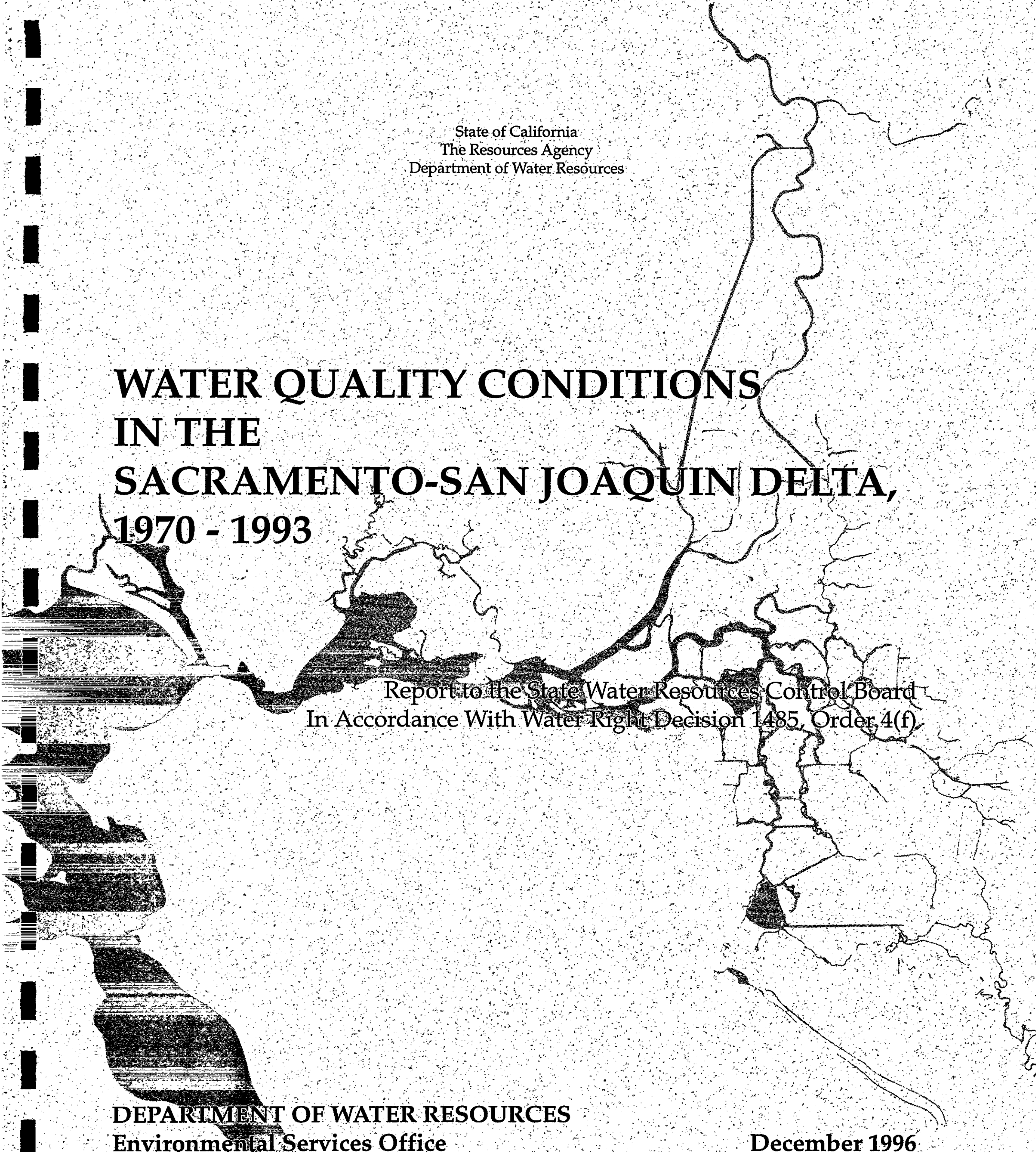
State of California
The Resources Agency
Department of Water Resources

WATER QUALITY CONDITIONS IN THE SACRAMENTO-SAN JOAQUIN DELTA, 1970 - 1993



Report to the State Water Resources Control Board
In Accordance With Water Right Decision 1485, Order 4(f)

DEPARTMENT OF WATER RESOURCES
Environmental Services Office

December 1996

Douglas P. Wheeler
Secretary for Resources
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FOREWORD

The California State Water Project and the federal Central Valley Project are multipurpose systems that, in addition to their primary role of providing a critical water supply, provide flood control, power, and recreation. Operation and management plans for these projects must provide for environmental needs and be developed with an understanding of potential impacts on the aquatic ecosystem.

These projects have been operated in accordance with State Water Resources Control Board Decision 1485 (August 1978) and its predecessor, Decision 1379 (July 1971). These decisions established water quality standards to protect beneficial uses of water supplies in the Sacramento-San Joaquin Delta and Suisun Marsh. They included a monitoring mandate to ensure compliance with these standards, identify changes potentially related to project operations, and determine the effectiveness of the Delta Water Quality Control Plan in preserving Delta and Suisun Marsh water quality.

This program and associated special studies have helped managers better understand the effects of project operations on the Delta's ecology and have provided information that will help determine future operation criteria.

This report is a summary of data collected by the Department of Water Resources and the U.S. Bureau of Reclamation during 1970-1993 and represents an extended version of the annual monitoring reports required by Decision 1485.

RL Brown

Randall L. Brown, Chief
Environmental Services Office

C-036476

CONTENTS

Appendixes

Figures

Tables

ORGANIZATION

STATE OF CALIFORNIA
Pete Wilson, Governor
THE RESOURCES AGENCY
Douglas P. Wheeler, Secretary for Resources
DEPARTMENT OF WATER RESOURCES
David N. Kennedy, Director

Steven L. Kashiwada
Deputy Director

Robert G. Potter
Chief Deputy Director

Carlos Madrid
Deputy Director

L. Lucinda Chipponeri
Assistant Director for Legislation

Susan N. Weber
Chief Counsel

ENVIRONMENTAL SERVICES OFFICE

Randall L. Brown .... Chief

This report was prepared under the supervision of

Harlan Proctor .... Chief, Compliance Monitoring and Analysis Branch

By

Peggy Lehman .... Environmental Specialist IV

With assistance from

Stephen P. Hayes .... Chief, Bay/Delta Section
Heather Peterson .... Environmental Specialist II
Katherine Triboli .... Water Resources Technician II
Murray Clayton .... Scientific Aid
Emiliano Gayton .... Scientific Aid
Jeannie Lee .... Student Assistant

GIS mapping was done by

Sheryl Baughman .... U.S. Bureau of Reclamation
Dave Sullivan .... U.S. Bureau of Reclamation
Ricardo Camarillo .... U.S. Bureau of Reclamation

Phytoplankton and benthic data were analyzed with assistance from

George Weber .... Consultant
Wayne Fields .... Hydrozoology

Data collection over the years has included many employees of the Department of Water Resources and U.S. Bureau of Reclamation.

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SUMMARY

The upper estuary has been characterized as a dynamic system, changing as it responds to the intricate interactions of physical, chemical, and biological forces. This was reflected by the lack of a consistent long-term trend for most variables between 1970 and 1993. Among the physical variables that did demonstrate a trend, total exports, Secchi disk depth, and wind velocity increased over the period of record. The increase in Secchi disk depth was accompanied by a decrease in suspended and volatile solids. Among the biological variables that demonstrated a trend, chlorophyll *a* concentrations decreased in the southern and Suisun Bay regions, and diatoms decreased throughout the upper estuary.

Most changes in physical, chemical, and biological variables were related to the magnitude of inflows. High inflows in the early 1970s and 1980s were associated with low water temperatures and high concentrations of total phosphorus, organic nitrogen, silica, ortho-phosphate, and chlorophyll *a* concentration. Dilution, however, probably produced the low nutrient concentrations during the record high streamflows in 1983. Drought periods in 1976-1977 and 1987-1992 were characterized by high water temperatures, high nutrient concentrations, low chlorophyll *a* concentrations (except in the southern Delta), few diatoms, and reduced aquatic vegetation. The drought periods were also characterized by increased densities of salt-water-tolerant benthic species downstream and by invasions of exotic benthic species.

Most patterns of change in physical, chemical, and biological variables over time were a function of changes in water year type. Water year types were characterized by different environmental conditions, which often varied between upstream and downstream stations. The most uniform conditions among stations occurred during wet and critical years. In wet years, wind velocity, turbidity, and total organic nitrogen were consistently higher than average and water temperature was consistently average. Specific conductance and total dissolved solids were lower than average. Nutrients were variable, but generally differed between upstream and downstream stations. During critical years, nearly all variables increased except for streamflow.

Conditions in normal and dry years were variable among stations. In normal years, water temperature, specific conductance, and concentrations of nitrate and ortho-phosphate were lower than average, but pH and concentrations of dissolved oxygen, volatile solids, and silica were higher than average. In dry years, variables differed between upstream and downstream stations; upstream stations generally had higher air temperature, Secchi disk depth, total organic nitrogen, and nutrient concentrations.

Changes in biological variables also coincided with changes in water year type. The highest chlorophyll *a* concentrations and diatom densities were usually associated with normal years. Chlorophyll *a* concentrations were low for most regions upstream during wet years, when high outflows flush phytoplankton downstream. However, high outflows also increase the downstream range of freshwater organisms like the clam *Corbicula fluminea*, the amphipod *Corophium stimpsoni*, and the diatom *Melosira granulata*. During critical years, chlorophyll *a* concentrations and diatom densities were low for most of the estuary except upstream along the periphery of the delta. In contrast, critical years were accompanied by an increase in marine and brackish water benthic organisms downstream, especially *Potamocorbula amurensis*. Dry years had consistently low chlorophyll *a* concentrations and diatom densities when low streamflows were accompanied by high exports.

Among water years, the magnitude of change among variables differed. Biological variables and streamflow had the largest variation, with maximum deviations from the mean near 200 percent. The next largest variations were for specific conductance and total dissolved solids, which varied up to 100 percent. Variation was about 30% for nutrients, Secchi disk depth, and wind velocity. The smallest variation was associated with air and water temperature, with deviations up to 9 percent.

Most physical, chemical, and biological variables varied with season. During spring, streamflows, wind velocities, and water temperatures were higher than average; Secchi disk depths and nutrient concentrations were lower than average; and diatoms were abundant and coincident with high chlorophyll *a* concentrations. Summer physical and chemical conditions were similar to spring, but deviations from the mean were higher. Diatom numbers were lower than average, and densities of brackish water benthic species and flagellates were higher than average. During winter, Secchi disk depths and concentrations of most nutrients were higher than average, but wind velocities, water temperatures, densities of phytoplankton and benthic organisms, and chlorophyll *a* concentrations were lower than average. Although in the fall most variables changed in the same fashion as in the winter, variation was lower. Aquatic vegetation demonstrated little seasonal variation.

Chapter 1

INTRODUCTION

The 1993 annual Water Quality Compliance Monitoring Report submitted to the State Water Resources Control Board in accordance with Water Right Decision 1485, Order 4(f), has been expanded to provide a comprehensive descriptive summary of the physical, chemical, and biological information collected for this program and its predecessors between 1970 and 1993. This monitoring program has been an integral part of the joint water right permit issued for operation of the State Water Project and Central Valley Project. The Board envisioned information from the monitoring program involving two concepts:

- Establishment of a long-term data base of physical, chemical, and biological data so that timely feedback of changes in environmental and biological variables caused by the projects could be reviewed.
- Specific studies of environmental and biological interactions so that changes revealed by the long-term data base could be better understood.

The wealth of information collected during this period has been helpful in characterizing the upper estuary over a wide range of conditions. The information has been used to:

- Determine compliance with established water quality standards.
- Maintain a comprehensive and available source of information.
- Determine if beneficial uses of Delta waters are being protected from potential impacts of CVP and SWP operations.
- Provide information for forming hypotheses on biological and physical mechanisms that could be tested through special studies.
- Provide data to develop a predictive capability through the use of models.
- Determine if objectives of the Delta Water Quality Control Plan are being met.

The roots of this monitoring program reach back to two earlier programs, the Delta Fish and Wildlife Protection Study and the San Luis Drain Surveillance Program. The Fish and Wildlife Protection Study was a cooperative effort initiated in 1961 between the Department of Water Resources and the Department of Fish and Game. It included a special studies agenda to improve understanding of fish and wildlife requirements in the estuary. The Drain Surveillance Program (1968) was a joint monitoring effort between DWR and the U.S. Bureau of Reclamation to measure baseline water quality in receiving waters prior to construction of a facility that would discharge agricultural drainage into the western Delta.

Both of these programs influenced the monitoring requirements of Water Right Decision 1379, the first water right permit addressing SWP and CVP operations, issued in 1971. During the D-1379 hearing process, a recommendation to extend the Fish and Wildlife Protection Study was approved and formalized by a memorandum of understanding between the Department of Fish and Game, Department of Water Resources, U.S. Bureau of Reclamation, and U.S. Fish and Wildlife Service. This 4-agency group evolved into the current Interagency Ecological Program with nine participating state and federal agencies. The scope of the Drain Surveillance Program was significantly expanded, and the most comprehensive Delta monitoring program to date became the required monitoring provision in D-1379. The compliance monitoring program has been periodically revised; the last major revision was in 1978, when Decision 1485 was issued. Although these two elements have always been closely coordinated, they were formally integrated in 1993 during the reorganization of Interagency Ecological Program. The program is now the Interagency Comprehensive Compliance Monitoring Program.

This report is a summary of the absolute and relative magnitude of change over time for physical, chemical, and biological variables measured at monitoring stations throughout the study area. A description of long-term trends is followed by a description of the relative water year type and seasonal variation. Whenever possible, results are discussed in relation to possible controlling factors. A detailed analysis of interactions among variables is beyond the scope of this report.

ENVIRONMENTAL VARIABLES

Water quality variables are sampled as a requirement of Decision 1485. Changes in water quality are often a function of streamflow, and both are used to assess the impacts of State Water Project and Central Valley Project operations on estuarine biota and compliance with water quality standards. This chapter presents some of these changes.

Data were indexed to water year types. A water year describes precipitation for a period beginning October 1 and ending September 30 of the following year. Water years since 1906 have been classified as wet, above normal, below normal, dry, or critical based on the Sacramento River Index (Figure 1). Because "above normal" or "below normal" years occurred only three times in 1970-1993, data for this study were grouped into wet, normal, dry, and critical years. Data were also indexed to seasons: fall (October-December), winter (January-March), spring (April-June, and summer (July-September).

Water year type and seasonal data were calculated as percent deviations from the long-term average. Deviations were calculated as the difference between the monthly average and the long-term average, divided by the long-term average times 100 for each variable at each station. Positive deviations indicate values are higher than average, and negative deviations indicate values are lower than average.

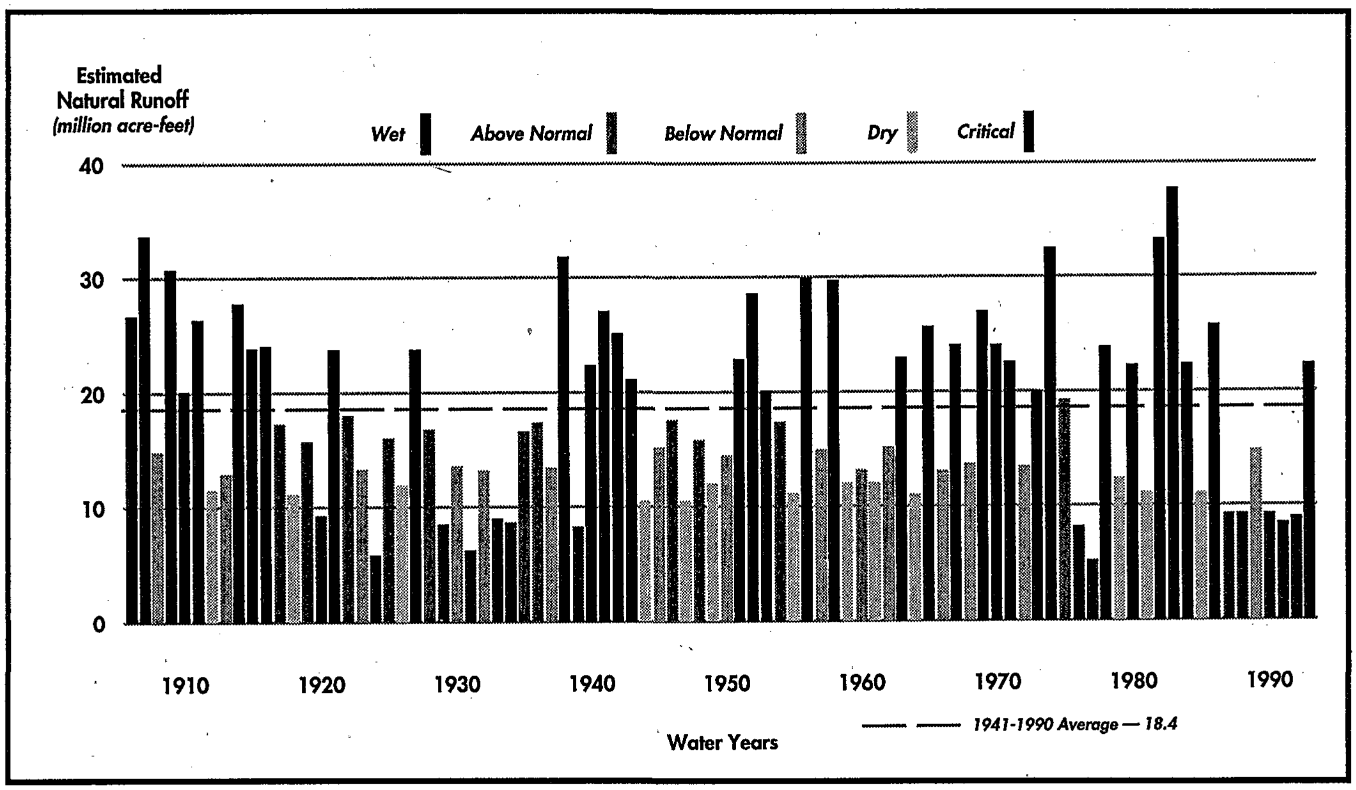

Figure 1
SACRAMENTO RIVER INDEX SINCE 1906
The Sacramento River Index is the sum of unimpaired runoff from the Sacramento River at Bend Bridge, Feather River inflow to Lake Oroville, Yuba River at Smartville, and American River inflow to Folsom Lake.

Streamflow

Long-Term Trends

Unimpaired runoff was variable over the past 100 years, with no indication of a long-term trend (Figure 1). Between 1970 and 1993, inflows were generally higher in the early 1970s and 1980s and decreased by at least a factor of 2 during the droughts of the mid-1970s and late 1980s. Compared with previous 24-year periods, 1970-1993 had more dry and critical years. Dry and critical years comprised ten of the years between 1970-1993, compared with six of the years in 1946-1969, eight of the years in 1922-1945, and three of the years before 1922.

The period 1970-1993 was also characterized by a shift in the relative number of normal and wet years. Wet years were nearly 4 times more frequent in 1970-1993 (Figure 1). This contrasted with 1946-1969, when the number of wet and normal years were equal; with 1922-1945, when normal years were twice as frequent as wet years; and with 1906-1921, when wet years were more frequent by a factor of 2.

The Sacramento and San Joaquin rivers provide most of the inflow to the upper estuary. Most of the inflow in 1970-1993 came from the Sacramento River, where streamflow ranged from 5,000 to 33,000 acre-feet (Figure 2). San Joaquin River streamflow was less than half that of the Sacramento River, with maximum streamflow of 10,000 acre-feet. Streamflows for both rivers were high in the early 1970s and 1980s and decreased by a factor of 2 during the 1976-1977 and 1987-1992 droughts. Overflow from the Sacramento River is diverted into the Yolo Bypass and eventually flows into the upper estuary. Yolo Bypass inflows are equivalent to inflows for the San Joaquin River and had the same pattern of change over time as did the two rivers.

The volume of freshwater inflow to the upper estuary is impacted by diversions of the Central Valley Project and the State Water Project. Exports have increased since the CVP began in the early 1950s and the SWP in the 1960s. In general, average annual exports were slightly higher at the CVP (Figure 2). Combined, the projects have exported 1,000 to 58,000 acre-feet per year, increasing over time from a minimum in the 1950s to a maximum in the late 1980s. Exports decreased in the early 1990s due to a combination of hydrologic conditions, environmental restrictions, and water quality standards.

Water-Year Trends

Streamflow differed among water year types. For wet years, flow in most rivers was higher than average and was associated with outflow that was 85 percent above average (Figure 3). In contrast, lower than average streamflows occurred in the central delta due to the reduced export flows in wet years. For critical years, flow in major streams and tributaries was low and associated with outflow 74% lower than average (Figure 4). Streamflows in the central delta were higher than average due to high export pumping and often produced reverse flow in the rivers, as indicated by the arrows. Streamflow was similar for normal and dry years, when flow was lower than average for most streams and tributaries (Figures 5 and 6). Normal years, however, had relatively higher inflow from the Sacramento River.

CVP/SWP exports also varied with water year type. Exports were highest during dry years when export flows were 14-15% higher than average and inflows were 22-29% lower than average (Figures 3-6). Exports were lowest during critical years when export flows were only 1-2% lower than average. However, exports represented a large portion of the total flow because inflows were 44-67% lower than average. Exports were also low in wet years, when they had little impact on inflows, which were 39-69% higher than average. Exports during normal years were variable, with higher flows at the CVP.

Seasonal Trends

Seasonal streamflow differences were large and associated with precipitation patterns. Only winter months had large and positive deviations for streamflow and precipitation (Figures 7 to 10). Precipitation was also generally higher than average in the fall, but this was coupled with lower than average inflow. Outflow was similar in spring and fall because snowmelt in the spring and precipitation in the fall produced nearly equal inflows. Summer had the lowest precipitation and inflows, which produced outflows 70% lower than average.

During winter and summer, total exports were 22-31% higher than the mean, providing water for agriculture during summer and for storage during winter. In spring and fall, total exports were 24-29% lower than the mean.

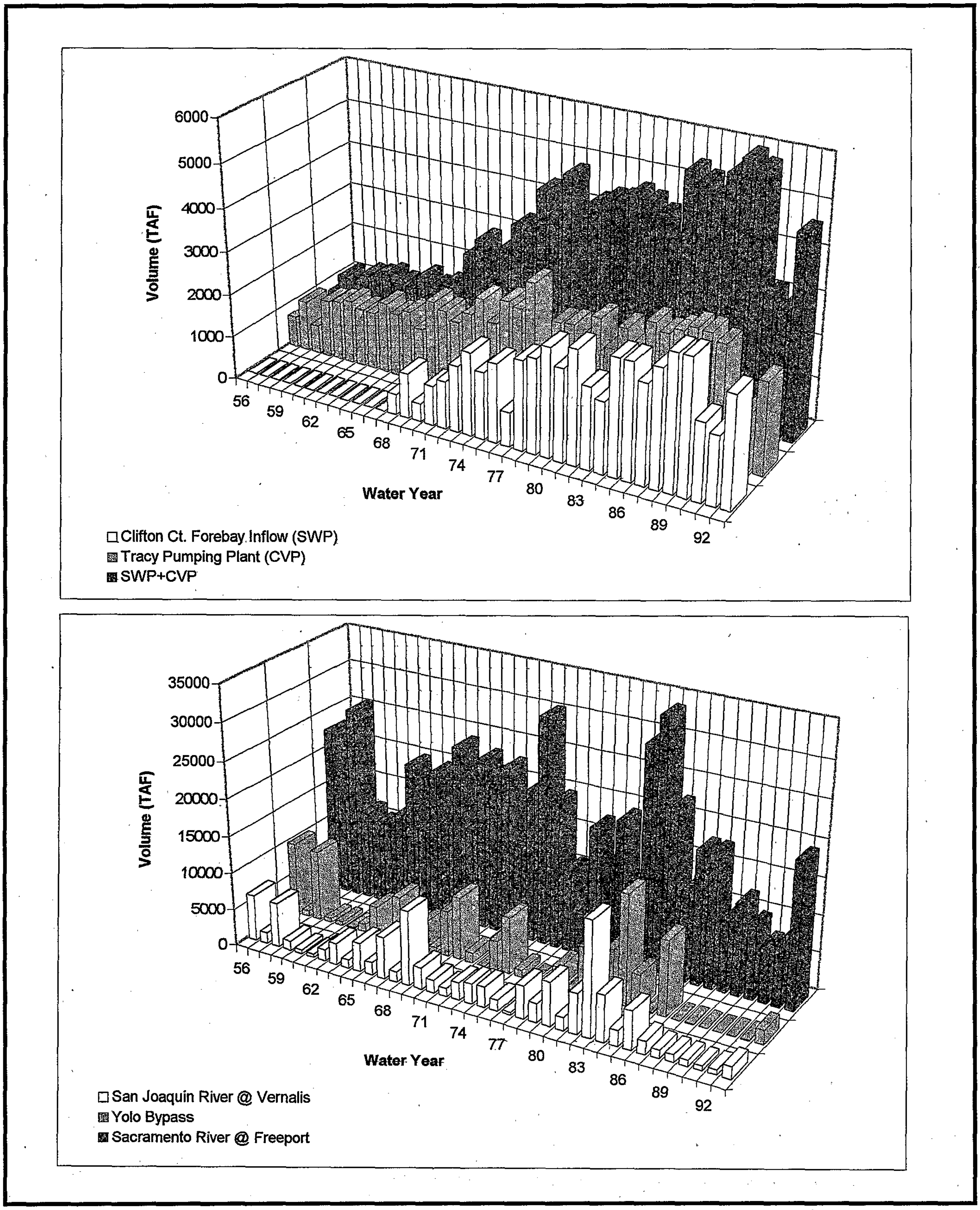

Figure 2
STREAMFLOWS AND EXPORT FLOWS FOR 1956-1993

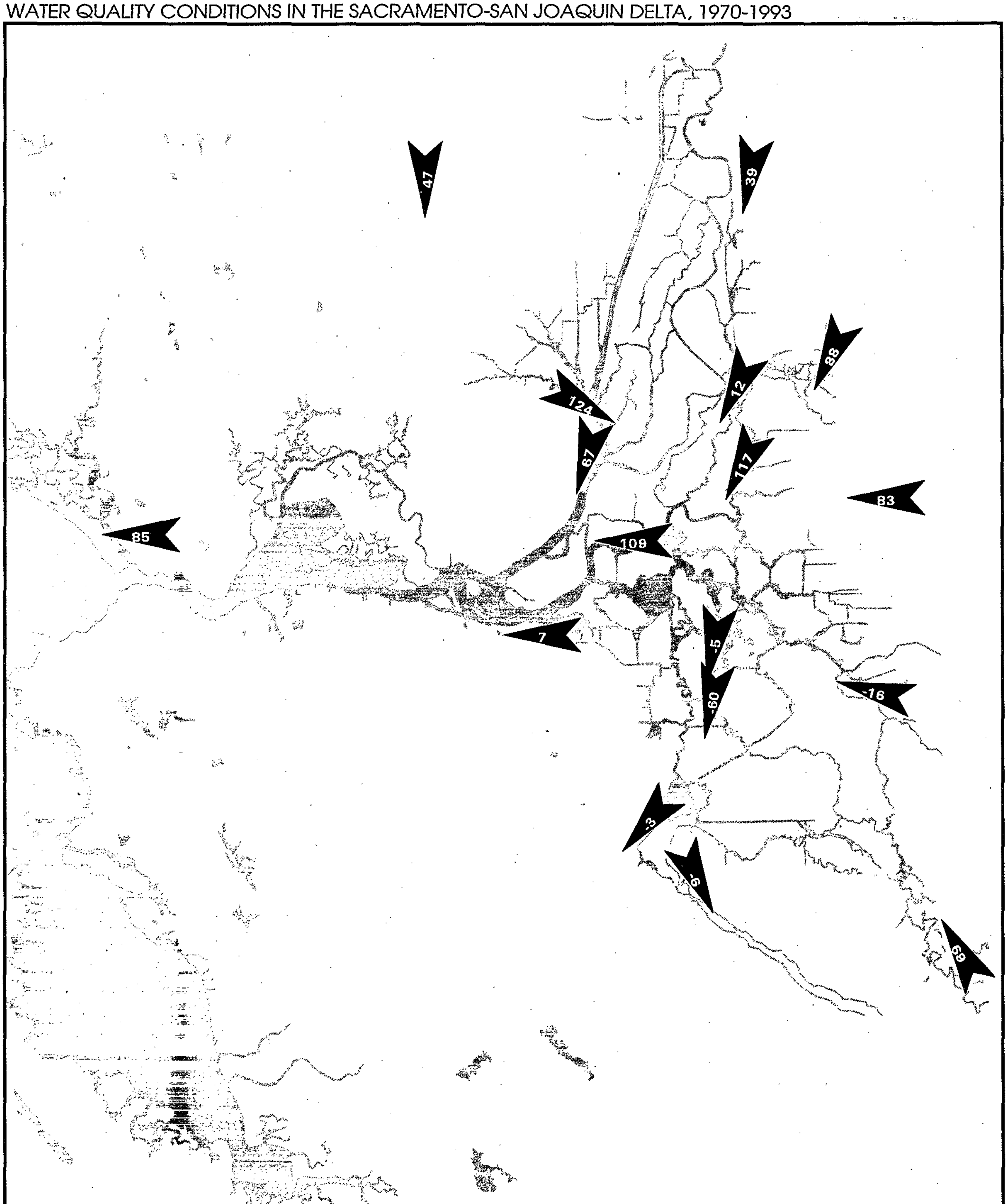

Figure 3
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR WET YEARS

Figure 4
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR CRITICAL YEARS

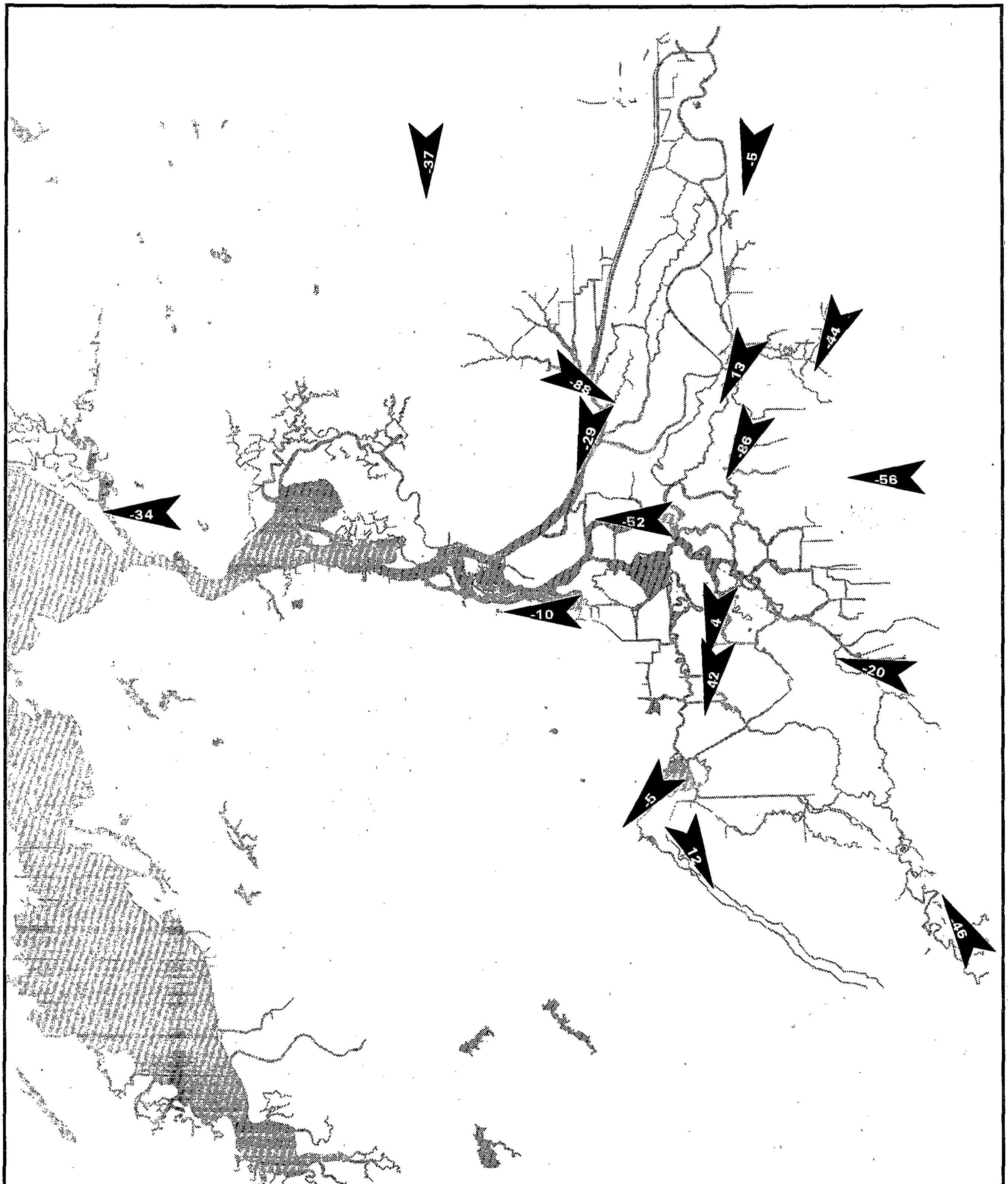

Figure 5
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR NORMAL YEARS

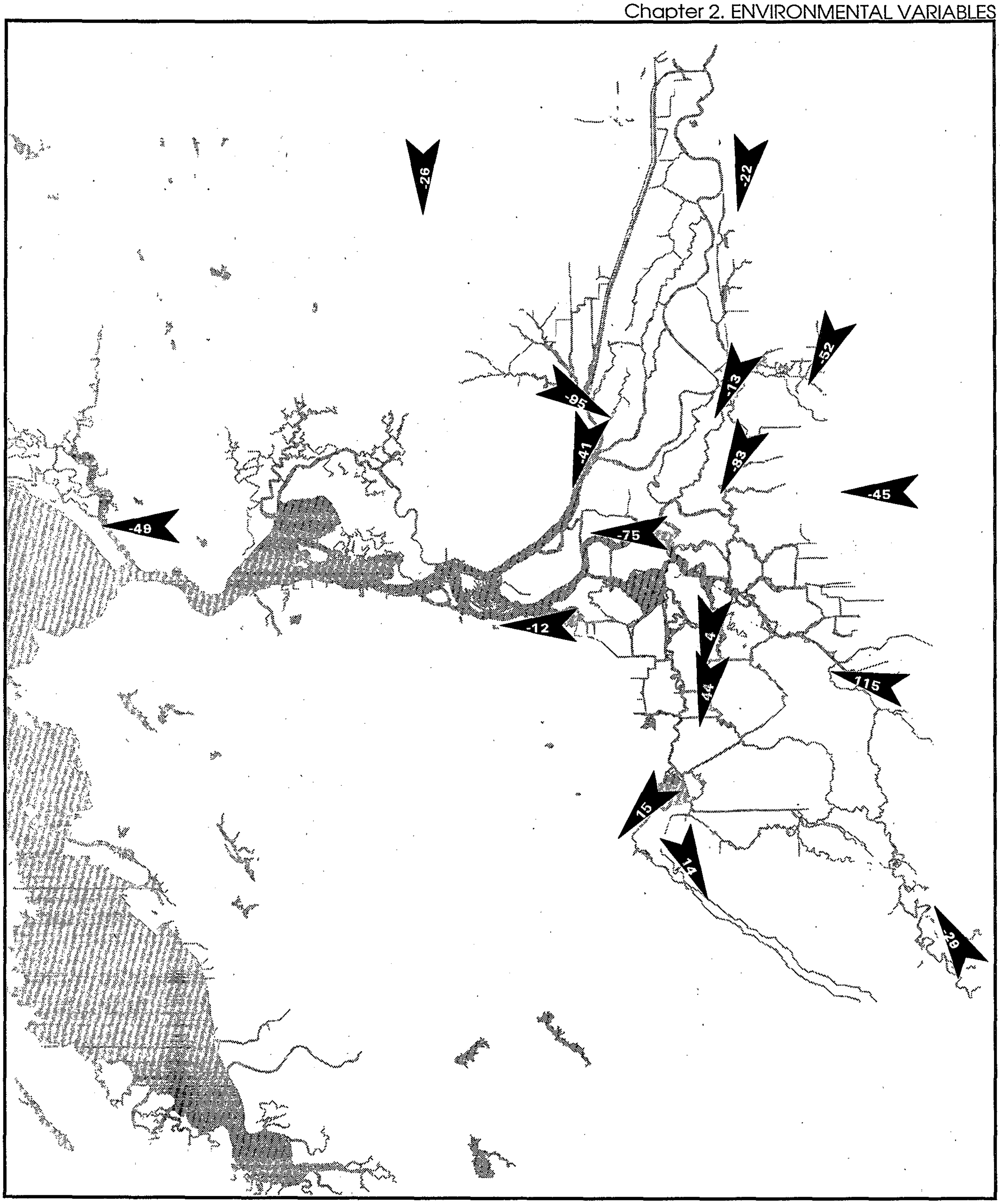

Figure 6
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR DRY YEARS

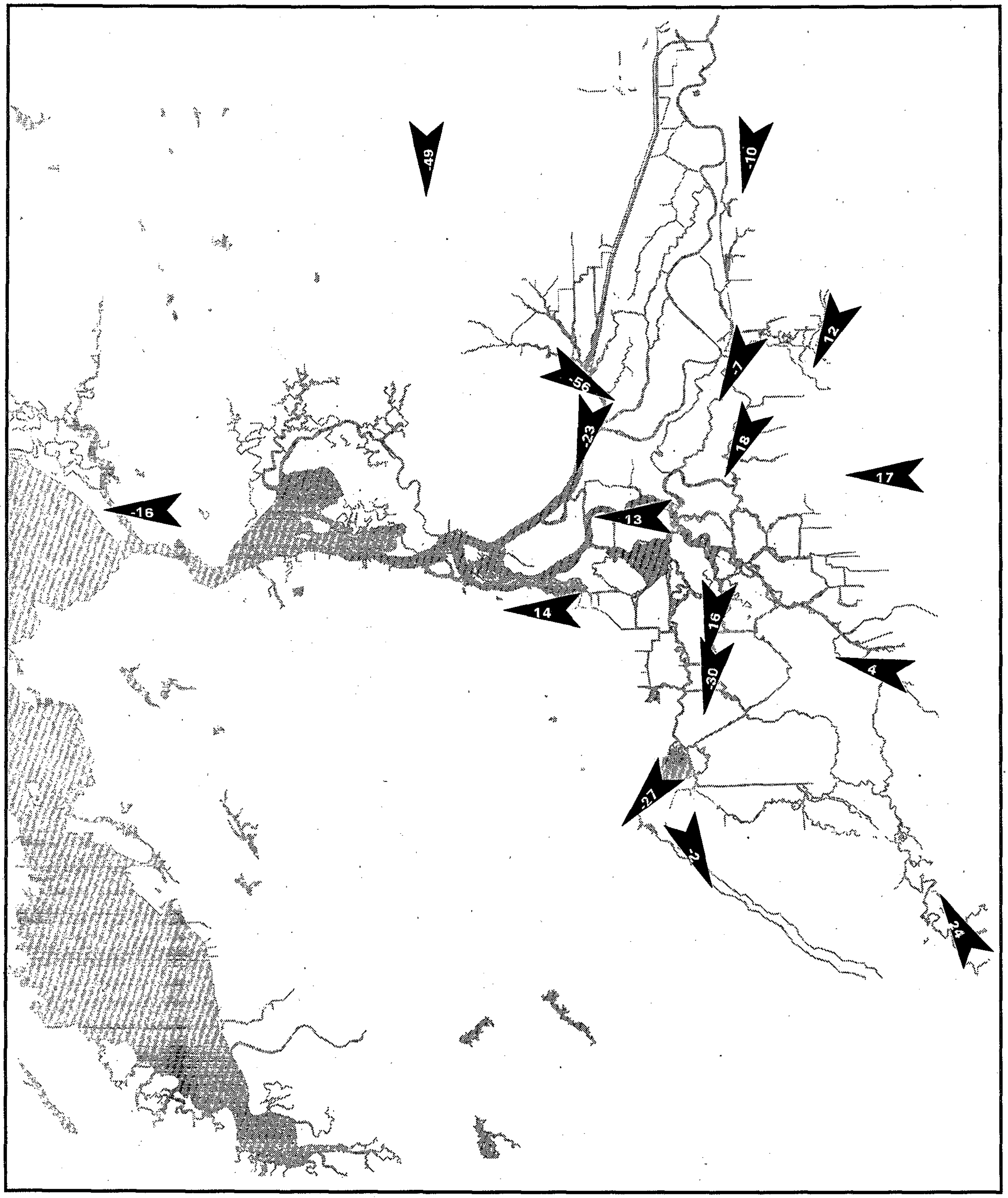

Figure 7
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR SPRING

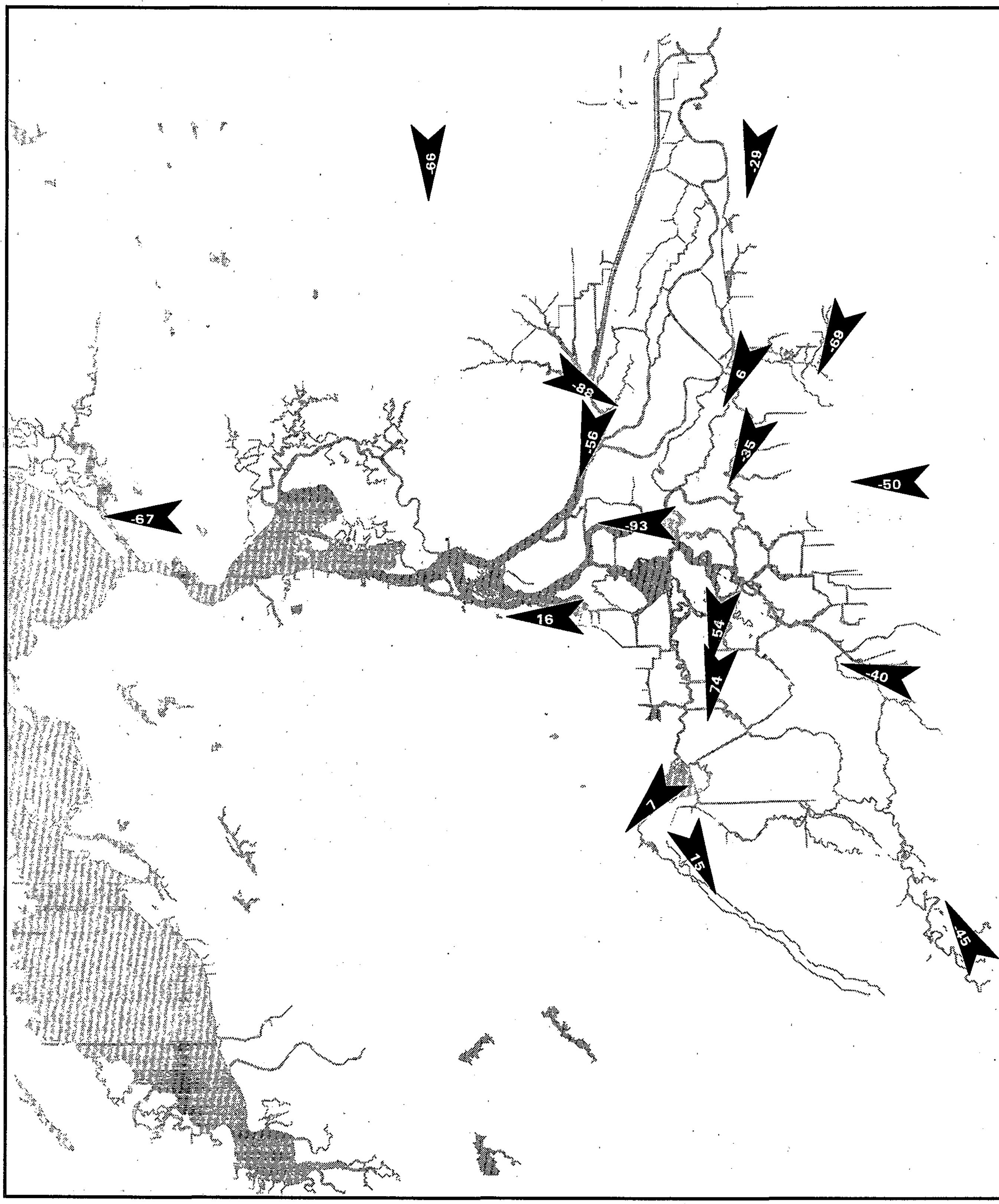

Figure 8
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR SUMMER

Figure 9
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR FALL

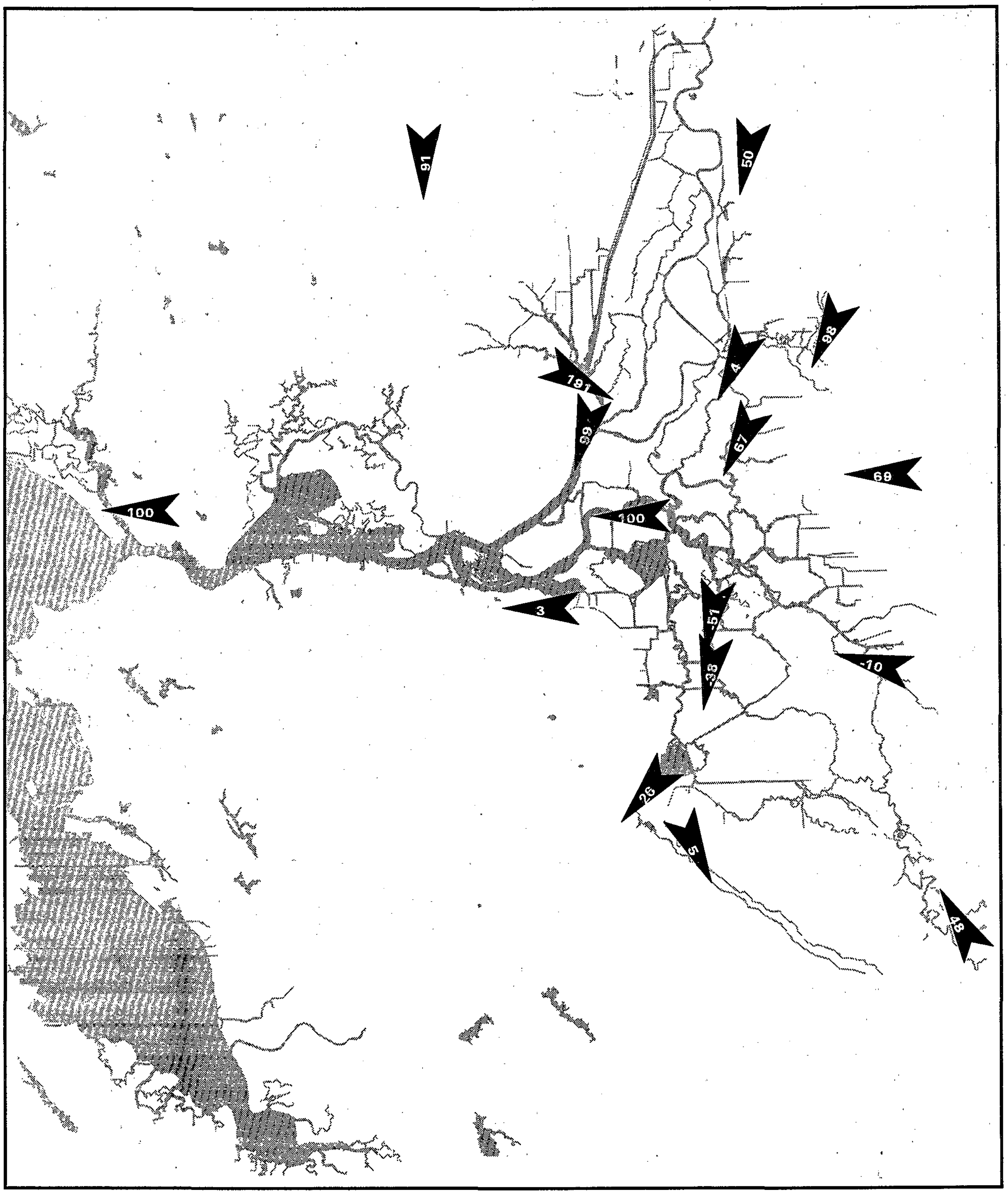

Figure 10
PERCENT DEVIATION OF STREAMFLOW VARIABLES FROM THE LONG-TERM AVERAGE FOR WINTER

Physical Variables

Physical variables were measured (along with chemical variables and nutrients) at 26 stations on a monthly or semi-monthly interval (Figure 11). Measurements were made within 1 hour of high slack tide, and the time of each sample was recorded to the nearest 5 minutes using Pacific Standard Time. A qualitative statement on weather conditions was recorded for each cruise.

Water Temperature — Water temperature (degrees C) was measured at 1 meter with either a mercury thermometer or a YSI telethermometer.

Water Transparency — Water transparency was measured to the nearest centimeter using a plastic Secchi disc, 20 centimeters in diameter. All measurements were done in the shade.

Air Temperature — Air temperature (degrees C) was measured in the shade with a hand-held mercury thermometer or a YSI telethermometer.

Wind Speed and Direction — Wind speed (km/hr) was measured with a Dwyer hand-held wind meter or a Danforth electronic wind speed indicator. Wind direction was determined with a hand-held or nautical compass.

Stations were grouped into regions based on individual and combined hierarchical cluster analysis of monthly data for 14 physical and chemical variables and chlorophyll *a* concentrations (Table 1). These regions are similar to those determined for an independent analysis of phytoplankton community composition (Lehman and Smith 1991).

Long-Term Trends

Figure 12 shows 1971-1993 trends in air temperature, water temperature, water transparency, water turbidity, and wind velocity.

Annual air temperature averaged 18.8°C and ranged from 15 to 22°C in the upper estuary. The highest average air temperatures measured (24-29°C) were in the southern and central regions; the lowest (12-13°C) were in the northern and San Pablo Bay regions. Air temperature fluctuated annually but was lowest during the high inflow years of the early 1970s and 1980s and highest during the 1976-1977 and 1987-1992 droughts. The highest temperatures, however, were measured in 1970 and 1971.

Water temperature followed a similar pattern to air temperature, but was slightly lower and had a narrower range. Average annual water temperature in the upper estuary was 16.6°C, ranging from 15 to 18°C among regions. Among regions, water temperature was highest (23°C) in the eastern region and lowest (13°C) in the northern region. Like air temperature, water temperature was lower in the early 1970s and 1980s and increased in the mid-1970s and mid- to late 1980s but was sometimes high in 1970 and 1971 . Increased water temperature after 1977 coincided with a 1977 climate shift, which was associated with changes in many physical variables in the estuary (Lehman and Smith 1991).

Table 1
REGIONS OF THE UPPER ESTUARY AND THEIR ASSOCIATED SAMPLING STATIONS

Region	Index	Representative Stations
Northern Delta	ND	C3
Western Delta	WD	D11, D12, D14, D15
Lower San Joaquin River	LSJ	D16, D19, D26
Lower Sacramento River	LS	D4, D22, D24
Southern Delta	SD	C7, C10, P8, P12
Eastern Delta	ED	MD7, MD10
Central Delta	CD	C9, D28A, P10
Suisun Bay	SB	D6, D7, D8, D9, D10
San Pablo Bay	SPB	D41

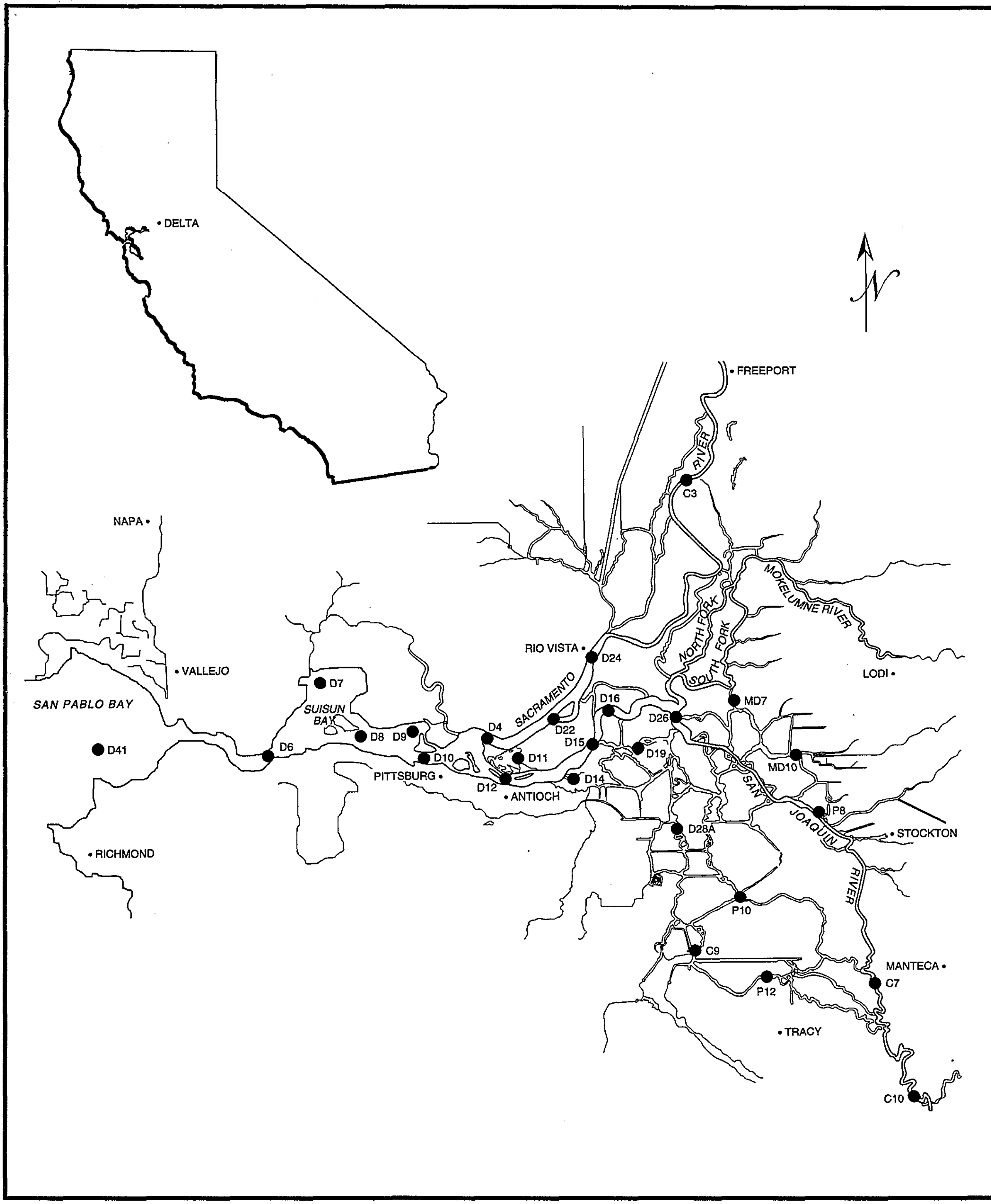

Figure 11
MONITORING STATIONS FOR PHYSICAL VARIABLES, CHEMICALS, AND NUTRIENTS
Sacramento-San Joaquin Delta

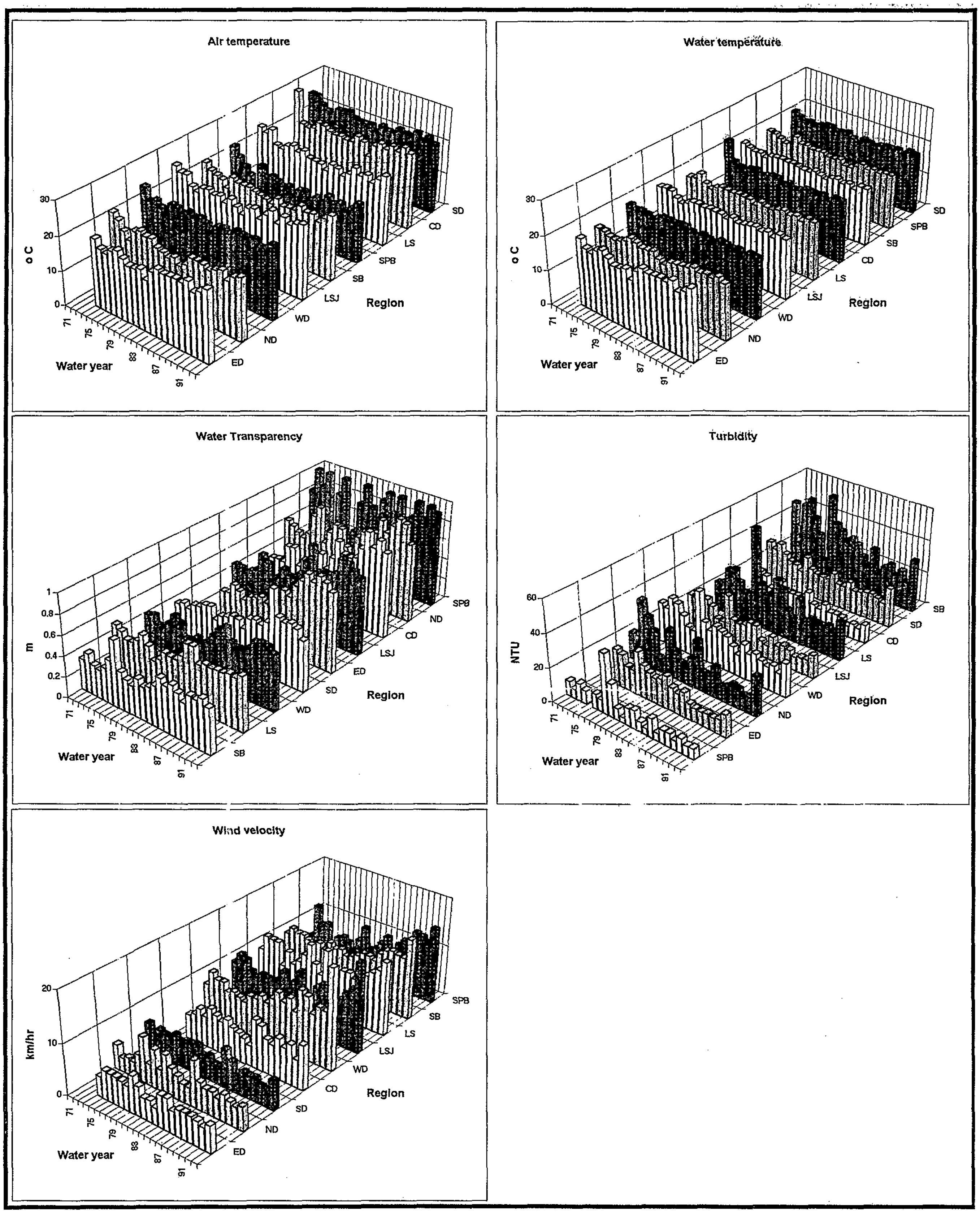

Figure 12
LONG-TERM TRENDS FOR AIR TEMPERATURE, WATER TEMPERATURE, WATER TRANSPARENCY, TURBIDITY, AND WIND VELOCITY

High suspended sediment load is characteristic of the San Francisco Bay estuary and produces continually low water transparency throughout the upper estuary. Annual average Secchi disk depth was 0.58m, ranging from 0.5 to 1.0m among regions. The lowest Secchi disk depth (0.3m) occurred frequently among regions. Water transparency was consistently low in Suisun Bay, where shallow water combined with tidal and wind mixing frequently resuspend bottom sediments. The highest Secchi disk depths (1-1.5m) were measured in the northern, central, and San Pablo Bay regions. Among years, water transparency increased continually after the late 1970s, accompanied by a decrease in turbidity.

Average wind velocity was 7.7 km/hr, ranging from 4.8 to 11 km/hr. Average annual wind velocity was higher in the western, lower Sacramento, lower San Joaquin, and Suisun Bay regions, which are affected by onshore and offshore winds. Wind velocities were low upstream in the northern, eastern, and southern regions, where there is little influence of onshore and offshore winds. The increase in wind velocity downstream after 1977 was probably produced by the 1977 climate shift, which was associated with increased wind velocities along the California coast (Lehman and Smith 1991).

Water-Year Trends

Figures 13 to 17 show percent deviation from the long-term mean for air temperature, water temperature, water transparency, water turbidity, and wind velocity for wet, normal, dry, and critical water years, all of which are discussed below.

During wet years, spatial variation was small for physical variables. Air and water temperatures were 1-2% higher than average in the western region and downstream and 1-9% lower than average upstream. Water transparency and turbidity varied in a uniform fashion. For most stations, water transparency was 7-30% lower than average and turbidity was 16-43% higher than average. Wind velocity was high in the western, lower Sacramento River, and San Joaquin River regions but low in the Suisun Bay and San Pablo Bay regions.

During normal years, environmental variables were spatially variable. Air temperatures were up to 9% lower than average at most stations but increased by up to 8% near the periphery of the upper estuary. Water temperature decreased by 2-6% at many stations. Lower-than-average water temperatures were probably produced by lower-than-average air temperatures and increased streamflows. Water transparency was 12-45% higher than average upstream in the eastern and southern regions but decreased by as much as 30% downstream. Turbidity was also spatially variable but did not differ for upstream and downstream regions. Wind velocity was higher than average downstream but up to 27% lower than the average upstream.

Dry years were characterized by downstream gradients in physical variables. Air and water temperature were up to 8% higher than average in the western region and 9% lower than average upstream. Low water transparency and high turbidity characterized the eastern Suisun Bay and western regions. Secchi disk depths were higher than average upstream of Suisun Bay and decreased farther upstream. In contrast, turbidity was lower than average for all stations upstream of Suisun Bay. Higher-than-average water temperatures in the western region and downstream were probably produced by a combination of increased air temperature and residence time. Cooler air temperatures upstream and warmer air temperatures downstream may have contributed to increased wind velocities upstream.

During critical years, air temperature, water temperature, and water transparency were high. At most stations, values were higher than average by up to 8% for air temperature, 6% for water temperature, and 45% for water transparency. As expected, turbidity varied in an opposite fashion to water transparency and was 9-46% lower than average. Low turbidity in Suisun Bay in critical years is partially a function of clam grazing, which increases the precipitation of suspended material. Wind velocity increased from 8% lower than average in the lower San Joaquin River to 7-28% higher than average in Suisun and San Pablo bays.

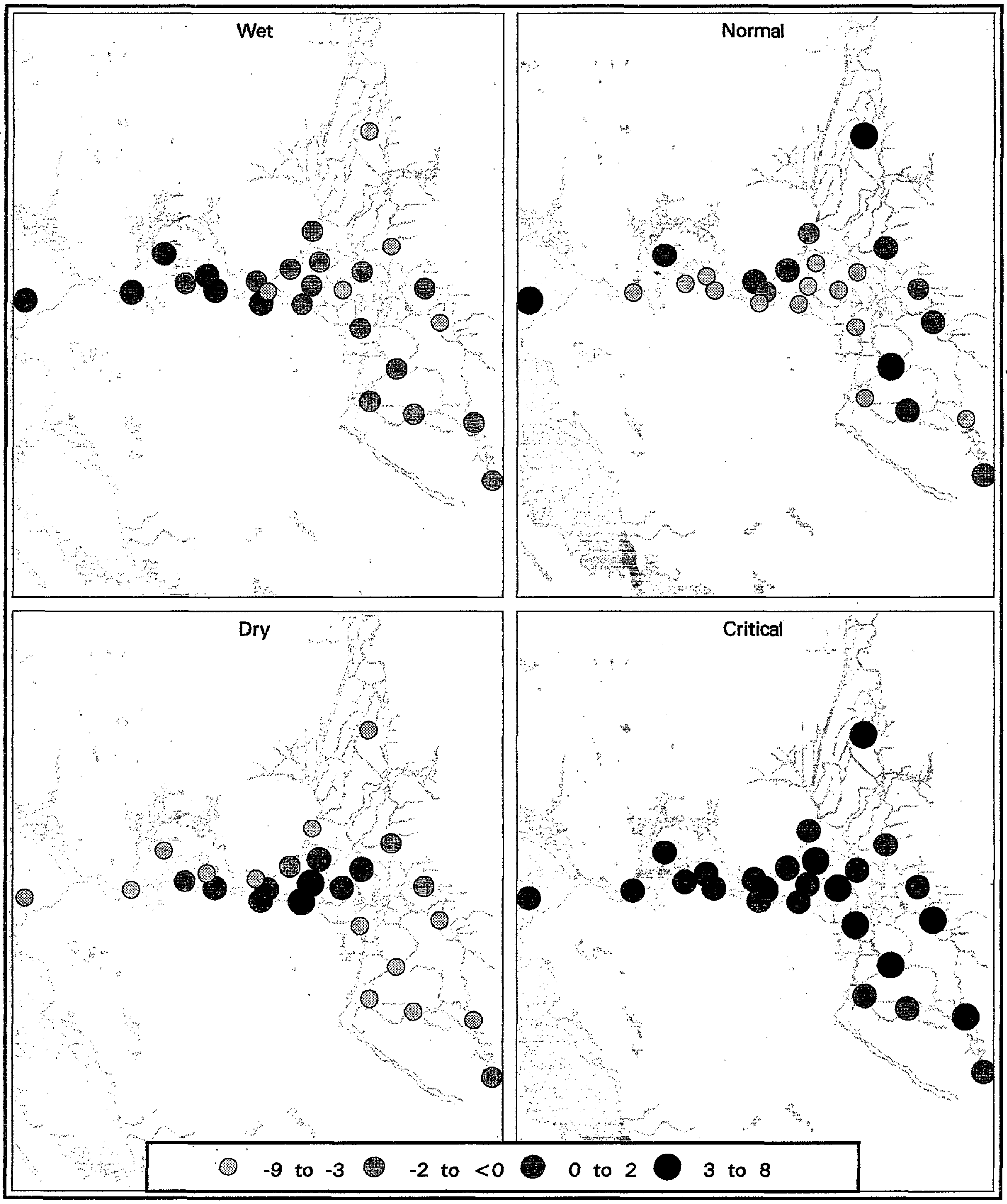

Figure 13
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR AIR TEMPERATURE, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

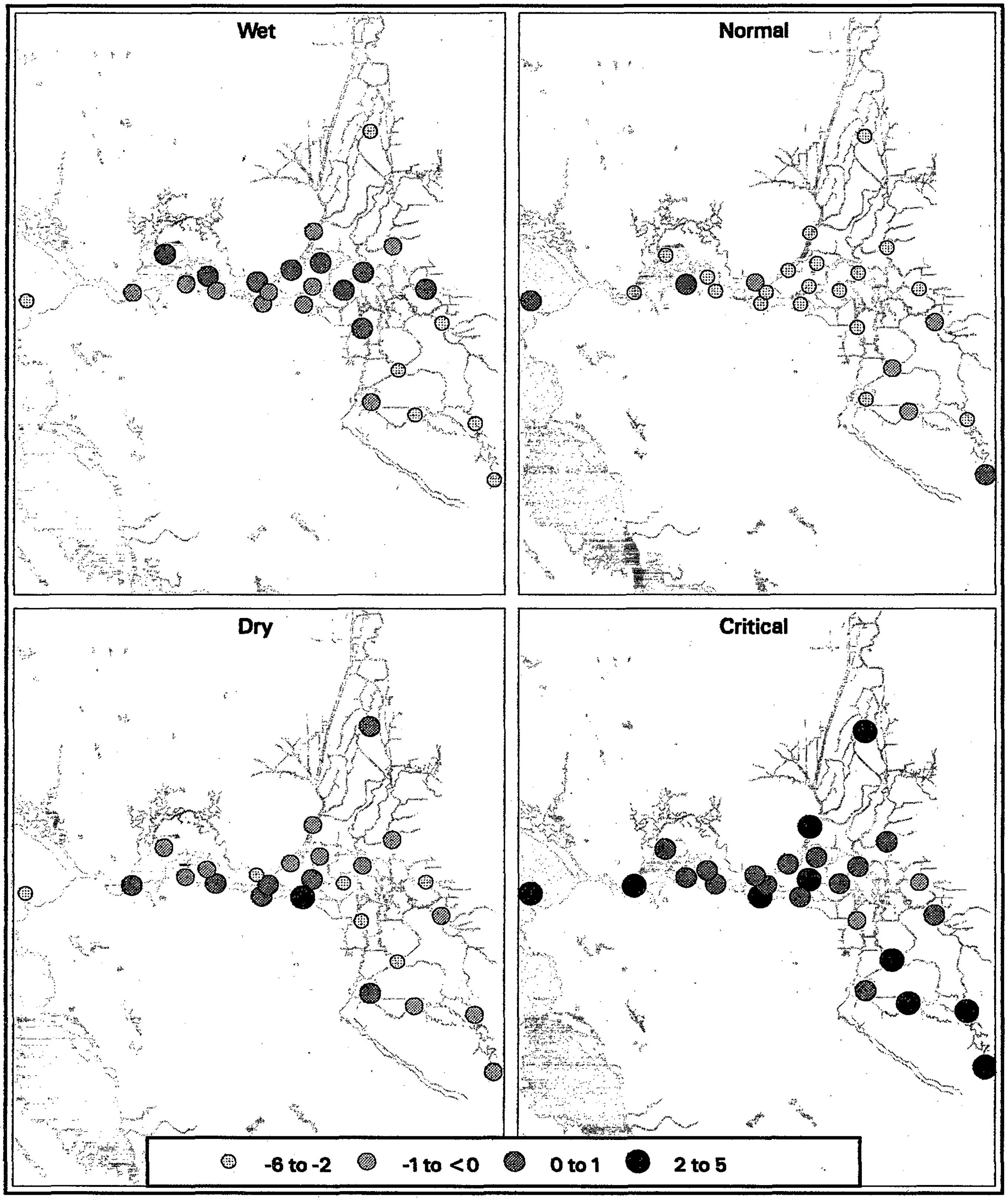

Figure 14
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WATER TEMPERATURE, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

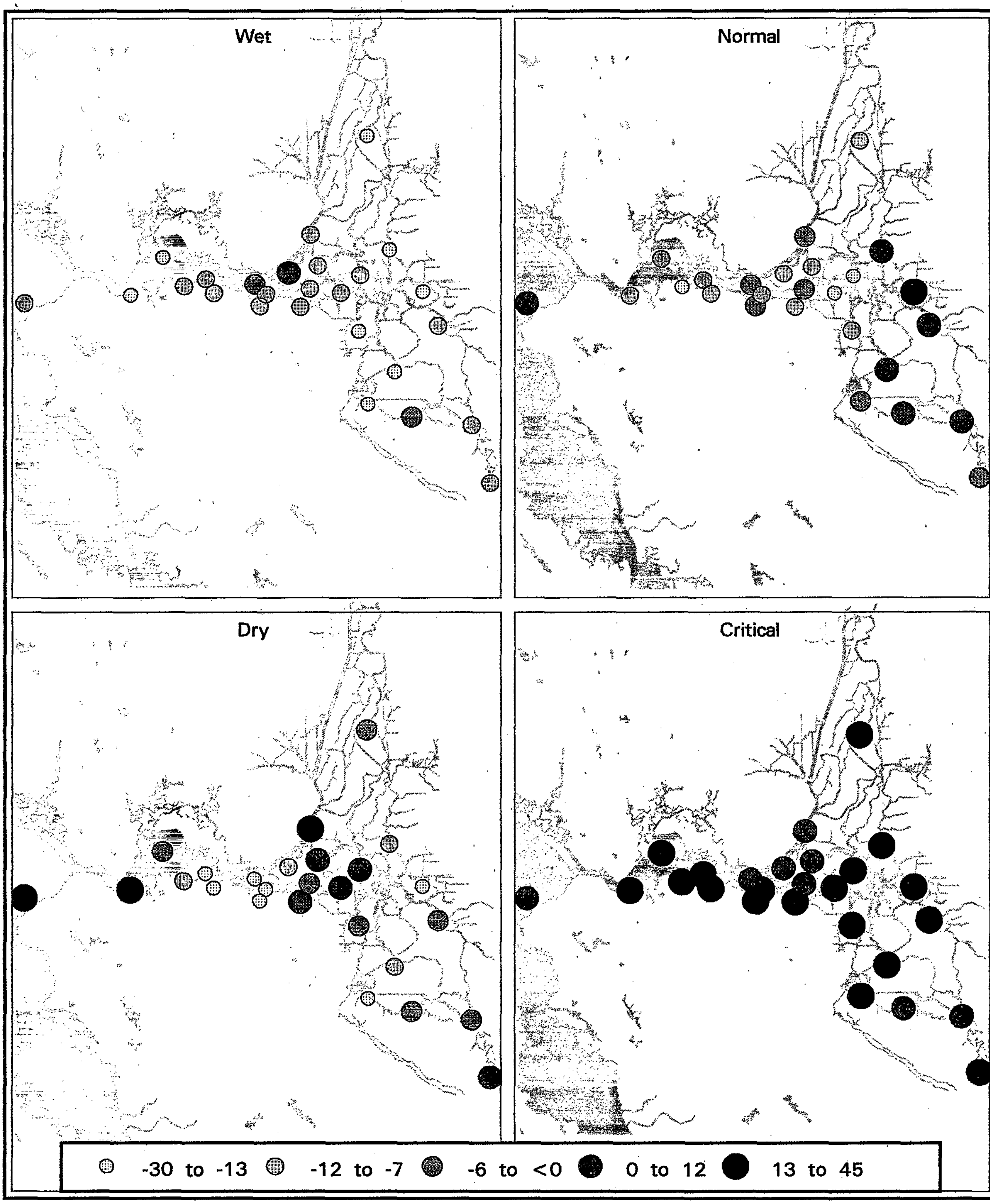

Figure 15
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WATER TRANSPARENCY, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

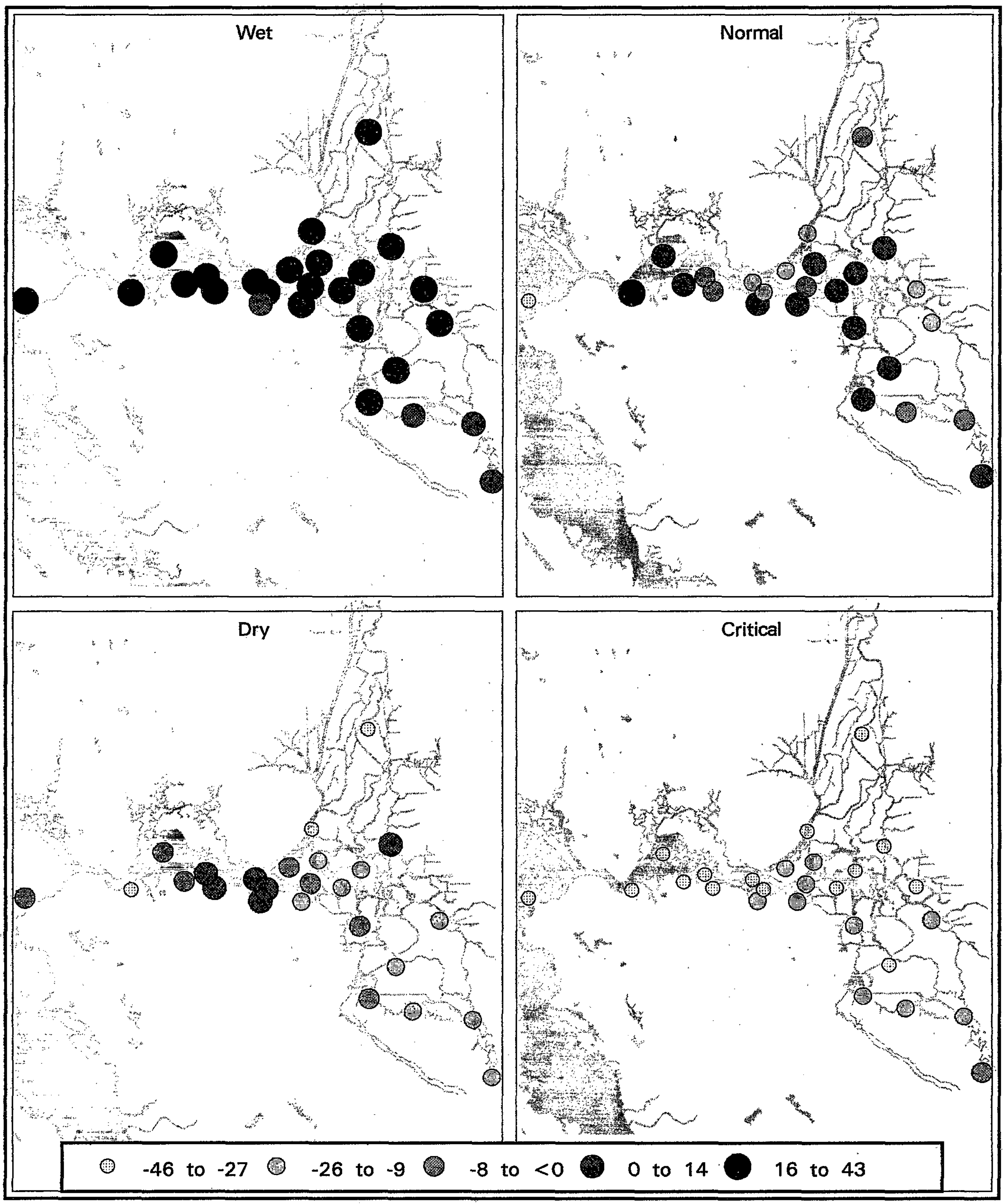

Figure 16
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TURBIDITY, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

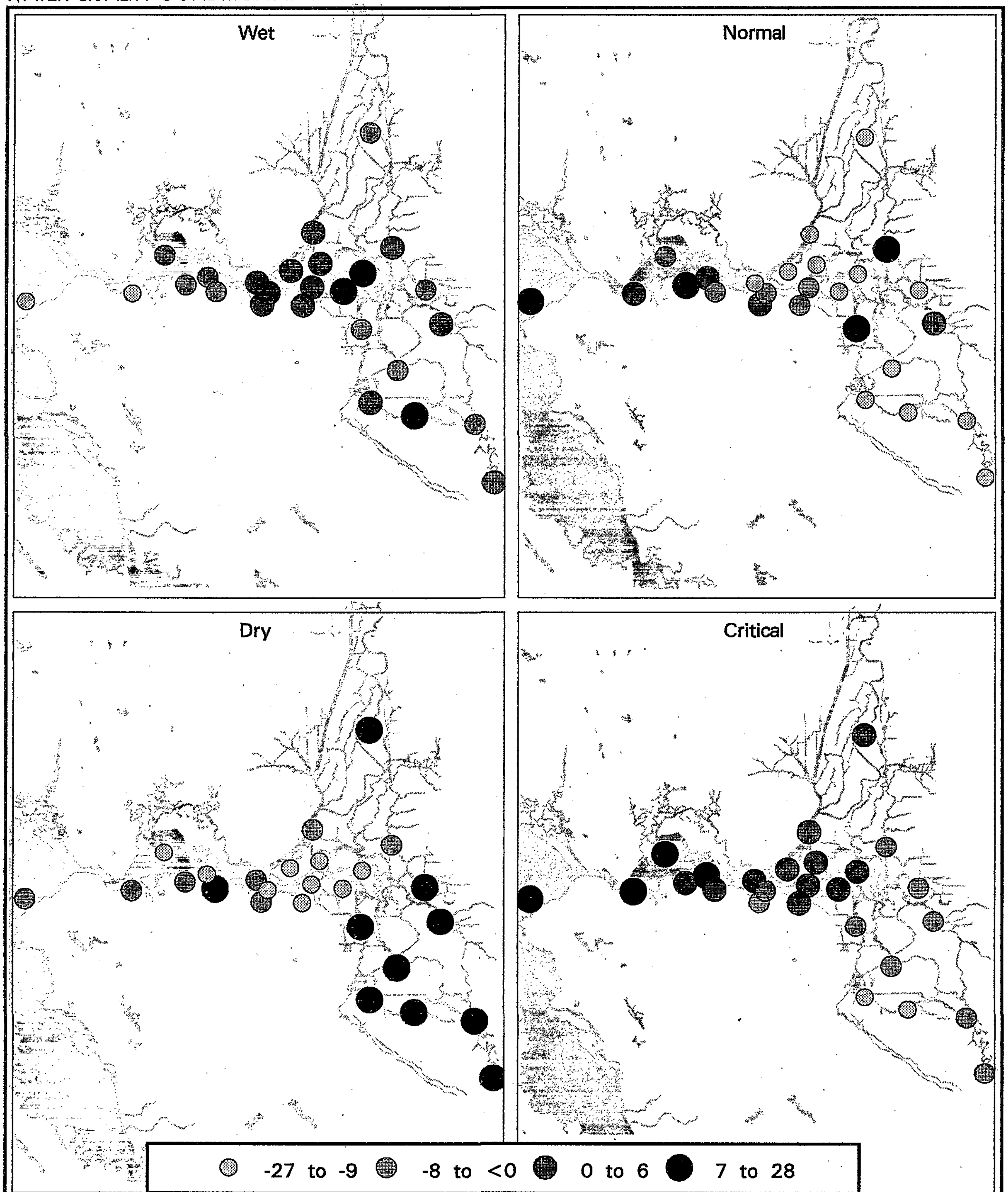

Figure 17

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WIND VELOCITY, BY WATER-YEAR TYPE

Gaps between data ranges in legend indicate no data.

Seasonal Trends

Figures 18 to 22 show percent deviation from the long-term mean for air temperature, water temperature, water transparency, wind velocity, and turbidity for fall, winter, spring, and summer.

Fall was characterized by low air temperature, water temperatures, and wind velocity and high water transparency. Air and water temperatures were up to 29% lower throughout the upper estuary and were accompanied by lower-than-average wind velocity, except along the Delta periphery. Wind velocity also had a downstream gradient, with lower velocities in the western region and downstream. Water transparency increased by as much as 48% throughout the upper estuary during the fall, when low streamflow coupled with low wind velocity facilitates sedimentation of suspended material. The increase in water transparency was smaller for Suisun and San Pablo bays, where water combined with tidal and wind mixing resuspends bottom sediments. Patterns of change in water transparency were similar to those for turbidity, which was as much as 51% lower upstream but only 22% lower downstream.

In winter, air and water temperatures and wind velocity were also lower than average. Air temperature is controlled by large-scale weather patterns, which during the winter reduce air temperatures by 30-41% and contribute to the 26-37% lower water temperatures. Reduced onshore winds during the winter also produced lower-than-average wind velocities at most upper estuary stations. Wind velocities also had a downstream gradient and were 20-43% lower than average in the lower Sacramento region and downstream but only up to 19% lower than average upstream. Water transparency was characterized by higher-than-average values in the San Joaquin River, while turbidity increased by as much as 100% above the long-term average downstream.

In spring, air and water temperature, wind velocity, and turbidity were high and water transparency was low. Air and water temperatures were up to 16% higher than average and were accompanied by 21-42% higher wind velocity. Vertical mixing associated with high winds plus transport of sediment with snowmelt runoff probably contributed to the up to 29% lower-than-average water transparency and high turbidity.

Summer was characterized by high air and water temperature and a strong downstream gradient for water transparency and wind velocity. Air and water temperatures increased by 17-42% at all stations and were accompanied by wind velocity at least 20% higher than average in the western region and downstream but 19% lower than the mean upstream. Decreased water transparency and high turbidity throughout most of the upper estuary was probably a function of resuspension of bottom sediments by wind and increased phytoplankton standing stock. Average chlorophyll *a* concentrations in the lower San Joaquin are often near 50 µg/L and can reach 300 µg/L (Lehman *et al* 1992).

C-036507

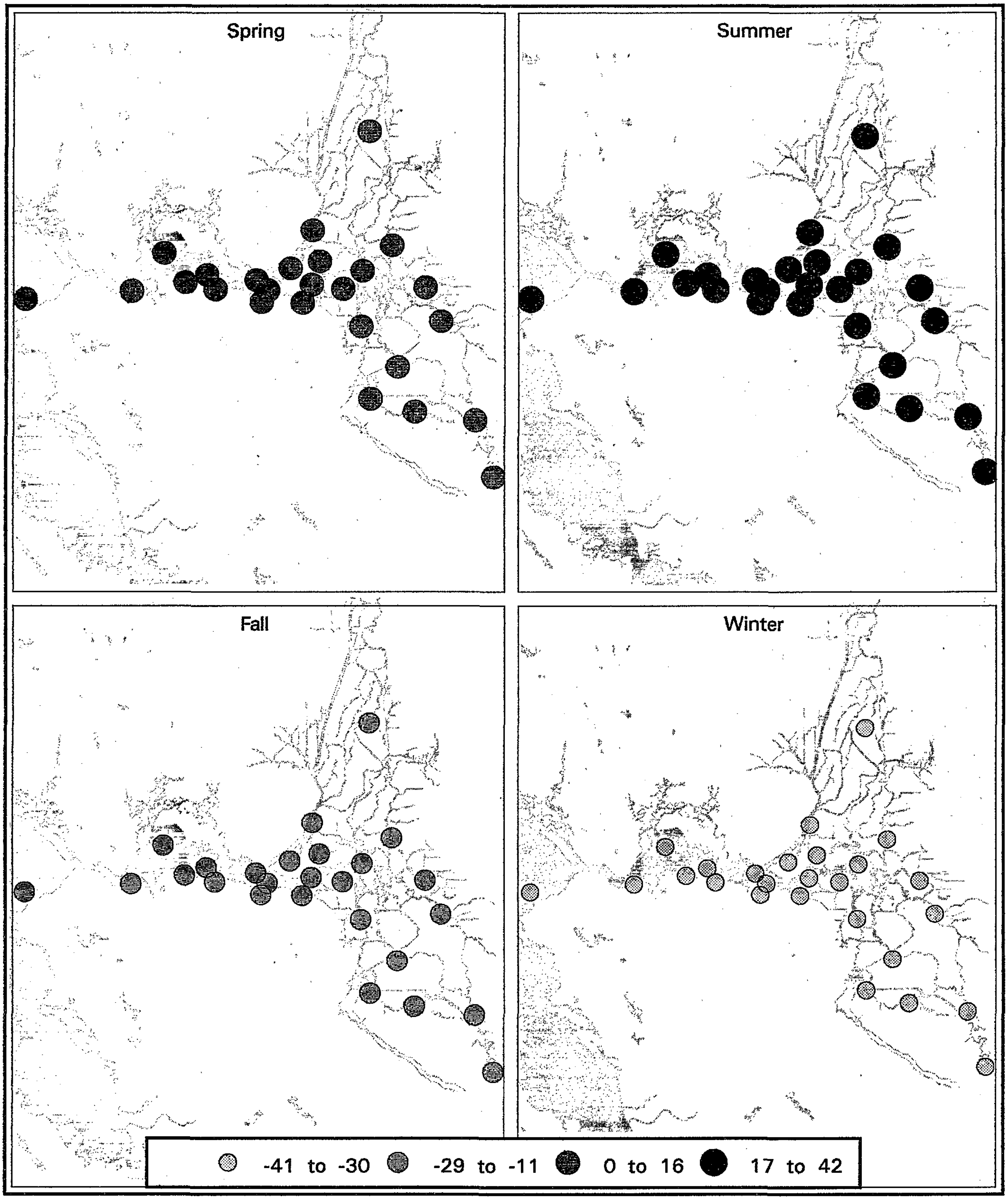

Figure 18
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR AIR TEMPERATURE, BY SEASON
Gaps between data ranges in legend indicate no data.

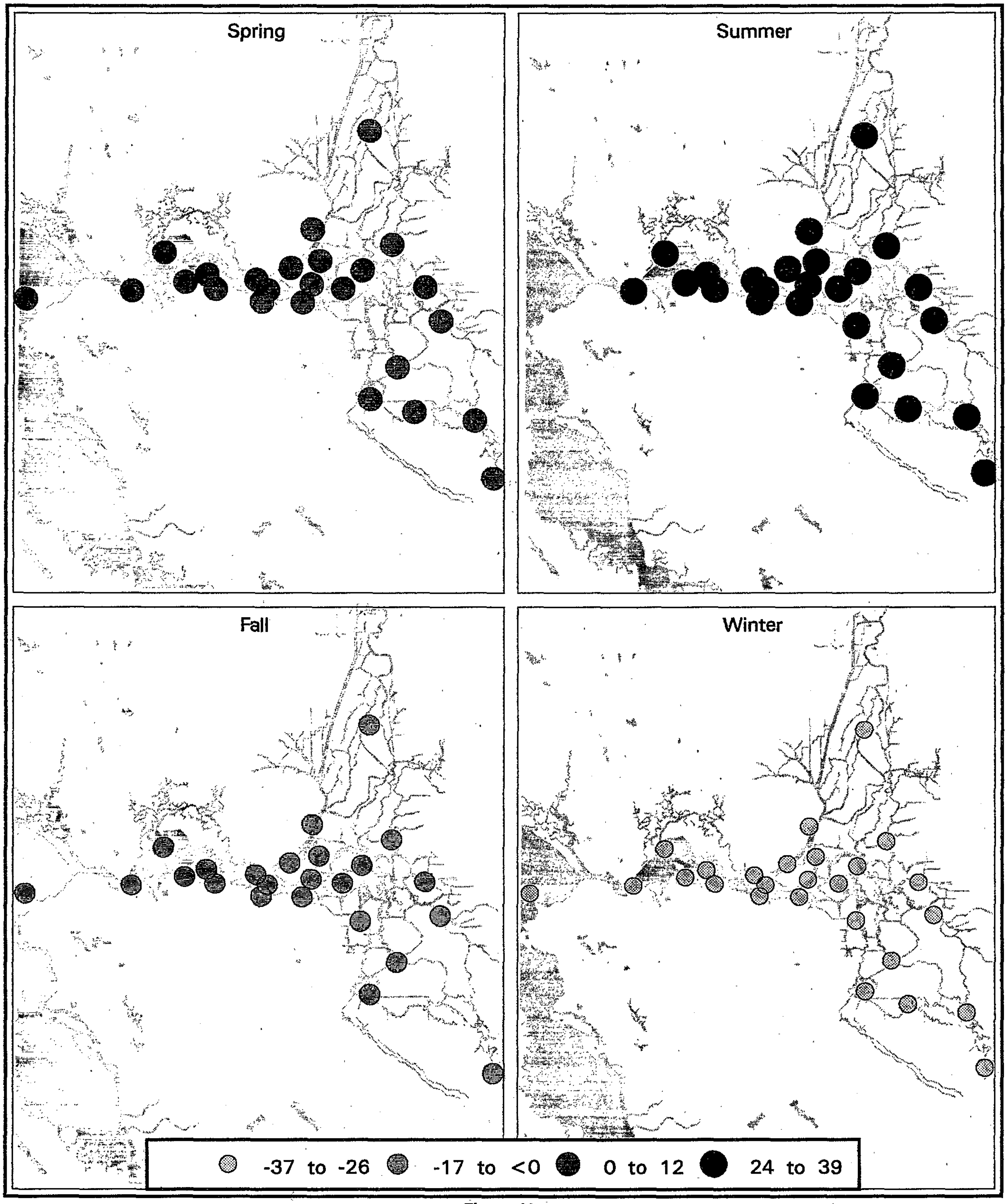

Figure 19
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WATER TEMPERATURE, BY SEASON
Gaps between data ranges in legend indicate no data.

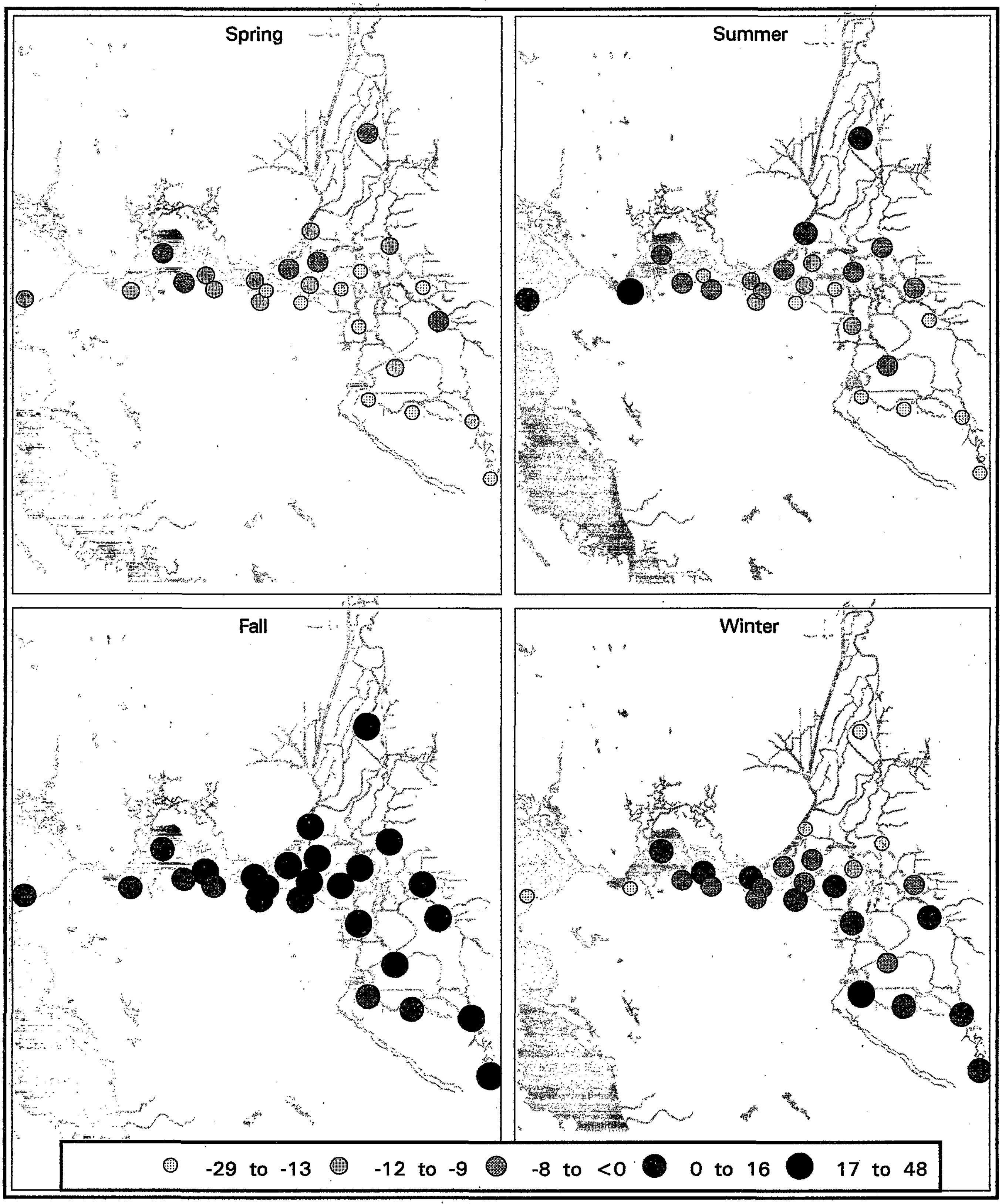

Figure 20
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WATER TRANSPARENCY, BY SEASON
Gaps between data ranges in legend indicate no data.

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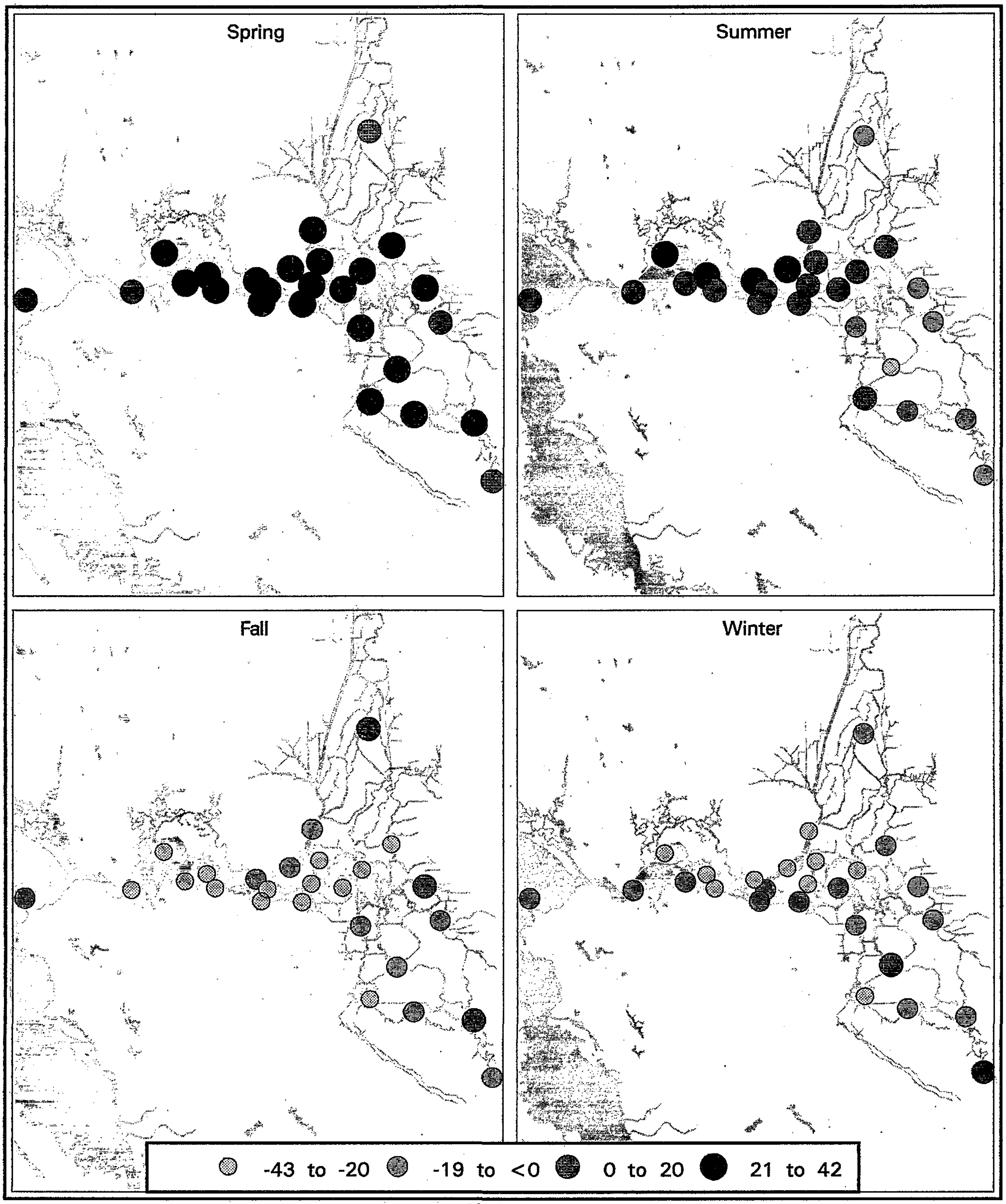

Figure 21
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR WIND VELOCITY, BY SEASON
Gaps between data ranges in legend indicate no data.

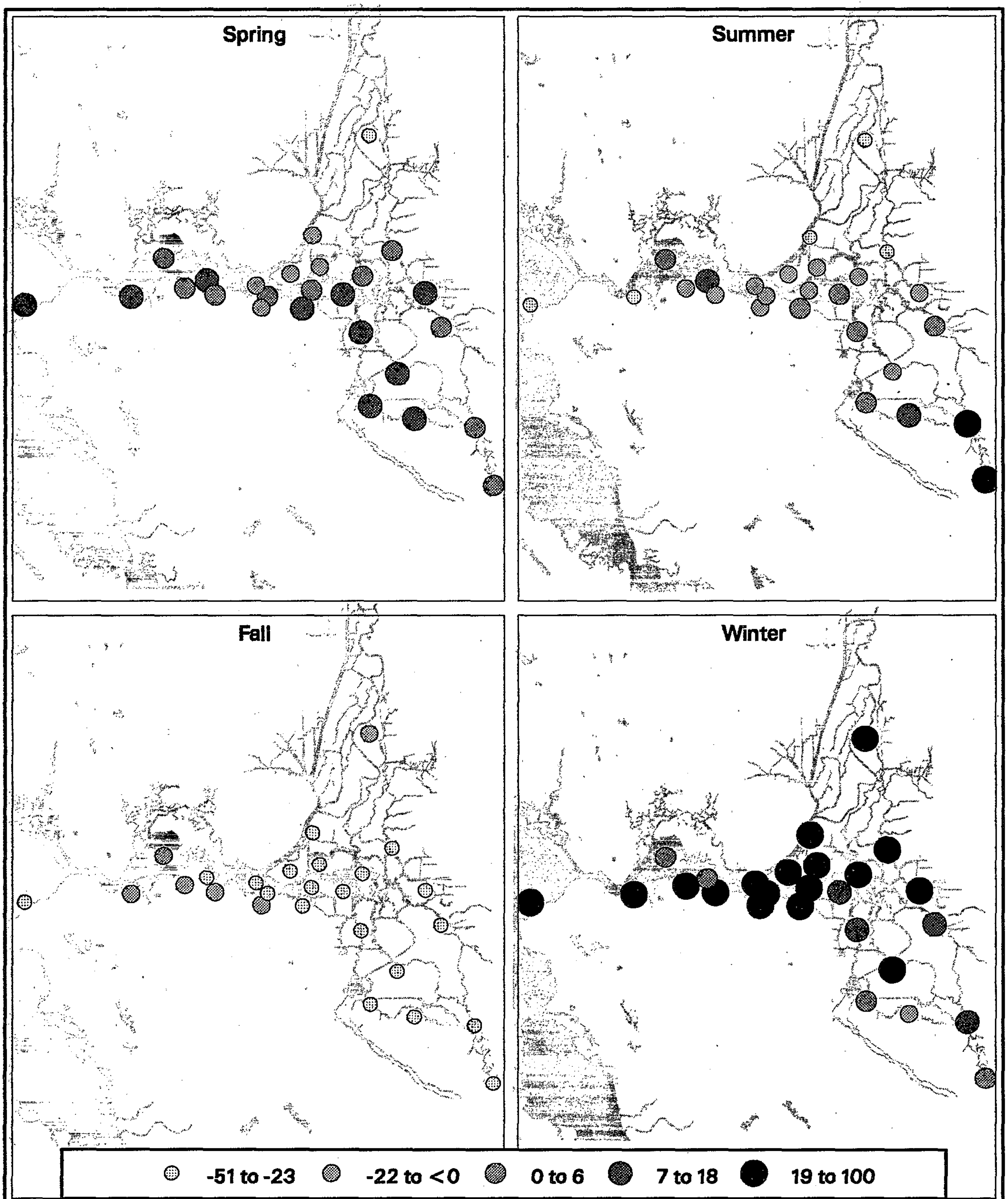

Figure 22
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TURBIDITY, BY SEASON
Gaps between data ranges in legend indicate no data.

Chemical Variables

Water samples for chemical variables were collected monthly or semi-monthly at the same 26 stations as for physical variables (Figure 11). Water samples were collected at 1 meter within 1 hour of high slack tide using a Van Dorn sampler or a submersible pump. The time of each sample was recorded to the nearest 5 minutes using Pacific Standard Time, and total water depth was measured to the nearest foot by an electronic depth-sounding instrument or a weighted measuring tape.

Dissolved Oxygen — The Winkler method was used to determine dissolved oxygen concentration to the nearest tenth mg/L.

Specific Conductance — Specific conductance (µS/cm) was measured with a Beckman RC-19 conductivity bridge equipped with a manual temperature compensator. Values were compensated to 25 degrees.

pH — pH was measured to the nearest tenth of a unit with a Beckman pHI 12 or pHI 71 meter with a Beckman Futura Plus, epoxy body electrode (EPA Method 150.1 [1983]).

Turbidity — A Hach Model 2100A turbidimeter was used to measure turbidity as nephelometric turbidity units (NTU). Samples with turbidity above 40 NTU were diluted with one or more volumes of turbidity-free distilled water before analysis (EPA Method 180.1 [1983]).

Suspended and Volatile Solids — Water samples for suspended and volatile solids (mg/L) were stored in polyethylene bottles and refrigerated at 4°C. Suspended and volatile solids were measured using EPA Methods 160.2 and 160.4 [1983]. Dissolved solids were analyzed by EPA Method 160.1 [1983].

Chloride — Water samples for chloride were filtered through a 0.45-micron pore size membrane filter. Chloride concentration (mg/L) was measured to a minimum of 1 mg/L using EPA Method 325.2 (1983).

Stations were grouped into the same regions as for physical variables, based on individual and combined hierarchical cluster analysis of monthly data (Table 1).

Long-Term Trends

Figure 23 shows long-term trends for 1971-1993 for specific conductance, pH, dissolved oxygen, suspended solids, volatile solids, and total dissolved solids, all of which are discussed below.

Annual average specific conductance was below 1000 µS/cm upstream and increased to 1000-35000 µS/cm downstream in Suisun and San Pablo bays. The lowest specific conductance was measured in the northern region, where values did not exceed 200 µS/cm. The highest values of 35000 µS/cm were measured in San Pablo Bay. Interannual variation in specific conductance among regions was consistent with changes in inflow. High values were measured during the critical water years in 1976, 1977, the late 1980s, and the early 1990s. Lower values were measured in the early 1970s and 1980s, when normal and wet years were common.

Annual average pH was 7.7, with a range of 6-8 among regions. A wider range of pH values occurred in the lower San Joaquin and southern region, where discharge of agricultural herbicides, pesticides, and fertilizers may affect pH. Changes in pH over the period of record reflect changes in inflow. Low pH characterized the 1982-1983 record inflow years, and high values characterized the drought years of the late 1980s and early 1990s.

Annual average dissolved oxygen concentration remained above the recommended standard of 5 mg/L for all regions. The range of annual average concentrations was about 6-10 mg/L; the average was 8.8 mg/L. Dissolved oxygen concentrations are consistently high for most regions due to high wind mixing, moderate streamflows, and low phytoplankton biomass. The largest variation was in the southern region, where daily concentrations ranged from near 0 to 22 mg/L. Large variations in dissolved oxygen in the southern region are produced by low streamflow, high water temperature, and phytoplankton blooms during summer and fall. Among water years, dissolved oxygen concentration varied in response to inflow — low during the warm, dry years of the mid-1970s, late 1980s, and early 1990s and high during the early 1970s and 1980s.

Suspended solids were highest in Suisun Bay and the southern regions. These shallow regions are susceptible to resuspension of bottom sediments from tidal and wind mixing. Among years, suspended solids decreased for all regions after the mid 1970s but were high during the 1977 drought.

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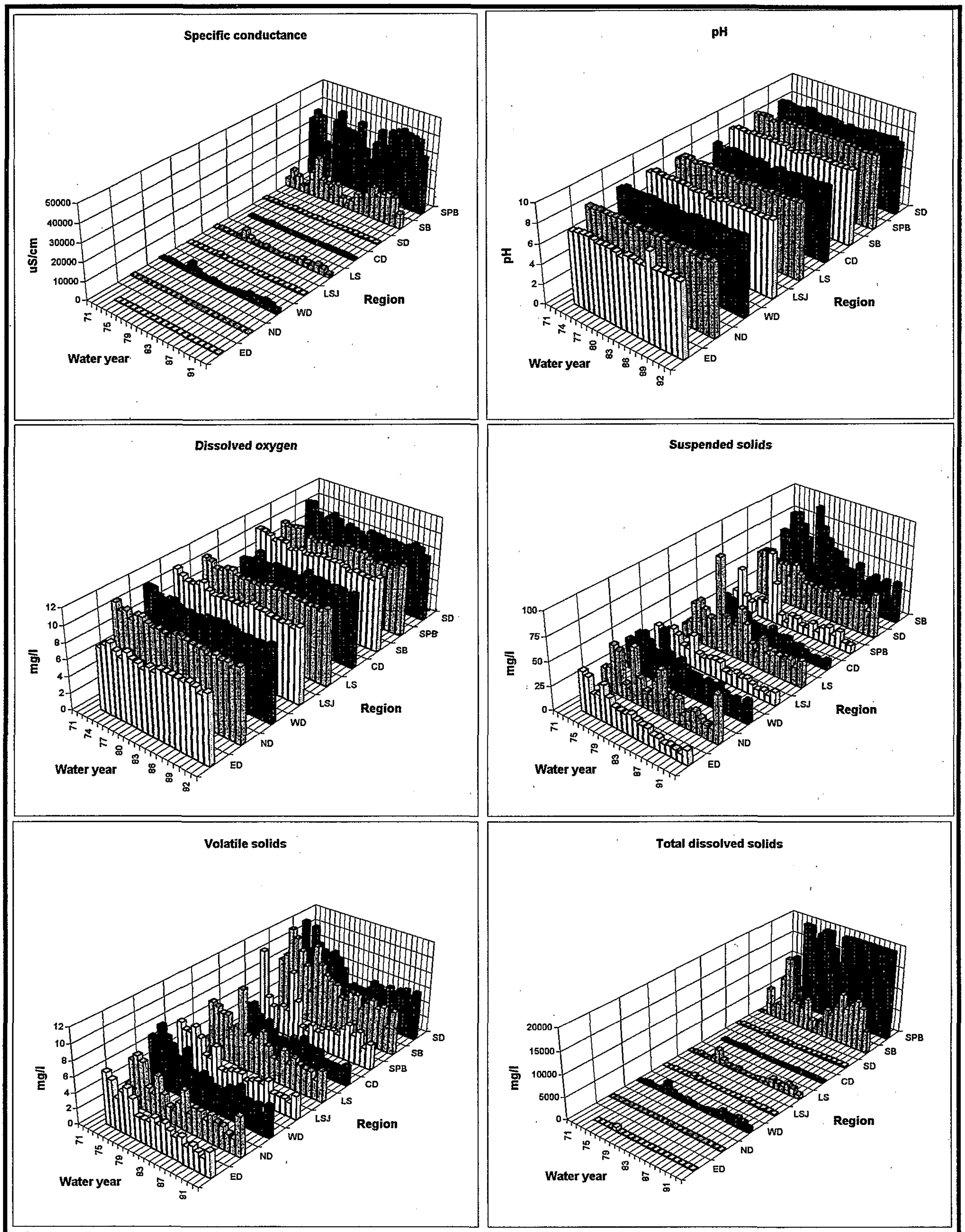

Figure 23
LONG-TERM TRENDS FOR CHEMICAL VARIABLES, 1971-1993

Annual average volatile solids decreased over the period of record and averaged 4.29 mg/L. The highest volatile solids concentrations were measured in the southern and Suisun Bay regions; the lowest were in the central region. After the mid-1970s, volatile solids gradually decreased by a factor of 2 for all regions.

Total dissolved solids increased with distance downstream. Annual average total dissolved solids were less than 10 mg/L in the northern region upstream and increased to 24000 mg/L downstream in San Pablo Bay. Among years, dissolved solids were high during droughts in the mid-1970s and late 1980s. The very low total dissolved solid concentrations in 1982-1983 were probably a function of dilution from record high inflows.

Water-Year Trends

Figures 24-28 show percent deviation from the long-term mean for specific conductance, pH, dissolved oxygen, volatile solids, and total dissolved solids for wet, normal, dry, and critical water years. Volatile solids and suspended solids were similar.

During wet years, specific conductance and total dissolved solids were lower than average for most stations but decreased more downstream. In contrast, volatile solids were above average for most stations but were lower in the southern region. Among stations, dissolved oxygen concentrations increased and pH decreased, and the difference between stations was small.

During normal years, specific conductance, pH, dissolved oxygen concentration, volatile solids, and total dissolved solids varied little among stations. Specific conductance and total dissolved solids were up to 58% lower than average, while dissolved oxygen concentration and pH were up to 13% higher than average. The largest deviations were calculated for volatile solids, which increased by 15-88%.

During dry years, variation was high among stations. Volatile solids and pH were above average downstream. Specific conductance and total dissolved solids were lower only in the western region. No pattern was apparent for dissolved oxygen, which varied between adjacent stations.

During critical years, specific conductance, pH, and total dissolved solids were high. Specific conductance increased by up to 58% upstream and 100% downstream due to saltwater intrusion. Total dissolved solids also increased more downstream (42-88%) than upstream (0-41%). The pattern was the opposite for dissolved oxygen concentration, which increased upstream where phytoplankton blooms develop. Volatile solids had a downstream gradient, with values 15-88% higher than average upstream and 14-45% lower than average downstream. Unlike the other variables, pH did not demonstrate a downstream gradient, increasing by up to 4% above average for most stations.

Seasonal Trends

Figures 29-33 show percent deviation from the long-term mean for specific conductance, pH, dissolved oxygen concentration, volatile solids, and total dissolved solids for fall, winter, spring, and summer. Volatile solids and suspended solids were similar.

High inflows in the spring reduced specific conductance and total dissolved solids by up to 43% at downstream stations. Higher-than-average volatile solids throughout the region suggest that high spring inflows may transport volatile solids from upstream or resuspend material from the bottom sediments. Low water temperatures in spring plus mixing associated with high inflows probably produced the 8% higher-than-average dissolved oxygen concentrations throughout the region, but inflows were not high enough to decrease pH to below average values.

Low streamflows plus increased marine water intrusion in the summer probably produce the 40-62% higher specific conductance and 30-120% higher total dissolved solids downstream of the confluence. The reduced downstream transport and increased residence time associated with low streamflows probably also produced the 68% higher-than-average volatile solids upstream. Most of this increase may be from living or decomposed phytoplankton, which increase during summer. Reduced mixing and inflows and high water temperatures in summer also kept dissolved oxygen concentrations at 8-18% below average and pH at 2-7% above average at most stations.

Low inflows in the fall increased total dissolved solids to 12-30% and specific conductance to 40-62% above average for all stations except those near the margin of the Delta. Low phytoplankton production in the fall may contribute to the lower-than-average volatile solids. In contrast, dissolved oxygen (8%) and pH (1%) remained near average for most stations.

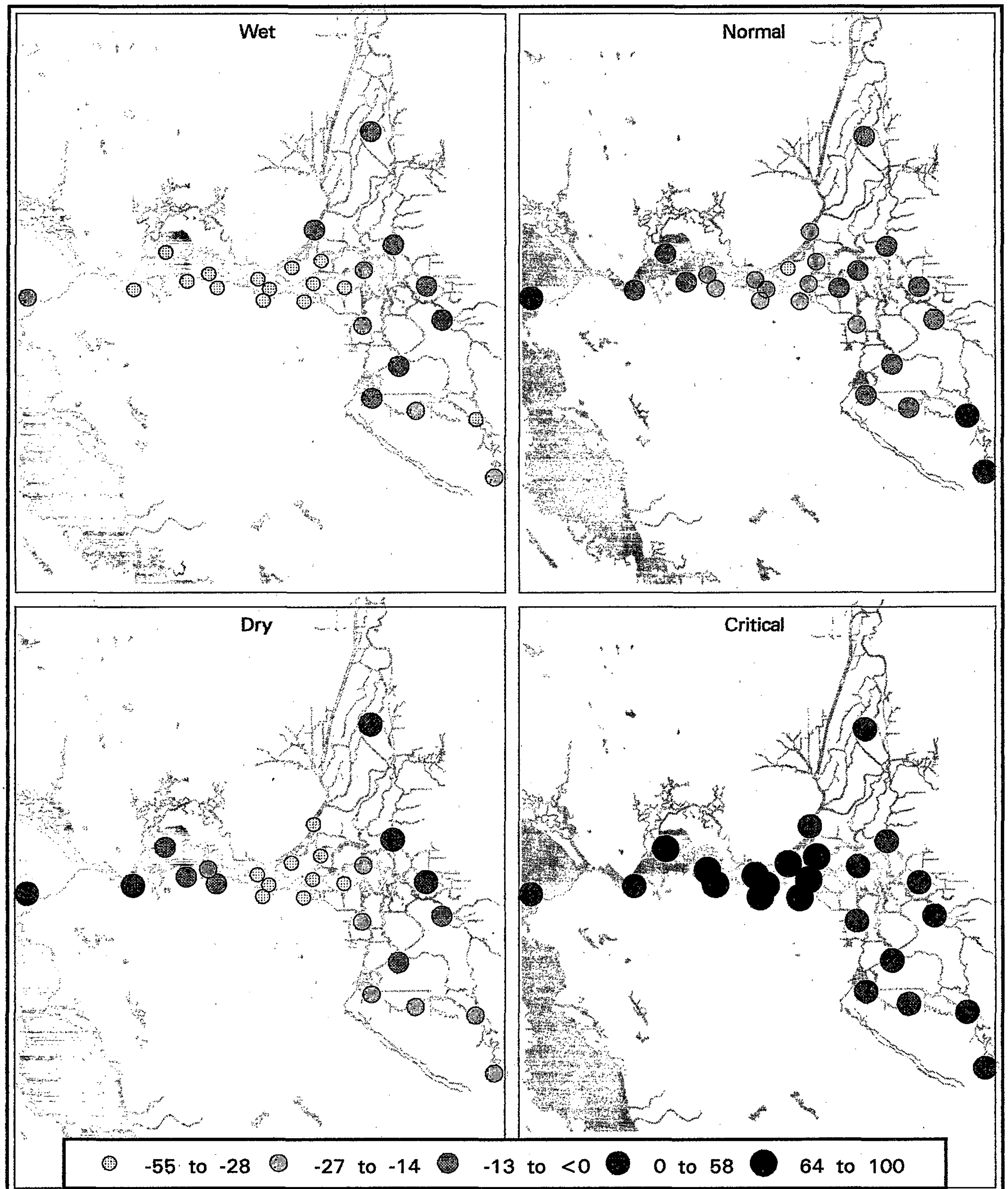

Figure 24

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR SPECIFIC CONDUCTANCE, BY WATER-YEAR TYPE

Gaps between data ranges in legend indicate no data.

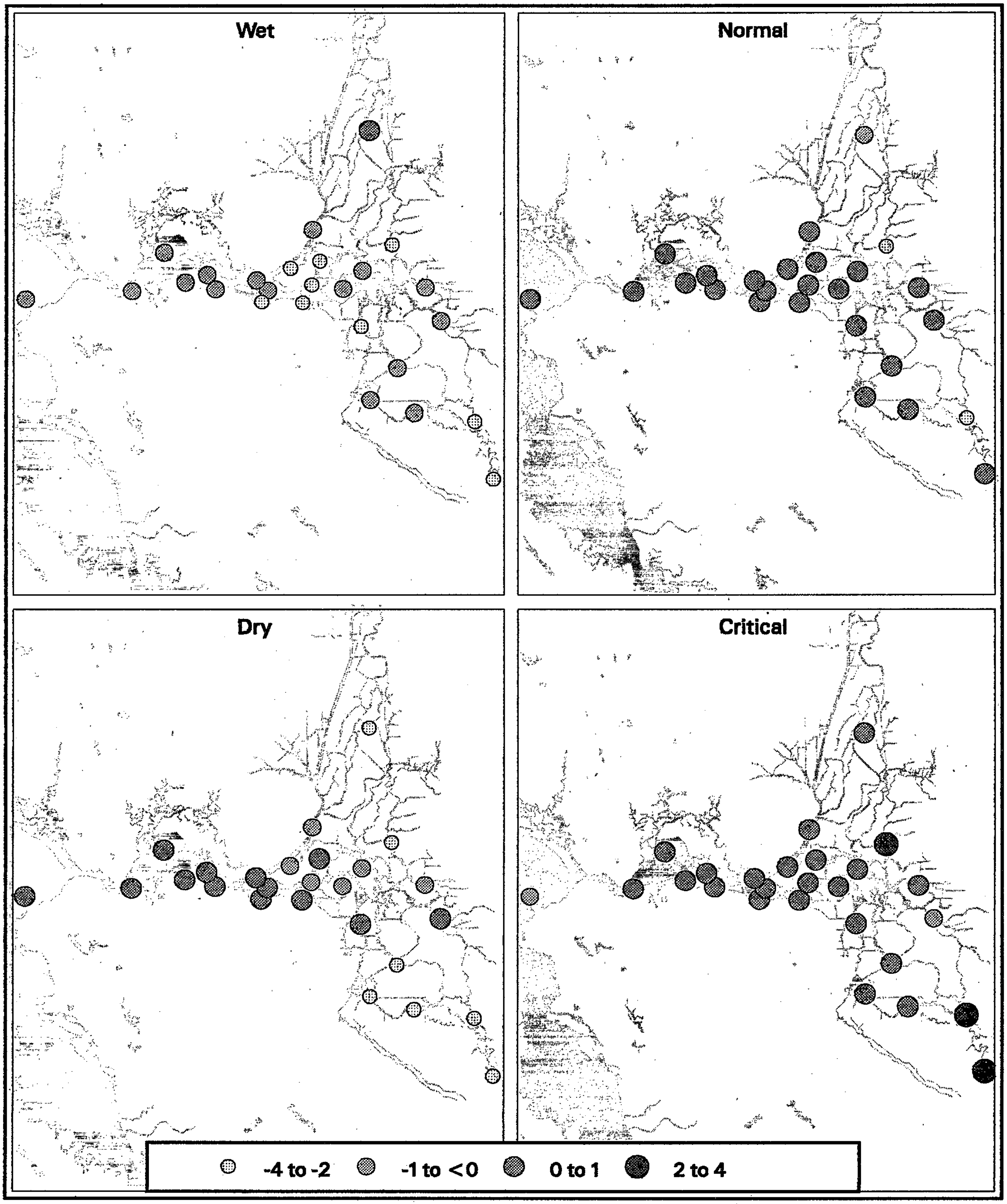

Figure 25
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR pH, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

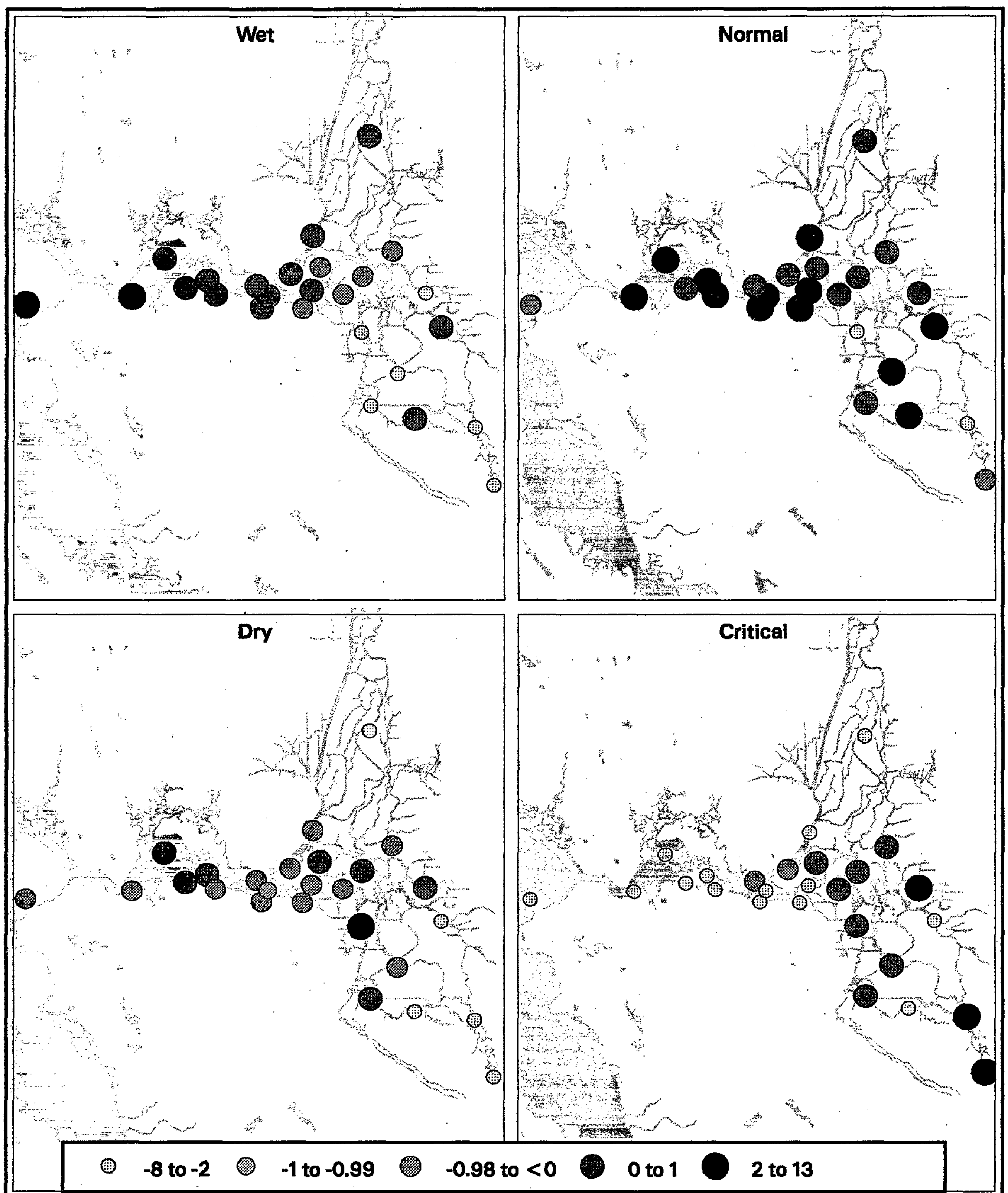

Figure 26
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR DISSOLVED OXYGEN CONCENTRATION, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

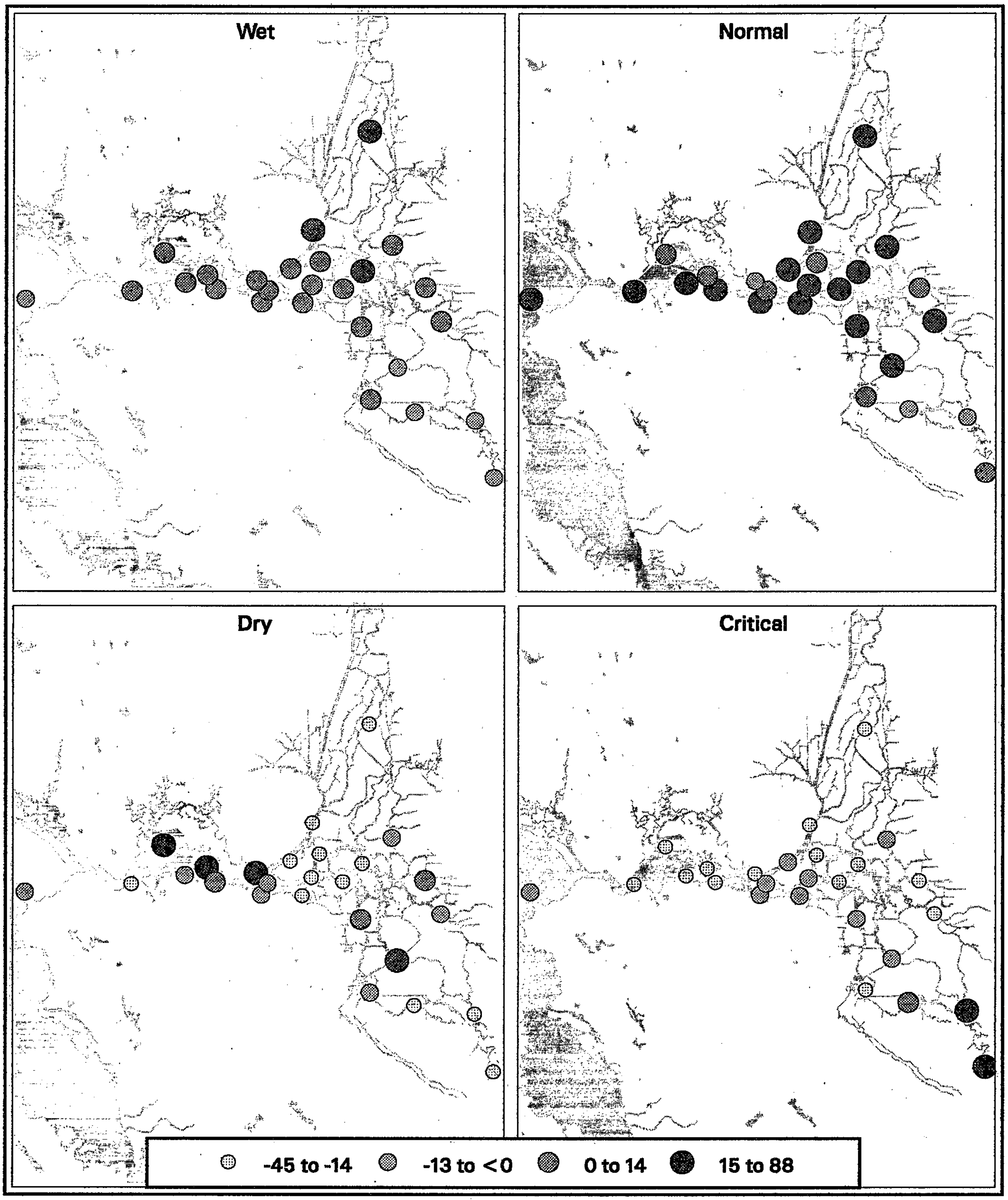

Figure 27
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR VOLATILE SOLIDS, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

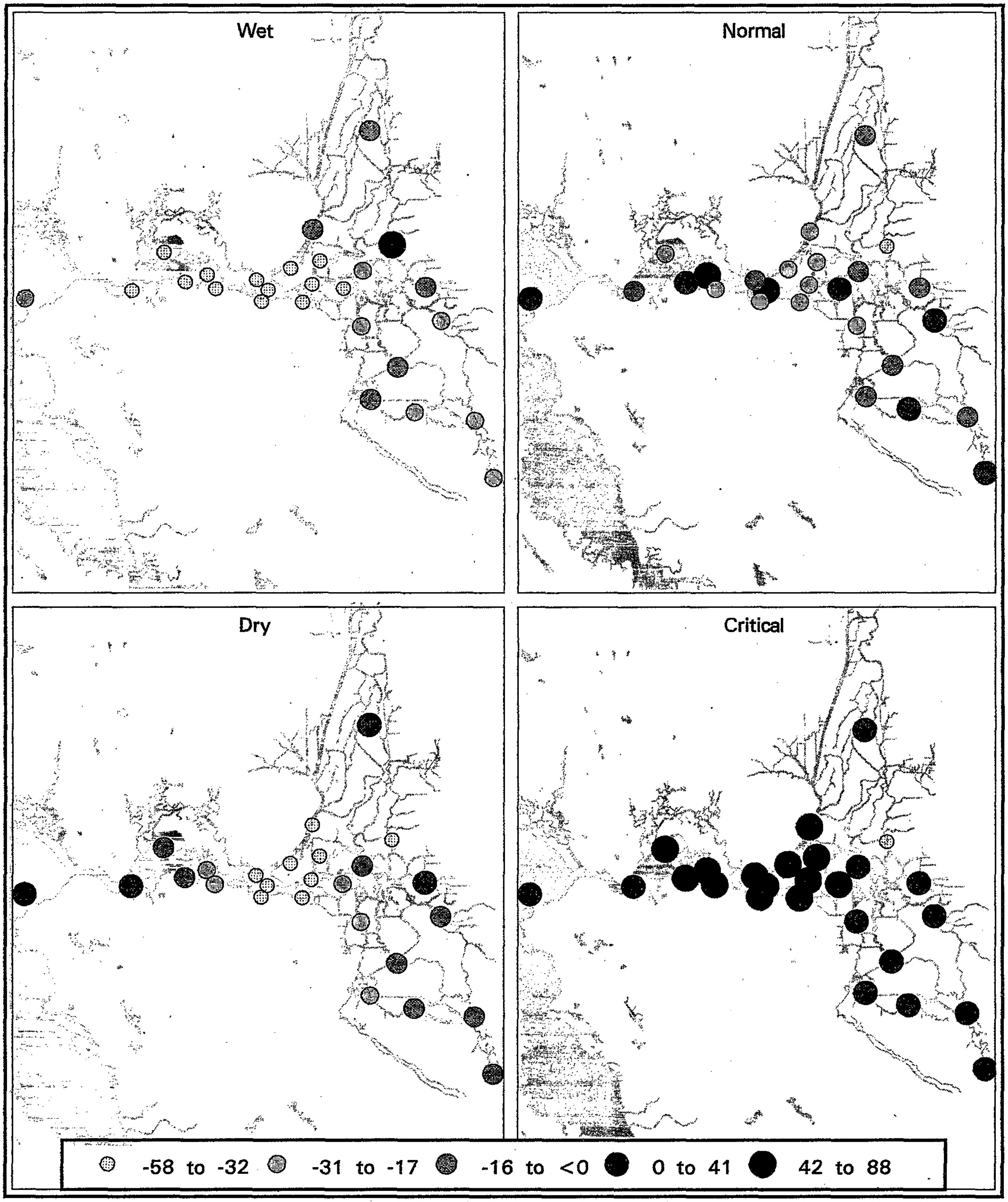

Figure 28

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL DISSOLVED SOLIDS, BY WATER-YEAR TYPE

Gaps between data ranges in legend indicate no data.

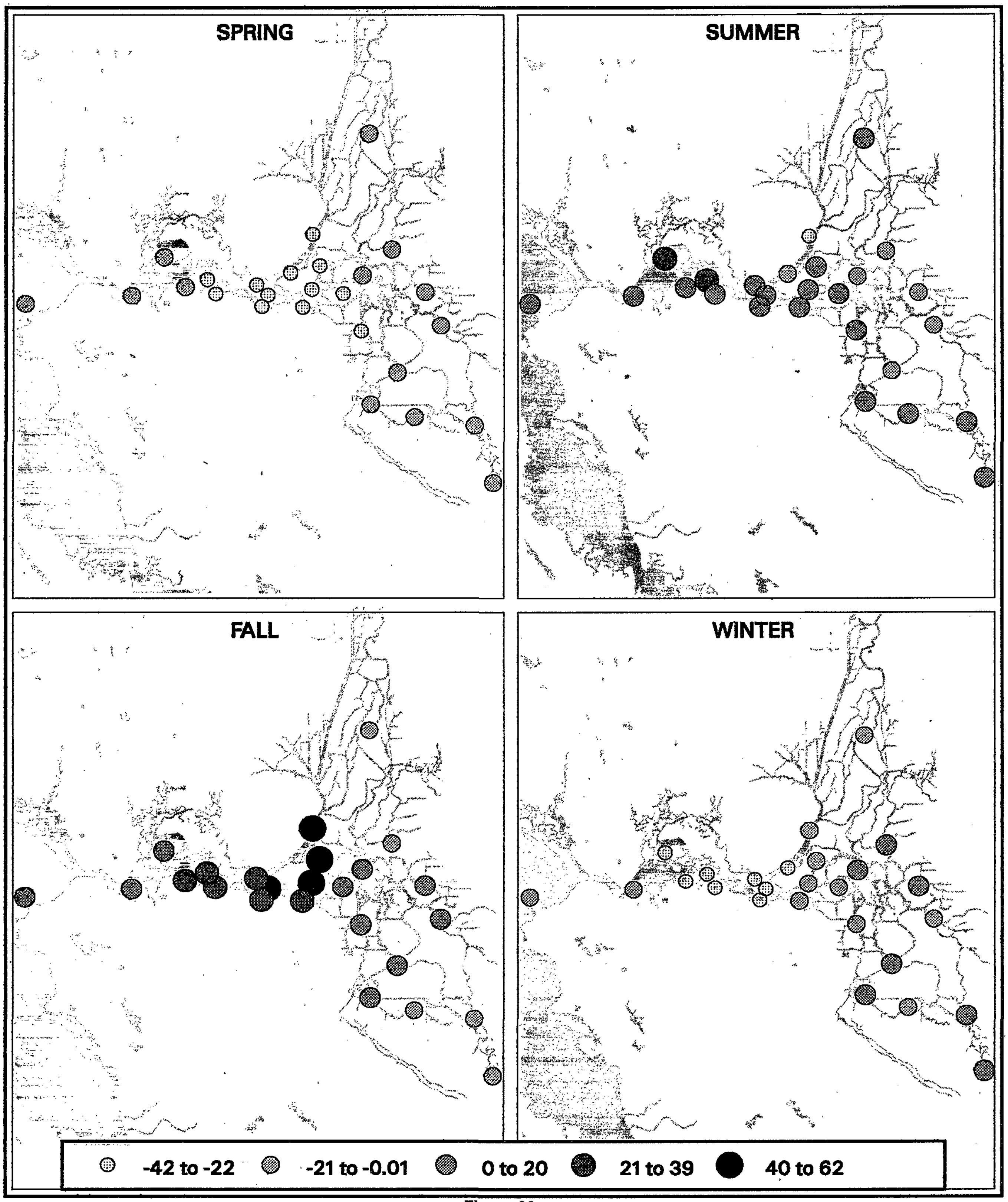

Figure 29
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR SPECIFIC CONDUCTANCE, BY SEASON
Gaps between data ranges in legend indicate no data.

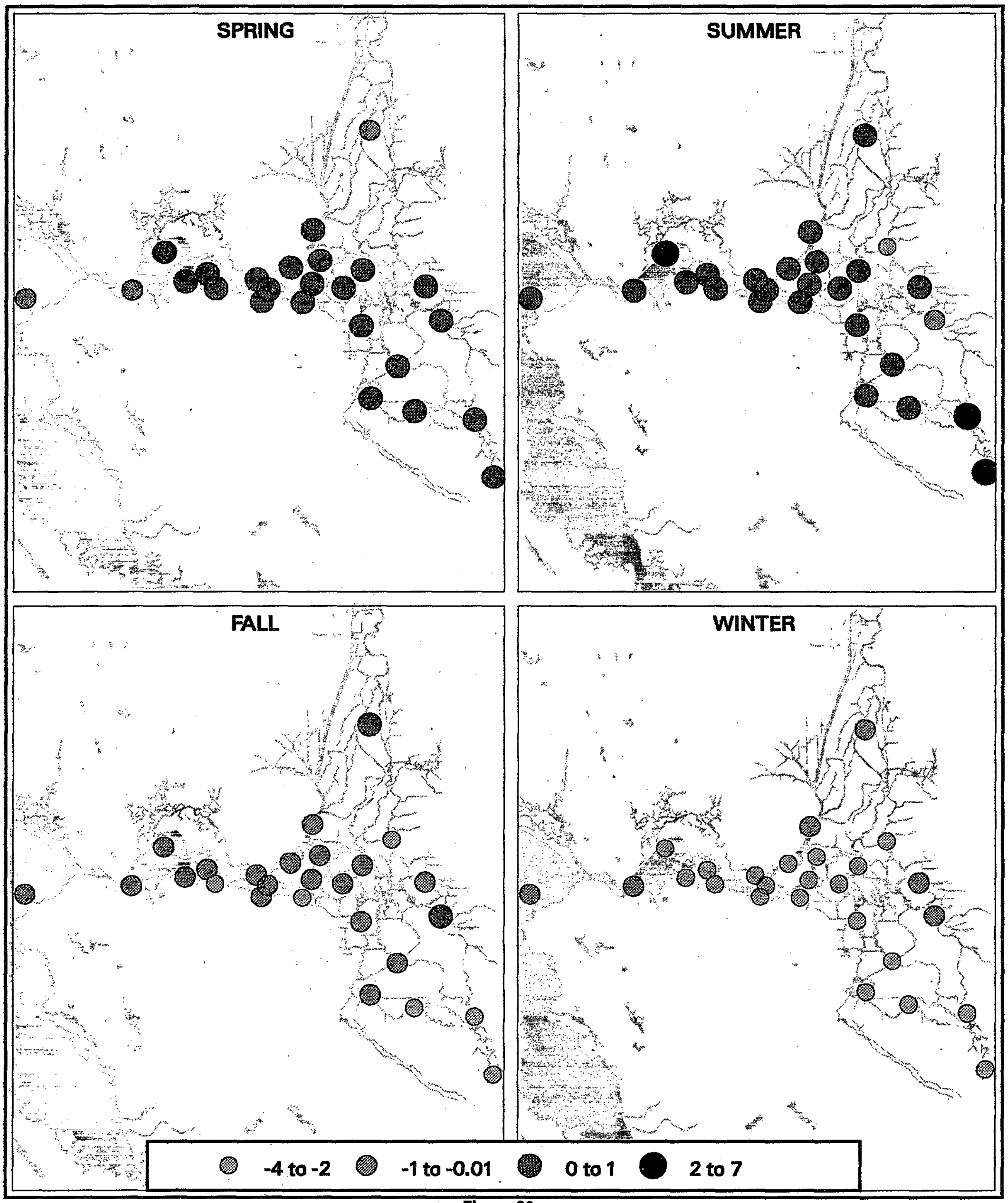

Figure 30
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR pH, BY SEASON
Gaps between data ranges in legend indicate no data.

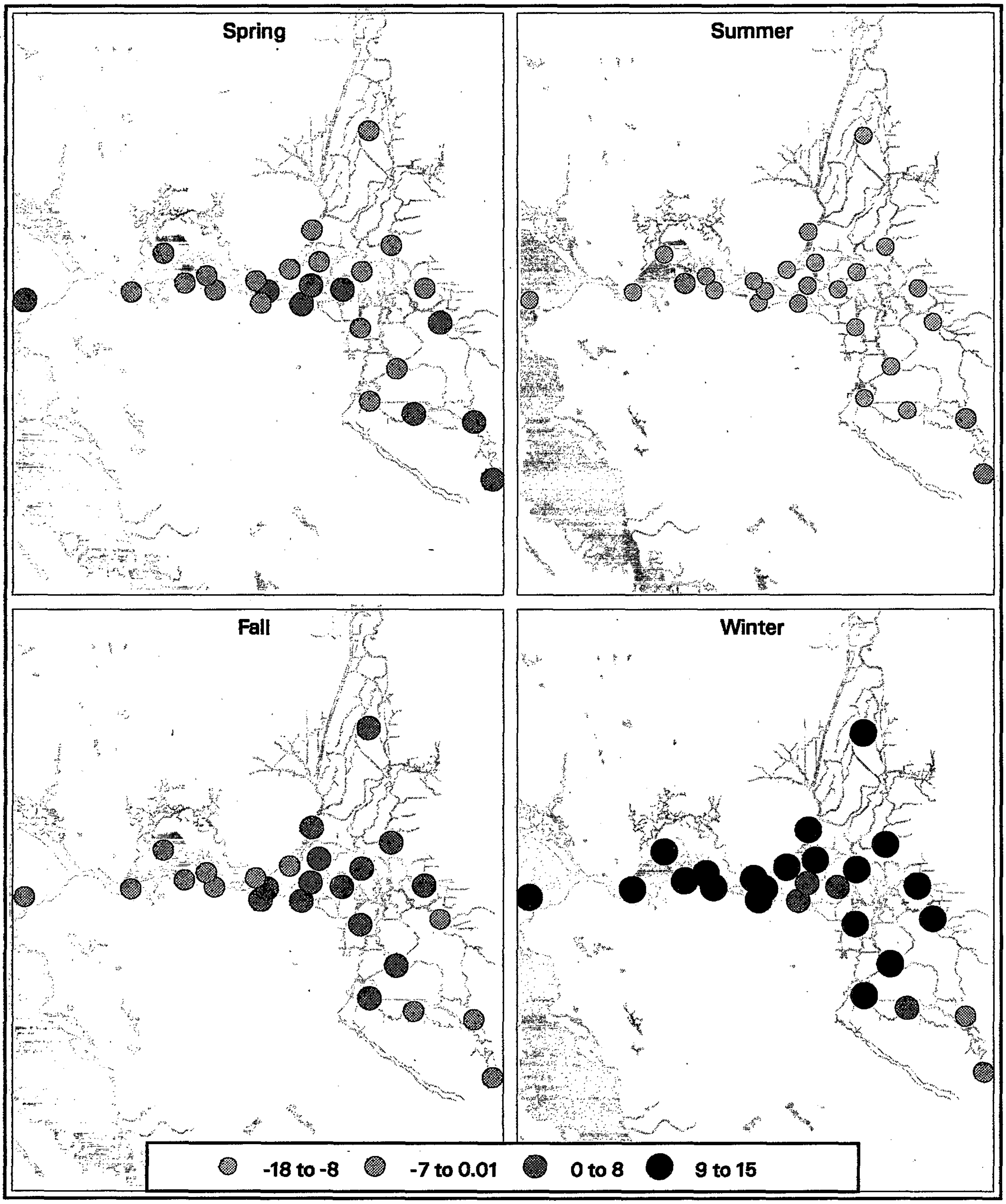

Figure 31
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR DISSOLVED OXYGEN, BY SEASON
Gaps between data ranges in legend indicate no data.

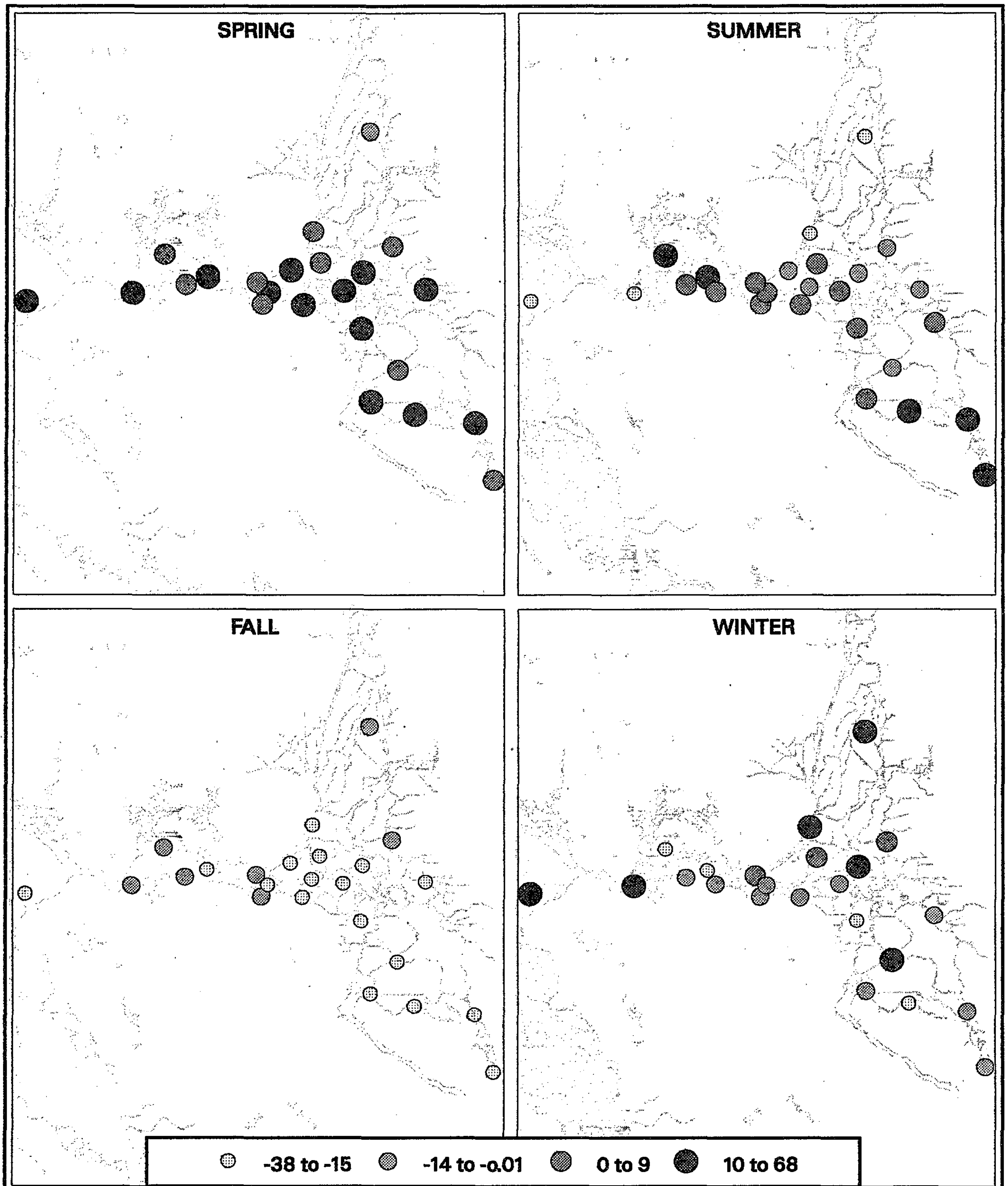

Figure 32
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR VOLATILE SOLIDS, BY SEASON
Gaps between data ranges in legend indicate no data.

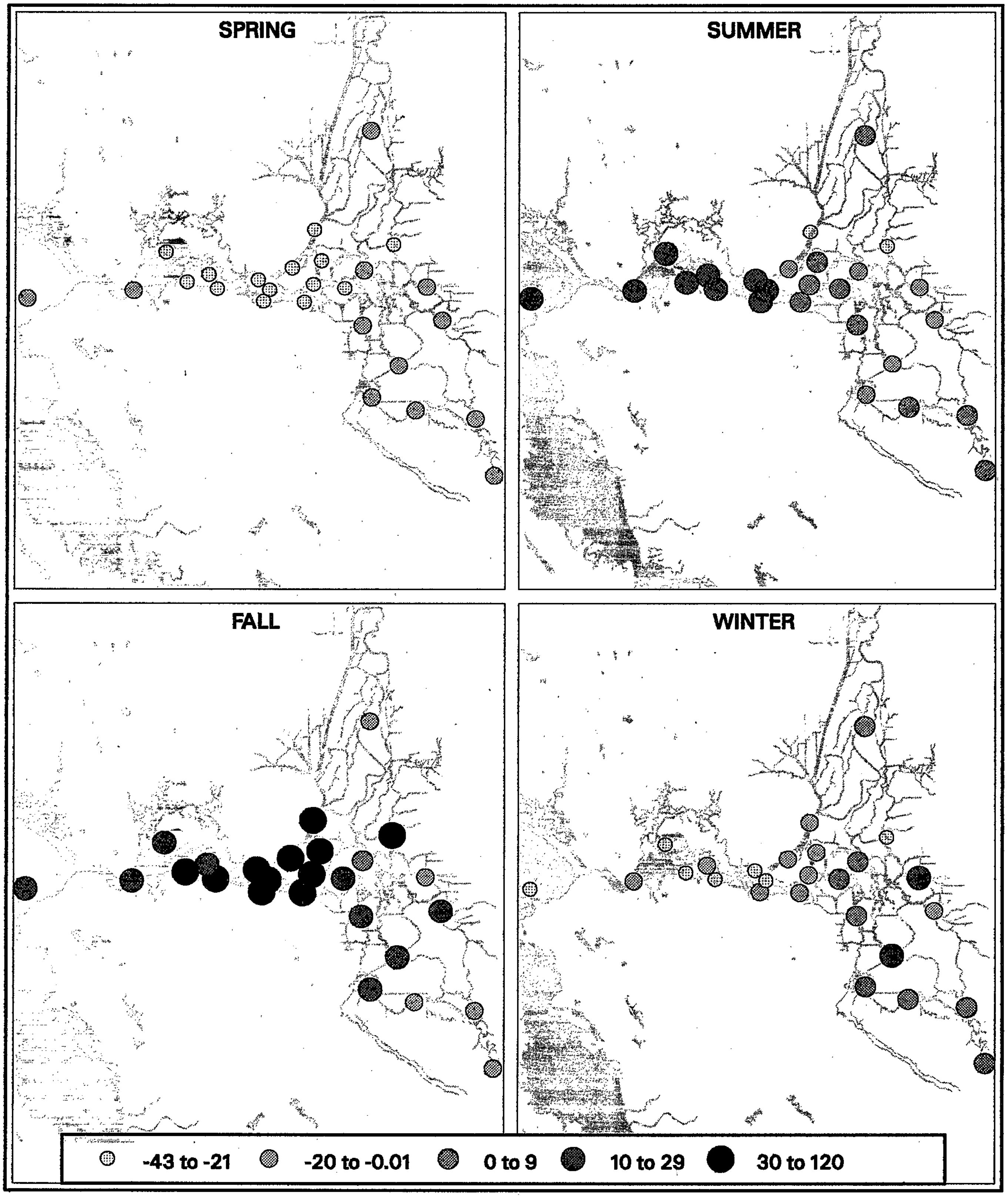

Figure 33
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL DISSOLVED SOLIDS, BY SEASON
Gaps between data ranges in legend indicate no data.

With high winter inflows, total dissolved solids and specific conductance decreased by up to 43% downstream. High inflows and associated mixing could also account for the higher-than-average dissolved oxygen concentrations (9-15%) and lower-than-average pH (2-4%) throughout the region. The increase of volatile solids by 10-68% suggests high inflows may also increase the organic loading from upstream at some stations.

Nutrient Concentrations

Nutrient concentrations were measured monthly or semi-monthly at the same 26 stations as for physical variables (Figure 11). Water samples were collected at 1 meter within 1 hour of high slack tide using a Van Dorn sampler or a submersible pump. The time of each sample was recorded to the nearest 5 minutes using Pacific Standard Time. Total water depth was measured to the nearest foot by an electronic depth-sounding instrument or a weighted measuring tape.

Water samples for nutrient analyses were stored in new polyethylene bottles that were rinsed twice with distilled water. A filtered and a nonfiltered sample were taken at each site. For the filtered sample, the water sample was passed through a prewashed 0.45-micron pore size membrane filter, and the filtrate was frozen immediately. Concentrations of combined nitrate and nitrite, ortho-phosphate, ammonia, silicate, and dissolved organic nitrogen were determined from filtered samples. Concentrations of total organic nitrogen and total phosphorus were analyzed from unfiltered samples.

The minimum reporting limit was 0.01 mg/L for all nitrogen and phosphorus species except organic nitrogen, for which it was 0.1 mg/L. Specific methods for each nutrient are listed in Table 2.

Stations were grouped into the same regions as for physical variables, based on individual and combined hierarchical cluster analysis of monthly data (Table 1).

Long-Term Trends

Figure 34 shows long-term trends for 1971-1993 for total phosphate, silica, organic nitrogen, ammonia nitrogen, nitrate, and ortho-phosphate, all of which are discussed below.

Average annual total phosphate concentration was 0.15 mg/L, with the highest values in the southern region. Total phosphate concentration followed the same pattern of change over time as phosphate concentration, with low concentrations during the record streamflows of the early 1980s, when dilution and phytoplankton uptake were high. High concentrations occurred during the droughts of the mid-1970s and late 1980s, when dilution and phytoplankton uptake were low.

Average annual silica concentrations were high in the upper estuary. Silica inputs from the Sacramento and San Joaquin rivers averaged 13.9 mg/L, with a range of 3-18 mg/L throughout most of the upper estuary. Silica concentrations were consistently lower than 10 mg/L in San

Table 2
NUTRIENT SAMPLING METHODS

Substance	Method	Method #	Reference
Silica	Colormetric, molybdate blue method	I-1700-85	USGS 1985
Ammonia	Colormetric, automated phenate method	350.1	EPA 1983
Nitrite plus nitrate	Colormetric, automated, cadium reduction	353.2	EPA 1983
Total and dissolved ammonium plus organic nitrogen	Colormetric, semi-automated method	351.2	EPA 1983
Ortho-phosphate	Colormetric, automated, ascorbic acid method	365.1	EPA 1983
Total phosphorus	Colormetric, semi-automated method	365.4	EPA 1983

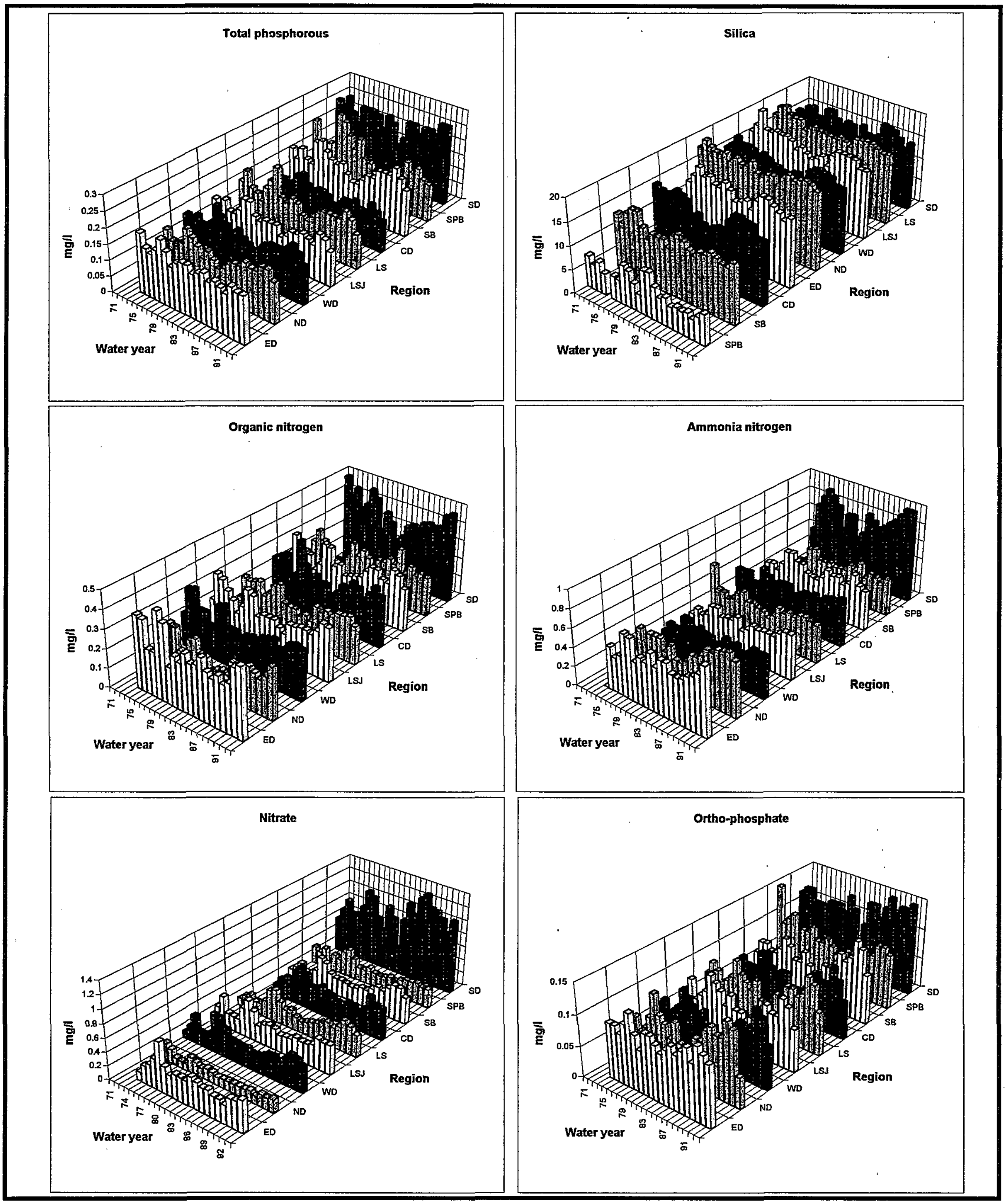

Figure 34
LONG-TERM TRENDS FOR NUTRIENTS, 1971-1993

Pablo Bay, where silica inputs from upstream sources are small except during wet years. Although phytoplankton blooms may occasionally remove silica to limiting levels, concentrations were usually not limiting. Over the period of record, silica concentrations varied with inflow and phytoplankton biomass. High concentrations coincided with high inflows in the early 1970s and low phytoplankton uptake during the late 1980s.

Average annual organic nitrogen concentration was 0.24 mg/L. In the southern region, concentrations were relatively high, at 0.20-0.45 mg/L. High concentrations in the lower Sacramento and western regions despite low concentrations in the upper Sacramento at the northern region suggest that some organic material comes from sources in the Delta. Carbon flux calculations suggest that about 38% of the upstream organic material reaching Suisun Bay is derived from phytoplankton in the Delta (Jassby *et al* 1995). Some organic nitrogen may also be discharged from sewage treatment facilities. Among years, concentrations were lower in the early 1980s, when high inflows diluted concentrations and flushed organic material downstream. High concentrations were also associated with the 1987-1992 drought.

Average annual ammonia plus organic nitrogen concentration for the upper estuary was 0.46 mg/L. Concentrations were a factor of 2 higher in the southern region, which also had the widest range in values. The high variability of ammonia plus organic nitrogen in the southern region may be partly due to fertilizer applications in this agricultural region. Over time, concentrations varied in a similar fashion to organic nitrogen, ortho-phosphate, and total phosphate, with high concentrations during the droughts in the mid-1970s, late 1980s, and early 1990s.

Average annual nitrate concentration for the upper estuary was 0.37 mg/L. Concentrations were variable among regions but were consistently higher in the southern region, where fertilizers may add nitrate into the San Joaquin River. Nitrate concentrations for the upper Sacramento River were low, at 0.2 mg/L or less, but higher in the lower Sacramento River, where nutrients are input from sewage treatment plants and the San Joaquin River. Nitrate concentrations are usually non-limiting to phytoplankton but can reach limiting levels during blooms. Among years, concentrations were high in the mid-1970s and late 1980s, when phytoplankton uptake was low.

Ortho-phosphate concentrations averaged 0.08 mg/L and were variable among regions. Concentrations were highest in the southern region, possibly due to inputs from the nutrient-rich San Joaquin River and agricultural fertilizers. Ortho-phosphate concentration is usually above limiting levels but may reach limiting levels during phytoplankton blooms. Ortho-phosphate concentrations were low during the early 1980s, when record streamflows diluted nutrient concentrations and phytoplankton uptake was high. Concentrations were high during the drought years of the mid-1970s and late 1980s, when dilution and phytoplankton uptake were low.

Water-Year Trends

Figures 35-40 show percent deviation from the long-term mean for total phosphate, silica, organic nitrogen, ammonia nitrogen, nitrate, and ortho-phosphate for wet, normal, dry, and critical water years, all of which are discussed below.

During wet years, nutrient concentrations varied by as much as 30% at upstream and downstream stations. Nitrate, total phosphate, and ortho-phosphate concentrations were lower than average downstream of the San Joaquin River. This may be partly a function of phytoplankton uptake, because chlorophyll *a* concentrations were higher than average downstream. In contrast, silica concentrations were higher than average in Suisun and San Pablo bays, which receive a higher loading from upstream during wet years. Ammonia plus organic nitrogen, total organic nitrogen, and nitrate concentrations were higher than average in the lower San Joaquin River.

During normal years, silica concentrations were 5-24% higher than average. Dilution is less than in wet years and moderately high streamflows transport silica downstream. Conversely, total phosphorous, ortho-phosphate, nitrate, and ammonia plus organic nitrogen were probably 7-30% lower than average for most stations due to phytoplankton uptake. Chlorophyll *a* concentrations were up to 175% higher than average during normal years, as demonstrated by higher-than-average concentrations of total organic nitrogen.

During dry years, inorganic nutrient concentrations were low and spatially variable. For most stations, nutrients were as much as 30% lower than average downstream, where reduced down-

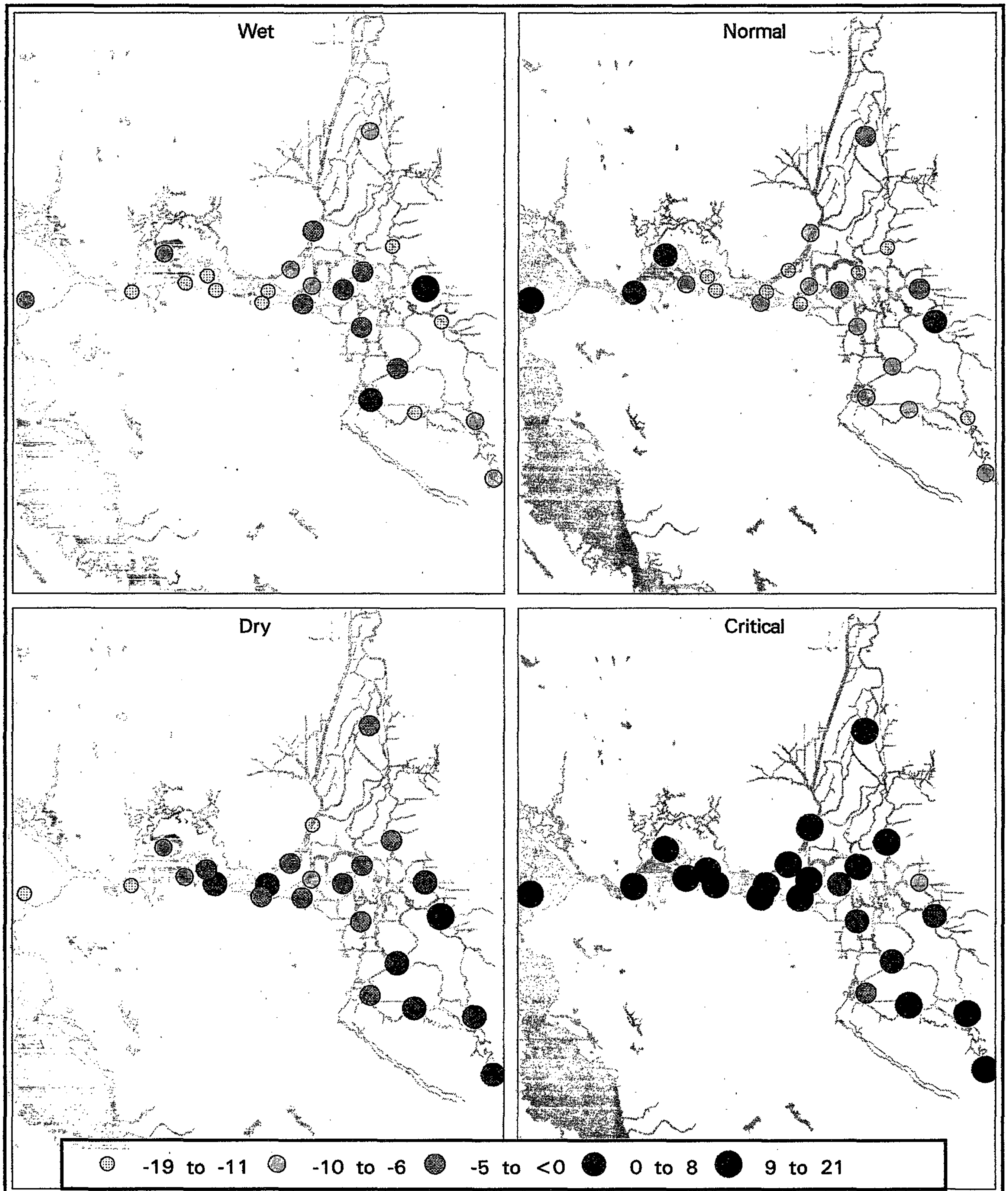

Figure 35
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL PHOSPHATE, BY WATER-YEAR TYPE
Gaps between data ranges in legend indicate no data.

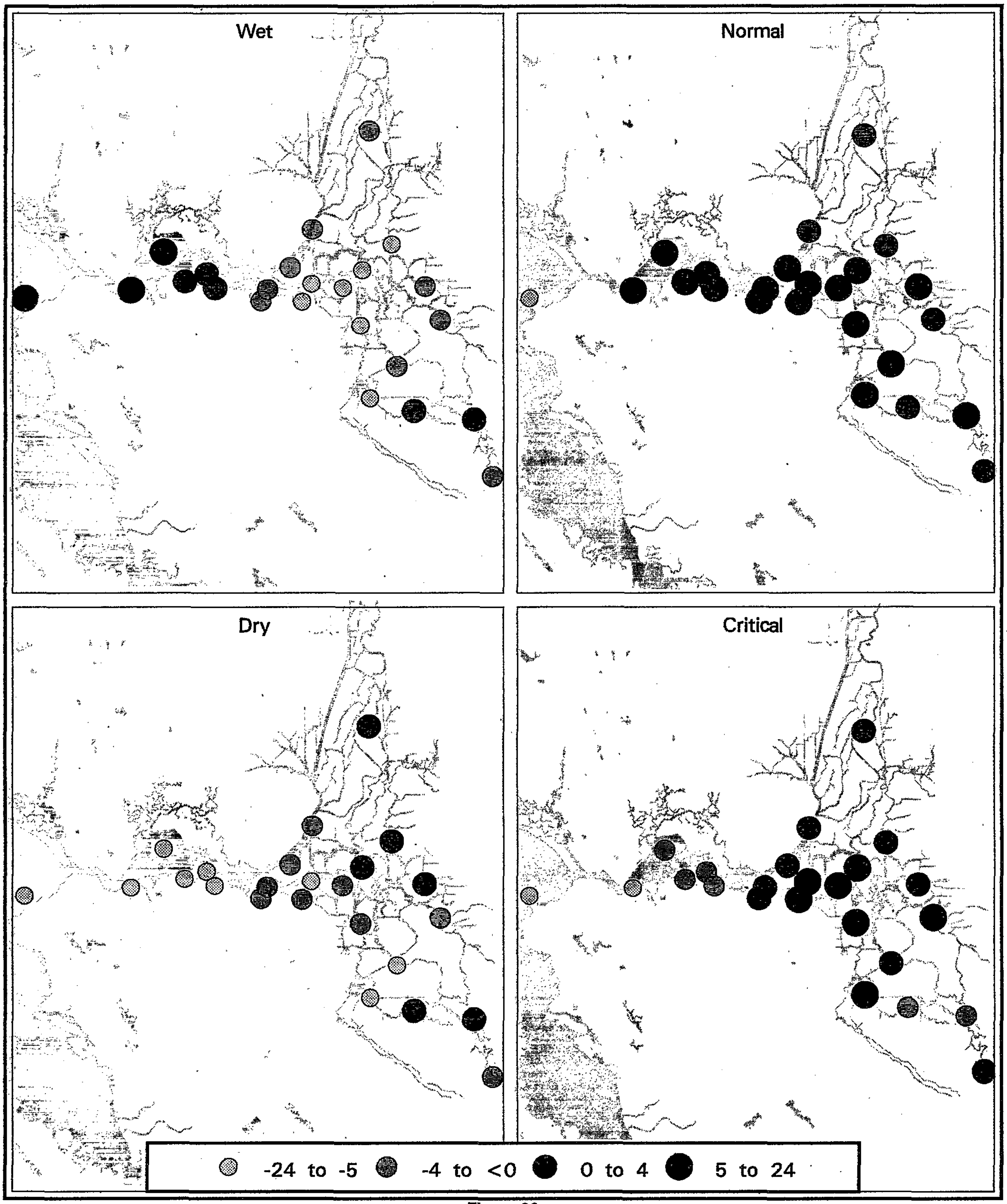

Figure 36
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR SILICA CONCENTRATION, BY WATER-YEAR TYPE
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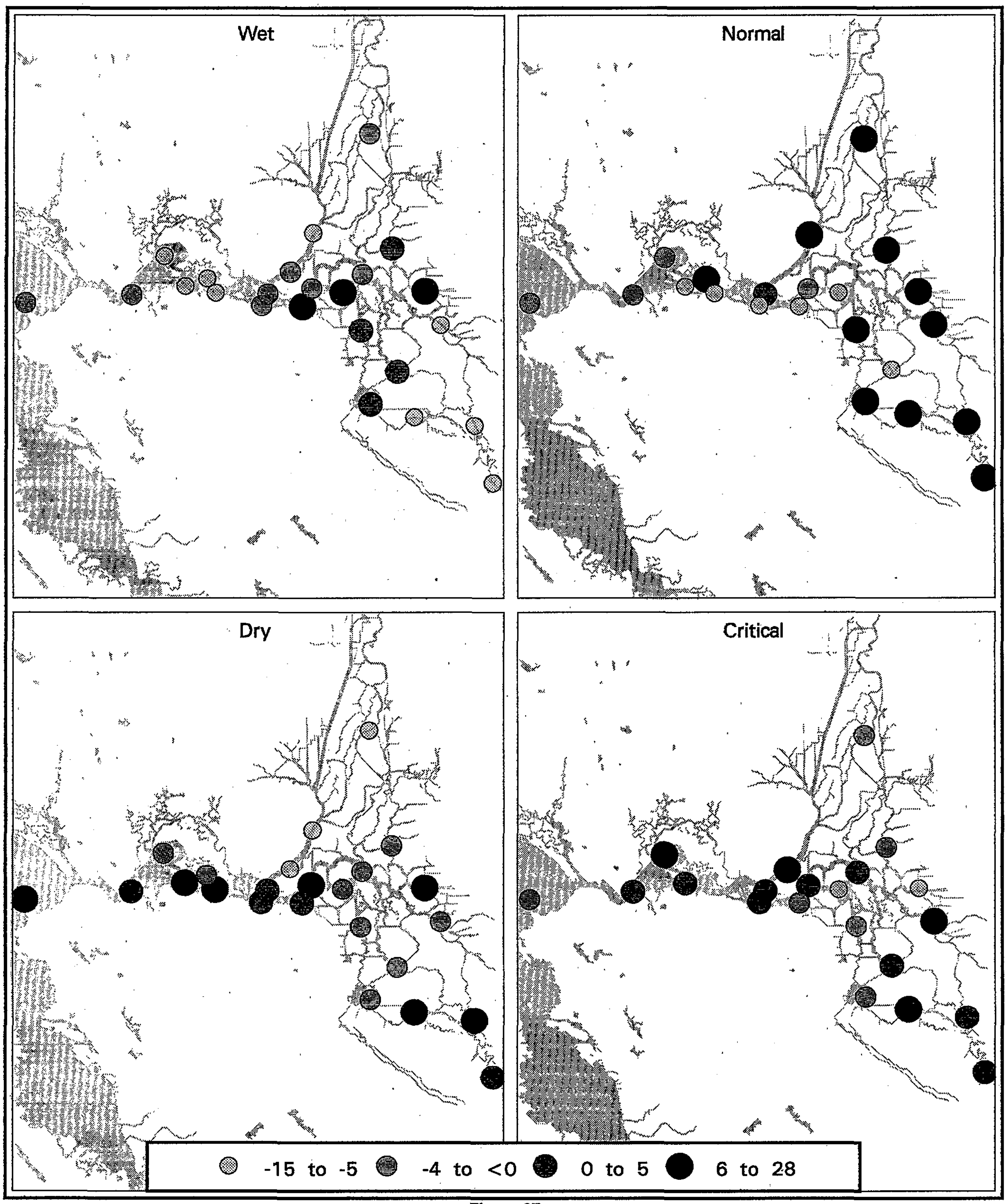

Figure 37
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL ORGANIC NITROGEN, BY WATER-YEAR TYPE
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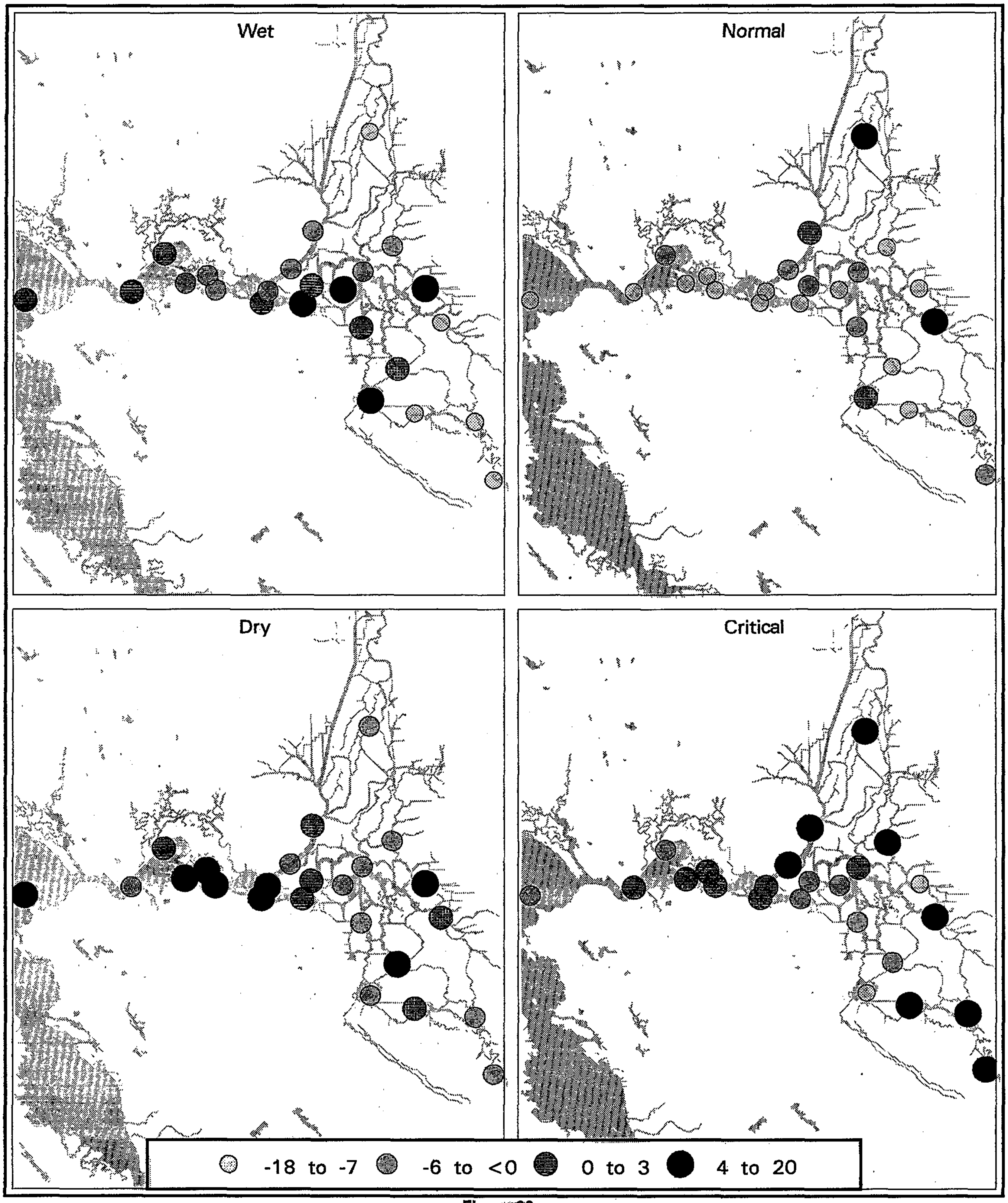

Figure 38
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR AMMONIA PLUS ORGANIC NITROGEN, BY WATER-YEAR TYPE
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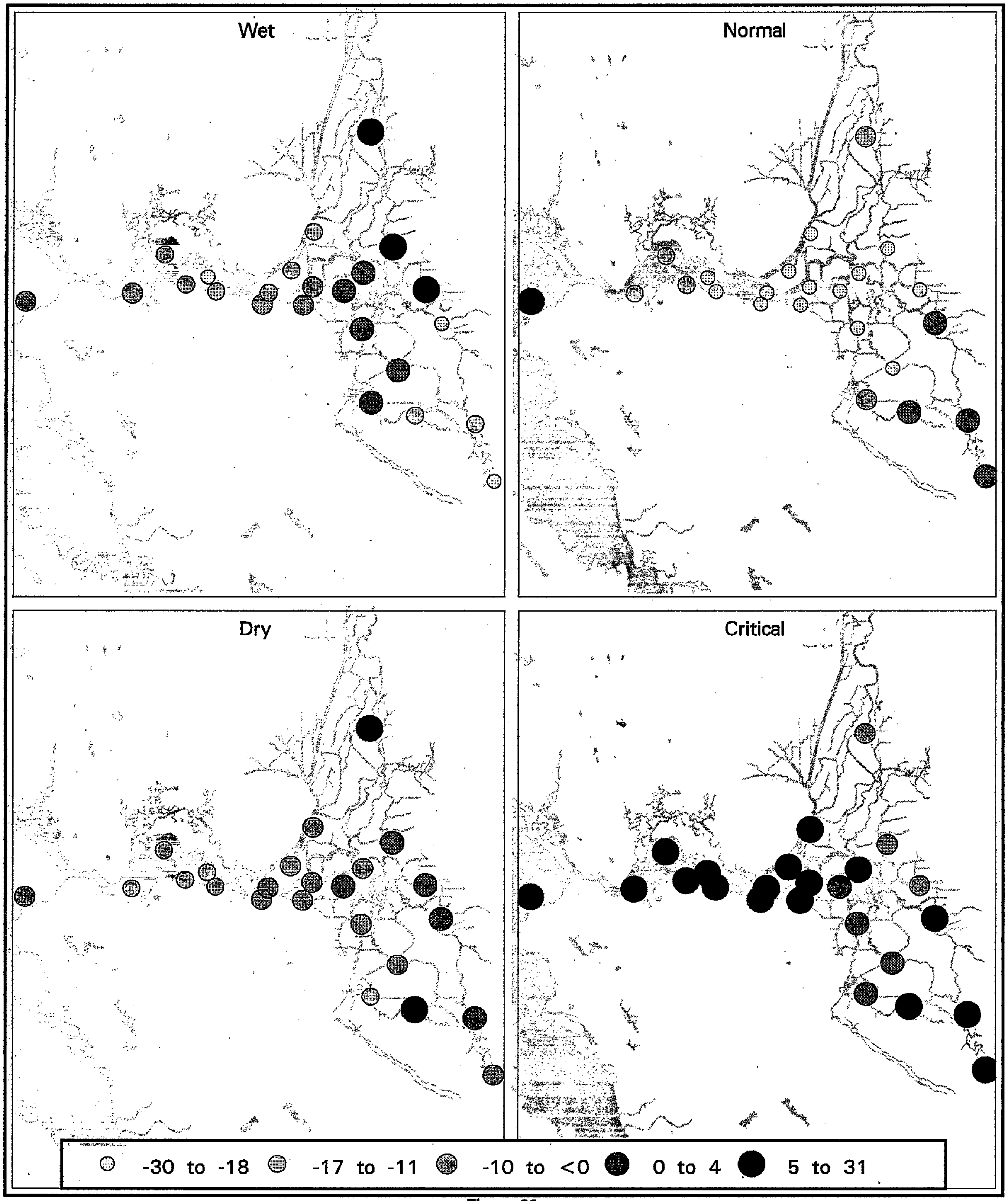

Figure 39

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR NITRATE CONCENTRATION, BY WATER-YEAR TYPE

Gaps between data ranges in legend indicate no data.

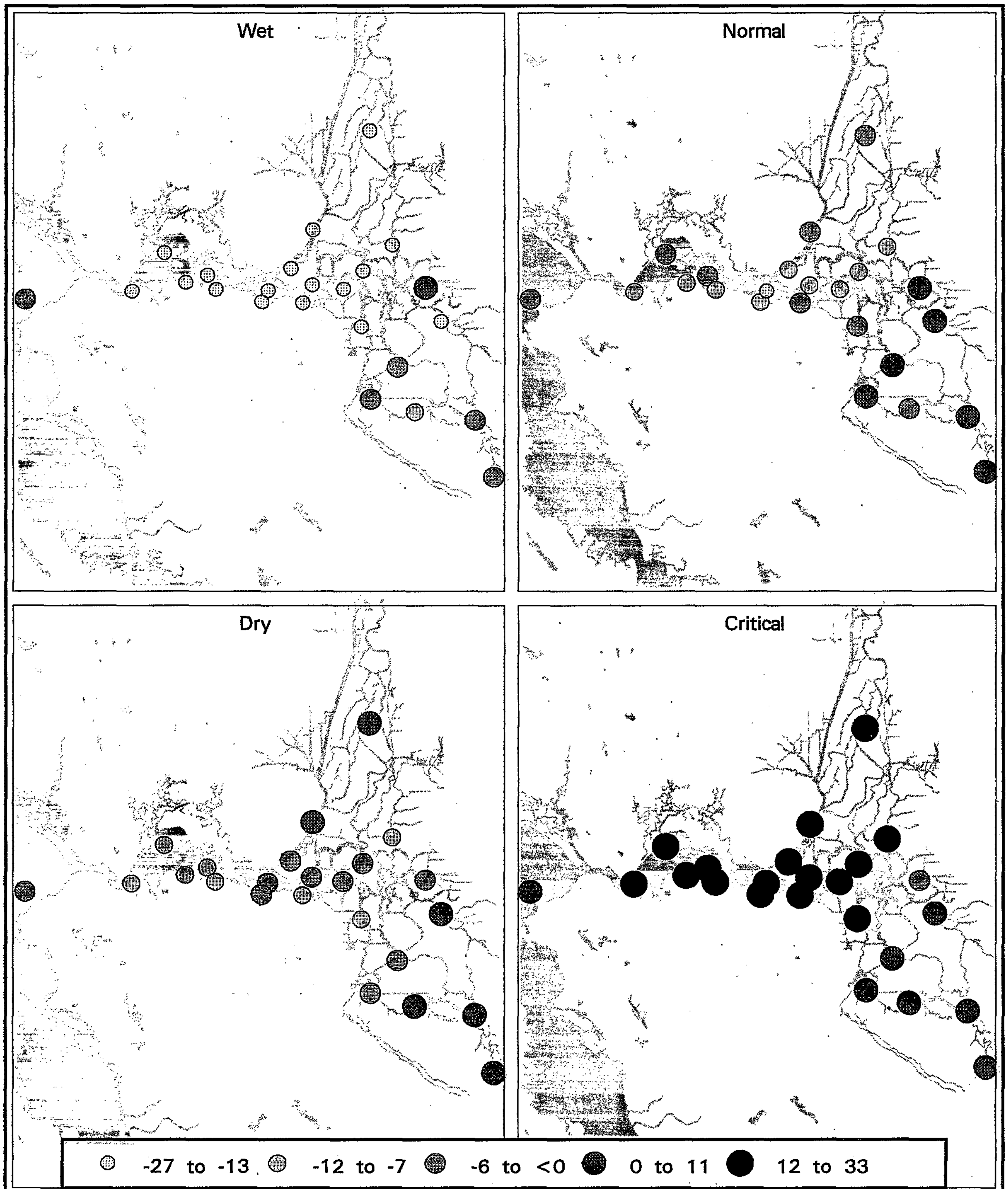

Figure 40

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR ORTHO-PHOSPHATE CONCENTRATION, BY WATER-YEAR TYPE

Gaps between data ranges in legend indicate no data.

stream transport plus management practices replace nutrient-rich San Joaquin River with comparatively nutrient-poor Sacramento River water. The lower-than-average concentrations of inorganic nutrients contrasted with the higher-than-average concentrations of total organic nitrogen and ammonia plus organic nitrogen. Shallow water during dry years may enhance resuspension of organic material from the bottom by wind and tide.

Critical years were characterized by higher-than-average ortho-phosphate, total phosphate, and nitrate concentrations. Concentrations increased by up to 33% among stations and were higher in the lower Sacramento River and seaward than in the lower San Joaquin River. Ammonia plus organic nitrogen and total organic nitrogen concentrations were variable among stations. The higher-than-average silica concentrations in the lower San Joaquin River and central Delta probably reflect reduced silica loading from upstream. Silica concentrations in the estuary are directly related to inflow (Peterson *et al* 1985).

Seasonal Trends

Figures 41-46 show percent deviation from the long-term mean for total phosphate, silica, total organic nitrogen, ammonia plus organic nitrogen, nitrate, and ortho-phosphate during fall, winter, spring, and summer.

During the fall, total phosphorous, ortho-phosphate, nitrate, and silica concentrations were up to 31% higher than average at many stations. Concentrations of total and ortho-phosphate were high downstream, and nitrate concentrations were high upstream. Reduced inflows in the fall also created a downstream gradient of silica concentration. The lower-than-average total organic nitrogen and ammonia plus organic nitrogen at most stations was probably due to decreased transport of organic material from upstream, increased sedimentation at low streamflows, increased export, and reduced phytoplankton growth.

During winter, nutrient concentrations were 7-122% higher than average at most stations and were significantly higher than for other seasons for all nutrients except dissolved ortho-phosphate and total phosphorous. Only ortho-phosphate and total phosphate had lower-than-average concentrations downstream. High nutrient concentrations during the winter were a function of high loading from upstream plus low phytoplankton uptake.

In the spring, nutrient concentrations were lower than average, but concentrations were relatively higher downstream, except for total phosphate. The difference between upstream and downstream was large for ammonia plus organic nitrogen and total organic nitrogen. High concentrations of organic material downstream may result from high inflows, which transport phytoplankton and land-derived organic material downstream. High spring streamflows are associated with increased phytoplankton biomass downstream (Lehman 1996). High spring streamflows also transport more silica downstream and produce the higher-than-average concentrations in Suisun Bay. Nitrate and ortho-phosphate concentrations were lower than average at most stations but were consistently higher downstream.

In summer, higher nutrient concentrations were measured downstream for phosphate, nitrate, total organic nitrogen, and ammonia plus organic nitrogen. Total organic nitrogen and ammonia plus organic nitrogen concentrations were probably higher downstream, because organic material is resuspended from the bottom of shallow bays by wind and tide. As expected, silica concentrations were higher upstream due to reduced downstream transport.

C-036535

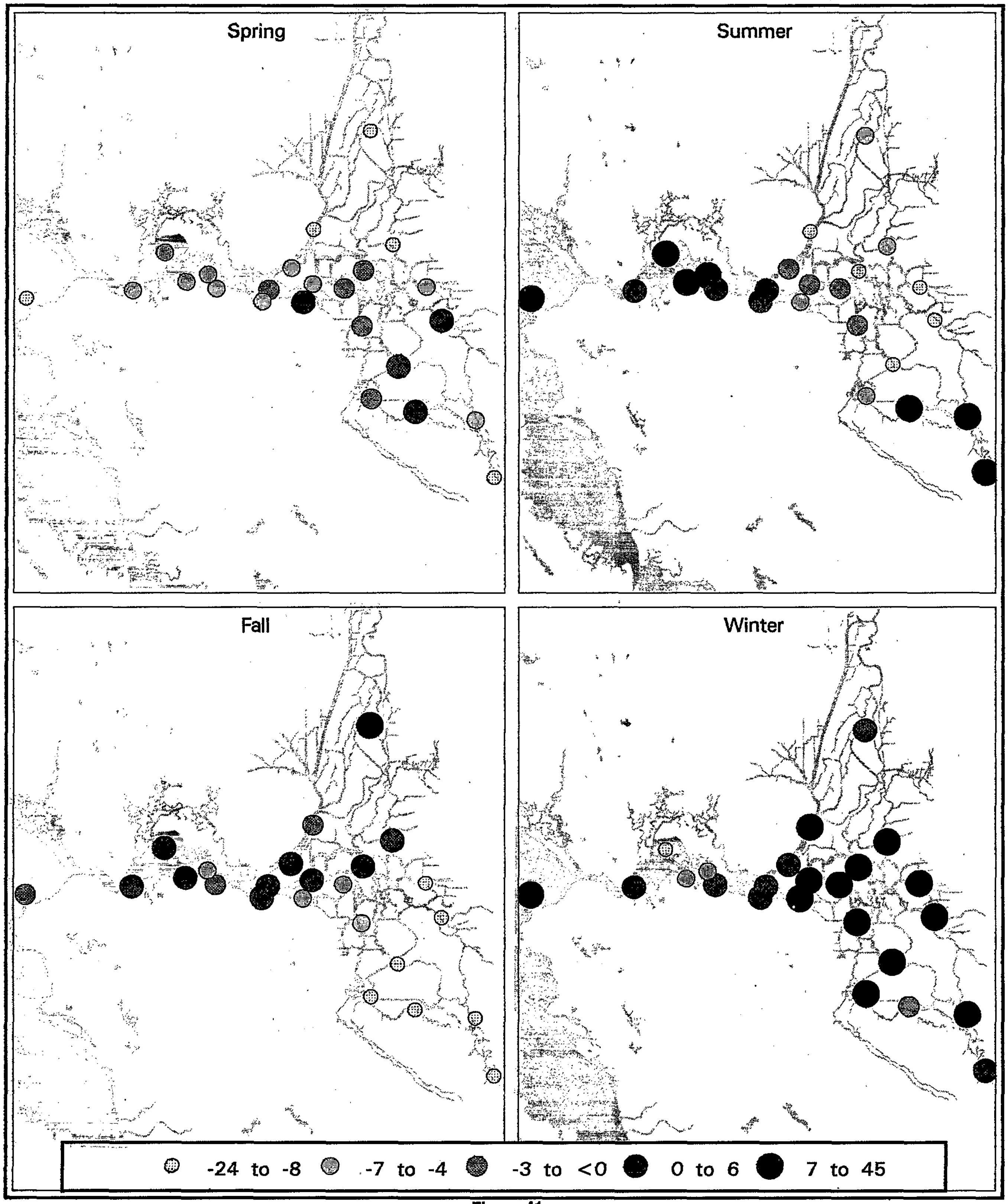

Figure 41

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL PHOSPHATE, BY SEASON

Gaps between data ranges in legend indicate no data.

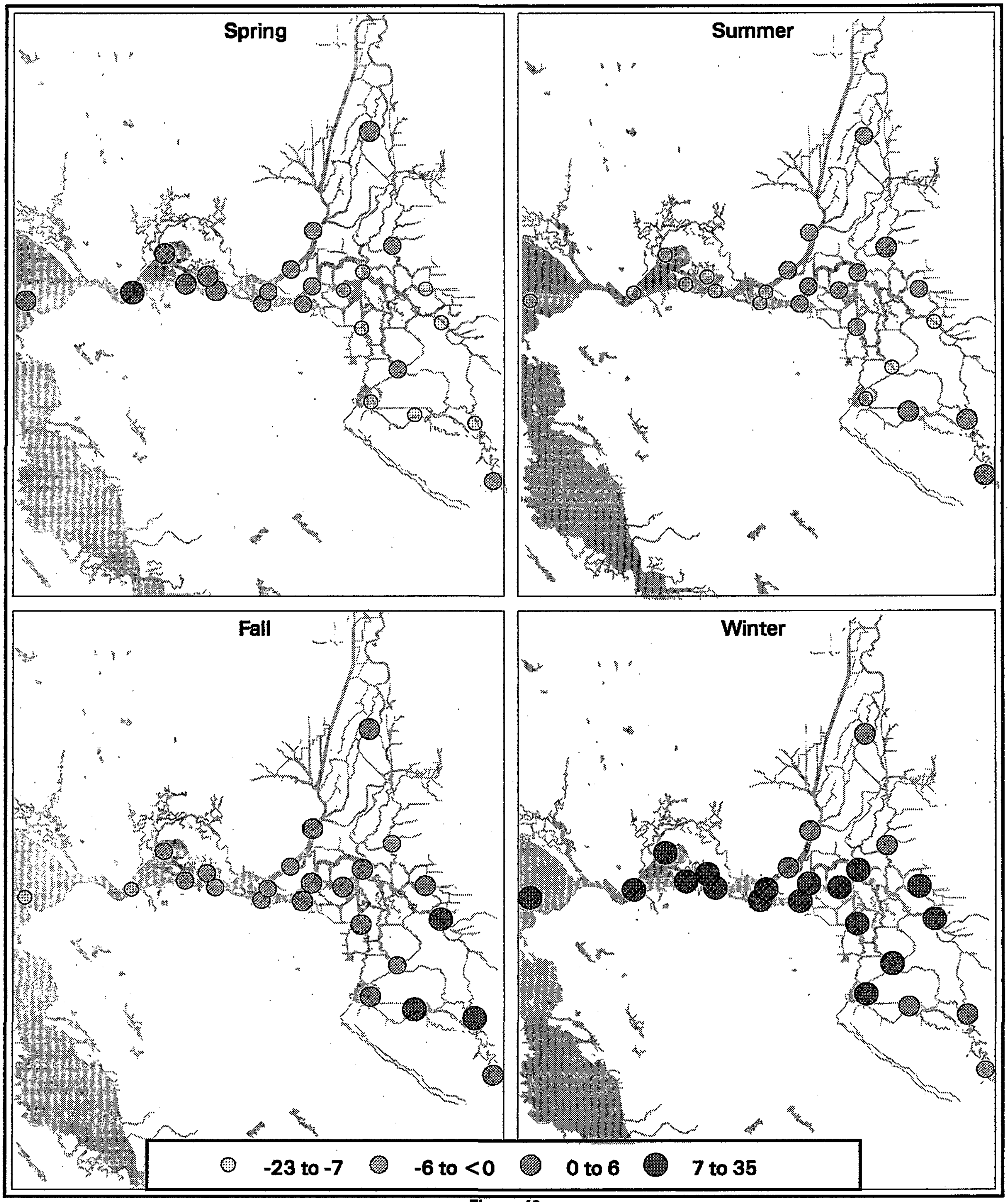

Figure 42
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR SILICA CONCENTRATION, BY SEASON
Gaps between data ranges in legend indicate no data.

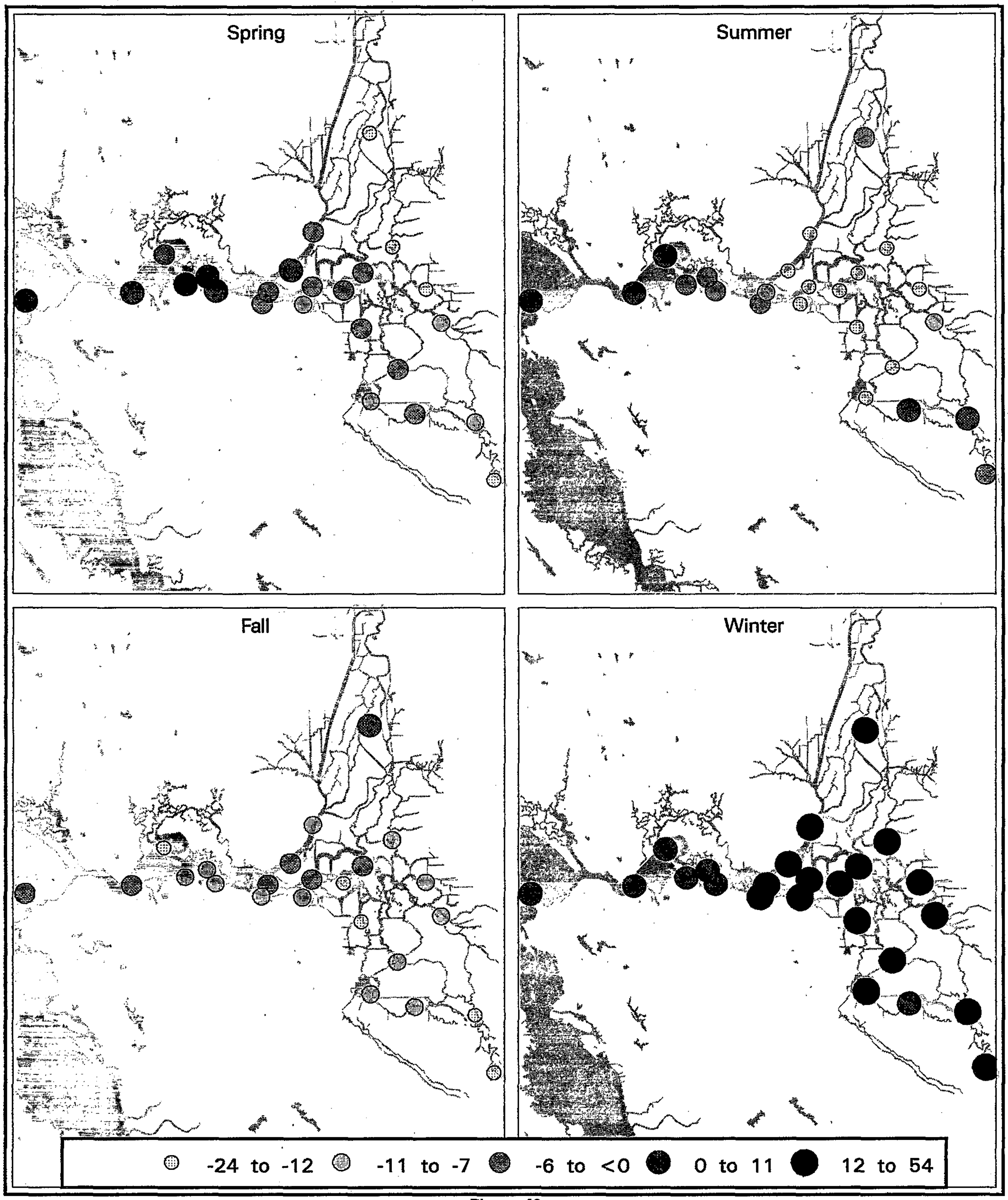

Figure 43
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR TOTAL ORGANIC NITROGEN, BY SEASON
Gaps between data ranges in legend indicate no data.

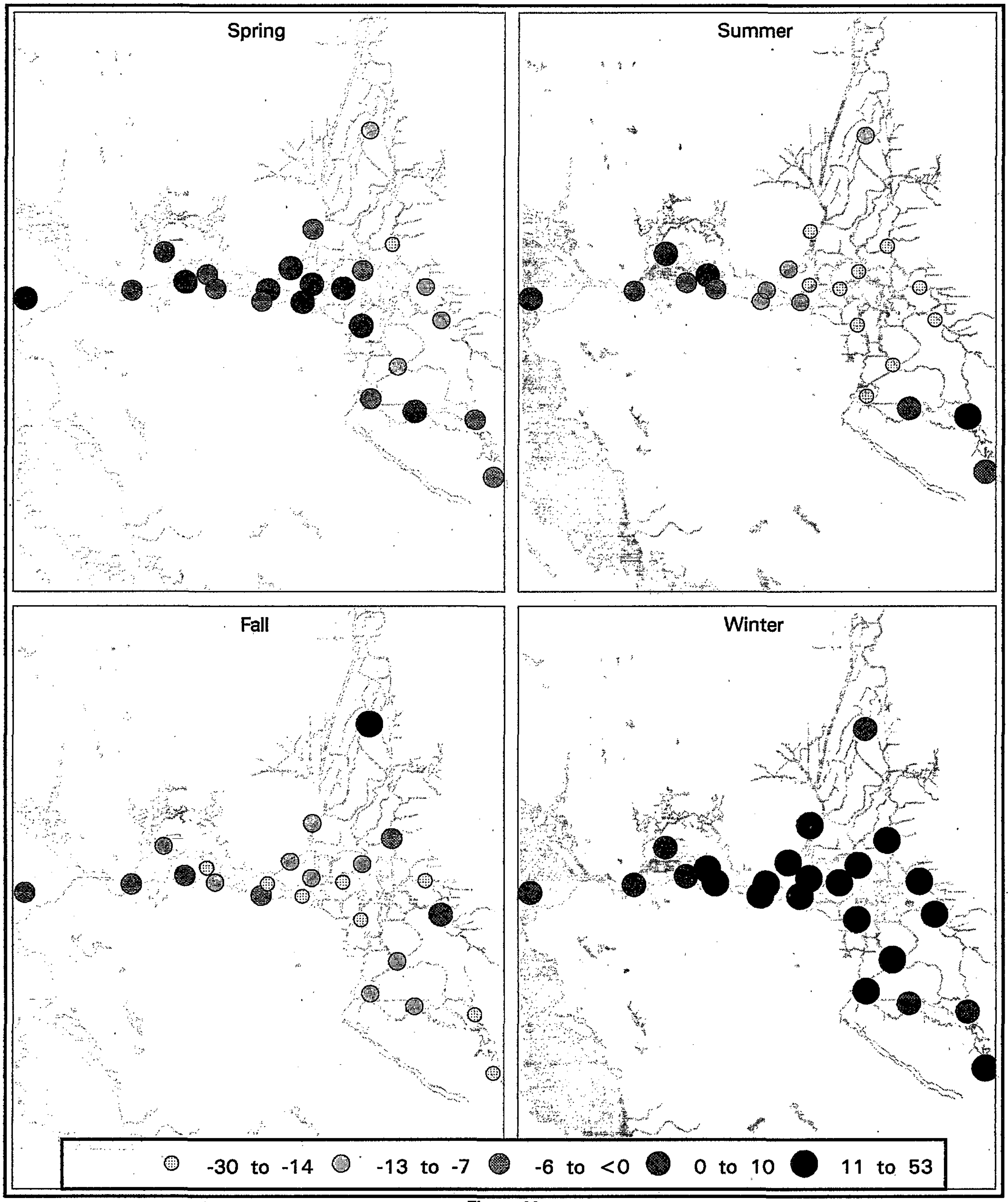

Figure 44
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR AMMONIA PLUS ORGANIC NITROGEN, BY SEASON
Gaps between data ranges in legend indicate no data.

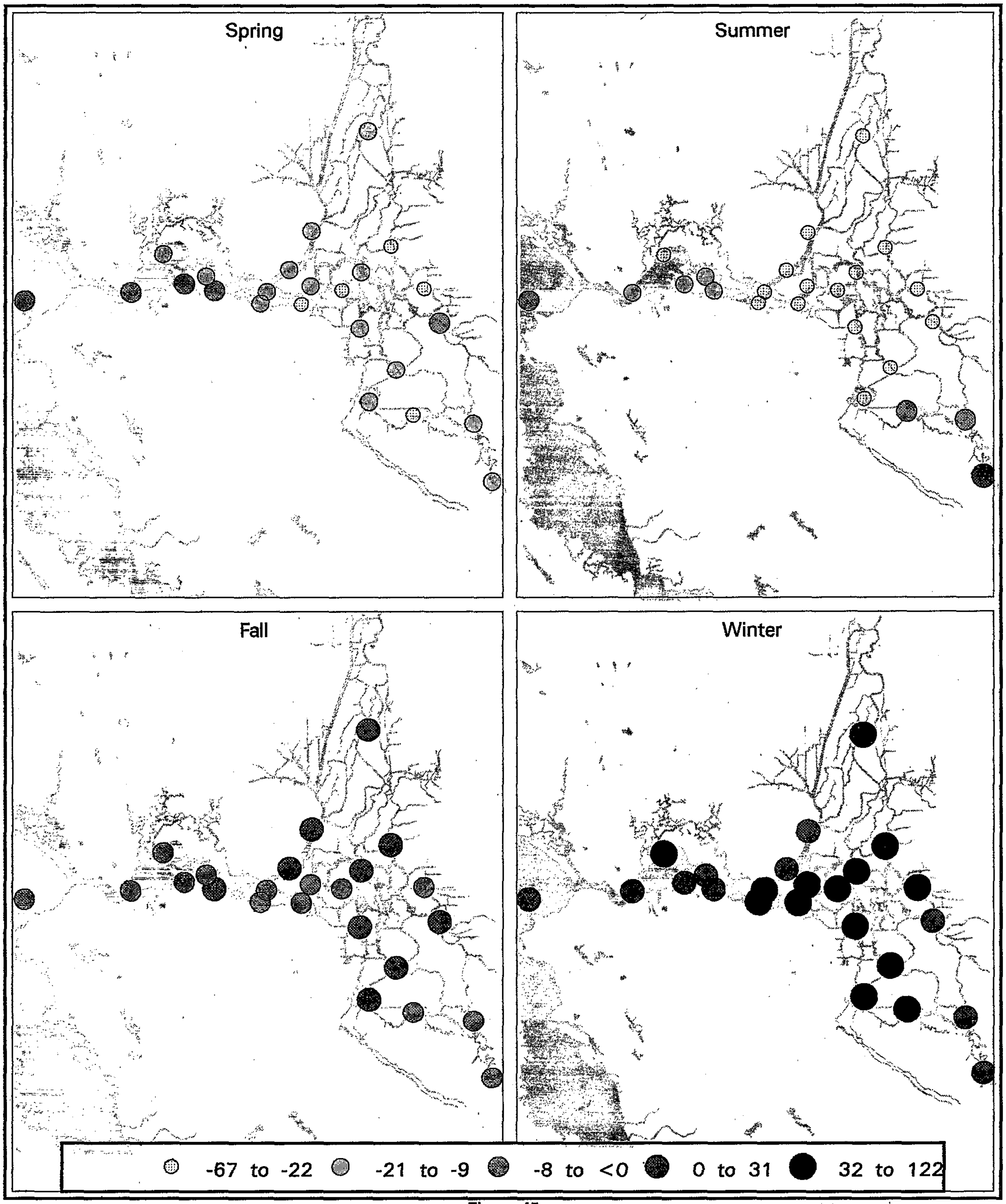

Figure 45

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR NITRATE CONCENTRATION, BY SEASON

Gaps between data ranges in legend indicate no data.

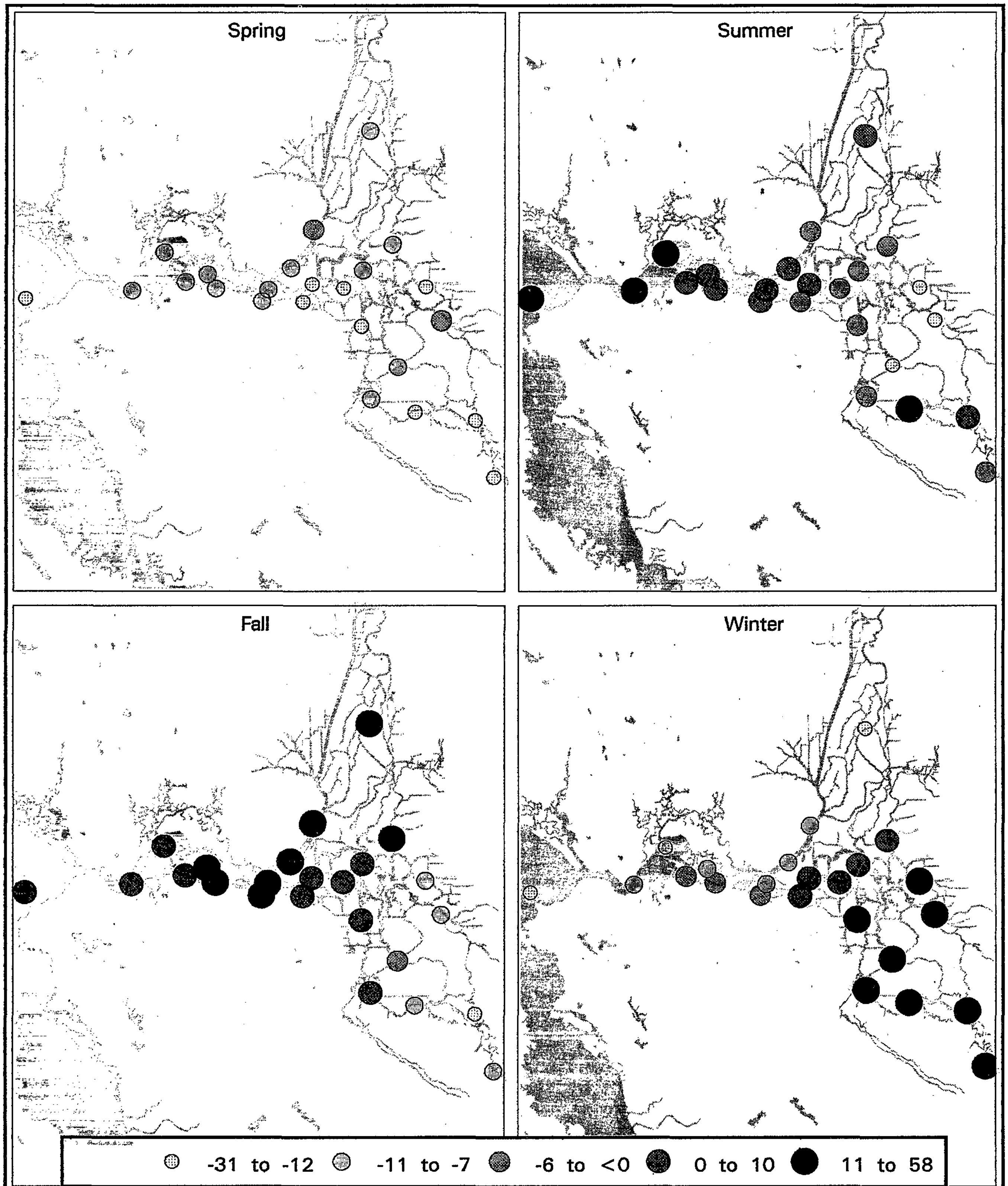

Figure 46
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR ORTHO-PHOSPHATE CONCENTRATION, BY SEASON
Gaps between data ranges in legend indicate no data.

Trace Metals

Small amounts of trace metals that occur naturally are important for physical, chemical, and biological processes in the upper estuary. Concentrations that become elevated as a result of agricultural and industrial activity may adversely affect water quality and biological resources. Trace metals are measured as part of the Decision 1485 monitoring program to observe long-term changes and detect potentially toxic substances.

Concentrations of nine trace metals were measured in May and September at 11 stations in the Delta and Suisun Bay from 1975 to 1993 (Figure 47). Water samples were collected in nonmetallic samplers. Dissolved trace metal concentrations were determined from water samples filtered through 0.45-micron pore size membrane filters directly into 16-ounce acid-washed polyethylene storage bottles. Total trace metals were determined from whole water samples. As a preservative, 1 mL of concentrated nitric acid was added to each bottle, and bottles were stored in a cool, dark location.

The lowest concentrations in this report are minimum reporting limits, or the smallest concentration that can be accurately measured by the particular laboratory method. Between 1975 and 1985, the minimum reporting limit was 10 µg/L for all trace metals. In 1986, the minimum reporting limit was changed to 5 µg/L for all trace metals except mercury and arsenic, which were 1 µg/L. In September 1993, detection limits changed to 10 µg/L for zinc and 50 µg/L for iron. Specific methods for trace metal analysis are listed in Table 3.

Potentially harmful concentrations of trace metals in the estuary were determined from standards and guidelines developed for the health of humans and aquatic biota (Table 4). Unhealthy concentrations for humans were determined from drinking water standards established by the Department of Health Services. Primary drinking water standards are based on National Primary Drinking Water Regulations (40CFR, Part 141) and are the maximum permissible contaminant levels to protect human health when the water is used continuously for drinking or cooking (Department of Health Services 1990, 1993). Secondary drinking water standards are based on the Secondary Drinking Water Regulations (40CFR, Part 143) and are the maximum permissible contaminant levels to assure that taste, odor, or appearance of drinking water are not adversely affected. Secondary drinking water standards are not based on health concerns. Potentially acute and chronic toxicity levels for aquatic biota are determined from water quality guidelines developed by the U.S. Environmental Protection Agency (U. S. Environmental Protection Agency 1986; 1991).

Figure 48 shows concentrations of trace metals throughout the upper estuary. All trace metals reached detection limits sometime between 1975 and 1993. Total and dissolved iron and manganese and total chromium, copper, and zinc were consistently higher than detection limits throughout the period of record at all stations. Total and dissolved arsenic became consistently above detection limits when the detection limits were lowered in 1986. Dissolved chromium, zinc, lead, and copper and total lead and mercury consistently exceeded detection limits at only some stations, and dissolved and total cadmium and dissolved chromium, lead, and total mercury rarely exceeded detection limits.

Total and dissolved concentrations of most trace metals decreased after 1985, but changes among stations were variable and did not demonstrate a trend. In contrast, little change occurred after 1985 at many stations for total iron and total and dissolved manganese and zinc.

Trace metals frequently exceeded guidelines for marine and freshwater toxicity and drinking water standards. Trace metals exceeded guidelines for freshwater acute and chronic toxicity 34 times. Most of these were after 1983 at station D6 for copper and before 1981 throughout the upper estuary for lead. Marine acute and chronic toxicity guidelines were exceeded 181 times, 160 of which were for dissolved copper. These exceedences occurred throughout the upper estuary in both May and September and were more common in the lower Sacramento, lower San Joaquin, and Honker Bay. The remaining exceedences were for lead at many stations before 1981. Drinking water standards were exceeded only 11 times for iron in the western region and downstream and for manganese in the San Joaquin River. Cadmium and zinc rarely exceeded toxicity or drinking water guidelines, and chromium never did.

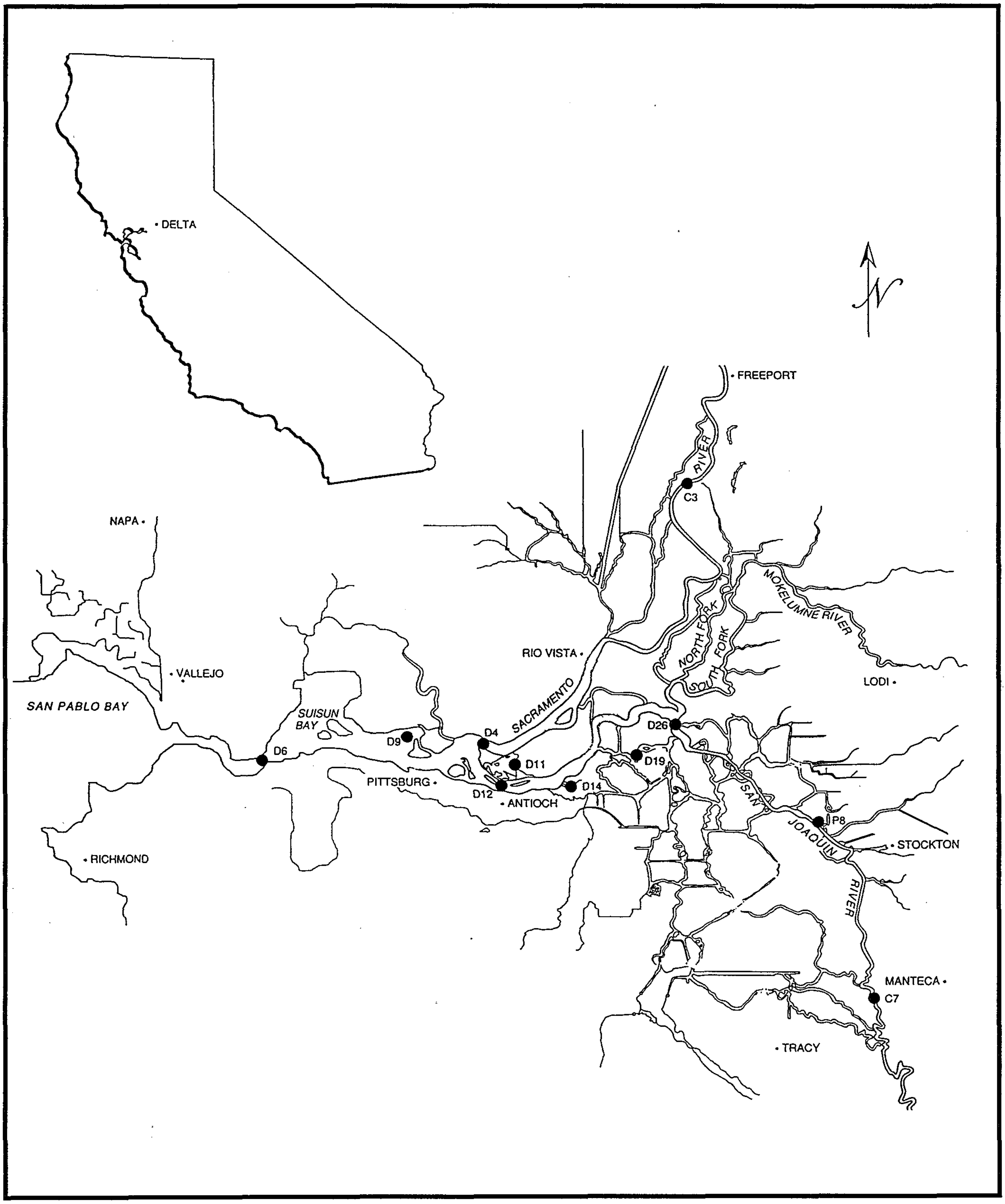

Figure 47
SAMPLING STATIONS FOR TRACE METALS
Sacramento-San Joaquin Delta

Table 3
TRACE METAL SAMPLING METHODS

Substance	Method	Method #	Dates	Reference
Arsenic	Atomic absorption, hydride method	206.3	6/86-9/93	EPA 1983
	Color (S.D.)-SM 3500 AsC		5/75-9/86	ALPHA 1989
Cadium	Atomic Absorption, furnace, Zeeman method	213.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Chromium	Atomic Absorption, furnace, Zeeman method	218.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Copper	Atomic Absorption, furnace, Zeeman method	220.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Iron	Atomic Absorption, furnace, Zeeman method	236.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Lead	Atomic Absorption, furnace, Zeeman method	239.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Manganese	Atomic Absorption, furnace, Zeeman method	243.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989
Zinc	Atomic Absorption, furnace, Zeeman method	289.2	1/87-9/93	EPA 1983
	AA-SM 3111-C		5/75-9/86	ALPHA 1989

Table 4
DISSOLVED TOXICITY LEVELS AND DRINKING WATER STANDARDS FOR TRACE METALS

Standard Type	As	Cd	Cr	Cu	Fe	Pb	Mn	Zn	Hg
Freshwater									
Acute Toxicity	360	3.9	16	18	1000	82	—	120	2.4
Chronic Toxicity	190	1.1	11	12	—	3.2	—	110	0.012
Marine									
Acute Toxicity	69	43	1100	2.9	—	220	—	95	2.1
Chronic Toxicity	36	9.3	50	2.9	—	8.5	—	86	0.025
Drinking Water Standards									
Primary	50	10	50	—	—	50	—	—	2
Secondary	—	—	—	1000	300	—	50	5000	—

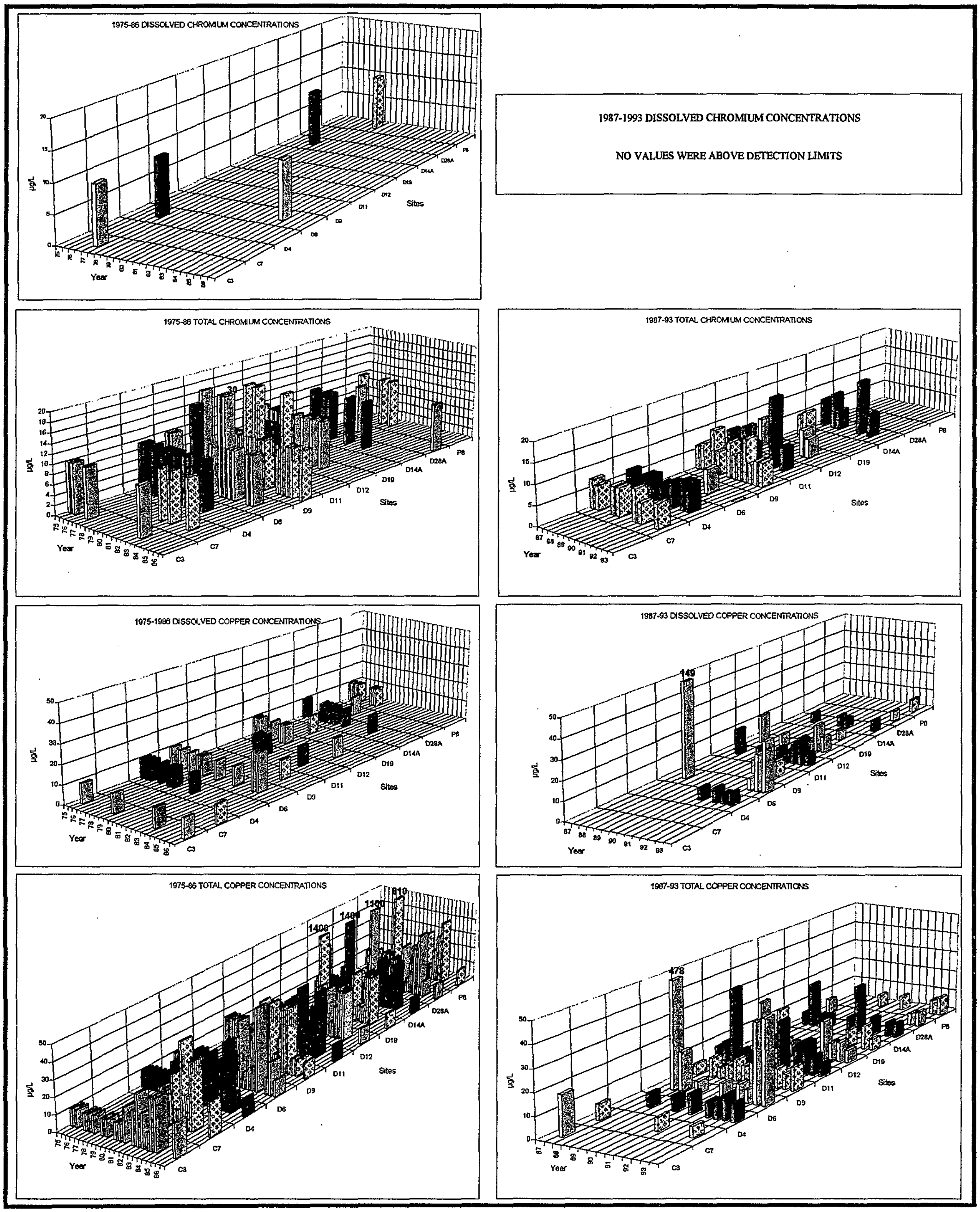

Figure 48
CONCENTRATIONS OF TRACE METALS, 1975-1993

C — 0 3 6 5 4 5

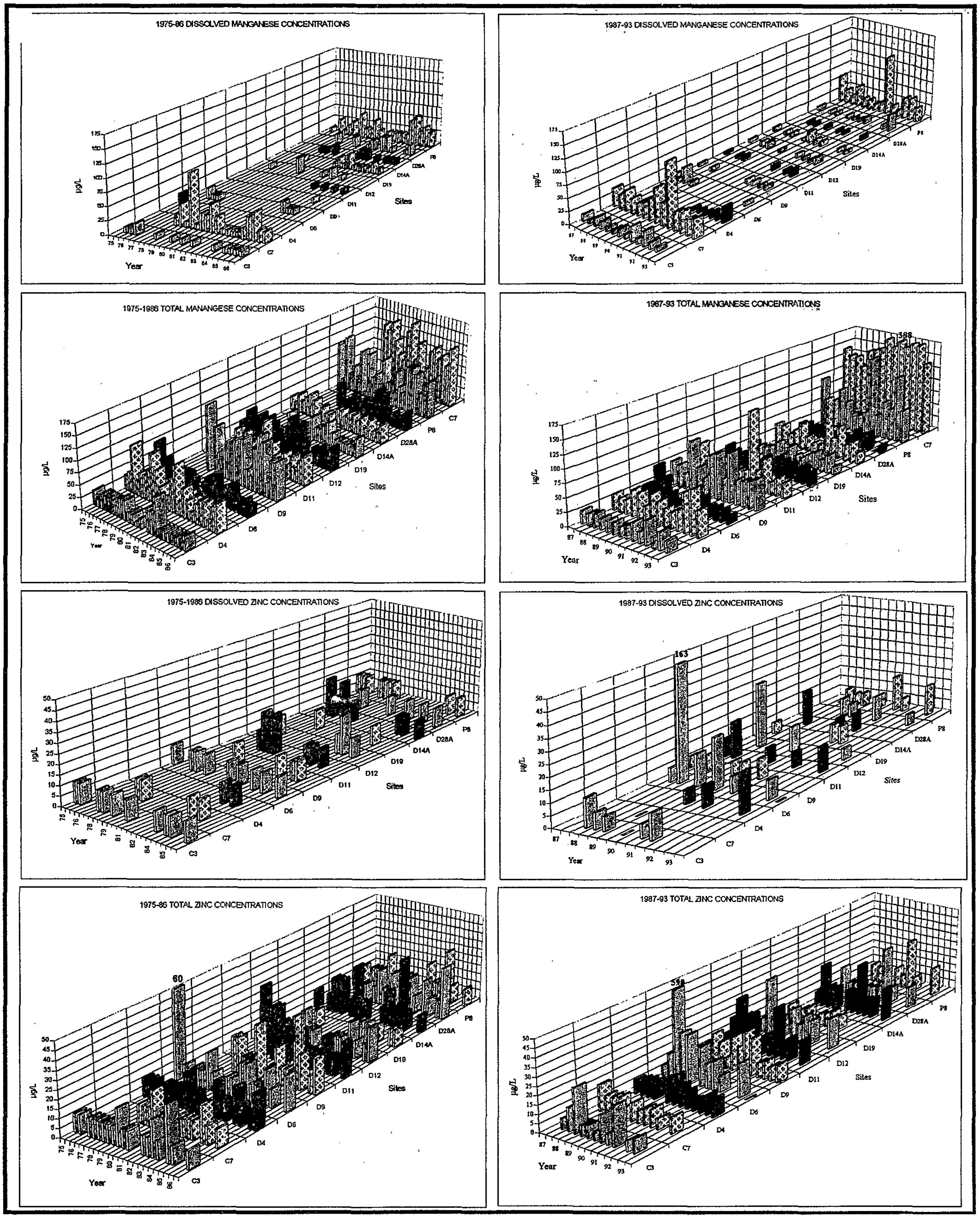

Figure 48(continued)
CONCENTRATIONS OF TRACE METALS, 1975-1993

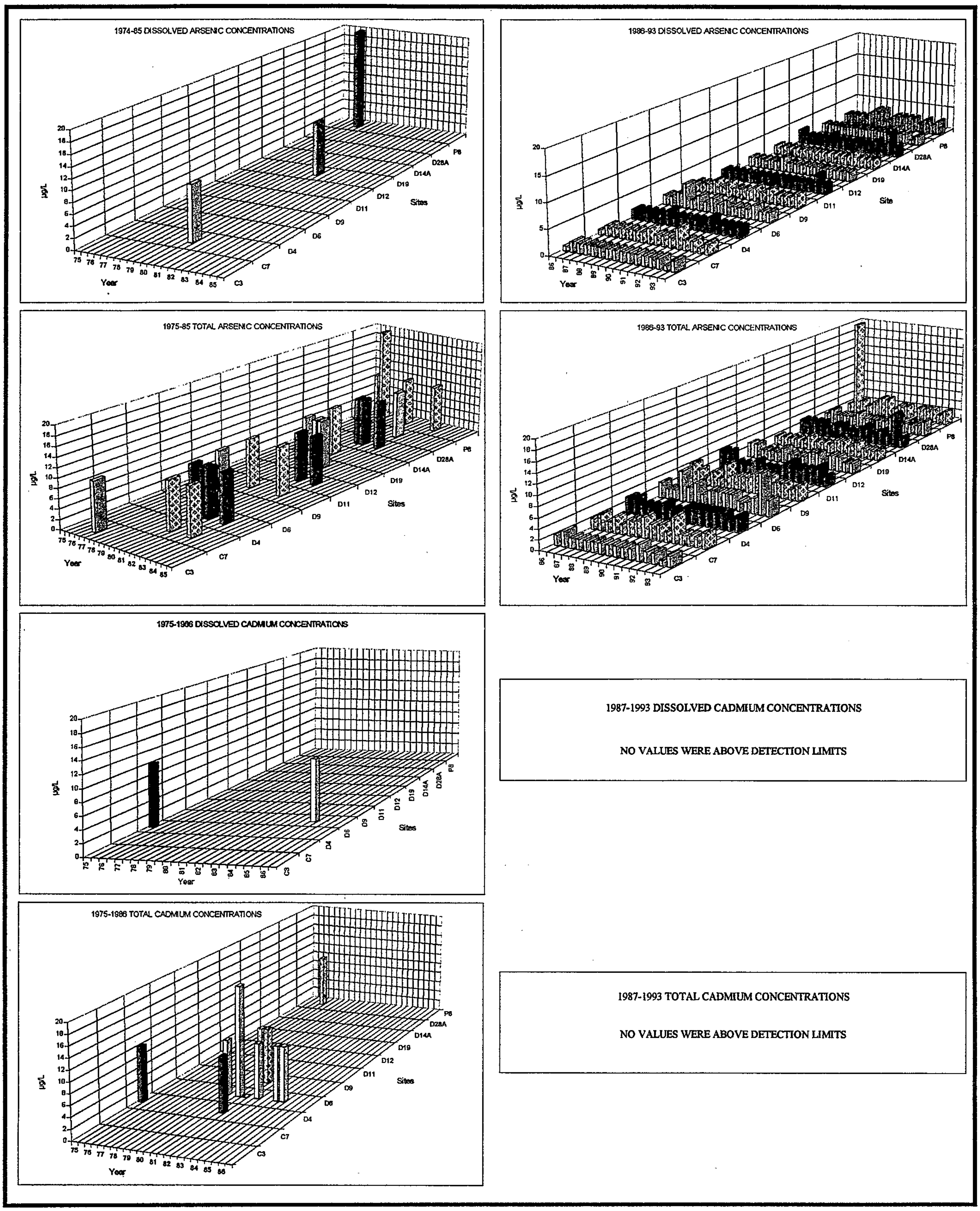

Figure 48 (continued)
CONCENTRATIONS OF TRACE METALS, 1975-1993

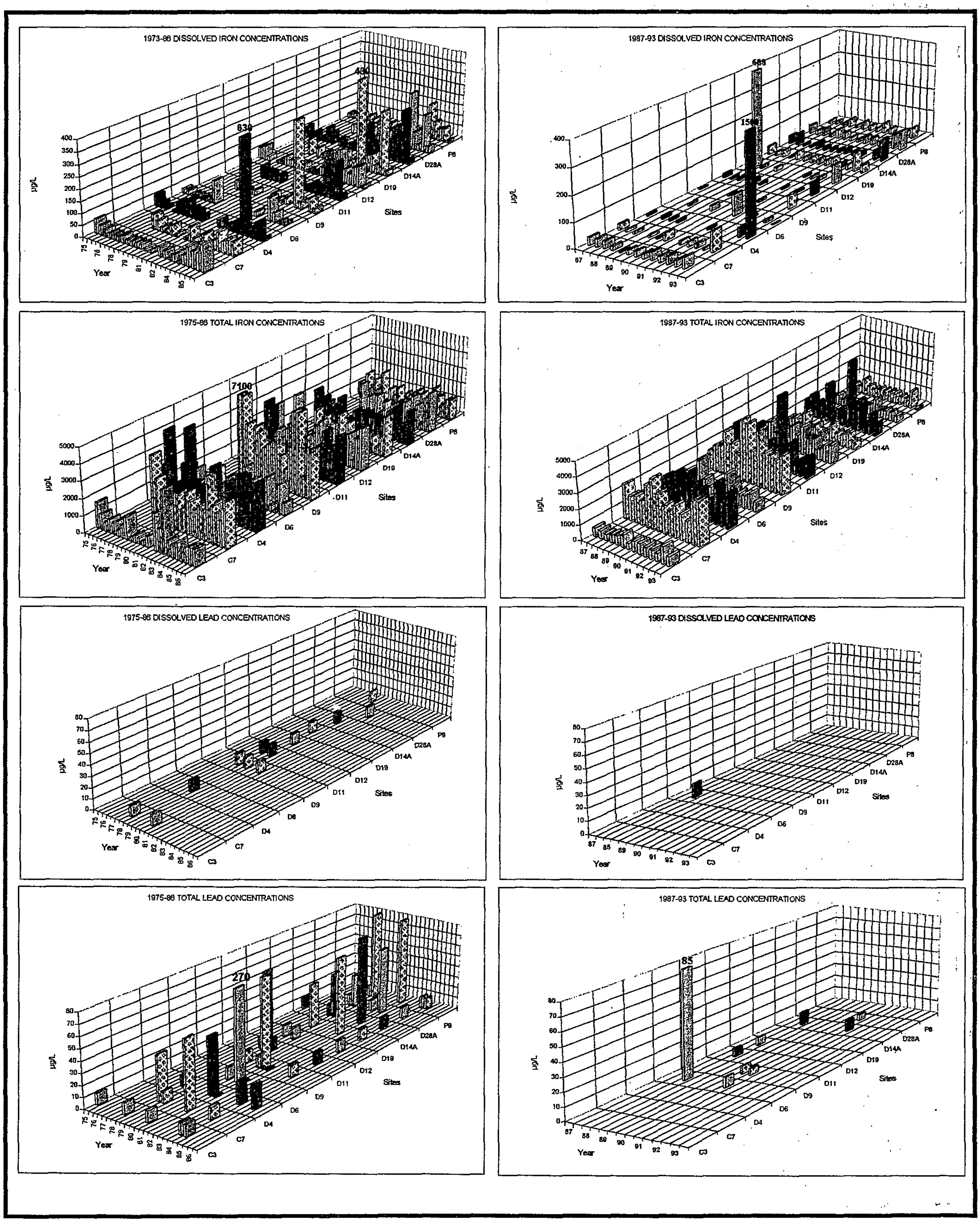

Figure 48 (continued)
CONCENTRATIONS OF TRACE METALS, 1975-1993

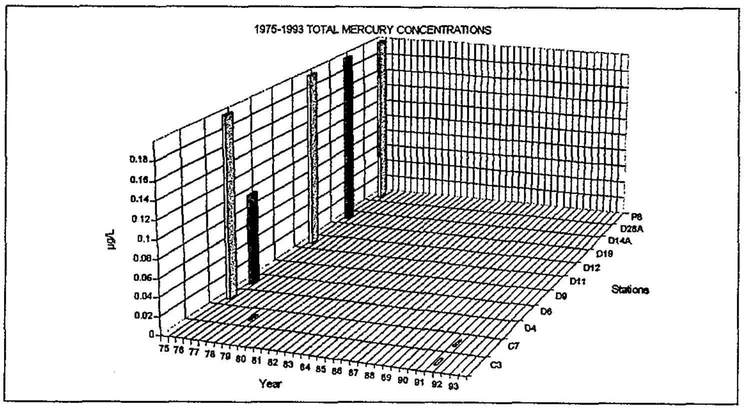

Figure 48 (continued)
CONCENTRATIONS OF TRACE METALS, 1975-1993

Organic Pesticides and Polychlorinated Biphenyls

Organic pesticides and polychlorinated biphenyls (commonly called PCBs) were measured semiannually from 1975 to 1993 at the same 11 stations as for trace metals (Figure 47). The number of organic pesticides monitored increased from 8 to 25 in 1987 and to 39 in 1988 because of increased analytical capability.

Water samples were collected in new, 2-quart glass bottles pre-cleaned with pesticide-grade solvents and fitted with Teflon-lined caps. Bottles were immersed directly into the water for collection. Water samples were stored in the dark under refrigeration until analysis using the liquid extraction method and gas chromatography with dual electrolytic conductivity detectors (EPA Method 608 [1982]). Concentrations for each pesticide are determined above minimum detection limits. The concentration of the pesticide is unknown below the minimum detection limit. Minimum reporting limits are listed in Table 5. When an unidentified chlorinated hydrocarbon was measured, the amount of the unidentified chlorinated hydrocarbon was quantified in equivalents of DDT.

All of the 8 chlorinated organic pesticides measured between 1975 and 1986 were above the minimum reporting limit at some time (Table 6). Atrazine plus simazine exceeded minimum reporting limits most frequently and at nearly every station. Unknown substances were also measured for most stations. In contrast, BHC, CHC, Dacthal, PCNB, Methoxychlor, and Difolatan were rarely above reporting limits.

Concentrations of chlorinated organic pesticides were rarely above the minimum reporting limit in 1987-1993 and differed from those measured before 1987 (Table 7). Diuron was detected on the San Joaquin River at Buckley Cove (P8) in 1989. Also in 1989, endosulfan sulfate was detected at Mossdale. In addition, unidentified compounds with concentrations between 0.02 and 0.05 µg/L were detected during 1989. Two of these compounds were detected at Buckley Cove and one compound each was detected at Franks Track (D19), Antioch Ship Channel (D12), and Sherman Lake (D11).

Pesticides detected in 1987-1993 were consistent with patterns of pesticide use in the estuary. Diuron and endosulfan sulfate are pesticides applied to many crops grown in the Delta or drainages to the Delta. Diuron is used to kill broadleaf weeds in crops such as alfalfa, cotton, grapes, barley, and wheat. Concentrations measured were below those known to affect fish. Chronic toxicity tests in flow-through systems indicate rainbow and cutthroat trout can survive indefinitely at concentrations of 140 and 500 µg/L, respectively (Johnson and Finley 1980).

Endosulfan sulfate is an insecticide applied to various vegetable, fruit, nut, and grain crops. Concentrations of endosulfan sulfate above the minimum reporting limits were below the reported freshwater acute and chronic toxicity levels but exceeded the marine acute and chronic toxicity levels (USEPA 1986).

C-036549

Table 5
TOXICITY LEVELS, DRINKING WATER STANDARDS, AND MINIMUM REPORTING LIMITS FOR SYNTHETIC ORGANIC COMPOUNDS

Compound	Toxicity Levels[1,2] (µg/L) Freshwater Acute	Freshwater Chronic	Marine Acute	Marine Chronic	Drinking Water Standards[3] (µg/L) Primary	Secondary	Minimum Reporting Limit (µg/L)
Alachlor							0.05
Aldrin	3.0		1.3				0.01
Atrazine					3.0		0.02
BHC, Alpha							0.01
BHC, Beta							0.01
BHC, Delta							0.01
BHC, Gamma	2.0	0.08	0.16		4.0		0.01
Captan							0.02
Chlordane	2.4	0.0043	0.09	0.004	0.1		0.05
Chlorothalonil							0.01
Chlorpropham							0.02
Chlorpyrifos							0.01
DCPA							0.01
DDD							0.01
DDE	1.05		14.0				0.01
DDT	1.1	0.001	0.13	0.001			0.01
Dichloran							0.01
Dicofol							0.01
Dieldrin	2.5	0.0019	0.71	0.0019			0.01
Diuron							0.25
Endosulfan Sulfate	0.22	0.056	0.034	0.0087			0.02
Endosulfan	0.22	0.056	0.034	0.0087			0.01
Endosulfan I	0.22	0.056	0.034	0.0087			0.01
Endosulfan II	0.22	0.056	0.034	0.0087			0.01
Endrin	0.18	0.0023	0.037	0.0023	0.2		0.01
Endrin Aldehyde							0.01
Heptachlor	0.52	0.0038	0.053	0.0036	0.01		0.01
Heptachlor Epoxide	0.52	0.0038	0.053	0.0036	0.01		0.01
Methoxychlor	0.012	0.03		0.03	100.0		0.05
PCB	2.0	0.014	10.0	0.03			0.1
PCB-1016	2.0	0.014	10.0	0.03			0.1
PCB-1221	2.0	0.014	10.0	0.03			0.1
PCB-1232	2.0	0.014	10.0	0.03			0.1
PCB-1242	2.0	0.014	10.0	0.03			0.1
PCB-1248	2.0	0.014	10.0	0.03			0.1
PCB-1254	2.0	0.014	10.0	0.03			0.1
PCB-1260	2.0	0.014	10.0	0.03			0.1
PCNB							0.01
Simazine					10.0		0.02
Thiobencarb					70.0	1.0	0.02
Toxaphene	0.73	0.0002	0.21	0.0002	5.0		1.0

1 EPA. "Water Quality Criteria Summary". Pages 6-10 in: *Quality Criteria for Water, 1986.* Office of Water Regulations and Standards. EPA 440/5-86-001. May 1, 1986.

2 EPA-Federal Register, 56(223), November 19, 1991. Proposed Rules.

3 Department of Health Services. "Federal and State Drinking Water Standards". Pages 3-10 in: *Compilation of Federal and State Drinking Water Standards and Criteria.* Technical Document 3. July 1993.

PESTICIDE CODES

BHC	Benzene hexachloride	CHC	Unknown
DCPA	Dacthal	DDD	1,1-dichloro-2,2-bis(4-chlorophenyl)ethane
DDE	Dichlorodiphenyldichloroethylene	DDT	Dichlorodiphenyltrichloroethane
PCB	Polychlorinated biphenyl	PCNB	Pentachloronitrobenzene

C-036550

Table 6
CHLORINATED ORGANIC PESTICIDE CONCENTRATIONS EXCEEDING MINIMUM REPORTING LIMITS, 1975 TO 1986
(μg/L)

Site	Date	BHC	CHC	Atrazine and Simazine	Dacthal	Unknown	PCNB	Methoxychlor	Difolatan
C10	05-09-75	-	-	-	-	0.05	-	-	-
	09-14-76	-	-	-	-	0.11	-	-	-
	05-06-76	0.02	-	-	-	-	-	-	-
	05-24-77	-	-	-	0.48	-	-	-	-
C3	01-20-76	-	-	-	-	0.02	-	-	-
	05-06-76	-	-	-	-	0.02	-	-	-
	05-19-80	-	-	-	-	0.01	-	-	-
	05-02-83	-	-	0.06	-	-	-	-	-
C7	09-11-78	-	-	0.02	-	-	-	0.03	-
	05-21-79	-	-	0.08	-	-	-	-	-
C9	01-24-77	-	-	0.05	-	-	-	-	-
D11	06-15-78	-	-	0.05	-	-	-	-	-
	07-15-82	-	-	0.06	-	-	-	-	-
	05-02-85	-	0.12	0.1	-	-	-	-	0.02
	05-22-86	-	-	0.24	-	-	-	-	-
D12	05-08-75	-	-	-	-	0.015	-	-	-
	09-03-75	-	-	-	-	0.04	-	-	-
	06-15-78	-	-	0.06	-	-	-	-	-
	05-24-79	-	-	-	-	0.44	-	-	-
	07-15-82	-	-	0.12	-	-	-	-	-
	05-05-83	-	-	0.11	-	-	-	-	-
	05-02-85	-	-	0.12	-	-	-	-	0.09
	05-22-86	-	-	0.2	-	-	-	-	-
D14A	05-27-77	-	-	0.03	-	-	-	-	-
	05-20-82	-	-	0.05	-	-	-	-	-
	07-15-82	-	-	0.07	-	-	-	-	-
	09-29-83	-	-	-	-	0.01	-	-	-
	05-02-85	-	-	0.07	-	-	-	-	0.02
	09-06-85	-	-	-	-	0.03	-	-	-
	05-22-86	-	-	0.15	-	-	-	-	-
D19	06-14-78	-	-	0.05	-	-	-	-	-
	09-12-79	-	-	-	-	0.02	-	-	-
	07-16-82	-	-	0.1	-	0.02	-	-	-
	05-01-85	-	-	0.06	-	-	-	-	-
	05-21-86	-	-	0.05	-	-	-	-	-
D22	09-15-76	-	-	-	-	0.01	-	-	-
D28A	06-13-78	-	-	0.08	-	-	-	-	-
	05-17-82	-	-	-	-	0.01	-	-	-
	05-13-85	-	-	0.03	-	-	-	-	-
D4	06-14-78	-	-	0.04	-	-	-	-	-
	05-23-79	-	-	0.03	-	-	-	-	-
	07-16-82	-	-	0.09	-	0.03	-	-	-
	05-04-83	-	-	0.15	-	-	-	-	-
	05-01-85	-	-	0.07	-	-	-	-	-
	05-21-86	-	-	0.06	-	-	-	-	-
D6	06-15-78	-	-	0.05	-	-	-	-	-
	07-15-82	-	-	-	-	0.22	-	-	-
	05-05-83	-	-	0.14	-	-	-	-	-
	05-02-85	-	-	0.15	-	-	-	-	0.02
	05-22-86	-	-	0.12	-	-	-	-	-
D9	06-14-78	-	-	0.04	-	-	-	-	-
	05-23-79	-	-	-	-	0.09	-	-	-
	07-16-82	-	-	0.09	-	0.02	-	-	-
	05-04-83	-	-	0.09	-	-	-	-	-
	05-01-85	-	-	0.15	-	-	-	-	-
	09-05-85	-	-	0.02	-	-	-	-	-
	05-21-86	-	-	0.03	-	-	-	-	-
P10	05-06-76	-	-	-	-	0.01	-	-	-
	01-25-77	-	-	0.05	-	-	-	-	-
P2	01-21-75	-	-	0.015	-	0.025	-	-	-
	05-07-76	-	-	-	-	-	0.035	-	-
	05-07-76	-	-	-	-	0.015	-	-	-
P8	02-03-75	-	-	-	-	0.03	-	-	-
	05-01-75	-	-	-	-	0.045	-	-	-
	09-11-75	-	-	-	-	0.075	-	-	-
	01-22-76	-	-	0.01	0.01	-	-	-	-
	01-25-77	-	-	0.09	-	-	-	-	-
	09-02-77	-	-	0.07	-	-	-	-	-
	09-12-78	-	-	0.05	-	-	-	-	-
	05-22-79	-	-	-	-	0.01	-	-	-
	09-11-79	-	-	0.04	-	-	-	-	-
	09-03-80	-	-	0.04	-	0.03	-	-	-
	05-12-81	-	-	0.1	-	-	-	-	-
	05-17-82	-	-	0.04	-	-	-	-	-
	09-07-82	-	-	-	-	0.12	-	-	-
	09-27-83	-	-	-	-	0.02	-	-	-
	05-16-84	-	-	0.05	-	0.01	-	-	-
	09-14-84	-	-	0.04	-	-	-	-	-
	05-13-85	-	-	0.2	0.03	-	-	-	-
	09-04-85	-	-	0.06	-	-	-	-	-

C —0 3 6 5 5 1

C-036551

Table 7
CHLORINATED ORGANIC PESTICIDE CONCENTRATIONS EXCEEDING MINIMUM REPORTING LIMIT, 1987-1993
(μg/L)

Site	Date	DDE	Dieldrin	Endrin	Diuron	Endosulfan Sulfate	PCB-1248	PCB-1254	PCB-1260
D19	9-6-88	0.05	0.05	0.05					
C7	9-5-89					0.040			
	5-1-90						1.00	1.00	1.00
P8	5-1-89				0.50				

The Bryte Laboratory minimum reporting limits for atrazine, BHC-Gamma, chlordane, DDT, dieldrin, endosulfan, endrin, heptachlor, methoxychlor, PCB, simazine, thiobencarb, and toxaphene exceeded the chronic freshwater and marine organism toxicity levels adopted in November 1991 by the Environmental Protection Agency as well as primary or secondary drinking water standards (Table 5). Therefore, non-detection of these pesticides by the Compliance Monitoring Program does not necessarily mean levels of those pesticide are below the EPA toxicity levels or drinking water standards.

C-036552

BIOLOGICAL VARIABLES

Biological variables are also sampled as a requirement of Decision 1485. Phytoplankton, benthos, and higher aquatic plant density and composition reflect changing conditions and serve as environmental indicators. Changes in the biota are used along with environmental variables to assess impacts of State Water Project and Central Valley Project operations on the estuary. This chapter summarizes some of the changes in phytoplankton, benthos, and higher aquatic plants in 1970-1993.

Chlorophyll *a* Concentration

Between 1970 and 1993, chlorophyll *a* concentration (used as an estimate of phytoplankton biomass) was measured monthly or semimonthly for 26 stations (Figure 49). Water samples were collected at 1 meter within 1 hour of high slack tide using a Van Dorn sampler or a submersible pump. The time of each sample was recorded to the nearest 5 minutes using Pacific Standard Time. Total water depth was measured to the nearest foot by an electronic depth sounding instrument or a weighted measuring tape.

Water samples (200-mL) were filtered through Gelman Type AE glass fiber filters. Filters were neutralized with a saturated solution of magnesium carbonate, folded, placed in manila envelopes, and frozen until analysis.

Chlorophyll *a* was extracted from filters using a Strickland and Parsons (1968) procedure. In 1979, the method was modified to sonification of the filter for 15 minutes and extraction for 2 hours in the dark. Chlorophyll *a* concentrations were calculated from absorbence at 663 nm and 665 nm, with a 750-nm filter background correction on a Perkin-Elmer 552 scanning spectrophotometer. Absorbence was measured before and after acidification with one drop of 1N hydrochloric acid.

Chlorophyll *a* and phaeophytin concentrations were calculated as:

$$\text{chlorophyll a, } \mu g/L = \frac{(24.7)(E_{663} - E_{665}\text{acid})(\text{mL of extract})(1000)}{(\text{mL filtered})(\text{cell path length, cm})}$$

$$\text{phaeophytin, } \mu g/L = \frac{(24.7)[(1.75)(E_{665}\text{acid}) - E_{663}](\text{mL of extract})(1000)}{(\text{mL filtered})(\text{cell path length, cm})}$$

where: E_{663} = absorbence value of acetone extract at 663 nm

E_{665}acid = absorbence value of acidified acetone extract at 665 nm.

Percent chlorophyll *a* concentration was calculated as:

$$\text{percent chlorophyll a} = \frac{(100)(\text{chlorophyll a, } \mu g/L)}{\text{chlorophyll a, } \mu g/L + \text{phaeophytin, } \mu g/L}$$

Data were indexed to water year types. A water year describes precipitation for a period beginning October 1 and ending September 30 of the following year. Water years since 1906 have been classified as wet, above normal, below normal, dry, or critical based on the Sacramento River Index (Figure 1, Chapter 2). Because "above normal" or "below normal" years occurred only three times in 1970-1993, data for this study were grouped into wet, normal, dry, and critical years.

Data were also indexed to season: fall (October-December), winter (January-March), spring (April-June), and summer (July-September).

Water year type and seasonal data were calculated as percent deviations from the long-term average. Deviations were calculated as the difference between the monthly average and the long-term average, divided by the long-term average times 100 for each variable at each station. Positive deviations indicate values are higher than average, and negative deviations indicate values are lower than average.

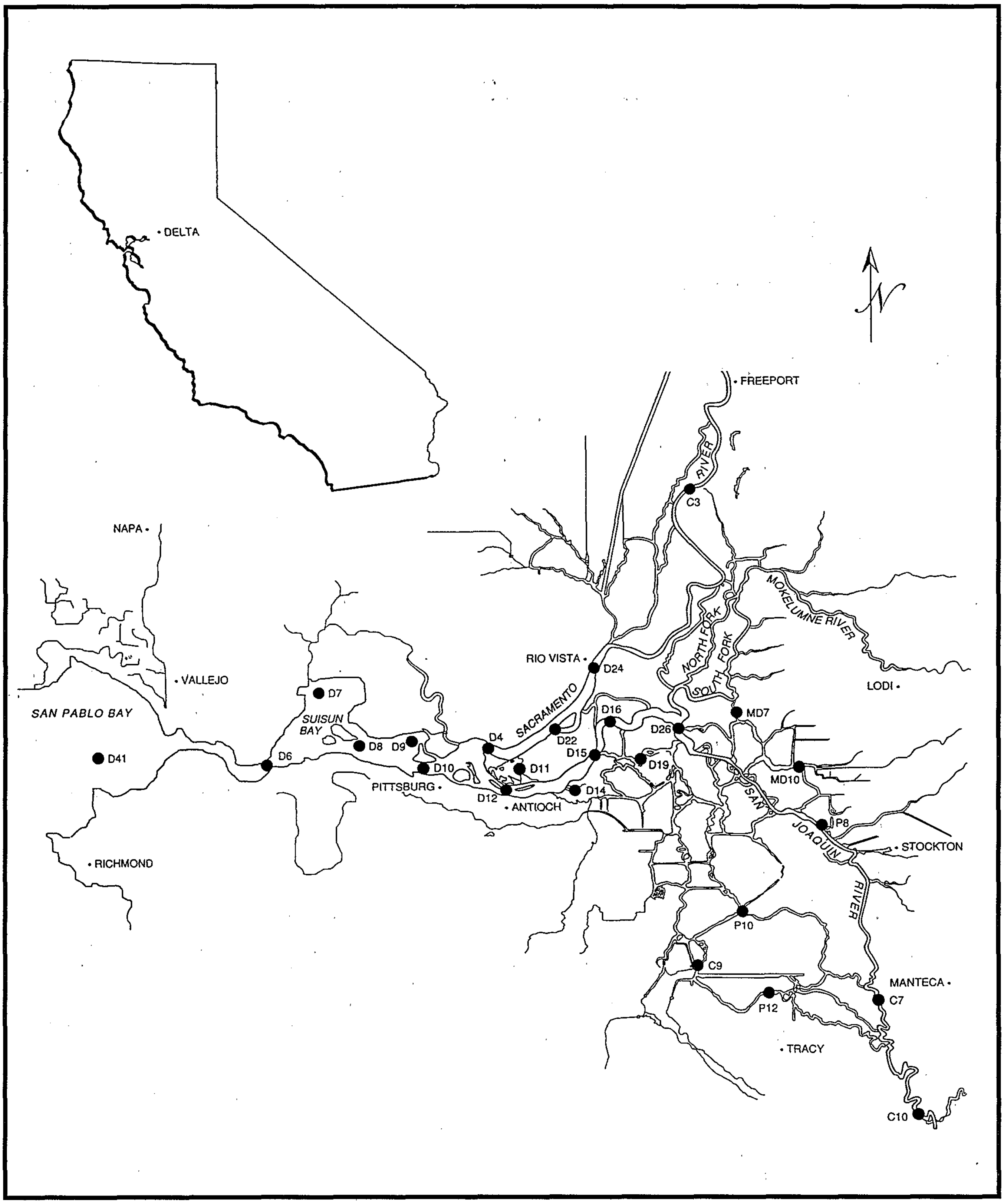

Figure 49
SAMPLING LOCATIONS FOR CHLOROPHYLL *a* CONCENTRATION AND PHYSICAL AND CHEMICAL VARIABLES
Sacramento-San Joaquin Delta

Stations were grouped into regions based on individual and combined hierarchical cluster analysis of monthly data for 14 physical and chemical variables and chlorophyll *a* concentrations (Table 1, Chapter 2). These regions are similar to those determined for an independent analysis of phytoplankton community composition (Lehman and Smith 1991).

Long-Term Trends

Figure 50 shows long-term trends for 1970-1993 for chlorophyll *a* concentration. Concentrations for the upper estuary were highly variable, averaging 1-47 μg/L among regions and with monthly values of 0-165 μg/L. Chlorophyll *a* concentrations were highest in the southern region, where annual average concentrations reached 180 μg/L. Concentrations were also high in the lower San Joaquin River, where freshwater phytoplankton blooms often occur. In the northern region, concentrations rarely exceeded 5 μg/L.

Among regions, chlorophyll *a* concentrations were variable over time. Little variation occurred in the northern region. In the southern region, chlorophyll *a* concentrations decreased by a factor of 3 after 1977 and increased again during the drought years in the early 1990s. For most regions, chlorophyll *a* concentrations were high in the early 1970s, 1980s, and 1993 when streamflows were moderate or high (Lehman 1996). Moderate to high streamflows are needed to transport phytoplankton downstream into Suisun Bay, where about 38% of the phytoplankton biomass is transported downstream from the Delta (Jassby *et al* 1995). Long-term changes in chlorophyll *a* concentration are also associated with changes in environmental conditions produced by the 1977 climatic shift (Lehman 1992). For Suisun Bay, decreased chlorophyll *a* concentrations after 1986 were also a function of increased grazing by the introduced clam, *Potamocorbula amurensis* (Alpine and Cloern 1992).

Water-Year Trends

Figure 51 shows percent deviation from the long-term average for chlorophyll *a* concentration for wet, normal, dry, and critical water years.

The highest chlorophyll a concentrations occurred during normal years, when concentrations were 58-175% above average for most stations. During wet years, chlorophyll *a* concentrations were significantly ($p<0.05$) lower than those during normal years. High concentrations occurred downstream during wet years, when high streamflows transport phytoplankton downstream. Even though concentrations at some stations were below average, chlorophyll *a* concentrations averaged 4% above the mean for dry years and were not significantly ($p<0.05$) different from wet years. Average chlorophyll *a* concentrations were 17% below average during critical years.

Chlorophyll *a* concentrations differed between upstream and downstream stations during critical years only. Concentrations were 58-175% higher than average along the periphery of the Delta and 27-77% lower than average downstream. This large difference was produced by low streamflows, which concentrated phytoplankton upstream (Lehman 1992; Lehman 1996), and increased benthic grazing (Nichols 1975; Alpine and Cloern 1992) and lysing of phytoplankton cells downstream.

Seasonal Trends

Figure 52 shows percent deviation of chlorophyll *a* concentrations from the long-term average during fall, winter, spring, and summer.

Chlorophyll *a* concentrations were up to 90% higher than average during the spring and summer, when high residence time, solar irradiance, and water temperature promote phytoplankton growth. Concentrations were somewhat higher during the spring, when concentrations were above average at most stations. Chlorophyll *a* concentrations were 25-75% lower than average during the fall and winter, when low residence time, solar irradiance, and water temperature reduce phytoplankton growth.

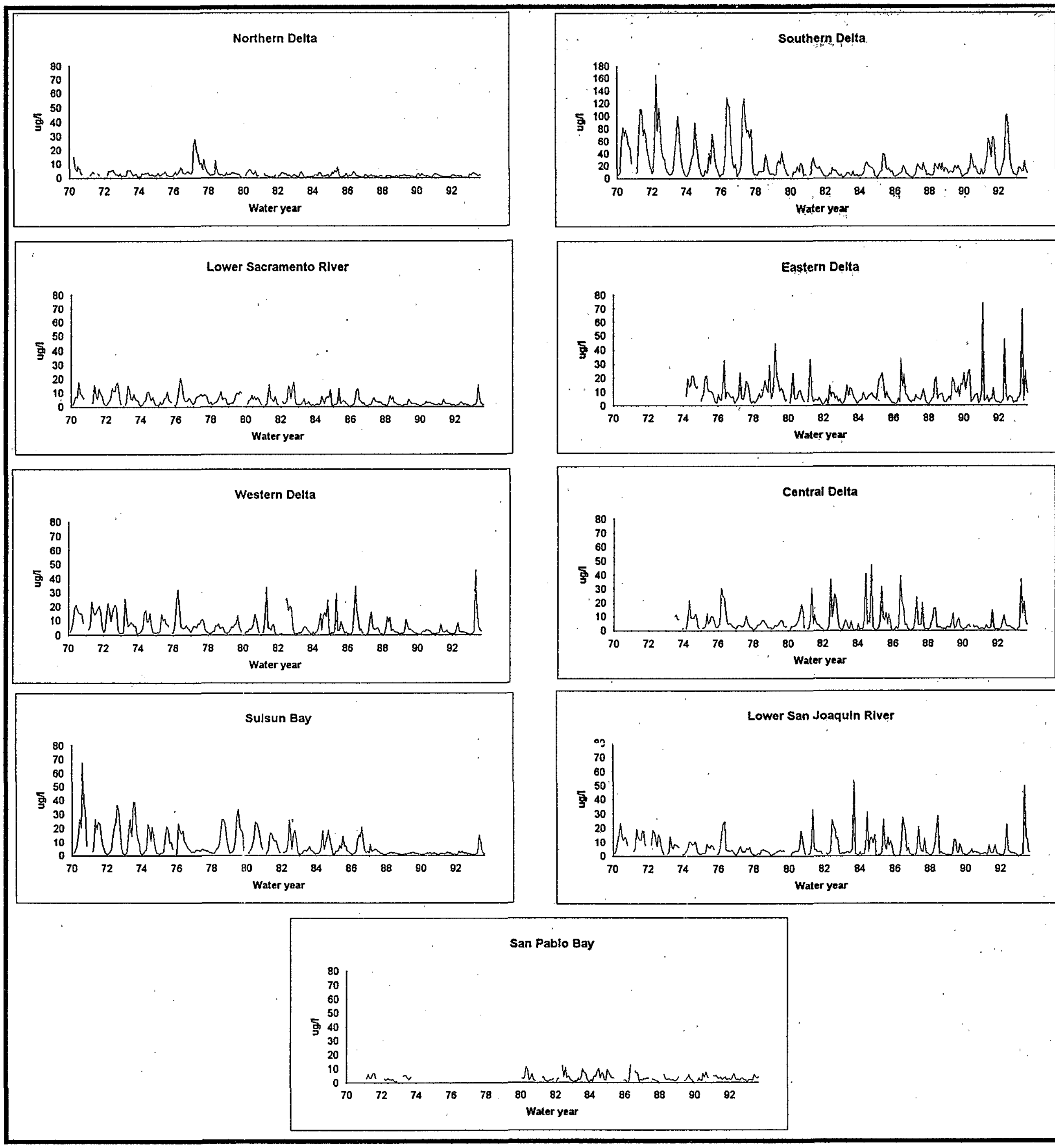

Figure 50
LONG-TERM TRENDS FOR CHLOROPHYLL *a* CONCENTRATION, 1970-1993

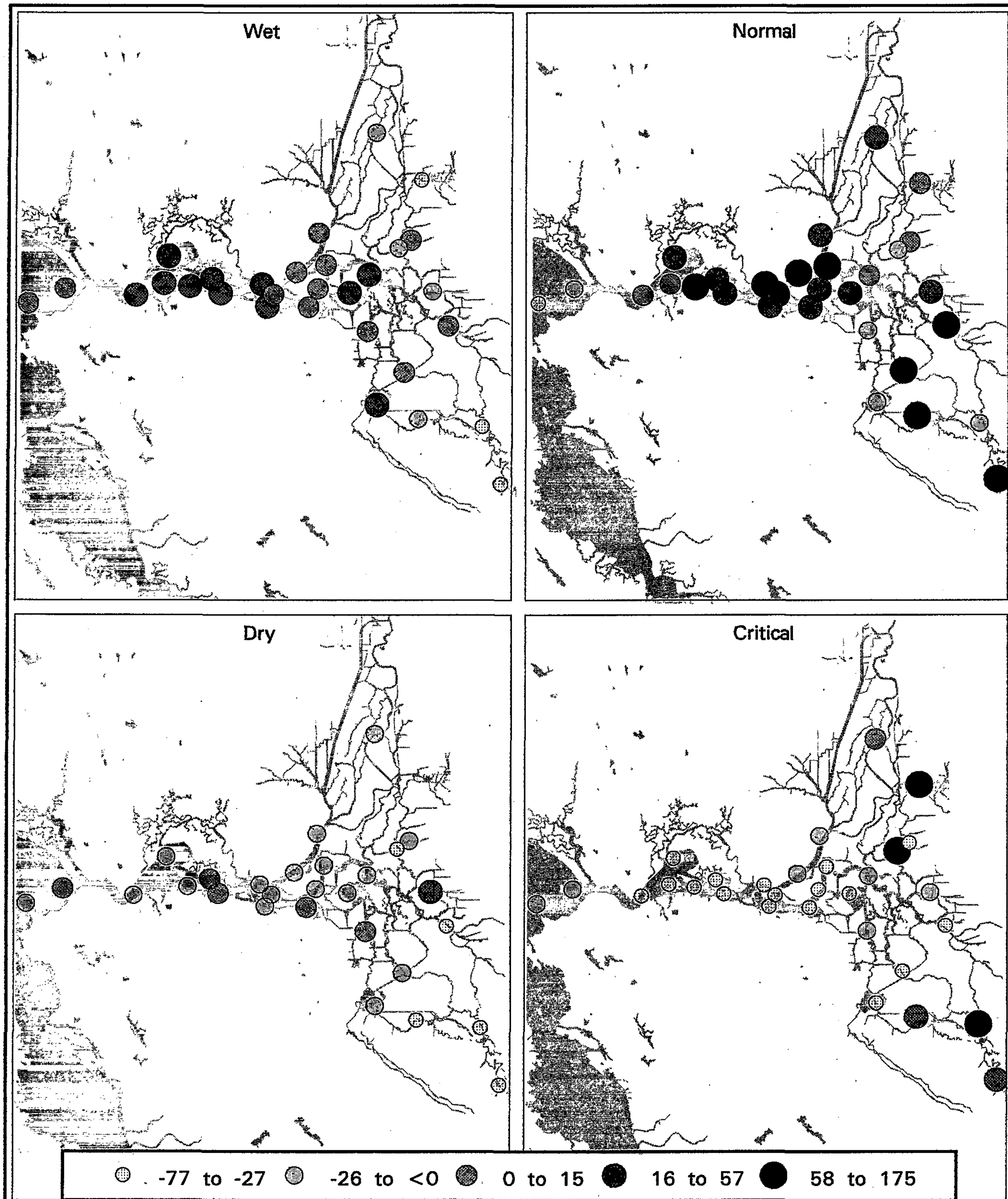

Figure 51
PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR CHLOROPHYLL *a* CONCENTRATION, BY WATER-YEAR TYPE

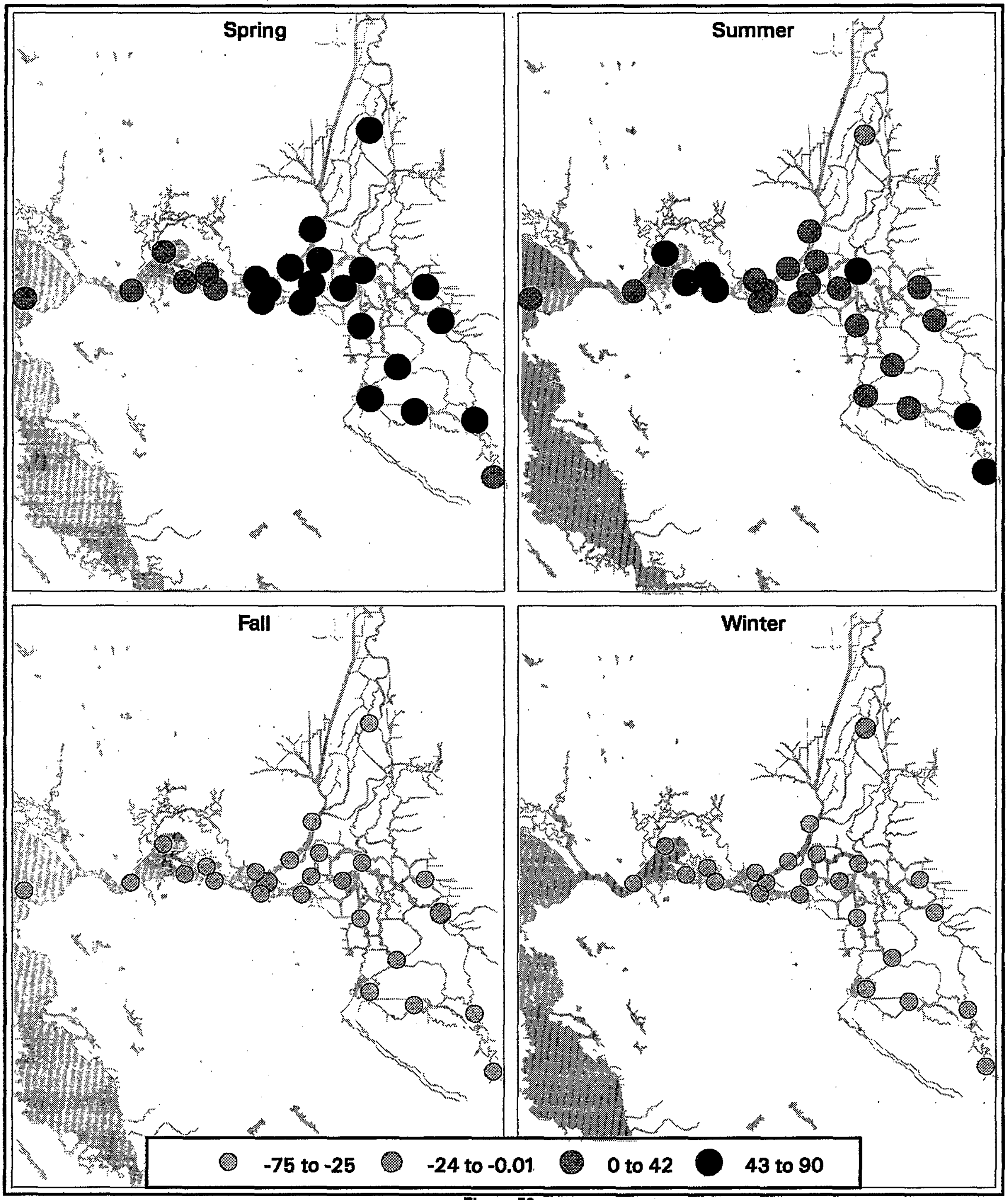

Figure 52

PERCENT DEVIATION FROM THE LONG-TERM AVERAGE FOR CHLOROPHYLL *a* CONCENTRATION, BY SEASON

Gaps between data ranges in legend indicate no data.

Phytoplankton Community Composition

Between 1975 and 1993, phytoplankton community composition was measured monthly or semimonthly at 16 stations (Figure 53). Water samples were collected at 1 meter within 1 hour of high slack tide using a Van Dorn sampler or a submersible pump. The time of each sample was recorded to the nearest 5 minutes using Pacific Standard Time, and total depth was measured to the nearest foot by an electronic depth sounding instrument or a weighted measuring tape.

For each station, a 50-mL water sample was stained and preserved with Lugol's solution and stored in the dark until analysis. Phytoplankton samples were prepared using the Utermohl method (1958), and phytoplankton cells were enumerated and identified to at least genus (Appendix A). Magnification changed over the 19-year interval: 280X in 1975-1981, 350X in 1982-1983, and 750X after 1984. An oil immersion objective was used to assist identifications. Also recorded were an estimate of cell dimensions and whether the cells occurred as individuals or colonies. Individual cells within colonies (*eg*, filaments) were not counted. Cell counts were converted to cells/mL using the following equation:

$$\text{cells / mL} = \frac{C \times Ac}{V \times Af \times F}$$

where: C = cell count
Ac = area of sampling chamber bottom (mm2)
Af = area of each field (mm2)
F = number of fields counted
V = volume settled (mL)

As for chlorophyll *a*, class and species composition were indexed to water year types, and water-year type and seasonal data were calculated as percent deviations from the long-term average. The long-term average for species composition was calculated as standard deviation units, which is the difference between the monthly average and the long-term average divided by the standard deviation of the long-term average over time.

Stations were grouped into the same regions as for chlorophyll *a*, based on hierarchial cluster analysis of monthly data.

Long-Term Trends in Group Composition

Figure 54 shows how phytoplankton community composition has changed since 1975. In 1975-1979, diatoms comprised about 80-100% of the phytoplankton community and were accompanied by up to 40% greens. Between 1980 and 1983, the community composition shifted to a more mixed assemblage, with fewer diatoms and more greens and other groups of phytoplankton. Percent diatom composition appeared to reach a minimum during the 1987-1992 drought and increased again in the early 1990s. Despite high variability, the same general trend was observed for all regions, with the strongest shift in the southern, Suisun Bay, and eastern regions. Average standard deviation units calculated from species data (Figure 55) indicate changes in percent diatom composition were a result of a loss of diatoms and an increase in other phytoplankton groups.

The loss of diatoms may be related to the decreased number of normal years and increased number of critical years after the mid-1970s (Lehman 1996). During normal years, streamflows are low enough to allow accumulation of phytoplankton biomass but high enough to transport diatoms downstream. During critical years, low streamflows reduce downstream transport and concentrate phytoplankton upstream where they are susceptible to pumping (Jassby *et al* 1995). In addition, downstream phytoplankton are more susceptible to grazing by benthic herbivores.

Water-Year Trends in Group Composition

Figure 56 shows percent deviation from the long-term average for phytoplankton community composition for the various water year types. All phytoplankton except dinoflagellates, green flagellates, and miscellaneous flagellates increased by at least 40% during normal years. All phytoplankton groups, except diatoms, also increased during critical years, but the increase was less than 50%. During wet and dry years, most phytoplankton decreased, but wet years had more chrysophytes, dinoflagellates, and miscellaneous flagellates than dry years. Most of these changes were statistically significant (Lehman 1996).

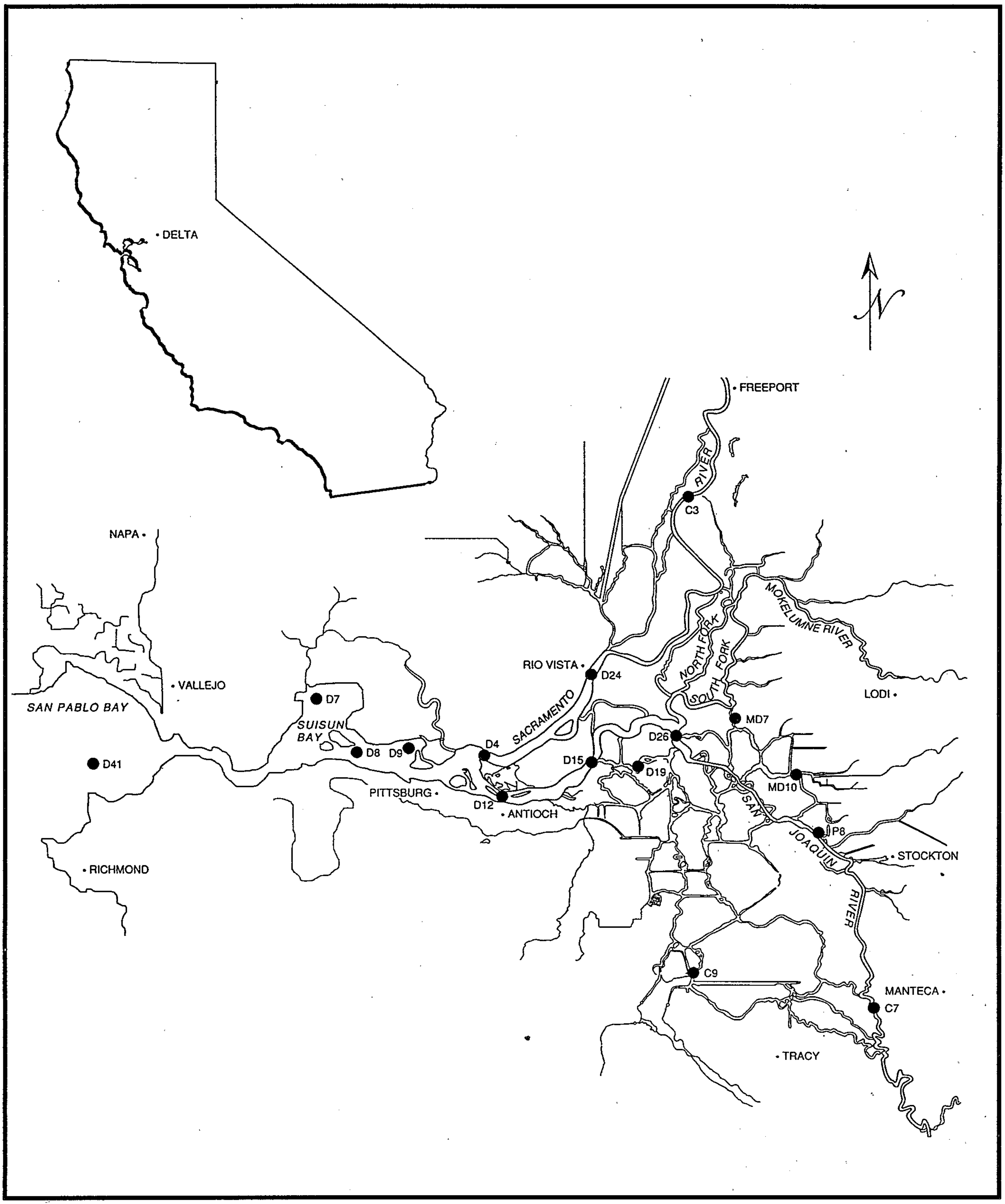

Figure 53
SAMPLING LOCATIONS FOR PHYTOPLANKTON COMMUNITY COMPOSITION
Sacramento-San Joaquin Delta

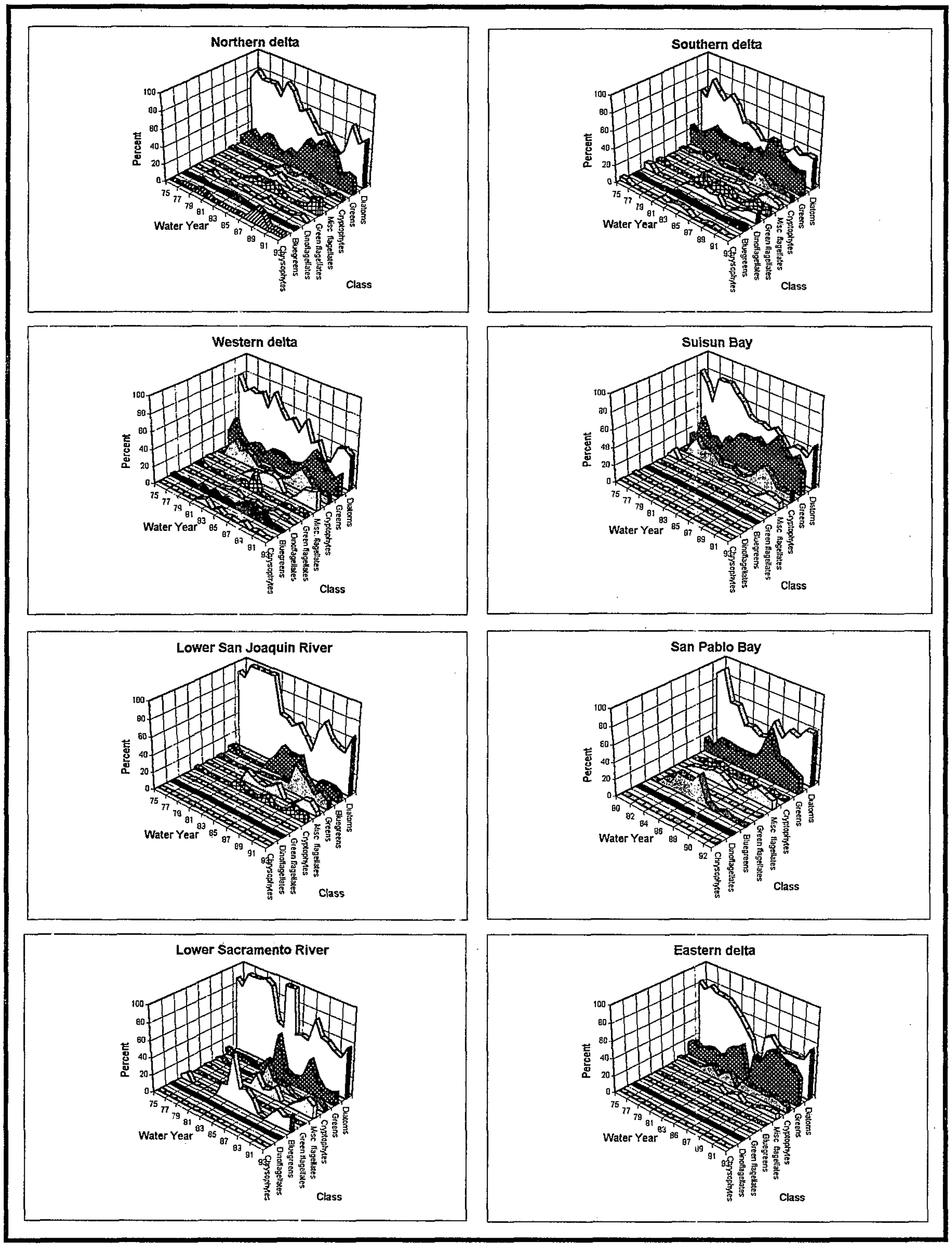

Figure 54
PHYTOPLANKTON COMMUNITY COMPOSITION, 1975-1993

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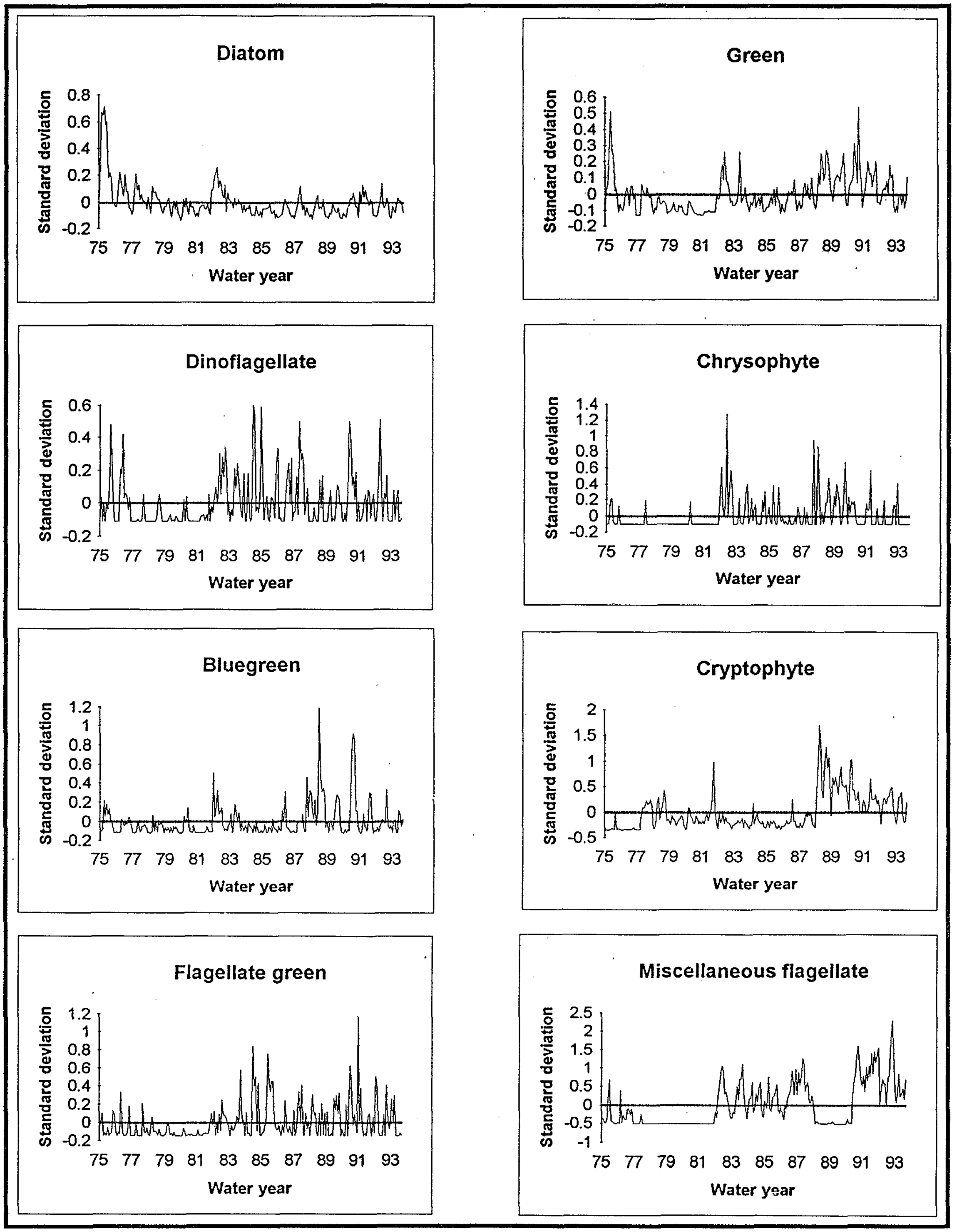

Figure 55
STANDARD DEVIATION UNITS CALCULATED FOR PHYTOPLANKTON GROUPS

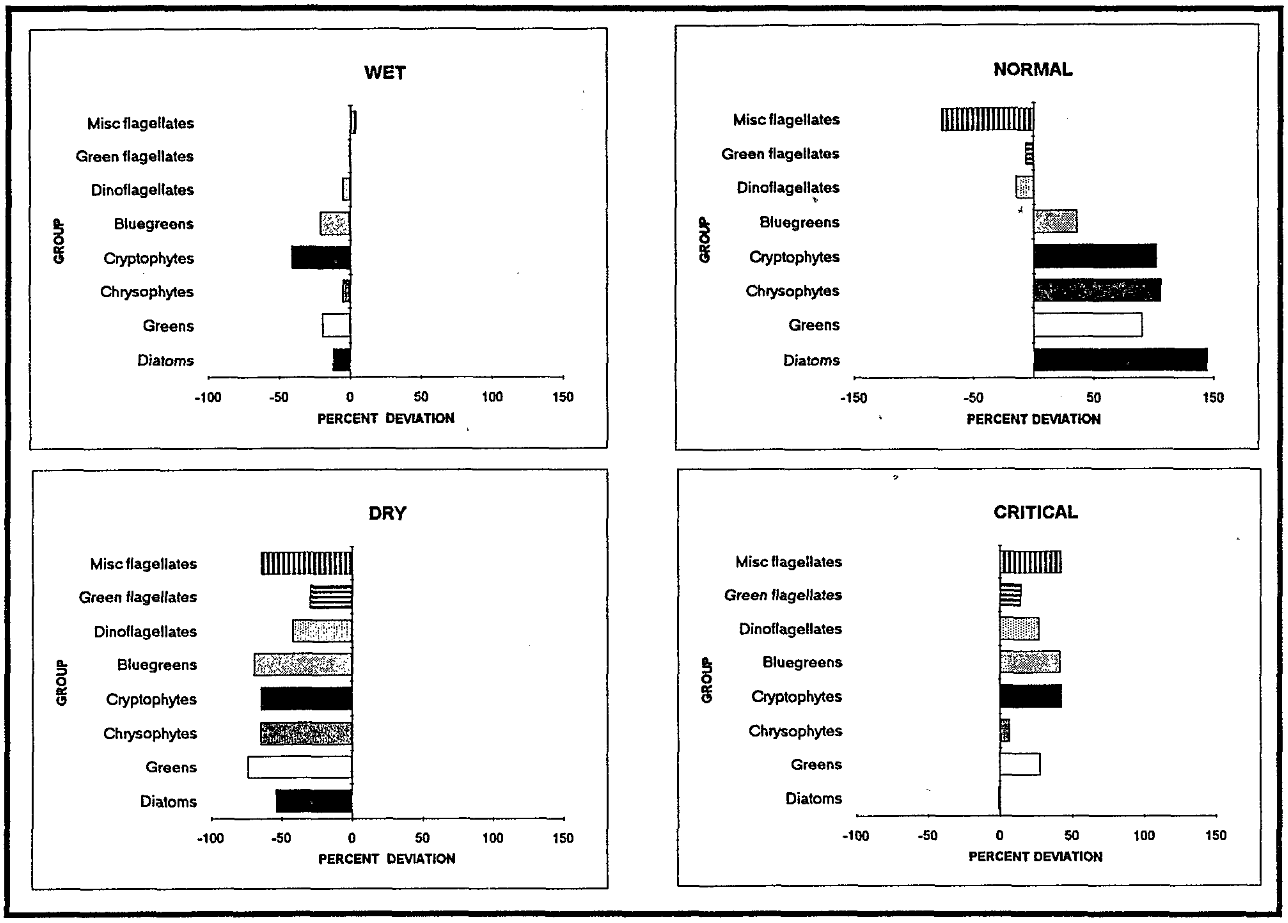

Figure 56
PHYTOPLANKTON COMMUNITY COMPOSITION AMONG WATER-YEAR TYPES

Seasonal Trends in Group Composition

Figure 57 shows percent deviation from the long-term average for phytoplankton community composition for fall, winter, spring, and summer.

Diatoms and greens were 40% more abundant in the spring. Cryptophytes were equally abundant in the spring and summer; and bluegreens, dinoflagellates, flagellate greens, and miscellaneous flagellates were more abundant in the summer. In general, most phytoplankton groups decreased during the fall and winter, except for some flagellates. Miscellaneous flagellates and flagellate greens increased during the fall, while chrysophytes increased during the winter. Seasonal changes were also dependent on water year type (Lehman 1996).

Long-Term Trends in Species Composition

Average standard deviation units for a few representative species are presented in Figure 58. Within groups of phytoplankton, the relative abundance of individual species differed. Freshwater diatoms like *Aulacoseira granulata*[1] or *Fragilaria crotonensis* increased during wet years. After 1985, diatoms like *Nitzschia sigmoidea* often decreased, while dinoflagellates like gymnodinium spp. or cryptophytes like *Rhodomonas lacustrus* increased. The diatoms Asterionella formosa and *Tabellaria fenestra* appeared to decrease after 1977. Other species were abundant during only a few years. Most of these changes were probably caused by the physiological response of each species to environmental conditions and the transport of species downstream with streamflow. These figures need to be viewed cautiously, because phytoplankton were not always identified to species.

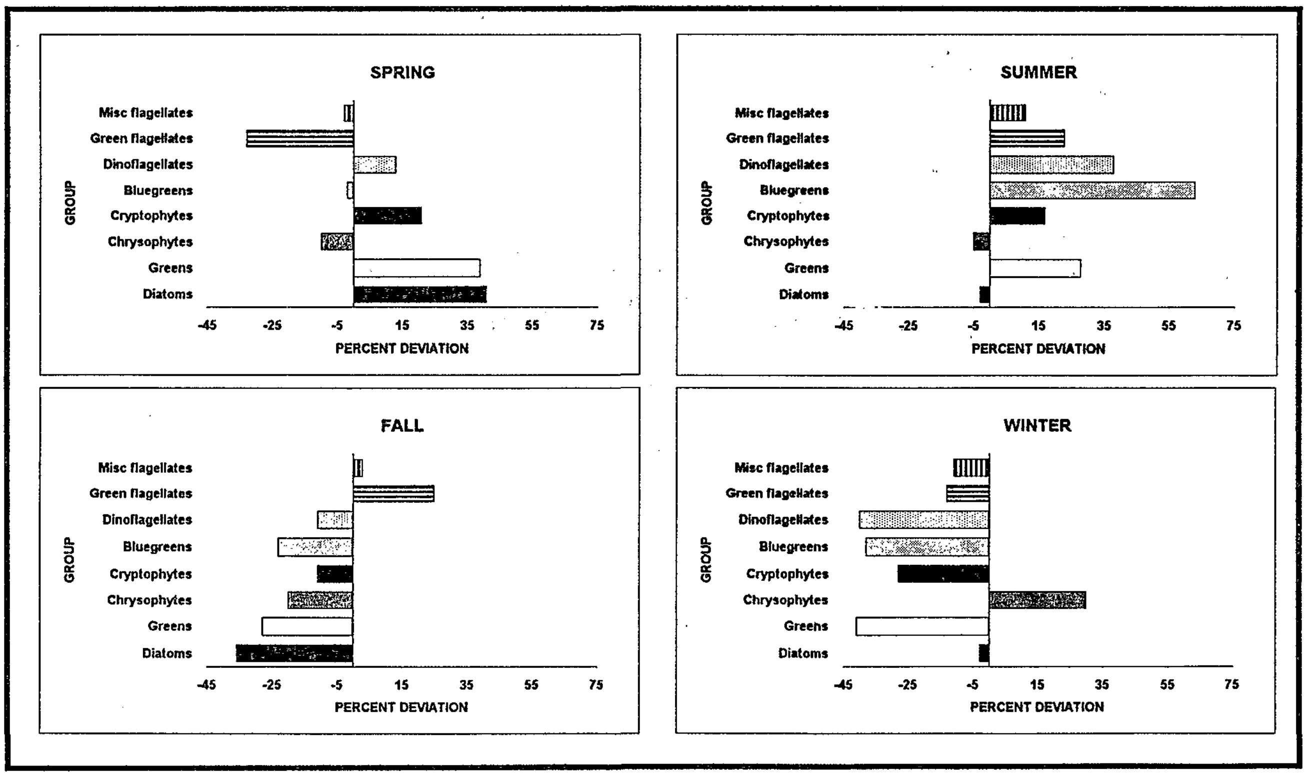

Figure 57
PHYTOPLANKTON COMMUNITY COMPOSITION AMONG SEASONS

1 Formerly *Melosira granulata.*

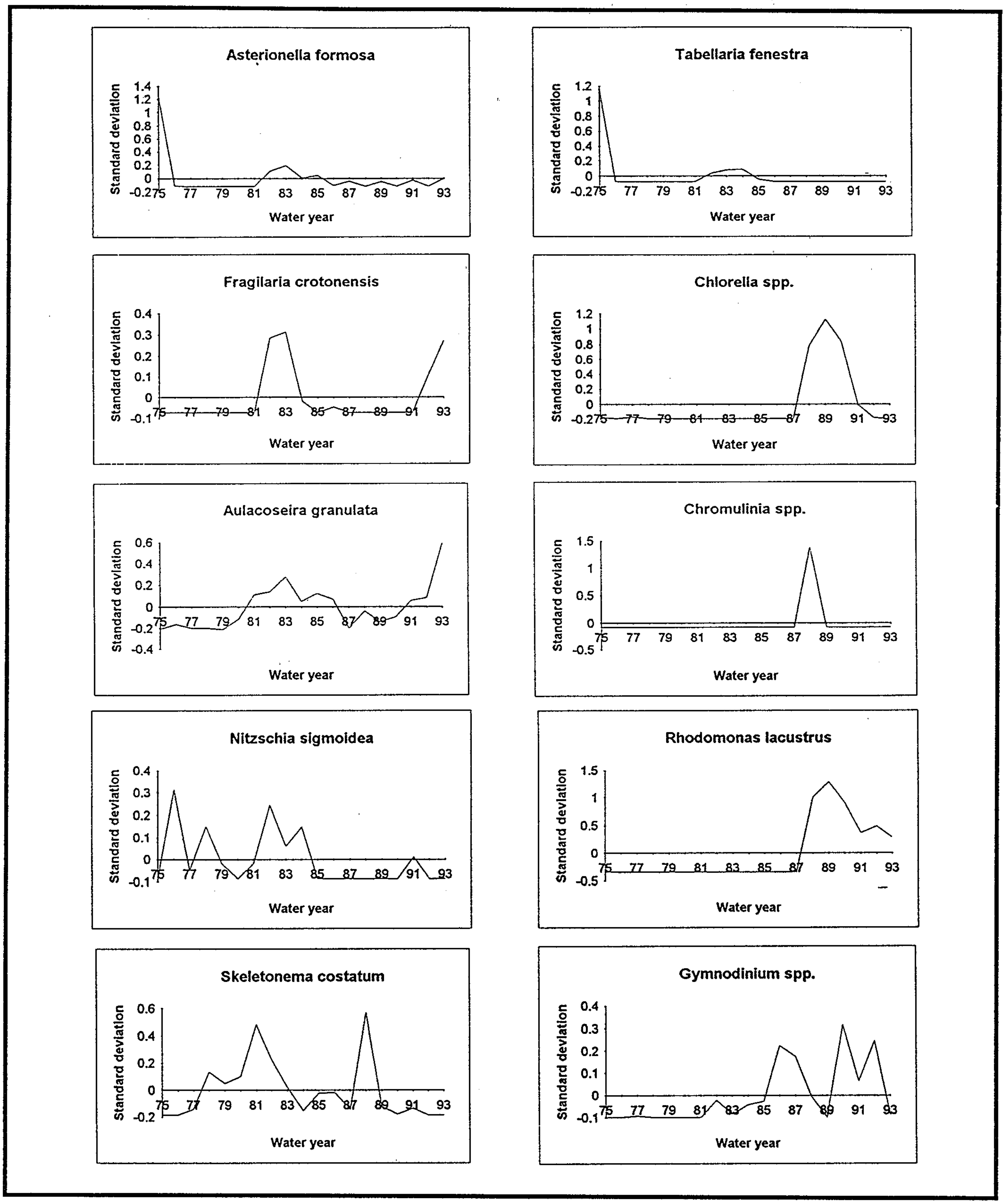

Figure 58
STANDARD DEVIATION UNITS FOR REPRESENTATIVE SPECIES

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Water-Year Trends in Species Composition

Because of the physiological response of species to physical conditions among years and the influence of streamflow on transport, individual species differed among water-year types (Figure 59). As with the long-term trend, these changes were often independent of phytoplankton groups. In normal years, abundance of the diatoms *A. formosa* and *T. fenestra*, the green *Chlorella* spp. and the cryptophyte *R. lacustrus* were higher than average. The diatoms *F. crotonensis, A. granulata,* and *N. sigmoidea* were more abundant in wet years, and the dinoflagellate *gymnodinium* spp. or chrysophyte *Chromulinia* spp. were more abundant in dry or critical years.

Seasonal Trends in Species Composition

Species composition also varied with season (Figure 60). Spring had relatively more freshwater diatoms, including *A. formosa, F. crotonensis,* and *A. granulata,* while the warmer and more saline summer had brackish water diatoms and greens like *Skeletonema costatum* and *Chorella* spp. Winter had diatoms and flagellates such as *N. sigmoidea* or *T. fenestra* and *Chromulinia* spp., but none of the selected species was more abundant in the fall.

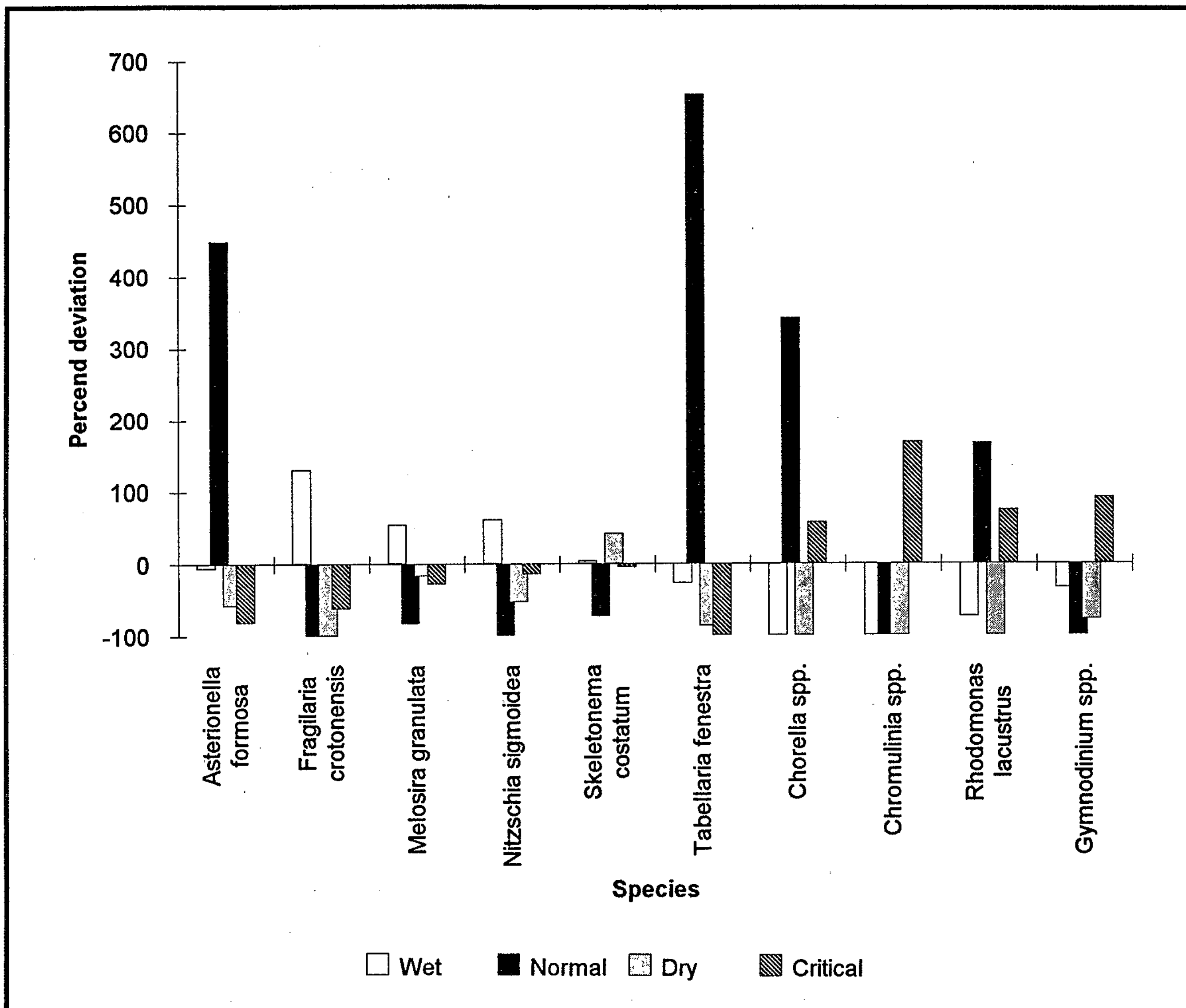

Figure 59
SPECIES COMPOSITION, BY WATER-YEAR TYPE

C —0 3 6 5 6 6

C-036566

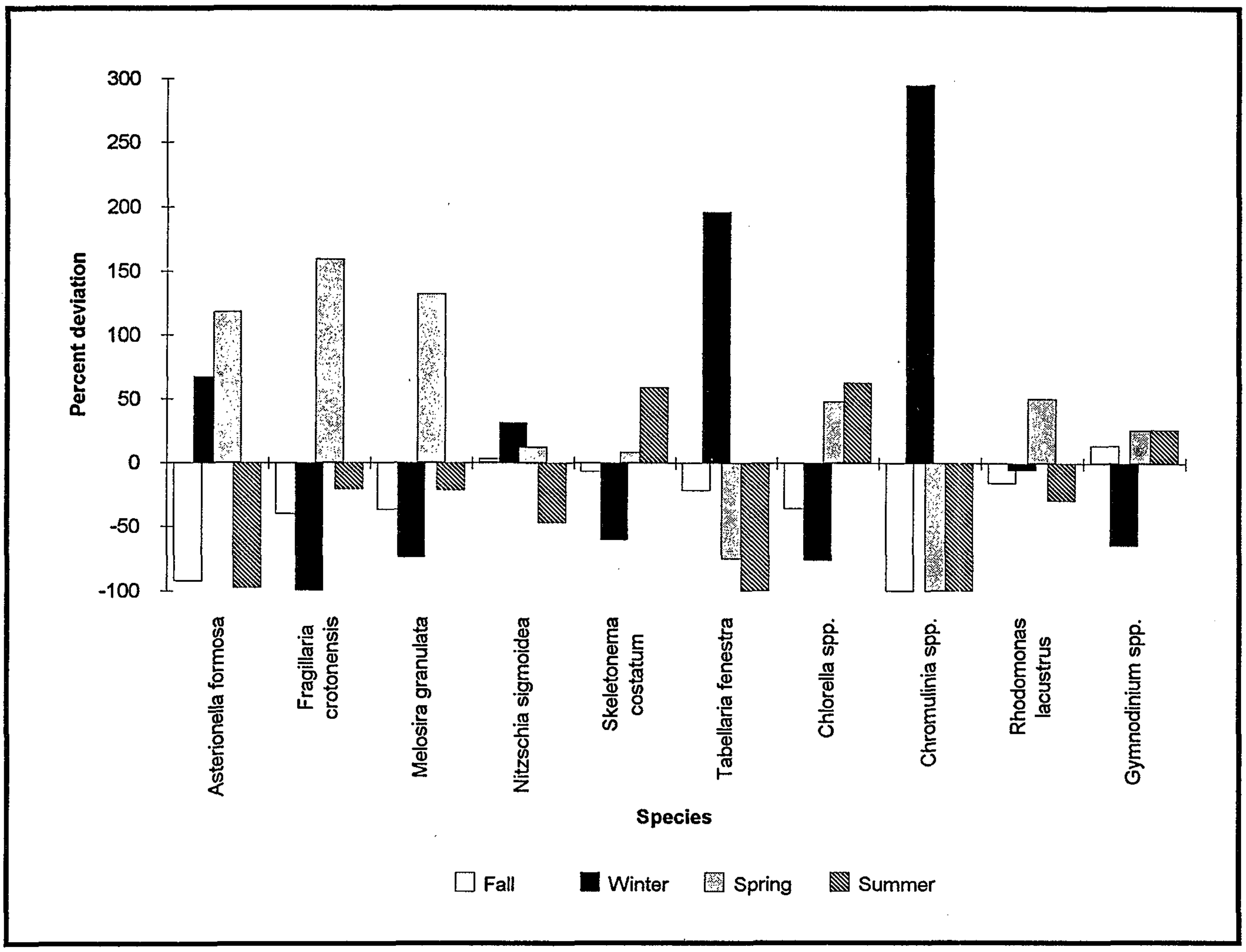

Figure 60
SPECIES COMPOSITION, BY SEASON

C —0 3 6 5 6 7

C-036567

Benthic Macrofauna and Substrate

Benthic macrofauna are organisms that live in or on or that are attached to bottom substrate and are larger than 0.5 millimeters. Benthic data are collected as part of the Water Quality Monitoring Program mandated by Decision 1485. The program monitors long-term trends in benthic macrofauna population density and distribution and substrate composition and detects introductions of exotic species.

Between 1975 and 1979, benthic and substrate samples were collected biannually during spring and fall at 18 stations. The program was revised in 1980, and between 1980 and 1992, benthic and substrate samples were collected monthly at five stations having different salinity and substrate conditions (Figure 61). A station was added in San Pablo Bay in 1992. Up to three sectors were sampled at each station (Table 8). For the benthic sample, triplicate grabs were taken with a Ponar dredge having an area of 0.053 m^2 (APHA 1989).

For benthic organisms, the contents of each sample were washed over 30-mesh screen (0.595 mm openings). Material remaining on the screen was washed into a plastic jar, preserved with 50% formalin, and stained with Rose Bengal dye. At the laboratory, the volume of settleable substrate in each sample and the composition of the substrate (eg, peat or sand) was estimated and recorded. The substrate was hand-picked for organisms under an illuminated magnifier (three diopters). Organisms were then placed in 70% ethyl alcohol for subsequent identification. A stereoscopic dissecting microscope (magnification of 7x to 120x) was used to identify most organisms to species. When taxonomic features were too small for accurate identification, the organism was permanently mounted on a slide and examined under a compound microscope. The number of organisms per square meter was calculated by the equation:

$$\text{organisms} / m^2 = \frac{(1.0\ m^2)\ C}{P}$$

where: C = organism count in sample
P = area of Ponar dredge (m^2)

Benthic samples were analyzed by Hydrozoology Laboratory at 8955 Langs Hill Road, Newcastle, CA 95658, which maintains a reference collection preserved in 70% ETOH.

One substrate sample was taken at each benthic sampling sector using the Ponar dredge. An additional substrate sample was taken at the right bank at D7 and at the right and left banks at D11 and D19. Each substrate sample was placed in a wide-mouth quart jar. Combustion at 440°C for 24 hours was used to determine the inorganic and organic content of the sample (APHA 1989). Inorganic material remaining after combustion was size-fractionated by passing particles through a series of screens (Manual of Testing Procedures for Soils, Department of Water Resources, 1962). Particle size was partitioned into three categories: fines (silt and clay 1-5 µm), sand (6-6000 µm), and gravel (7000-150000 µm).

The benthic database consists of several hundred individual species, many of which are only found at one or two stations (Appendix B). To document long-term trends in species density and distribution in the upper estuary, species selected for analysis occurred throughout most of the upper estuary and made up a large portion of the total community. Newly introduced species were also selected to document the spread of new species. In addition, tubificid worms with similar morphology, habitat, and population dynamics were

Table 8
BENTHIC AND SUBSTRATE SAMPLING SITES

Site	Sector*	Type of Sample**	Habitat
D4	R	Substrate/Benthos	River Channel
	C	Substrate/Benthos	
	L	Substrate/Benthos	
D7	R	Substrate	Shallow Bay
	C	Substrate/Benthos	
D11	R	Substrate	Flooded Tract
	C	Substrate/Benthos	
	L	Substrate	
D19	R	Substrate	Flooded Tract
	C	Substrate/Benthos	
	L	Substrate	
D28A	R	Substrate/Benthos	River Channel
	L	Substrate/Benthos	
D41A	R	Substrate/Benthos	Shallow Bay

* Sectors are determined while facing downstream (Right, Center, Left).
** Substrate samples consist of one random grab. Benthic samples consist of three grabs.

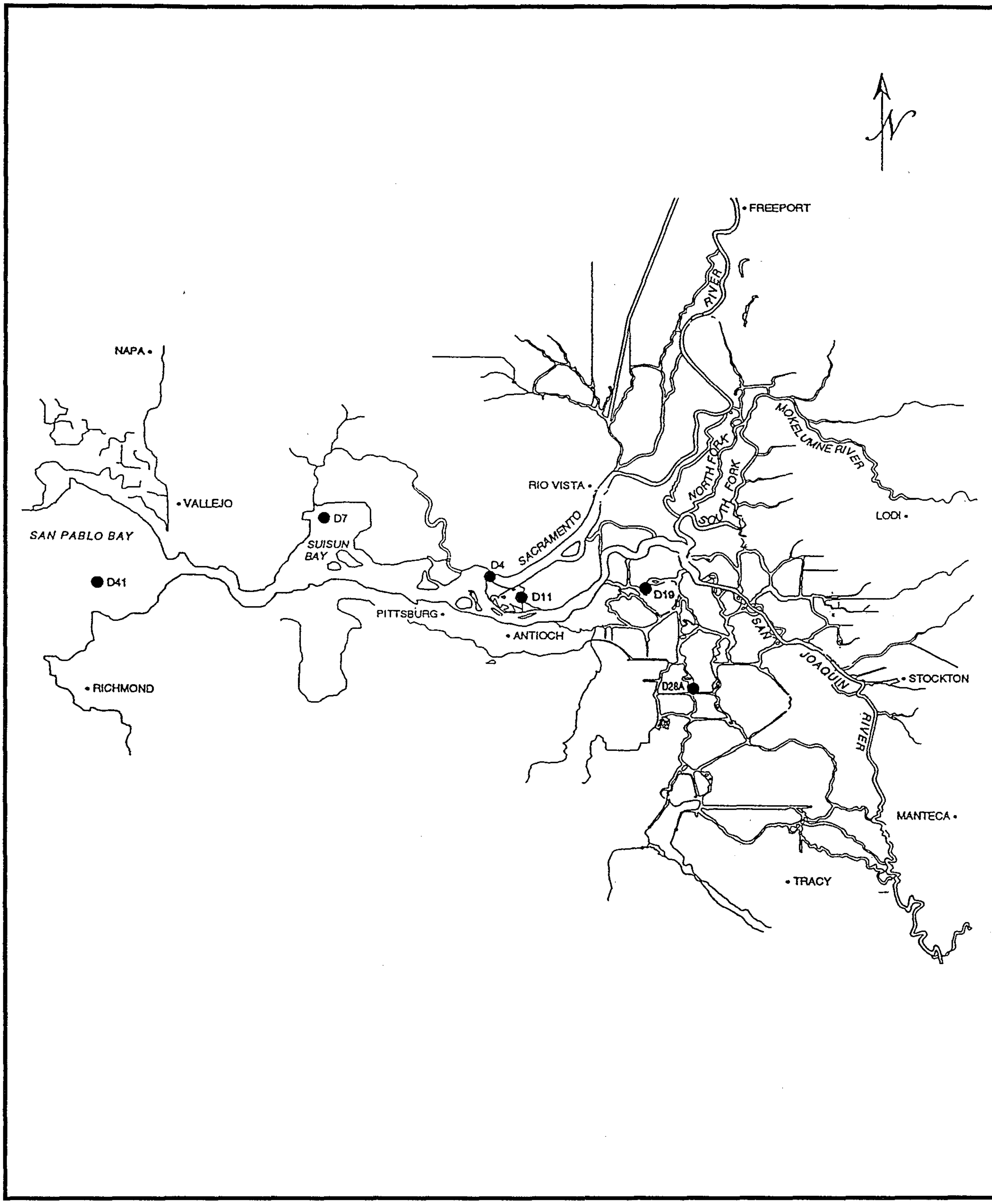

Figure 61
SAMPLING LOCATIONS FOR BENTHIC MACROFAUNA AND SUBSTRATE
Sacramento-San Joaquin Delta

grouped for analysis: *Bothrioneurum vejdovskyanum, Limnodrilus hoffmeisteri, Limnodrilus udekemianus,* and *Varichaetadrilus angustipenis.* They are primarily freshwater worms that tolerate brackish water and are found in the muddy substrate of low-elevation reservoirs and slow-moving streams.

Benthic Substrate

Figure 62 shows substrate composition at the six benthic monitoring sites during the period of record. At most benthic stations, the most common substrate was fines, except at Pt. Sacramento, where sand was often more common. Pt. Sacramento was also the only station where the substrate differed between channel and bank locations. Among years, sand was more common in the early 1980s and 1990s, with consistently low percentages in 1987-1989. A decreasing trend was also measured for sand at Sherman Lake.

Benthic Macrofauna

Figure 63 shows abundance of the more significant organisms analyzed during the period of record.

Hemileucon hinumensis

Hemileucon hinumensis is a salinity-tolerant arthropod. It was found in the Delta during August 1986 and may originate near Japan. During the 1987-1992 drought, *H. hinumensis* increased in Suisun Bay and the lower Sacramento and San Joaquin rivers. It replaced other cumacean species that increased during the 1976-1977 drought.

Dorylaimus species *a*

Dorylaimus species *a* is a carnivorous nematode or roundworm. Because it is intolerant of saline or brackish water, it is restricted to the central Delta (D19 and D28A). It moved downstream during wet years such as 1986 or 1993 with the increase in freshwater habitat downstream. Over time, densities were high during the wet year 1986, decreased during the 1987 drought, and increased during the normal and wet years, 1989 and 1993. Densities of *Dorylaimus* may be underestimated since many of these tiny organisms can wash through the collection screen. Their absence in the record before 1985 may be due to misidentification.

Potamocorbula amurensis

The introduced Asian clam *Potamocorbula amurensis* was first detected in December 1986 (Nichols *et al* 1990). This salinity-tolerant species is now abundant in the brackish and saline waters downstream and extends upstream to margin of the freshwater zone. During the 1987-1992 drought, it increased in the brackish water regions downstream (peak density can reach 48,000 clams/m^2) and rarely occurred in the freshwater regions of the central and eastern Delta. The invasion of this clam has changed the ecology of Suisun Bay, where it has shifted the food web from planktonic to benthic. Recent special studies suggest these clams are less abundant during high-outflow years, but residual clams and upstream movement of larvae allow these clams to quickly re-establish (Hymanson 1991).

Corbicula fluminea

Corbicula fluminea is an introduced freshwater clam. It lives in the freshwater regions upstream and is abundant in the central Delta, lower San Joaquin River, and western Delta (D28, D19, D11). *C. fluminea* comprises a large portion of the benthic organisms in the southeastern regions of the Delta, with peak densities near 20,000 clams/m^2. High densities and the widest distribution occurred during high streamflow years like 1982, when freshwater habitat increases throughout the Delta. Over time, maximum densities of *C. fluminea* have decreased in the lower Sacramento River (D4) and in the western and central Delta (D11 and D28A).

Tubificid Worms

Tubificid worms were found at all stations between 1980 and 1993. They were abundant at freshwater stations upstream and decreased at stations D11, D7, and D4 after 1988 when salinity increased during a series of critically dry years. High densities were associated with the wet years 1982, 1986, and 1993. The maximum density of tubificid worms increased after 1970 at all stations, with the largest increase at station D11. Lower maxima, however, were probably influenced by the reduced sampling frequency.

Corophium stimpsoni

C. stimsoni is an amphipod that is common in freshwater habitats throughout the upper estuary during times of high outflow. Between 1980 and

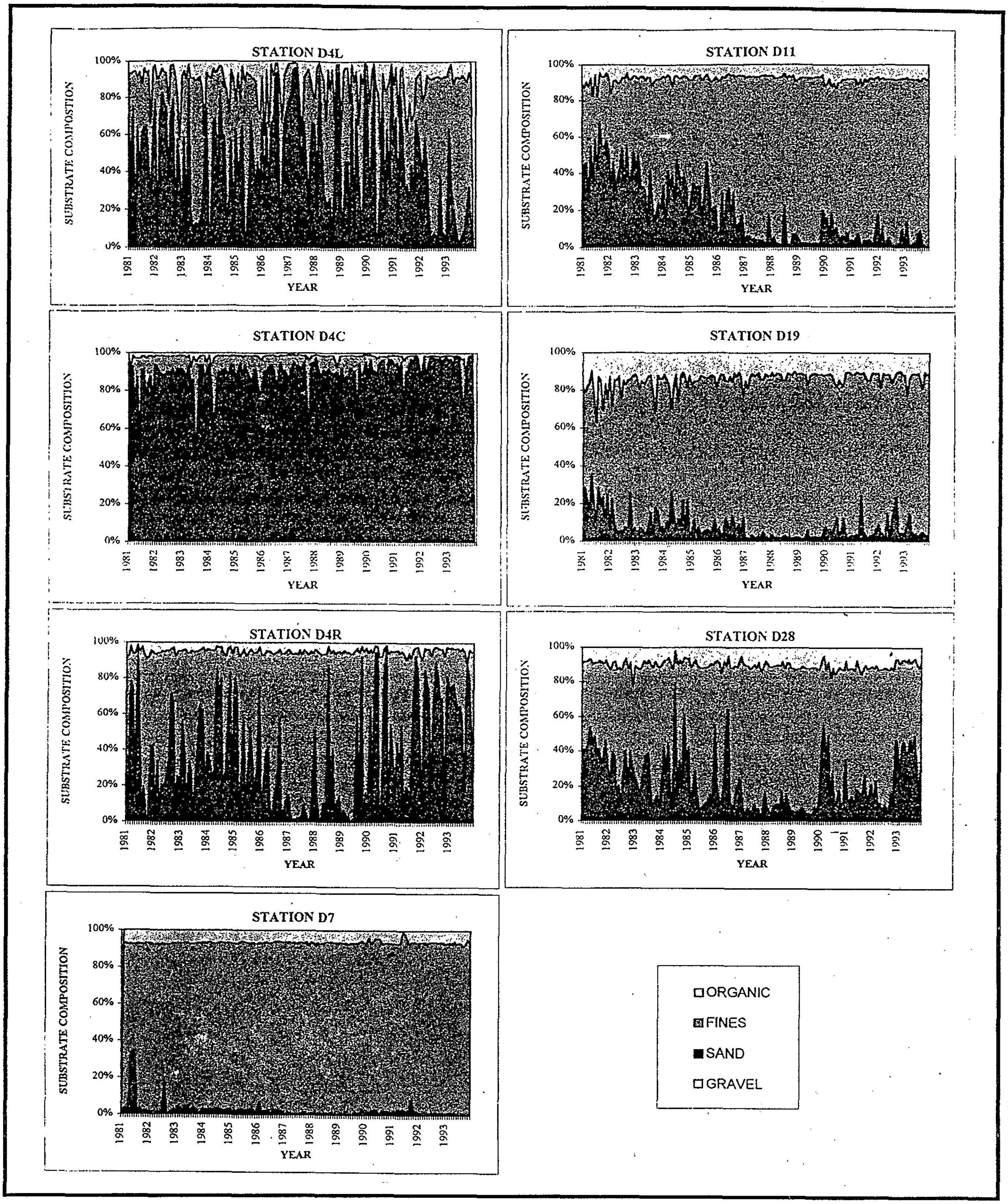

Figure 62
SUBSTRATE COMPOSITION, 1981-1993

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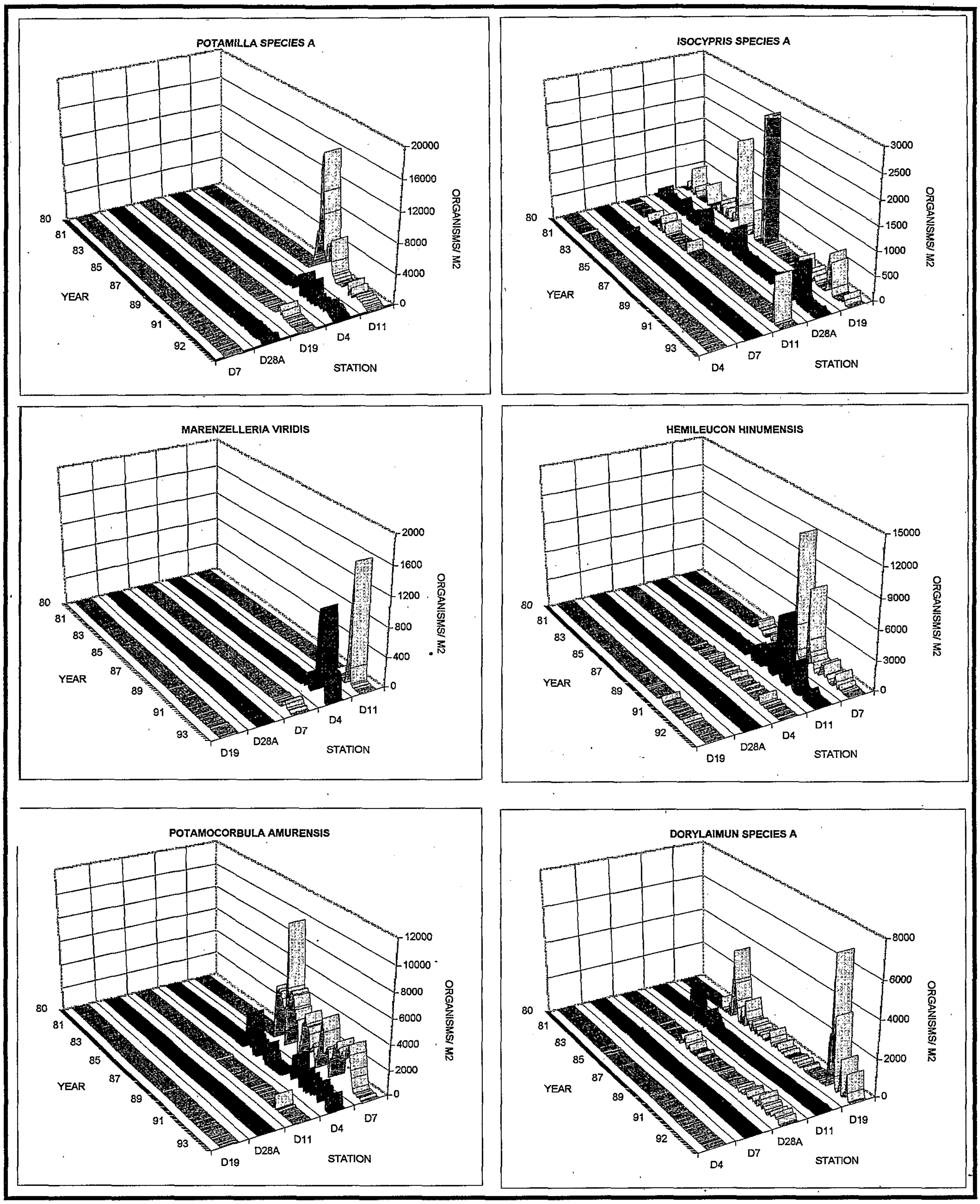

Figure 63
ABUNDANCE OF BENTHIC ORGANISMS, 1980-1992

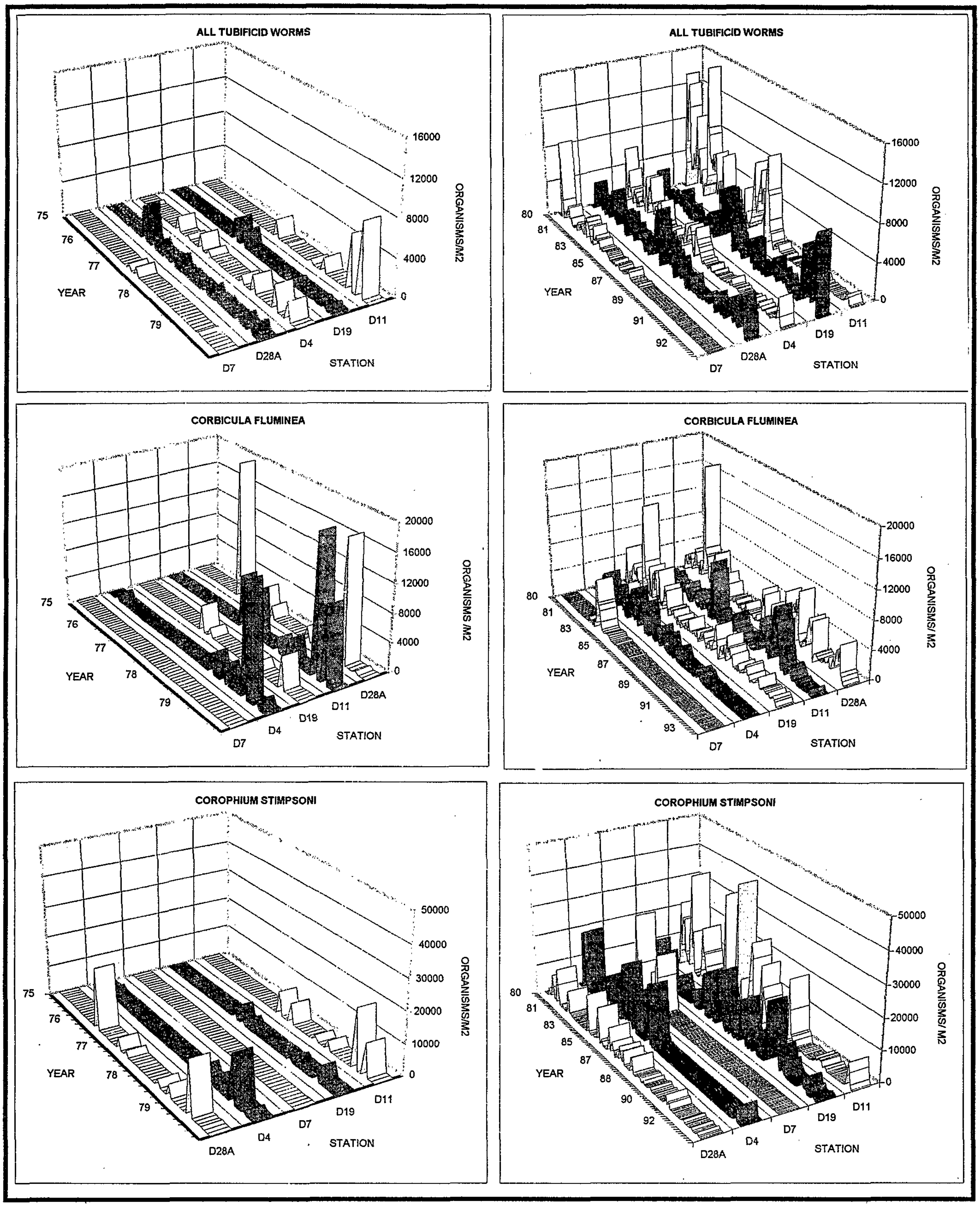

Figure 63 (continued)
ABUNDANCE OF BENTHIC ORGANISMS, 1980-1992

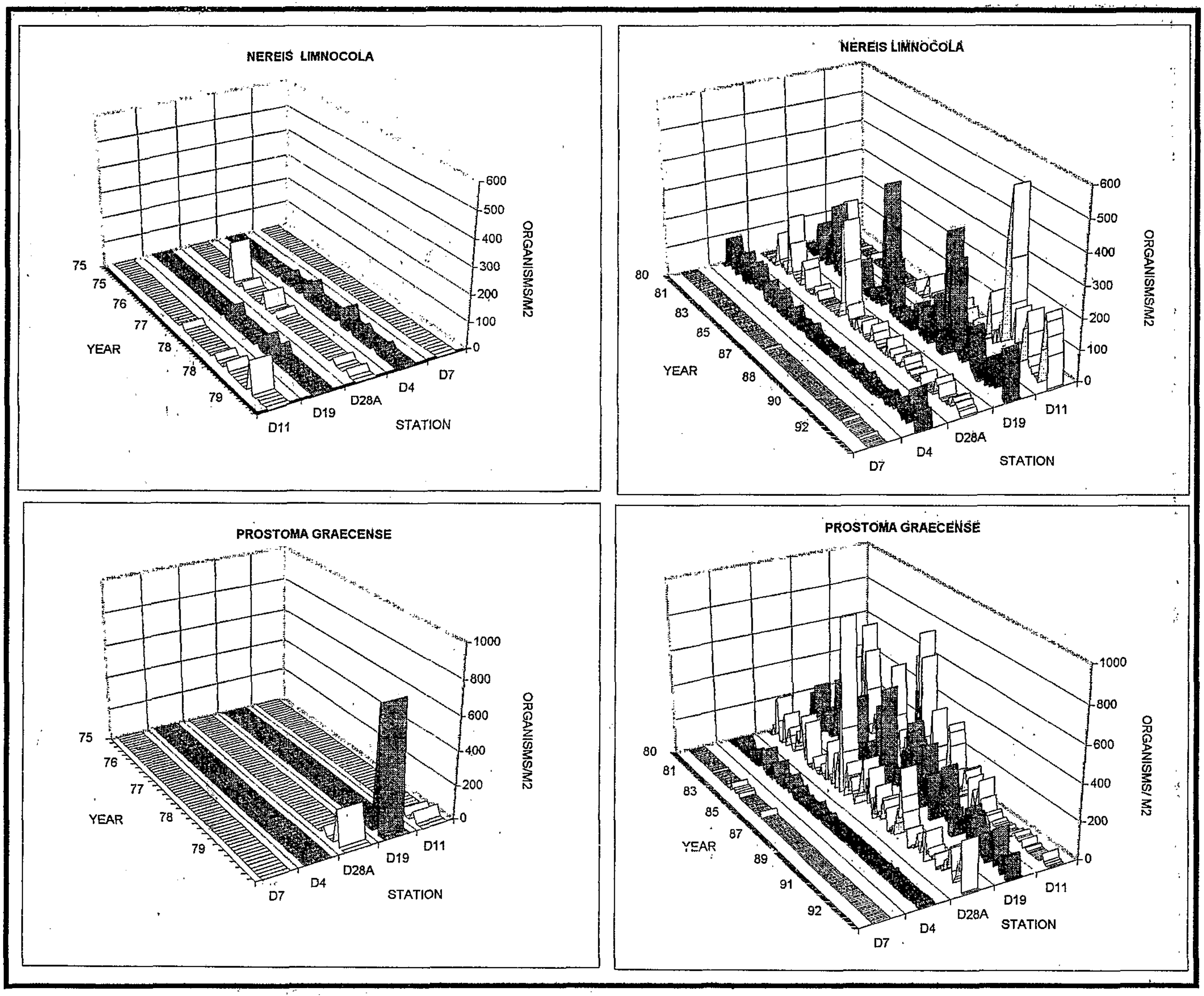

Figure 63 (continued)
ABUNDANCE OF BENTHIC ORGANISMS, 1980-1992

1993, maximum density reached 40,000 organisms/m^2 at D11, near the confluence of the Sacramento and San Joaquin rivers. *C. stimpsoni* was only found in the brackish water habitat in Grizzly Bay (D7) in 1982 and 1983, when high outflow decreased salinity in the lower estuary. A decline in *C. stimpsoni* in the lower Sacramento River (D4) and central Delta (D28A) after 1987 was probably due to the 1987-1992 drought, which reduced freshwater habitat in the Delta. Persistently high densities in the lower San Joaquin River (D19) in 1987-1990 despite the drought may be related to water management practices, which divert Sacramento River into the lower San Joaquin River during the summer. Maximum densities in 1980-1993 were somewhat lower than in 1975-1979 at most stations, with the largest difference at D11.

Potamilla species *a*

Potamilla species *a* is an introduced sabellid worm, believed to have been introduced from India or Asia. These tube worms live in fresh to brackish water and use water currents to feed on suspended organic matter. *P.* species *a* was found at D11 in the western Delta during June 1989. It became abundant near the confluence of the Sacramento and San Joaquin rivers in 1990-1991 but has since declined. It mainly occurs at D4 and D11 but increased at D19 and D28 during 1992.

Prostoma graecense

This nemertean worm occurred at all benthic sampling stations but densities were higher in freshwater habitats in the lower San Joaquin River and central Delta (D11, D19, D28A). Densities were higher at stations with slow-moving water such as D11 than at stations with fast-moving water such as D4. *P. graecense* reached maximum densities in the early to mid-1980s, when outflows were high. Few *P. graecense* were recorded before 1980, but 1979 densities were similar to those in 1980-1993. The absence of *P. graecense* in the early record is unclear and may be due to changes in identification. In general, this is a common species throughout estuaries.

Nereis limnicola

The common freshwater polychaete *Nereis limnicola* occurs frequently in the central Delta (D28), lower San Joaquin River (D19), and western Delta (D11). It has increased in the slow-moving and freshwater stations (D11 and D19) since 1985. Although it occasionally occurs in Grizzly Bay (D7), it is usually restricted to freshwater stations upstream. Over time, densities probably increased at D11 and D19.

Marenzelleria viridis

Marenzelleria viridis is an introduced annelid worm that became abundant in the western Delta in 1992. It is commonly found in the North Atlantic and the East Coast of the United States, where it prefers saline and brackish-water habitat. *M. viridis* was first found at D4 and is now common at D4 and D11. It has not yet spread into the central Delta as other introduced freshwater species have done. Density maxima occurred in early 1993.

Isocypris species *a*

Isocypris is an ostracod commonly called a seed or bean shrimp due to its semi-opaque bivalve carapace. *Isocypris* often occurs in the central Delta (D19, D29A). Density appears to be related to salinity. It was abundant in the early 1980s when outflows were high, decreased during the 1987-1992 drought, and increased again in late 1992-1993 when outflow was high.

C-036575

Aquatic Vegetation

A littoral zone vegetation survey of the Delta was conducted between 1988 and 1992. The objectives were to monitor long-term trends in the type and extent of aquatic vegetation and to detect the presence of nuisance aquatic plants such as *Hydrilla verticillata* and *Eichhonia crasspies* (water hyacinth). Aquatic vegetation can affect operation of the State Water Project by restricting flow, accelerating the rate of sediment accumulation, or clogging trash racks and filter screens.

This survey augments the annual Delta survey by the Department of Food and Agriculture to detect the spread of the exotic aquatic weed *Hydrilla verticillata. Hydrilla* is a native to Africa and was introduced into California in 1976 by the aquarium plant industry (Yeo and McHenry 1977). Its reproduction by vegetative fragmentation, tuber sprouts, and turion formation broadens the habitat of this plant and renders eradication of established populations extremely difficult. The presence of hydrilla is a concern for operation of the State Water Project, because once established, it clogs waterways and pumps.

Ten stations were sampled in the central and southern Delta during the fall of each year and the spring of low outflow years (Figure 64). Station selection was based on a surface photograph reconnaissance survey conducted in October 1988. The following information was collected at each station: water temperature and turbidity at 1 meter, Secchi disc depth, water depth at the edge of the submerged vegetation, estimated distance the aquatic vegetation extended from shore (vegetative extent), and a general station description (including photographs).

In addition, a minimum of three random grab samples for aquatic vegetation were made using a hydrilla hook. Any plant species not previously collected were sorted, washed, and pressed for preservation as herbarium specimens. The initial collection of each species was identified by the Botany Laboratory of the Department of Food and Agriculture. Subsequent collections were verified using the voucher herbarium specimens.

Long-Term Trend

The aquatic vegetation collected or observed during the surveys were typical members of the Delta flora. Plants encountered frequently were anachoris (*Egeria densa*), milfoil (*Myriophyllum spicatum*), hornwort (*Ceratophyllum demersum*), western pondweed (*Potamogeton latifolius*), crisp-leaved pondweed (*P. crispus*), water hyacinth (*Eichhorina crassipes*), and the common tule (*Scirpus acutis*) (Table 9). The red algae *Compsopogon coeruleus* was also found growing epiphytically on submerged plants. *Hydrilla verticillata* was not observed or collected during the surveys.

Aquatic vegetation was generally stable over the study period. The aquatic plant community was similar among surveys for each station and demonstrated little seasonal variation. The most common plants among stations were the submerged plants anachoris and milfoil and the emergent plant the common tule. However, these plants were not common at stations 1, 7, or 9. Instead, aquatic vegetation was nearly absent at stations 1 and 7, and western pondweed and crisp-leaved pondweed characterized station 9. Epiphytic algae were usually collected in the fall. The stability of the aquatic plant community during this time was probably because the sampling period was during the 1987-1992 drought, when water quality and streamflow conditions were similar.

Vegetative extent was low and similar at most stations prior to the 1992 fall vegetation survey, which had exceptionally high vegetative extent values. In 1988-1991, vegetative extent was 10 meters or less at most stations. The pattern was the same for the spring. In fall 1992, vegetative extent increased by at least a factor of two at many stations. At Franks Tract (station 1) and Mildred Island (station 7), the vegetative extent of anachoris was orders of magnitude higher. The large increase in vegetative extent in the fall of 1992 was probably a function of high streamflows that increased freshwater habitat in the upper estuary.

Figure 65 shows Secchi disc depth, surface water temperature, turbidity, and estimated distance of vegetation to shore for fall and spring measurements. Measured environmental variables were poorly related to changes in vegetative extent. Secchi disk depths were high and turbidity was low in the fall when decreased turbulence at low streamflows reduces the resuspension of bottom sediments. Otherwise, water transparency and turbidity were variable among years and stations. The lowest Secchi disk depths were at stations 9 and 10, where depths were consistently less than

C-036576

100 cm. At station 3, Secchi disc depths ranged from 100 to 160 cm.

Water temperature was also poorly related to vegetative extent. Surface water temperatures were similar among stations during each survey. Among years, water temperatures were cooler in the fall in 1990-1992 than in 1988-1989.

Summary

Species composition of aquatic vegetation throughout the Delta in 1988-1992 was similar among stations and seasons. The most common plants were anachoris, milfoil, and common tule, which had low vegetative extent during the drought and increased with streamflows in the 1993 wet year. Vegetative extent was poorly associated with coincident measurements of Secchi disk depth, surface water temperature, and turbidity.

Table 9
PRESENCE OF AQUATIC PLANT SPECIES WITHIN THE LITTORAL ZONE

Sample Site	NOV 1988	MAY 1989	NOV 1989	MAY 1990	NOV 1990	NOV 1991	NOV 1992
1							Ed
2	Ed, Ms, Cd Ec, Sa	Ms, Cd, Pn, Ec		Ed, Cd, Sa	Ed, Ms, Cd, Sa	Ed, Ms, Cd, Sa	Ed, Ms, Sa
3	Ed, Sa	Sa	Ec, Sa	Sa	Ec, Sa	Ed, Ms, Sa	Ec, Sa
4		Ed, Ms		Ed, Ms	Ed	Ed, Ms	NS
5	Ed, Ms, Sa	Ms, Pc, Sa	Ed, Ms, Sa	Ed, Sa	Ms, Ec, Sa	Ed, Ms, Sa	Ed, Ms, Sa
6	Ed, Ms, Sa	Ed, Sa	Ed, Ms, Sa				Ed
7				Ed			Ed
8	Ms, Cd, Sa	Ms, Sa	Ms, Cd, Sa	Ms, Sa	Ed, Ms, Cd, Sa	Ed, Ms, Sa	Ed, Ms, Cd, Ec, Sa
9	Pl, Pc	Pl, Pc	Pl, Pc, Sa	Pl, Pc	Pl, Pc	Pl, Pc	Pl, Pc
10	Ed, Sa	Ed, Sa	Ed, Ms, Pn	Ed, Cd, Sa	Ec, Sa	Ed, Sa	Sa

The vegetative species are: Ed, *Egeria densa*; Ms, *Myriophyllum specatum*; Cd, *Ceratophyllum demersum*; Pl, *Potamogeton latifolius*; Pc, *Potamogeton crispus*; Pn, *Potamogeton nodosus*; Ec, *Eichhonia crasspies*; Sa, *Scirpus acrutis*.

NS- Not Sampled.

C-036577

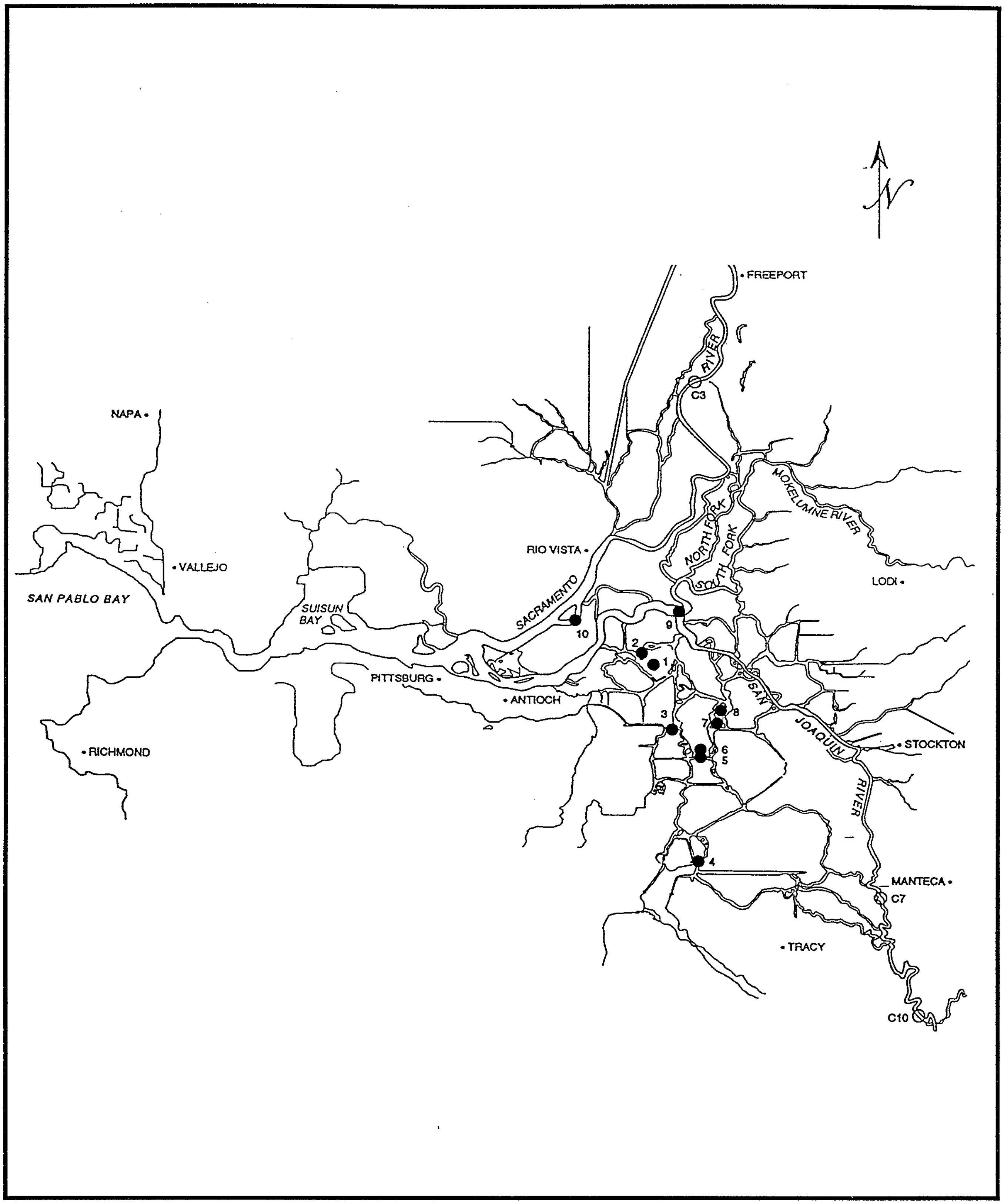

Figure 64
SAMPLING STATIONS FOR AQUATIC PLANT SURVEY
Sacramento-San Joaquin Delta

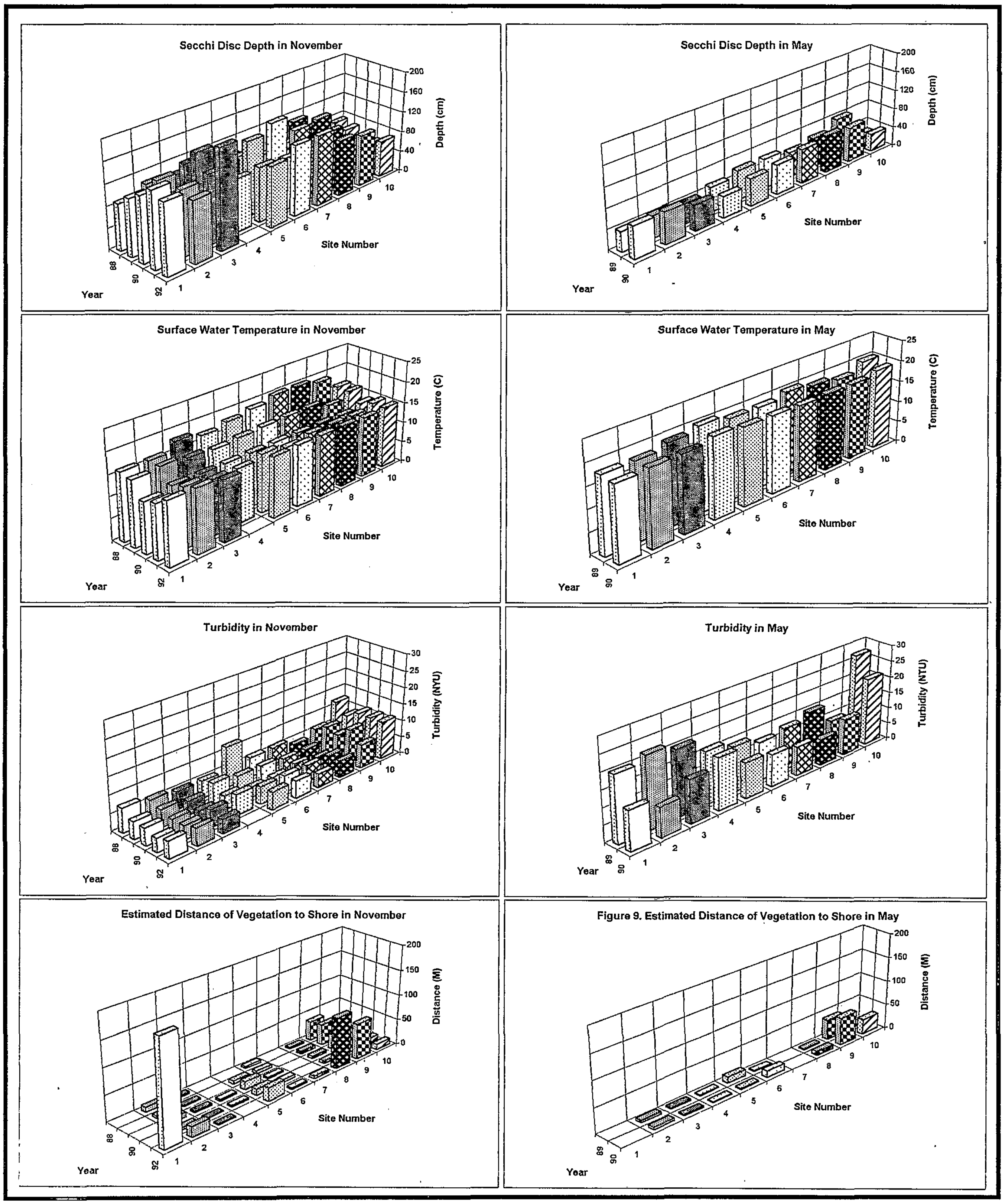

Figure 65
PHYSICAL CHARACTERISTICS AT TIME OF AQUATIC PLANT SURVEYS

C-036579

CONTINUOUS MONITORING NETWORK

Monthly to semi-monthly sampling cruise data are supplemented by continuous water quality monitoring at six stations in the upper estuary (Figure 66). These data provide rapid detection of short-term water quality changes that can be used to assess impacts of State Water Project and Central Valley Project operations and to adjust operations to comply with water quality standards.

In 1983-1990, specific conductance, pH, dissolved oxygen concentration, air temperature, and water temperature were measured continuously by a Schneider model RM25C multi-parameter robot monitor at six locations in the Delta. In April through September 1988-1990, chlorophyll *a* concentrations at Mallard Island and Antioch were estimated using hourly fluorometric measurements. Tidal-day averages were calculated based on a 24-hour day.

Long-Term Trends

Tidal-day averages varied primarily at yearly or seasonal time scales (Figure 67). Among stations, specific conductance varied yearly (360d) or seasonally (90d). pH also varied at the yearly or seasonal time scale but was accompanied by high variation at the 30d and 8d time scales. Dissolved oxygen concentration, air temperature, and water temperature varied primarily at the seasonal and 45d time scale. Combined, these two time scales accounted for up to 90% of the variance in air and water temperature. Chlorophyll *a* concentrations followed the same patterns as the environmental data and varied on a yearly time scale, with some seasonal or 45d variation. Differences among time scales were significant at the 0.05 level or higher.

Water-Year Trends

Water quality varied among water years (Figure 68). Specific conductance and pH were significantly higher ($p<0.05$) for the drought years 1987, 1988, and 1990. Air and water temperature were poorly related to water year type and were significantly ($p<0.05$) higher during both wet and critical years: 1983, 1987, and 1990. Dissolved oxygen concentration varied little among water years, while chlorophyll *a* concentrations were higher in 1988 and 1989 than in 1990.

Seasonal Trends

Mean monthly changes in water quality variables are shown in Figure 69. Specific conductance and pH increased during the year. Air and water temperature increased during summer, and reached a maximum in June. Both were significantly ($p<0.05$) different among all months except the pairs May/August and November/January. Changes in dissolved oxygen concentration were inversely associated with those of air and water temperature. A seasonal pattern was not evident for chlorophyll *a* concentrations, which were low throughout the year.

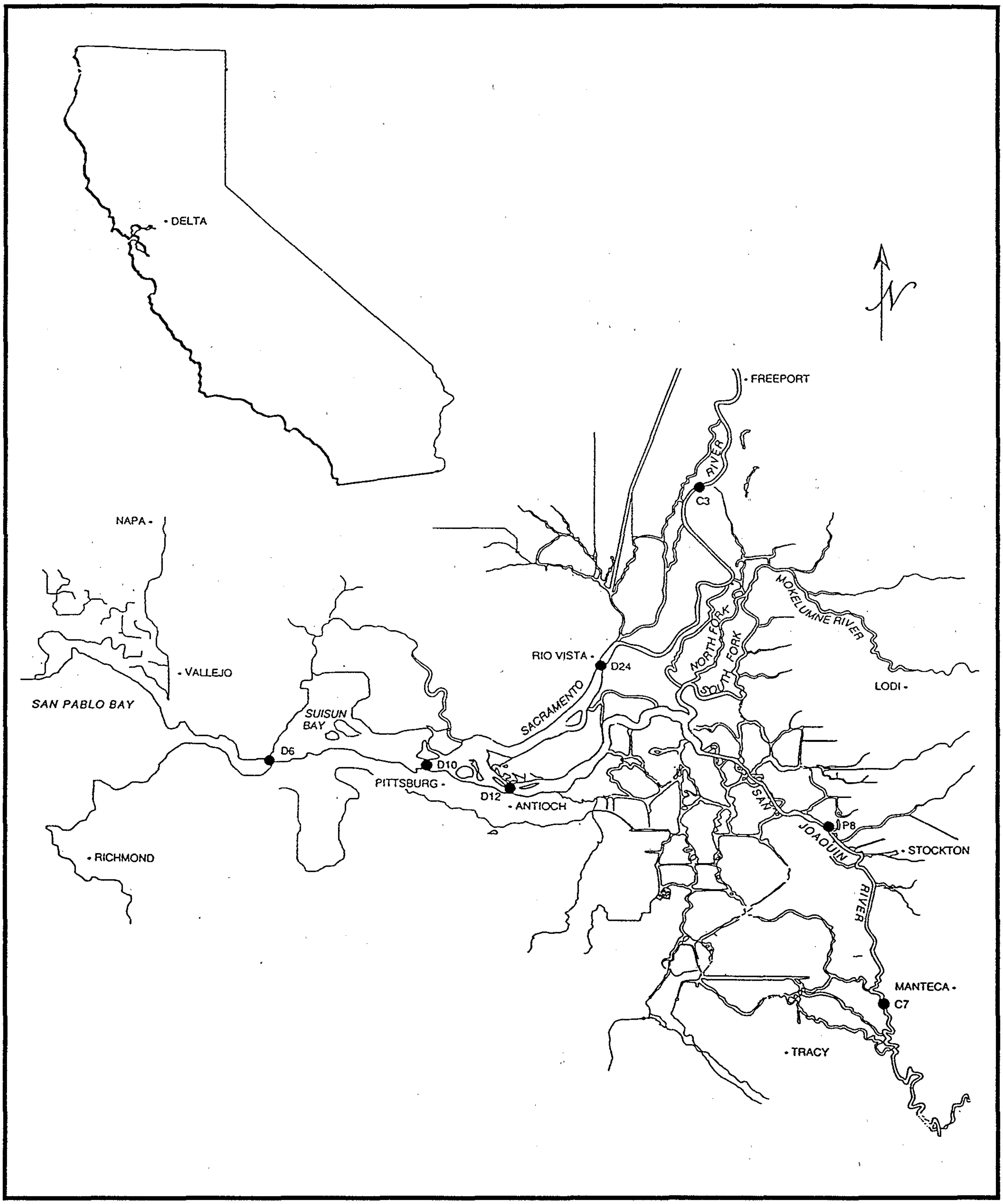

Figure 66
SAMPLING STATIONS FOR CONTINUOUS MONITORING NETWORK
Sacramento-San Joaquin Delta

C-036581

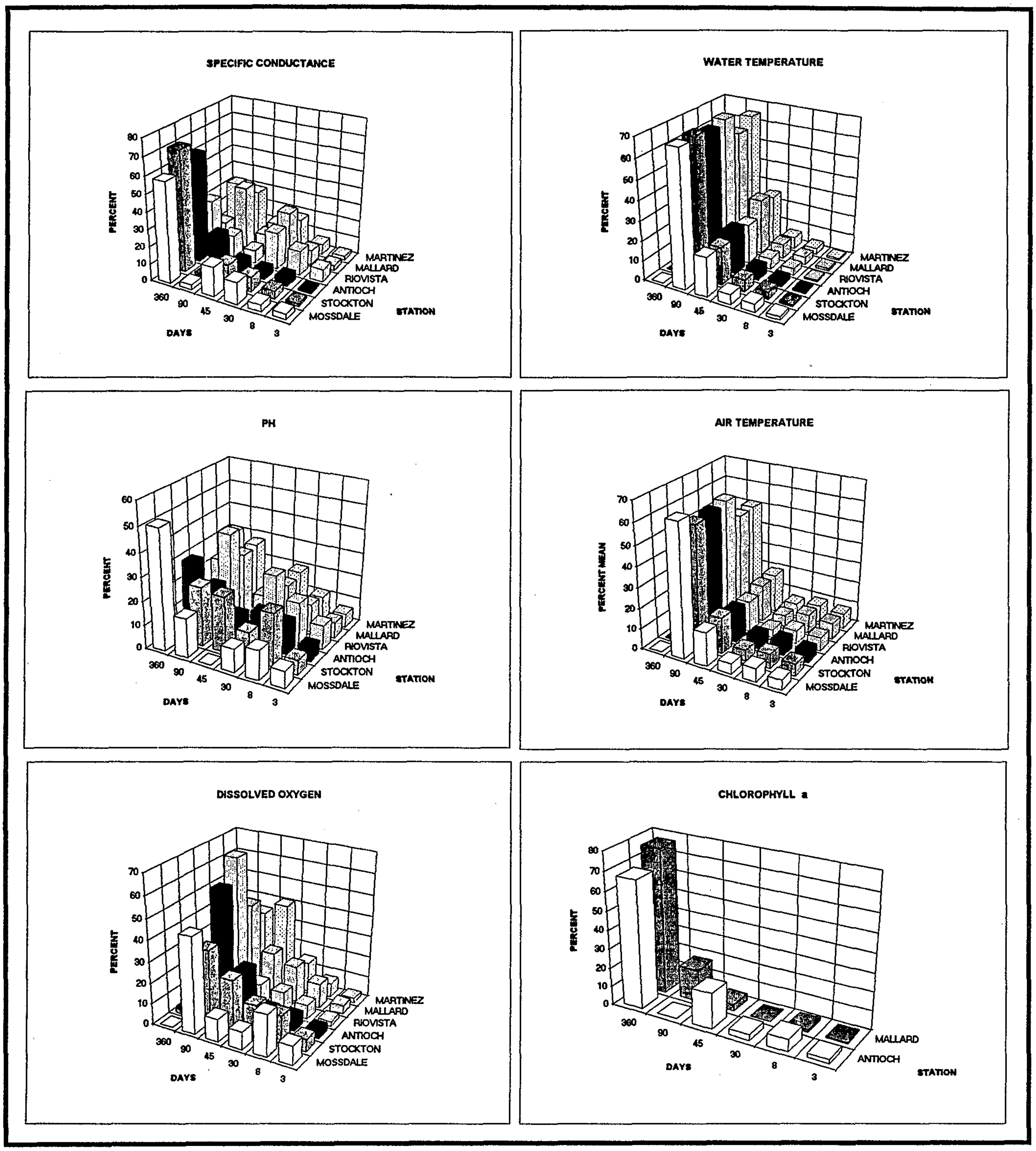

Figure 67
LONG-TERM TRENDS MEASURED BY THE CONTINUOUS MONITORING NETWORK

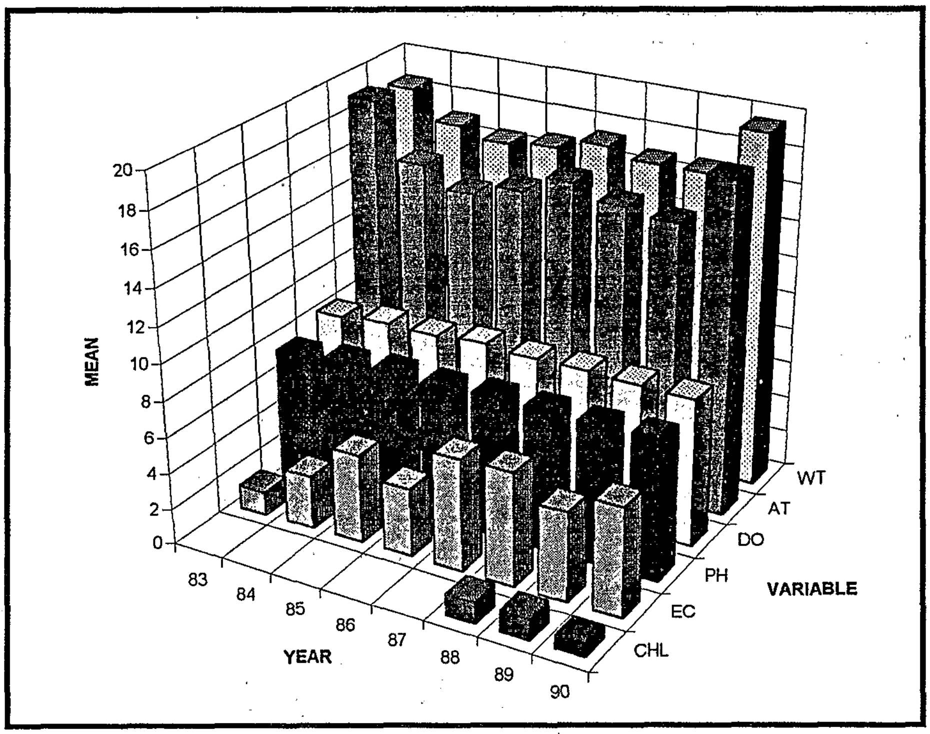

Figure 68
YEARLY MEANS FOR WATER QUALITY VARIABLES

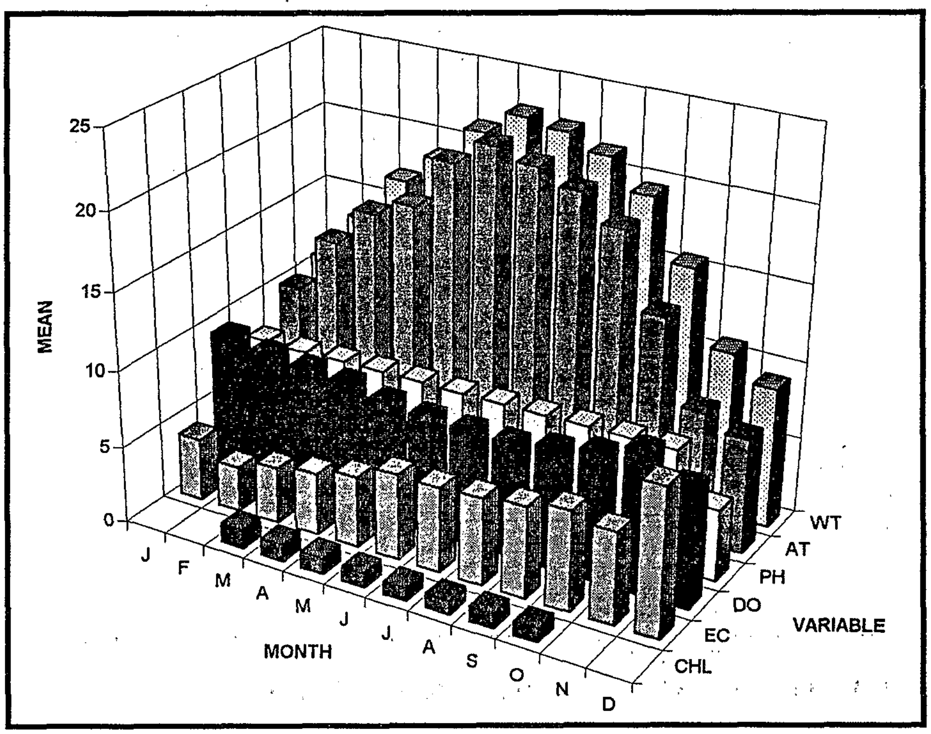

Figure 69
MONTHLY MEANS FOR WATER QUALITY VARIABLES

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Appendix A

PHYTOPLANKTON ANALYSIS MNEUMONICS WITH CORRESPONDING GENUS AND SPECIES

C-036586

1 — BACILLARIOPHYCEAE (DIATOMS)

ACHN	EXIG	Achnantes exigua
	GIBB	A. gibberula
	LANC	A. lanceolata
	LINE	A. linearis
	MINU	A. minutissima
ACTE		Actinella
ACTP		Actinoptychus
AMPC		Amphicampa
AMPL		Amphipleura
AMPO	COFF	Amphora coffaeiformis
	OVAL	A. ovalis
	NORM	A. normanii
AMPR	ORNA	Amphiprora ornata
	PALU	A. paludosa
ANOM		Anomoeoneis
ASTE	FORM	Asterionella formosa
ATTH	ZACH	Attheya zachariasi
BACI	PAXI	Bacillaria paxillifer
BIDD	LAEV	Biddulphia laevis
CALO	AMPB	Caloneis amphisbaena
CAMP		Campylodiscus
CENO		Centronella
CERT	ARCU	Ceratoneis arcus
CHAE	ELMO	Chaetocerus elmorei
COCN	DISC	Coconeis discula
	PLAC	C. placentula
CORE		Corethron
COSC	DENA	Coscinodiscus denarius
CYCL	BODA	Cyclotella bodanica
	COMT	C. comta
	GLOM	C. glomerata
	MENE	C. meneghiniana
	OCEL	C. oscellata
	STEL	C. stelligera
	STRI	C. striata
CYMA	SOLE	Cymatopleura solea
CYMB	LANC	Cymbella lanceolata
	MEXI	C. mexicana
	SINU	C. sinuata
	TUMI	C. tumida
	TURG	C. turgida
	VENT	C. ventricosa
DENT		Denticula
DIAT	VULG	Diatoma vulgare
	ELON	D. elongatum
	HIEM	D. hiemale
DIPL	BOMB	Diploneis bombus
	ELLI	D. elliptica
	SMIT	D. smithii
EPIT	SORE	Epithemia sorex
	TURG	E. turgida
	ZEBR	E. zebra
EUCA	ZOOD	Eucampia zoodiacus
EUNO	FORM	Eunotia formica
FRAG	ARCU	Fragilaria arcus
	BREV	F. brevistriata
	CONS	F. construens
	CROT	F. crotonensis
	INTE	F. intermedia
	PINN	F. pinnata
	VAUC	F. vaucheriae
FRUS	VULG	Frustulia bulgaris
GOMA	ACUM	Gomphonema acuminatum
	ANGU	G. angustatum
	PARV	G. parvulum
GOMS	HERC	Gomphoneis herculeana
GYRO		Gyrosigma
HANT	AMPX	Hantzschia amphioxys
HYDR	WHAM	Hydrosera whampoensis
ISTH	NERV	Isthmia nervosa
LICM		Licmophora
MAST	SMIT	Mastogloia smithii
MELO	AMBI	Melosira ambigua
	BIND	M. binderana
	GRAN	M. granulata
	HERZ	M. herzogii
	ISLA	M. islandica
	ITAL	M. italica
	VARI	M. varians

MERI	CIRC	Meridion circulare
NAVI	BACI	Navicula bacillum
	CUSP	N. cuspidata
	CRYP	N. cryptocephala
	HUNG	N. hungarica
	PUPU	N. pupula
	RADI	N. radiosa
NEID		Neidium
NITZ	ACIC	Nitzschia acicularis
	AMPH	N. amphibia
	DISS	N. dissipata
	FILI	N. filliformis
	LACU	N. lacunarum
	LINE	N. linearis
	LORE	N. lorenziana
	PALE	N. palea
	PARV	N. parvula
	SIGM	N. sigmoidea
	TRYB	N. tryblionella
	VERM	N. vermicularis
OEST	POWE	Oestrupia powelli
OPEP	MART	Qpephora martyi
PERO		Peroniopsis
PINN	SUDE	Pinnularia sudetica
PLEU		Pleurosigma
RHIZ	ERIE	Rhizoselenia eriensis
RHOI	CURV	Rhoicoshpenia curvata
RHOP	GIBB	Rhopalodia gibba
SKEL	COST	Skeletonema costatus
	POTA	S. potamos
STAS		Stauroneis
STEN		Stenopterobia
STEP	ASTR	Stephanodiscus astraea
	NIAG	S. niagarae
SURI	OVAT	Surirella ovata
SYNE	ACTI	Syndera actinastroides
	ACUS	S. acus
	CYCL	S. cyclopum
	FILI	S. filiformis
	INCI	S. incisa
	RADI	S. radians
	RUMP	S. rumpens
	TABU	S. tabulata
	ULNA	S. uina
	VAUC	S. vaucheriae
TABE	FENE	Tabellaria fenestrata
TERP		Terpsinoe
THAL		Thalassionema
THAS	ROTU	Thalassiosira rotula
	ECCE	T. eccentrica
THAT		Thalassionthrix
TROP		Tropidoneis

2 — CHLOROPHYCEAE (GREENS)

ACAN		Acanthosphaera
ACTI	GRAC	Actinastrum gracillimum
	HANT	A. hantzschii
ANKI	BRAU	Ankistrodesmus braunii
	CONV	A. convolutus
	FALC	A. falcatus
	APRI	A. spiralis
ARTO		Arthrodesmus
AULA	SUBM	Aulacomonas submarina
BOTR	SUDE	Botryococcus sudeticus
	BARU	B. braunii
CART	CORD	Carteria cordiformis
	KLEB	C. klebsii
	LUCE	C. lucerna
CHAR		Characium
CHAT		Chaetophora
CHLA	ANGU	Chlamydomonas angulosa
	GLOB	C. globosa
	DINO	C. dinobryoni
CHLG		Chlorogonium
CHLO		Chlorella
CHLR		Chlorosphaeralean
CHOD	LONG	Chodatella longiseta
	QUAD	C. quadriseta
CLAD		Cladophora
CLOM	SETA	Closterium setaceum
CLOP	LONG	Closteriopsis longissima
COCC	ORBI	Coccomonas orbicularis

C-036588

C-036588

COEL	MICR	Coelastrum microporum
COLE		Coleochaete
COSM		Cosmarium
CRUC	CRUC	Crucigenia crucigenioides
	IRRE	C. irregularis
	QUAD	C. quadrata
	TETR	C. tetrapedia
DACL	INFU	Dactylococcus infusionum
DICT	PULC	Dictyosphaerium pulchellum
DIDY	INCO	Didymocystis inconspicua
DIMO	LUNA	Dimorphococcus lunatus
DIPS		Diplostrauron
DISP	CRUC	Dispora crucigenioides
DRAP		Draparnaldia
ELAK	GEL	Elakatothrix gelatinosa
EUAS		Euastrum
EUDO	ELEG	Eudorina elegans
EXCE		Euastrum
FRAN	DROE	Franceia droescheri
GEMI		Geminella
GLOE		Gloeocystis
GOLE	RADI	Golenkinia radiata
GONI	SOCI	Gonium sociale
GONY	SEME	Gonystomum semen
HAEM		Haematococcus
HORM	SUBT	Hormidium subtile
HYAL		Hyalotheca
HYDT	RETI	Hydrodictyon reticulatum
JURA		Juranyiella
KIRC	OBES	Kirchneriella obesa
MESO		Mesotaenium
MICA		Micrasterias
MICR		Micractinium
MICS		Microspora
MONO		Monostroma
MOUG		Mougeotia
NEPH		Nephrocytium
OEDO		Oedogonium
OOCY	BORG	Oocystis borgei
	CRAS	O. crassa
	SUBM	O. submarina
OURO		Ourococcus
PALD		Palmodictyon
PALL	MINI	Palmellococcus miniatus
PALM		Palmella
PAND	MORU	Pandorina morum
PAUL		Paulschulzia
PEDI	BORY	Pediastrum boryanum
	DUPL	P. duplex
	OBTU	P. obtusum
	SIMP	P. simplex
	TETR	P. tetras
PENI		Penium
PHAT	LENT	Phacotus lenticularis
PHYT		Phytoconis
PITH		Pithophora
PLET		Pleurotaenium
POLY		Polyedriopsis
PROD		Protoderma
PTER	ACUL	Pteromonas aculeata
	ANGU	P. angulosa
PYRA	TETR	Pyramimonas tetrarhyncus
QUAD		Quadrigula
RADI		Radiococcus
RHIC		Rhizocionium
ROYA		Roya
SCEN	ABUN	Scenedesmus abundans
	ACUM	S. acuminatus
	ARCU	S. arcuatus
	ARMA	S. armatus
	BIJU	S. bijuga
	DIMO	S. dimorphus
	QUAD	S. quadricauda
SCHL		Schroederiella

SCHR	JUDA	Schroederia judayi
	SETI	S. setigera
SELE	MINU	Selenastrum minutum
SORA		Sorastrum
SPER	EXUL	Spermatozopsis exultans
SPHA	SCHR	Sphaerocystis schroeteri
SPHO		Sphaerozosma
SPIY		Spirogyra
SPON	PLAN	Spondylosium planum
STAD		Staurodesmus
STAM	PARA	Staurastrum paradoxum
	SEBA	S. sebaldi
STIG		Stigeoclonium
STYL		Stylosphaeridium
TEDS		Tetradesmus
TEEN	CAUD	Tetraedron caudatum
	MINI	T. minimum
	REGU	T. regulare
	TRIG	T. trigonum
TESM	ELEG	Tetrastrum elegans
	STAU	T. staurogeniaeforme
TETR		Tetraspora
TOMA	CATE	Tomaculum catenatum
TREU	SETI	Treubaria setigerum
TROC		Trochiscia
ULOT		Ulothrix
VOLV		Volvox
WEST	BOTR	Westella botryoides
WISL	PLAN	Wislouchiella planctonica
XANT		Xanthidium
ZYGN		Zygnema

3 — CHRYSOPHYCEAE (YELLOW-BROWNS)

CHRA		Chrysamoeba
CHRE		Chrysosphaerella
CHRM		Chromulina
CHRP	PLAN	Chrysocapsa planctonica
CHRU		Chrysococcus
CHRY	PARV	Chrysochromulina parva
DICE		Diceras
DINO	BAVA	Dinobryon bavaricum
	CYLI	D. cylindricum
	DIVE	D. divergens
	SERT	D. sertularia
KEPH	RUBR	Kephyrion rubrii
LAGY		Lagynion
MALL	ALPI	Mallomonas alpina
	CAUD	M. caudata
	PROD	M. producta
STIP		Stipitococcus
SYNU	UVEL	Synura uvella

4 — CRYPTOPHYCEAE (CRYPTOMONADS)

CHIL		Chilomonas
CHRO	NORD	Chroomonas nordstedtii
CRYP	EROS	Cryptomonas erosa
	OVAT	C. ovata
RHOD	LACU	Rhodomonas lacustris

5 — CYANOPHYCEAE (BLUE-GREENS)

AGME	ELEG	Agmenellum elegans
	TENU	A. tenuissima
ANAB	AFFI	Anabaena affinis
	CIRC	A. circinalis
	FLOS	A. flos-aquae
	HELI	A. helicoidea
	OSCI	A. oscillariodes
	SPHA	A. sphaerica
	SPIR	A. spiroides
ANAC	CYAN	Anacystis cyanea
	DIMI	A. dimidiata
	INCE	A. incerta
	LIMN	A. limneticus
	MARG	A. marginata
	NIDU	A. nidulans
	THER	A. thermalis
ANAP	ELEN	Anabaenopsis elenkinii
APHA	FLOS	Aphanizomenon flos-aquae
ARTH		Arthrospira
COCH		Coccochloris
CYAN	HAMI	Cyanarcus hamiformis
DACT		Dactylococcopsis
GLOT		Gloeotrichia
GOMI		Gomphosphaeria
LYNG	BIRG	Lyngbya birgei
MARS	ELEG	Marssoniella elegans
NOST		Nostic
OSCI		Oscillatoria
PHOR		Phormidium
PSEU		Pseudanabaena
RHAB	SIGM	Rhabdoderma sigmoidea
RIVU		Rivularia
SCHZ		Sshizothrix
SPIL	MAJO	Spirulina major
STAU		Staurocladia
STOG		Stigonema
SYNY		Synechocystis
TRIC		Trichodesmium

6 — DINOPHYCEAE (DINOFLAGELLATES)

CERA	CORN	Ceratium cornutum
	HIRU	C. hirundinella
CYST		Cystodinium
GLEN	GYMN	Glenodinium gymnodinium
	QUAD	G. quadridens
GYMN		Gymnodinium
GYRD		Gyrodinium
HEMI		Hemidinium
MASS		Massartia
PERI	ACIC	Peridinium aciculiform
	WILL	P. willei
	WISC	P. wisconsinense
	LIMB	P. limbatum
	BIPE	P. bipes

7 — EUGLENOPHYCEAE (EUGLENOIDS)

COLA		Colacium
EUGL	DESE	Eugiena deses
LEPO		Lepocinclis
PERA		Peranema
PHAS	NORD	Phacus nordstedtii
TRAC	HISP	Trachelomonas hispida

8 — UNIDENTIFIED FLAGELLATES

FLAG		Unidentified flagellates

9 — XANTHOPHYCEAE (YELLOW-GREENS)

ARAC	MINO	Arachnochloris minor
BOTD		Botrydiopsis
BUMA		Bumilleria
BUMP		Bumilleriopsis
CENT		Centritractus
CHLT		Chloroallanthus
GONO	SCUL	Goniochloris sculpta
HARP		Harpochytrium
OCHR		Ochromonas
OPHI		Ophiocytium
TRAY	BICO	Trachychloron biconicum
TRIB		Tribonema
VAUC		Vaucheria

0 — RHODOPHYCEAE (REDS)

AUDO		Audouinella
BACT		Bactrachospermum
COMP		Compsopogon
RHOC		Rhodochorton
POLS		Polysiphonia

Appendix B

DELTA BENTHOS 1993 SPECIES LIST

ANNELIDA

Branchiobdellidae	Cambarincola species a
Capitellidae	Heteromastus filiformis
Cirratulidae	Cirriformia spirabrancha
Enchytraeidae	UID enchytraeid species a UID enchytraeid species b
Erpobdellidae	Mooreobdella microstoma
Glossiphoniidae	Helobdella stagalis H. triserialis Placobdella montana
Glossoscolecidae	Sparganophilus eiseni
Goniadidae	Glycinde armigera
Lumbriculidae	Lumbriculus variegatus L. species a
Lumbrineridae	Lumbrineris species a
Maldanidae	Asychis elongata
Megascolecidae	UID megascolecid species a
Naididae	Bratislava bilongata Chaetogaster diaphanus C. limnaei Dero digitata D. trifida Nais communis / variabilis N. pardalis N. pseudobtusa N. simplex N. elinguis Ophidonais serpentina Paranais frici Pristina brevista P. leidyi Slavina appendiculata Stylaria lacustris Vejdovskyella comata V. intermedia
Nephtyidae	Nephtys caecoides N. cornuta franciscana
Nereidae	Nereis limnicola N. procera N. succinea
Orbiniidae	Haploscoloplos elongatus
Phyllodocidae	Eteone californica E. lighti
Polynoidae	Harmothoe imbricata
Sabellidae	Fabricia berkeleyi Manayunkia speciosa Potamilla species a
Spionidae	Boccardia ligerica Polydora ligni Pseudopolydora kempi UID spoinid species a Streblospio benedicti
Syllidae	Sphaerosyllis californiensis
Tubificidae	Aulodrilus limnobius A. pigueti A. pluriseta Bothrioneurum vejdovskyanum Branchiura sowerbyi Hyodrilus frantzi capillatus I. templetoni Limnodrilus hoffmeisteri L. udekemianus Potamothrix bavaricus Psammoryctides californianus Quistadrilus multisetosus Spirosperma ferox Teneridrilus mastix UID tubificid species a Tubificoides brownae T. fraseri T. species a Varichaetadrilus angustipenis

ARTHROPODA

Ampeliscidae	Ampelisca abdita
Asellidae	Asellus occidentalis
Astacidae	Pacifistacus leniusculus
Baetidae	Baetis bicaudatus
Balanidae	Balanus improvisus
Caenidae	Caenis amica
Callianassidae	Upogebia pugettensis
Candonidae	Canona species a
Caprellidae	Caprella species a
Ceratopogonidae	Palpomyia species a
Chaoboridae	Chaoborus albatus

Chironomidae	Ablabesmyia species a Chironomus attenuatus C. species a Cladotanytarsus species a Cricotopus bicinctus C. species a Cryptochrinonomus species a C. species b Demicryptochironomus species a Einfeldia species a Endochironomus species a E. species b Epoicocladius species a Harnischia curtilamellata Micropsectra species a Monodiamesa species a Nanocladius distinctus N. species a Nimbocera species a Parachironomus species a Paracladopelma species a Paralauterborniella species a Paratendipes species a Paratanytarsus species a Phaenopsectra species a Polypedilum species a Procladius species a Psectrocladius species a Robackia claviger Stenochironomus species a Stictochironomus species a Tanytarsus species a Tanypus stellatus
Coenagrionidae	Zoniagrion exclamationis
Corixidae	Corisella inscripta Trichocoixa verticalis
Corophiidae	Corophium acherusicum C. alienense C. insidiosum C. oaklandense C. spincorne C. stimpsoni C. heteroceratum Grandidierella japonica
Crangonidae	Crangon franciscorum
Cylindroleberididae	Sarsiella zostericola
Cyprididae	Eucypris species a Herpetocypris brevicaudata Isocypris species a Cyprideis species a
Ephemeridae	Hexagenia limbata californica
Gammaridae	Elasmopus antennatus Gammarus diaberi Melita nitida
Gomphidae	Gomphus olivaceus
Grapsidae	Herigrapsus nudus
Heptageniidae	Heptagenia rosea
Hydropsychidae	Hydropsyche species a
Hydroptilidae	Hydroptila species a Oxyethira species a
Idoteidae	Synidotea laticauda
Leptoceridae	Nectopsyche gracilis Oecetis species a
Leptophlebiidae	Paraleptophlebia species a
Leuconidae	Hemileucon hinumensis
Limnesiidae	Limnesia species a
Majindae	Pyromaia tuberculata
Munnidae	Munna species a
Mysidae	Neomysis mercedis
Nannastacidae	Cumella vulgaris
Naucoridae	Ambrysus species a
Palaemonidae	Palaemon macrodactylus
Phoxocephalidae	Paraphoxus milleri
Pionidae	Forelia species a
Pleustidae	Parapleustes pugettensis
Sphaeromatidae	Gnorimosphaeroma insulare G. oregonesis Sphaeroma pentodon
Talitridae	Hyalella azteca
Tanaidae	Tanais species a
Tricorythidae	Tricorythodes minutus
Unionicolidae	Unionicola species a U. species b
Xanthidae	Rhithoropanopeus harrisii

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CHORDATA

Molgulidae	Molgula manhattensis

CNIDERIA

Hydridae	Hydra species a

MOLLUSCA

Anclyidae	Ferrissia rivularis
Assimineidae	Assiminea californica
Corbiculidae	Corbicula fluminea
Corbulidae	Potamorcorbula amurensis
Myidae	Mya arenaria
Mytilidae	Musculista senhousia Mytilus edulis
Physidae	Physa gyrina
Planorbidae	Gyraulus species a G. species b
Pyramidellidae	Odostomia fetella
Semelidae	Theora lubrica
Sphaeriidae	Pisidium casertanum P. compressum Sphaerium species a
Tellinidae	Macoma balthica
Thiardae	Melanoides tuberculata
Thiardae	Melanoides tuberculata
Unionidae	Anodonta wahlamatensis
Unknown	Nudibranch species a
Veneridae	Gemma gemma Protothaca staminea

NEMATODA

Dorylaimidae	UID actinolaiminae species a Dorylaimus species a
Plectidae	Teratocephalus species a
Unknown	UID nematoda species a

NEMERTEA

Tertastemmatidae	Prostoma graecense
Unknown	UID nemertean species a UID paleonemertean species a

PLATYHELMINTHES

Planariidae	Dugesia tigrina
Unknown	UID species a UID species b UID species c UID species microturbellarian species a

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