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# A Conceptual Model of Estuarine Freshwater Inflow Management

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**ABSTRACT: As humans continue to influence the quantity, timing, and quality of freshwater input to estuaries, it is becoming increasingly common for policies to be enacted that mandate the establishment of freshwater inflow criteria that will serve to preserve and protect estuarine ecosystems. This paper reviews the scientific literature describing how changes in freshwater inflow affect estuaries, proposes a conceptual model that explores the roles of scientists, citizens, politicians, and managers in the management of freshwater inflow to estuaries, and uses the model to explore the ways in which freshwater inflow is managed in a variety of estuaries. The scientific review is organized to provide an overview of the connections between freshwater inflow (in terms of the quantity, quality, and timing of water delivery), estuarine conditions (such as salinity and concentrations of dissolved and particulate material), and estuarine resources (such as the distribution and abundance of organisms), and to highlight our understanding of the causative mechanisms that underlie the relationships among these variables. The premise of the conceptual model is that the goal of estuarine freshwater inflow policy is to protect those resources and functions that we as a society value in estuaries, and that management measures use scientific information about the relationships among inflow, conditions, and resources to establish inflow standards that can meet this goal. The management approach can be inflow-based (flow is kept within some prescribed bounds under the assumption that taking too much away is bad for the resources), condition-based (inflow standards are set in order to maintain specified conditions in the estuary), or resource-based (inflow standards are set based on the requirements of specific resources), but each of these is carried out by regulating inflow. This model is used as a framework to describe the development of freshwater inflow criteria for estuaries in Texas, Florida, and California.**

''Water may flow in a thousand channels, but it all returns to the sea.''—African proverb

## Introduction

There are very few estuarine systems in the world unaffected by upstream manipulation of their freshwater inflow. Approximately 60% of the global storage of freshwater is behind registered dams (Vörösmarty and Sahagian 2000), and Dynesius and Nilsson (1994) concluded that 77% of the total water discharged by the 139 largest river systems in the northern third of the world are strongly or moderately affected by dams, interbasin transfers, and surface water withdrawals. Demand for freshwater is only expected to increase as world population continues to grow (Postel 1998). In light of these pressures, the evaluation of various flow regimes for sustainable river management and the analysis of the environmental effects of hydrologic alteration are both areas of active investigation (e.g., Sparks 1992; Poff et al. 1997), but it is also important to examine the consequences of freshwater flow regulation for coastal ecosystems.

Decreases in freshwater inflow can have far reaching, sometimes disastrous, consequences downstream. The Aswan High Dam in Egypt led to large changes in the discharge of Nile River flood water; after construction of the dam there was a considerable reduction in overall discharge, a decrease in peak flows, an increase in low flows, and a shift in the timing of the hydrograph (Vörösmarty and Sahagian 2000). Impoundment of water has led to a substantial decrease in the loading of nutrients to the Mediterranean Sea and the sediment load is now virtually nonexistent (Hallim 1991). These changes in inflow have had serious impacts on marine life, resulting in a 95% decrease in phytoplankton and an 80% decrease in fish catch: *Sardinella* catch dropped from 15,000 tons in 1964 (pre-dam) to 554 tons in 1966 (post-dam; Aleem 1972; Hallim 1991). In the Seekoei estuary in South Africa, a drought in 1988–1989 coupled with high upstream withdrawal rates resulted in no freshwater inflow to the estuary at all. Salinities in the upper portion of the estuary reached 98 psu, resulting in massive fish mortality (Whitfield and Bruton 1989). These are extreme cases, but they point out the importance of establishing the freshwater requirements of estuaries in comparison with competing needs.

For years, estuarine ecologists have been bemoaning the lack of attention paid to this issue. In 1966 B. J. Copeland published a paper titled ''Effects of decreased river flow on estuarine ecology'' that ended with the statement: ''As has been shown in the previous discussion, freshwater input to estuaries is an important factor. Without it, estuaries

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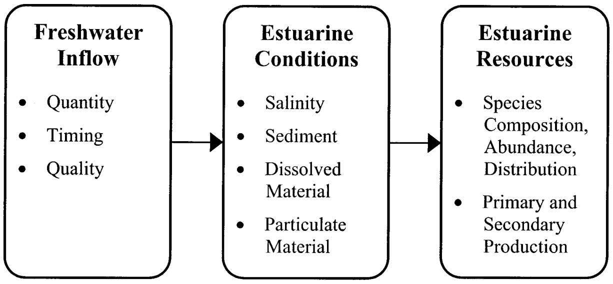


Fig. 1. Schematic diagram of the effects of freshwater inflow on estuaries.

become hypersaline and species composition can be altered drastically. With continuation of man's activities in allowing less and less fresh water downstream to the estuary, man may have to pave the estuarine areas and sell them for real estate.'' (p. 1837). A year later H. Dickson Hoese (1967, p. 259) wrote: ''This paper will end in a plea, which is already partly evidenced and answered. The pressure of rising salinities [due to decreased freshwater inflow] will increase, as first Texas and then the northeast have experienced, and while marshes may withstand the change better than most estuarine waters, all interests should make certain of this. After all estuaries are only downstream from the whole nation.''

The purpose of this paper is to propose a conceptual model that explores the roles of scientists, citizens, politicians, and managers in the management of freshwater inflow to estuaries. As is shown throughout this special issue, considerable progress has been made in understanding the consequences of changing inflow patterns to estuaries and the mechanisms that underlie these relationships. Applying this information in a management context requires establishing inflow standards that can meet the policy goal of protecting estuarine resources that are valued by society. This paper describes the scientific framework for evaluating inflow effects and provides an overview of the impacts of decreases in freshwater inflow to estuarine ecosystems. It then describes the types of societal values that are reflected in inflow policy and focuses on the management approaches that can be used to protect estuarine resources, and uses this as the context for presenting the development of estuarine freshwater inflow criteria in Texas, Florida, and California.

## Scientific Framework

Figure 1 provides a simple overview of the scientific framework for evaluating estuarine inflow, which involves determining the linkages between freshwater inflow, estuarine conditions, and resources. Basic research on this issue is geared towards assessing how changes in freshwater inflow affect estuarine conditions, and how these changes in turn affect different components of the ecosystem. This framework is in keeping with the model developed by Sklar and Browder (1998), and my compartments for inflow, conditions, and resources are roughly parallel to their compartments for freshwater discharge, chemistry, and plants and animals, respectively.

Although the model begins with freshwater inflow to the estuary, it is important to recognize that the quantity, timing, and quality of these inputs are all determined by events that occur upstream. Dams, diversions, and upstream withdrawals directly affect the amount of water that reaches the coast, and to the extent that these are consumptive uses there is reduced inflow to the estuary. In extreme cases no water reaches the sea. The timing of water delivery is also subject to upstream modification. Where dams are managed for flood control they tend to dampen the magnitude of flooding and can also result in reduced variations in inflow and modulation of seasonality. In the San Francisco Bay Estuary, reservoirs capture much of the spring snow melt and store it for later use in the summer when water demand for agriculture and power requirements are highest, effectively truncating the normal spring peak in the hydrograph (Kimmerer and Schubel 1994). The timing and location of freshwater delivery can also be affected by shifts in land use, such as conversion of land from forest to urban, that result in changes in runoff patterns. Channelization and the isolation of rivers from riparian buffers can also affect the timing of freshwater inflow to estuaries; in the Everglades the construction of control structures has completely changed the pattern of freshwater inflow to Florida Bay (McIvor et al. 1994).

To the extent that nutrients, pollutants, sediment, and organic material are all carried along with freshwater, any upstream changes in inflow will affect the amount and timing of their delivery to the estuary as well. Ustach et al. (1986) documented how clearing and draining land for agriculture resulted in a 10% increase in freshwater flow to a portion of the Neuse River estuary and a consequent increase in nutrients and turbidity. Estuarine concentrations of nutrients, organic matter, pollutants, and sediments have all been correlated with inflow (e.g., Jordan et al. 1991; Mallin et al. 1993; Jassby et al. 1995), and there are numerous examples of how year-to-year changes in river inflow influence the loading of materials to estuaries (e.g., Boynton et al. 1995).

Dams can also affect the water quality characteristics of estuarine inflow. Dams tend to trap sediment and decrease the downstream delivery of particles and associated materials such as particle-active metals and other pollutants. The presence of upstream dams on the Danube River reduced the load of silt and associated silica to the Black Sea (Ittekkot et al. 2000). Silica concentrations decreased from 140 $\mu$mol $L^{-1}$(pre-dam) to 58 $\mu$mol $L^{-1}$ (post-dam), with a concurrent change in the Si : N ratio from 42 to 2.8. The amount of time that water spends behind dams can result in substantial delays in its delivery to the estuary (Vörösmarty and Sahagian 2000), with consequent impacts on the quality and availability of organic matter. Townsend et al. (1996) provided evidence for increased photodegradation of dissolved organic material in reservoirs with longer residence times, and Mousset et al. (1997) measured a higher proportion of humic material in reservoir as compared to river water.

It is important to note that loading is the product of inflow and concentration. Although the above discussion focused on changes in inflow, changes in upstream water quality will clearly affect the delivery of materials to an estuary, regardless of flow conditions. A discussion of water quality change is outside the scope of this review, but both point and non-point source discharges can affect downstream water quality. Changes in upstream land use such as deforestation can lead to changes in both nutrient and sediment concentrations (Sklar and Browder 1998), and many coastal systems are showing symptoms of eutrophication as a consequence of increased nutrient concentrations (Rabalais et al. 1996; Howarth 1998). These types of water quality changes, when coupled to changes in discharge, can result in greatly altered patterns of loading to an estuary.

Changing the amount of freshwater input by any of the perturbations described above will have profound effects on estuarine conditions. One of the most obvious consequences of decreased freshwater input is that salt water may intrude farther upstream, resulting in increased salinity along the estuarine gradient. In extreme cases of high evaporation coupled with low rainfall, the estuary can become hypersaline. The Kariega estuary in South Africa had no rainfall for more than a year, resulting in the complete absence of river inflow, and salinities in the upper reaches were greater than 40 psu (Whitfield and Woolridge 1994). In addition to an upstream shift in salinity, decreased outflow can also lead to expansion of the zone of transition from zero salinity to full seawater, hence lengthening the estuary. This can be seen by comparing the upstream extent of the estuarine zone in rivers with high versus low flow. Although the mouths of the Altamaha and Satilla River estuaries in Georgia are located only 37 km apart and experience similar tidal regimes, median flow in the Satilla is 10 times lower than that in the Altamaha (25 versus 250 $m^3$ $s^{-1}$). As a consequence, one encounters freshwater only 20 km upstream in the Altamaha as compared to 50 km upstream in the Satilla (Smith 2001).

Alterations in freshwater inflow can also change the hydrodynamic regime of an estuary. Decreases in discharge will serve to increase the influence of the tide on circulation patterns, such that a stratified system with well-developed gravitational circulation can shift to a well-mixed system where tidal exchange increases in importance. Ingram et al. (1985) reported that diversion of the Eastmain River, Quebec, led to a 90% decrease in mean flow and a significant increase in tidal amplitude in the estuary. In the San Francisco Bay Estuary, Cloern (1984) found that the ratio of river discharge to tidal current speed could be used to explain stratification. At high river flows, South San Francisco Bay stratifies, turbidity and nutrient concentrations decline, phytoplankton biomass and production are high, and residual currents accelerate. A change in stratification as the result of changes in inflow can in turn affect bottom water hypoxia, as has been observed in Chesapeake Bay (Malone et al. 1988). Another hydrodynamic effect that can result from high freshwater discharge is the creation of fronts along the longitudinal axis of the estuary, which can serve as places for the accumulation of particles and surface active material (Bowman and Iverson 1978). To the extent that circulation patterns interact with local topography to create entrapment zones, changes in inflow can displace or eliminate the location of the estuarine turbidity maximum (ETM). In the Eastmain River, described above, the shift in circulation patterns also led to the generation of a turbidity maximum in the estuary (Ingram et al. 1985). In the San Francisco Bay Estuary, when discharge is in the appropriate range it positions the ETM adjacent to shallow bays and diatom biomass increases in that area (Cloern et al. 1983).

Another consequence of decreased freshwater inflow is that it results in an increase in flushing or freshwater transit time (Alber and Sheldon 1999; Sheldon and Alber 2002). The transit time provides a measure of the time it takes river water to transit through the system and has consequences for the ability of an estuary to flush out materials. As transit times increase, the concentrations of pollutants and pathogens can increase as well. The transit time also sets the time frame for conservative mixing and can thus be compared against the time scales of biogeochemical and other nonconservative processes to determine whether transformations may occur within estuaries. Freshwater transit time has been positively correlated with the fractional export of nitrogen from estuaries and negatively correlated with the amount of denitrification (Nixon et al. 1996; Dettmann 2001).

Changes in inflow can also lead to alterations in estuarine geomorphology. Because freshwater is generally also a source of sediment to an estuary, decreased inflow can result in losses for tidal deltas, benthic communities, and intertidal habitat (e.g., Boesch et al. 1994). These effects can be exacerbated due to the presence of upstream dams that trap sediment. The operation of dams can also have the opposite effect by reducing sediment scouring. To the extent that dams or other upstream perturbations artificially decrease flood discharge or increase the interval between flooding events, the erosive capacity of river floods will diminish. Under these conditions, the estuary tends to shrink its channel dimensions due to sediment accumulation. This has been well documented by Reddering (1988), who describes how changes in inflow can affect the depth profile, the configuration of the mouth, and the tidal prism of an estuary. At the extreme, the reduced scouring can lead to the closure of the mouth of a tidal inlet.

Changes in either the timing or quantity of freshwater input can have important effects not only on the delivery of dissolved and particulate material, but also on their concentrations in the estuary itself. Drinkwater and Frank (1994) summarized data from the receiving waters of six rivers that had significant freshwater flow regulation. In every case, decreased inflow was coupled to changes in nutrient and sediment concentrations. These relationships are generally positive, such that increased inflow brings in more material. Grange et al. (2000) measured a 20-fold increase in the nutrient concentration of the Kariega Estuary in the wet as compared to the dry season. In cases where inflow is not the main source of materials, the opposite relationship has been observed. In the estuary of the Fraser River in Canada decreased inflow led to decreased stratification and increased mixing of benthic nutrients (Beamish et al. 1994). Although these relationships are complicated, the point remains that inflow can have profound effects on water quality.

The final portion of the scientific framework is the connection to estuarine resources (Fig. 1). Estuarine ecology is almost by definition a study of the linkages between estuarine conditions and the distribution and abundances of estuarine biota and the resultant implications for such things as community structure, food web interactions, rates of primary and secondary production, and material cycling. Rather than provide an exhaustive review of this topic, my purpose here is to highlight those ecosystem-level changes that result directly from changes in freshwater inflow.

Salinity is a critical determinant of the habitat characteristics of an estuary, and shifting isohalines caused by variations in freshwater inflow can affect the distribution of both rooted vegetation and sessile organisms. Upstream movement of *Spartina* species in both the Delaware River and Chesapeake Bay has been linked to long-term increases in salinity (Schuyler et al. 1993; Perry and Hershner 1999), and we have also documented large differences in the distribution of marsh vegetation along two Georgia estuaries with different river flows (Smith 2001). This has implications in the context of the overlap concept described by Sklar and Browder (1998), who pointed out that the changing spatial distribution of appropriate habitat is important to consider when evaluating changes in inflow. As a given isohaline moves upstream, the channel width and the extent of intertidal habitat are often different, with consequent effects on the suitability of the new location for benthic organisms.

Changes in salinity structure affect the distribution of motile organisms as well. Most of the biota found in estuarine environments occurs within focused salinity ranges, and different stages in the life histories of many estuarine organisms have specific salinity requirements. Bulger et al. (1993) found nonrandom discontinuities in the distributions of fish along the estuarine gradients in Chesapeake and Delaware Bays. In their review of the impact of flow regulation, Drinkwater and Frank (1994) found changes in the species composition, distribution, abundance, and health of fish and invertebrates attributable to changes in freshwater flow. They also linked changes in river flow to changes in migration patterns, spawning habitat, and fish recruitment. Whitfield (1994) identified the longitudinal salinity gradient as the single most important factor linked to successful recruitment of larval and juvenile marine fish in South African estuaries and has gone on to develop a fish recruitment index that relates estuarine fish to inflow (Quinn et al. 1999). Abundances of anadromous fish such as striped bass and salmon have also been correlated with inflow (e.g., Stevens and Miller 1983; Rulifson and Manooch 1990).

Changes in the timing of water delivery can also affect estuarine resources. The life histories of many fish and shellfish are cued to high spring runoff, such that changes in timing can affect spawning and nursery cycles. Sutcliffe (1973) found a positive correlation between spring runoff in the St. Lawrence River and lobster landings in the Gulf of St. Lawrence 9 years later. In Sabine Lake, Texas, the presence of a dam shifted peak flows from spring to summer, reducing the availability of both low salinity nursery habitat for brown shrimp in the spring and high salinity nursery habitat for white shrimp in the summer (White and Perret 1974, referenced in Sklar and Browder 1998). In a study of the impact of salinity variability on estuarine organisms, Montague and Ley (1993) found a negative correlation between the standard deviation of salinity and the density of plants and benthic animals and suggested that frequent salinity fluctuations result in increased physiological stress. On the other hand, Flint (1985) found that episodic freshwater input stimulated production of both benthic infauna and shrimp in Corpus Christi Bay. These conflicting reports suggest that organisms have a complex response to inflow variability, and it is likely that the interaction of salinity and other dynamic characteristics determine habitat suitability in a given area.

Discharge-associated changes in the delivery of nutrients, organic matter, and sediment have implications for estuarine productivity rates and trophic structure. The relationship between nutrients and inflow is generally positive, and many investigators have found a correlation between nitrogen loading and phytoplankton production (Flint et al. 1986; Nixon 1992; Mallin et al. 1993; Boynton et al. 1995). The converse is also true: decreased inflow can often be linked to decreased rates of both primary and secondary production (Drinkwater and Frank 1994). Increased inflow also generally brings increased sediments, which can affect the light environment of the estuary via turbidity effects and result in reduced phytoplankton production. A drought in the San Francisco Bay Estuary was linked to increased water clarity and high chlorophyll concentrations (Lehman 1992), and in the Hudson River estuary, Howarth et al. (2000) also found a negative relationship between inflow and primary production, due in part to the fact that high discharge rates serve to both decrease light penetration and reduce flushing times, resulting in less opportunity for phytoplankton to grow within the estuary. In two South African estuaries with different riverine inflow, decreased inflow resulted in better light penetration and a concurrent increase in the importance of aquatic macrophytes, which resulted in a switch from a pelagic to a benthic food web and a change in the balance between detritivory and herbivory (Whitfield and Wooldridge 1994; Grange et al. 2000).

Organic matter input is also important for food web dynamics. In the Mbashi estuary in South Africa the presence of an upstream dam led to a reduction in the input of silt and organic detritus, which was correlated with a decrease in fish abundance (Plumstead 1990). The decline in fish was thought to be the result of decreased organic material as a food resource both for the fish themselves and for their prey. We also know from stable isotope evidence that terrestrially-derived organic matter is used in estuarine food webs, particularly in upstream reaches (e.g., Day et al. 1994; Riera and Richard 1996). In a comparative study of two Maine estuaries, Incze et al. (1982) showed that bivalves had increased dependence on terrestrially-derived material in an estuary with a high river discharge compared with one with little river input.

A good example of the propagation of changes in inflow through an ecosystem was observed in Appalachicola River estuary in Florida, where a two-year drought led to an approximately 50% reduction in river flow (Livingston et al. 1997). This resulted in an initial increase in primary production (due to reduced turbidity), followed by a long-term decrease in production, which they postulated was due to decreased delivery of nutrients to the estuary. There were also dramatic effects on trophic structure: overall trophic diversity decreased and there were increases in some groups (herbivores, detritivorous omnivores, primary and secondary carnivores) and decreases in others (tertiary predators were virtually absent). These responses were seen as changes in trophic structure (i.e., as an emergent property of the community) and not in terms of individual species (Livingston 1997). The effects of the drought took several years to make their way through the food web of the estuary (Livingston et al. 1997).

Considering the interplay of factors described above, it should come as no surprise that the relationship between inflow and secondary production is difficult to predict. In many systems, an increase in inflow results in increased catch of fish (Sutcliffe et al. 1983; Skreslet 1986) and shellfish (Browder 1985; Gracia 1991; Gammelsrød 1992; Galindo-Bect et al. 2000). The mechanisms that underlie these relationships are not always understood, but increased secondary production is generally attributed to increased nutrient inflow resulting in increased primary production. Indeed, Solis and Powell (1999) found a positive relationship between nitrogen loading and fisheries harvest for five Texas estuaries. Gammelsrød (1992) suggested three additional mechanisms for the relationship between increased freshwater inflow in the Zambezi River and shrimp catch: increased inflow leads to greater flooding by brackish water resulting in an increase in the area of habitat suitable for successful recruitment, increased inflow leads to greater dispersion of larvae, and increased inflow results in increased estuarine turbidity, which provides protection from predators. Inverse rela-

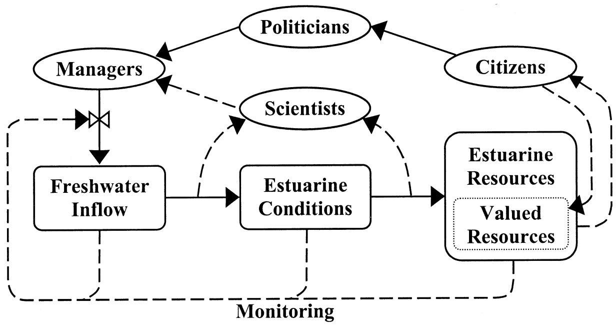


Fig. 2. Model of estuarine inflow management depicting the primary roles of citizens, politicians, scientists, and managers. Solid lines denote direct control; dashed lines denote information transfer; the gate on the arrow between managers and inflow signifies that managers can modify inflow based on the information they receive.

TABLE 1. Valued resources. Examples of the types of estuarine resources considered valuable by different stakeholders.

| |
|---|
| Intrinsic value |
| Navigation |
| Assimilative capacity |
| Fish and shellfish production |
| Wildlife habitat |
| Aesthetic, recreational value |
| Intertidal wetlands |
| Rare and endangered species |
| Essential fish habitat |

tionships between runoff and the catch of fish (Sutcliffe et al. 1983; Beamish et al. 1994), shellfish (Turner 1992), and other organisms (Ardisson and Bourget 1997) have also been observed. The mechanisms are not always understood, but they likely involve a decrease in the availability of suitable nursery habitat or a negative relationship between inflow and nutrients that results in lowered rates of primary production. Estuarine organisms may also be stressed by conditions of lowered salinity during periods of high freshwater inflow.

Freshwater inflow effects are far-reaching, and changes in inflow can result in changes to the biological, chemical, and physical attributes of an estuary. Estuarine scientists seek to identify relationships between inflow, conditions, and resources, and then to understand the causative mechanisms that underlie these relationships. It is this type of information that serves as the basis of scientific input to estuarine inflow management.

## Management Framework

A simplified conceptual model for estuarine inflow management is shown in Fig. 2; links not represented in the model include interactions among different groups (e.g., scientists and citizens; scientists and politicians; managers and politicians, etc.) and links via education. What is emphasized here are the primary connections among the various groups and how they relate to management. My premise is that the goal of estuarine freshwater inflow policy is to protect those resources and functions that we as a society value in estuaries, and that management measures are geared toward establishing inflow standards that can meet this goal.

I begin my discussion of the model with the compartment labeled Valued Resources, because a perceived threat to these resources is often the impetus for the development of a freshwater inflow policy. Valued resources are depicted as a subset of the estuarine resources box, as these are the resources and functions that people care about in estuaries. This does not usually include all of the natural resources of an estuary, and what is in this box will not be the same for each stakeholder. Examples of the types of resources that are often identified as valuable in estuaries are listed in Table 1. Intrinsic value is listed in recognition of the fact that some groups (e.g., many environmental organizations) consider estuaries intrinsically valuable and they are willing to see decisions made on this basis alone. Other groups value estuaries for their commercially important fisheries or the presence of wildlife habitat. Some of the terms listed in Table 1 connote legal value. Through passage of the Endangered Species Act society has asserted that the continued presence of rare and endangered species is valuable. The Magnuson-Stevens Act (although as yet untested) assigns value to essential fish habitat.

The reason for the emphasis on societal values is to focus attention on the role that citizens play in setting inflow policy. Policy making is by definition a political endeavor, and elected officials will respond to pressure from citizens to establish inflow policies that protect those resources that they care about. As long as there is no perceived threat to valued resources, it is difficult to muster the political will to pass legislation or to enforce estuarine inflow requirements. Even with a perceived threat, the effort is unlikely to be successful unless there are stakeholders who are concerned enough to attend public hearings or write to their representatives. In practice, the broader the base of support (e.g., the combined strength of the wildlife lobby, commercial fishermen, and the presence of endangered species), the more likely a requirement will prevail. Note that there are two arrows in Fig. 2 connecting citizens and valued resources. This is to denote that on one hand citizens act to determine which items fall into the valued resources category, and on the other they keep track of the status of these resources and can in turn exert an

TABLE 2. Upstream regulations. Examples of the types of policies and management decisions that affect freshwater inflow to estuaries.

| |
|---|
| Withdrawal permitting |
| Discharge permitting |
| Instream flow requirements |
| Reservoir management |
| Diversions and interbasin transfers |
| Flood plain modification |
| Sediment and erosion controls |
| Water quality standards |

influence on policymakers in the face of a perceived threat.

There is some overlap between my usage of valued resources and the Valued Ecosystem Component approach used by the Environmental Protection Agency (EPA) in the National Estuary Program (EPA 1995), which itself was an adaptation of the Valued Environmental Component (VEC) approach of Clark (1986). Clark (p. 17) defined VECs as ''attributes of the environment that some party to the assessment believes to be important'' and noted that ''Which components are valued will depend on specific social, political, and environmental circumstances.'' He sought to develop causal relationships between VECs and potential sources of environmental change, and devised a matrix to evaluate the importance of the various sources of disturbance on the identified VECs. As applied in the National Estuaries Program, VECs became Valued Ecosystem Components, which have been variously defined as ''1. A resource or environmental feature that is important (not only economically) to a local human population, or has national or international profile, or if altered from its existing status, will be important for the evaluation of environmental impacts of development and the focusing of administrative efforts'' and ''2. Any part of the environment that is considered important by the proponent, public, scientists and government involved in the assessment process. Importance may be determined on the basis of scientific concern or based on cultural values'' (SFWMD 2001, p. 50). These definitions are in keeping with valued resources as I described them above. VECs can develop, however, into lists of ecosystem components (e.g., microheterotrophs, phytoplankton, and soft-bottom benthos were all listed as VECs in the Southern California Bight; National Research Council 1990), and are perhaps more appropriate as scientific rather than societal considerations.

In order to meet policy goals and protect valued resources, it is important to recognize that freshwater inflow is the primary point where humans exert control over an estuary. The implications of this statement for inflow management are that any regulations that affect upstream flow will affect estuarine inflow, and any management actions that are put in place to protect estuaries will be focused on inflow regulation. The factors listed in Table 2 highlight the components of upstream management that influence the quantity, quality, and timing of freshwater inflow to estuaries. The connection between upstream policies and the downstream delivery of freshwater, although straightforward, is not generally made explicit. Except in situations where there is a dedicated effort to manage estuarine inflow, such as in the case studies discussed below, it is rare for decisions regarding upstream resources to be made in light of potential estuarine effects, and there is little recognition that upstream regulation is, by default, setting estuarine inflow. Although it is primarily on the inflow side that management practices can most influence estuarine conditions, the authority to make decisions regarding such things as permits for water withdrawal or point source discharges is given to agencies with jurisdiction over freshwater resources, which are generally independent of those agencies responsible for coastal resource protection. Note that there are other regulations that do directly affect estuarine conditions and resources (e.g., regulations affecting dredging, dock construction, fish catch, etc.), but these are not considered here because they do not usually influence freshwater inflow (although wastewater discharge can be an important source of freshwater to an estuary).

Management plans that are instituted with the goal of protecting estuarine resources are carried out by regulating inflow, and this is depicted as an arrow in Fig. 2 that runs directly from managers to inflow. Inflow then influences estuarine conditions, which in turn influence resources, as described in the scientific framework (Fig. 1). Management can be focused on different boxes in the scientific framework, and I therefore distinguish between an inflow-based, condition-based, or resource-based approach to estuarine inflow management. In an inflow-based approach, flow is kept within some prescribed bounds under the assumption that taking too much away is bad for the resources. A condition-based approach is one in which inflow standards are set in order to maintain a specified condition (e.g., salinity) at a given point in the estuary. In a resource-based approach, inflow standards are set based on the requirements of specific resources. Regardless of which approach is used, connections are usually made, either directly or indirectly, between inflow and valuable estuarine resources.

The model has numerous routes for information collection that can feed back to managers, causing a potential modification of inflow regulation. Monitoring of either inflow itself, estuarine conditions, or estuarine resources will provide direct feedback, allowing managers to determine if the decisions being made are actually effective in terms of meeting management goals. A minimum inflow level might be chosen that is geared towards maintaining the average high tide salinity below a certain threshold at a specific point in an estuary, and salinity data collected at that point could be used to determine if the target is being met. If average salinity is higher than expected, minimum inflow levels can then be modified accordingly. This is an example of what Johnson (1999) characterized as the ''monitor-and-modify'' approach to management.

Scientists are linked to the arrows between the boxes in Fig. 2 to indicate that the scientist (whether in academia or in an agency) studies the linkages between inflow, estuarine conditions, and resources. The connection between scientists and managers is drawn as a one-way arrow to underscore the point that understanding the ways in which inflow affects estuarine conditions and resources is critical for the establishment of scientifically-defensible inflow management. As the ones that have to make decisions about upstream flow (e.g., whether to grant permits for increased water withdrawals in a system), managers are on the front lines of the issue and it is important that they have timely access to scientific results.

Another potential series of connections that are not depicted in the model fall under the heading of adaptive management. Adaptive management represents a more extensive and integrated approach to management that has the potential to join managers, scientists, citizens, and politicians in an effort to manage natural resources. This idea has several components, including the development of alternate conceptual models for how a system works and therefore could be managed; the performance of large-scale manipulations in the field, replicated if possible, posed as scientific hypotheses and designed to understand the different outcomes that result from different management strategies; and both the ability and the flexibility to learn from these management experiments and incorporate the results into new management policies (Holling 1978; Walters and Holling 1990; Johnson 1999). Although there are some examples of the use of adaptive management in coastal and riparian systems (Walters 1997; Gilmour et al. 1999), there are also obstacles to its success (Walters 1997; Gray 2000). There have been calls for the use of adaptive management in establishing instream flow standards (Castleberry et al. 1996; Van Winkle et al. 1997), and some aspects of adaptive management are being applied in the San Francisco Bay Estuary (see case study, below). If this type of approach becomes more widely used in estuarine inflow management, it would then be appropriate to draw two-way arrows connecting scientists, citizens, managers, and politicians in the conceptual model (Fig. 2).

## Case Studies

The conceptual model can be used to examine how inflow is managed in different estuaries, including many of the systems detailed later in this volume. Although the strength of the connections may vary from place to place, my contention is that once a problem is recognized, politicians direct managers to protect estuarine resources, and they respond by establishing freshwater inflow guidelines geared toward meeting this goal. The experiences of Texas, Florida, and California support these generalizations and provide examples of resource-based, inflow-based, and condition-based approaches to estuarine inflow management.

### Texas

1948 marked the beginning of one of the worst droughts in Texas history. The drought lasted for nearly 10 years, and by 1956 the combined river discharge to the estuaries of the state was 86% below average (Longley 1994). There were declines in the harvest of oysters, white shrimp, and blue crabs (Copeland 1966); invasion of the bays by stenohaline marine organisms (Hoese 1960); and negative effects on fish such as black drum (Longley 1994). In 1957 the Texas Water Planning Act was passed, and it contained a legislative directive to give consideration to the effects of upstream development on coastal waters. This initial plan, formally adopted in 1969, included the establishment of a cooperative Bays and Estuaries Program by the Texas Water Development Board. In 1975, Texas Senate Bill 137 was passed which required comprehensive studies of the effects of freshwater inflows on the bays and estuaries. These early studies brought out the need for further information to support water management, and bills passed in 1985 and 1987 directed the Texas Water Development Board and the Texas Parks and Wildlife Department to conduct studies to determine the bay conditions necessary to support a sound ecological environment (Texas Water Code Ann. § 16.058 [2000]). The results of this effort, along with a description of an analytical methodology for developing estuarine freshwater inflow requirements, were written up in a report jointly produced by the two departments (Longley 1994).

The 1985 bill also assigned responsibility for water rights permitting to the Texas Natural Resource Conservation Commission (now the Texas Commission on Environmental Quality) and gave the Texas Parks and Wildlife Department the authority to be a party to hearings on applications for permits to change the pattern or quantity of freshwater inflow. The legislation mandated that ''For permits issued within an area that is 200 river miles of the coast . . . the commission shall include in the permit . . . those considerations necessary to maintain beneficial inflows to any affected bay or estuary'' (Texas Water Code § 11.147(b) [2002]). Beneficial inflows were defined as ''a salinity, nutrient, and sediment loading regime adequate to maintain an ecologically sound environment in the receiving bay and estuary system that is necessary for the maintenance of productivity of economically important and ecologically characteristic sport or commercial fish and shellfish species and estuarine life upon which such fish and shellfish are dependent'' (Texas Water Code § 11.147(a) [2002]). The legislation specifically recognized inflow effects (salinity, nutrient, and sediment loading), which in turn affect estuarine resources (sport or commercial fish and shellfish species and the life upon which they depend). Note that the identified resources are ones that are generally considered valuable by society.

The management approach used in Texas, which has evolved over the past 50 years, is an example of a resource-based approach, in which freshwater inflow is linked directly to valued resources. As described in Longley (1994), the legislative language was used to guide the development of the Texas Estuarine Mathematical Programming (TxEMP) model. The model, which is now in use as a management tool in Texas (Powell and Matsumoto 1994; Powell et al. 2002), uses a series of relationships between historic monthly inflow and the catch of various fish (black drum, red drum, sea trout), crustaceans (blue crab, white shrimp, brown shrimp), and mollusks (clams, eastern oyster; Matsumoto et al. 1994). The salinity ranges of each organism are considered, and as information on nutrients and sediments becomes available it can be added as well. The model itself is a nonlinear, stochastic, multi-objective optimization model of salinity-inflow and inflow-fishery harvest equations. Although it was developed based on historic catch records, there is a recognition that the model could be improved with fishery-independent data (Longley 1994).

Running the TxEMP model requires input from managers in terms of which species are included, the relative weighting of the species, fishery harvest targets, and constraints on inflow, salinity, nutrient loading, and sediment loading (Powell and Matsumoto 1994). Model results are in the form of a performance curve, which is a series of solutions that seeks to optimize inflow-harvest relationships. Variability in the inflow-salinity relationship is used to set statistical bounds on salinity. The inflow relationships necessary to run the model have now been developed for each of the 7 major Texas bays and estuaries (TWDB 2002). A full treatment of the model, including a description of the extensive monitoring and verification steps that have been developed to support the program, can be found elsewhere (Longley 1994; Powell et al. 2002).

One of the advantages of the Texas approach is that it is keyed to commercially-important fisheries and is easily understood by a range of constituents. It is also straightforward in that it works directly with both inflow and resources, rather than depending on relationships among different compartments. Although these correlations do not get at mechanisms, a direct link offers firm ground for establishing inflow requirements. A disadvantage of this approach is that decisions based on a limited number of species and their habitat requirements can invite solutions that protect the specified resource without regard for the rest of the ecosystem. Conversely, what is good for the ecosystem may not consistently benefit individual species (Sparks 1992). Although it can be argued that the Texas model avoids this in that it simultaneously optimizes the harvest of several species, the focus on commercial and recreational catch may still overlook other resources with different inflow requirements.

### Florida

Florida is divided into 5 Water Management Districts, which were established by the Water Resources Act of 1972 (1972 Fla. Laws ch 72-299). The genesis of this legislation is tied to the story of the management of water flow in south Florida. The development of drainage channels and the construction of levees as part of the Central and Southern Florida Flood Control Project, together with the development of water conservation areas north of Everglades National Park, resulted in large changes in flow patterns and substantial decreases in total freshwater flow to the southern Everglades (Light et al. 1995). In the early 1960s the park received little or no water, and between 1963 and 1965 rookeries of wood stork failed for 3 successive years (Farb 1965). The plight of the water-dependent animals in the Everglades (e.g., otters, alligators, wading birds) received national attention, and in 1970 the U.S. Congress passed an act that established minimum flows to the park (Light et al. 1995). A severe drought also began in 1970, which resulted in water shortages in both central and southeastern Florida and underscored the problem that rapid growth was bringing in terms of competing water demands. The Governor convened a conference of resource managers, policy makers, and stakeholders in 1971 to address the water crisis in south Florida (Blake 1980). The report of the Governor's conference resulted in four pieces of environmental legislation, one of which was the Water Resources Act, which established the regional water districts throughout the state and gave them control of surface water management and water allocation.

The Water Resources Act directly addressed the issue of freshwater inflow by requiring that the Water Management Districts establish minimum flows and levels for surface waters and aquifers within their jurisdiction (Florida Statute Ann. § 373.042 [2000]). The minimum flow is defined as ''the limit at which further withdrawals would be significantly harmful to the water resources or ecology of the area.'' Steps in the development of minimum flows and levels (MFLs) include identifying water resource functions, defining significant harm, and providing standards to protect these functions against significant harm. Water resource functions protected under Chapter 373 are broad, and include flood control, water quality protection, water supply and storage, fish and wildlife protection, navigation, and recreation. Although this language was in place in 1972, a report written for the Florida Legislature in 1995 concluded that none of the water management districts had fully established minimum flows and levels (OPPAGA 1995). The Act was amended in 1997 to require that the governing board of each of the five Water Management Districts develop a priority list and schedule for the establishment of MFLs for water within their jurisdiction. Once MFLs are established, they are to be periodically reviewed (at least every five years) and updated as new information becomes available (an example of the monitoring-feedback loop in Fig. 2). The legislation also stated that proposed MFLs should be based on the best available scientific information, and could be subject to independent peer review when deemed necessary by the District or when requested by affected parties.

The water management districts are taking a variety of approaches to comply with the MFL directive, some of which are detailed later in this volume (Doering et al. 2002; Flannery et al. 2002; Mattson 2002). Here I use the strategy of the Southwest Florida Water Management District to highlight an inflow-based approach to management, and that of the South Florida Water Management District to further explore the resource-based approach.

The Southwest Florida Water Management District (SWFWMD) provides a good example of inflow-based management. In the late 1980s they established what was called the 10% presumption in reviewing applications for water withdrawal permits, which stated that: ''The District presumes that the withdrawal of water will not cause unacceptable environmental impacts if the withdrawal, combined with other withdrawals, does not reduce the rate of daily flow by more than 10% at any point in the drainage system at the time of withdrawal'' (Section 4.2.C.2 of the Basis of Review, SWFWMD 1998). The District had conducted studies to demonstrate that reducing inflow by 10% or less had a minimal impact on estuarine conditions and resources (see Flannery et al. 2002). This is an interesting approach in that it links withdrawal to daily flow, thereby preserving natural streamflow variations. The use of the 10% presumption was successfully challenged in an administrative hearing in 1995, in part because it was considered arbitrary. Although the 10% presumption is no longer in effect, the District still limits withdrawals to a percentage of streamflow. A full account of the development of the percent of flow approach by the SWFWMD, along with a review of supporting studies that relate inflow to estuarine resources, can be found in Flannery et al. (2002).

Inflow-based management is very much in keeping with the approach that is often advocated for rivers, where flow is considered a master variable because it is correlated with many other factors in the ecosystem (Poff et al. 1997; Richter et al. 1997). In the case of the SWFWMD, the emphasis is on maintaining the natural flow regime with the premise that maintaining inflow will also maintain complex estuarine interactions regardless of whether scientists understand them. One limitation to the percent of flow method is that it depends on natural variation and so may not be very useful for highly altered systems. Inflow-based approaches are attractive because they are straightforward, but without supporting studies their link to resources is weak, rendering them less compelling to the general public and therefore more difficult to sustain in the face of opposition.

In contrast to the SWFWMD, the South Florida Water Management District (SFWMD) takes a resource-based approach for setting inflow requirements. Rather than focusing on the requirements of key commercial species, such as was described for Texas, the SFWMD applies the Valued Ecosystem Component method to choose an important resource or set of resources in each water body and then works to provide suitable environmental conditions for that resource. There is a distinction to be made between indicators, which are key species or habitat types that are particularly sensitive to estuarine conditions, versus valued resources, as identified by society. Although they can be one and the same, those resources that are sensitive and might be considered good indicators of estuarine conditions are not always the ones that the public values, and those resources that the public values might be less sensitive to change. The proposed MFLs for the Loxahatchee River and estuary and the Caloosahatchee estuary illuminate this contrast.

In the Loxahatchee River and estuary, bald cypress, *Taxodium distichum,* was identified as the key species to be protected against significant harm. The upstream freshwater portion of the river is comprised of largely pristine cypress-river swamps, including a number of trees within the 300–400 year old range (SFWMD 2001). Many people enjoy canoeing and other recreational activities in this part of the river and identify cypress with the system. The trees serve to stabilize the shoreline, and they provide habitat for many other plants and animals, including epiphytic plants and nesting birds. However, there is evidence that increasing encroachment of saltwater has led to an upstream shift in cypress and a concurrent invasion by mangroves (SFWMD 2001). In proposing a minimum flow for this area, the assumption was made that maintaining suitable environmental conditions for cypress would also be important for other desirable species (SFWMD 2001).

The proposed minimum flow for the Loxahatchee sought to maintain salinities at less than 2 psu (identified as a critical value for cypress) at a given position in the estuary in order to prevent further upstream encroachment of mangroves. In keeping with the requirements of the MFL legislation, this proposal was evaluated by an external scientific review panel. The panel identified a potential problem in that, although high salinity can kill cypress, their response is not well-quantified. Because cypress are long-lived and slow-growing, it may also be many years before they would show a change in response to a change in inflow. Therefore, cypress, although a valued resource, is not necessarily a good choice as a management tool. The MFL proposal has undergone extensive revision by the District and is now focused on sensitive plants in the floodplain-cypress community. The revised proposal is currently under review.

In the Caloosahatchee estuary, the proposed minimum flow is instead based on the distribution of a suite of sensitive indicator species (see Doering et al. 2002). In this case three species of seagrasses (*Vallisneria americana, Halodule wrightii,* and *Thalassia testudinum*) were identified as key species that provide important benthic habitat for juvenile estuarine and marine species. These seagrasses have different salinity requirements, and maintaining their distribution patterns along the longitudinal axis of the estuary was proposed as an overall indicator of estuarine health. The SFWMD did a combination of field and laboratory research to determine the salinity sensitivity of the various seagrasses, and their results were then combined with modeling and hydrologic studies to determine the flow rates needed to maintain target salinities within the estuary (Doering et al. 2002). Although the plants used are sensitive indicators of estuarine salinity, and do in fact offer protection and foraging for many other organisms, they are not readily identified by the public and do not represent a resource that is highly valued by society. The case is made that if the plants are protected conditions will also be suitable for other organisms. Chamberlain and Doering (1998) describe how the optimal flows determined for the seagrasses will also be beneficial for fish, shrimp, crabs, and other resources. Once again, it was necessary to link the indicator chosen by the scientists to resources valued by society and to provide this information to the public.

### San Francisco Bay Estuary

The final case study is that of the San Francisco Bay Estuary, California, which receives flow from the Sacramento and San Joaquin rivers. The San Francisco Bay Estuary represents a system where inflow has been extensively modified by humans: diversion of freshwater for irrigation and municipal use has frequently exceeded 50% of the inflow to the estuary, especially during drought years (Jassby et al. 1995). Decreased or altered inflow has been linked to decreases in species of economic interest in the estuary (including Chinook salmon and striped bass), and several species of fish have been listed under endangered species legislation (Kimmerer and Schubel 1994). Against this backdrop, a drought began in 1987 and lasted through most of 1992.

A description of the complex regulatory environment with regard to water management in the San Francisco Bay Estuary is beyond the scope of this review, but by the early 1990s it was clear that the State Water Resources Control Board, the agency responsible for determining water distribution, was not making suitable progress on the issue (Kimmerer and Schubel 1994). In 1991, the San Francisco Estuary Project initiated a series of workshops to provide scientific input for the development of an effective inflow management strategy (Tuohy 1993; Kimmerer and Schubel 1994). At these workshops, which were attended by scientists from both agencies and academia, monitoring information on estuarine resources was compiled and related to salinity and inflow, and a scientific consensus was reached on the basis for an estuarine standard. The recommendations of the workshop were incorporated into the inflow standard implemented by the State Water Resources Control Board. This standard, which is based on salinity, was adopted in 1994 as part of the Bay-Delta Accord, an agreement among agencies and stakeholders with interests in the San Francisco Bay Estuary. The CALFED (California Water Policy Council and Federal Ecosystem Directorate) Bay-Delta program, a forum for state and federal cooperation in the region, was also established at that time (CALFED 2002).

Although it can be linked to both inflow and resources, the freshwater inflow standard in use for the San Francisco Bay Estuary is based on the location of water of a given salinity and represents a condition-based approach to inflow management. The water quality standard adopted in the Bay-Delta Accord states that inflow must be managed so that the so-called $X_2$ (the distance from the Golden Gate Bridge to the 2 psu isohaline, measured 1 m off the bottom and averaged over more than 1 d) is positioned where it may be ''beneficial to aquatic life'' (CALFED 2002, p. 1). Maintaining the 2 psu isohaline downstream positions the salinity gradient of the estuary in such a way as to allow increases in numerous estuarine resources. Indeed, investigators found significant statistical relationships between $X_2$ and the supply of phytoplankton and phytoplankton-derived detritus; the abundance of mysids and shrimp; the survival of striped bass and striped bass year class strength; the survival of salmon smolts; and the abundance of planktivorous, piscivorous, and bottom-foraging fish (Kimmerer and Schubel 1994; Jassby et al. 1995). These connections are thought to reflect the availability of suitable habitat, but the causal mechanisms remain largely unresolved (although see Kimmerer 2002).

In addition to relating $X_2$ to resources, it was also necessary to relate it to freshwater flow. As might be expected, $X_2$ can be correlated with estimated net outflow to the Bay from the delta of the Sacramento and San Joaquin Rivers. However, there was concern because outflow estimates can be uncertain, particularly at low flows when the relative importance of losses such as the consumption of freshwater within the delta increases (Jassby et al. 1995). $X_2$ was therefore selected as the basis of the standard because it is a reflection of net outflow and its location can be determined through salinity measurements.

The approach taken in the San Francisco Bay Estuary has many of the components of adaptive ecosystem management. Scientists from both academia and agencies were involved in the original workshops, and many of these individuals also participate in the Interagency Ecological Program, a long-standing consortium of federal and state agencies working in the estuary that provides technical advice to CALFED. CALFED is committed to stakeholder participation, as evidenced by the fact that representatives from agricultural, environmental, and urban groups were all a party to the Bay-Delta Accord. The fact that $X_2$ was linked to a range of estuarine components at all trophic levels and that there was general understanding among stakeholders that it was a meaningful indicator of habitat quality enabled the adoption of a standard broadly geared toward protection of the ecosystem rather than one focused more narrowly. This fits the definition of ecosystem management, which is another concept often discussed in the context of adaptive management (Christensen et al. 1996). What is missing from this example in terms of adaptive management is the development of alternative models that can be explicitly tested (see Kimmerer 2002). The experience in the San Francisco Bay Estuary still provides a useful model of the joint power of scientific consensus and public involvement in influencing both management and policy decisions with regard to estuarine freshwater inflow.

## Summary

Citizens, politicians, managers, and scientists all have roles in addressing the issue of freshwater inflow, as depicted in the conceptual model (Fig. 2). Policy making is by definition a political endeavor and inflow policies are set in response to pressures from constituents, more often than not as a reaction to a crisis. It is interesting that this crisis came in the form of prolonged droughts in all three of the cases considered here. The language used in these policies: ''to maintain an ecologically sound environment'' in Texas; to ensure that no harm comes to ''the water resources or ecology of the area'' in Florida; and to ''restore ecological health'' in the San Francisco Bay Estuary, is broad, and it is up to resource managers to determine how to meet these goals.

To establish a management strategy to meet these broad policy objectives for freshwater inflow requires an understanding of the connections between inflow itself, estuarine conditions (such as salinity and concentrations of dissolved and particulate material), and resources (such as the distribution and abundance of organisms). Although any of these components can be used as the basis of a management approach, there is always an implicit or explicit recognition that they are linked. Managing inflow on the basis of flow alone depends on the assumption that if inflow is protected the conditions (and therefore resources) will be protected as well. In the case of the Southwest Florida Management District, the inflow-based standard is backed up by extensive observations linking inflow to resources (Flannery et al. 2002). Managing inflow based on resources, as in Texas or in the South Florida Water Management District, works backwards to relate resources to conditions (salinity), and in turn to inflow. Managing inflow based on conditions, as in the San Francisco Bay Estuary, requires information on both sets of linkages.

The role of scientists in this process is to ensure that inflow management is based on the best science available. Given that there is generally very little quantitative information about the relationships among inflow, conditions, and resources for a given estuary, it is critical that managers have timely access to research on these topics. Close involvement of scientists in the development and evaluation of the technical aspects of a management strategy is equally important, as is evidenced by the case studies presented here: Part of the reason that Texas, California, and Florida have been able to develop scientifically-defensible management strategies is their commitment to involving scientists. Florida's expert review process is particularly effective in that outside scientists are brought in to critically review the technical basis for proposed standards.

Finally, it is worth remembering that freshwater inflow to estuaries comes from upstream. Freshwater delivery defines almost all of the physical, biological, and chemical properties of an estuary. Inflow is equally important to managers; no matter what the focus of a management strategy, carrying it out requires regulating inflow. In the end, it is not possible to establish freshwater inflow protection for estuaries unless they are viewed as the recipients of any and all upstream perturbations that influence the quantity, quality, or timing of freshwater delivery.


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