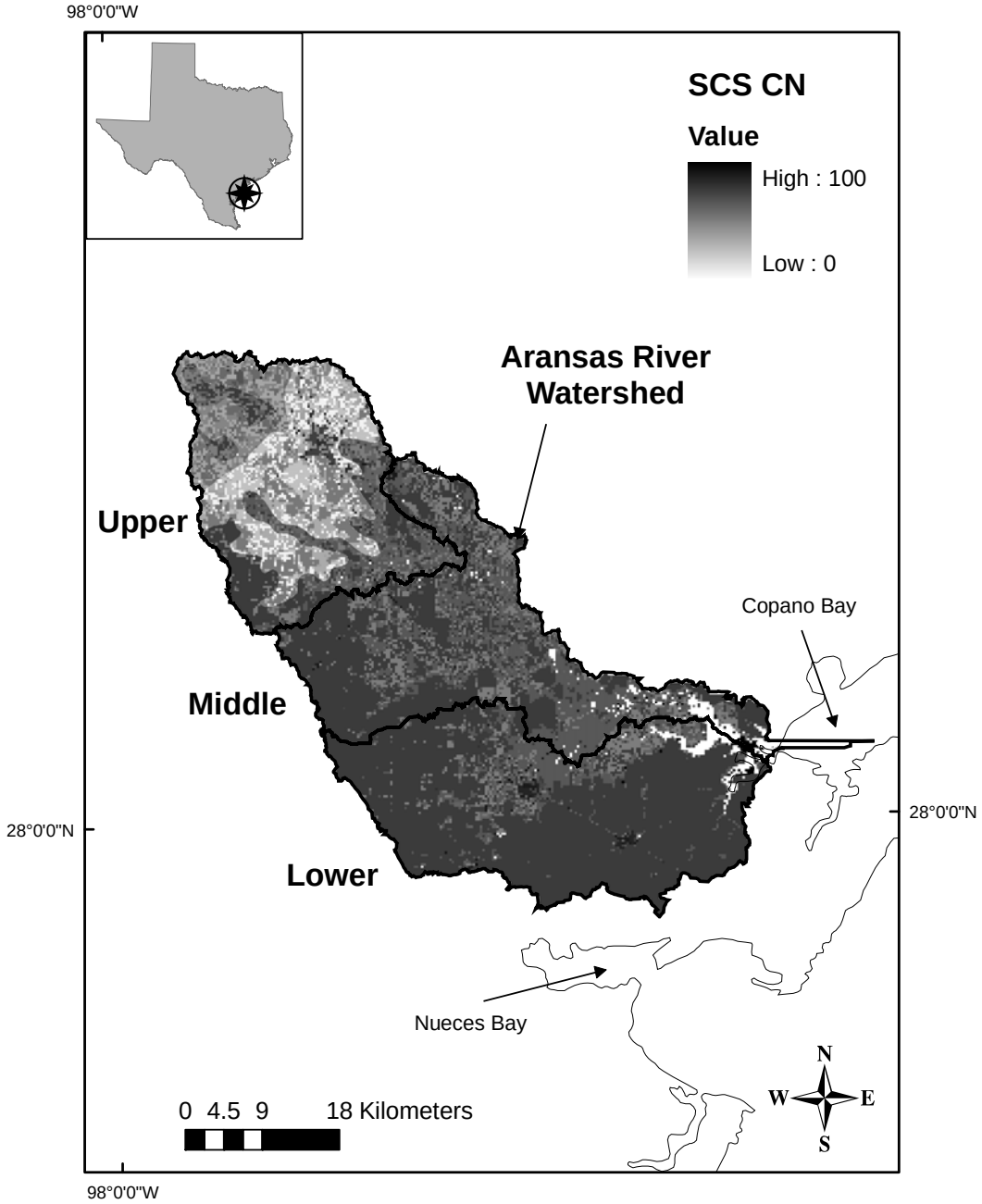

Figure 2 Curve numbers by soil type in the Aransas River watershed. Urbanized and cropland yield the highest CN numbers. Areas with high CN numbers have less precipitation infiltration capacity.

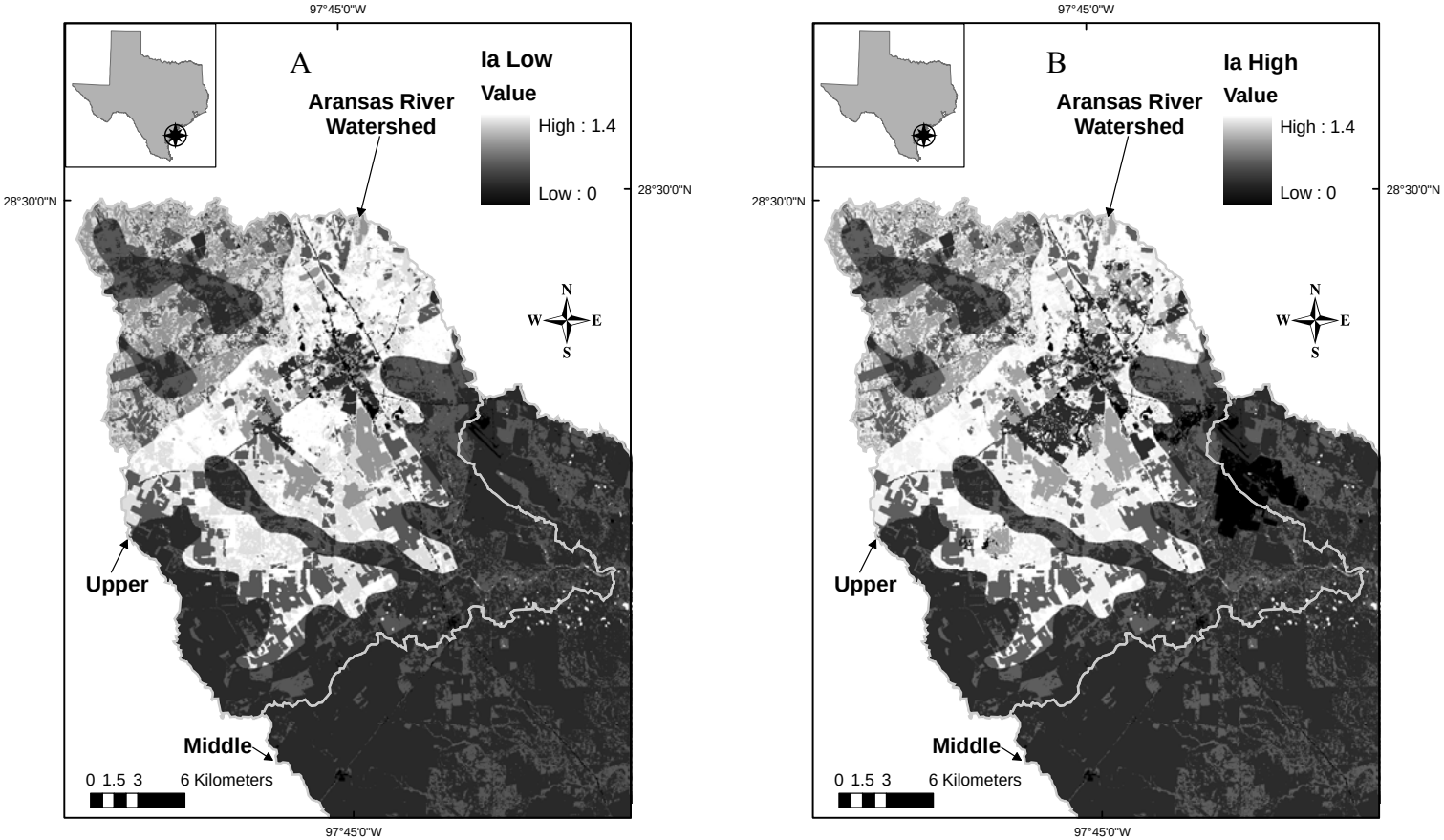

Figure 3 Estimated land use / land cover due to two different percent increases in upper Aransas River watershed urbanized areas with initial abstraction values used in watershed development scenarios. A) 100%. B) 200%.

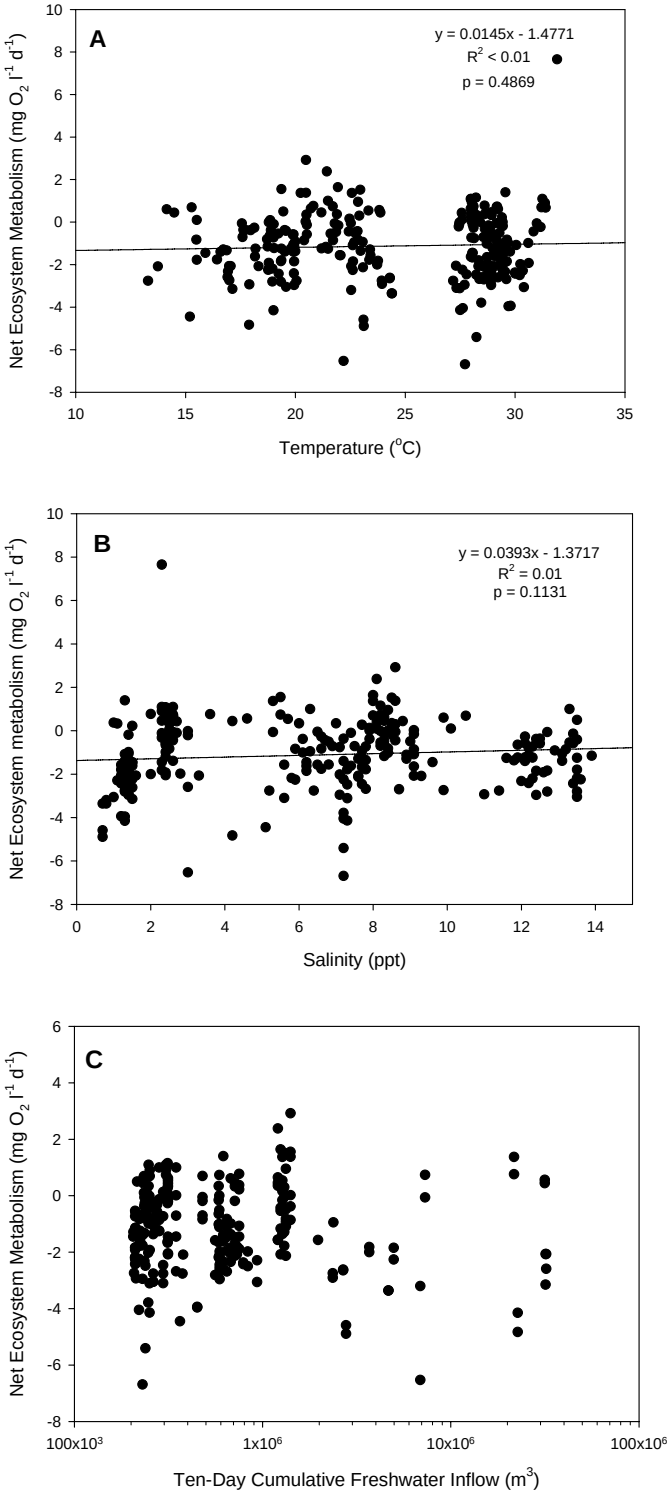

Figure 4 The response of net ecosystem metabolism to changing environmental conditions in Copano Bay. A) Temperature. B) Salinity. C) Freshwater inflow.

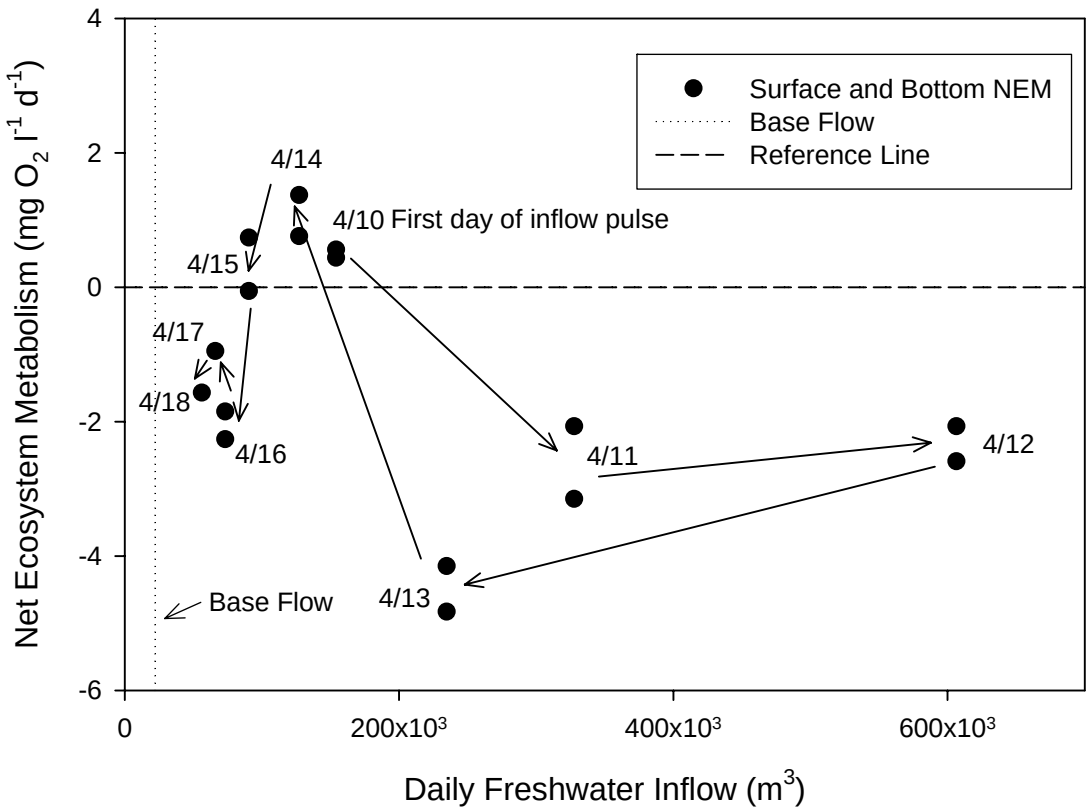

Figure 5 The temporal response of net ecosystem metabolism to rapidly changing daily freshwater inflow. Net ecosystem metabolism is autotrophic during the first day of increased inflow, is net heterotrophic over the following three days, and then becomes autotrophic again for a couple of days before returning to more typical heterotrophic values during base flow conditions (Aransas River 2002-2005 USGS median gauged freshwater inflow $= 22\times10^3$ m^3 d^{-1}).

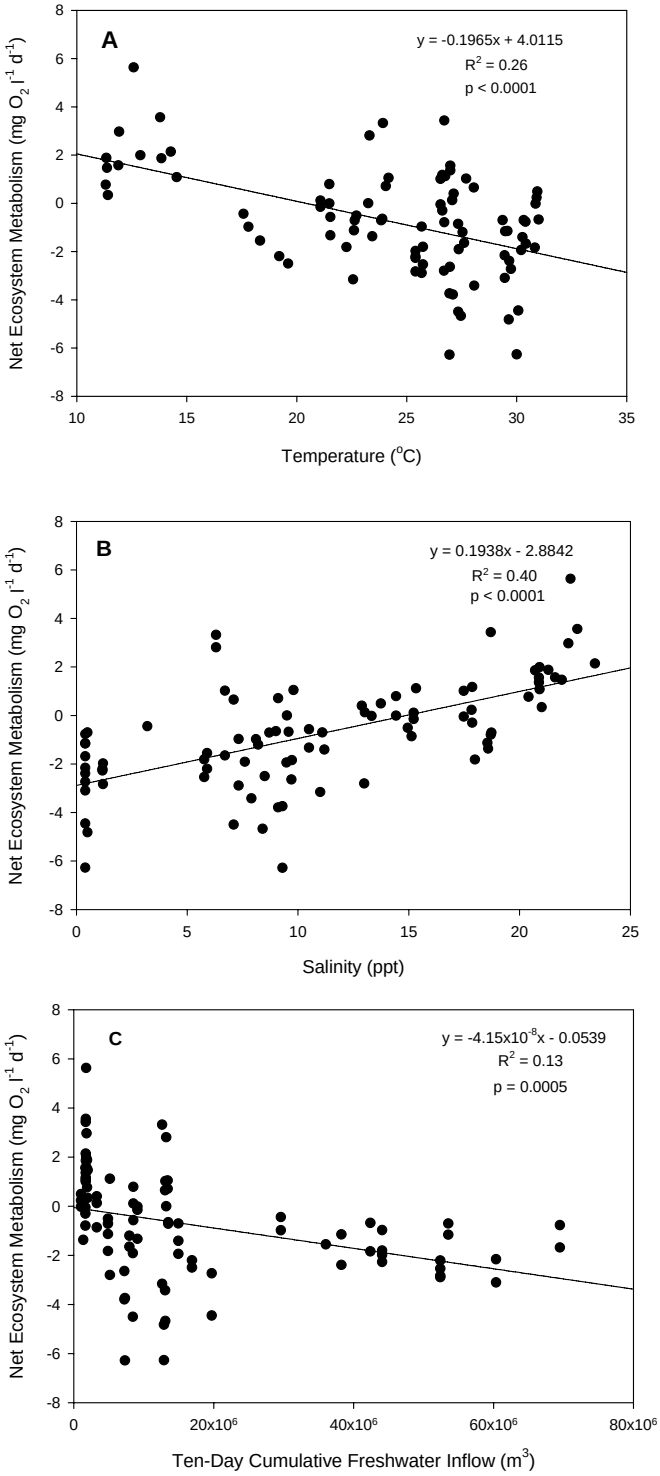

Figure 6 The response of net ecosystem metabolism to changing environmental conditions in Lavaca Bay. A) Temperature. B) Salinity. C) Freshwater inflow.

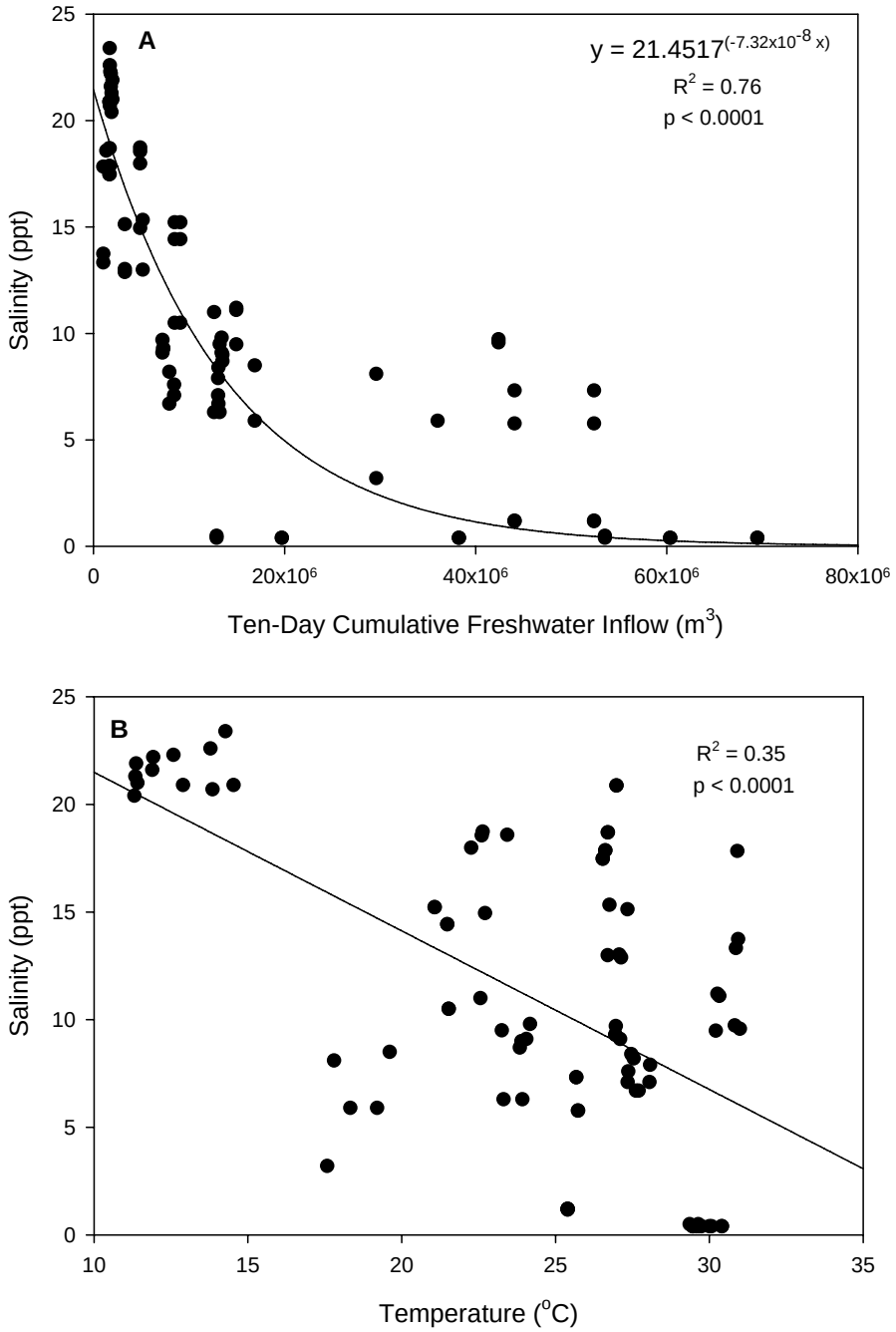

Figure 7 Relationships between A) freshwater inflow and salinity, and B) temperature and salinity in Lavaca Bay

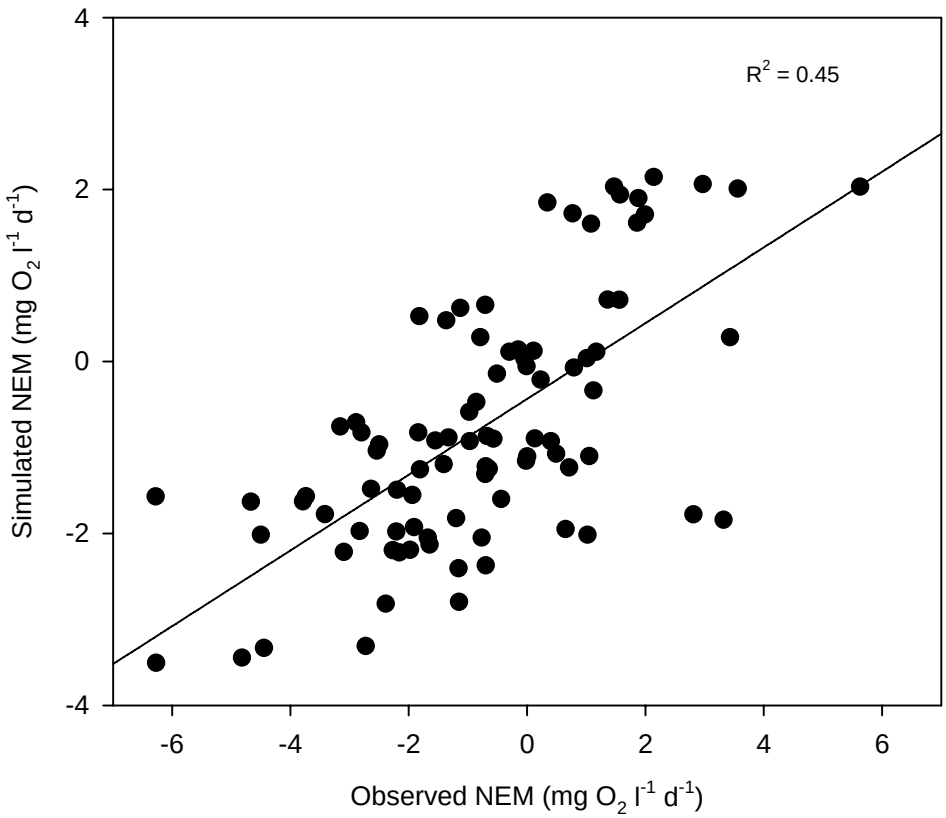

Figure 8 Lavaca Bay NEM model. Simulated NEM values match fairly well with those observed during 2002-2004.

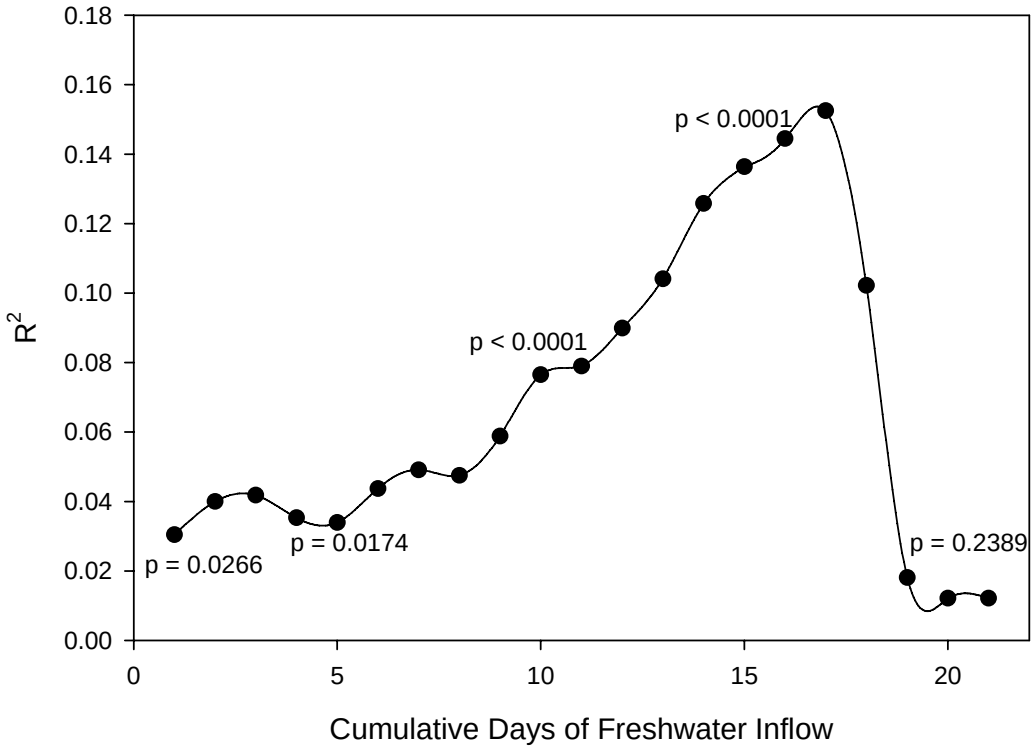

Figure 9 The fit of the model to observed NEM results in Copano Bay was assessed at one to twenty-one days of cumulative freshwater inflow. The best fit was obtained when the model was fit to NEM results at 17-day cumulative flow.

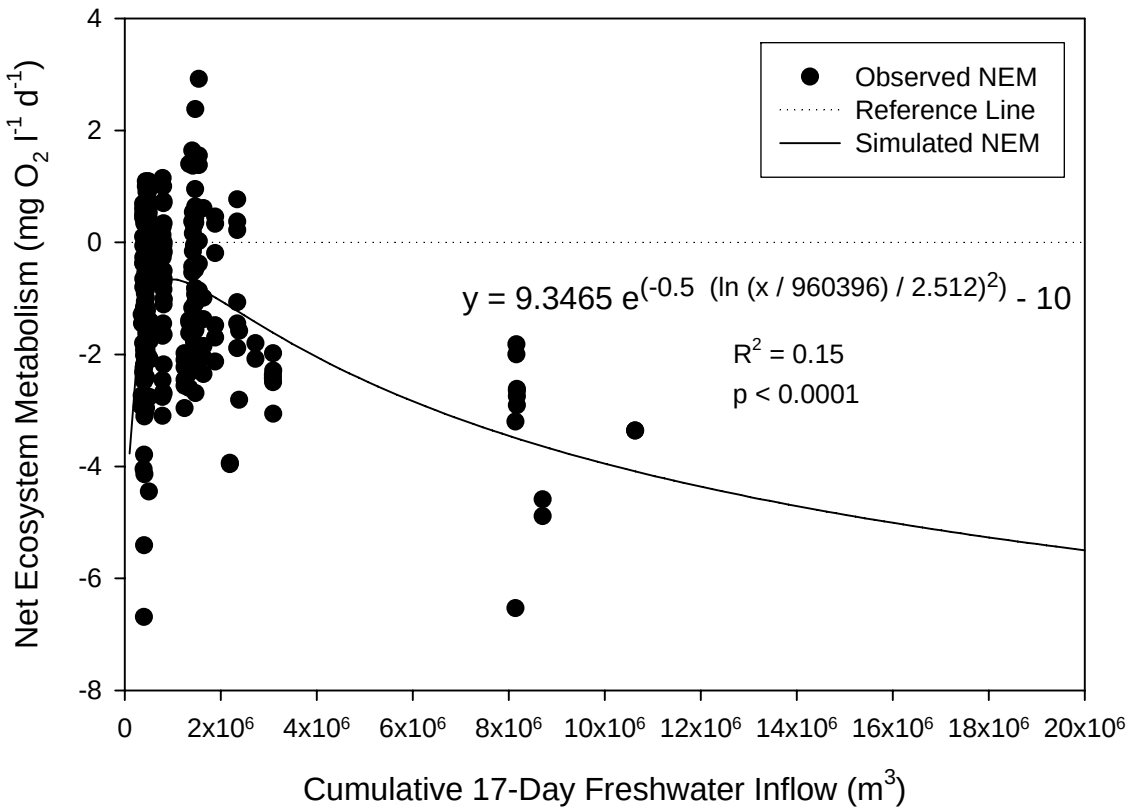

Figure 10 The relationship between observed and model predicted net ecosystem metabolism in Copano Bay, 2004. The non-linear net ecosystem metabolism model curve increases, peaks, and then steadily decreases as 17-day cumulative freshwater inflows increase.

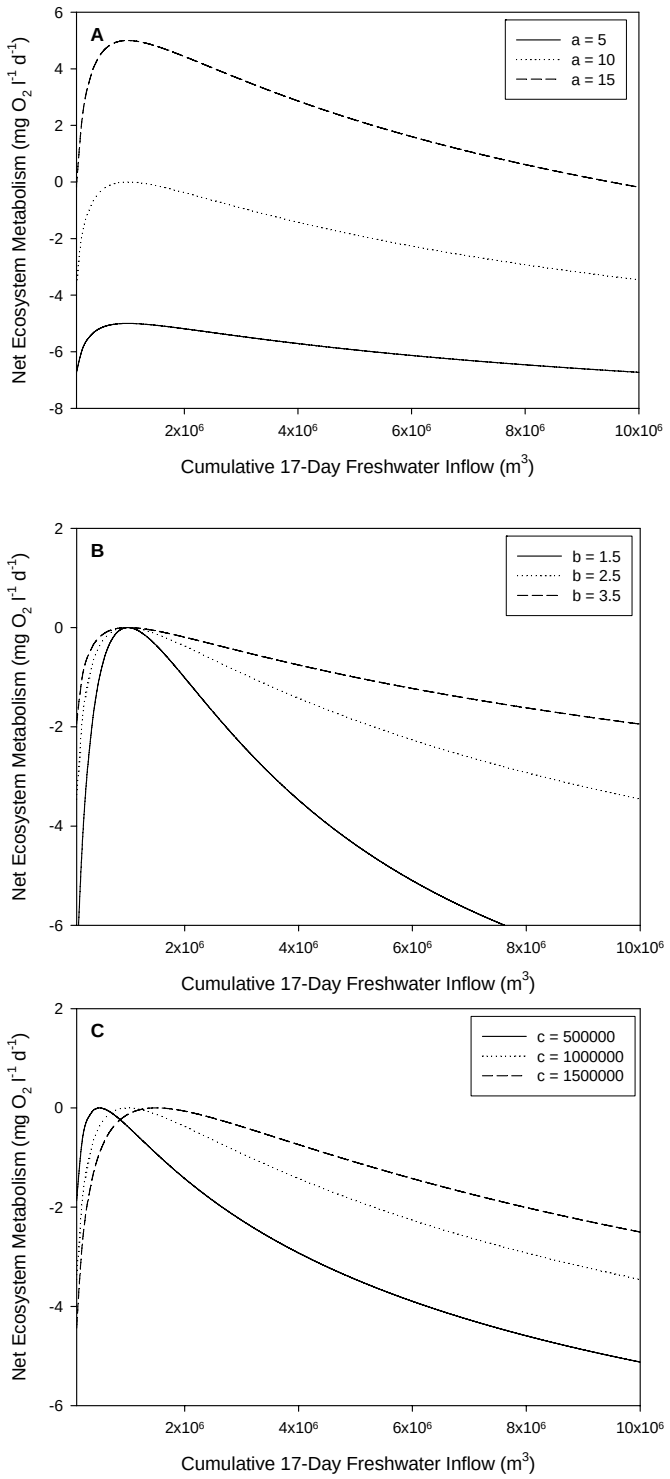

Figure 11 Sensitivity analysis of the three parameters in the net ecosystem metabolism model over a range of 17-day cumulative freshwater inflows. A) Parameter a. B) Parameter b. C) Parameter c.

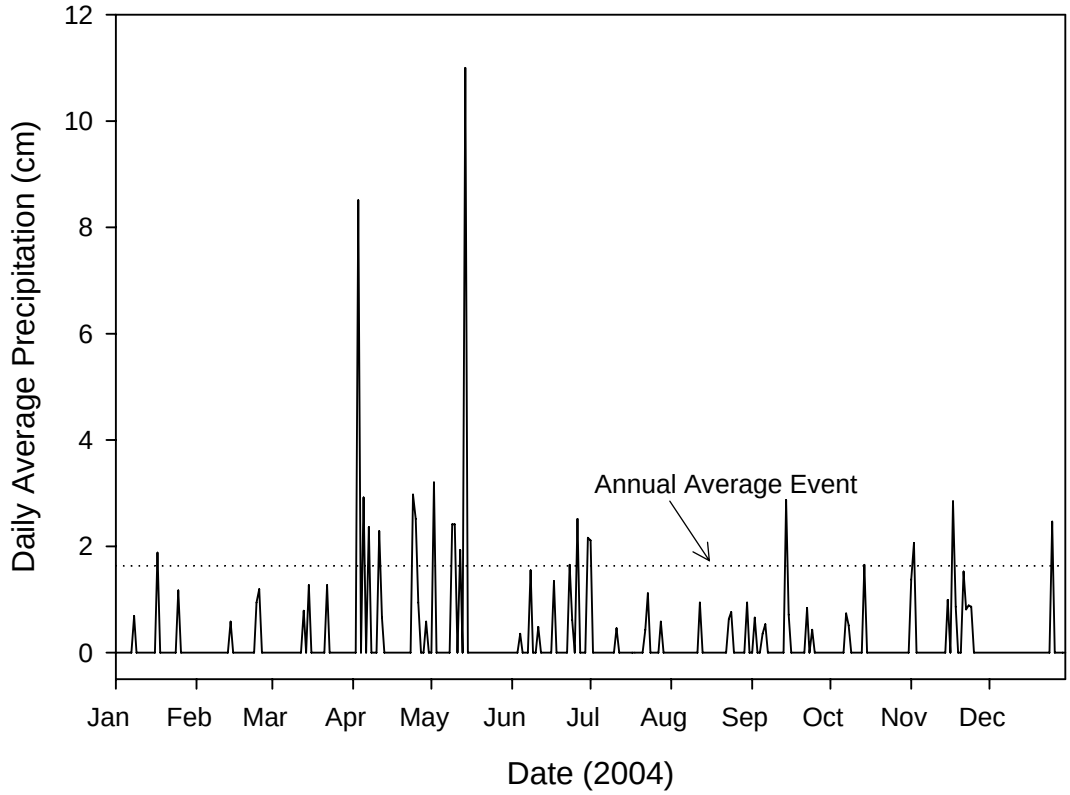

Figure 12 Daily average upper Aransas River watershed precipitation calculated from Nexrad data. Annual average precipitation (1.63 cm d^{-1}) during events denoted by dotted line.

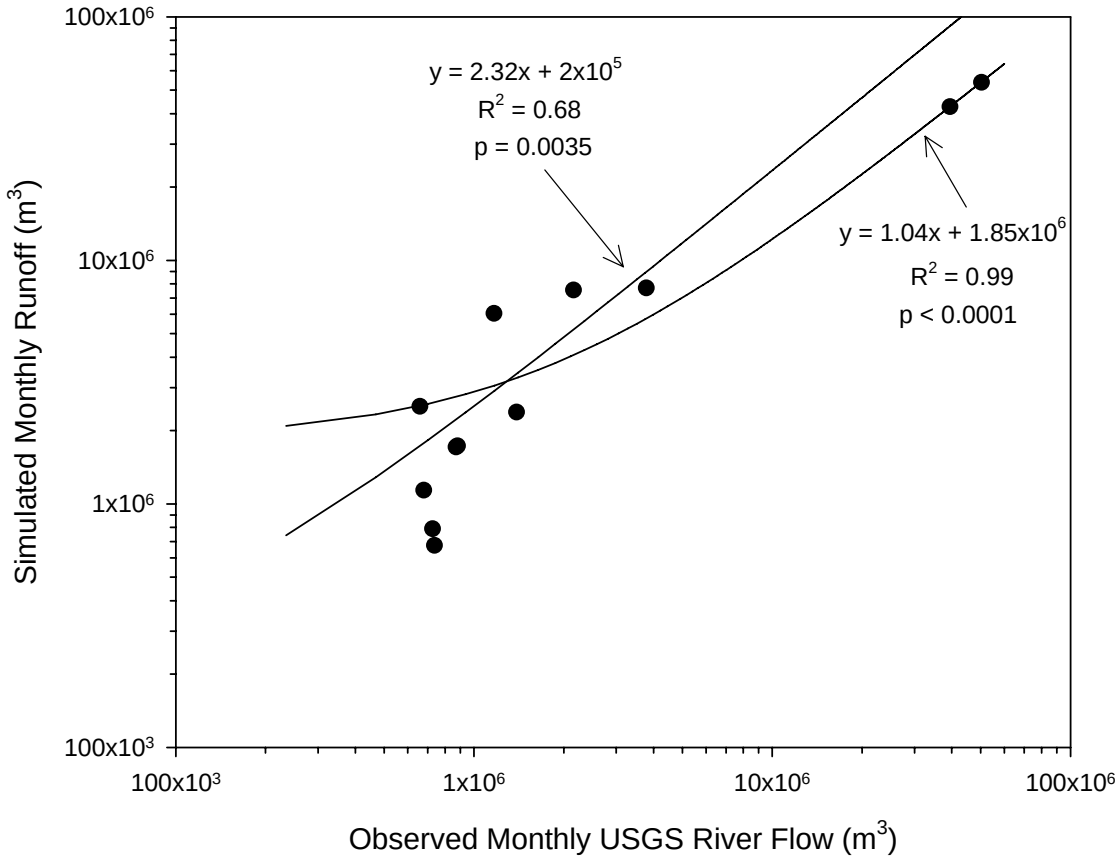

Figure 13 A comparison of simulated monthly runoff from the upper Aransas River watershed and observed monthly Aransas River flow. The best fit line is shown with (R^2 = 0.99) and with out (R^2 = 0.68) inclusion of the two extreme flow events. More scatter is present at lower flows, possibly due to the effects of differences in evaporation rates of runoff.

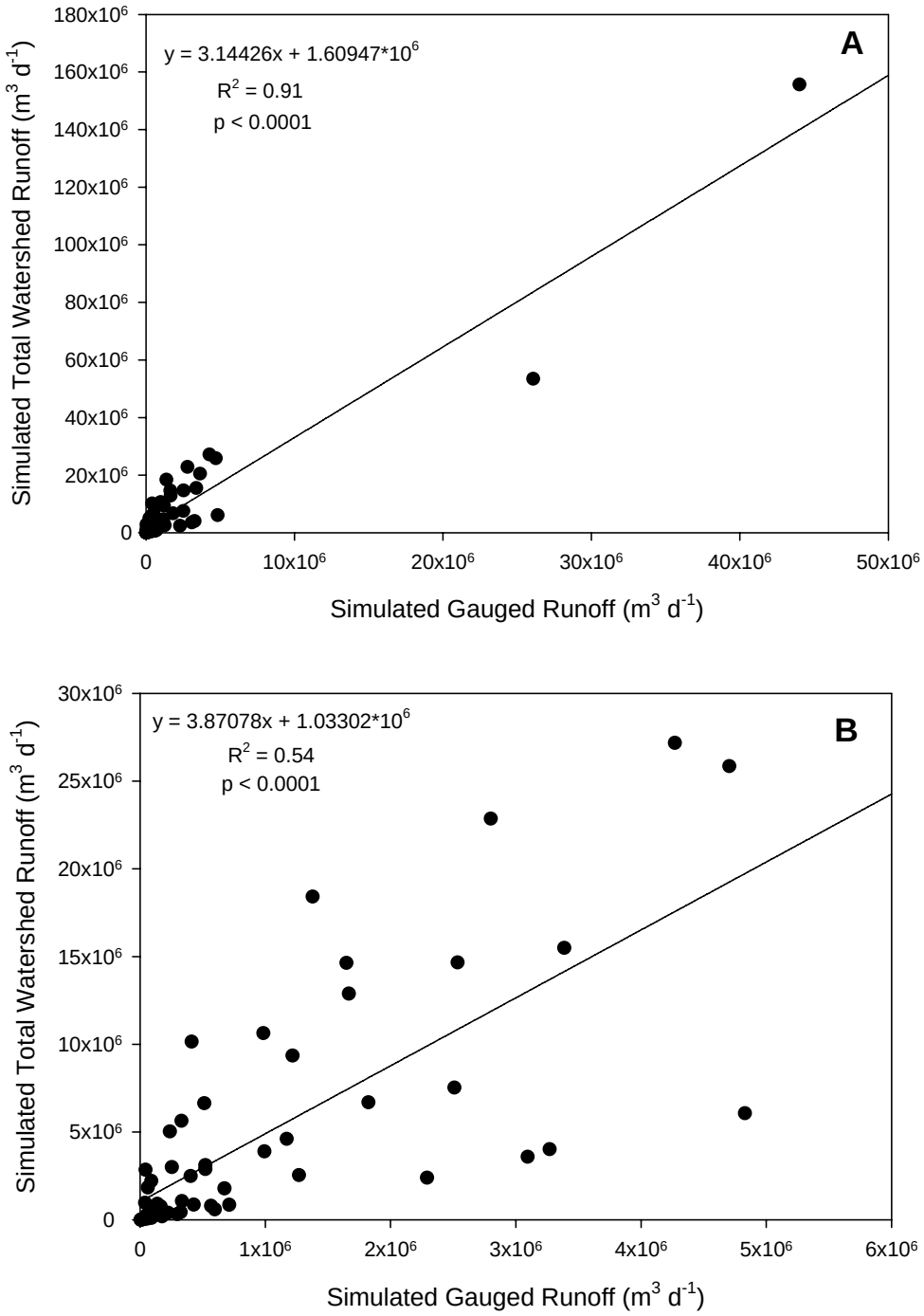

Figure 14 Comparison of simulated runoff in the upper sub-basin and simulated runoff from the entire Aransas River watershed. A) Full data-set. B) Without outliers.