Ecological and Socio-Economic Impacts of Watershed Development on Coastal and Marine Areas

Paul A. Montagna, Principal Investigator Hae-Cheol Kim, Assistant Research Scientist Terry Palmer, Research Assistant Jennifer Beseres Pollack, Assistant Research Scientist

Harte Research Institute for Gulf of Mexico Studies Texas A&M University - Corpus Christi 6300 Ocean Drive, Unit 5869 Corpus Christi, Texas 78412-5869

> Phone: (361) 825-2040 Email: paul.montagna@tamucc.edu



Final report submitted to:

Korea Maritime Institute 1652, Sangam-dong, Mapo-gu Seoul, 121-270 KOREA

November 13, 2010

Cite as:

Montagna, P.A., H.-C. Kim, T. Palmer, and J. B. Pollack. 2010. Ecological and Socioeconomic Impacts on Coastal and Marine Areas from Watershed Development. Final Report to the Korea Maritime Institute. Harte Research Institute, Texas A&M University-Corpus Christi, Corpus Christi, Texas, 106 p.

Table of Contents

List of Figures	iii
List of Tables	vi
Acknowledgements	vii
Abstract	viii
Introduction	1
Problem and Needs	1
Conceptual Model of Estuary Ecosystems	5
Estuaries	5
Human Interactions	
Hydrology and the Water Cycle	
Tides and Residence Times	
Estuarine Condition and Water Column Effects	
Salinity	
Sediments	
Nutrients	
Biological Indicators	
Case Studies	
Case Study 1: Texas, USA	
Physical Background	
History of Freshwater Management	
Recent Relevant Research	
Case Study 2: Florida, USA	
Water Management District Profiles	
Florida Water Resource Policy	
District Management Approaches	
Study 1: Southwest Florida Mollusks	
Study 2: St. John's River Macroinvertebrates	

Study 3: Tri-State Water Sharing Issues Within the Apalachicola-Chattahoochee-Flint Bas	in
	54
Case Study 3: California, USA	59
Regional Water Quality Control Board Profiles6	50
Study 1: San Francisco Bay6	52
Study 2: Colorado River	56
Case Study 4: Australia	71
The Murray- Darling River Basin	77
History of Murray-Darling Management	79
Case Study 5:South Africa	33
Mtata Estuary	35
Summary: Water Supply, People, and the Future	37
References	39

List of Figures

Figure 1. Intercepted continental runoff entrains 3-6 times more water in reservoirs than exis natural rivers (MEA 2005).	ts in 1
Figure 2. Effects of altered inflow on estuaries (Montagna et al. 1996).	7
Figure 3. Conceptual model of inflow effects (modified from Alber 2002).	8
Figure 4. Relationship between conceptual model of inflow effects and the water quality risk assessment paradigm.	9
Figure 5. Location of Texas estuaries (Longley 1994).	23
Figure 6. Catchments of the seven major Texas estuaries and two major River estuaries (Rio Grande and Brazos River). Only portions of the catchments that are in Texas are included in map.	24
Figure 7. Precipitation gradient within Texas (data source: Texas General Land Office)	25
Figure 8. Mean annual inflow balance from 1941-1994 in Texas estuaries from northeast to southwest.	27
Figure 9. Average monthly inflow balance (1941 - 1994) in Texas estuaries. Sabine-Neches Estuary, Trinity-San Jacinto Estuary, Lavaca-Colorado Estuary, Guadalupe Estuary Mission-Aransas Estuary, Nueces Estuary, and Laguna Madre Estuary	y, 28
Figure 10. Total inflow to Texas Coast and Southern Oscillation Index (SOI). Bottom panel contains trends smoothed using polynomial regression and weights computed from Gaussian density.	n the 30
Figure 11. The Texas environmental flows allocation process overview. Arrow direction indicates flow of information. (adapted from Hess, 2010)	34
Figure 12. Map of the areas assigned to each biological indicator for evaluation of flow effect on the suitability of the salinity conditions (from TJBBEST 2009)	ts 36
Figure 13. Principal component analysis (PCA) showing relationships between environmenta variables in Lavaca-Colorado Estuary, Texas (from Pollack et al. 2009)	ıl 38
Figure 14. Predicted percent change from mean biomass for both deposit and suspension feed (solid circles) based on changes (-50 % to 50 %) of long-term mean salinity and nutrients in (a) Lavaca Bay and (b) Matagorda Bay (from Kim & Montagna 2008).	lers
Figure 15. Macrofauna community response as a function of salinity in Rincon Bayou, Texas (from Montagna et al. 2002b). Abundance (A), biomass (B), and N1 diversity (C). Nonlinear response to salinity (solid line) at each time period	42

Figure 16	. Cumulative inflow versus macrofaunal biomass in Rincon Bayou, Texas from Montagna et al. (2002b)
Figure 17	. Location of Water Management Districts, Florida, USA
Figure 18	. Map of the west coast of Florida showing the study sites
Figure 19	. Relationship between total mollusc diversity (A) and abundance (B) vs. salinity at Myakka River and diversity (C) and abundance (D) versus salinity at Peace River. Circles, Hill's N1 diversity index; squares, abundance. From Montagna et al. (2008a).
Figure 20.	. Mollusc species abundances versus salinity in southwestern Florida estuaries (from Montagna et al. (2008a). A) Corbicula fluminea; B) Polymesoda caroliniana; C) Rangia cuneata; and D) Tagelus plebeius
Figure 21	. Location of sample sites along the St. John's River, Florida, USA (from Montagna et al. 2008b)
Figure 22.	. Figure 21. Salinity versus A) Abundance and B) Hills N1 diversity in the St. John's Estuary
Figure 23.	Non-metric Multi-dimensional Scaling plot of SAV stations labeled by station and substrate type; inside (In) and outside (Ou) grass beds. Stations listed in Figure 20. Similarity contours from cluster analysis are overlaid
Figure 24	. Location of Apalachicola-Chattahoochee-Flint River system and drainage basin in Georgia, Alabama, and Florida, USA
Figure 25.	. Rainfall gradient in California. Average annual precipitation (inches) from 1961- 1990. Map from Oregon Climate Service: http://www.wrcc.dri.edu/pcpn/ca.gif 59
Figure 26	. Location of the 9 California Regional Water Quality Control Boards
Figure 27.	. San Francisco Bay Estuary, California, showing the different subregions (Suisun Bay, North Bay / San Pablo Bay, Central Bay, and South Bay)
Figure 28.	Colorado River, stretching through the southwestern US states of Colorado, Utah, Arizona, Nevada and California en route to Mexico and the Gulf of California (from Varady et al. 2001)
Figure 29	. Mean annual rainfall in Australia from 1961 – 1990 (from Australian Bureau of Meteorology)
Figure 30.	. Mean evapotranspiration in Australia from 1961 – 1990 (from Australian Bureau of Meteorology)
Figure 31.	. The geographical distribution of estuaries based on climate and tidal range (from Digby et al. 1999)

Figure 32. Murray Darling Basin showing states and major rivers. ACT = Australian Capital	
Territory.	78
Figure 33. Generic procedures for the determination of Resource Directed Measures	84
Figure 34. Mtata esturay, South Africa	86
Figure 35. Conceptual model of inflow management with roles of stakeholders. Solid lines denote control and dashed lines denote information flow. Switch denotes adaptive	
management.	87

List of Tables

Table 1.	Selected references for salinity effects on estuarine macrobenthic and epibenthic organisms.	15
Table 2.	Climatic gradient in Texas Estuaries	26
Table 3.	Identified Biological Indicators for Evaluating Freshwater Inflow Needs to Galveston Bay (from TJBBEST 2009)	35
Table 4.	Linear correlations between sediment and hydrologic principal component factor score and biological variables in the Lavaca-Colorado Estuary from Pollack et al. (2009)	es 39
Table 5.	St. John's River oligohaline macrobenthic indicators ($B = bivalves$, $P = polychaetes$).	53
Table 6.	Timeline of Apalachicola-Chattahoochee-Flint water use issues	56
Table 7.	Potential impacts of surface water allocation on water, biological, socioeconomic and cultural resources.	58
Table 8.	Colorado River Allocations	69
Table 9.	Regional classification of Australian Estuaries (from Tomczak 2000)	74
Table 10	Economically important fishes of coastal freshwaters threatened by changes in drainage to estuaries (From Cappo et al. 1998b after Merrick and Schmidt 1984)	76
Table 11	. Evolution of the River Murray Waters Agreement, 1914 to 1981 (from Clarke 1982 and www2.mdbc.gov.au/about/history_mdbc.html)	81
Table 12	2. Potential uses for different levels of Reserve determinations	83

Acknowledgements

This work was supported by Cooperative Research Activities between the Harte Research Institute for Gulf of Mexico Studies and Korea Maritime Institute. The authors thank Drs. Jong-Hee Ghang, Chin-Soo Lim, Jongdeog Kim for organizing this opportunity. Authors also give special thanks to Mr. Jungho Nam for his inspiring leadership in developing the concept for this paper.

When you drink the water, remember the spring. Chinese Proverb

Abstract

Large water development projects in Korea could have effects on coastal resources. Like all other countries in the World, water is an important resource to Korea, and water needs will grow as the population grows.

It is useful to review what is known about the importance of freshwater inflow to estuaries so that coastal resources can be managed wisely. Fresh water is important in diluting marine water in estuaries, which are the most productive environments on Earth. Fresh water also delivers nutrients and sediments to the coast. Estuaries are characterized by salinity and nutrient gradients, which are important in regulating many biological processes.

As water is diverted for human consumption, it is common for many environmental problems to appear. While many countries have water quality programs, few are dealing with water quantity alterations. The first step is to define marine resources to protect, and the water quality conditions those resources need to thrive. The second step is to determine the flow regimes needed to maintain the desired water quality conditions.

Finally, many regions are using adaptive management programs to manage freshwater resources. These programs set goals to protect ecosystem resources, identify indicators, and monitor the indicators over time to ensure that the goals are appropriate and resources are protected.

By conducting *in situ* monitoring and research, scientists can more accurately determine the ecological and socio-economical impacts of changing freshwater inflows. In turn, stakeholders and managers can make well-informed decisions to successfully manage freshwater inflows to their local coasts.

Introduction

Korean society has recognized that revision of its' national marine policy is required to effectively respond to changing conditions around the marine policy realm. The new administration has a primary interest in the emerging issues of "low carbon footprint and green growth." The Korean government recently began to develop four major rivers: the Han, Nakdong, Keum, and Youngsan rivers. Coastal societies and other indirect stakeholders are concerned about the ecological and economic effects that could occur downstream from these large projects. Potential effects are associated with changes in river flow to estuaries, changes in land use patterns, and riparian modification. These effects could cause negative impacts on Korean society as a result of the river development. In addition there is uncertainty about future precipitation and the water cycle because of climate change. It is important to define potential impacts of the development and plan for adaptation or mitigation in the coastal zone. The purpose of the current document is to describe what is currently known about the ecological and socio-economic impacts of riparian development on coastal and marine areas.

Problem and Needs

Demand for freshwater by human populations is large and continues to grow. Half of the world's major cities are within 50 km of the coast, and coastal populations are 2.6 times more dense than those further inland (Crossland et al., 2005). Because of this, water scarcity and limited or reduced access to water are major challenges facing society and limiting economic development in many countries (MEA 2005). As humans further develop technologies for diverting and capturing freshwater from rivers and streams, greater reductions in freshwater delivery to the coastal zone will occur. The amount of water impounded behind dams globally has quadrupled since 1960 and three to six times as much water is held in reservoirs as in natural rivers (Figure 1; MEA 2005). One of the most severe anthropogenic impacts on coastal areas in the near future will likely be through continued interference with hydrology and water flows to the coast (Pringle et al. 2000). The potential for change in coastal ecosystems due to altered flow regimes is enormous and a growing concern world-wide.



Figure 1. Intercepted continental runoff entrains 3-6 times more water in reservoirs than exists in natural rivers (MEA 2005).

Two major forces are reshaping freshwater flows to estuaries worldwide: demographics and engineering. The coastal population is large and continues to grow, resulting in increasing demand for freshwater. Approximately 60% of the people in the United States live within 60 km of the coast, and 17 of the 20 fastest growing counties are located in coastal areas (Culliton 1998). Fresh water use in the United States has doubled since 1940 and is likely to double again by 2015 (Naiman et al. 1995). Globally, humans use about 54% of the runoff that is spatially and temporally available (Postel et al. 1996) and are having a profound effect impact on the water cycle (Vörosmarty and Sahagian 2000). As the global population continues to grow, less water will be available for instream flows, inflow to estuaries, or outflows to the coastal zone.

Dams have been constructed throughout human history, but large engineering marvels are more recent. Large dams were first built in the 1920's through 1930's to provide hydroelectric power, not water resources. Since then many large reservoirs have been built to meet an increasing population's needs for water and energy. Excepting Alaska, the hydrology of nearly every body of freshwater in the United States has been modified by dams, diversions and withdrawals (Naiman et al. 1995), and similar trends are apparent worldwide (Dynesius and Nilsson 1994; Pringle et al. 2000). For the first time in human history these large watershedscale structures have severely limited inflow to many of the world's estuaries and consequently altered functioning of these ecosystems (Montagna et al. 2002).

An estuary is defined as a semi-enclosed body of water where salt water from the ocean mixes with fresh water from rivers and land. Nothing is more fundamental to the functioning of an estuary than the amount of freshwater delivery to the mixing zone (Dahms 1990, Montagna et al. 2002a). Freshwater inflow regimes vary, but inflows are usually delivered in pulses that arrive in stochastic and complex long-term cycles. The pulses of inflow regimes have four characteristics: frequency, timing, duration, and volume. Altered freshwater inflow has driven changes in coastal ecosystem hydrology, downstream transport of nutrients and sediments, salinity regimes, and has resulted in losses of habitat, biodiversity, and productivity (Montagna and Kalke 1992, Longley 1994, Attrill et al. 1996, Mannino and Montagna 1997, Montagna et al. 2002b, Tolley et al. 2006). Maintaining the hydrological regime and natural variability of an estuary is necessary to maintain its ecological characteristics, including biodiversity.

Because freshwater inflow to estuaries is a major influence on coastal ecosystems, it is important to understand the effects caused by altered freshwater inflow and to create effective management strategies for water resource development and coastal resource management. International attention has become focused on the importance of preserving freshwater flows and the need to develop and employ standards on limitations to the reduction or alteration of flows (Istanbul Water Guide 2009). The European Union (EU) has undertaken several initiatives in recent years, the most important being the European Water Framework Directive (2000/60/EC) which aims to achieve "good ecological status" for all inland and coastal waters by 2015 through the establishment of environmental objectives and ecological targets for surface waters (WFD 2000). The South African National Water Act of 1998 requires that, for any given water

resource, sufficient water be set aside to provide for basic human needs and the protection and maintenance of aquatic ecosystems (Republic of South Africa 1998; Thompson 2006). The National Water Policy of India (2002) directs that minimum flow should be ensured in perennial streams for maintaining ecological and social considerations. Within the United States, states with large coastal populations (e.g., Texas, Georgia, California) were among the first to face the issue of environmental flows by passing legislation to protect coastal species and resources (Montagna et al. 2002a). This international attention indicates that water shortages, and the consequent reductions of environmental flows, are emerging global issues.

Climate change threatens to change precipitation and temperature patterns in vast regions of the globe. Even with no change in precipitation, increased temperature will increase evapotranspiration, thus creating water deficits in many regions. Although dewatering of estuaries at the current time is driven largely by coastal development and human demand for freshwater, current water management practices may not be adequate to cope with the impacts of climate change. Despite the uncertainty associated with global climate models, the tendency towards more widespread drought increases concomitantly for many arid and semi-arid regions of the globe, including the African Sahel and southern Africa, Central America, the Mediterranean basin, western USA, southern Asia, eastern Australia, and northeastern Brazil (Bates et al. 2008). One immediate threat of reduced precipitation is food security, which depends on irrigation. However, the greater water deficits will lead to greater dewatering of the coastal zone. If river discharge decreases, salinity of coastal ecosystems will increase and the amount of sediment and nutrient delivery will decrease, thereby altering the zonation of plant and animal species as well as the availability of freshwater for human use (Bates et al. 2008; Pollack et al. 2009).

Given the unprecedented change in the water cycle caused by human and climate systems, there are clear needs to manage water resources in the coastal zone using an ecosystembased approach to protect human health and well-being by sustaining coastal resources. Considerable scientific information is needed to manage coastal ecosystems, such as: What affect will altered freshwater inflow have on coastal resources? What are the relative magnitudes of effects driven by human activities versus climate change? The focus of management initiatives must shift to land planning efforts that conserve water, prevent polluted runoff and groundwater contamination, restore the physical integrity in aquatic ecosystems by increasing natural flow regimes, and promote and protect ecosystem services that could potentially be produced (Ruhl et al. 2003). Despite the growing consensus that the key to maintaining healthy aquatic ecosystems and the services that they provide is to preserve or restore some semblance of a natural flow regime to protect the native flora and fauna, we have continued to implement a piece-meal policy approach making such efforts exceedingly difficult (Katz 2006). The issues of what to do about environmental flows will increase in importance worldwide as developing nations further develop water resources for cities, irrigation, and industry. Creating answers to the above questions will provide policy makers and resource managers with science-based

ecosystem information and an array of options to manage environmental flows and water quantities.

Conceptual Model of Estuary Ecosystems

Estuaries

An estuary is a semi-enclosed coastal body of water, which has a free connection with the open sea and within which, sea water is measurably diluted with fresh water from land drainage (Pritchard, 1967). Most estuaries have a series of landscape subcomponents: a river (or fresh water) source, a tidal-estuarine segment, marshes (or mangroves depending on latitude), bays, and a pass (or inlet) to the sea. However, all estuaries are quite different; the landscape of each subcomponent can vary, combinations and connections of these subcomponents can vary, and some subcomponents can be missing. The interaction of three primary natural forces causes estuaries to be unique and different:

- Climate causing variability in the freshwater runoff and evaporation regimes.
- Continental geology causing variability in elevation, drainage patterns, landscapes, and seascapes.
- Tidal regime -causing differences in the degree of mixing and elevation of the mixing zone.

Because each of these three physical drivers can vary in a large number of ways, it is easy to imagine how the various combinations of these forces can combine to create a vast array of estuarine typologies. Further variability in estuarine typology is caused by the interactions of these physical drivers.

The physical differences amongst estuaries are the key to predicting the effects of fresh water alterations. Thus classifying estuarine typologies is an important first step toward understanding the need for riparian connections to the sea. In spite of the unique signatures of most estuaries, several classification schemes have been presented (Pritchard, 1967; Davies, 1973; Day et al., 1989).

Based on geomorphology, Pritchard (1952) recognized four estuary typologies: (1) drowned river valleys created by sea level change or sediment starvation in coastal plains, (2) fjords formed by glaciations, (3) bar-built estuaries formed by sediment deposition by winds and tides, and (4) tectonic estuaries caused by faults in the coastal zone. Davies (1973) recognized that there is a continuum of inlet types based on the energy expended on the coast by waves. On one end of the spectrum are lagoons that are enclosed by sandy spits and at the other end of the spectrum are deltas that are muddy and formed by river processes. Day et al. (1989) recognized that all previous definitions still do not encompass all estuarine typologies and suggested that an estuary is any coastal indentation that remains open to the sea at least intermittently and has any amount of freshwater inflow at least seasonally.

Water balance is the second important defining characteristic of estuaries. The freshwater balance is simply the sum of the water sources minus the sum of the water losses. The many sources of fresh water to the coastal zone include: rivers, streams, groundwater, direct precipitation, and non-point-source runoff. There are fewer mechanisms that cause losses of fresh water, but these primarily include evaporation and freshwater diversions for human use. Pritchard (1952) recognized three classes of estuaries based on natural hydrological processes: (1) positive estuaries where freshwater input from rain, runoff, rivers and groundwater exceeds evaporation; (2) neutral estuaries where the sources and sinks are in balance; and (3) negative or inverse estuaries where evaporation exceeds the combined sources of fresh water. Depending on climate, some systems change seasonally, being positive during rainy seasons and negative during dry seasons. Many estuaries in the world have strong year-to-year variability caused by interannual climatic variability.

Human Interactions

Human activities and water resource development can change the freshwater balance in estuaries dramatically (Figure 2). Freshwater diversions used as water supplies for large humans populations or large agricultural areas are large sinks or losses to systems. However, return flows (e.g., wastewater or industrial water) add a source of fresh water to ecosystems. In many cases the diversions and return flows can be roughly in balance if they are planned as a unit using integrated water planning. But this is rarely, if ever the case. Because many water systems depend on gravity feeds to save pumping expenses, diversions are often taken upstream and returns (minus losses to leaks and use) are put in downstream. Depending on intervening elevation and geomorphology, return flows can even be put into different watersheds. When the demand for water is large relative to the supply, the water balance can be altered significantly.

Clearly, the estuaries most at risk from human activities are those that already have a negative water balance throughout the year or during certain seasons or times. Those estuaries that are neutral but have large upstream water demands are also at great risk of degradation due to altered flow regimes. The change of fresh water volume will have profound effects on salinity in a shallow estuary (e.g., coastal plain estuaries or lagoons), but a smaller effect on a deeper estuary (e.g., fjords or tectonic estuaries). This difference of effect is often caused by shallow estuaries having smaller water volumes than deeper estuaries.



Figure 2. Effects of altered inflow on estuaries (Montagna et al. 1996).

Given that humans can now alter many factors of the water cycle, it is imperative that freshwater resources be managed effectively to protect downstream ecological resources. Beginning in the 1960's, scientists began to investigate how altered freshwater flows to the coast might affect biological resources (Copeland 1966; Hoese 1967). Since then, there have been at least two major compilations of papers on the topic: Cross and Williams (1981) and Montagna et al. (2002a). As a result of these two symposia and other work there have been two important reviews (Alber 2002; Estevez 2002) from which a conceptual model has emerged that helps us to identify inflow effects (Figure 3).



Figure 3. Conceptual model of inflow effects (modified from Alber 2002).

Following a review of the practices in three states (California, Florida, and Texas) where there is a long history of inflow studies, Alber (2002) defined the scientific framework for identifying the effects of inflow on estuarine resources. Historically, all freshwater inflow methodologies started from the perspective of hydrology or resource protection. The earlier approaches were all focused on resources such as protection of fish, charismatic, or iconic species. The problem quickly encountered is that the relationship between biology and hydrology is complex and embedded in the food web and material flow dynamics of estuaries. For example, one cannot grow fish by simply adding water to a fish tank. These experiences led to a generic framework that inflow hydrology drives estuarine condition and estuarine condition drives biological resources (Figure 3).

Ultimately, biological resources in estuaries are affected by salinity more than flow by itself. Salinity is affected by flow, but there are complexities because of the interactions between tides and geomorphology. Consequently all salinity-flow relationships are characterized with very high variance or scatter, especially in the low flow end of the spectrum. Because of the links between flow, salinity and biology, all the resource based approaches are multi-step. First, the resource to be protected is identified. Second, the salinity range or requirements of that resource are identified in both space and time. Third, the flow regime needed to support the required distribution of salinity is identified, usually using hydrodynamic and salinity transport models.

The usefulness of the environmental flow framework (Figure 3) is that estuarine resources are categorized into the familiar framework used to describe ecological health (i.e., integrity, function, and sustainability). Two new terms are added: valued resources and ecosystem services. The resources are typically called "valued ecosystem components" or VECs. These are resources that are identified by stakeholders as having aesthetic, ecological, economic, or social value. These resources include bioengineers (or foundation species) that create habitat, fisheries species, and birds. These species are typically charismatic, characteristic, or iconic to an area. Ecosystem services are the benefits provided by the environment to human health and well-being (Costanza et al 1997). It is clearly in the socio-

economic interest to sustain ecosystem services, especially those provide by VEC habitats such as oyster reefs, marshes, and seagrass beds.



Figure 4. Relationship between conceptual model of inflow effects and the water quality risk assessment paradigm.

Another important feature of the environmental flow conceptual model (Figure 3) is that it is analogous to the well accepted environmental risk assessment (or risk management) paradigm (Figure 4) that has evolved in water quality management since the 1970's. The risk assessment paradigm is also known as the pressure-state-response (PSR) model. In the water quality PSR model, the pressure is applied from a toxic substance, the state represents the presence or concentration of a substance, and response is the biological response to that state. Management actions are another way in which "Response" can be thought of. The analogy here is that flow is the pressure, estuarine condition is the state, and change in estuarine resources is the biological response (Figure 4). This is a very powerful way to think about the effects of inflow on estuarine resources, because it helps us to define the ecological health of estuaries. Assessing risk by defining health is often the first step in managing environmental resources.

Defining ecological health is a vexing issue. Consider the analogy with human health. Scientists have proven that the normal human body temperature range is 36 to 38 °C. If a person's body temperature is above this range then they have a fever, and are likely sick. This example illustrates several important principles about human health as it relates to defining ecological health and how the definition has evolved for water quality assessment (Montagna et al. 2009). It is easy to integrate the conceptual model of inflow effects and the risk assessment paradigm to provide a general reference frame to define "ecological health" (see Box). Indicators of health have to be identified and the indicators must be within an acceptable range. Two difficulties are that there are no simple indicators of ecological health; and when an indicator can be measured, there is seldom sufficient data to know what the acceptable ranges are. Also, the definition of ecological health is complex because it depends on definitions of other

Defining Ecological Health

- Ecological health is assessed by determining if <u>indicators</u> of ecological <u>conditions</u> are in an acceptable range.
- Indicators are measures (or metrics) of ecological health for which sufficient information exists to establish an acceptable range of responses across broad spatial and temporal scales.
- Ecological condition is the status of ecological <u>function</u>, <u>integrity</u>, and <u>sustainability</u>.
- Ecological function is judged acceptable when the ecosystem provides important ecological processes.
- Ecological integrity is acceptable when the ecosystem has a balanced, resilient community of organisms with biological diversity, species composition, structural redundancy, and functional processes comparable to that of natural habitats in the same region.
- Ecological sustainability is acceptable when an ecosystem maintains a desired state of ecological integrity over time.

terms (those underlined terms in the box). But in the end, the most important indicator is likely ecological sustainability. Sustainability is the ultimate definition of ecological health because an environment that is sustainable is healthy in the strict sense.

Hydrology and the Water Cycle

Water is the most widely used natural resource on Earth. However, less than one percent (0.7%) of the water on earth is fresh and of sufficient quality to be classified as drinkable. Only two-one thousandths of one percent (0.00002) is readily available in streams and lakes for humans to use to drink, bathe, or irrigate crops. The same amount of water is available today as 2,000 years ago, yet the world's population was just three percent of what it is today, thus water availability is an extreme limit to growth and prosperity (Lane et al. 2003)

The earth is often referred to as the blue planet because water covers about two-thirds of its surface. Because water is so plentiful on earth, the water cycle influences most climatic and surface geological processes. Two dominant processes drive the water cycle: evapotranspiration and precipitation. Water resource planners, however, appear to be concerned mainly with

precipitation because they can manipulate runoff. Rain over large areas interacts with land elevation to form drainage patterns and familiar landscapes, e.g., tributaries, streams, rivers, and wetlands. These drainage systems are watersheds. If the watershed is adjacent to the coastal zone, then the ultimate drainage basin is the estuary, where fresh water mixes with sea water. Freshwater runoff can, and has been, manipulated to develop hydroelectric power or create reservoirs for water storage. Altering watercourses alters riparian, wetland, and estuarine habitats (Figure 2). In coastal regions, the effect of freshwater inflow alteration depends on the type of estuary, the biological components present, and the climatic setting.

It is important to be able to build water budgets in order to manage flow to estuaries. In most places in the World, there are already abundant data on water flow and availability because of the importance of water to agriculture, cities, and industry. However, rainfall and river flow alone are not always sufficient to calculate total inflow to estuaries. Fresh water can also enter estuaries via runoff from land and through seepage of ground water. The degree to which these are important will likely be unique to each system. Initially, runoff and ground water seepage can be assumed to be small or insignificant, but eventually estimates of these inputs will be needed. Runoff is usually calculated based on models of land elevation, drainage patterns, and rainfall. Unlike surface water, ground water is difficult to observe and measure. Runoff estimates are found commonly, but groundwater inputs are rarely used in water cycle planning budgets.

Evapotranspiration is the water loss from direct evaporation of the water surface and water lost through plants. Water loss from a system due to evapotranspiration must also be known, especially in hot, dry areas where the volumes can be large. Evaporation can be measured directly by placing water in a pan and measuring the volume lost daily. Total water loss can be calculated as the product of the evaporation rate and the surface area of the water body.

Human activities and water use and reuse must also be accounted for. Water authorities usually record the amount of water withdrawn from ecosystems for human use, thus it should be relatively easy to obtain this information over a long period of time. Water is also returned to the environment after use. This is usually in the form of wastewater, but sometimes it is agricultural runoff, or industrial cooling water. These quantities should also be known and accounted for in determining the total inflows to estuaries.

Once all the basic parameters are known, the water balance can be calculated. The water balance for an estuary is simply the sum of the inputs, minus the sum of the outputs:

In this case, precipitation is directly falling on the surface of the estuary. This value will be small and insignificant in drowned river valleys, but large and significant in large coastal bays.

The natural hydrological parameters (i.e., rain, inflow, runoff, and evaporation) are driven by climate, which varies considerably within years (i.e., seasons) and among years. It is clear from the recent debate on climate change that different climatic cycles range from decades to millennia. The natural variability in river flow and levels is essential information to know for water resource management. In the end, environmental flows are likely most beneficial if they mimic the natural flow regime. The regime is composed of several characteristics including the variability of flow rates and levels. Storms create floods, which in turn create events that drive change in ecosystems. The size of these events (or disturbances) has several important characteristics: frequency, duration, extent and timing. Each of these characteristics will have a statistical distribution with a mean and variance. Frequency is how often flood events occur. Duration is how long the event lasts. Extent is the magnitude or size of the event. Timing is the seasonality of events. For example, during a recent demonstration project to divert flow back into a marsh restoration area, the frequency, timing, and duration of floods was restored; but the extent was not because large volumes of water were trapped behind a dam (Ward et al., 2002). This project demonstrates that all characteristics of freshwater inflow to estuaries have to be characterized so that the total regime can be understood.

Tides and Residence Times

In all estuaries, the dilution of seawater by the volume of fresh water inflow is affected by the amount of salt water in the system. The volume of salt water is controlled by two main factors: tides and the volume of the receiving body.

Tides are the rise and fall of the sea around the edge of the land. Tides are driven by the gravity of the moon and sun, which causes a bulge in the water on the Earth's surface.. Far out at sea, tidal changes go unnoticed, but are very important to the plants and animals that live on edge of sea, in the "intertidal zone." Tides occur in primarily hree different patterns. Diurnal tides are where there is one high and one low tide each day. Semidiurnal tides are where the rise and fall is repeated twice each day. Semidiurnal mixed tides occur where there are unequal tidal heights each day. Tidal levels are different in different parts of the world, but in general the tidal ranges can be small (less than 1 m) or great (more than 3 m). Thus, tidal range can be categorized as microtidal, mesotidal, or macrotidal.

The combination of river flow and tides means there are different types of circulation patterns in estuaries. Salt wedge estuaries are where river flow dominates the mixing and typically freshwater overflows salty water giving rise to strong stratification at a specific point. Well-mixed estuaries are where the wind dominates mixing and there is a strong salinity gradient from the river to the sea, but little to no stratification of the water column. Partially mixed estuaries are where tides dominate the mixing patterns so that there is stratification of salt water on the bottom and fresher water on the top, but this gradient can be variable along the axis of the estuary. Fjord-type estuaries are where sea water is trapped in deep parts of the bay, typically behind a sill, and there is little to no exchange with the surface water. Because of the mixing of salt and freshwater, water budgets and salt budgets are very important to understanding flow dynamics. In a water budget, the total volume transported out in a unit of time is the sum of the volume transported in by tides plus the volume of the freshwater inflow transported into the estuary from the river. Thus, the residence time (or flushing time) is the volume of the estuary ($V_{estuary}$) divided by the rate of flow of water leaving the estuary (T_{out}), i.e., Flushing Time = $V_{estuary} / T_{out}$. The flushing time is very important because it controls the carrying capacity of wastes, the flushing time for fresh water, the flushing by tidal action, the effects of mixing, and it can be affected by coastal upwelling and downwelling. Thus, flushing rate is the master variable that controls nearly all estuarine processes.

Estuarine Condition and Water Column Effects

Watershed development such as the construction of dams and withdrawal of water for human use has changed flow regimes, transport of sediments and nutrients, modified habitat, and disrupted migration routes of aquatic species (MEA 2005). These modifications to the hydrologic cycle affect the quantity, quality, and timing of freshwater inflows, and the health of estuaries. Understanding the cascading link between inflow, condition, and response (Figure 2) is the key to understanding how change driven by human and climate systems can drive resistance and resilience of biological communities.

Salinity

The salinity at any point within an estuary reflects the degree to which seawater has been diluted by freshwater inflows. Estuaries are transitional zones between freshwater and marine environments, and as such, display gradients of salinity (0 in freshwater to 35 parts per thousand in seawater) and nutrients (high in freshwater, low in seawater; Montagna et al. 2010). When less dense freshwater flows into more dense saltwater, the freshwater has a tendency to remain primarily on the surface layer (Kjerfve 1979). However, winds and tides tend to mix the water column, creating longitudinal and vertical salinity gradients within estuaries (Day et al. 1989). Estuaries can be classified based on their water balance: a) positive estuaries have freshwater input and evaporation, and c) negative estuaries have evaporation that exceeds freshwater input (Pritchard 1952). Depending on the hydrologic cycle, a system may change seasonally from being a positive to a negative estuary, or vice versa.

Water development projects can reduce the delivery of freshwater to estuaries and also affect the timing of inflow pulses, which can affect organisms adapted to the original salinity conditions. Although estuarine organisms generally have a wide salinity tolerance (euryhaline), most are located only within a portion of their salinity range. Thus, salinity gradients play a major role in determining the distribution of estuarine organisms (Table 1). Secondary production by estuarine benthic macrofauna in particular is known to increase in with increases in freshwater inflow (Montagna and Kalke 1992). Salinity gradients also can act as barriers to predators and disease. Two important oyster predators in Gulf of Mexico estuaries, the southern oyster drill *Thais haemastoma* and the stone crab *Menippe mercenaria* are intolerant of sustained salinities below 15 psu (Menzel et al. 1958, MacKenzie 1977). Freshwater inflow, depending on the volume, can dilute or even eliminate infective *Perkinsus marinus* oyster disease particles in low salinity areas (Mackin 1956, La Peyre et al. 2009). The timing of freshwater inflows is also important to estuarine organism abundance and distribution because the organisms have evolved over long periods to particular regimes of freshwater inflow and associated hydrological conditions (Montagna et al. 2002).

Authors	Organism(s) Studied	Study Location	Salinity Tolerance Results	
Chadwick & Feminella (2001)	Burrowing mayfly Hexagenia limbata	USA (Alabama)	Laboratory bioassays showed that <i>H. limbata</i> nymphs could survive elevated salinities (LC50 of 6.3 ppt at 18 °C, 2.4 ppt at 28 °C). Similar growth rates at 0,2,4, & 8 ppt.	
Saoud & Davis (2003)	Juvenile brown shrimp Farfantepenaeus aztecus	USA (Alabama)	Growth significantly higher at salinities of 8 & 12 ppt than at salinities of 2 and 4 ppt.	
Tolley et al. (2006)	Oyster reef communities of decapod crustaceans & fish	USA (Florida)	Upper stations (~20 ppt) and stations near high-flow tributaries (6- 12 m ³ s ⁻¹) were typified by decapod <i>Eurypanopeus depressus</i> & gobiid fishes. Downstream stations (~30 ppt) and stations near low- flow tributaries (0.2-2 m ³ s ⁻¹) were typified by decapods <i>E</i>	
Montagna et al. (2008a)	Southwest Florida mollusc communities	USA (Florida)	<i>Corbicula fluminea, Rangia cuneata, & Neritina usnea</i> only species to occur < 1 psu. <i>R. cuneata</i> good indicator of mesohaline salinity zones with tolerence to 20 psu. Gastropod <i>N. usnea</i> common in fresh to brackish salinities. <i>Polymesoda caroliniana</i> prese	
Montague & Ley (1993)	Submersed vegetation & benthic animals	USA (Florida)	Mean salinity ranged from ~11-31 ppt. Standard deviation of salinity was best environmental correlate of mean plant biomass and benthic animal diversity. Less biota at stations with greater fluctuations in salinity. For every 3 ppt increase in standard	
Rozas et al. (2005)	Estuarine macrobenthic community	USA (Louisiana)	Increased density and biomass with increases in freshwater inflow and reduced salinities. Salinity ranged from 1-13 psu.	
Finney (1979)	Harpacticoid copepods Tigriopus japonicus, Tachidius brevicornis, Tisbe sp.	USA (Maryland)	All species tested for response to salinities from 0-210 ppt. <i>Tigriopus</i> became dormant at 90 ppt died at 150 ppt. <i>Tachidius</i> became dormant at 60 ppt, died at 150 ppt. <i>Tisbe</i> died shortly after exposure to 45 ppt.	

Table 1. Selected references for salinity effects on estuarine macrobenthic and epibenthic organisms.

Authors	Organism(s) Studied	Study Location	Salinity Tolerance Results
Kalke & Montagna (1991)	Estuarine macrobenthic community	USA (Texas)	Chironomid larvae & polychaete <i>Hobsonia florida</i> : increased densities after freshwater inflow event (1-5 ppt). Mollusks <i>Mulinia lateralis & Macoma mitchelli</i> : increased densities & abundance during low flow event (~20 ppt). <i>Streblospio benedicti & Medioma</i>
Keiser & Aldrich (1973)	Postlarval brown shrimp <i>Penaeus</i> aztecus	USA (Texas)	Shrimp selected for salinities between 5-20 ppt.
Montagna et al. (2002b)	Estuarine macrobenthic community	USA (Texas)	Macrofauna increased abundances, biomass & diversity with increased inflow; decreased during hypersaline conditions. Macrofaunal biomass & diversity had nonlinear bell-shaped relationship with salinity: maximum biomass at ~19 ppt
Zein-Eldin (1963)	Postlarval brown shrimp <i>Penaeus</i> aztecus	USA (Texas)	In laboratory experiments with temperatures 24.5-26.0 °C, postlarvae grew equally well in salinities of 2-40 ppt.
Zein-Eldin & Aldrich (1965)	Postlarval brown shrimp <i>Penaeus</i> aztecus	USA (Texas)	In laboratory experiments with temperatures < 15 °C, postlarval survivial decreased in salinities < 5 ppt.
Allan et al. (2006)	Caridean shrimp Palaemon peringueyi	South Africa	At constant salinity of 35 ppt, respiration rate increased with increased temperature. At constant temperature of 15 °C, respiration rate increased with increased salinity.
Ferraris et al. (1994)	Snapping shrimp Alpheus viridari, Polychaete Terebellides parva, sipunculan Golfingia cylindrata	Belize	Organisms subjected to acute, repeated exposure to 25, 35, or 45 ppt. <i>A. viridari</i> hyperosmotic conformer at decreased salinity, but osmoconformer at increased saliniry. <i>G. cylindrata</i> always osmoconformer. <i>T. parva</i> always osmoconformer; decreased surviv
Lercari et al. (2002)	Sandy beach	Uruguay	Abundance, biomass, species richness, diversity & evenness

Authors	Organism(s) Studied	Study Location	Salinity Tolerance Results	
	macrobenthic community		significantly increased from salinity of ~6 ppt to salinity of ~25 ppt.	
Chollett & Bone (2007)	Estuarine macrobenthic community	Venezuela	Immediately after heavy rainfall (~25 psu), spionid polychaetes showed large increases in density & richness versus normal values (~41 psu).	
Dahms (1990)	Harpacticoid copepod Paramphiascella fulvofasciata	Germany (Helgoland)	After 2 hours, no mortality in salinities of 25-55 ppt. Almost all displayed dormant behavior < 20 ppt and > 55 ppt.	
McLeod & Wing (2008)	Bivalves Austrovenus stutchburyi & Paphies australis	New Zealand	Sustained exposure (> 30 d) to salinity < 10 ppt significantly decreased survivorship.	
Rutger & Wing (2006)	Esturaine macroinfaunal community	New Zealand	Infaunal community in low salinity regions (2-4 ppt) showed low species richness & abundance of bivalves, decapods, & Orbiniid polychaetes, but high abundance of amphipods & Nereid polychaetes compared to higher salinity regions (12-32 ppt).	
Drake et al. (2002)	Estuarine macrobenthic community	Spain	Species richness, abundance, and biomass decreased in the upstream direction, positively correlated with salinity. Highly significant spatial variation in macrofaunal communities along the salinity gradient. Salinity range: 0-40 ppt.	
Normant & Lamprecht (2006)	Benthic amphipod Gammarus oceanicus	Baltic Sea	Low salinity basin (5-7 psu). Physiological performance examined from 5-30 psu. Feeding & metabolic rates decreased with increasing salinity; nutritive absorption increased. Faeces productoin & ammonia excretion rates decreased strongly from lowest to	

Sediments

In addition to changing salinity levels, freshwater inflow provides nutrients, sediments and organic material that are important for overall productivity of the estuary. Thus, any upstream changes in inflow will affect the amount and timing of their delivery to the estuary as well (Alber 2002). High estuarine turbidity is generally observed during high-flow periods due to elevated sediment inputs. Sediments are delivered to estuaries from rivers and streams by freshwater inflow, which helps to build and stabilize wetlands, tidal flats, and shoals (Olsen et al. 2007). Particulate matter carried by rivers also provides the primary energy source for organisms living in the estuarine environment (Day et al. 1989).

Freshwater diversion from estuaries is decreasing the delivery of water and sediment to the coastal zone. Within the continental U.S., approximately 90% of the sediment being eroded from land is stored somewhere between the river and the sea (Meade et al. 1990). Changes in sediment discharge over the past 200 years are primarily due to anthropogenic factors including a) deforestation and agriculture, b) changes in land management strategy, and c) construction of dams, diversions and levees (McKee and Baskaran 1999). Worldwide, reservoirs and water diversions have resulted in a net reduction of sediment delivery to estuaries by roughly 10%, and prevent about 30% of sediments from reaching the oceans (Syvitski et al. 2005; Vörösmarty et al. 2003).

Nutrients

The nutrient content of freshwater flows entering estuarine waters is important since it is closely linked to primary production (Valiela 1995). In estuarine systems, nitrogen is the principal limiting element, followed by phosphorus. The addition of nutrients to estuaries is a natural process which has been greatly enhanced by human activities. In recent decades, population growth, agricultural practices, wastewater treatment plants, urban runoff, and the burning of fossil fuels have greatly increased nutrient inputs over the levels that occur naturally (Bricker et al. 1999). The concentrations of nutrients in estuaries are dynamic in space and time as a function of inputs and outputs from river flows and oceanic exchange as well as biological uptake and regeneration (Day et al. 1989). Salinity is generally an inverse indicator of the availability of land-derived nutrients, with low salinities (high freshwater inflow) linked to high nutrient concentrations (Pollack et al. 2009; Montagna et al. 2010).

Freshwater inflow can enrich estuarine nutrients and increase primary and secondary production (Livingston et al. 1997, Brock 2001). Conversely, decreased inflow has been linked to decreased rates of both primary and secondary production (Drinkwater and Frank 1994). Excess loading of nutrients to coastal waters can cause dense, long-lived algal blooms that block sunlight to submerged aquatic vegetation. The decay of these blooms consumes oxygen that was once available to fish and shellfish, which, can result in anoxic or hypoxic conditions (Rabalais and Nixon 2002). Excess nutrients can thus cause degraded water quality and affect the use of estuarine resources such as fishing, swimming, and boating (Bricker et al. 1999).

Biological Indicators

Change in freshwater inflow to an estuary not only changes the salinity of an estuary, but also nutrient concentrations. Increases in freshwater inflow usually lead to an increase in bioavailable nutrients, which in turn stimulate primary production. This primary production is often in the form of phytoplankton growth. The phytoplankton growth in turn stimulates secondary production by organisms such as zooplankton and benthic suspension feeders. Following the increase in secondary production, there is often an increase in tertiary production by organisms such as shrimps and fishes. This process is an oversimplification of the biological response to an increase in freshwater inflows. Every estuary and coastal zone is different and complex food webs exist rather than simple food chains. Because it is impossible to determine the exact changes in every population as they respond to changes in freshwater inflow, we instead approximate biological effects using biological indicators.

Biological indicators are individual species or communities of species that integrate changes in the environment so that when monitored, can indicate changes or stability in a particular environment. We expect indicator organisms to do for us today what canaries did for miners in the 18th and 19th centuries. Indicator organisms should have at least five characteristics that make them useful in applied research (Soule, 1988). 1) They should direct our attention to qualities of the environment. 2) They should give us a sign that some characteristic is present. 3) They should express a generalization about the environment. 4) They should suggest a cause, outcome or remedy. 5) Finally, they should show a need for action.

Benthic invertebrate communities have been widely used as indicators of ecological health in environmental assessment, pollution detection, and ecological monitoring studies. The benthic community is unique among coastal and marine organisms for several reasons. First, they are predominantly permanent residents of estuaries, unlike much of the more visible nekton that are made up of large populations of migratory organisms. Second, they are relatively longlived compared to plankton. Third, the benthos are relatively immobile and fixed in space, unlike nekton and plankton that move freely or with currents. In addition, everything dies and ends up in the detrital food chain, which is utilized by the benthos. Because of gravity, there is a record of all environmental change in the sediments, and benthos are commonly referred to as the "memory" of the ecosystem because this record of past events is layered in the sediments. This combination of characteristics means that the benthic community integrates changes in ecosystems over long time scales. Benthos are therefore the best sentinel group responding to changes in external conditions without the complication of movement to different regions of the coastal zone. Because benthic organisms are relatively immobile, they are usually the first organisms affected by environmental stress. Many ecological monitoring programs use benthic abundance, biomass, and diversity as ecological indicators of the state, productivity, or health with respect to changes in the environment.

Diverse and abundant populations of benthic invertebrates provide a necessary food source for many aquatic and terrestrial species. Because of the importance of benthic organisms

in the estuarine food-chain, fluctuations in their abundance can influence recruitment patterns in coastal fisheries and avian migratory behavior. Therefore, it is important to continuously monitor the abundance and diversity of benthic infauna within an estuarine system.

There are good ecological conceptual models that provide a scientific basis for interpreting the data generated in benthic monitoring and detection studies. These approaches utilize many single species, community studies, and statistical models. One of the most important concepts is the succession model proposed by Rhoads et al. (1978). They applied scientific theories of ecological succession and its relation to productivity to suggest ways that dredge-spoil could be managed to enhance productivity. The same year, Pearson and Rosenberg (1978) published a review showing how benthic community succession changed in relation to organic enrichment. The central tenant of this theory is that distance from a pollution source is analogous to time since a natural disturbance. Thus, the sequence of colonization and succession events that occur after a disturbance are similar to the changes in communities observed with distance from a pollution source. There is typically a gradient from smaller, less diverse, pioneering species limited to surface sediments to larger, more diverse, climax assemblages of deeper dwelling organisms. The gradient changes over both distance from a pollution source or is represented by community development over time after a disturbance. Thus, we have a scientific justification for benthic community structure and biodiversity studies as an assessment tool. Since these two classic studies, numerous other studies have demonstrated the value of benthic communities as an excellent indicator of environmental health

Ecological health can be defined by benthic metrics employing this series of linked definitions. The condition of integrity is assessed when community structure and diversity are stable over long periods of time. The condition of function is assessed when biomass (the best indicator of productivity) is stable over long time periods. These three metrics: abundance, biomass, and species diversity are easily obtainable in routine sampling programs.

An important objective of many resource agencies is to quantify the relationship between bioindicators of marine resource populations and freshwater inflows to bays and estuaries. However, there is year-to-year variability in population densities and successional events in estuarine communities. This year-to-year variability is apparently driven by long-term, and global-scale climatic events. For example, El Niño affects rates of precipitation and concomitantly rates of freshwater inflow along the Texas coast, which in turn influences salinity patterns in Texas bays (Tolan, 2007). Therefore, the best approach is to document long-term changes in populations and communities that are influenced by freshwater inflow. The best indicator of productivity is the change in biomass of the community over time (Banse and Moser 1980). Based on initial sampling of one to four years of benthic data in Texas bays, it was originally concluded that inflow does increase benthic productivity (Kalke and Montagna, 1991; Montagna and Kalke, 1992; 1995). However, further analysis of the data set over a 5-year period demonstrated that the largest effect may not be on productivity, but may be on community structure (Montagna and Li, 1996). This implies that reduced inflows may not only reduce

productivity (a measure of ecosystem function), but may also change the composition of species in an estuary (a measure of ecosystem structure).

Texas Coastal Bend estuaries were studied over a 20 year period by Montagna (2008) to determine the long-term response of benthic organisms to freshwater inflow. Results show that the biological effects on benthic communities appear to be driven by the El Niño cycle. Flood conditions introduce nutrient rich waters into the estuary that result in lower salinity. During El Niño periods the lowest salinities and highest nutrient values were recorded. During these periods the spatial extent of the freshwater fauna is increased, and the estuarine fauna replaced the marine fauna in the lower end of the estuary. The high level of nutrients stimulated a burst of benthic productivity (of predominantly freshwater and estuarine organisms), which lasted about six months. This was followed by a transition to a drought period with low inflow resulting in higher salinities, lower nutrients, dominance by marine fauna, decreased productivity, and decreased abundances.

Florida Bay was examined to determine the relationship between commercially important pink shrimp (*Farfantepenaeus duorarum*) recruitment and freshwater inflow characteristics (Browder et al., 2002). Experiments were conducted to determine the rates of juvenile shrimp growth and survival at varying temperatures and salinities and the results were used to refine an existing model of potential pink shrimp recruitment. Results showed high survival over a wide salinity range except at extreme temperatures. In particular, shrimp were least tolerant of high salinity at low temperatures and low salinity at high temperatures. Maintenance of freshwater inflow was found to be key in providing favorable salinities over the greatest amount of suitable and accessible habitat. Timing of flows in relation to arrival of postlarvae from offshore spawning grounds was also found to be important (Browder et al., 2002).

Case Studies

There have been at least two major compilations of research on the topic of freshwater inflow. There was a symposium convened in 1980 in San Antonio, Texas to identify the issues regarding "Freshwater Inflow to Estuaries" (Cross and Williams 1981). The goal of the symposium was to identify potential solutions and recommendations to deal with the issues of altered inflow regimes. A second symposium was convened in 2001 in St. Pete Beach, Florida entitled "Freshwater Inflow: Science, Policy and Management" (Montagna et al. 2002a). The second symposium is notable because in the intervening 21 years, many agencies began to implement freshwater inflow rules and regulations, performed research on the effects of the rules, and even attempted to restore estuaries where inflow was reduced. One important aspect of nearly all the minimum freshwater inflow rules is that they are amenable to adaptive management. Results from ongoing monitoring and assessment programs are used to modify and optimize the operating decisions. This is very important. Following is a brief summary of case studies in Texas, Florida, California, Australia, and South Africa.

Case Study 1: Texas, USA

Physical Background

There are seven major estuarine systems along 600 km of coastline (Figure 5, Longley, 1994). All seven Texas estuaries have similar geomorphic structure and physiography. Barrier islands are parallel to the mainland along the coast. Between the islands and the mainland there are lagoons. The lagoons are interrupted with drowned river valleys that form the bay and estuarine systems. There are Gulf inlets through the barrier islands, which connect the sea with the lagoon behind the island. The lagoon opens to a large primary bay. There is a constriction between the primary bay and the smaller secondary bay. Most bays are fed by just one or two rivers draining watersheds (Figure 6). The river generally flows into the secondary bay and thus secondary bays have greater freshwater influence. Primary bays provide the connection with the Gulf of Mexico and thus have greater marine influence.

The Texas coast is bounded by Sabine River (border to Louisiana) in the northeast and the Rio Grande (border with Mexico) in the southwest (Figure 5). From northeast to southwest, the major bay-estuarine systems are the Sabine-Neches Estuary, Trinity-San Jacinto Estuary, Lavaca-Colorado Estuary, Guadalupe Estuary, Mission-Aransas Estuary, Nueces Estuary, and Laguna Madre Estuary. Laguna Madre is actually two different systems: Upper Laguna Madre/Baffin Bay and Lower Laguna Madre. Texas follows the traditional system of naming an estuary for the river(s) that dilute sea water (Longely 1994). In NOAA publications (e.g., Orlando et al. 1993), these systems are named after the primary bay (Sabine Lake, Galveston Bay, Matagorda Bay, San Antonio Bay, Aransas Bay, Corpus Christi Bay, and Laguna Madre, respectively). There are also two riverine estuaries, the Brazos River and the Rio Grande, which flow directly to the Gulf of Mexico (Figure 6).



Figure 5. Location of Texas estuaries (Longley 1994).



Figure 6. Catchments of the seven major Texas estuaries and two major River estuaries (Rio Grande and Brazos River). Only portions of the catchments that are in Texas are included in map.



Figure 7. Precipitation gradient within Texas (data source: Texas General Land Office).

The estuaries of Texas are remarkably hydrologically diverse in spite of similar geomorphology. This is due to a climatic gradient, which influences freshwater inflow to estuaries. The gradient of decreasing rainfall, and concomitant freshwater inflow, from northeast to southwest, is the most distinctive feature of the coastline (Figure 7, Table 2). Along this gradient, rainfall decreases by a factor of two, but inflow balance decreases by almost two orders of magnitude. Inflow balance is the sum of freshwater inputs (gaged, modeled runoff, direct precipitation, plus return flows) minus the outputs (diversions and evaporation). The net effect is a gradient with estuaries with similar physical characteristics but a declining salinity gradient.

Table 2. Climatic gradient in Texas Estuaries.

Listed from north to south: area at mean low tide (Diener, 1975), average annual precipitation (1951-1980; Larkin and Bomar, 1983), average annual freshwater inflow balance (1941-1994; Texas Water Development Board), average salinity (Orlando et al., 1993), and average annual commercial harvest (1962-1998; McEachron and Fuls 1996, Texas Parks and Wildlife Department 1988, Robinson et al. 2000).

					Commercial Harvest	
Estuary	Area (km ²)	Rainfall (cm y ⁻¹)	Inflow $(10^6 \text{ m}^3 \text{ y}^{-1})$	Salinity (ppt)	Finfish (10^3 kg y^{-1})	Shellfish (10^3 kg y^{-1})
Sabine-Neches	183	142	16,894	8	3	341
Trinity-San Jacinto	1,416	112	13,495	16	176	4352
Lavaca-Colorado	1,158	102	3,679	18	59	2531
Guadalupe	551	91	2,677	16	63	1846
Mission-Aransas	453	81	278	15	140	1947
Nueces	433	76	346	23	173	840
Laguna Madre	1,139	69	-880	36	677	163


Figure 8. Mean annual inflow balance from 1941-1994 in Texas estuaries from northeast to southwest.

SN=Sabine-Neches Estuary, TJ=Trinity-San Jacinto Estuary, LC=Lavaca-Colorado Estuary, GE=Guadalupe Estuary, MA=Mission-Aransas Estuary, NC=Nueces Estuary, and LM=Laguna Madre Estuary. Data sourced from the Texas Water Development Board (http://midgewater.twdb.state.tx.us/bays_estuaries/hydrologypage.html)

Freshwater inflow balance patterns appear to group into four distinct climatic subregions, which vary by about an order of magnitude each (Figure 8). The northeastern most subregion is composed of the Sabine-Neches Estuary (containing Sabine Lake) and the Trinity-San Jacinto Estuary (containing Galveston Bay). This northeastern subregion has the highest rainfall and inflow balance greater than 10^{10} m³ y⁻¹. The next three climatic subregions form the largest area, the Coastal Bend, which is composed of five estuaries linked by large lagoons. The Coastal Bend estuaries are in an area bounded by the Colorado River and Rio Grande. The most well known lagoonal estuary is Laguna Madre. The climatic subregions are distinct in several ways. Most important is a lack of connection between the watersheds, thus each bay system is fed by different rivers. The Intracoastal Water Way provides a man-made, dredged channel linking all subregions. The Lavaca-Colorado Estuary (containing Matagorda Bay) and Guadalupe Estuary (containing San Antonio Bay) have an average inflow rate of about 10^9 m³ y⁻¹. The Mission-Aransas Estuary (containing Aransas Bay) and Nueces Estuary (containing Corpus Christi Bay) have an average inflow rate of about 10^8 m³ y⁻¹.

evaporation exceeds inputs and has an average negative inflow rate of about $10^8 \text{ m}^3 \text{ y}^{-1}$. Thus the region spans positive, neutral and negative estuaries.



Figure 9. Average monthly inflow balance (1941 - 1994) in Texas estuaries. Sabine-Neches Estuary, Trinity-San Jacinto Estuary, Lavaca-Colorado Estuary, Guadalupe Estuary, Mission-Aransas Estuary, Nueces Estuary, and Laguna Madre Estuary.

There is also a concomitant gradient of different timing of peak inflow events (Figure 9). The northern estuaries receive peak inflow during the spring, the central estuaries are bimodal receiving peak inflows during the spring and fall, and the southern-most estuaries receive peak inflows during the fall. These distinct patterns are very important ecologically, because growth, reproduction, and migration of many species is keyed to seasonal events. The timing and magnitude of inundation is believed to regulate finfish and shellfish production (Texas Dept. Water Resources, 1982). The differences within and among the subregions and estuaries of Texas provides a sufficiently broad scale to examine effects of climate change and variability on ecological processes.

The latitudinal gradient of decreasing inflow into estuaries regulates salinity. As well as a latitudinal climatic gradient, there is a longitudinal salinity gradient within each estuary. The salinity gradient within and among the estuaries has already been demonstrated to regulate the

infaunal molluscan community (Montagna and Kalke, 1995). There are also salinity gradients within the estuaries from the river mouth to the sea, which influences the zonation of communities found within the estuaries (Kalke and Montagna 1991; Montagna and Kalke, 1992; 1995). The interactions among the geophysical factors of climate, estuarine physiography and diversity of habitat types in the Gulf of Mexico are factors that influence diversity of the region.

Another characteristic of Texas estuaries is the extreme year-to-year variability in inflow (Figure 10). Consequently, salinity gradients within estuaries vary from year-to-year. The southwestern estuaries in particular appear to be in a nearly desert climate that is punctuated by flood events. The floods are caused by tropical storms or larger global climate patterns. The El Nińo Southern Oscillation (ENSO) has a strong influence on inflow to Texas estuaries. The Southern Oscillation Index (SOI; http://www.cgd.ucar.edu/cas/catalog/climind/soi.html) is negative during El Nińo events. There is an inverse correlation between SOI and total inflow to the Texas coast (-0.14, p=0004). The inverse correlation between smoothed SOI and smoothed total inflow to the Texas coast is strong (-0.47, p<0001; Figure 5). The ENSO phenomenon is only one climactic factor affecting inflow. Inflow is also influenced by tropical waves, which affect the coast from the east.



Figure 10. Total inflow to Texas Coast and Southern Oscillation Index (SOI). Bottom panel contains trends smoothed using polynomial regression and weights computed from the Gaussian density.

The Texas coast is likely an ideal area to study climate change effects on estuaries, because it already is a natural experiment. There is physical similarity among Texas estuaries, each is simple draining only one or two watersheds, and they lie in a climatic gradient that is influenced by large-scale climate patterns in the Pacific and Atlantic Oceans. Being semi-arid and semi-tropical, small changes in global temperature will likely have large effects. It is simple to posit hypotheses, e.g., drier conditions will result in estuaries more like the southwestern estuaries, and wetter conditions will result in estuaries more like those to the northeast. It will be possible to design stratified sampling programs where statistical control can be used on confounding factors, e.g., watershed drainage basins, anthropogenic inputs, Gulf of Mexico exchange, specific habitats, circulation patterns, and alterations by man.

History of Freshwater Management

The state of Texas first made efforts to examine the coast-wide problem of freshwater inflows following an almost 8-year drought in the 1950's that was so severe that 244 of 254 Texas counties were declared disaster areas (TWDB 2009) and many Texas rivers finally stopped flowing. The drought resulted in hypersalinity of estuaries, fish kills, loss of blue crabs and white shrimp, and invasions by stenohaline species (Copeland 1966; Hoese 1967). The most positive effect of the drought was the passing of the Texas Water Planning Act by Texas State Legislature in 1957. This act was amended over the next 10 years and led to the creation of a Texas Water Plan that was adopted in 1969. The plan called for an estimated 2.5 million acrefeet (3 billion m³) of supplemental freshwater inflows annually to Texas bays and estuaries. In 1967, a cooperative Bays and Estuaries program was created to collect physical, chemical, and biological data to inform water planning. This program was expanded by legislative mandate in 1975 (Senate Bill 137), which required comprehensive studies of the effects of freshwater inflows on bays and estuaries. The results of these studies were published in a series of reports and summarized by the Texas Department of Water Resources (1982). Although the reports included preliminary estimates of freshwater inflow needs by each bay, it was generally accepted that the databases underlying those estimates were too limited to be used for water management (Longley 1994).

The need for further research on water management was supported by bills that were passed in 1985 (House Bill 2) and 1987 (Senate Bill 683) and as a consequence recommendations were developed to continue estuarine monitoring programs to provide better data to inform state water planning and permitting decisions (Longley 1994, Powell et al., 2002). In 1995, the state legislature extended the monitoring mandate and added new language to the Texas Water Code to "maintain those conditions considered necessary to maintain beneficial inflows to any affected bay or estuary system", where 'beneficial inflows' describe "a salinity, nutrient, and sediment loading regime adequate to maintain and ecologically sound environment" (Texas Water Code 11.147 (a)(b)). Currently, state agency staff coordinates with water authorities, consultants, and university scientists to investigate each major estuary system using a standard protocol to develop freshwater inflow recommendations for implementation in state water planning and permitting processes. The Texas Parks and Wildlife Department, Coastal Fisheries Division, has an extensive monitoring program for fish in all Texas bays. The Texas Water Development Board monitors and collates river inflow and bay hydrographic data

to estimates flows to the coast. These data are used in periodic assessments to revise inflow targets.

Texas legislators did not give a definition of an "ecologically sound environment", but many interpretations have been made. An estuary can be considered ecologically sound when the physical, biological, and chemical parameters that are measured fall within the range of values that existed before human interference (Longley, 1994). Ecological soundness can also be characterized as an environment that has several trophic levels through which nutrients are cycled, so that production of commercially and recreationally important fisheries is maintained. With these goals in mind, Texas state agencies have been working to investigate the relationships between freshwater inflow and salinity, nutrient and sediment loadings, to determine the conditions required for and ecologically sound environment, and to develop methods to quantify freshwater inflow needs (Longley, 1994).

There is also more complex way to define ecological soundness based on the concepts the U.S. Environmental Protection Agency uses for defining assessment of ecological health in the water quality context. Ecological health is assessed by determining if <u>indicators</u> of ecological <u>conditions</u> are in an acceptable range. <u>Indicators</u> are measures (or metrics) of ecological health for which sufficient information exists to establish an acceptable range of responses across broad spatial and temporal scales. Ecological <u>condition</u> is the status of ecological <u>function</u>, <u>integrity</u>, and <u>sustainability</u>. Ecological <u>function</u> is judged acceptable when the ecosystem provides important ecological processes. Ecological <u>integrity</u> is acceptable when the ecosystem has a balanced, resilient community of organisms with biological diversity, species composition, structural redundancy, and functional processes comparable to that of natural habitats in the same region. Ecological <u>sustainability</u> is acceptable when an ecosystem maintains a desired state of ecological integrity over time.

Ecological health or soundness can be defined by benthic metrics employing this series of linked definitions. The condition of integrity is assessed when community structure and diversity are stable over long time periods. The condition of function is assessed when biomass (the best indicator of productivity) is stable over long time periods. These three metrics: abundance, biomass, and species are easily obtainable in routine sampling programs.

There was great concern that the existing policies were not protective of environmental flows, so a study was commissioned to assess environmental flows state-wide (SAC 2004). When no action was taken in the 2005 Legislative session, the Governor requested an update to the study (SAC 2006). In 2007, the Texas Legislature made a substantial change to water policy with the passage of Senate Bill 3 (SB3). Senate Bill 3 requires that any new water permits contain, to the extent possible, a set aside for an environmental flow regime. The new regulatory approach to protect water courses requires environmental flow standards to be developed by the Texas Commission on Environmental Quality (TCEQ) rulemaking process. Senate Bill 3 directed the use of an environmental flow regime in developing flow standards and defined an environmental flow regime as a schedule of flow quantities that reflects seasonal and yearly

fluctuations that typically would vary geographically, by specific location in a watershed, and that are shown to be adequate to support a sound ecological environment and to maintain the productivity, extent, and persistence of key aquatic habitats.

As a part of SB3, the Environmental Flows Advisory Group (EFAG) was created 'in recognition of the importance that the ecological soundness of our riverine, bay, and estuary systems and riparian lands has on the economy, health, and well-being of the state' (Texas Water Code, Section 11.0236). The EFAG is composed of policy makers: three Texas House members, three Texas Senate members, and the three heads of the natural resource agencies. A Science Advisory Committee (SAC) was created to advise the EFAG (Figure 11). The main role of the SAC is in recommending technical guidance to assist in a consistent approach state-wide. To fulfill this role, the SAC has created nine guidance documents (SAC 2009a, 2009b, 2009c, 2009c, 2009c, 2009c, 2010b, and 2010c).

SB3 created an adaptive management process whereby scientific and stakeholder groups are formed for individual estuary/basin systems called Basin and Bay Expert Science Team (BBEST), and Basin and Bay Stakeholder Committee (BBSC) respectively. These estuary-focused groups interact with each other to inform the Texas state agencies, and ultimately the Texas Commission on Environmental Quality, of their recommendations for the quantity, quality and sources of freshwater inflow that reach each estuary (Figure 9). The science teams from the two northern most estuary systems; the Trinity and San Jacinto Rivers and Galveston Bay BBEST (Trinity-San Jacinto Estuary; TJBBEST) and Sabine and Neches Rivers and Sabine Lake BBEST (Sabine-Neches Estuary; SNBBEST) both submitted Final Environmental Flow Recommendations Reports in November 2009. However, as of March 2010, the flow recommendation process is just beginning for Nueces, Mission-Aransas, Guadalupe, and Lavaca-Matagorda BBESTs (Hess 2010).



Figure 11. The Texas environmental flows allocation process overview. Arrow direction indicates flow of information. (adapted from Hess, 2010).

The approach of the Trinity and San Jacinto Rivers and Galveston Bay BBEST (TJBBEST) is to use salinity as a proxy for freshwater inflow and therefore provide suitable salinity criteria rather than inflow volumes for the estuary (TJBBEST 2009). Salinity criteria were identified for several biological indicator species (Table 3) in the regions that they currently occur (Figure 12). The TJBBEST also identified applicable time-periods to implement salinity criteria for each species (period of concern) rather than apply salinity criteria permanently. The spatial and temporal salinity ranges of the biological indicator species were determined from long-term monitoring databases from Texas Parks and Wildlife Department (a state resource agency) or other monitoring studies. Most biological indicator species selected were sessile organisms, as these organisms integrate changes in water quality over time while remaining in the same place.

Table 3. Identified Biological Indicators for Evaluating Freshwater Inflow Needs to Galveston Bay (from TJBBEST 2009).

•				
	Common Name	Scientific Name	Criterion	Period of Concern
Habitat Indicator	Wild Celery	Vallisneria	<5 psu for	Spring
		americana	germination and	
			establishment	
	u	u	<10 psu for survival	Summer and Fall
Low Salinity	Atlantic Rangia	Rangia cuneata	2 – 10 psu for	Spring and Fall
Indicators			spawning and	
			larval survival	
	Gulf menhaden	Brevoortia	5 – 15 psu for	Winter and Spring
		patronus	occurrence as	
			forage fish	
	Blue catfish	Ictalurus furcatus	<10 psu for	Single pulse in
			occurrence as	winter or spring
			predator	
High Salinity	Mantis shrimp	Squilla empusa	>25 psu for	Summer - Fall
Indicators			abundance	
	Pinfish	Lagodon	>25 psu for	Summer - Fall
		rhomboides	abundance	
Oyster Health	Dermo and oyster	Dermo= Perkinsus	10 – 20 psu to	July - September
Indicators	drill impacts on	marinus	prevent excessive	
	oyster	Oyster drill=	parasitism and	
		Stramonita	predation	
		haemastoma		
		Oyster=		
		Crassostrea		
		virginica		
	u	u	<5 psu to remove	2 weeks at 10 year
			parasite load from	intervals
			central reefs	



Figure 12. Map of the areas assigned to each biological indicator for evaluation of flow effects on the suitability of the salinity conditions (from TJBBEST 2009).

The Sabine and Neches Rivers and Sabine Lake BBEST (SNBBEST) used an instream flow focus to determine optimal inflow to the Sabne-Neches Estuary (Hess 2010). The Sabine-Neches Estuary, located on the Texas-Louisiana border, is unique among Texas Estuaries in that it is the smallest major Estuary in Texas, receives the most freshwater inflow and has a median salinity of only 8 psu (Longely 1994, SNBBEST 2009). The Sabine-Neches estuary is also unique because its' salinity regime has changed more from deepening of the ship channel than because of a reduction in freshwater inflows. SNBBEST created subcommittees for research on gaging, hydrology, biology, water quality, and geomorphology. SNBBEST recommended implementing seasonal subsistence flows as the minimum amount of inflow to the estuary. Seasonal subsistence flows were calculated using existing gage flow data as being equivalent to the median of the lowest 10 % of historical base flows by season. ENBBEST also recommended seasonal base flows depending on the hydrologic condition (wet / average / dry) of the season. Seasonal pulse flows in to the estuary are also recommended but at a rate less than the current one to two events per season. The inflow approach by SNBBEST also stresses revisions of the current criteria in the future as more knowledge is attained and consequences of such recommendations are implemented.

Recent Relevant Research

An important objective of many resource agencies is to quantify the relationship between bioindicators of marine resource populations and freshwater inflows to bays and estuaries. However, there is year-to-year variability in population densities and successional events in estuarine communities. This year-to-year variability is apparently driven by long-term, and global-scale climatic events, e.g., El Niño, which affects rates of precipitation and concomitantly rates of freshwater inflow (Tolan 2007). Therefore, the best approach is to document long-term changes in populations and communities that are influenced by freshwater inflow. The best indicator of productivity is the change in biomass of the community over time (Banse and Moser 1980). Based on initial sampling of one to four years of benthic data in Texas bays, it was originally concluded that inflow does increase benthic productivity (Kalke and Montagna 1991; Montagna and Kalke 1992, 1995). However, further analysis of the data set over a 5-year period demonstrated that the largest effect may not be on productivity, but may be on community structure (Montagna and Li 1996). This implies that reduced inflows may not only reduce productivity, but may also change the composition of species in an estuary.

Texas Coastal Bend estuaries were studied over a 20 year period by Montagna (2008) to determine the long-term response of benthic organisms to freshwater inflow. Results show that the biological effects on benthic communities appear to be driven by the El Niño cycle. Flood conditions introduce nutrient rich waters into the estuary that result in lower salinity. During El Niño periods the lowest salinities and highest nutrient values were recorded. During these periods the spatial extent of the freshwater fauna is increased, and the estuarine fauna replaced the marine fauna in the lower end of the estuary. The high level of nutrients stimulated a burst of benthic productivity (of predominantly freshwater and estuarine organisms), which lasted about six months. This was followed by a transition to a drought period with low inflow resulting in higher salinities, lower nutrients, dominance by marine fauna, decreased productivity and abundances.

The Lavaca and Matagorda Bay system on the central Texas coast was studied to investigate the relationship between benthic macrofaunal community structure and sediment and hydrologic parameters (Pollack et al. 2009). A principal component analysis (PCA) was used to illustrate the relationships among water column environmental variables (Figure 13). The PC 1 variable loads have high negative values for silicate (SiO₄), dissolved inorganic nitrogen (DIN; sum of nitrate, nitrite and ammonium), and phosphate (PO₄); and high positive values for salinity. High salinity is correlated with low nutrients (and vice versa), thus PC 1 represents a linear scale of freshwater inflow effects. High temperature is correlated to low dissolved oxygen (DO) along PC 2, thus PC 2 represents seasonal effects. PC 1 (freshwater inflow effects on hydrology) was significantly correlated with macrobenthic diversity and evenness, while sediment characteristics did not have a significant relationship to any biological variables (Table 4).



Figure 13. Principal component analysis (PCA) showing relationships between environmental variables in Lavaca-Colorado Estuary, Texas (from Pollack et al. 2009).

Principal components 1 and 2 (PC 1 and PC 2) explained 47% and 20% of the variance. Sal =salinity, DIN dissolved inorganic nitrogen (sum of nitrate, nitrite and ammonium), PO4 = phosphate, SiO4 = silicate, Chl = chlorophyll, DO = dissolved oxygen, Temp = temperature.

Table 4. Linear correlations between sediment and hydrologic principal component factor scores and biological variables in the Lavaca-Colorado Estuary from Pollack et al. (2009).

D 41-1 - M-4-1 -	Sedir	nent	Water Column		
Benthic Metric	PC1	PC2	PC1	PC2	
Biomass (g m ⁻²)	-0.166	0.406	-0.164	-0.305	
	0.510	0.094	0.075	0.001	
	18	18	119	119	
Abundance (n m ⁻²)	-0.290	0.311	-0.055	0.059	
	0.242	0.209	0.556	0.525	
	18	18	119	119	
Diversity (N1)	0.055	0.445	-0.289	-0.407	
	0.829	0.064	0.001	<.0001	
	18	18	119	119	
Diversity (H')	0.159	0.452	-0.310	-0.408	
	0.528	0.060	0.001	<.0001	
	18	18	119	119	
Evenness (J')	0.193	0.291	-0.208	0.122	
	0.442	0.242	0.023	0.187	
	18	18	119	119	
FIBI PC1	0.320	0.179	-0.198	-0.068	
	0.196	0.478	0.033	0.464	
	18	18	117	117	
FIBI PC2	-0.302	0.386	-0.150	-0.515	
	0.222	0.114	0.106	<.0001	
	18	18	117	117	

Abbreviations: r = Pearson product correlation coefficient, P = probability of the null hypothesis, <math>n = number of sample pairs.

In a related study of the Matagorda Bay system, a bioenergetic model was created to predict the benthic biomass effects that could result from changes in freshwater inflow (Kim & Montagna 2009). Model simulation results were interpreted to assess the role of freshwater inflow in controlling benthic productivity. In general, when salinity increased with decreasing nutrient concentrations, deposit feeder biomass increased while suspension feeder biomass decreased (Figure 14). The change in trophic groups also indicates that functional diversity will decrease with increasing salinity. In other words, reduced inflow will change the character of the bay into one that produces mainly worms, and not worms and shellfish. Estuary-wide, the model predicts that reduced freshwater inflow could increase benthic productivity overall as upstream communities acclimate to elevated salinities and take on characteristics of downstream polyhaline communities.



Figure 14. Predicted percent change from mean biomass for both deposit and suspension feeders (solid circles) based on changes (-50 % to 50 %) of long-term mean salinity and nutrients in (a) Lavaca Bay and (b) Matagorda Bay (from Kim & Montagna 2008).

The error bars represent the standard error. The number of simulated data points during 1988 - 2005 was 69 data points for Lavaca and 72 for Matagorda Bay. The long-term mean salinity is 15.1 for Lavaca Bay and 24.1 for Matagorda Bay, and these values represent long-term mean salinity with 0% change. The long-term mean biomass for all feeders was 2.36 g dw m⁻² for Lavaca Bay, and 12.78 g dw m⁻² for Matagorda Bay, respectively, and these values represent long-term mean biomass under 0% of salinity. Labels the X-axis represent salinity in % and corresponding changes in nutrients are not shown here but were considered in the model simulations. Dashed lines represent the position of reference simulation (0% change).

The 50-month Rincon Bayou Demonstration Project is an example of a recent project investigating freshwater inflow needs of bays and estuaries in the state of Texas. The demonstration project increased the amount of freshwater into the upper Nueces Delta by ~ 732 %, reversing the hypersaline salinity gradient (> 50 ppt) to a more natural pattern (21 - 28 ppt,

Bureau of Reclamation 2000). Benthic organisms were better able to tolerate brackish salinity conditions than hypersaline conditions. As a result, total macrofauna abundance and size of individuals increased, particularly within the salinity range of 10 - 45 ppt. Macrofauna characteristics demonstrated strong nonlinear relationships with salinity (Figure 15; Montagna et al. 2002b). Macrofauna abundance appeared to peak at a salinity of 33 ‰, biomass peaked at a salinity of 19 ‰, and diversity peaked at a salinity of 9 ‰. As a secondary result, the abundant and diverse benthic community of the demonstration project provided conditions favorable for feeding and growth of fish and shrimp. Fish and shrimp utilize estuaries as seasonal nursery areas (Riera et al. 2000). A novel log-normal regression model was use to identify the salinity range and inflow range that yields the peak abundance, biomass, and diversity (Montagna et al. 2002b). When macrofaunal characteristics were regressed against non-zero cumulative quarterly inflow rates, only biomass was significantly correlated, with a peak macrofauna biomass of 4 g m⁻³ occurring at flow rates of 125,000 m³ per quarter (Figure 16). So, in this ecosystem, a little water went a long way to increase ecosystem health. Increased freshwater inflow was found to also affect the processes of nutrient cycling, increasing water column and marsh plant production.



Figure 15. Macrofauna community response as a function of salinity in Rincon Bayou, Texas (from Montagna et al. 2002b). Abundance (A), biomass (B), and N1 diversity (C). Nonlinear response to salinity (solid line) at each time period.

Circles represent periods of rising salinity and diamonds represent periods of falling salinity. Dashed line is not significant.



Figure 16. Cumulative inflow versus macrofaunal biomass in Rincon Bayou, Texas from Montagna et al. (2002b).

Circles (\circ) *represent periods of rising salinity and diamonds* (\diamond) *represent periods of falling salinity.*

At this point in time, there is no question as to whether freshwater inflow is important to coastal ecosystems. Rather, the important questions are how, when, where, and in what quantities inflow should be utilized for environmental purposes (SCWEF 2004). Scientific methods and protocols have advanced over the past 40 years to improve our understanding of the importance of freshwater inflow. One of the main conclusions of this work is that adaptive management and precautionary principle methods need to be incorporated into the scientific study, management strategy implementation, and regulatory permitting phases of future freshwater inflow studies (SCWEF 2004). This important conclusion should be applied coastwide (throughout the United States) to develop future approaches of managing freshwater inflow for adequate protection of coastal ecosystems.

Case Study 2: Florida, USA

Florida has a long history of water flow management, particularly in the southern portion of the state. In 1972, the Florida Water Resource Act was established, which created five Water Management Districts throughout the state (1972 Laws of Florida Ch. 72-299): 1) Northwest Florida Water Management District, 2) Suwannee River Water Management District, 3) St. John's River Water Management District, 4) Southwest Florida Water Management District and 5) South Florida Water Management District (Figure 17). Florida defines the need for inflows broadly as "the limit at which further withdrawals would be significantly harmful to the water resources or ecology of the area." To allow for successful planning and management of regional water resources, the districts were created with boundaries defined by watersheds rather than political margins. Each district develops a water management plan for water resources within its region that identifies their role in water resource management and provides long-term (20-year) direction. The plans are updated at least every 5 years. The result is that freshwater inflow studies and rules are unique to each estuary and the different Florida water management districts have adopted different strategies for regulating freshwater flows in response to different intensities of human engineering and societal expectations of water management (e.g. flood control versus water supply).



Figure 17. Location of Water Management Districts, Florida, USA.

Water Management District Profiles

The Northwest Florida Water Management District comprises 29,280 km² in the Florida panhandle, encompassing 16 counties and 6 major drainage basins (NFWMD 2010). The district's population in 2007 was 1.3 million residents, or approximately 7.5% of Florida's population.

The Suwannee River Water Management District is the smallest in Florida, encompassing 19,788 km² in the north-central region, and representing 15 counties and 13 river basins (SRWMD 2010). The total population of the district is expected to increase from 320,000 in 2010 to 750,000 by 2050 (SRWMD Strategic Plan 2010).

The St. John's River Water Management District comprises 32,116 km² in northeast and east-central Florida, including 18 counties and 10 major drainage basins (SJWMD 2010). The St. John's River is the longest river in Florida, measuring 499 km and flowing south to north with a downhill gradient of less than 5 cm per mile (1.6 km) from head to mouth. From 1995 to 2000, the population of the St. John's River Water Management District increased from approximately 3.5 million to 3.9 million people, an increase of about 11% (SJRWMD DWMP 2005).

The Southwest Florida Water Management District is 25,900 km², encompassing 16 counties and 11 major drainage basins in west-central Florida (SWFWMD 2010). In 2000, the district's population was approximately 4 million people (SWFWMD DWMP 2005).

The South Florida Water Management District covers 16 counties and 2 major drainage basins in the southern half of the state (SFWMD 2010). The district includes Lake Okeechobee, the Everglades Agricultural Area, and the Florida Keys. The South Florida Water Management District's population is approximately 7.5 million people.

Florida Water Resource Policy

Chapter 373 of the Florida Statutes identifies four areas of responsibility for the districts: water supply, water quality, flood protection and floodplain management, and natural systems. The Water Management Districts were given a mandate to establish minimum flows and levels for surface waters and aquifers. Minimum flow was defined as "the limit at which further withdrawals would be significantly harmful to the water resources of ecology of the area."

According to Florida Administrative Code (FAC), minimum flows and levels should be evaluated to ensure consideration of natural seasonal fluctuations in water flows or levels, nonconsumptive uses, and environmental values associated with coastal, estuarine, riverine, spring, aquatic, and wetlands ecology, including the following 10 natural resource and environmental values: 1) recreation in and on the water, 2) fish and wildlife habitats and the passage of fish, 3) estuarine resources, 4) transfer of detrital material, 5) maintenance of freshwater storage and supply, 6) aesthetic and scenic attributes, 7) filtration and absorption of nutrients and other pollutants, 8) sediment loads, 9) water quality, and 10) navigation (Section 62-40.473 F.A.C.).

In 1995, the Florida legislature concluded that none of the water management districts had fully completed the task of establishing minimum flows and levels (OPPAGA 1995), and the Florida Water Resource Act was amended in 1997 to require the development of a priority list for the establishment of minimum flows. This legislation is significant because 1) it states that resource-based criteria are central to the determination of minimum flows, 2) the objective of the minimum flow must be to sustain the resource, and 3) the determination of minimum flows is a bottom-up process with scientists determining the criteria and definitions of sustainability (Estevez 2002). The management districts have taken a variety of approaches in complying with this mandate (Alber 2002).

District Management Approaches

The Southwest Florida Water Management District has been utilizing a management approach for unimpounded rivers that limits withdrawals to a percentage of streamflow at the time of withdrawal (Flannery et al. 2002). The natural flow regime of a river is the baseline for identifying the effects of increased withdrawal levels; various streamflow parameters are then evaluated to determine changes in river flow regimes. This approach to water supply planning and regulation is designed to maintain the physical structure and ecological characteristics of unimpounded rivers. Relationships between freshwater inflow and estuarine characteristics are then examined to determine withdrawal limits that will not result in negative environmental impacts. This percent-of-flow approach was supported by initial findings that showed a curvilinear response of isohaline locations to freshwater inflow and the influence of inflow on catch per unit effort for a number of key organisms (Flannery et al. 2002). Studies of the Alafia River estuary in particular demonstrated that the abundances of many estuarine-resident and juvenile estuarine-dependent species decline during low-flow periods (Flannery et al. 2002). The Southwest Florida Water Management District is using the percent-of-flow approach to inform adaptive management, such that continued data collection is being used to improve management strategies over time.

In contrast, the South Florida Water Management District and the Suwannee River Water Management District use resource-based approaches for setting inflow requirements with the aim of protecting biodiversity, productivity, and connections to adjacent floodplains and downstream habitats (Alber 2002; Mattson 2002). In this approach, critical habitats (e.g., oyster reefs, vegetated communities, or tidal creeks), that are also referred to as valued ecosystem components (VEC), are identified as are the range of salinities experienced in these areas. The freshwater inflow regime needed to establish these salinity regimes is then determined and correlations between salinity and biological variables are examined (Mattson 2002). This management approach is tied to the concept that growth and survival of larval and juvenile fish and shellfish can be enhanced if favorable salinity and suitable physical habitat overlaps spatially at the appropriate time of year (Browder and Moore 1981). Results from oyster monitoring in the Suwannee River Estuary showed that oyster reef development is best in areas influenced by freshwater inflow, with salinity requirements and timing of inflow being important for

reproduction and successful spat settlement (Mattson, 2002). In the Caloosahatchee Estuary, the proposed minimum inflow rule is based on protecting the habitat for three seagrass species that are sensitive indicators of inflow effects (Doering et al. 2002). The most vexing problem in Florida is the Everglades, because sheet flow to this icon of the state has been reduced dramatically over the last few decades. Planning is underway for a large re-plumbing of south Florida to resolve this problem.

The St. John's River Water Management District evaluated topographic, soil, and vegetation data collected within the plant communities associated with the river in conjunction with hydrologic modeling to determine minimum flows and levels (ECT 2008). Minimum flows and levels were evaluated for 5 criteria: inundation of riparian wetlands for stream biota, saturation of hydric hammocks, maintenance of riparian hydric soils, adequate depths for fish passage and eelgrass beds, and protection of eelgrass beds from boat and canoe traffic (CH2M Hill 1999). Salinity targets were developed to protect various biological resources including hard clams *Mercenaria mercenaria*, eastern oysters *Crassostrea virginica*, fish species red drum *Sciaenops ocellatus*, snook *Centropomus undecimalis*, and spotted seatrout *Cynoscion nebulosus*, and seagrasses such as *Halodule* and *Syringodium* (Estevez and Marshall 1993).

The Northwest Florida Water Management District has not yet established minimum flows and levels for its drainage basins. Seven water bodies have been identified as priority areas for establishing minimum flows and levels: Floridian Aquifer, Inland Sand & Gravel Aquifer, Deer Point Lake, Wakulla Spring, Jackson Blue Spring, Yellow River, and Morrison Spring (NWFWMD 2010). Criteria for identification of the existence or potential for significant harm to these water bodies included: 1) potential migration of saline water due to aquifer drawdown, 2) potential future water supply, 3) reduced discharge to coastal areas, 4) aquatic resources needs, and 5) consumptive demands. Predicted dates of establishment for the minimum flows and levels to these water bodies range from 2010-2015.

Study 1: Southwest Florida Mollusks

Six tidal rivers in southwest Florida along the Gulf of Mexico were examined using a meta-analysis of existing data sets of mollusk communities to determine salinity-mollusk relationships at regional scales (Figure 18) (Montagna et al. 2008a).



Figure 18. Map of the west coast of Florida showing the study sites.

The most important variable correlated with mollusc communities was salinity, which is a proxy for freshwater inflow. Mollusk abundance and diversity relationships with salinity were examined with a non-linear model. The assumption behind the model is that there is an optimal range for salinity and values decline prior to and after meeting this maximum value. That is, the relationship resembles a bell-shaped curve. The shape of this curve can be predicted with a three-parameter, log normal model:



The model was used to characterize the nonlinear relationship between a biological characteristic (Y) and salinity (X). 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 salinity, and c the location of the peak response value on the salinity axis.

In the Peace and Myakka Rivers, there were strong relationships between mollusk diversity and abundance with salinity, where diversity and abundance increased with increasing salinity, peaked, and then declined (Figure 19).



Figure 19. Relationship between total mollusc diversity (A) and abundance (B) vs. salinity at Myakka River and diversity (C) and abundance (D) versus salinity at Peace River. Circles, Hill's N1 diversity index; squares, abundance. From Montagna et al. (2008a).

Looking at individual mollusc species, *Corbicula fluminea, Rangia cuneata, Polymesoda caroliniana, Tagelus plebius* all exhibited significant bell-shaped responses to salinity, with *C. fluminea* abundance peaking at 0.6 ppt, *R. cuneata* at 3.7 ppt, *P. caroliniana* at 4.9 ppt and *T. plebius* at 7.3 ppt (Figure 20). These salinity ranges can be used to predict changes in mollusk



assemblages in response to alterations in salinity that result from actual or simulated changes in freshwater inflow.

Figure 20. Mollusc species abundances versus salinity in southwestern Florida estuaries (from Montagna et al. (2008a). A) Corbicula fluminea; B) Polymesoda caroliniana; C) Rangia cuneata; and D) Tagelus plebeius.

Symbol key: Al, Alafia River; Cu, Curry Creek; Do, Dona/Roberts Bay; My, Myakka River; Pe, Peace River; Sk, Shakett Creek; Sh, Shell Creek; We, Weeki Wachee River.

Mollusk species appeared to be controlled more by water column hydrography than by sediment composition, with salinity as the most important environmental variable and proxy for freshwater inflow.

Study 2: St. John's River Macroinvertebrates

In 2008, an analysis was conducted using existing data in the St. John's River to investigate the relationship between freshwater inflows and estuarine macrobenthic communities. The analysis approach was similar to the meta-analysis of southwest Florida mollusc data (Montagna et al. 2008a). The spatial scale of the study included eight SAV stations within the St. John's Estuary (Figure 21).

Macrofaunal abundance and diversity peaked when salinity was 1.5 and 1.0 ppt respectively (Figure 22). The rate of change of abundance and diversity with salinity was greater when salinities were below 1.5 and 1.0 ppt respectively then when salinities were above these values. The correlations between salinity and both abundance and diversity were high but not significant.

Although the salinity ranges are small in the study area, specific species and taxa were found that are indicators of the various salinity regimes (Table 5). Diversity increases as salinity increases. No specific bivalve or polychaete indicator was found in the fresh water zone (such as Crescent Lake). Only one indicator, the bivalve *Rangia cuneata*, was an indicator for the fresh oligohaline zone (such as Rice Creek and Scratch Ankle Creek). But there were three additional species that were

found to be indicators of the salty oligohaline zone (found in the remaining stations).



Figure 21. Location of sample sites along the St. John's River, Florida, USA (from Montagna et al. 2008b).



Figure 22. Figure 21. Salinity versus A) Abundance and B) Hills N1 diversity in the St. John's Estuary.

There was a clear correlation between salinity and macrofauna characteristics at these eight stations. The macrofaunal community structures of each station were grouped more by salinity zone then whether the communities are inside or outside of grass beds (Figure 23). Crescent Lake, with lowest mean salinities (0.2 ppt), has a community structure significantly different than all other communities. The macrofaunal community structures of Rice Creek and Scratch Ankle are similar to each other but significantly different from all others. Rice Creek and Scratch Ankle have the second lowest mean salinities (0.5 to 0.6 ppt). The four stations with the highest salinities (2.4 to 4.2 ppt) have similar macrofaunal community structure except for the Doctors Lake macrofaunal community that is found outside the grass beds. There was a greater change in community structure with salinity when salinities were below 1 ppt then when salinities are between 2 and 4 ppt.

	Salinity Zone						
Indicators	0.3 ppt Crescent Lake	0.4 - 0.6 ppt Rice Creek/ Scratch Ankle	20 3.9 ppt All Other Stations				
Species		Rangia cuneata (B)	Marenzelleria viridis (P) Laeonereis culveri (P) Polymesoda caroliniana (B) Rangia cuneata (B)				
Higher Taxa	No bivalves		More polychaetes More bivalves				

Table 5. St. John's River oligohaline macrobenthic indicators (B = bivalves, P = polychaetes).



Figure 23. Non-metric Multi-dimensional Scaling plot of SAV stations labeled by station and substrate type; inside (In) and outside (Ou) grass beds. Stations listed in Figure 20. Similarity contours from cluster analysis are overlaid.

There was also a greater change in macrofaunal abundance and diversity at lower salinities then higher ones. Macrofaunal abundance and diversity increase sharply to peaks at 1.5 and 1.0 ppt respectively before slowly decreasing as salinity increases. The variability of macrofaunal abundance and macrofauna around the correlation curves created indicates that other variables in addition to salinity influence macrofaunal communities.

The investigation established that there is a clear change in macrofauna communities with salinity. However results are based on the limited salinity range within which the SAV stations are located. If additional macrofauna and salinity data from a wider salinity range further downstream of the existing SAV stations were incorporated with the existing data, then the results of the non-linear model that determined the optimum salinities would likely be very different.

Study 3: Tri-State Water Sharing Issues Within the Apalachicola-Chattahoochee-Flint Basin

The Apalachicola River is the largest river by volume in Florida, and the 4th largest river in the southeastern United States (FL DEP 2010). The river is formed by the confluence of the Chattahoochee and Flint Rivers at the Florida-Georgia Border (Figure 24). The Apalachicola River discharges into Apalachicola Bay, one of the most productive estuarine systems along the US gulf coast. The Apalachicola-Chattahoochee-Flint (ACF) basin is 51,800 km² in area and is shared by Georgia, Florida and Alabama (Lin 2010). Land use is primarily agriculture within the state of Georgia and silviculture within Florida.



Figure 24. Location of Apalachicola-Chattahoochee-Flint River system and drainage basin in Georgia, Alabama, and Florida, USA.

There has been a long history of negotiation and litigation between Florida, Georgia, Alabama, and the US Army Corps of Engineers over water sharing issues within the ACF basin (Table 6). As of July 2009, there were seven separate lawsuits (FL DEP 2010). At the center of the debate is the US Army Corps' operation of four major hydropower dams on the Chattahoochee River: Buford Dam (north of the city of Atlanta), Jim Woodruff Dam (at intersection of Chattahoochee and Flint Rivers), West Point Dam (south of the city of Atlanta), and George Dam (north of Woodruff Dam). Lake Sydney Lanier, on the Chattahoochee River near Atlanta, is the largest reservoir on the system, providing 65% of total conservation storage (Wilber 1992). The conflict stems from three main issues: 1) urban and agricultural development within Georgia that has increased upstream water withdrawals, 2) reduced instream flows to Florida affecting the productivity of fisheries in Apalachicola Bay, 3) Alabama's concerns about downstream and cumulative impacts of water resource actions (Lin 2010, NWFWMD 2010).

Table 6. Timeline of Apalachicola-Chattahoochee-Flint water use issues.

1946	Congress authorizes creation of 5 dams (4 on Chattahoochee River, 1 on
	Apalachicola River) in an effort to develop the nation's waterways
1956	Buford Dam completed, creates Lake Sydney Lanier
1957	Jim Woodruff Dam completed, creates Lake Seminole
1963	Walter F. George Dam completed, creates Lake Eufala;
	Andrews Dam (navigation only) completed, creates Lake Andrews
1975	West Point Dam completed, creates West Point Lake
1988	Apalachicola Bay declared a federal disaster area after reduced flow in the
	Apalachicola-Chattahoochee-Flint Rivers damages oyster harvest
1990	Florida and Alabama sue Georgia and US Army Corps of Engineers
1992	Governors of Georgia, Florida, and Alabama (and the US Army Corps of
	Engineers) sign a 5-year memorandum of agreement (MOA), puts the lawsuit
	on hold;
	States initiate Comprehensive Study of Resources in the ACF basin as directed by
	MOA
1997	Apalachicola-Chattahoochee-Flint interstate compact negotiated, approved by
	congress
1998	Deadline for compact: December 1998 \rightarrow extended 12 times
2003	Compact expired with no settlement. Lawsuit reactivated, focus on endangered
	species act
2007	Governors of Georgia, Florida and Alabama meet with President Bush regarding
	drought in southeastern US, develop exceptional drought operations (EDO)
	plan
2008	US Army Corps of Engineers begins implementing EDO plan
2010	Litigation continues; Georgia, Florida and Alabama file motions to make all
	settlement negotiations confidential

The water use needs of Georgia, Alabama and Florida have primarily been met by the construction of dams and reservoirs. Georgia dominates consumptive water use in the ACF basin with municipal and industrial uses exceed 1 million cubic meters per day (Carter et al. 2008). In addition, Georgia's agricultural uses of ground and surface waters that affect ACF flows can exceed 2.4 million cubic meters per day during dry summer growing seasons. Direct withdrawal of water from federal ACF reservoirs for agricultural water supply is not authorized. The city of Atlanta, Georgia obtains 99% of its water from surface water supplies, with 72% of the water derived from Lake Lanier and the Chattahoochee River. Alabama's water use needs from the ACF basin are considerably less than Georgia's. Alabama averages less than 190,000 cubic meters per day, primarily for municipal and industrial uses (Carter et al. 2008). Florida withdraws less than 38,000 cubic meters per day from the Apalachicola River. In 2006, the Northwest Florida Fisheries Management District reserved most of the flows from the

Apalachicola River and one of its tributaries, the Chipola River, for protection of fish and wildlife in Apalachicola Bay.

Although dam construction in the ACF basin was primarily for the purposes of power generation, navigation and flood control, the dams have also become important for recreation, water quality and protection of fish and wildlife resources (Carter et al. 2008). In 1998, a Draft Environmental Impact Statement (DEIS) was prepared focusing on the potential impacts of surface water allocation within the ACF basin in Alabama, Georgia and Florida, including Apalachicola Bay (DEIS 1998). In particular, the DEIS examined potential effects of water allocation on water, biological, socioeconomic, and cultural resources from 1995-2050. The DEIS provided a summary of potential impacts of low, moderate and high river flow scenarios on each major resource area (Table 7).

Freshwater inflows to Apalachicola Bay help support a productive coastal ecosystem including a variety of commercial and recreational fisheries. In particular, the oyster fishery in Apalachicola Bay provides over 90% of Florida's oyster harvest and 10% of the US oyster harvest (Andree 1983). Freshwater inflows are important to the maintenance of the oyster fishery. A 1992 study demonstrated that low river flows in the ACF system were positively correlated with oyster catch per unit effort (CPUE) 2 years later, indicating that increased bay salinities allow for increased predation on newly settled spat, thus reducing the population of marketable oysters 2 years later (Wilber 1992). From a socioeconomic perspective, more than 1,000 people are employed by the oyster industry in Florida's Franklin County (Carter et al. 2008).

Dams and reservoirs alter the natural flow regimes of rivers and streams, fragment the free-flowing riverine system, alter physical and chemical processes, and disrupt biological communities (Cowie 2002). Recent attention has been focused on the role of freshwater from the ACF basin in protecting and maintaining threatened and endangered species. Because of the habitat alteration caused by dams and reservoirs, 34 fish species and 16 freshwater mussel species are imperiled in Georgia (Cowie 2002). Particular attention has been focused on four species listed under the US Endangered Species Act: the Gulf sturgeon *Acipenser oxyrinchus desotoi*, fat threeridge mussel *Amblema neislerii*, Chipola slabshell mussel *Elliptio chipolaensis*, and purple bankclimber mussel *Elliptoideus sloatianus*. All four species are susceptible to changes in water flow, temperature and dissolved oxygen levels (Carter et al. 2008). In addition, the Gulf sturgeon is anadromous and needs to migrate upstream from the Gulf of Mexico to spawn. The three mussel species live in sand and gravel stream bottoms. Dams alter their free-flowing water habitats and restrict larval dispersal, resulting in small and isolated populations (Carter et al. 2008).

Table 7. Potential	impacts	of surface	water	allocation	on v	water,	biological,	socioecono	omic and
cultural resources	•								

Resource	Potential Impacts
Water	During moderate and high flow years, water quantities should be sufficient to meet demands. During low flow years, river flows would be substantially reduced and reservoir levels maintained near full pool. Navigation flows would be reduced slightly.
Biological	Impacts on terrestrial vegetation, floodplain freshwater wetlands would be minimal. Estuarine wetlands would experience greatest impacts in low flow scenarios. Wildlife expected to have only limited habitat loss for brief periods during low flow periods. Reduced inflow to Apalachicola Bay increases salinity, predatory oyster drill abundance, and oyster mortality.
Socioeconomic	Population growth is expected to follow water availability. Low and moderate flow periods will create less desirable conditions for recreational activities such as wade fishing, canoeing and rafting. Municipalities at the headwaters of the ACF basin will experience substantially greater adverse impacts than areas downstream. No water supply shortages for irrigation are predicted.
Cultural	Three potential effects of water level management may affect cultural and archaeological resources: erosion, deposition of sediment, and access. Of the three flow scenarios, the low flow scenario would produce downstream flows with lower velocity, resulting in fewer opportunities for erosion.

Drought conditions plagued much of the southeastern US beginning in 2006. By mid-2007, only one third of the system conservation storage was left in ACF reservoirs (Zeng et al. 2009). In response, the states of Georgia, Florida and Alabama, and the US Army Corps of Engineers developed an Exceptional Drought Operations plan (EDO). The EDO plan replaced all previous flow provisions with a minimum flow requirement of 134.5 cubic meters per second. All flows above this level could be stored in the ACF reservoirs. The basic idea was to maintain water in ACF reservoir storage to protect against further drought. Maintaining low flows was predicted to cause less harm to species during drought years than if the water was completely depleted.

The issue of shared water in the ACF basin is far from resolved, and the coming year will be filled with intense activity as litigation continues between Georgia, Florida and Alabama at the federal level. Emerging issues of concern will include future climate change and clean water act jurisdiction and the impacts of these issues on water sharing and biological resources.

Case Study 3: California, USA

California is the third largest state in the United States, covering approximately 414,000 km² in area. California is also the most populous in the United States, with an estimated 37 million people (US Census Bureau 2010). The state is highly variable in terms of climate and precipitation. Seasons in California are either wet (October to March) or dry (April to September). Two-thirds of the average annual precipitation falls in the northern third of the state, and the eastern side of California's mountain ranges create rain shadow deserts with extremely dry climates (Figure 25).



Figure 25. Rainfall gradient in California. Average annual precipitation (inches) from 1961-1990. Map from Oregon Climate Service: http://www.wrcc.dri.edu/pcpn/ca.gif

The regulation, protection and administration of water quality in California are carried out by the State Water Resources Control Board and nine Regional Water Quality Control Boards (Figure 26). These groups are tasked with protecting and enhancing the quality of California's waters for present and future generations (State Water Resources Control Board 2010). The State Water Resources Control Board is one of five departments within the California Environmental Protection Agency and is responsible for allocating surface water rights in California, as well as setting statewide policy and regulations for water quality control (State Water Resources Control Board 2010). The State Board comprises five full-time members appointed by the governor for 4-year terms.

Because of regional differences in water quality and quantity, the state is divided into nine regions for administration of California's water quality control program: 1) North Coast, 2) San Francisco Bay, 3) Central Coast, 4) Los Angeles, 5) Central Valley, 6) Lahontan, 7) Colorado River Basin, 8) Santa Ana, and 9) San Diego (Figure 24) (State Water Resources Control Board 2010). Each Regional Board is responsible for local implementation of policy and regulations, and development of water quality decisions for its region. The Regional Boards each consist of nine part-time members appointed by the governor for 4-year terms. Each Regional Board has developed a Basin Plan (or Water Quality Control Plan) for their region. The Basin Plans are developed to preserve and enhance water quality and protect the beneficial uses of regional waters (State Water Resources Control Board 2010). These plans are designed to be more than an abstract set of goals and policies; rather they set forth water quality standards for surface and ground waters, identify water quality problems and recommend control measures (Lahontan Basin Plan 1995). The Basin Plans also serve to implement the US Clean Water Act in California.

Regional Water Quality Control Board Profiles

The North Coast Region (1) is approximately $50,220 \text{ km}^2$ in area in northwestern California. The region is divided into two drainage basins, the Klamath River and North Coastal Basins. Precipitation in the North Coast Region (64.1 km³) is greater than any other region in California. The population and economy of the basin have developed much more slowly than other areas in the state (North Coast Basin Plan 2007).

The San Francisco Bay Region (2) is approximately 11,922 km² in area and is located directly south of the North Coast Region along the Pacific Coast. The dominant feature is the San Francisco Bay Estuary, the largest estuary on the west coast of the United States. The region also forms the backdrop of the fourth largest metropolitan region in the United States (San Francisco Basin Plan 2007).

The Central Coast Region (3) is approximately 29,200 km² in area and includes all or portions of 9 counties. The region is located directly south of the San Francisco Bay Region along the Pacific Coast and is considered arid in terms of precipitation. The total population of

the region is estimated to be 1.22 million people. Agriculture and food production activities are the major regional industries (Central Coast Region Basin Plan 2006).

The Los Angeles Region (4) is approximately 10,130 km² in area and includes 5 major watersheds. The region is located along the Pacific Coast, directly south of the Central Coast Region. The main precipitation occurs during a few major storms each year. The total population of the region is estimated to be around 12 million people (Los Angeles Region Basin Plan 1995).



Figure 26. Location of the 9 California Regional Water Quality Control Boards. Map from California State Water Resources Control Board: http://www.waterboards.ca.gov/publications_ forms/publications/factsheets/docs/boardoverview.pdf

The Central Valley Region (5) covers the entire area included in the Sacramento River Basin (70,474 km²) and the San Joaquin River Basin (41,129 km²), about ¹/₄ of California's total area. These rivers provide more than half of the state's water supply. Surface waters from these two drainage basins converge to form the Sacramento-San Joaquin River Delta, which drains into San Francisco Bay. The Central Valley Region forms the eastern (inland) boundary of the North Coast, San Francisco Bay, and Central Coast Regions (Central Valley Region Basin Plan 1998).

The Lahontan Region (6) is approximately $85,809 \text{ km}^2$ in area and includes both the highest (Mount Whitney) and lowest (Death Valley) points in the continental United States. The region is located in a rain shadow, but precipitation in the higher elevations can be high (up to 178 cm per year). The Lahontan Region covers almost the entire eastern edge of the state and is divided into 12 major watersheds. The region has relatively low resident population and an agricultural emphasis on livestock grazing rather than crops (Lahontan Basin Plan 1995).

The Colorado River Region (7) covers approximately 51,800 km² in southeastern California and includes all or part of 4 counties. The Colorado River is the most important waterway in the region, which contains a small portion of the total Colorado River drainage area. The most significant feature in the region is the Salton Trough, an extension of the Gulf of California that contains the Salton Sea. The region has relatively low population density (Colorado River Basin Plan 2006).

The Santa Ana Region (8) is the smallest of the nine water quality control board regions in the state (approximately 7250 km²). The region located along the Pacific coast directly south of the Los Angeles Region, with its eastern boundary formed by the Colorado River Region. The most significant hydrologic feature in the region is the Santa Ana River. Although geographically small, the estimated population of the region is approximately 4 million people (Santa Ana Basin Plan 2008).

The San Diego Region (9) covers approximately 10,100 km² in the southwest corner of California. The region is located directly south of the Santa Ana Region along the Pacific Coast and is bounded on the eastern edge by the Colorado River Region. The region is divided into 11 major watersheds. The estimated population of the San Diego Region is approximately 9 million people (San Diego Basin Plan 1994).

Study 1: San Francisco Bay

The San Francisco Bay Estuary on the central coast of California is the largest estuary on the west coast of the United States, roughly 2,575 km² in area (Figure 27). Almost all the freshwater to the estuary is supplied by the Sacramento and San Joaquin Rivers, which drain via the Sacramento-San Joaquin Delta into the estuary and mix with the saline waters of the Pacific Ocean. The estuary is commonly divided into four subregions: Suisun Bay, North Bay/San Pablo Bay, Central Bay, and South Bay. Freshwater inflow to the San Francisco Bay Estuary supports a wide range of habitats including deepwater channels, tidelands, marshlands, and
freshwater streams (San Francisco Bay Basin Plan 2007). These habitats in turn support diverse communities of crabs, clams, fish, and birds, as well as migrating waterfowl. However, much of the freshwater is captured upstream by the dams, canals and reservoirs that are part of California's water diversion projects.

California has been dealing with water allocation issues for decades. The Sacramento-San Joaquin River Delta is the largest source of California's freshwater, providing water to approximately 23 million people, or two-thirds of all Californians (CALFED 2010). The Delta drains approximately 100,000 km², or 40% of the area of California (Kimmerer 2002). Precipitation falls mainly to the north of the Sacramento-San Joaquin Delta in winter and spring, but the greatest demand for freshwater is in the south during the summer. During the dry summer and fall months, the Sacramento-San Joaquin River Delta system is used to move water from water-rich reservoirs in the north to water-poor farms and cities to the south (Kimmerer 2002). Water is exported for irrigation and municipal use by large diversion pumps operated by the Central Valley Project and the California State Water Project. These facilities represent one of the world's largest man-made water systems (Nichols et al. 1986). Since 1977, about half of the total possible flow into the estuary has been diverted, with about 80% used for agricultural irrigation, and the rest used for municipal and industrial purposes (Pierson et al. 2002).



Figure 27. San Francisco Bay Estuary, California, showing the different subregions (Suisun Bay, North Bay / San Pablo Bay, Central Bay, and South Bay).

The Sacramento-San Joaquin Rivers and Delta are located east of the estuary. Map from NOAA San Francisco Bay Watershed Database and Mapping Project, available online at: http://mapping2.orr.noaa.gov/portal/sanfranciscobay/sfb_html/sfbenv.html

Freshwater diversion has altered the hydrograph more through a change in seasonal patterns than in total flow (Kimmerer 2002). Water is released from reservoirs to prevent salt water intrusion into the Sacramento-San Joaquin Delta from the Pacific Ocean. Both the

physical process of diverting water and the resulting alterations in flow have affected biological communities of the San Francisco Bay Estuary (Nichols et al. 1986). In particular, concerns have increased over the effects of altered freshwater flows on several fish species, including the delta smelt (*Hypomesus transpacificus*), threadfin shad (*Dorosoma petenense*), longfin smelt (*Spirinchus thaleichthys*) and striped bass (*Morone saxatilis*) (Feyrer et al. 2007, Sommer et al. 2007). The delta smelt is endemic to the low-salinity habitats of the northeastern San Francisco Bay Estuary. Although the species was harvested commercially in the 19th and 20th century, recent declines have resulted in its listing as threatened under the US Endangered Species Act (Bennett 2005). Another ecological consequence of altered freshwater flows is the absence of a summer phytoplankton bloom in northern San Francisco Bay during extremely low inflows, which can suppress the pelagic food web, fish recruitment, and fisheries yields (Nichols et al. 1986).

The State Water Resources Control Board adopted an inflow standard based on a desired salinity gradient within the bay. Freshwater inflow to the San Francisco Bay Estuary is regulated in part using the variable X2. This variable represents the longitudinal position in the estuary of the 2 psu salinity isohaline, measured 1 m from the bottom of the water column, averaged over a suitable time period (> 1 day) (Pierson et al. 2002). This low salinity zone moves primarily in response to flow (i.e., located further upstream during low flow periods). The direction and magnitude of fluctuations is measured as the distance in kilometers from the Golden Gate Bridge. When freshwater inflows to the estuary are high, X2 is located seaward in direction (the value of X2 is lower). The standard was chosen as an indicator of estuarine habitat and significant relationships have been found between X2 and phytoplankton, shrimp, mysids, and desirable fish larvae (Kimmerer 2002). This approach is important because the rule can be linked to many ecosystem components and all trophic levels within the estuary. California's regulations allow the salinity guideline to change seasonally.

Dissatisfaction with water resource management led to the establishment in 1995 of the CALFED Bay/Delta Science Program. The mission of the joint federal-state program is to provide the best possible scientific information for water and environmental decision-making in the San Francisco Bay / Sacramento-San Joaquin River Delta system (CALFED 2010). In November 2009, the California legislature passed a comprehensive water package to fund water issues including Delta sustainability, conservation and watershed protection, and water recycling and conservation programs. California Senate Bill SBx7 1 created a new Delta Stewardship Council, an independent state agency tasked with developing a Delta Plan by January 2012 with goals of providing a more reliable water supply and protecting and restoring the Delta ecosystem. Under the same bill, the CALFED Bay/Delta Science Program became the Delta Science Program, reporting to the new Council (Delta Council 2010).

As part of the recent legislation, SBx7 1 Division 35: the Sacramento-San Joaquin Delta Reform Act of 2009, the California State Water Resources Control Board was tasked with developing new flow criteria for the Delta ecosystem necessary to protect public trust resources.

The flow criteria must include the volume, quantity and timing of water necessary for the Delta ecosystem under different conditions. The compliance deadline for development of these criteria is nine months. In addition, the California Department of Fish and Game, with assistance from the US Fish and Wildlife Service and the National Marine Fisheries Service, was tasked with developing and recommending flow criteria and quantifiable biological objectives for species of concern that are dependent on the Delta. The deadline for these recommendations is 12 months. Thus, flow criteria and recommendations should be forthcoming sometime in mid-late 2010. The comprehensive management plan for the delta (Delta Plan) will be completed by January 1, 2012.

Study 2: Colorado River

The Colorado River forms the southeastern border of California (Figure 28). The watershed includes parts of seven US states and two Mexican states. The Colorado River is the primary water source for the southwestern United States, irrigating more than 14,900 km² of farmland and supplying water to nearly 30 million people (Luecke et al. 1999). The natural course of the river flows from the headwaters in northwestern Colorado south and west through Utah, Nevada, Arizona, California, and into Mexico en route to the Gulf of California. River flows are highly variable, ranging from maximum annual natural flows of 30.5 km³ per year to minimum annual natural flows of 6.2 km³ per year (California's Colorado River Water Use Plan 2000).

The Colorado River has a long history of water rights negotiations. The statutes, court decisions and decrees, contracts, interstate compacts, and administrative rules and regulations that define the apportionments of the Colorado River to US states and Mexico are collectively known as "the Law of the River" (Varady et al. 2001). Most significantly, the Colorado River Compact of 1922 was negotiated among seven states in the basin of the Colorado River to govern water allocation. For the purposes of the compact, the river basin is divided into an Upper Basin that includes Colorado, Utah, Wyoming, New Mexico and Arizona, and a Lower Basin that includes California, Arizona and Nevada. The compact apportioned 9.25 km³ of water per year to each basin (Colorado River Compact 1922). In 1944, the United States also signed a treaty agreeing to deliver an annual quantity of 1.85 km³ of water per year to Mexico (Treaty Series 994 1944). The amount of water allocated was based on the expectation that the river's average flow was 20.2 km³ of water per year. Unfortunately, the long-term average river flow is considerably less (approximately 18.0 km³ per year). As a result the sum of the compact apportionments and the Mexican treaty exceed the flow of the Colorado River in most years (Woodhouse et al. 2005) (Table 8). Because of intense human alteration in the form of dams and reservoirs, little or no water has reached the Gulf of California since 1960.

Historically, the Colorado River Delta comprised a vast area (7,810 km²) of riparian, freshwater, brackish and tidal wetlands, and served as an important migratory bird stopover and nursery for endemic marine species (Luecke et al. 1999). However, decades of dam and irrigation projects have diverted water from the Colorado River for human use, resulting in a

much reduced delta of small wetlands and brackish mudflats. In the 1930s, following the construction of the Hoover Dam, virtually no freshwater reached the delta for 6 years as the Lake Mead reservoir filled behind the dam. From 1963-1981 a similar situation occurred following the construction of the Glen Canyon Dam and subsequent filling of the Lake Powell reservoir. Floodwaters are released only when the US Bureau of Reclamation predicts flows that exceed the system's capacity for use and storage (Luecke et al. 1999). During the 20th century, river flows to the Colorado River Delta have been reduced nearly 75% (Fradkin 1981, Glenn et al. 1999).



Figure 28. Colorado River, stretching through the southwestern US states of Colorado, Utah, Arizona, Nevada and California en route to Mexico and the Gulf of California (from Varady et al. 2001).

Reductions of Colorado River flow to the delta have resulted in significant physical and biological impacts. On the physical side, reduced delivery of silt and nutrients as well as higher salinity and pollutant concentrations have been recorded (Luecke et al. 1999). In addition, the delta has taken on the highly unusual condition where erosion (versus accretion) is the dominant

physical process (Thompson 1968). Biologically, organism abundance is linked to Colorado River flow. The relative abundance of postlarval blue shrimp *Litopenaeus stylirostris*, which constitute the dominant shrimp fishery, was significantly higher when flow from the Colorado River reached the upper Gulf of California (Aragón-Noriega & Calderón-Aguilera 2000). Declines in the catch of *Totoaba macdonaldi*, a once abundant and now endangered fish species in the Gulf of California coincide with increased fishing effort and decreases in Colorado River flow (Cisneros-Mata et al. 1995). During historical periods of natural river flow, conservative estimates of bivalve mollusc abundances are around $6x \ 10^9$ (density ~ 50 m⁻²), whereas the present abundance is ~94% lower, or around 3 m⁻² (Kowalewski et al. 2000). The delta is home to the largest populations of two endangered species, the desert pupfish (*Cyprinodon macularius*) and the Yuma clapper rail (*Rallus longirostris yumanensis*) (Glenn et al. 1996). Future reductions in flow could have strong impacts on these and other vulnerable species.

Allocation	Km ³ Per Year			
Upper Basin				
Colorado	4.8 km ³			
Utah	2.1 km ³			
Wyoming	1.2 km ³			
New Mexico	1.0 km^3			
Lower Basin				
Arizona	3.5 km ³			
California	5.4 km ³			
Nevada	0.4 km ³			
Additional Allocations				
Mexico	1.9 km ³			
Total Allocations	20.3 km^3			
Long-term average flow	18.0 km ³			

Table 8. Colorado River Allocations

The flashiness of flow conditions can also influence species persistence. In particular, non-native species may not be behaviorally adapted to avoid displacement by sudden floods (Poff et al. 1997). The historical cycles of scour and fill in the Colorado River prevented encroachment of vegetation onto the river banks, whereas recent flow reduction has results in increased wetland and woody vegetation, dominated by the exotic salt cedar tree (*Tamarix* sp.) (Stevens et al. 1995).

At the heart of present-day Colorado River water allocation issues is the fact that "the Law of the River" is based on 19th century needs, which differ fundamentally from current issues. However, negotiating water reallocations can be tricky because the gains of one user must be balanced by the losses of another. One option that has been proposed as an environmentally sustainable mechanism for reallocations is to develop a water market system, which would offer several advantages such as: 1) allowing water to be traded on a voluntary basis, 2) allowing water to be directed to the most productive activities thus optimizing the economic returns, and 3) allowing for the creation of water taxes or other allocation systems to ensure that environmental and consumptive uses can be balanced (Varady et al. 2001). However, water markets that need to cross state boundaries involve additional layers of complexity. The water market option will require more research and time to determine if they offer a reasonable solution to interstate and international water allocation issues.

In the past 30 years, partial restoration of river flow has been accomplished by floodwater, periodic reservoir releases, agricultural return flows and municipal wastewater (Kowalewski et al. 2000). Agricultural drain water has created Cienega de Santa Clara, a 42 km² Typha domengensis marsh containing the largest remaining population of the endangered Yuma clapper rail and many other species of migratory and resident waterfowl (Glenn et al. 2001). Attempts at river restoration have also been conducted via planned flood events. In 1996, a 7day test flood (approximately 35% of the pre-dam average spring flood) was released from Glen Canyon Dam. As a result, over 53% of the beaches increased in size and just 10% decreased in size (Poff et al. 1997). The flood was of not large enough to significantly reduce populations of nonnative fishes, but studies indicate that similar managed floods can impair nonnative predator and competitor populations, enhancing the survival of native fishes (Valdez et al. 2001). Totoaba macdonaldi has experienced an increase in annual survival of in recent years, suggesting recovery of the stock (Cisneros-Mata et al. 1995). It is important to consider the timing and magnitude of planned floods in reference to organism life cycles, biological interactions and habitat preferences. Although the test flood successfully restored sandbars and was timed to prevent impacts to species of concern, an estimated 10.7% of the habitat of the endangered Kanab ambersnail (Oxvloma havdeni kanabensis) and 7.7% of the snail population were lost to the flood (Stevens et al. 2001).

Involvement of and cooperation between many stakeholder groups is important to the success of any restoration program. Scientists studying Colorado River water issues have interacted with numerous stakeholder groups including federal and state agencies, energy production organizations, Native American tribes, and recreational and environmental organizations (Poff et al. 2003). Identification of consistent funding streams is also important. For example, ecosystem research and monitoring can be supported via allocation of a small portion of the revenues generated at federal hydroelectric power dams. Hydropower revenues from the main dams of the Colorado River Storage Project (Richter et al. 2003) support both the Grand Canyon Monitoring and Research Center and the monitoring element of the Recovery Implementation Program for Endangered Fish Species in the Upper Colorado River Basin (Poff

et al. 2003). Monies from environmental damage taxes and mitigation banking may also be developed to secure water or money for the purpose of ecosystem restoration (Pitt et al. 2000).

Large ecosystem restoration is a relatively new practice where science plays an integrative rather than analytic role and the process is generally political (Luecke 2000). In addition, restoration may not equate to reestablishment of the pre-development ecosystem, but rather the restoration of ecosystem function (Pitt 2001). Thus, establishing metrics of restoration success can be a complex process. For the future, some combination of applied science and management, better-adapted institutions, bi-national interests, and community-level participation are needed to help resolve the problems facing the Colorado River watershed and Delta (Varady et al. 2001). The United States and Mexico appear to be committed to protect the Colorado River and Delta ecosystems. In 2000, the countries signed an agreement known as Minute 306, which amends the 1944 treaty between the US and Mexico. The agreement seeks to develop joint studies that include possible approaches to ensure use of water for ecological purposes... with a focus on defining habitat needs of fish and marine and wildlife species of concern to each country (IBWC 2000). Because the agreement does not specifically dedicate instream flows to the Colorado River Delta, the future of the delta ecosystem remains in jeopardy. Time will tell whether recent increases in public interest related to Colorado River water issues can drive existing water problems toward resolution in a timely fashion.

Case Study 4: Australia

Australia is the driest inhabited continent in the world. Rainfall is also highly variable over space (Figure 29) and time in Australia causing Australia to impound twelve times as much water as would be impounded in similar latitudes in the United States of America (Cappo et al. 1998a). Throughout Australia, especially in northern regions, a large amount of the rainfall is lost through evapotranspiration (Figure 30). The average population density of the approximately 22 million people in Australia is less than 3 people per square kilometer. However, 90 percent of people live within 100 km of the coast (World Resources Institute: http://earthtrends.wri.org). Most people are concentrated on the east and south-east coasts (including Adelaide, Melbourne, Sydney, Brisbane, Townsville), and the south-west coast (including Perth).



Figure 29. Mean annual rainfall in Australia from 1961 – 1990 (from Australian Bureau of Meteorology).

Estuaries vary greatly on the Australian coast based on precipitation, evaporation and tide regimes (Table 9, Figure 31). Irregular flood and fire regimes that occur in the catchments of temperate parts of Australia (mid-southern latitudes) strongly influence the water quantity and quality of the estuaries into which the catchments drain (Roy et al. 2001). Many Australian estuaries are susceptible to full or partial closure of their mouths because of the small and variable amount of precipitation in their catchments. Additional removal of freshwater for human uses can potentially harm the estuary by:

- altering the estuarine salinity gradient;
- altering estuarine flushing and water quality; and,
- obstructing diadromous fish and crustacean migration (Peirson 2002).



Figure 30. Mean evapotranspiration in Australia from 1961 – 1990 (from Australian Bureau of Meteorology).

 Table 9. Regional classification of Australian Estuaries (from Tomczak 2000).

Location	Tidal Range	Rainfall		Estuary Type		Example	
		Summer	Winter	Summer	Winter		
East and Southeast	moderate	moderate		slightly stratified if the catchment is large enough to suppress rainfall variability; intermittent otherwise		Hawkesbury River, Derwent River (slightly stratified), Port Hacking(intermittent)	
South	small	nearly none; large evaporatio n	moderate; large evaporatio n	inverse		Spencer Gulf, St. Vincent Gulf	
Southwest	small	a little	moderate	slightly stratified	highly stratified	Swan River	
North and Northeast	large to extreme	very large	nearly none; large evaporatio n	slightly stratified	salt plug	South Alligator River, Wenlock River	



Figure 31. The geographical distribution of estuaries based on climate and tidal range (from Digby et al. 1999).

Many fish and crustacean species have been impacted by changes in freshwater inflows. Economically important fishes that have been affected by salinity changes include Barramundi (*Lates calcarifer*), king threadfin (*Polydactylus macrochir*) jungle perch (*Kuhlia rupestris*), Australian bass (*Macquaria novemaculeata*), estuary perch (*Macquaria colonorum*), Black Bream (*Acanthopagrus butcheri*) and blue catfish (*Arius graeffei*; Table 10; Cappo et al. 1998b, Robins et al. 2006, Milton et al. 2008, Halliday et al. 2008,). Grazing in wetlands, sometimes in addition to barriers to fish movement, have caused water retention and increased evaporation in some Australian Estuaries, especially in Northern Territory (Cappo et al. 1998b). The resulting increase in salinity has caused a reduction in fish diversity, freshwater wetlands, and nutrient cycling.

Species	Drainage	Adult Habitat	Juvenile Habitat	Spawning site	Spawning salinity
Barramundi		tidal, nontidal, longshore to headlands	uppper tidal limits and non-tidal freshwater	sheltered estuary mouths, sandbars <2 m deep	17-31
Jungle Perch	NE Coast	headwaters	headwaters	lower estuaries	brackish
Australian Bass	SE Coast, NE coast (lower)	tidal to non- tidal	brackish water	lower estuaries	12-15 (larvae best at >= 20)
Estuary Perch	SE Coast, Murray Darling Indian Ocean	tidal, non- tidal	brackish water	estuary mouths	
Blue Catfish (<i>Arius</i> graeffii)	Timor Sea, Gulf, NE Coast, SE Coast (upper)	tidal, non- tidal	tidal, non- tidal		

Table 10. Economically important fishes of coastal freshwaters threatened by changes in drainage to estuaries (From Cappo et al. 1998b after Merrick and Schmidt 1984).

The location and qualities of the salinity gradient in an estuary or coastal zone is a key factor in determining spawning success, migration and recruitment of estuarine species (Cappo et al 1998b). Other ecological effects of reducing freshwater inflows have been researched in many estuaries among the different regions of Australia.

Freshwater inflow is the most important factor in determining the position of the saltwedge in some microtidal estuaries such as the Swan River Estuary (Western Australia; Kurup et al. 1998). Changing land use in the Swan River catchment has historically increased inflows resulting in a decrease in salinity and increase in chlorophyte biomass (Chan et al. 2002). An increase in nutrient concentrations also as a result of land-use changes has allowed an increase in biomass of the four main phytoplankton species found there. However seasonal changes in salinity as a result of altered seasonal inflows have a small impact on estuarine and freshwater phytoplankton dynamics (Chan et al. 2002).

In nine permanently open eastern Australian estuaries, dusky flathead (*Platycephalus fuscus*), luderick (*Girella tricuspidata*), sand whiting (*Sillago ciliata*) and sea mullet (*Mugil cephalus*) all increased in abundance with an increase in freshwater inflow and decreased in abundance during drought periods (Gillson et al. 2009). Seasonal flows were the most important factors in determining variability in abundance for the four species. The minima and maxima of flows were more important than other aspects of inflow.

Increases and decreases of cohorts of the apex fish Mulloway (*Argyrosomus japonicas*) have been related to increases and decreases in freshwater inflows in the Murray River estuary suggesting that the estuary may provide important refuge for juveniles (Ferguson et al. 2008).

In the Upper Derwent Estuary, a drowned river estuary in Tasmania, hydroelectric storage operations have decreased the frequency of flows needed to flush the already severely degraded estuary (Davies and Katish 1994). This lack of flushing has allowed low dissolved oxygen concentrations and high hydrogen sulfide concentrations in bottom waters to persist for seven months of the year.

Different methods have been used in an attempt to determine environmental flow requirements in different regions of Australia (Pierson et al. 2002). Pierson et al. (2002) provided a multidisciplinary adaptive management methodology for assessing freshwater inflows using a checklist of major ecological processes that needs to be developed for each estuary of coastal zone. This methodology relies on the availability of high quality ecological and water quality data, which is often lacking in estuarine systems. In tropical (northern) Australia, fisheries production has been promoted as a suitable indicator of freshwater inflows such as in the Fitzroy River Estuary in northern Queensland (Robins et al. 2005). Fisheries can be a useful indicator both because they are generally considered by the public to be important and there is generally available historical data in the form of commercial catch. The spatial and temporal extent of biological and water quality data in Victoria forced preliminary analyses of inflow requirements for the state to be based on physical measures such as GIS-based hydrological modeling and an EFLOWS hydrology modeling (Hardie et al. 2006, Barton et al. 2008). Other potential and already implemented environmental flow requirements are summarized by Gippel (2002) and Gippel et al. (2009).

The Murray- Darling River Basin

The Murray Darling Basin (MDB) comprises the catchments of Australia's three largest rivers, the Darling River (2740 km), the Murray River (2530 km; also known as River Murray and Mighty Murray) the Murrumbidgee River (1,690 km) and their tributaries (Australian

Bureau of Statistics 2008; Figure 32). The MDB covers over one million square kilometers (14 percent of Australia's area) across five Australian states and territories (Australian Capital Territory, New South Wales, Queensland, South Australia and Victoria). Precipitation in the MDB is spatially variable, with 1400 mm y⁻¹ falling in the highlands and only 300 mm y⁻¹ falling in the northwest (MDMC 1987). The high spatial variability in rainfall combined with high temporal variability that occurs, results in considerable seasonal and year-to-year variability in river flows within the MDB (Hatton MacDonald and Young 2002). Of the over half a million gigaliters of rainfall per year that falls in the MDB, 94 percent evaporates, 2 percent evaporates and 4 percent ultimately becomes runoff. As a consequence of the high evaporation rate, 80 percent of land in the MDB is classified as having an arid or semi-arid climate (GWP 2003). Two-thirds of the land in the MDB is used for growing crops or is in pasture, which provides 39 percent of the Australian national income that is derived from agricultural production. Two percent of the agricultural land in the MDB (1.65 million hectares) is irrigated. This equates to 65 percent of all irrigated land in Australia occurring in the MDB. The 2 million people that inhabit the MDB use 52 percent of Australia's total water consumption through agricultural, industrial and household uses.



Figure 32. Murray Darling Basin showing states and major rivers. ACT = Australian Capital Territory.

The high demand of water has consequently degraded many parts of the of the MDB including widespread soil erosion, river siltation, accelerated recharge of groundwater aquifers and subsequent discharge of saline groundwaters to rivers, increased dry-land salinity, loss of flora and fauna habitat (GWP 2003) and changes to the geomorphology of the Murray River mouth (Harvey 1988, 1996; Walker and Jessup 1992; Bourman and Barnett 1995). Increases in salinities in the Murray River Estuary have been linked not only to changes in physio-chemical processes, but to changes in the abundance and distribution of flora and fauna. Changes in estuarine salinity have been linked to changes in abundance and distribution of fish species such as hardyhead species (*Craterocephalus* sp; Wedderburn et al. 2007) and mulloway (*Argyrosomus japonicas*; Ferguson et al. 2008), abundance, distribution and diversity of plant and macroinvertebrate communities (Geddes 2004), and the spatiotemporal variation in community structure of wetland birds (Paton et al. 2009). The degradation has resulted in ecological, cultural, social and economic consequences. Management problems in the MDB as identified by GWP (2003) include:

- increasing competition for scarce water resources
- resistance to further land clearing controls by State Governments
- increasing conflict over who should pay for remediation of degraded common resources
- how to best mobilize and target the use of available resources for on-ground action and
- how to address poorly specified institutional arrangements for common property resource management.

History of Murray-Darling Management

After a series of droughts and increasing conflict between those concerned with using the river for navigation and those concerned with using the river for irrigation, the River Murray Waters Agreement (RMWA) was signed in 1915 by the governments of Australia, New South Wales, Victoria and South Australia. As a result of this agreement, several lakes, dams, weirs and locks were created along the Murray River over the next 70 years by the River Murray Commission (RMC). The increasing awareness of riverine water quality (in addition to water quantity), and the connectivity between practices in the watershed and water quality on the Murray River from the 1960s to the 1980s brought about new water management responsibilities for the RMC (Table 11). Despite amendments to the RMWA up until the early 1980s, it became increasingly obvious that the RMWA was insufficient in managing both the dwindling resources and growing environmental degradation in the Murray River. The insufficiency of the RMWA to address especially water quality especially problems with salinity, can be partially attributed to its' 'limited geographical and functional scope' (Crabb 1988). To mitigate this problem, the Murray-Darling Basin Ministerial Council (MDBMC) was formed in 1985.

The MDBMC was comprised of parliamentary ministers (federal level), who were primarily responsible for the land, water and environmental sectors of Australia, and the three relevant state governments (New South Wales, Victoria and South Australia). In 1987, the MDBMC-directed Murray-Darling Basin Environmental Resources Study was completed (MDBMC 1987). This study was completed to determine the extent of degradation within the MDB. Also in 1987, the Murray-Darling Basin Agreement (MDBA) was signed, however this was replaced by an entirely new agreement of the same name in 1992. The purpose of the MDBA was to 'promote and co-ordinate effective planning and management for the equitable efficient and sustainable use of the water, land and other environmental resources of the Murray-Darling Basin'. The new MDBA was given full legal status by the Murray-Darling Basin Act in 1993. The MDBA now involved three separate institutions to manage the MDB:

-the Murray-Darling Basin Ministerial Council (to provide policy and direction)

-the Murray-Darling Basin Committee (to manage tributaries within the catchment and advise the ministerial council) and

- the Community Advisory Group (to advise the MDBMC from a community viewpoint).

The state of Queensland and the Australian Capital Territory were included in the Agreement in 1996 and 1998 respectively.

Table 11. Evolution of the River Murray Waters Agreement, 1914 to 1981 (from Clarke 1982 and www2.mdbc.gov.au/about/history_mdbc.html).

Matters beyond the powers of the Commission in 1914

Problems arising on tributary rivers. Problems caused by adjacent land-use. Problems of flood mitigation and protection. Problems of erosion and catchment protection. Problems of water quality and pollution from agricultural and other sources. Problems of influent and effluent waters. The needs of flora and fauna. Possible recreational, urban or industrial use. The environment or aesthetic consequences of particular proposals.

Matters permitted by previous amendments and informal practice before 1976

Limited powers of catchment protection. Power to initiate future proposals. Provision of certain dilution flows to maintain water quality. Lock maintenance work, improving navigability. Provision of recreational facilities. Expenditure on salinity investigations. Expenditure on re-designed works to protect fish life. Construction and operation of storages on tributaries.

Principal innovations in agreement reached in October, 1981

Power to consider any or all relevant water management objectives including water quality, in the investigation, planning and operation of works.

Power to monitor water quality.

Power to co-ordinate studies concerning water quality in the River Murray.

Power to recommend water quality standards for adoption by the states.

Power to make recommendations to any government agency or tribunal on any matter which may affect the quantity or quality of River Murray waters.

Power to make representations to any government agency concerning any proposal which may significantly affect the flow, use, control or quality of River Murray waters.

Power to recommend future changes to the Agreement.

New water accounting provisions.

Restructuring of the management system occurred in 2008 with the implementation of the Murray-Darling Basin Authority, which assumed the responsibilities of the former Murray-Darling Basin Committee. The Murray-Darling Basin Authority is directed by decisions made by the new six member Authority, Ministerial Council and the Basin Officials Committee. The key functions of the Murray Darling Basin Authority as stated by the Authority include:

- Preparing the Basin Plan for adoption by the Minister for Climate Change, Energy Efficiency and Water, including setting sustainable limits on water that can be taken from surface and groundwater systems across the Basin
- Advising the minister on the accreditation of state water resource plans
- Developing a water rights information service which facilitates water trading across the Murray–Darling Basin
- Measuring and monitoring water resources in the Basin
- Gathering information and undertaking research
- Engaging the community in the management of the Basin's resources. (MDBA 2010)

Currently the Murray Darling Basin Authority is developing a Basin Plan as required by the (Australian) Water Act 2007. Functions of the plan include determining and enforcing sustainable limits of surface- and ground waters, set basin-wide water quality (including salinity) objectives, set requirements to be met by state resource plans, develop efficient water trading schemes, and improve water security for all uses of basin water resources (MBDA 2010). These functions will be met after considerable scientific and socioeconomic evaluations of environmental, economical and cultural (see Morgan et al. 2004, Venn and Quiggin 2007, Steenstra 2009) basin resources.

There has been continuing speculation about the effects of the plan, particularly on agricultural practices of irrigating farmers, water allocation buy-back programs, and effects of climate change on water availability. The quantity of water that is necessary to provide satisfactory environmental conditions in the MDB is widely known to be over allocated (DEWHA 2010). This over-allocation has prompted several willing-seller water rights buy-back schemes such as the Murray Darling Basin Authority's 'Living Murray Water Purchase Project' and the Australian Government's 'Small Block Irrigators Exit Grant Package'. The Living Murray Water Purchase Project's aim to recover 500 gigaliters of water permits from large scale water users (mostly irrigating farmers) involved the reviews of seller-proposed water permit prices over a 2-month period in mid 2009. The Small Block Irrigators Exit Grant Package is a plan primarily for smaller irrigating farmers (< 40 ha) that was available from late 2008 until mid 2009. For further information on water buy-back in the MDB see Qureshi et al. 2010, Crase (2009) and the Department of the Environment, Water, Heritage and the Arts website (www.environment.gov.au/water).

The negative effects of the forecasted drier conditions as a result of global climate change have also been discussed (CSIRO 2008, Quiggin 2008, Adamson et al. 2009, Connor et al. 2009). Scenarios of surface water reductions in MDB include a median decline of 11 percent throughout the basin, which would result in a 24 percent reduction in current outflow from the Murray river mouth, and 30 percent of the flow that would occur without development within the MDB (CSIRO 2008). The combined effects of climate change, surface water reductions, and over-allocation provides a further stress on the already stressed MDB environment.

Case Study 5:South Africa

South African law recognizes basic human water requirements as well as the need to sustain healthy freshwater and estuarine ecosystems. The Water Law Principles of 1996 set the direction of the future of water resources management (Department of Water Affairs and Forestry 2004). The twin principles of sustainability and equity run through the South Africa National Water Policy of 1997 and the National Water Act (Act 36 of 1998). The key to balancing sustainability and equity lies in the provisions for the Reserve, the ability to quantify a Reserve, and the ability to manage water uses to meet the Reserve.

The concept of the Ecological Reserve is central to identifying environmental flow requirements. The policy identifies Four different levels of assessment of the Ecological Reserve have been identified (Table 12):

- Desktop estimate (to obtain a low confidence value for the reserve of a water resource for use in the Water situation assessment model) (Not applicable to estuaries)
- Rapid determination
- Intermediate determination
- Comprehensive determination.

Criteria for the selection of the appropriate level of RDM determination include:

- Degree to which the catchment is already utilised
- Sensitivity and importance of a catchment, and
- Potential impact of proposed water use.

LEVEL	USE
Desktop estimate	For use in Water Situation Assessment Model (WSAM) as part of planning processes only.
Rapid determination	Individual licensing for small impacts in unstressed catchments of low importance & sensitivity; compulsory licensing "holding action"
Intermediate determination	Individual licensing in relatively unstressed catchments
Comprehensive determination	All compulsory licensing. In individual licensing, for large impacts in any catchment. Small or large impacts in very important and/or sensitive catchments.

Table 12. Potential uses for different levels of Reserve determinations.

The generic procedures for determining an Ecological Reserve are shown in Figure 33.



Figure 33. Generic procedures for the determination of Resource Directed Measures.

Mtata Estuary

The1998 South Africa National Water Act requires a reserve to satisfy basic human needs and to protect aquatic ecosystems. The basic human needs reserve is the right of every person to 25 liters of water of adequate quality per day. In addition, the Act establishes the Ecological Reserve to protect rivers, wetlands, estuaries and groundwater.

An example to implementing provisions of the Act is the approach used to identify inflow requirements for the Mtata Estuary (Figure 34). In the watershed, there is a storage capacity that is 50% of mean runoff and only 8% of the runoff now reaches the sea. The approach to determine the Ecological Reserve is an interesting example of adding explicit value judgments to the process of setting minimum flows (Adams et al. 2002). A multi-step process was adopted where values (i.e., expert opinions) are of primary consideration. The method includes documenting geographical boundaries of the estuary and determining estuarine health by comparing it to a national rating system. Altogether a seven step process was used:

- 1. Delineate geographical boundaries.
- 2. Ecoregional typing.
- 3. Assess present state and reference condition.
- 4. Determine present ecological status and importance using ecological health and importance indices.
- 5. Determine ecological management class
- 6. Set the quantity of the reserve and resource quality objectives.
- 7. Design resource monitoring program.

The more pristine, or healthy, an ecosystem, the more inflow is required. The evaluations are based on best expert opinion and value judgments, which are rendered as scores. Results are used to select an acceptable river flow regime that represents the highest reduction in inflow that will still protect the estuary. The Mtata Estuary had high scores because it was an Ecological Reserve so it was assigned a high inflow requirement.



Figure 34. Mtata esturay, South Africa.

Summary: Water Supply, People, and the Future

At this point in time, there is no question as to whether freshwater inflow is important to coastal ecosystems. Rather, the important questions are how, when, where, and in what quantities inflow should be utilized for environmental purposes (Study Commission on Water for Environmental Flows 2004). Scientific methods and protocols have advanced over the past 40 years to improve our understanding of the importance of freshwater inflow (Figure 2). One of the main conclusions of this work is that adaptive management and precautionary principle methods need to be incorporated into the scientific study, management strategy implementation, and regulatory permitting phases of future freshwater inflow studies (Study Commission on Water for Environmental Flows, 2004; Montagna et al. 2009). This important conclusion should be applied coast-wide throughout the World to develop future approaches of managing freshwater inflow for adequate protection of coastal ecosystems.

As the famous Frenchman Victor Hugo said "science has the first word on everything, but the last word on nothing". This satiric yet true statement outlines the importance of always having a stakeholder driven, integrated ecosystem-based management process. So, if the key to using the scientific framework in Figure 4 is to work backwards, then stakeholder involvement in choosing the estuarine resources to be protected is a key first step. Again, Alber (2002) recognized that managing freshwater flows has roles for politicians, resource managers, scientists, and the public (Figure 4). An important feature of Alber's conceptual model is that it is essentially an adaptive management process because managers can modify decisions based on information flow (Figure 35).



Figure 35. Conceptual model of inflow management with roles of stakeholders. Solid lines denote control and dashed lines denote information flow. Switch denotes adaptive management.

The process outlined in Figure 34 is essentially an adaptive management process, and it has been applied to determining the environmental flows for Nueces Delta, Texas, USA that are needed to maintain estuarine productivity (Montagna et al., 2009). When the Choke Canyon Reservoir was constructed on the Nueces River in 1982, a special condition required the City of Corpus Christi to provide not less than 185 million cubic meters (151,000 ac-ft) of water per year to the Nueces Estuary through a combination of spills, releases, and return flows to maintain ecological health and productivity of living marine resources. However, no releases were made and salinities in Nueces Bay increased to hypersaline conditions during the drought period of 1988 – 1990. The estuary was inverted where fresher seawater entered the delta and diluted the hypersaline parts of the estuary only on high tides. In contrast, the bay historically supported populations of shrimp and ovsters generally requiring salinities in the range of 10 - 20 ppt. Since 1990, a series of restoration activities, monitoring and experimental studies, and other amendments to the water permit has resulted in a stakeholder driven process that increased environmental health while providing a sustainable water supply to the region. The lesson learned is that the goal of achieving consensus can occur with a blend and balance of science and a stakeholder decision process (Figure 35).

The relationship between inflow and estuarine resources is surely a problem of biocomplexity. Further adding to the complexity are the long period climate cycles (e.g., El Niño Southern Oscillation, the Pacific Decadal Oscillation, and the North Atlantic Multi-decadal Oscillation), which cause lags in biological responses to current events (Esteves 2002). After reviewing many inflow studies, Estevez (2002) came to three conclusions: 1) flow-effects studies will be improved if they occur at multiple trophic levels, multiple scales, and multiple temporal periods, 2) within-estuary salinity gradients have to be exploited because they control structure and function, but care must be take because of non-linear relationships, and 3) it must be determined if instream analyses can inform inflow recommendations, that is to what extent are riverine flows protective of estuarine resources.

Whatever classification scheme is used for the methodologies, the important considerations in choosing an appropriate approach include: (1) whether data are available to apply the methodology in a scientifically sound manner; (2) whether results of applying the methodology are reasonably accurate; and (3) whether the results can be translated into flow recommendations that can be implemented by the responsible parties. By conducting studies that meet these three considerations, scientists will be able to more accurately determine the ecological and socio-economical impacts of changing freshwater inflows. In turn, stakeholders and managers will then be able to make well-informed decisions to successfully manage freshwater inflows to their local coasts.

References

- Adams, J.B., G.C. Bate, T.D. Harrison, P. Huizinga, S. Taljaard, L.Van Niekerk, E.E. Plumstead, A.K. Whitfield, and T.H. Wooldridge. 2002. A method to assess the freshwater inflow requirements of estuaries and application to the Mtata Estuary, South Africa. *Estuaries* 25:1382-1393.
- Adamson, D., T. Mallawaarachchi and J. Quiggin. 2009. Declining inflows and more frequent droughts in the Murray–Darling Basin: climate change, impacts and adaptation. *The Australian Journal of Agricultural and Resource Economics* 53: 345–366.
- Alber, M. 2002. A conceptual model of estuarine freshwater inflow management. *Estuaries* 25: 1246-1261.
- Allan, E. L., P. W. Froneman, and A. N. Hodgson. 2006. Effects of temperature and salinity on the standard metabolic rate (SMR) of the caridean shrimp *Palaemon peringueyi*. *Journal of Experimental Marine Biology and Ecology* 337:103-108.
- Andree, S. (ed.). 1983. Apalachicola Oyster Industry: Conference Proceedings. Florida Sea Grant College, 85 pp.
- Aragón-Noriega E.A. and Calderón-Aguilera L.E. 2000. Does damming of the Colorado River affect the nursery area of blue shrimp *Litopenaeus stylirostris* (Decapoda: Penaeidae) in the Upper Gulf of California? *Rev. Biol. Trop.* 48: 867-71
- Atrill, M. J., S. D. Rundle, and R. M. Thomas. 1996. The influence of drought-induced low freshwater flow on an upper-estuarine macroinvertebrate community. *Water Resources* 30: 261-268.
- Australian Bureau of Statistics. 2008. 4610.0.55.007 Water and the Murray-Darling Basin A Statistical Profile, 2000-01 to 2005-06. Released 15 August 2008. <u>www.abs.gov.au</u>
- Banse, K. and Mosher, S. 1980. Adult body mass and annual production/biomass relationships of field populations. *Ecological Monographs* 50:355-379.
- Barton, J.L., A.J. Pope, G.P. Quinn and J.E. Sherwood. 2008. Identifying threats to the ecological condition of Victorian estuaries. Department of Sustainability and Environment Technical Report. Victoria, Australia.
- Bates, B.C., Z.W. Kundzewicz, S. Wu and J.P. Palutikof (eds.). 2008: Climate Change and Water. Technical Paper of the Intergovernmental Panel on Climate Change, IPCC Secretariat, Geneva, 210 pp.
- Bennett W.A. 2005. Critical assessment of the delta smelt population in the San Francisco Estuary, California. San Francisco Estuary and Watershed Science [online serial] 3(2), Article 1. Available at: http://www.escholarship.org/uc/item/0725n5vk?display=all

- Bourman, R.P. and E.J. Barnett. 1995. Impacts of river regulation on the terminal lakes and mouth of the River Murray, South Australia. *Australian Geographical Studies* 33: 101–115.
- Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando, and D.R.G. Farrow. 1999. National estuarine eutrophication assessment: effects of nutrient enrichment in the nation's estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science, Silver Spring, Maryland.
- Brock, D.A. 2001. Nitrogen budget for low and high freshwater inflows, Nueces Estuary, Texas. *Estuaries* 24: 509-521.
- Browder, J. A., and Moore, D. 1981. A new approach to determining the quantitative relationship between fishery production and the flow of fresh water to estuaries. Pages 403-430 in R. D. Cross and D. L. Williams, editors. Proceedings of the national symposium on freshwater inflow to estuaries. Volume 1. U.S. Fish and Wildlife Service, U.S. Department of Interior, Washington, D.C.
- Browder, J. A., Z. Zein-Eldin, M. M. Criales, M. B. Robblee, S. Wong, T. L. Jackson, and D. Johnson. 2002. Dynamics of pink shrimp (*Farfantepenaeus duorarum*) recruitment potential in relation to salinity and temperature in Florida Bay. *Estuaries* 25:1355-1371.
- Bureau of Reclamation. 2000. Concluding Report: Rincon Bayou Demonstration Project. Volume I: Executive Summary. United States Department of the Interior, Bureau of Reclamation, Oklahoma-Texas Area Office, Austin, Texas.
- CALFED Bay-Delta Program. 2010. Available online at: http://calwater.ca.gov/index.aspx
- California's Colorado River Water Use Plan. 2000. Colorado River Board of California. The Resources Agency, State of California. Available online at: http://www.crb.ca.gov/Calif_Plan%20May%2011%20Draft.pdf
- Cappo, M., D.M. Alongi, D.M. Williams and N. Duke. 1 998a. A review and synthesis of Australian fisheries habitat research: Major threats, issues and gaps in knowledge of coastal and marine fisheries habitats - Volume 1: A prospectus of opportunities for the FRDC "ecosystem protection program" FRDC 95/055. Australian Institute of Marine Science.
- Cappo, M., D.M. Alongi, D.M. Williams and N. Duke. 1998b. A review and synthesis of Australian fisheries habitat research: Major threats, issues and gaps in knowledge of coastal and marine fisheries habitats - Volume 2: Scoping Review FRDC 95/055. Australian Institute of Marine Science.
- Carter N.T., Corn M.L., Abel A., Kaplan S.M., Buck E.H., Brougher C., and Alexander K. 2008. Apalachicola-Chattahoochee-Flint (ACF) drought: federal water management issues. CRS report for congress. Congressional Research Service. Order code RL34326.

Central Coast Region Basin Plan. 2006. Central Coast Regional Water Quality Control Board.

Central Valley Basin Plan. 2009. Central Valley Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/centralvalley/water issues/basin plans/

- CH2M Hill. 1999. Water supply needs and sources assessment: Alternative water supply investigation: Review of established minimum flows and levels for the Wekiva River system. Special Publication SJ99-SP1. St. John's River Water Management District, Palatka, Florida.
- Chadwick, M. A., and J. W. Feminella. 2001. Influence of salinity and temperature on the growth and production of a freshwater mayfly in the Lower Mobile River, Alabama. Limnology and Oceanography 46:532-542.
- Chan, T.U., D.P. Hamilton, B.J. Robson, B.R. Hodges and C.J. Dallimore, 2002. Impacts of hydrological changes on phytoplankton succession in the Swan River, Western Australia, *Estuaries* 25: 1405-1415.
- Chollett, I., and D. Bone. 2007. Effects of heavy rainfall on polychaetes: Differential spatial patterns generated by a large-scale disturbance. J Exp Mar Biol Ecol 340:113-125
- Cisneros-Mata M.A., Montemayor-Lopez G., Roman-Rodriguez M.J. 1995. Life history and conservation of *Totoaba macdonaldi*. Conserv Biol 9:806-814
- Clark, S.D. 1982. "The River Murray Waters Agreement: down the drain or up the creek?" *Civil Engineering Transactions* 24: 201-208.
- Colorado River Basin Plan. 2006. Colorado River Basin Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/coloradoriver/publications_ forms/publications/docs/basinplan_2006.pdf
- Colorado River Compact. 1922. US Department of the Interior, Bureau of Reclamation. Available online at: http://www.usbr.gov/lc/region/g1000/pdfiles/crcompct.pdf
- Connor, J., K. Schwabe, D. King, D. Kaczan and M. Kirby. 2009. Impacts of climate change on lower Murray irrigation. *The Australian Journal of Agricultural and Resource Economics* 53: 437–456.
- Copeland, B.J. 1966. Effects of decreased river flow on estuarine ecology. *Journal Water Pollution Control Federation* 38: 1831-1839.
- Costanza, R., d'Arge, R., de Groots, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., Raskins, R.G., Sutton, P., van den Belt, M. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253-260.
- Cowie G (ed.) 2002. Reservoirs in Georgia: meeting water supply needs while minimizing impacts. River Basin Science and Policy Center, University of Georgia. 34 pp.

- Crabb, P. 1988. Managing the Murray-Darling Basin. Australian Geographer 19: 64-88.
- Crase, L., S. O'Keefe and B. Dollery. 2009. Water Buy-Back in Australia: Political, Technical and Allocative Challenges. A Contributed Paper to the Australian Agricultural & Resource Economics Society's Annual Conference, Cairns, February 11-13, 2009.
- Cross, R., and D. Williams (eds.). 1981. Proceedings of the National Symposium on Freshwater Inflow to Estuaries. U.S. Fish and Wildlife Service, Office of Biological Services. FWS/OBS-81/04. 2 vols.
- Crossland, C. J., H. H. Kremer, H. J. Lindeboom, J. I. Marshall Crossland, and M. D. A. Le Tissier, editors. 2005. Coastal Fluxes in the Anthropocene. Springer-Verlag, Berlin, Germany.
- CSIRO. 2008. Water availability in the Murray-Darling Basin. Summary of a report to the Australian Government from the CSIRO Murray-Darling Basin Sustainable Yields Project. CSIRO, Australia. 12 pp. <u>www.csiro/mdbsy</u>
- Dahms, H.U. 1990. Salinity, temperature, pH tolerance and grain-size preference of *Paramphiascella fulvofasciata* (copepoda, harpacticoida). *Biol Jb Dodonaea* 58:48-62.
- Davies, J.L. 1973. Geographical variation in coastal development. Hafner, New York.
- Davies, P.E. and S.R. Katish. 1994. Influence of River Hydrology on the Dynamics and Water Quality of the Upper Dewent Estuary, Tasmania. *Australian Journal of Marine and Freshwater Resources* 45: 109-130.
- Day, J.W. C.A.S. Hall, W.M. Kemp, and A. Yáñez-Arancibia. 1989. Estuarine Ecology. John Wiley, New York.
- Department of Water Affairs and Forestry. (2004. Water resource protection and assessment policy implementation process. Resource directed measures for protection of water resource: Methodology for the Determination of the Ecological Water Requirements for Estuaries. Version 2, Final Draft. Pretoria.
- DEIS. 1998. Draft Environmental Impact Statement. Water Allocation for the Apalachicola-Chattahoochee-Flint River Basin: Alabama, Florida and Georgia. US Army Corps of Engineers, Mobile District. http://www.sam.usace.army.mil/pd/actacfeis/
- Delta Science Program. 2010. Delta Stewardship Council. Available online at: http://www.deltacouncil.ca.gov/delta science program/
- DEWHA (Department of the Environment, Water, Heritage and the Arts). 2010. Murray Darling Basin. www.environment.gov.au/water/locations/murray-darling-basin
- Diener, R.A. 1975. Cooperative Gulf of Mexico Estuarine Inventory and Study-Texas: Area Description. U.S. Department of Commerce, NOAA Tech. Rep. NMFS Circ. 393, 125 pp.

- Digby, M.J. P. Saenger, M.B. Whelan, D. McConchie, B. Eyre, N. Holmes and D. Bucher. 1999. A Physical Classification of Australian Estuaries. LWRRDC Occasional Paper 16/99 (National River Health Program, Urban Sub–Program, Report No. 9), Land and Water Resources Research and Development Corporation, Canberra, ACT, Australia. au.riversinfo.org/library/nrhp/estuary_clasifn/
- Doering, P.H., R.H. Chamberlin, and D.E. Haunert. 2002. Using submerged aquatic vegetation to establish minimum and maximum freshwater inflows to the Caloosahatchee Estuary, Florida. *Estuaries* 25:1343-1354.
- Drake, P., A. M. Arias, F. Baldo, J. A. Cuesta, A. Rodriguez, A. Silva-Garcia, I. Sobrino, D. Garcia-Gonzalez, and C. Fernandez-Delgado. 2002. Spatial and temporal variation of the nekton and hyperbenthos from a temperate European estuary with regulated freshwater inflow. Estuaries 25:451-468.
- Drinkwater, K.F., and K.T. Frank. 1994. Effects of river regulation and diversion on marine fish and invertebrates. *Aquatic Conservation: Marine and Freshwater Ecosystems* 4: 135-151.
- Dynesius, M. and C. Nilsson. 1994. Fragmentation and flow regulation of river systems in the northern third of the world. *Science* 266: 753-762.
- ECT. 2008. Environmental evaluations for the development of minimum flows and levels for the St. John's River near Deland at State Road 44, Volusia County. Special Publication SJ2004-SP30. St. John's River Water Management District, Palatka, Florida.
- Estevez, E.D. 2002. Review and assessment of biotic variables and analytical methods used in estuarine inflow studies. *Estuaries* 25: 1291-1303.
- Estevez E.D. and Marshall M.J. 1993. Sebastian River salinity regime. Mote Marine Laboratory Special Publication SJ94-SP1. Mote Marine Laboratory, Sarasota, Florida.
- Ferguson, G.J., T.M. Ward, and M.C. Geddes. 2008. Do historical catches and recent age structures of mulloway, *Argyrosomus japonicus*, reflect freshwater inflows in the remnant estuary of the Murray River, South Australia? *Aquatic Living Resources* 21: 145-152.
- Ferraris, J. D., K. Fauchald, and B. Kensley. 1994. Physiological responses to fluctuation in temperature or salinity in invertebrates - Adaptations of Alpheus viridari (Decapoda, Crustacea), Terebellides parva (Polychaeta) and Golfinigia cylindrata (Sipunculida) to the mangrove habitat. Mar Biol 120:397-406.
- Feyrer F., Nobriga M.L., Sommer T.R. 2007. Multidecadal trends for three declining fish species: habitat patterns and mechanisms in the San Francisco Estuary, California, USA. Can J Fish Aquat Sci 64:723-734
- Finney, C. M. 1979. Salinity Stress in Harpacticoid Copepods. Estuaries 2:132-135.

- FL DEP. 2010. Florida Department of Environmental Protection. Apalachicola-Chattahoochee-Flint River System Timeline of Action as of July 27, 2009. http://www.dep.state.fl.us/ mainpage/acf/timeline.htm
- Flannery, M.S., E.B. Peebles, and R.T. Montgomery. 2002. A percent-of-flow approach for managing reductions of freshwater inflows from unimpounded rivers to southwest Florida estuaries. *Estuaries* 25:1318-1332.
- Florida Administrative Code. 2005. Section 62.40-473. Minimum Flows and Levels. https://www.flrules.org/gateway/ruleNo.asp?ID=62-40.473
- Fradkin, Philip L. 1981. A River No More: The Colorado River and the West. University of California Press.
- Geddes, M. 2004. Survey to investigate the ecological health of the North and South Lagoons of the Coorong, June/July 2003. Report submitted to Upper South East Environmental Management Advisory Group. South Australian Research and Development Institute. 21 pp.
- Gillson, J., J. Scandol and I. Suthers. 2009. Estuarine gillnet fishery catch rates decline during drought in eastern Australia. *Fisheries Research* 99: 26–37.
- Gippel, C., 2002. Workshop on environmental water requirements for Australian estuaries. Outcomes and further directions. Report to Environment Australia.
- Gippel, C.J., B. Anderson, C. Harty, N. Bond, J. Sherwood and A. Pope. 2009. Gap analysis and strategy development for national level estuary environmental flows policies. Waterlines report, National Water Commission, Canberra, Australia.
- Glenn E., Garcia J., Tanner R., Congdon C., and Luecke D. 1999. Status of Wetlands Supported by Agricultural Drainage Water in the Colorado River Delta, Mexico. Horticultural Science 34(1): 16-21.
- Glenn E.P., Zamora-Arroyo F., Nagler P.L., Briggs M., Shaw W., Flessa K. 2001. Ecology and conservation biology of the Colorado River delta, Mexico. J Arid Environ 49:5-15
- Glenn E., Lee C., Fegler R., Zengel S. 1996. Effects of Water Management on the Wetlands of the Colorado River Delta, Mexico. Conservation Biology 10(4):1175-1186.
- GWP (Global Water Partnership). 2003. 'The Murray-Darling Basin Commission, Australia (Case # 25)' *in* Integrated Water Resources Management Toolbox. www.gwptoolbox.org
- Halliday, I.A., J.B. Robins, D.G. Mayer, J. Staunton-Smith and M.J. Sellin. 2008. Effects of freshwater flow on the year-class strength of a non-diadromous estuarine finfish, king threadfin (*Polydactylus macrochir*), in a dry-tropical estuary. *Marine and Freshwater Research* 59: 157–164.

- Hardie, R., L. Lloyd and J. Sherwood. 2006. Determining the Environmental Water
 Requirements of Victoria's Estuaries Development of a Module to extend the FLOWS
 Methodology Draft Method for Pilot Trials. Earth Tech, Melbourne, Australia.
- Harvey, N. 1988. Coastal management issues for the mouth of the River Murray, South Australia. *Coastal Management* 16: 139–149.
- Harvey, N. 1996. The significance of coastal processes for management of the River Murray estuary. *Australian Geographical Studies* 34: 45–57.
- Hatton MacDonald, D. and M.D. Young. 2001. A case study of the Murray-Darling Basin: Final report for the International Water Management Institute. Policy and Economic Research Unit, CSIRO Land and Water, Adelaide, Australia.
- Hess, M. 2010. Freshwater inflow management issues in Texas: What's being done?.Presentations at Freshwater Inflows 2010 and Beyond Conference, Corpus Christi, February 2010.
- Hoese, H.D. 1967. Effects of higher than normal salinities on salt marshes. *Contributions to Marine Science* 12: 249-261.
- Istanbul Water Guide. 2009. Outcomes of the 5th World Water Forum. Istanbul. http://www.worldwatercouncil.org/fileadmin/wwc/World_Water_Forum/WWF5/global_w ater_framework_part_1_final.pdf
- IBWC. 2000. Minute 306: Conceptual framework for United States—Mexico studies for future recommendations concerning the riparian and estuarine ecology of the Limitrophe section of the Colorado River and its associated delta. Amends United States-Mexico, 59 Stat. 1219. Treaty with Mexico Respecting the Utilization of the Waters of the Colorado and Tijuana Rivers and of the Rio Grande, Feb. 3, 1944.
- Kalke, R. D., and P. A. Montagna. 1991. The effect of freshwater inflow on macrobenthos in the Lavaca River Delta and Upper Lavaca Bay, Texas. *Contributions to Marine Science* 32:49-71.
- Katz, D. 2006. Going with the flow: Preserving and restoring instream water allocations. In P.H. Gleik (ed.), The World's Water 2006-2007. Washington D.C.: Island Press 29-49.
- Keiser, R. K., and D. V. Aldrich. 1973. Gradient apparatus for study of salinity preference of small benthic and free swimming organisms. Contrib Mar Sci 17:153-162.
- Kim, H.-C., and P.A. Montagna. 2009. Implications of Colorado River freshwater inflow to benthic ecosystem dynamics: a modeling study. Est Coast Shelf Sci. 83(4) 491-504.
- Kimmerer, W.J. 2002. Physical, biological, and management responses to variable freshwater inflow into San Francisco Estuary. *Estuaries* 25:1275-1290.

- Kjerfve, B. 1979. Measurements and analysis of water current, temperature, salinity and density.In: Dyer KR (ed) Hydrography and Sedimentation in Estuaries. Cambridge University Press, United Kingdom, pp 186-216.
- Kowalewski M., Serrano G.E.A., Flessa K.W., Goodfriend G.A. 2000. Dead delta's former productivity: Two trillion shells at the mouth of the Colorado River. Geology 28:1059-1062
- Kurup, G.R., D.P. Hamilton and J.C. Patterson. 1998. Modelling the effect of seasonal flow variations on the position of salt wedge in a microtidal estuary, *Estuarine, Coastal and Shelf Science* 47: 191–208.
- Lahontan Region Basin Plan. 1995. Lahontan Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/lahontan/water_issues/programs/ basin plan/references.shtml
- Lane, N.F., R.M. Bierbaum, and M.T. Anderson. 2003. Science and water policy for the United States. In: Lawford, R.G., D.D. Fort, H.C. Hartmann, and S. Eden (eds.), Water: Science, Policy, and Management. American Geophysical Union, Water Resource Monograph 16, Washington, DC., pp. 207-222.
- La Peyre M.K., Gossman B., La Peyre, J.F. 2009. Defining optimal freshwater flow for oyster production: effects of freshet rate and magnitude of change and duration on eastern oysters and Perkinsus marinus Infection. *Estuaries and Coasts* 32:522-534.
- Larkin, T.J. and G.W. Bomar. 1983. *Climatic Atlas of Texas*. Texas Department of Water Resources. Austin, Texas. 151 pp.
- Lercari, D., O. Defeo, and E. Celentano. 2002. Consequences of a freshwater canal discharge on the benthic community and its habitat on an exposed sandy beach. Marine Pollution Bulletin 44:1397-1404.
- Lin C.-I. 2010. Apalachicola-Chattahoochee-Flint Tri-State Negotiation. Aquapedia. Tufts University. https://wikis.uit.tufts.edu/confluence/display/aquapedia/Apalachicola-Chattahoochee-Flint+Tri-State+Negotiation
- Livingston, R.J., X.F. Niu, F.G. Lewis, and G.C. Woodsum. 1997. Freshwater input to a Gulf estuary: Long-term control of trophic organization. *Ecological Applications* 7: 277-299.
- Longley, W.L. (ed.). 1994. Freshwater inflows to Texas bays and estuaries: ecological relationships and methods for determination of needs. Texas Water Development Board and Texas Parks and Wildlife Department, Austin, Texas. 386 p.
- Los Angeles Region Water Quality Control Basin Plan. 1995. Los Angeles Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/losangeles/ water_issues/programs/basin_plan/basin_plan_documentation.shtml

- Luecke, D. 2000. An environmental perspective on large ecosystem restoration processes and the role of the market, litigation, and regulation. Arizona Law Review, 42: 395-410.
- Luecke, D.F., Pitt, J., Congdon, C., Glenn, E., Valdés-Casillas, C., and Briggs, M. 1999. A delta once more: Washington, D.C., Environmental Defense Fund Publications. Available online at: http://www.edf.org/documents/425_delta.pdf
- MacKenzie, C.L., Jr. 1977. Development of an aquaculture program for rehabilitation of damaged oyster reefs in Mississippi. U.S. Fish and Wildlife Service Marine Fisheries Review, 39(8): 1-13.
- Mackin, J. 1956. *Dermocystidium marinum* and salinity. *Proceedings of the National Shellfish* Association 46:116-128.
- Mannino, A., and P.A. Montagna. 1997. Small-scale spatial variation of macrobenthic community structure. *Estuaries* 20:159-173.
- Mattson, R. A. 2002. A resource-based framework for establishing freshwater inflow requirements for the Suwannee River estuary. Estuaries 25:1333-1342.
- McEachron, L. W. and Fuls, B., 1996. Trends in relative abundance and size of selected finfishes and shellfishes along the Texas coast, Coastal Fisheries Division, Management Data Series, Austin, Texas.
- McKee, B.A., and M. Baskaran. 1999. Sedimentary processes of Gulf of Mexico estuaries. In: Bianchi TS, Pennock JR, Twilley RR (eds) Biogeochemistry of Gulf of Mexico estuaries. John Wiley & Sons, New York.
- McLeod, R. J., and T. R. Wing. 2008. Influence of an altered salinity regime on the population structure of two infaunal bivalve species. Est Coast Shelf Sci 78:529-540.
- Meade, R.H., T. Yuzyk, and T. Day. 1990. Movement and storage of sediment in rivers of the United States and Canada. In: Riggs WH (ed) The geology of North America. Geological Society of America.
- Menzel, R.W., N.C. Hulings, and R.R. Hathaway, 1958. Causes of depletion of oysters in St. Vincent Bay, Apalachicola Bay, Florida. *Proceedings of the National Shellfisheries Association*, 48: 66-71.
- Merrick, J.R. and G.E. Schmida 1984. Australian Freshwater Fishes: Biology and Management. Griffin Press, Sydney.
- Millennium Ecosystem Assessment. 2005. Ecosystems and human well being: current state and trends. Findings of the condition and trends working group. Island Press. 948 pp.
- Milton, D., I. Halliday, M. Sellin, R. Marsh J. Staunton-Smith and J. Woodhead. 2008. The effect of habitat and environmental history on otolith chemistry of barramundi *Lates calcarifer* in estuarine populations of a regulated tropical river. *Estuarine, Coastal and Shelf Science* 78: 301-315.

- Montagna, P.A. 2008. Long-term response of benthic organisms to freshwater inflow in Texas coastal bend estuaries. Final Report to Texas Parks and Wildlife Department. Available at: http://harteresearchinstitute.org/montagna/publications.html
- Montagna, P.A., M. Alber, P. Doering. and M.S. Connor. 2002a. Freshwater inflow: Science, policy, management. *Estuaries* 25:1243-1245.
- Montagna P.A., Estevez E.D., Palmer T.A., and Flannery M.S. 2008a. Meta-analysis of the relationship between salinity and molluscs in tidal river estuaries of southwest Florida, USA. Am Malacol Bull 24:101-115
- Montagna, P.A, E.M. Hill, and B. Moulton. 2009. Role of science-based and adaptive management in allocating environmental flows to the Nueces Estuary, Texas, USA. In: Brebbia, C.A. and E. Tiezzi (eds.), *Ecosystems and Sustainable Development VII*, WIT Press, Southampton, UK, pp. 559-570.
- Montagna, P.A. and R.D. Kalke. 1992. The effect of freshwater inflow on meiofaunal and macrofaunal populations in the Guadalupe and Nueces Estuaries, Texas. *Estuaries* 15:307-326.
- Montagna, P.A. and R.D. Kalke. 1995. Ecology of infaunal Mollusca in south Texas estuaries. *American Malacological Bulletin* 11:163-175.
- Montagna, P. A., R. D. Kalke, and C. Ritter. 2002b. Effect of restored freshwater inflow on macrofauna and meiofauna in upper Rincon Bayou, Texas, USA. *Estuaries* 25: 1436-1447.
- Montagna, P.A., and J. Li. 1996. Modeling And Monitoring Long-Term Change In Macrobenthos In Texas Estuaries. Final Report to the Texas Water Development Board. University of Texas at Austin, Marine Science Institute, Technical Report No. TR/96-001, Port Aransas, Texas, 149 pp. Available at: http://harteresearchinstitute.org /montagna/docs/TWDB1996.pdf
- Montagna P.A., Palmer T.A., and Pollack J.B. 2008b. St. John's Estuary: estuarine benthic macroinvertebrates. Phase 1 final report. Submitted to St. John's River Water Management District.
- Montagna, P.A., Li, J., Street, G.T. 1996. A conceptual ecosystem model of the Corpus Christi Bay National Estuary Program Study Area. Publication CCBNEP-08, Texas Natural Resource Conservation Commission, Austin, Texas. 114 pp. Available at: http://cbbep.org/ publications/virtuallibrary/ccbnep08.pdf
- Montagna, P.A., G. Ward, and B. Vaughan. 2010. The importance and problem of freshwater inflows to Texas estuaries. In: Water Policy in Texas: Management for a Diverse Land. In press.
- Montague, C. L., and J. A. Ley. 1993. A possible effect of salinity fluctuation on abundance of benthic vegetation and associated fauna in northeastern Florida Bay. Estuaries 16:703-717.
- Morgan, M., L. Strelein and J. Weir. 2004. Indigenous Rights to Water in the Murray Darling Basin. AIATSIS Research Discussion Paper, vol. 14. AIATSIS, Canberra.
- MBBMC (Murray-Darling Basin Ministerial Council). 1987. Murray-Darling Basin Environmental Resources Study. Sydney, NSW, Australia. 426 pp.
- MDBA (Murray-Darling Basin Authority). 2010. Website: www.mdba.gov.au . Accessed March 2010.
- Naiman, R.J., J.J. Magnuson, D.M. McKnight, J.A. Stanford, J.R. Karr. 1995. Freshwater ecosystems and their management: A national initiative. *Science* 270:584-585.
- National Water Policy of India. 2002. Government of India, Ministry of Water Resources. New Delhi. http://wrmin.nic.in/writereaddata/linkimages/nwp20025617515534.pdf
- Nichols, F.H., J.E. Cloern, S.N. Luoma and D.H. Peterson. 1986. The modification of an estuary. *Science* 231: 567-573.
- Normant, M., and I. Lamprecht. 2006. Does scope for growth change as a result of salinity stress in the amphipod Gammarus oceanicus? Journal of Experimental Marine Biology and Ecology 334:158-163.
- North Coast Region Water Quality Control Basin Plan. 2007. North Coast Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/northcoast/ water_issues/programs/basin_plan/083105-bp/02_introduction.pdf
- NWFWMD. 2010. Northwest Florida Water Management District. Available online at: http://www.nwfwmd.state. fl.us/
- Office of Program Policy Analysis and Government Accountability (OPPAGA). 1995. Performance Review of the Consumptive Use Permitting Program Administered by the Department of Environmental Protection and the Water Management Districts. Report No. 94-34. http://www.oppaga.state.fl.us/Summary.aspx?reportNum=94-34
- OPPAGA (Office of Program Policy Analysis and Government Accountability). 1995.
 Performance Review of the Consumptive Use Permitting Program Administered by the Department of Environmental Protection and the Water Management Districts. Report No. 94-34. http://www.oppaga.state.fl.us/Summary.aspx?reportNum=94-34
- Olsen, S.B, T.V. Padma, and B.D. Richter. 2007. Managing Freshwater Inflows to Estuaries: A Methods Guide. U.S. Agency for International Development, Washington, D.C. 44 pp.
- Orlando, S.P., Jr., L.P. Rozas, G.H. Ward, and C.J. Klein, 1993. Salinity Characteristics of Gulf of Mexico Estuaries. Silver Spring, MD: National Oceanic and Atmospheric Administration, Office of Ocean Resources Conservation and Assessment. 209 pp.

- Paton, D.C., D.J. Rogers, B.M. Hill, C.P. Bailey and M. Ziembicki. 2009. Temporal changes to spatially stratified waterbird communities of the Coorong, South Australia: implications for the management of heterogenous wetlands. *Animal Conservation* 12: 408-417.
- Pearson, T.H. and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichement and pollution in the marine environment. *Oceanography and Marine Biology Annual Reviews* 16:229-311.
- Pierson W.L., Bishop K., Van Senden D., Horton P.R. and Adamantidis C.A. 2002. Environmental water requirements to maintain estuarine processes. Environmental flows initiative technical report number 3, Commonwealth of Australia, Canberra.
- Pitt J. 2001. Can we restore the Colorado River delta? J Arid Environ 49:211-220
- Pitt J., Luecke D., Cohen M., Glenn E., and Valdes-Casillas C. 2000. Two countries, one river: managing ecosystem conservation in the Colorado River delta. Natural Resources Journal, 40: 819-864.
- Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R.E. Sparks and J.C. Stromberg. 1997. The natural flow regime; a paradigm for river conservation and restoration. *BioScience* 47: 769-784.
- Poff, N. L., J. D. Allan, M. A. Palmer, D. D. Hart, B. D. Richter, A. H. Arthington, K. H. Rogers, J. L. Meyers and J. A. Stanford. 2003. Fiver flows and water wars: emerging science for environmental decision making. *Frontiers in Ecology and the Environment* 1: 298-306.
- Pollack, J.B., J.W. Kinsey, and P.A. Montagna. 2009. Freshwater Inflow Biotic Index (FIBI) for the Lavaca-Colorado Estuary, Texas. *Environmental Bioindicators* 4:153–169.
- Postel, S.L., G.C. Daily, P.R. Ehrlich. 1996. Human appropriation of renewable fresh water. *Science* 271:785-788.
- Powell, G.L., J. Matsumoto, and D.A. Brock. 2002. Methods for determining minimum freshwater inflow needs of Texas bays and estuaries. *Estuaries* 25:1262-1274.
- Pringle, C.M., M.C. Freeman, and B.J. Freeman. 2000. Regional effects of hydrologic alterations on riverine macrobiota in the New World: Tropical-Temperate comparisons. *BioScience* 50: 807-823.
- Pritchard, D.W. 1952. Estuarine hydrography. In Advance in Geophysics, Vol. I Academic Press, New York, pp. 243-280.
- Pritchard, D.W. 1967. What is an estuary: physical viewpoint. In: G.H. Lauff (ed.) Estuaries. American Association for the Advancement of Science, Washington, D.C., pp.52-63.
- Rabalais, N.N., and S.W. Nixon. 2002. Preface: Nutrient over-enrichment of the coastal zone. *Estuaries* 25: 639-639

- Republic of South Africa, 1998. National Water Act, No 36 of 1998. Government Gazette, Pretoria.
- Rhoads, D.C., P.L. McCall, and J.Y. Yingst. 1978. Disturbance and production on the estuarine seafloor. *American Scientist* 66:577-586.
- Richter B.R., Mathews R., Harrison D.L., and Wigington R. 2003. Ecologically sustainable water management: managing river flows for ecological integrity. Ecol Appl. 13: 206–24.
- Riera P., P. A. Montagna, R. D. Kalke and P. Richard. 2000. Utilization of estuarine organic matter during growth and migration by juvenile brown shrimp *Penaeus aztecus* in a South Texas estuary. *Mar. Ecol. Prog. Ser.* 199:205-216.
- Robins, J.B., I.A. Halliday, J. Staunton-Smith, D.G. Mayer and M.J. Sellin. 2005. Freshwaterflow requirements of estuarine fisheries in tropical Australia: A review of the state of knowledge and the application of a suggested approach. *Marine and Freshwater Research* 56: 343-360.
- Robins, J., D. Mayer, J. Staunton-Smith, I. Halliday, B. Sawynok and M. Sellin. 2006. Variable growth rates of the tropical estuarine fish barramundi *Lates calcarifer* (Bloch) under different freshwater flow conditions. *Journal of Fish Biology* 69: 379-391.
- Robinson, L., P. Campbell, and L. Butler. 2000. Trends in Texas commercial fishery landings, 1972-1998. Management Data Series No. 173. Texas Parks and Wildlife Department. Austin, Texas.
- Rozas, L. P., T. J. Minello, I. Munuera-Fernandez, B. Fry, and B. Wissel. 2005. Macrofaunal distributions and habitat change following winter-spring releases of freshwater into the Breton Sound estuary, Louisiana (USA). Est Coast Shelf Sci 65:319-336.
- Ruhl, J.B., C. Lant, T. Loftus, S. Kraft, J. Adams, and L. Duram. 2003. Proposal for a model state Watershed Management Act. *Environmental Law* 33: 929-947.
- Rutger, S. M., and S. R. Wing. 2006. Effects of freshwater input on shallow-water infaunal communities in Doubtful Sound, New Zealand. Mar Ecol Prog Ser 314:35-47.
- Sabine and Neches Rivers and Sabine Lake Bay Basin and Bay Expert Science Team. 2009. Environmental Flows Recommendations Report: Final Submission to the Sabine and Neches Rivers and Sabine Lake Bay Basin and Bay Area Stakeholder Committee, Environmental Flows Advisory Group, and Texas Commission on Environmental Quality. Submitted Novmber 2009.
- San Diego Basin Plan. 1994. San Diego Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/sandiego/water_issues/programs/basin_plan/ index.shtml
- San Francisco Basin (Region 2) Water quality control plan (Basin plan). 2007. California Regional Water Quality Control Board, San Francisco Bay Region. Available online at:

http://www.swrcb.ca.gov/sanfranciscobay/water_issues/programs/basin_plan/docs/ bp_ch1withcover.pdf

- Santa Ana Region Basin Plan. 2008. Santa Ana Regional Water Quality Control Board. Available online at: http://www.waterboards.ca.gov/santaana/water_issues/programs/basin_plan/index.shtml
- Saoud, I. P., and D. A. Davis. 2003. Salinity tolerance of brown shrimp Farfantepenaeus aztecus as it relates to postlarval and juvenile survival, distribution, and growth in estuaries. Estuaries 26:970-974.
- Science Advisory Committee, 2004. Final Report Science Advisory Committee Report on Water for Environmental Flows. Presented to the Study Commission on Water for Environmental Flows. October 26, 2004.
- Science Advisory Committee. 2009a. Use of Hydrologic Data in the Development of Instream Flow Recommendations for the Environmental Flows Allocation Process and The Hydrology-Based Environmental Flow Regime (HEFR) Methodology. Report # SAC-2009-01-Rev1. April 20, 2009.
- Science Advisory Committee. 2009b. Methodologies for Establishing a Freshwater Inflow Regime for Texas Estuaries Within the Context of the Senate Bill 3 Environmental Flows Process. Report # SAC-2009-03-Rev1. June 5, 2009.
- Schmitt, R.J. and C. W. Osenberg. 1996. Detecting ecological impacts: concepts and applications in coastal habitats. Academic Press, San Diego, California, USA.
- SFWMD (South Florida Water Management Distric). 2010. Available online at: http://www.sfwmd.gov/
- SJRWMD (St. John's River Water Management District). 2005. District Water Management Plan, September 2005. Available online at: http://sjr.state.fl.us/dwmp/index.html
- Sommer T., Armor C., Baxter R., Breuer R., Brown L., Chotkowski M., Culberson S., Feyrer F., Gingras M., Herbold B., Kimmerer W., Mueller-Solger A., Nobriga M., Souza K. 2007. The collapse of pelagic fishes in the Upper San Francisco Estuary. Fisheries 32:270-277
- Soule, D. F. 1988. Marine organisms as indicators: reality or wishful thinking? In: Soule, D. F. and G. S. Kleppel (eds.) Marine Organisms as Indicators, Springer-Verlag, New York, pp. 1-11.
- SRWMD (Suwannee River Water Management District). 2010. Available online at: http://www.srwmd.state.fl.us/
- State Water Resources Control Board. 2010. California Environmental Protection Agency. Available online at: http://www.waterboards.ca.gov/about_us/water_boards_structure/ whoweare.shtml

- Steenstra, A. 2009. Accommodating Indigenous Cultural Values in Water Resource Management: The Waikato River, New Zealand; the Murray- Darling Basin, Australia; and the Colorado River, USA. Australian Agricultural & Resource Economics Society's Annual Conference, Cairns, February 11-13, 2009.
- Stevens L.E., Ayers T.J., Bennett J.B., Christensen K., Kearsley M.J.C., Meretsky V.J., Phillips A.M., Parnell R.A., Spence J., Sogge M.K., Springer A.E., Wegner D.L. 2001. Planned flooding and Colorado River riparian trade-offs downstream from Glen Canyon Dam, Arizona. Ecol Appl 11:701-710
- Stevens L.E., Schmidt J.C., Brown B.T. 1995. Flow regulation, geomorphology, and Colorado River marsh development in the Grand Canyon, Arizona. Ecological Applications 5: 1025-1039
- Science Advisory Committee. 2004. Report On Water For Environmental Flows. Prepared for Senate Bill 1639, 78th Legislature, Study Commission On Water For Environmental Flows, October 26, 2004. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/txefsac8132 008article4.pdf
- Science Advisory Committee. 2006. Recommendations of The Science Advisory Committee. Presented To Governor's Environmental Flows Advisory Committee, August 21, 2006. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/txefsac8132

http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/txefsac8132 008article5.pdf

- Science Advisory Committee. 2009a. Use of Hydrologic Data in the Development of Instream Flow Recommendations for the Environmental Flows Allocation Process and The Hydrology-Based Environmental Flow Regime (HEFR) Methodology. Report # SAC-2009-01-Rev1., April 20, 2009. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/hydro logicmethods04202009.pdf
- Science Advisory Committee. 2009b. Geographic Scope of Instream Flow Recommendations. Report # SAC-2009-02, April 3, 2009. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/geogr aphicscope.pdf
- Science Advisory Committee. 2009c. Methodologies for Establishing a Freshwater Inflow Regime for Texas Estuaries Within the Context of the Senate Bill 3 Environmental Flows Process. Report # SAC-2009-03-Rev1., June 5, 2009. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/fwi20 090605.pdf

Science Advisory Committee. 2009d. Fluvial Sediment Transport as an Overlay to Instream Flow Recommendations for the Environmental Flows Allocation Process. Report # SAC-2009-04, May 29, 2009. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/sac_2 009 04 sedtransport.pdf

Science Advisory Committee. 2009e. Essential Steps for Biological Overlays in Developing Senate Bill 3 Instream Flow Recommendations. Report # SAC-2009-05, August 31, 2009. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/biolo gyoverlay.pdf

Science Advisory Committee. 2009f. Nutrient and Water Quality Overlay for Hydrology-Based Instream Flow Recommendations. Report # SAC-2009-06, November 3, 2009. Available at

http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/wqov erlay_200911039.pdf

Science Advisory Committee. 2010a. Discussion Paper: Moving From Instream Flow Regime Matrix Development to Environmental Flow Standard Recommendations. February 17, 2010. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/sac_d iscussionpaper.pdf

- Science Advisory Committee. 2010b. Lessons Learned from Initial SB3 BBEST Activities. Report # SAC-2010-01, July 14, 2010. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/2010 0714sac_lessonslearned.pdf
- Science Advisory Committee. 2010c. Considerations in the Development of an SB3 Work Plan for Adaptive Management. Report # SAC-2010-02, August 20, 2010. Available at http://www.tceq.state.tx.us/assets/public/permitting/watersupply/water_rights/eflows/2010 0820sac_guidance_workplan.pdfSwarzenski, P.W., C.D. Reich, R.M. Spechler, J.L. Kindinger, and W.S. Moore. Using multiple geochemical tracers to characterize the hydrogeology of the submarine spring off Crescent Beach, Florida. *Chemical Geology* 179:187-202.
- SWFWMD (Southwest Florida Water Management District). 2005. District Water Management Plan, July 2005. Available online at: http://www.swfwmd.state.fl.us/about/watermanagementplan/
- SWFWMD (Southwest Florida Water Management District). 2010. Available online at: http://www.swfwmd.state.fl.us/

- Syvitski, J.P.M., C.J. Vörösmarty, A.J. Kettner, and P. Green. 2005. Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science* 308: 376-380.
- Texas Department of Water Resources. 1982. The influence of freshwater inflows upon the major bays and estuaries of the Texas Gulf coast. Texas Department of Water Resources, report LP-115, Austin, Texas. 53 p. plus appendices.
- Texas Parks and Wildlife. 1988. Trends in Texas Commercial Fishery Landings, 1977-1987. Management Data Series, No. 149. Texas Parks and Wildlife Department, Coastal Fisheries Branch. Austin, Texas.
- Thompson, H., 2006.Water Law. A Practical Approach to Resource Management and the Provision of Services. Juta & Co Ltd, Cape Town, 769 pp.
- Thompson R.W. 1968. Tidal Flat Sedimentation on the Colorado River Delta, Northwestern Gulf of California. Memoir 107. Geological Society of America.
- Tolan, J.M. 2007. El Niño-Southern Oscillation impacts translated to the watershed scale: estuarine salinity patterns along the Texas Gulf coast, 1982 to 2004. *Estuarine Coastal and Shelf Science* 72:247-260.
- Tolley, S. G., A. K. Volety, M. Savarese, L. D. Walls, C. Linardich, and E. M. Everham. 2006. Impacts of salinity and freshwater inflow on oyster reef communities in Southwest Florida. *Aquatic Living Resources* 19:371-387.
- Treaty Series 994. 1944. Utilization of waters of the Colorado and Tijuana Rivers and of the Rio Grande. US Department of the Interior, Bureau of Reclamation. Available online at: http://www.usbr.gov/lc/region/g1000/pdfiles/mextrety.pdf
- Trinity and San Jacinto Rivers and Galveston Bay Basin and Bay Expert Science Team. 2009. Environmental Flows Recommendations Report: Final Submission to the Trinity and San Jacinto Rivers and Galveston Bay Basin and Bay Area Stakeholder Committee, Environmental Flows Advisory Group, and Texas Commission on Environmental Quality. Submitted Novmber 2009.
- USACE (United States Army Corp of Engineers). 1998. Draft Environmental Impact Statement. Water Allocation for the Apalachicola-Chattahoochee-Flint River Basin: Alabama, Florida and Georgia. US Army Corps of Engineers, Mobile District. http://www.sam.usace.army.mil/pd/actacfeis/
- U.S. Census Bureau. 2010. Available online at: http://www.census.gov/
- Valdez R.A., Hoffnagle T.L., McIvor C.C., McKinney T., Leibfried W.C. 2001. Effects of a test flood on fishes of the Colorado River in Grand Canyon, Arizona. Ecol Appl 11:686-700
- Valiela, I. 1995. Marine Ecological Processes. Second Edition. Springer-Verlag, New York. 686 pp.

- Varady R.G., Hankins K.B., Kaus A., Young E., Merideth R. 2001. ... to the Sea of Cortes: nature, water, culture, and livelihood in the Lower Colorado River basin and delta - an overview of issues, policies, and approaches to environmental restoration. J Arid Environ 49:195-209
- Venn, T. J. and J. Quiggin. 2007. Accommodating Indigenous Cultural Heritage Values in Resource Assessment: Cape York Peninsula and the Murray-Darling Basin, Australia. *Ecological Economics* 61: 334-344.
- Vörösmarty, C.J., M. Meybeck, B. Fekete, K. Sharma, P. Green, and J.P.M. Syvitski. 2003. Anthropogenic sediment retention: major global impact from registered river impoundments. *Global and Planetary Change* 39: 169-190.
- Vörösmarty, C.J. and D. Sahagian. 2000. Anthropogenic disturbance of the terrestrial water cycle. *BioScience* 50: 753-765.
- Walker, D.J. and A. Jessup. 1992. Analysis of the dynamic aspects of the River Murray mouth, South Australia. *Journal of Coastal Research* 8: 71–76.
- Ward, G.H., M.J. Irlbeck, and P.A. Montagna. 2002. Experimental river diversion for marsh enhancement. *Estuaries* 25: 1416-1425.
- Water Framework Directive. 2000. Directive 2000/60/EC of the European parliament and of the council establishing a framework for Community action in the field of water policy. http://eurlex.europa.eu/LexUriServ/LexUriServ.do?uri=CONSLEG:2000L0060:20011216: EN:PDF
- Wedderburn, S.D., K.F. Walker, B.P. Zampatti. 2007. Habitat separation of *Craterocephalus* (Atherinidae) species and populations in off-channel areas of the lower River Murray, Australia. *Ecology of Freshwater Fish* 2007: 16: 442–449.
- Wilber D.H. 1992. Associations between freshwater inflows and oyster productivity in Apalachicola Bay, Florida. Est Coast Shelf Sci 35:179-190
- Woodhouse C.A., Gray S.T., and Meko D.M. 2006.Updated streamflow reconstructions for the Upper Colorado River Basin, Water Resources Research: 42, W05415. DOI: 10.1029/2005WR004455.
- Zein-Eldin, Z. P. 1963. Effect of salinity on growth of postlarval penaeid shrimp. Biol Bull 125:188-196.
- Zein-Eldin, Z. P., and D. V. Aldrich. 1965. Growth and survival of postlarval Penaeus aztecus under controlled conditions of temperature and salinity. Biol Bull 129:199-216.
- Zeng W., Jiang F., and Zhang Y. 2009. Reservoir management in the Apalachicola-Chattahoochee-Flint (ACF) river system during the interim operation plan (IOP) during the ongoing drought. Proceedings of the 2009 Georgia Water Resources Conference, April 27-29, University of Georgia.