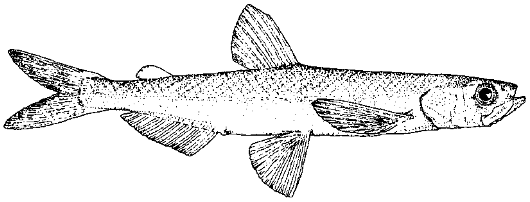
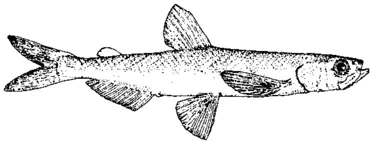
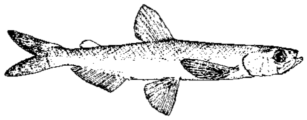
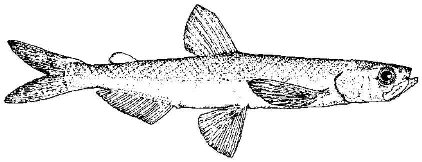
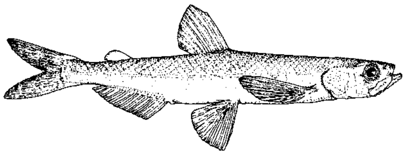


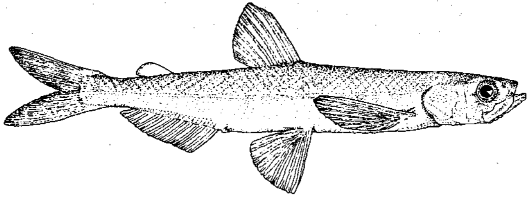


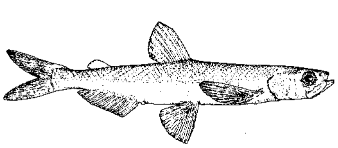






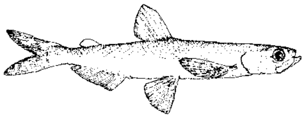
San Francisco Estuary Project

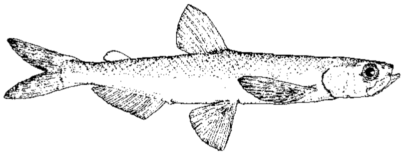


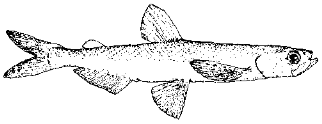


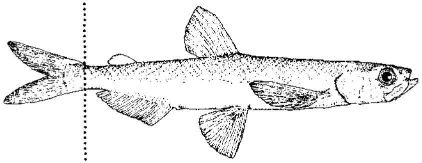
MANAGING FRESHWATER DISCHARGE TO THE SAN FRANCISCO BAY/SACRAMENTO-SAN JOAQUIN DELTA ESTUARY: THE SCIENTIFIC BASIS FOR AN ESTUARINE STANDARD

1993

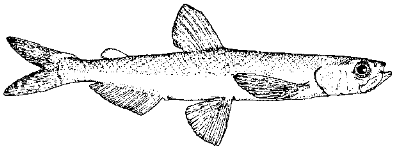








Conclusions and Recommendations of Members of the Scientific, Policy, and Management Communities of the Bay/Delta Estuary



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CONCLUSIONS AND RECOMMENDATIONS OF MEMBERS OF THE SCIENTIFIC, POLICY, AND MANAGEMENT COMMUNITIES OF THE BAY/DELTA ESTUARY

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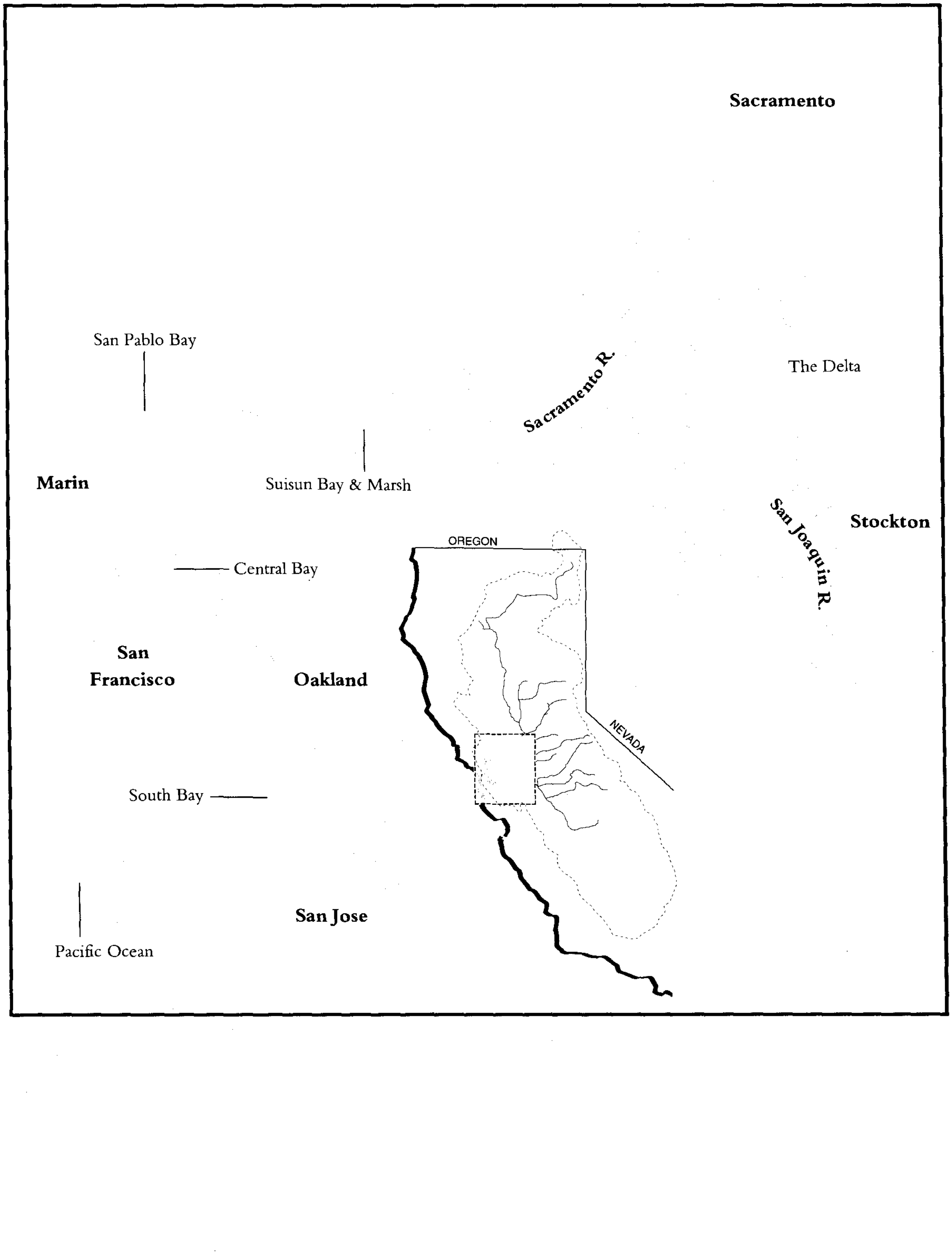

A PERSONAL OBSERVATION

The San Francisco Bay/Sacramento-San Joaquin Delta estuary comprises the San Francisco, San Pablo, and Suisun bays, and the Delta of the Sacramento-San Joaquin rivers, two large rivers that drain the Central Valley of California.

As a result of diversions of freshwater from the Sacramento-San Joaquin Delta for agriculture and for urban use, the river discharge into the estuary is significantly lower than natural levels. According to some estimates, river inflow to the estuary is only 50 to 70% of what it was in 1800. Among the primary responses of the estuary to this decrease in freshwater inflow have been an upstream (landward) displacement of the low salinity transition zone between the estuary proper and the tidal reaches of the rivers, and a compression of low salinity habitat.

It is well established scientifically that the extent — the area and volume — of low salinity habitat in estuaries is important to the success of a number of species, such as anadromous and semi-anadromous fishes, and to the success of other estuarine ecosystem components such as tidal marshes. The contribution of each of the different processes and properties characteristic of low salinity zones of estuaries, and the contributions of different combinations of these processes and properties, to the success of different ecosystems components, are complex and have not been successfully evaluated for any estuary.

I know of no other estuary that has as complicated a situation in the low salinity transition zone as does the San Francisco Bay/Sacramento-San Joaquin Delta estuary. The complex engineering system in the Delta region for withdrawing and diverting freshwater away from the estuary acts like a giant "predator", particularly for ichthyoplankton and young-of-the-year fish. This confounding leads to debate and disagreement over the relative importance of the benefits of low salinity habitat and therefore of flow, on the one hand, and of the liabilities of the physical diversion of a portion of that flow and the associated processes of entrainment of organisms, on the other. The debate and the demands for scientific certainty are intensified because of the economic importance of water, particularly for agriculture. Are total flow and the extent of low salinity habitat the most important factors for a healthy estuarine ecosystem? Or, are the places, times and mechanisms by which a fraction of that flow is diverted the most important factors? Could stresses on the estuarine ecosystem related to water-use be reduced sufficiently by changing the timing and mechanisms of withdrawal without reducing the total amount of water diverted? If so, what specific water management strategies would be most effective? What would be the associated biological benefits? Until these questions can be answered with a degree of scientific certainty acceptable to the State Water Resources Control Board, estuarine standards are needed to protect the estuarine ecosystem against further degradation.

This complex situation argues strongly for the development of *estuarine standards* as part of a set of management tools for protecting the Bay/Delta estuarine ecosystem. At a series of technical workshops convened to address this situation, the vast majority of the workshop participants, and virtually all of the participating scientists, concluded that one set of estuarine standards should be salinity standards which can be used to position the low salinity zone relative to the "predator". Different standards should be developed for different seasons to reflect the presence and vulnerability of "prey" that are particularly sensitive to the actions of this "predator". One appropriate index proposed by these workshops for the development of salinity standards is the upstream limit of the near bottom 2‰ (parts per thousand) isohaline. An isohaline is defined as a line in the estuary connecting all points of equal salinity. The salinity standards based upon this index should be expressed as specific upstream limits — one for each of several periods (seasons) of the year and averaged over that period. The locations for the standards would be selected to attain an appropriate level of ecosystem protection to achieve an appropriate environmental goal.

Given the present state of knowledge, the principal method of selecting salinity standards for the Bay/Delta estuary will be through the use of statistical relationships. A number of these relationships were developed during the technical workshops to evaluate the responses of estuarine organisms at different trophic levels to seasonal changes in the position of the 2‰ isohaline. This approach "lumps together" a number of factors including the ecological effects of low salinity habitat and the physical effects of entrainment losses.[1] Research efforts should be enhanced to provide the knowledge needed to disaggregate the cause-effect relationships between biological success and salinity, flow, diversion, and a variety of other environmental factors. Until that understanding is developed, salinity standards can provide a valuable tool for protecting the ecosystem because they integrate the effects of all these processes and phenomena upstream from the specified location of the 2‰ isohaline.

While the confounding of the ecologically positive effects of habitat by the negative effects of entrainment provides value-added to the proposed salinity standard (which integrates the effects of both), it also raises a caution flag. Any proposed changes to the water withdrawal and distribution system, or in the way

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[1] *This issue is explored in the technical papers that accompany this report.*

in which this system is operated—particularly within the Delta region—should trigger a re-evaluation of the standards. And, if any changes are actually made to that system, or to its operation, the biological responses to those changes should be monitored carefully to produce the data needed to formulate new salinity standards; standards to achieve the desired level of environmental protection.

Depending upon the nature of the changes made to the "plumbing system" or to its operation, and the resulting changes in "predation rates", the upstream limit of the 2‰ isohaline (the salinity standard) might have to be moved farther downstream by providing more fresh water or might be allowed to migrate farther upstream and still maintain a level of environmental protection equivalent to that before the changes were made. This report proposes a method of setting salinity standards that provides for such adjustments.

Each conclusion and recommendation in this report was reviewed and voted upon in the final workshop. In no case did the final number of dissenting votes exceed three and in only a few cases did the number exceed two. In spite of this endorsement, a number of participants subsequently requested that their names be removed from the cover of the report. I have honored those requests. I thank each participant for his or her hard work and creativity in a search for solutions to a complex problem, a problem with a variety of dimensions: environmental, economic and socio-political.

A number of other people contributed to the success of the technical workshops. I thank Maureen Flynn for her patient and careful typing of the many drafts of this final report as well as the other reports. She never lost her good humor. I thank Liz Blair for making the necessary preparations for each of the workshops. Her attention to detail contributed to productive workshops. I thank Susan Schubel for assisting in the facilitation of the first workshop. Finally, I thank Tim Vendlinski for his overall project support and management. He did a superb job and was largely responsible for maintaining an even keel even in some troubled seas.

J. R. Schubel
Dean and Director
Marine Sciences Research Center at SUNY
Stony Brook, NY
26 October 1992

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INTRODUCTION

Aquatic resources of the Sacramento-San Joaquin Delta and upper portions of San Francisco Bay have undergone significant declines over the past several decades. Species characteristic of the Delta and rivers, such as striped bass and salmon, began to decline during the late 1970s. Prolonged drought, large diversions of fresh water, and dramatic increases in populations of introduced aquatic species during the 1980s and 1990s brought a number of indigenous aquatic species to extremely low levels. Species that spend more of their lives downstream of the Delta, including Delta smelt, longfin smelt, and many zooplankton, maintained large populations through the 1970s, but declined sharply after the mid-1980s. Declines in aquatic resources have led to curtailed fishing seasons, to petitions for endangered species status, and general concern about the health of the estuarine ecosystem.

Concern over the impacts of increased salinity produced from the combination of drought and high diversion rates is not limited to aquatic communities. The few remaining fragments of brackish and freshwater tidal marshlands are particularly vulnerable to increased salinity or to reduced variability in salinity. Under natural conditions, these tidal marsh communities would move upstream with the changing salinity. But the flood plains and other lowlands suitable for the evolution of tidal marshes are absent upstream. Tidal marshes provide important habitat for numerous plants and animals of special concern.

Large demands for water by the agricultural community and by California's burgeoning urban areas make it difficult to allocate additional freshwater for the protection of dwindling aquatic and wetland resources of the estuary. Management of the State's water resources necessitates a delicate balancing of needs, given the intense and growing competition for water. If the freshwater needs of the estuary are to be considered seriously they must be based on sensitive, straightforward, and diagnostic indicators of the responses of the estuarine ecosystem to patterns of freshwater inflow.

An extensive body of scientific evidence indicates that flows into, within, and through the estuary are extremely important to organisms that depend on the estuary for at least a portion of their life cycles. However, the mechanisms by which flows affect different elements of the ecosystem are not well understood. In the Bay/Delta estuary, many chemical and physical properties and processes are tightly linked to flow, including proportion of water diverted, salinity at a given point, the longitudinal position of a particular salinity range, and alteration of the effects of toxicants through dilutions. Any of these phenomena could be controlling a particular species, but each will also vary with the other variables that are closely correlated with flow.

At present, the complex configuration of the Delta and the estuary, combined with the complex withdrawal and diversion network, preclude any simple, directly monitored measure of freshwater discharge to the estuary. Effective protection and management of the estuary requires an index of the estuary's response to freshwater inflow that (1) can be measured accurately, easily and inexpensively; (2) has ecological significance; and (3) has meaning for nonspecialists. Net Delta outflow, which is calculated from various measures and estimates of water inflow and use, has been a useful tool but it does not satisfy all of these requirements. Because of the high correlations among the flow-related variables, the choice of a suitable index does not need to be based on any presumed mechanism.

The San Francisco Estuary Project convened a series of technical workshops to evaluate the responses of estuarine biota and habitats to various conditions of salinity and flow. The workshops involved approximately 30 scientists and policy makers with expertise in estuarine oceanography and ecology, and in water and living resource management. The group focused its attention on Suisun Bay area, the portion of the estuary downstream of the confluence of the Sacramento and San Joaquin rivers and upstream of Carquinez Strait. Internal Delta issues (such as gate closures, water exports, and internal flows) or problems with downstream portions of San Francisco Bay (such as urban and industrial discharges) were not directly addressed by the group. No attempt was made to incorporate all management actions that might benefit biological communities, nor to identify what level of environmental restoration and protection should be set based on salinity and flow.

Identification of freshwater needs of aquatic resources has caused conflict for a variety of reasons. Debate of scientific issues is fundamentally different from other kinds of debate in that it should yield to scientific investigation. Participants developed issue papers that delineated areas of scientific agreement. Several issue papers showed that conditions in Suisun Bay largely reflected the abundance, recruitment, or survival not only of local species, but also of habitat conditions for species upstream and downstream. A primary result of the issue papers produced for this group was that almost all species *studied* increased in abundance as a simple function of increased outflow and decreased salinity. The absence of a plateau or peak in the relationship of species abundances and outflow conditions means that science alone cannot identify an optimal outflow. Furthermore, the similar response of species at all ecological (trophic) levels argues strongly that the estuary should be managed using an ecosystem approach rather than on a species by species basis.

The technical workshops concentrated on developing the scientific rationale for an estuarine index to measure the estuary's response to different levels and patterns of freshwater input. Participants recognized that economic and socio-political considerations should be accounted for at other points in the deliberations. The needs of society, as well as the needs of the environment, should be considered in determining appropriate allocations of freshwater. However, the premise of the workshops was that one should start with the best scientific and technical judgements possible.

Many large-scale changes in the structure of the Delta have been proposed to facilitate water-use and to reduce impacts of water withdrawal on aquatic resources. There was general recognition by the group that the present Delta withdrawal and distribution system is a major contributor to the declines of important species. The conclusions and recommendations of the workshops are based upon the present water withdrawal and distribution system and would need to be re-evaluated if any significant alterations to that system are considered.

The conclusions and recommendations in this report were developed by the estuarine scientists and managers who participated in one or more of the workshops. The complete list of participants and their affiliations are listed in Appendix D. All conclusions and recommendations in this report were reviewed, voted on, and endorsed by a consensus of the estuarine scientists and managers who participated in the fourth and final workshop in the series (26 August 1992). The term consensus is used to represent group solidarity on an issue; a judgement arrived at by most of the scientists and managers present. In all cases, the consensus was unanimous or nearly unanimous. The conclusions and recommendations are arranged in a sequence that "tracks" the evolution of thinking of the participants. The conclusions and recommendations reached by the group reflect the participants' best scientific and technical judgements, not necessarily the positions of their affiliated agencies or organizations.

The following conclusions and recommendations are intended to provide guidance and information on *how* estuarine standards could be developed and how *different levels of protection* of estuarine resources could be selected.

The full justifications for these conclusions and recommendations are contained in technical papers that accompany this report and in other documents prepared for the San Francisco Estuary Project. (Appendix E).

IMPORTANT CONCLUSIONS AND RECOMMENDATIONS

(1) CONCLUSION

Because of the complex nature of the freshwater delivery and distribution system in the San Francisco Bay/Sacramento-San Joaquin Delta estuary, there is at present no single, simple, accurate measure of freshwater input to the estuary that conveys information important to resource managers and to the public, and that is meaningful to those with special concerns about how fluctuations in freshwater inflow to the estuary affect habitat and the condition of the estuarine ecosystem.

RECOMMENDATION

Estuarine standards should be developed to be used in conjunction with flow standards. One set of standards should be based upon an index of the physical response of the estuary to fluctuations in the input of fresh water. These standards should have diagnostic value in providing, throughout the year, a level of protection to the estuary and to important ecosystem values and functions consistent with environmental goals and objectives for the Bay/Delta estuary.

(2) CONCLUSION

Estuarine standards to be used in conjunction with flow standards should be based upon an index that is simple and inexpensive to measure accurately, that has ecological significance, that integrates a number of important estuarine properties and processes, and that is meaningful to a large number of constituencies.

RECOMMENDATION

Salinity should be used as an index for the development of some estuarine standards.

JUSTIFICATION

In the first workshop (August 1991), participants identified and assessed a number of indices of the estuary's responses to flow to use in managing freshwater discharge to the estuary. The preliminary, pre-workshop, choice was the position of the entrapment zone. This index was abandoned quickly, however. The entrapment zone is important to estuarine ecosystem processes and functions, but at present there is no single, straightforward "entrapment zone index" suitable for monitoring the position or strength of the entrapment zone as a function of freshwater input.

Salinity was selected as the most appropriate index because: (1) the salinity distribution is of direct ecological importance to many species; (2) the salinity distribution is a result of the interplay of freshwater input, geometry of the estuarine basin, diversion of fresh water in the Delta, and the tidal regime; and (3) salinity measurements can be made accurately, directly, easily, and economically. Moreover, since most of the major concerns about reductions in the freshwater input to the estuary are associated either directly or indirectly with the loss or alteration of low salinity habitat, salinity is an ideal index for keeping track of the extent — both area and volume — of low salinity habitat. The salinity distribution represents the response of the estuary to different combinations of river discharge, diversions and withdrawals, tidal regime, and basin geometry.

(3) CONCLUSION

Salinity measured at about 1m above the bottom[1] is an index upon which estuarine standards should be developed. The index is a practical way of tracking changes in habitat.

RECOMMENDATION

Standards should be developed using an index that establishes an upstream limit of the position of the 2‰ near-bottom isohaline, averaged over different periods of the year.

(4) CONCLUSION

Analysis of the available historical data indicates that, throughout the year, the farther downstream the 2‰ near-bottom isohaline is displaced, the greater the abundance or survival of most species examined.

RECOMMENDATION

The downstream position of the 2‰ isohaline should be unconstrained.

[1] *Because the difference between surface and near-bottom salinities is small and because the relationship between them is reasonably well known, surface salinity could also be used. Near-bottom salinity is recommended, however, because it is a more stable indicator.*

JUSTIFICATION

From the environmental perspective — an important perspective, but not the only one — scientific uncertainty dictates taking an environmentally conservative approach, i.e. providing enough Delta outflow to the estuary to push the 2‰ isohaline farther downstream than might be required with greater scientific certainty. It is anticipated, and preliminary analysis supports it, that the salinity standard — the upstream limit of the 2‰ near-bottom isohaline — will vary from season to season to provide the desired level of protection.

(5) CONCLUSION

Estuarine systems are characterized not only by short-term responses to the mean salinity at any given location, but also by responses to longer-term seasonal, annual and interannual variability in salinity and other properties.

Recent advances in scientific understanding indicate that this dynamic character of healthy estuarine ecosystems is particularly true for the distribution and abundance of wetland vegetation, but also holds for other aquatic organisms.

RECOMMENDATION

The potential importance of variations in salinity on different time scales to the structure and dynamics of estuarine ecosystems should be considered in developing salinity standards. Deviations from the patterns of salinity variability in the historical data set could increase the risk of not achieving environmental goals and objectives even if mean positions of the 2‰ near-bottom isohaline were matched with the historical data sets.

JUSTIFICATION

There is strong biological evidence from a number of estuaries throughout the world that variability in flow, in circulation and mixing, in the salinity distribution, and in the distribution of other important properties and processes is important in maintaining a healthy estuarine ecosystem. Therefore, variability in flow above the threshold needed to meet the seasonal salinity standard is encouraged.

(6) CONCLUSIONS

Empirical statistical relationships were developed between a variety of estuarine properties and resources, and the position of the near-bottom 2‰ isohaline and other flow-related variables. *The relationships developed are statistical relationships. They are not proof of cause-effect.* The relationships indicate clearly, however, that the position of the near-bottom 2‰ isohaline can serve as a powerful diagnostic indicator of the condition of biological "units" (communities, populations) across a range of different trophic levels.

With the information these relationships can provide, water managers will be in a far better position to regulate freshwater discharge to the estuarine system to produce, on the average[2], predictable and desirable ecological responses of the estuary consistent with goals selected for the estuarine ecosystem. If this strategy is followed, the probability of the desired ecological response will be enhanced and the chances of undesirable ecological surprises in the estuary will be reduced.

Because the statistical relationship between net Delta outflow and the position of the near-bottom 2‰ isohaline is strong, the position of the near-bottom 2‰ isohaline is an excellent surrogate for net Delta outflow in managing freshwater input to the estuary. The relationship may be improved further through routine direct monitoring of the position of the 2‰ isohaline and a suite of biological responses.

RECOMMENDATIONS

The salinity distribution should be monitored continuously at a series of at least six stations spaced approximately five kilometers apart and located along the channel between about Emmaton and Carquinez Bridge. Measurements should be made at least near the surface and near the bottom at each station. The data should be telemetered to a convenient location for timely analysis and interpretation. These continuous monitoring data should be supplemented with detailed surveys to map the distribution of salinity in three dimensions. The data should be readily available in a timely way to all interested parties.

An appropriate biological monitoring program should determine responses of a variety of organisms to changes in position of the 2‰ isohaline.

JUSTIFICATION

During the second and third workshops, and during intersessions between workshops, a systematic search was made to select the most powerful tools of analysis to describe how diagnostic biological indicators respond to changes in position of the near-bottom 2‰ isohaline. When data were rich enough, other variables were included in the analyses.

[2] *Over a period of several years.*

The first task was to specify the most diagnostic resource variables — the responses of indicators that would convey the maximum amount of environmental/ecological information. In every case, the objective was to demonstrate how these diagnostic environmental/ecological indicators responded to changes in the position of the near-bottom 2‰ isohaline and to a variety of other flow measures. In every case, experts on the particular biological response were consulted in selecting the appropriate averaging time for the position of the 2‰ isohaline.

(7) CONCLUSION

The position of the near-bottom 2‰ salinity isohaline is an index of habitat conditions for estuarine resources at all trophic levels, including the supply of organic matter to the food web of Suisun Bay, an important nursery area. In other words, well-behaved statistical relationships exist between the near-bottom 2‰ isohaline and many estuarine resources for which sufficient data exist to make appropriate analyses. Moreover, at least a rudimentary understanding exists for the causal mechanisms underlying many of these relationships. The location of the near-bottom 2‰ isohaline is important either because it is a direct causal factor or because it is highly correlated with a direct causal factor (e.g. diversions).

Preliminary analyses show that errors in prediction using models which incorporate only the position of the 2‰ isohaline are comparable to the errors using more complex models which incorporate additional flow-related variables. In other words, given the present data sets, predictive models using only the position of the near-bottom 2‰ isohaline perform as well as more complex models that incorporate other variables. However, some of these other variables may be very important in affecting habitat and the condition of biological resources of the estuary.

RECOMMENDATIONS

At this time, the most appropriate basis for setting salinity standards for the portion of the estuary on which this report concentrates is the position of the near-bottom 2‰ isohaline *alone,* unless it can be shown either that another variable is the controlling variable or that incorporation of additional variables improves the predictive capability.

Further research should be conducted to improve prediction of the responses of important estuarine resources to variations in the position of the near-bottom 2‰ isohaline. That research should incorporate other variables where they can be shown to contribute significantly.

(8) Conclusion

A number of key species are subject not only to the biological effects of the location of the near-bottom 2‰ isohaline, and therefore the effects of freshwater inflow to the estuary, but also to the physical effects of entrainment and diversion by the various water projects.

Recommendations

Salinity standards should be keyed to the existing city, county, regional, state, and federal water diversion and distribution system. Proposed changes to that system should trigger a re-evaluation of the salinity standards to ensure that they will continue to provide the desired level of environmental protection while retaining as much flexibility as possible in meeting the state's other needs for water.

Since a broad class of models can be constructed, including mechanistic and statistical models that incorporate both biological and physical parameters and other factors such as diversions, exports, and antecedent conditions, efforts should be enhanced to ensure a consistent, long-term accurate measurement program to enhance these models and to decrease the uncertainties in their application. The ultimate goal is to have a predictive model that incorporates the position of the 2‰ isohaline and other appropriate physical and biological variables.

(9) Conclusion

Salinity standards should be based upon the best scientific and technical knowledge. A method is needed to summarize and to advance the state of scientific and technical knowledge of the complex relationships between variations in the position of the near-bottom 2‰ isohaline during different periods of the year (and associated Delta outflow) and a variety of diagnostic ecosystem responses.

Recommendations

Salinity and flow-response matrices should be developed for different biologically important periods of the year. The matrices should summarize the existing state of knowledge of the responses of a rich variety of estuarine organisms and communities as well as estuarine properties and processes, to the location of the near-bottom 2‰ isohaline and associated freshwater discharge to the estuary. The estuarine properties and biological responses initially identified for inclusion in these matrices are summarized in Exhibit A.

A Matrix Manager should be appointed to oversee the development of the summary matrices and to ensure quality control. The Matrix Manager should orchestrate the analyses of relevant data and ensure that the results of the analyses are cast into forms appropriate for the intended uses.

Because estuarine habitat suitability and, therefore, estuarine ecosystem health are not simply a function of the instantaneous salinity distribution, the entry in each response cell of the matrix, whenever possible, should be based upon the development of functional relationships of estuarine properties to isohaline positions (and freshwater input to the estuary) that incorporate lagged terms, seasonal variability, and other water management variables. Ideally, the input to each matrix cell would include a directory of the appropriate model, or models, that could be used for prediction.

The proposed matrices are shorthand methods for keeping track of advances in the state of scientific knowledge and for ensuring that the most up-to-date scientific knowledge is used in decision-making. They are not intended to be used as isolated regulatory tools. They are a summary of the state of development of those tools, a guide to which tools to use during different times of the year, and an index of where to find them. The responsibility for development of the matrices and for periodically updating them should be institutionalized. One appropriate agency might be the Interagency Ecological Studies Program.

JUSTIFICATION

The proposed matrices are an effective shorthand way of summarizing in a convenient format the status of a large amount of data and information relating the responses of the estuary to fluctuations in freshwater inflow and to other water management variables. The matrices are a useful vehicle for summarizing the biological benefits — using a broad array of response indicators — of positioning the near-bottom 2‰ salinity isohaline at various distances upstream (inland) from the Golden Gate Bridge during different periods of the year. The proposed matrices would provide the first quantitative and comprehensive summary of how the San Francisco Bay/Sacramento-San Joaquin Delta estuary ecosystem responds to fluctuations in freshwater inflow to the estuary (Delta outflow) and to the estuary's changing salinity regime. The matrices have further advantages. They will provide managers, policy-makers and the public with: (1) a clear statement by the scientific community of the current status of understanding of the effects of different freshwater discharge-diversion scenarios on the estuarine ecosystem; (2) an identification of critical gaps in scientific knowledge that can be used to guide future research and monitoring activities; and (3) a summary that is easily updated on a cell-by-cell basis as new knowledge is developed.

The models upon which the matrices are based can serve as tools for regulatory agencies to use in incorporating the environmental needs of the estuary into a set of management prescriptions for storing, releasing, and diverting water for consumptive uses. Selection of the level or degree of biological response to be achieved — the level of environmental protection — is the responsibility of regulatory bodies acting in response to society's priorities.

(10) CONCLUSION

The actual setting of salinity standards — specifying the upstream locations of the near-bottom 2‰ isohaline for different periods of the year — should be keyed to environmental goals: to achieving and sustaining some desired biological response level specified in terms of habitat protection or abundance and survival rates of important and diagnostic estuarine and wetland species.

RECOMMENDATIONS

Goals should be expressed in terms of desired conditions for some future time. Progress toward those goals should be monitored and reported widely. Environmental goals for the estuary will be most effective if they are expressed in terms of restoring conditions to those that existed at specific historical times such as those summarized in Exhibit B.

(11) CONCLUSIONS

At prevailing patterns of the position of the near-bottom 2‰ isohaline, the biological resources of the low salinity portion of the estuary, including the Delta, have been seriously depleted. Data from the Interagency Ecological Studies Program and the University of California at Davis indicate clearly that species at every trophic level are now at, or near, record low levels in the Delta and in Suisun Bay. This is not surprising considering the recent drought, the introduction of exotic species, and the increased diversion of water.

Analyses of the data indicate that the abundance or survival of a number of important species at a variety of life history stages and from a variety of trophic levels is related to the position of the near-bottom 2‰ isohaline. Of the organisms whose response to salinity has been analyzed, the farther downstream the 2‰ isohaline is, the higher their abundance or survival.

Almost all of the components of the estuarine community analyzed during the workshops (e.g., organisms, habitats, and processes) show a strong, coherent, and negative monotonic response to increased penetration (upstream movement) of the near-bottom 2‰ isohaline. There is no well-defined break point that can be reliably identified statistically in the composite relationship between the abundance or survival of these components and the position of the 2‰ isohaline. In other words, the biological benefits of downstream displacement of the 2‰ continue to increase over the range of positions of the 2‰ near-bottom isohaline reflected in the historical data set.

If one selects a certain level of restoration and biological response as a goal, then one can develop statistical relationships to prescribe the appropriate range of the position of the near-bottom 2‰ isohaline and the amounts of water necessary to achieve these salinity distributions during different periods of the year. *While such action will not guarantee achieving a desired level of resource recovery or protection, it would increase the probability of attaining these goals.*

RECOMMENDATIONS

A range of environmental/ecosystem restoration goals should be selected, and analyses should be made, to determine the distribution of the 2‰ near-bottom isohaline throughout the year consistent with those goals. Historical flow and salinity data should be examined to determine how frequently these conditions would have been met before construction of the Central Valley Project; the State Water Project; a variety of city, county, and regional projects that divert water; and before the large-scale reclamation of historical tidal marshlands. The results of these analyses would provide a valuable context within which to evaluate the amounts of water needed to achieve a range of ecological goals.

EXHIBIT A

A PRELIMINARY LIST OF DIAGNOSTIC ESTUARINE PROPERTIES AND COMMUNITIES TO BE INCLUDED IN THE SALINITY AND FLOW-RESPONSE MATRICES FOR DIFFERENT BIOLOGICALLY-IMPORTANT PERIODS OF THE YEAR

Estuarine Property/Community

I. Water Quality for Human Use
  - A. *taste & odor*
  - B. *THM content*
  - C. *salinity*

II. Bathymetry Changes

III. Hydrodynamic Processes
  - A. *transport/circulation*
  - B. *structure*
  - C. *bay-ocean exchange*
  - D. *residence times*

IV. Habitat Area and Volume

V. Suspended Sediment Dynamics

VI. Water Properties
  - A. *light availability*
  - B. *temperature*
  - C. *salinity distribution*
  - D. *nutrient distributions*

VII. Fates & Effects of Toxins

VIII. Algal Biomass, Primary Productivity, Species
  - A. *bay*
  - B. *Delta*

IX. Nuisance Blooms
  - A. *macroalgal*
  - B. *microalgal*

X. Organic Carbon as Food

XI. Planktonic/Neritic Crustaceans: Copepods & Mysids

XII. Fish Abundance
  - A. *estuarine residents*
  - B. *estuarine spawners*
  - C. *euryhaline estuarine species*
  - D. *anadromous species*
  - E. *euryhaline marine species*

XIII. Benthic Faunal Abundance

XIV. Invasion Likelihood, Success

XV. Marsh and Mudflat Communities

A. plant species
B. migratory, transient, and resident waterfowl; shorebirds, raptors, and passerine species
C. mammal species
D. amphibian and reptilian species
E. invertebrate species as prey

EXHIBIT B

SOME ALTERNATIVE LEVELS OF RESTORATION AND BIOLOGICAL RESPONSE (I.E. MAINTENANCE, ENHANCEMENT, RESTORATION) THAT COULD BE USED IN FORMULATING GOALS FOR THE ESTUARY

Of the possible alternative biological goals, the following could be expressed in terms of *average* historical levels of abundance or survival rates of aquatic resources during different periods of time:

1984-89: In selecting this period, the goal would be to *maintain* aquatic resources at current levels and to prevent further declines. This period encompasses wet and dry years, including the first three years of the recent extended drought, during which full export demands for water were met.

1970-75: In selecting this period, the goal would be to *restore* aquatic resources to levels that existed during a series of years that encompass 1975, the benchmark year for the anti-degradation standard for water quality parameters under the federal Clean Water Act. There were no critically dry years in this sequence period.

1973-77: In selecting this period, the goal would be to *restore* aquatic resources to levels that existed during a representative period of years encompassing 1975, the benchmark year for the federal anti-degradation standard. This period includes two critically dry years (1976 and 1977).

1956-68: In selecting this period, the goal would be to *restore* aquatic resources to levels that existed before major environmental impacts of the State Water Project. This period covers a broad range of hydrologic and hydrodynamic conditions and provides a reasonable estimate of impacts of the state-operated project that should be subject to the State's non-degradation policy (1968 is considered the base year for this policy according to Resolution 68-16 of the State Water Resources Control Board). Resource agencies have identified the probable habitat conditions of the Delta for salmon through this period, and factors that controlled populations of striped bass are reasonably well understood.

1922-44: In selecting this period, the goal would be to *restore* aquatic resources to levels that existed before the federal and state water projects were constructed and operated. Selection of this period reflects the policy of USEPA discussed in its bio-criteria guidance paper which suggests restoring biological parameters in impaired water bodies to levels that existed under reasonably unimpaired conditions.

PRE-PROJECT AND FORECASTED CONDITIONS

Environmental goals could also be formulated directly in terms of the position of the near-bottom 2‰ isohaline. Using this approach, two alternative goals would be to restore the movement of the near-bottom 2‰ isohaline to average pre-project conditions; and to maintain the movement of the near-bottom 2‰ isohaline at average conditions forecasted to include the demands of 1995.

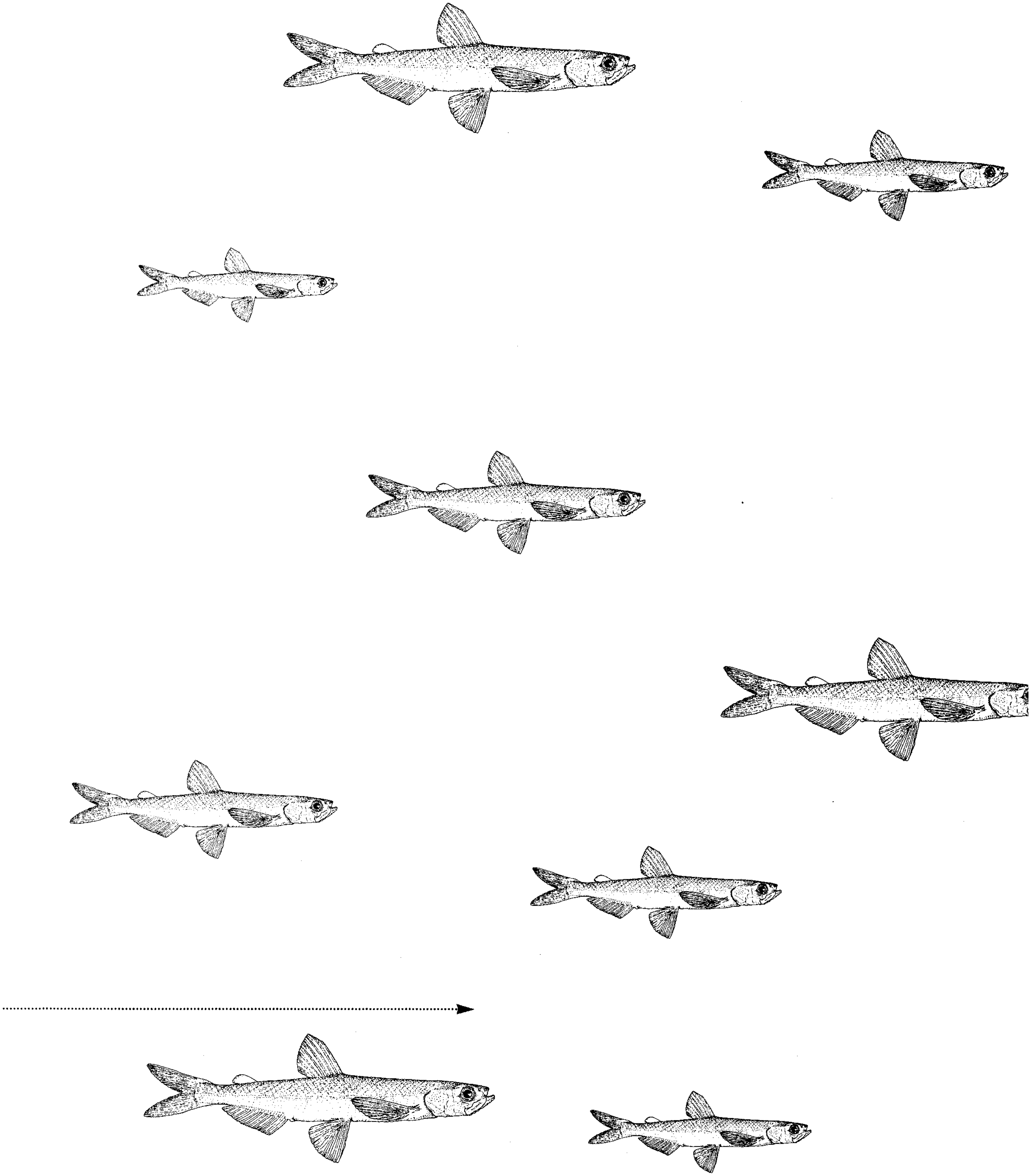
- *Pre-project conditions*: the goal would be to *restore* variation in the position of the near bottom 2‰ isohaline (and/or net Delta outflow) to conditions that would exist today without operation of the federal and state projects. The rationale for this alternative is that the Central Valley Project and State Water Project have a mitigation obligation which, if enforced, would require restoration to these conditions.
- *Forecasted level of diversions*: the goal would be to *maintain* the position of the near-bottom 2‰ isohaline (and/or net Delta outflow) at conditions that would probably exist if the levels of demand for exports via the project pumps and in-Delta diversions continue as forecasted to 1995. This alternative would illustrate the response of biological resources if the regulatory agencies take no action.

1860-1900: The alternative goals outlined above reflect the goal of protecting or restoring *existing* tidal marshlands, but they do not reflect the separate goal to restore or create *additional* amounts of tidal marshland to recover the values and functions lost when most of the historical tidal marshlands were reclaimed for agricultural prior to 1900. Therefore, this goal is designed to restore key wetland resources to levels that existed prior to the large-scale reclamation; this goal and the others for aquatic resources are not mutually exclusive.

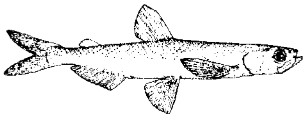
SUMMARY

Estuarine scientists and managers of the San Francisco Bay/Sacramento-San Joaquin Delta estuary recommend development of salinity standards for different periods of the year to be used in conjunction with flow standards. An appropriate index upon which to base salinity standards is an upstream position of the near-bottom 2‰ isohaline, averaged over the period of interest, to provide a prescribed level of environmental protection. Selection of the appropriate average upstream positions — the salinity standards — should be based upon environmental goals and the development of predictive models that relate diagnostic responses of organisms and processes to the position of the near-bottom 2‰ isohaline, and associated Delta outflow.

When appropriate, other flow-related variables should be incorporated into the models. Existing data and information are adequate to make a first cut at defining upstream limits of the 2‰ isohaline consistent with a range of environmental goals. The downstream limit of the near-bottom 2‰ isohaline should not be controlled; variability in flow and, therefore, in salinity response should be encouraged. While such policies and practices can not guarantee recovery and maintenance of important living resources, they will increase the probability of restoring and sustaining populations of a number of the estuary's important estuarine species.



AN ESTIMATE OF THE HISTORICAL POSITION OF 2-PPT SALINITY IN THE SAN FRANCISCO BAY ESTUARY



AUGUST 1992

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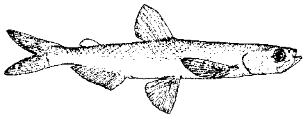
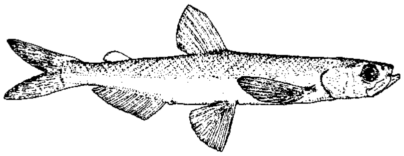
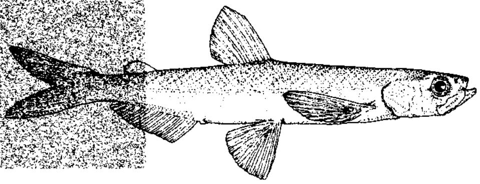
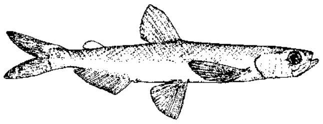
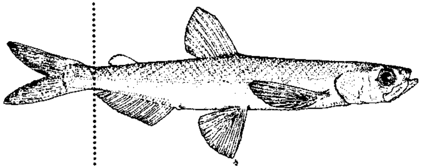


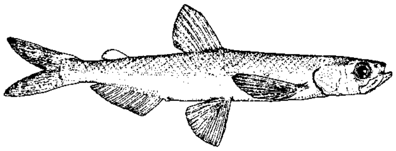








AN ISSUE PAPER PREPARED FOR THE FOURTH TECHNICAL WORKSHOP ON SALINITY, FLOWS, AND LIVING RESOURCES OF THE SAN FRANCISCO BAY/DELTA ESTUARY



INTRODUCTION

This report describes the development of a robust series of historical positions of the 2 parts per thousand (ppt) isohaline one-meter off the bottom of the estuary. This value is being considered for use in a salinity standard for the estuary. For simplicity we refer to this position as X_2.

X_2 was chosen as an index of the flow-salinity relationship because it is easy to measure and understand, and there is a long historical record that can be used to determine its value in the past. A number of descriptors of the habitat are related to X_2, including:

- Net outflow from the delta.
- The position of the estuarine entrapment zone.
- This issue is explored in the technical papers that accompany this report.
- The position of habitat for estuarine species.
- Salinity at other locations in the estuary.
- The mean depth, surface area, and volume between any two salinities.
- The amount of estuarine habitat exposed to export pumping.
- The proportion of inflow to the Delta that is exported
- The amount of estuarine habitat exposed to withdrawal within the Delta.
- The proportion of inflow to the Delta that is consumed in the Delta
- The amount of net upstream flow in the lower San Joaquin River.

Jassby's companion report describes relationships between X_2 and a number of estuarine responses, including the abundance or survival of several species of interest. Estuarine species in all trophic levels have responses that are strongly negatively correlated with X_2. It is important to recognize that the relationships could arise through the effects of any one or more of the variables listed above. Thus, X_2 is an *index* of habitat conditions, and can be used as a *predictor* in statistical models, but we do not assert that it is the direct *cause* of any of the responses observed.

OBJECTIVE

The objective of this report is to provide an estimate of the historical value of X_2 using the best available data, and to describe its relationship to Delta outflow.

GENERAL APPROACH

The principal source of data was from continuous monitoring sites maintained by the U.S. Bureau of Reclamation. These data, from sites at 56-92 km from the Golden Gate Bridge (Figure 1), consisted of daily mean salinity values at the surface, uncorrected for tidal variation. Our approach was to: select the appropriate surface salinity corresponding to 2 ppt at the bottom, interpolate between the monitoring stations to determine the daily position of that salinity, (which was assumed to represent X_2), calculate daily outflow from the delta; determine the best regression to fill in the gaps in the X_2 record, determine monthly and longer-term means and their relationships to flow, and compare the resulting values with independent grab sample data.

TIDAL EFFECTS

The data used were daily averages, and the original hourly data are no longer available in electronic format. The tidal cycle, with periods of about 12.5 and 25 hours, would have been aliased with daily mean data. This may have introduced an erroneous cycle into the data with a ca. 14-day period. To examine this, we used hourly data from the DWR Mallard Slough station and compared daily means of the raw data with data passed through a tidal filter (Godin filter obtained from Larry Smith, USGS). Differences in the daily means appeared small except when salinity was changing slope rapidly (i.e. reversing direction of change). Residuals had a significant autocorrelation with about a 2-week lag. However, monthly means of these values differed by less than 1% (Table 1). Since the ultimate use of the X_2 data is in monthly or seasonal means, the daily means are adequate. This may not be the case for other uses of the data.

Table 1. Effects of tidal filtration of salinity data on monthly mean values. Each value is a monthly mean of daily means of hourly raw or filtered data.

Year	Month	Unfiltered	Filtered	Percent difference
84	5	2.13	2.12	-0.2%
84	6	3.73	3.73	-0.1%
89	1	10.58	10.55	-0.3%
89	2	10.72	10.66	-0.6%
89	3	3.21	3.18	-0.8%
89	4	1.33	1.33	0.0%
89	5	4.23	4.23	-0.0%

SURFACE SALINITY VALUE

The relationship of surface to bottom salinity is not simple. At a bottom salinity of 2 ppt, surface salinity is constrained to 0-2. However, this range includes a vast part of the estuary. We have used the available surface and bottom grab sample data to estimate stratification and its dependence on flow for a bottom salinity close to 2 ppt.

We used the data for surface and bottom salinity for all points (DFG, USBR, and USGS data) where bottom salinity was between 1.5 and 2.5. The difference between surface and bottom salinity has a weakly quadratic relationship with delta outflow above 6300 cfs ($r^2=0.50$, Figure 2). The median value for outflow below that is 0.24 ppt. Therefore an equation for target salinity, i.e. salinity at the surface corresponding to 2 ppt at the bottom, is:

$$S_2 = \mathrm{MIN}\,\{\,(\,-0.46 + 1.87\,LQ - 0.33\,LQ^2\,),\, 1.76\,\}, \qquad (1)$$

where S_2 is the surface salinity where the bottom salinity is 2, and LQ is the $\log_{10}$ of freshwater outflow into the estuary.

However, using this equation presented serious problems at high flows. The principal problem was that at high flows the target surface salinity was so low that its slope with distance was low as well, since the salinity in the rivers is about 0.1-0.2 ppt. This meant that the predicted position of 2 ppt became excessively sensitive to thc stratification. The second problem was that at high flows, 2 ppt at the bottom was downstream of the seaward continuous monitoring station (56 km), so its position could be determined from these data only by extrapolation, further exacerbating uncertainty regarding stratification.

We therefore decided to use the median value of surface salinity for 2 ppt at the bottom, and to use flow to estimate X_2 below 56 km. As we will show, this resulted in a robust estimate of X_2 when compared with the grab data.

INTERPOLATING THE SALINITY DATA

The objective of this exercise was to determine a transform of the data that would linearize the salinity-distance relationship at around the 1-3 ppt range, then interpolate to get X_2 for each date. Linearizing was considered necessary for interpolation, especially for those time periods when data were missing from stations in the required salinity range. Data for this was from the USBR continuous monitoring data at 6 locations, with the DWR Mallard Slough station (75 km) replacing the 77 km station in 1984-1990 (Figure 1). The 77 km station apparently was not in operation after 1978.

Richard Denton suggested an exponential formulation for salinity at a given point vs. flow. Dave Peterson suggested that the volume of estuary upstream of a given station, to where tidal influences cease, is an important variable in examining the distribution of salt. Presumably the reason is that longitudinal dispersion depends on the size of the tidal prism upstream of the sampled location.

Upstream volume was determined by trapezoidal integration of cross-sectional areas of the estuary determined from nautical charts at approximately 1-km intervals. A more-or-less arbitrary upstream end of the distribution was set on the San Joaquin River at the mouth of the Mokelumne, at 112 river kilometers, and at 100 km on the Sacramento River. Qualitative sensitivity analysis showed that the exact value made little difference to our ability to linearize the data.

By solving a steady-state diffusion equation we get the following relationship:

$$\ln (S - S_r) = \ln (S_o - S_r) + a\, Q\, x / (K_x\, A). \qquad (2)$$

where S is salinity, S_r is river salinity, S_o is ocean salinity, a is a constant over distance (which may vary over time), Q is freshwater flow, x is distance up the estuary, K_x is the longitudinal dispersion coefficient, and A is cross-sectional area.

We made several simplifying assumptions to (2). Upstream salinity S_r is small (ca. 0.1 ppt), so it is eliminated for simplicity. Q and oceanic salinity are assumed constant. K_x is believed to vary as the volume upstream of the sampling station, so it is replaced by volume. Then the equation to be fitted for a given date is:

$$\ln (S) = b + c\, x / (V_x\, A) \qquad (3)$$

where b and c are constants and V_x is upstream volume as defined above. In fitting the data to this equation, it became apparent that including the cross-sectional area A made the fit worse, perhaps because K_x varies inversely with A. Therefore this was dropped from the equation. The resulting curves are illustrated by a random sample of data from individual dates in several years (Figures 3 and 4).

We interpolated log (S) vs. x/V_x for each date in the sample series from USBR, which ran from 1 October 1967 to 30 November 1991. In some cases we extrapolated, but did not do so beyond 5 km from the nearest station. Of a total of 8827 days in that series, we were able to obtain 7794 values from the interpolation. In most cases missing data arose either because X_2 was downstream of 56 km, or because gaps appeared in data from a critical station. Data were interpolated separately for the Sacramento and San Joaquin Rivers. Data from the Sacramento only were used when the value fell below 83 km (since there was a station at 81 km in the Sacramento, at about the confluence of the two rivers); when it was upstream of that point the two values were averaged. Given that there was only one station on each river upstream of the confluence, these data should not be used to make statements about the differences in salt distributions in the two rivers. The result of this calculation was an interpolated value of X_2 for the 7794 dates.

DAILY DELTA OUTFLOW

The DWR Dayflow model calculates the water budget of the delta on a daily basis using a combination of gaged and ungaged inflows, consumption and precipitation within the delta, and measured export flows. The biggest uncertainty, particularly at low delta outflow, is consumption within the delta. The value used until recently has been a long-term mean for each month, with daily values adjusted to vary smoothly from day to day. However, actual delta consumption varies from year to year depending on cropping patterns and precipitation.

DWR has computed revised estimates of delta consumption on a monthly basis. We corrected the daily Dayflow values by replacing the gross delta consumption with the revised monthly values, converted from acre-feet per month to cfs. We did not attempt to smooth the values.

The log (base 10) of corrected outflow was used in all of the models described below. There were a few days on which net outflow was negative, and the log of a negative number does not exist. We inspected the X_2 data for those dates and found that delta outflow was being underestimated. We therefore set the value of log outflow for those days to 2.5, equivalent to a minimum outflow of 316 cfs.

FILLING IN THE GAPS

Numerous gaps existed in the data set. Nearly all of these occurred when X_2 was clearly downstream of the westernmost station (Martinez at 56 km), so X_2 could not be determined reliably. We filled in these gaps using a regression model of the longest unbroken data record, 1975-77, comprising over 1000 data points.

Numerous models have been used to predict salinity or the position of a salinity value from flow, most involving some sort of averaging or lag term. Alan Jassby examined a number of models containing autoregressive terms and lagged flow terms. The best model was also one of the simplest: X_2 on any day is a function of log outflow on that day and the previous day's value of X_2. This is an autoregressive model of lag 1 with an additional independent variable (log outflow). The model is:

$$X_2 (t) = 10.16 + 0.945\, X_2 (t-1) - 1.487\, LQ\ (t)$$

where X_2 (t) and X_2 (t-1) are the 2 ppt positions at time t and t-1, respectively, and LQ is the $\log_{10}$ of the revised net delta outflow (daily mean, cfs). The R^2 was 0.986, and the standard error of the regression was 1.32 km for predicting each value using the previous predicted (rather than actual) value.

This model has the disadvantage for filling gaps that the value at any time depends on that from previous times, so any errors could accumulate as the prediction moved further into the gap. To test for this in the worst case, we calculated a regression prediction for every day in the series, setting the starting point to equal the first measured value. The resulting prediction had an error with a mean of 0.06 km and a standard deviation of 3.54 km (this is equivalent to the standard error of the regression). Note that the predicted values are computed using no knowledge of actual X_2 except for that on the first day. Figure 5 shows the time course of the residuals from this regression. Although there is some apparent autocorrelation present, the residuals are otherwise well behaved, with an approximately normal distribution and no time trend.

The best regression based on flow alone, which included 29 lag terms in a polynomial lag arrangement, had a standard error of the regression of 2.93 on the original set of data used to establish the model (A.D. Jassby, pers. comm.). However, this model was much more cumbersome to use than the AR model, and did not use the available X_2 values at each end of the gaps.

Tidal range lagged 7 days was slightly correlated with the residual from the AR model. However, the r^2 value is small (<5%) except when the predicted value is between 60 and 70 km, when r^2 is 12%. Since this would provide little overall improvement to the model, and since we will use monthly or seasonally averaged data, we did not pursue this further. In addition, some of this tidal signal could arise from the use of daily mean data (see above). The gaps in the data record were filled in by using the autoregressive model stepwise to predict each value in turn. However, this generally resulted in relatively large jumps in X_2 from the last value in each gap to the next measured value. In principle each X_2 value should be equally correlated to the succeeding value as to the preceding value. Therefore we forecast the data in the gaps in both directions: working forward from the last known value, and working backward from the first known value following the gap. Each value was then calculated as a weighted mean of the two calculated forecasts, the weighting factor being the time in days between the individual date and the previous or following known value. The resulting values varied smoothly at either end of the gaps.

Figures 6 through 11 show the values of the interpolated data, the values filled in as described above, and the values determined by the autoregressive model starting at the beginning of the entire time series. The latter regression reproduces general patterns well, but does not appear as useful in filling in gaps as the method described above. Of a total of 8827 days of data, 1033 days or 12% were missing (Figure 12), mostly because high flows pushed X_2 downstream of the most seaward monitoring station.

MONTHLY AND SEASONAL MEANS, AND RELATIONSHIP TO FLOW

Once the filled-in daily values of X_2 had been obtained, we calculated values by month and by season. Monthly values (Figure 13) varied with flow as has been observed before, and the regression line was close to that obtained previously (Williams and Hollibaugh 1987). The best regression to predict monthly X_2 was structurally the same as that for daily values:

$$X_2(t) = 122.2 + 0.3278\, X_2(t-1) - 17.65\, \mathrm{LOG}\, [Q_{out}(t)], \quad (3)$$

where t is now in months. The R^2 value was 0.96 and the standard error of the estimate was 2.30 km. The standard error of the estimate for predicting the entire time series from flow and the first value was 2.45 km. This regression has a tight fit with no obvious outliers or departure from linearity (Figure 14). If the lag term in X_2 is removed by assuming steady state and setting $X_2(t) = X_2(t-1)$, the slope of the relationship with log (Q_{out}) is -26.3.

Although there may be a slight nonlinearity in the response of X_2 to flow in Suisun Bay (around 60-75 km), using a cubic spline curve to fit the data did not improve the predictability substantially (A.D. Jassby, pers. comm.). Therefore, the linear formulation is the best estimate of the relationship between flow and X_2.

Seasonal mean values (Figure 15) show somewhat lower variability in summer (July-September) than the other seasons. Except during droughts, when X_2 has been high in all seasons, winter values generally fall below 70 km and autumn values below 80 km, while summer values tend to exceed 80 km most of the time.

The frequency distribution of monthly mean values (Figure 16) shows that upstream maxima are about the same throughout the year; that is, drought conditions can occur in any month. The median, other percentiles, and minimum of the monthly mean values are variable throughout the year, with highest values occurring in summer.

COMPARISON WITH PREVIOUS ESTIMATES

Grab sample data used in estimating stratification (CDFG, USGS, USBR) were also used in a comparison of the predicted daily mean X_2 with grab sample values. This comparison reveals a large amount of scatter and apparent bias at values of X_2 above about 80 km (Figure 17). Below 80 km the residual is not significantly different from 0 (t-test), while at or above 80 km it averages 2.9±0.6 km (mean ± 95% CL of the mean). The scatter may be largely due to the effects of tidal variation, while the bias could be due to the common practice of sampling while attempting to follow the high tide upstream.

Additional grab sample data were obtained from Jim Cloern (USGS, pers. comm.), and from the CDFG midwater trawl data. The Cloern data comprised bottom samples for salinity taken on a series of transects up the North Bay. To get the data in the correct format we interpolated linearly between adjacent stations, using only those series of samples from the north bay in which bottom data were taken. We obtained times for the same samples and used those in a tidal prediction program to get an estimate of tidal height. The time was increased by three hours to account for the difference in phase between the Golden Gate and the north bay, and the difference between high tide and slack flood; both of these were estimated from a book of tide tables.

Figure 18 shows the relationship between Cloern's grab sample data and daily X_2 data. The standard error of the estimate was 2.8 km, less than that for the predictions of X_2 from daily outflow, and the slope of the regression was not significantly different from 1 ($p>0.1$). The difference between the mean grab sample and X_2 data was not significant (paired-sample t-test, $p>0.1$). Note that the apparent outliers in Figure 18 generally occur at high tidal values, with the sign in a direction consistent with the sign of the residual. In other words, correcting for tidal stage at the time of sampling would have tightened the fit of these data further.

The CDFG MWT data consisted of monthly surface salinity values from about 85 stations during September-December from 1967-1990. Data were prepared by Steve Obrebski (SFSU, pers. comm.) for a different purpose. For each date, the value for each km of distance was either taken from the station data or interpolated linearly between the nearest stations. Data were then smoothed using a 9-point running mean, and the point with surface salinity closest to 2 ppt was selected (note that this should be slightly upstream of X_2). The relationship between this estimate and X_2 (Figure 19) had a slope not significantly different from 1 ($p>0.05$) and a standard error of the estimate of 4.0 km, provided the two points below X_2=56 km are deleted. These points are in a range of X_2 at which flow should produce significant stratification (i.e. $\log_{10}(Q_{out})=4.86$; see Figure 2). At this point, the position of a given surface salinity is a poor predictor of X_2. Note also that the horizontal difference between X_2 and the position of 2 ppt at the surface is lost in the variability of these data.

CONCLUSIONS

The current X_2 estimates provide a value that is easily determined from the data and whose residuals appear well-behaved. The differences between the best estimates and other values (i.e. previous estimates and grab sample data) are explainable. Thus this data set is probably adequate for setting a salinity standard.

The model could be improved in several ways that are probably not warranted now. First, the small but significant tidal effect could be included. Second, it is evident from Figures 6-11 that some hysteresis may exist in the signal; that is, the response to a declining flow differs from that to an increasing flow. Furthermore, there appear to be differences in response at high and low flows, to the extent that these are observable with the present data set.

Based on the regression of monthly X_2 to log outflow (Figure 13), the steady-state outflow necessary to maintain a given X_2 can be readily determined (Table 2). Moving X_2 downstream by 8 km from a point in this range requires about a 2-fold increase in outflow.

Table 2. Flow requirements for steady-state X_2 values from 60 to 110 km from the Golden Gate, based on regression in Figure 13.

X_2	Outflow, cfs	Water required, million acre feet/month
60	43000	2.60
65	28000	1.67
70	18000	1.08
75	12000	0.70
80	7500	0.45
85	4800	0.29
90	3100	0.19
95	2000	0.12
100	1300	0.08
105	840	0.05
110	540	0.03

An important consequence of the nonlinear relationship of X_2 to delta outflow is the asymmetry in water requirement implied by Table 2. A change in X_2 takes the same proportional change in flow at any initial position, but the actual quantity of flow can vary. For example, it takes 18,000 acre-feet of water per month to move X_2 downstream from 110 to 105 km, and 921,000 acre-feet per month to move it from 65 to 60 km. This has serious implications for management: keeping X_2 at precisely the position set by the standard will always cost less water than allowing it to move about that position. Since one of the recommendations of the workshop is to allow for variability, it is important that the standard be set in such a way as to prevent constancy of position.

REFERENCES

Williams, P.B. and J.T. Hollibaugh. 1987. A salinity standard to maximize phytoplankton abundance by positioning the entrapment zone in Suisun Bay. Phillip Williams & Associates Report No. 412-4

Figure 2. Difference between surface and bottom salinity for grab sample data for which bottom salinity was 1.5-2.5 ppt, vs. log net delta outflow averaged over previous 30 days. The regression line is a quadratic relationship with log Q_{out} *constrained to a minimum of 0.24 ppt.*

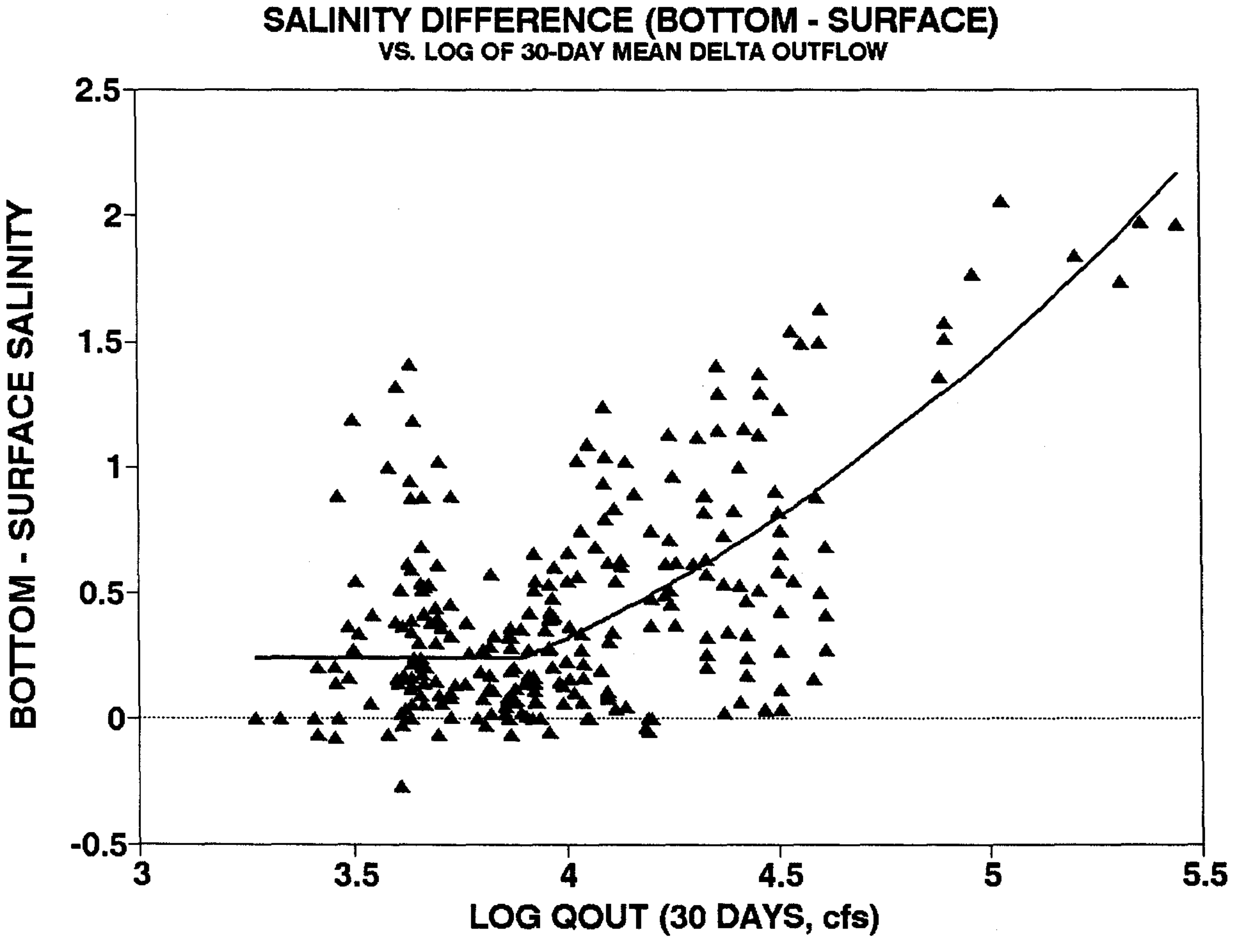


FIGURE LEGEND

Figure 1. This map of the upper estuary displays sites of the continuous monitoring stations (triangles) within the study area, and indicates hypothetical positions of a 2ppt isohaline measured at 5 kilometer increments upstream from the Golden Gate Bridge.

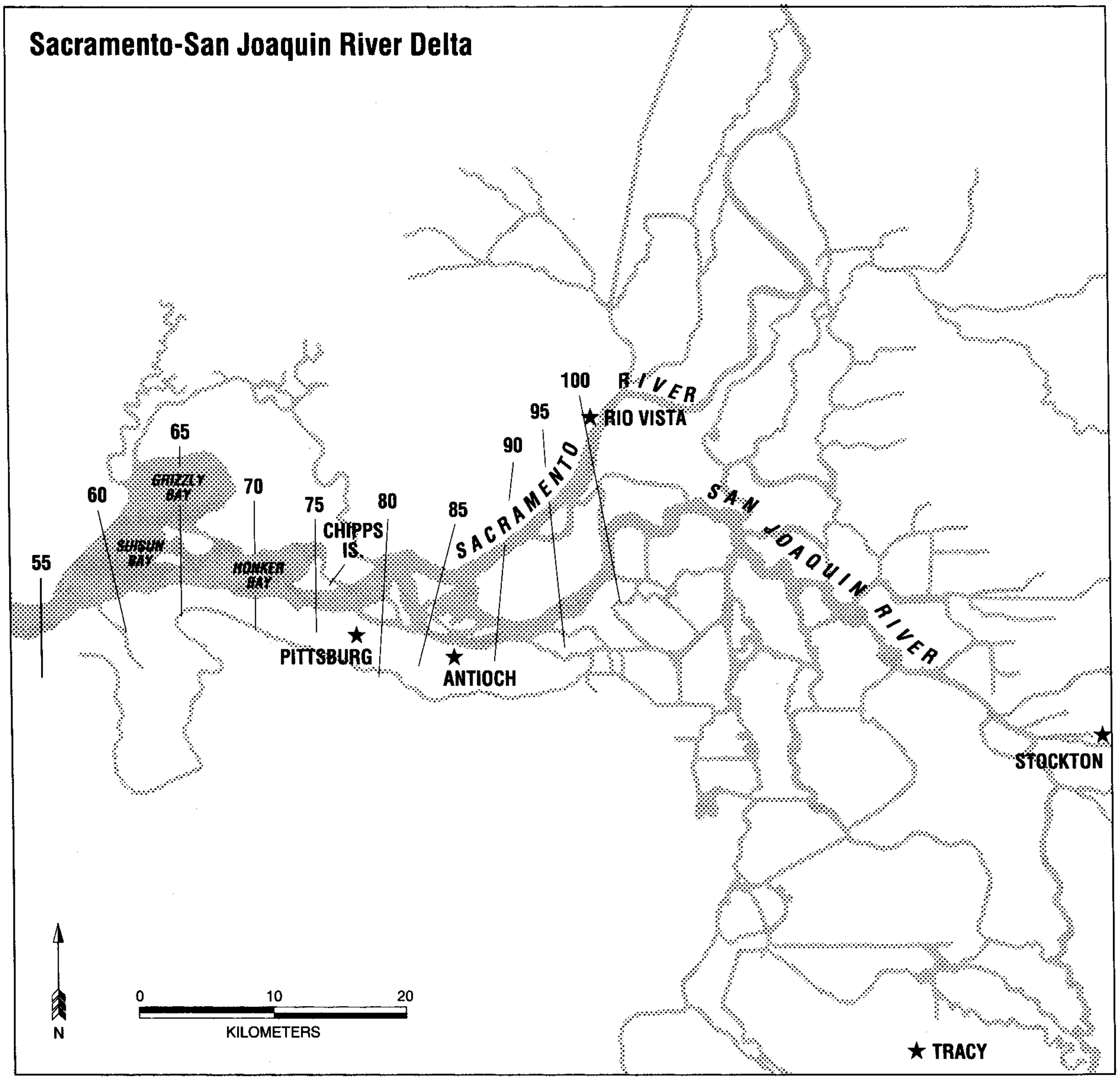


Figure 3. Representative continuous monitoring data used in interpolation. The data have been straightened out to the degree possible by log-transforming salinity and converting the distance measure to distance/upstream volume.

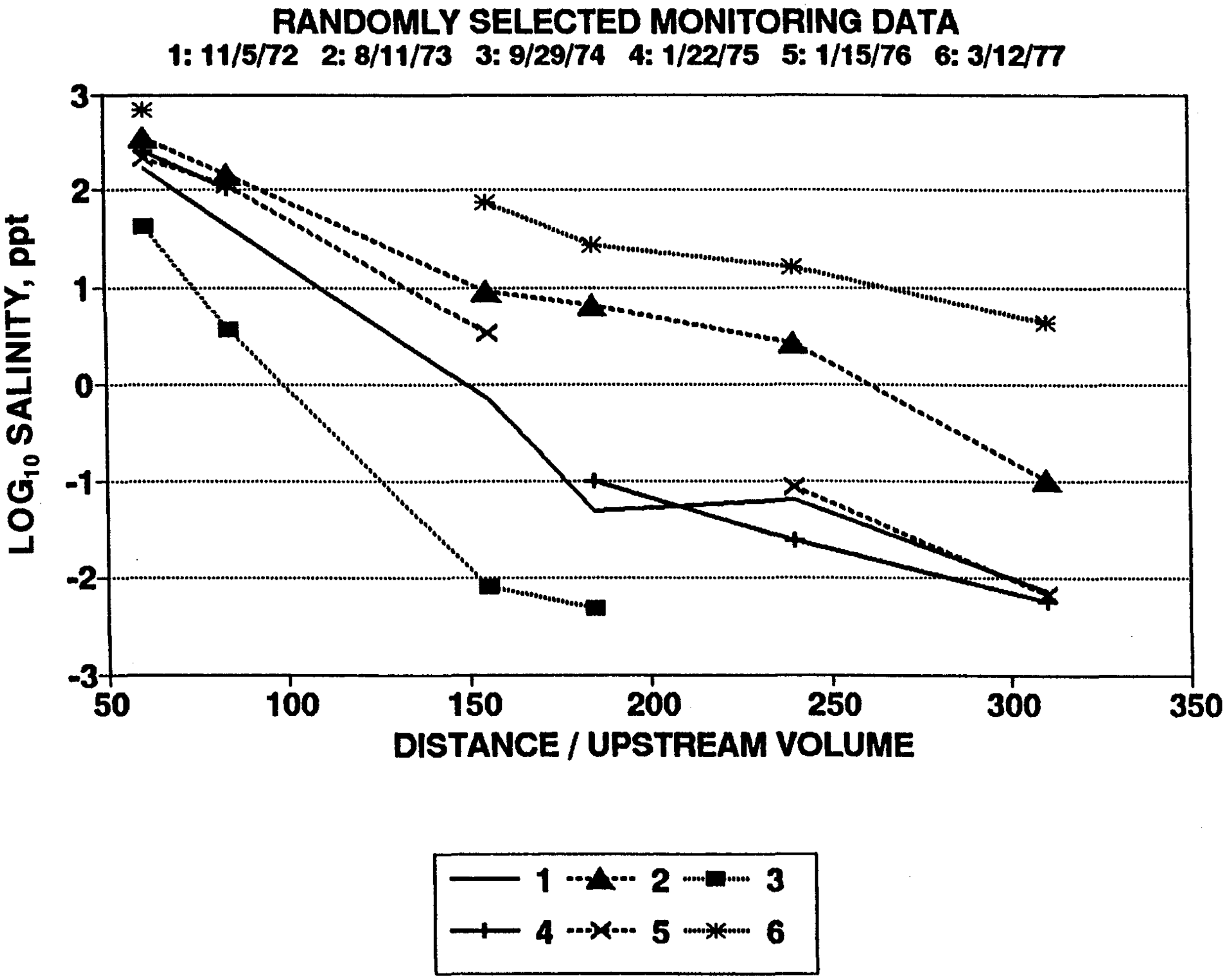


Figure 4. Another example as in Figure 2.

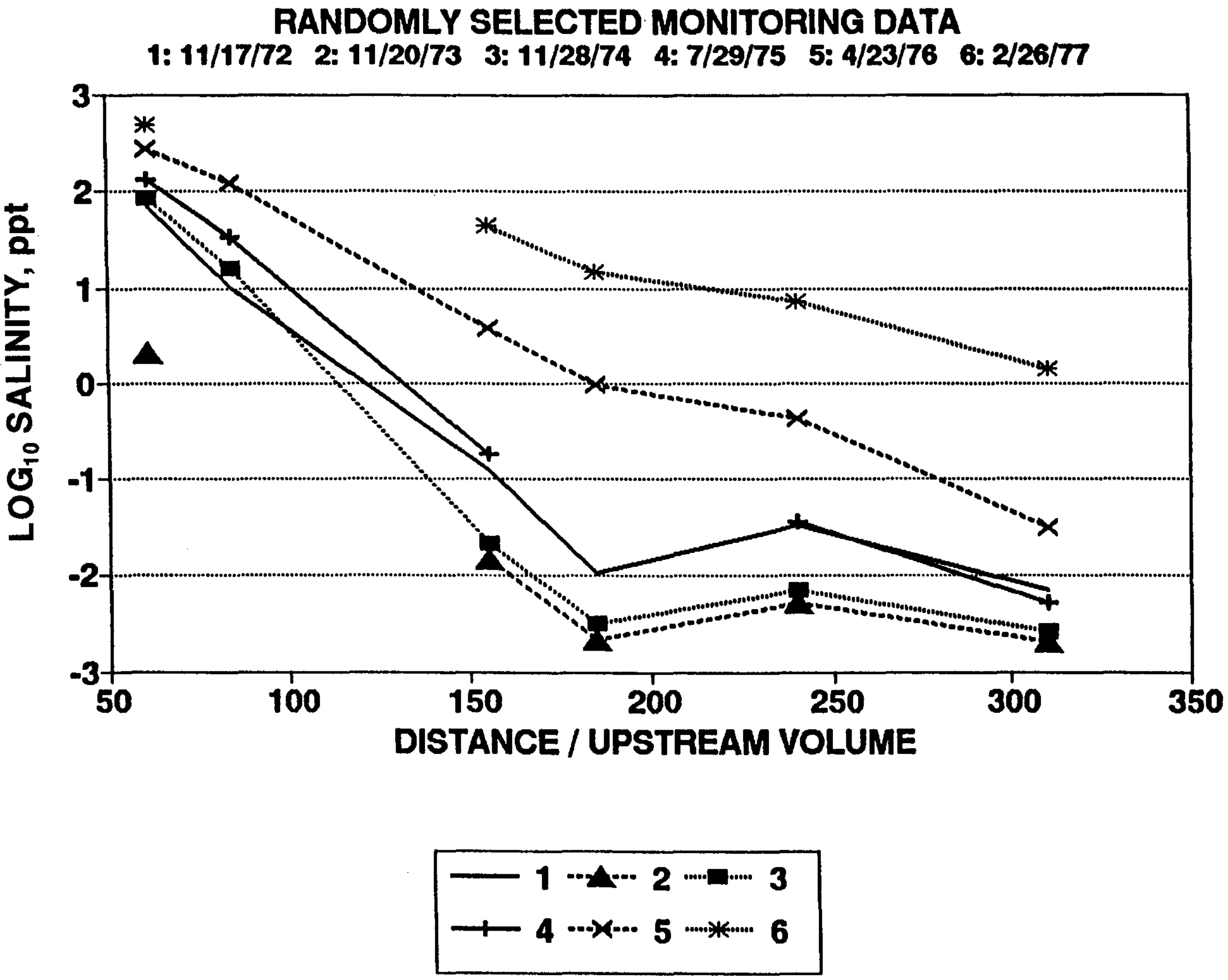


Figure 5. Residuals from autoregressive (AR) regression to predict daily X_2, calculated using the interpolated values.

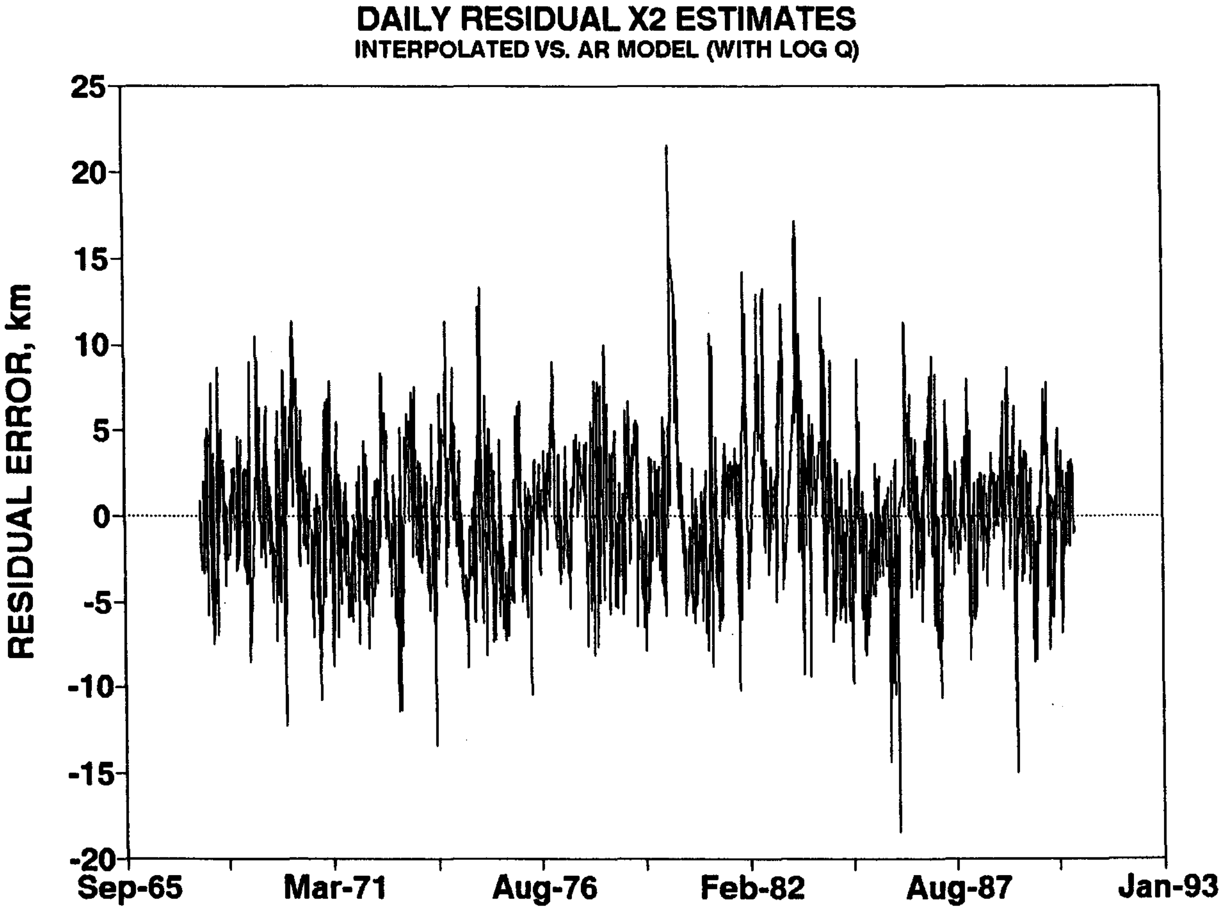


Figure 6. Time series of X_2 estimates, including the interpolated values, the filled values, and the values from the AR model.

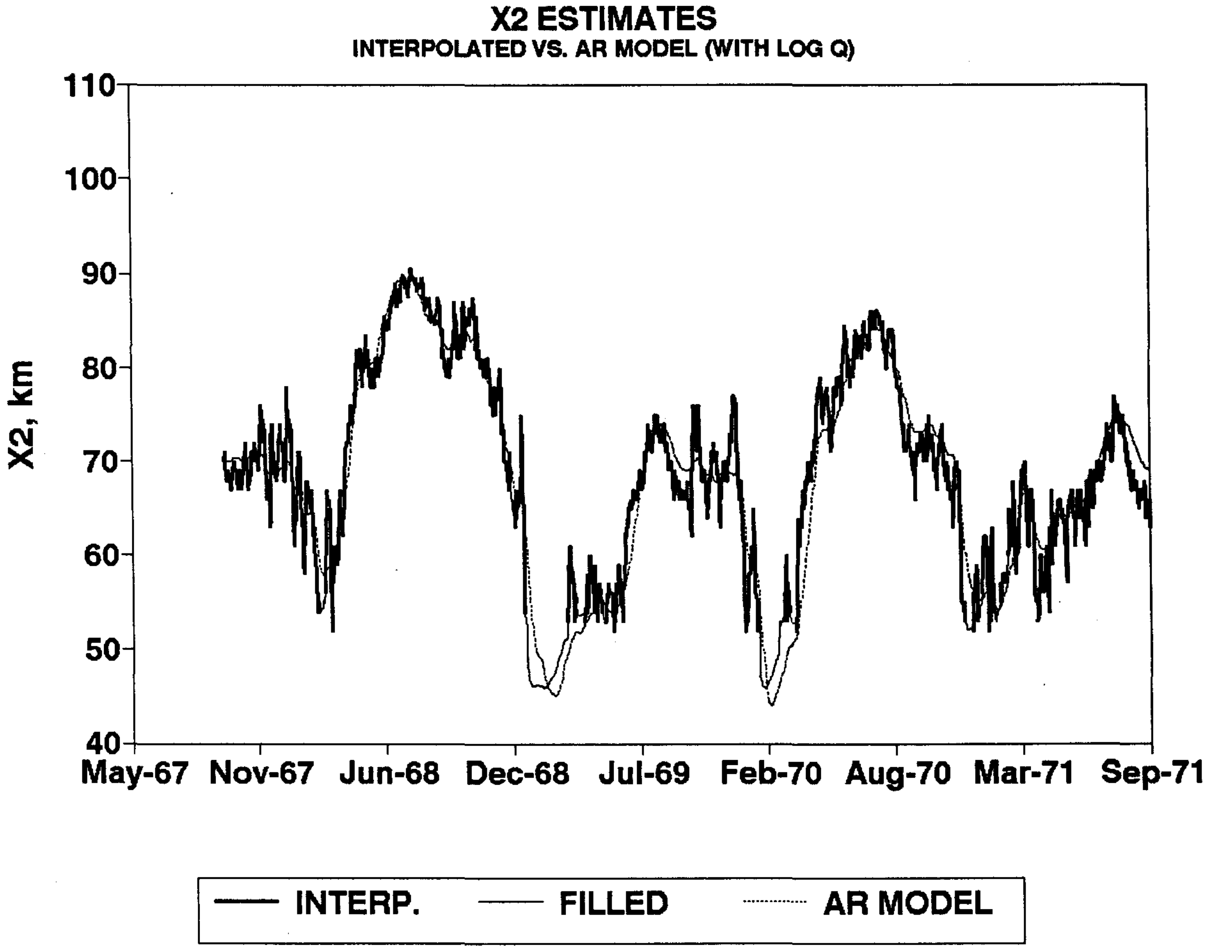


Figure 7. Time series of X_2 estimates, including the interpolated values, the filled values, and the values from the AR model.

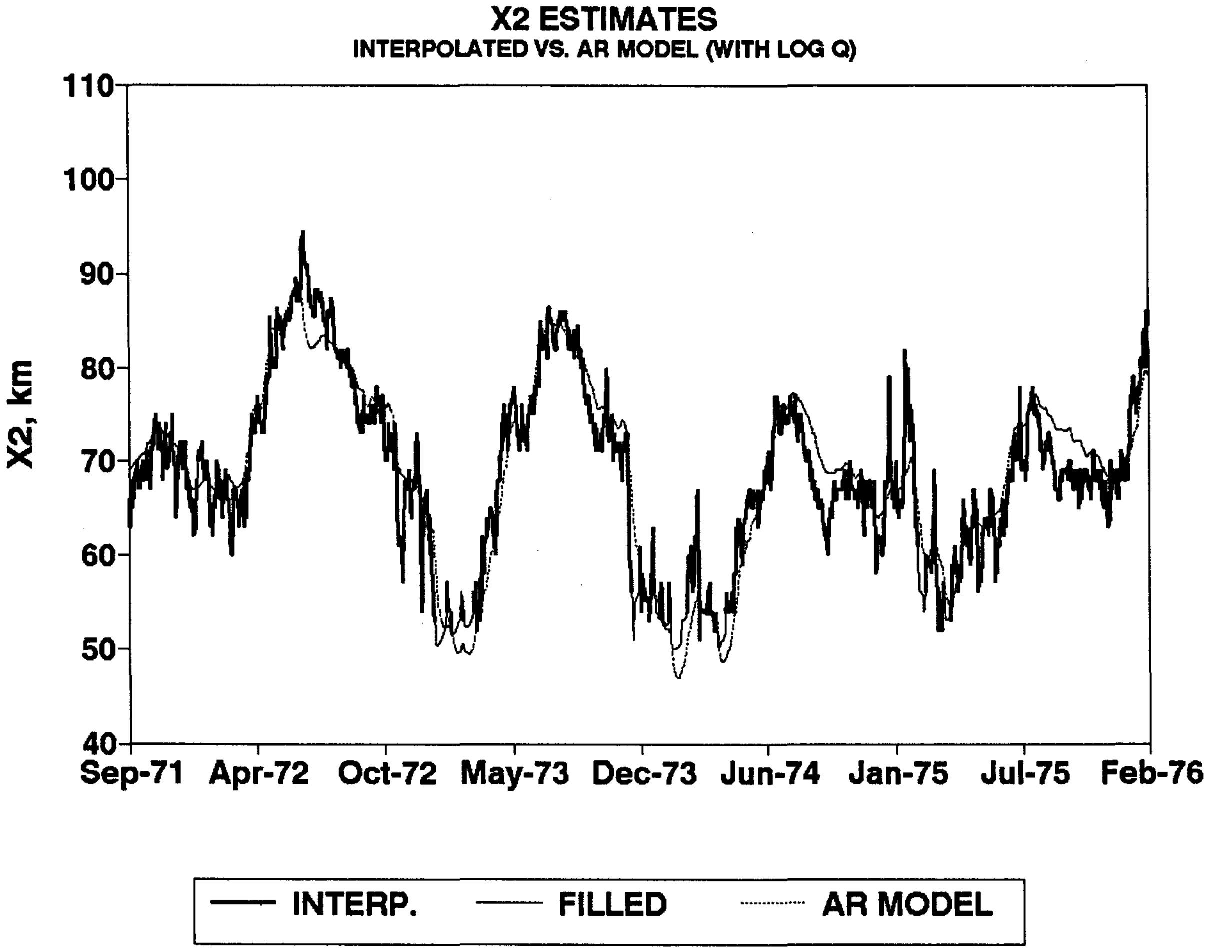


Figure 8. Time series of X_2 *estimates, including the interpolated values, the filled values, and the values from the AR model.*

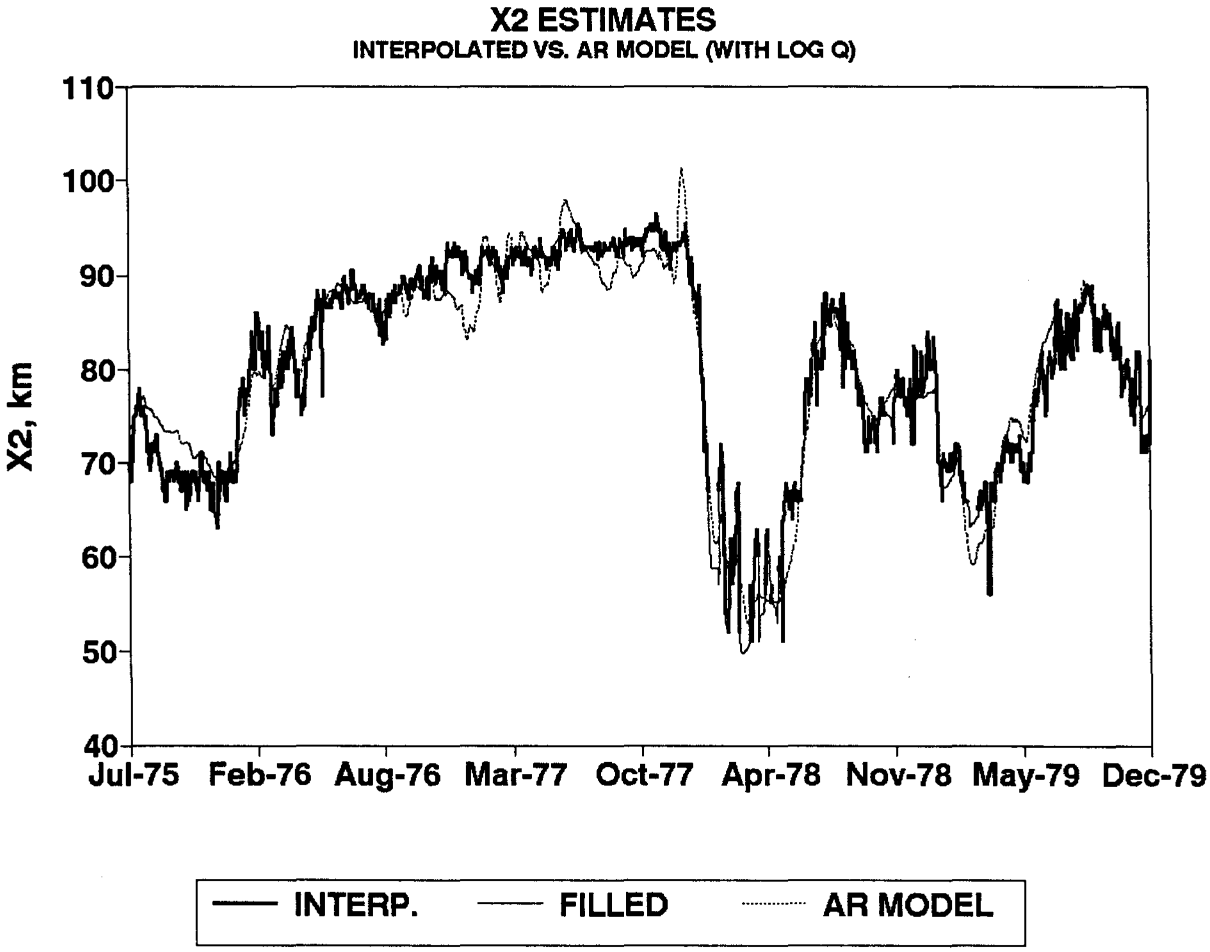


Figure 9. Time series of X_2 estimates, including the interpolated values, the filled values, and the values from the AR model.

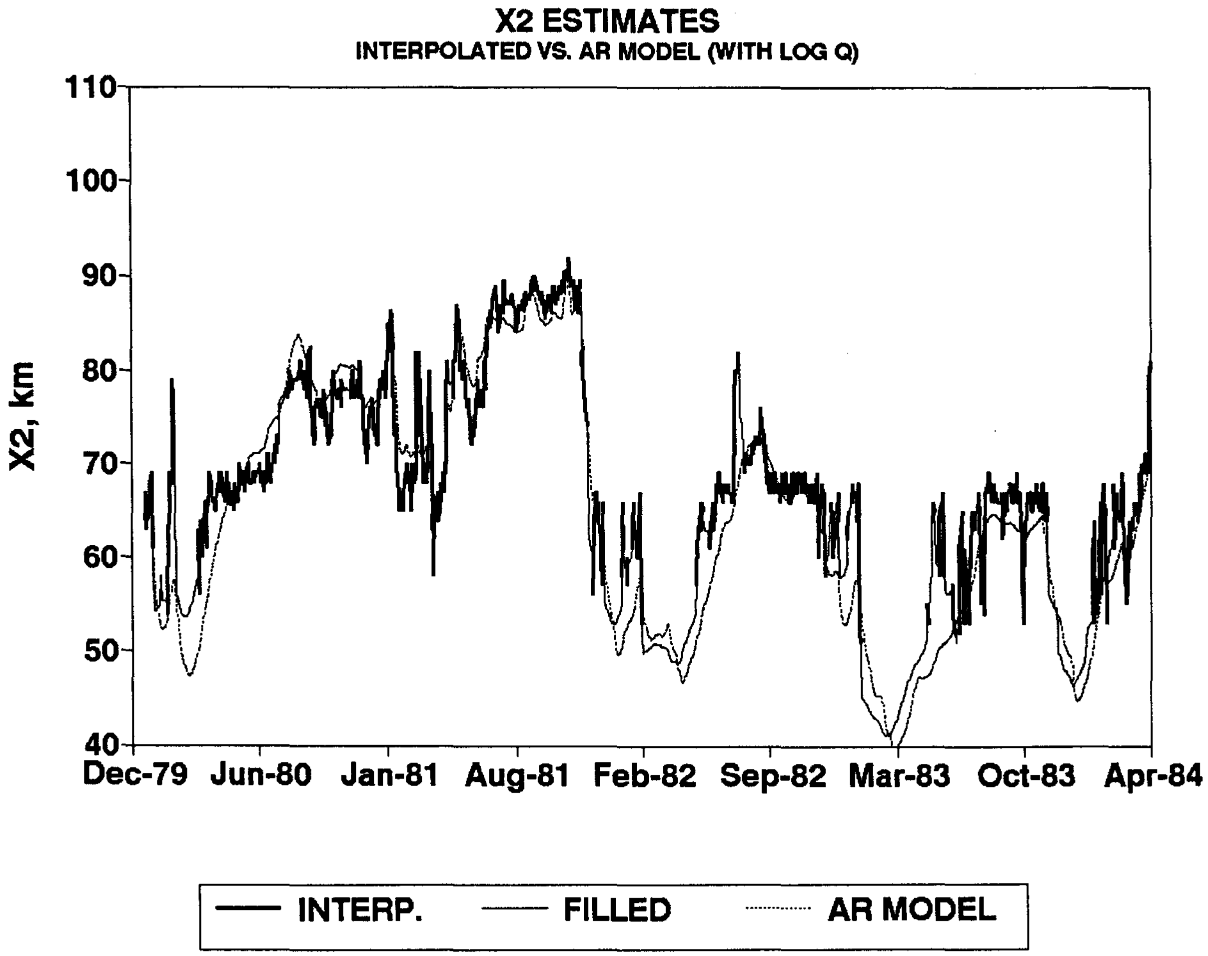


Figure 10. Time series of X_2 estimates, including the interpolated values, the filled values, and the values from the AR model.

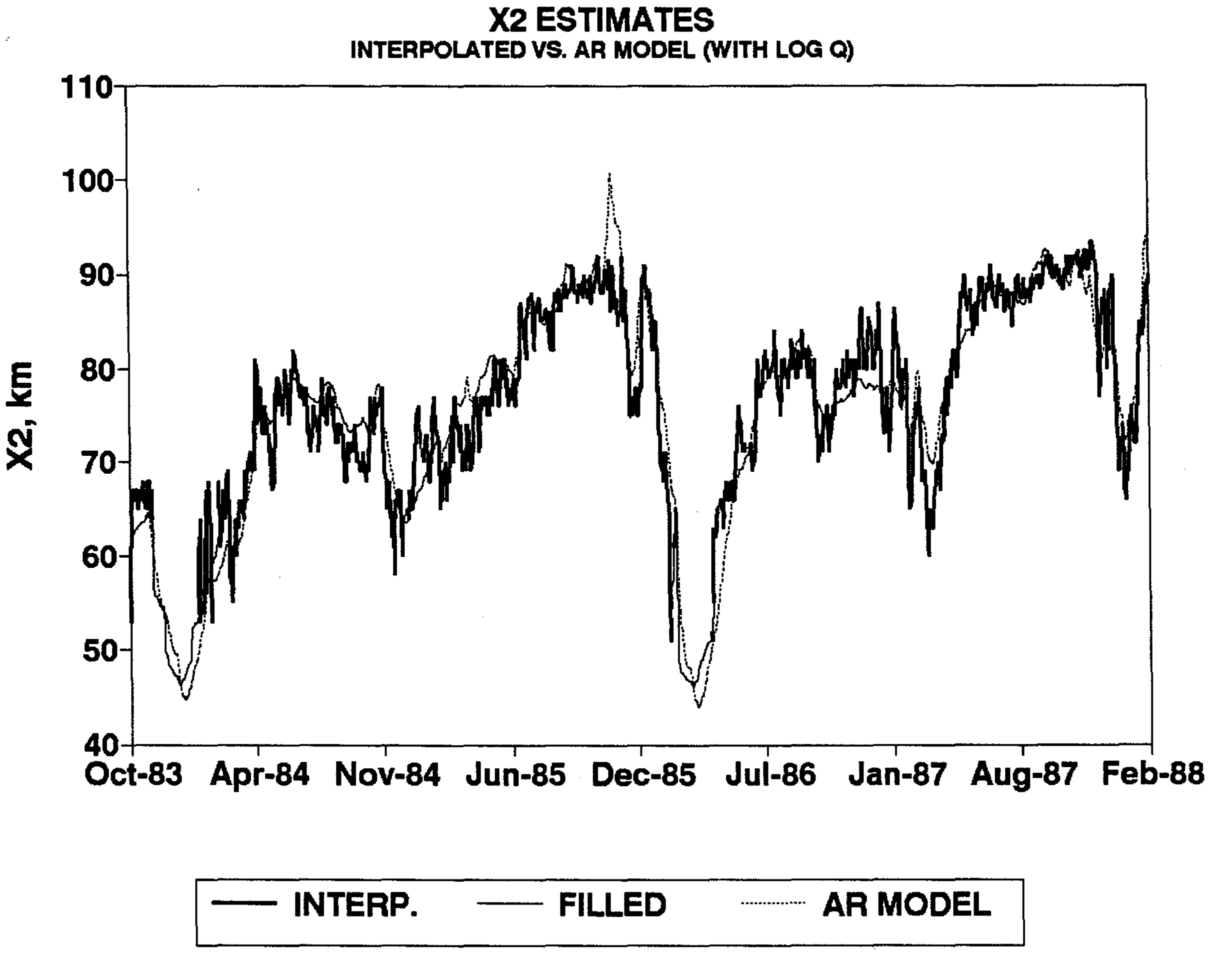


Figure 11. Time series of X_2 estimates, including the interpolated values, the filled values, and the values from the AR model.

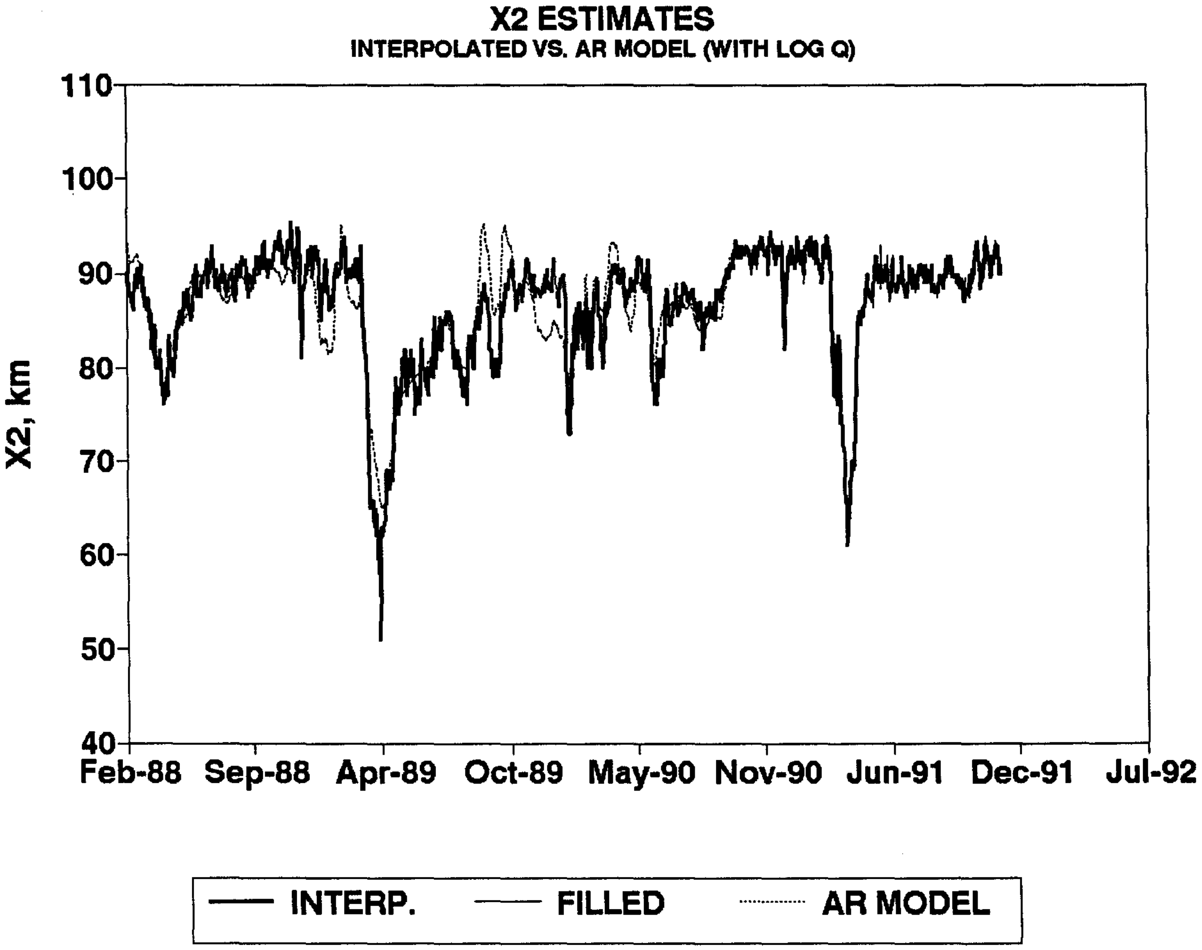


Figure 12. Frequency distribution of missing data in the interpolated series.

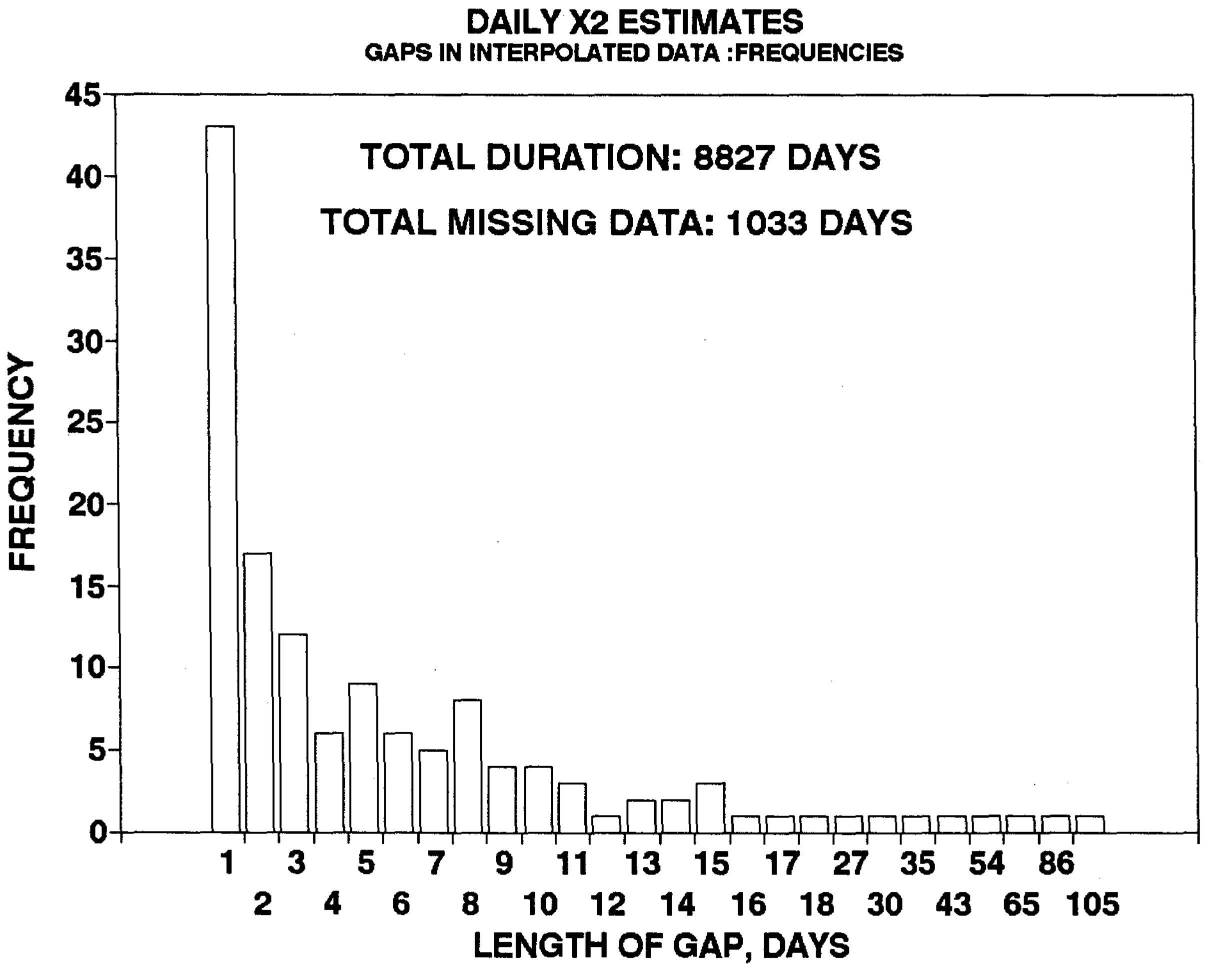


Figure 13. Monthly means of X_2 plotted against mean Q_{out} for the same month. Solid line, geometric mean regression; dashed line, equation of Williams and Hollibaugh (1987 Figure 23).

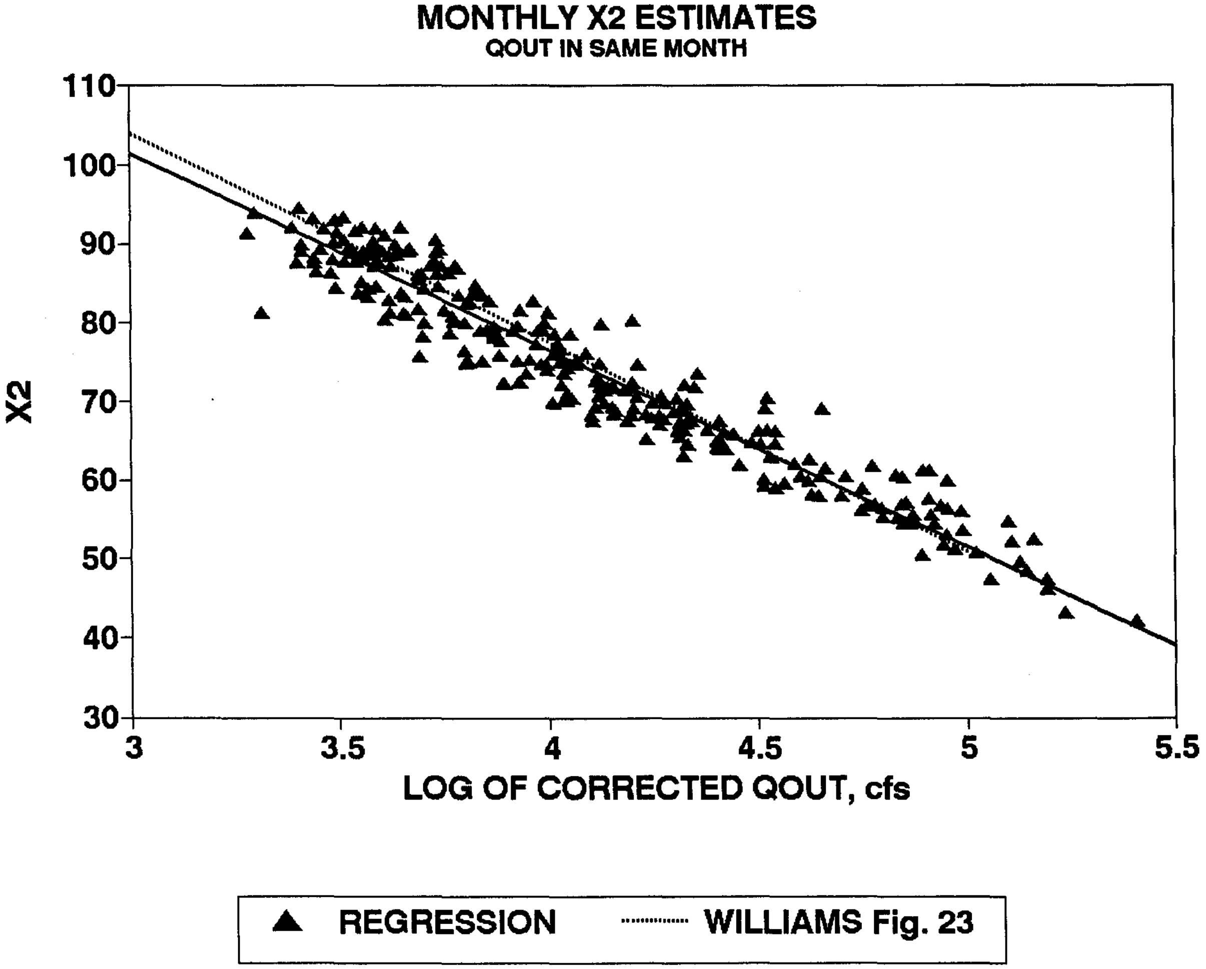


Figure 14. Monthly means of X_2 vs. values predicted from monthly AR model with flow (Model statistics shown).

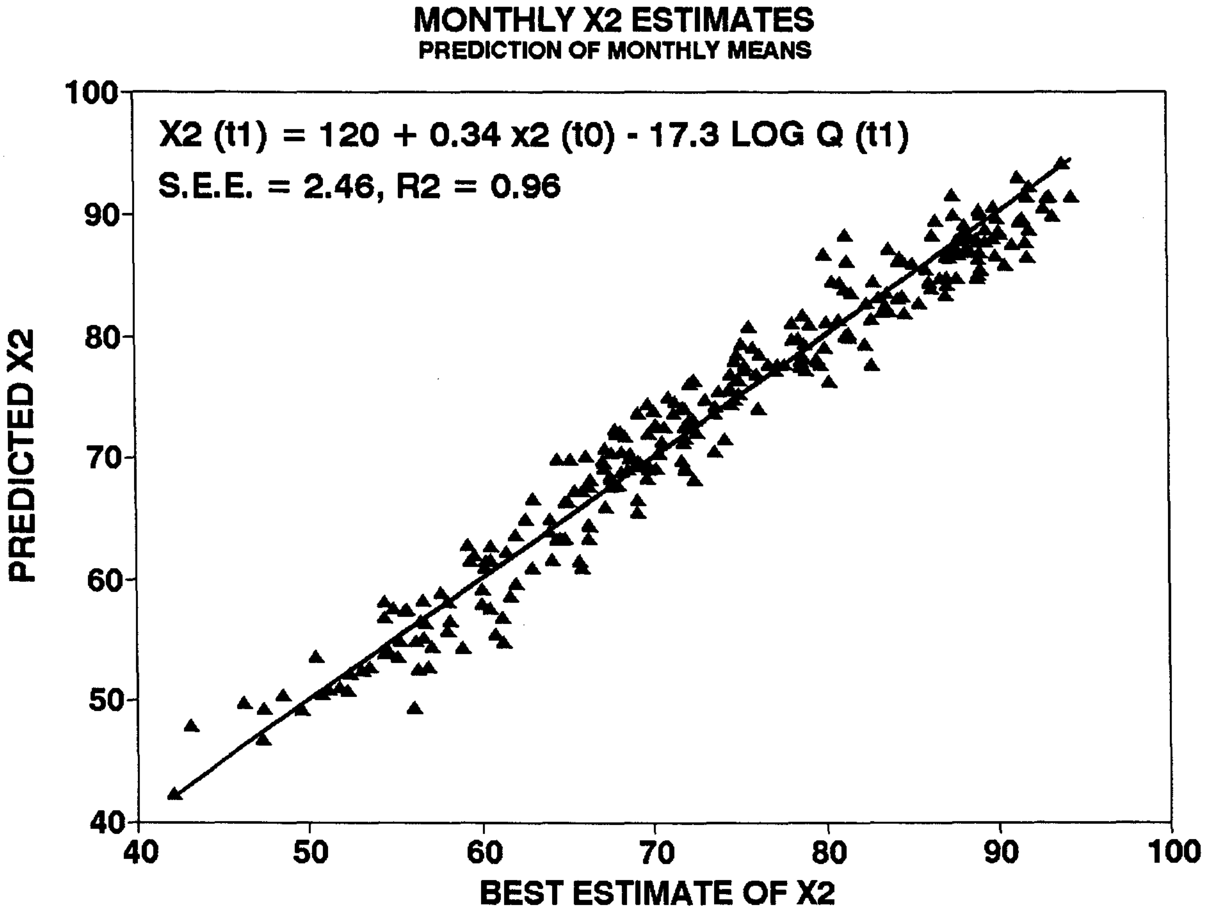


Figure 15. Seasonal mean values of X_2.

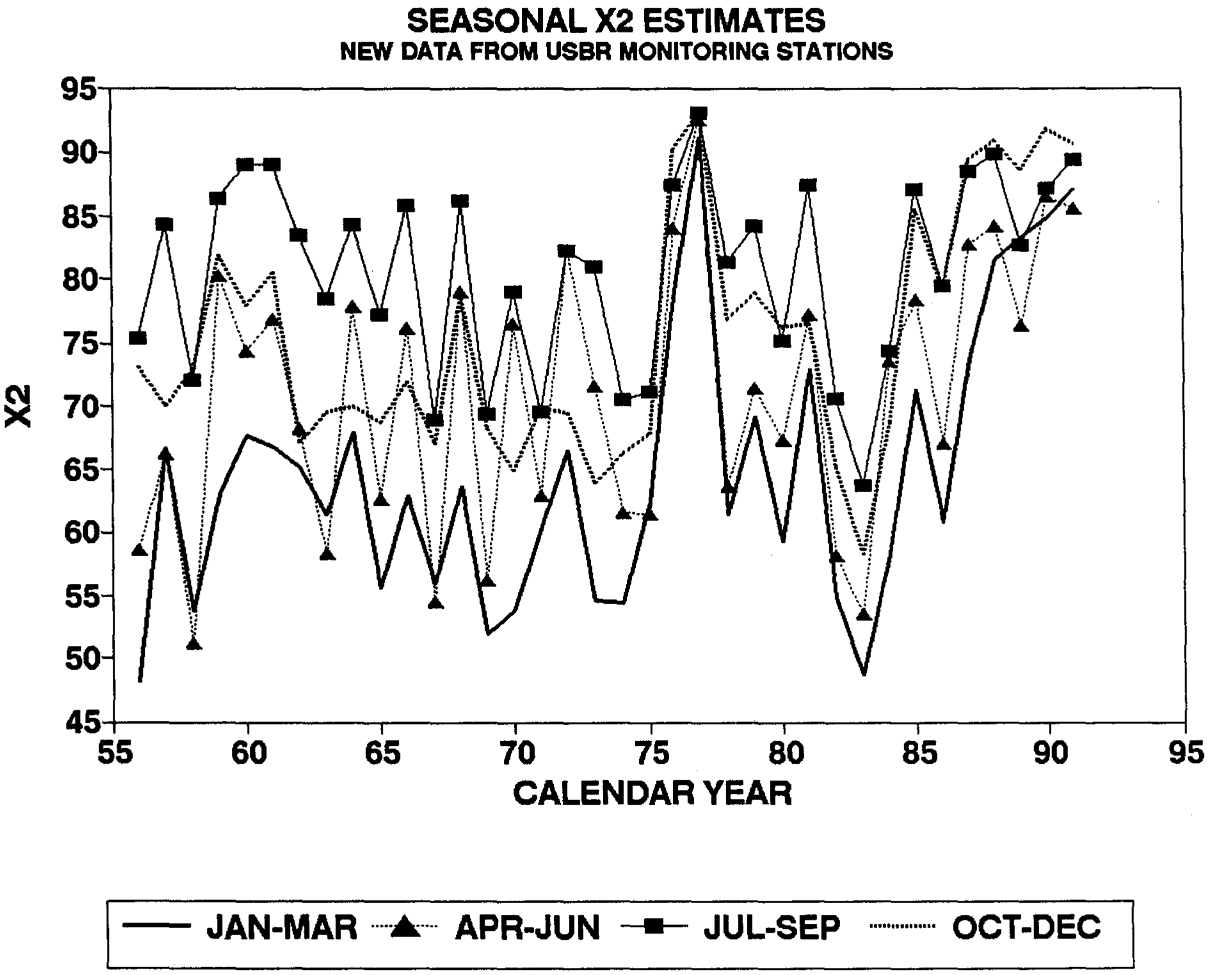


Figure 16. Maximum, minimum, and quartiles of the mean X_2 for each month during water year 1968-1991.

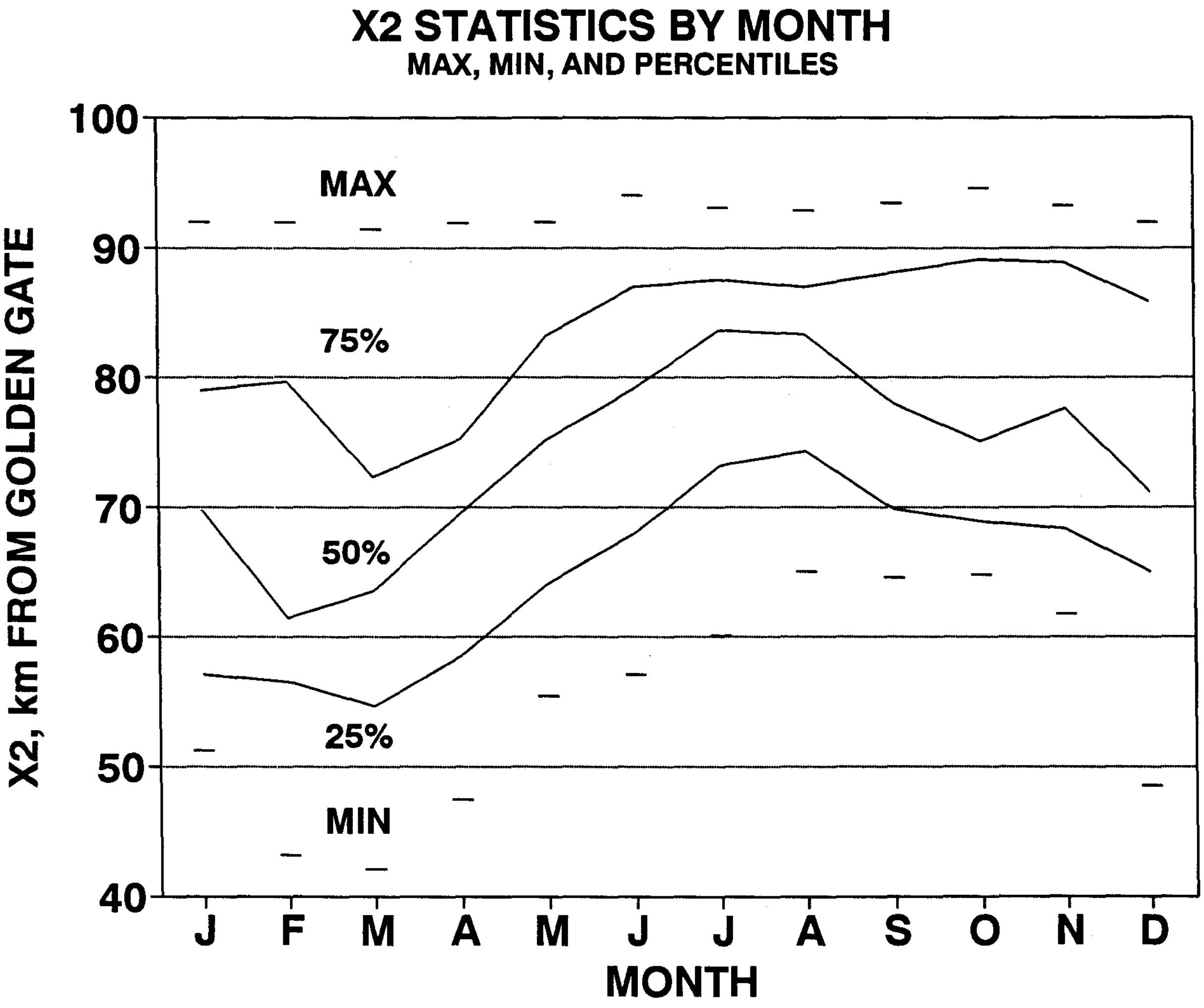


Figure 17. Analysis of residual of grab sample data vs. X_2 from interpolation. Grab samples were taken at salinity of 1.5-2.5, except for CDFG surface samples which were at salinities of 1.76 ± 0.3.

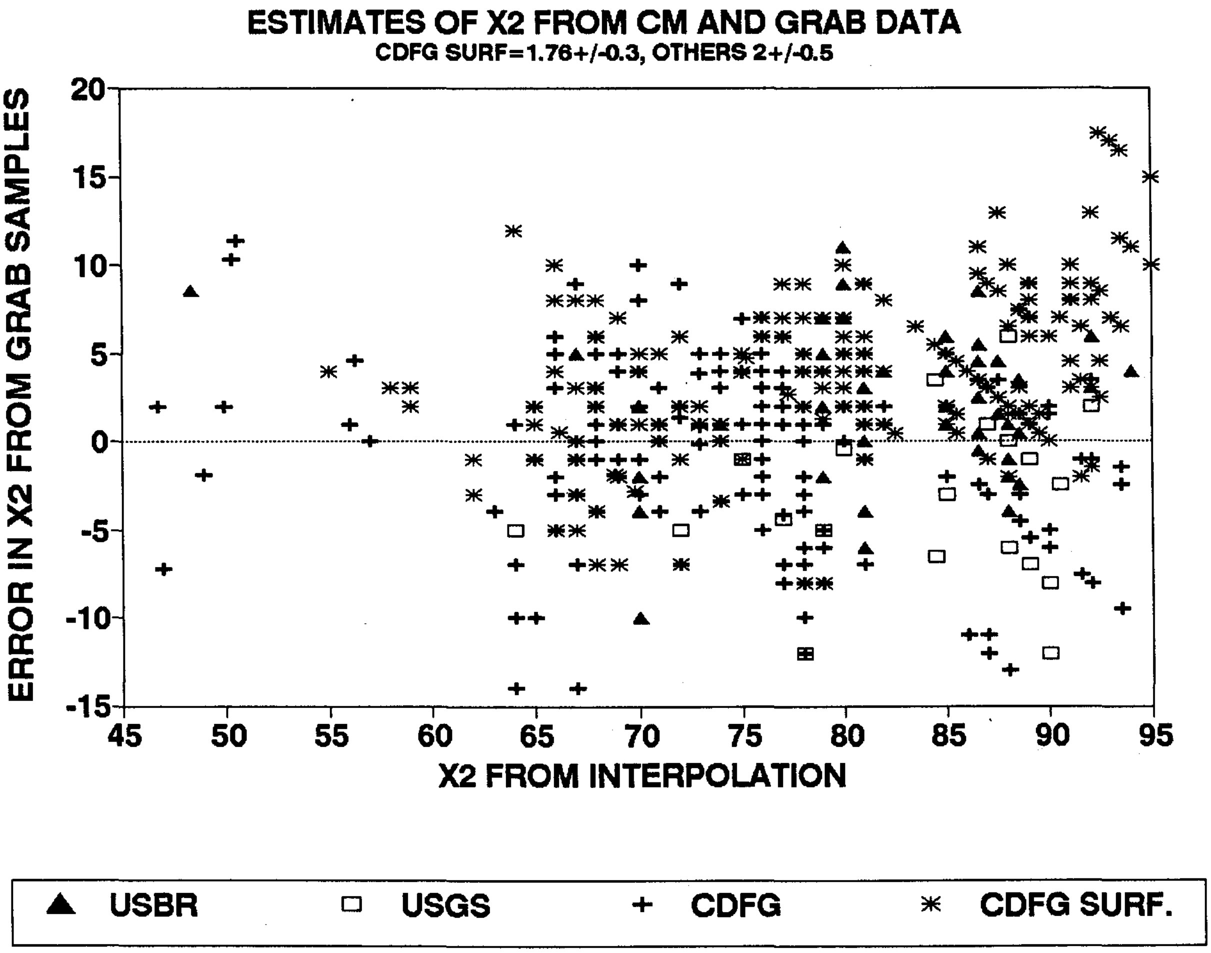


Figure 18. Grab sample data from Cloern (pers. comm.) vs. X_2 *data from interpolation. Numbers are tidal height estimated at time of sampling. Solid line, 1:1; dashed line, geometric mean regression.*

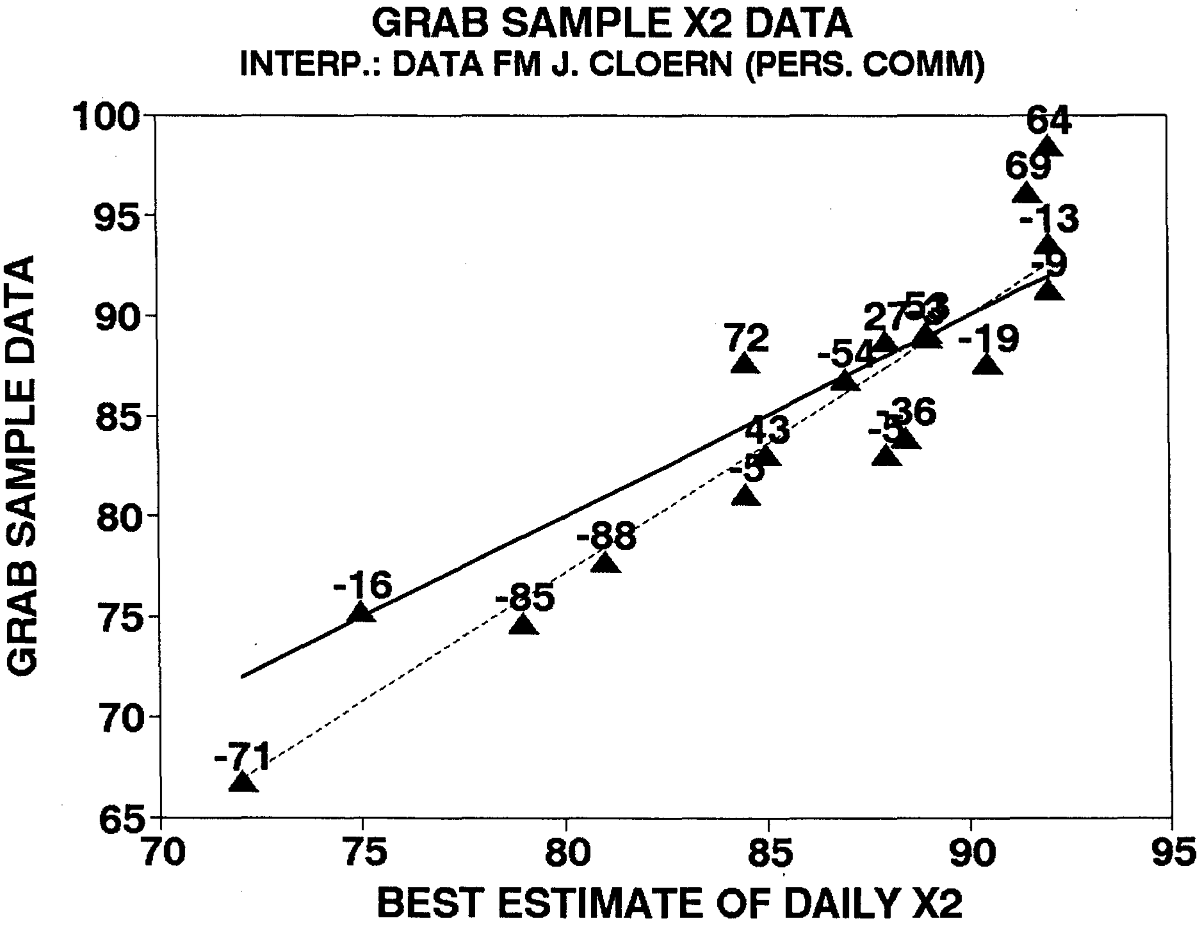
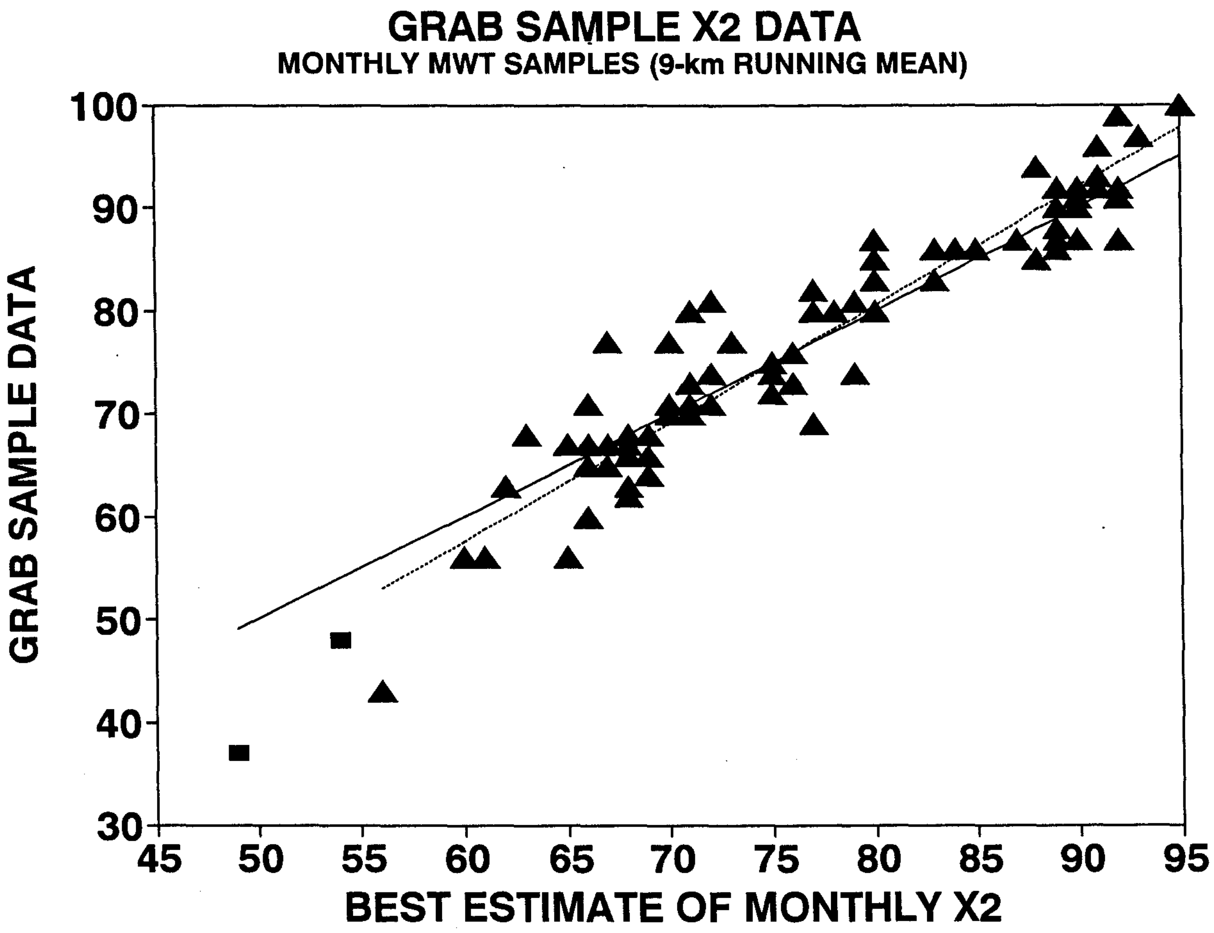
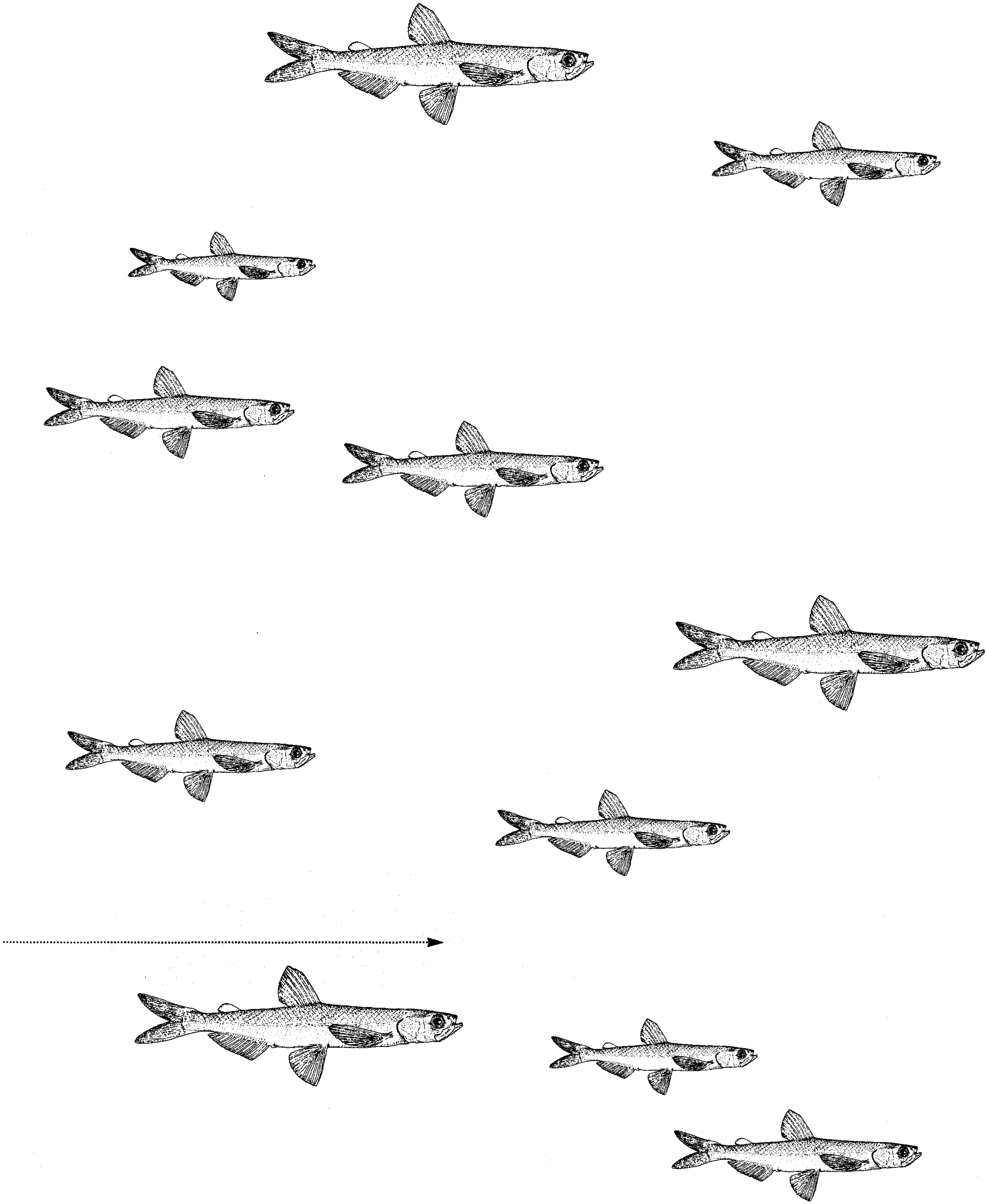
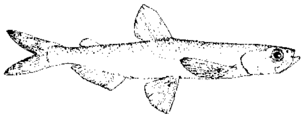


Figure 19. Grab sample data from CDFG MWT data set vs. X_2 data from interpolation. Solid line, 1:1; dashed line, geometric mean regression. The two points indicated by squares are below 56 km and are not included in the regression.





ISOHALINE POSITION AS A HABITAT INDICATOR FOR ESTUARINE RESOURCES: SAN FRANCISCO ESTUARY



AUGUST 1992

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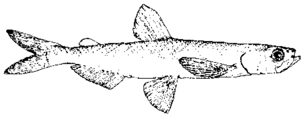
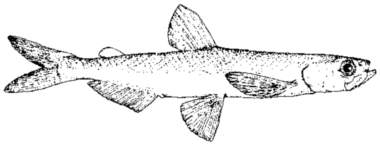
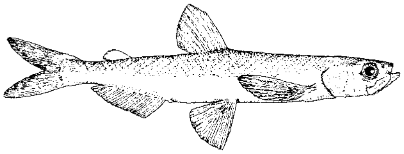
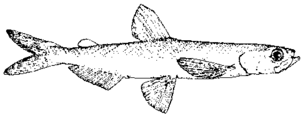
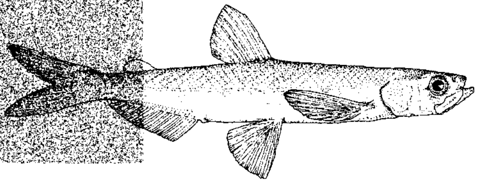
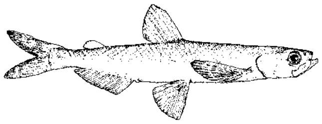


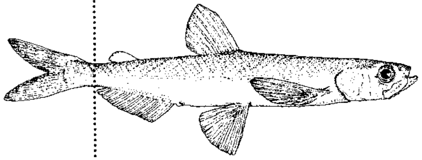


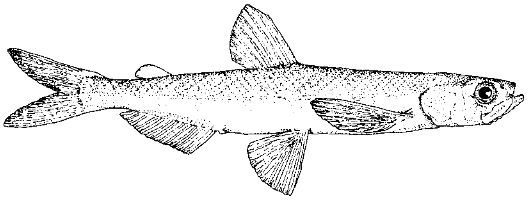




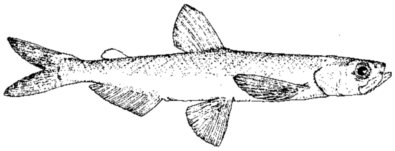








AN ISSUE PAPER PREPARED FOR THE FOURTH TECHNICAL WORKSHOP ON SALINITY, FLOWS, AND LIVING RESOURCES OF THE SAN FRANCISCO BAY/DELTA ESTUARY



INTRODUCTION

The term *habitat indicator* is used here to mean an environmental attribute that is well-defined and measurable, and that can be used to characterize the suitability of environmental conditions for a population, community or ecosystem. This definition corresponds to terminology used in the USEPA Environmental Monitoring and Assessment Program (EMAP) and the determination of useful indicators is indeed one of the major goals of that program. Habitat indicators are of particular interest for the San Francisco Estuary because of striking long-term trends and interannual variability in the levels of many estuarine populations. The simultaneous declines in many truly estuarine species suggest that they are responding to common stresses, and the question naturally arises as to whether one or a small number of habitat characteristics can serve as an index of the net effect of these stresses.

Salinity is of course a critical factor for estuarine organisms, and much of their distribution can be understood in terms of their salinity tolerances. But the salinity field also embodies other information, such as the size of freshwater inflows into estuaries and the location of turbidity maxima in which planktonic organisms and detrital particles may become concentrated. Simple characterizations of the salinity field are thus prime candidates for estuarine habitat indicators. Here, we examine the location of 2 ppt bottom salinity as a habitat indicator for the San Francisco Estuary. This particular position, denoted by $X2$ and measured as distance from the Golden Gate, was chosen in large part on the basis of data availability. X2 also has some ecological justification, however, as it is often associated with an important nursery area (Suisun Bay) and the location of an estuarine turbidity maximum.

The assessment of $X2$ presented here consists of two separate parts. First, we portray the empirical relationships between several biological resources in the San Francisco estuary and $X2$. As is widely understood, statistical relationships are not proof of causal connections, and it is not the intention of this report to suggest that $X2$ itself or, more generally, the salinity field controls biological resources in the estuary. Rather, the particular hypothesis investigated here is that $X2$ can serve as an index of those habitat characteristics that *do* underly the variability in biological resources. This hypothesis is of interest because of the well-defined nature of $X2$, its relative ease of measurement, and the existence of a historical database. As will be seen, $X2$ has significant and pervasive relationships with populations at all trophic levels.

Second, we examine how other variables such as diversions can affect the above simple relationships between $X2$ and various biological resources. Predictions or standards that are based on $X2$ alone may give erroneous results if additional variables not highly correlated with $X2$ have large enough effects. In order to explore this possibility, we examine a model that incorporates X2 plus an additional variable. We show that the additional variable has profound effects.

EMPIRICAL RELATIONS BETWEEN X2 AND ESTUARINE RESOURCES

The associations between estuarine resources (e.g., longfin smelt abundance) and environmental factors such as X2 are expressed here using *generalized linear models*, which, as the name implies, are flexible extensions of classical linear models (McCullagh and Nelder 1989). If Y is the response variable, the Z_i are predictor variables, and $E(Y)=\mu$, a generalized linear model takes the following form:

$$g(\mu) = \alpha + \sum_{i=1}^{p} \beta_i Z_i ,$$

where g is the *link function* describing how the mean depends on the linear combination of predictors, and α and the β_i are constants; g can be any monotonic differentiable function. The dependence of the variance of Y on the mean μ is specified independently of the link function:

$$\text{var } Y = \phi V(\mu),$$

where V is the *variance function* and ϕ is known as the *dispersion parameter.* If $V = 1$ and $g = I$, the identity function, the model is equivalent to a classical linear model. The models are estimated by maximum-likelihood, using an iteratively reweighted least-squares algorithm.

The response variables were chosen so that populations at a number of trophic levels would be represented. The predictor variables were determined as follows: For each biological resource, a specialist was consulted to recommend the averaging period over which X2 was likely to be related to the resource. In the case of longfin smelt, for example, the average of X2 for the period February-May was used. These periods were chosen on the basis of the biology of the resource in question, not by trying to optimize some statistic. The variables used, observations available, and sources for the data for each model are summarized in Table 1. Where noted, the observation corresponding to 1983 flows was eliminated. In these cases, a significant portion of the population may have been seaward of the sampling stations, causing an underestimate of the annual abundance.

Table 1. Response variables, associated predictor variables, observations available, and data sources.

Response	X2 Period	Observations	Source
POC supply in Suisun Bay, annual primary production plus river load of algal-derived POC (Gg yr^{-1})	Jan-Dec	75-89	Herbold et al 1992; A. Jassby, J. Cloern and T. Powell, unpublished MS.
Neomysis mercedis, Mar-Nov abundance index (no.)	Mar-Nov	72-88[a]	DWR[b]
Crangon franciscorum, annual abundance index (no.)	Mar-May	80-90	DFG[c]
Molluscs in Grizzly Bay, annual abundance (no. m^{-2})	3-yr average Jan-Dec	81-90	DWR
Striped bass egg survival, 38 mm index: Peterson egg production	Apr-Jul	69-82, 84-91	DFG
Starry flounder, annual abundance index (no.)	Previous year Mar-Jun	80-91[a]	DFG
Longfin smelt, annual abundance index (no.)	Jan-Jun	68-73, 75-78, 80-91[a]	DFG
Striped bass, fall MWT index (no.)	Jul-Nov	68-73, 75-78, 80-91	DFG

[a] *1983 intentionally omitted*
[b] *California Department of Water Resources*
[c] *California Department of Fish and Game*

Because of the small amount of data available (number of observations $n \leq 22$), models requiring estimation of more than 2 parameters (aside from the intercept) were not considered. For each resource, two types of models were estimated: (1) using *X2* alone, averaged over some suitable period, and (2) using a natural spline in *X2* with 1 interior knot (2 degrees of freedom). Generally speaking, a natural spline is superior to a polynomial for representing nonlinearities; with the few degrees of freedom permitted here, however, the difference between the two may be unimportant. If more than one model was "well-behaved" (each coefficient individually significantly different from zero and residuals consistent with model assumptions), the final model was selected on the basis of the *AIC* statistic (Hastie and Tibshirani 1990).

The results for each biological resource are summarized in Table 2 and in Figs. 1-8. All models are well-behaved in the above sense. The data demonstrate that simple and statistically significant relationships exist between X2 and biological populations at many trophic levels. Moreover, the supply of energy to the base of the food web, as represented by phytoplankton carbon, is also associated with X2. Except for mollusc density in Grizzly Bay, all responses show a decline as X2 increases, i.e., as flows decrease. Molluscs exhibit an increase at extreme values of X2, whether high or low.

Table 2. Summary of relationships between response variables Y and X2: n, number of observations; g, link function; V, variance function; df, degrees of freedom for X2 in model (1=linear, 2=natural spline with 1 interior knot); r, multiple correlation between Y and the predictors.

Y	*n*	*g*	*V*	*df*	*r*
Phytoplankton POC	15	I	1	1	.85
Neomysis	16	I	μ	1	.79
Crangon	11	I	1	1	.91
Molluscs	10	I	μ	2	.80
Striped bass survival	22	I	1	1	.62
Starry flounder	11	log	μ	2	.92
Longfin smelt	21	log	μ	1	.86
Striped bass	22	I	1	2	.84

Although the causal mechanisms underlying these associations are not at issue here, the distinctive response of the mollusc community deserves a few comments. Persistent high values of X2 (persistent low flows) permits the colonization of Suisun Bay by marine benthic macroinvertebrates. In times past, the main colonizing species was *Mya arenaria*, but this role has been usurped by the invader *Potamocorbula amurensis* during the current drought. In a similar manner, persistent low values of X2 (persistent high flows) leads to colonization by freshwater benthic macroinvertebrates, particularly *Corbicula fluminea*. The net effect of these "high-density" colonizations from both the seaward and landward directions under persistent low or high flows, respectively, is a minimum in mollusc density at intermediate values of X2 (Fig.4).

X2 is clearly a viable candidate for indexing estuarine habitat conditions. In addition to its well-defined and measurable nature and its interpretation as an indicator of the salinity field, it has a pervasive and clear relationship with many estuarine biological properties. Relationships exist between X2 and an important component of the food web base in Suisun Bay (phytoplankton POC), zooplankton consumers (*Neomysis* and *Crangon*), a major group of benthic consumers in Suisun Bay (molluscs), bottom-foraging fish (starry flounder), and both survival (striped bass) and abundance (longfin smelt, striped bass) of fish that feed in the water column. The statistical significance of these simple relationships and the wide variety of trophic levels involved reflects the ability of X2 to act as a surrogate for the effects of net delta outflow and other hydrodynamic variables.

Relationships between net delta outlfow and estuarine resources can also be demonstrated and, considering the close association between X2 and net delta outflow, these relationships may be as pervasive as those with X2. One could expect, however, an advantage to using *measured* values of X2 compared to net delta outflow. The latter has high uncertainty, particularly at low flows. The more noise in the predictor variables, the weaker the apparent relationship between the response and predictors; we are thus more likely to discover subtle relationships when using measured X2 than when using outflow. In this report, the X2 values are determined in part from a time series model relating X2 to log outflow (see accompanying issue paper by Kimmerer and Monismith), so the difference between the two data sets may not be pronounced. Aside from the postulated differences in uncertainty for the two variables, the time series model demonstrates that X2 incorporates lag effects of outflow. In principle, then, the relationships with other estuarine resources may be better for X2 than for outflow, or vice-versa. An explicit comparison has not been made, but should be done.

EFFECTS OF ADDITIONAL VARIABLES

X2 is not the only variable affecting estuarine resources. The previous history of the resource, as well as other environmental forces, may exert some influence. Although the empirical relationships offer evidence that X2 summarizes estuarine conditions for many resources of interest, they are not necessarily adequate for forecasting purposes in their present form. It is true that some of these other variables are correlated with X2, particularly hydrological ones such as fraction of water diverted (*DIVER*). It is *not* true, however, that these correlations will remain intact in the future, particularly if the estuary is managed to achieve a certain X2. Furthermore, some of these variables may not *now* be correlated with X2. We therefore have to examine the implications of these other variables. Relationships that acknowledge only the influence of X2 must have *some* predictive value, as evidenced by the previous discussion. The apparent relationship between X2 and an estuarine resource may change, however, when other important variables are explicitly included. As shown in what follows, the choice of a "target" X2 level can change as well.

In our example, striped bass survival index (38 mm index: Peterson egg production) is the response variable. This variable should be independent of the population's past history, and so we exclude previous values of the index from the set of predictor variables. In addition to X2, *DIVER*, the fraction of total inflow diverted, is used as a predictor. In practice *DIVER* is correlated with X2, but it is nevertheless an independent mechanism in principle. Whereas X2 affects the *susceptibility* of a population to entrainment, as well as its food supply and other environmental conditions, *DIVER* represents the relative *intensity* of entraining forces. On the basis of DFG recommendations, an averaging period of April-July was used for each of these variables. Because of the small number of data points, no additional predictors were considered.

The nature of the relationship was explored with generalized additive models (Hastie and Tibshirani 1990). These are an extension of generalized linear models in which the effects of individual predictor variables are additive but the form of each effect is relatively unconstrained and determined by a smoothing of the data. More formally, (1) and (2) are still appropriate but the Z_i are replaced by $f_i(Z_i)$, where the f_i are functions determined by an iterative smoothing process. Once a form has been established for each effect, the individual effects can be parameterized and their significance tested in more conventional ways. When applied to the current problem, the analysis suggested a nonlinear effect for X2 but a more or less linear one for *DIVER* (the middle two plots of Fig.9): the X2 effect was unimodal with a peak between 75 and 80 km, while *DIVER* had a monotonic negative effect. Maximal survival is therefore attained when X2 is situated approximately in Suisun Bay, while *DIVER* is deleterious to survival at all values of *DIVER*. Both forms are consistent with existing views on the actual mechanisms at work. In this analysis, the link function was taken to be $g = \log$, and the variance function $V = \mu$.

In order to quantify the dynamics in more familiar terms, a classical linear model was constructed using the generalized additive model results as a guide. The response was log-transformed and the nonlinearity in X2 was represented with a quadratic term. The diagnostic once again show consistency with the underlying assumptions (Fig. 10) and the model produces a convincing fit, except in 1980 and 1982 (Fig.11). Note that the coefficients are individually significant at the .05 level (Table 3). The multiple correlation coefficient between actual survival index and the predictors is $r = .71$ ($p<.001$), compared to .62 when X2 alone is used.

To see the effects of an additional variable (*DIVER*) on the choice of X2, consider the model of Table 3:

$$\ln S_t = \alpha + \beta_1 X_t + \beta_2 X_t^2 + \beta_3 D_t ,$$

where S_t = striped bass survival index, X_t = X2, D_t = *DIVER*, α and the β_i are constants, and t denotes the year. If we wish to ensure that

$$S_t \geq S_{med} ,$$

where S_{med} is the long-term median survival index, then (3) and (4) imply that X_t must lie within the parabola (Fig. 12):

$$D_t = -\frac{1}{\beta_3} [\beta_2 X_t^2 + \beta_1 X_t + (\alpha - \ln S_{med})].$$

Table 3. Linear model of log striped bass survival index. $R^2 = .64$*, residual standard error* $s = 0.56$ *on 18 df.*

Term	Coefficient ± SE	t	Pr (>|*t*|)
1 (intercept)	-38 ± 16	-2.4	.030
X2	1.0 ± 0.4	2.4	.027
$X2^2$	$(6.4 \pm 2.5) \times 10^{-3}$	-2.6	.020
DIVER	-2.5 + 1.2	-2.1	.051

One obvious implication is that too high an *X2* can be *harmful* to survival, a feature not present in a single-predictor model using only *X2*. Also, when diversions are low enough (e.g., *DIVER* < 0.5), survival is high even when *X2* is upstream of Suisun Bay. On the other hand, for *DIVER* > 1.6, no *X2* position can ensure high survival. If we used a single-predictor model, we would have deduced that 73 km is an appropriate threshold value for attaining median survival. According to the analysis here, 73 km would have been too stringent a requirement in 12 of the year and *no requirement* would have been effective in the remaining years except insofar as it forced *DIVER* to have been lower. Note that any threshold lower than 78 km is unnecessarily stringent, according to this model. The effects of including *DIVER* in the model are thus profound.

CONCLUSIONS

The location of 2 ppt bottom salinity is closely associated with the population size of estuarine organisms at all trophic levels, as well as with the supply of organic matter to the food web from primary production and riverine loading. In principle, forecasts of these population sizes must also take into account additional variables for the most accurate setting of standards.

More generally, *X2* has many properties that render it a suitable habitat indicator in diverse estuarine systems. Although the necessity of calibration for each estuary and the need for more than a single annual sample appear to be at odds with EMAP requirements for habitat indicators, it has not yet been demonstrated that these requirements are realistic for estuaries. Temporal (and spatial) gradients are unusually intense in estuaries compared to other ecosystems and interannual variability in the seasonal pattern is also high. *X2* at least can be measured with some ease on a seasonal basis. Furthermore, although we have emphasized here the mean values of *X2* during specified periods, it may turn out that the variance and higher moments also contain valuable information about conditions for estuarine populations.

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Herbold, B., A. D. Jassby, and P. M. Moyle, 1992. *Status and trends report on aquatic resources in the San Francisco Estuary.* San Francisco Estuary Project, U.S. Environmental Protection Agency, San Francisco.

McCullagh, P., and J. A. Nelder, 1989. *Generalized linear models*, 2nd. ed. Chapman and Hall, London.

FIGURE LEGEND

Figure 1. Supply of particulate organic carbon (POC) to Suisun Bay from phytoplankton production and riverine loading of algal-derived particulate matter, compared to annual average X2.

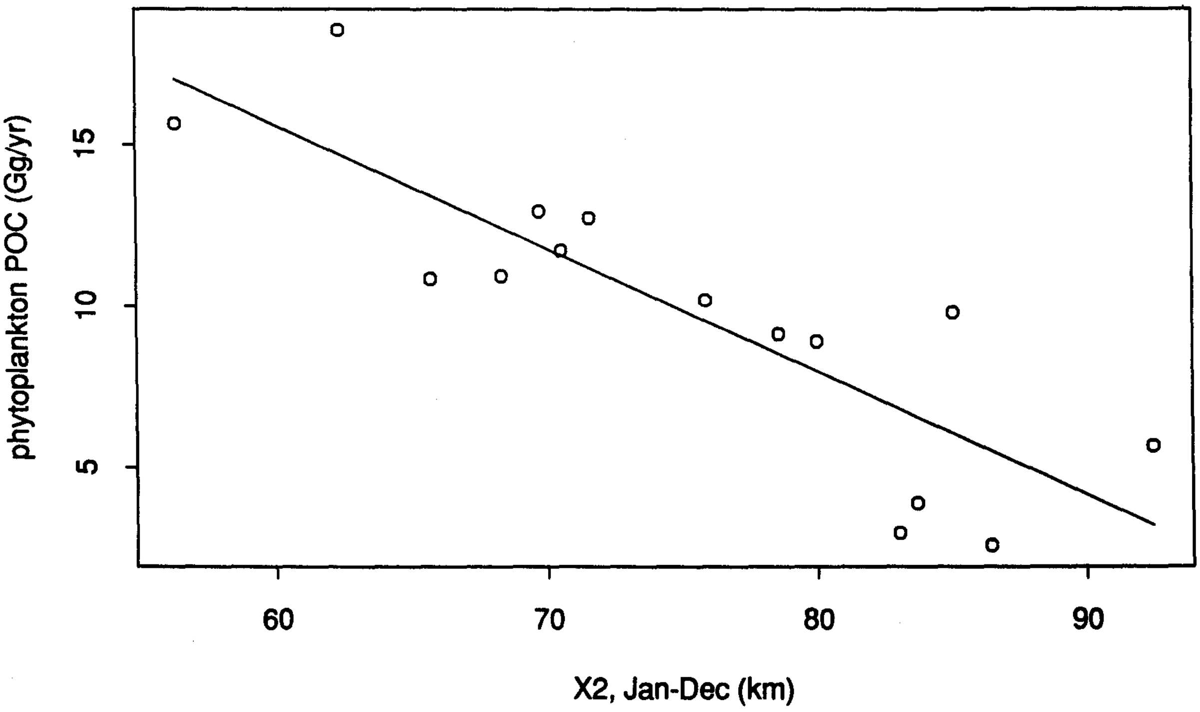


Figure 2. March-November abundance index of Neomysis mercedis.

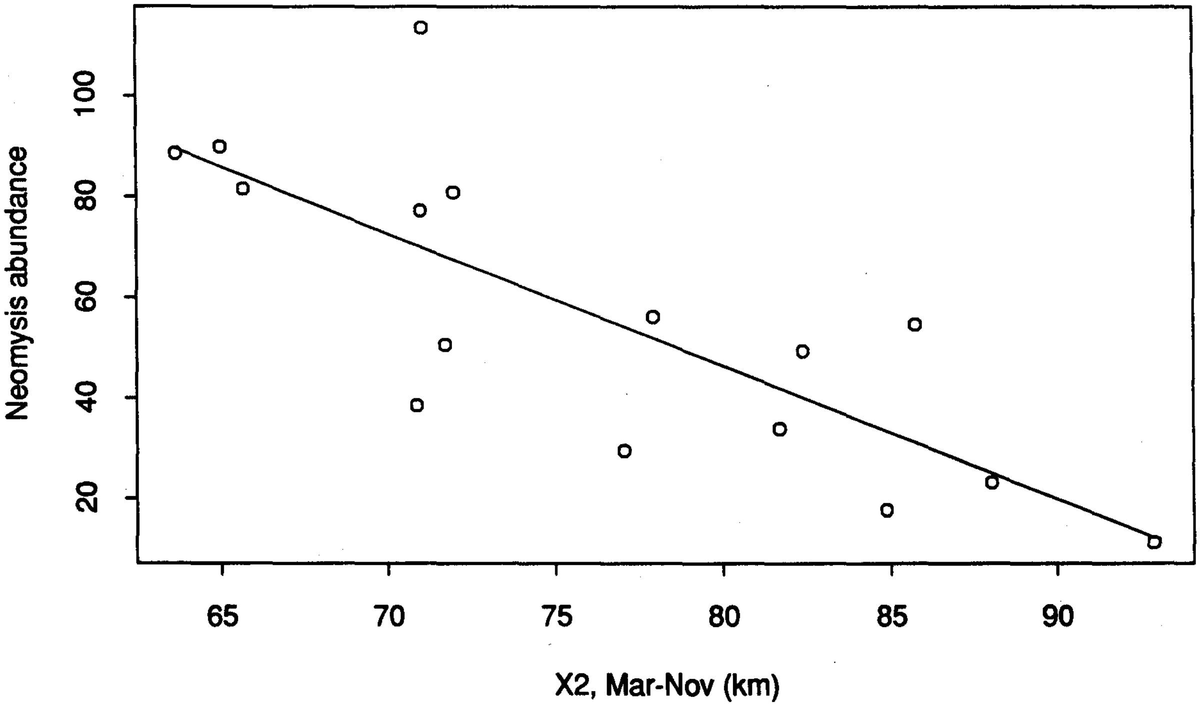


Figure 3. Annual abundance index of Crangon franciscorum *in the San Francisco estuary.*

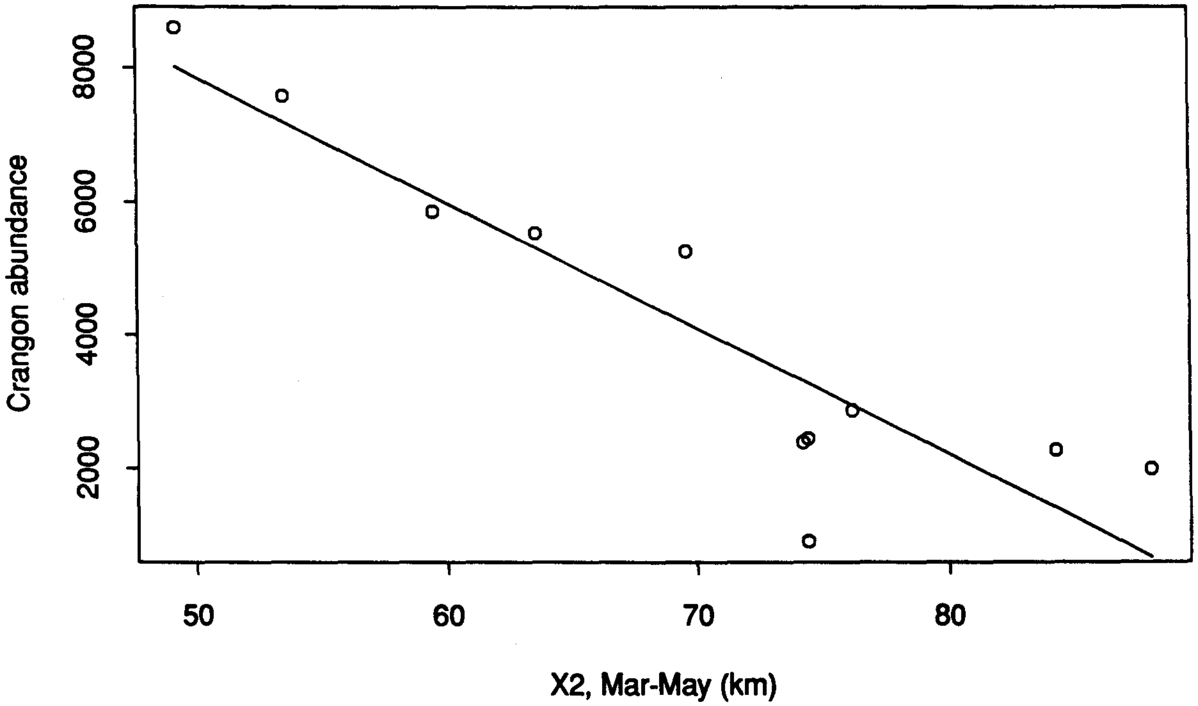


Figure 4. Annual abundance of molluscs in Grizzly Bay, compared to X_2 averaged over the current and previous two years.

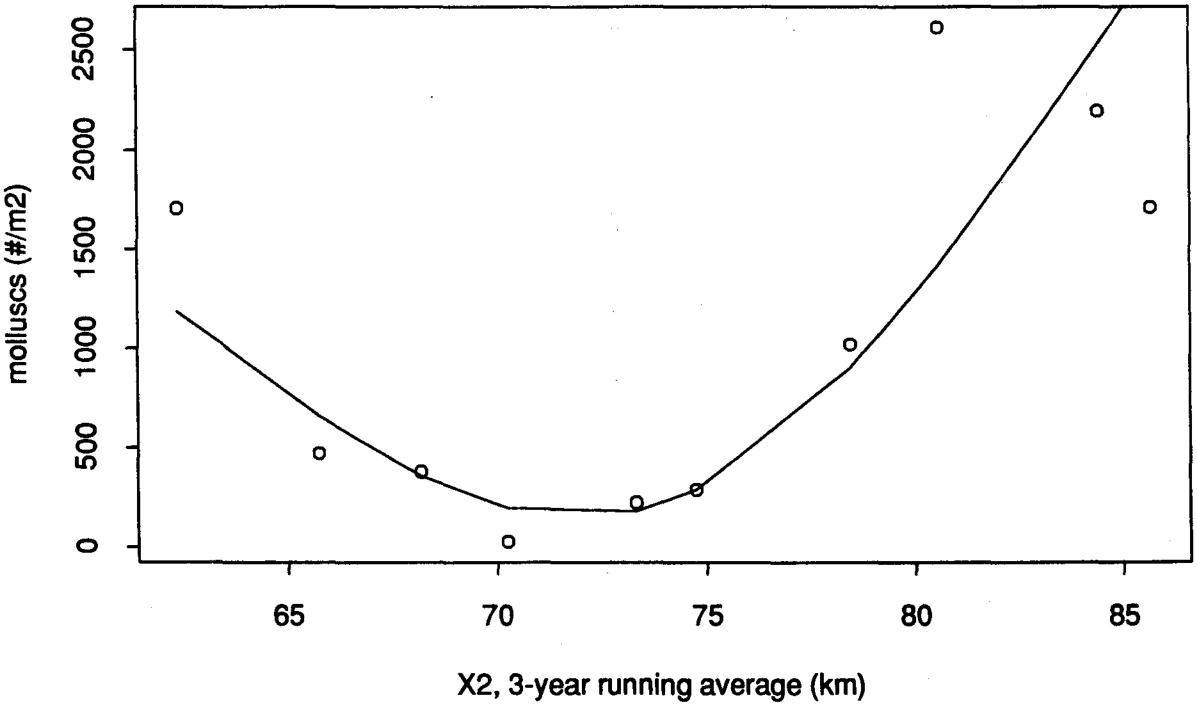


Figure 5. Survival of striped bass (Morone saxatilus) *from egg to adult (38 mm index: Peterson egg index:).*

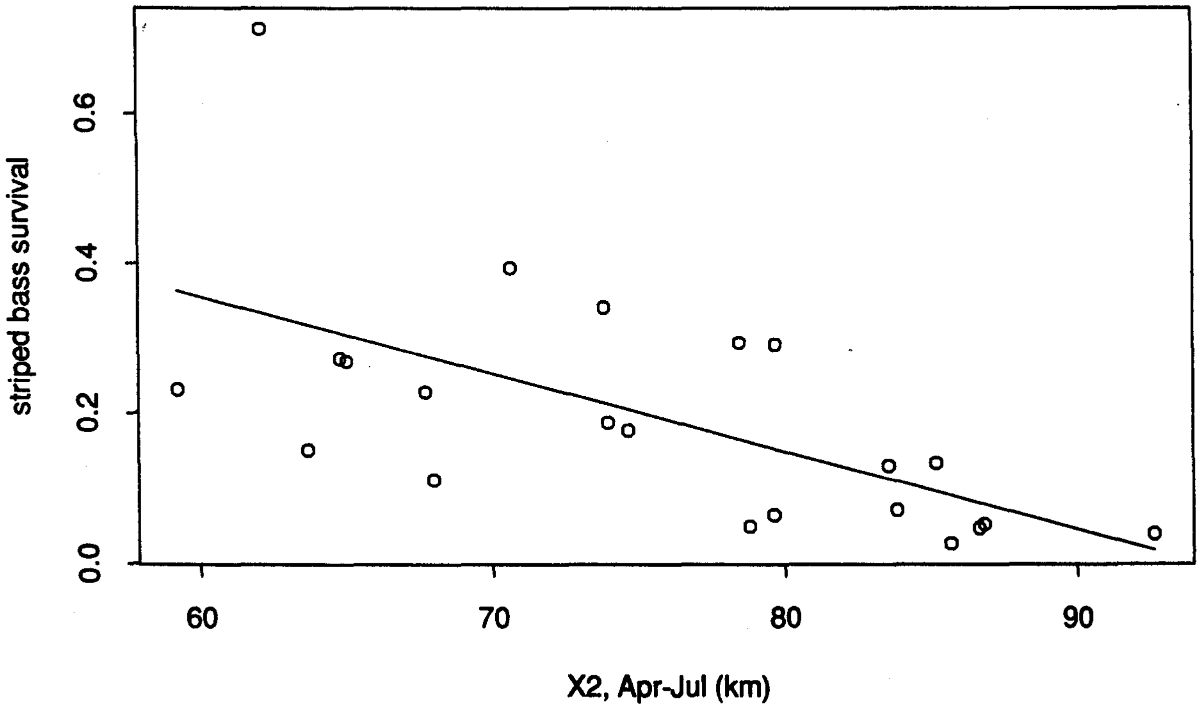


*Figure 6. Annual abundance index of starry flounder (*Platichthys stellatus*).*

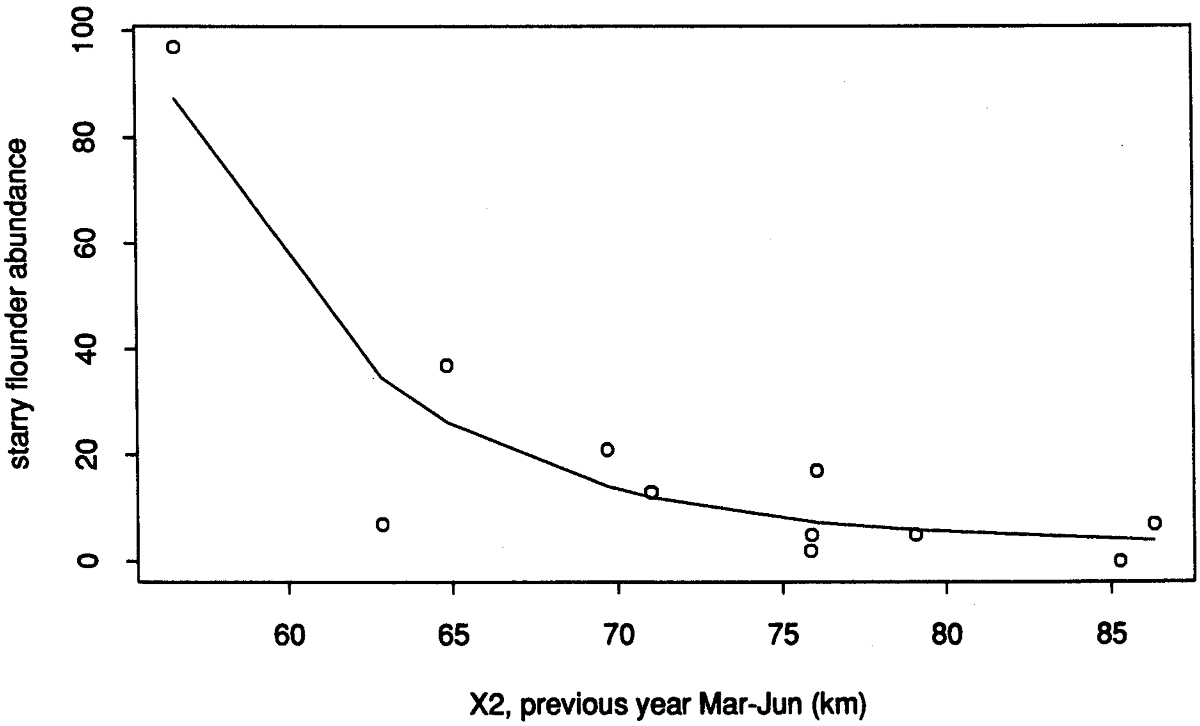


*Figure 7. Annual abundance index of longfin smelt (*Spirinchus thaleichthys*).*

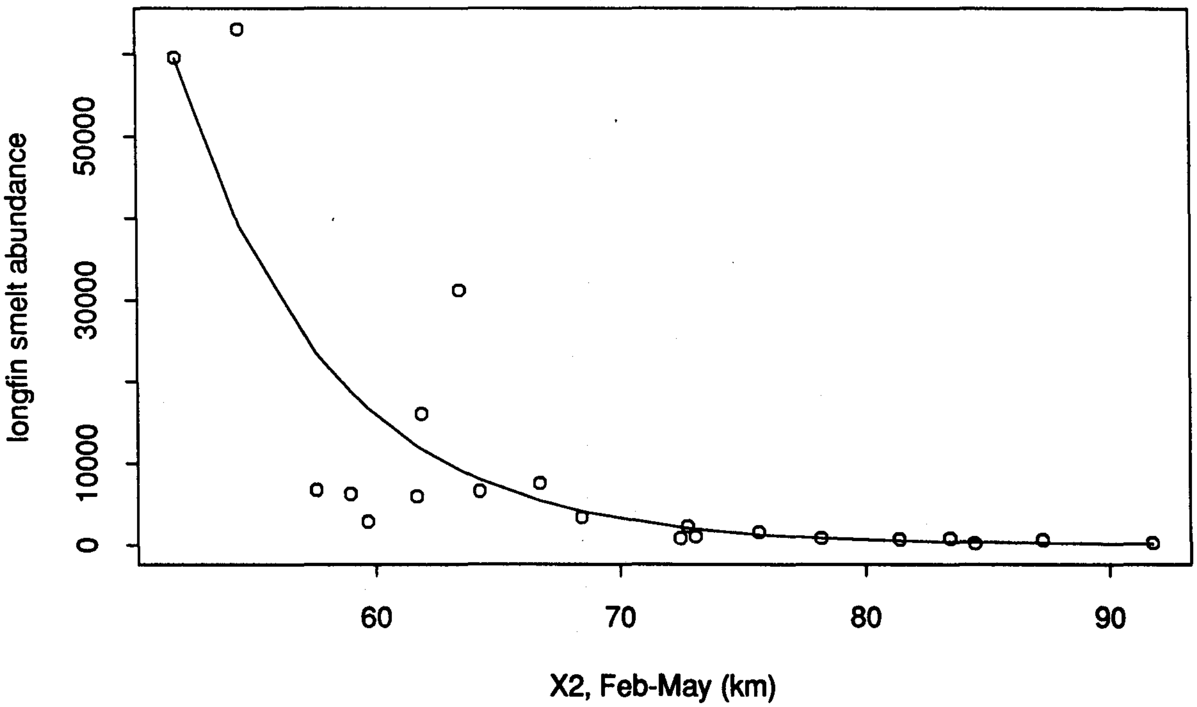


Figure 8. Striped bass (Morone saxatilus) *fall midwater trawl index.*

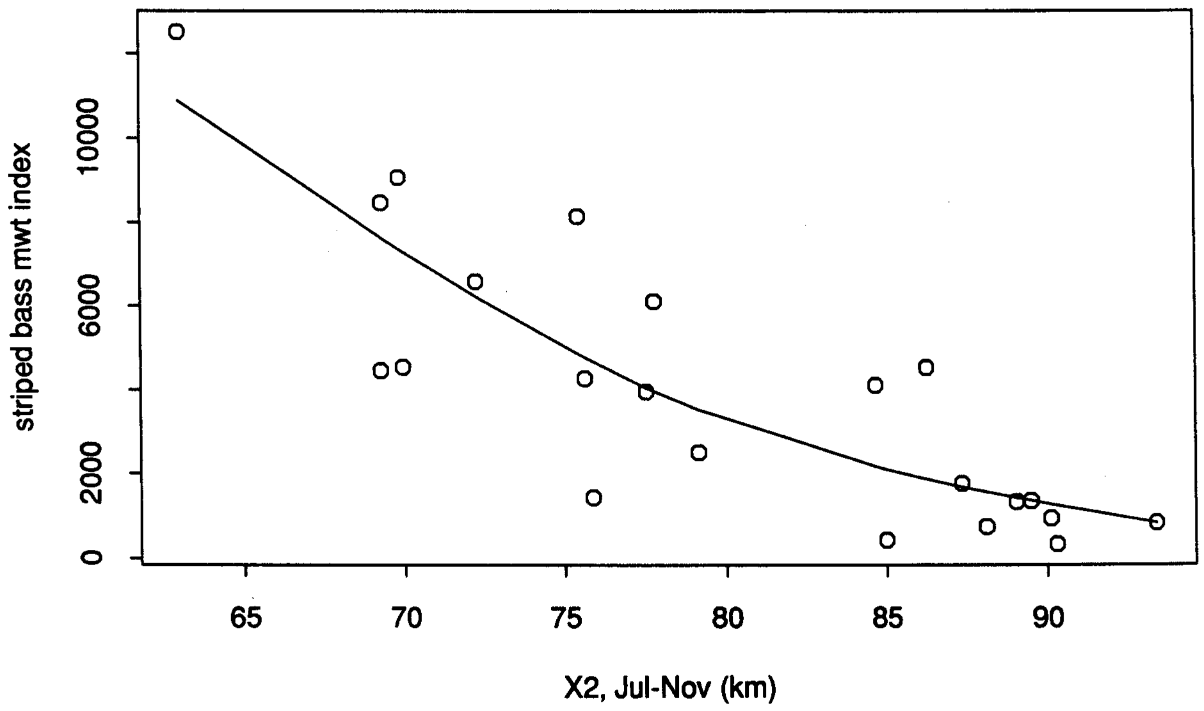


Figure 9. Diagnostics for generalized additive model of striped bass survival index with g = log and V = I: top panels, response and absolute value of residuals vs. fitted values; middle panels, partial residual plots for the two predictors X2 and DIVER; bottom panels, time and box plots of residuals. "Response" refers to the actual value of the index, not its transformed value. Note that residuals increase with the fitted values, in conformity with the variance function.

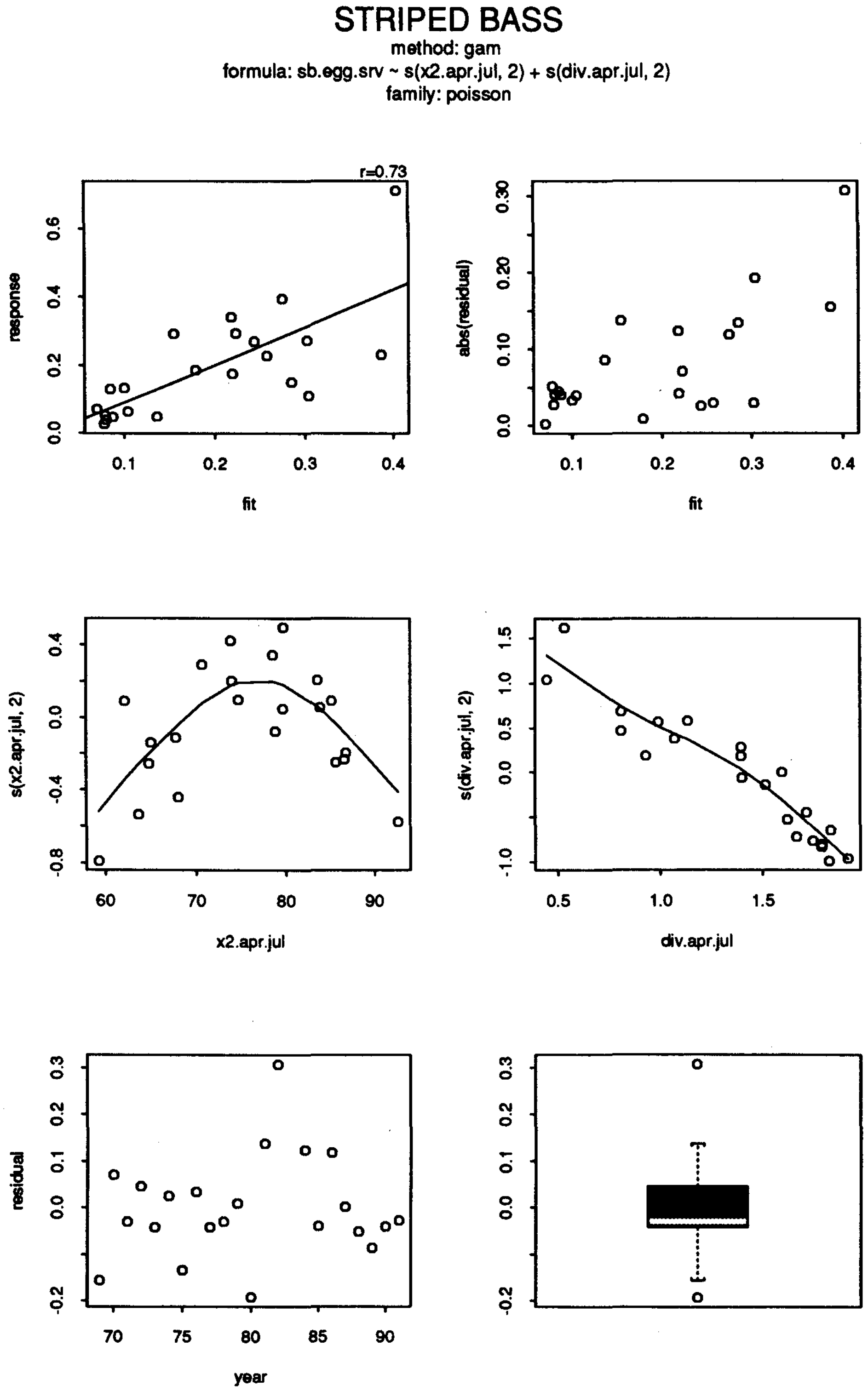
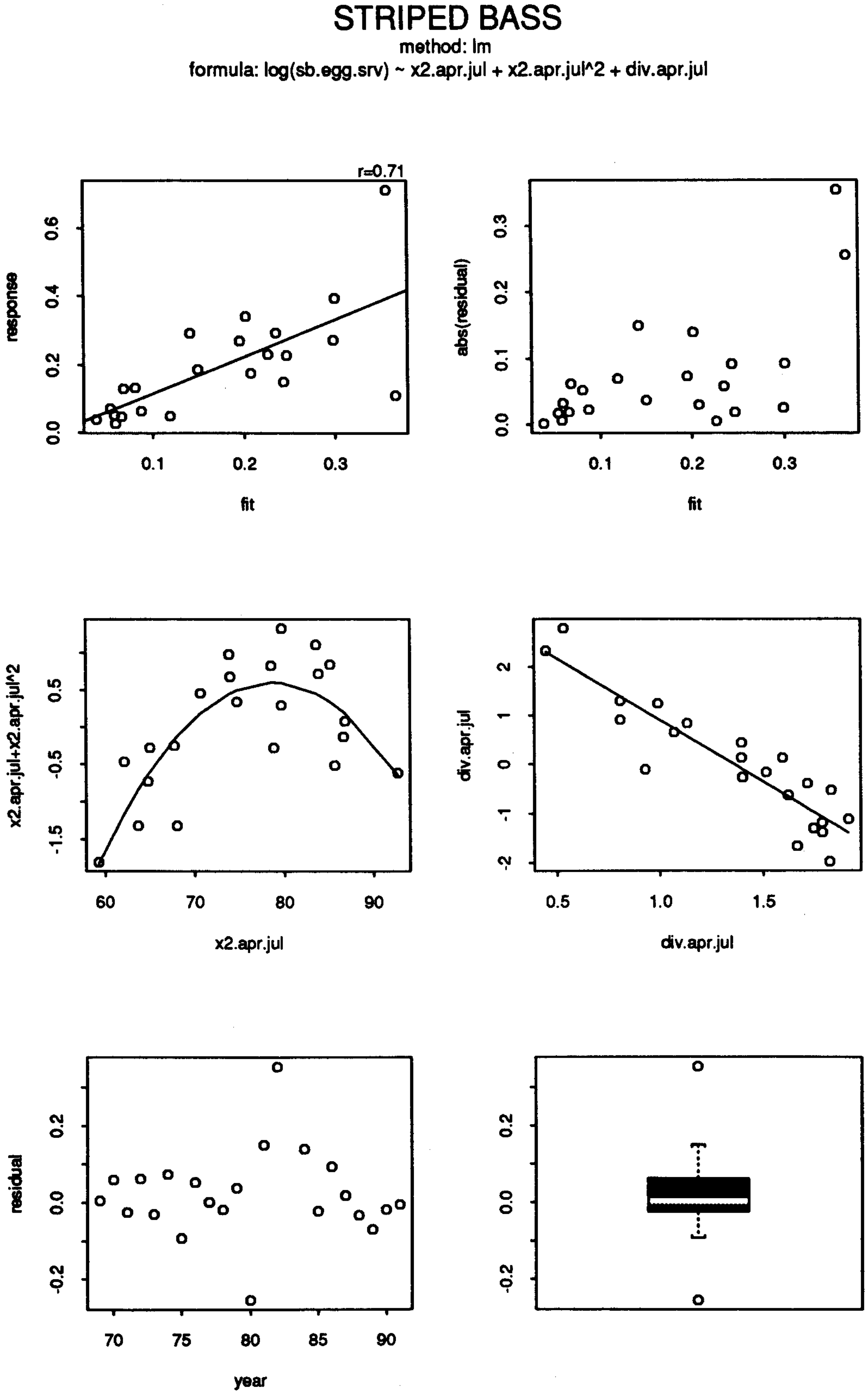


Figure 10. Diagnostics for linear model of striped bass survival index (see Figure 9).



STRIPED BASS
method: lm
formula: log(sb.egg.srv) ~ x2.apr.jul + x2.apr.jul^2 + div.apr.jul

Figure 11. Linear model of striped bass survival index: (A) Response vs. linear predictor; (B) Response and fitted values vs. year.

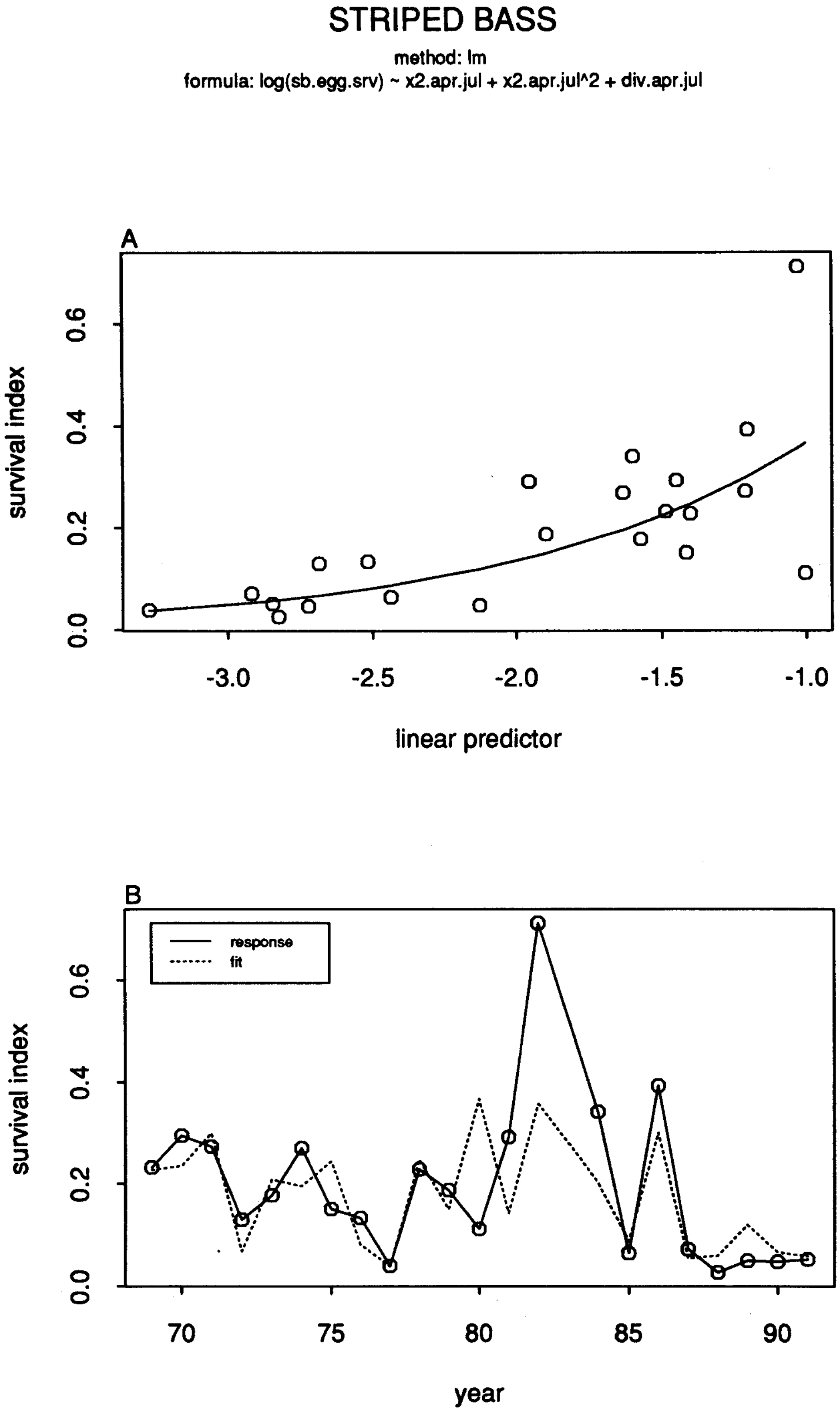
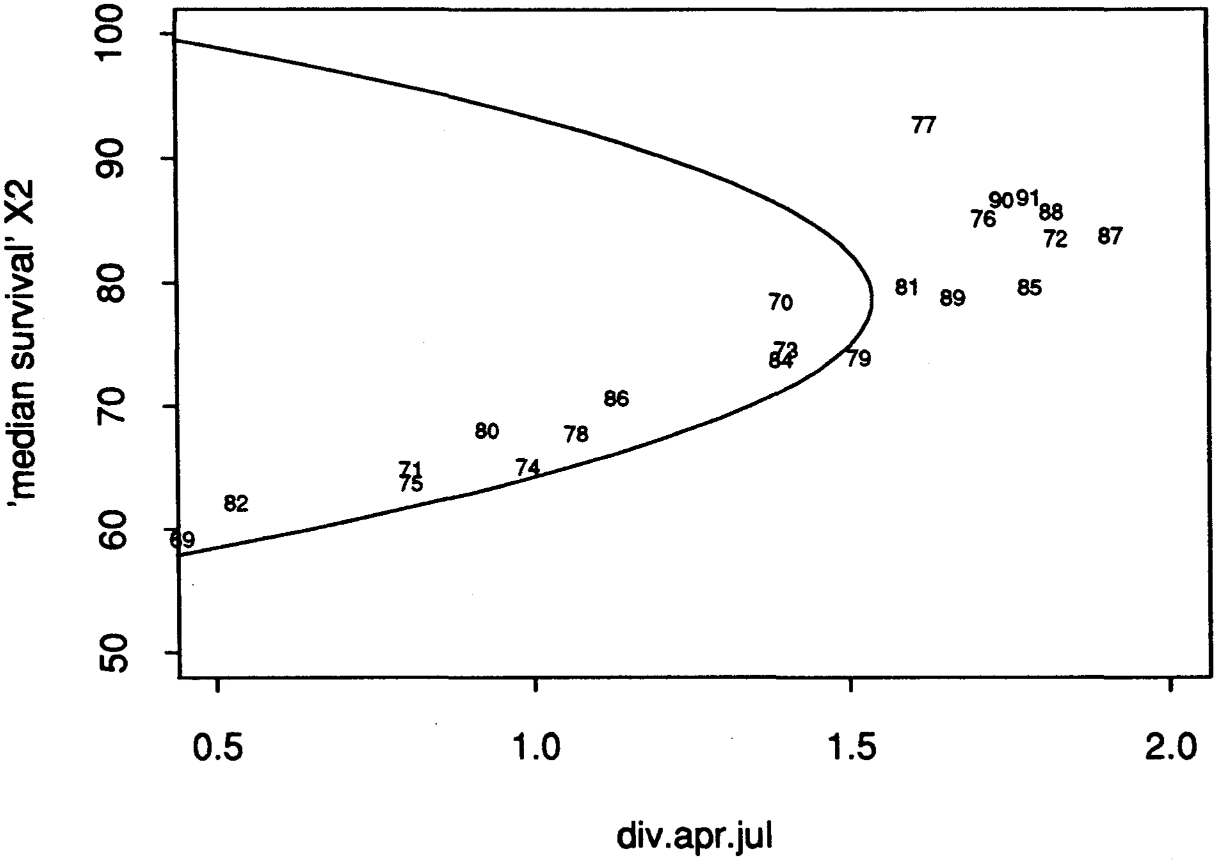
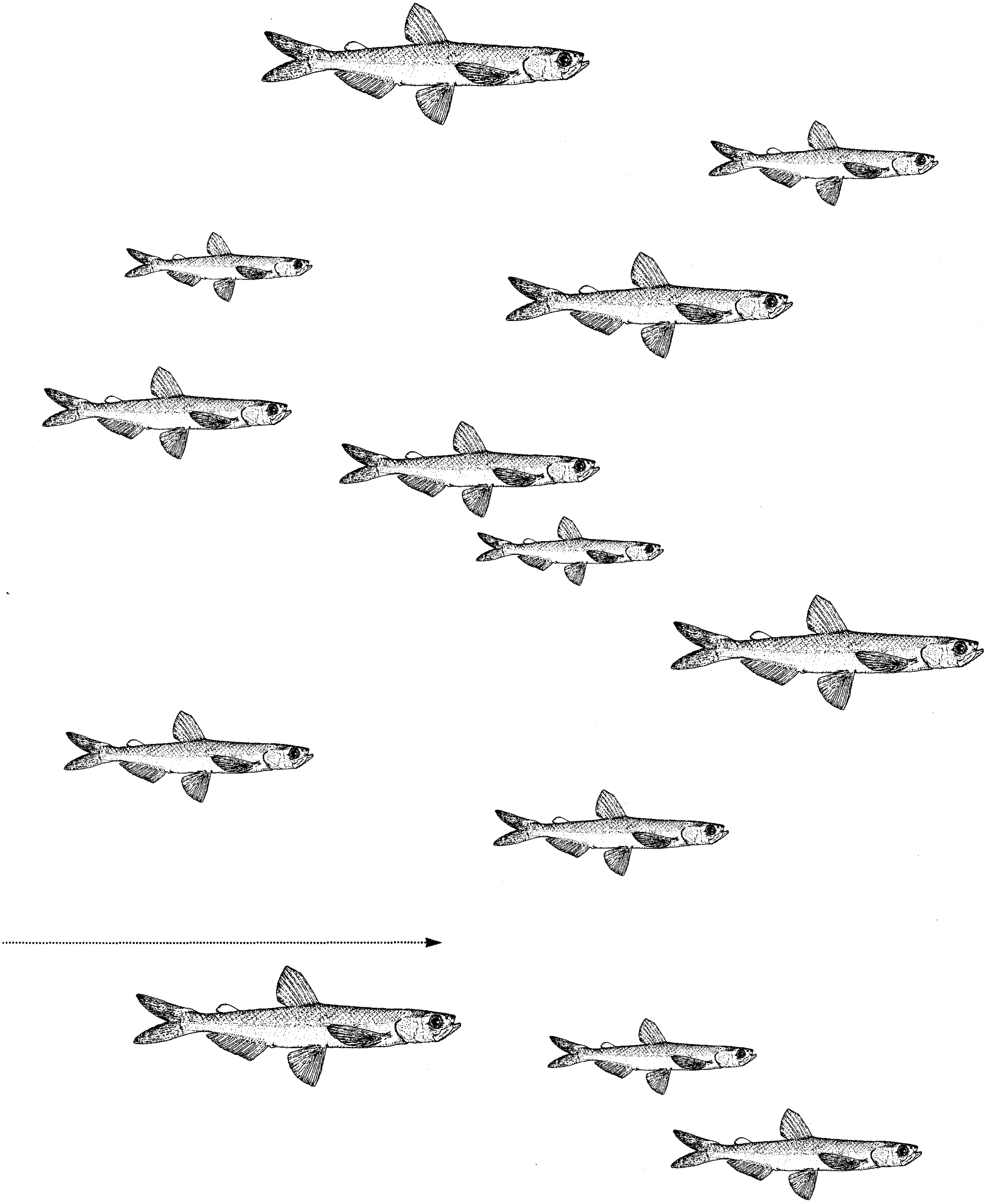
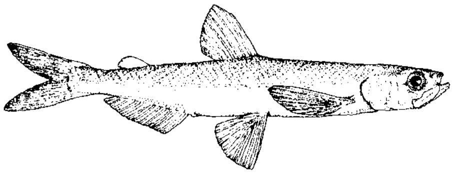


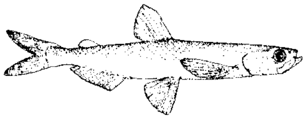
Figure 12. X2 needed to ensure median survival of striped bass, as a function of DIVER. Status for individual years also indicated on the plot.





EVALUATION OF THE IMPACTS OF AQUEOUS SALINITY ON THE SHORELINE VEGETATION OF TIDAL MARSHLANDS IN THE SAN FRANCISCO ESTUARY





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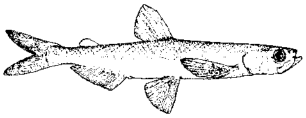
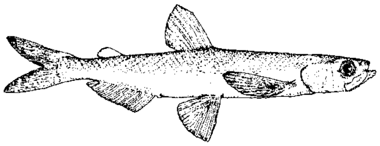
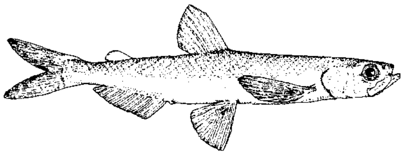
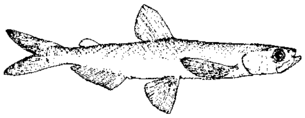
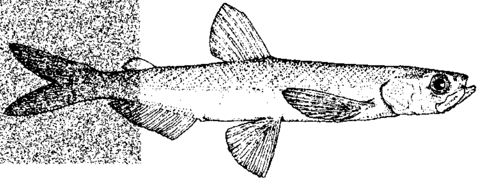
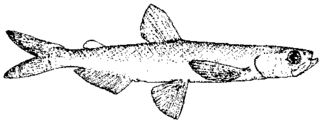


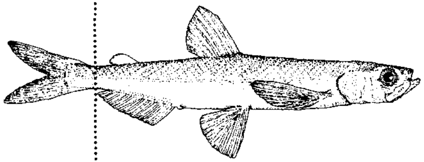


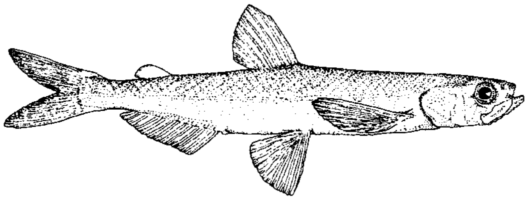




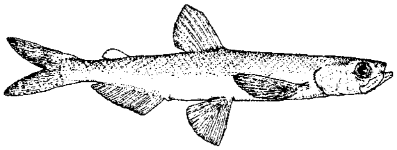


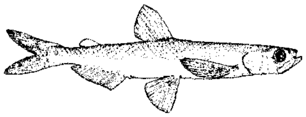






AN ISSUE PAPER PREPARED FOR THE FOURTH TECHNICAL WORKSHOP ON SALINITY, FLOWS, AND LIVING RESOURCES OF THE SAN FRANCISCO BAY/DELTA ESTUARY





SUMMARY

We surveyed the literature and began to document ecological evidence of salinity changes in tidal marshlands of the San Francisco Estuary. To examine the ideas presented in the literature, we conducted a survey of vegetation in the field. Our survey focused on recent changes in the spatial distribution of common plant species of the shorelines and channel margins of brackish tidal marshland. In order of decreasing tolerance to salinity, these species are California cordgrass (*Spartina foliosa*), alkali bulrush (*Scirpus robustus*), and tules (*Scirpus acutus*). In addition, we conducted a synoptic survey of tule stature for this year and last to assess the response of vegetation to interannual variation in salinity regime.

There is abundant evidence to suggest that the ecology of tidal marshlands in the Estuary has been affected by regional increases in salinity. A continuum of ecological effects has been observed. Extreme responses have involved a loss of glycophytes, especially tules, in the lower intertidal zone, and their replacement in part by halophytes, especially cordgrass. Lesser responses have included the replacement of some tules by alkali bulrush. Minimal responses have involved decreases in the stature of tules, especially near the downstream limits of their estuarine distribution. The pattern of vegetation response observed in the Suisun Marsh Area was confirmed by the similar pattern observed along the Napa River. These similarities were apparent despite local influences on water quality.

There is little or no evidence to suggest that regional salinity has decreased. Although the stature and relative abundance of some plant species, especially tules, have probably increased during years of abundant freshwater input, the overall or net changes in plant species distributions strongly suggest an upstream increase in salinity as affected by sea level rise, drought, and regulation of river inputs.

Experimental tests of the relationships between aqueous salinity and the stature and distribution of major plant species, with further investigation of estuarine transgression and related ecological succession, are required to validate our preliminary findings, and to determine the relative influences of natural versus regulated changes in salinity regime. Based upon such research, the response of vegetation to salinity changes could be represented as mechanistic models to predict the ecological effects of sea level rise or changes in throughputs of river water.

STUDY OBJECTIVES

The purpose of this study was to examine the response of estuarine marsh vegetation to changes in salinity regime, as required to develop scientific rationale for estuarine salinity standards. The findings are presented in three parts: (1) a review of the pertinent literature and data bearing on the response of tidal marsh vegetation to changes in salinity, (2) new field studies of correlations between salinity and tidal marsh vegetation, and (3) conclusions and recommendations about using vegetation to monitor the ecological effects of salinity changes.

PREVIOUS WORK RELATING TO THE EFFECTS OF AQUEOUS SALINITY ON TIDAL MARSH VEGETATION

(1.1) THE IMPORTANCE OF TIDAL MARSHLAND AND ITS VEGETATION

Since the time of California statehood, about 85% of the historical amount of marshland downstream of the Delta has been destroyed by conversion to agriculture or urban landuses (Dedrick 1989). The regional citizenry recognizes that tidal marshlands are critically important for the ecological health of the region and expects government to restore and protect tidal marsh resources.

Numerous studies have established that tidal marshlands can have significant geomorphic and ecological values in a regional context, including flood control, shoreline stabilization, sediment entrapment, water quality improvement, and food chain support for aquatic, semi-aquatic, and terrestrial plants and animals (e.g., see review by Sather and Stuber 1984, Zedler et al. 1985, PERL 1990). The "Status and Trends Reports" for the Estuary (ABAG et al. 1991, Herbold et al. 1992, USFWS 1992) indicate that more species of fish, mammals, and birds of special concern (i.e., species that are endangered, threatened, or proposed for such status) are associated with tidal marshlands than any other habitat type of the region.

Vascular vegetation plays a major role in the evolution of tidal marshlands. They evolve from intertidal mudflats when supplies of water and inorganic sediment are suitable for plant colonization. During periods of rapid sea level rise or river flooding, the frequency and duration of inundation and hence the rate of sediment delivery increases. Accretion of the youthful marshland depends upon the entrapment of sediment by the pioneering vegetation. As tidal marshland matures, vascular vegetation plays an increasingly important role as the source of sediment to maintain marshland elevations, relative to the tides. Youthful tidal marshland is relatively low and inorganic, whereas mature tidal marshland is relatively high and more organic. The tendency of plants to maximize production maintains the mature marshlands near the upper limit of the local tidal range. Tidal control of habitat conditions for marshland plants prevents the evolution of terrestrial conditions.

Dynamic interactions between vascular vegetation and the flow of tidal water maintain tidal marshlands. Although the tides ultimately control the distribution and abundance of intertidal plant species, plant growth controls the distribution of tidal energy and the rate of geomorphic work conducted by the tides, especially near the upper limit of the tidal range.

The plants strongly influence the quality and quantity of habitats for themselves and for many resident species of native wildlife. The ecological values and functions of tidal marshland are therefore largely determined by the nature of the plant community.

(1.2) THE IMPORTANCE OF SALINITY

In the context of an estuary as a whole, including its landward transgression over time, the structure of plant communities in tidal marshland is strongly correlated to salinity regime (Meyerson 1972; Atwater and Hedel 1976, Harvey et al. 1977; MacDonald 1977, Mudie and Byrne 1980; ABAG et al. 1991). The obvious correlations between species distributions and salinity at these large scales of space and time account for the rather casual designations of saline, brackish, or freshwater estuarine zones. Most intertidal plant species show continuous spatial and temporal distributions along the salinity gradient of an estuary, however, with shifts in sympatry or allopatry among species. The boundaries between salinity zones are therefore arbitrary.

Spatial and temporal variations in plant community structure within tidal marshlands are not closely correlated with salinity. The traditional view amongst ecologists working in tidal marshes on the West Coast is that the arid climate makes salinity the dominant factor that controls the distribution and abundance of plant species. However, despite the demonstrable influence of salinity on photosynthesis, growth, and survival of tidal marsh vegetation, especially in culture (Penfound and Hathaway 1938, Phleger 1971, Vogl 1976, Mahall and Park 1976, Barbour 1978, Smart and Barko 1978, Pearcy et al. 1982, Ustin et al. 1982, Dacey and Howes 1984, Pearcy and Ustin 1984), salinity is not necessarily the dominant factor affecting natural community structure, Mahall and Park 1976 and Zedler and Beare 1986 notwithstanding. The recent literature regarding the interaction of salinity and oxygen tension (Mendelssohn and McKee 1987, Pennings and Calloway 1991, Brenchey-Jackson 1992a,b) indicates that hypoxia is more important in tidal marshland than previously estimated. Furthermore, the interactions of physical gradients within marshland in combination with natural disturbance continuously alter competitive relationships, causing extreme site variability and complex patchiness of the plant mosaic (Bertness 1990, 1991, 1992). The emerging synthesis about estuarine plant ecology is that the interplay among water supply, soil salinity, light, and hypoxia, operating partly through alteration of competitive relationships and partly through physiological tolerance, and slowly enough to transcend several growing seasons, might be the only way to fully explain the structure of natural plant communities of tidal marshland.

The degree of correlation between plant community structure and estuarine salinity therefore increases with scale through space and over time. Mechanisms operating at small scales that account for the patchy distribution and abundance of plant species within tidal marshland can be disconnected from the larger gradients of tidal and salinity regimes that delimit an estuary as a whole. The large-scale analyses of ecological trends that integrate across the small-scale causal mechanisms can result in correlations between salinity and plant community structure that might be either evidence of the pervasive influence of salinity or simply scale effects.

The state of understanding about the effects of water salinity on tidal marsh vegetation permits the formulation of four basic hypotheses: (1) salinity is not important (the null hypothesis); (2) salinity dominates all other factors; (3) salinity interacts with water volume, especially as related to hydroperiod of habitats; and (4) interactions of many factors must be considered (the comprehensive hypothesis). The existing evidence from the literature does not support the null hypothesis. A strong correlation between salinity and plant community structure might be expected for hypotheses 2-4, however. The second and third hypotheses, if unfalsifiable, would provide useful rationale to establish salinity standards. Although hypothesis 4 is perhaps true, it is not very operational with regard to water quality regulation.

(1.3) THE IMPORTANCE OF CLIMATE

Under natural conditions, climate controls sea level and river discharge. As sea level rises, the Estuary moves inland, while river discharge affects the salinity regime and the extent and duration of flooding. Landward transgression of the Estuary has continued at variable rates throughout the later Holocene, and in the future may increase dramatically due to unnatural climate change (i.e., the "Greenhouse Effects" sensu EPA 1988, Williams 1988).

Marine influences proceeded upstream through the Golden Gate about 10 thousand years ago, at a maximum rate of about 30 m/yr (Atwater et al. 1977), and reached the Delta about 7 thousand years ago (Schlemon and Begg 1975). Since then the average rate of sea level rise has probably decreased from about 2.0 cm/yr to about 0.15 cm/yr (Atwater et al 1977; Byrne and Collins unpublished data), which suggests that the average rate of transgression of the Estuary has decreased from about 30 m/yr to about 2 m/yr.

Rates of estuarine transgression and upstream ecological succession are not necessarily the same, however. Given that a prominent halophyte, *Salicornia virginica*, arrived at tidal marshland along the Petaluma River about 1.8 thousand years ago (Byrne and Collins unpublished data) and at Browns Island (western edge of the Delta about 50 km upstream from the Petaluma River) within 0.2 thousand years (Atwater 1980, personal communication: Brian Atwater), a coarse estimate of the average rate of upstream migration of saline marshland conditions during recent millennia is 30 m/yr. This is greater than 10x the estimated rate of transgression for the same time period, and coincidentally equivalent to the rate of transgression when the Estuary first formed.

The rate of upstream migration of plant species indicative of saline conditions is probably accelerated by drought. Most plant species that inhabit estuarine marshes are adapted to broad ranges of salinity (see especially Penfound and Hathaway 1938, Pearcy et al. 1982, Pearcy and Ustin 1984). Colonies of salt tolerant species that are established upstream during periods of low river discharge and high salinity are not necessarily eradicated when discharge is increased. The average rate of upstream migration of salt tolerant species can therefore exceed the average rate of estuarine transgression, especially where interannual variations in discharge are extreme. In the absence of drought, the projected rate of upstream migration of saline conditions might equal the recent rate of estuarine transgression.

Differences in climate among subregions of the Estuary apparently do not affect the distribution of common species of plants that inhabit tidal marshland. For example, the distribution of halophytes does not correspond to the regional pattern of potential evapotranspiration or rainfall. The halophytic plant community around San Pablo Bay and San Francisco Bay is subject to lower temperatures and greater amounts of precipitation that the freshwater plant community of the tidal marshlands of the Delta.

FIELD SURVEYS

We conducted intensive field surveys in search of correlations between aqueous salinity regime and plant community structure that could be used as indices for the ecological response of tidal marshland to sea level rise or changes in throughputs of river water. We have not investigated the causal mechanisms that explain the correlations observed.

(2.1) METHODS

(2.1.1) SELECTION OF STUDY SITES

Our sampling universe is the expected distribution of vegetation response to recent past or future changes in tidal salinity regime, as affected by sea level rise and regulated or natural variations in river throughputs. The most noticeable responses might range in severity from local extinctions of shoreline plant species to declines in their vigor and stature. Given the rising stand of the seas, the effects of salinity will progress upstream, most noticeably at the ecotone between glycophytic and halophytic communities. In general terms, the sampling universe therefore corresponds to the brackish zones of the Estuary.

The breadth of a brackish zone is partly a function of bathymetry of the Estuary. The large, shallow bays and distributaries upstream of Carquinez Strait, permit extensive daily excursions of the tides and seasonal variations in salinity along the axis of the Estuary and thus elongate the axial salinity gradient. The continuum of plant community structure that correlates to the salinity gradient is therefore also elongated, and can only be quantified with a comparable array of study sites. A similar situation at a smaller scale exists upstream of Mare Island, along the Napa River. The salinity regimes in both areas are affected by discharges from tributary streams, in addition to the combined discharges from the

Sacramento and San Joaquin rivers. We therefore established sites and conducted surveys throughout much of the tidal marshlands of the greater Suisun Marsh (termed the Suisun Marsh Study Area) and along the Napa River (Figure 1).

Our study of the Napa River marshlands also helped us to generalize our findings and to assess the potential value of major tributary systems of the Estuary as model systems to predict the overall ecological effects of changes in freshwater outflow from the Delta.

Much of the brackish marshlands have been subject to disturbance by agriculture and the operations of industry and cities. Since the history of disturbance is not well recorded for most of the tidal marshlands, the selection of undisturbed sites cannot be assured. This further indicates the value of an extensive array of study sites that permits integration of ecological surveys across a broad range of habitat conditions.

(2.1.2) SELECTION OF STUDY PLOTS

The time constraints imposed upon our study required us to focus upon areas within tidal marshland where correlations between salinity regime and plant stature or community structure should be most obvious. Our study sites were therefore located along the immediate shorelines of the bays and margins of tidal marsh channels, where soil salinities are most responsive to changes in salinity of the tides.

In general, fluctuations in moisture content and salinity of tidal marsh soils decrease rapidly with distance away from shorelines or channel margins. The fine silts and clays transported by the tides impregnate the channel banks and greatly decrease their hydraulic conductivity, such that tidal flushing of the bank sediments is restricted to a narrow zone along the channel margin (Howland 1976, Nuttle 1988). Drawdown and recharge of the free-standing pieziometric surface at the interior reaches of tidal marshland between channels is mainly a function of evapotranspiration and inundation by tides that exceed bankfull stage, with no advective movement of the watertable toward the channel network. The interior areas of tidal marshlands therefore tend to trap salts delivered by overbank tides. Where the tides are saline, the tidal marsh sediments range from saline to hypersaline between the channel margins and interior marsh plains. Where the tides are fresh, the tidal marsh sediments are uniformly not saline. But within the brackish marshlands, where tidal salinities vary seasonally from fresh to saline, soil salinities increase rapidly from fresh or brackish to saline or hypersaline with elevation and distance away from channel margins (Balling and Resh 1983).

Spatial variations in plant community structure parallel these steep salinity gradients of brackish marshlands. The glycophytes are mainly restricted to the shoreline or channel banks and the halophytes are mainly restricted to natural levees and drainage divides. Along the axial salinity gradient of the Estuary, plant community structure varies less among drainage divides than channel margins or marshland shorelines.

(2.1.3) SELECTION OF STUDY SPECIES

The plant species selected for study are dominant components of the plant assemblages of shorelines and channel margins of the brackish tidal marshlands. In order of their increasing tolerance to salinity, these species are tules (*Scirpus acutus* or *Scirpus californicus*), alkali bulrush (*Scirpus robustus*), and California cordgrass (*Spartina foliosa*). The latter two species grow best at low salinities but tolerate high salinities, whereas *S. acutus* and *S. californicus* tolerate low salinities. Pickleweed (*Salicornia virginica*) actually requires saline conditions for growth. It can form extensive monocultures in saline marshes, but is mainly restricted to drainage divides and upland boundaries of brackish marshlands. Cattails (*Typha* spp.) are abundant and often occur with tules. The hybrids among the local species of cattails (*T. latifolia, T. augustifolia, T. domingensis,* and *T. glauca*) are difficult to identify in the field, however, which complicates the analysis of their regional distribution or stature. Both Olney bulrush (*Scirpus olneyi*) and reed (*Phragmites communis*) are locally abundant in the Suisun Marsh Study Area. The replacement of Olney bulrush by alkali bulrush should be examined as a possible measure of upstream migration of the ecotone between brackish and saline tidal marsh conditions.

(2.1.4) ASSESSING PLANT SPECIES DISTRIBUTIONS

In the absence of paleo-ecological studies (see our Conclusions and Recommendations), our analysis of the response of tidal marshland to regional changes in salinity has depended upon historical observations of changes in the distribution, abundance, or stature of tidal marshland vegetation. A regional survey of intertidal plant species conducted in 1975 and 1976 (Atwater and Hedel, 1976), as annotated by additional investigations (Atwater et al. 1979; personal communication: Brian Atwater), provided the baseline data for our study.

Qualitative surveys of the distribution and relative abundance of the selected plant species were conducted daily on the ground during periods of low tide from 6 July through 16 July 1992. We thoroughly examined the northern shoreline of the Suisun Marsh Study Area from the Benicia Bridge to the mouth of Suisun Slough, and the southern shoreline from the Benicia Bridge to Hastings Slough, to map the upstream limits of cordgrass and die-back of tules. Similar surveys were conducted by power boat from the Highway 37 Bridge to downtown Napa along the Napa River, and from the Napa River upstream about 3 km along Napa Slough and South Slough. The results of these surveys were compared to the results of other surveys conducted by Atwater and Hedel (1976) and Mendelssohn and Winfield (1989-1991). In addition, accounts of past conditions of tidal marsh plant communities were solicited from people especially familiar with the areas we surveyed.

The die-back of tules is obvious at low tide as the presence of dead or dormant tule rhizomes with characteristic culm scars and stumps that protrude above the sediment. The width of the tule die-back zone was rejected as a possible indicator of regional salinity effects because it depends in part upon the shoreline gradient. The die-back zone must be narrower where the shorelines are steep than where the shoreline slopes gradually, regardless of the tidal salinity regime.

(2.1.5) ASSESSING TULE STATURE

Minimal salinity effects were assessed as changes in average stature of mature tule culms between this year and last, in relation to distance upstream throughout the Suisun Marsh Study Area and Napa River. Stature was assessed for the interior reaches of the most robust stands of tules at each study site. The measure of stature was mean culm diameter per 0.25 m^2 of tule stand, at 15 cm above the substrate. Culm diameter at this height was a useful proxy for average culms height, where height is measured as the distance between the substrate and the base of the inflorescence (Figure 2). This assessment assumes that live culms represented growth for this year, and that standing dead culms represented growth from last year.

The easiest method to collect these kinds of data is also accurate. All live and standing dead culms within a 0.25 m^2 plot are cut cleanly and squarely with shears or a knife at the same distance along the culm 15 cm above the substrate. This is simplified by placing a 0.25 m^2 PVC frame with legs 15 cm long onto the substrate over culms initially cut waist high. Then the edges of the square frame can be used as a height reference for the final culm cuts. Care must be taken to assure that dead culms bowed to the ground or leaning over are not cut too short or too long. A white metric rule marked in black is used to measure the maximum diameter of each culm across its cut surface. Values for live and dead culms for each plot are recorded separately.

We collected data for three replicate 0.25 m^2 plots at each site. When possible, plots were located in separate stands or at least 3 m apart in a single stand. Most plots provided at least 30 measures each of live and dead culms. Data were pooled for all three plots at each site. the estimates of mean diameter for live and dead culms at each site were treated as paired values.

The use of calipers to measure the diameter of uncut culms was rejected because it required separate measurement of distance above the substrate for each culm, and maximum diameter of uncut culms was not immediately obvious. These slight difficulties in concert substantially increased the time required to collect data, during a study that began with severe time constraints. The use of calipers and non-destructive sampling could be incorporated into future assessments of this kind.

The meaning of tule stature as an indicator of salinity change is somewhat diminished by local taxonomic uncertainties. The two species, *S. californicus* and *S. acutus*, apparently hybridize within the brackish marshlands, although pure stands of each species also occur (Harvey et al. 1977). According to the taxonomies of Mason (1957), Munz (1959), and

Howell (1970), we expect that the tule stands that we selected for study were mostly *S. acutus*. We used the round cross-section of the distal end of the culm, its dark green color, and the tightly branched panicle as field characters, which in most cases were validated by the presence of barbed bristles of the perianth. These are in some cases the same stands that Atwater and Hedel (1976) concluded were *S. californicus*, however, although their taxonomic identifications were perhaps more tenuous (personal communication: Brian Atwater).

(2.1.6) ASSESSING TIDAL SALINITY

A method to describe salinity regimes for the Suisun Marsh Study Area that has been adopted by the "Salinity, Flows, and Living Resources Workshop" is outlined by Kimmerer and Monismith (1992). In essence, X_2 (the position of the benthic 2 ppt isohaline contour relative to the Golden Gate) is derived from measures of river discharge (i.e., Delta outflow), and past values of monthly, seasonal, and annual values of X_2 have been calculated (Figure 3). These calculations provided a basis to assess salinity changes since the previous surveys of tidal marsh vegetation.

Values of X_2 can also be estimated based upon the position of a "target salinity", which is the salinity value at the water surface that corresponds to the 2 ppt benthic isohaline contour (Kimmerer and Monismith 1992). For low to moderate Delta outflows, the target salinity is about 1.8 ppt. This value is not suitable for high Delta outflows that force vertical stratification of the Estuary. Locating the position of the target salinity involves interpolation among surface salinity stations. Given that monthly Delta outflow has been average or less this year, surface salinity values (USBR stations) were used to estimate the position of the target salinity, and hence X_2, for this spring. The difference in salinity between this spring and last was used to help explain the observed changes in tule stature.

A synoptic survey of tide water salinity for the Suisun Slough system was examined for evidence of local salinity controls (personal communication: Bruce Herbold). The data set comprises almost weekly measures of water salinity at stations throughout the dendritic network of Suisun Slough and its major tributaries, including downstream and headward stations on small channels (second-order) in fully tidal marshland, for the decade 1980 to 1990. Although the data were not standardized with regard to tidal stage or phase, they represent well the seasonal and interannual variations in salinity, including effects of the recent drought, for a major subordinate system of the Estuary. One of us (T.C. Foin) has also received salinity data collected by DWR (California Department of Water Resources) at a variety of stations in the Suisun Marsh Study Area. But these data have not been analyzed.

(2.2) RESULTS AND DISCUSSION

The ability of all estuarine plant species to tolerate conditions unsuitable for growth, and to tolerate such conditions for several years, greatly complicates the use of short-term field surveys to deduce the mechanisms that control vegetation change. The information content of field surveys increases with the period of record, especially if the record comprises frequent entries.

Our intensive study can be referenced to similar studies of the recent past and therefore our study provides useful information about changes in the distribution and stature of tidal marsh vegetation. The results of our study of both the Suisun Study Area and the Napa River, developed over a period of just 10 days, provide evidence of change in tidal marsh plant stature representing at least two years, and changes in the distribution of plant species that span nearly 3 decades.

(2.2.1) TIDAL SALINITY

The average salinity of water within the fully tidal marshlands of the Suisun Bay Study Area was statistically related to position of X_2 (Figure 4). As Delta outflow decreases, and X_2 migrates upstream, the salinity range within the tidal marshlands increases.

The results of the synoptic survey of tide water salinity within the Suisun Slough system clearly indicate local controls on salinity, however (Figures 5A and 5B). Salinities within Boynton Slough (Figure 5A) generally increased with distance downstream, but were lower overall, and the apparent drought effect has been slight, relative to the downstream reaches of Suisun Slough. This can be attributed to the sewage outfall that supplies non-saline effluent to Boynton Slough upstream of the salinity stations. The seasonal pattern of salinity has been variable along Goodyear Slough (Figure 5A). The salinity regime at the downstream station, which is more distant from the numerous connections to diked marshlands, was more similar to the salinity regime at Suisun Slough. Seasonal salinity and the apparent drought effect were more similar among stations at Suisun Slough and the tributaries not subject to local salinity controls, including Peytonia Slough, Cutoff Slough (western reaches), and Spring Branch (Figure 5B). The salinity regime at Peytonia Slough, which is influenced by unknown amounts of urban runoff from Fairfield and Suisun City, were generally low relative to Spring Branch, which is at a comparable distance upstream from the mouth of Suisun Slough but does not directly receive urban runoff.

These results suggest that various local controls on salinity can influence any assessment of regional salinity changes. Sites selected to monitor ecological changes of tidal marshland in relation to regional salinity should not be subject to local salinity control. In this regard, we note the great value of Spring Branch as a headward extension of a relatively natural tidal drainage system with a salinity regime that reflects regional conditions.

(2.2.2) PLANT SPECIES DISTRIBUTIONS

The results of our survey of some common plant species agree with the results of a similar survey conducted in part this year by DWR. The survey by DWR has been more extensive, however, because it has involved more plant species and it has encompassed the natural islands of Suisun Bay. We have included in some cases the results of the DWR survey as provided to us for a more comprehensive description of plant species distributions than would be possible based upon our study alone.

(2.2.2.1) CORDGRASS

Within the Napa River Estuary, the upstream limit of the distribution of cordgrass (*Spartina foliosa*) was discovered about 0.5 km upstream of the Stanley Street Bridge, and cordgrass was encountered throughout the reaches of tributary sloughs that we surveyed (Figure 1). Steep and rocky conditions probably make the river bank unsuitable for colonization by cordgrass for about 2 km further upstream than the Stanley Street Bridge. Thus, saline conditions might extend further upstream than indicated by the distribution of cordgrass.

Within the Suisun Marsh Study Area, the upstream limit of the distribution of cordgrass was discovered less than 0.5 km west of Pacheco Slough, and less than 0.5 km west of the mouth of Sulphur Springs Creek, along the southern and northern shorelines, respectively (Figure 1). These are similar locations along the estuarine gradient, relative to the Golden Gate. In both cases, the limit of the distribution is indicated by a single pioneering colony smaller than 1 m^2 at the immediate margin of the marshland. About 100 m downstream along either shoreline, colonies of similar size are separated from each other by 10 m or less. With increasing distance downstream, colonies coalesce into a continuous band of cordgrass 1-2 m wide. We did not observe cordgrass along the tidal channels within marshland bordering Suisun Bay east of the Benicia Bridge.

Information about the previous distribution of cordgrass along the southern shoreline is equivocal. Cordgrass is abundant along the immediate shoreline of the accreting delta of Alhambra Creek, but no certain records about the colonization at this area have been recovered. The Martinez shoreline, including Alhambra Creek, was not included in the previous survey of Atwater and Hedel (1976).

Changes in the distribution of cordgrass along the northern shoreline of Suisun Bay are better documented. Atwater and Hedel (1976) did not observe cordgrass at Southampton Bay, although it might have been present as a pioneer (personal communication: Brian Atwater). Atwater did observe cordgrass at Southampton Bay in 1977, however, (Atwater et al. 1979), which he has attributed to the severe drought of that year. This record suggests that the distribution of cordgrass has extended upstream about 9 km in about 15 yr, at an average rate of about 0.6 km/yr.

This is probably a conservatively low estimate of the rate of upstream invasion by cordgrass. Ecological succession, as possibly affected by changes in salinity regime, is unlikely to be constant. Interannual variations in sea level rise and river discharge affect variations in the rate of salinity change. Assuming that the rate of upstream invasion by cordgrass is positively correlated to the maximum upstream excursion of saline tides, then the invasion was probably retarded between 1982 and 1985, when salinity values in the Suisun Bay Area were perennially low (Figure 3). Most of the observed upstream migration of cordgrass probably occurred during fewer than 15 years.

The DWR survey has also discovered cordgrass on the southern shorelines of Ryer, Roe, and Seal Islands (Grewell 1992). These populations are represented by pioneer colonies smaller than 1 m^2, and suggest that upstream migration of cordgrass has been more rapid along the mainstem course of tidal excursion through Suisun Bay than along the more peripheral courses.

We expect that invasion by cordgrass reflects competitive release by the tule die-back, and that the pattern of tule replacement by cordgrass might be reversed with sufficient numbers of successive years of low salinity. Cordgrass is probably not eradicated by decreases in salinity, however. Reciprocal increases in cordgrass and decreases in tules observed at Southampton Bay during the drought of 1977 reversed during subsequent years of abundant rainfall (personal communication: Brian Atwater), but tule abundance has again decreased during the more recent drought (this study and Grewell 1992).

The extended distribution of cordgrass upstream within the Estuary might indicate an expansion of habitat for the California Clapper Rail, a Federal and State Endangered Species. Surveys of Clapper Rails conducted during spring 1992 indicate that their upstream distribution corresponds closely to the upstream limits of cordgrass. Clapper Rails are not restricted to cordgrass, however, and they have been recorded during previous springs as far upstream as Hill Slough (Collins and Evens 1992, Evens and Collins 1992).

(2.2.2.2.) TULES

The die-back of tules (*Scirpus acutus* and *S. californicus*) that we observed supports the speculation by Atwater and Hedel (1976) that the maximum amount of tidal submergence tolerated by tules is inversely proportional to tidal salinity regime. In general, the amount of die-back decreases with distance upstream.

The die-back was apparent along 0.8 km of the Napa River from the southeast shoreline of Coon Island upstream to about 0.5 km beyond Rockstram (Figure 1). Longtime residents of the lower Napa Valley do not recall any sudden changes in water quality since the construction of oxidation ponds by the Napa Sanitation District during the late 1960's, which was reported to return the Napa River to "normalcy" (p. 23, CDFG 1977), with markedly improved water quality. An historical trend of increasing salinity is expected for the Napa River due to the large increases in consumption of Napa River water by agriculture and urbanization during this century (Faye, 1972).

Most of the tule die-back along the Napa River is represented by a zone less than 0.5 m wide on steep levees. At the eastern entrance to Napa Slough, near the downstream limit of the die-back, the shoreline slopes gradually to a broad mudflat, and the die-back zone is about 2 m wide.

Within the Suisun Marsh Study Area, the die-back of tules was obvious at Southampton Bay and along the northern shoreline from the Benicia Bridge to about 1 km west of the Navy wharf at Lake Herman Road (Figure 1). These reaches of the shoreline slope gently and rather uniformly to broad mudflats. At Southampton Bay and the Benicia Bridge, the average width of the die-back zone exceeded 3 m. With increasing distance upstream from the Benicia Bridge to the Navy Wharf, the width of the die-back zone gradually decreased.

The die-back was also apparent along the southern shoreline of Suisun Bay from the mouth of Alhambra Creek to the Mouth of Pacheco Creek (Figure 1). At Alhambra Creek, the die-back was diffuse, and sparsely interrupted by live tule culms and cordgrass. The shoreline east of the Benicia Bridge is apparently eroding, and the die-back is discontinuously evident in a narrow zone and as blocks of tule rhizomes that have broken from the shoreline.

Any recent assessment of the tule die-back along the southern shoreline as an effect of increasing salinity could be confounded by the impacts of an oil spill and related clean-up efforts that occurred along the shoreline in spring 1988. The Napa River and the northern shoreline east of Benicia Bridge was not affected by the oil spill, however (Mendelssohn and Winfield 1989-1991), which therefore could not account for the die-back observed there.

For both the Suisun Marsh Study Area and the Napa River, erosion is associated with tule die-back. Where the shoreline is steep, horizontal erosion is evident as undercuts below the existing stands of emergent vegetation, and as fallen blocks of tule rhizomes. Where the shoreline slopes gradually, exposure of the dead rhizomes indicates a vertical erosion of about 15 cm of sediment, since this is the average depth of living rhizomes beneath the sediment surface. This vertical erosion occurs where the lower areas of the die-back zone (i.e., the areas of lowest tidal elevation) are below the areas colonized by other vascular plant species, including cordgrass and alkali bulrush.

We observed evidence of substantial recent declines in abundance of tules in the interior regions of some youthful marshlands, including the northern area of Coon Island along the Napa River, and the marshlands just east of the Benicia Bridge. These declines were evident as abundant tule detritus (i.e., dead culms) amongst dead or dormant tule rhizomes, extending onto the marsh plain away from channels large and small. Since the youthful marshlands are low in tidal elevation, relative to mature marshlands, they are subject to greater tidal inundation. This might account for the extension into the interior reaches of marshland of any tidal controlled conditions that cause tule die-back.

In the Suisun Marsh Study Area, declines in the stature or amount of cover of tules along tidal channels have probably affected declines in abundance of the Suisun Song Sparrow, a Federal Candidate for Endangered Status and a State Species of Special Concern (Marshall and Collins, in preparation). This is because nesting and feeding by Suisun Song Sparrows is mainly restricted to tidal marshland with dense stands of tules (Marshall 1948). The total extent of declines in habitat due to changes in vegetation have not been assessed.

(2.2.2.3) ALKALI BULRUSH

The distribution of alkali bulrush (*Scirpus robustus*) is rather discontinuous along the immediate shoreline of the downstream reaches of the Suisun Bay Area and all along Napa River. A general pattern is apparent that suggests correlation to intermediate changes in salinity regime, lesser than the salinity changes relating to upstream invasion of cordgrass, subequal to the changes relating to die-back of tules, and greater than changes in salinity that relate to interannual decreases in tule stature.

Areas of local extinction or severe die-back of tules, as indicated by the presence of abundant dead or dormant tule rhizomes, near the downstream limits of brackish conditions have been colonized in the upper intertidal zone by alkali bulrush. In these areas, such as Southampton Bay at Carquinez Strait and Coon Island along the Napa River, the alkali bulrush is accompanied by sparse pickleweed (*Salicornia virginica*) and is bordered at lower elevations by cordgrass. Further upstream, beyond the upstream limits of cordgrass and where the tule die-back is less severe, the alkali bulrush and tules of slight to moderate stature occur together along the shoreline, with pure stands of alkali bulrush just upslope. Horizontal erosion in these areas can leave alkali bulrush as the dominant shoreline vegetation. Accreting shorelines are usually not bordered by alkali bulrush alone. With increasing distance upstream, alkali bulrush decreases in abundance along either eroding or accreting shorelines, and it occurs less frequently in mixed stands with tules. Further upstream, alkali bulrush is restricted to the headward reaches of rills or first-order channels in the interior reaches of the tidal marshland.

We did not discover the exact upstream limit of alkali bulrush for the shorelines of either the Suisun Marsh Study Area or the Napa River (Figure 1). Along the southern shoreline of Suisun Bay, we encountered alkali bulrush at the mouth of Hastings Slough, but not at Tug Boat Pier at Port Chicago. The DWR survey encountered alkali bulrush further upstream at the McAvoy Harbor (Grewell 1992). Along the northern shoreline, we encountered alkali bulrush near the mouth of Suisun Slough, but not at Spring Branch of First Mallard Slough. Alkali bulrush was also observed at the toe of a new flood control levee along the downstream reach of Ledgewood Creek, upstream of Peytonia Slough. We found abundant alkali bulrush throughout most of our survey of the Napa River.

Based upon physiological experiments of the effects of salinity on growth, Pearcy et. al (1982) predicted that alkali bulrush would be found where wet season salinities were low enough to permit growth and dry season salinities were high enough to exclude glycophytes. To the best of our knowledge, this prediction has not been tested.

(2.2.3) CHANGES IN TULE STATURE

The mean diameter of tule culms was greater this year than last, and increased with distance upstream for both years. This pattern was evident along the Napa River and throughout the Suisun Marsh Study Area. Along the Napa River, from the mouth of the River to the city of Napa, the upstream increase in culm diameter was exponential, but the difference in culm diameter between this year and last decreased linearly (Table 1 and Figure 6). Within the Suisun Bay Study Area, from Carquinez Strait to the confluence of the Sacramento and San Joaquin Rivers, both the upstream increase in culm diameter (Table 1 and Figure 7) and the change in culm diameter between this year and last were linear (Figure 8). Although the salinity data are few, positive correlations between spatial or temporal changes in salinity and tule stature are evident.

The obvious implication of these data is that maximum culm diameter is sensitive to salinity. The rate of change in culm diameter with distance upstream might be related to spring and summer salinity gradients. Interannual variations in culm diameter, especially within the downstream limits of the brackish region, might reflect interannual variations in salinity.

However, the observed changes in tule stature between this year and last might involve more than just annual variability in salinity effects. The small stature of the culms for last year probably reflect the cumulative effects of 5 years of drought. The change in culm stature between this year and last was probably greater than would be expected for most years.

Table 1. Codes, names, and estuarine positions (i.e., distance in km from Golden Gate) of shoreline stations used to assess changes in tule stature, as indicated by mean values of mature culm diameter for this year (with standard error of mean given parenthetically), and annual change in mean culm diameter (i.e., this year minus last year).

Code	Name	Distance (km)	Stature (mm) 1992	Stature 1992-1991
N1	White Slough	50	11.7 (1.8)	5.0
N2	Coon Island	60	12.3 (3.2)	4.5
N3	Cutting's Wharf	65	13.2 (2.8)	4.8
N4	Napa Town	70	18.2 (3.3)	3.8
S1	Southampton Bay	45	11.4 (4.6)	5.5
S2	Alhambra Creek	50	15.1 (4.3)	6.9 *
S3	Benicia Bridge	55	10.2 (3.4)	4.9
S4	Pacheco	58	14.5 (4.0)	7.0 *
S5	Goodyear Cut	60	11.1 (3.1)	5.0
S6	Port Chicago	60	15.6 (3.4)	4.5
S7	Simmons Island	70	13.4 (1.9)	3.3
S8	Power Plant	70	15.4 (2.4)	2.3
S9	Montezuma West	80	14.9 (5.2)	3.8
S10	Spring Branch	85	14.9 (3.3)	1.3
S11	Control Gate	80	15.5 (3.3)	2.1
S12	Collinsville	80	16.2 (3.7)	1.5
S13	Antioch	85	18.7 (3.9)	-0.8

* *Tidal salinity regime is subject to local fluvial influence during the wet season.*

(2.2.4) RATE OF UPSTREAM ECOLOGICAL SUCCESSION

As discussed previously, a reasonable estimate of the average rate of estuarine transgression during recent millennia is about 2 m/yr. The rate of upstream ecological succession can be accelerated by drought, however, such that the rate of succession exceeds the rate of estuarine transgression. According to the paleo-record for the millennia, the rate of upstream ecological succession has averaged about 30 m/yr. During the most recent droughts, the upstream invasion by cordgrass indicates that the rate of upstream ecological succession can accelerate to almost 600 m/yr or 20x the average! Whether this accelerated rate of upstream migration of saline tidal marsh conditions has precedent in the natural record is unknown.

CONCLUSIONS AND RECOMMENDATIONS

(3.1) *CONCLUSIONS*

The basic conclusion supported by the literature and our field studies is that the major vascular plant species of the tidal marshlands are arrayed along the axial salinity gradient of the Estuary. The similarity of vegetation patterns along the Napa River and throughout the Suisun Marsh Study Area suggests similar dynamic control of plant community structure. This also suggests that studies of vegetation can be used to predict and monitor the ecological effects of changes in salinity regime.

Not all plant species are so neatly arrayed, however. The functional role of salinity as a causative agent for the distribution and abundance of plant species may differ within and between marshlands. In fact, although aqueous salinity is important, it has not been identified as the direct causative factor. For example, although upstream invasion of cordgrass and tule die-back have been observed commonly, the role of competition between these species and the possible dormancy of the tule rhizomes has not been investigated. Furthermore, the relative importance of sea level rise, drought, or regulation of riverine throughputs as causative agents for the observed changes in tidal marsh vegetation has not been determined.

While clear and unambiguous standards for water salinity in tidal marshlands might be difficult to set, the evidence suggests that salinity does play a major role in plant abundance and zonation. Indeed, variation in salinity could be essential to maintain natural diversity of the tidal marsh plant community.

Both the theoretical framework and practical methodology exists to address these topics now.

(3.2) *RECOMMENDED RESEARCH*

The protection and restoration of tidal marshland are patently desirable. The existing marshlands are mostly small and isolated fragments that support remnants of native communities including small populations of plant and animal species that are threatened or endangered by the loss of habitat. The reclamation of marshlands and floodplains has greatly constrained the opportunities for upstream evolution of intertidal habitats. Even in the absence of estuarine transgression, the amount of existing tidal marshlands is inadequate to assure the conservation of the endemic tidal marsh communities.

Conservation of the living natural resources of the tidal marshlands will require understanding the evolution of habitats as well as the immediate ecological interactions that momentarily sustain plants and wildlife. To achieve such understanding we recommend the following research.

1. A detailed chronology that relates estuarine transgression to sea level rise and river discharge should be developed based upon a study of the paleo-migration of brackish vegetation. This research is critical to establish a baseline for predicting the effects of accelerated sea level rise or sustained change in river discharge. The possibility of creating a record of historical changes in tule stature, based upon the size of detrital culms that accumulate as peat beneath living tule stands, should also be explored.

2. Future changes in the distribution and stature of native vegetation along the shoreline of the Estuary should be used to monitor the effects of changes in aqueous salinity. A continuum of effects could be monitored. The greater effects could be assessed as changes in the upstream limits and total acreage of cordgrass, tule die-back, and alkali bulrush along accreting shorelines of tidal marshlands between the Benicia Bridge and Browns Island. These effects should be assessed once each decade, and perhaps more frequently during periods of drought. The lesser effects could be assessed as interannual changes in the stature of tules. This effect should be assessed every summer at fixed locations along the southern and northern shoreline of the Estuary, between the Benicia Bridge and Browns Island. Monitoring stations should not be subject to local salinity controls.

3. The functional relationships among biotic and abiotic processes that control the physical form and ecological functions of tidal marshland as a whole should be assessed. An understanding of the functional relationship between growth of vascular vegetation and the evolution and maintenance of tidal marsh channels is especially important.

4. The program to predict or monitor salinity effects should be combined with experimental manipulation of plant species in the field, including especially reciprocal transplantations with respect to tidal elevation, and experimental estimation of competitive interactions and rates of displacement among major plant species. This research is critical to understand the maximum and minimum responses of shoreline vegetation to expected changes in salinity regimes.

5. Experiments should be conducted to quantify the effects of salinity and inundation regimes on food chain support functions of tidal marshlands, including especially the support of rare and endangered species.

6. The potential effects of large-scale restoration of tidal marshland on tidal prism and salinity intrusion within the Estuary should be evaluated. In combination with hydrodynamic models of the Estuary, the recommended studies of paleo-ecological succession and the salt tolerance of shoreline vegetation could be used to estimate the amount of river discharge required to counter sea level rise and the increase in tidal prism that might result from the necessary restoration of tidal marshlands.

(3.3) RECOMMENDED MONITORING PLAN

As a means of assisting policy makers and resource managers in decisions regarding the ecological health of the Estuary, relationships between seasonal salinity regime and diagnostic estuarine properties should be quantified. Each relationship should represent one property as a continuous function of the position of X_2. A diverse array of relationships representing all trophic levels and major processes of the estuarine ecosystem should be developed. Assuming that vegetation strongly influences the food web support functions, nutrient dynamics, and sediment budgets of tidal marshlands, and that the marshlands influence these same functions for the Estuary as a whole, then relationships between X_2 and tidal marsh vegetation should be part of the basis for any comprehensive by any management or monitoring plan.

Some particular aspects of the nature of tidal marsh vegetation can complicate the development and interpretation of models relating vegetation to salinity. Since tidal marsh plants can discount short-term conditions in favor of long-term smoothing, instantaneous responses by plants are not reliable indicators of trends, and temporal lags in response can be expected. Values of X_2 position for perhaps a large number of consecutive growing seasons with corresponding measures of vegetation might be required to account for lags in continuous functions that serve as predictive models. As we previously suggested, the length of record required to determine the relative influence of sea level rise, drought, and regulation of river inputs on plant community structure can be achieved in the near future by an investigation of the distant past, through paleo-ecological studies. Furthermore, salinity effects would involve spatial shifts in species distributions rather than regional extinctions, and this might blur the distinction between negative and positive salinity effects.

In the absence of data that explain how the survivorship of plant or animal species is actually affected by salinity, we recommend quantification of statistical correlations between changes in plant community structure and either salinity regime or river throughputs. The correlations should be quantified through time and with distance along the axis of the Estuary. Statistical relationships that could be represented in a regional monitoring plan include the following:

1. Average rate of change in tule culm diameter with upstream distance as a function of average springtime X_2 position (minimum salinity effect);

2. Rate of upstream replacement of tules by alkali bulrush as a function of average springtime X_2 position (intermediate salinity effect);

3. Rate of upstream invasion of cordgrass or tule die-back as a function of average springtime X_2 position (maximum salinity effect).

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PERSONAL COMMUNICATIONS

Brian Atwater, PhD., USGS (United States Geological Survey), University of Washington, Seattle, WA.
1980 to present: various communications and field trips regarding geomorphology and ecology of tidal marshlands, including field trip on 17 July 1992 dedicated to discussion of salinity effects on tidal marsh vegetation.

Bruce Herbold, PhD., EPA (Environmental Protection Agency), 75 Hawthorne St., San Francisco, CA.
5 June 1992: reviewed and acquired his data record for salinity in Suisun Slough and its tributaries.

FIGURE LEGEND

Figure 1. Map of Napa River Estuary and the Suisun Marsh Study Area.

This figure shows the general shoreline of the Napa River Estuary and the Suisun Marsh Study Area, with the reaches of tule die-back indicated in bold line, the reaches with some pure stands of alkali bulrush as the only shoreline vegetation indicated by dotted lines, and the approximate upstream distribution of cordgrass indicated by asterisks. These distributions are described more comprehensively in the text. Question marks indicate that the limit of die-back of alkali bulrush as a shoreline dominant is uncertain. Alpha-numeric codes refer to shoreline stations used to assess regional patterns of tule stature. These stations are described in Table 1.

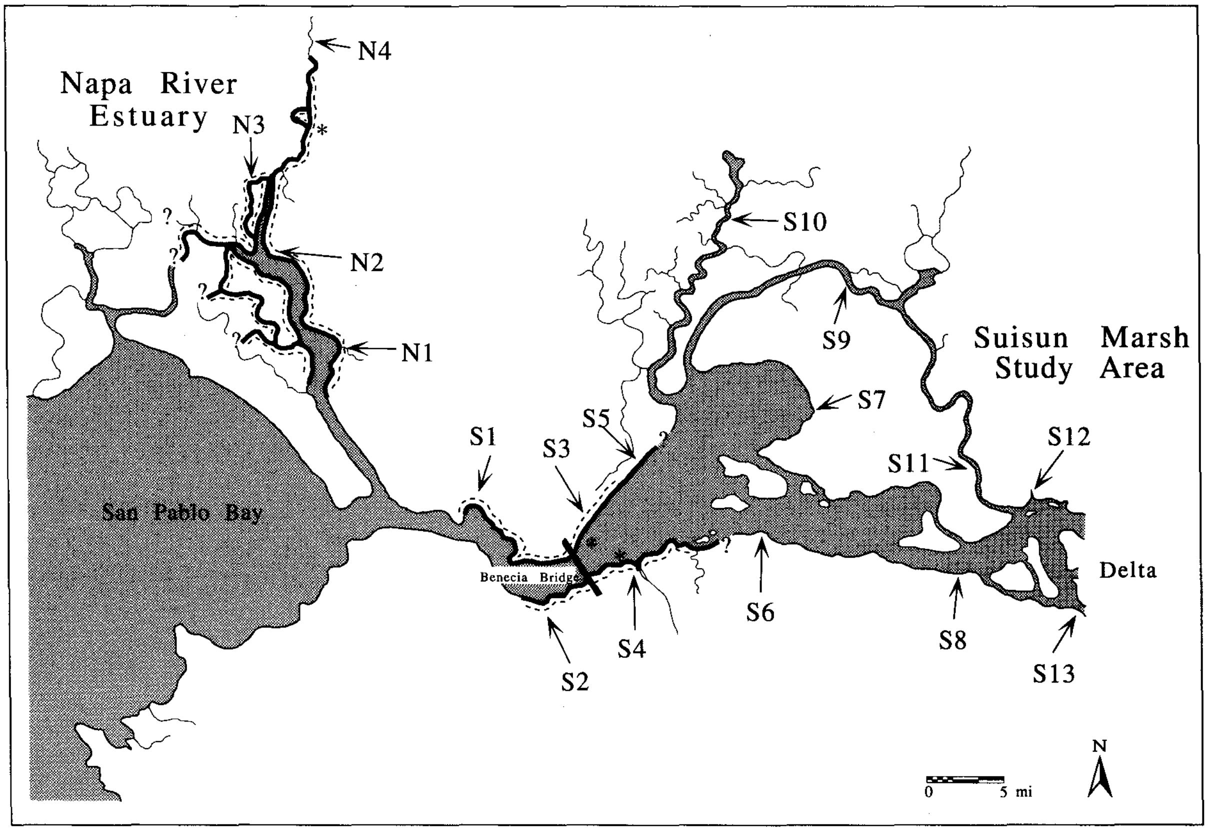


Figure 2. Correlation between Tule culm height and culm diameter.

This figure shows that tule culm height and diameter are similarly related for standing dead and live culms, which suggests that the dead culms retain their stature. The data set was developed from samples of culms collected throughout the Suisun Marsh Study Area.

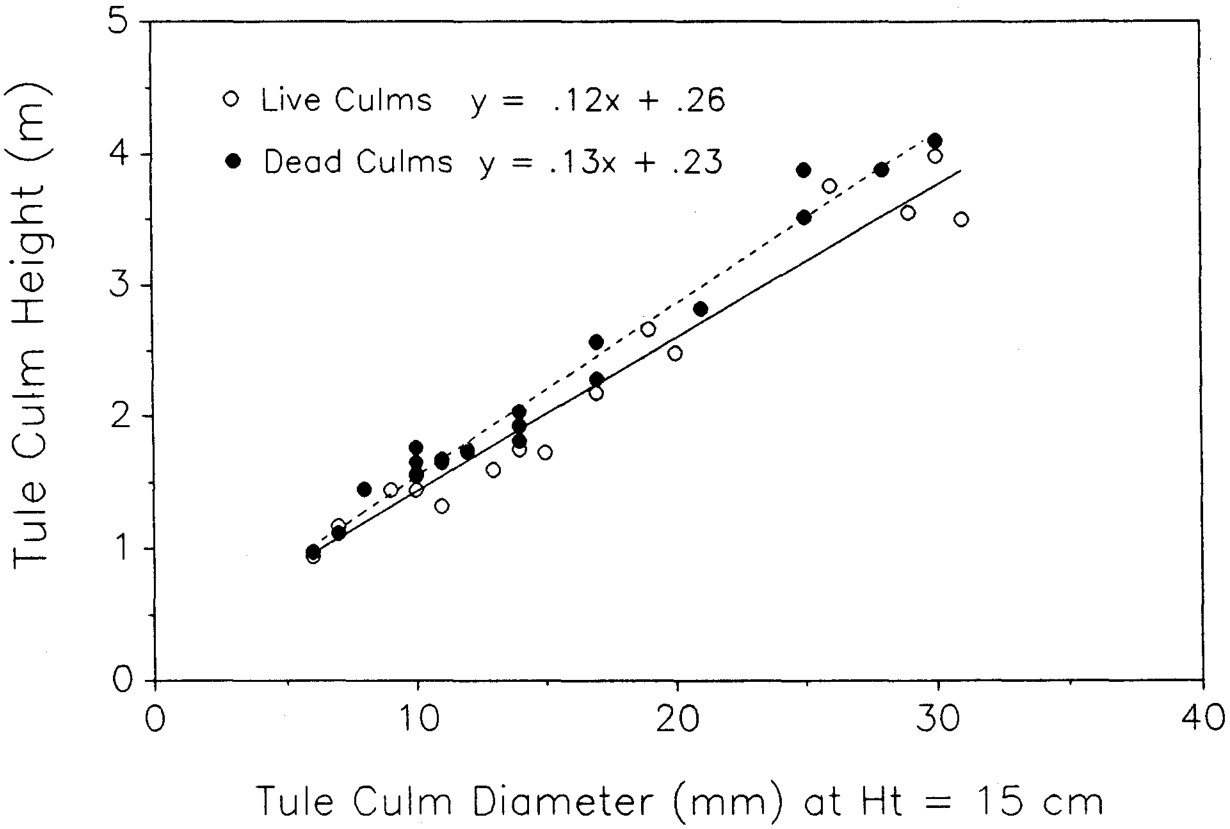


Figure 3. Springtime salinity regimes for the Suisun Marsh Study Area.

This figure shows the springtime salinity regime as X2 for years 1968 through 1992. Lower values (positions) of X2 indicate fresher conditions. Note that the regime was much fresher this year (1992) than during the previous 5 years. This decrease in springtime salinity might account for the greater stature of tules this year compared to last year (see Table 1).

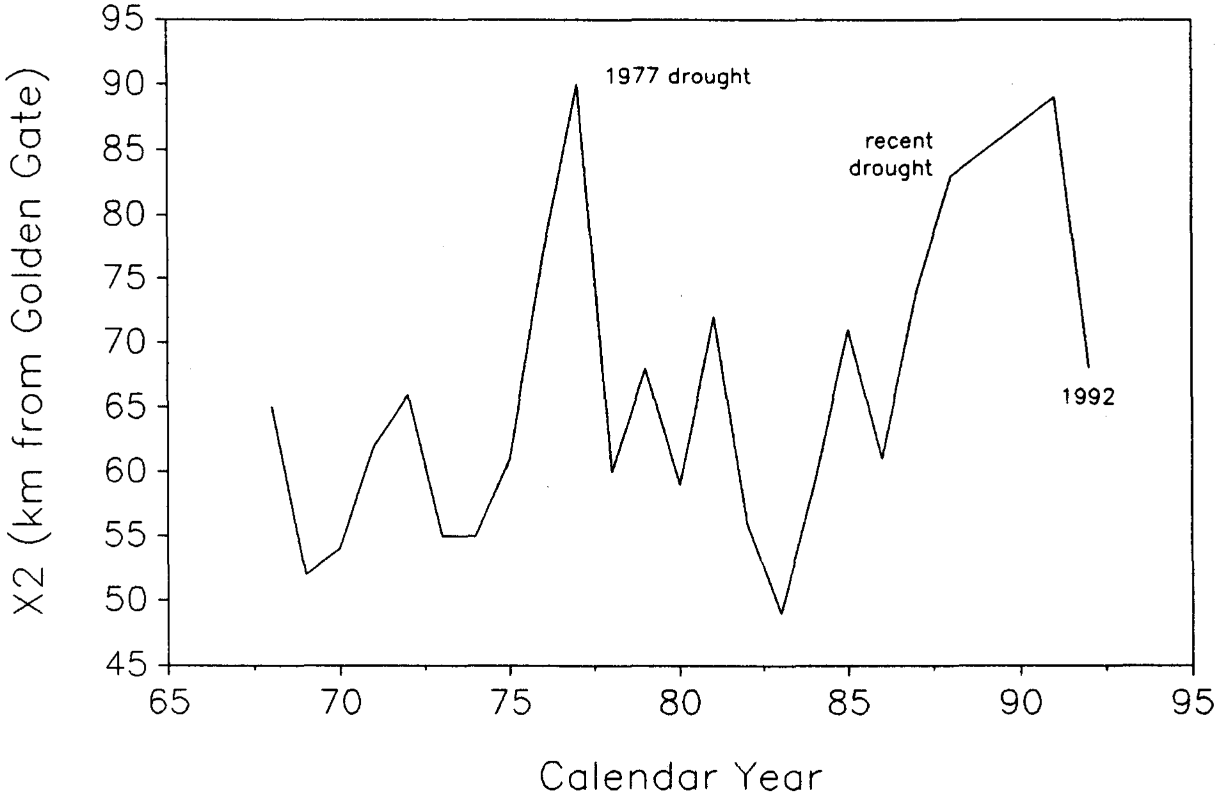
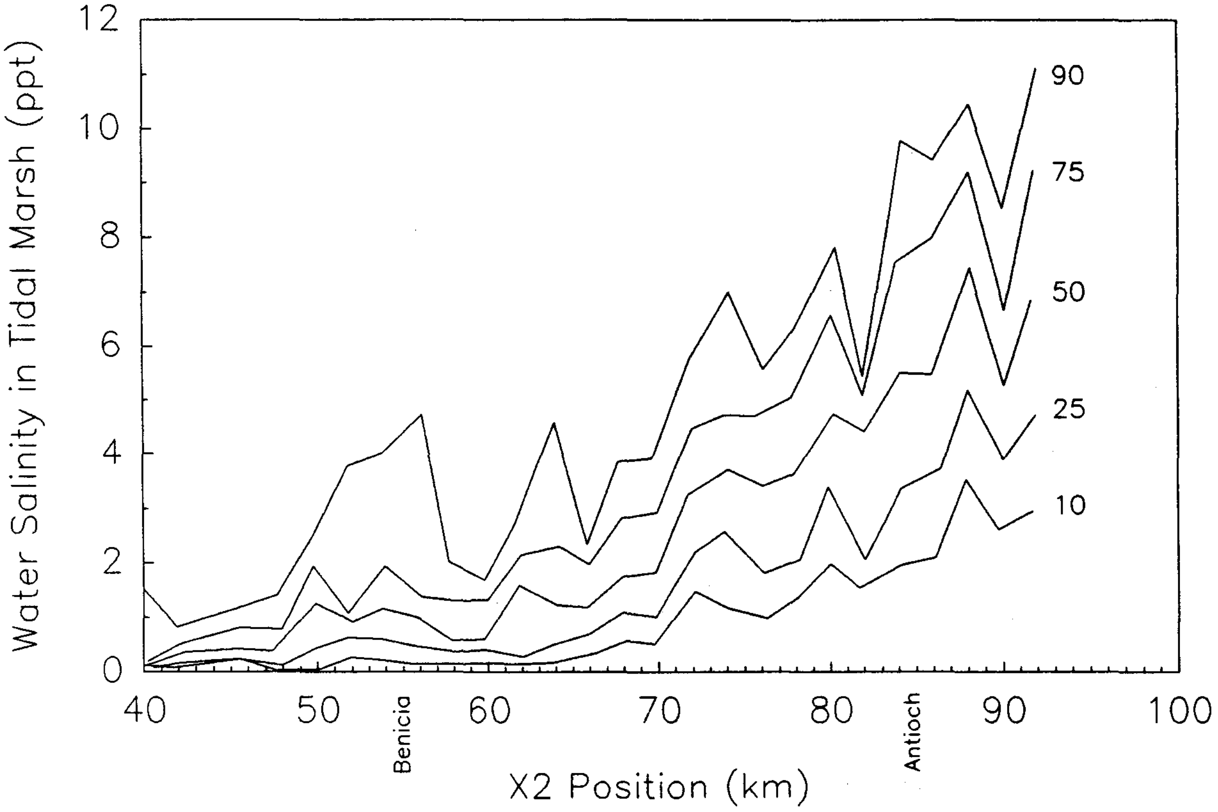


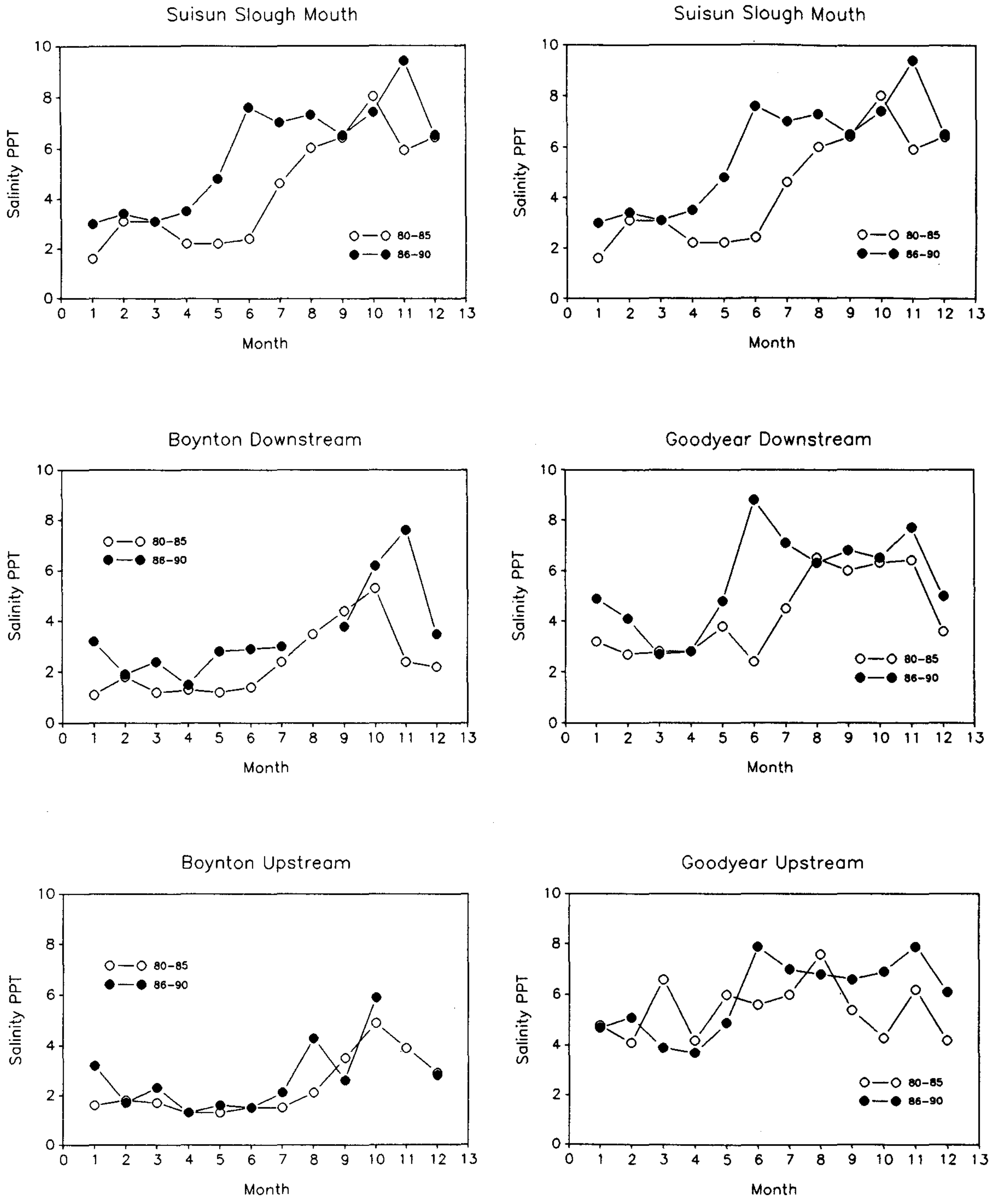
Figure 4. Relationship between water salinity in Suisun Marsh and X2.

This figure shows the 10-90 percentiles of average salinity for tidal areas of Suisun Marsh as a function of X2. Note that as X2 migrates upstream (increasing values of X2), the range of salinity increases.



Figures 5A. Evidence of local salinity controls.

Figure 5A shows mean monthly salinity patterns at the mouth of Suisun Slough (top two graphs) and at two locations within two tributary systems, Boynton Slough and Goodyear Slough (bottom four graphs), for periods before and during the recent drought. Salinities within Boynton Slough are generally less and the apparent drought effect is slight. This can be attributed to the sewage outfall that supplies non-saline flows upstream of the sampling stations. The erratic pattern of seasonal change in salinity within Goodyear slough is probably due to the exchange of water between Goodyear slough and adjacent diked lands.



Figures 5B. Evidence of local salinity controls.

Figure 5B shows obvious drought effects and similar seasonal trends in salinity throughout natural tributary systems of Suisun Slough that are not substantially influenced by water management. All data were provided by Bruce Herbold (see list of personal communications).

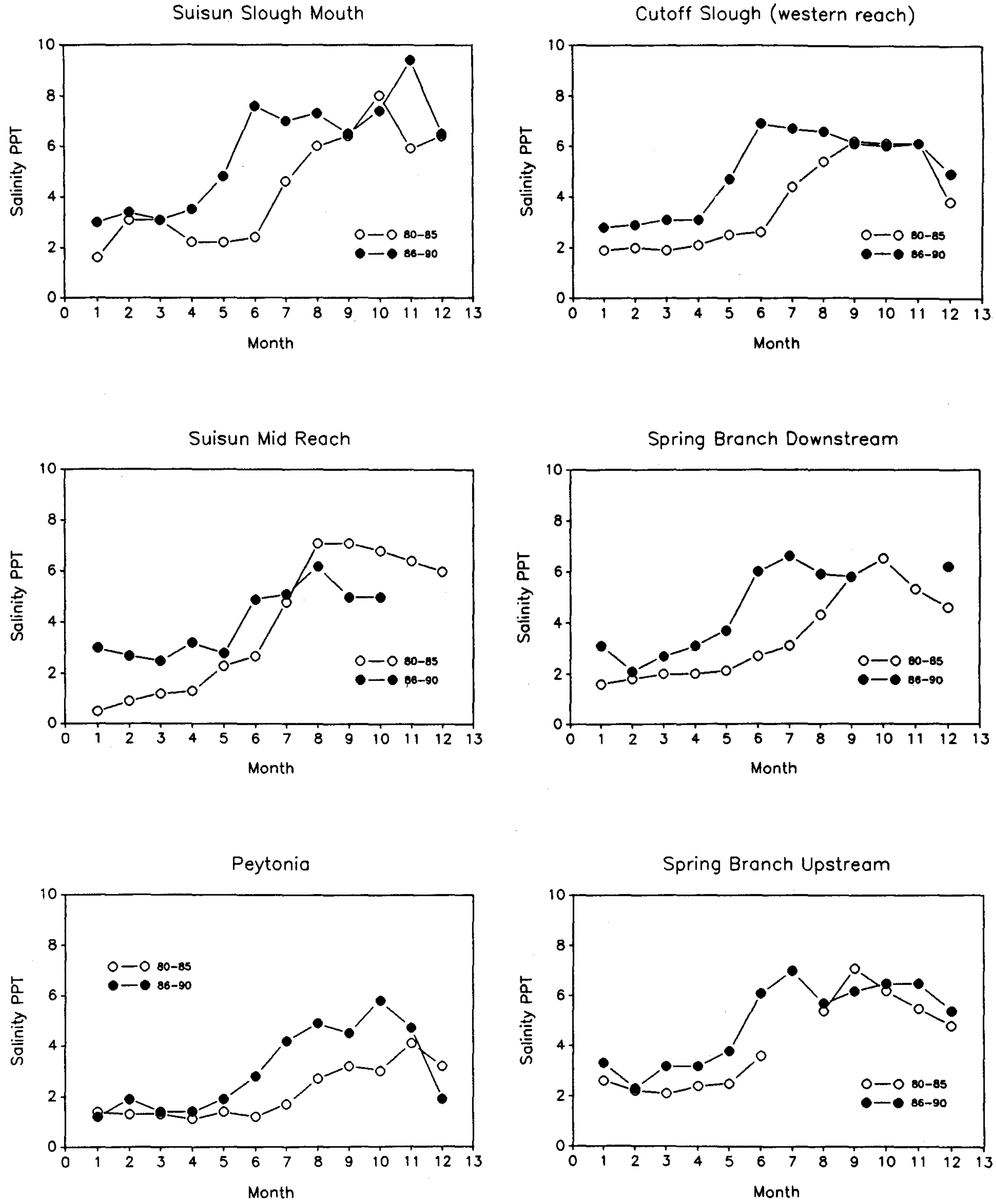


Figure 6. Tule stature along the Napa River Estuary.

This figure shows the trend of change in tule culm diameter with distance upstream along the Napa River during 1992, and the upstream change in culm diameter between 1991 and 1992. Error bars represent one standard deviation of the mean.

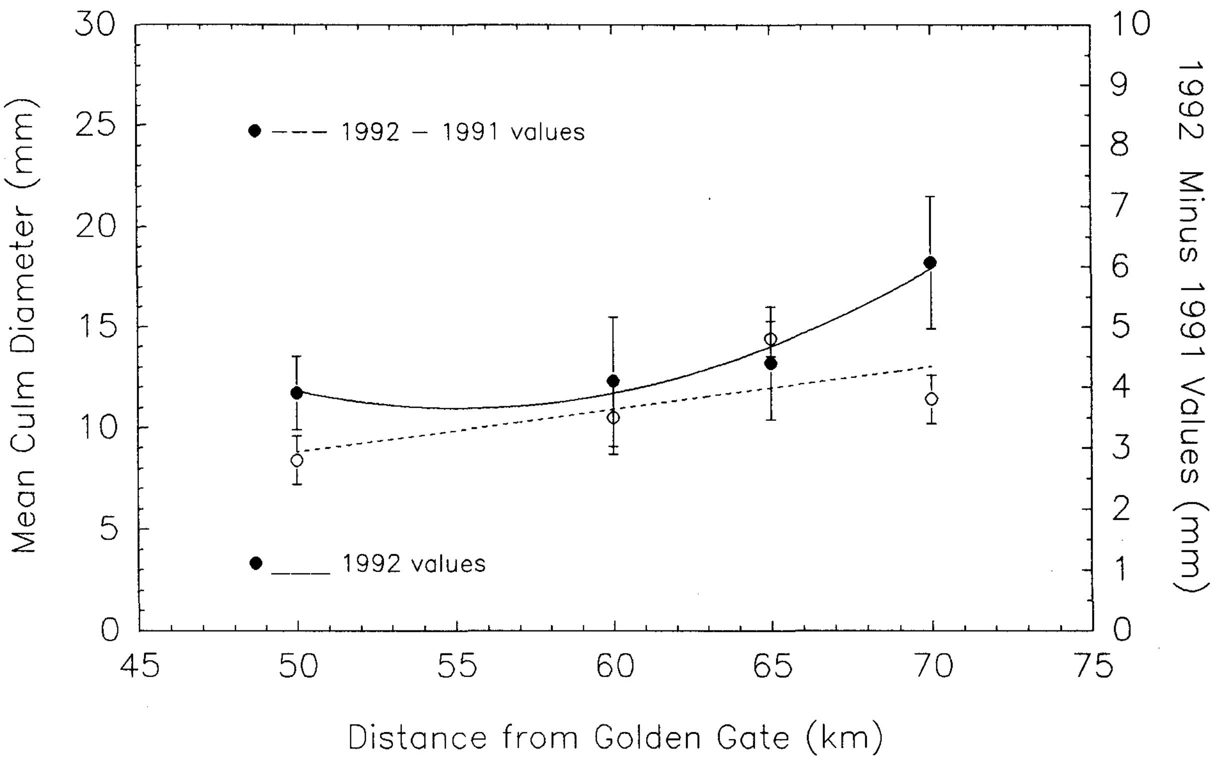


Figure 7. Tule stature for the Suisun Bay Study Area.

This figure shows the trend of change in tule culm diameter and water salinity with distance upstream from Carquinez Strait during spring 1992. Each value of mean culm diameter is based upon three replicate sample units at a single station. Each mean salinity value is based upon the monthly means for March through May 1992. Error bars represent one standard deviation of the mean.

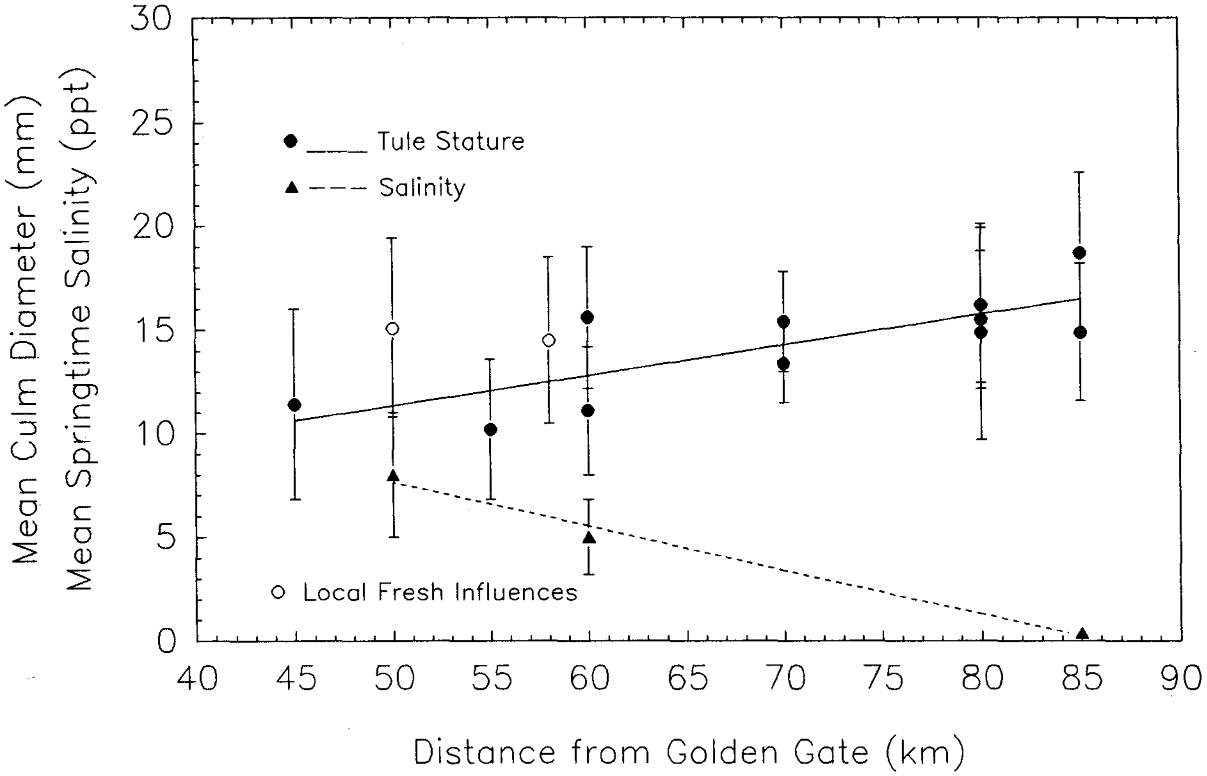
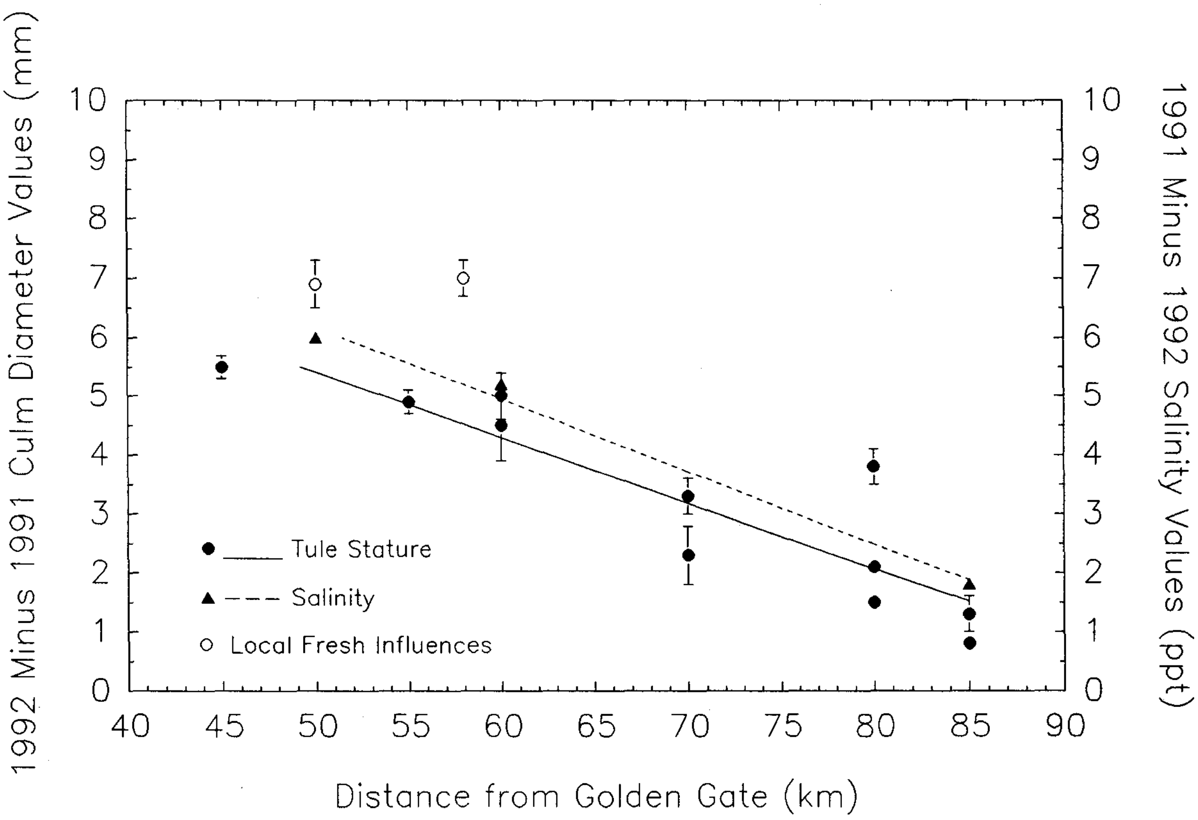


Figure 8. Change in maximum tule stature and springtime water salinity between 1991 and 1992 for the Suisun Bay Study Area.

This figure shows that the trend of upstream change in tule culm diameter and springtime water salinity between 1991 and 1992 were similar. Each mean value for change in culm diameter is based upon paired values for dead and live culms for three replicate sampling units at a single station. Stations subject to local freshwater influences were excluded from the analysis. Error bars represent one standard deviation of the mean change in culm diameter. Variability of change in salinity could not be assessed because monthly values were available for only a single year (1992).



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APPENDIX E

SOME KEY DOCUMENTS WHICH PROVIDE THE SCIENTIFIC RATIONALE FOR THE CONCLUSIONS AND RECOMMENDATIONS OF THIS REPORT

Cloern, J.E., et al., 1992. Summary of 21 January 1992 Meeting Regarding the Rationale/Utility of a "Matrix" Format to Describe the Functional Responses of the San Francisco Bay-Delta System to Fluctuations in Delta Outflow. Prepared for the San Francisco Estuary Project. 3 pp.

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