MARINE SCIENCE INSTITUTE
THE UNIVERSITY OF TEXAS AT AUSTIN
750 Channel View Drive • Port Aransas, Texas 78373-5015 • Phone (361) 749-6711 • Fax (361) 749-6777

February 1, 2006

Carla G. Guthrie
Natural Resource Specialist
Texas Water Development Board
1700 North Congress Ave.
P.O. Box 13231
Austin, TX 78711-3231

RE: Submission of Final Report entitled,
"Verification of Bay Productivity Measurement by Remote Sensors"
Interagency Cooperative Contract Number: IA03-483-003

Dear Dr. Guthrie,

Enclosed please find copies of the referenced final report. As required, I have enclosed one electronic copy, one single-sided hard copy, and nine double-sided hard copies. This final report is a revised version of the draft report sent in July 2004. I have revised the report to include all suggestions made by the review team. As such, this report closes this study project.

I would like to thank you and the Board for your past and continued support of my research. I find this relationship very gratifying, and hope that you have gotten information that is directly applicable to your management needs.

If you need any further information, please call me at (361)749-6779, or FAX (361)749-6777, or e-mail paul@utmsi.utexas.edu.

Sincerely,

Paul Montagna, Ph.D.
Research Professor

Left running head: M. J. Russell et al.

Right running head: Estuarine Health and Function

Title: Effect of Freshwater Inflow on Estuarine Health and Function: Estimated by Whole Ecosystem Metabolism

Names and address of authors:

Marc J. Russell[1]

Paul A. Montagna

Richard D. Kalke

University of Texas Marine Science Institute

750 Channel View Dr., Port Aransas, TX 78373

Submitted to: Estuaries

Draft date: June 2, 2004

[1]Corresponding author (361) 749-6817, marcr@utmsi.utexas.edu

Abstract:

Freshwater inflow is necessary to maintain health and productivity in estuarine ecosystems. There are no standard criteria to set inflow levels, however. Also, freshwater inflow rates are changing due to changing land use patterns, water diversions for human consumption, and climate effects. There is a need to be able to predict how changing hydrology might affect estuary health. One indicator of estuarine health is ecosystem function of which whole ecosystem metabolism is a major component. It was hypothesized that whole ecosystem metabolism in shallow estuaries will depend on freshwater inflow. To test this hypothesis, whole ecosystem metabolism was calculated in Lavaca Bay, Texas and its relationship to freshwater inflow determined. We calculated a significant indirect relationship between whole ecosystem metabolism and freshwater inflow near to the freshwater source in the upper bay, with more negative whole ecosystem metabolism occurring after higher freshwater inflow events. No significant relationship was found between whole ecosystem metabolism and freshwater inflow in the lower bay. The relationship between freshwater inflow and net ecosystem metabolism could be useful in total maximum daily load (TMDL) programs for dissolved oxygen impairment. We conclude that freshwater loading i.e., the combination of water quality and quantity, drives ecosystem function in shallow water estuaries. The location of freshwater inflow sources within an estuary, however, is important in regulating this relationship.

Introduction

Freshwater inflow is necessary to maintain both primary and secondary productivity in coastal estuary ecosystems. Minimum freshwater inflow levels are required by many states to protect estuarine health, but there is no standard approach or criterion to set inflow levels (Montagna et al. 2002). Also, freshwater inflow rates are changing because of changes in land use, water diversion for human consumption, and climate change effects. These anthropogenic changes result in decreased freshwater inflow and changes in the capture and reduction of flood events. There is a need to be able to predict how these anthropogenic changes in hydrology might affect estuarine health. Estuarine health is the ecological integrity of an entire system. Ecological integrity can be defined as a condition of ecosystems that is fully developed when the network of biotic and abiotic components and processes is complete and functioning optimally (Campbell, 2000). A reliable and accurate indicator of estuarine health is ecosystem function.

An important component of ecosystem function is whole ecosystem metabolism. Whole ecosystem metabolism is calculated by subtracting respiration from primary production for all biological components contained in a defined body of water. A positive whole ecosystem metabolism indicates that primary production exceeds respiration. A negative whole ecosystem metabolism means that respiration exceeds primary production. In the aquatic environment, whole ecosystem metabolism depends on a variety of physical and biological factors. Physical factors that influence whole ecosystem metabolism include depth, surface wind speed, freshwater inflow, turbidity, substrate type, salinity, temperature, flow rates, nutrient concentrations, and tidal cycles. Biological factors that influence whole ecosystem metabolism include chlorophyll-a, amount of live biomass in the water column and sediment, photosynthesis rates, and respiration rates. Changes in whole ecosystem metabolism may be driven by short term events, seasonal, or annual cycles of environmental conditions. Freshwater inflow, by delivering nutrients and organic matter from the watershed, may be the most important of these environmental conditions by affecting the health, function, and productivity of estuarine ecosystems.

Whole ecosystem metabolism is linked to dissolved oxygen dynamics through the processes of photosynthesis and respiration. Dissolved oxygen concentrations must remain sufficiently high to preserve ecosystem health. There are currently 4641 impaired water bodies in the United States listed on the Environmental Protection Agency's 2002 303(d) list for organic enrichment/low dissolved oxygen. Low dissolved oxygen ranks 5th on the top 100 impairments list. Low dissolved oxygen is responsible for the approval of 947 total maximum daily load (TMDL) programs, representing over 10% of the total number currently approved. One effect of dissolved oxygen dynamics that has received recent interest is bottom water hypoxia events during summer months. Causes of bottom water hypoxic conditions include water column stratification, nutrient enrichment, and organic matter decomposition (Officer et al., 1984; Pokryfki and Randall, 1987; Rabalais et al. 2001). The balance between water/sediment interface photosynthesis and respiration can determine whether these waters become hypoxic or anoxic. Large areas of shallow water estuaries can become hypoxic during summer months when high levels of water column primary production, stratification, benthic respiration, and reduced flushing by freshwater inflow reduce bottom water dissolved oxygen levels to dangerous levels (<2.0 mg O_2 l^{-1}). Over one half of the estuaries in the Gulf of Mexico exhibit moderate to severe dissolved oxygen depletion (hypoxia/anoxia), a key indicator of aquatic ecosystem health

(Bricker et al. 1999). Hypoxia in Corpus Christi Bay was documented in the summer months of 1988 (Montagna and Kalke 1992) and has occurred every summer since (Montagna and Morehead 2003). Organic matter and nutrients delivered by freshwater inflow not only effect estuarine health but also estuarine function.

Ecosystem function in Texas shallow water estuaries may be altered by anthropogenic modifications of Texas watersheds and the subsequent changes in freshwater inflow dynamics. Restored inflow to Rincon Bayou Texas, after damming reduced freshwater inflow by 55%, resulted in infauna abundance, biomass, and diversity increases (Montagna et al, 2002). Increased freshwater inflow restored the ecosystem function of this salt marsh nursery habitat for estuarine dependent, commercially important species such as the brown shrimp, *Farfante penaeus aztecus* (Riera et al, 2000). Ecosystem function often translates into ecosystem productivity.

Ecosystem productivity may be related to freshwater inflow by supplying nutrients and organic matter from the watershed. Freshwater inflows to South Texas estuaries are limited (~0-800 million m^3 y^{-1}). An analysis of open water dissolved oxygen measurements to calculate ecosystem metabolism over the past 20 years concluded that some Texas estuaries have low amounts of gross primary productivity with only 200 g C m^{-2} y^{-1} (Ward, 2003). Low gross primary production may be due to lack of freshwater inflow. Both organic matter and nutrients can be used to fuel primary and secondary production in an estuary either directly by incorporation into new biomass or indirectly by re-mineralization.

Open water dissolved oxygen measurements have been used to estimate whole ecosystem metabolism, providing spatially and temporally integrated estimates of metabolic processes since Odum's seminal work in the 1950's (Odum 1956). Whole ecosystem metabolism is a calculation of the change in dissolved oxygen concentration resulting from biological processes in an aquatic ecosystem over a period of 24 hours. Atmospheric oxygen flux must be estimated to separate physical and biological influences on dissolved oxygen concentration (Odum and Wilson 1962). Atmospheric oxygen flux is influenced by a combination of dissolved oxygen concentration gradients and near surface turbulence dynamics. The physical factors driving near surface turbulence must therefore be accounted for during calculations of whole ecosystem metabolism.

It was hypothesized that whole ecosystem metabolism in shallow estuaries will depend on freshwater inflow. To test this hypothesis, whole ecosystem metabolism was calculated in Lavaca Bay, Texas and its relationship to freshwater inflow determined. We calculated whole ecosystem metabolism from continuous oxygen measurements and compared them to freshwater inflow amounts.

Materials and Methods

A monitoring plan was designed to assess both the spatial and temporal variability in whole ecosystem metabolism using dissolved oxygen concentrations in Lavaca Bay. Fifty-eight 24-hour water quality monitoring samples, 20 water column nutrient samples, 43 water column chlorophyll-a, and 50 sediment samples were taken over a two year period (2002-2003) (Table 1a and 1b). Six different Texas Commission of Environmental Quality (TCEQ) sites were sampled to provide spatial coverage (Table 2) (Fig. 1) (http://www.tceq.state.tx.us). Sites were divided into upper bay (stations 1-3), and lower bay (stations 4-6) groups. The upper, lower bay groups are subdivided by a constriction caused by the Highway 35 overpass (Fig. 1). Dissolved oxygen and other water quality parameter measurements were taken every 15 minutes 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°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 is automatically corrected to 25°C.

The relatively high wind speeds that occur across the shallow water estuaries of Texas imply that wind will dominate the physical control of atmospheric oxygen flux. Texas estuaries experience sustained wind speeds commonly around 7-8 m s^{-1} (~13-18 mph), but can have daily variations in wind speed from 1-10 m s^{-1} (~2-23 mph) (Texas Coastal Ocean Observation Network data at http://lighthouse.tamucc.edu/TCOON/HomePage). Estuaries in other regions of the U.S. tend to have wind speeds in the range of 0-6 m s^{-1} (~0-12 mph) with maximum atmospheric oxygen exchanges measured at 8.6 m s^{-1} (~19 mph) (Kemp and Boynton 1980; Marino and Howarth 1993). Meteorological forcing dominates water exchange and circulation in South Texas estuaries because of shallow water depths (medium depth ~2-4 m), small tidal range (~0.25 m), little freshwater inflow (~0-800 million m^3 y^{-1}), and long over-water fetches (Orlando et al. 1993). These characteristics when combined with ample sunlight, high temperatures, and relatively steady South-east winds make South Texas estuarine ecosystems particularly amenable to open water methods of estimating whole ecosystem metabolism. Biological processes can still dominate dissolved oxygen concentration changes in South Texas estuaries even with the prevalence of high wind speeds. The physical features of South Texas estuaries, when combined with the highly dynamic and large influence of wind speed on surface turbulence, require that estimates of whole ecosystem metabolism in this region adjust for changes in atmospheric oxygen flux because of changing wind speeds.

The wind dependent diffusion coefficients given by D'Avanzo et al. (1996) were applied to calculations of whole ecosystem metabolism in Lavaca Bay. D'Avanzo et al.'s diffusion coefficients allowed for diffusion corrected calculations of dissolved oxygen concentration change that could vary over short temporal scales (hourly). The major physical influence on whole ecosystem metabolism calculations was thus removed by adjusting for atmospheric oxygen flux generated during undersaturated or supersaturated dissolved oxygen concentration conditions. Removal of the physical influences on dissolved oxygen concentration left just the biologically driven changes in dissolved oxygen concentration.

Net ecosystem metabolism was calculated using open water diurnal methods. Dissolved oxygen concentrations were taken every 15 minutes and converted to a rate of change in dissolved oxygen concentration. These rates of change were then adjusted to control for diffusion of oxygen between the water column and the atmosphere by using percent saturation of dissolved oxygen in the water column and the wind dependent diffusion coefficient K (g O_2 m^{-2} h^{-1}) at 0% saturation proposed by D'Avanzo et al. (1996) using the equation:

$R_{dc} = R - ((1-((S_1 + S_2) / 200)) * K / 4)$; where

R_{dc} = diffusion corrected oxygen concentration rate of change per 15 minutes,

R = observed oxygen concentration rate of change,

S_1 and S_2 = dissolved oxygen percent saturations at time one and two respectively,

K = diffusion coefficient at 0% dissolved oxygen saturation.

To calculate daily net ecosystem metabolism the 15-minute diffusion corrected rates of dissolved oxygen change were then summed over a 24-hour period, starting and ending at 8AM. Open water dissolved oxygen methods similar to those used here have been used in a variety of estuaries to calculate net ecosystem metabolism (Kemp et al 1992; D'Avanzo et al. 1996; Borsuk et al. 2001; Caffrey 2003).

Net ecosystem metabolism was regressed against freshwater inflow, salinity, water temperature, water column depth, water column chlorophyll-a, water column nutrients, and sediment characteristics. Freshwater inflow was calculated by summing all daily USGS gauged river flow (millions of cubic feet day^{-1}) into the bay during the ten days prior to sampling (http://waterdata.usgs.gov/tx/nwis/rt). A ten day period was assumed to be the time interval needed to capture an estuary's response to relatively recent freshwater inflow. Salinity, water temperature, and depth daily means were calculated from multiparameter sonde measurements. Chlorophyll-a was sampled by modifying the TCEQ's *Surface Water Quality Monitoring Procedures Volume 1* (2003) (http://www.tnrcc.state.tx.us/admin/topdoc/rg/415/415.html) methods for collection of routine water chemistry samples. Two 10-ml sub-samples from a 1-L van Doran bottle were collected and filtered on site. Chlorophyll-a concentration was determined using non acidification fluorometric techniques (Welschmeyer 1994). Water column nutrient analyses for ammonium, phosphate, silicate, and nitrate plus nitrite were run on a Lachat Quikchem 8000 using standard colormetric techniques (Parsons et al 1984, Diamond 1994).

Sediment and macrobenthos were sampled by taking five 6.7 cm diameter cores per station. Three cores were divided into 0-3 cm and 3-10 cm sections, and preserved in formalin until macrobenthic analysis. One core was divided into 0-3 cm and 3-10 cm sections for sediment grain size analysis; all of the 0-3 cm section and a vertical slice of the 3-10 cm section were collected in the field, but only 20 cm^3 were used in analysis. Zero to 1 cm and 2-3 cm sections from the final core were placed in sterile Petri dishes for total carbon, total nitrogen, and total organic carbon analyses.

Results

Principle component analysis (PCA) of site specific environmental variables yielded two relatively distinct groups of stations located in upper and lower Lavaca bay. Two groups of stations; 1, 2, and 3 in upper Lavaca bay and station 4, 5, and 6 in lower Lavaca bay were identified from salinity, temperature, and depth measurements taken during every 24-hour dissolved oxygen deployment (Fig. 2a). Salinity and temperature had the highest loading values with depth being similar to salinity (Fig. 2b). Principle components 1 and 2 explained 56.3% and 28.1% respectively of the total variability. The station groups resulted from a gradient of high salinity conditions at station 6 in the upper left to lower salinity conditions at station 1 in the lower right (Fig. 2a). Temperature depended on time of year when samples were collected with lower temperatures corresponding to the lower left and higher temperatures in the upper right (Fig. 2a).

Chlorophyll-a measurements resulted in similar station groups as the environmental condition analysis (Fig. 3). Stations grouped together into three sets; 1 in upper bay, 3, 5, and 6 in lower bay, and stations 2 and 4 made up a transitional group. Significant differences were seen between station 1 and the group of stations 3, 5, and 6. Stations 2 and 4 grouped with both upper and lower bay groups. The discrepancy between site 3 and 4 falling in an alternate group than during the environmental condition analysis may be due to resuspension of benthic algae by turbulence generated as water moves past an overpass located down estuary from station 3 and up estuary of station 4. Chlorophyll-a did not have a significant relationship with net ecosystem metabolism (linear regression, $p = 0.5821$) (Fig. 4).

Water column principle component nutrient analysis separated stations along a gradient from upper to lower bay. The large change in nutrient concentrations during a large pulse of freshwater inflow implies that the main driving force behind nutrient concentrations is freshwater inflow (Fig. 5a). Upper bay stations encounter slightly higher concentrations of nutrients than lower bay stations under lower freshwater inflow conditions (Fig. 5b). Principle component 1 and 2 accounted for 83.3% and 7.9% respectively of the total variance (Fig. 5c).

Sediment characteristic PCA resulted in a separation between upper and lower bay stations (Fig. 6a). Principal component 1 and 2 accounted for 60% and 24% respectively of the total variability (Fig. 6b). Stations were vertically separated on PC 2 by a gradient of sandy sediment in upper bay to clay dominated sediments in lower bay. Lower bay stations also had more total sediment nitrogen. Station 5 separated from the rest of the stations on PC 1 because of the large quantities of total carbon, total organic carbon, and rubble measured there. The rest of the stations were characterized by a larger percentage of silt and higher concentrations of total nitrogen. No significant relationship was found between any sediment characteristic and net ecosystem metabolism (linear regression, $p = 0.076\text{-}0.106$).

Linear regression analysis comparing net ecosystem metabolism with freshwater inflow, salinity, temperature, and depth resulted in only salinity ($p < 0.001$, $R^2 = 0.400$) or freshwater inflow ($p < 0.001$, $R^2 = 0.374$) being significant depending on which was entered into the model first. Freshwater inflow will be used during the rest of the analysis instead of salinity since freshwater inflow is more manageable by anthropogenic modification of watersheds than salinity.

Freshwater inflow correlated with net ecosystem metabolism in upper Lavaca bay (linear regression $p \leq 0.0001$, $R^2 = 0.41$) (Fig. 7). The largest net ecosystem metabolism residuals occurred during the lowest levels of freshwater inflow into upper Lavaca bay. The most negative net ecosystem metabolism values were calculated in upper Lavaca bay.

Lower Lavaca bay net ecosystem metabolism had an insignificant correlation with freshwater inflow (linear regression $p = 0.3497$, $R^2 = 0.03$) (Fig. 8). The largest response in net ecosystem metabolism to freshwater inflow, however, was seen in lower Lavaca bay. The two large positive values of net ecosystem metabolism in Lower Lavaca bay occurred at station 6 during higher freshwater inflows. The lack of data during moderate freshwater inflows stems from the pulsing nature of precipitation events in Texas watersheds which are characterized by extended periods of drought punctuated by flood events (Fig. 9).

Discussion

Freshwater inflow and salinity were determined to be the only factors to have a relationship with net ecosystem metabolism in Lavaca Bay. Freshwater inflow and salinity, however, have a fairly strong inverse relationship to each other (linear regression, $p < 0.0001$, $R^2 = 0.43$) (Fig. 10). Freshwater inflow is much more manageable than salinity because freshwater inflow is not as affected by tidal and meteorological changes. The large variability in estuarine environmental factors means that care must be taken to control for effects these factors may have on one's response variable of interest, in this case net ecosystem metabolism. Separation of stations into two groups located in upper and lower Lavaca Bay, even though no significant relationships were found, allowed us to remove most of the effects on net ecosystem metabolism from station differences in temperature, depth, chlorophyll-a, water column nutrients, and sediment characteristics. The only other environmental factor that needed to be controlled for was atmospheric water column oxygen diffusion.

The large influence that diffusion coefficients have on atmospheric water column oxygen diffusion and the resulting net ecosystem metabolism values meant that we needed to choose an appropriate diffusion equation for our specific ecosystem of study. Caffrey (2004) concluded that 25% of daily measured oxygen concentration changes at 42 National Estuarine Research Reserve (NERR) sites were due to atmospheric oxygen flux in water depths of approximately 1 meter. Estimates of diffusion coefficients and their relationship to wind speed have been calculated using a variety of methods. Odum and Hoskin (1958) used a method based entirely on the rate of change of dissolved oxygen concentration in South Texas estuaries during night time periods experiencing constant or near constant wind velocities. Their results suggest for Texas shallow water estuaries the volumetric diffusion coefficient k (in mg O_2 l^{-1} hr^{-1} at 100% saturation deficit) increases linearly from 0-3 as wind increases from 0-12 m s^{-1} (0-30 mph) (Odum and Wilson 1962). Hartmon and Hammond (1984) working in San Francisco Bay had similar results and derived an area based wind-dependent diffusion coefficients K (in g O_2 m^{-2} h^{-1} at 100% saturation deficit) that ranged from approx. 0-1.5 with wind speeds of 0-10 m s^{-1}. Kemp and Boynton (1980) assumed that atmospheric flux in relatively deeper systems varied as a constant function of the oxygen gradient between surface water dissolved oxygen and atmospheric gas with a diffusion coefficient that varied with both air and water turbulence. Their estimates of gas transfer across the air-water interface from measurements using the floating dome method (Copeland and Duffer 1964; Hall 1970) yielded area based diffusion coefficients of 0.9 to 9.7 g O_2 m^{-2} h^{-1}. Boynton et al (1978) also found a similar range of K's (0.4-10.7 g O_2 m^{-2} h^{-1}) using a variety of methods. With more use of the floating dome method and comparisons between different system types (i.e., estuaries, open ocean, and lakes) a more complete picture of wind speed influence on atmospheric oxygen flux became available (Marino and Howarth 1993). A general exponential relationship suggested by Smith (1985) was used to model oxygen transfer velocity as a linear function of wind speed. Smith's log linear model explained 55% of the atmospheric oxygen flux variability in a combined data set compiled from a wide range of systems and measurement techniques (Marino and Howarth 1993). A recent comparison of three wind-dependent diffusion coefficients with a constant coefficient of 0.5 g O_2 m^{-2} h^{-1} concluded that the constant coefficient was only similar to the wind-dependent coefficients at wind speeds from 0-5 m s^{-1} and greatly underestimated air-sea exchange at winds greater than 8 m s^{-1} (Caffrey 2004) (Table 3). The three wind-dependent diffusion coefficient equations are similar when plotted over wind speeds from 0-10 m s^{-1} (Fig. 11). D'Avanzo et al. (1996), studying a shallow estuarine system in Waquoit Bay, Cape Cod, Massachusetts, estimated relatively higher air-sea exchanges over the entire range of wind speeds than that found for the wide range of systems used by Marino and Howarth (1993) which included deep open ocean waters. A wind dependent diffusion coefficient similar to that proposed by D'Avanzo et al. (1996) or Marino and Howarth (1993) is therefore preferable to assuming a constant diffusion coefficient in systems encountering strong and highly variable wind speeds. We chose to use D'Avanzo et al.'s (1996) diffusion coefficients in our calculations of net ecosystem metabolism's relationship to freshwater inflow because both of our estuarine systems have shallow water depths.

Freshwater inflow alone is not driving whole ecosystem metabolism in estuaries, it is the organic and inorganic loads contained in that inflow. We can define freshwater loading as the combination of water quantity and quality. Freshwater inflow into an estuary contains organic matter and nutrients from an estuary's corresponding watershed. Freshwater inflow rates can be used as a proxy for freshwater loading from a specific watershed and will integrate watershed level processes that effect both water quality and quantity. The relationship between freshwater inflow and whole ecosystem metabolism was found to differ depending on location within a shallow water estuary.

In the upper bay, net ecosystem metabolism becomes more negative as freshwater loading increases. A negative net metabolism value implies that an allochthonous source of organic matter is being respired, and that daily respiration is higher than photosynthesis. This organic matter sink may result in higher secondary production, but an extremely large negative net ecosystem metabolism could lead to dissolved oxygen impairment as large amounts of oxygen are converted to carbon dioxide during oxidation of organic matter. Upper Lavaca bay, being located in close proximity to freshwater point sources, had the largest negative net ecosystem metabolism response to increased freshwater inflow. Multiple freshwater point sources present at Lavaca Bay (i.e. rivers and streams) may have led to the relatively larger variability in net ecosystem metabolism during lower freshwater inflow periods. Shallow depths in the upper bay may also have contributed to variability due to the effects of changing daily irradiance on benthic primary production during low inflow periods when water clarity tends to increase. Upper bay health and function, even with the increased variability at lower freshwater inflows, seem to be primarily driven by levels of freshwater loading, but causality cannot be drawn from these results due to use of correlation statistical analysis.

The lower bay, which likely receives less organic matter, has a more balanced to slightly positive net ecosystem metabolism with increased freshwater loading. A balanced net ecosystem metabolism implies that lower Lavaca bay doesn't act as a sink or source of organic matter. A positive net metabolism value implies that autochthonous organic matter is being produced, and the ecosystem is a net source of organic matter. Autochthonous matter production may be the result of increased nutrient input from periods of increased freshwater flow. The two large positive net ecosystem metabolism values during a period of high freshwater inflow occurred at station 6. Net ecosystem values closer to zero were found at station 4 during the same freshwater inflow period. Upper bay conditions may push down into the lower bay where station 4 is located during very high freshwater inflows. Station 4 may act as a transition between upper and lower bay results during high freshwater inflows. If we separated the station 4 results from stations 5 and 6 we could tentatively conclude that the lower bay has a large positive net ecosystem response during high freshwater periods. The lack of replicate samples at station 5 and 6 during high freshwater inflows, however, means that further research will be needed before valid conclusions about lower bay net ecosystem dynamics can be made. Autochthonous matter production in lower Lavaca bay could, if severe, lead to eutrophic conditions and occurrences of harmful algal blooms, but this is usually prevented in Lavaca bay by wind and tidal flushing, and a well mixed water column. The deeper depths of the lower bay and the spatial separation from freshwater inflow point sources implies that water column processes will dominate and tidal forcing may be more important here than in the upper bay. The lack of significance in the relationship between freshwater loading and whole ecosystem metabolism implies that other factors are more important than freshwater loading this far away from freshwater inflow point sources. Which factors are important, however, are still unknown.

These findings conclude that freshwater loading drives ecosystem function in shallow water estuaries. The location within an estuary, however, is important in describing this relationship. Whole ecosystem metabolism provides an indicator of ecosystem health and function but is also a direct estimate of the biological processing of oxygen. Total maximum daily load programs for dissolved oxygen impairment could use the techniques and relationships between freshwater inflow and net ecosystem metabolism generated during this study and apply them to keep estuarine ecosystem metabolism in balance. Future research efforts include conducting broader scale studies to quantify the temporal and spatial variability in net ecosystem metabolism's relationship with freshwater inflow. The larger range of environmental conditions captured during this future research will be used to produce a practical integrated watershed level modeling tool for management of estuarine dissolved oxygen concentrations, health, and function.

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Table 1a. Monitoring dates by station with results for net ecosystem metabolism (NEM), mean daily salinity (Sal.), temperature (Temp.), and depth, as well as chlorophyll-a (Chl.-a).

Date	Sta	NEM	Sal. (ppt)	Temp. (°C)	Depth (m)	Chl.-a (ug l^{-1})	Date	Sta	NEM	Sal. (ppt)	Temp. (°C)	Depth (m)	Chl.-a (ug l^{-1})
4/24/2002	1	-2.8	12.99	26.70	0.72	15.48	4/15/2003	6	-0.95	22.58	22.08	1.22	
4/24/2002	2	1.12	15.33	26.75	0.76	8.84	5/28/2003	1	1.17	17.86	26.62	0.83	5.47
4/24/2002	4	-1.43	22.74	26.36	0.85	8.64	5/28/2003	1	1.01	17.48	26.54	0.80	5.47
4/24/2002	5	-0.48	22.74	26.36	0.85	6.64	5/28/2003	2	3.43	18.70	26.70	0.84	7.15
4/24/2002	6	-0.55	24.58	26.51	1.28	3.88	5/28/2003	3	1.36	20.87	26.98	1.02	6
5/22/2002	1	-1.37	18.58	23.43	0.85	15.84	5/28/2003	4	1.37	22.33	27.05	1.06	6.85
5/22/2002	4	-1.35	24.81	23.51	1.23	13.24	5/28/2003	4	1.52	22.30	27.03	1.07	6.85
5/22/2002	5	-0.49	24.82	23.42	0.86	9.26	5/28/2003	5	1.13	23.41	27.23	0.86	2.33
5/22/2002	5	-1.31	25.18	23.44	0.89	9.26	5/28/2003	6	1.61	24.21	27.28	1.19	7.69
5/22/2002	6	-1.07	26.90	23.62	1.22	10.96	7/22/2003	2	-0.68	9.58	30.99	0.63	
8/21/2002	1	-1.94	9.48	30.21	0.65	14.16	7/22/2003	3	-1.84	9.72	30.83	1.00	
8/21/2002	2	-0.7	11.10	30.32	0.71	15.38	7/22/2003	4	-2.10	10.49	30.88	0.84	
8/21/2002	2	-1.41	11.19	30.26	0.74	15.38	7/22/2003	4	-1.76	10.48	30.88	0.84	
8/21/2002	4	-1.30	12.74	30.37	0.87	9.71	7/22/2003	6	-0.53	22.00	30.68	0.92	
8/21/2002	4	-0.70	12.81	30.31	0.91	9.71	8/19/2003	1	-0.02	13.33	30.86	0.63	
8/21/2002	5	-1.05	17.87	30.33	0.77	8.41	8/19/2003	1	0.49	13.74	30.94	0.64	
8/21/2002	6	-0.06	18.97	30.31	1.12	9.36	8/19/2003	2	0.23	17.83	30.91	0.60	
10/9/2002	1	0.4	12.88	27.13	0.79	20.4	8/19/2003	5	0.27	24.54	30.80	0.78	
10/9/2002	1	0.13	13.01	27.07	0.86	20.4	8/19/2003	6	0.43	25.54	30.71	0.99	
10/9/2002	2	-0.86	15.12	27.34	0.80	17.83	9/23/2003	1	-2.83	1.20	25.40	0.68	6.32
10/9/2002	4	-0.80	17.89	27.31	1.06	12.92	9/23/2003	1	-2.21	1.18	25.40	0.68	6.32
10/9/2002	5	-0.82	19.87	27.46	0.89	8.62	9/23/2003	2	-2.54	5.77	25.73	0.77	10.08
10/9/2002	6	-0.44	21.03	27.62	1.20	10	9/23/2003	3	-2.89	7.32	25.68	1.10	10.64
3/18/2003	1	-1.33	10.50	21.53	0.78	17.82	9/23/2003	4	-0.26	8.50	25.52	1.07	10.99
3/18/2003	2	-0.01	14.42	21.48	0.74	6.24	9/23/2003	4	-0.90	8.42	25.61	1.06	10.99
3/18/2003	3	-0.15	15.22	21.07	1.20	6.33	9/23/2003	6	3.1	19.66	26.06	1.11	12.28
3/18/2003	6	-0.13	19.71	20.89	1.12	5.62							
4/15/2003	1	-0.51	14.95	22.71	0.80								
4/15/2003	2	-0.71	18.73	22.63	0.83								
4/15/2003	2	-1.13	18.55	22.60	0.85								
4/15/2003	3	-1.82	17.98	22.26	1.13								
4/15/2003	4	-1.15	20.70	22.12	1.18								

Table 1b. Monitoring dates by station listing results for water column ammonium (NH_4), phophate (PO_4), silicate (SIO_4), and nitrate plus nitrite (NN) in umol l^{-1}, sediment total nitrogen (Tot.N), total carbon (Tot.C), and total organic carbon (TOC) in percent of total sediment, and sediment composition as a proportion of total sediment.

Date	Sta	NH_4	PO_4	SIO_4	NN	Tot.N %	Tot.C %	TOC %	Rubble	Sand	Silt	Clay
4/24/2002	1	0.81	1.05	75.39	2.68	0.068	0.786	0.628	0.009	0.457	0.346	0.188
4/24/2002	2	0.01	0.46	65.73	0.41	0.057	1.242	0.596	0.017	0.243	0.495	0.245
4/24/2002	4	0.04	0.6	63.34	0.46	0.097	1.421	0.882	0.006	0.147	0.524	0.323
4/24/2002	5	0	0.63	45.19	0.42	0.039	12.401	10.454	0.943	0.023	0.013	0.021
4/24/2002	6	0.75	2.3	26.27	2.31	0.098	1.468	0.805	0.005	0.122	0.559	0.314
3/18/2003	1	0.28	0.4	41.6	0.4							
3/18/2003	2	0.28	0.49	46.38	0.53							
3/18/2003	3	0.31	0.34	55.22	0.11							
3/18/2003	6	0.66	0.01	4.05	0.3							
4/15/2003	1					0.094	1.082	0.813	0.012	0.545	0.395	0.048
4/15/2003	2					0.047	0.950	0.528	0.017	0.375	0.529	0.078
4/15/2003	3					0.127	1.647	1.132	0.015	0.241	0.649	0.094
4/15/2003	4					0.103	1.428	0.866	0.006	0.116	0.749	0.129
4/15/2003	6					0.134	1.662	1.047	0.008	0.112	0.753	0.127
5/28/2003	1	0.28	0.62	69.79	0.53							
5/28/2003	2	0.27	0.66	76.83	0.62							
5/28/2003	3	1.14	0.51	50.49	0.45							
5/28/2003	4	0.44	0.33	31.68	1.12							
5/28/2003	5	2.06	0.62	35.4	1.52							
5/28/2003	6	0.41	0.47	31.88	0.79							
9/23/2003	1	6.005	5.515	266.885	7.38							
9/23/2003	2	9.46	3.135	220.76	6.135							
9/23/2003	3	8.545	3.468	194.615	8.34							
9/23/2003	4	7.788	2.78	187.18	5.51							
9/23/2003	6	1.84	1.975	145.49	3.165							

Table 2. Stations sampled for net ecosystem metabolism. T. C. E. Q. descriptions and locations.

Assessment Unit	**Station No.**		**Short Description**	**Latitude (N)**	**Longitude (W)**
	TCEQ	**UTMSI**			
Upper-Bay	17552	LB 1	Lavaca Bay So. of Garcitas Cove	28.69683456	96.64499664
Upper-Bay	17553	LB 2	Lavaca Bay West of Point Comfort	28.67436218	96.58280182
Upper-Bay	13383	LB 3	Lavaca Bay at SH 35	28.63888931	96.60916901
Lower-Bay	17554	LB 4	Lavaca Bay East of Noble Point	28.63933372	96.58449554
Lower-Bay	13384	LB 5	Lavaca Bay at 'Y' at CM 66	28.59583282	96.56250000
Lower-Bay	17555	LB 6	Lavaca Bay South of Rhodes Pt.	28.59769440	96.51602173

Table 3. Wind dependent and constant diffusion coefficient (K) equations. Diffusion coefficients (K) are in g O_2 m^{-2} h^{-1}. Odum and Wilson; and Marino and Howarth estuarine subset equations estimated from graphs.

Author(s)	Location(s)	Wind Speed Range (m s^{-1})	Equation X = Wind Speed	Variability Explained (%)
Odum and Wilson, 1962	Texas Gulf Coast	0-12	0.2x	NA
Marino and Howarth, 1993	World Wide Full data set	0-12	$0.1098e^{(0.249x)}$	55
Marino and Howarth, 1993	Estuarine data subset	0-12	$e^{(1.00 + 0.4x)}$	NA
D'Avanzo et al., 1996	Waquoit Bay	NA	$0.56e^{(0.15x)}$	NA
Caffrey, 2004	NERR sites	0-10	0.5	NA

Fig. 1. Map of 24 hour data sonde deployment at U. T. M. S. I. stations in Lavaca Bay.

Fig. 2a. Environmental condition PCA scores.

Fig. 2b. Environmental condition PCA loads.

Fig. 3. One way anova of chl.-a by station with Tukey's minimum significant difference = ± 3.7 as error bars.

Fig. 4. Net ecosystem metabolism vs. chlorophyll-a linear regression.

Fig. 5a. Water column nutrient PCA scores (Circled area contains scores during high freshwater inflow).

Fig. 5b. Water column nutrient PCA scores close up.

Fig. 5c. Water column nutrient PCA loads.

Fig. 6a. Sediment characteristics PCA scores.

Fig. 6b. Sediment characteristics PCA loads.

Fig. 7. Upper Bay net ecosystem metabolism vs. freshwater inflow.

Fig. 8. Lower Bay net ecosystem metabolism vs. freshwater inflow.

Fig. 9. Cumulative ten day prior to date gauged freshwater inflow into Lavaca Bay, Texas. (Circles denote sample dates.)

Fig. 10. Mean daily salinity vs. cumulative freshwater inflow from ten days prior to sample date. (Labeled by U. T. M. S. I. station number.)

Fig. 11. Wind dependent and constant diffusion coefficients (K) vs. wind speed.

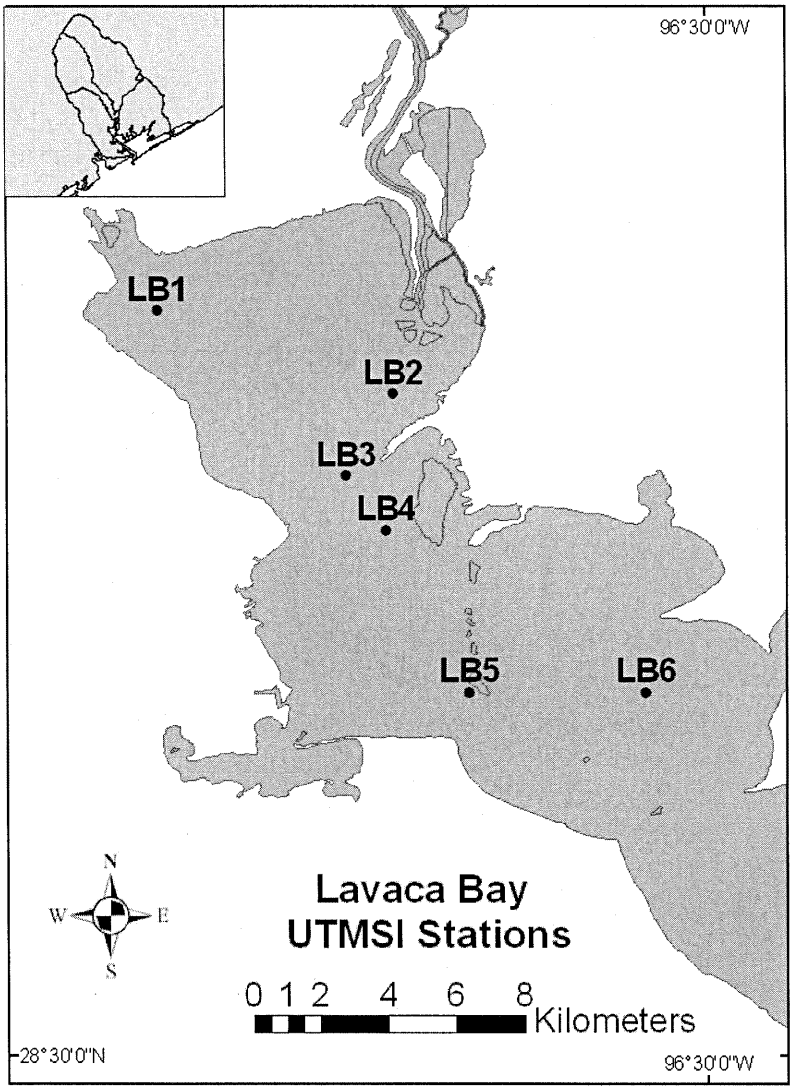


Fig. 1. Russell et al.

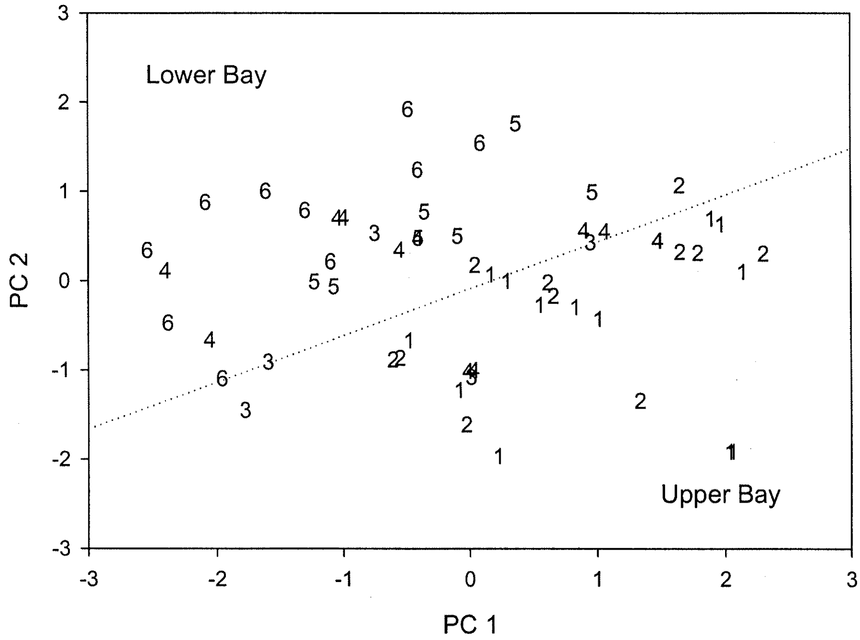


Fig. 2a. Russell et al.

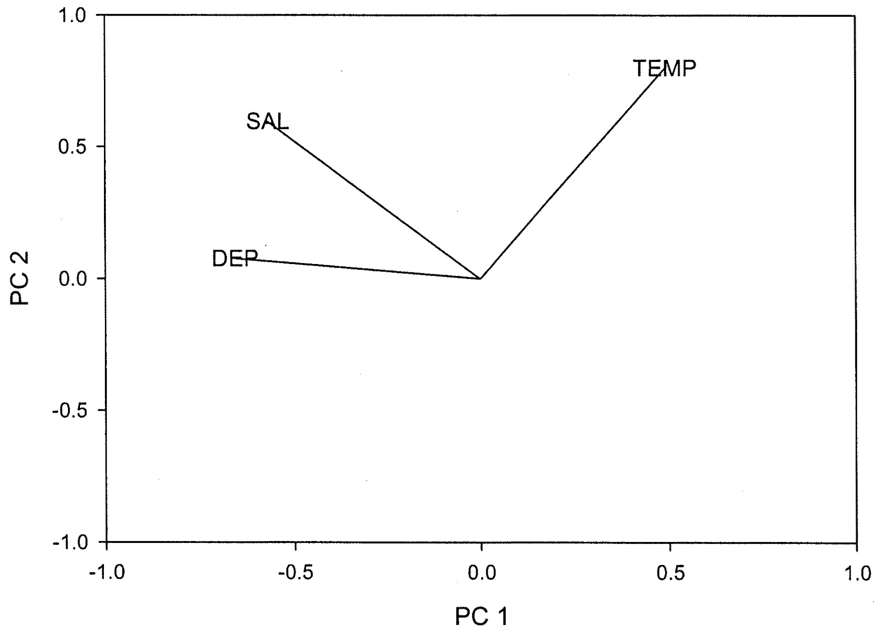


Fig. 2b. Russell et al.

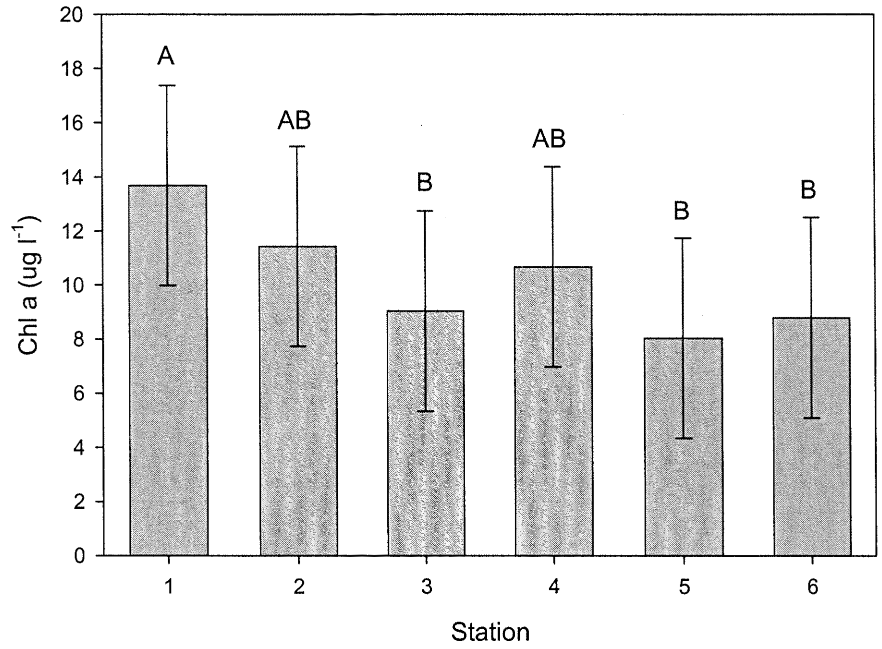


Fig 3. Russell et al.

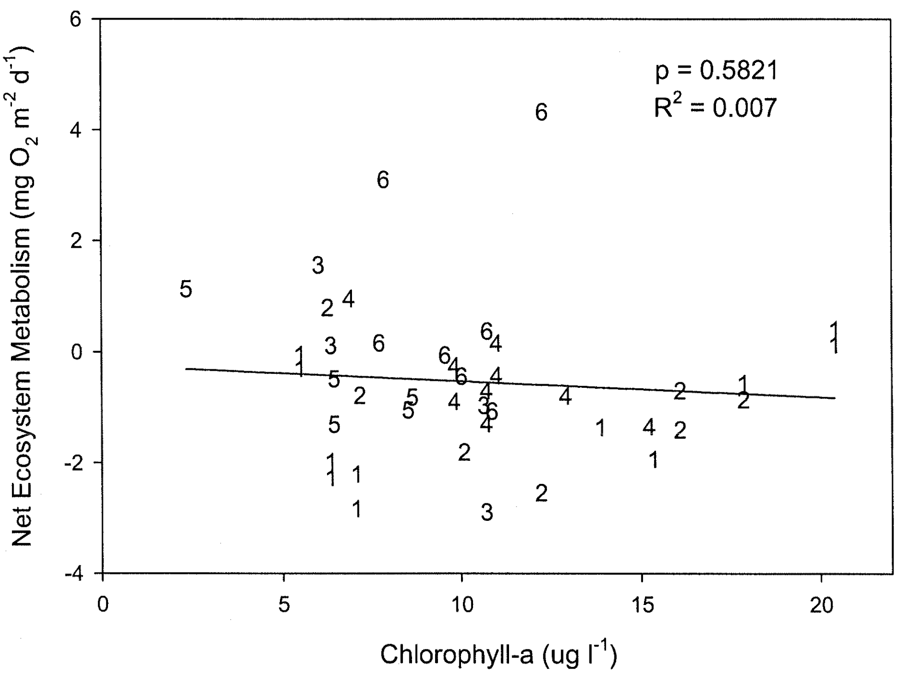


Fig. 4. Russell et al.

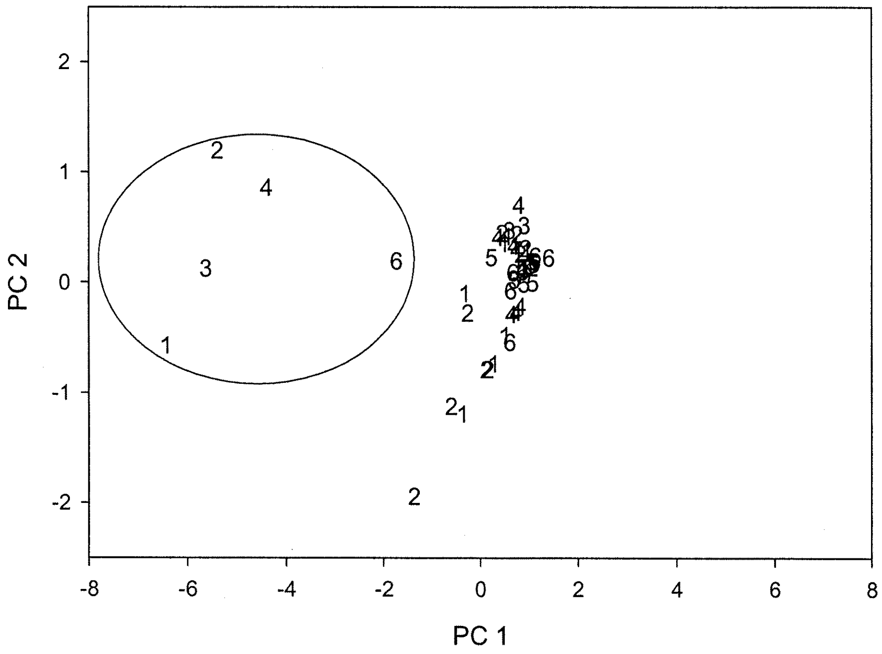


Fig. 5a. Russell et al.

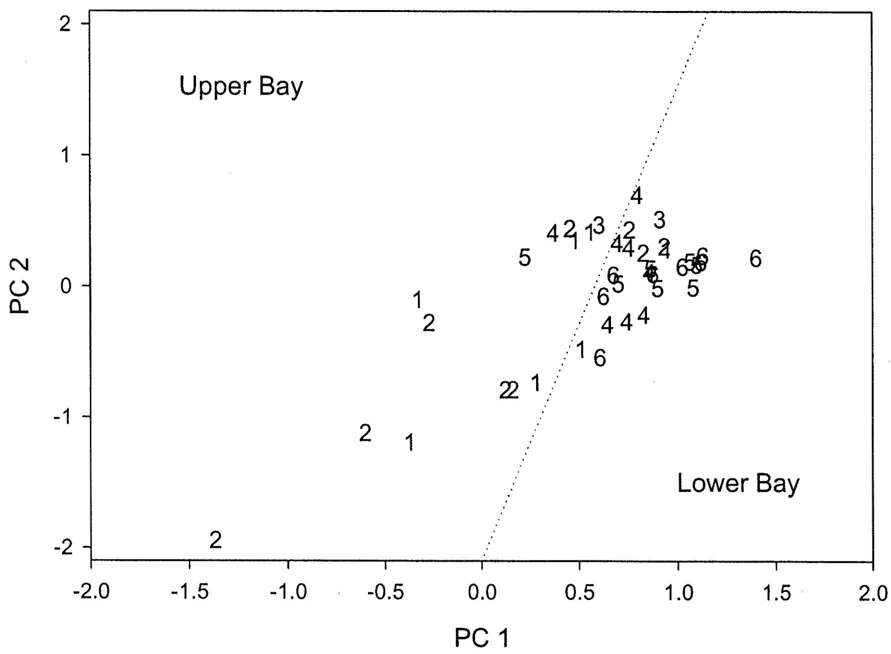


Fig. 5b. Russell et al.

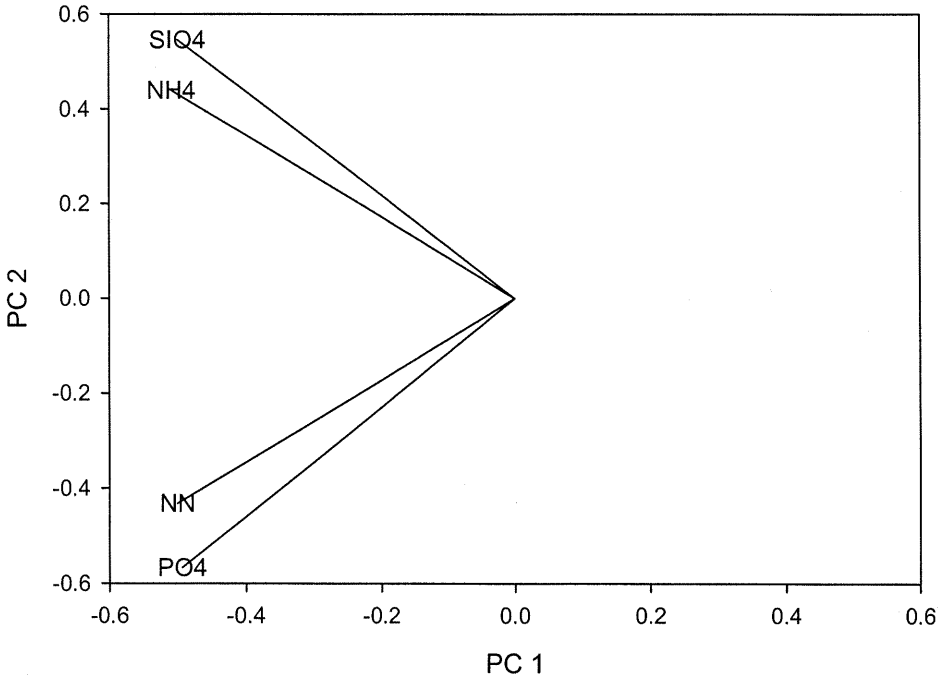


Fig. 5c. Russell et al.

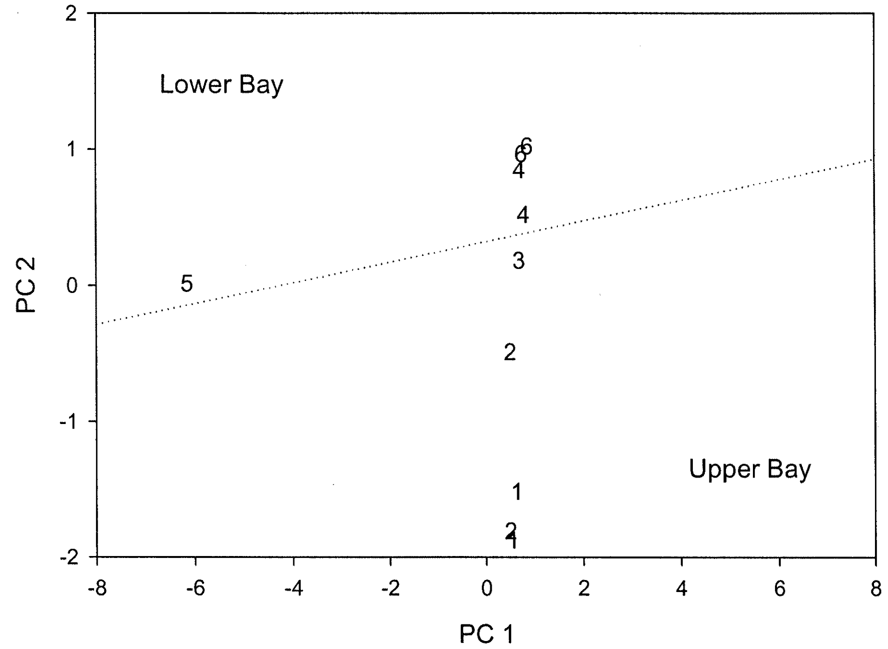


Fig. 6a. Russell et al.

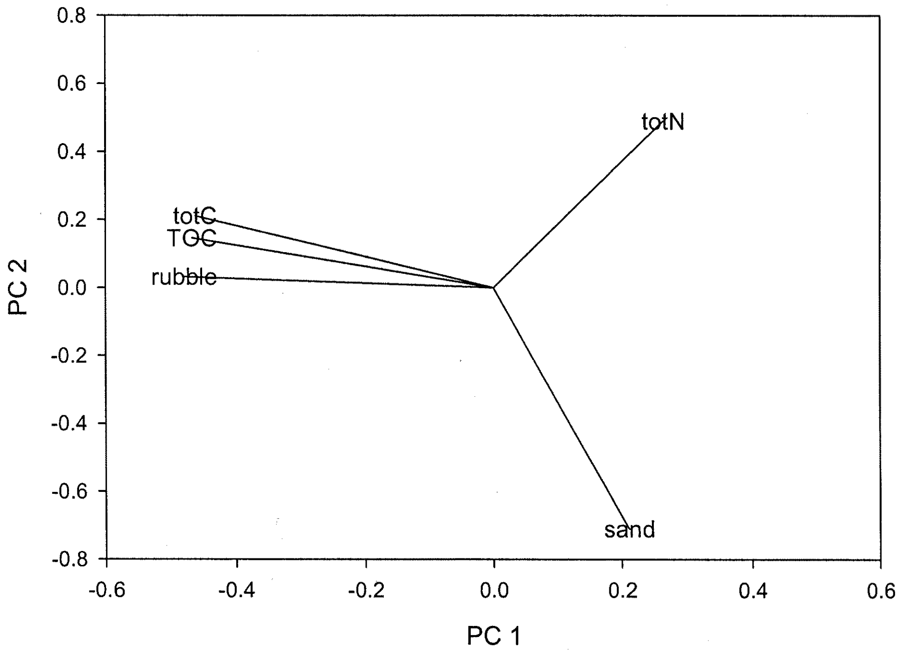
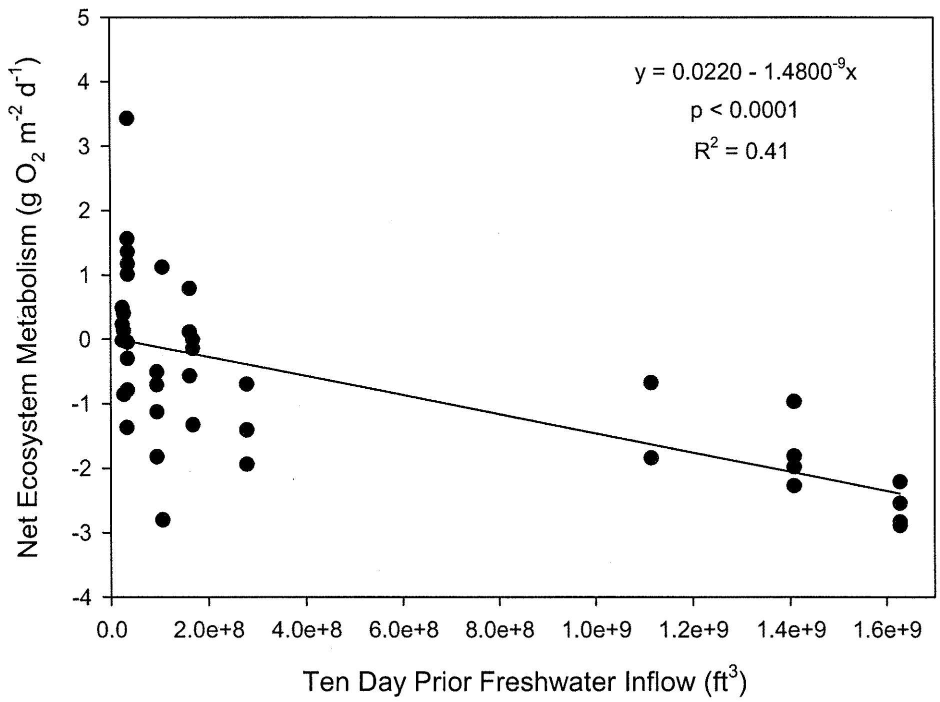


Fig. 6b. Russell et al.



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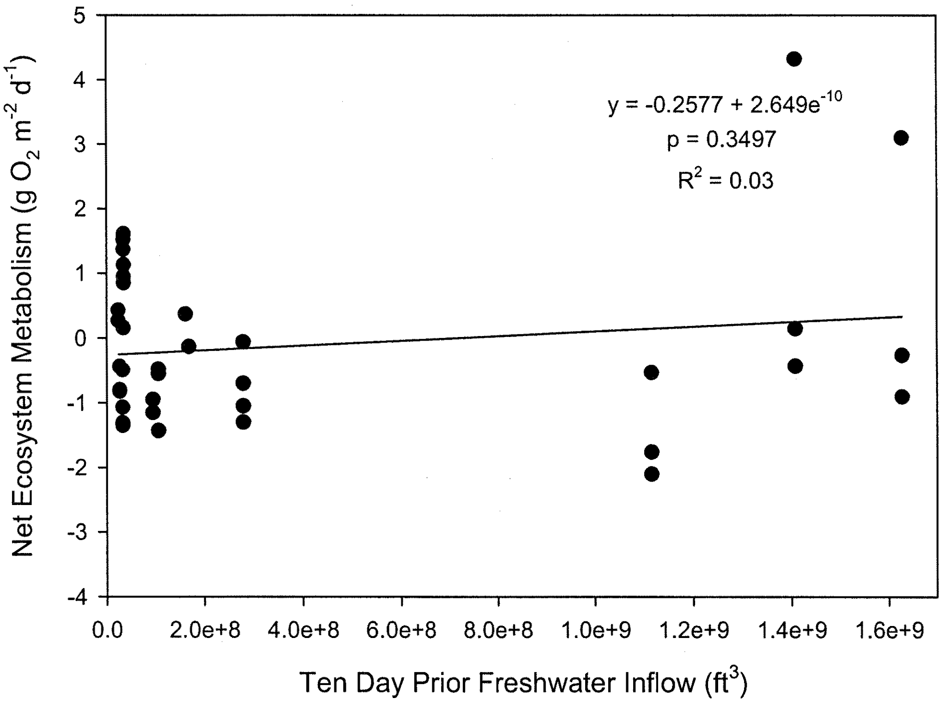

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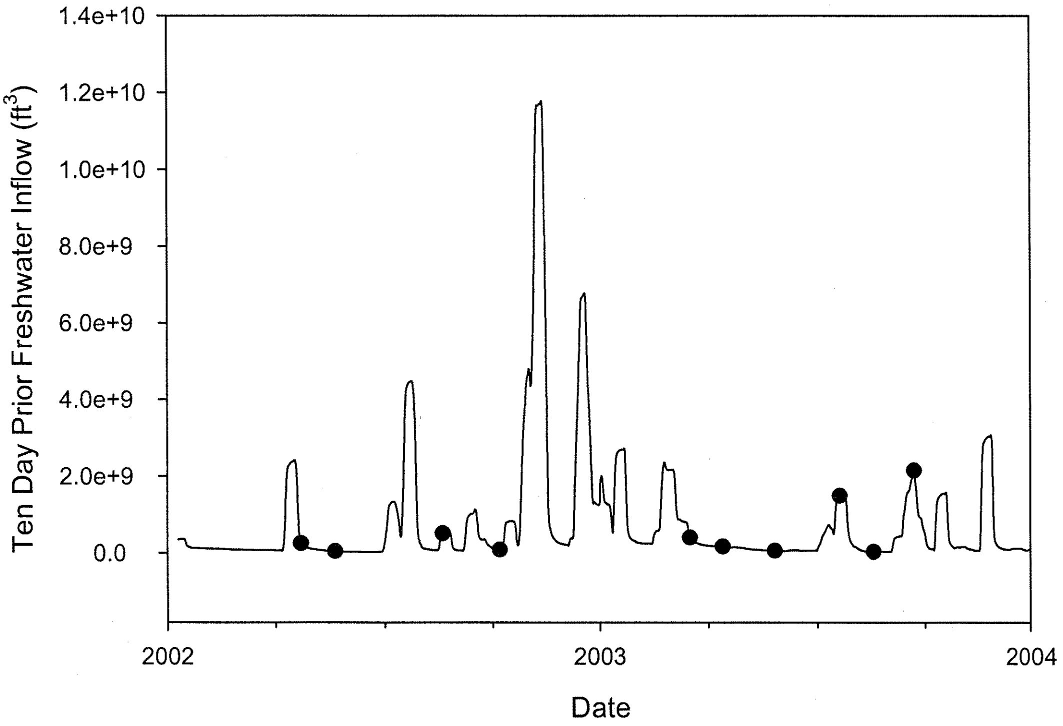

Fig. 9. Russell et al.

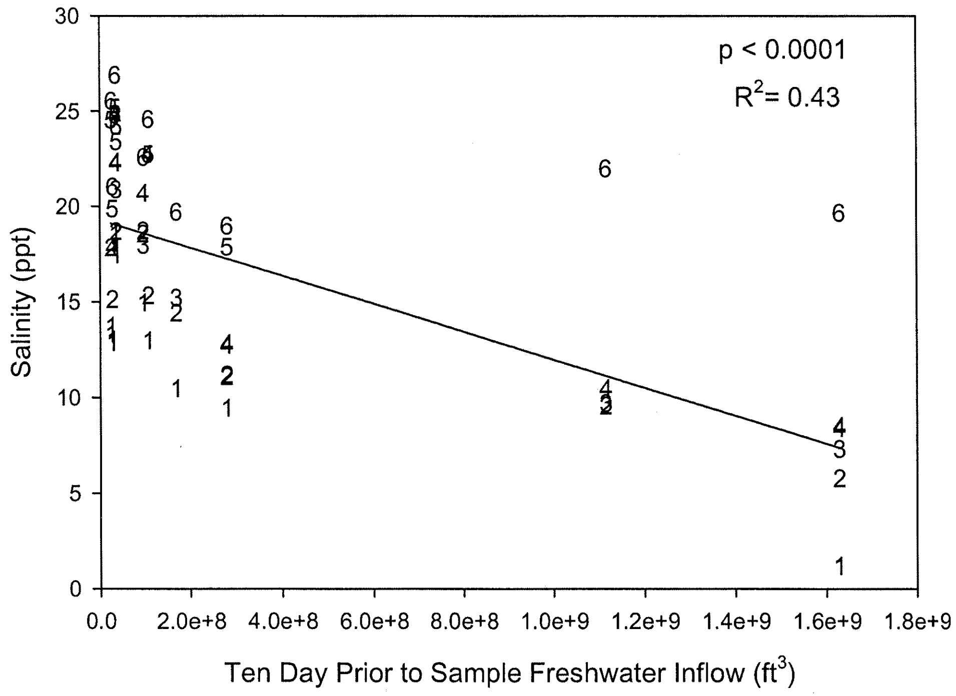

Fig. 10. Russell et al.

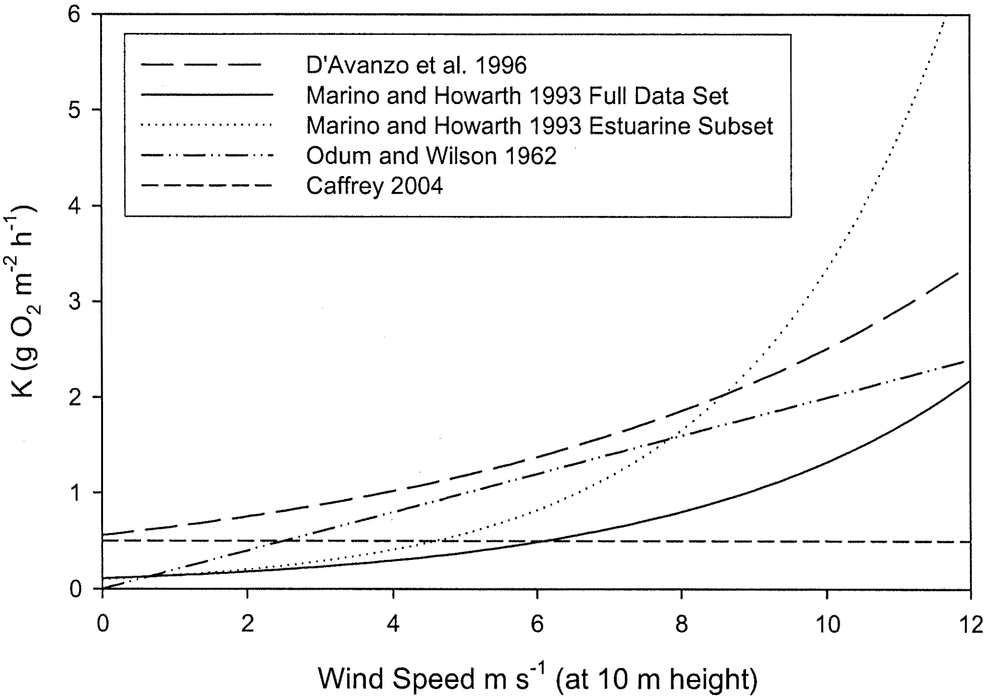

Fig. 11. Russell et al.