

### 3. THE MEASUREMENT OF WATER AND SEDIMENT QUALITY IN GALVESTON BAY

#### 3.1 Water and Sediment Quality Parameters

The quantification of the quality of water and sediment in an estuary is accomplished by determination of a suite of parameters, some of which are indicator variables, such as coliforms and BOD, some of which are constituents which *per se* have major rôles in biochemical processes, such as process compounds, e.g. nitrogen and phosphorus species, or toxic contaminants such as PAH's and pesticides, and some of which serve in both capacities, such as salinity. This study focused on the following categories of parameters:

- temperature,
- salinity and related parameters,
- suspended sediments and turbidity,
- pH,
- dissolved oxygen,
- nutrients, *viz.* nitrogen, phosphorus and organic carbon,
- organics as measured by oil & grease, volatile solids and biochemical oxygen demand,
- chlorophyll-a and phaeophytin,
- coliforms,
- metals (total and dissolved), and
- trace organics, including pesticides, herbicides, PAH's, PCB's, and priority pollutants

Temperature, salinity and pH have been routinely measured in the field for some time, therefore for point measurements, the data base is most extensive for these variables. Temperature and salinity, moreover, exhibit considerable variability, temperature due to the local heat-exchange processes at the surface, and salinity due to watermass movement within the estuary in conjunction with high spatial gradients. Generally, pH exhibits less variability, due to the high buffering capacity of seawater, but for this reason departures from the range 7-9 are especially significant.

Dissolved oxygen (DO) is the traditional and ubiquitous indicator of aquatic health. Biochemical oxygen demand (BOD), oil & grease, and volatile solids are tests which have developed from situations dominated by oxygen-demanding pollutants, and while their merit as water-pollutant parameters continues to be debated, the fact is that these parameters enjoy the longest period of record in most aquatic systems. Trace metals and pesticides are more recent arrivals, whose utility continues to be vexed by uncertain analytical procedures.

Finally, the EPA Priority Pollutant List, per Section 307 of the Clean Water Act, is a means to specify which individual compounds are to be given detailed study due to their high pollutant or toxicological potential. Table 3-1 presents the EPA

TABLE 3-1  
PRIORITY POLLUTANTS

<i>General Contaminants</i>				
Asbestos				
Total Cyanide				
Total Phenols				
<i>Metals</i>		<i>Nominal Detection Limits (mg/L)</i>		
Arsenic (As)		5		
Cadmium (Cd)		5		
Chromium (Cr)		20		
Lead (Pb)		100		
Mercury (Hg)		0.2		
Selenium (Se)		5		
Silver (Ag)		25		
Antimony (Sb)		30		
Beryllium (Be)		5		
Copper (Cu)		20		
Zinc (Zn)		5		
Nickel (Ni)		40		
Thallium (Tl)		10		
<i>Organics - Volatiles</i>		<i>Method detection limits</i>		
	<u>624</u>	<u>524.1</u>	<u>601</u>	<u>602</u>
Acrolein	100	20.0		
Acrylonitrile	100	20.0		
Benzene	4.4			0.2
Bromoform	4.7	0.94	0.20	
Carbon tetrachloride	2.8	0.56	0.12	
Chlorobenzene	6.0	1.20	0.25	0.2
Chlorodibromomethane	3.1	0.62	0.09	
Chloroethane	10	2.0	0.52	
2-Chloroethylvinyl ether	10	2.0	0.13	
Chloroform	1.6	0.32	0.05	
Dichlorobromomethane	2.2	0.44	0.10	
1, 1-Dichloroethane	4.7	0.94	0.07	
1, 2-Dichloroethane	2.8	0.56	0.03	
1, 1-Dichloroethylene	2.8	0.56	0.13	
1, 2-Dichloropropane	6.0	1.20	0.04	
cis-1, 3-Dichloropropylene	5.0	1.00	0.34	
Ethylbenzene	7.2	1.44		0.2
Methyl bromide	10	2.00	1.18	
Methyl chloride	10	2.00	0.08	
Methylene chloride	2.8	0.56	0.25	

TABLE 3-1  
(continued)

<i>Organics - Volatiles (continued)</i>	<i>Method detection limits</i>			
	<u>624</u>	<u>524.1</u>	<u>601</u>	<u>602</u>
1, 2, 2, 2-Tetrachloroethane	6.9	1.38	0.03	
Tetrachloroethylene	4.1	0.82	0.03	
Toluene	6.0	1.20		0.2
1, 2-Trans-dichloroethylene	1.6	0.32		
1, 1, 1-Trichloroethane	3.8	0.76	0.03	
1, 1, 2-Trichloroethane	5.0	1.00	0.02	
Trichloroethylene	1.9	0.38	0.12	
Vinyl Chloride	10	2.00	0.18	
<i>Organics - Pesticides</i>	<i>Method detection limits</i>			
	<u>625</u>	<u>608</u>		
Aldrin	1.9	0.004		
Alpha-BHC	10	0.003		
Beta-BHC	4.2	0.004		
Gamma-BHC (Lindane)	10	0.004		
Delta-BHC	3.1	0.004		
Chlordane	10	0.014		
P, F-DDT	4.7	0.012		
P, F-DDE	5.6	0.004		
P, F-DDD	2.8	0.011		
Dieldrin	2.5	0.002		
Endosulfan I	10	0.014		
Endosulfan II	10	0.004		
Endosulfan sulfate	5.6	0.066		
Endrin	10	0.006		
Endrin aldehyde	10	0.023		
Heptachlor	1.9	0.003		
Heptachlor epoxide	2.2	0.24		
PCB-1242	36	0.065		
PCB-1254	36	0.065		
PCB-1221	30	0.065		
PCB-1232	36	0.065		
PCB-1248	36	0.065		
PCB-1260	36	0.065		
PCB-1016	36	0.065		
Toxaphene	10	0.24		

TABLE 3-1  
(continued)

*Organics - Base-Neutrals with Method 625 detection limits*

Acenaphthene	1.9	Diethyl phthalate	1.9
Acenaphthylene	3.5	Dimethyl phthalate	1.6
Anthracene	1.9	Di-n-butyl phthalate	2.5
Benzidine	44	2, 4-Dinitrotoluene	5.7
Benzo (a) anthracene	7.8	2, 6-Dinitrotoluene	1.9
Benzo (a) pyrene	2.5	Di-n-octyl phthalate	2.5
3, 4-Benzofluoranthene	4.8	1, 2-Diphenylhydrazine	10
Benzo (ghi) perylene	4.1	Fluoranthene	2.2
Benzo (k) fluoranthene	2.5	Fluorene	1.9
bis (2-Chloroethoxy) methane	5.3	Hexachlorobenzene	1.9
bis (2-Chloroethyl) ether	5.7	Hexachlorobutadiene	0.9
bis (2-Chloroisopropyl) ether	5.7	Hexachlorocyclopentadiene	10
bis (2-Ethylhexyl) phthalate	2.5	Hexachloroethane	1.6
4-Bromophenyl phenyl ether	1.9	Indeno (1, 2, 3-c,d) pyrene	3.7
Butyl benzyl phthalate	2.5	Isophorone	2.2
2-Chloronaphthalene	1.9	Naphthalene	1.6
4-Chlorophenyl phenyl ether	4.2	Nitrobenzene	1.9
Chrysene	2.5	N-Nitrosodimethylamine	10
Dibenzo (a,h) anthracene	2.5	N-Nitrosodi-n-propylamine	10
1, 2-Dichlorobenzene	1.9	N-Nitrosodiphenylamine	1.9
1, 3-Dichlorobenzene	1.9	Phenanthrene	5.4
1, 4-Dichlorobenzene	4.4	Pyrene	1.9
3,3'-Dichlorobenzidine	16.5	1, 2, 4-Trichlorobenzene	1.9

*Organics - Acid with Method 625 detection limits*

2-Chlorophenol	3.3	4-Nitrophenol	2.4
2, 4-Dichlorophenol	2.7	p-Chloro-m-cresol	3.0
2, 4-Dimethylphenol	2.7	Pentachlorophenol	3.6
4, 6-Dinitro-o-cresol	24	Phenol	1.5
2, 4-Dinitrophenol	42	2, 4, 6-Trichlorophenol	2.7
2-Nitrophenol	3.6		

Priority Pollutants, a list which continues to mutate as constituents are listed and de-listed by EPA. While data for all of these were sought as a part of this study, generally for only a small minority are there sufficient data to even allow statistical judgements to be made.

Complete tabulations of the water quality and sediment quality parameters of this study are given in Tables A-1 and A-2 of the appendix, including sources of data, and uncertainty measures.

### **3.2 Relationships between Parameters**

In estuarine water quality, there are several classes of parameters that measure (or can be interpreted to measure) the same essential property. For example, salinity can be estimated from measurements of chlorides concentration, total dissolved solids, density, conductivity, and light refraction. Different data collection programs in the Bay may employ different measures, depending upon objective, convenience and tradition.

The relations between parameters are considered here, for two purposes. First, from an analytical viewpoint, the use of one parameter may have conceptual advantages over another, e.g. DO deficit may be more indicative of oxygen conditions than the concentration of dissolved oxygen itself. Second, while related parameters are technically distinct, the fact that they can be associated and may be converted from one to another means that a much denser and longer-duration data set can be compiled by converting these to a common parameter. To establish whether such relations are justified for a given class of parameters is a central step in a long-term trend analysis. These are referred to as "proxy" relationships, and the creation of proxy data sets is treated in the following chapter as an element of data processing. Here we present that basis (or lack of) for the formulation of such a relationship. In particular, we address the parameters salinity, BOD, DO, DDT, turbidity, nitrogen and phosphorus, and coliforms.

#### *3.2.1 Salinity*

Salinity is one of the quintessential quality elements of estuarine waters, being determined fundamentally by the intermixing of fresh and oceanic waters. As a virtually conservative parameter, easily measured, and ubiquitous, it is an excellent watermass tracer. It is also a key ecological indicator, as it affects the suitability of habitat due to varying osmoregulation capabilities of organisms. Salinity further affects many chemical reactions and sedimentation. Any direct impact on salinity has the potential of indirect consequences for ecosystem structure and function.

Since there are large spatial gradients in salinity and it exhibits high temporal variability, for work in estuaries a lower degree of precision in salinity determination can be accepted than the case either in totally fresh or oceanic

systems. This means that data can be employed from a variety of protocols and parameters.

One of the most common methods of salinity measurement is via conductivity. (In fact, in oceanography, the new practical salinity scale *defines* salinity in terms of conductivity.) Conductivity is a strong function of temperature so the temperature at which the measurement applies is essential. Generally, a reported conductivity will be either at ambient temperature (i.e., the temperature of the water when the measurement was taken) or compensated to a standard reference temperature of 25°C. (Modern inexpensive conductivity meters perform this compensation internally.) For a fixed temperature, conductivity varies nearly linearly with salinity. A regression based upon the data of USNHO (1956) is

$$\begin{aligned} S &= 0.000588 \cdot C && \text{for } C < 17,000 \text{ } \mu\text{mhos} \\ S &= 0.000679 \cdot C - 1.543 && \text{for } C > 17,000 \text{ } \mu\text{mhos} \end{aligned} \tag{3-1}$$

where C is conductivity at 25°C and S is salinity in ‰.

While most of the field data from Texas is compensated to 25°C, occasionally some are encountered at ambient temperature, requiring a means to correct conductivity at an arbitrary temperature to that at 25°C. A parabola was fitted to the data in *Standard Methods* (APHA, 1971, 1985), yielding:

$$f(\delta T) = 0.0002286 \cdot \delta T^2 - 0.02114 \cdot \delta T + 1 \tag{3-2}$$

Here  $\delta T = T - 25$ , where T is the ambient temperature in °C, and  $f(\delta T)$  is the ratio of conductivity at 25° C to that at ambient, i.e. the factor one would multiply the ambient conductivity by to convert it to conductivity at 25° C.

Salinity originally measured the dissolved solids in seawater, which are dominated by halogen salts. A simpler measure was to determine the salts of a single halogen, *viz.* chlorine, and employ the empirical law of constant proportions (Forchhammer's Law). The relation between salinity and chlorinity based upon early work of Knudsen is approximately (Defant, 1961, Wallace, 1974)

$$S = 0.03 + 1.805 \cdot Cl$$

for S and C in ‰. A century later, this relation was re-evaluated as

$$S = 1.807 \cdot Cl \tag{3-3}$$

Certainly to the accuracy necessary for estuarine work, this is a satisfactory means of interconverting.

While the conventional viewpoint is to regard density as a fundamental physical property that varies as a function of temperature and salinity, in the present context we regard density (and specific gravity) as an alternative measure of

salinity and seek a relationship by which salinity can be expressed as a function of density. Again, the oceanic relation is basic. The equation of state for seawater is empirical, and has most recently (UNESCO, 1981) been expressed as a best-fit multinomial with 15 coefficients. For present purposes, we retain only the higher-order terms, to obtain the approximate relation:

$$S = \frac{\rho - (a_1 + b_1T + a_2T^2)}{(a_4 + b_4T + a_5T^2)} \quad (3-4)$$

where

$$\begin{aligned} a_1 &= 999.8426 & b_1 &= 6.794 \times 10^{-2} \\ a_2 &= -9.0953 \times 10^{-3} & a_4 &= 8.245 \times 10^{-1} \\ a_5 &= 7.644 \times 10^{-5} & b_4 &= -4.090 \times 10^{-3} \end{aligned}$$

for salinity S in parts per thousand, temperature T in degrees Celsius, and density  $\rho$  in kg/m<sup>3</sup>. This approximation is more than adequate for the accuracy necessary in estuary work, as evidenced by the numerical comparison in Table 3-2 between the complete empirical equation of state and that given by the above equation.

One additional measure of salinity is the refractive index of water. The field instrument used for this purpose is a portable refractometer that is calibrated for a direct read-out of salinity (the Goldberg refractometer). For present purposes, therefore, the conversion is unnecessary, but in the data processing procedures of Chapter 4 note is made of when this methodology is employed, for establishing a level of uncertainty in the data.

Generally, in the field data from Galveston Bay, one of the above measures is employed for determination of salinity, so the only decision available in analyzing the data is the proper conversion. On occasion, there is a choice. In the case of the Galveston Bay Project data (see Section 3.3), both laboratory titrations and conductivity determinations were carried out, so this offers a means of directly comparing alternative methodologies over a wide range of variation. Because chlorides were measured rather than salinity, the above relations must be combined to relate conductivity and chlorinity, *viz.*

$$C = \begin{cases} 2661 \cdot Cl + 2272 & Cl > 5.5 \text{ ‰} \\ 3073 \cdot Cl & Cl < 5.5 \text{ ‰} \end{cases} \quad (3-5)$$

The conductivity and chlorinity data from the Galveston Bay Project are plotted in Figs. 3-1 through 3-4. The Galveston Bay Program was initiated in 1968 and immediately placed a heavy workload of samples on the City of Houston laboratory performing the water chemistry. Further, many of the parameters were foreign to the routine of the lab personnel and required a period of familiarization. The variation in lab performance is indicated by the separate years of data shown in Figs. 3-1 *et seq.* The solid line in these figures is the relation given by (3-5), a relation which should be expected to hold except for the very lowest values of salinity. While this line provides an excellent fit to the data, there is considerable

TABLE 3-2

Numerical variation of computed density with temperature and salinity

Temperature (°C)	Salinity (ppt)						
	0	5	10	15	20	25	30
<i>International equation of state for seawater</i>							
0	999.843	1003.913	1007.955	1011.986	1016.014	1020.041	1024.072
5	999.967	1003.949	1007.907	1011.858	1015.807	1019.758	1023.714
10	999.702	1003.612	1007.501	1011.385	1015.269	1019.157	1023.051
15	999.102	1002.952	1006.784	1010.613	1014.44	1018.279	1022.122
20	998.206	1002.008	1005.793	1009.576	1013.362	1017.154	1020.954
25	997.048	1000.809	1004.556	1008.301	1012.050	1015.806	1019.569
30	995.651	999.380	1003.095	1006.809	1010.527	1014.252	1017.985
35	994.036	997.740	1001.429	1005.118	1008.810	1012.509	1016.217
<i>Lower-order approximation [Equation (3-4)]</i>							
0	999.843	1003.965	1008.088	1012.210	1016.332	1020.455	1024.577
5	999.955	1003.985	1008.014	1012.044	1016.074	1020.104	1024.134
10	999.612	1003.569	1007.525	1011.481	1015.437	1019.393	1023.350
15	998.815	1002.717	1006.619	1010.520	1014.422	1018.324	1022.226
20	997.563	1001.430	1005.296	1009.162	1013.029	1016.895	1020.761
25	995.857	999.707	1003.557	1007.405	1011.257	1015.107	1018.957
30	993.695	997.548	1001.401	1005.254	1009.107	1012.960	1016.813
35	991.079	994.954	998.829	1002.703	1006.578	1010.453	1014.328

scatter about the line, despite the nominal precision in the measurements of  $\pm 0.1$  ‰.

In the 1968 data (Fig. 3-1), at project initiation, there are several freak points, and there is a systematic departure of the data from the relation (3-5) for chlorinities above about 12‰. A close examination of the data led us to conclude that many of the conductivity measurements are in error, and that the errors originated in the lab rather than in data entry. That the conductivities should be so frequently in error is not surprising given the nature of conductivity meters in the mid-1960's, which were difficult to read with multiple scales on the same readout, and were subject to drift and calibration problems. Most of the freak points appeared to be the result of misplaced decimals or dropped leading digits. Further, the meter appears to have been reading systematically low, with an increasing error at the higher chlorinities, which would have been near or at the upper limit of the range of the conductivity meter. By 1969 (Fig. 3-2), this bias seems to have been substantially reduced (perhaps, the meter was serviced, or a more modern instrument acquired). But the occasional errant conductivity value is still apparent. Accordingly, we used the chlorinity data preferentially for the 1968-70 GBP data. In 1971, the GBP program suffered a hiatus of several months, then resumed with fewer stations and a reduced suite of parameters. More significantly for present purposes, the lab was changed, with the samples now being sent to a commercial lab for analysis. The data from this period, Fig. 3-4, display different characteristics, in which the conductivity data are better behaved, but now the occasional aberrant values appear in the chlorides. Therefore, for the 1971-72 data, we employed the conductivity data preferentially.

In the TWC Statewide Monitoring Network (SMN) data (see Section 3.3, below), both field and laboratory conductivity measurements may be available for a given sample, and occasionally there may be a laboratory determination of chlorides as well. To test the above relations, as well as to clarify the variability and relation among these different parameters, we focused on those data records in which all three variables were measured, and further restricted the analysis to the bay samples only, so as to have a wider range of salinity values and increased leverage on the regressions. The plot of field versus laboratory conductivity shown in Fig. 3-5 does not create a warm feeling for the precision of the conductivity measurement. Part of this widespread discrepancy is due to degraded accuracy in the laboratory determinations, and part is due to the fact that a significant proportion of the reported laboratory values are not really measurements, but are "supplied data."

We first examine the latter. Fig. 3-6 displays a plot of laboratory measured conductivity versus chlorides. There is considerable scatter, surprising because both measurements are made in the laboratory, presumably to the precision intrinsic in both types of analyses: we would have expected better agreement. More significantly, there seem to be two separate regressions, some of the data falling along a line with almost zero scatter. This line proved to be the relation  $y = x \cdot 15/4$ . The conclusion is inescapable: for this subset of the data, only one of chlorides or conductivity was actually measured, and the other was computed based upon the (incorrect) rule that conductivity is 15/4 times chlorides. In this

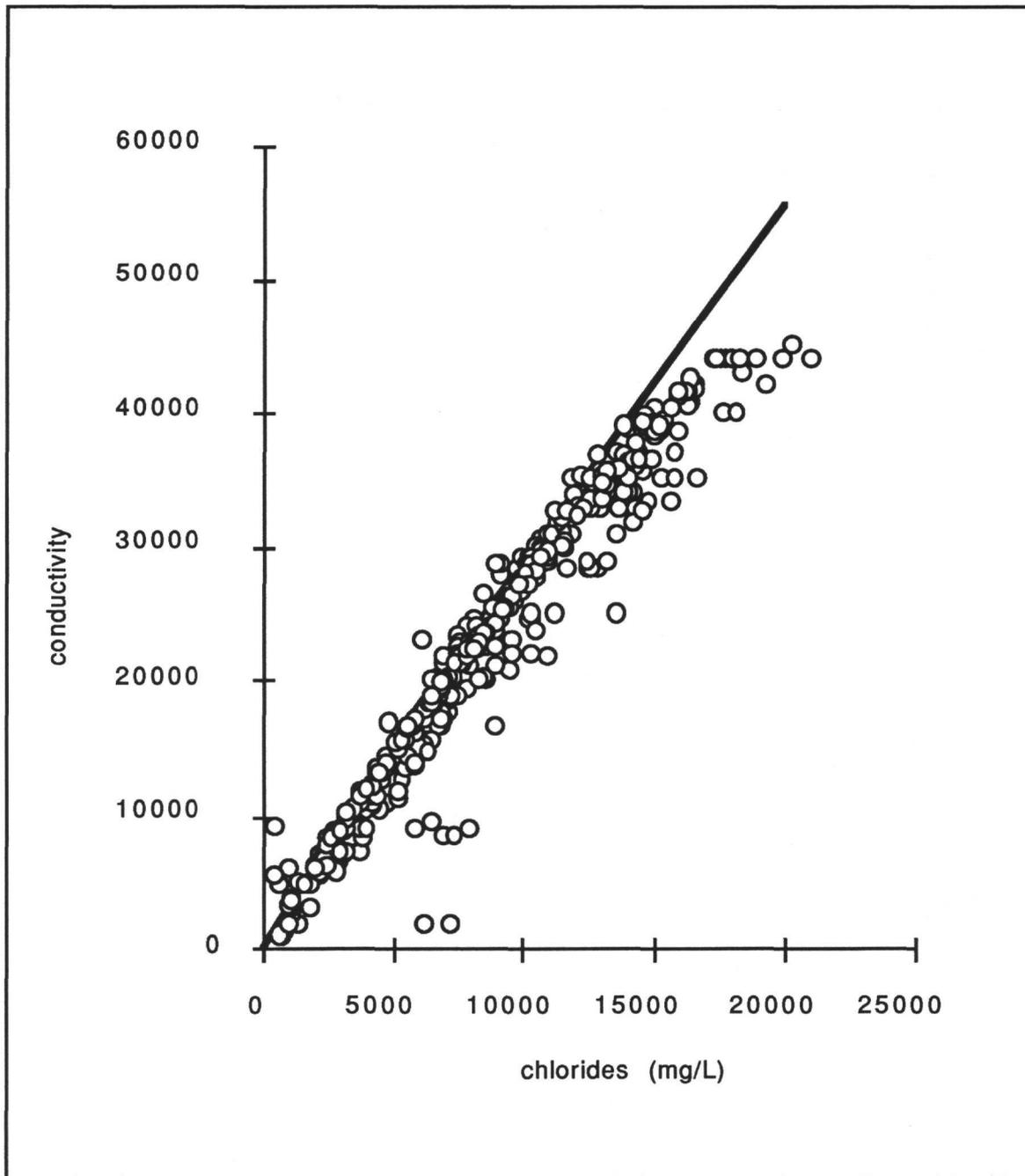


Fig. 3-1. Scatterplot of conductivity versus chlorides from 1968 data of Galveston Bay Project

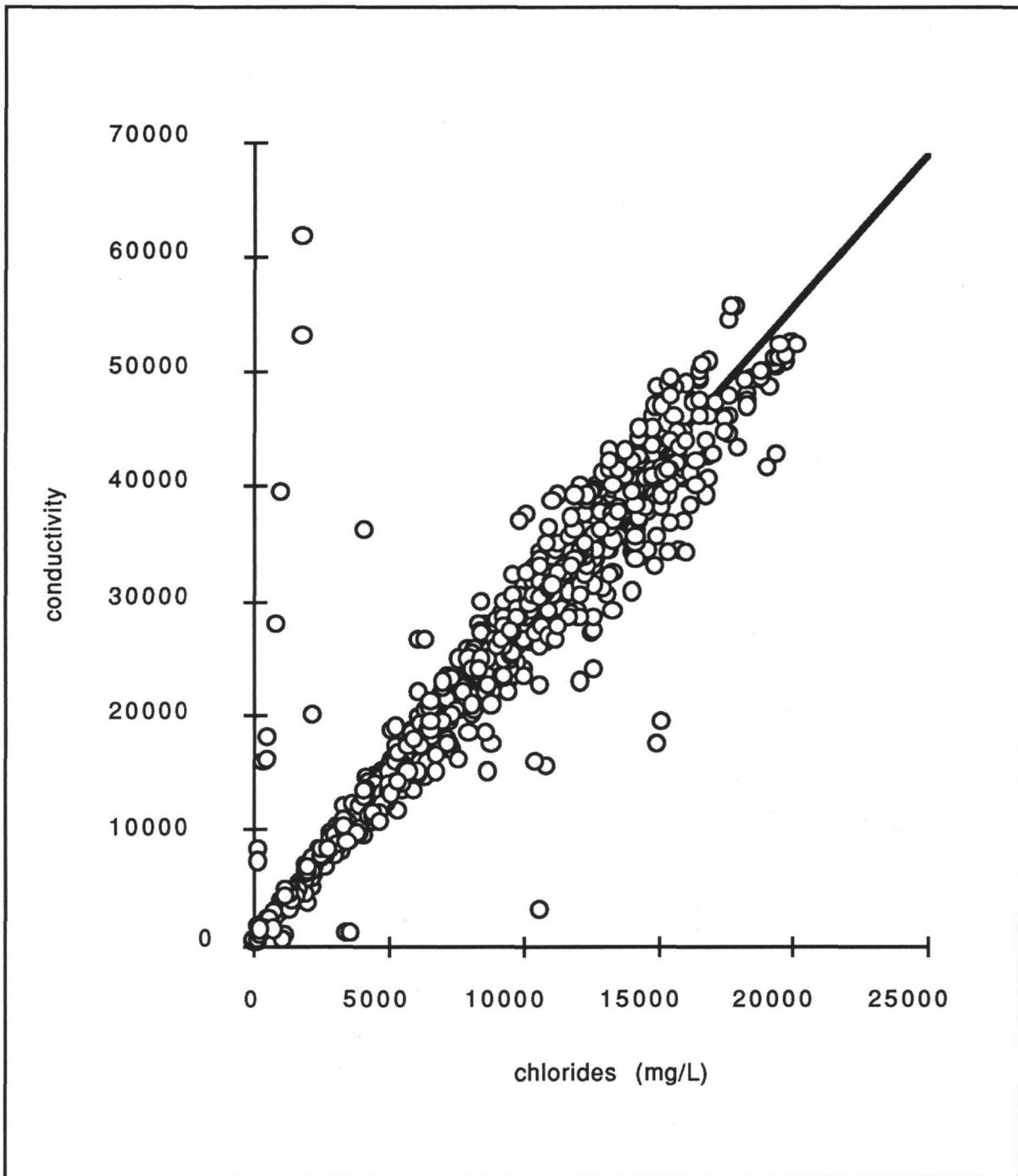


Fig. 3-2. Scatterplot of conductivity versus chlorides from 1969 data of Galveston Bay Project

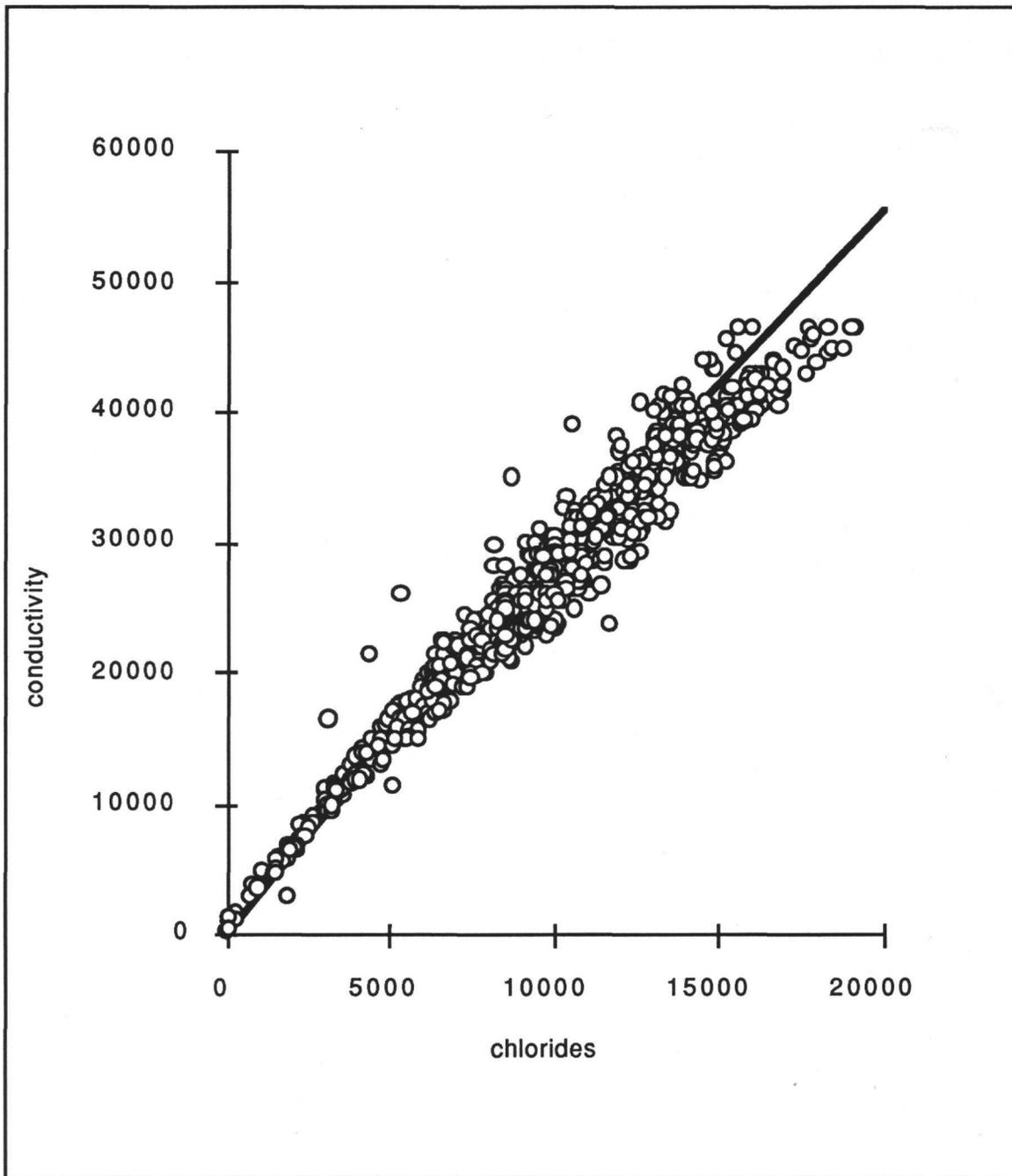


Fig. 3-3. Scatterplot of conductivity versus chlorides from 1970 data of Galveston Bay Project

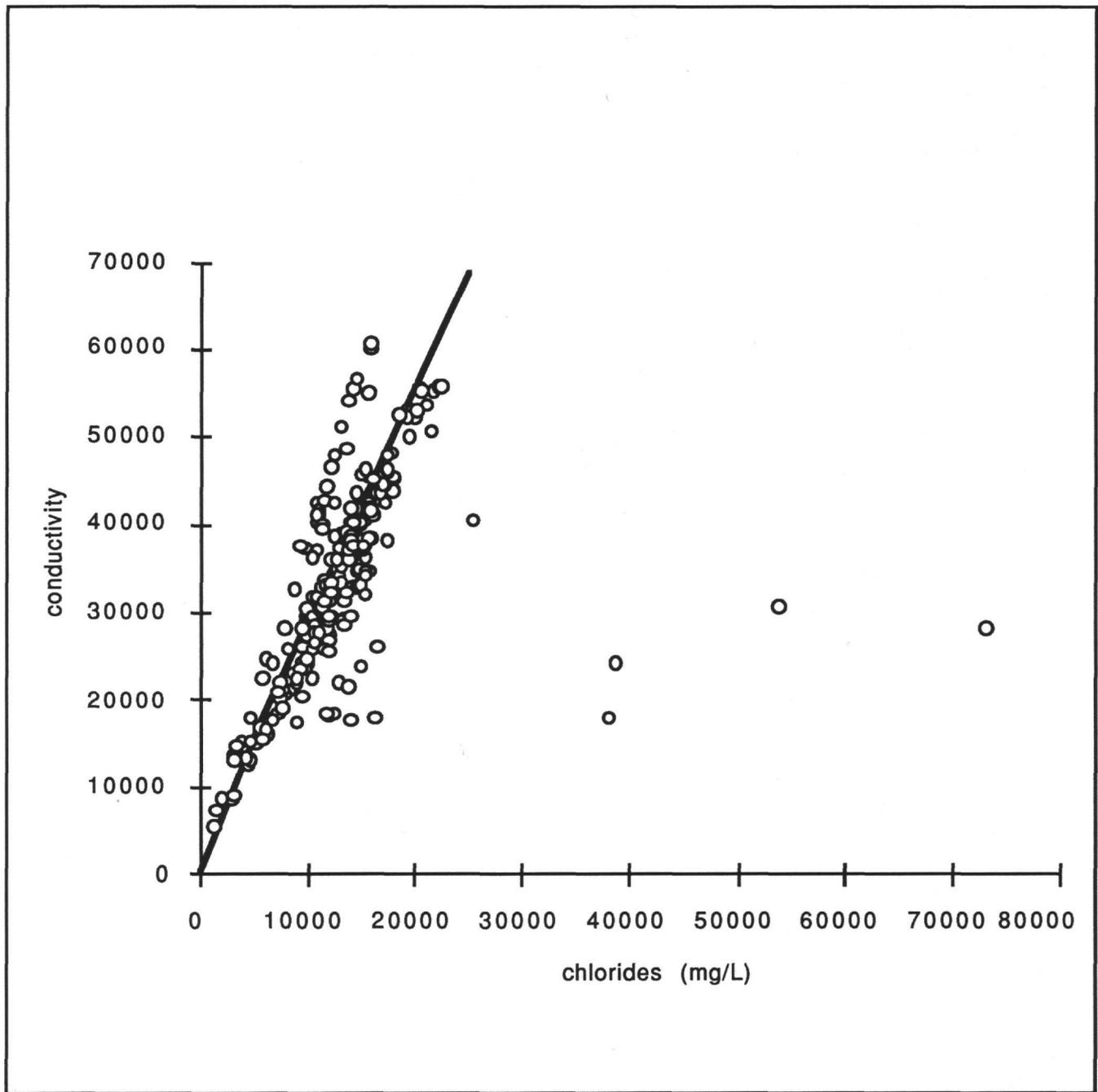
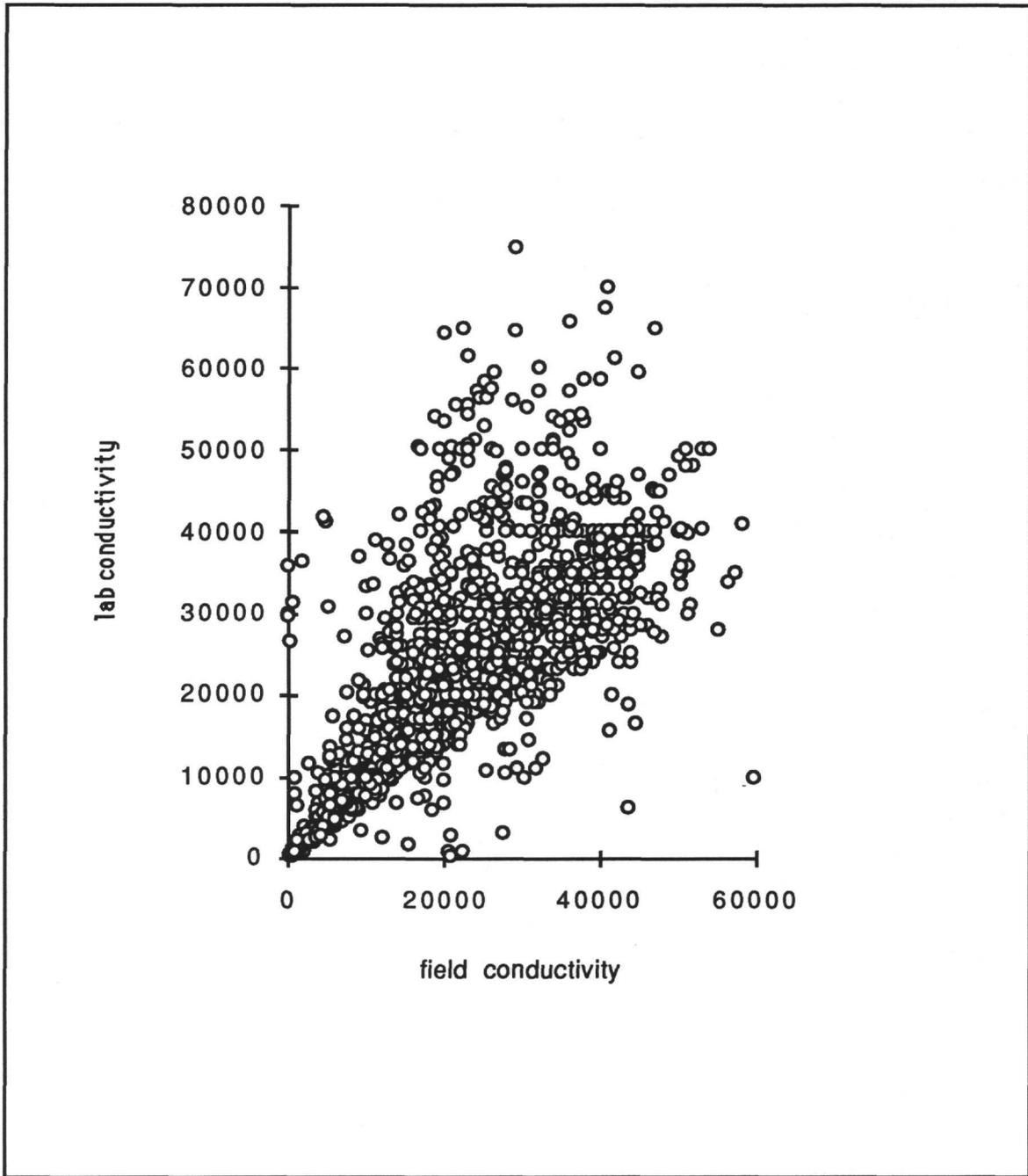


Fig. 3-4. Scatterplot of conductivity versus chlorides for 1971-72 data of Galveston Bay Project



**Fig. 3-5.** Scattergram of laboratory versus field measurements of conductivity from SMN data base, bays and surface data only.

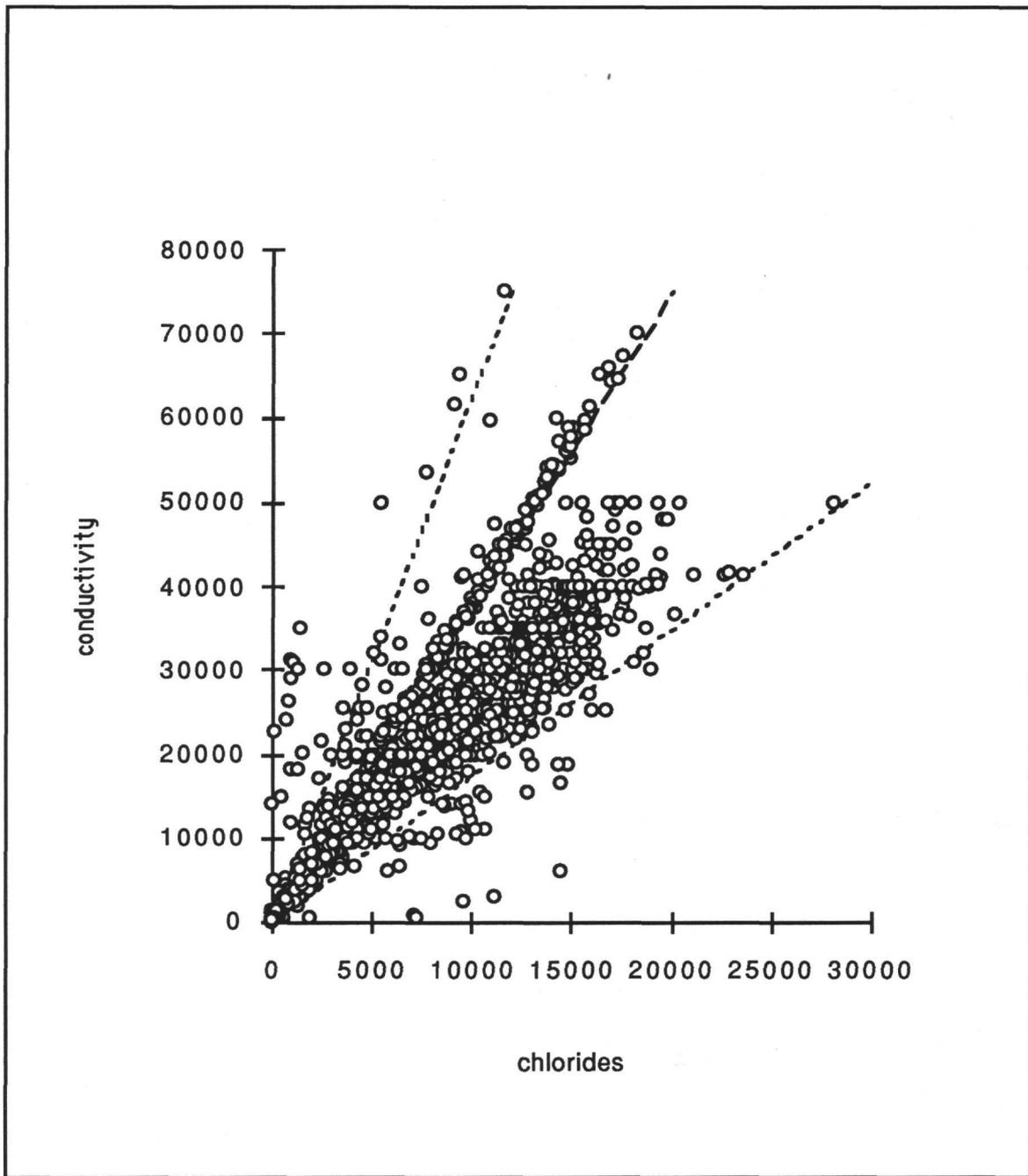


Fig. 3-6. Scattergram of laboratory measurements of conductivity and chlorides from SMN data base. (Lines are  $y = \frac{25}{4} \cdot x$ ,  $y = \frac{15}{4} \cdot x$ ,  $y = \frac{7}{4} \cdot x$ .)

project, we could not look into the source of this procedure, whether it originated at the lab, the data entry clerk or *post facto* in data processing, but in our opinion the "supply" of data in place of actual measurements is bad practice, even if the correct relation is used. We also note that this practice offers one more degree of freedom for human error, and indeed a close inspection of the data reveals a few tens of points that fall on the lines  $y=7/4 \cdot x$  and  $y= 25/4 \cdot x$ : apparently the "rule-of thumb" was occasionally blown. For present purposes, we must somehow determine which "measurement" is bogus and expunge it from our data base. We examine the statistics of each of these versus the independent field measurement of conductivity, for those data sets whose points fall on the  $y=15/4 \cdot x$  line, as summarized in Table 3-3. Of course, since the bogus data is a constant multiple of the real measurement, the extent of linear correlation versus an independent variable will be the same, and statistics alone cannot determine which of the two is the actual measurement. However, the linear regressions can be compared to external information as how the relation ought to behave. In Table 3-3, we see first that the lab conductivity systematically exceeds the field data with a slope of 1.43, in comparison to the expected slope of 1. Second, we see that the regression of chlorides versus field conductivity falls very close to the oceanographic relation (3-5): a regression slope of 0.3764 versus a slope from (3-5) of 0.3758. We conclude that the lab chlorides values are real and the lab conductivities are bogus.

The second problem with the TWC SMN lab conductivity measurements noted above is more fundamental, and seems to lie in the measurement procedures themselves. The lab conductivities are systematically at variance with the field

TABLE 3-3  
Linear Regression Statistics on TWC SMN Data with  
Relation: Conductivity= $15/4$  x Chlorides

Number of data points (bays and surface data only)		366	
field conductivity mean	15900	Standard deviation	10171
A. Lab conductivity versus field conductivity			
lab conductivity mean	24180	standard deviation	17195
linear correlation	0.847	explained variance	0.718
OLS regression slope	1.432		
standard error	9139		
B. Lab chlorides versus field conductivity			
chlorides mean	6404	standard deviation	4500
linear correlation	0.851	explained variance	0.724
OLS regression slope	0.3764		
standard error	2364		

values, and are much more prone to aberrant values, as depicted in Fig. 3-7. The lab samples analyzed by the Texas State Department of Health are subjected to extreme dilution to bring the conductivity into the range of the laboratory meter, then the measured value is scaled back up by the reciprocal of the dilution. This introduces potential sources of error relative to the direct field measurements (which use a submerged probe and direct on-deck readout): the dilutions may be performed imprecisely; nonlinear variation of conductivity with salinity—though slight—may be sufficient to corrupt the measurement when large dilutions are needed; the calculations necessary are subject to arithmetical mistakes. Beyond this, there are other anomalies. The points aligning with the relations  $y=25/4 \cdot x$  and  $y=7/4 \cdot x$  were noted above. There are groups of several tens of points for which the field values range a factor of 2, but the lab values are constant at an even multiple of 5000 or 10000. These lines of data points may be seen at lab values of 20000, 30000 and 40000, though they are somewhat obscured on Fig. 3-7 due to the nearly 1500 data points plotted. In summary, there is suspicion attaching to the SMN lab conductivities. Given this, it will probably be little surprise that similar problems were encountered with the Texas State Department of Health data base. Details are given in Ward and Armstrong (1992). In summary, all lab conductivity data before 1973 were determined to be completely unreliable and expunged from the data base; those after 1973 were used only when other measures were unavailable (about 50 measurements in all).

This hyperdetailed discussion is presented to reinforce two points. First, the precision of the methodology notwithstanding, it is the procedures and technique of the field crew, the laboratory and the data entry personnel that are controlling in the level of accuracy attained, especially the laboratory. Even for as straightforward and commonplace a measurement as reading a conductivity meter or titrating for chlorides, the potential for error is substantial, as shown above. What then can be expected of more complex and demanding analyses of trace metals or organics? Second, we rarely have the luxury of simultaneous determinations of two related variables, by which we can evaluate the consistency and probable error of the data, as is the case in Figs. 3-1 *et seq.* What then of the many programs in which only a single measure of salinity was made, and there is no means of cross-checking the data? Any data point should be regarded with suspicion, and the cross-comparison with other nearby, contemporaneous measurements, even from different programs, should be an indispensable guide to weighing the reality of a measurement.

### 3.2.2 Dissolved oxygen

As noted above, DO is one of the fundamental indicators of aquatic health, since it determines the ability of aerobic organisms to survive. With the development of electrometric probes for DO—a welcome technology for anyone who has ever performed Winklers in a rocking boat—field measurements of DO have increased geometrically, and are now a routine component of most *in situ* monitoring. The data base for DO is therefore approaching that for temperature and salinity, especially in the last two decades, though the data from the 1950's and 1960's are principally laboratory determinations on water samples.

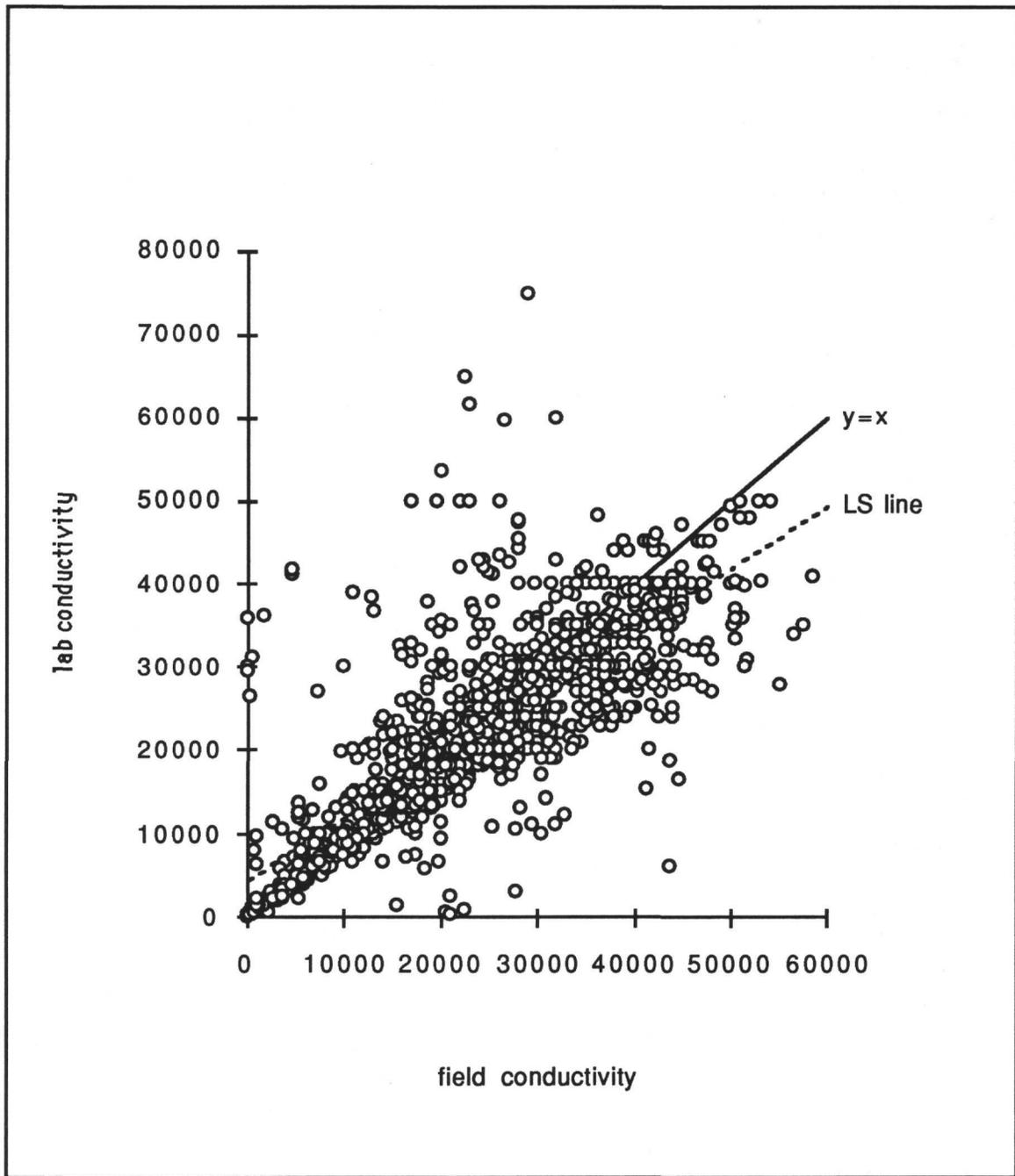


Fig. 3-7. Relation of lab conductivity to field conductivity for SMN data, bays and surface values only, bogus data removed.

DO is introduced into the water column principally through "reaeration," the mechanical process of surface transport from the atmosphere, and through photosynthesis. Therefore DO can serve as an indicator of both mechanical aeration and the intensity of primary production. The primary depletion of DO is due to biochemical stabilization of organics through the respiratory process of the biological community (see Section 3.2.4 below), and low DO's are traditionally linked to the presence of oxygen-demanding pollutants and/or very high rates of primary production.

One of the key controls on the concentration of DO is its solubility, which is a strong function of temperature and salinity. APHA (1985) compiled data on oxygen solubility and offered a nonlinear regression equation with eight coefficients for direct calculation. This expression is much more precise than is required here, even if the coefficients were correct, which they are not. A simpler functional equation was sought, and after several various forms were evaluated by regression and analysis, the Fair-Geyer (1954) expression proved to be simplest and most accurate:

$$C_s = 5 (100 - Cl) / (T + 35) \quad (3-6)$$

where  $C_s$  is DO saturation in mg/L, Cl is chlorinity in parts per thousand and T is temperature in degrees Celsius. The coefficients were re-evaluated using the data in APHA, 1985. The general accuracy of this relation is shown by the sample computations in Table 3-4 for the normal estuarine range of chlorinity and temperatures. Clearly, the formula is accurate to less than 0.1 mg/L.

Table 3-4  
Measured and computed DO saturation concentration (mg/L)

Chlorides (ppt):		0		10		20	
Temperature (°C)	Measured	Computed	Measured	Computed	Measured	Computed	
0	14.621	14.286	12.888	12.857	11.355	11.429	
5	12.770	12.500	11.320	11.250	10.031	10.000	
10	11.288	11.111	10.058	10.000	8.959	8.889	
15	10.084	10.000	9.027	9.000	8.079	8.000	
20	9.092	9.091	8.174	8.182	7.346	7.273	
25	8.263	8.333	7.457	7.500	6.728	6.667	
30	7.559	7.692	6.845	6.923	6.197	6.154	

Measured data from APHA (1985)  
Computations from Equation (3-6)

The functional dependence of solubility on temperature and salinity given by (3-6) illustrates that saturation—and hence DO concentration—will vary substantially over the year, as temperatures range from perhaps 5° to 35°C and chlorinity from 0 to in excess of 20 ‰. If the cold temperature is correlated with low salinity and the highest temperature with highest salinity, a not-unreasonable assumption for Galveston Bay, the total excursion in solubility is from 6 to 14 mg/L. This high range of natural variability can mask variations in DO of importance in diagnosing water-quality problems. Accordingly, two associated parameters of dissolved oxygen are defined: the oxygen *deficit*

$$D = C_s - C \quad (3-7a)$$

and fraction of *saturation*

$$\text{Sat} = 100 C / C_s \quad (3-7b)$$

where C is DO concentration and D is DO deficit, both in mg/L, and Sat is saturation in per cent. The use of these parameters effectively removes the influence of varying temperature and salinity, and allows a more direct interpretation of the (transformed) DO measurements in terms of water quality. The more important of these is the deficit, because it can be shown that if DO kinetics are first-order (or, more generally, are approximately linear in DO concentration) then the total temperature and salinity variation is absorbed in the solubility: the corresponding DO deficit has no temperature/salinity dependency. Interpretation of the DO "climate" requires any two of these three parameters. Deficit, by itself, cannot be interpreted biologically: a deficit of a given magnitude may be biologically limiting in summer and biologically unimportant in winter. For present purposes, we employ the DO concentration and the DO deficit.

### 3.2.3 Turbidity

Turbidity refers to the interference with the passage of light by suspended matter in the water, and is therefore an indirect indicator of the concentration of such suspended matter. Both turbidity and suspended solids have an important rôle in quantifying light penetration and therefore primary productivity in the water column. Further, there are methods of making turbidity-related observations in the field. While turbidity has value in itself as a water-quality indicator, our present interest is in its use as a surrogate measure of suspended solids.

Laboratory turbidity measures are calibrated by standard silica suspensions, so as to eliminate the source of variation due to suspended particles of different constituency and geometry. The traditional method of viewing a candle flame through a vertical tube containing the water sample motivated the definition of the Jackson Turbidity Unit (JTU), see APHA (1985). Modern electrometric optics offer an alternative to the traditional Jackson turbidimeter (e.g., APHA, 1985, Lamont, 1981, Kirk, 1983). The reduction in transmitted light intensity, as measured by a transmissiometer, is expressed as a fraction of the source intensity (per cent transmittance) or in terms of an extinction coefficient. A few

measurements of this type exist from Galveston Bay, primarily associated with small-scale research projects. Alternatively, nephelometers measure light scattering at 90° and the measurement is reported in Nephelometric Turbidity Units, which are defined to be numerically about the same as JTU's. Unfortunately, this numerical equivalence holds only for the calibration compound. For different types and distributions of suspended matter, NTU's and JTU's depart. Further, each is an index and does not *per se* correspond to a physical property of the water. When the reference suspension in the nephelometric procedure is the formazin polymer, the results are often reported as FTU; for present purposes, we regard these as equivalent to NTU.

The depth of the Secchi disc has for many years been the limnologist's and oceanographer's standard means for field measurement of turbidity (Hutchinson, 1957). Unfortunately, the relation between Secchi depth and conventional measures of turbidity is murky at best. Hutchinson (1957) comments, "we should not expect to find more than a rough correspondence between the transmission and Secchi disk transparency of a series of lakes," on the basis of a brief analysis of the differing responses of the transmissiometer and the Secchi disk. A deeper analysis offers somewhat more optimism; Preisendorfer (1986) is the last word on the subject. Effler (1988) combined literature optical theory with field measurements from a number of lakes to arrive at the relation

$$SD = N / [\sqrt{(a^2 + 0.256 ab)} + (a + b)]$$

where  $a$  and  $b$  are the absorption and scattering coefficients, resp., in dimensions  $[L^{-1}]$ . Here  $N$  is a constant, probably a weak function of other optical properties including spectral distribution of light, in the range 8.0-9.6. Since  $b \gg a$  usually, for nephelometric turbidity  $T$  (roughly proportional to  $b$ , with a constant  $\approx 1 \pm 25\%$  for  $b$  in  $m^{-1}$  and  $T$  in NTU) the approximate relation becomes:

$$SD = N'' / T \tag{3-8}$$

where  $N''$  ranges about 5-10 for  $SD$  in meters and  $T$  in NTU's, depending on other optical properties of the water. Vis-a-vis application of the relation (3-8) to the turbid waters of a shallow estuary, in contradistinction to the lakes addressed by Effler (1988), the water is muddied if the water is muddy, because  $SD$  becomes decreasingly sensitive to  $T$  as  $T$  becomes large. (Holmes, 1970, developed an inverse exponential relation between  $SD$  and an optical parameter related to an extinction coefficient, which has the same asymptotic behavior.)

Relating turbidity to suspended matter is even more opaque. From Mie theory, we might anticipate a relation between suspended particles and the scattering coefficient  $b$  of the form

$$b = \pi \sum n_i r_i^2 / 4$$

where  $n_i$  is number of particles of mean radius  $r_i$  per unit volume. This implies  $b = A \cdot SS$  for  $SS$  the suspended solids concentration. (Actually,  $b$  may be taken as

the total extinction coefficient.) From British coastal waters A lies in the range 0.25-0.50 for SS in mg/L (Jones and Willis, 1956). Di Toro (1978) found A=0.40 for San Francisco Bay. Since  $T \approx b (\pm 25\%)$ , we have T proportional to SS.

In summary, there is good reason to expect an inverse relation between suspended solids and Secchi depth,  $SS = B/SD$ , and a direct proportional relation between suspended solids and turbidity,  $SS = A T$ , with constants (from the above literature values) on the order of  $10 < B < 50$  and  $0.2 < A < 0.5$ , for SD in meters, SS in mg/L, and T in NTU. The problem now is to test these relations against data from Galveston Bay to verify the functional form, and to determine the appropriate values of the constants. Virtually the only extensive paired measurements of Secchi depth, turbidity and TSS are those of the TWC SMN data base. The SMN data of paired turbidity and TSS measurements are shown in panel (a) of Fig. 3-8. The data of this figure are from the bays of the Galveston system only; i.e. the tributaries are excluded so that a wider range of turbidities is realized, hence better leverage in the regression. (Also, the dozen or so very high values are excluded from the plot to give a better resolution for the majority of the data, but are included in all of the statistics.) That each is a noisy measurement is immediate from these figures. However, the data (1350 data points) of panel (a) are consistent with the hypothesis of a proportional relation. The zero-intercept least-squares line has practically the same standard error as the ordinary LS line and is physically better based. The slope of this zero-intercept line is about 0.93, which is about twice the value to be expected from the literature values given above. The lower panel of Fig. 3-8 addresses the inverse relation between TSS and SD, by plotting TSS versus  $(SD)^{-1}$ . (The 400 data points are a random sample of measurements from the bays only.) Again, the data are noisy, not quite so much as panel (a), but consistent with a linear relation between the variates. Again, the physically-based zero-intercept LS line is as good a regression as the OLS, differing in standard error by only 1%. The slope of this line is 13.0, within but near the lower limit of the range of the literature values.

In conclusion, for the present study, the following forms are adopted to serve as proxy relations giving TSS in terms of turbidity measurement:

$$\begin{aligned}
 SS &= 3 \log\{Tr/100\} / z \\
 SS &= 0.9 T \\
 SS &= 13 / SD
 \end{aligned}
 \tag{3-9}$$

where SS is suspended solids in mg/L, z is the optical path length in meters, Tr is percent transmission, T is turbidity in JTU, NTU or FTU, SD is Secchi depth in meters, and log denotes the natural logarithm. These relations are at best approximate and much is left unsettled.

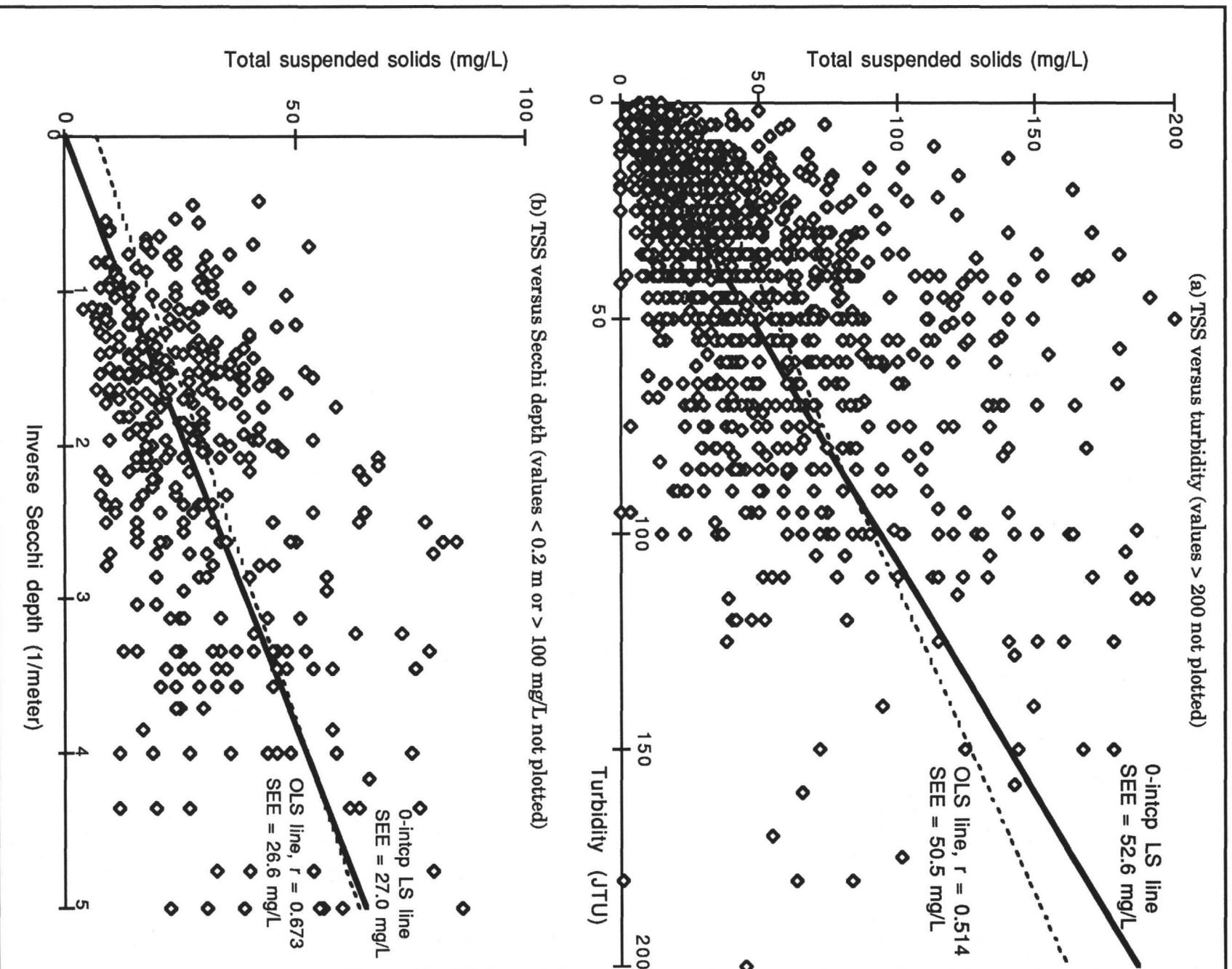


Fig. 3-8. TSS dependency on turbidity measures, from TWC SMN data base

### 3.2.4 Biochemical Oxygen Demand

Since the classical work of Phelps and Streeter the biochemical oxygen demand (BOD) has become one of the fundamental parameters for estimating the presence of oxygen-demanding organics in a water sample (either from a sewage effluent or from a natural watercourse) and is one of the central parameters in the mathematical modeling of dissolved oxygen in the watercourse. Despite this long history of use, which can be traced back to the 19th Century, the BOD test is in many respects still controversial and is a continuous source of debate regarding the correct laboratory procedures and interpretation of the results.

Fundamentally, the BOD is the amount of dissolved oxygen (DO) consumed in a sample of water during some period of time. The standardized test for the 5-day BOD is familiar and detailed in, e.g., *Standard Methods* (APHA, 1971, 1985) and HMSO (1983). We also note that the BOD, as established in a laboratory test such as *Standard Methods*, is a direct measure of a biochemical process at work within a stoppered bottle in equilibrium, a trivial observation at this point but which looms large in the interpretation of BOD data. The basic concept of BOD was to measure the potential oxygen depletion in a natural stream due to an injected waste. Since then the concept has evolved in two separate directions, both of which are referred to as BOD, to compound the confusion. The first is the oxygen consumed within the watercourse by the degradation of organic wasteloads, for which the ultimate BOD is the crucial quantity, with the oxygen depletion directly related through Phelps Law as described below. The second is the evolution of the BOD bottle test as a measure of the organic wasteload of an effluent, and therefore, as a direct monitor of the operation of a waste-treatment facility and the key design parameter for treatment processes. Generally, analytical procedures fall into three classes: manometric, including the classical Warburg device (Heddele and Tavener, 1981; Elmore, 1955), multiple BOD bottles analyzed sequentially (Parisod and Schroeder, 1978), and electrometric-probe-monitored BOD. For the latter two, a modification of the technique is to reaerate the sample during the course of the "BOD" progression.

The amount of oxygen consumed as consequence of aerobic biochemical processes in a water parcel, whether it be a laboratory BOD bottle on a shelf or a moving parcel of water embedded within the flow of a natural watercourse, is directly dependent upon a number of variables, as follows:

- (1) Types of bacteria present in the water;
- (2) Initial quantities of each type of bacteria present;
- (3) Multiplication or growth rates for each type of bacteria present;
- (4) Chemical characteristics of the substrate, i.e., the oxidizable organic constituents within the water;
- (5) The quantity, or concentration, of the oxidizable constituents;
- (6) Constituents which act as an inhibitor or a stimulant for the bacterial metabolism;
- (7) Environmental parameters, most notably pH and temperature;
- (8) Other aerobic organisms in the water, notably phytoplankton.

It is apparent that there is a multiplicity of factors that can affect the BOD in the water parcel.

There are two broad categories of bacteria contributing to oxidation within the water parcel, the heterotrophs and the chemautotrophs. The former is dominated by, and therefore practically synonymous with carbonaceous bacteria. The heterotrophic bacteria indigenous to a natural watercourse comprise a numerous, complex, and diverse community of organisms with a wide range of specific metabolic capabilities (see the review by Costerton and Colwell, 1979). Further, the regular presence of wasteloads not only provides a rich and varied source of organic carbon to stimulate and support this bacterial community, but also represents in itself a significant source of bacteria. The chemautotrophs derive their cell carbon from inorganic carbon compounds, e.g., CO<sub>2</sub> or carbonates. Of particular concern are the nitrifying chemautotrophs, *Nitrosomonas* spp. and *Nitrobacter* spp. which are the principal organisms responsible for the oxidation of ammonia to nitrate.

The carbonaceous ecosystem of the BOD sample (be it a laboratory bottle or a parcel of water in a stream) is comprised of a multiplicity of species of heterotrophic bacteria and predators (primarily protozoa), which when subjected to introduced organic carbons undergo a definite succession as the carbons are stabilized. At first there may be an acclimation or adaptation period, then the bacteria will begin to exhibit an exponential growth accompanied by an exponential decline in the carbon source. As the source begins to be depleted, the bacterial population passes through a period—perhaps brief—of stable population, then enters a declining exponential growth phase. The O<sub>2</sub> uptake curve and the bacterial curve have essentially the same shape, the familiar S-shape of an autocatalytic process. (NB, the reaction agrees with the classical first-order kinetics only when the increasing exponential growth phase is entered.) Actually, this description is even oversimplified, being applicable only to a monospecific bacteria culture responding to a monomolecular carbon source. In a real system, the nutrient source will be made up of many organic carbon constituents, some more labile than others, and different species of bacteria in the community will attack that component of the substrate most appropriate for an energy source in terms of availability and the specific biochemical abilities of the species. All of these processes of stabilization and oxygen uptake will be going on simultaneously, at different individual kinetic rates, thereby blurring the autocatalytic nature of the process (Parisod and Schroeder, 1978).

The identification and separation of the carbonaceous stage of oxidation from the nitrogenous in the BOD measurement has been a matter of considerable study. Bird (1981) summarizes the effects of nitrification upon the results of the standard BOD. The frequently observed lag between the nitrogenous stage and the carbonaceous has been ascribed to a prejudicial effect of organic matter on the activity of the nitrifiers. Thus, occurrence of nitrification is suppressed until the carbonaceous oxidation is practically complete. This is an older view forwarded by, among others, Phelps, but is now generally dismissed in favor of the simpler explanation that the concentrations of nitrifiers in the initial diluted sample are merely too small to affect the oxidation rate, but rather a period on the order of 15

days is required before the nitrifiers proliferate to a concentration capable of producing measureable oxygen reduction (Stones, 1976). The fact that carbonaceous organisms have a much shorter doubling time, and are therefore capable of immediately exerting an oxygen demand, is fortuitous in allowing the separation of the two stages. The most important aspect of this lag for our present purposes is that the 5-day value is unlikely to exhibit any significant nitrifier activity.

Because of the desirability of unequivocally separating the carbonaceous and nitrogenous stages, much has been made of the use of nitrification "inhibitor" constituents, and these have become an optional step in the laboratory procedure (HMSO, 1983). Any such substance should, of course, be employed with great care to be certain that nitrification is in fact inhibited, not simply retarded, and that inadvertent effects upon the heterotrophic organisms do not occur. There is considerable debate in the literature as to the specificity of various nitrification suppressors.

Given the above, it should be no surprise that there is a variety of data from Galveston Bay, all labeled "BOD". The measurements include dilution-series 5-day BOD with cultured seed, dilution-series 20-day BOD with cultured seed, aerated BOD, dilution-series 5-day BOD with natural seed, dilution-series 20-day BOD with natural seed, nitrogen-suppressed 5-day BOD and nitrogen-suppressed 20-day BOD. Further, there is no meaningful way to interconvert from one to the other, except (perhaps) use of the first-order relation to convert from 5-day N-suppressed to 20-day N-suppressed, if one has a good estimate of the rate coefficient, which one does not.

The choice of which of these to employ therefore is based upon two criteria: (1) the parameter most utilitarian for the purposes of the study, (2) the parameter affording the greatest data record. With respect to the latter, the greatest amount of data, both in spatial coverage and period of record, is for dilution-series 5-day BOD. Most of the N-suppressed BOD data has been obtained by TWC, generally since about 1980. Aerated-BOD data were taken during the first two years of the Galveston Bay Project (with a scattering since in a few academic research studies). With respect to the former criterion, it is the prime objective of this study to establish long-term trends in BOD as a measure of organic loading and labile carbon sources. For this purpose, the 5-day BOD will serve as a suitable index. Were this a modeling study, for which the BOD would be used as a sink term in a DO budget, then much more stringent requirements would be necessary on the measurement to ensure that it bore some relation to a real physical process. But that is not our purpose here. Accordingly, this study focused on the 5-day BOD. For consistency, it would be preferable to limit this to BOD<sub>5</sub> without nitrification suppression, since most of the historical data is of this type. The TWC SMN data base contains both types, but unfortunately there are no paired measurements of "total" and "carbonaceous" BOD<sub>5</sub> (i.e., of nonsuppressed and nitrogen-suppressed BOD<sub>5</sub>). We therefore assume that for a 5-day duration the two will generally be equivalent, and use both in the BOD<sub>5</sub> data base.

The precision of a BOD test is exactly that of the dissolved oxygen measurement compounded by the number of independent DO measurements (there are at least two). The accuracy is another matter, particularly with respect to replicability. The manifold processes underway in a BOD bottle render the oxygen depletion quite complex and problematic. One ubiquitous source of error is in the dilution itself. In many studies employing BOD, the phenomenon has been encountered of increasing BOD (per unit volume) as the sample is subjected to greater dilutions. Thirty-five years ago, Elmore (1955) remarked with respect to dilution-series BOD:

When several dilutions are completed on the same sample and the B.O.D. curves are plotted after correction for dilution and blank, normally a family of curves is obtained with the more dilute samples indicating a higher value. This basic discrepancy has been attributed variously to toxicity of the samples, inhibition of nitrite formation below 2 p.p.m. dissolved oxygen, variations of rate of B.O.D. with concentration of dissolved oxygen available, and most recently to small errors in determination of the proper blank. Often the B.O.D. to be used must be selected from two B.O.D. values that appear equally satisfactory but are considerably divergent.

The situation is still much the same. This same problem has been found in Galveston Bay as well. In the Galveston Bay Project, the BOD test was given intensive study. Routine measurements were made of BOD at various multiple dilutions, and later aerated. The increase of BOD with dilution was attributed to toxicity in the sample water, inhibiting the metabolism of the bacteria. In Espey et al. (1971), data from Galveston Bay on the ratio of BOD<sub>5</sub> in 4:1 dilution to that of an undiluted sample were tabulated, and shown to range from 1.5 to 4.5, with the highest values in Galveston Harbor and the lower bay. However, no association with other parameters was evident. Reynolds and Eckenfelder (1970) also investigated the phenomenon and evaluated six causal hypotheses with inconclusive results. Still later in the same project, the question was re-investigated by Oppenheimer et al. (1973), who carried out long-term (up to 45 days) incubations of 1:5 dilutions. While the diluted BOD<sub>5</sub> measurements were 3-6 times that of undiluted samples, these authors seem to have been more concerned with a search for correlation with TOC (which was unsuccessful), and were unable to comment conclusively on the cause of the dilution discrepancy.

A possible explanation for the phenomenon lies in the Monod equation for bacterial growth (e.g. Monod, 1949), formally equivalent to Michaelis-Menten kinetics. This equation gives the growth rate as a function of substrate concentration  $C$ , which also governs the oxygen consumption rate. If  $C$  is initially high, the growth rate is essentially its maximum value. As  $C$  is reduced, the growth rate declines. The rate constant for BOD exertion is, therefore, a nonlinear function of the dilution factor, which of course contradicts the basic assumption underlying the dilution approach, that the BOD depletion is simply proportional to dilution.

For present purposes, when dilution is reported, we use data from the smallest dilution sample available. Generally, however, the BOD is reported without any

further information on dilution, so we must regard the dilution factor as a (considerable) source of uncertainty in the measurement.

The controversy attending the BOD measurement has led some researchers to propose alternatives to the parameter. One such is the total organic carbon (TOC), a suggestion motivated by the notion that BOD measures the organic carbon substrate (Busch, 1966). However, the nature of the carbon compounds as well as the capabilities of the bacteria dictate the oxygen demand. For example, Maier and McConnell (1974) report TOC:BOD ratios in the range 0.5-1.0 for "simple" carbon compounds (e.g., glucose, stearic acid), and in the range 0.5-4.0 for wastewaters of varying treatment levels. Note was made above to the unsuccessful attempts to correlate BOD and TOC during the Galveston Bay Project. We believe there to be too much uncertainty in the relation for TOC to serve as a suitable proxy for BOD<sub>5</sub>, or *vice versa*. Further, there is very little TOC data from Galveston Bay which could be used as a basis to test a proxy relationship.

### 3.2.5 Nitrogen and Phosphorus Species

Two of the principal nutrients, nitrogen and phosphorus in their various forms play an essential rôle in aquatic biological processes. Further, their concentrations can be significantly augmented by the activities of man, especially through point discharges of municipal and industrial wastes, and through runoff from modified watersheds. While nitrogen exists in four principal species, not all of these are routinely measured. The most frequently measured forms, and therefore the best data base, are ammonia and nitrate; though wherever possible we also present analyses of organic nitrogen and Kjeldahl nitrogen (the sum of ammonia and organic). Nitrite is much less frequently measured, and in the watercourse is rather unstable, being readily oxidized or reduced to other forms (and therefore nitrite concentrations are much less than those of the other species). As the relative proportions of these species are a strong function of origin, transport and microbial kinetics, no relation among them is meaningful for serving as the basis of a proxy equation. (The relations typically used in mass budgeting, e.g. Meybeck, 1982, are strictly applicable to natural waters, not waters potentially subject to abnormal sources of nitrogen or abnormally vigorous populations of chemautotrophs.) Therefore, separate analyses were carried out for each of these. It should be noted that the measurement of organic nitrogen refers in fact to the least refractory components of the organic pool (McCarty et al., 1970).

The most common measures of phosphorus concentration are orthophosphates and total phosphorus. Generally, the latter is predominant in the Galveston Bay data, hence was selected as the principal measure of phosphorus for analysis. As is the case with nitrogen species, there is no consistent relation among the forms of phosphorus that would serve as a proxy. The proportion of orthophosphate may be quite low in natural waters, but as much as 90% of the total phosphorus in municipal effluents. Also, the natural concentrations in seawater are much different from those in fresh water. Accordingly, we have chosen to focus the

analysis specifically upon total phosphorus. One significant source of uncertainty in this measurement is the treatment of particulate (versus dissolved) phosphorus. Phosphorus is sorptive and has an affinity for fine-grained suspended sediments. In some of the data sets, it is not clear whether the total-phosphorus analyses are restricted to the dissolved fraction (i.e., whether the sample is filtered) or includes the particulate. (In the case of recent TWC data, total phosphorous is digested and filtered before analysis.)

### 3.2.6 DDT

Analysis of chlorinated organic pesticides, and trace organic chemicals in general, is a relative newcomer to water and sediment quality monitoring. Protocols and procedures are still evolving, and this is reflected in a confusion of data acquisition. Some of the problem originates in the multiple forms a specific organic can assume: various isomers, analogs and metabolites. Further, the nomenclature for many of these is nonstandard and contributes to the confusion, particularly in data reporting. Lindane, for example, is reported variously as gamma-BHC (benzene hexachloride), gamma-HCH (hexachlorocyclohexane) and lindane. Most of these problems were mooted in this project because the amount of data available was so limited that little meaningful analysis could be performed. One exception was the insecticide DDT, which is certainly the most prominent of the chlorohydrocarbons and for which the available data base is greatest.

Dichlorodiphenyltrichloroethane (DDT) as a technical product is comprised of as many as 14 analogs and isomers. By far the most important are p,p'-DDT and o,p'-DDT. The relative proportion of the two is a function of the proportion in the initial source and of the relative kinetics and metabolism in the receiving water. Neither of these is particularly well-defined, though the former is probably better established than the latter, to be about 70% p,p'-DDT and 20% o,p'-DDT in technical grade DDT (Buechel, 1983). This is roughly consistent with the rule-of-thumb of a 3:1 ratio of p,p'-DDT to o,p'-DDT that seems to be current now. Both forms are hygroscopic and sorb readily to fine particulates, both sediments and phytoplankton (Crompton, 1985). Apparently microbial assimilation is stimulated by sorption (Chau and Afgan, 1982, Crompton, 1985) but appears to affect both isomers equally. Treatments of the kinetics (including volatilization) of DDT make no differentiation between the isomers (e.g., Moore and Ramamoorthy, 1984) so we assume that their ratios will be preserved in the receiving water. Accordingly, the relation between total DDT and p,p'-DDT is taken to be:

$$\text{Total DDT} = 1.4 \cdot (\text{p,p'-DDT}) \quad (3-10)$$

While this appears to be a workable proxy relation, we have no reported paired measurements by which we can test it, therefore in the subsequent analyses, we present separate treatments of the total DDT data as reported, and the extended data set using the above proxy relation.

### 3.2.7 Chlorophyll

As a surrogate indicator of phytoplankton biomass, chlorophyll-a has become a common water quality parameter. There are two basic analytical methods applied to water sampling in the Galveston Bay data base, the trichromatic and the spectrophotometric. The former can be corrupted by its response to the metabolic product phaeophytin and thus overestimate the concentration of chlorophyll a. The TWC SMN data base includes both types of analyses, though none are paired. However, for each of the spectrophotometric measurements of phaeophytin there is also a spectrophotometric measurement of chlorophyll-a (though not the converse). The statistical association between the two for this data base is summarized in Table 3-5.

TABLE 3-5  
Statistics of chlorophyll-a and phaeophytin data in TWC data base

number of observations			2546
chlorophyll-a	mean	16.12	standard deviation 23.43
phaeophytin	mean	5.46	standard deviation 12.65
linear correlation	phaeophytin on chlorophyll-a		0.035
explained variance			0.12%

It is noted that the coefficient of variation of phaeophytin is 230%, no doubt due to the significant space-time variability of phytoplankton in Galveston Bay as well as imprecision in the measurement, and that chlorophyll-a and phaeophytin are virtually uncorrelated. Therefore, the presence of phaeophytin represents an uncertainty in the measurement of chlorophyll-a over and above the imprecision of the basic measurement. Accordingly, the effect of phaeophytin in this data analysis was incorporated into the uncertainty specification for trichromatic chlorophyll-a (see Section 4.2.3 below).

### 3.2.8 Coliforms

The reader has no doubt noticed that this section has addressed the potential for proxy relationships in general order of increasing uncertainty. Therefore, it should be no surprise that coliforms are treated last. The specification of two basic classes of bacterial growth-response referred to as "total coliforms" and "fecal coliforms" is a controversial, low-precision measure, originally intended to provide an index to the extent of contamination by pathogens of enteric origin. There is, due to the extensive oyster industry, a considerable data set for Galveston Bay. Sometimes, total coliforms are measured, sometimes fecal, occasionally both. The question is whether there is a stable relationship between the two that will allow us to proxy a data set for one or the other.

To the extent that both are dominated by an origin in discharge of sewage, the answer would be anticipated to be affirmative. However, both—especially total coliforms—are the result of a large, varied community of microorganisms with various non-sewage, non-anthropogenic, and even non-mammalian sources. (This, in fact, is the nub of the controversy surrounding the efficacy of coliforms as an indicator organism.) There is a rule-of-thumb about, that fecal coliforms are approximately one-fifth of total coliforms (e.g. Kenner, 1978) but there seems to be little published support. Fortunately, in the Galveston Bay data set, there are hundreds of paired measurements of total and fecal coliforms, which will allow us to determine whether this or any other proportion is appropriate. The most convenient data source is that of the Galveston Bay Project, where 1779 paired 5-tube MPN's with a wide range of dilutions were performed throughout the bay over a five-year period. Further, most of the laboratory testing was carried out at the same lab by the same personnel.

The obvious approach to determining the validity of a relationship is to carry out a correlation analysis. The GBP data are depicted in Fig. 3-9 as a scattergram. ("Depicted" because the bulk of the 1779 data points plot on top of each other in the vicinity of the origin; the lower panel of Fig. 3-9 shows an expansion of the range up to  $1 \times 10^6$ , but still the majority of the points are clustered below the resolution of the figure.) A linear regression analysis discloses a relation of

$$F = 0.352 C + 1056$$

(where F denotes fecal coliforms and C total coliforms per 100 mL) with a correlation coefficient of  $r = 0.889$ . In water quality analysis, such a high linear correlation is rare, and one would feel justified in taking it to the bank. One would also be wrong. This is an excellent example of how the uncritical use of statistics can lead one astray. The problem is that total and fecal coliforms are not quite independent, but by the nature of the coliform test,  $0 \leq F \leq C$ . This relation confines the data points to lie between the rays  $F=0$  and  $F=C$ , and this induces a correlation that is irrelevant to any physical relationship. For two variates X and Y related by  $Y=cX$  where c is a uniformly distributed random variable  $0 \leq c \leq 1$ , we have  $r^2 = 3/[1+4(\sigma_X/\mu_X)^2]$  where  $\sigma_X/\mu_X$  is the coefficient of variation of X. If X is uniformly distributed then  $r^2 = 0.43$ . I.e., there is an artificial linear correlation of 0.65. For coliforms, the natural distribution is nonnegative and highly skewed, with most of the data clustering in the smaller values. As a first approximation, take X to be lognormal, whence  $r^2 = 0.75$ . With this much spurious correlation, the  $r^2 = 0.79$  of the GBP coliform data is seen to have no significance at all.

Instead, we examine the ratio F/C. For the GBP data, this ratio has a mean of 0.325 and a standard deviation of 0.306. This is too large a degree of scatter to attach any significance to the mean. In fact, for practical purposes, the ratio approximates a uniform distribution over the range 0 to 1. Consequently, we conclude that there is no useful ratio by which fecal and total coliform may be related, and the two should be treated as independent measures.

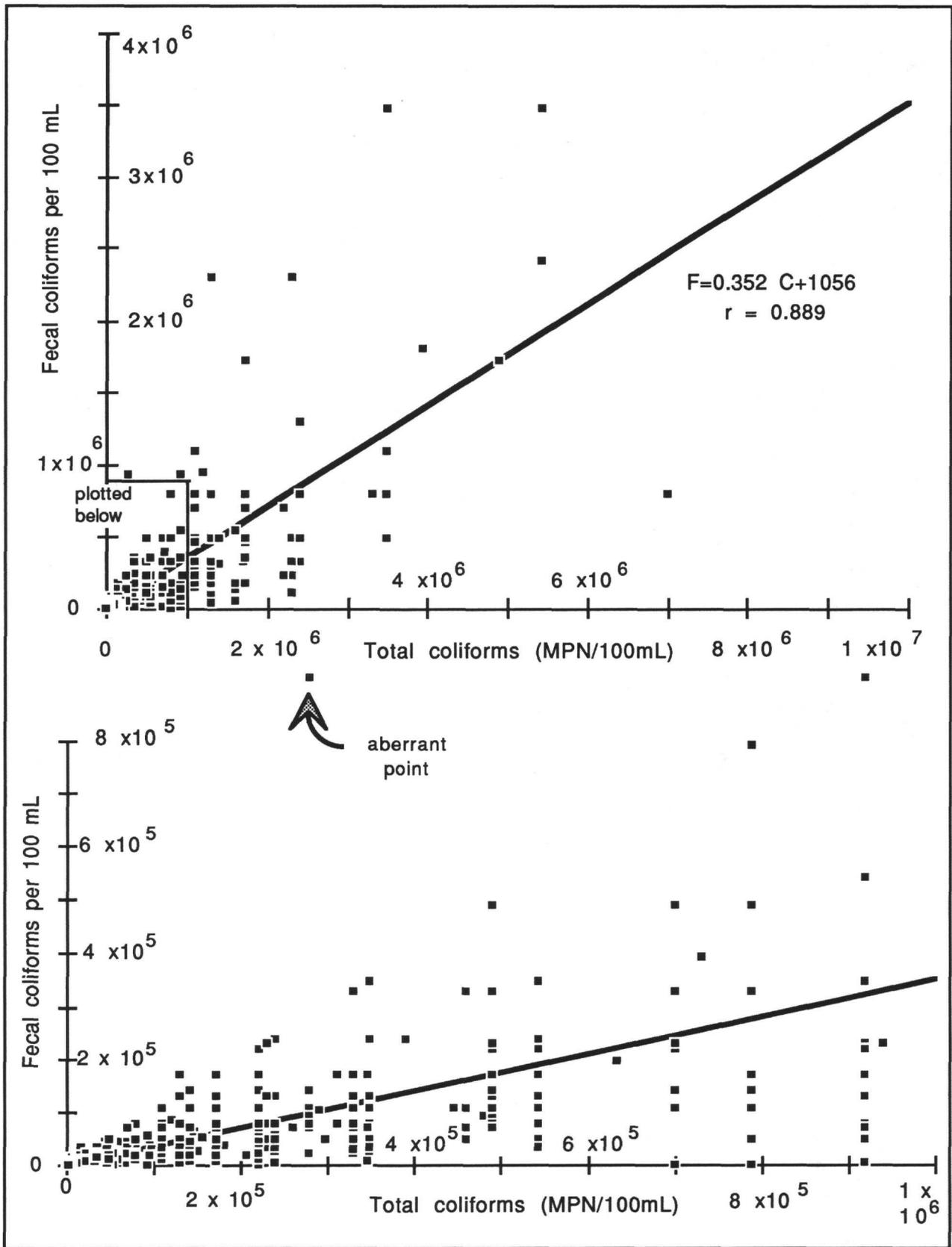


Fig. 3-9. Regression of fecal coliforms versus total coliforms, GBP data

### 3.3 Data collection in Galveston Bay

The data analyzed in this project were drawn from numerous past programs in Galveston Bay. These programs are summarized in Table 3-6. Each of these comprises measurement of some of the water or sediment quality variables within a part of Galveston Bay for some definite sampling interval and period. Apart from this general statement, the programs differ in objectives and procedures.

At the most basic level, we differentiate between the objectives of monitoring, survey and research. Monitoring programs are put in place for a protracted or indefinite period for the purposes of sampling a suite of variables. The data from such a program generally serves more than one purpose. A key characteristic of a monitoring program is consistency in the suite of variables acquired, since it is the accumulation of a long-period data base that is the purpose of the program. Important monitoring programs in Galveston Bay include the Texas Water Commission Statewide Monitoring Network sampling, the Texas Parks and Wildlife Coastal Fisheries Program, the Bays and Estuaries Program of the Texas Water Development Board, and the Texas State Department of Health Shellfish Sanitation surveys.

A survey, in contrast, is characterized by a definite limit in time. It may be a one-time sampling run, or may be a few such runs carried out within a relatively short calendar period. The objective generally emphasizes spatial distribution, and the characterization of the suite of parameters at a point in time. An example is the Bureau of Economic Geology Submerged Lands Project, in which each of a network of stations in the bay was sampled once to determine a basic suite of chemistry and texture. A survey and a monitoring program share the feature that a suite of measurements is obtained that can be used to support different analyses and studies.

Finally, a research program is formulated to address a specific hypothesis, that in turn dictates the suite of measurements. As a practical matter, most research programs are limited in time, due to the nature of the funding process. While we differentiate between these three general strategies of sampling, it must be noted that there is considerable overlap: many monitoring programs provide data for research, many research programs comprise monitoring, and either can contain surveys as a part of the program. The emphasis here is on the difference in philosophy underlying the program strategy, which can clarify the management and prosecution of various programs.

Extremely important to the present project are the presentation and dissemination of the basic data. Monitoring programs generally have provision for data storage and dissemination, nowadays digital. Surveys usually have some form of hard-copy presentation, and research programs may not publish or even preserve the basic measurements, but rather present analyzed or reduced data in a professional publication. Since this project seeks to compile and analyze a combined data set, machine processing is indispensable, and we therefore require all data in a machine-readable format. Where digital databases existed we sought copies from the managing agencies. In some instances, the digital record has

TABLE 3-6

Sampling programs in Galveston Bay  
used in GBNEP Status and Trends analysis

Abbreviation	Agency or source	Project or Program	Source of Data	Project Code	Format of source	Comments
SMN	Texas Water Commission	Statewide Monitoring Network	TWC USGS others	1	mag tape line image of report forms	40 M tape download by special purpose mainframe codes
CDS	Texas Water Development Board	Coastal Data System	TWDB USGS contractors	2	ASCII files	different format from SMN. Line/site stations
TPWD	Texas Parks & Wildlife Dept.	Coastal Fisheries Hydrographic obs	TPWD field labs	3	ASCII	location by lat/long
GBP	Texas Water Quality Board	Galveston Bay Project 1968-72	archival tape of Espey, Huston, Inc.	5	BCD card images	Re-built file during GBNEP Data Inven- tory; original tape lost
TSDH EST	Texas State Department of Health	Estuarine Data File	TSDH Shellfish Sanitation	6	ASCII	Includes most of data from TSDH Project of 63-67
USCE7	Corps of Engineers Galveston District	Operations & Main- tenance Div. 1970s data	USCE O&M	8	hard-copy tabulations	Keyboarded by this project
USCE8	Corps of Engineers Galveston District	Operations & Main- tenance Div. 1980s data	USCE O&M	9	hard-copy, some LOTUS	Keyboarded by this project

TABLE 3-6  
(continued)

Abbreviation	Agency or source	Project or Program	Source of Data	Project Code	Format of source	Comments
USCE9	Corps of Engineers Galveston District	Operations & Maintenance Div. 1989 data	USCE O&M	10	LOTUS spreadsheets	
USBCF	Bureau of Commercial Fisheries	Hydrography & water quality program 57-66 of Trent & Pullen	old tape	11	BCD or hard-copy printout	Half of file re-built by GBNEP Data Inv. The rest keyboarded
BEG	Bureau of Economic Geology, UT	Submerged Lands Study sponsored by GLO	BEG project	12	hard copy tables & maps	Keyboarded by this project
UTD DMRP	University of Texas at Dallas	USCE Dredged Materials Research Project	DMRP reports	13	hard copy tables	Keyboarded by this project
TAMU ESP	Texas A&M Univ. Civil Engr.	Estuarine Systems Project Sediment Study	ESP reports	14	hard copy tables	Keyboarded by this project
USCE WAL	Corps of Engineers Galveston District	Trinity Marsh Biological & Hydrological/Wallisville project	USCE warehouse	15	field sheets & tables	Keyboarded by this project
TAMU METS	Texas A&M Univ. Oceanogr.	Metals survey of Davis (68)	Ph.D. thesis	16	tabular	Keyboarded by this project
POG	Galveston Wharves (Port of Galveston)	Environmental Study for Pelican Is. Terminal	Project Reports	17	tabular	Keyboarded by this project
NOS	National Ocean Service of NOAA	National Status & Trends Project	Project reports	18	tabular	Keyboarded by this project

TABLE 3-6  
(continued)

Abbreviation	Agency or source	Project or Program	Source of Data	Project Code	Format of source	Comments
EHWES	Espey, Huston & Assoc., Inc.	West Bay Env. Studies contract to USCE/Galv	Project reports	19	tabular	Keyboarded by this project
TSDH50	Texas State Dept of Health	50-52 Surveys of Galv. Bay w/ Galveston Cnty	Project reports	20	tabular/ field sheets	Keyboarded by this project
TSDH58	Texas State Dept of Health	58 Survey of Galveston Bay w/ Galveston Cnty	Project report	21	tabular	Keyboarded by this project
CEMI	Coastal Ecosystem Management, Inc.	Trinity Bay surveys, contract USCE/Ft.Worth	Project reports	22	tabular	Keyboarded by this project
HUMB	Humble Oil & Refining Co.	Hydrographic & ecological study of HSC	Project reports	23	tabular	Keyboarded by this project
HSCAERN	Espey, Huston & Assoc., Inc.	Aeration Study of HSC contract from GCWDA	Project report	24	tabular	Keyboarded by this project
HSCNIT	Espey, Huston & Assoc., Inc.	Nitrogen budget study of HSC, TDWR contract	Project report	25	tabular	Keyboarded by this project
CLCND	Chambers, Liberty Cnty Navign Distr	Salinity monitoring in Trinity Bay	USCE files	26	tabular	Keyboarded by this project
TWRI TRIN	Texas A&M Univ.	Hydrological & biological study of Trinity marsh	TWRI report	27	tabular	Keyboarded by this project
PHR NTAK	Texas A&M Univ.	Intake studies at P.H. Robinson SES	M.S. theses	28	tabular	Keyboarded by this project

been lost or destroyed. For a few of these we were able to recover the data record, in whole or in part. Where only hard copy or field notes existed, the data were keyboarded.

### 3.3.1 Principal Data-Collection Programs

Of central importance to Galveston Bay are the existing monitoring programs, since these are the vehicles for continued, routine acquisition of data, and therefore form the backbone for determining the present water quality and any time trends. There are four major monitoring programs under way which contribute information on water and sediment quality of the bay, operated by the following agencies:

Texas Water Commission  
Texas Parks & Wildlife Department  
Texas Department of Health  
U.S. Geological Survey

The Texas Water Commission Statewide Monitoring Network (SMN) is a principal continuing source of a broad spectrum of data. The SMN sampling program is a program of sampling at fixed stations at regular intervals, usually carried out by headquarters, field and/or District offices of the Texas Water Commission (TWC). Generally, field parameters are obtained *in situ*, by means of electrometric probes or portable analytical kits, and water/sediment samples are shipped to the laboratories of the Texas Department of Health for analysis. (Since 1983, some of the analyses are performed by TWC labs, though organics and metals are analyzed by TDH.) Parameters have been expanded from conventional variables in the early 1970's to trace constituents, pesticides and priority pollutants in recent years. The term Statewide (a.k.a. Stream) Monitoring Network also refers to a data management system. The SMN data base is a digitized comprehensive data management program implemented on the TWC mainframe computer and operated in coordination with the Texas Natural Resources Information System of the Texas Water Development Board. The SMN data base includes all sampling activities of the Statewide Monitoring Network, as well as special studies (including microbiology and benthos) and Intensive Surveys. It also includes data from other agencies, notably Texas Water Development Board and the U.S. Geological Survey. There are over 1200 separate constituents with entries in the SMN data base, including water and sediment parameters, and biological parameters.

The Texas Parks & Wildlife Department or its predecessor agencies, the Texas Game and Fish Commission and the Texas Game, Fish and Oyster Commission, has monitored the fishery resources of the system for many years (see Ward and Armstrong, 1991), and in association with this obtains a limited suite of water-quality variables. These tend to focus on estuarine habitat characteristics, e.g. salinity, dissolved oxygen, turbidity and temperature. While the range of variables is obviously much more limited than that of the SMN, the temporal intensity of the program is much greater. The TPWD program obtains data

somewhere in the system on virtually a daily basis, in contrast to the sampling interval of the SMN of one to several months. Further the spatial intensity is also greater. On the other hand, the TPWD samples a random network of stations, so there is no time continuity at a fixed point in the bay. The data is now entered into a digital data base at TPWD headquarters for detailed statistical analyses.

In order to regulate the harvesting of oysters in Galveston Bay, the Division of Shellfish Sanitation Control of the Texas Department of Health (TDH) samples the bay at regular stations at varying temporal intensity, depending upon the season of year and upon the antecedent hydrological conditions. For the purpose of this program, the sampling is now limited to coliforms and a few associated hydrographic variables, salinity, temperature and pH. Like the TPWD, this program samples more intensely in space and time than the SMN and has accumulated data from many years from Galveston Bay. The collected data is maintained in a digital data base at TDH headquarters in Austin.

The activities of the U.S. Geological Survey (USGS) emphasize the inflows to the bay, though the Houston office has and does perform sampling within the estuary itself to help meet the needs of other federal and state agencies. The routine programs are described thoroughly in the publications of USGS (e.g., USGS, 1991). This data is published annually and is maintained in a digital data base, the National Water Data Storage and Retrieval System WATSTORE. Data collected by USGS in support of other agencies, e.g., Texas Water Development Board, may be managed differently, depending upon the nature of the data and the preferences of the sponsoring agency.

In addition, there are important recent or ongoing data collection programs in Galveston Bay, as listed in Table 3-6, however these are not *monitoring* programs because they do not exhibit the regularity and time continuity implied by that term. One of the more important of these is the sampling performed by Galveston District Corps of Engineers in association with its Operations and Maintenance Program on navigation projects. This is intense sampling emphasizing sediment quality that is performed in association with dredging activities. The sampling interval is therefore dictated by the condition of the channel, i.e. sediment accumulation, and may be as long as several years. The Corps data program has been subdivided in Table 3-6 according to the suite of parameters obtained. Generally, there has been an evolution from an emphasis on conventional chemistry and metals to specific hydrocarbons.

Of the historical programs available, there are several which are noteworthy. Most important is the Galveston Bay Project (GBP), a comprehensive study of the system conducted by the Texas Water Quality Board. The GBP routine monitoring program involved monthly sampling at a network of fixed stations, and was conducted in two phases. The first extended from July 1968 through October 1970. After a hiatus, sampling resumed in March 1971 and continued through August 1972. The first phase involved sampling multiple depths at 35 stations. The second phase was considerably reduced, 15 stations being occupied, only in the main bay (i.e. not in the upper Houston Ship Channel) and sampled at mid-depth for most stations, surface and 2/3 depth for the deep channel stations. One of the

important accomplishments of the Galveston Bay National Estuary Program is the recovery of the digital data file from this signal program, a data file that had been lost for years. (In addition, there was a "high-frequency" component of the program, for which no record, hard-copy or otherwise, had survived. The digital data set for this program was also recovered by the GBNEP. However, the nature of the sampling—intensive collection of a few parameters for two or three tidal cycles, every six months or so—precluded its use in a large-scale trends analysis.)

Another noteworthy program is the Submerged Lands Study of the University of Texas Bureau of Economic Geology, sponsored by the Texas General Land Office. This program, which focused entirely upon sediment, falls into the category of a survey, because it involved one-time only sampling. However, it is the only data set extant which samples the *entirety* of Galveston Bay at a uniform station distribution (1-mile), irrespective of the location of shoals, channels, navigation aids and reefs (which tend to spatially bias most measurements from the system).

Older studies performed by the Texas State Department of Health (TSDH, now TDH) entailed a broader suite of samples and more widely distributed sampling stations than is the case for the current program. Especially for the period 1963-67, TSDH carried out intensive sampling throughout the system. Even earlier, the TSDH in cooperation with local county agencies performed sampling of coliforms, salinity, temperature, BOD and pH in the system, providing data records back to as early as 1952.

In the period 1958-1967 the U.S. Bureau of Commercial Fisheries undertook an extended and intensive sampling of the Galveston Bay system, primarily directed at biological sampling, especially shrimp, but also including limited hydrographic and water quality data, viz. temperature, dissolved oxygen, salinity, phosphorus and organic nitrogen. The sampling interval ranged from weekly to monthly, usually at least twice monthly. This is one of the most intensive continuous, consistent hydrographic surveys ever performed on the Galveston Bay system as a whole. It is also the first to employ large-scale digital data manipulation as an intrinsic part of the program. All data were entered onto punched cards. Although several copies of the card deck were disseminated, as both cards and tapes, all are now lost. Only a few copies of the printout for water quality data exist. The loss of this valuable data set is discussed in detail in Ward and Armstrong (1991). During the present project, about half of the digital data were located in an archive tape of a consulting company, the rest of the file was re-keyboarded from a copy of the printout (see Ward and Armstrong, 1992). A few "patently obvious" typographical errors in the original publication (Pullen & Trent, 1969) were discovered in this process and corrected.

### 3.3.2 *Summary of the data base*

The data programs of Table 3-6 formed the basis for the analysis here. Most of these programs, it will be noted, are small-scale research activities, though most of the data is dominated by the few large-scale programs summarized above. The approach of this project is to combine and merge these programs to synthesize a

more comprehensive data base for the system. Details on the data sets of these individual programs are given in the companion data base report (Ward and Armstrong, 1992), along with any problems encountered in the data and how those problems were resolved (or reconciled). Particular note should be made of the programs which were keyboarded into a digital format for this project. As noted earlier, this digital data set, which is capable of much more analysis than it is subjected to here, is considered one of the chief products of this project.

In addition, three more significant data sets were digitized, but were completed too late to be included in the analysis of this project. The first of these is field hydrographic data collected by the TPWD Seabrook laboratory, extending back to the early 1960's. This data set, which only exist in a single hard copy at the lab, was photocopied for this project, and keyboarded by the project staff. This will augment the data record for salinity, temperature and (occasionally) dissolved oxygen. Some of the sampling locations are still being determined. The second set is the routine monitoring of the TAMU Estuarine Systems Project. Dr. Hann and one of the PI's located the only remaining record of raw data from this extensive study of the Houston Ship Channel, *viz.* a single computer printout and files of the raw field sheets, in a warehouse of Texas A&M during summer 1991. This data, primarily weekly salinity/temperature/DO profiles, and less frequent BOD determinations, has now been completely keyboarded. Finally, the magnetic tapes containing the Houston Lighting and Power study of Trinity Bay and the Cedar Bayou Station discharge have at last been decoded by this project, yielding numerous measurements of conventional parameters in northern Trinity Bay. (These tapes were provided by HL&P during the GBNEP Data Inventory Project, but there was no record as to format, and until now they could not be read and interpreted.)

The data bases for water quality and for sediment quality are summarized on a baywide basis in Tables 3-7 through 3-10. (Parameter names are abbreviated for compactness; their definitions are given in Table 3-11.) These tables are information-dense, and largely self-explanatory. Also, it is important to note that these tables characterize the data *sets* rather than the *parameters*, being compiled from unscreened data. That is, these data are as obtained from the source and have not been screened for bad data. Indeed, the range of each variable discloses the presence of obviously spurious entries in the data set. For example, pH ranges from 0 to 17000, and the largest measured salinity is 50000 ppt. Many of the zero values appear to be blank entries (i.e., no measurement) that in the process of agency transcription and digitization were replaced with a zero.

For many of the parameters, such as metals and pesticides, the lower range of measurement is delimited by the detection limits of the procedure, and detection limits are generally reported as part of the data set. Despite this, several of the data sets include zero values. The data in these tables have not been screened for such anomalies, either; therefore the minimum values are given both for the entire data set and for only nonzero values. These tables do provide a ready index to the relative intensity with which different variables have been measured, and the extant period of record. Because of the large spatio-temporal variability in

most of these parameters, the baywide means have little significance, however they are useful in typifying the magnitudes of the different variables (provided the spurious values do not seriously corrupt the mean).

Figures 3-10 *et seq.* display graphically the sampling intensity throughout the bay for the more important of the water and sediment parameters. Sampling intensity is measured by the number of observations within each Hydrographic Segment (see Section 2.3.3) for the period of record. The amount of data available is strongly dependent upon the parameter. For salinity and temperature, two of the more easily measured variables, the data holdings are large, approaching 80,000 independent measurements, while for some metals and most pesticides only a few hundred measurements are available. Further, what might appear to be a large number of historical samples for a given parameter on a baywide basis, from Tables 3-7 through 3-10, is shown to be quite modest—even inadequate—when related to specific areas of the bay. It is apparent at once from Figs. 3-10 *et seq.* that sampling intensity is highly heterogeneous in space, some areas of the bay having been subjected to relatively frequent sampling, and some rarely sampled. There is a particular bias, as might be expected, for the main channels and for those areas with historical pollution problems. The period of record generally ranges over many years so the number of samples *per year* is a considerably smaller number. There is roughly an order of magnitude less sediment data from Galveston Bay than water quality data.

It should be noted that most of the variables of Table 3-1 do not appear in Tables 3-7 *et seq.* Very few measurements have been made in Galveston Bay of most of the priority pollutants. In some instances, there may be a scattering of measurements, but not enough to use in any meaningful way in a status-and-trends analysis. For example, there were two measurements of water-phase Endosulfan-I from the entire Galveston Bay system. Similarly, most of the individual PAH's were represented only by a handful of data. Those variables for which the sample base is totally lacking or inadequate are excluded from these tables. For those parameters for which there is at least a minimum analyzable data base, most of those measurements are below detection limits (BDL), as indicated in these tables.

The treatment of detection limits in analysis of water quality is particularly vexing. There are three logical alternatives, each of which has a rational basis. First, the measurements BDL can be simply ignored, as providing essentially no quantitative information. Second, the BDL values can be replaced with zero in the analyses, on the argument that for practical purposes the parameter is not present. This is probably the most commonly elected alternative. It is, for example, the approach adopted by the National Ocean Service in its National Status & Trends Program (NOS, 1991). Third, the BDL values can be taken to be the reported detection limits, on the basis that the actual concentration could be as high as the detection limit.

TABLE 3-7

SUMMARY OF WATER QUALITY DATA FROM GALVESTON BAY:  
CONVENTIONAL PARAMETERS  
(Unscreened)

Parameter (Table 3-11)	units	No. of obs	Avg >DL	St dev	No. > DLs	% > DLs	Min >DL	date	Min >0	date	Max	date	Avg w/ BDL=0	Avg w/ BDL=DL
wqtemp	deg C	75993	22.4	8.4	75993	100	0	760108	0.1	661022	460	710930	22.4	22.4
wqsal	ppt	77376	15.5	360	77375	100	0	560810	0.00065	780307	50000	800709	15.5	15.5
wqdo	ppm	59181	7.28	6.4	59179	100	0	580224	0.01	880519	910	590512	7.28	7.28
wqph		42106	8.31	83	42106	100	0	870211	0.1	580603	17000	731210	8.31	8.31
wqturb	JTU	13815	40.5	56	13815	100	0	690528	0.9	750318	2000	500605	40.5	40.5
wqtss	ppm	8221	43.6	71	8126	98.84	0	761018	1	770110	2000	741216	43.1	43.2
wqsecchi	m	5388	0.545	0.31	5385	99.94	0.0072	761020	0.0072	761020	7.5	860915	0.544	0.552
wqammn	ppm	12713	1.18	4.3	11575	91.05	0	671001	0.001	870409	170	790511	1.08	1.1
wqorgn	ppm	6508	1.16	1.5	6482	99.6	0	680716	0.01	790612	57	770927	1.15	1.15
wqkjl	ppm	7059	2.73	14	7022	99.48	0	680716	0.01	770824	340	790511	2.72	2.72
wqno3n	ppm	12003	0.462	2.2	10555	87.94	0	700707	0.002	820427	140	730911	0.406	0.409
wqtotp	ppm	12291	0.941	1.3	12282	99.93	0	711228	0.01	690731	48	760707	0.941	0.941
wqvols	ppm	984	2990	2500	984	100	0	760419	2	750203	17000	730113	2990	2990
wqvss	ppm	10663	11.8	13	10308	96.67	0	630402	1	631015	500	730426	11.4	11.5
wqo&g	ppm	1245	5.12	8.3	596	47.87	0	750403	0.2	750709	100	860822	2.45	3.03
wqtoc	ppm	7278	12.4	11	6615	90.89	0	740212	0.13	890719	230	840507	11.2	11.8
wqbod5	ppm	9520	5.93	110	9051	95.07	0	500605	0.1	500831	10000	740710	5.64	5.75
wqcobd5	ppm	308	2.67	1.6	294	95.45	0.5	671001	0.5	671001	16	820804	2.55	2.59
wqchla	ppb	4705	18.8	25	4058	86.25	0	760831	0.2	770915	460	780322	16.2	16.4
wqtcoli	/100 ml	17061	61000	770000	16263	95.32	0	770125	0.8	710316	8x10 <sup>7</sup>	750219	58200	58200
wqfcoli	/100 ml	19745	12000	110000	18032	91.32	0	740515	2	680402	3.5x10 <sup>6</sup>	690819	11000	11000

TABLE 3-8

SUMMARY OF WATER-QUALITY DATA FROM GALVESTON BAY: METALS  
All units in micrograms per liter  
(Unscreened)

Parameter (Table 3-11)	No. of obs	Avg >DL	St dev	No. > DLs	% > DLs	Min >DL	date	Min >0	date	Max	date	Avg w/ BDL=0	Avg w/ BDL=DL
wqmetast	1830	17.2	48	965	52.7	0	730911	0.0029	730911	1200	800212	9.1	13.4
wqmetasd	33	4.67	1.4	5	15.2	2.4	890516	2.4	890516	6	900524	0.708	5.34
wqmetbat	396	161	130	347	87.6	0.07	880406	0.07	880406	800	741119	141	171
wqmetbad	24	84	76	24	100.	30	900815	30	900815	430	900606	84	84
wqmetb	2	300	-	2	100.	300	840925	300	840925	300	840925	300	300
wqmetcdt	1988	14.2	14	505	25.4	0	730911	0.002	730911	120	750828	3.62	10.1
wqmetcdd	65	1.36	0.64	26	40.	0.5	750412	0.5	750412	3.8	750611	0.543	1.47
wqmetcrt	1953	26.5	30	721	36.9	0	730911	0.013	730911	610	761122	9.78	22.8
wqmetcrd	48	7.56	3.5	8	16.7	2.3	750507	2.3	750507	11	750328	1.26	8.57
wqmetcut	2030	98.7	130	1297	63.9	0	730911	0.03	880406	1400	840925	63.1	68.1
wqmetcud	80	3.76	3.2	38	47.5	0.24	671015	0.24	671015	16	901008	1.78	5.73
wqmetfet	379	1430	2600	379	100.	1	760518	1	760518	28000	750930	1430	1430
wqmetfed	57	76.9	91	42	73.7	5	750328	5	750328	490	900524	56.7	58.1
wqmetpbt	2023	58.6	130	809	40.	0	730911	0.06	730911	1900	791204	23.4	51.2
wqmetpbd	80	9.03	12	31	38.8	1	750328	1	750328	64	890516	3.5	4.73
wqmetmnt	1041	124	88	926	89.	0	760907	0.065	880406	1000	750709	111	114
wqmetmnd	63	35	45	30	47.6	0.3	671015	0.3	671015	170	671215	16.7	20.3
wqmethgt	2044	1.49	7.8	882	43.2	0	750709	0.02	820428	210	740716	0.641	0.873
wqmethgd	62	0.828	3.4	44	71.	0.003	750920	0.003	750920	23	900719	0.588	0.653
wqmetnit	1726	72.9	100	894	51.8	0	730911	0.045	880406	2000	760609	37.8	46.8
wqmetnid	70	12.8	9.4	33	47.1	1.5	671215	1.5	671215	40	900222	6.02	9.84
wqmetset	344	21.4	23	72	20.9	2	730711	2	730711	100	880223	4.48	6.83
wqmetсед	35	-	-	0	0.	-	-	-	-	-	-	0	5
wqmetagt	441	16.1	13	215	48.8	0	730911	0.006	730911	130	730911	7.83	13.9
wqmetagd	35	16	-	1	2.9	16	900606	16	900606	16	900606	0.457	18.7
wqmetznt	1818	145	270	1345	74.	0	730911	0.061	730911	4600	750227	107	113
wqmetznd	78	20.6	23	71	91.	1.3	750611	1.3	750611	140	900402	18.8	19.3

TABLE 3-9

SUMMARY OF WATER-QUALITY DATA FROM GALVESTON BAY: ORGANICS  
 All units in micrograms per liter  
 (Unscreened)

Parameter (Table 3-11)	No. of obs	Avg >DL	St dev	No. > DLs	% > DLs	Min >DL	date	Min >0	date	Max	date	Avg w/ BDL=0	Avg w/ BDL=DL
wq-abhc	24	-	-	0	0	-	-	-	-	-	-	0	0.0267
wq-lind	235	0.032	0.004	5	2.1	0.03	850716	0.03	850716	0.04	850716	0.000681	0.183
wq-ddd	81	-	-	0	0	-	-	-	-	-	-	0	0.106
wq-dde	81	3.01	3	2	2.4	0.024	750715	0.024	750715	6	740521	0.0744	0.153
wq-ddt	81	5.41	7.3	3	3.7	0.092	731011	0.092	731011	16	740521	0.2	0.311
wq-pdde	278	0.014	0.0049	5	1.8	0.01	800619	0.01	800619	0.02	800619	0.000252	0.023
wq-pddt	461	0.0811	0.094	9	1.9	0	851003	0.03	800619	0.3	891115	0.00158	0.0248
wq-xddt	542	1.44	4.3	12	2.2	0	851003	0.042	800619	16	740521	0.0318	0.0759
wq-aldr	338	1.08	2.7	8	2.4	0.018	750715	0.018	750715	8.3	740521	0.0256	0.0478
wq-chlr	437	0.1	-	1	0.2	0.1	851018	0.1	851018	0.1	851018	0.000229	0.594
wq-diel	339	0.16	0.3	9	2.6	0.04	800619	0.04	800619	1	710628	0.00425	0.0403
wq-endo	124	0.01	0	5	4.0	0.01	800619	0.01	800619	0.01	800619	0.000403	0.0242
wq-endr	82	0.504	0.5	2	2.4	0.007	750522	0.007	750522	1	710628	0.0123	0.115
wq-toxa	427	-	-	-	-	-	-	-	-	-	-	0	0.966
wq-hept	279	0.0458	0.1	6	2.2	0	740507	0.005	750520	0.27	780815	0.000986	0.0207
wq-hepx	97	0	0	4	4.1	0	740507	-	-	-	-	0	0.0377
wq-mthx	95	0.1	0.2	5	5.3	0	740507	0.5	810805	0.5	810805	0.00526	0.345
wq-pcb	533	2.43	2.7	25	4.7	0	740425	0.03	751023	10	860325	0.114	0.696
wq-mala	65	0	-	2	3.1	0	760803	-	-	-	-	0	0.459
wq-para	65	0	-	2	3.1	0	760803	-	-	-	-	0	0.235
wq-diaz	56	1.04	0.98	6	10.7	0.24	860911	0.24	860911	3.2	860711	0.112	0.468
wq-mthp	51	0	-	-	-	-	-	-	-	-	-	0	0.262
wq-24d	58	0	-	2	3.4	0	760803	-	-	-	-	0	19
wq-245t	58	0.02	0.028	3	5.2	0	760803	0.06	860806	0.06	860806	0.00103	4.57
wq-pah	238	12.5	9.8	22	9.2	1.6	870825	1.6	870825	41	851003	1.16	5.09
wq-napt	180	0	0	48	26.7	0	890615	-	-	-	-	0	1.47
wq-acen	180	0	0	48	26.7	0	890615	-	-	-	-	0	1.47
wq-flra	180	92.9	81	2	1.1	12	870909	12	870909	170	861001	1.03	1.93
wq-bnza	180	9.08	65	53	29.4	0	890615	0.2	870825	480	861001	2.67	3.03

TABLE 3-10

SUMMARY OF SEDIMENT DATA FROM GALVESTON BAY  
(Unscreened)

Parameter (Table 3-11)	units	No. of obs	Avg >DL	St dev	No. > DLs	% > DLs	Min >DL	date	Min >0	date	Max	date	Avg w/ BDL=0	Avg w/ BDL=DL
sedorgn	mg/kg	457	1100	4400	446	97.6	0.8	731011	0.8	731011	56000	780907	1080	1080
sedtotp	mg/kg	401	577	550	400	99.8	1.5	750520	1.5	750520	5200	870929	575	575
sedo&g	mg/kg	1233	1250	2600	1184	96.0	0	750220	0.049	731024	32000	751106	1200	1200
sedkjln	mg/kg	724	1080	1200	714	98.6	3	801211	3	801211	10000	770524	1060	1060
sedtoc	mg/kg	895	8.6	9.1	895	100.0	0.1	720410	0.1	720410	200	761003	8.6	8.6
sedvols	mg/kg	889	57100	84000	889	100.0	80	760712	80	760712	980000	740515	57100	57100
sedmetas	ug/kg	1407	4.73	4.5	1251	88.9	0	740703	0.1	780418	50	800825	4.21	4.32
sedmetba	ug/kg	616	225	170	616	100.0	1	751112	1	751112	1600	761003	225	225
sedmetb	ug/kg	239	54.8	28	236	98.7	3.3	760912	3.3	760912	120	761003	54.1	54.2
sedmetcd	ug/kg	1531	1.71	2.8	727	47.5	0	840731	0.01	750520	41	750412	0.81	1.18
sedmetcr	ug/kg	1705	30.3	36	1614	94.7	0.02	750520	0.02	750520	650	750328	28.7	28.9
sedmetcu	ug/kg	1783	19.6	37	1692	94.9	0.01	750520	0.01	750520	830	671215	18.6	18.8
sedmetfe	ug/kg	364	15900	11000	364	100.0	800	720410	800	720410	94000	750412	15900	15900
sedmetpb	ug/kg	1790	36.9	70	1584	88.5	0.05	750520	0.05	750520	2000	771115	32.6	33.8
sedmetmn	ug/kg	826	390	320	826	100.0	21	740703	21	740703	4700	750328	390	390
sedmethg	ug/kg	1526	0.329	1.2	755	49.5	0	740703	0.004	720410	19	780815	0.163	0.223
sedmetni	ug/kg	1641	17.2	22	1542	94.0	0	750711	0.5	780414	720	750328	16.1	16.5
sedmetse	ug/kg	404	0.827	0.89	89	22.0	0	840731	0.08	821206	3.8	830105	0.182	0.897
sedmetag	ug/kg	63	1.01	2.5	47	74.6	0	840731	0.049	840731	12	720123	0.755	1.14
sedmetSr	ug/kg	250	156	200	250	100.0	13	760911	13	760911	1400	761003	156	156
sedmetzn	ug/kg	1785	202	4700	1704	95.5	1.9	870909	1.9	870909	190000	790802	193	194
sed-abhc	ug/kg	68	0.761	1.1	19	27.9	0.01	719994	0.01	719994	3.3	719994	0.213	0.717
sed-lind	ug/kg	505	4.7	13	65	12.9	0	731024	0.02	719994	100	760915	0.604	1.73
sed-ddt	ug/kg	211	435	1400	39	18.5	0	731024	0.15	780525	6100	740515	80.3	84.4
sed-dde	ug/kg	210	190	610	38	18.1	0	731024	0.29	780525	2300	740515	34.4	37.2
sed-ddd	ug/kg	190	43.3	89	17	9.0	0	731024	1.9	780515	300	760915	3.88	7.89
sed-aldr	ug/kg	527	356	1500	51	9.7	0	731024	0.04	711228	7300	740515	34.5	35.5
sed-chlr	ug/kg	617	164	360	47	7.6	0.06	711228	0.06	711228	2000	760915	12.5	14.3
sed-diel	ug/kg	527	10.1	38	66	12.5	0	731024	0.06	711228	300	760915	1.27	2.7
sed-endo	ug/kg	55	0.606	0.3	2	3.6	0.3	780525	0.3	780525	0.91	780525	0.022	7.09
sed-endr	ug/kg	231	18.6	70	17	7.4	0	731024	0.12	720713	300	760915	1.37	5.01

Table 3-10  
(continued)

Parameter (Table 3-11)	units	No. of obs	Avg >DL	St dev	No. > DLs	% > DLs	Min >DL	date	Min >0	date	Max	date	Avg w/ BDL=0	Avg w/ BDL=DL
sed-toxa	ug/kg	632	561	1600	9	1.4	0	731024	50	760916	5000	760915	7.99	25.6
sed-hept	ug/kg	443	3.29	16	36	8.1	0	731024	0.02	720411	100	760915	0.268	1.35
sed-hepx	ug/kg	251	5.51	20	25	10.0	0	731024	0.06	711228	100	760915	0.549	1.27
sed-mthx	ug/kg	190	184	570	11	5.8	0	731024	20	760916	2000	760915	10.6	19.7
sed-pcb	ug/kg	282	442	1100	49	17.4	0	731024	7.8	770915	6400	800124	76.8	91.1
sed-mala	ug/kg	121	-	-	0	0.0	-	-	-	-	-	-	0	5.55
sed-para	ug/kg	152	253	250	2	1.3	5	760916	5	760916	500	760915	3.32	6.47
sed-diaz	ug/kg	141	68.9	150	9	6.4	1.4	860930	1.4	860930	500	760915	4.39	9.93
sed-mthp	ug/kg	147	110	200	5	3.4	5	760916	5	760916	500	760915	3.73	6.94
sed-24d	ug/kg	52	2.58	0	1	1.9	2.6	870917	2.6	870917	2.6	870917	0.0496	33.7
sed-245t	ug/kg	52	-	-	0	0.0	-	-	-	-	-	-	0	6.8
sed-pddd	ug/kg	249	6.05	13	37	14.9	0.21	870731	0.21	870731	71	840106	0.899	1.26
sed-pdde	ug/kg	361	5.2	9.2	45	12.5	0.04	721229	0.04	721229	51	810805	0.649	1.1
sed-pddt	ug/kg	542	13.2	25	31	5.7	0	890615	0.24	711228	120	880727	0.753	1.2
sed-pah	ug/kg	261	5.21	14	99	37.9	0.1	880328	0.1	880328	130	870420	1.98	7.61
sed-napt	ug/kg	230	31.6	49	14	6.1	1.2	870718	1.2	870718	150	870825	1.92	43.2
sed-acen	ug/kg	232	18.1	21	13	5.6	5.9	870825	5.9	870825	83	870825	1.02	42.2
sed-flra	ug/kg	232	123	320	85	36.6	1	870226	1	870226	1900	870825	45.2	51.4
sed-bnza	ug/kg	232	132	380	74	31.9	1	870226	1	870226	2400	870825	42	48.5

TABLE 3-11

ABBREVIATIONS FOR WATER AND  
SEDIMENT QUALITY PARAMETERS  
(See Table A-1 for units and data sources)

*Conventional Parameters*

WQTEMP	temperature of water
WQSAL	salinity of water, converted from various proxy measures
WQDO	dissolved oxygen in water
WQDODEF	dissolved oxygen deficit in water
WQPH	pH of water
WQTURB	turbidity of water
WQSECCHI	Secchi depth of water
WQTSS	total suspended solids in water
WQXTSS	extended total suspended solids in water, based on proxy data
WQAMMN	ammonia nitrogen in water
WQORGN	total organic nitrogen in water
WQKJLN	total Kjeldahl nitrogen in water
WQNO3N	nitrate nitrogen in water
WQTOTP	total phosphorus (as P) in water
WQVOLS	total volatile solids in water
WQVSS	volatile suspended solids in water
WQO&G	oil & grease in water
WQTOC	total organic carbon in water
WQBOD5	5-day biochemical oxygen demand
WQCBOD5	5-day biochemical oxygen demand, nitrification-suppressed
WQXBOD5	extended record of 5-day BOD, based on proxy relationships
WQCHLA	chlorophyll-a in water
WQTCOLI	total coliforms in water
WQFCOLI	fecal coliforms in water

*Metals*

WQMETAST	total arsenic in water (i.e., unfiltered)
WQMETASD	dissolved arsenic in water
WQMETBAT	total barium in water
WQMETBAD	dissolved barium in water
WQMETB	total boron in water
WQMETCDT	total cadmium in water
WQMETCDD	dissolved cadmium in water
WQMETCRT	total chromium in water
WQMETCRD	dissolved chromium in water
WQMETCUT	total copper in water
WQMETCUD	dissolved copper in water
WQMETFET	total iron in water
WQMETFED	dissolved iron in water
WQMETPBT	total lead in water
WQMETPBD	dissolved lead in water

TABLE 3-11  
(continued)

WQMETMNT	total manganese in water
WQMETMND	dissolved manganese in water
WQMETHGT	total mercury in water
WQMETHGD	dissolved mercury in water
WQMETNIT	total nickel in water
WQMETNID	dissolved nickel in water
WQMETSET	total selenium in water
WQMETSED	dissolved selenium in water
WQMETAGT	total silver in water
WQMETAGD	dissolved silver in water
WQMETZNT	total zinc in water
WQMETZND	dissolved zinc in water
<i>Organics</i>	
WQ-ABHC	alpha-BHC in water
WQ-LIND	lindane (gamma-BHC) in water
WQ-DDT	Total DDT in water
WQ-DDE	Total DDE in water
WQ-DDD	Total DDD in water
WQ-PDDD	p,p'-DDD in water
WQ-PDDE	p,p'-DDE in water
WQ-PDDT	p,p'-DDT in water
WQ-XDDT	extended DDT in water, based on proxy relation
WQ-ALDR	Aldrin in water
WQ-CHLR	Total chlordane in water
WQ-DIEL	Dieldrin in water
WQ-ENDO	Endosulfan I in water
WQ-ENDR	Endrin in water
WQ-TOXA	Toxaphene in water
WQ-HEPT	Heptachloride in water
WQ-HEPX	Heptachloride epoxide in water
WQ-MTHX	methoxychlor in water
WQ-PCB	Total PCB's in water
WQ-MALA	Malathion in water
WQ-PARA	Parathion in water
WQ-DIAZ	Diazinon in water
WQ-MTHP	methyl parathion in water
WQ-24D	2,4 D in water
WQ-245T	2,4,5 T in water
WQ-PAH	total PAH's in water
WQ-NAPT	naphthalene in water
WQ-ACEN	acenaphthene in water
WQ-FLRA	fluoranthene in water
WQ-BNZA	benzo(a)pyrene in water

TABLE 3-11  
(continued)

*Conventional Parameters*

SEDORGN	total organic nitrogen in sediment
SEDTOTP	total phosphorus (as P) in sediment
SEDO&G	oil & grease in sediment
SEDKJLN	total Kjeldahl nitrogen in sediment
SEDTOC	total organic carbon in sediment
SEDVOLS	volatile solids in sediment

*Metals*

SEDMETAS	arsenic in sediment
SEDMETBA	barium in sediment
SEDMETB	boron in sediment
SEDMETCD	cadmium in sediment
SEDMETCR	chromium in sediment
SEDMETCU	copper in sediment
SEDMETFE	iron in sediment
SEDMETPB	lead in sediment
SEDMETMN	manganese in sediment
SEDMETHG	mercury in sediment
SEDMETNI	nickel in sediment
SEDMETSE	selenium in sediment
SEDMETAG	silver in sediment
SEDMETSR	strontium in sediment
SEDMETZN	zinc in sediment

*Organics*

SED-ABHC	alpha-BHC in sediment
SED-LIND	lindane (gamma-BHC) in sediment
SED-DDT	Total DDT in sediment
SED-DDE	Total DDE in sediment
SED-DDD	Total DDD in sediment
SED-ALDR	Aldrin in sediment
SED-CHLR	Total chlordane in sediment
SED-DIEL	Dieldrin in sediment
SED-ENDO	Endosulfan I in sediment
SED-ENDR	Endrin in sediment
SED-TOXA	Toxaphene in sediment
SED-HEPT	Heptachloride in sediment
SED-HEPX	Heptachloride epoxide in sediment
SED-MTHX	methoxychlor in sediment
SED-PCB	Total PCB's in sediment
SED-MALA	Malathion in sediment
SED-PARA	Parathion in sediment
SED-DIAZ	Diazinon in sediment
SED-MTHP	methyl parathion in sediment

TABLE 3-11  
(continued)

SED-24D	2,4 D in sediment
SED-245T	2,4,5 T in sediment
SED-PDDD	p,p'-DDD in sediment
SED-PDDE	p,p'-DDE in sediment
SED-PDDT	p,p'-DDT in sediment
SED-PAH	total PAH's in sediment
SED-NAPT	naphthalene in sediment
SED-ACEN	acenaphthene in sediment
SED-FLRA	fluoranthene in sediment
SED-BNZA	benzo(a)pyrene in sediment

In our view, the selection is dependent upon the purpose at hand. The non-BDL statistics can provide some insight into the precision and variability of the parameter, which the more constant DL values would corrupt or even mask. However, to completely ignore BDL results is to lose information, albeit non-quantitative. The fact is that a water or sediment sample was obtained (usually at great effort), a careful analysis performed, and an upper bound established on the concentration of the parameter. This information should not be dismissed cavalierly. The latter two alternatives use that information, either optimistically or pessimistically, depending upon the intent of the analyst. In this project, with typical equivocation, we decided to employ all three, i.e. to compute *appropriate* statistics with only above-DL data, with the BDL values set to zero and with the BDL values set to the DL, thereby establishing a probable *range* of the statistic. The "appropriate" statistics include averages and variability for the above-DL data, but do not include calculations of variability for the latter two, since the largely invariant values of either end of the the range (i.e. either value assumed for a BDL measurement) would distort the results. Even in a trends analysis (which is variability in time), to incorporate 0 or DL values might either mask any vestige of a real trend by padding the data with zeroes or displace the real trend with a trend of measurement sensitivity. The user of these results therefore can choose among them whichever best serves the purpose of the analysis.

### 3.3.3 Deficiencies

The general adequacy of coverage in Galveston Bay for each parameter is an important dimension of this compilation. This includes identification of data gaps in the record and their associated implications for application of the record to long-term trend analysis (seasonal bias is a common problem, for example). It also includes procedural shortcomings that prejudice or limit the applicability of data collected. These are given here for specific parameters and classes of parameters, and suggestions are offered on alterations in procedures for monitoring in future programs to assist filling data gaps or repairing data deficiencies. Emphasis is on procedural modifications that can be implemented with little or no cost, and that will not interfere with the objectives of the primary agency but will greatly augment the value of the data.

We first propose four key precepts of data collection, because it is the violation of these principles that contribute to data deficiencies in the record that are avoidable or correctable at little cost.

(1) We re-emphasize that Galveston Bay is a highly variable environment, subject to many external factors, each of which contributes a degree of "noise" in any measured parameter. To filter this noise, and expose variations in time and space, requires that sufficient independent measurements be available over the range of variation of the external factors. For time variability, continuity of data record is an all-important property of any data base. For space variability, a high density of sampling stations repeatedly sampled is necessary. (The actual

intensity of sampling is determined by the intrinsic variability of the parameter of concern.)

(2) Incremental cost, not project objective, should be the governing criterion for delineation of a suite of measurements. Generally, a large investment is required to obtain the basic sample. This cost is dominated by operations: putting a sampling crew (and usually a boat) on a specific station, or installing an automatic data logger on a platform in the bay. The incremental cost in acquiring additional measurements (including loss of efficiency) must be weighed against the cost of occupying the station, in specifying the suite of parameters to be obtained. Whether additional parameters have direct application in the project is unimportant; they may be peripheral or irrelevant to the objective of the project, but have great value for other objectives and therefore justify the small incremental cost for their acquisition. The same principle of incremental cost versus benefits should be considered in specifying laboratory analyses. Many procedures, e.g. mass spectrometry or grain-size by settling tube, are cost-loaded in technician training and sample preparation, and can admit additional parameters or greater resolution with minor incremental cost. Again, a certain altruistic philosophy is necessary in the sampling agency, to acquire measurements that may be irrelevant to the immediate objective, but from which others will benefit.

(3) What parameters are measured should also be determined from the historical perspective. Extending a past data record may be sufficient to justify including a parameter, even if modern analysis and technology suggest a more useful variate. In particular, when a new parameter is inducted into an ongoing survey to replace a less satisfactory parameter, measurements of both the new and the old parameters should be performed in order to establish (or falsify) the relation between them.

(4) Recording and processing of the data should be performed with great sensitivity to potential loss of information. Replacing a series of raw measurements over time or space by an average, failing to preserve information on sampling time, position or conditions, or intermixing actual measurements with "estimated" values without any means of separation, all represent losses of information, and are all practices that can be avoided with a care and forethought. Data entry (i.e., transcription) errors are a prime cause of information loss, and any data entry procedure should include a means of verification. Again, there is a benefit-to-cost issue that must be considered in its largest dimension. The implementation of routine data checking procedures may be viewed (myopically) as a redundant cost item in the data management process, but shrinks to negligibility compared to the unit cost of acquiring and analyzing a water sample, and is an absolutely indispensable investment to preserve the information in that sample analysis.

Specific deficiencies in the Galveston Bay record include the following:

(A) Even after the efforts of this project to locate, compile and synthesize a "complete" digital data base, the period of record extends back only to about 1965 for conventional indicators, and 1975 for metals and organics. For some traditional parameters, and for certain areas of the bay, e.g., the confined reach of the Houston Ship Channel, the record can be extended back to the late 1950's. Beyond this, what data still survive are sporadic in time, e.g. a few coliform and salinity measurements in the early 1950's, a few salinity/temperature measurements from 1939, a handful of dissolved oxygen measurements from Offatts Bayou in the early 1930's. This earlier data is generally inadequate in space and time continuity for any reliable characterization or trends analysis: data prior to 1950 are therefore excluded from the data base.

(B) As salinity and temperature are the most easily measured variables, they represent the densest and longest data record. Even at this, past and present sampling practice does not permit analysis of time scales of variation shorter than a few days. For temperature and dissolved oxygen, especially, there is a known diurnal variability which is virtually unsampled in Galveston Bay. Moreover, the extant measurements are nearly all for daytime hours, which must be considered a source of potential bias in the data. For salinity, in many areas of the bay (depending upon the presence of steep gradients), there is a tidal oscillation that is unsampled as well, but certainly a source of variability. The use of automatic data logging and electrometric sensing now permit the recovery of nearly continuous, fine-scale time signals of these parameters, and should be incorporated into routine monitoring of the bay, perhaps in association with tide gauging. NB, such data acquisition should not replace routine sampling, since such sampling provides far better spatial continuity and is not subject to vandalism and sensor degradation, which plague automatic monitors.

(C) One of the principal properties of the water of Galveston Bay is its turbidity. Suspended solids are particularly important in characterizing water quality because of the rôle particulates play in habitat quality, and in the sorption of nutrients and contaminants on the finer particulates. However, some programs do not obtain any measure of turbidity, and those few that do obtain suspended solids do not measure the grain-size distribution. Even a simple sequential filtration to determine partitioning of clays-and-finer would be of immense value in interpreting the data. The understanding of the behavior of most nutrients, metals, pesticides and priority pollutants is limited by the lack of information on suspended solids in the water column.

(D) One of the central problems in constructing a sufficiently dense and long-term data base for analysis for the conventional parameters is the inconsistency in measurements and analytical methodologies from one program to another. Some programs emphasize COD and sulfides, say, while another examines phytoplankton and TOC, and a third may analyze BOD and chlorophyll-a. (Certainly, research demands that the utility of different indicator variables be explored, and the specific objectives of a given program may necessitate non-conventional analyses. On the other hand, as noted in (2) above, the major

investment is in acquiring the sample. Some consistency with other sampling programs could be reasonably attained at minor cost.) The nutrients data of the extensive U.S. Bureau of Commercial Fisheries Program in the 1960's proved to be incompatible with the later data of the Texas Water Commission. The Galveston Bay Project did not obtain any suspended solids data whatever. The PAH suite obtained by the Corps of Engineers is different from the PAH measurements of the Water Commission; the National Ocean Service Status & Trends Program obtain a different suite from either of these. And so it goes. The net effect is limited data coverage in a specific parameter that makes spatio-temporal analysis uncertain. Even for salinity, were it not for reliable proxy relationships, our ability to synthesize a comprehensive data set would be seriously truncated.

(E) Metals data are dominated by total (unfiltered) analyses. So little measurement has been made of the dissolved phase that no characterization or trends analyses are possible. This practice is perhaps not inappropriate because of the known affinity of trace metals for particulates, but underscores the problem of not having paired measurements of suspended solids to which the total concentrations could be related. Future sampling should include routine measurement of suspended solids with every metals sample. It would be even better to include a determination of grain-size distribution, as noted in (C) above. Much of the historical data for metals has been corrupted by inattention to detection limits. Frequently detection limits are reported in error (perhaps not determined as a part of the analysis) or, worse, zero values of concentration are reported. As noted above, a well-defined detection limit provides at least some information as an upper bound on concentration.

(F) Pesticides and other organic contaminants are a recent addition to the suite of measurements, and the water-quality data base is presently inadequate for any detailed analyses. The best record is the extended DDT, obtained by combining reported "total" values with those estimated from the pp'-DDT isomer using the proxy relation (3-10). Even at this, only 12 observations are above detection limits in the entire bay. Interpretation of organic-contaminant data generally is based upon normalization to organic carbon (e.g., Karickhoff, 1981, Moore and Ramamoorthy, 1984a). For Galveston Bay, there are practically no paired measurements of organic carbon and organic contaminants.

(G) Sediment data is extremely limited for the bay. This is unfortunate because (1) the shallow nature of the bay would suggest that sediment interactions should be a significant factor in the quality of the overlying water and its habitat value, (2) sediment is considered to be a long-time-constant integrator of bay quality, compared to the variable and evanescent nature of the overlying water. While the number of observations given in Table 3-8 might appear to be large, they reduce to at most 100 observations (per parameter) per year over the period of record (and this is misleading, since the samples concentrate in a few specific years), which amounts to one sample per 5 square miles per year. For many metals and most organic pollutants, the data base is even smaller, and, moreover, only about 10% of the measurements are above detection limits.

(H) In recent years, there has been a shift of emphasis from rather gross and imprecise measurements such as oil & grease, volatile solids and total PAH's, to specific hydrocarbon parameters. In most cases, this has involved a replacement of the old parameter with the new, so that the data record for the older parameter terminates, the data available for the new parameter is extremely limited, and there is no information as to the probable association between the two. An example is the replacement of total PAH with a suite of specific PAH's such as naphthalene, acenaphthene and fluoranthene. (Further, in some programs, the specific PAH's vary from run to run.) Noting Precept (3) above, we suggest a continuation of data for each of the older variables together with the new parameters for at least several hundred samples (above detection limits).

(I) A major deficiency of the sediment data base is that there are almost no measurements of sediment texture (i.e. grain-size distribution). As noted in (C) above, many of the parameters of concern, such as heavy metals and pesticides, are known to have an affinity for fine-grained sediment, and moreover probably enter the system through run-off, also the source for most of the fine-grain fraction of sediment. Therefore, analysis of the variability of these quality parameters in the sediment must consider the grain-size fractions. Considering that sediment texture is an inexpensive measurement, especially compared to gas-liquid chromatography and spectrometry, it is inexplicable that texture data has not been routinely obtained for sediment samples. The BEG Submerged Lands Project is the only instance in which texture and metals data were both obtained, and even in this program only a minority of the samples were *paired*, i.e. chemical and textural analysis run on aliquots of the same sample. We recommend that texture analysis be instituted as a routine aspect of any chemical analysis of a sediment sample.

(J) Data management is generally poor. Reference is made to the conclusions of Ward and Armstrong (1991) concerning data management practices and data loss in general. We were most surprised to encounter major data management problems in the TWC SMN data base, which must be considered the central data repository for the system. These problems include data entry errors, position errors, and incorporation of "bogus" measurements into the data base together with real measurements. These "bogus" values are supplied in place of a variable that was not measured directly, and are: (a) derived from other measurements by "rules of thumb," which may or may not be approximately correct, but in any event should not be presented as a measurement, or (b) rough values, perhaps constant for an entire data run, that are "order of magnitude." We do not know the origin(s) of these practices, though indirect evidence suggests they originate at the TDH laboratory, but they should be terminated immediately, and the existing data base given careful study to expunge the bad entries, even if this requires tracking back to the original data forms. Certainly, part of the problem is the cumbersome data retrieval and transmittal practices of the SMN which frustrate graphical display and statistical processing of the data.



Fig. 3-10 Sampling density in Galveston Bay for WQTEMP

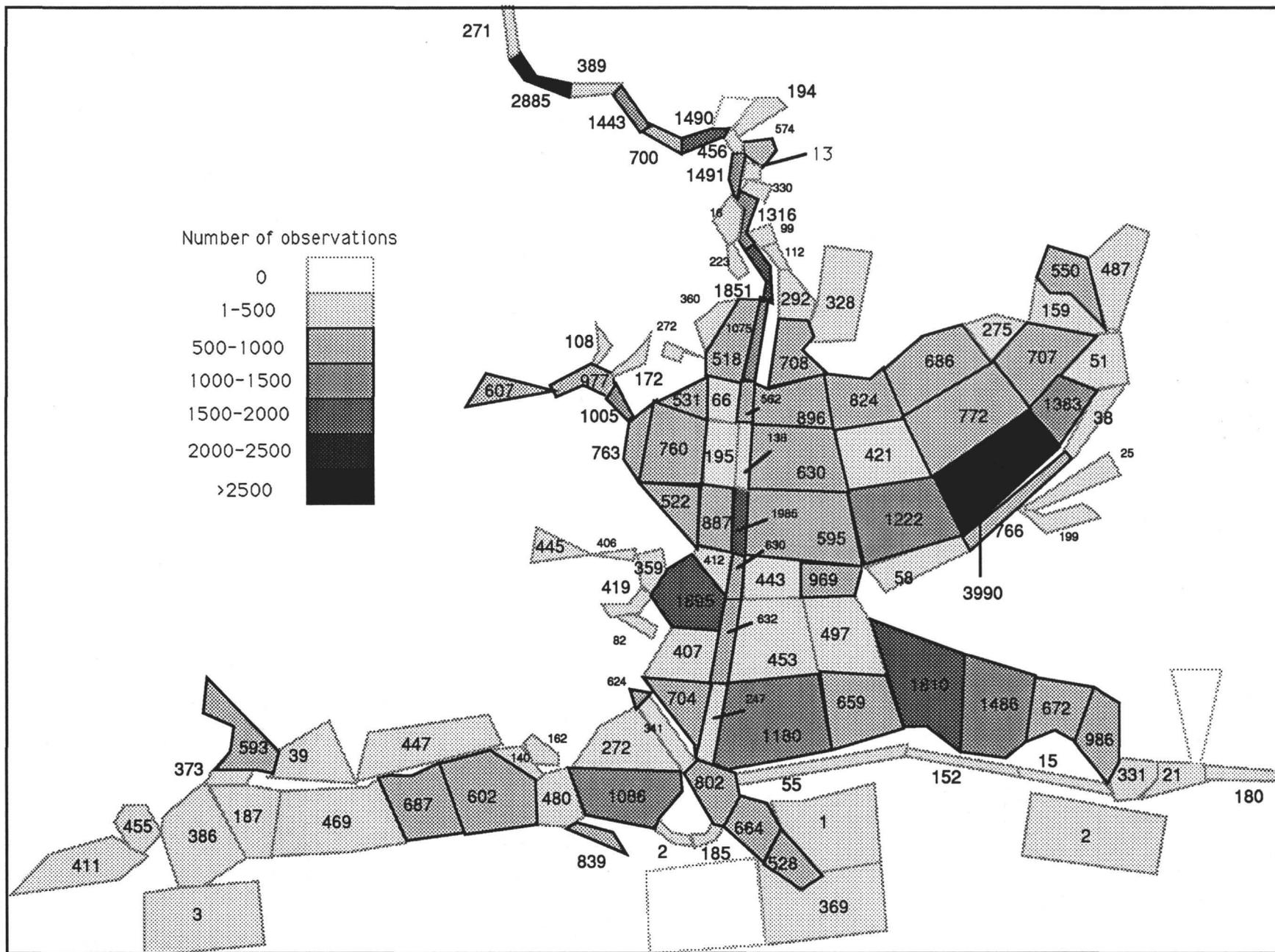


Fig. 3-11 Sampling density in Galveston Bay for WQSAL



Fig. 3-12 Sampling density in Galveston Bay for WQDO





Fig. 3-14 Sampling density in Galveston Bay for WQXTSS

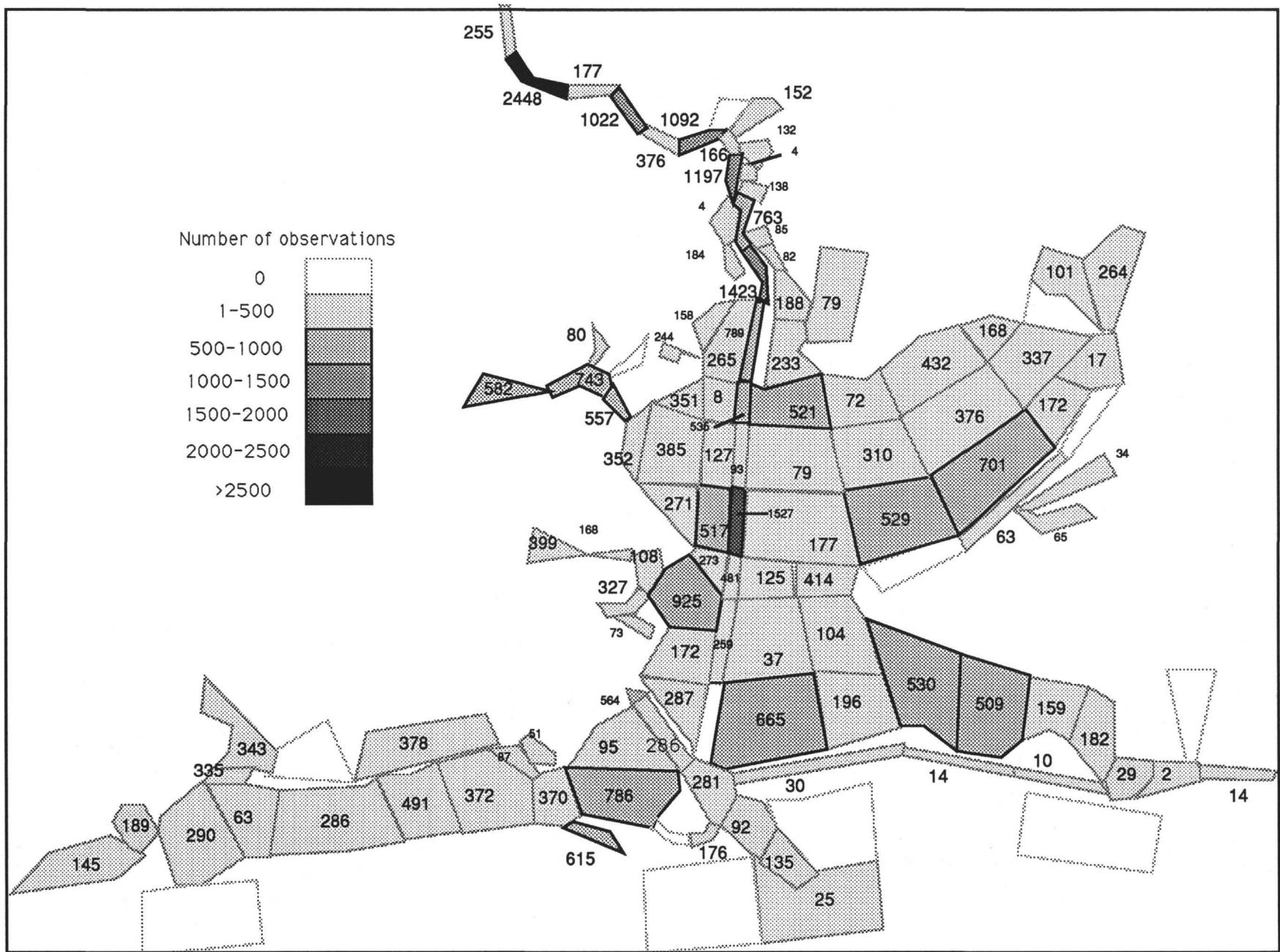


Fig. 3-15 Sampling density in Galveston Bay for WQPH





Fig. 3-17 Sampling density in Galveston Bay for WQKJLN



Fig. 3-18 Sampling density in Galveston Bay for WQAMMN



Fig. 3-19 Sampling density in Galveston Bay for WQTOC

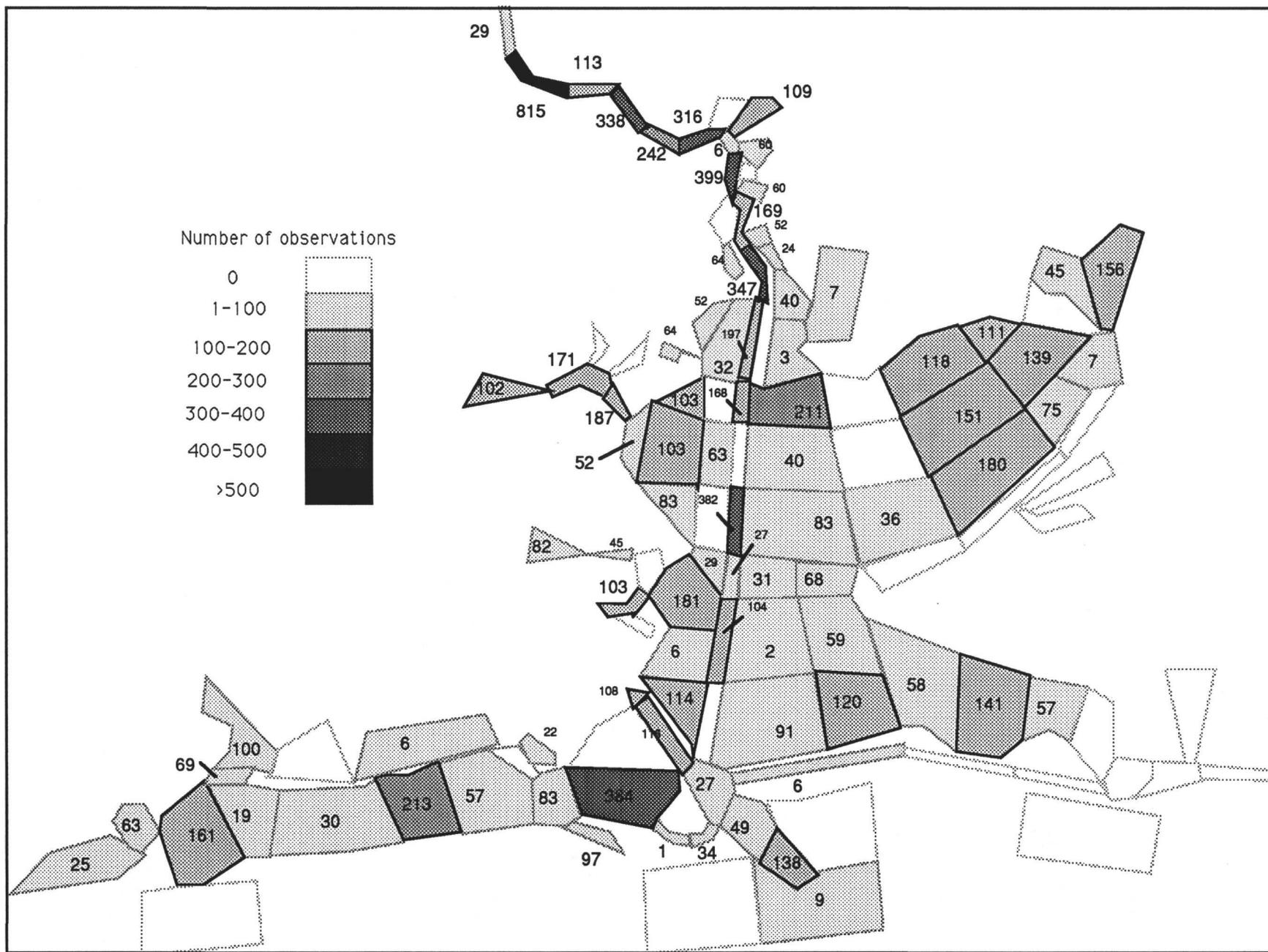


Fig. 3-20 Sampling density in Galveston Bay for WQTOTP

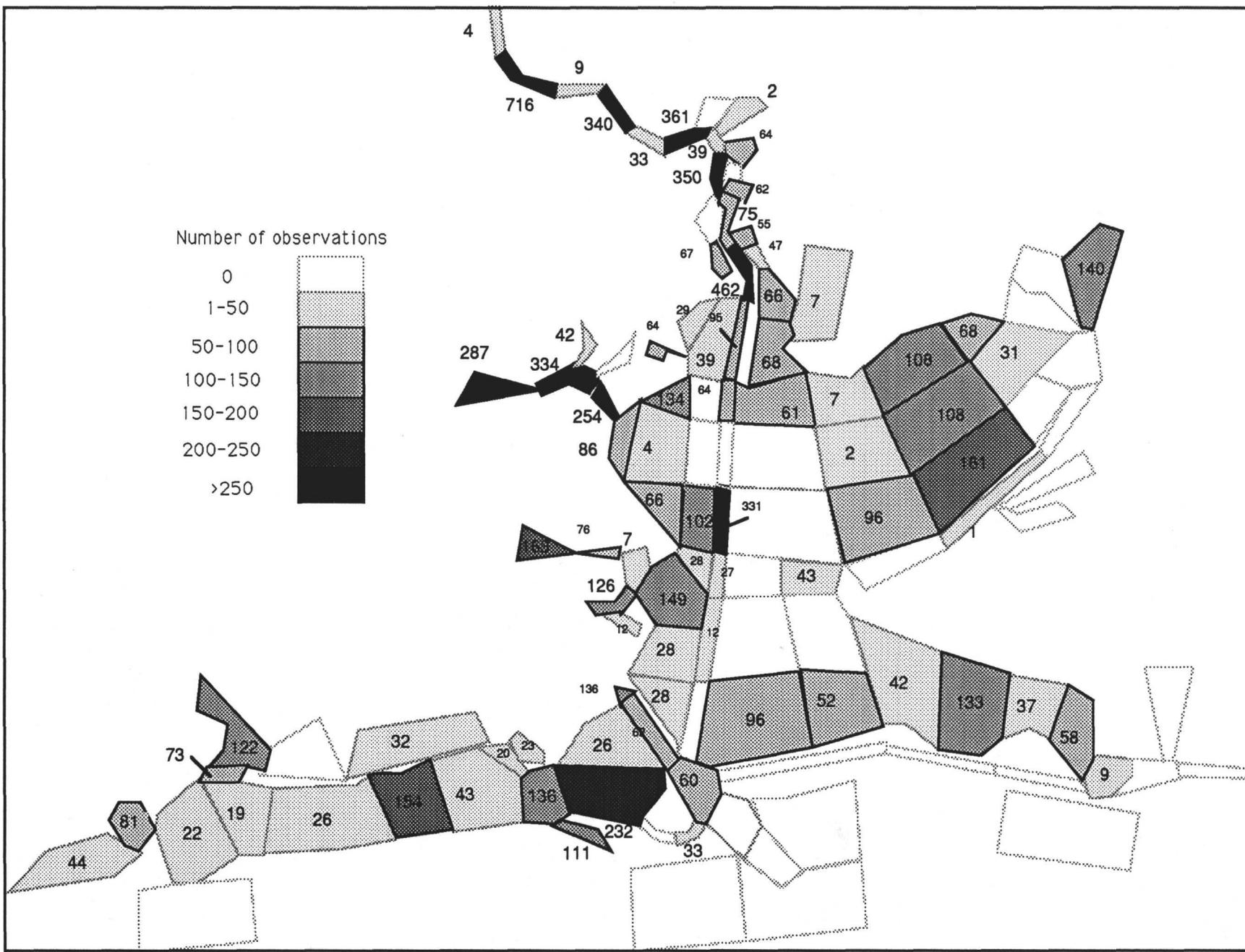


Fig. 3-21 Sampling density in Galveston Bay for WQVSS

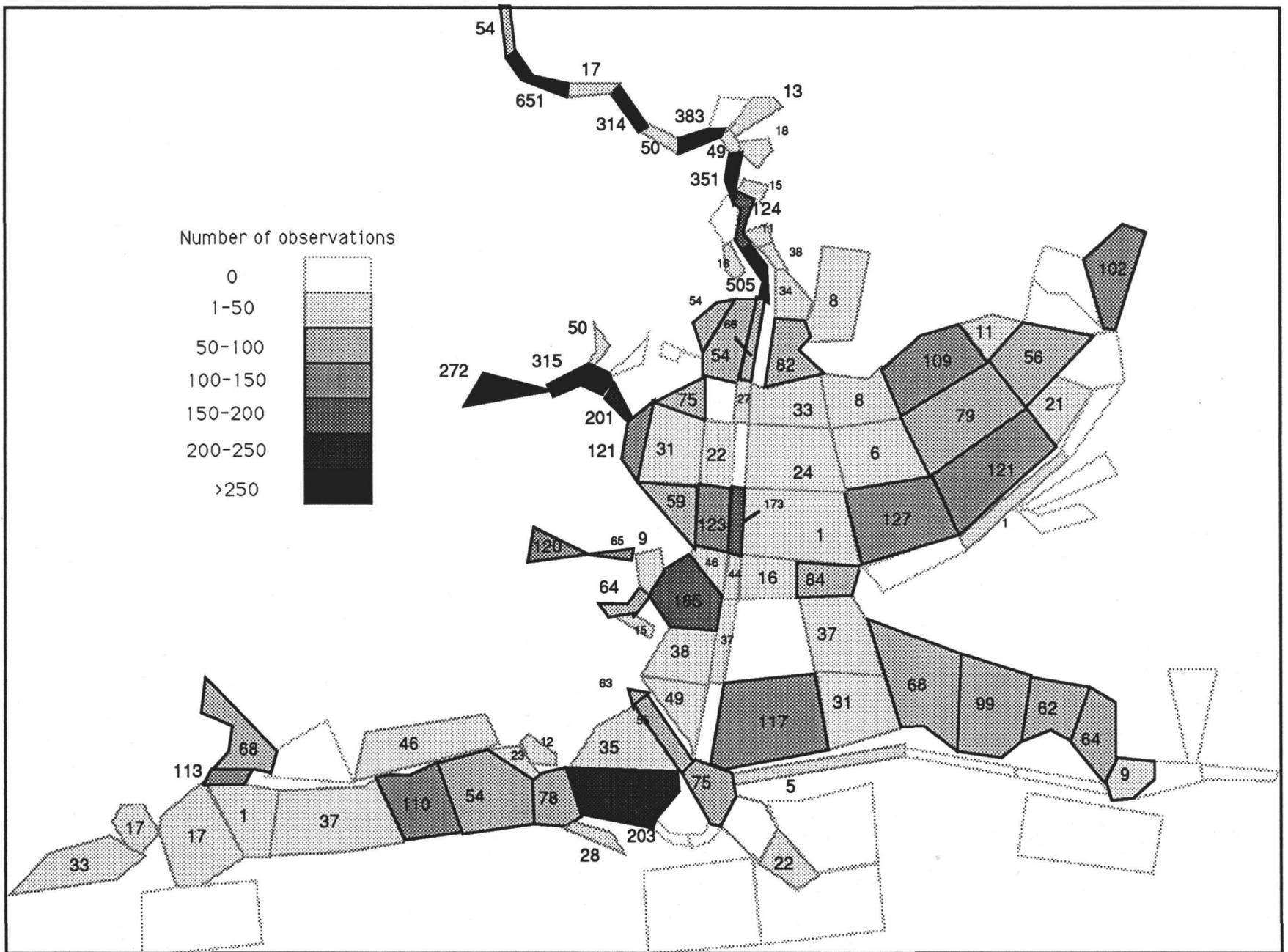


Fig. 3-22 Sampling density in Galveston Bay for WQXBOD5

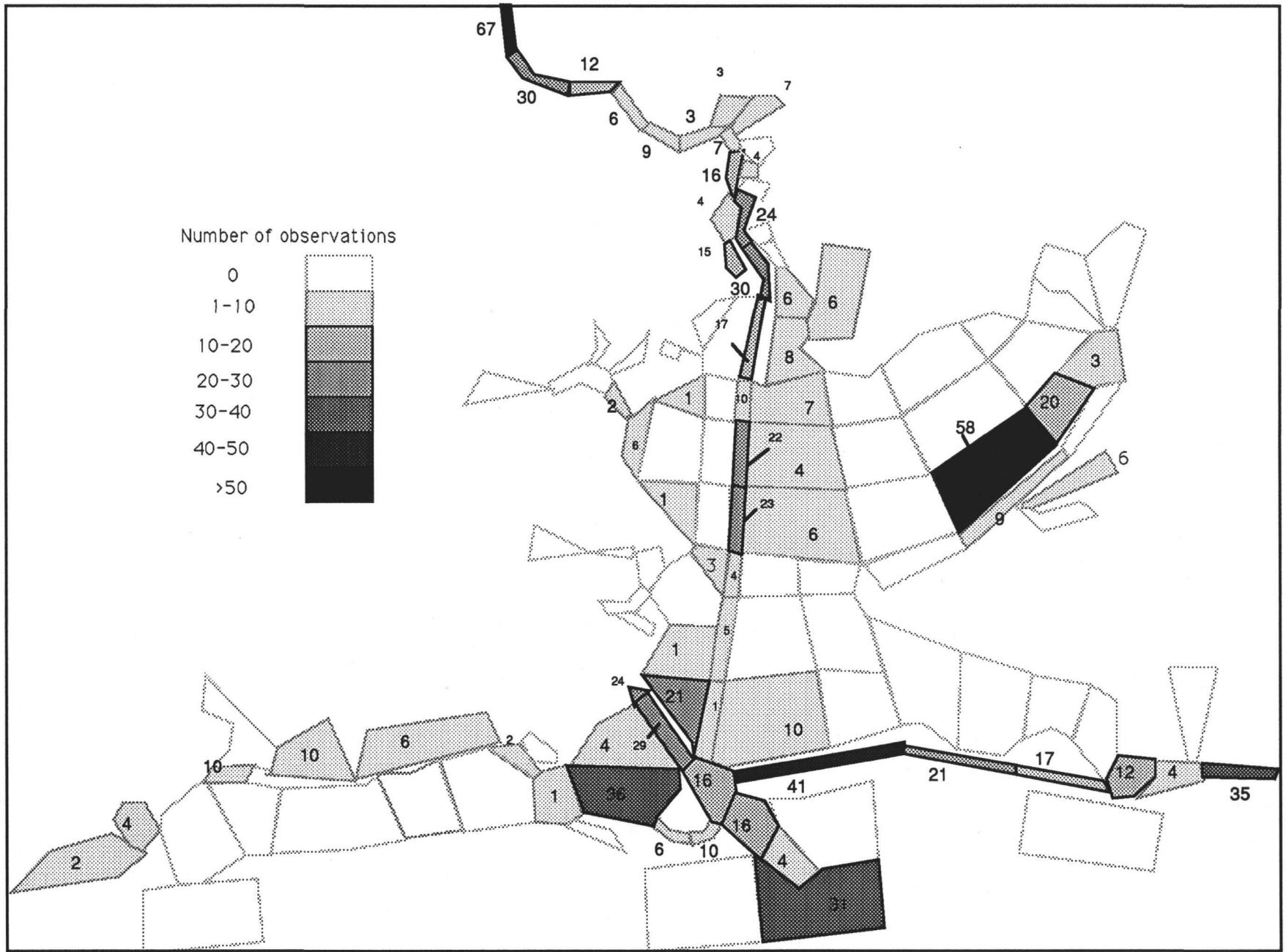


Fig. 3-23 Sampling density in Galveston Bay for WQO&G



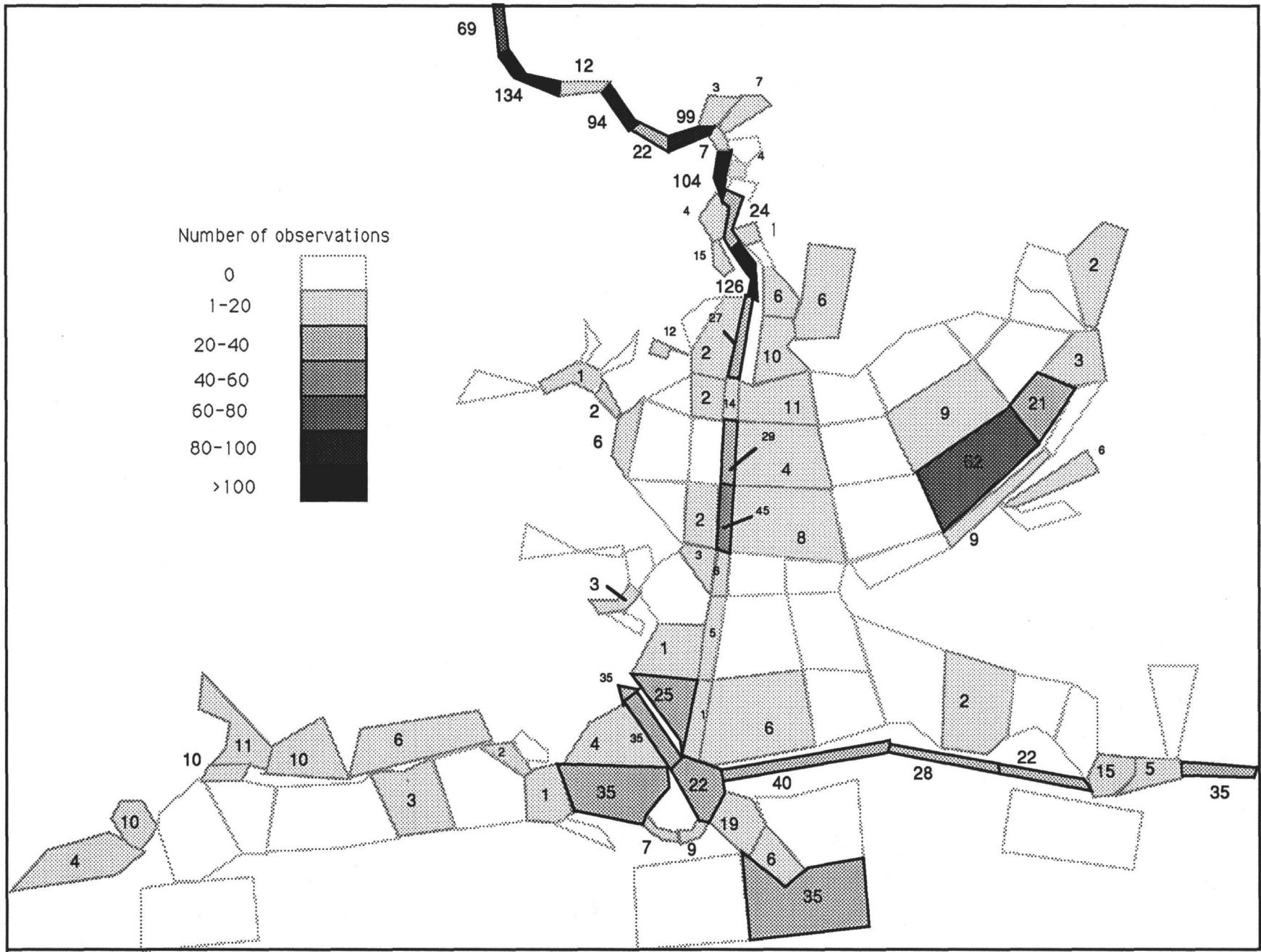


Fig. 3-25 Sampling density in Galveston Bay for WQMETCDT

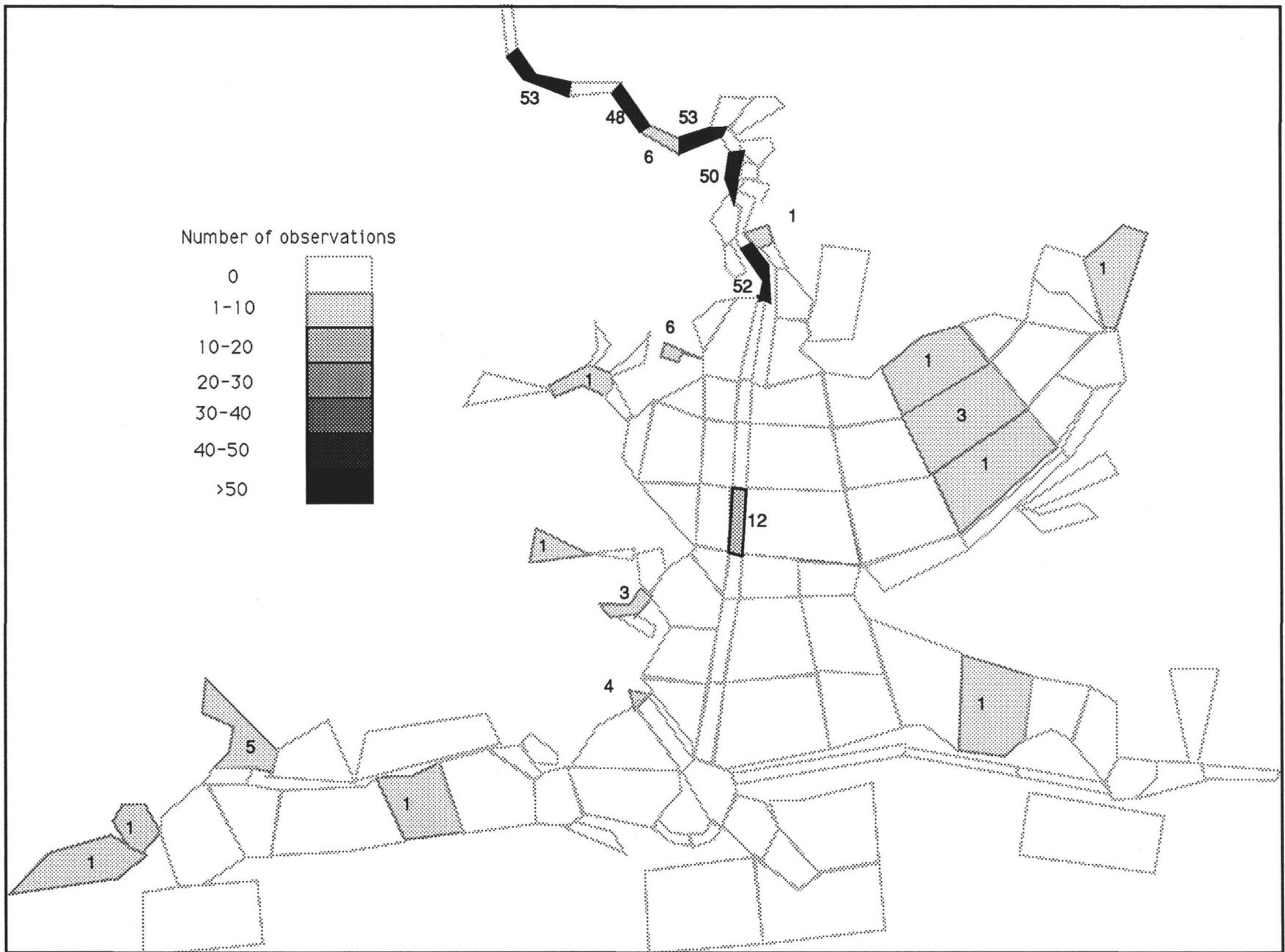


Fig. 3-26 Sampling density in Galveston Bay for WQMETFET



Fig. 3-27 Sampling density in Galveston Bay for WQMETPBT



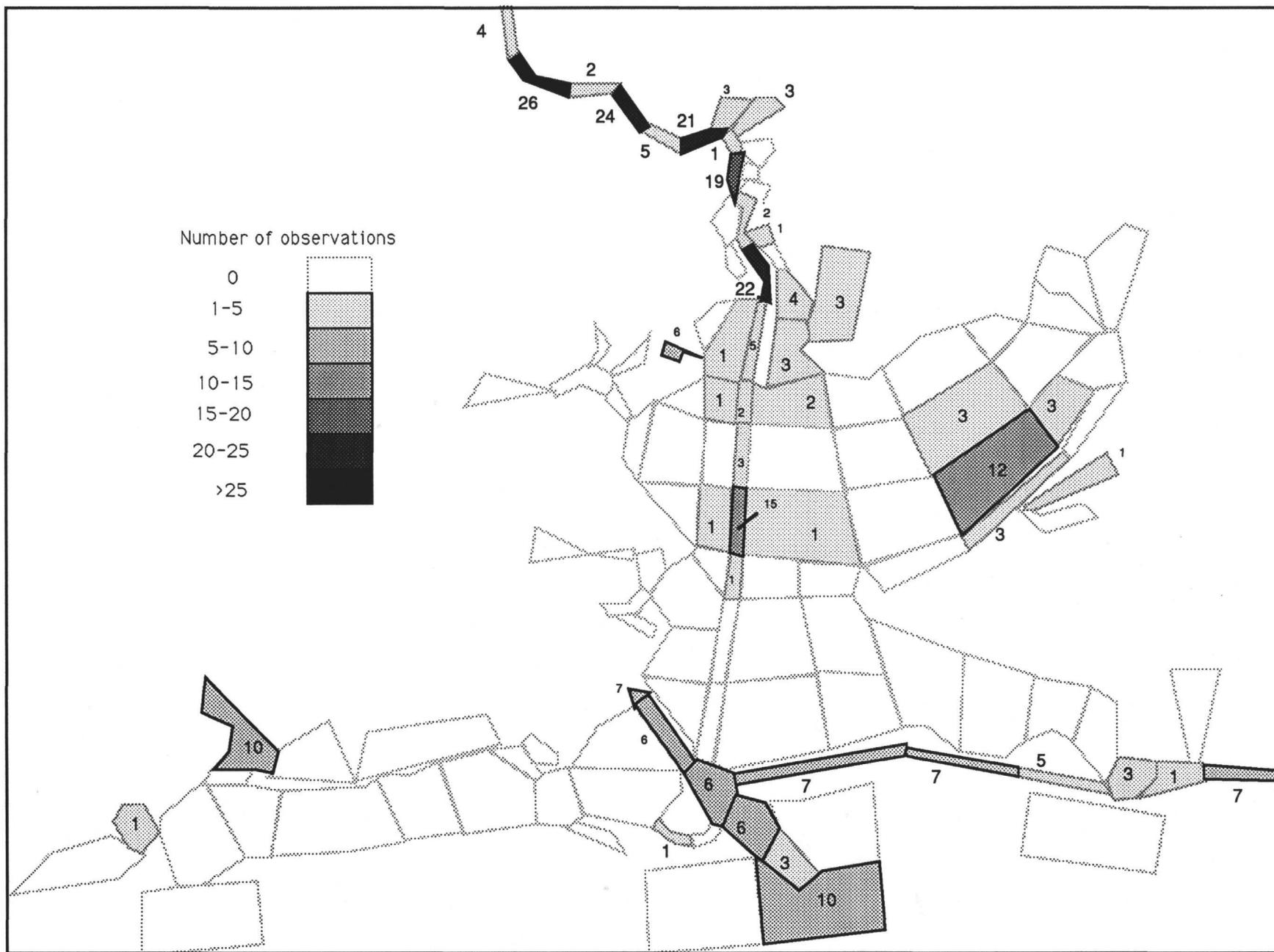


Fig. 3-29 Sampling density in Galveston Bay for WQMETSSET



Fig. 3-30 Sampling density in Galveston Bay for WQ-LIND





Fig. 3-32 Sampling density in Galveston Bay for WQ-CHLR

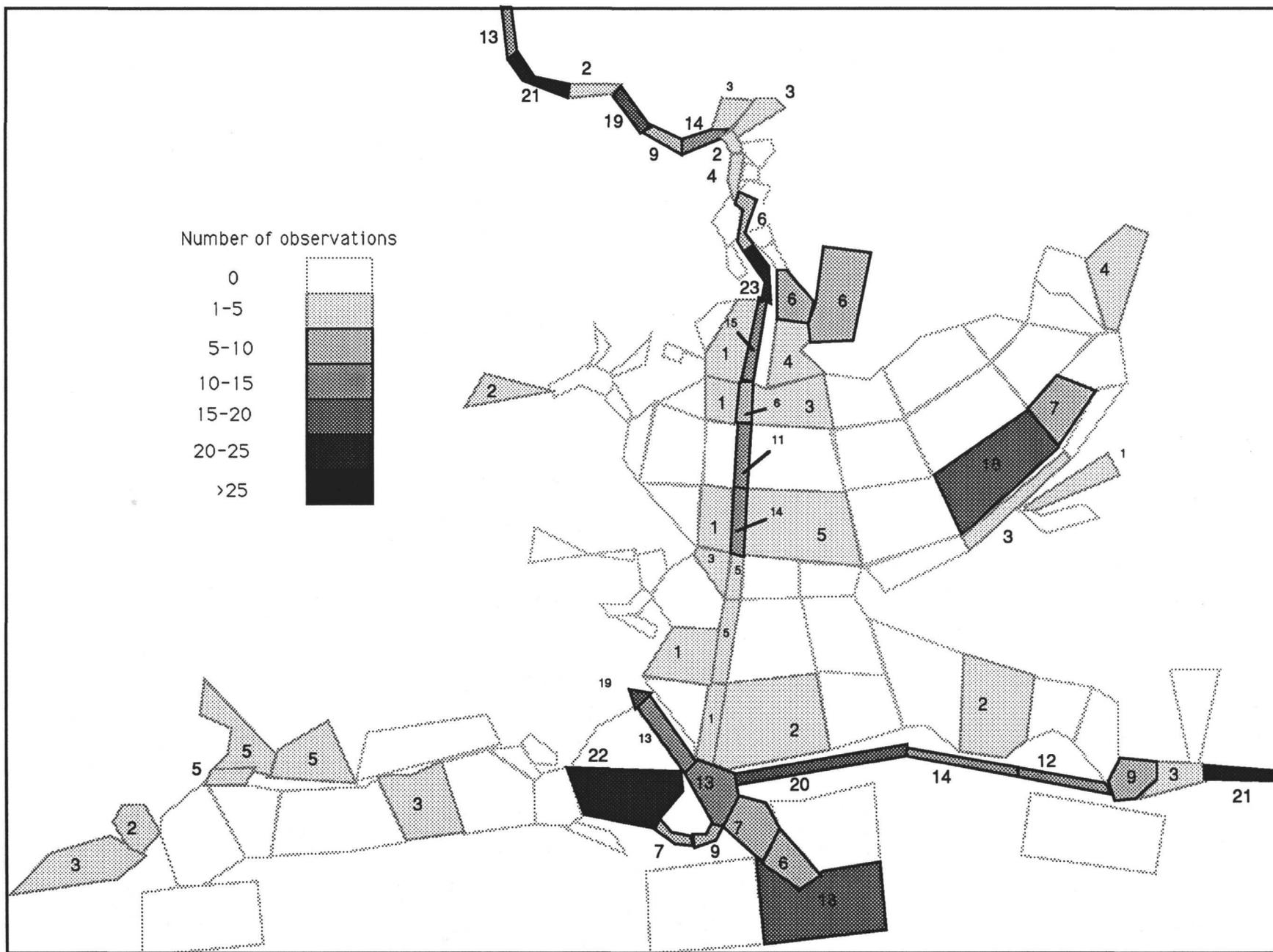


Fig. 3-33 Sampling density in Galveston Bay for WQ-PCB

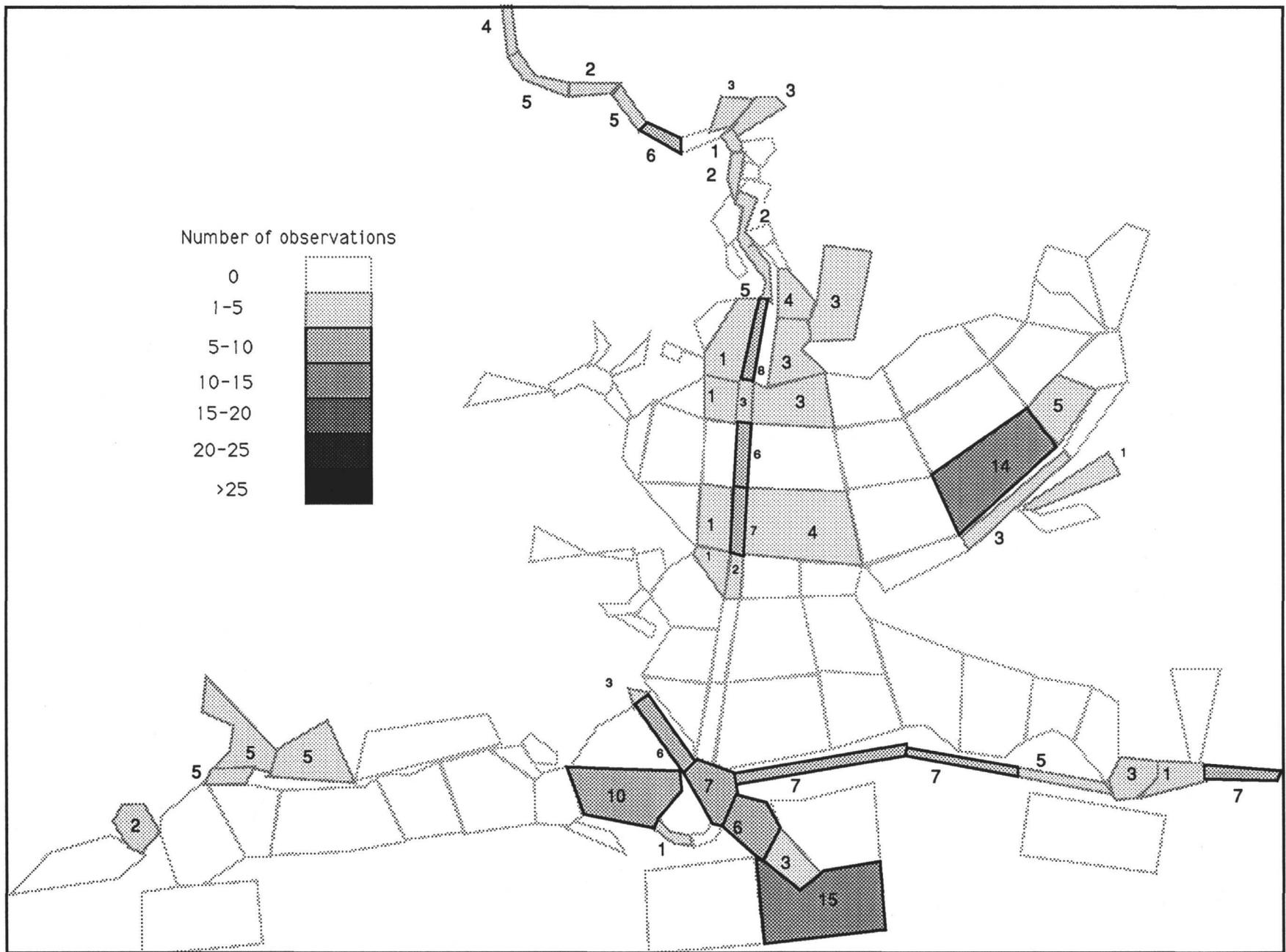


Fig. 3-34 Sampling density in Galveston Bay for WQ-PAH

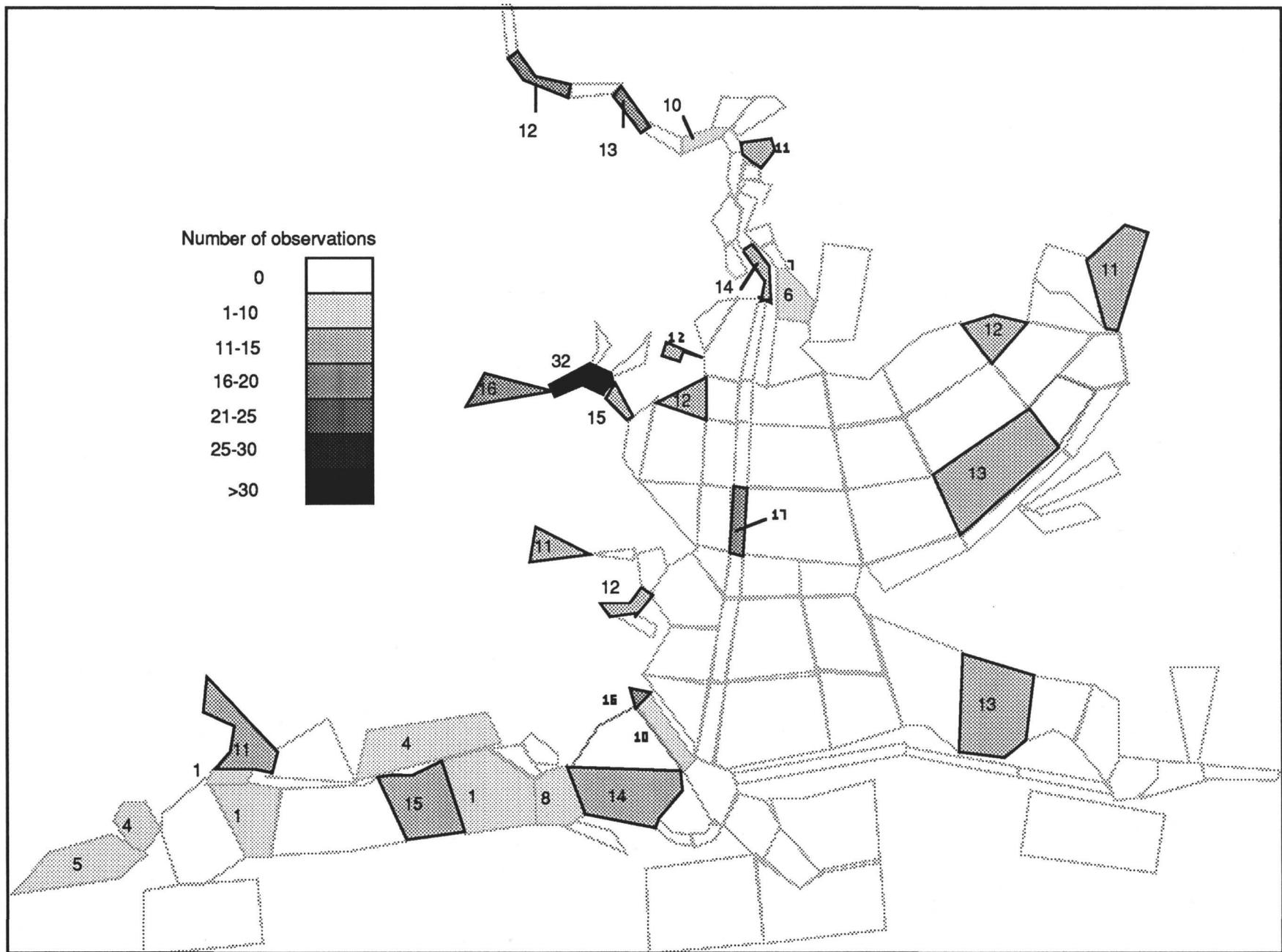


Fig. 3-35 Sampling density in Galveston Bay for SEDTOTP

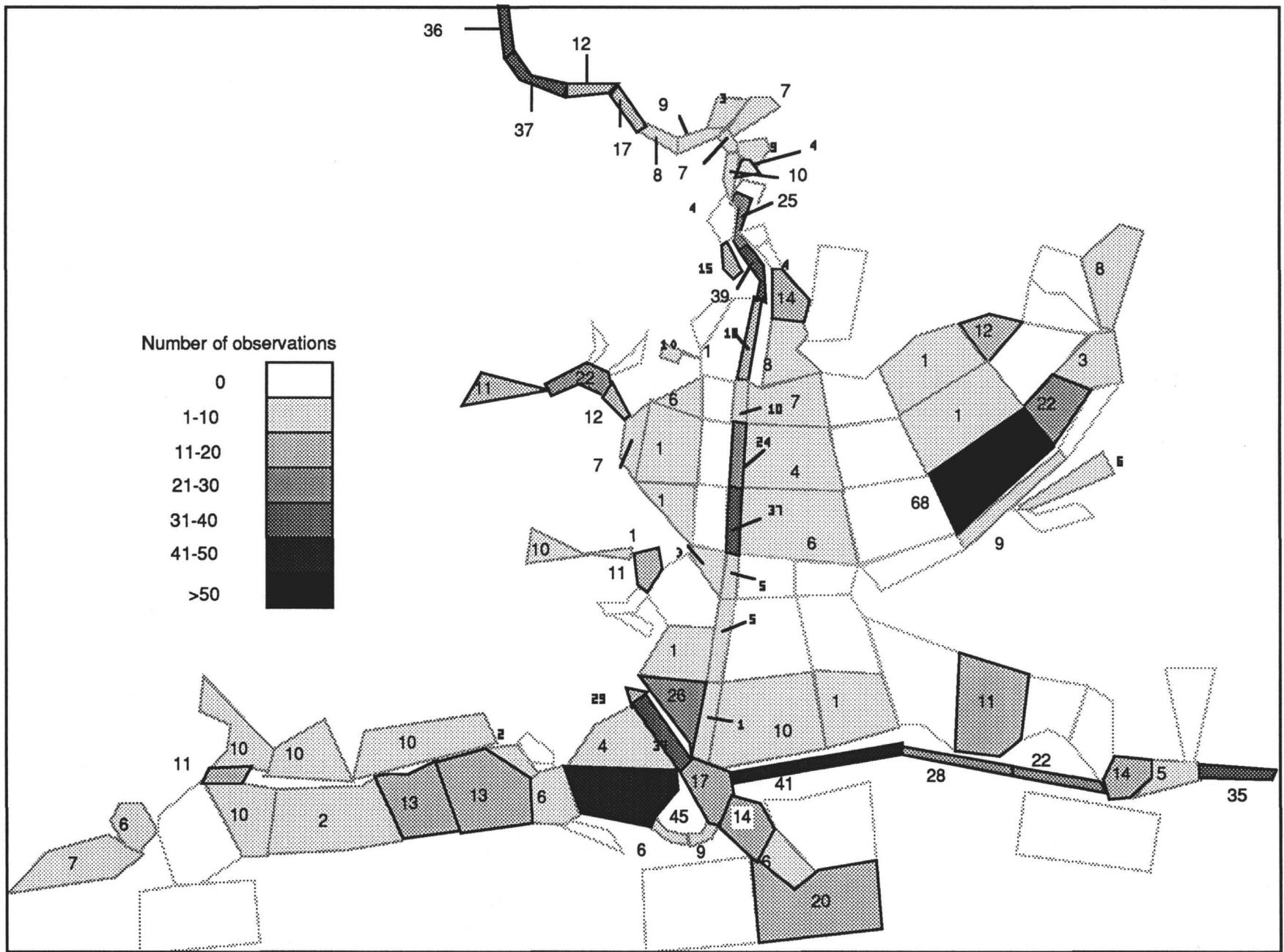


Fig. 3-36 Sampling density in Galveston Bay for SEDO&G

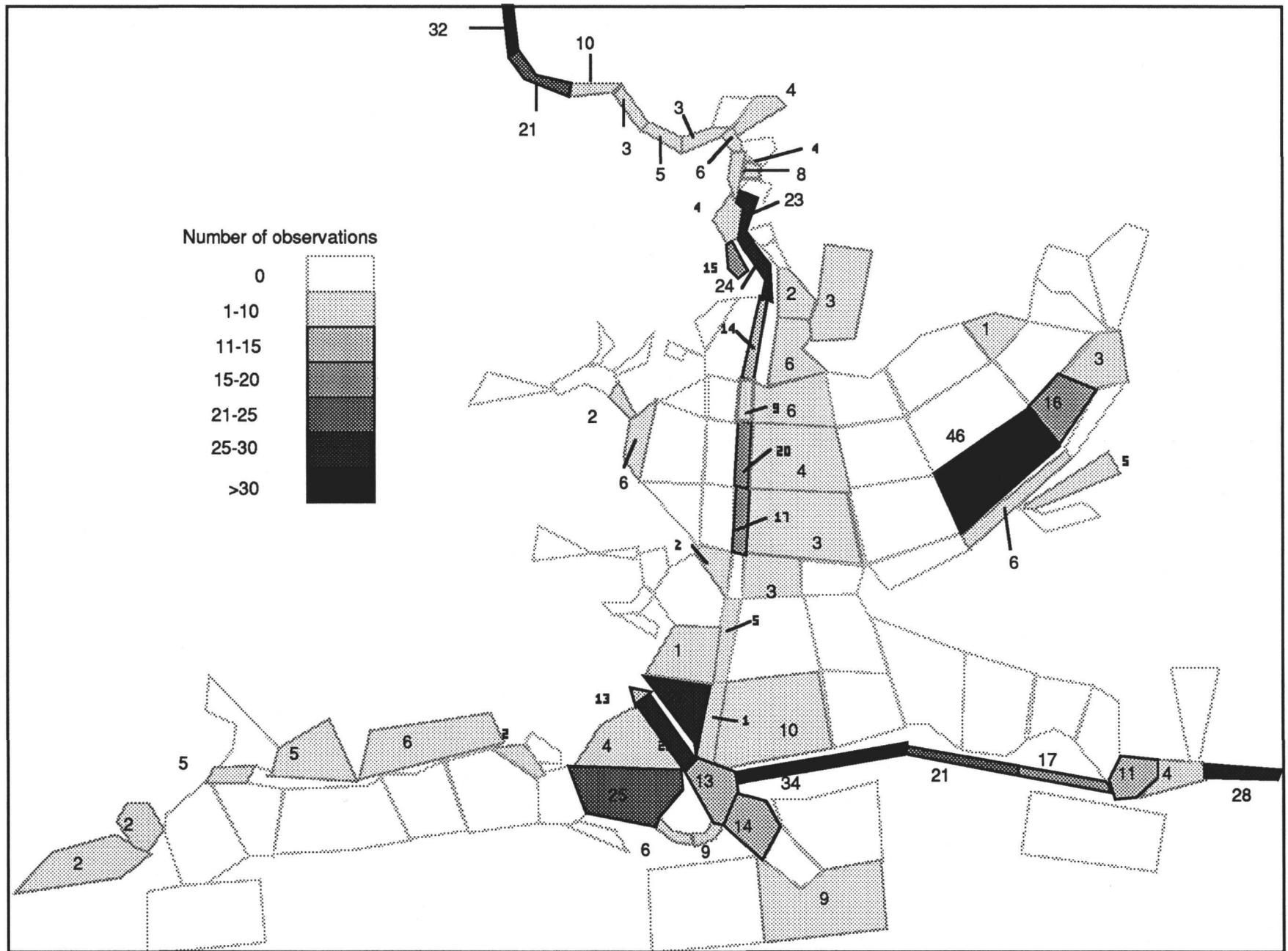


Fig. 3-37 Sampling density in Galveston Bay for SEDKJLN



Fig. 3-38 Sampling density in Galveston Bay for SEDTOC



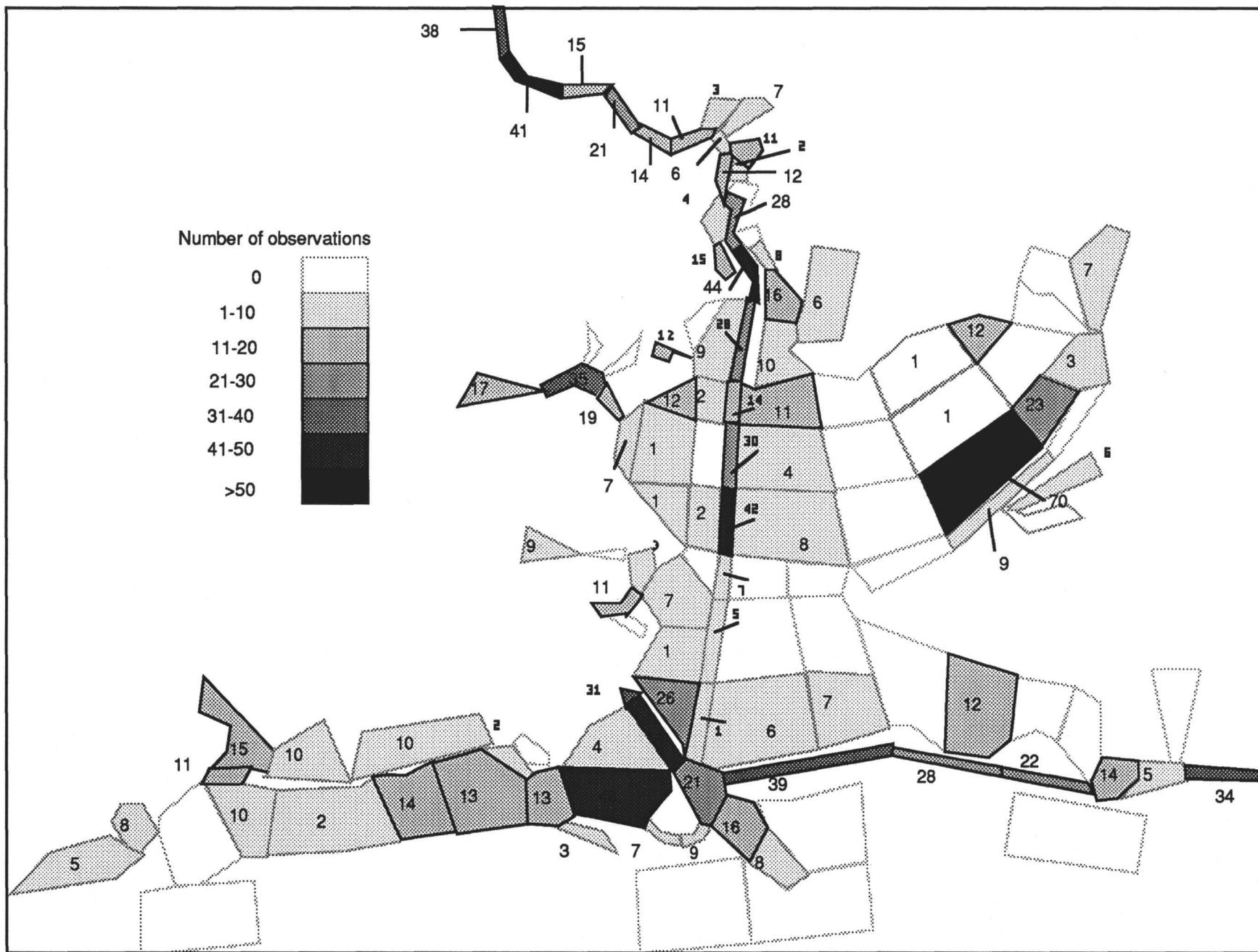


Fig. 3-40 Sampling density in Galveston Bay for SEDMETAS

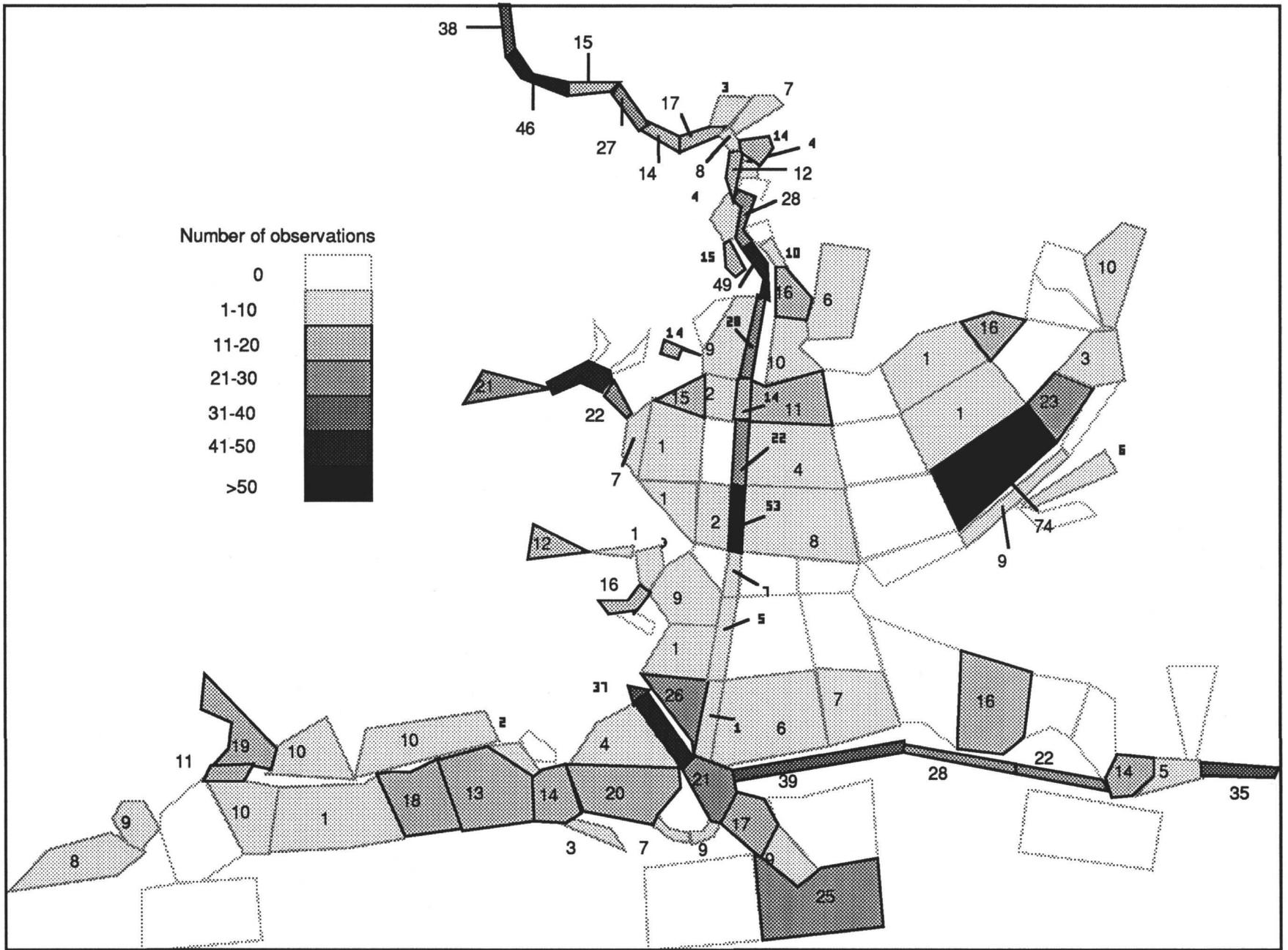


Fig. 3-41 Sampling density in Galveston Bay for SEDMETCD

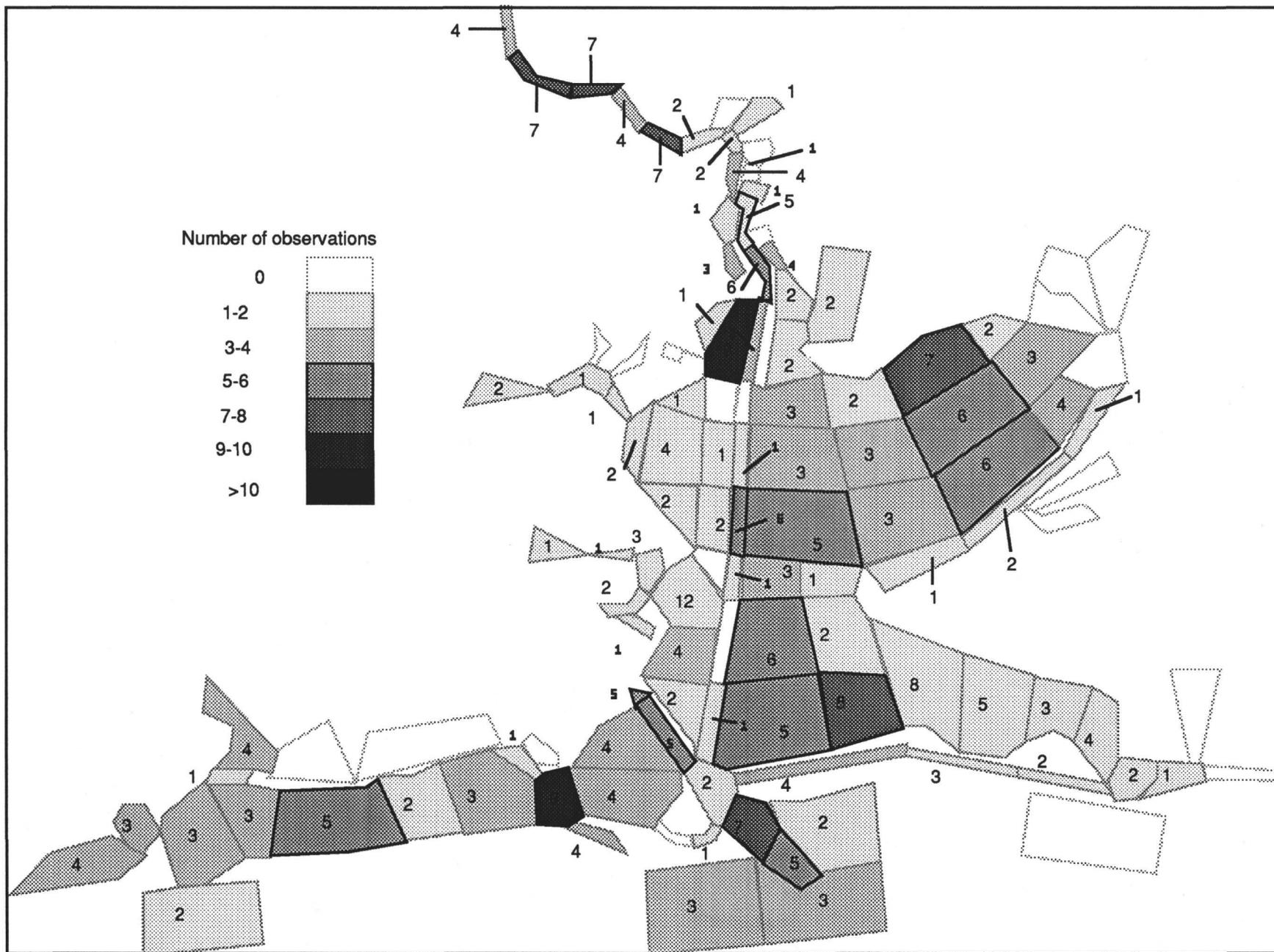


Fig. 3-42 Sampling density in Galveston Bay for SEDMETFE



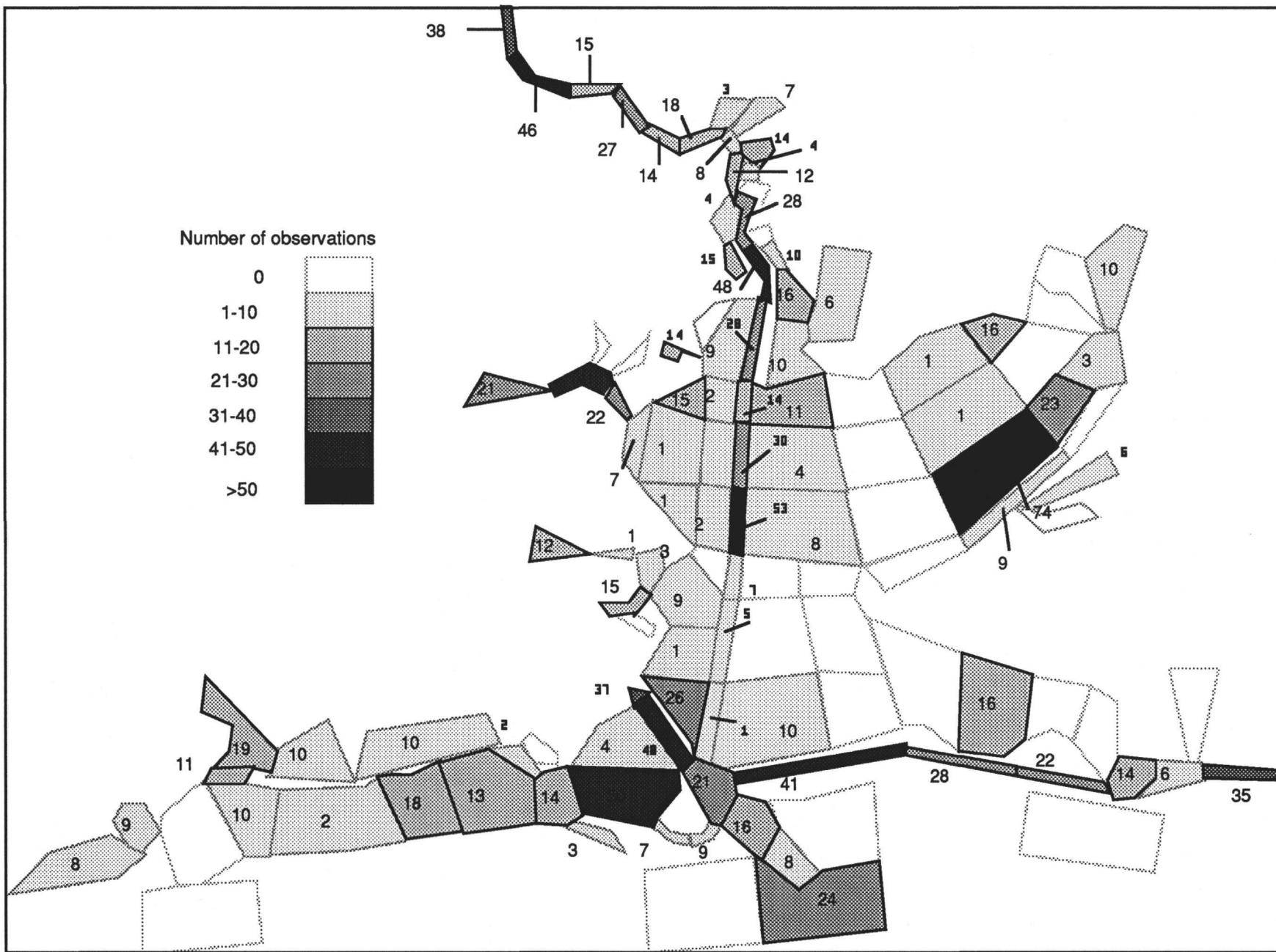


Fig. 3-44 Sampling density in Galveston Bay for SEDMETHG

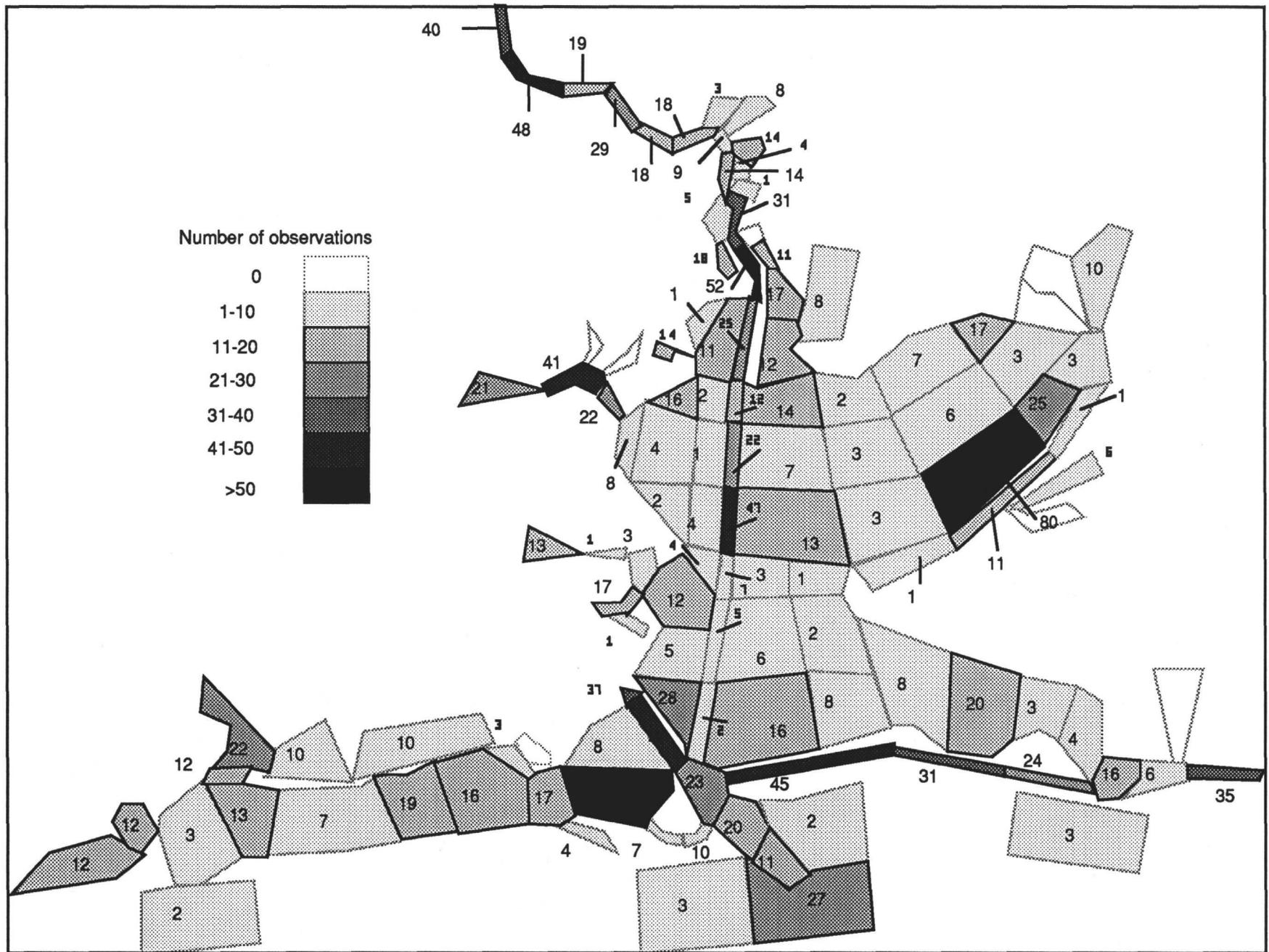


Fig. 3-45 Sampling density in Galveston Bay for SEDMETZN

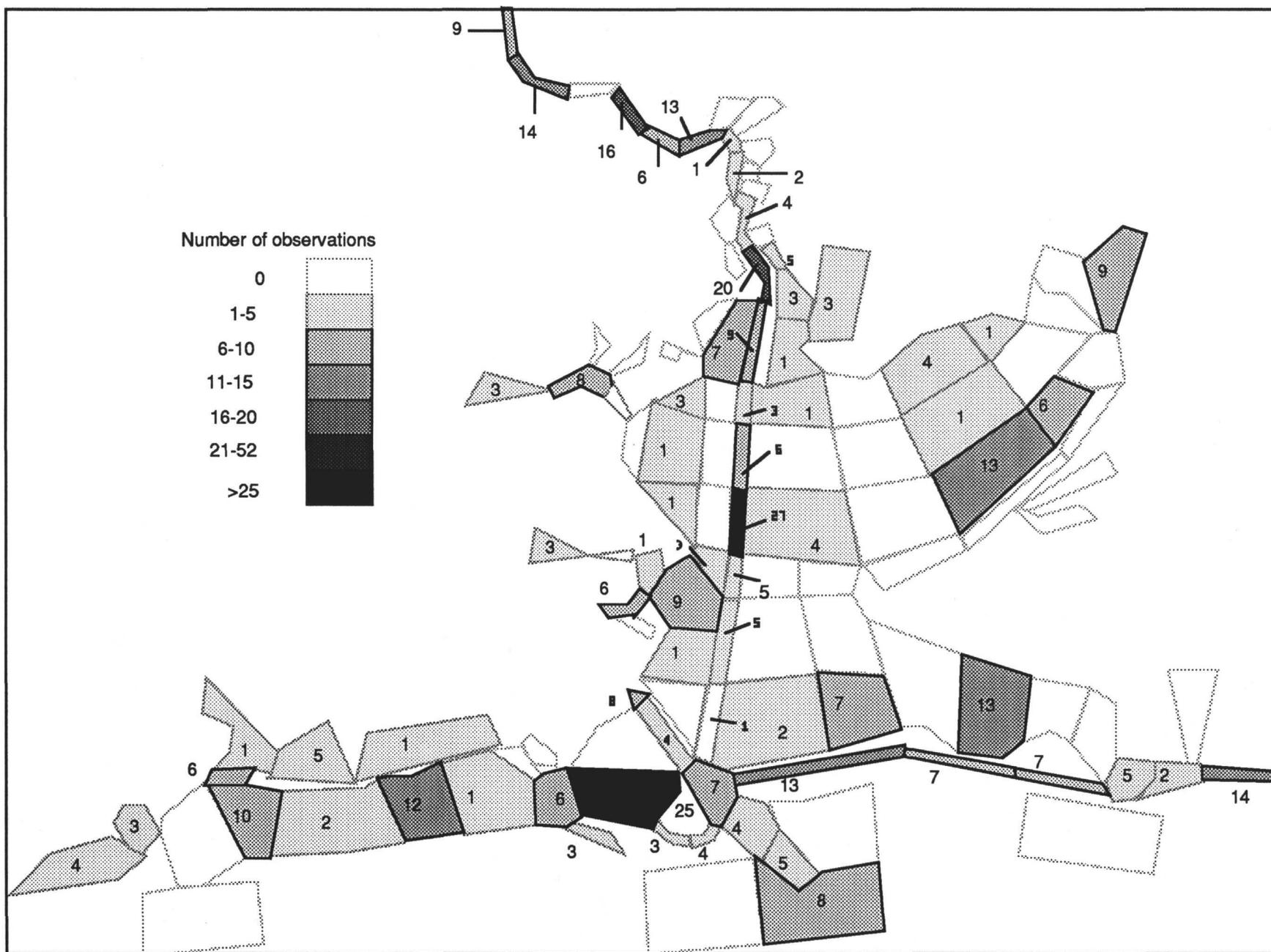


Fig. 3-46 Sampling density in Galveston Bay for SED-LIND

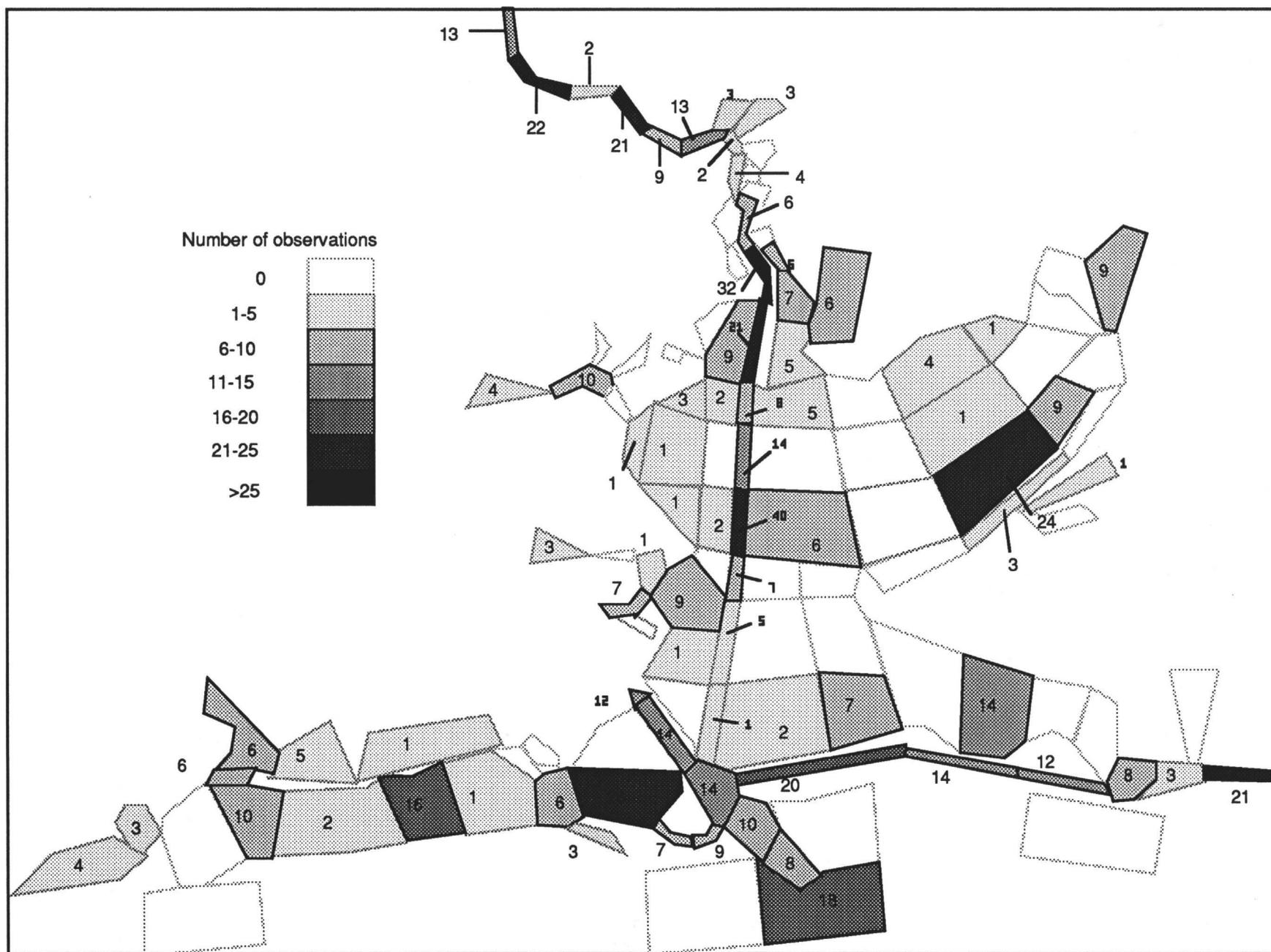


Fig.3-47 Sampling density in Galveston Bay for SED-XDDT

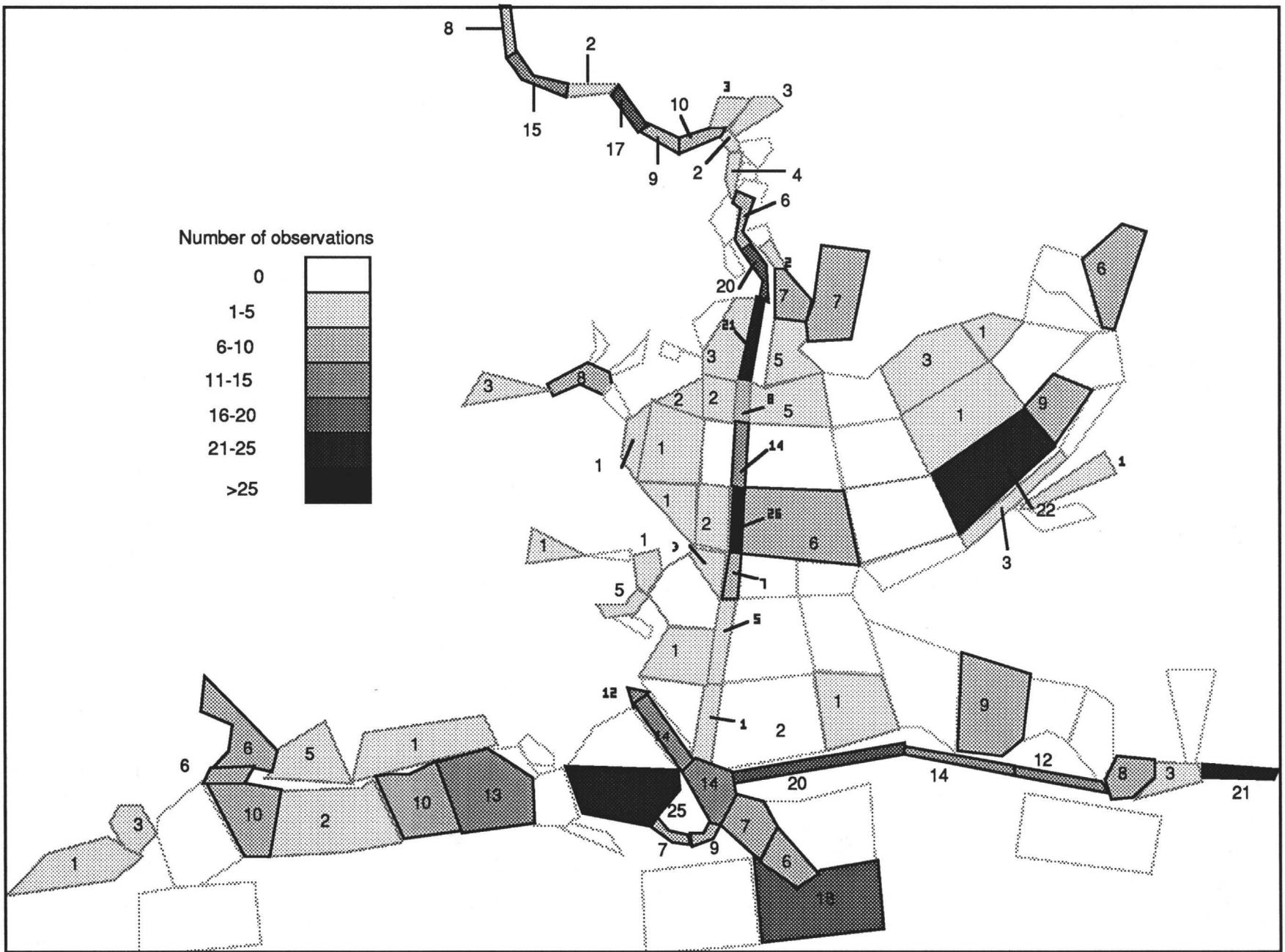


Fig. 3-48 Sampling density in Galveston Bay for SED-CHLR



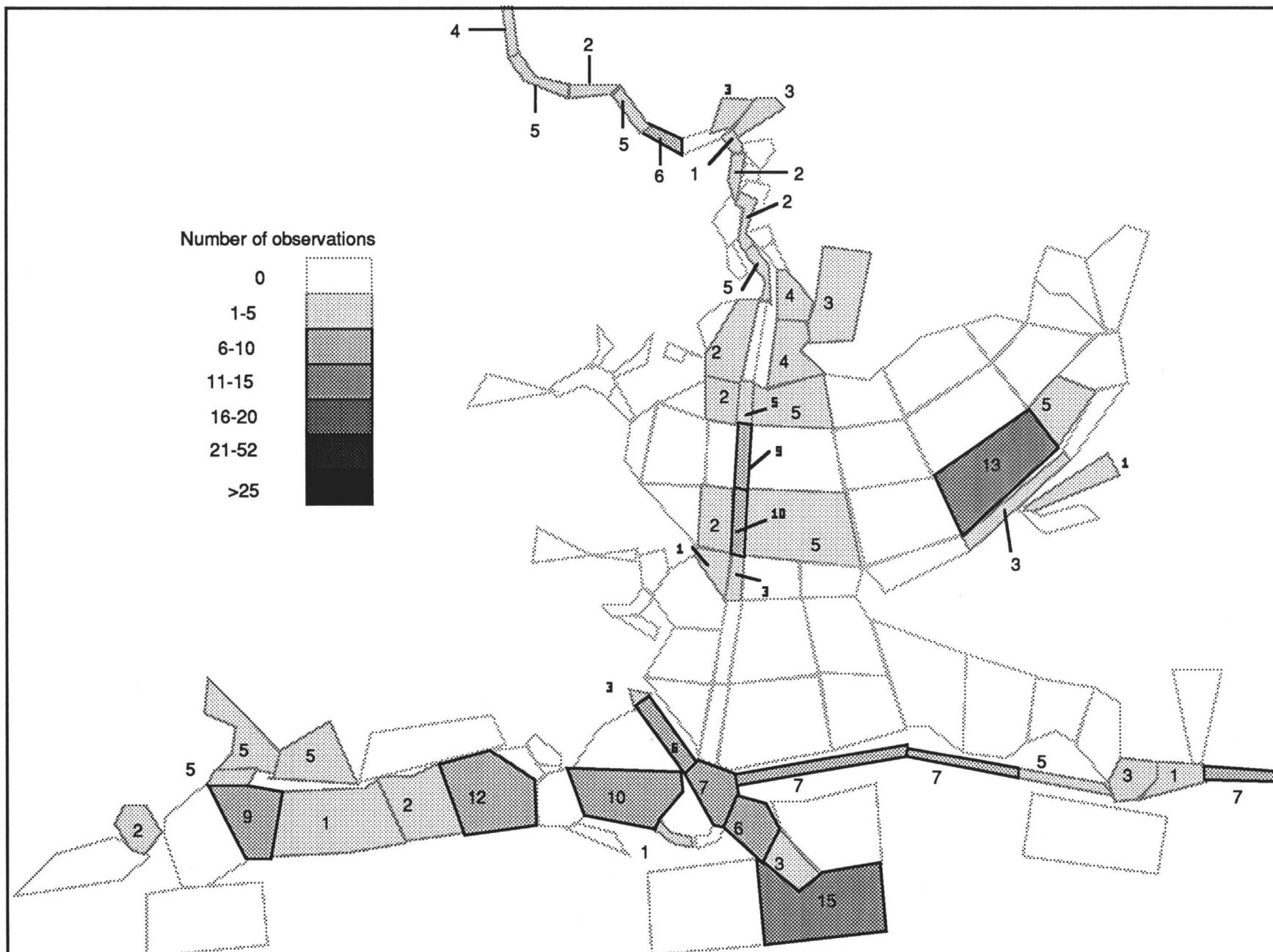


Fig. 3-50 Sampling density in Galveston Bay for SED-PAH