Marine Research Education Advisory Services

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Florida Sea Grant

SHORT- AND LONG-TERM EFFECTS OF FORESTRY OPERATIONS ON WATER QUALITY AND THE BIOTA OF THE APALACHICOLA ESTUARY (NORTH FLORIDA, U.S.A.)

by

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TECHNICAL PAPER NO. 5
May 1978

The information contained in this paper was developed under the auspices of the Florida Sea Grant College Program, with support from the NOAA Office of Sea Grant, U.S. Department of Commerce, grant number 04-8-M01-76. This document is a Technical Paper of the State University System of Florida Sea Grant College Program, 2001 McCarty Hall, University of Florida, Gainesville, FL 32611. Technical Papers are duplicated in limited quantities for specialized audiences requiring rapid access to information, which may be unedited.

Contents

Summary of Results and Conclusions

An integrated (laboratory-field) study was carried out to determine the short- and long-term effects of forestry operations (clearcutting, ditching, roadbuilding, draining) in the Tate's Hell Swamp on the water quality and biota of the Apalachicola estuary. There were indications that various activities associated with site preparation caused increased runoff and reduced water quality in receiving systems. Such effects were noted in East Bay (East and West Bayous) during periods of high local rainfall. The study included short- and long-term evaluations of grass-bed assemblages, benthic infauna, litter-associated organisms, and epibenthic fishes and invertebrates in East Bay. It also included an analysis of key physico-chemical forcing functions (river flow, rainfall) and the relationships of such factors with biotic trends in the Apalachicola estuary over long periods (6-20 years).

Rainfall and Apalachicola River flow are important determinants of bay functions and tend to follow 5-8 year cycles of peak activity. During the period of intensive biological study (1972-1978), annual river flow and local rainfall peaked during 1973-75. Important episodic events during this time included extensive river flooding (winter, 1973), peaks of local rainfall (summer, 1974 and 1975), and low water temperature (winter, 1976-77). Salinity in upper portions of East Bay remained low after 1974 despite decreases in river flow and rainfall during the last 3 years of study. There were general increases in daytime dissolved oxygen and nutrients (N, P) in East Bay relative to the outer system (Apalachicola Bay). The eastern portions of East Bay were particularly affected by storm water runoff from Tate's Hell Swamp during the winter and summer months of 1974 and 1975; such runoff caused temporary water quality changes which included increased color, reduced pH and salinity, and increased turbidity. These changes were associated with heavy local rainfall and forestry operations such as construction of roads and drainage ditches and clearcutting in areas contiguous with the bay. Although such changes do occur naturally, a comparison of various stations indicated that forestry operations exacerbated the low quality conditions and contributed to pulsed influxes of upland runoff through associated estuarine areas. The exact changes in water quality depended on various factors such as the timing, extent, and location of clearing operations, the sequence, location and extent of local rainfall, various (local) drainage characteristics, and revegetation processes in cleared areas. A comparative analysis of long-term trends of pH in the upper portions of East Bay (West Bayou) indicated sharp decreases of pH in bay areas affected by upland runoff. These changes were corroborated by water quality trends in former control areas of the bay. The relatively short-term changes in water quality should be viewed from the perspective of long-term trends of local rainfall, and further work is necessary to determine the long-term implications of forestry operations in Tate's Hell Swamp on the Apalachicola Bay system, particularly with respect to potential cultural eutrophication due to increased dissolved nutrients and changes in salinity in upper portions of East Bay.

A laboratory-field effort was made to determine the avoidance reaction of blue crabs (*Callinectes sapidus*), a dominant species in East Bay, to highly colored, acidic runoff from cleared portions of the Tate's Hell Swamp. Blue crabs of two age groups (juvenile, adult) showed a marked laboratory avoidance response to such runoff (pH 4.6, 5.8) and to test water with induced reduction of pH. There was significant avoidance ($p < 0.001$) of water with pH experimentally reduced to below 6.0; generally, within a pH range of 4.5-7.0, there was an inverse relationship between pH and avoidance while the color of the water appeared to play only a minor role in the avoidance reaction. It was concluded that pH was a primary determinant of the avoidance reaction of this species. The field data, however, gave divergent results; small crabs reached peak abundance in areas characterized by low pH (approximating 4.0 in East and West Bayous). However, large crabs appeared to avoid areas of the bay having low pH, thus indicating a potential avoidance reaction. Factors other than pH were thought to determine the field distribution of this species. These could include ontological variation in reactions to specific inhibitors, intraspecific competition and predation, habitat-specific reactions, and/or differential trophic response to runoff conditions. Despite the apparent contradiction between the laboratory and field results, avoidance of areas in the bay affected by storm water runoff may well be an important mechanism in the response of estuarine biota to such factors. Data concerning the blue crab indicate that laboratory studies without the benefit of associated field observations may be misleading when applied directly to impact analysis.

A comparative analysis of the benthic infauna was carried out to determine potential response to storm water runoff in East Bay. Seasonal peaks of infaunal biomass usually occurred during winter-early spring months (stations 1X, 2, 3, 4, 5A, 5B) and fall months (4A, 6). The bay-wide infaunal biomass and numbers of individuals tended to undergo a significant decline over the 3-year period of study; infaunal species richness also declined with time. This could have been related to changes in energy relationships associated with declining river flows during this period. Areas associated with grassbeds, marshes, or direct river flow (stations 1X, 5A, 3) were most productive in terms of infaunal biomass. Generally, areas in upper East Bay were characterized by low infaunal productivity; in areas such as West Bayou, relative dominance was highest and biomass lowest during summer-fall periods of high local rainfall, reduced salinity, dissolved oxygen, and pH, and high temperature. Numbers of individuals, biomass, relative dominance, species richness and species diversity of benthic infauna reflected short-term changes in water quality that were directly related to upland forestry activities and the incidence of intense local rainfall. These changes were temporary, and there was a relatively rapid recovery of the infaunal assemblages with time. Again, short-term fluctuations should be viewed within the context of the long-term variation of the principal physico-chemical forcing functions, trends in the overall productivity of the bay system, and the reaction of the infauna to stress which is natural to upland portions of the bay.

An experiment was carried out in East Bay in 1974 to determine short-term changes in assemblages of organisms associated with leaf-litter as a response to alterations of water quality. Stations influenced by runoff from local rainfall (5A) were compared to river-dominated (3) and barrier island (1X) areas. Salinity and temperature were primary determinants of the spatial and temporal distribution of the litter-associated fauna, which was dominated by various forms of small crustaceans (amphipods, decapods, etc.). Increased salinity was closely associated with increases in most of the biological indices. The lowest numbers of such organisms occurred during the late summer of 1974 at station 5A and was attributed to stress due to low water quality associated with heavy local rainfall. In this area, there were decreases in species numbers, Margalef richness, and species diversity. Recovery of the biota was relatively rapid with increases in all biological parameters during the following fall period.

Grassbeds (dominated by Vallisneria americana) in upper East Bay (station 4B), receiving runoff from areas affected by forestry activities, were also studied from November, 1975 through October, 1976. Temperature was the principal limiting factor for Vallisneria production and productivity of grassbeds affected by upland runoff did not appear to differ from that in control areas. Invertebrate biomass and abundance in Vallisneria beds were dominated by the gastropod mollusk Neritina reclivata and various community indices were affected by such dominance. Peak animal biomass in such areas occurred during spring (March-May) and late fall (November-December), and was associated with seasonal changes in Vallisneria growth and death. However, increased runoff caused short-term reductions in the grassbed fauna in West Bayou relative to Round Bay. There were indications that the benthic macrophytes buffered rapid changes in water quality parameters such as pH, and that this could have been responsible for the observation of little adverse impact of storm water runoff on the grassbed assemblages of East Bay.

An intensive sampling program was carried out to analyze the short-term responses of epibenthic fishes and invertebrates to episodic influxes of storm water runoff in upper East Bay during summer-fall periods of 1976 and 1977. Rapid changes in salinity and reduced water quality (low pH and dissolved oxygen, high color) led to immediate, short-term decreases of numbers of dominant fishes and invertebrates. This should be viewed within the context of seasonal and annual fluctuations of individual populations, changes in various external forcing functions that are unrelated to the storm water runoff, and the relatively low level of local rainfall during the 1976-77 period of study. However, short-term decreases in fish and invertebrate species richness and diversity indices were directly associated with patterns of episodic rainfall and known changes in upland areas affected by forestry operations. Such episodic influxes of low-quality runoff reduced the numbers and biomass of key nurserying species such as white shrimp (_Penaeus setiferus_) although species-specific reaction to runoff (e.g., of _C. sapidus_, as noted above) precluded broad generalizations. Various changes in population and community functions were reviewed within the context of seasonal variability and the influence of key physico-chemical factors such as temperature, salinity, river flow, local rainfall, sediment or substrate type, and upland conditions. Maximal adverse impact of storm water runoff on the habitat of nurserying populations occurred during periods of peak estuarine productivity.

The overall impact analysis was viewed in the context of long-term biotic trends in the Apalachicola Bay system. Since 1975, there has been a general decrease of various biological functions of epibenthic assemblages. This could be associated with annual fluctuations and long-term cycles of river flow and organic matter in the bay system. Productivity peaks for fishes and invertebrates occurred during 1974-75. This included various community indices and individual populations of penaeid shrimp and blue crabs. Close correlations of long-term commercial catches in Franklin County (oysters, blue crabs, penaeid shrimp) with annual levels of Apalachicola River flow corroborated the importance of river flow to overall bay productivity. Six-year trends of fish and invertebrate community indices in eastern portions of East Bay differed from those in other portions of the estuarine system or the system as a whole. Areas receiving drainage from the Tate's Hell Swamp did not show peaks during the 1974-75 period of generally high bay-wide productivity. Such changes in the long-term biotic trends were related to episodic shocks of low quality runoff from Tate's Hell Swamp during this period.

The data indicate that forestry operations in wetlands systems can cause severe short-term declines in water quality which can then have adverse effects on the aquatic biota in receiving areas. In upper portions of East Bay, this is actually an exacerbation of natural stress caused by rapid changes in key physico-chemical functions. While such effects appear to be short-term, depending on patterns and intensity of local rainfall, the impact occurs during seasons and years of peak estuarine productivity. Also, certain long-term trends that may affect the salinity and eutrophication potential remain poorly defined and deserve further attention. Revegetation appears to be an important factor in the temporal aspects of impact, and permanently cleared areas could contribute to chronic reductions of bay productivity through periodic habitat destruction and impairment due to influxes of low quality runoff water. Management objectives should include efforts to minimize the flashiness of runoff associated with upland development and to eliminate situations whereby such water (of natural or anthropogenic origin) is flushed directly into areas of high aquatic productivity.

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I. Introduction

Scientific background

There is little published information on the effects of upland clearcutting operations on estuarine biota. Watershed alterations due to timber operations have been noted in various areas (Tebo, 1955; Hewlett and Hibbert, 1961; Swank and Douglass, 1974). The long-term studies in the Hubbard Brook Experimental Forest (New Hampshire) have led to a series of papers concerning geology and hydrography (Likens et al., 1967; Johnson et al., 1968, 1969; Hornbeck et al., 1970), nutrient relationships (Fisher et al., 1968; Bormann et al., 1969; Likens et al., 1967, 1972; Hobbie and Likens, 1973), and various effects of forest cutting (Bormann et al., 1968; Likens et al., 1969, 1970; Smith et al., 1968; Pierce et al., 1970; Hornbeck et al., 1970). Clearing operations were associated with significant changes in the quantity and quality of runoff water. During the first 2 years after clearcutting, such flow exceeded the expected by 33% the first year, and 29% the second. There were also substantial increases in stream water of various major ions and nutrients. The weighted average pH dropped by 0.8 units subsequent to clearcutting.

Various studies have determined the highly complex contributing factors to rainfall-runoff relationships. Following a rainstorm, there is surface drainage and lateral movement of percolating water in soil (Barns, 1940), with relative flow rates that depend largely on watershed characteristics.

Forest cover tends to control such flow and is responsible for lateral movement through interception, infiltration (basin recharge) and evapotranspiration (Sokolovskii, 1968); this process is modified by antecedent soil water capacity as a function of the intensity and distribution of previous rainfall. Lull and Reinhart (1972) showed that the porosity of litter and humus usually causes high infiltration rates in a forested system, which is also a factor in reduced water yield due to high evapotranspiration (Ziemer, 1964; Harr et al., 1975). Clearcutting thus causes increased runoff (Hewlett and Helvey, 1970; Lull and Reinhart, 1972) because of insufficient storage ability of the affected soils (Lull and Reinhart, 1972; Hornbeck, 1973). Roadbuilding and associated ditching also contribute to increased runoff and erosion of the cleared areas because of increases in compacted (i.e. less permeable) area and channelization of the flow. Often, the drainage area itself is expanded relative to immediate flow into receiving systems. Thus, past studies indicate that deforestation leads to increased runoff due to the reduced capacity of the system to intercept the flow. In addition, there are changes in rates of evapotranspiration and an inhibited infiltration ability of the resident soils (Patric and Reinhart, 1971); these changes can cause increased water yields of up to 40 cm of water in clearcut areas (Heikurainen, 1965; Satterlund, 1965). However, downstream areas may show reduced effects because of channel storage and lag (Helvey, 1970). System-specific variability would qualify generalization from one area or region to another.

Water quality is often altered by deforestation. There is sometimes increased erosion leading to high levels of suspended solids in runoff

(Packer, 1965; Dickenson et al., 1967; Lull and Reinhart, 1972), which can cause direct injury to aquatic life (Phillips, 1971) and/or indirect effects such as spawning inhibition (Phillips, 1971) and reduced primary productivity. Benthic biota can be inhibited and, in addition to increased nutrients, there can be reduced levels of dissolved oxygen in receiving areas. These changes, in addition to low pH, can cause problems with respect to benthic and column productivity. Marine organisms are generally not exposed to reduced pH, although fresh and brackish water organisms may be periodically affected by low pH under natural conditions. The lethal limit for fishes may approximate 5.0 (Bishai, 1960; Jones, 1964). According to the European Inland Fisheries Advisory Commission (1969), pH levels below 5.0 can cause considerable reductions in the productivity of aquatic systems. Juvenile populations can be highly sensitive to low pH (Lloyd and Jordan, 1964; Kwain, 1975), which can cause variation in reproduction effectiveness, population structure, and fish distribution (Powers, 1941; Collins, 1952). According to Calabrese and Davis (1967), oyster (Crassostrea virginica) eggs require a pH range of 6.75 to 8.75. Since pH is only one of the parameters affected by upland runoff (in addition to color, turbidity, salinity, etc.), such laboratory observations concerning pH remain oversimplified with respect to the impact of upland runoff on estuarine systems. Although such episodic changes in estuaries have been considered as a source of stress, the exact impact of runoff on estuarine systems attributable to upland clearcutting activities remains undocumented in the scientific literature.

During the winter and early spring of 1974, there were visible changes in water quality in East Bay with considerable increases in the water color.

There were reports from commercial fishermen that extensive areas of Tate's Hell Swamp (Fig. 1) had been clearcut, plowed, ditched, and drained into several creeks (Whiskey George, Cash, Sandbank, High Bluff) that lead directly into East Bay. Subsequent investigation indicated that much of the upland area is owned by pulp interests (Fig. 2), and that thousands of acres of the swamp had been clearcut since 1968; from 1970-74, much of the cleared land was above and immediately adjacent to East Bay (Fig. 3). The clearcut lands were largely drained by canals which emptied directly into the above named creeks. A general view of the areas involved is shown in Fig. 4, while various portions of the clearcut areas are shown in Figs. 5-8. In addition, such areas were routinely fertilized (phosphate-base) during the winter.

Together with personnel from a local paper-pulp mill, a field trip was made in the Tate's Hell Swamp. Water samples were taken in various drainage areas (Fig. 9, Table 1). At the time (summer of 1974), extensive rainfall occurred locally and was evident in the runoff from the upland areas. Turbidity was relatively low while color levels varied with location. Although too few data were taken for a definitive study of the upland drainage features, certain trends were evident. The lowest color values were found in areas adjacent to recently cutover forests; several of these ditches drained directly into East Bay. Often, such fields were littered with wood particles. Observations made during rainstorms confirmed that extensive quantities of highly colored water washed off the cutover fields and directly entered the various creeks in the East Bay drainage system. These observations indicated that newly cleared and ditched areas were subject to substantial runoff and that the color of such runoff was higher than that from adjacent natural swamps. Local reductions in pH were also evident.

According to a recent study of the effects of forest management on Apalachicola Bay (Hydroscience, Inc., 1977), the following findings and conclusions were made:

1. Ditching does not drastically affect the quantity of runoff. There is 15% increase in direct runoff during the first year after clearcutting (2.8 inches compared to a range of literature values from 1.3 to 17.7 inches).

2. Clearcutting and site preparation cause increases in nitrogen, color, pH, and suspended solids and decreases in dissolved oxygen. Beyond month ten (after clearing), impact on most water quality variables is minor although total nitrogen and dissolved oxygen are affected for at least 22 months.

3. Watershed characteristics were changed by road construction and drainage ditches with major impact in the Cash Bayou drainage basin in terms of increased drainage area and extensive clearcutting from 1970-1974. It was here that the water yield increased the most (estimated 60% greater than natural yield).

4. Short-term (several weeks) water quality effects were noted following fertilization or clearcutting with maximum color impacts noted for Cash (station 5C) and West (station 5B) Bayou basins as a result of clearcutting. Such impacts were sustained for 10 weeks following clearcutting while impacts on nitrogen and phosphorus were sustained for 15 weeks following clearcutting.

5. Overall, water quality impact due to forestry management was confined to Cash and West Bayous as concerned salinity, color, total nitrogen, and total phosphorus. Cash Bayou salinity decreased by less than 10% while color increased by 20%. Such increases were considered small compared to the natural range of these variables in the bay.

6. Salinity impacts on penaeid shrimp were insignificant. Clearcutting did cause increased levels of suspended solids and decreased dissolved oxygen which was most evident during periods of high runoff. Clearcutting had no effect on pH levels in the bay.

7. The bayou areas are nursery grounds for a number of fish and invertebrate species. There were no significant differences in the species composition and abundance of Cash and West Bayou when compared to other parts of East Bay. Although such bayous had a distinctive species composition, it was not clear how changes in water quality due to forest management may have affected the biota. The biological sampling program did not reveal a aramatic impact attributable to local runoff although a more extensive data base is needed before any definitive statement could be made regarding biological impact.

8. Runoff from recently clearcut areas inhibited the feeding response of pinfish (Lagodon rhomboides) and grass shrimp (Palaemonetes pugio). Such inhibition occurred for pH levels below 5 and was diminished considerably after aging 2 to 5 days. Runoff from an undisturbed forest area caused greater inhibition of the feeding response of grass shrimp than runoff from clearcut areas. Thus, it appears that forest management is not responsible for this inhibition.

9. "Overall, present management practices do influence the quantity and quality of runoff from the Carrabelle area. However, the water quality of Apalachicola Bay is confined to Cash and West Bayous. The subsequent impact on the aquatic biota of this area cannot be distinguished with the available data base."

An integrated laboratory-field study was carried out to determine the potential short- and long-term impact of forestry operations in Tate's Hell Swamp on the biota of the Apalachicola Estuary, including a series of laboratory experiments (avoidance bioassays). Six years of field collections were completed, in addition to special projects designed to test hypotheses concerning the influence of land runoff from Tate's Hell Swamp on the East Bay system. This included an analysis of grassbed (Vallisneria americana) assemblages, benthic infauna, litter-associated organisms, juvenile (epibenthic) fishes and invertebrates, and larger fishes. Short-term (day-months) and long-term (years) trends have now been analyzed and are presented in this report. As part of this program an area of the Tate's Hell Swamp was monitored for water quality before, during, and after a clearcutting operation. This clearcutting experiment was carried out with the help of local forestry interests.

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TABLE 1

Physico-chemical monitoring in upper portions of Tate's Hell Swamp drainage leading to East Bay (30 July, 1974). Stations are shown in photographs (Fig. 9).

Station	(Station Description)	Temp.	D.O.	Color	Turbidity	Salinity
5H	Natural swamp area -- uncleared	31	5.8	280	3	0
5F	Drainage from 6 year old plantation	32	7.2	155	3	0
5J	Drainage from area cut 3 years ago and planted 3 years ago	32	5.2	325	2	0
5G	Cut 1 year ago and planted 6 months ago	32	6.4	1260	2	0
5K	Cleared and ditched 30 days ago	32	5.0	350	6	0
5L	Ditch running into Cash Creek -- drains recently cleared and plowed lands	31	5.2	600	5	0
5M	Ditch running into Cash Creek -- drains recently cleared and plowed lands	31	5.6	310	4	0

Fig. 1: The Apalachicola Bay System showing oyster bars, marshes, and station placement for field collections in the bay and upland areas. Also shown is the experimental clearcutting area in the Tate's Hell Swamp.

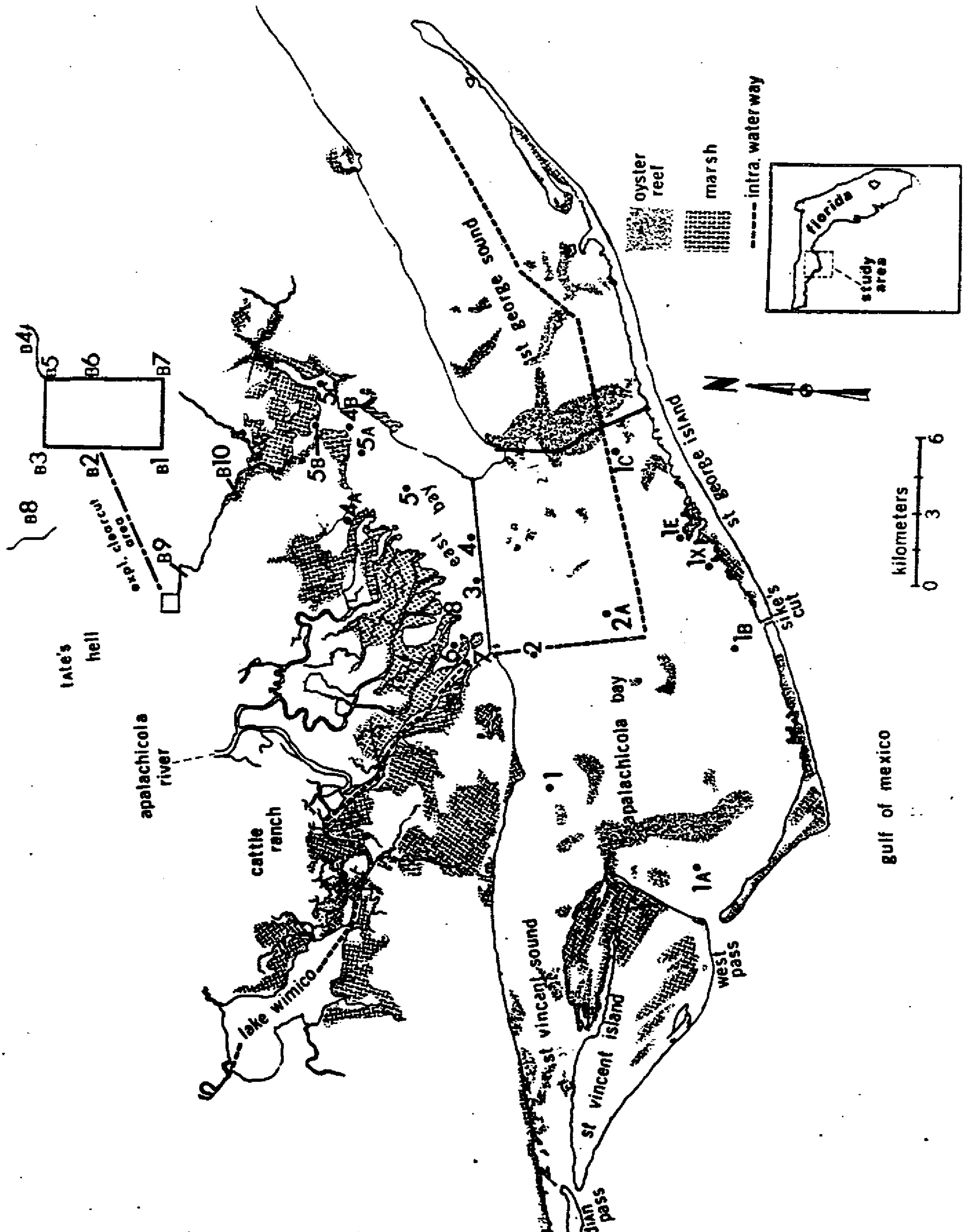

tate's hell
apalachicola river
cattle ranch
lake wimico
expl. clearcut area
B1
B2
B3
B4
B5
B6
B7
B8
B9
B10
east bay
st george sound
apalachicola bay
apalachicola
st vincent sound
st vincent island
st george island
sike's cut
west pass
indian pass
gulf of mexico
1
1A
1B
1C
1E
1X
2
2A
3
4
4A
4B
5
5A
5B
oyster reef
marsh
intra. waterway
florida
study area
N
kilometers
0
3
6

Fig. 2: Map showing the extent of paper-pulp interests in the Tate's Hell Swamp.

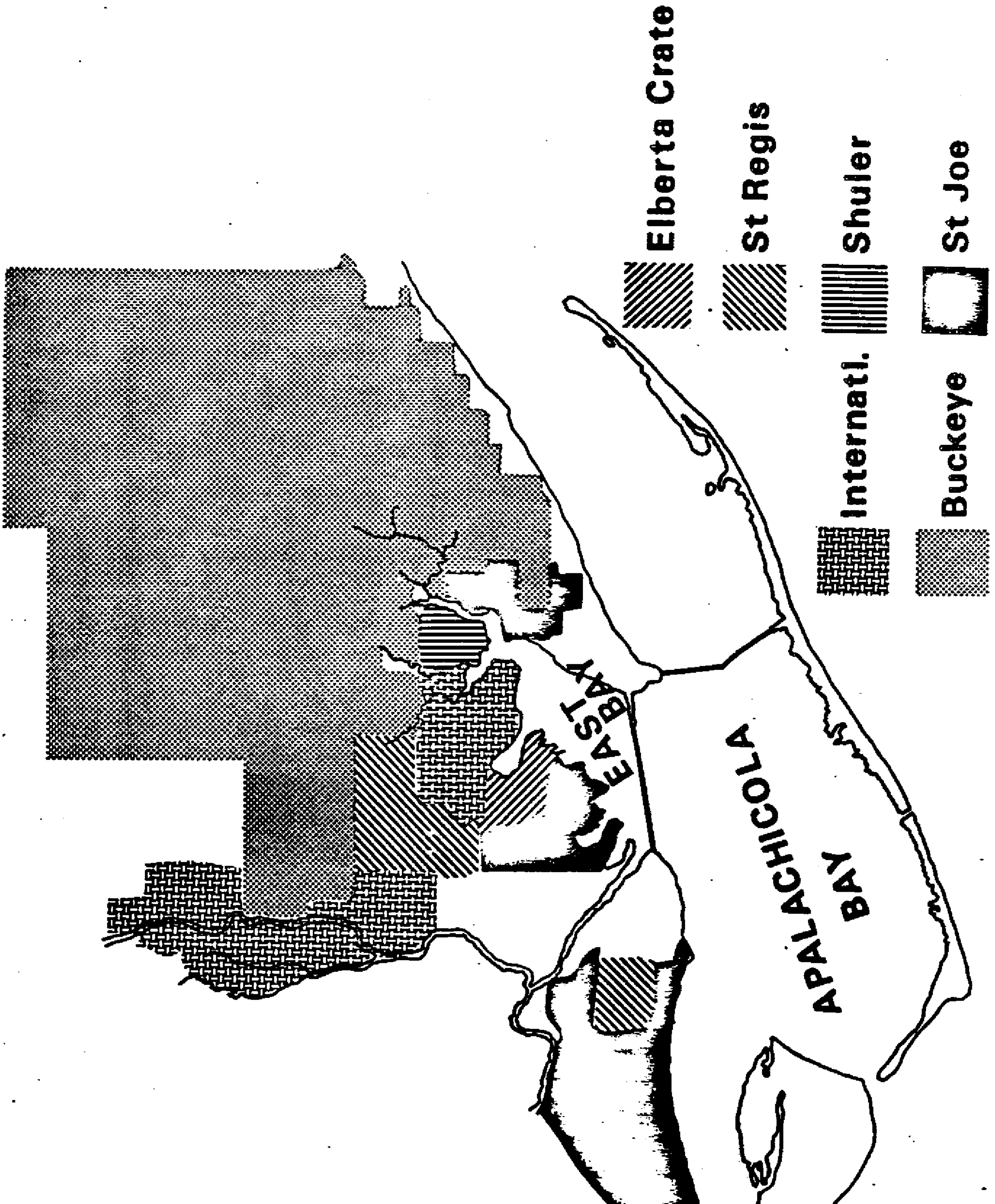
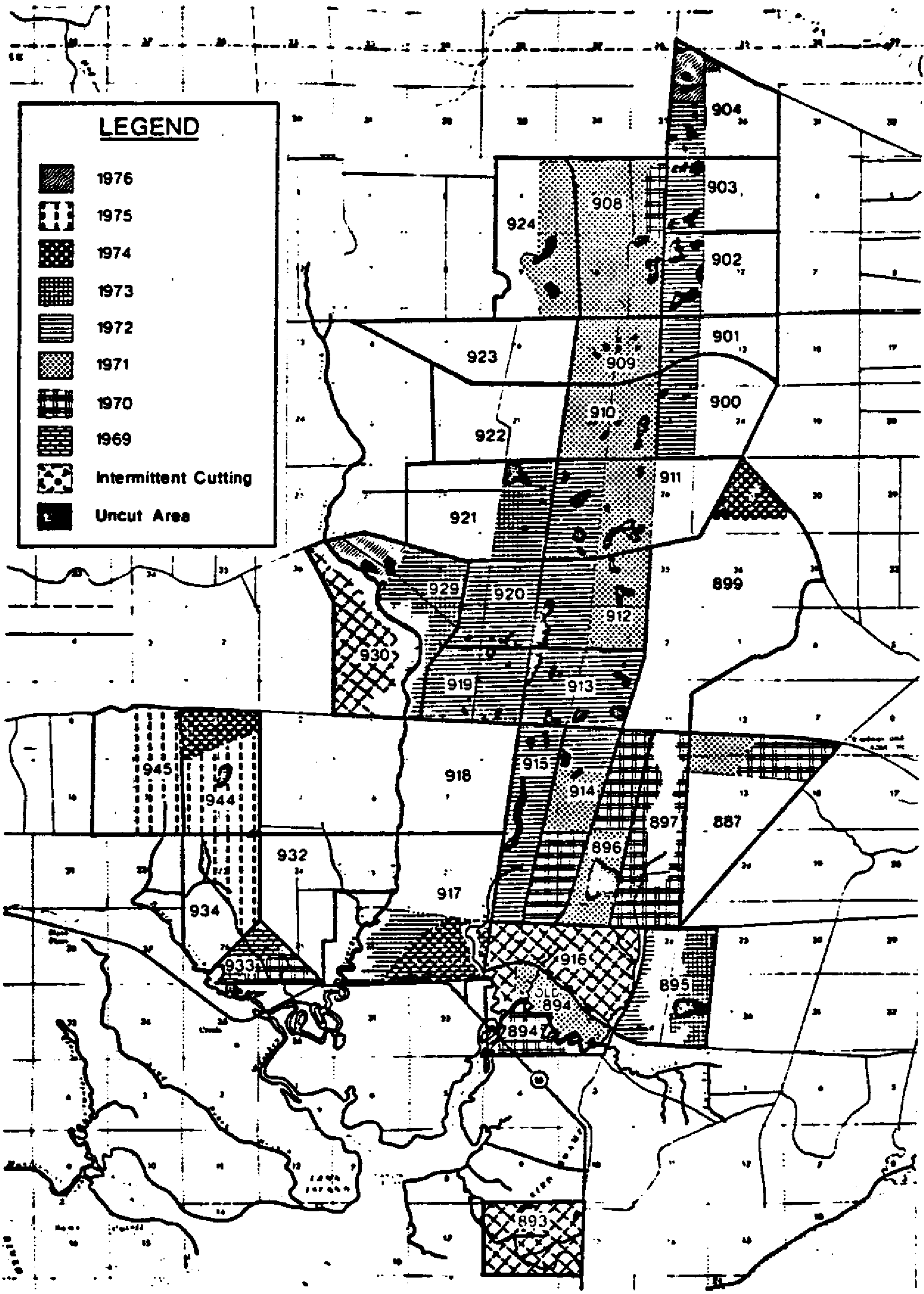

Elberta Crate
St Regis
Internatl.
Shuler
Buckeye
St Joe
EAST BAY
APALACHICOLA BAY

Fig. 3: A rough approximation of clear-cut areas in the Tate's Hell Swamp in chronological order (1969 - 1976). Map and information are supplied by a local paper-pulp mill (Walter L. Beers, Jr., pers. comm.).


LEGEND
1976
1975
1974
1973
1972
1971
1970
1969
Intermittent Cutting
Uncut Area
904
903
908
924
902
901
923
909
900
910
922
911
921
929
920
899
930
912
919
913
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916
895
OLD 894
894
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Fig. 4: LANDSAT photograph of the Apalachicola Valley (April, 1976) showing extent of cleared areas in the Tate's Hell Swamp (photo supplied by the National Aeronautics and Space Administration).

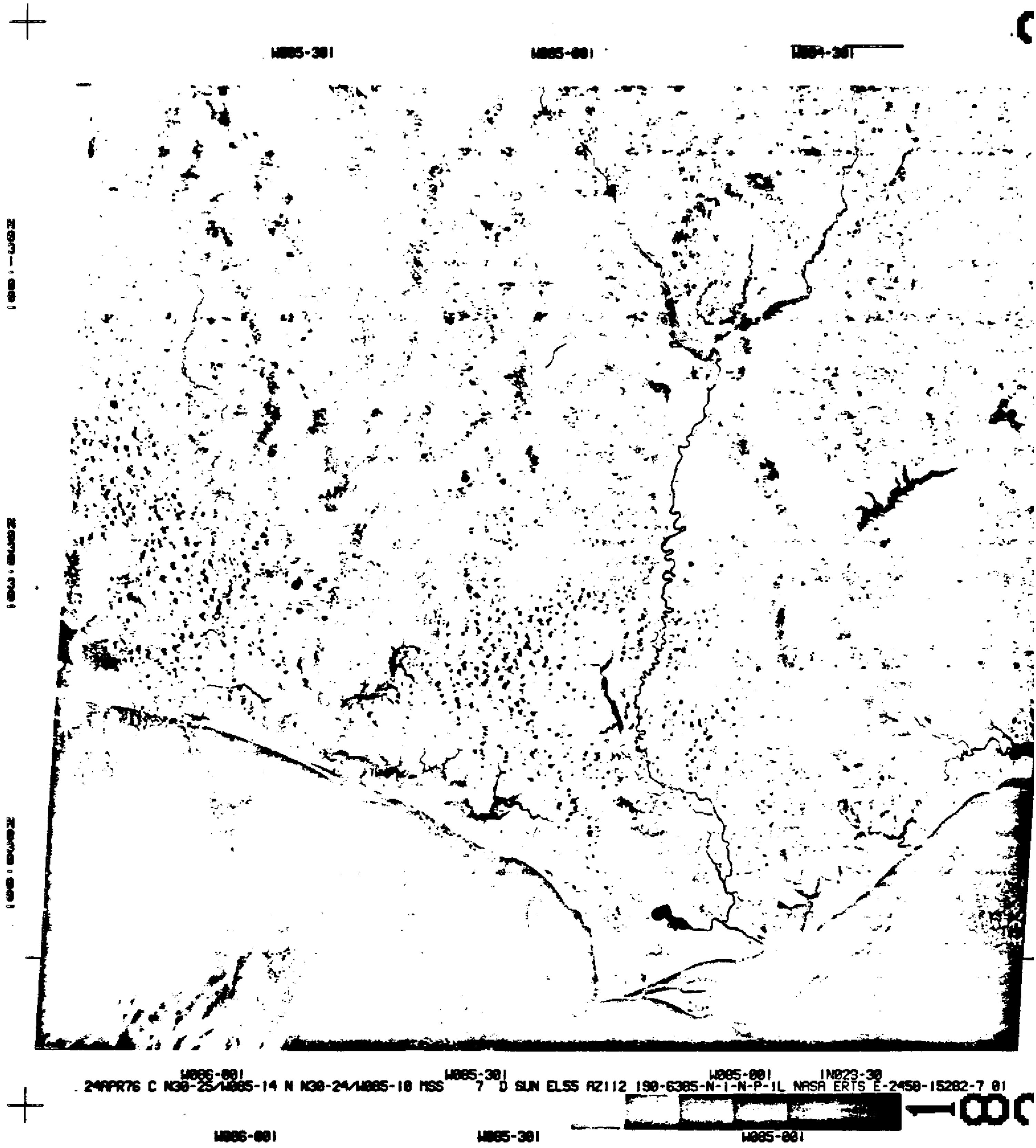
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Fig. 5: Clearcut portion of Tate's Hell Swamp with East (Cash) and West Bayous and Round Bay in the background.

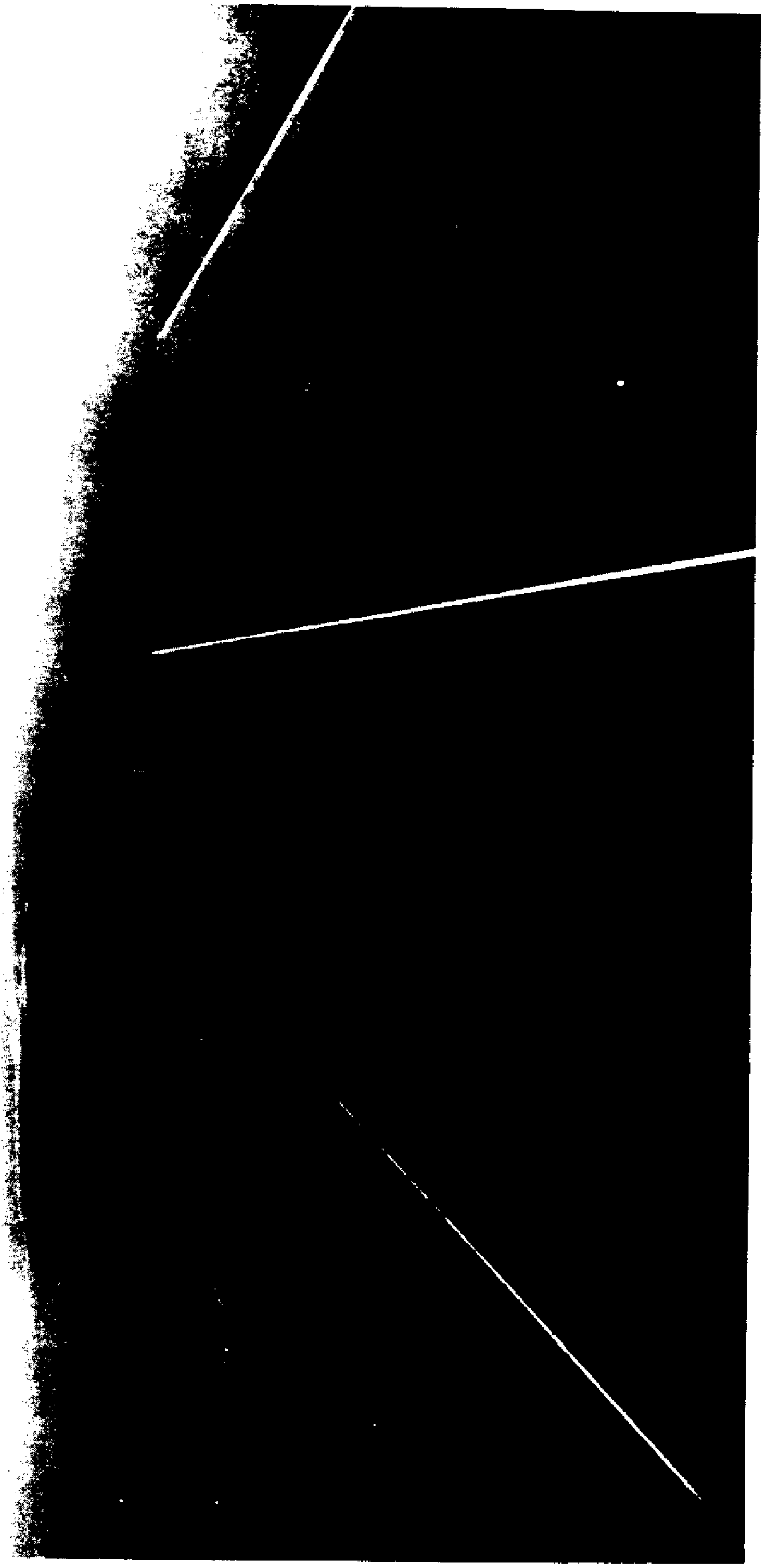

Fig. 6: Ditches draining clear-cut area in Tate's Hell Swamp into upper reaches of the West Bayou System.

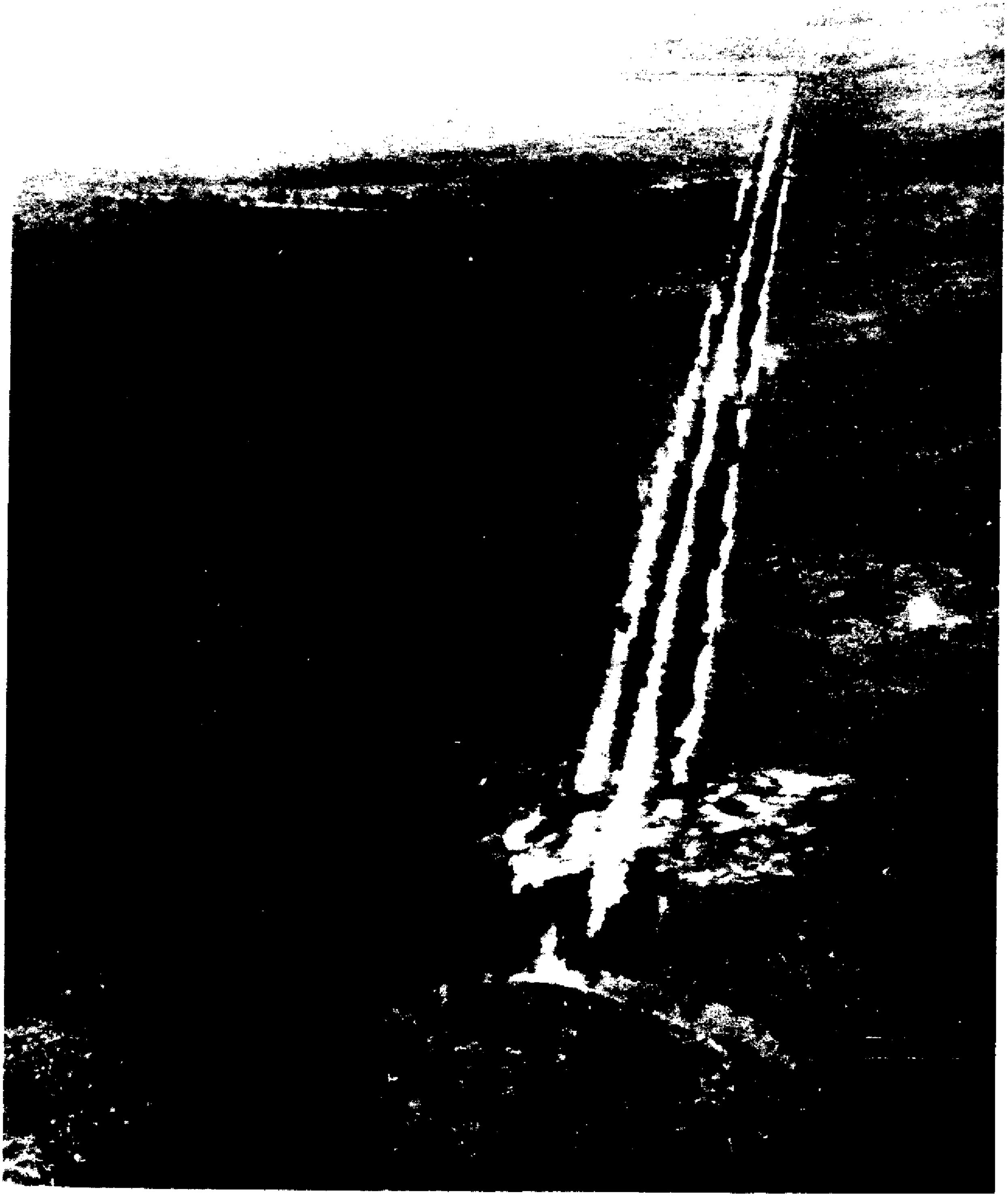

Fig. 7: Close-up of drainage into the West Bayou System.

Fig. 8: Close-up of drainage into West Bayou showing the "plug" between the drainage ditch and the Bayou and the actual connection (to the right) which leads directly into the natural system from upland clear-cut areas.

Fig. 9: Photographs corresponding to water quality stations (Table 1: 5F-5M) in Tate's Hell Swamp (30 July, 1974). This includes natural swamp (5H), newly cleared (5K) and ditched (5L, 5M) areas, recently planted areas (5G), and 3-year (5J) and 6-year (5F) plantations.

NATURAL SWAMP

6 YEARS AFTER
PLANTING

6 MONTHS AFTER
PLANTING

3 YEARS AFTER
PLANTING

PLOWED

NEWLY CLEARED

TATE'S HELL SWAMP

II. Materials and Methods

Station placement

Permanent stations (Fig. 1, Introduction) were established in several ways. The original station determinations were based on diving surveys and analysis of previous studies. There was an effort to sample representative habitats in the bay. Additional stations were added as new studies were undertaken to answer other questions or test different hypotheses.

Sediment analysis

Sediment samples were taken with a corer (d. 7.62 cm) monthly from March, 1975 through February, 1976. This sampling was carried out at fixed stations around the bay. Analyses were conducted on the top 5-10 cm of each core.

Two established methods were used, including a standard geological analysis which eliminates biological functions. At monthly intervals, a sample of 50-150 g was wet-sieved through a series of U. S. Standard sieves. Each fraction was dried at 100°C for 24 hours and weighed. Sieve-class weights were then used to construct cumulative percent particle size curves (Inman, 1952) on arithmetic probability paper. A second analysis involved a supplementary subset of the above samples (Ingram, 1971). A 30-50 g sample was dried at 100°C for 24 hours and then treated with 10% HCl for 12 hours to remove carbonates. After redrying of the sample, organic matter was removed by treatment with 30% H_2O_2 for 12 hours. The sample was then dried, and dry-sieved through a series of sieves on a mechanical shaker for 30 minutes. Sieve class weights were analyzed by the method of moments (Folk, 1966) using a computer program developed by J. P. May (Dept. of Geology, Florida State University). Sediment organic matter was analyzed monthly by drying a subsample at 100°C for 24 hours and ashing at 500°C for 4 hours (Cummings and Waycheck, 1971).

Physico-chemical determinations

Surface and bottom water samples were taken monthly at fixed stations in the Apalachicola Estuary (Fig. 1) with a 1-liter Kemmerer bottle. Dissolved oxygen and temperature were measured with a Y. S. I. dissolved oxygen meter and a stick thermometer. Salinity was taken with a temperature-compensated refractometer calibrated periodically with standard sea water. All pH measurements were made using several field metering devices. River flow data taken at Blountstown, Florida were provided by the U. S. Army Corps of Engineers (Mobile, Alabama). Computerized summaries of the river data were provided by Mr. Roger Ruminick of the U. S. Geological Survey (Tallahassee, Florida). Local rainfall, wind, and air temperature data were provided by the National Oceanic and Atmospheric Administration (Environmental Data Service, Apalachicola, Florida). East Bay rainfall information was provided by the East Bay forestry tower. Turbidity was determined using a Hach Model 2100-A turbidimeter and was expressed as Jackson Turbidity Units (J.T.U.). Water color was measured using an A.P.H.A. platinum-cobalt standard test.

Light penetration was estimated with a standard Secchi disk. Data concerning chlorophyll a, orthophosphate (inorganic, soluble, reactive), nitrite, nitrate, and silicate were provided through a Florida Sea Grant Program directed by Dr. Richard L. Iverson (Department of Oceanography, Florida State University); these parameters were measured according to standard procedures (Livingston et al., 1974).

Biological sampling

Chronic laboratory (avoidance) bioassays

Laboratory avoidance experiments were conducted in a Y-maze avoidance trough from January to September 1975. The trough tests the reactions of animals to steep gradients of water quality parameters. The apparatus was housed in a sound-proof plywood room with two 1.2 m (4 ft.) 40 watt fluorescent bulbs installed 1.3 m (4.2 ft.) above the Y-maze. A television camera and monitor were used to observe and record crab movements in the trough.

Control (normal) water for the avoidance experiments was taken from station 3, located at the southernmost limit of East Bay, a part of the Apalachicola Bay System (north Florida, USA) (Fig. 1). The primary source of this water was the Apalachicola River, which is considered to have a major influence on the environmental conditions of Apalachicola Bay (Livingston et al., 1974). The experimental "dark" runoff was taken from Sand Bank Creek (near 5C, Fig. 1) which drains adjacent clearcut areas and empties directly into the upper margins of East Bay. Water from both areas was placed in permanent 3600 liter plywood storage tanks where it was recycled through dacron filters. All water was stored for at least 24 hours before being pumped to a delivery system adjacent to the test room (Fig. 2).

All experiments were carried out during the day (0900-1800 H) under controlled conditions of temperature, dissolved oxygen, pH, and light intensity. Water quality parameters were determined using a mercury thermometer, a temperature-compensated refractometer, a colorimeter (APHA-Platinum-Cobalt Standard test; APHA, 1971), a model 2100 A turbidimeter, an oxygen meter, and a portable pH meter.

Small (20-60 mm wide) and large (60-140 mm) blue crabs, *Callinectes sapidus* Rathbun, were tested for avoidance. Crabs were taken with a 5 m (16 ft.) otter trawl from station 4A in East Bay (Fig. 1). Crabs were transported to the Marine Laboratory at Turkey Point (about 50 km east of collecting site), placed in aerated aquaria with undergravel oyster filtration, and acclimated for 6 to 12 hours prior to testing.

Small crabs were tested for one hour in groups of ten whereas the larger crabs were tested for 30 minutes and singly to avoid disruptive social interactions. Crabs were removed from the aquaria and placed in the holding area of the Y-maze (Fig. 2) for a 10 minute acclimation period. A barrier was then remotely raised and crabs were presented with a choice between two types of water (control and experimental). Halfway through each experiment (30 min. with the small crabs, 15 min. with the larger ones), the two types of water flowing into the trough were transposed to avoid effects of preferential selection of the trough arms. Any large crabs not moving out of the holding area within ten minutes were discarded. All crabs were measured and sexed after each experiment.

Experiments were run initially with control water and with highly colored (acidic) runoff from the upland creek. Subsequently, the effects of pH alone were tested in a series of experiments in which control water, its pH reduced to levels approximating those in the field through the metered addition of dilute (0.5-10% V/V) hydrochloric acid (Fig. 2), was substituted for the experimental runoff. This acid had been used previously for such purposes because it is highly dissociated and its anions have low toxicity (Jones, 1947, 1948; Bishai, 1960, 1962). Additional tests were made with the large crabs to determine the influence of water color alone. These involved buffering the experimental runoff to a pH equivalent to the control water by the addition of dilute NaOH (1% V/V).

The avoidance responses of the small crabs were evaluated with the test statistic Z, which tests for the equality (null hypothesis) of two binomial proportions (P1, P2, below) (Remington and Schork, 1970). Avoidance was indicated when the number of crabs in the control arm at the end of the first 30 minute interval (immediately before transposing the test waters) was significantly larger than that found in the same arm, under reversed conditions, at the end of the second 30 minute interval. This results in a high Z value and the rejection of the null hypothesis. Net significant avoidance was computed by the following formula:

$$\text{Avoidance Index (AI)} = \frac{P1-P2}{P1} \times 100$$

where:

$$P1 = \frac{\text{No. of crabs in the control arm at the end of the first 30 min. interval}}{\text{No. of crabs in both arms at the end of the first 30 min. interval}}$$

$$P2 = \frac{\text{No. of crabs in the experimental arm at the end of second 30 min. interval}}{\text{No. of crabs in both arms at the end of second 30 min. interval}}$$

Crabs in the holding area at the base of the Y-maze were excluded from the counts. Because of the technique of transposing the flows of water halfway through each experiment, the results test the effect of the type of water (control, experimental).

The threshold pH level for the small crabs was calculated by regressing the mean avoidance indices (%) on pH and then extrapolating for the pH value which elicited avoidance by 50% of the crabs (AI = 50%). A confidence interval for each point in the regression line was computed by Daniel and Wood's (1971) formula.

Avoidance by the larger crabs was measured as the amount of time spent in the control water as a percentage of the total test time of 30 minutes (minus the time spent in the holding area). Values higher and lower than 50% indicated avoidance and preference, respectively. A one-way analysis of variance tested statistical differences between time-responses of the large crabs.

Standards of measurement in field collections

All field samples (biological) were analyzed in the laboratory according to established methods. Organisms were routinely counted and measured (where possible: standard length for fishes, carapace width for blue crabs, etc.). Weight conversions were also made as described by Livingston et al. (1977).

Dry weights were obtained by oven-drying samples for 48 hours at 105°C. Ash-free dry weights were obtained by ignition of the specimens in a muffle furnace for 1 hour at 550°C. Preliminary samples indicated less than 1% error was introduced by reducing the ignition time from the recommended 3 hours (Cummings and Waycheck, 1971) to 1 hour.

Linear regression equations utilizing a log-log (natural logs) transformation were calculated for each species where data were available. These were calculated according to the following general equation:

$$\ln(\text{weight}^*) = \ln(\text{length}^{**})a - b$$

where:

a and b = regression coefficients

*weight = dry weight (fishes)
= ash-free dry weight (invertebrates)

**length = standard length (fishes)
= total or carapace width (invertebrates)

For those invertebrate species where no length or width measurements were taken, a representative grouping according to size was dried and/or ashed; a single mean weight per individual was given for that species. For those species collected so rarely that no length-weight relationship could be established, regression equations or average weights of similar species (similar body shape, size, etc.) were substituted.

Benthic infauna

Permanent stations were chosen in established areas of study (Fig. 1; 1, 1X, 3, 4, 4A, 5A, 6). A hand-operated corer (d. 7.7 cm) was used and 10 subsamples were taken monthly to depths of 15 cm at each station (1, 1X, 3, 6: from March, 1975 to February, 1976; 4, 4A, 5A, 5B; from February, 1975 to present). All samples were washed through a 0.5 mm screen and fixed in 10% formalin. Rose bengal was added at a concentration of 200 mg/l (Mason and Yevich, 1967). Animals were rough-sorted and placed in 40% isopropyl alcohol, identified to species, and counted. Biomass (ash-free dry weight) was determined by oven drying each sample at 100°C for 12 hours, then, after weighing it, heating it at 500°C for four hours. Standard determinations for each species were made using 100-200 individuals for computations of mean dry weight/individual. This was then used for all conversions to biomass.

Leaf litter assemblages

Stations were established on the basis of previously determined salinity regimes (Fig. 1). Station 5A, a predominantly freshwater habitat during spring and early summer, is characterized by salt water intrusion during late summer and fall periods. Station 3 is a river-dominated area with frequent increases in benthic salinity during summer and fall periods. Station 1X has relatively high salinities throughout the year except during periods of high river discharge when intermediate salinity levels prevail.

Experiments in the field were carried out with specially designed detritus baskets. These baskets were constructed of plastic-coated hardware cloth (6.5 mm^2 mesh) shaped into cubes (30.5 mm/side) with hinged tops. An inner fiberglass screen liner (2 mm^2) covered the sides and bottom of each basket. This allowed organisms access to the inside of the basket; when the basket was pulled to the surface, organisms were trapped inside. Baskets were weighted for stability. Leaf litter was collected along the banks of the lower Apalachicola River. Species composition of this litter was mixed, but it consisted primarily of water oak (Quercus nigra), over-cup oak (Q. lyrata), red maple (Acer rubrum), and sweetgum (Liquidambar styraciflua). The leaves were air dried and placed in baskets (400 g dry weight per basket), which were then situated at the various sampling sites. Sampling times were set according to seasonal fluctuations of key environmental parameters in the Apalachicola Bay system. Three periods were chosen (spring, April-May; summer, August-September; fall, October-November). During the spring series, seven baskets (containing leaves) and two controls (containing no leaves) were placed at stations 1X, 3, and 5A. At weekly intervals over a four to six week period, the baskets were retrieved and rinsed in a bucket of sea water. During each sampling, leaf matter was removed, placed in the water a second time, and swirled to remove all organisms. The leaves were then replaced in the respective baskets and returned to the bay. Organisms in the buckets were strained through a 297 micron sieve, washed into jars, and preserved in 10% formalin. In the laboratory, they were identified to species, counted and weighed (wet weight). Ash free dry weights were determined as described above.

Multiple samples (7) were used to evaluate the method of collection. A composite species accumulation was determined. Each point represented the mean number of species found in the 7 subsamples taken at weekly intervals from 9 April to 14 May. In each instance, an asymptote of species accumulation was reached by the fourth sample. Further analysis was carried out using a modification of a program described by Livingston et al. (1976). At each sampling period, fifty random draws were made of the 7 possible combinations of species. Numbers of species accumulated with each sample were averaged and plotted as a percentage of the total number of species taken for the 7 samples. The cumulative distribution function showed that at station 3, between 90 and 95% of all species were taken by the fourth sample. At station 5A, these figures ranged from 90 to 97% during the sampling period with asymptotes routinely established by the fourth sample. An analysis was also made of the variability in the determination of total numbers of individuals (N) taken within a group of subsamples. Analysis of variance of N was determined from week to week. A theoretical standard error was calculated with confidence limits established to determine variation by sample ($\frac{S.E.}{X}$ x 100%) for a given set of samples. This permitted a comparison of the true mean of any number of samples with the mean for the total number of samples (42). At station 5A, four samples of a given time period were within ±30.8% of the mean ($p < 0.05$). At station 3, the ANOVA results indicated marked differences in N from week to week. Consequently, data were analyzed on a weekly basis. The four samples taken in each period were within ±51.0% of the mean ($p < 0.05$). Thus, the data indicate that in terms of the number of species taken in a given set of samples, by the fourth sample, a representative S value was achieved at each site. At station 5A, relatively uniform N values were noted from sample to sample, so that four samples would again allow adequate sampling effort. However, at station 3, because of higher variability of N, more samples were necessary to achieve the same confidence level. Based on these data, it was determined that four samples were adequate for analysis. All further operations were carried out using sets of four baskets for each collection.

Grassbed assemblages (East Bay)

Macrophyte samples were taken in two grassbed areas (stations 4A and 4B; Fig. 1). These areas were dominated by Vallisneria americana. A detailed analysis of sampling criteria is given by Livingston et al. (1976). Samples were taken monthly from November, 1975 to October, 1976. Vegetation was sampled by haphazardly throwing 8 0.25 m^2 hoops at each station and gathering all plant matter within each hoop. The plant matter was placed in plastic bags, and the samples were taken to the laboratory, where they were washed, sorted to species, and identified.

Collections were dried in ovens at 105°C for about 12 hours (until there was no further weight loss). Total (whole plant) dry weight for each species was determined and recorded by station, and data were entered into the computer files as biomass (dry weight)/m^2.

The species Vallisneria americana composed 99% (+) of the overall biomass. Consequently, an effort was made to estimate the productivity of this species from periodic standing crop measurements according to a method described in Livingston et al. (1977). Vallisneria, as a perennial, dies back in the late fall of each year. Minimal biomass was determined by averaging the dry weight figures taken during the dormant period (i.e. the winter months) and subtracting this from (summer) biomass figures at each station. The confidence limits were broad because of extreme seasonal and spatial variability; maximal biomass for station 4A was calculated from June rather than September (which is when biomass peaks were actually observed).

Physico-chemical data were taken according to methods described earlier in this report. Sampling for grassbed organisms was carried out in identical fashion at stations 4A and 4B during the day and the succeeding night (about one hour after sunset). Six one-minute trawl tows (at speeds of about 1.5 knots) were made using a 32 cm dredge net (D-net) (nylon bag: 1 mm mesh) for benthic sampling and a 30 cm plankton net (1 mm mesh) for the surface biota. Sampling was carried out in such a way that the same volume of water (15 m^3/6 samples) was sampled by each net. All organisms were preserved in 10% formalin in the field and later washed and transferred to 40% isopropyl alcohol in the laboratory. Samples were sorted, identified to species, measured, and counted.

Short-term changes in epibenthic fishes and invertebrates

Apalachicola Bay seasonally undergoes considerable biological changes due in part to variability of physico-chemical parameters. Peak levels of productivity occur during specific periods (Livingston, 1976; Livingston et al., 1976). A seven month sampling period (May-November) was chosen to coincide with increased rainfall and periods of peak productivity to study the short-term effects of the increased input of highly colored, acidic runoff on the epibenthic fauna of East Bay. This sampling program was designed to test the immediate response of the bay to upland runoff, and was carried out over a 2-year period (1975-1977). Thus, regular collections were supplemented with intensive sampling efforts during periods of precipitation.

Day and night collections at stations 4A, 5B, and 5C were taken following periods of intensive rainfall using a 5 meter (16 foot) otter trawl (1.9 cm wing mesh and 0.6 cm mesh liner) at speeds of 2-5 knots. During dry periods, collections were taken at least every two weeks for baseline data. Seven repetitive 2-minute trawls were made at each station. This trawling technique has been proven adequate for representative sampling by Livingston (1976). Fishes and invertebrates were preserved in 10% formalin solution for later identification and measurement. Fishes were measured by standard length, shrimp from tip of telson to end of rostrum, and crabs were measured laterally across the carapace. Snails and bivalves were counted.

Surface and bottom physico-chemical parameters (dissolved oxygen, temperature, pH, salinity, depth, and light penetration measured by standard Secchi disk) were taken at each station prior to trawling.

Water samples (top and bottom) were collected with a 1-liter Kemmerer bottle for color and turbidity analysis. Wind direction and velocity were estimated and relative tidal changes recorded. Local precipitation was obtained from the East Bay forestry tower and river flow from the Army Corps of Engineers (Mobile, Alabama).

Long-term changes in epibenthic fishes and invertebrates

Biological sampling was carried out in the bay at fixed stations (Fig. 1) with 5-m (16 foot) otter trawls (1.9 cm mesh wing and body; 0.6 cm mesh liner) towed at speeds of 2.0-2.5 knots. The determination of station placement and sampling procedures has been described by Livingston (1974, 1976). Day and night samples were taken at monthly intervals from March, 1972 to May, 1974. Only day samples were taken thereafter. Complete data were not taken during 3 summer months of 1974. All collections were preserved in 10% formalin and identified to species.

The following schedule was followed from March, 1972 to February, 1977.

7 (2-minute) trawl tows per month
stations 1, 5, and 6

2 (2-minute) trawl tows per month
stations 1A, 1B, 1C, 2, 3, 4, 5A

This program was carried out in conjunction with more intensive sampling in East Bay (stations 4A, 5B, 5C) and St. George Island (1E, 1X). Analysis was carried out on composite samples including stations 1, 5, and 6. Varying combinations of stations and time periods were used for calculations, which generally were performed on composite data at monthly intervals. In addition, seine and trammel net collections were made in various areas according to methods described by Livingston (1974).

Statistical methods and computations

Most of the quantitative analyses were made using an interactive computer program (the Special Program for Ecological Science: Livingston and Woodsum, 1977; see Livingston et al., 1977) under the KRONOS operating system on a Cyber 73 computer (Florida State University Computer Center).

In all computations, numbers of individuals (N), dry weight biomass (B), and number of species (S) were used. Various indices were determined from the invertebrate and fish data. These included the Margalef Index (MA) (Margalef, 1958), the Simpson Index (SI) (Simpson, 1949), and the Shannon Index (H') (Shannon and Weaver, 1963; Pielou, 1966 a, b, 1967, 1969). Relative dominance (D_1) was determined by dividing the number of individuals of the single most dominant species by the total number of individuals (McNaughton, 1968; Berger and Parker, 1970). The rationale for the use of these indices has been developed elsewhere (Livingston, 1975, 1976) and will not be detailed here. The ρ measure of affinity (van Belle and Ahmad, 1973) was used for the cluster analysis with a locally modified version of a program furnished by Dr. D. F. Boesch. All other statistical calculations were run with programs taken from the Statistical Package for the Social Sciences (S.P.S.S., 1975) and the Biomedical Computer Program (B.M.D., 1973).

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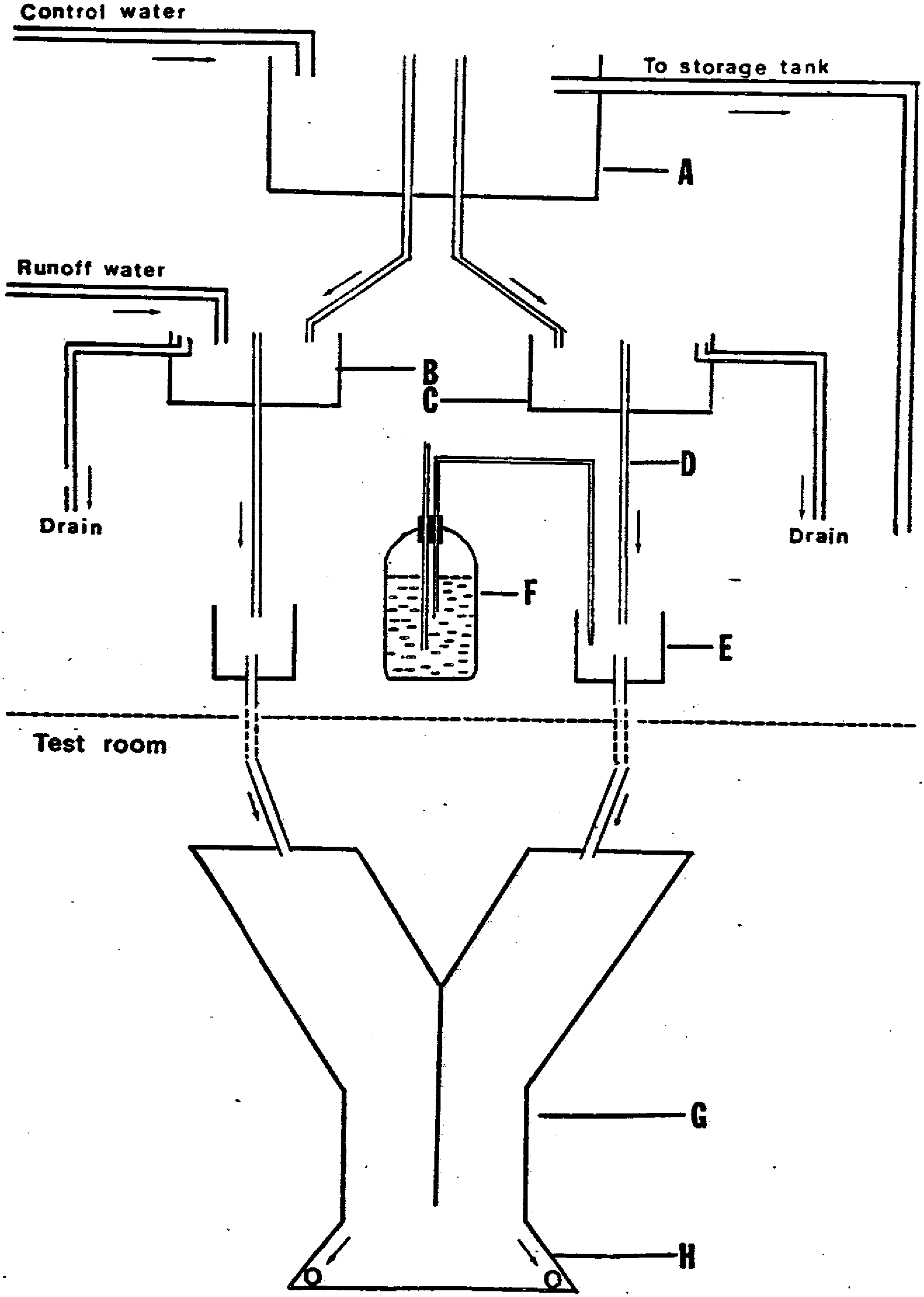
Fig. 1: See Chapter 1 (Fig. 1) for chart of station locations.

Fig. 2: Apparatus for the determination of blue crab avoidance


Control water
To storage tank
A
Runoff water
B
C
D
Drain
Drain
F
E
Test room
G
H

III. Physico-chemical Relationships

Habitat characteristics: sediments

The Apalachicola Bay System is characterized by sand, silt, and shell components in various mixtures; St. Vincent Sound and northern portions of Apalachicola Bay are silty areas that grade into sand/silt and shell gravel as St. George Island is approached. Relict coarse (quartz) sands are covered by fine-grained material deposited by the Apalachicola River and biological processes in the bay. East Bay is composed of silty sand and sandy shell. Relatively high turbidity and sedimentation have significantly reduced benthic macrophyte distribution in all but the shallowest (fringing) portions of the bay. Details of the sediment type and distribution are already available (Livingston et al., 1977).

Station 1

This is a mid-bay station approximately 2 m in depth. The bottom is somewhat loose, barren of vegetation, with occasional large wood and shell fragments. There are scattered coarse, sandy deposits in an otherwise fine sand area. The monthly average grain size is 2.60 ø units and contains 6.52% organic matter. There was considerable variation between samples both for grain size and organic content, with no obvious trends. The concurrent decrease in grain size and increase in organic content noted in February, 1976, coincided with maintenance dredging activities nearby. Recent dredging (early 1978) may have a considerable effect on the benthic areas around station 1.

Station 1X

This station is situated in a shallow (1 m), protected grass bed, composed mainly of Halodule wrightii. The bottom is very firm sand with scattered oyster bars in the area. The average monthly grain size is 2.02 ϕ units and the organic content averages 2.06%. There was little between-sample variation in grain size but the sediment organic content increased from July to January, coinciding with the die-off and decomposition of Halodule blades. Various forms of benthic macrophytes such as Ulva lactuca and Gracilaria spp. are found here. A barrier oyster bar lies just offshore; inside this reef, detritus is deposited in the protected embayments by northerly and westerly winds. Considerable amounts of such detritus are found in this area.

Station 3

Station 3, approximately 0.5 km north of the Gorrie Bridge, is a shallow area (1-1.5 m) subject to strong river action and tidal currents. Various forms of detritus (branches, logs, leaves, etc.), brought in by (winter-early spring) river flooding, are commonly found here. During summer months, there is extensive colonization and deposition of various species of blue-green and green algae. Water hyacinth (Eichornia crassipes) is found along the shore. Marsh grasses in this area include Phragmites communis, Typha latifolia, and Juncus roemerianus. The bottom is firm, fine sand with beds of Ruppia maritima and Vallisneria americana in the vicinity. The average grain size is 2.83 ϕ units and organic content averages 3.52%. There has been some variability between samples for grain size and organic content, probably resulting from the river-deposited debris.

Station 6

This station is located in the middle of a shallow (1 m), protected embayment close to the Apalachicola River with seasonally dense beds of *Ruppia* nearby. The bottom is a loose, fine sand-silt. Woody debris is almost always noted in the samples. The monthly average grain size is 3.64 ø units and organic content averages 5.60%. Samples have been variable with respect to grain size and organic content, with no trends observed.

Station 4A

This station is in a shallow (1-2 m) *Vallisneria* bed in upper East Bay. The bottom is fairly loose silty-sand. The monthly average grain size is 3.98 ø units and organic content averages 8.61%. A fall peak in organic content probably results from the die-off of *Vallisneria* blades.

Station 5A

Station 5A, approximately 1 km south of the upper marshes of East Bay, has a monotonous silty-sand bottom with sparse (scattered) growth of *Ruppia maritima*. Trawl catches indicate the presence of *Gracilaria foliifera*. The upper coastline is fringed by beds of *Vallisneria americana* and upland marshes. The average grain size is 1.82 ø units and organic content averages 2.58%. Between-sample variation in grain size is low, but organic content increases from summer through winter, to some extent because of the *Vallisneria* die-off.

Station 5B

This station is located in an upper East Bay tributary. The bottom is loose silt with Vallisneria fringing the shoreline. The average grain size is 4.22 ø units and organic content averages 11.23%. Grain size and organic content (relatively high) show relatively little variability throughout the year.

In general, grain size decreases and organic content increases as one moves from the outer Apalachicola Bay area into the upper reaches of East Bay. The observed late summer-fall die-off of benthic macrophytes coincides generally with an increase in sediment organic content.

Long-term changes in local rainfall and Apalachicola River flow

As part of the overall research effort in the Apalachicola Estuary, and with the cooperation and help of Dr. Duane A. Meeter (Department of Statistics, Florida State University) and Mr. Glenn C. Woodsum (Department of Biological Science, Florida State University), an analysis of long-term meteorological data (temperature, rainfall, river flow) is presently under way. This review is anticipatory of a publication now in preparation (Meeter et al., 1978), and will serve to place the present 6-year biological study of the Apalachicola Bay System in perspective with respect to the long-term trends of the primary meteorological forcing functions. Since the Apalachicola River appears to be one of the key physico-chemical components of the bay system, this background is considered important to the present impact analysis.

Apalachicola River flow data (1958 - present) were provided by the U. S. Army Corps of Engineers (Mobile, Alabama). Data were taken near Blountstown, Florida and initial analyses (mean flow rates, etc.) were provided by Mr. Roger P. Ruminik (U. S. Geological Survey; Tallahassee, Florida). River flow information from 1920 - 1957 was provided by the Army Corps based on Blountstown river gauges monitored by the U. S. Weather Service. Local climatological data (temperature, rainfall) for Apalachicola, Florida (1937 - present) were provided by the Environmental Data Service (NOAA) in Apalachicola. The rainfall data for the East Bay area were provided by the East Bay Forestry Tower located in the Tate's Hell Swamp (Franklin County, Florida).

Raw data (monthly means) are shown in Fig. 1. An inordinately high level of river flow occurred during 1929. River flow and rainfall (Apalachicola) showed considerable seasonal and annual variability. The air-temperature data (Apalachicola) were relatively constant with respect to the annual fluctuations of temperature peaks although the seasonal low temperature values did vary from year to year. Our biological sampling period (1972 - present) went from peaking river flow (1973 - 1975) and rainfall (1974 - 1975) to present low flow and drought conditions. Temperature reached an extreme low during 1976 after several years of relatively moderate winter conditions. Composite monthly means (river flow, 1958 - 1977; rainfall, temperature, 1937 -1977) are given in Fig. 2 while $\log_{10}$ monthly means ± 1 standard deviation are shown in Fig. 3. The river flow usually peaks in the winter or early spring (March-April); low flows occur from June to November; there is a relatively high level of monthly variability during winter periods of high flow. There is considerable variability with respect to monthly river flow peak placement from year to year, and this variability is superimposed on the extreme level of annual variation throughout the 50-year period of study. There is a minor peak of local rainfall which tends to coincide with the river flow pattern during winter and early spring. The highest levels of local rainfall occur from July through September, coinciding with peaking temperatures. The lesser rainfall peak coincides with maximal winter river flows while the major summer rainfall tends to follow the flow peaks by 5 to 6 months. The trends from 1972 to 1977, with monthly river flow ranges, are shown in Fig. 4. Since the extreme flooding during the winter of 1973, there has been a general decrease in river flow until the present time. During the winter of 1976, there was only minor peaking of river flow and this trend appears to have continued through the winter of 1978.

A moving average plot (6-month, 36-month) of river flow and rainfall is shown in Fig. 5. Peak flows in both parameters tended to occur in 5- to 8-year cycles and there was a general similarity in the patterns of rainfall and river flow. The possibility of even longer term cycles of rainfall with a period of 40 or more years remains open according to these data. There was a generally consistent level of low river flow conditions with the exception of the mid 1950's (1955 - 1957) at which time flow rates seemed to be extremely low. A spectral analysis was carried out on the river flow data (Fig. 6) to determine the range of smoothing (60 lags, 240 lags) and to develop an optimal level for peak resolution and confidence intervals. Peaks at 4- to 6-month intervals were interpreted as harmonics of the 12-month cycle (the river flow is not sinusoidal). There was another, less determinate peak at an interval from 40 to 120 months.

Further analysis was based on what was considered to be a reasonable compromise of 120-month lags (Fig. 7). The logged rainfall data showed high standard deviations during dry periods, so square root transformations were used in the analysis. The 6-month and 12-month intervals of the bi-peaked rainfall pattern were evident and the relationships of river flow and rainfall described above were evidently real according to this analysis. There was also a well-defined rainfall peak at 80 to 100 months, thus confirming a 6- to 8-year rainfall cycle which corresponds generally with the results of the moving average analysis. These data tend to confirm long-term trends in both parameters which, though not exactly correlated, tend to occur in the same area of the spectrum.

Analysis of coherence (level of correlation between the two series at a specific frequency or period) and phase (amount by which sine waves of two series are separated) of river flow and rainfall data from 1937 to 1977 are shown in Fig. 8. The long-term period phase shift approximation close to 0 indicates that the long-term waves in the two series occur nearly simultaneously, and that the approximate period is around 7 years. The coherence estimates at 6, 12, and 60-80 months (5 to 7 years) reinforce our previous results. The (winter) peaks of the two series co-occur in time while the summer rainfall peak follows the river flow peak by around 5 months (120-150° phase shift). It should be added that if the coherence is 0, the phase estimate is random and stronger coherence values indicate enhanced precision of the phase estimate. Work is continuing concerning long-term drainage patterns and the potential influence of the Jim Woodruff Dam on the Apalachicola drainage system.

The physico-chemical environment of the bay

Data were analyzed with respect to general changes at two representative stations (1, Apalachicola Bay; 5, East Bay). Water temperatures are shown in Figs. 9 and 10. There was little spatial variation (depth, area) of water temperature at a given time. Although temperature peaks tended to remain stable from year to year, there was a general decrease in winter lows from 1974 to 1977 which was particularly pronounced during the winter of 1976-77 (Fig. 11). During this period, the seasonal range of variability was extended as compared to the previous 6 years. This is shown, along with a recent warming trend, in Fig. 12.

Salinity changes in inner and outer portions of the bay are shown in Figs. 13-16. River flow was a major determinant of the salinity regimes in the system. East Bay was less saline than Apalachicola Bay, and there was considerable stratification, especially during summer and fall periods. The secondary decreases in salinity during summer and fall months appeared to reflect surface runoff from local rainfall patterns. Such changes were more pronounced in East Bay than in Apalachicola Bay. Low salinities occurred during winter and spring months (associated with river flow), followed by increasing salinity during the summer. There was then a rapid decline in the late summer or fall (coincident with increased local precipitation), and this was followed by a fall or winter salinity peak just prior to the ensuing decrease in salinity with renewed increases in river flow. River flow and rainfall fluctuations from 1970 to the present are shown in Figs. 17, 18, and 18A; Table 1. Peak flows occurred in 1973, since which time total flow, peak flow, and seasonal variability have decreased, especially during the 1976-78 period. The rainfall data show peak monthly values during the summers of 1970 and 1974-75, while total annual rainfall peaked in 1970 and 1975. The relatively higher values in Tate's Hell Swamp relative to the Apalachicola area are noteworthy. During the period of study (1972-1978), the major water flows into the system thus occurred during the period from 1973-1975. Moving average analyses of East Bay and Apalachicola salinities (Fig. 19) reflect this trend, with lowest salinities occuring during the 1973-75 period. The salinity is now on the increase (1976-present). These changes are reflected in baywide salinity data (Figs. 20-24) with the exception of Sike's Cut (station 1B; bottom salinities remain high and stable) and upper portions of East Bay (station 5A; salinities have remained low since the end of 1974).

The influence of the major river flooding during the winter of 1973 is apparent in various physico-chemical functions such as color, turbidity, and Secchi readings (Figs. 25-27). On the whole, there has been a general decrease in turbidity in inner and outer bay areas since 1973. Color followed this decrease at station 1; however, station 5 had peak color levels during 1974-75 and color was generally higher in East Bay than in Apalachicola Bay except during the 1973 river flooding; this reflects the relative influence of river flow and local rainfall on various portions of the bay. Secchi readings have continued to go down in East Bay while Apalachicola Bay has had relative increases in light penetration during the last 2 years of sampling. These data are presented in more detail in Figs. 28-42. East Bay was thus more highly colored than Apalachicola Bay, and showed a trend which appeared to be linked to patterns of local rainfall and runoff in the Tate's Hell Swamp area. Color levels became most pronounced during the summers of 1974 and 1975. This was not the case with respect to turbidity (Figs. 12-15), which tended to decrease during the study period in the bay as a whole. Turbidity seemed to be closely correlated with river flow. Turbidity peaks usually occurred during winter and spring months. One notable exception to this tendency occurred during the summer of 1974 in areas of East Bay. These trends in color and turbidity were generally reflected by the Secchi disk data, where significant decreases with time occurred in East Bay relative to the Apalachicola Bay area. Dissolved oxygen data are shown in Figs. 43-51. There was considerable seasonal variation at most stations with peak levels generally occurring during winter and spring months indicating the usual relationship with water temperature and salinity. In East Bay, there was a significant increase in dissolved oxygen with time relative to those levels in Apalachicola Bay.

A statistical treatment was carried out with the first four years of physico-chemical data. The seasonal changes in various physico-chemical variables in the Apalachicola Bay System have already been described (Livingston, 1974, 1976; Livingston et al., 1974; Livingston et al., 1976). Overall, this is a shallow barrier island estuary dominated physically by the fluctuations of the Apalachicola River; this is especially true of parameters such as salinity, color, turbidity, and nutrient levels. Generally, this is a highly turbid bay with considerable oyster bar development and little benthic macrophyte productivity except in shallow (fringing) areas. Tides in the Apalachicola Estuary are semi-diurnal (mixed, unsymmetrical) with a small range (up to 1 m). Winds in the area follow no clear directional trend although during fall and winter there is a northerly flow which becomes southerly during the rest of the year. In June, 1972, Hurricane Agnes came ashore near the Apalachicola region with winds gusting to 55 knots and tides around 2 m above the norm. The results of a factor analysis (Table 2) indicate that high river flow is usually associated with increased color and turbidity and reduced Secchi readings, and low levels of salinity, temperature, and chlorophyll a. These results are consistent with the known seasonal pattern of these factors, and indicate the important influence of the Apalachicola River on the physical environment of the Apalachicola Estuary. While the river dominates the seasonal fluctuations of parameters such as salinity, long-term changes in the overall salinity of the bay appear to be related also to other functions, such as local rainfall and runoff. This would indicate that long-term salinity levels are determined by multiple interactions which lead to apparently contradictory results when viewed as short-term (as distinct from long-term) trends.

Physico-chemical changes in East Bay: visual observations

The background river flow and rainfall conditions (Figs. 52-53) should be kept in mind when specific water quality functions in the upper portions of the Apalachicola Estuary are analyzed. Visual observations of water color (Fig. 54) indicated movement of highly colored water into eastern portions of East Bay following locally heavy rainfall (summer, winter). Affected areas include our stations 5, 5A, 5B, 5C, and, occasionally, 4 or 1C. Tidal currents and wind dominated the movement of such water through the system. These observations have been corroborated by LANDSAT images of the area (Jack Hill, U. S. E. P. A., Las Vegas; pers. comm.). Water quality maps of East Bay from November, 1974 to October, 1975 (Figs. 55-56) add further support to these observations. Conditions of high color and low pH usually occurred in eastern portions of East Bay during winter and summer periods of high local rainfall and runoff into the bay.

Water quality in Tate's Hell Swamp: experimental studies

According to data supplied by a local paper-pulp mill (Dr. Walter L. Beers, Jr., pers. comm.), overall forestry activities in Tate's Hell Swamp (i.e., total cleared acreage) peaked during 1971-1972. Portions of the drainage system immediate to the East and West Bayous (upper East Bay) were cleared largely during a period from 1973-75. Since this time there has been little clearcutting or forestry activity in this area, with one notable exception. Originally, the Round Bay drainage (station 4A, Fig. 1 of Chapter I) was used as a control station since it was out of the immediate drainage associated with cleared areas of the East and West Bayou systems. However, during the late spring of 1976, a drainage ditch was dug which emptied into Round Bay, and, during the next 12 to 18 months, considerable upland portions of this drainage system were clearcut (Figs. 57-59). Also, during 1975, a cooperative watershed study was started in a managed area of the East Bay system whereby sections 9 and 16 were managed according to the following schedule (Dr. Walter L. Beers, Jr., pers. comm.):

Activity	Date
chopping	9/15/75-10/3/75
harvesting	9/17/75-11/10/75
intermittent burning	12/8/75-12/25/75
chopping and bedding	2/2/76-2/25/76
hand planting	2/25/76-3/18/76
fertilization	3/6/76-3/7/76
canal re-excavation	3/24/76-5/15/76
no site activity	5/15/76-present

This area has been studied by scientists from the University of Florida and a local forestry operation. In a cooperative effort from January, 1975 to the present, our group has monitored water quality functions in this drainage system on a monthly basis. Station placement is shown in Fig. 1, Chapter I of this report. The purpose has been to sample the area before, during, and after forestry management operations with a comparison in areas above the cleared sections (the so-called control stations, B4 and B8). As shown, this area drains directly into the West Bayou arm of East Bay (station 5B).

Results of this ongoing study are shown in Figs. 60-68. The pH in the Tate's Hell Swamp drainage is relatively low. The temporal change in pH with time has been modest. Stations closer to the bay (B10) are characterized by generally higher pH although, during periods of rainfall, such levels remain relatively low. Color tends to increase in areas receiving runoff from newly cleared areas relative to control stations (B4, B8). This is especially apparent during winter and summer periods of high rainfall. Stations closer to the bay (B9, B10) are characterized by relatively lower levels of color. Turbidity follows this trend with particularly high levels during the summer of 1977 at certain stations (B1, B7). Turbidity is generally quite low at stations closer to the bay (B10). It is possible that turbidity has been affected by the construction of a weir and settling basin at station B7. Nutrient levels are shown in Table 3. Whereas the control stations showed relatively uniform levels of various nutrients through time, there were periodic increases in PO_4 and especially NO_3 at various other stations, especially during the summer of 1975 and the winter of 1976. During periods of drought, the drainage ditches usually dry up; however, during heavy rainfall, water runs through the drainage ditches directly into the West Bayou area. High color levels usually occur only when water has been standing in upland (cleared) portions of an adjacent drainage area for a certain period of time.

Overall, these data would lead to the following conclusions:

1. The construction of drainage ditches in Tate's Hell Swamp constitutes a form of channelization whereby upland swamp water is drained directly into receiving bodies of water (in this case, the eastern portion of East Bay). The temporal pattern and total amount of rainfall, together with levels of clearcutting and other forestry operations, determine the quality of the runoff. During periods of drought there is no such flow, and repeated episodes of heavy rainfall are usually needed before water quality changes in receiving systems are noted.

2. The pH levels in aquatic portions of Tate's Hell Swamp are naturally low and this parameter does not appear to be directly affected

by the clearcutting operation. The color and turbidity of runoff from cleared areas is usually high and, as water moves toward the bay, the pH goes up, turbidity is substantially reduced and color undergoes moderate decreases. All such functions are strongly influenced by the timing, duration, and intensity of local rainfall and the associated forestry operations as outlined above.

3. Impact analysis should include consideration of various aspects of forestry operations such as clearing, draining, fertilization, and revegetation. Although the clearing phases do not seem to affect the overall levels of pH, the draining of the swamp into receiving waters does shorten the residence time of runoff water in upland areas. Extending the drainage area and clearing the land of vegetation increases the rapid lateral movement of water through the system during periods of heavy rainfall and, consequently, aggravates the water quality situation. The data indicate that such effects on water movement and quality are seasonal and entirely dependent on local rainfall patterns and surface runoff features of the immediate upland areas. The loss of nutrients from such lands is part of this general water quality situation.

Long-term water quality changes in the Apalachicola Estuary

The relationships of river flow and local rainfall to water quality functions of the Apalachicola Estuary have already been discussed. While river flow has a dominant effect on salinity and other physico-chemical functions, local runoff is also an important determinant of water quality, especially in East Bay. Proximity to river flow and upland drainage systems such as Tate's Hell Swamp are important considerations in any evaluation of causal relationships of water quality in the Apalachicola Estuary.

Using data taken by the Florida Division of Health (John Taylor, Jr., pers. comm.) from October, 1970 to November, 1971, a comparison was made of long-term changes (1970 -present) of salinity and pH in East Bay (Figs. 69-78). Salinity changes at stations 4 and 5 reflected river flow and rainfall variations over the years with high salinities during the initial period of sampling giving way to generally reduced salinities (1974-76) and, finally, to a trend of gradual recovery of seasonally high salinity levels during the period from 1976 to the present. The major river flow event during the winter of 1972-73 is evident in the salinity regime throughout the bay. At station 4A, the subsequent recovery of increased salinity was evident during 1976 but there was a general decrease in salinity in 1977. This runs counter to trends in other portions of East Bay. At stations 5B and 5C, these trends in salinity showed progressive decreases in salinity from 1974 to the present time. This decrease does not resemble the patterns in any other portion of the bay and is not congruous with the long-term river flow and rainfall conditions (Figs. 52-53; Table 1). The pH levels at station 4 have not varied to any significant degree during the study period. At station 5, the pH range was maximal during 1976 although readings did not go below 6 and there was no general trend, up or down. At station 4A, there was a tendency for the pH to increase from 1975 to mid-1976 and then generally to decrease from mid-1976 to the present. The lowest pH ever recorded at this station was taken during the summer of 1977. Low pH levels were taken during the summer of 1975 at stations 5B and 5C. The pH levels during this period were tested statistically and found to be significantly lower than those during the 1970-71 period (prior to intensive local clearing activities). At both stations (5B, 5C) there has been a recent (1975-76) trend of increasing pH despite the fact that salinities were falling at station 4B during this period. This could be interpreted as indicative of a recovery of water quality in this area of the bay.

These data indicate that local forestry operations do have an effect on water quality conditions in aquatic (downstream) receiving systems. Recent clearing and draining activites have caused pH decreases in the Round Bay area, a former control area. Impact due to low pH was maximal in East and West Bayous following extensive local forestry operations and heavy summer rainfall during the summers of 1974 and 1975. Recovery of the pH levels appeared to be relatively rapid (1-2 years) although related effects on local salinity conditions did not follow the water quality recovery. The salinity relationships remain relatively unclear at this time. The extent of the water quality changes reflects input from a number of variables: extent, type and location of forestry operations, timing and intensity of local rainfall, river flow trends, and temporal (sequential) successions of other contributing factors. Water quality changes in upper East Bay can be related directly to upland forestry management activity. The quantitative aspects of water movement from drained portions of upland areas remain poorly understood and appear to be more long-lasting than the changes in water quality as indicated by pH.

Changes in other water quality functions in upper portions of East Bay are shown in Figs. 79-90. Dissolved oxygen levels at station 4A became more variable with time; the lowest levels occurred during the summer of 1977 and the highest levels occurred during the winter of 1977-78. At stations 5B and 5C, the lowest D. O. levels were recorded during the summers of 1974 and 1975. In both areas, there was a general trend of increased levels of dissolved oxygen after the 1974-75 period. Color levels tended to decrease at all 3 stations with time until an unexplained peak occurred throughout the bay during the winter of 1978. This peak was probably caused by high local rainfall and high winds at this time. Overall, water color tended to reach its highest levels during the 1974-75 period. Turbidity peaked at station 4A during the winter of 1976-77 and peaks of turbidity occurred during the winter and early spring months of 1975 at stations 5B and 5C. These trends are apparent in the Secchi data where the trend has been downward at station 4A during the 1977-78 period while it has remained relatively stable at stations 5B and 5C.

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Table 1: Annual Apalachicola River Flow (mean monthly, C.F.S.) and Local Rainfall (cm) from 1970 to 1977.

Year (1 Jan.-31 Dec.)	River Flow	Rainfall Apalachicola	Rainfall East Bay Tower
1970	19,500	159.56	
1971	28,561	98.09	155.70
1972	23,620	121.31	203.96
1973	34,114	133.30	216.41
1974	23,575	147.17	197.87
1975	31,407	176.10	214.63
1976	26,206	121.28	
1977	22,040	98.12	

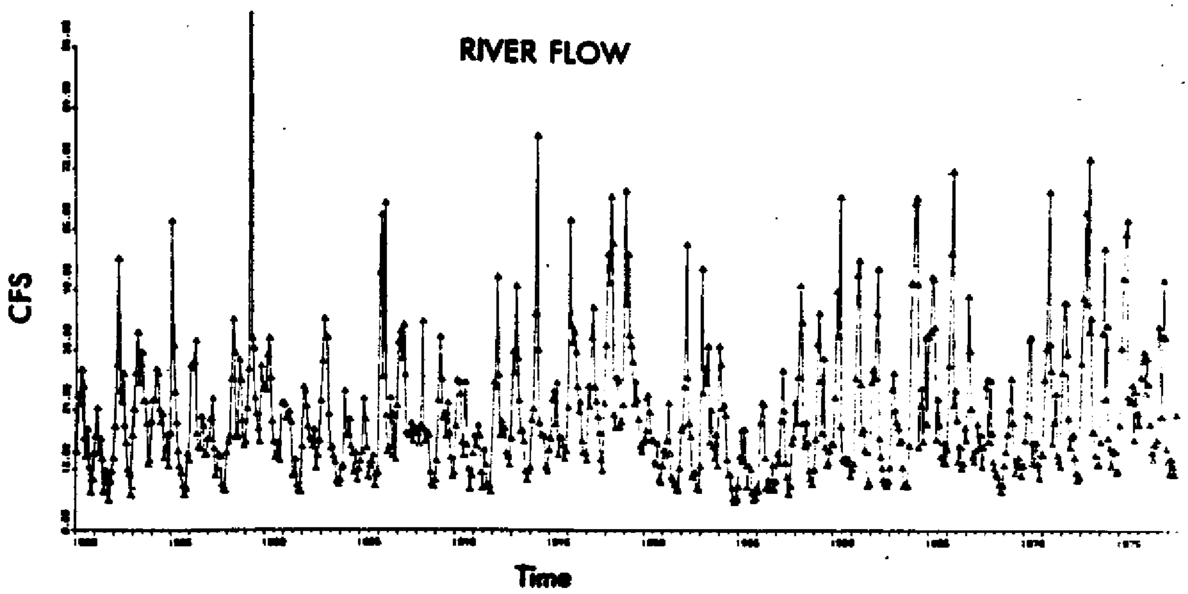
Table 2: Factor analysis of a set of physicochemical variables taken from March, 1972 to February, 1976. Color, turbidity, Secchi readings, salinity, temperature, and chlorophyll A were noted at Station 1 in the Apalachicola Estuary Tidal Data included the stages of the tide on the day of collection while the wind variable was represented by 2 vector components.

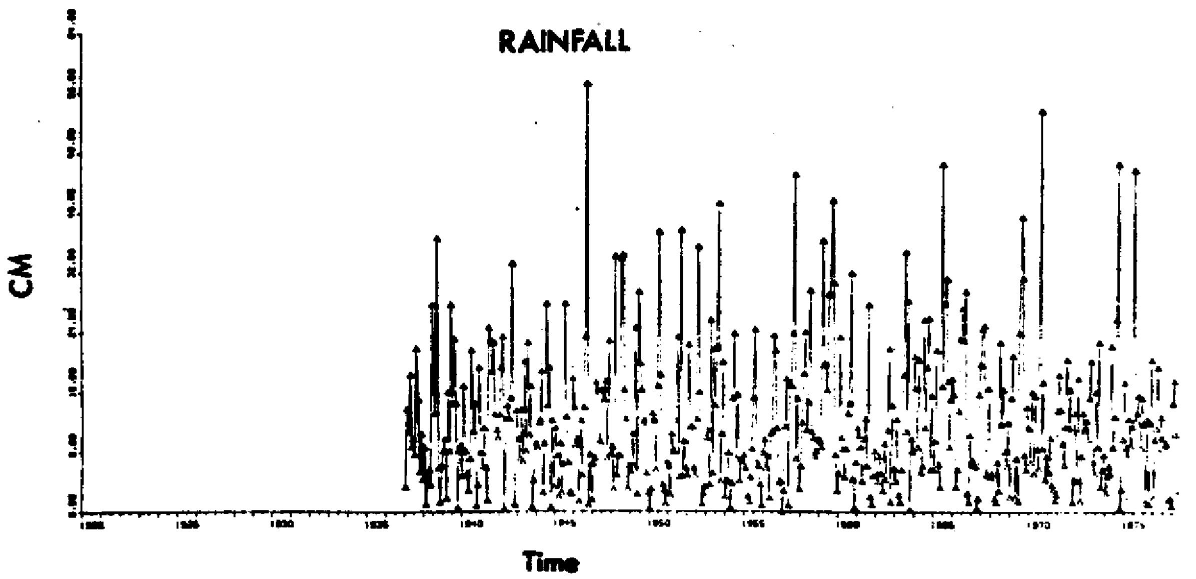
Variable	Factor 1 (49.0% of the variance)	Factor 2 (22.3% of the variance	Factor 3 (17.9% of the variance	Factor 4 (10.8% o the varia
River flow	-0.82	-0.08	-0.07	-0.08
Local rainfall	-0.04	-0.30	-0.09	0.20
Tide (incoming or outcoming	0.26	0.61	-0.68	0.06
Tide (high or low)	0.09	0.39	0.61	-0.37
Wind direction (E-W)	-0.02	0.09	0.36	0.37
Wind direction (N-S)	0.10	-0.20	0.22	0.31
Secchi	0.57	-0.07	-0.17	0.24
Color	-0.80	0.33	0.01	0.07
Turbidity	-0.73	0.54	0.08	0.23
Temperature	0.38	0.15	0.02	-0.18
Salinity	0.68	0.21	0.23	-0.02
Chlorophyll A	0.47	0.51	0.09	0.31

Table 3: Water Quality data taken during experimnetal clearcutting in Tate's Hell Swamp

		5/23/75	6/26/75	8/27/75	10/20/75	11/24/75	12/15/75	1/28/76	2/23/76	3/29/76	4/29/76
						PO_4 (μg P/ℓ)					
B	1	5.60	16.8	5.86	4.23	5.66	7.25	10.4	----	----	----
B	3	2.65	7.62	12.2	10.4	5.96	6.07	14.2	----	----	----
B	4	----	6.40	4.01	0.56	7.99	4.74	14.2	9.86	19.5	14.4
B	5	3.39	2.28	0.93	5.64	1.02	----	18.8	2.53	36.2	3.38
B	7	5.01	17.5	1.54	----	9.30	17.5	13.8	----	----	----
B	8	3.39	6.09	----	0.43	2.47	2.37	15.5	2.27	17.1	14.4
B	9	3.34	3.81	0.62	1.97	4.94	9.77	7.19	9.60	52.2	----
B	10	7.08	9.90	3.39	0.70	2.47	2.07	66.7	6.67	27.1	----
						NO_2 (μg N/ℓ)					
B	1	7.10	12.8	5.11	----	----	13.0	9.19	----	----	----
B	3	0.87	2.17	2.75	9.50	10.97	12.1	4.64	----	----	----
B	4	----	1.24	3.20	2.28	2.02	2.59	3.25	1.15	3.62	2.37
B	5	0.87	12.5	2.87	----	1.07	----	8.75	1.09	----	0.745
B	7	6.08	12.8	2.92	----	----	----	12.4	----	----	----
B	8	3.68	2.79	----	3.11	2.74	3.97	3.25	4.88	5.55	----
B	9	1.81	2.35	2.98	----	7.94	6.50	10.3	1.06	4.31	----
B	10	1.87	2.94	7.42	----	2.28	0.70	5.88	6.94	12.3	----
						NO_3 (μg N/ℓ)					
B	1	3.19	----	----	----	----	91.5	8.93	----	----	----
B	3	0.94	12.1	----	----	----	33.7	31.8	----	----	----
B	4	----	35.2	3.52	----	5.54	4.26	5.75	4.66	7.49	6.29
B	5	41.0	40.8	547.0	----	0.33	----	203.0	248.0	----	38.1
B	7	3.61	17.3	----	----	----	----	17.2	----	----	----
B	8	3.37	32.0	----	----	----	1.44	14.0	13.5	8.51	----
B	9	1.52	32.7	----	----	----	44.9	14.2	6.25	4.41	----
B	10	18.3	15.6	----	----	19.0	----	23.6	37.2	27.2	----

Fig. 1: Raw data concerning long-term changes of air temperature and rainfall (Apalachicola, Florida; 1937-1977) and Apalachicola River flow (Blountstown, Florida; 1920-1977). Monthly mean figures are given in degrees centigrade, centimeters, and cubic feet per second.


RIVER FLOW
CFS
Time


RAINFALL
CM
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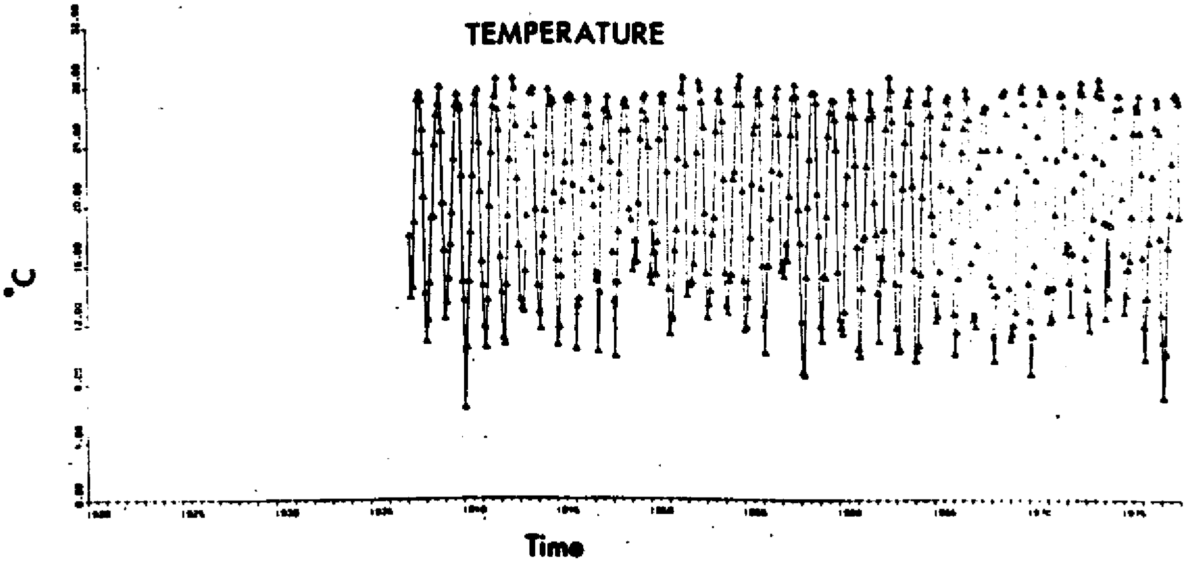
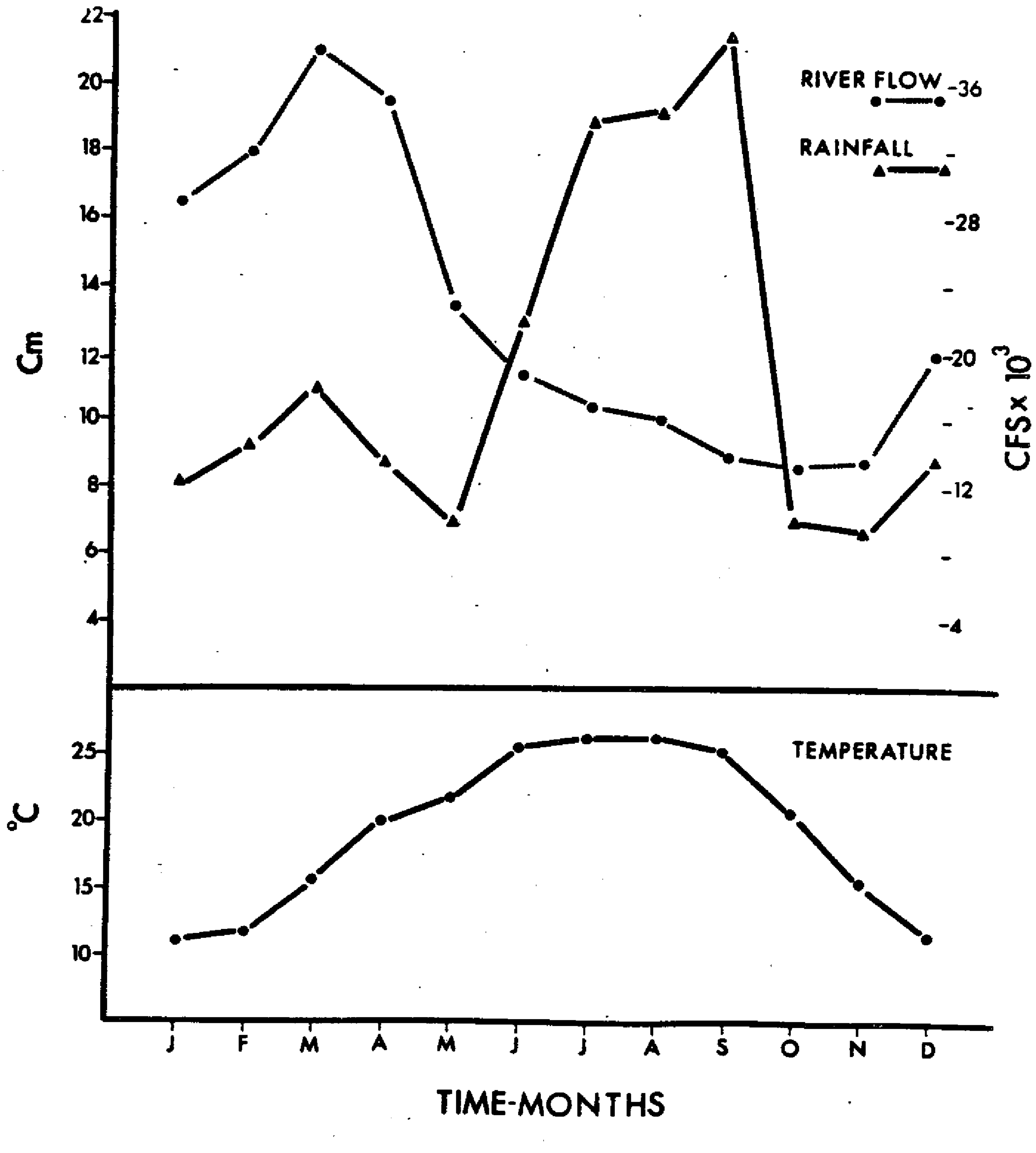

TEMPERATURE
°C
Time

Fig. 2: Mean figures by month for Apalachicola River flow (1958-77), local (Apalachicola) rainfall (1937-77) and local temperature.



22
20
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Cm
RIVER FLOW
RAINFALL
36
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20
12
4
CFS x 10^3
TEMPERATURE
25
20
15
10
°C
J
F
M
A
M
J
J
A
S
O
N
D
TIME-MONTHS

Fig. 3: Mean ($\log_{10}$) figures by month for Apalachicola River flow (1958-77) and local (Apalachicola) rainfall (1937-77) ± one standard deviation (river flow, April-September; rainfall, August-October).

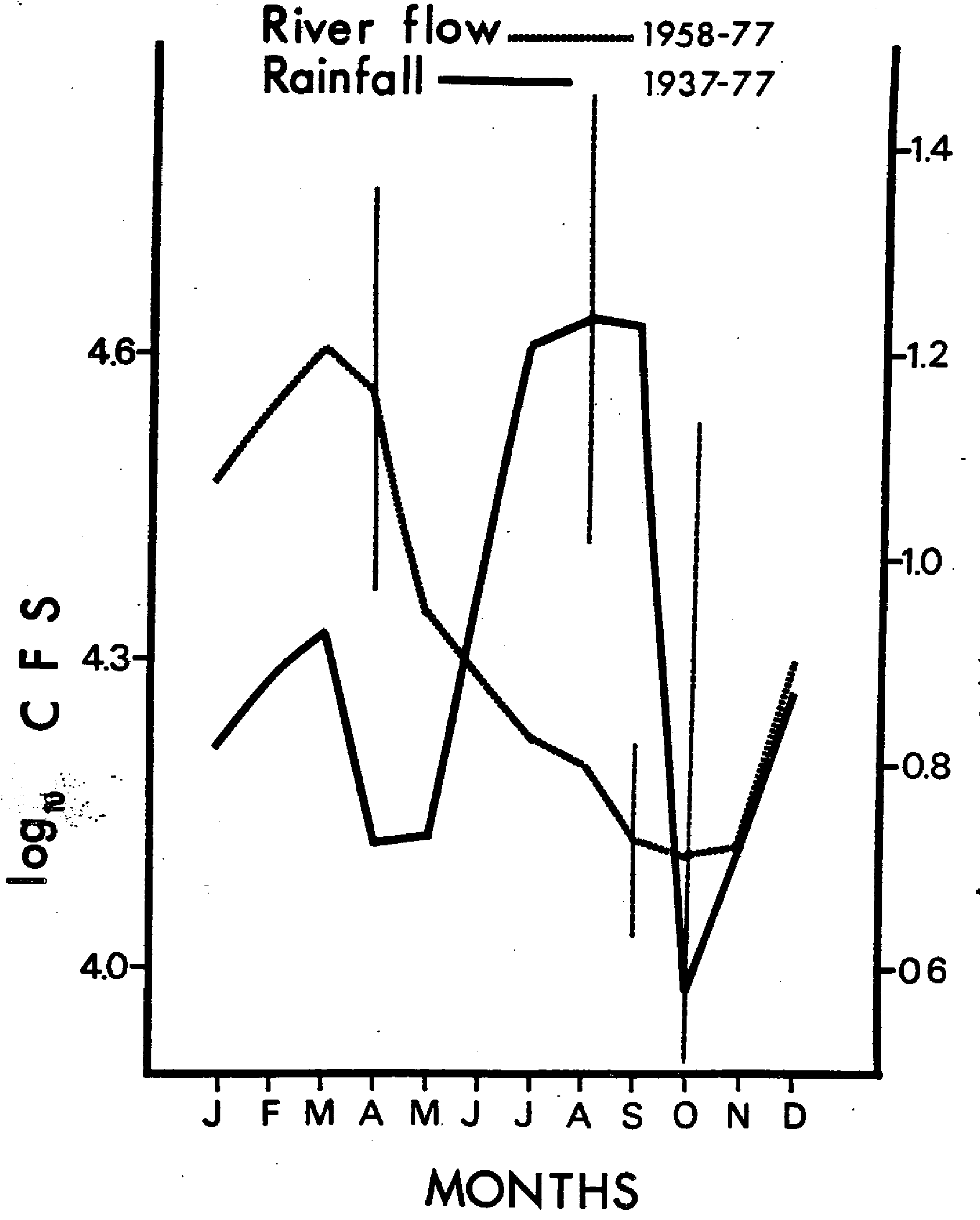

River flow 1958-77
Rainfall 1937-77
log10 CFS
4.6
4.3
4.0
log10 CM
1.4
1.2
1.0
0.8
0.6
J
F
M
A
M
J
J
A
S
O
N
D
MONTHS

Fig. 4: Monthly mean levels of Apalachicola River flow (C.F.S.) and local (Apalachicola) rainfall (cm) from March, 1972- February, 1977).

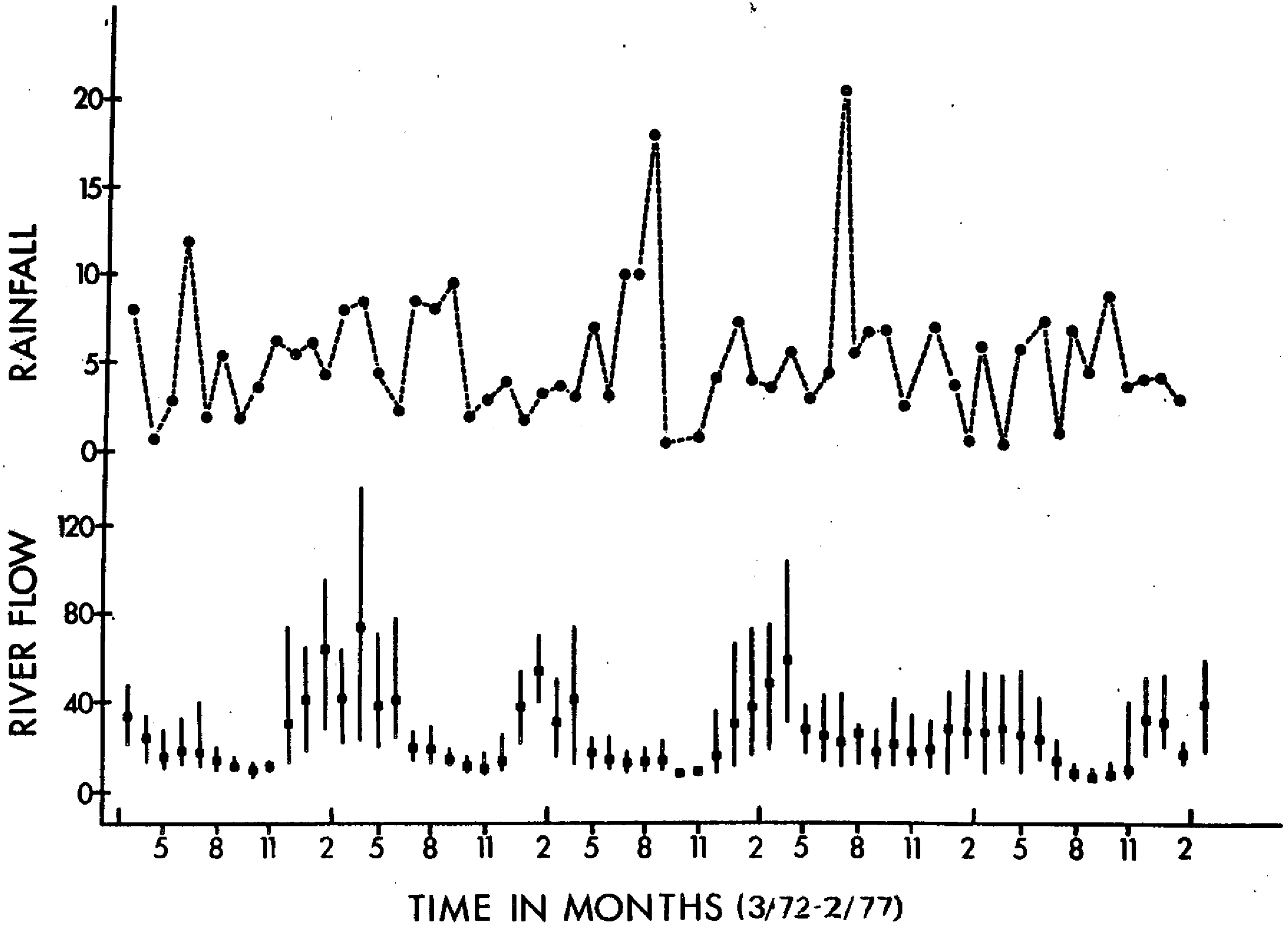

RAINFALL
20
15
10
5
0
RIVER FLOW
120
80
40
0
5
8
11
2
5
8
11
2
5
8
11
2
5
8
11
2
5
8
11
2
TIME IN MONTHS (3/72-2/77)

Fig. 5: Moving averages (6-month, 36-month) of monthly mean levels of Apalachicola River flow (1920-1977) and local (Apalachicola) rainfall (1937-1977).

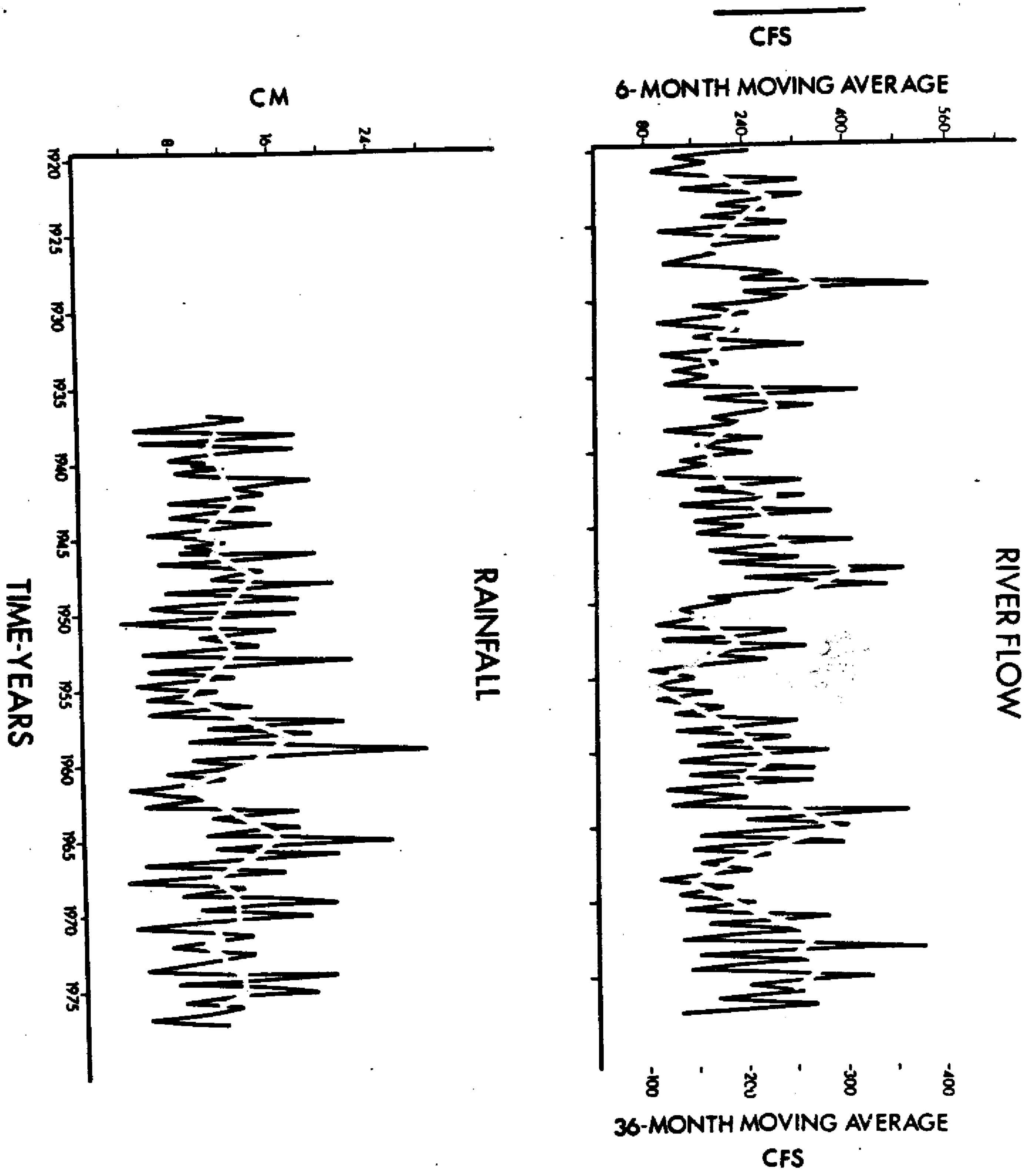

CFS
6-MONTH MOVING AVERAGE
80
240
400
560
RIVER FLOW
100
200
300
400
36-MONTH MOVING AVERAGE
CFS
CM
8
16
24
RAINFALL
1920
1925
1930
1935
1940
1945
1950
1955
1960
1965
1970
1975
TIME-YEARS

Fig. 6: Spectral analysis of Apalachicola River flow (mean monthly figures, 1920-1977) at 60 and 240 lags with 95% confidence intervals.

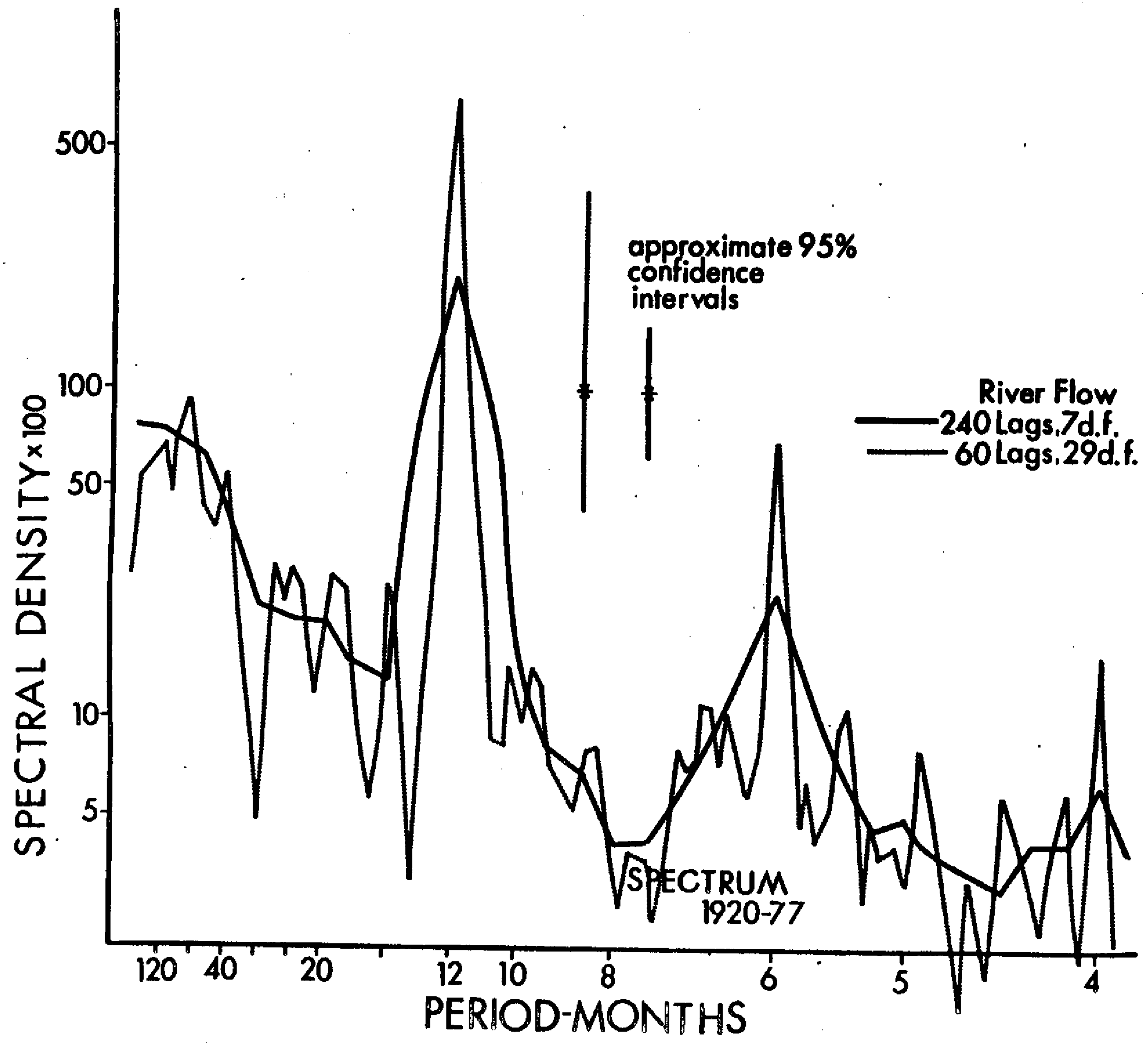

approximate 95%
confidence
intervals
River Flow
240 Lags, 7d.f.
60 Lags, 29d.f.
SPECTRUM
1920-77
SPECTRAL DENSITY × 100
500
100
50
10
5
PERIOD-MONTHS
120
40
20
12
10
8
6
5
4

Fig. 7: Spectral analysis of log monthly mean values of Apalachicola River flow (1937-77) and $\sqrt{\text{local rainfall}}$ (Apalachicola, 1937-77), using 120 lags.

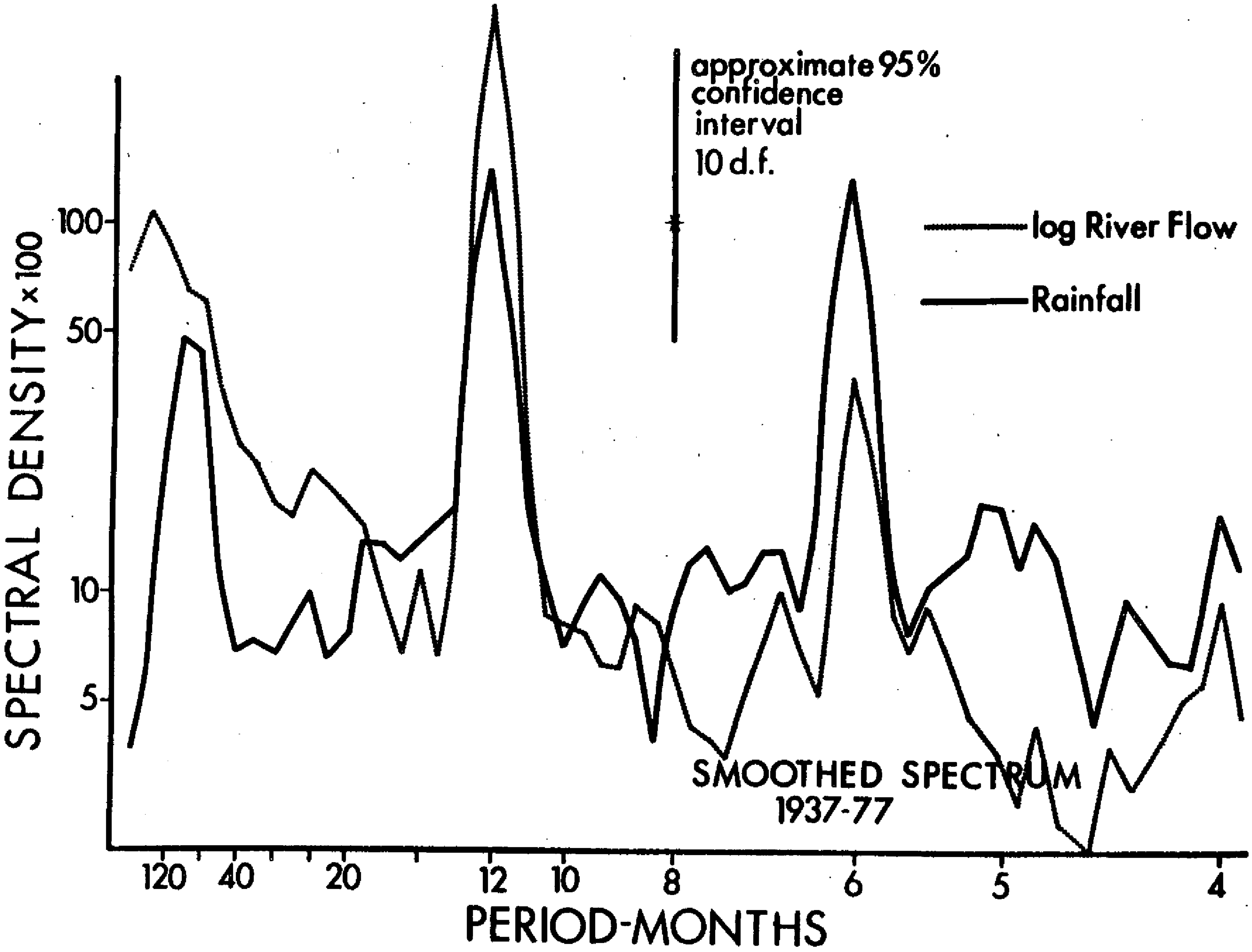
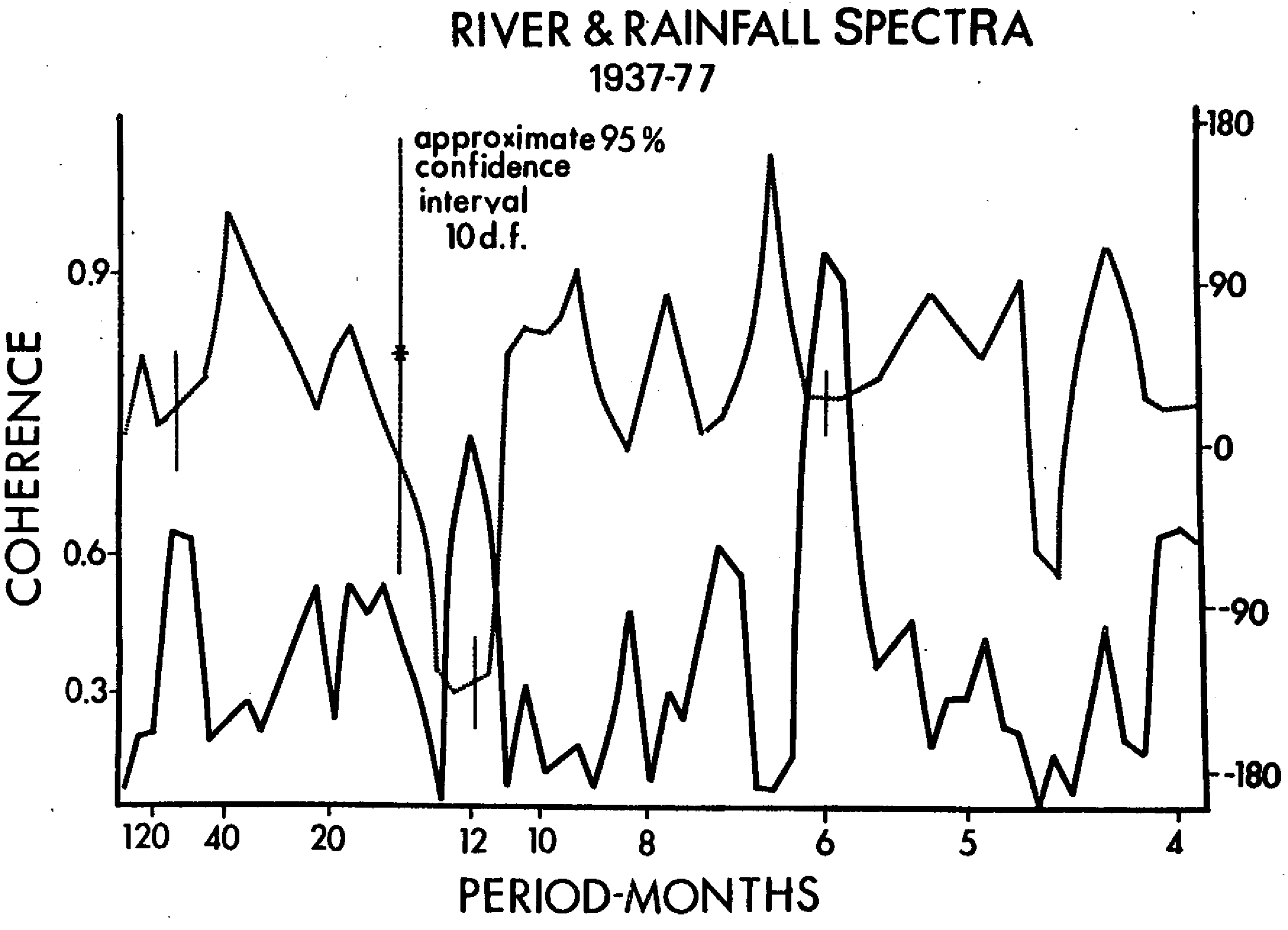
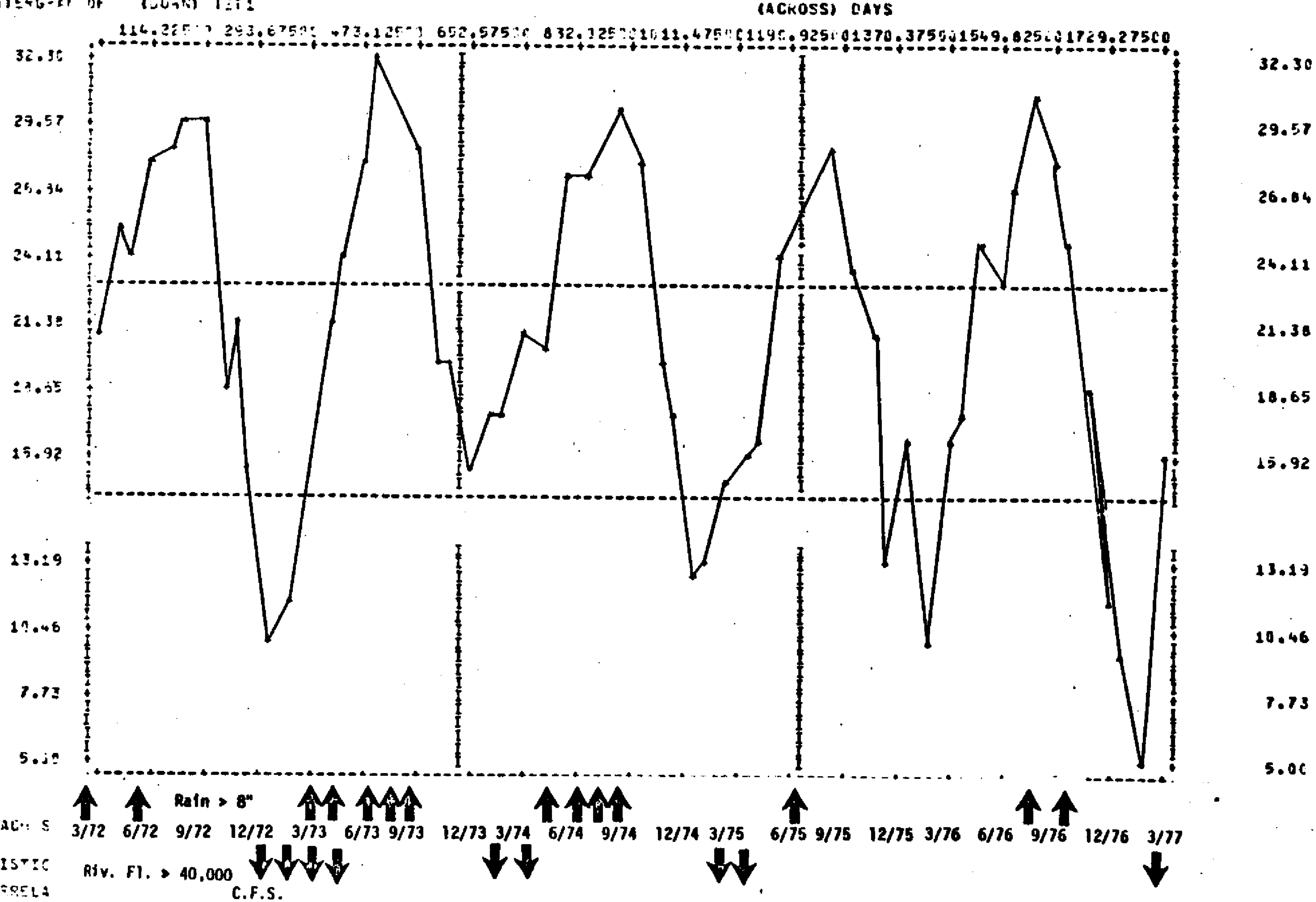

approximate 95%
confidence
interval
10 d.f.
log River Flow
Rainfall
SMOOTHED SPECTRUM
1937-77
SPECTRAL DENSITY × 100
100
50
10
5
120
40
20
12
10
8
6
5
4
PERIOD-MONTHS

Fig. 8: Spectral analysis showing coherence and phase (in degrees) of Apalachicola River flow and local (Apalachicola) rainfall (monthly means) taken from 1937-77.


RIVER & RAINFALL SPECTRA
1937-77
approximate 95 %
confidence
interval
10 d.f.
COHERENCE
0.9
0.6
0.3
PHASE DEGREE
180
90
0
-90
-180
120
40
20
12
10
8
6
5
4
PERIOD-MONTHS

WATER TEMPERATURE (SURFACE), STATION 1, °C

```
APALACH SCATTERGRAMS 1                                                77/03/11.     PAGE    3
FILE    NONAME   (CREATION DATE =  77/03/11.)
SCATTERGRAM OF   (DOWN) TET1
```



```
APALACH S
STATISTIC
  CORRELA
  STD ERR OF EST -        [illegible]      INTERCEPT (A)   -      24.41248      STD ERROR OF A -     1.67141
  SIGNIFICANCE A -        [illegible]      SLOPE (B)       -      -.00335       STD ERROR OF B -      .00157
  SIGNIFICANCE B -        [illegible]
```

Fig. 9

```
FILE    NONAME    (CREATION DATE = [illegible])
SCATTERGRAM OF    (DOWN) TEMP                                    (ACROSS) DAYS
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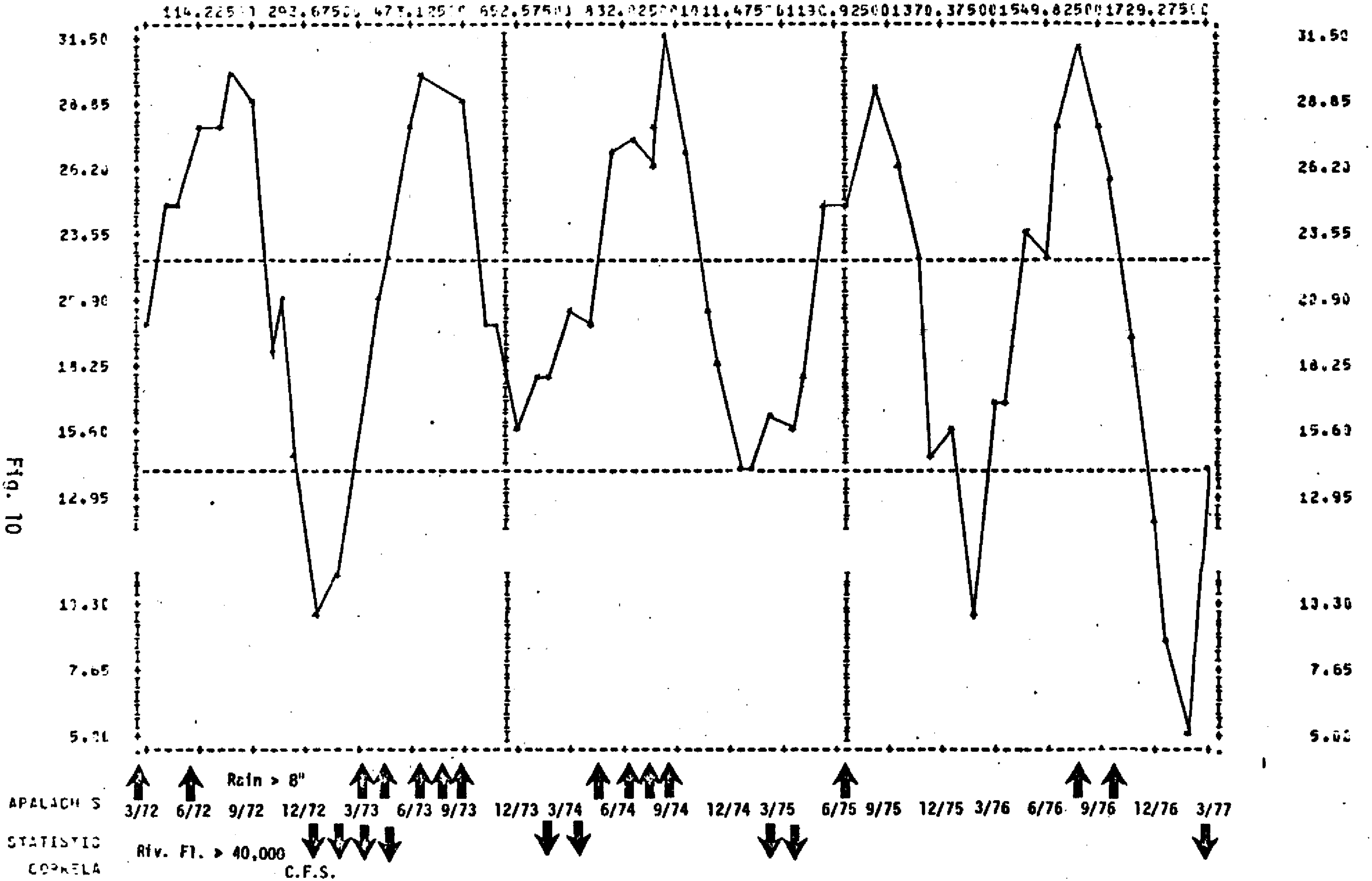

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STATISTIC
  CORRELA
  STD ERR OF EST -      6.27119       INTERCEPT (A)   -      24.33222       STD ERROR OF A -      1.06159
  SIGNIFICANCE A -      [illegible]   SLOPE (B)       -      -.00297        STD ERROR OF B -      .00156
  SIGNIFICANCE B -      [illegible]
  PLOTTED VALUES -      59            EXCLUDED VALUES-       0              MISSING VALUES -      47

              ******** IS PRINTED IF A COEFFICIENT CANNOT BE COMPUTED.
```

Fig. 10

Fig. 11: LOCAL AIR TEMPERATURE (APALACHICOLA), MONTHLY MEANS FROM 1970-1977, °C

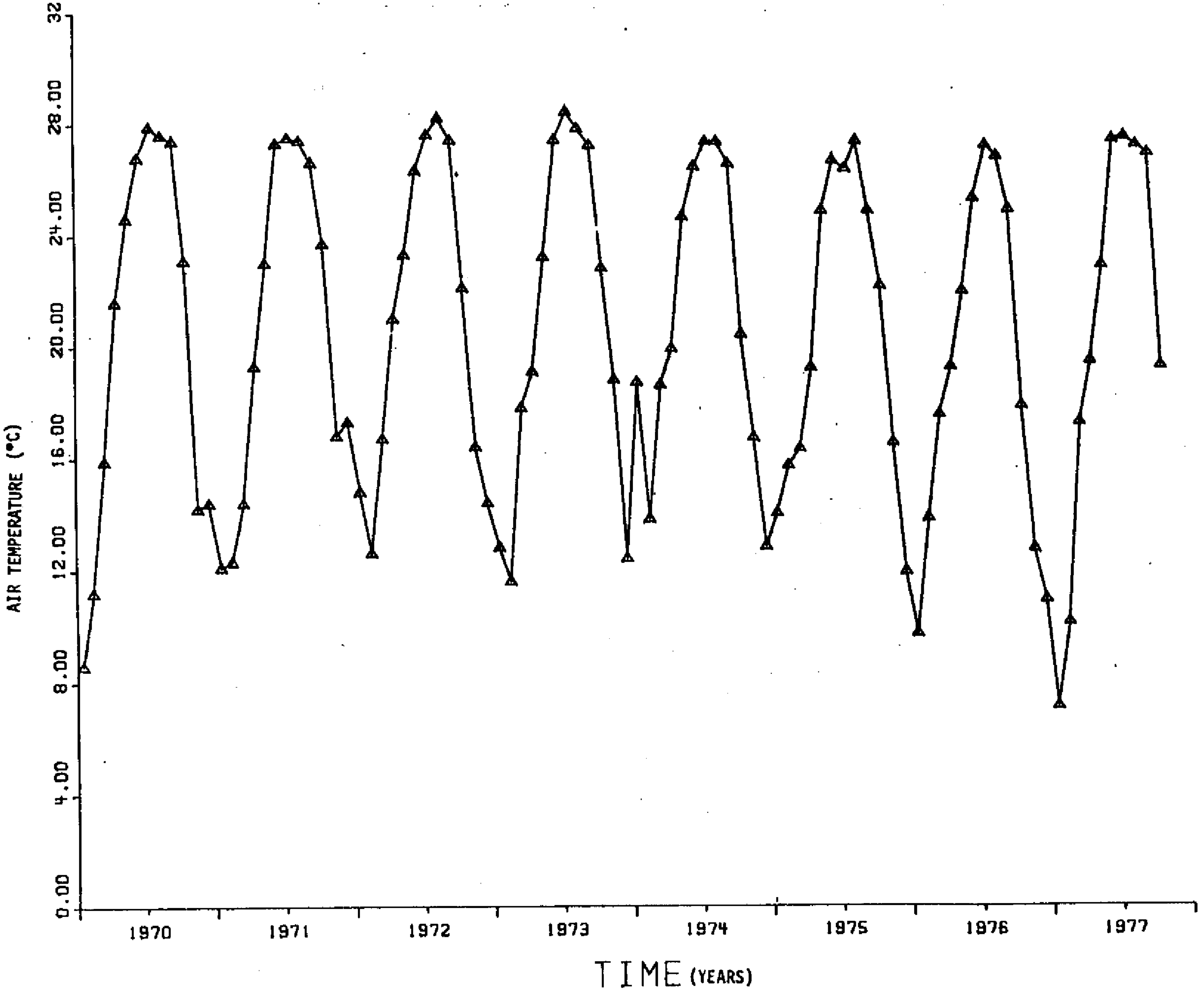
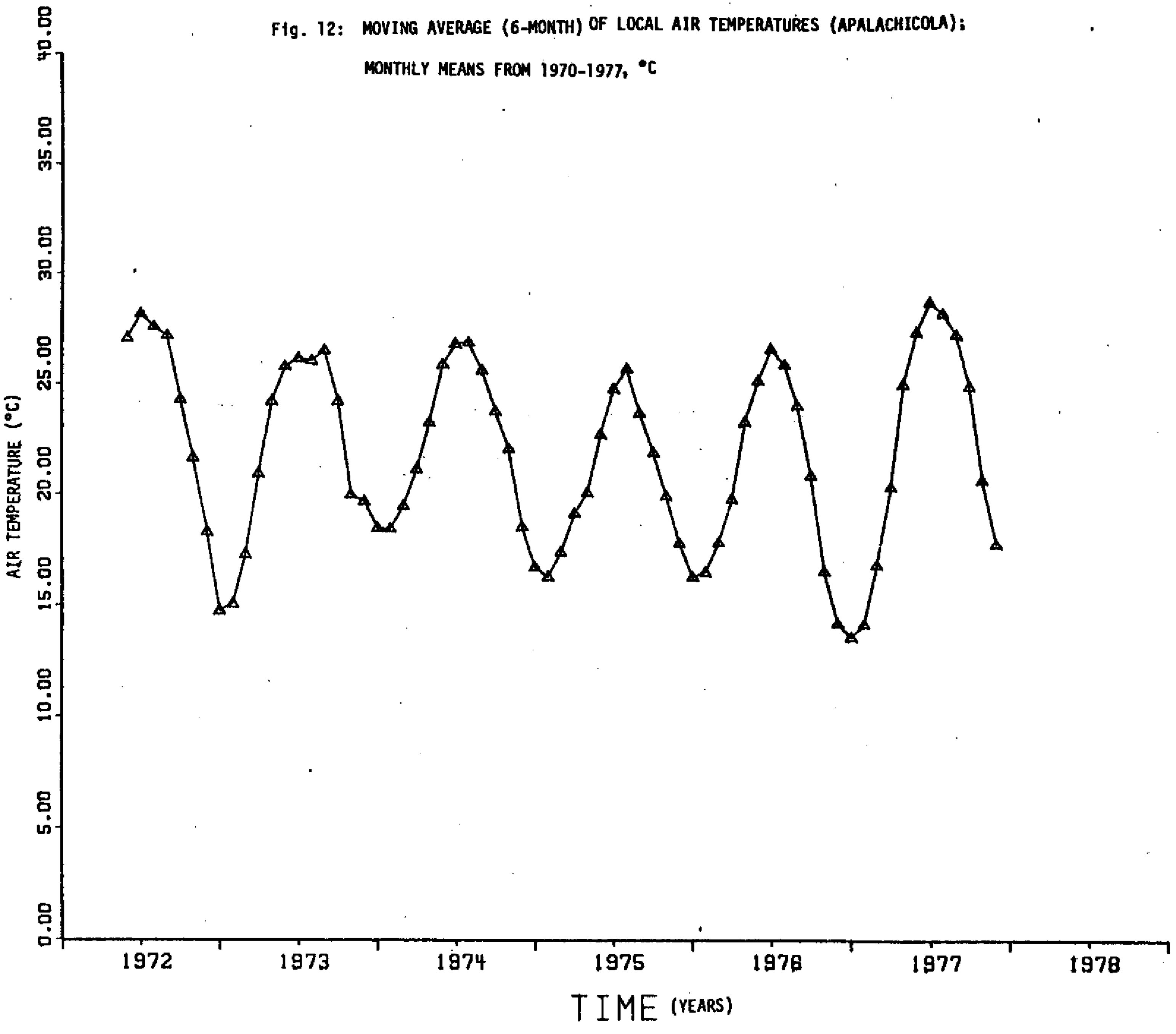
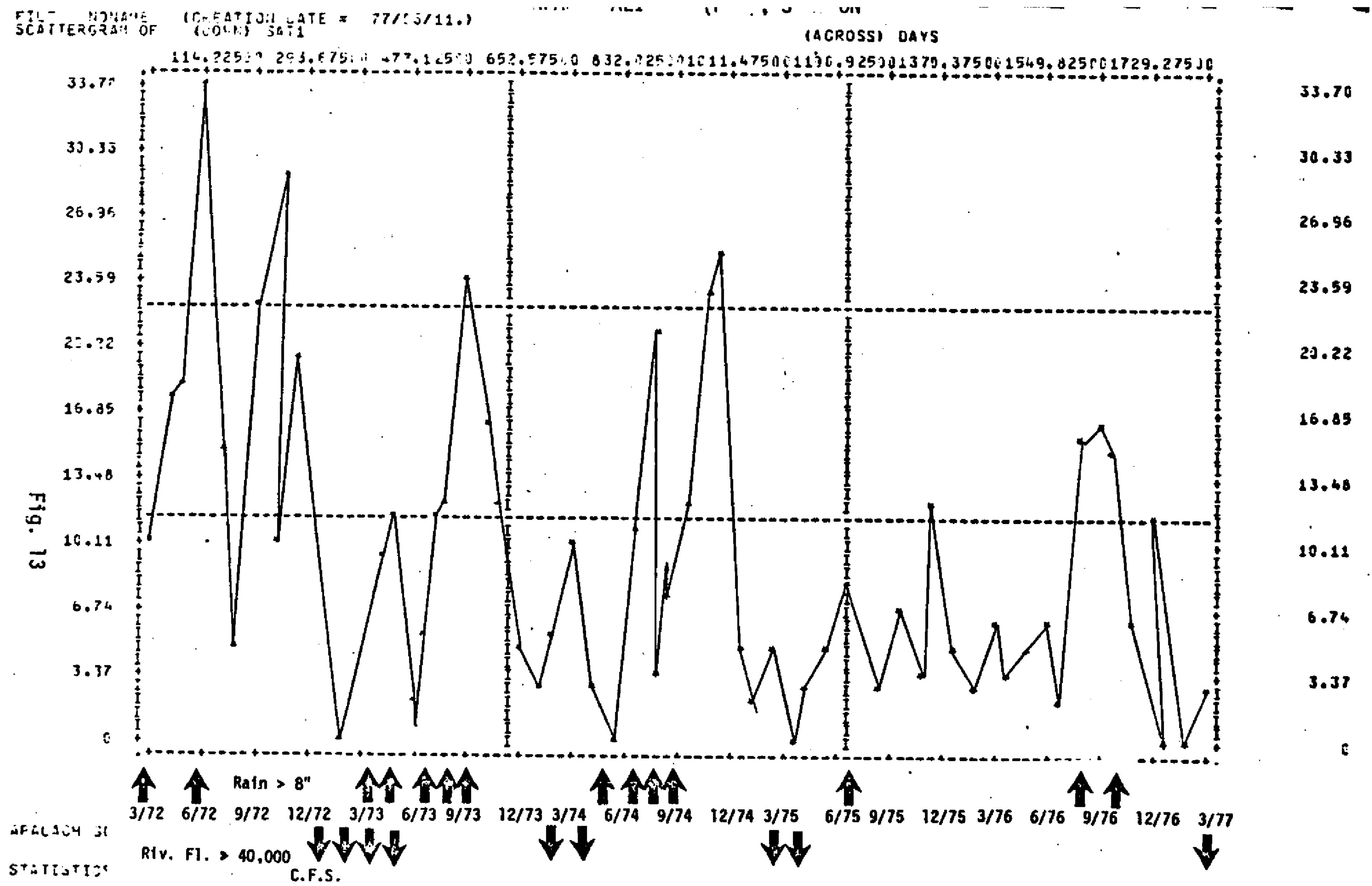


Fig. 12: MOVING AVERAGE (6-MONTH) OF LOCAL AIR TEMPERATURES (APALACHICOLA);

MONTHLY MEANS FROM 1970-1977, °C





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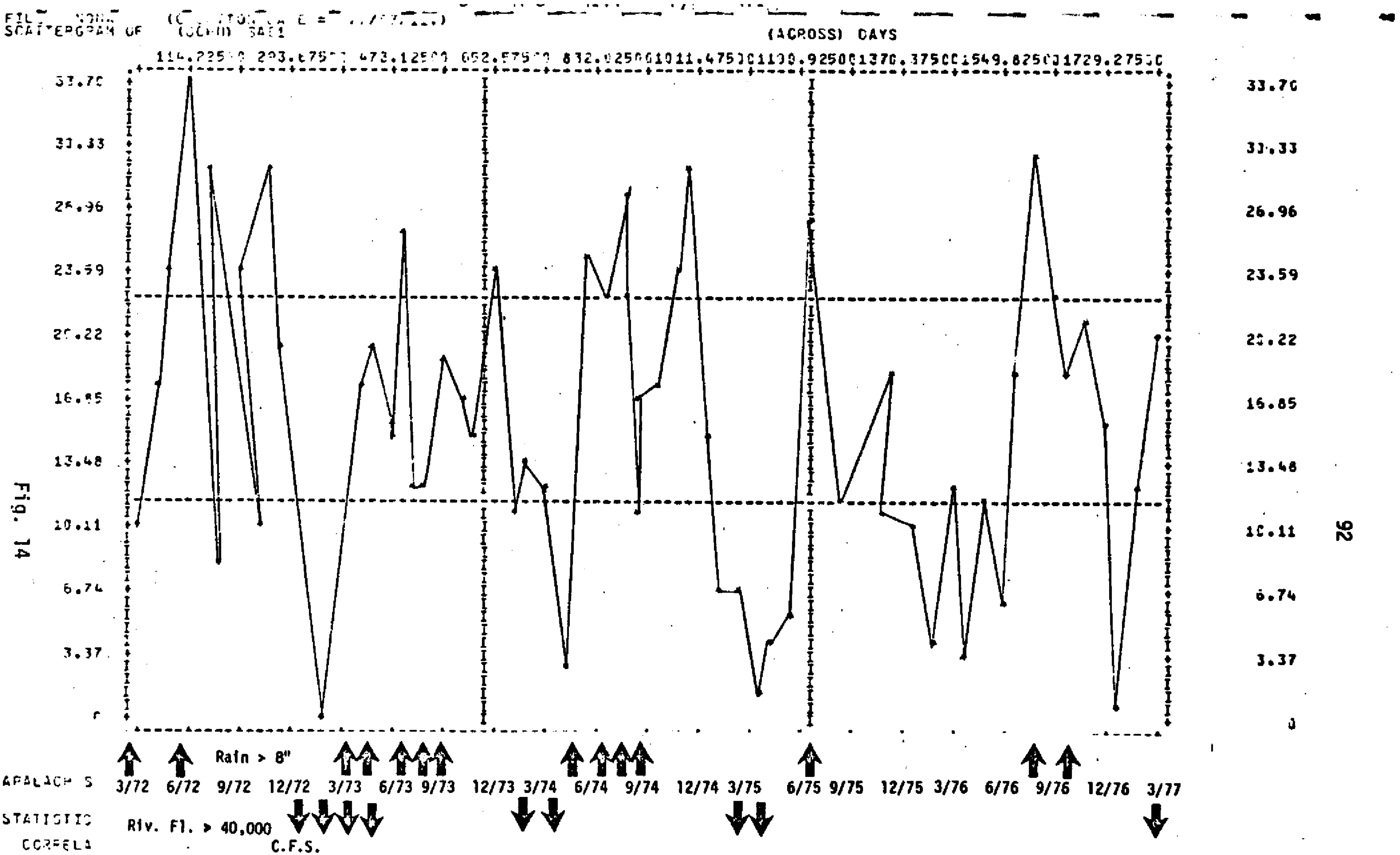
STATISTICS
  CORRELATION (R)-        -.39376     R SQUARED       -        .15506     SIGNIFICANCE R -      .00085
  STD ERR OF EST -        7.25242     INTERCEPT (A)   -      15.64093     STD ERROR OF A -     1.88468
  SIGNIFICANCE A -         .00001     SLOPE (B)       -       -.00593     STD ERROR OF B -      .00179
  SIGNIFICANCE B -         .00165
  PLOTTED VALUES -        61          EXCLUDED VALUES-      0             MISSING VALUES -     44
```

Fig. 13



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CORRELA
STD ERR OF EST -          8.04624        INTERCEPT (A)     -        19.11606       STD ERROR OF A -        2.09143
SIGNIFICANCE A -           .0001         SLOPE (B)         -         -.00347       STD ERROR OF B -         .00200
SIGNIFICANCE B -           .04411
PLOTTED VALUES -          6[illegible]   EXCLUDED VALUES-              0           MISSING VALUES -        45


        ******** IS PRINTED IF A COEFFICIENT CANNOT BE COMPUTED.
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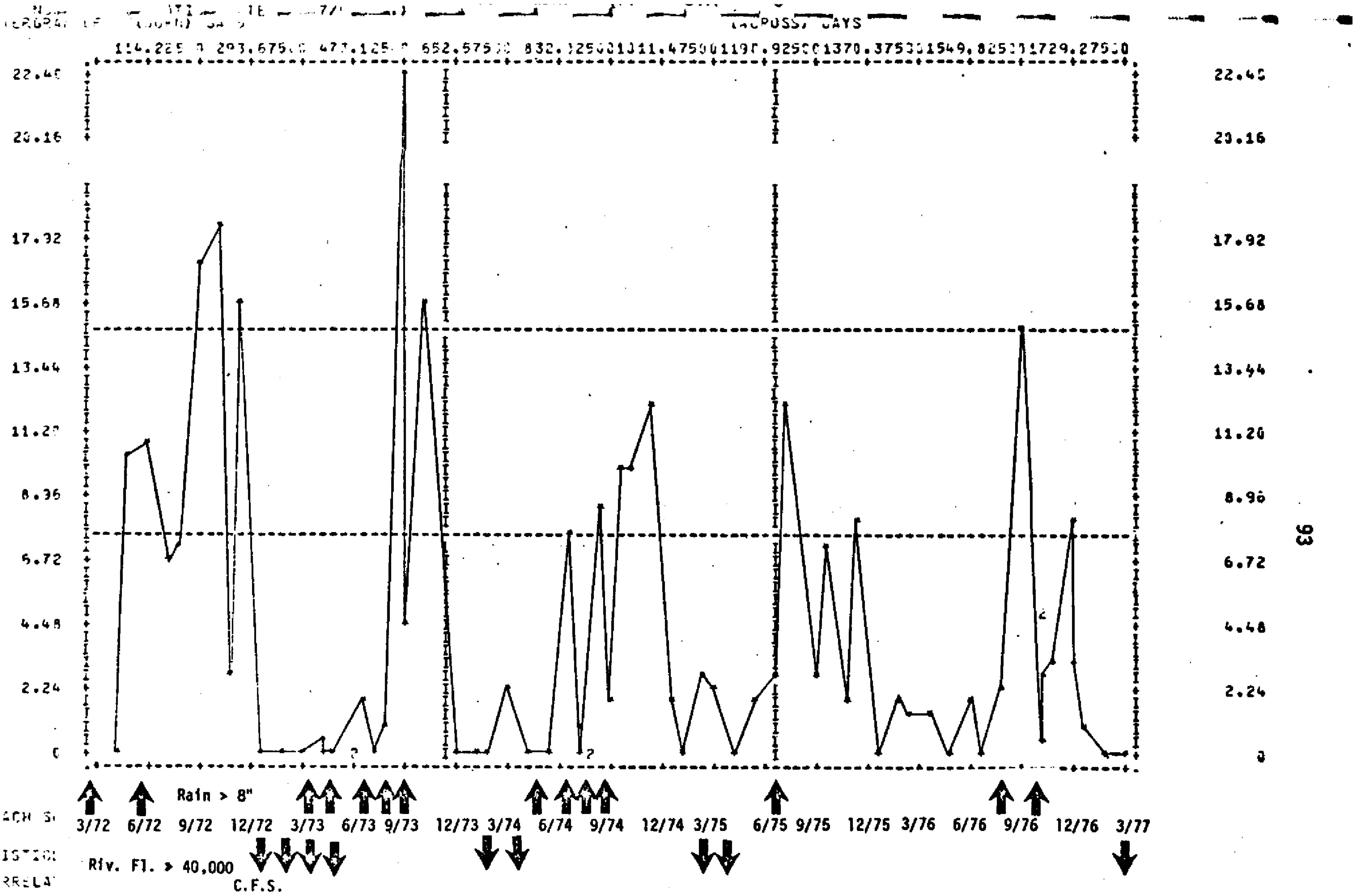
Fig. 14



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STATISTICS
  CORRELA
  STD ERR OF EST -        9.37693         INTERCEPT (A)   -          6.86213       STD ERROR OF A -          1.33902
  SIGNIFICANCE A -         .00011         SLOPE (B)       -          -.00196       STD ERROR OF B -           .00123
  SIGNIFICANCE B -         . 5851
  PLOTTED VALUES -        7               EXCLUDED VALUES-           0             MISSING VALUES -          35

            ******** IS PRINTED IF A COEFFICIENT CANNOT BE COMPUTED.
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Fig. 15

```
F  E  MONAME   (CREATION DATE = ///7/31.)      OM     NITL  FI     ATE   D
SCATTERGRAM OF    (DOWN) SAPS                                        (ACROSS) DAYS
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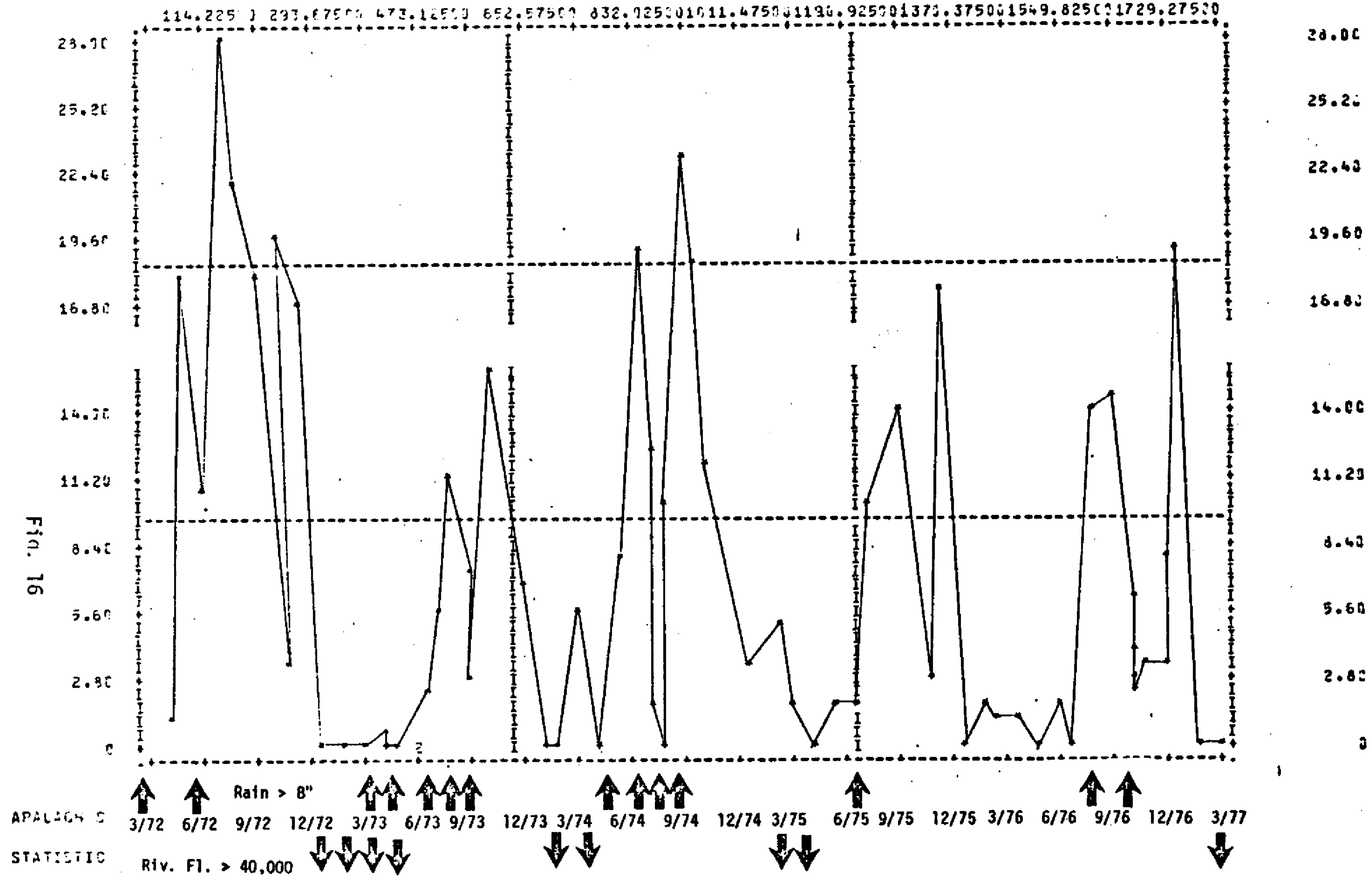

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  CORRELA
  STD ERR OF EST -          7.37546        INTERCEPT (A)   -          9.48788        STD ERROR OF A -          1.54479
  SIGNIFICANCE A -           .00001        SLOPE (B)       -          -.00279        STD ERROR OF B -           .00170
  SIGNIFICANCE B -           .05288
  PLOTTED VALUES -         66              EXCLUDED VALUES-           0              MISSING VALUES -          39

               ******** IS PRINTED IF A COEFFICIENT CANNOT BE COMPUTED.
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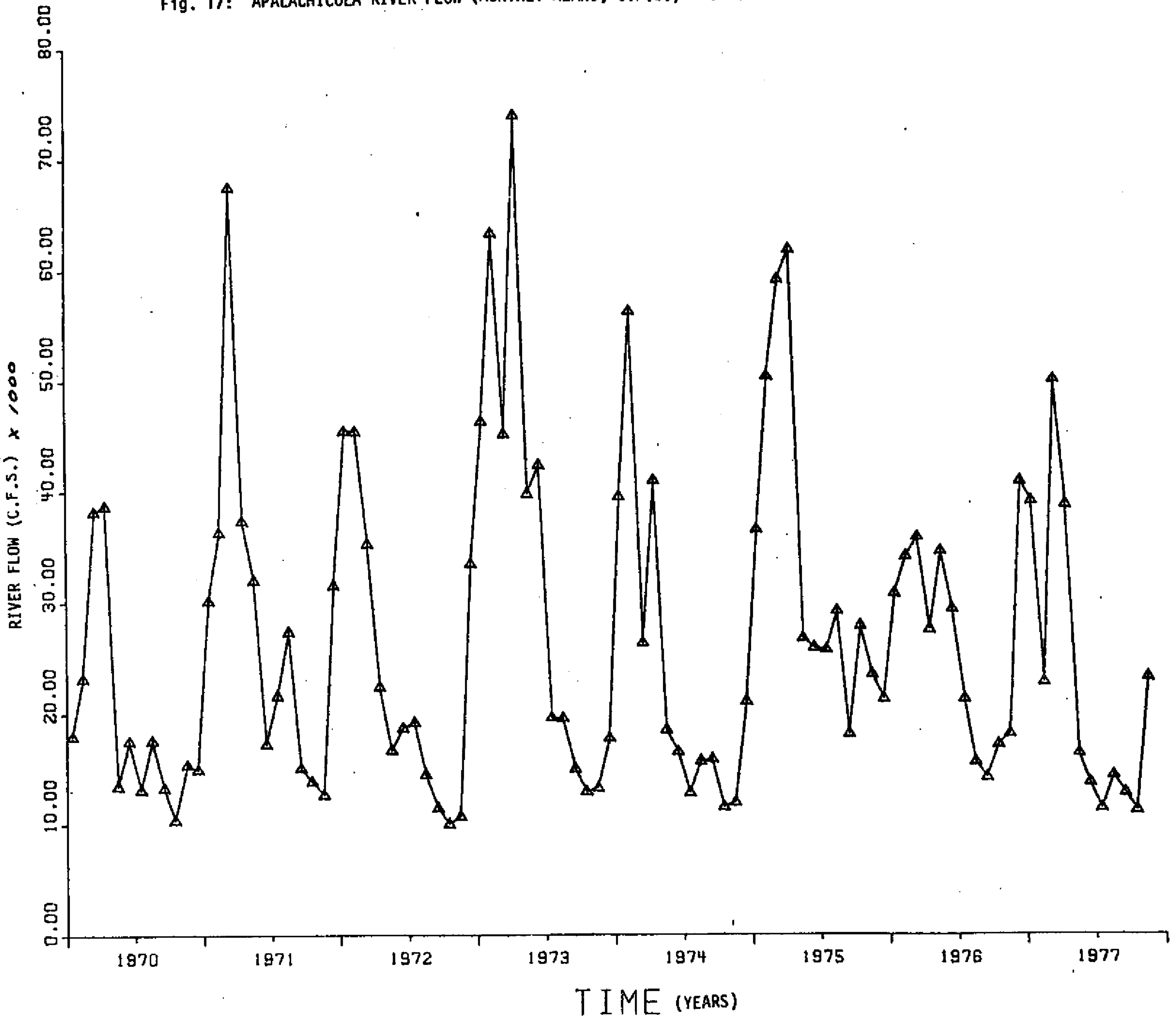
Fig. 16

Fig. 17: APALACHICOLA RIVER FLOW (MONTHLY MEANS, C.F.S.) FROM 1970-1977



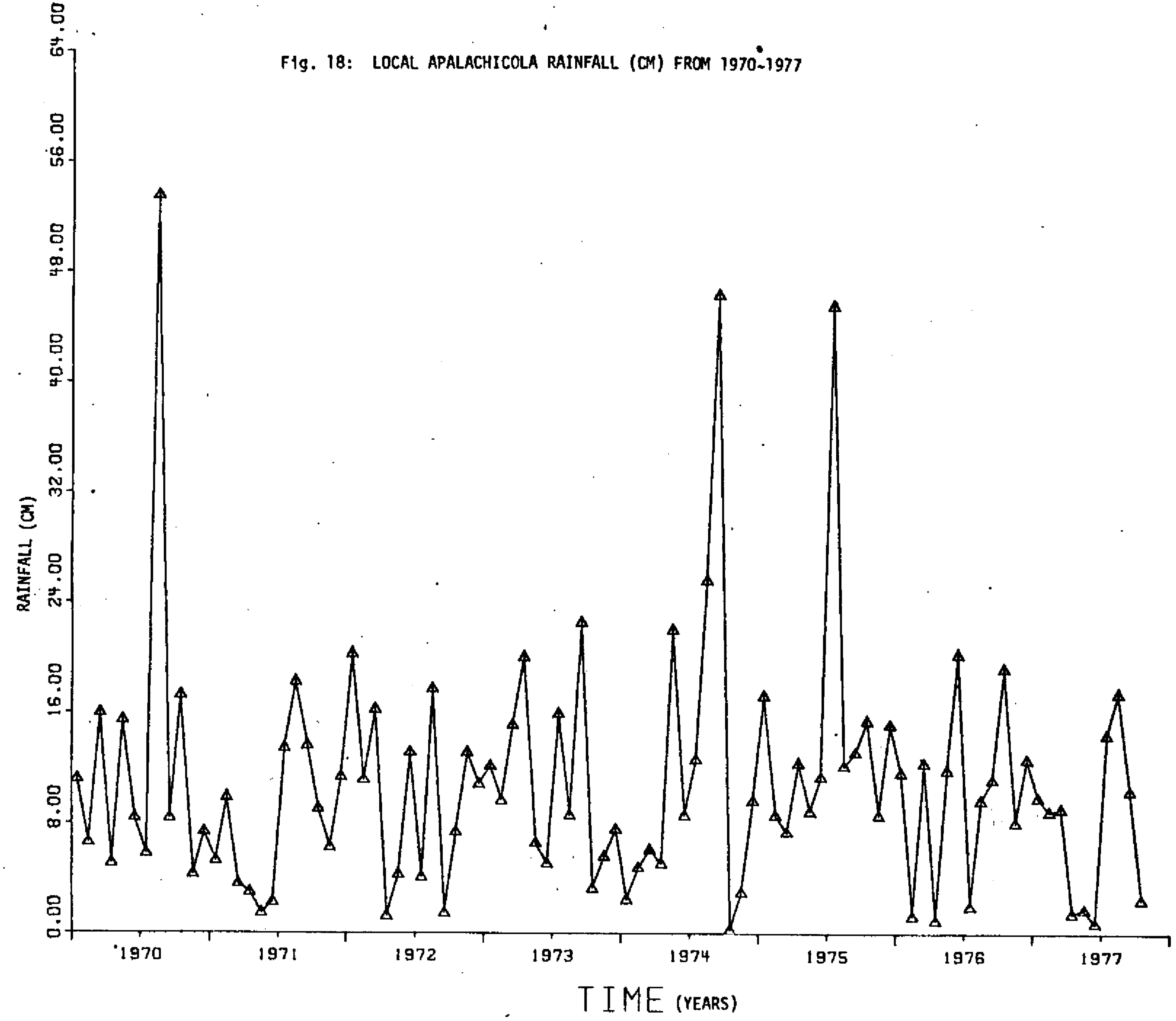


Fig. 18: LOCAL APALACHICOLA RAINFALL (CM) FROM 1970-1977

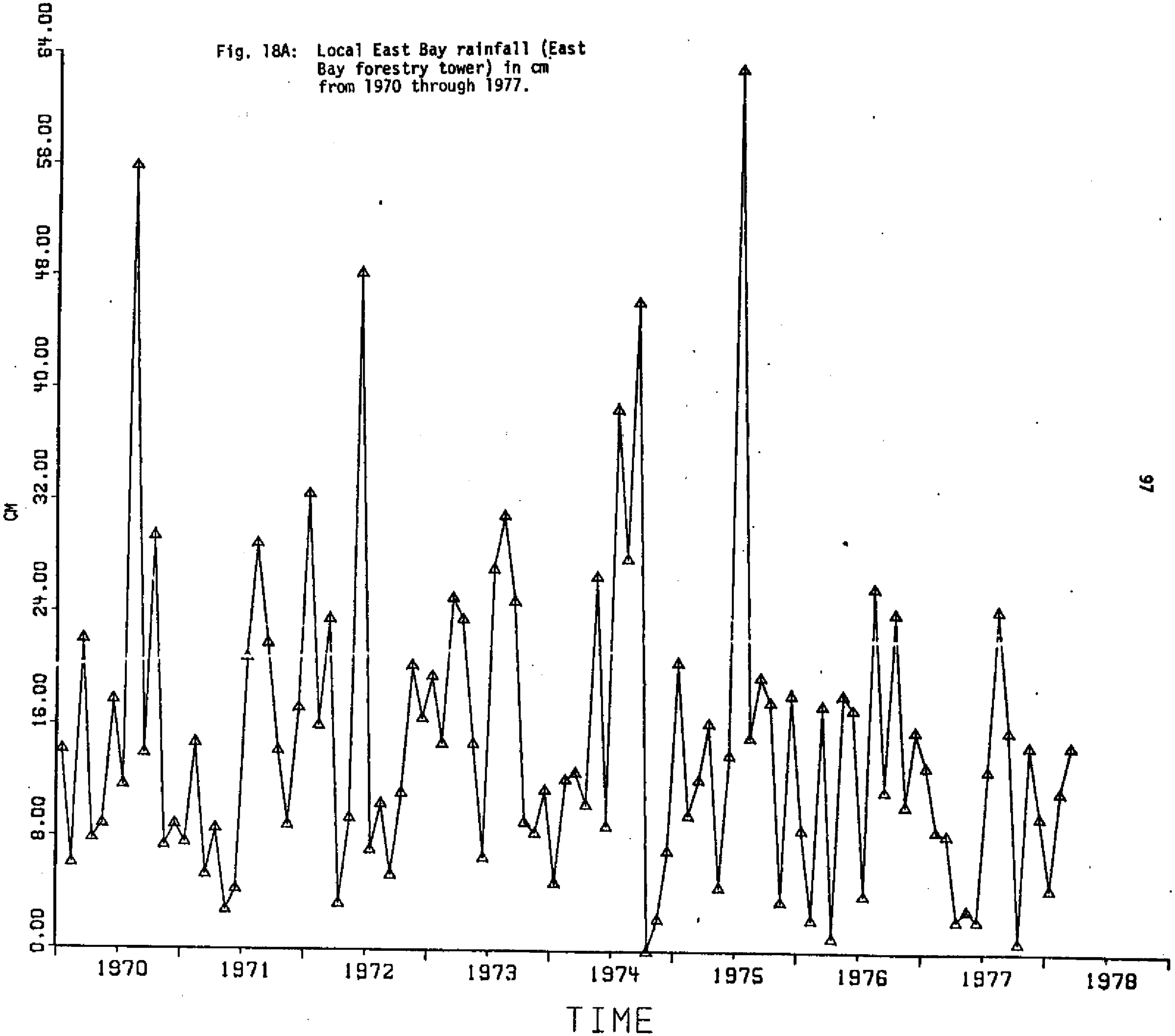


Fig. 18A: Local East Bay rainfall (East Bay forestry tower) in cm from 1970 through 1977.

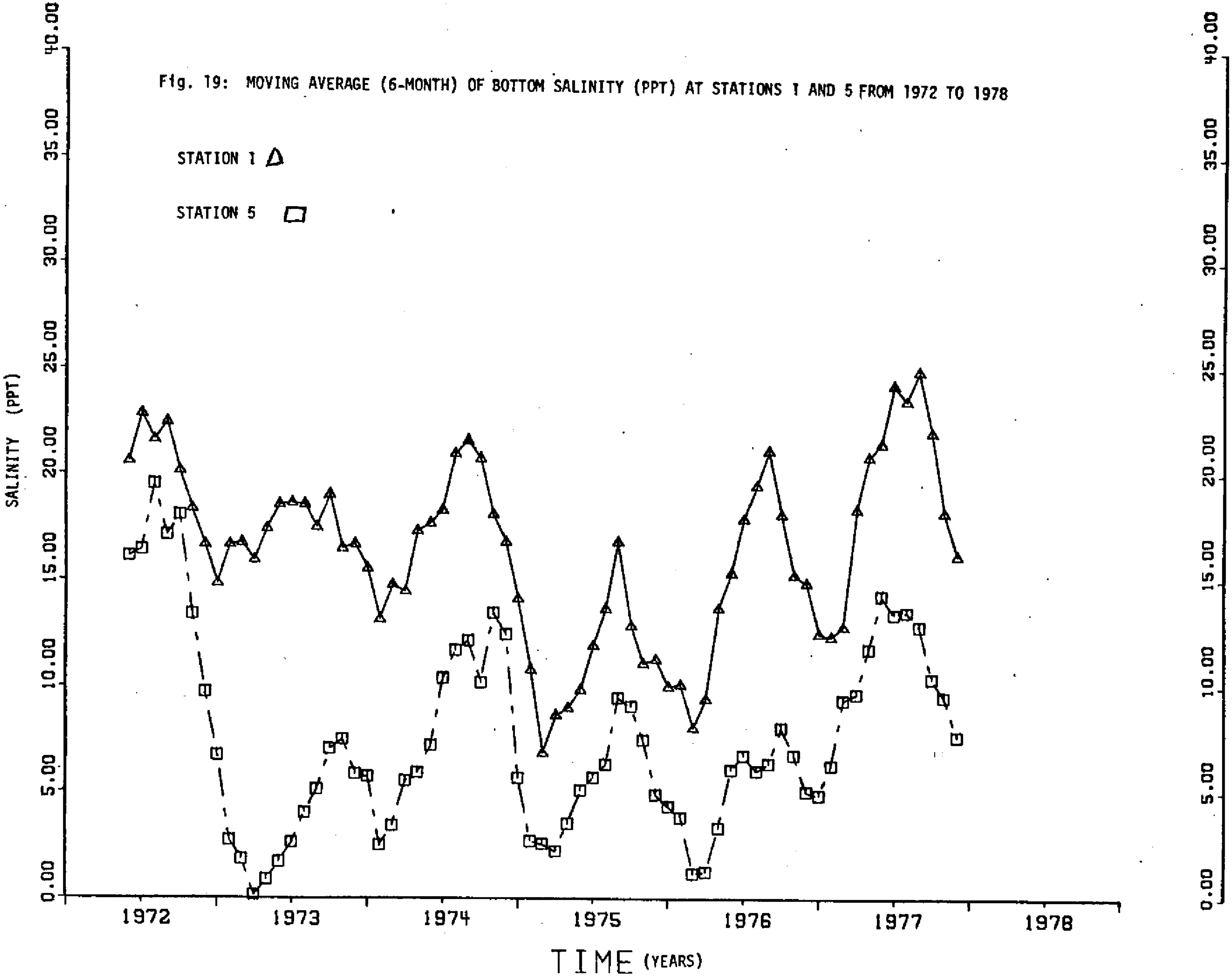


Fig. 19: MOVING AVERAGE (6-MONTH) OF BOTTOM SALINITY (PPT) AT STATIONS 1 AND 5 FROM 1972 TO 1978

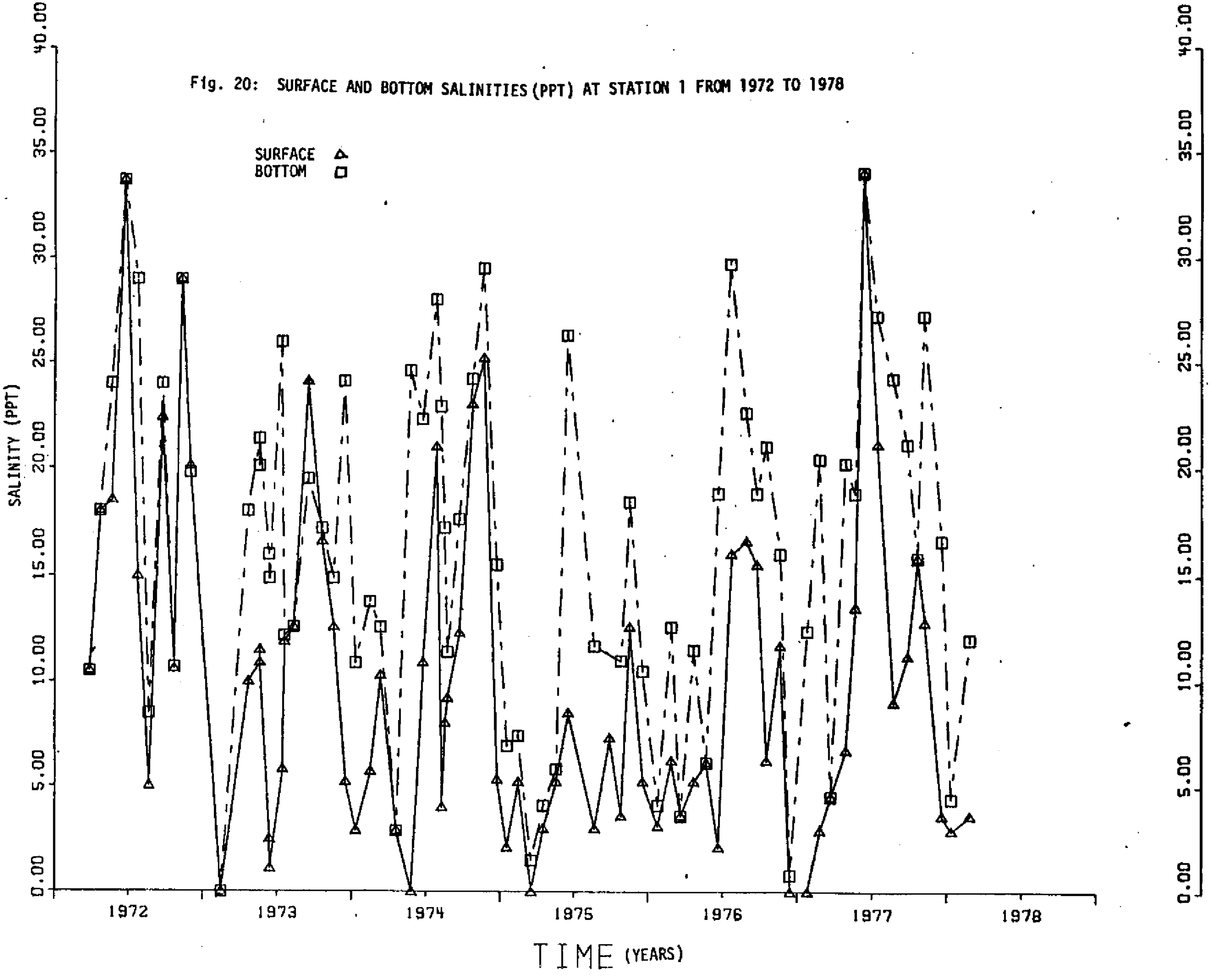


Fig. 20: SURFACE AND BOTTOM SALINITIES (PPT) AT STATION 1 FROM 1972 TO 1978

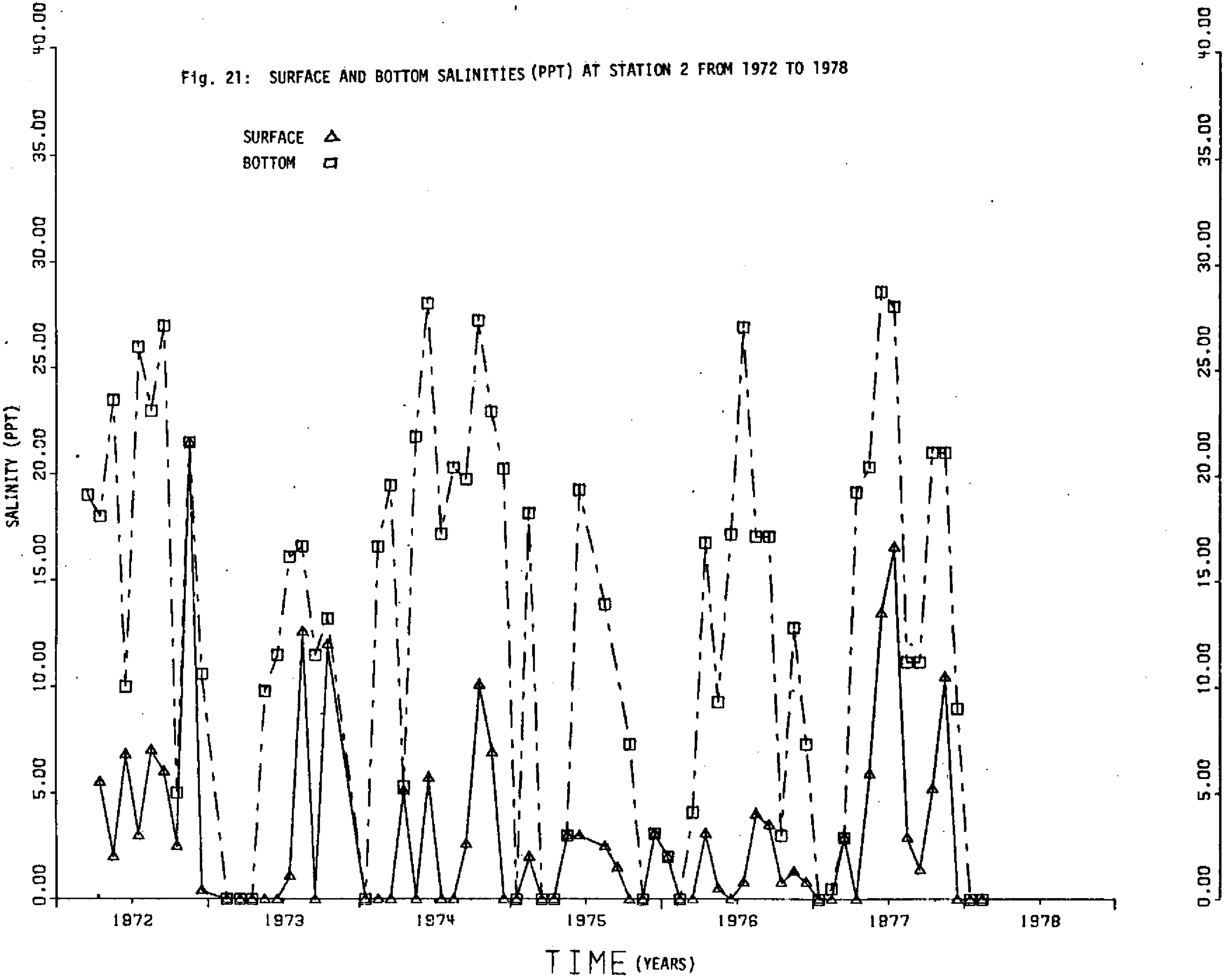


Fig. 21: SURFACE AND BOTTOM SALINITIES (PPT) AT STATION 2 FROM 1972 TO 1978

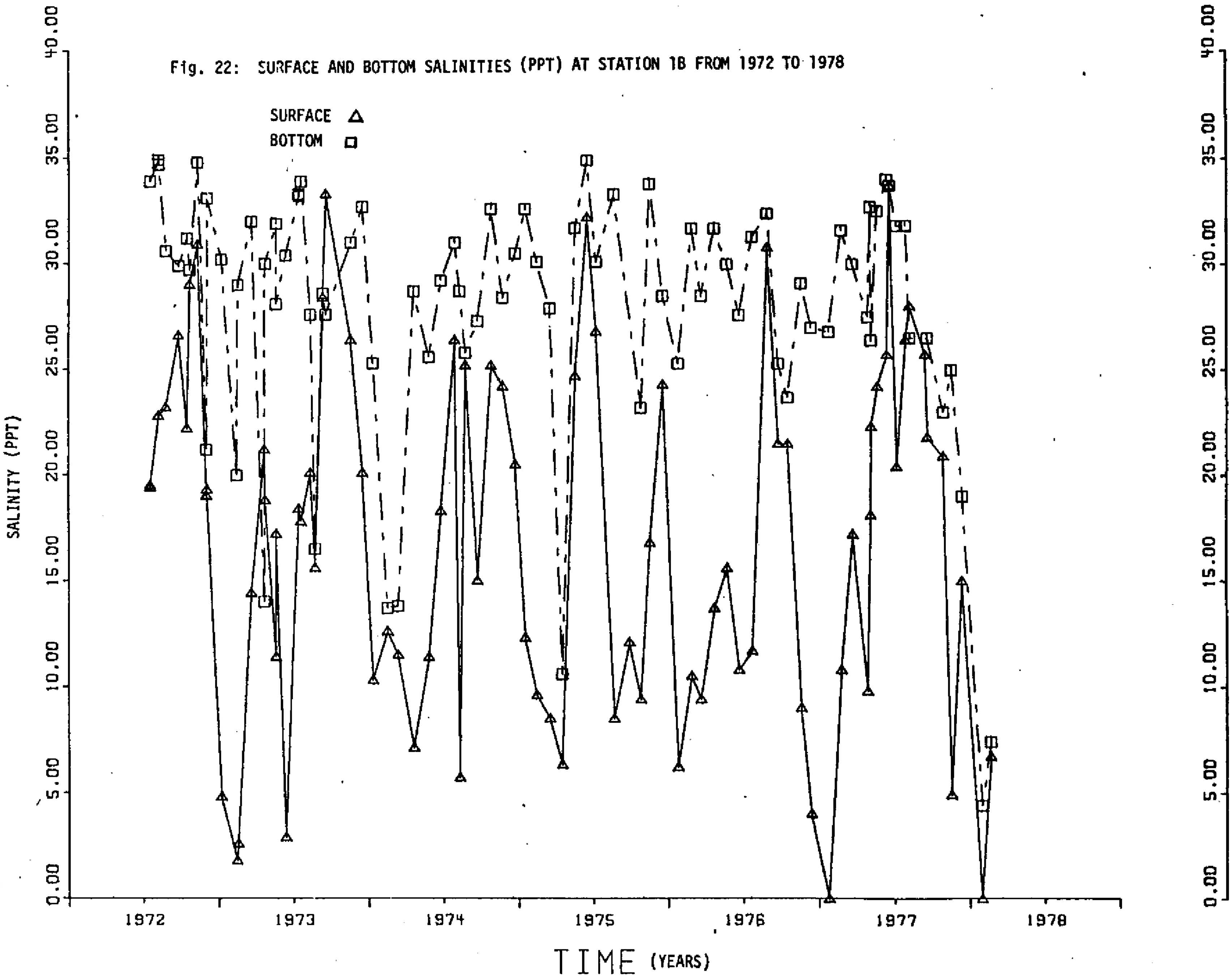


Fig. 22: SURFACE AND BOTTOM SALINITIES (PPT) AT STATION 1B FROM 1972 TO 1978

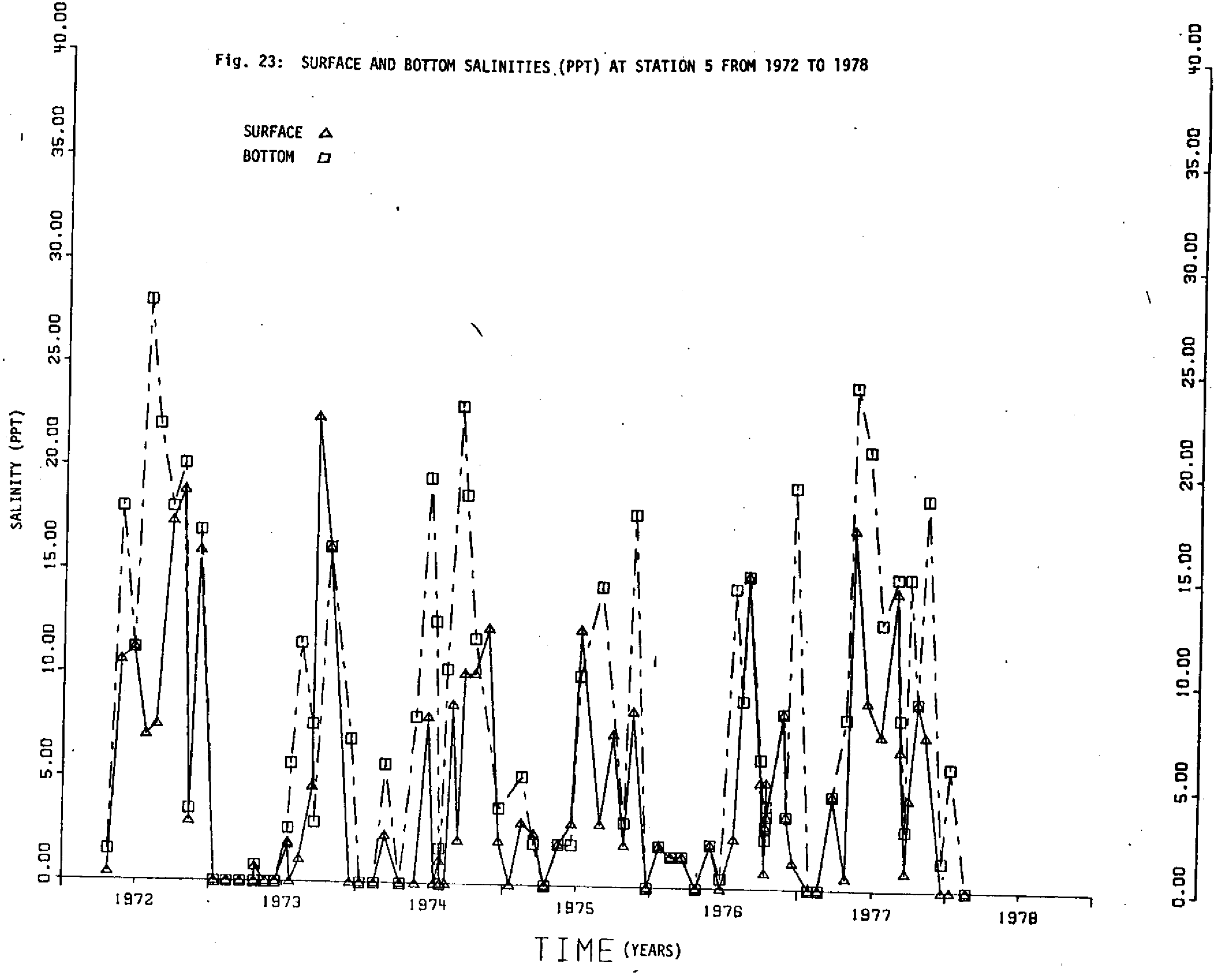


Fig. 23: SURFACE AND BOTTOM SALINITIES (PPT) AT STATION 5 FROM 1972 TO 1978

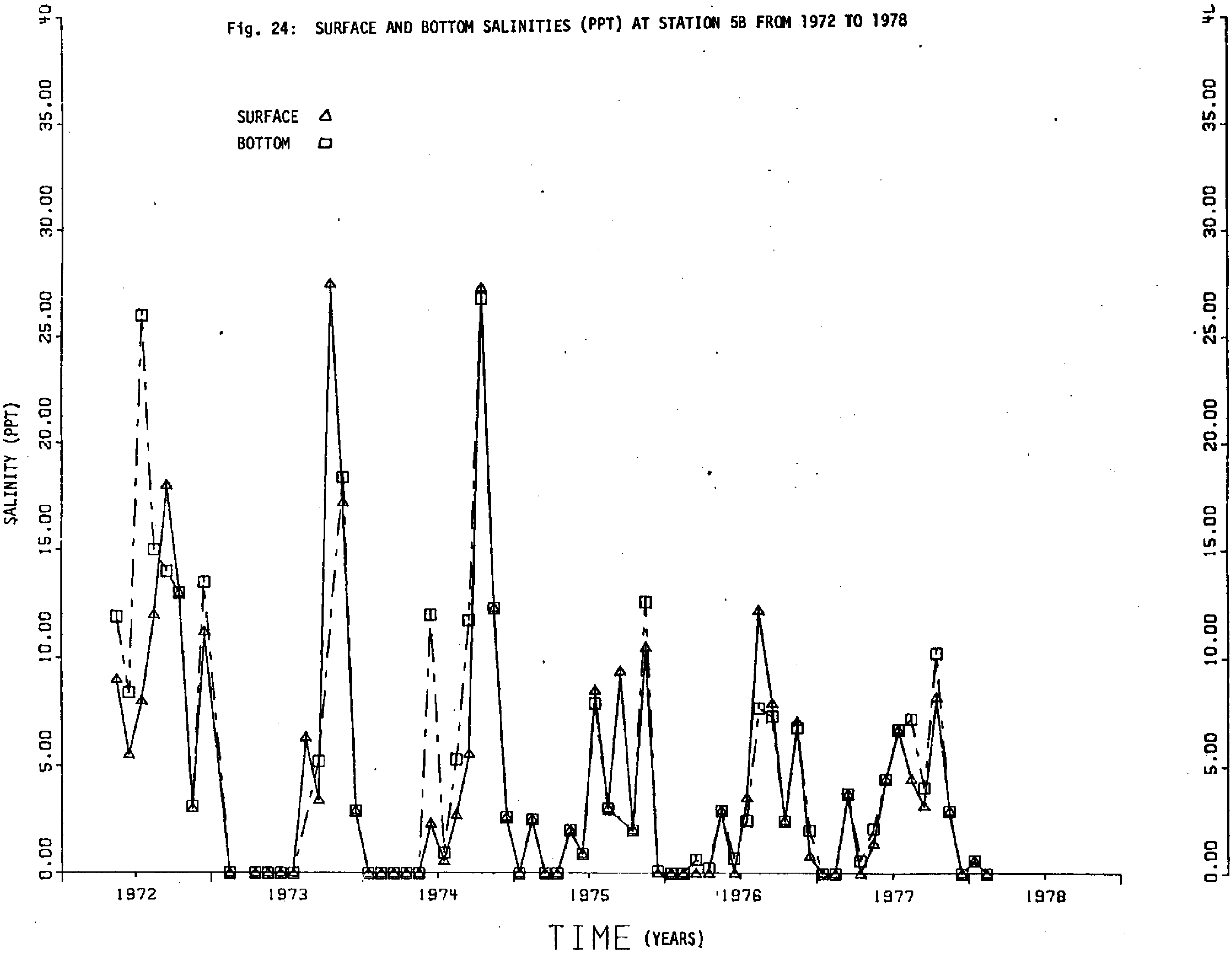
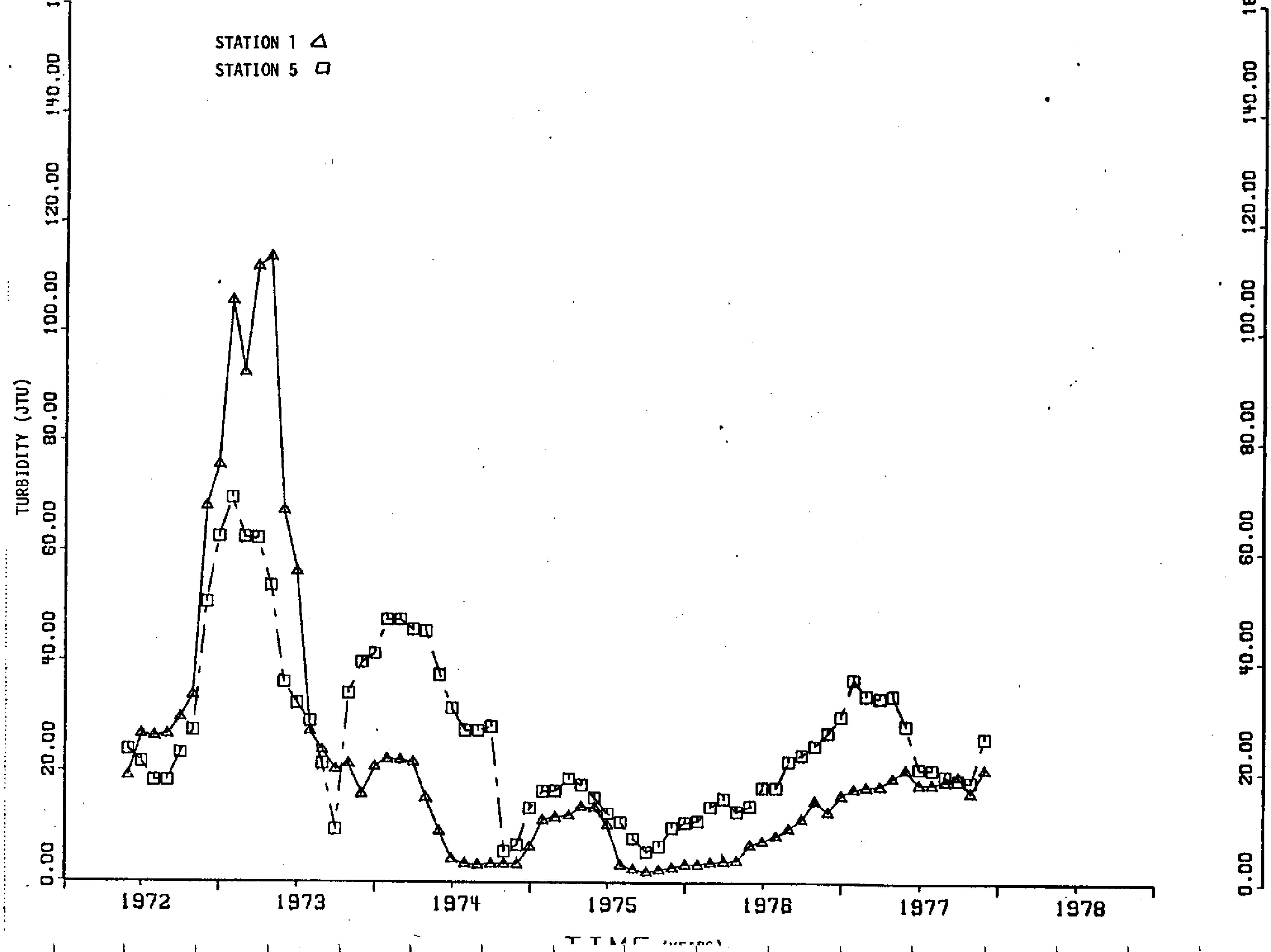


Fig. 24: SURFACE AND BOTTOM SALINITIES (PPT) AT STATION 5B FROM 1972 TO 1978

Fig. 25: MOVING AVERAGE (6-MONTH) OF BOTTOM TURBIDITY (JTU) AT STATIONS 1 AND 5 FROM 1972 TO 1978



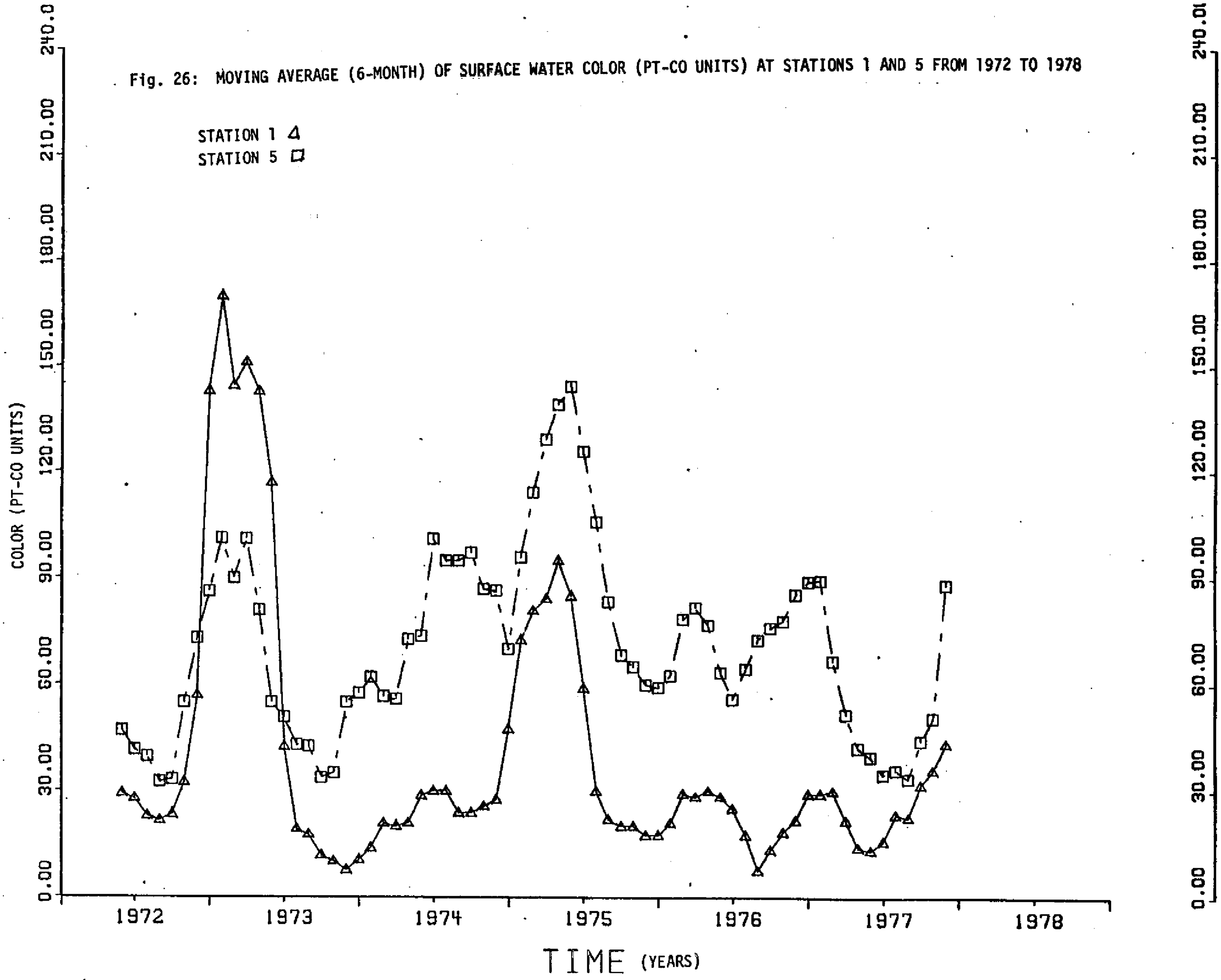


Fig. 26: MOVING AVERAGE (6-MONTH) OF SURFACE WATER COLOR (PT-CO UNITS) AT STATIONS 1 AND 5 FROM 1972 TO 1978

Fig. 27: MOVING AVERAGE (6-MONTH) OF SECCHI READINGS (M) AT STATIONS 1 AND 5 FROM 1972 TO 1978

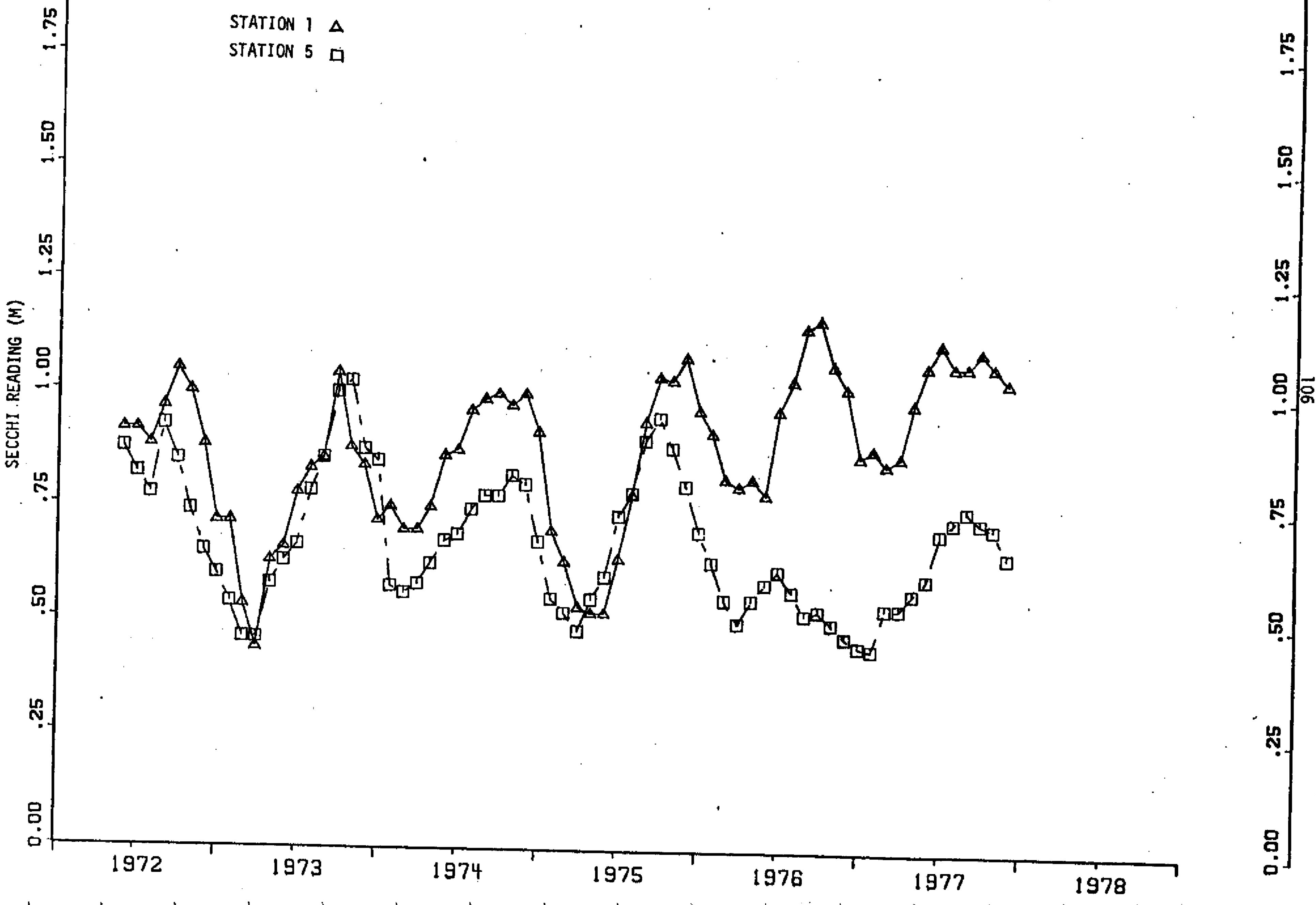


Fig. 28: WATER COLOR (SURFACE AND BOTTOM; PT-CO UNITS) AT STATION 5A FROM 1972 TO 1978

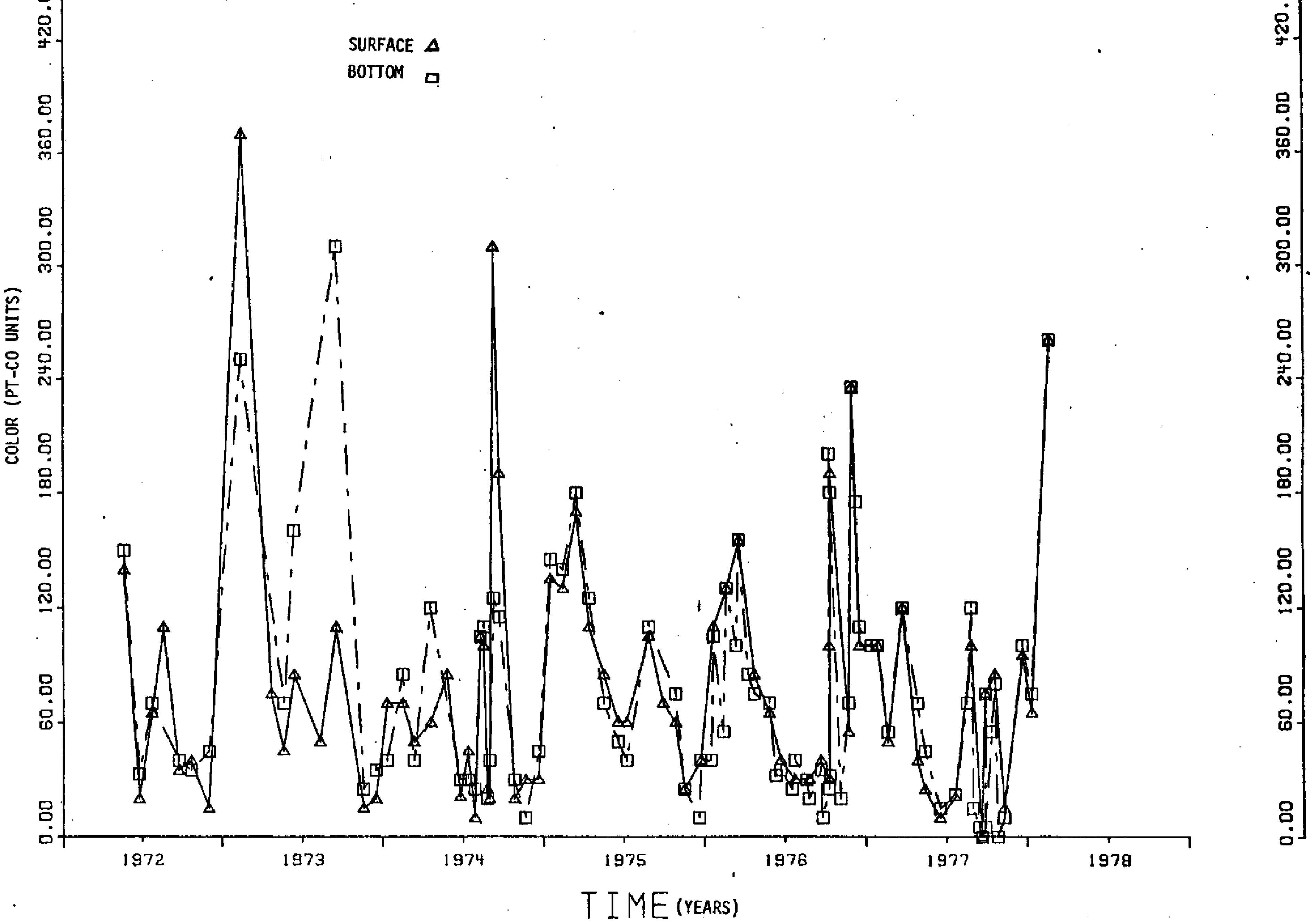
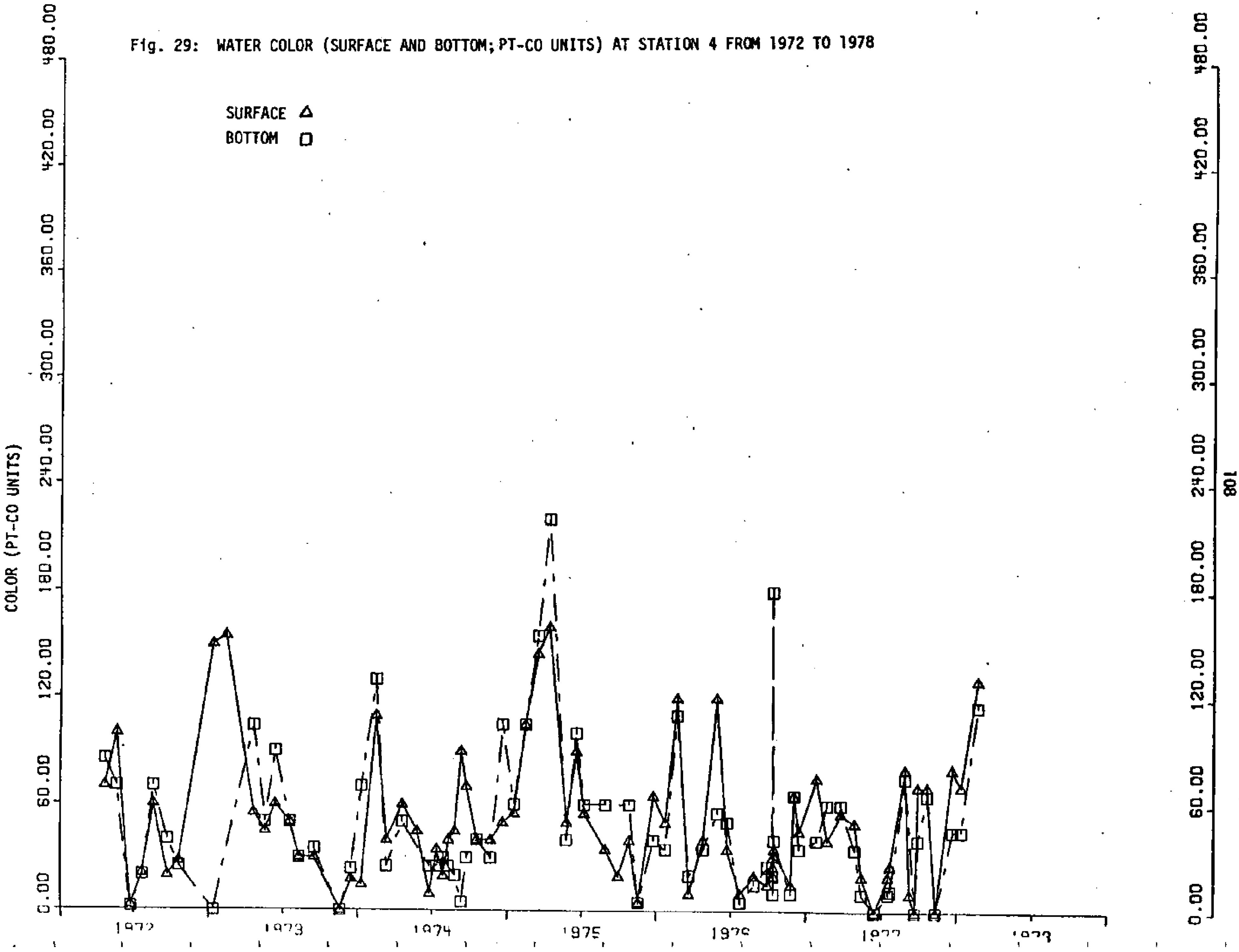


Fig. 29: WATER COLOR (SURFACE AND BOTTOM; PT-CO UNITS) AT STATION 4 FROM 1972 TO 1978



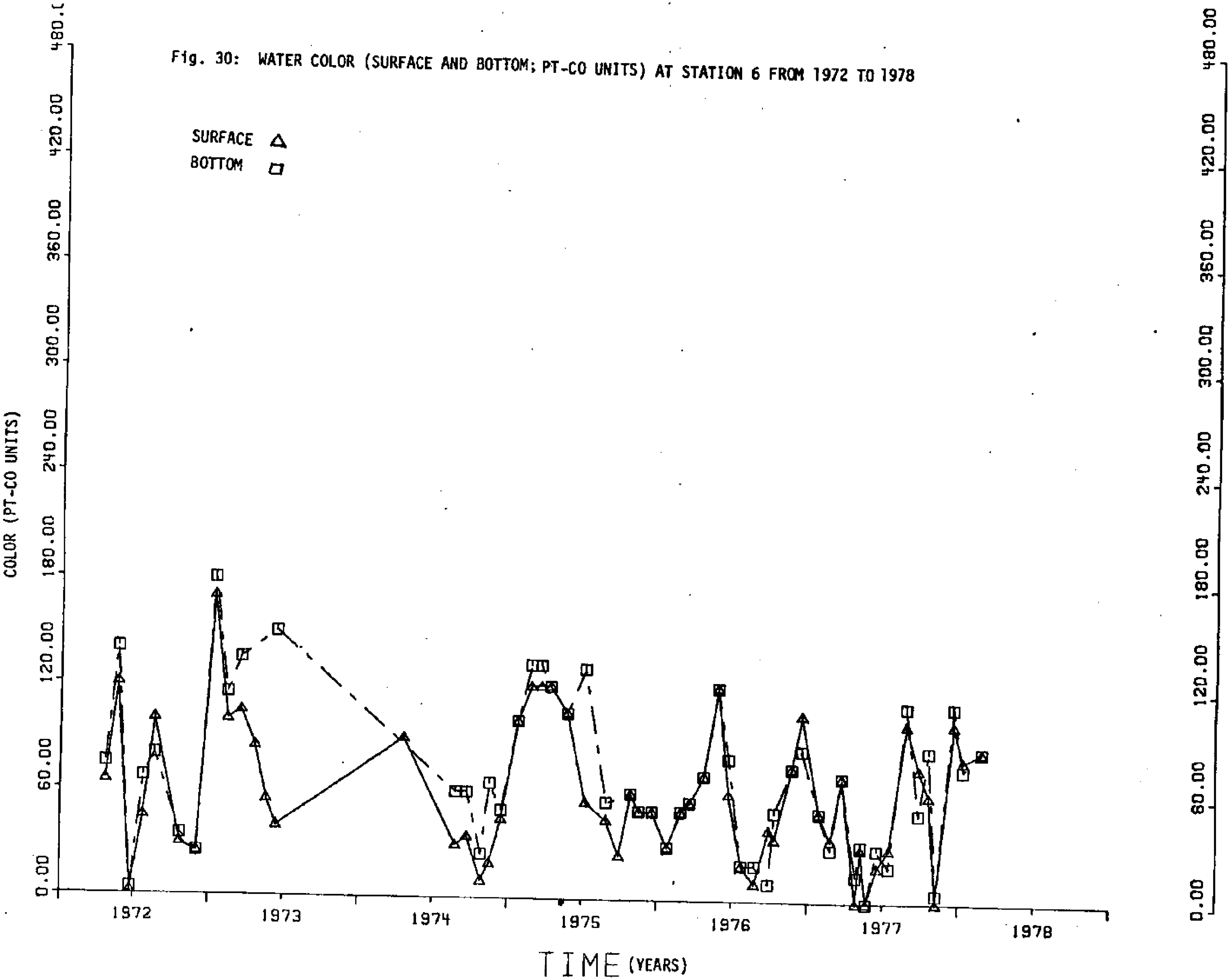


Fig. 30: WATER COLOR (SURFACE AND BOTTOM; PT-CO UNITS) AT STATION 6 FROM 1972 TO 1978

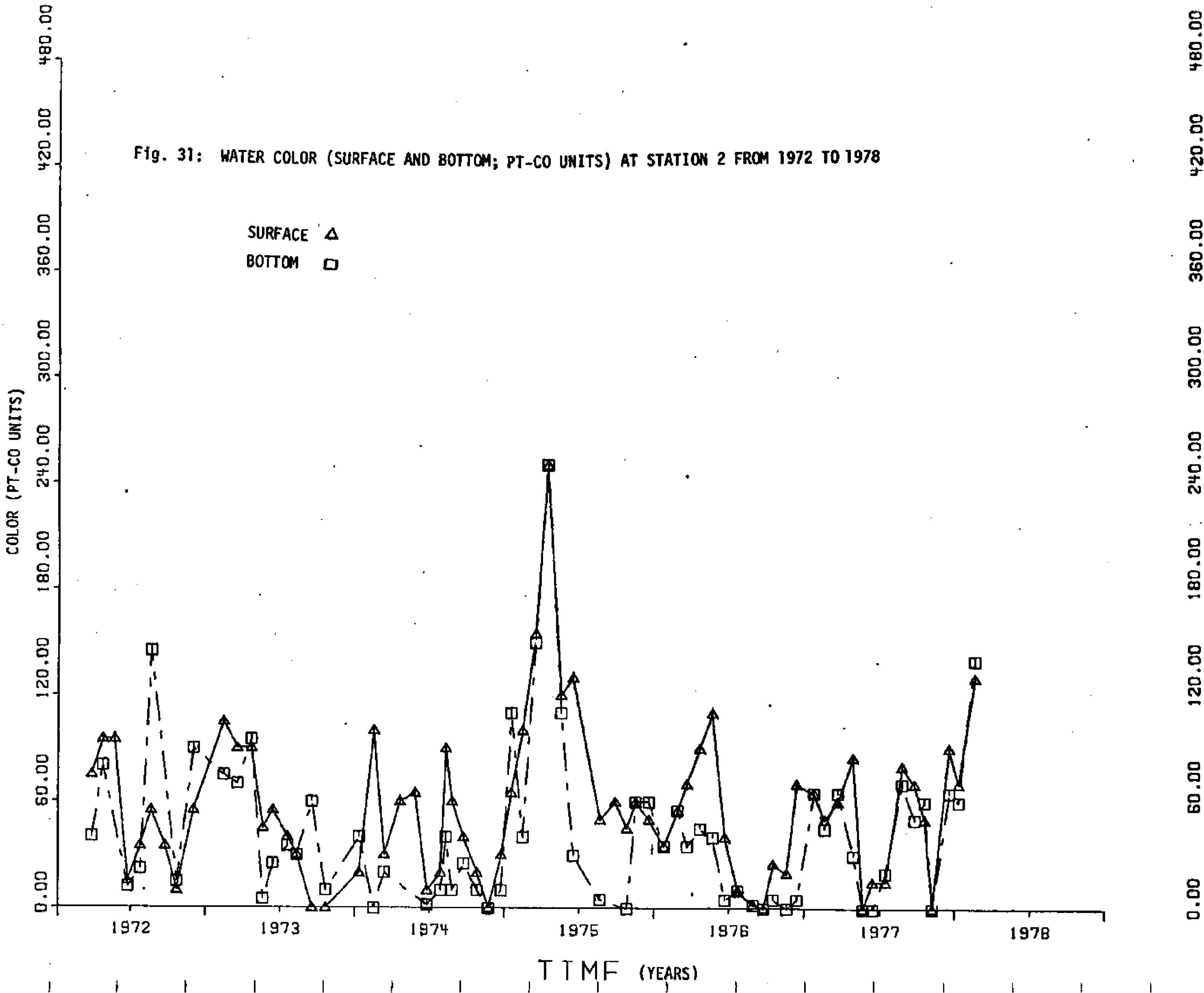
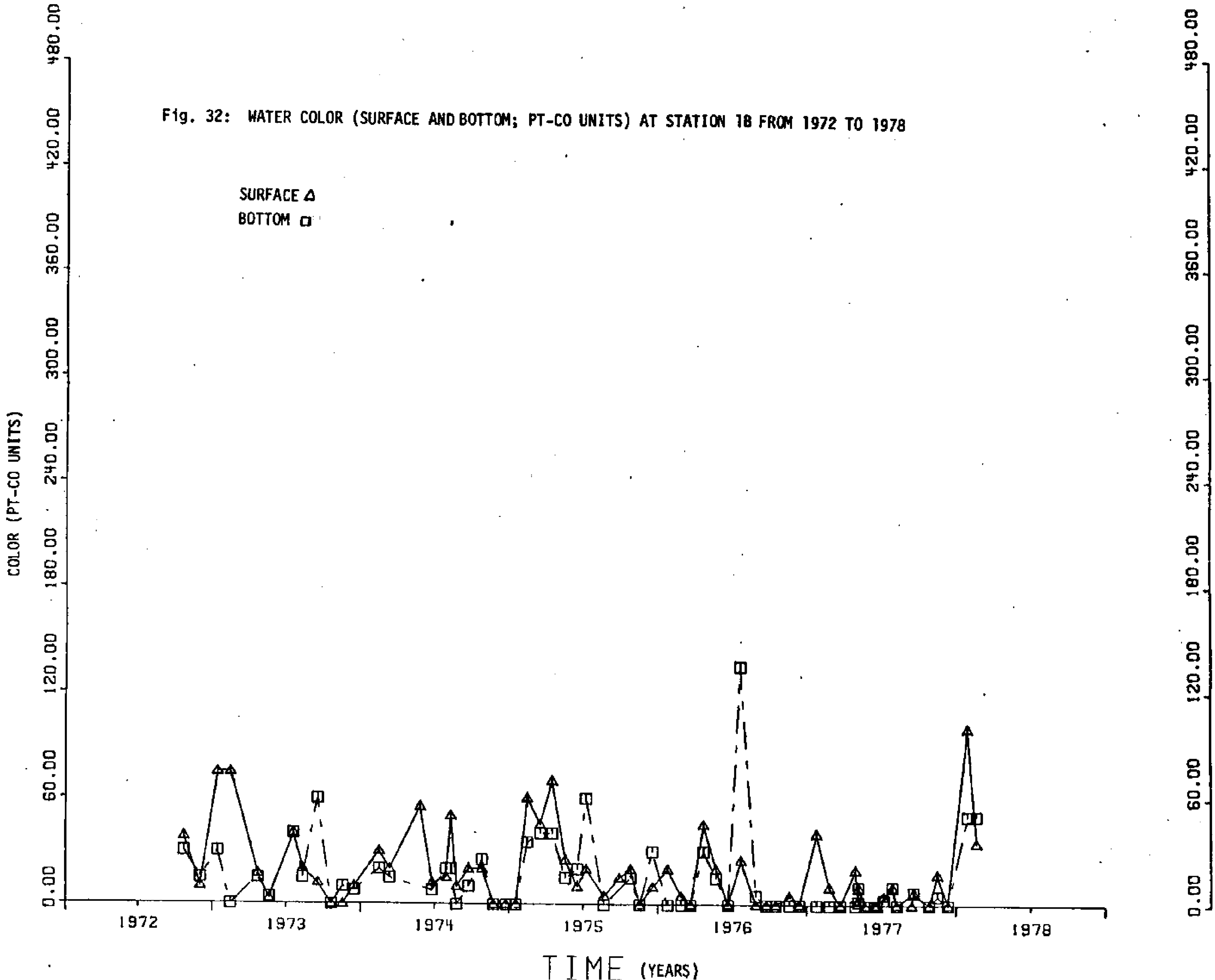


Fig. 31: WATER COLOR (SURFACE AND BOTTOM; PT-CO UNITS) AT STATION 2 FROM 1972 TO 1978

Fig. 32: WATER COLOR (SURFACE AND BOTTOM; PT-CO UNITS) AT STATION 18 FROM 1972 TO 1978



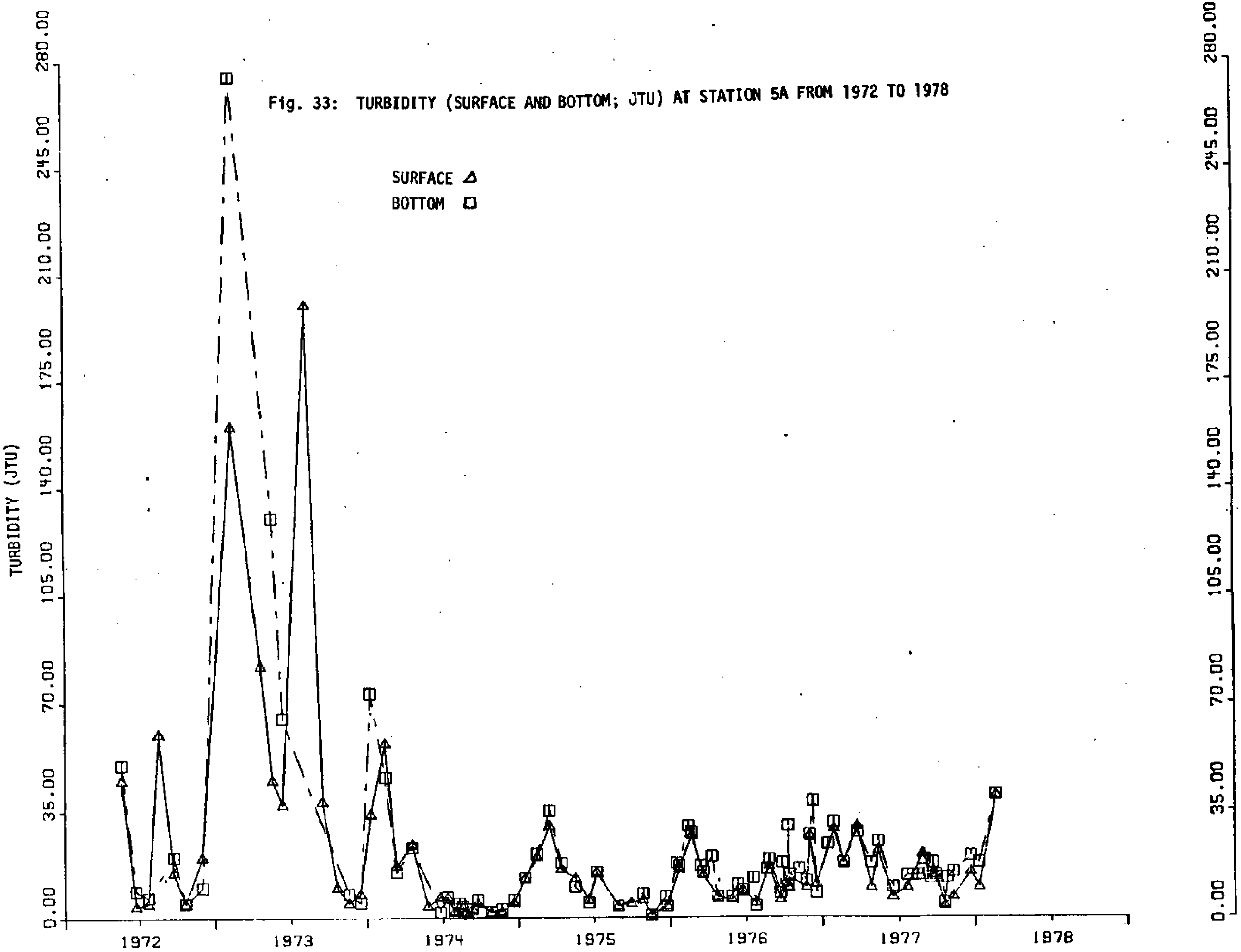


Fig. 33: TURBIDITY (SURFACE AND BOTTOM; JTU) AT STATION 5A FROM 1972 TO 1978

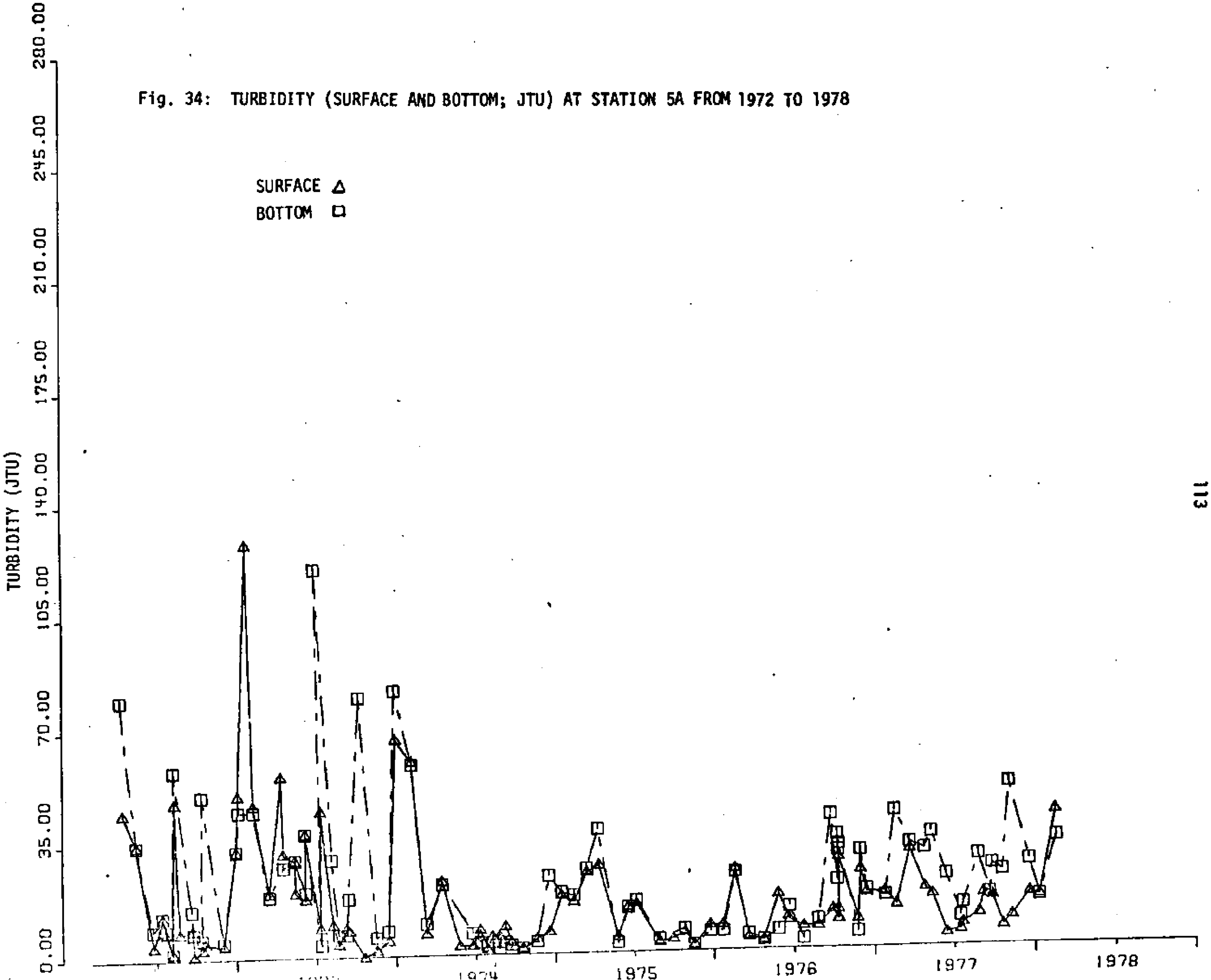


Fig. 34: TURBIDITY (SURFACE AND BOTTOM; JTU) AT STATION 5A FROM 1972 TO 1978

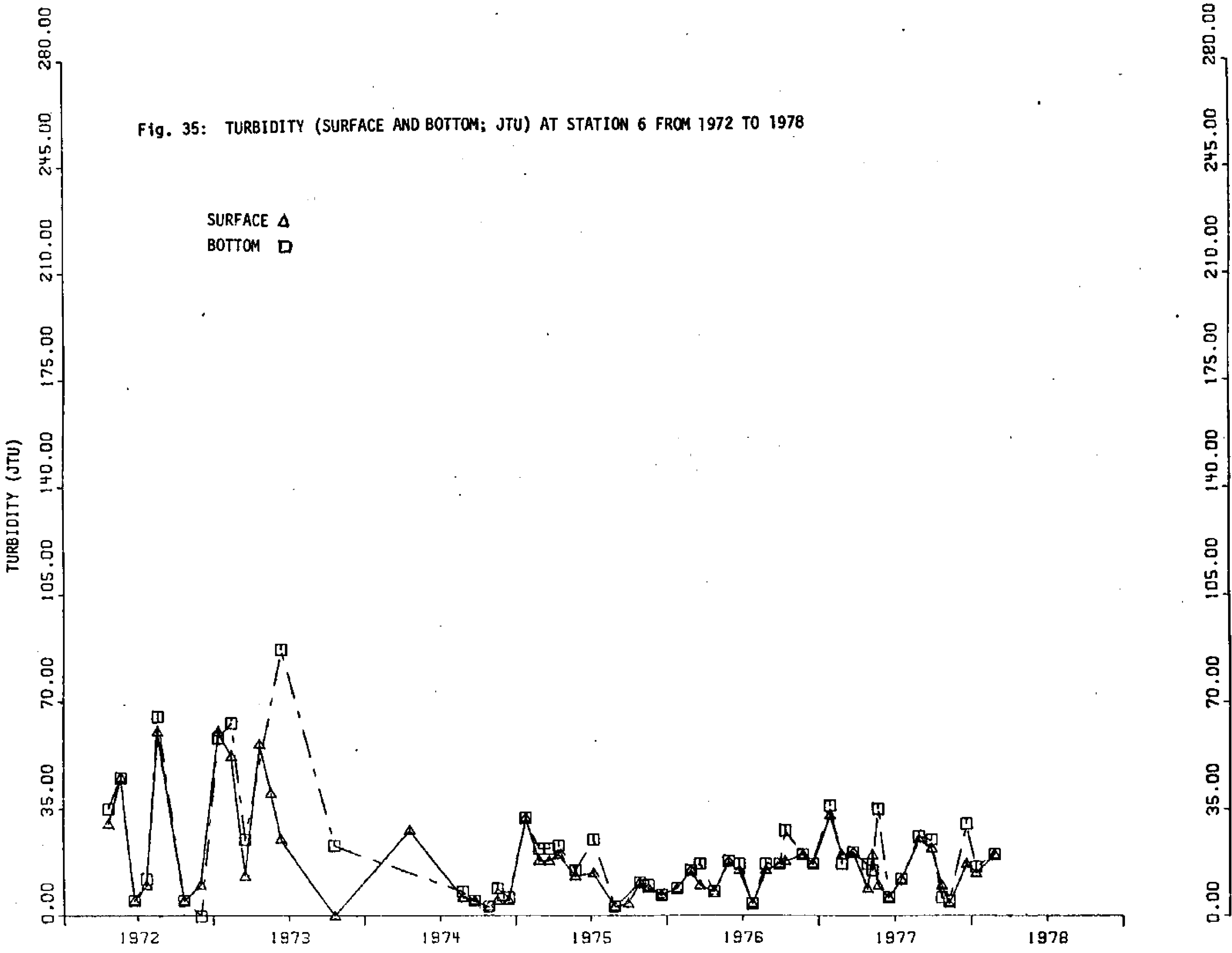


Fig. 35: TURBIDITY (SURFACE AND BOTTOM; JTU) AT STATION 6 FROM 1972 TO 1978

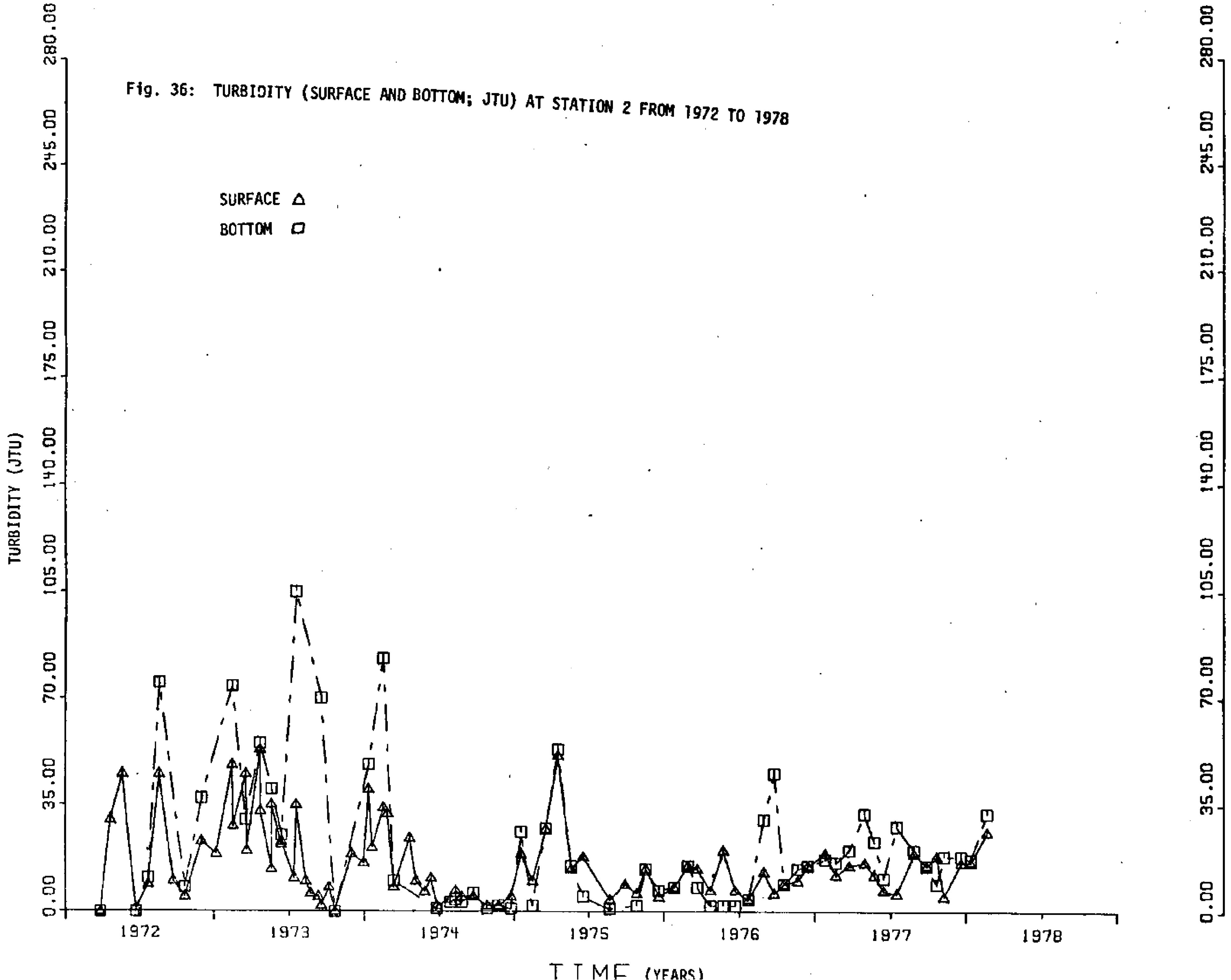


Fig. 36: TURBIDITY (SURFACE AND BOTTOM; JTU) AT STATION 2 FROM 1972 TO 1978

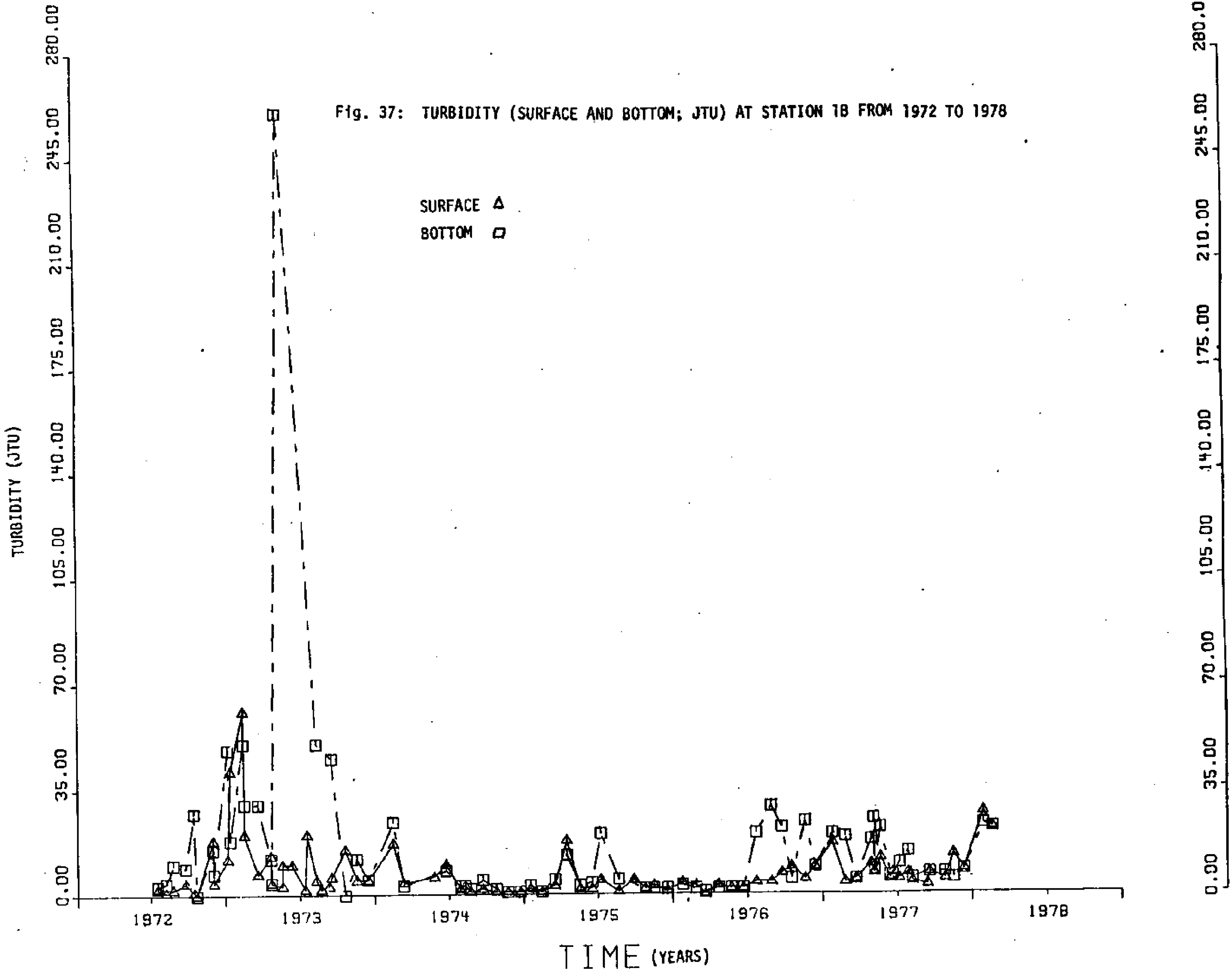


Fig. 37: TURBIDITY (SURFACE AND BOTTOM; JTU) AT STATION 1B FROM 1972 TO 1978

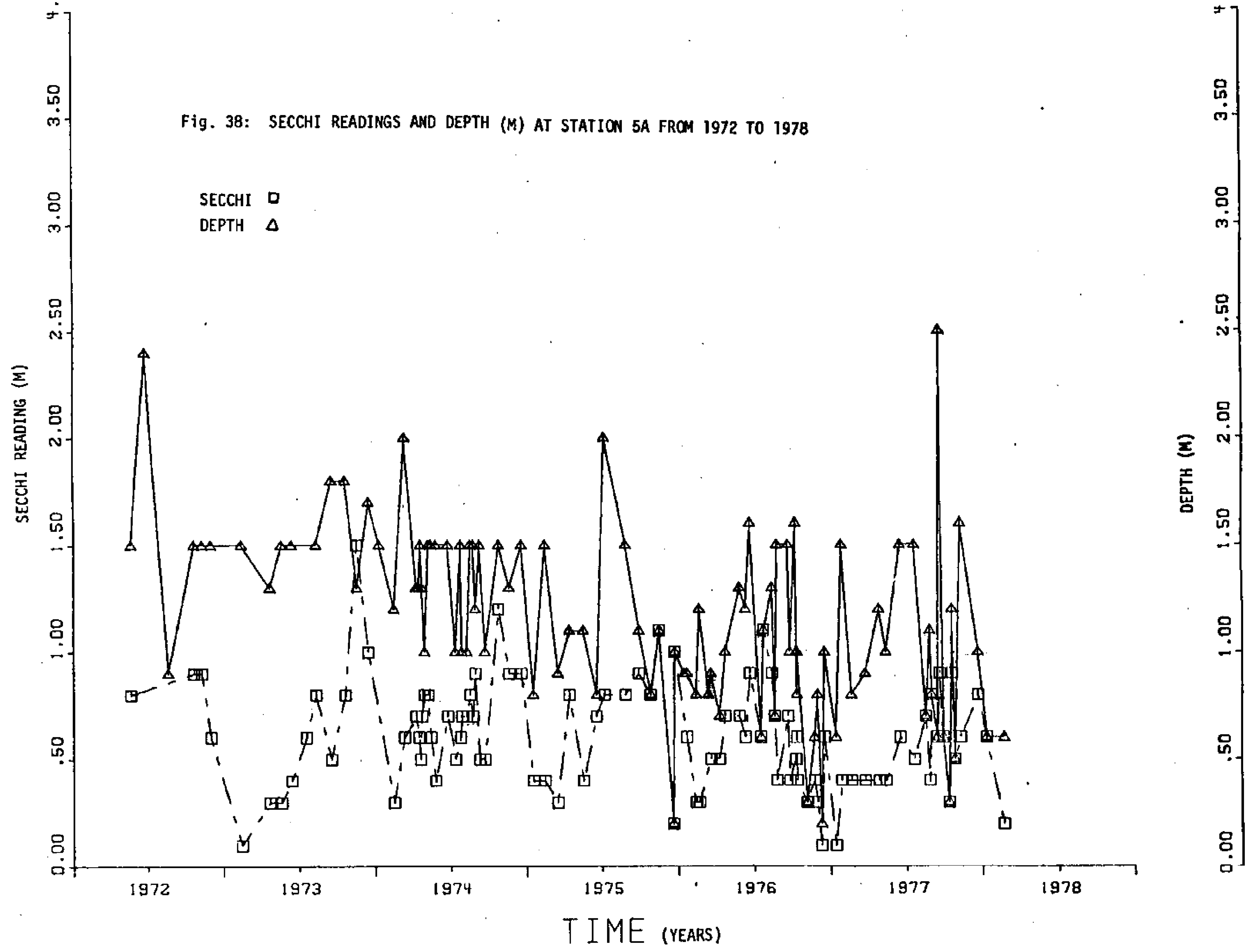


Fig. 38: SECCHI READINGS AND DEPTH (M) AT STATION 5A FROM 1972 TO 1978

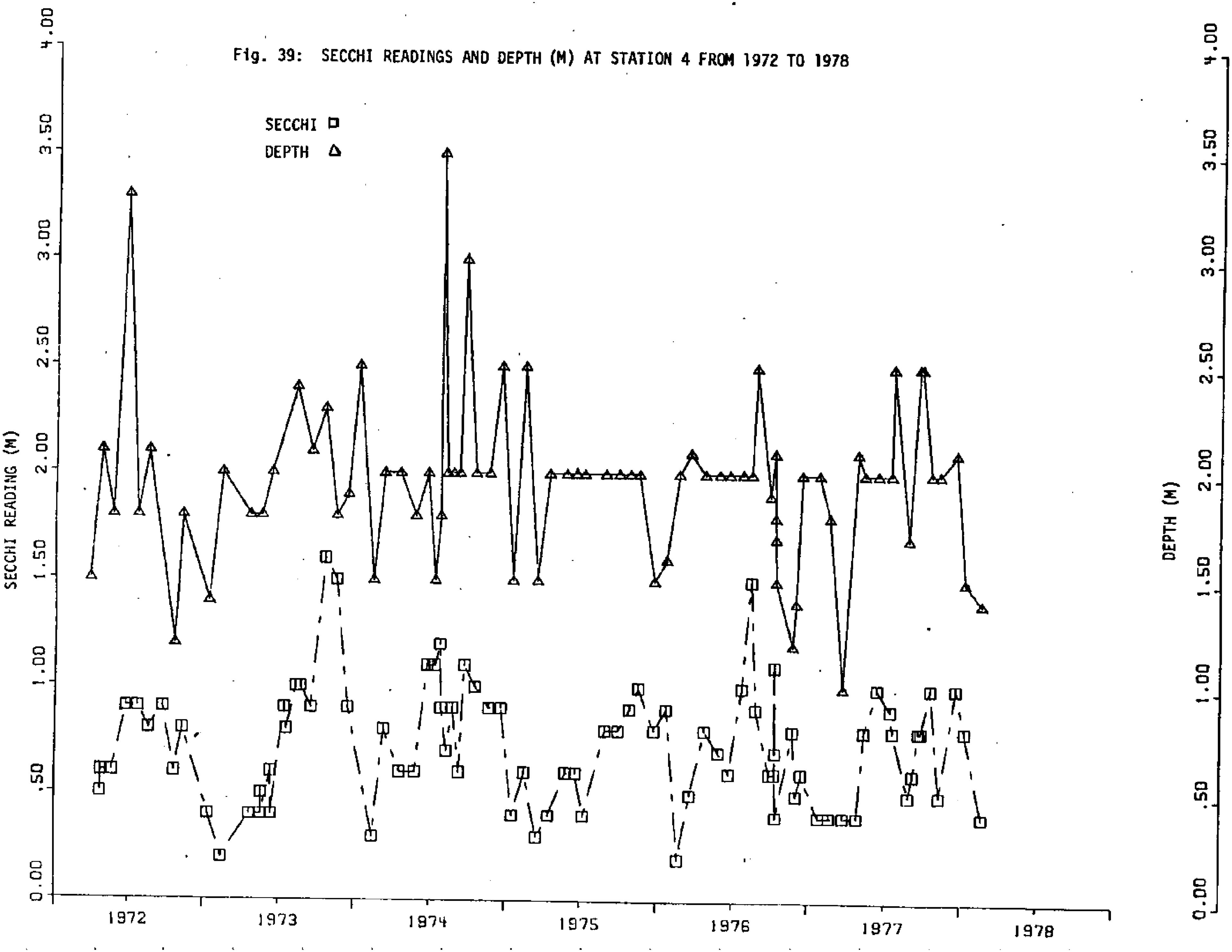


Fig. 39: SECCHI READINGS AND DEPTH (M) AT STATION 4 FROM 1972 TO 1978

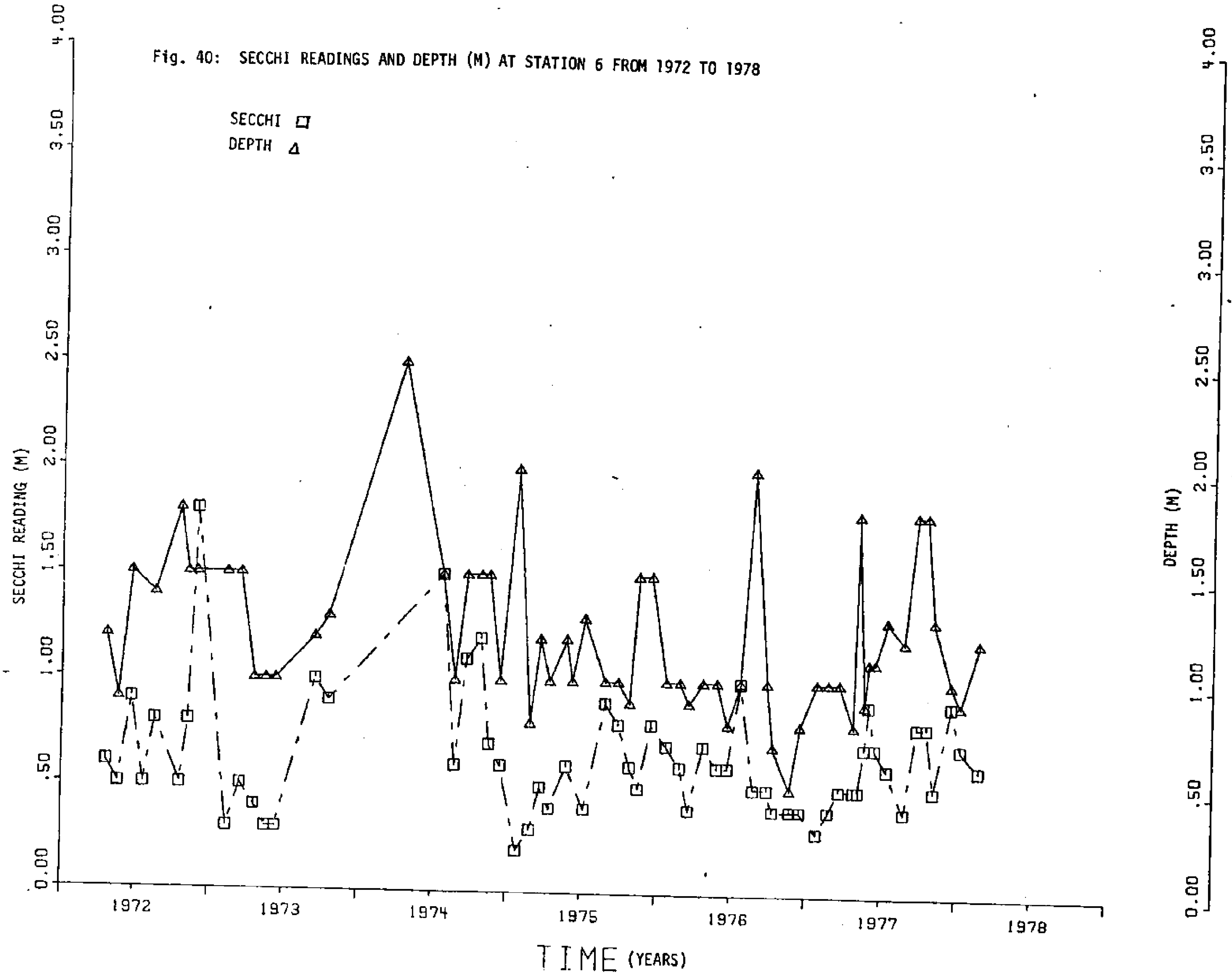


Fig. 40: SECCHI READINGS AND DEPTH (M) AT STATION 6 FROM 1972 TO 1978

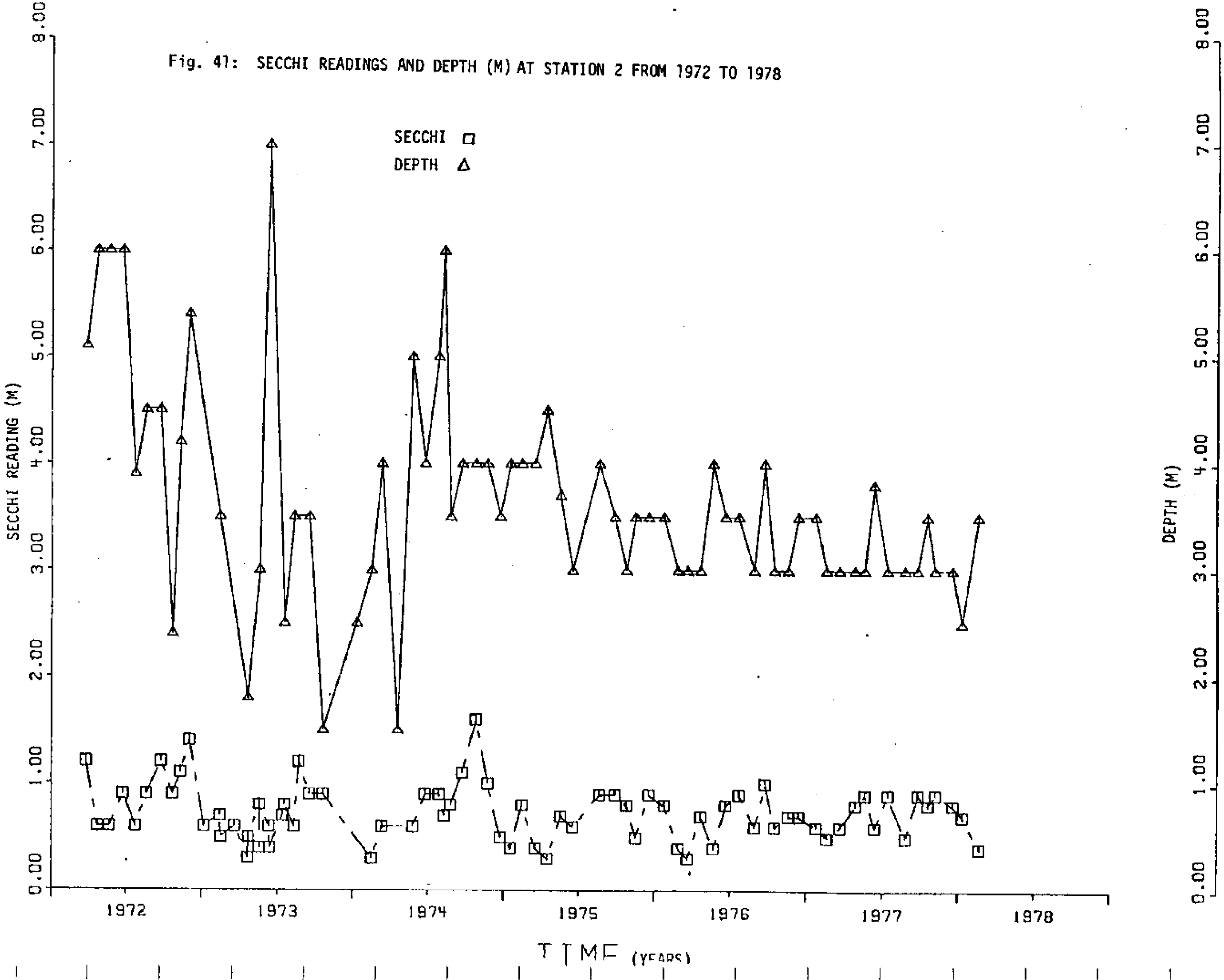
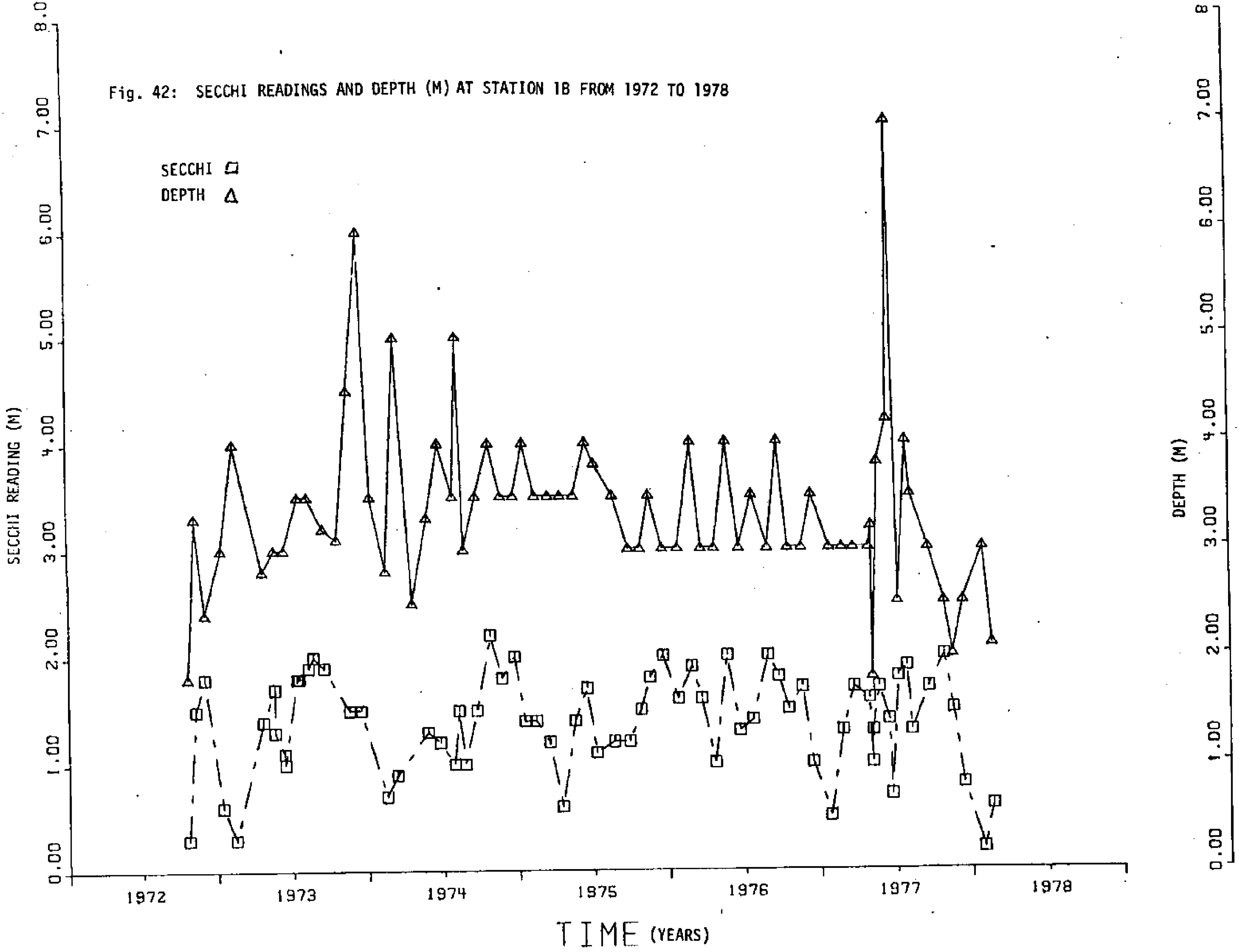


Fig. 41: SECCHI READINGS AND DEPTH (M) AT STATION 2 FROM 1972 TO 1978

Fig. 42: SECCHI READINGS AND DEPTH (M) AT STATION 1B FROM 1972 TO 1978



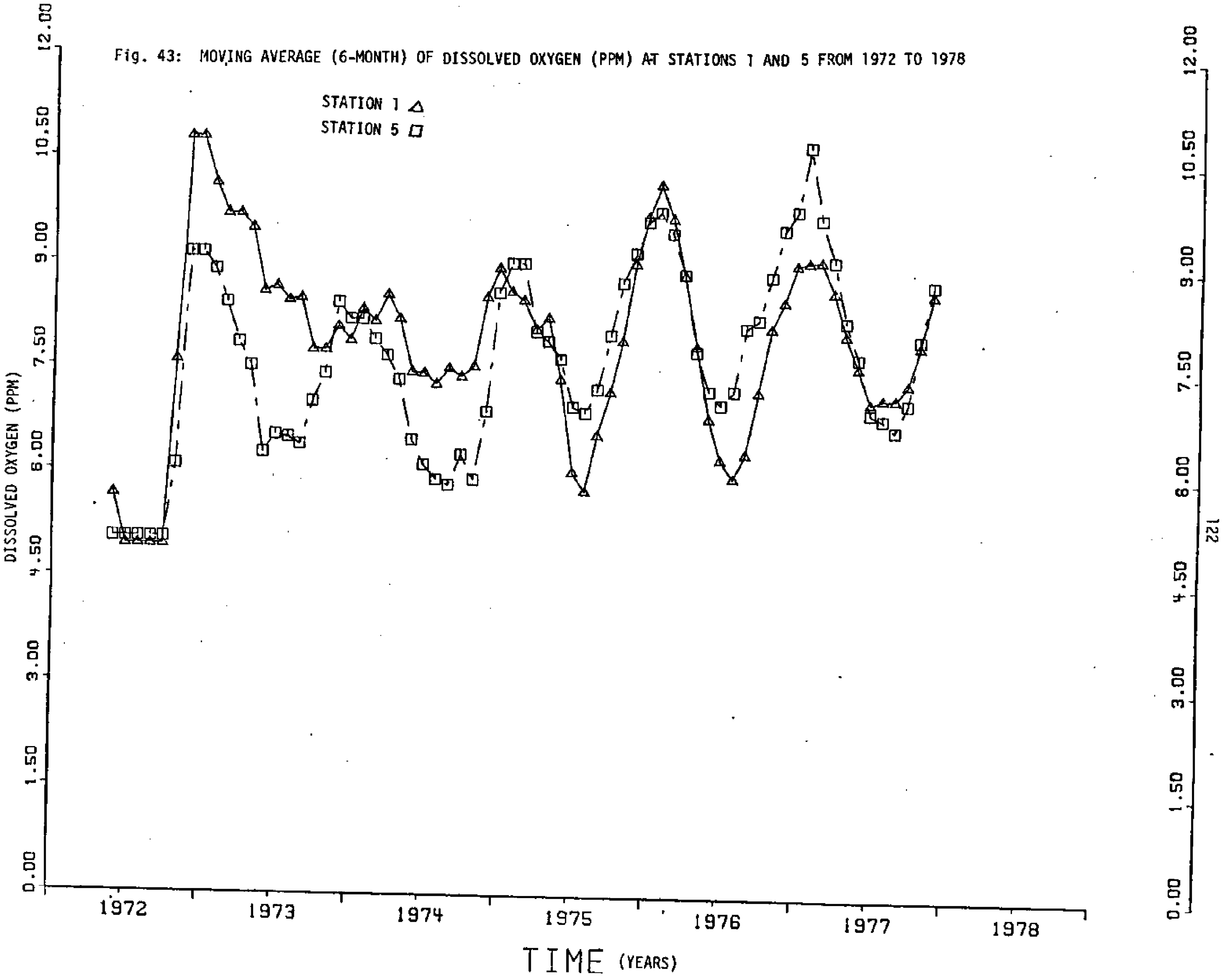


Fig. 43: MOVING AVERAGE (6-MONTH) OF DISSOLVED OXYGEN (PPM) AT STATIONS 1 AND 5 FROM 1972 TO 1978

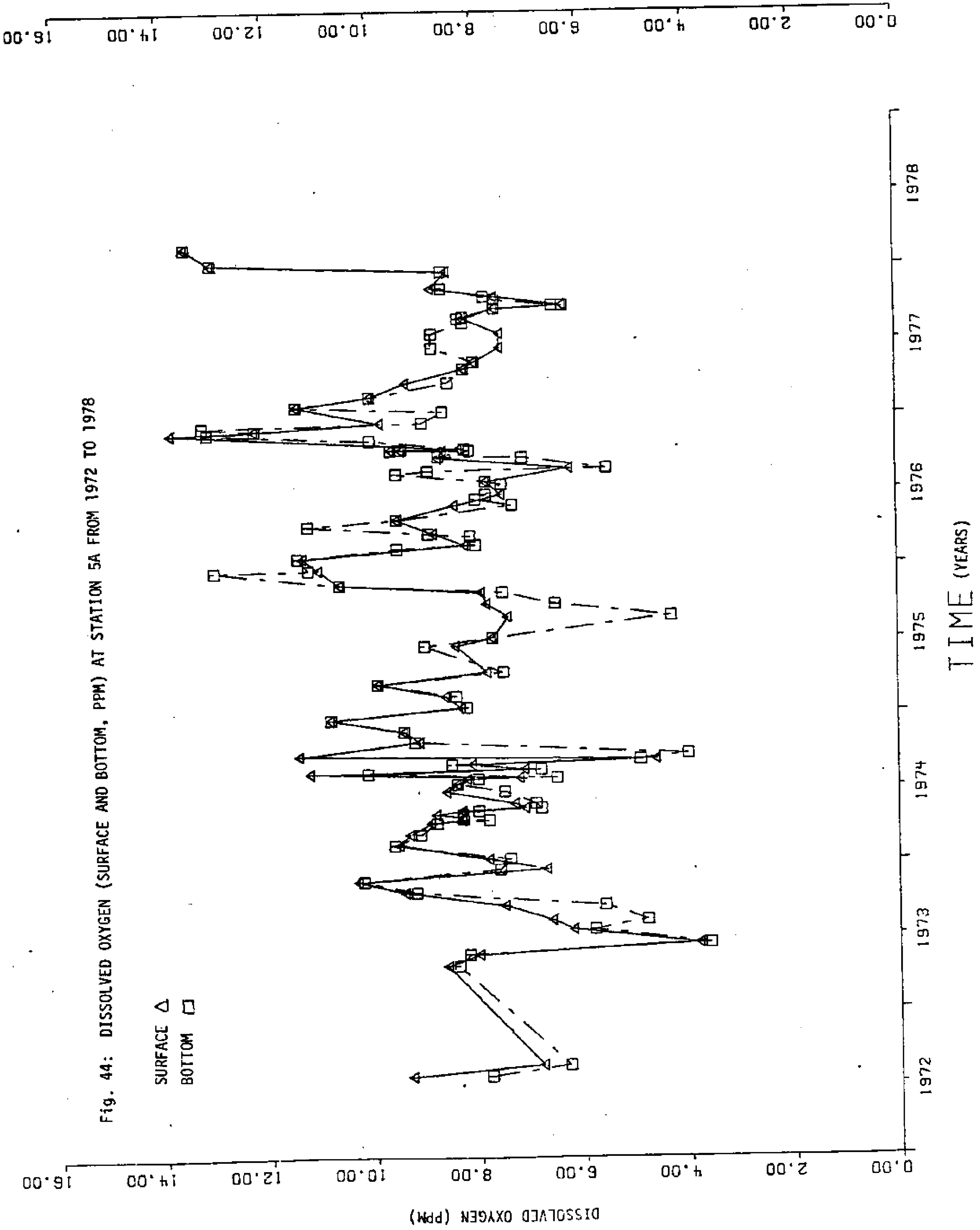


Fig. 44: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 5A FROM 1972 TO 1978

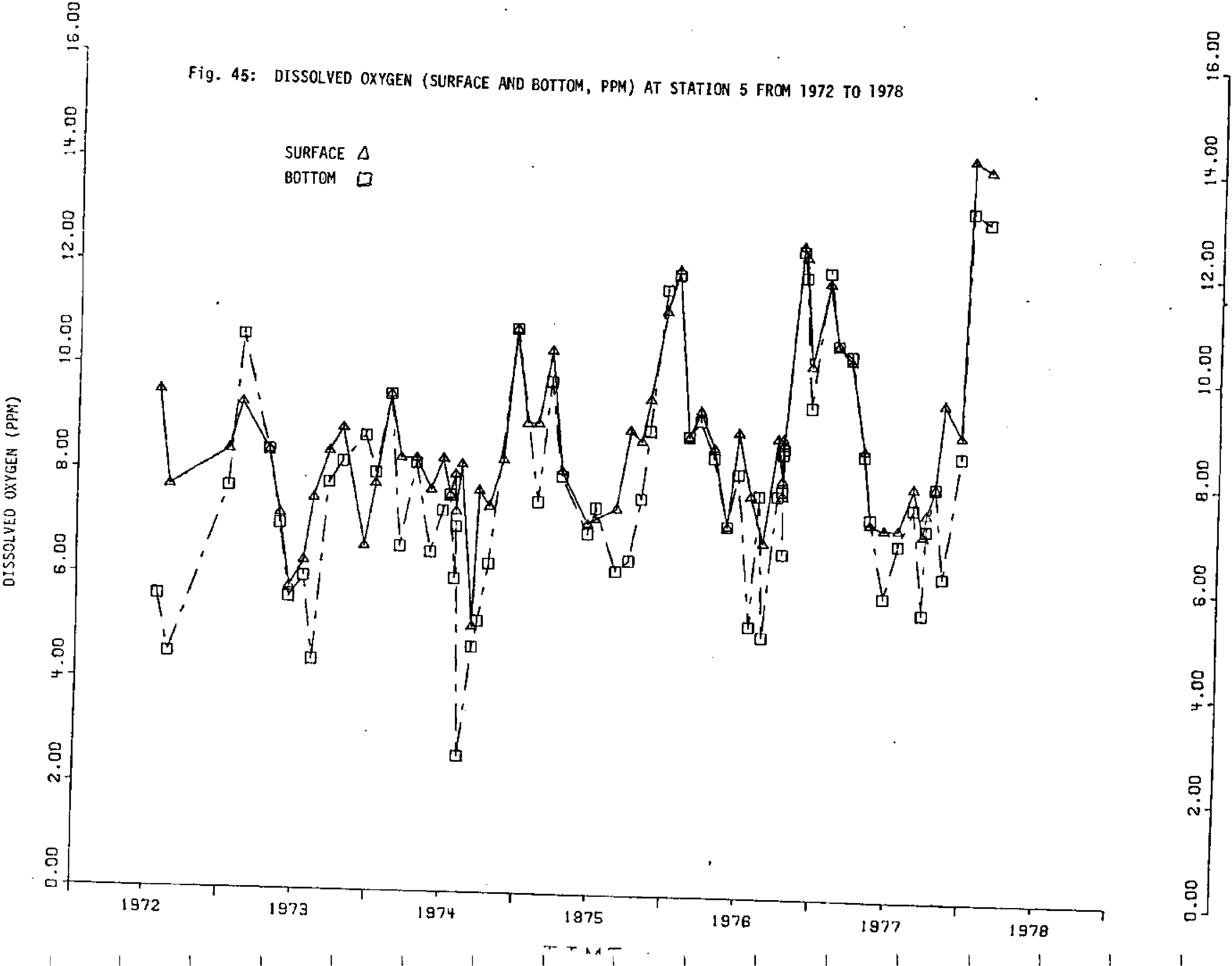


Fig. 45: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 5 FROM 1972 TO 1978

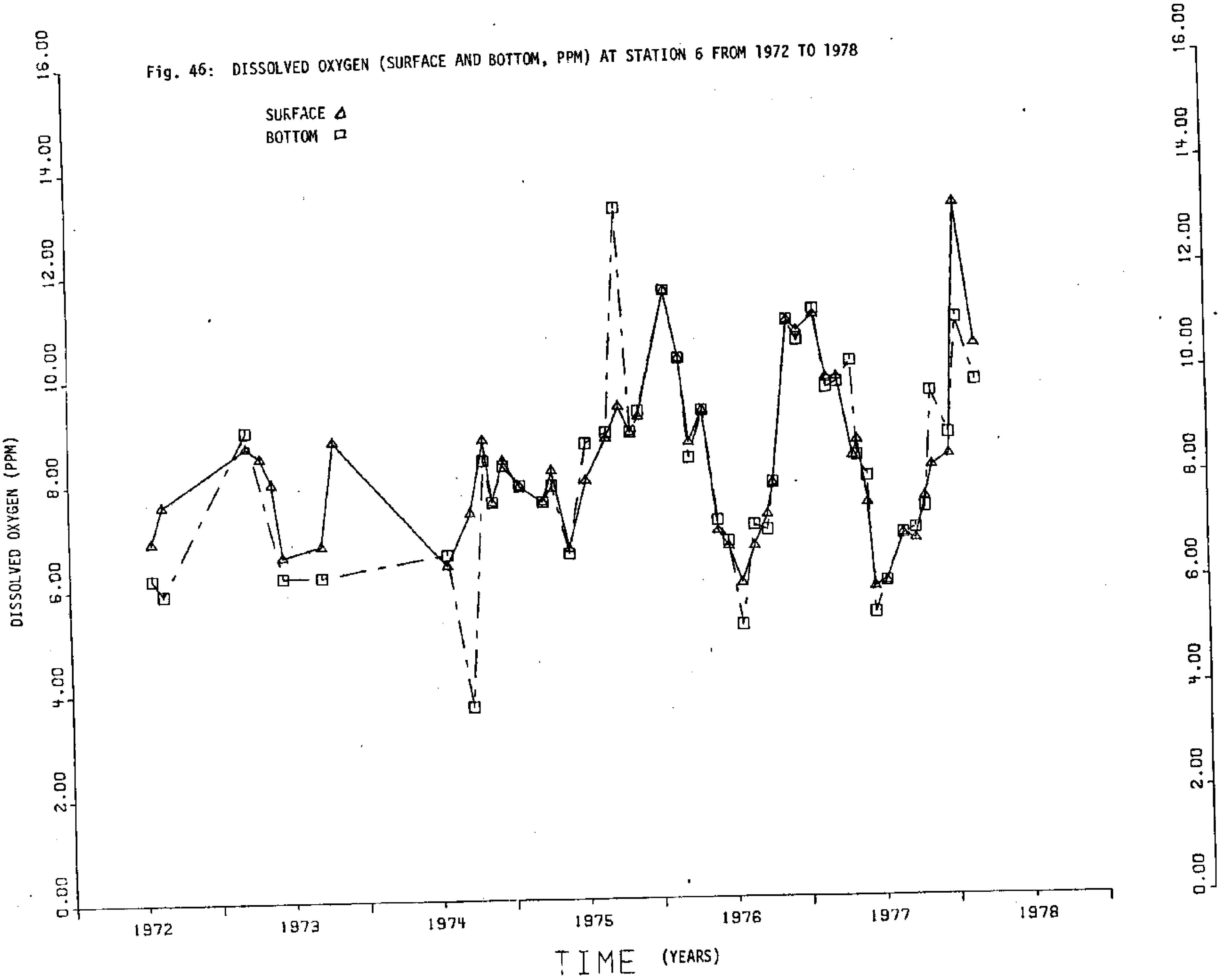


Fig. 46: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 6 FROM 1972 TO 1978

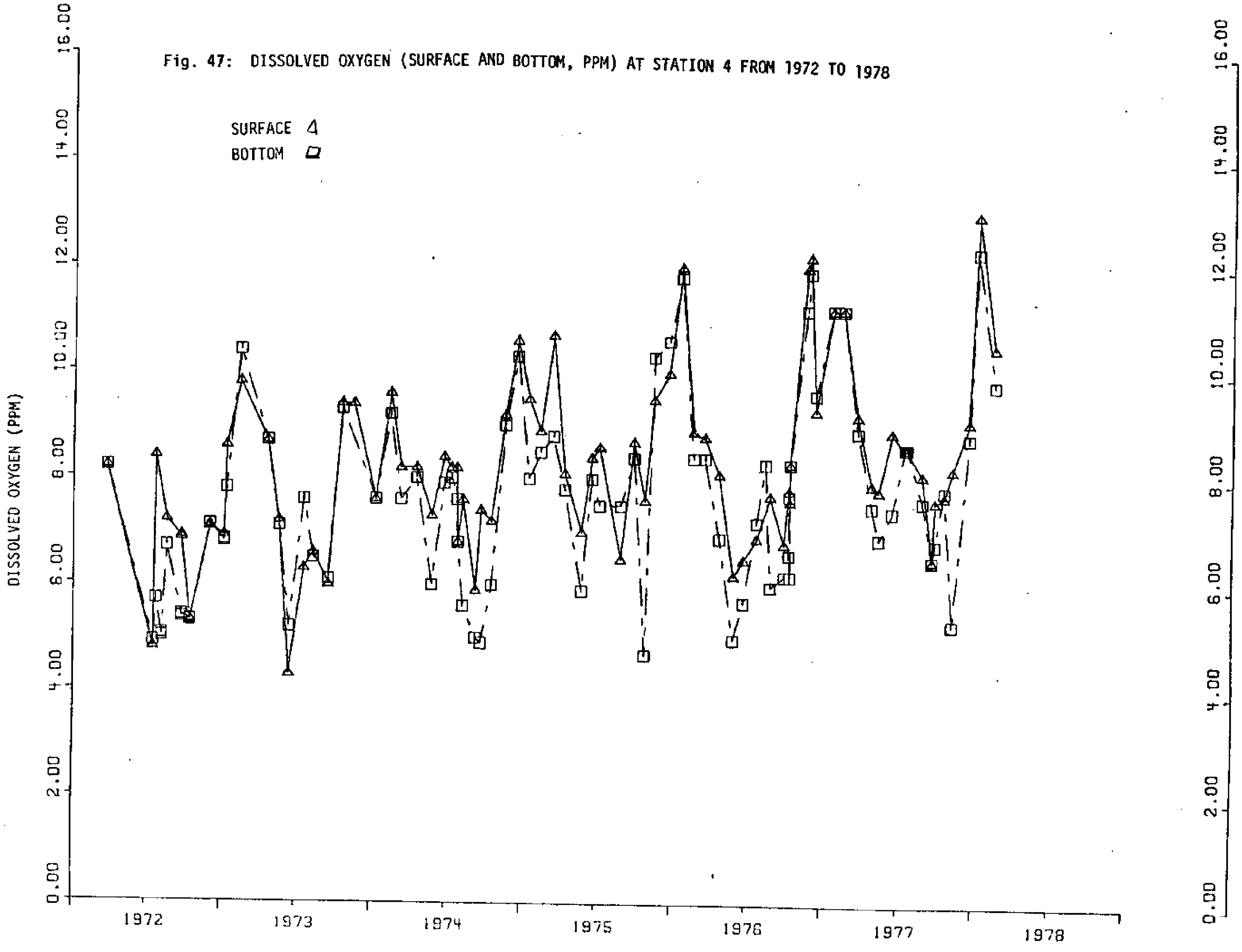


Fig. 47: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 4 FROM 1972 TO 1978

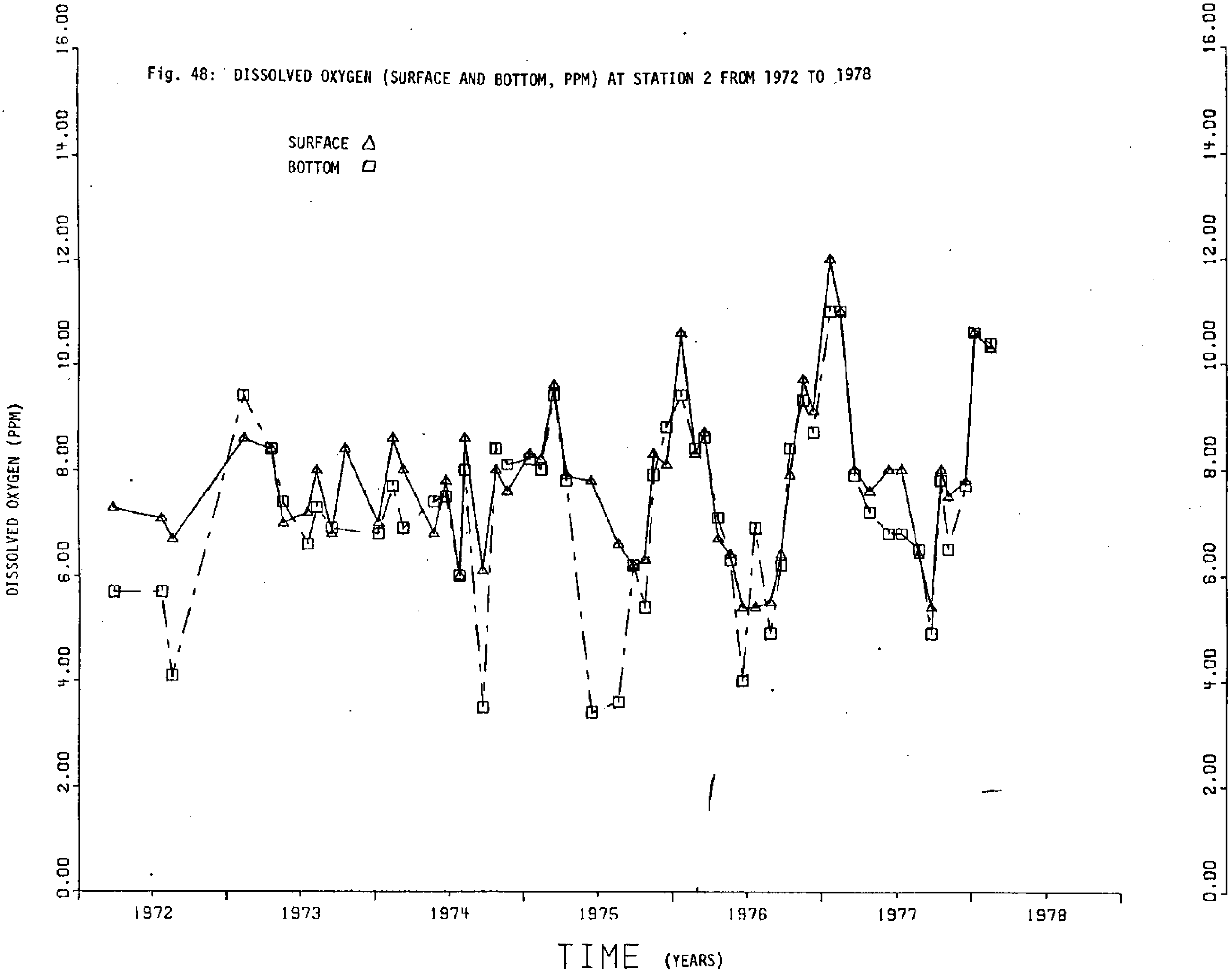


Fig. 48: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 2 FROM 1972 TO 1978

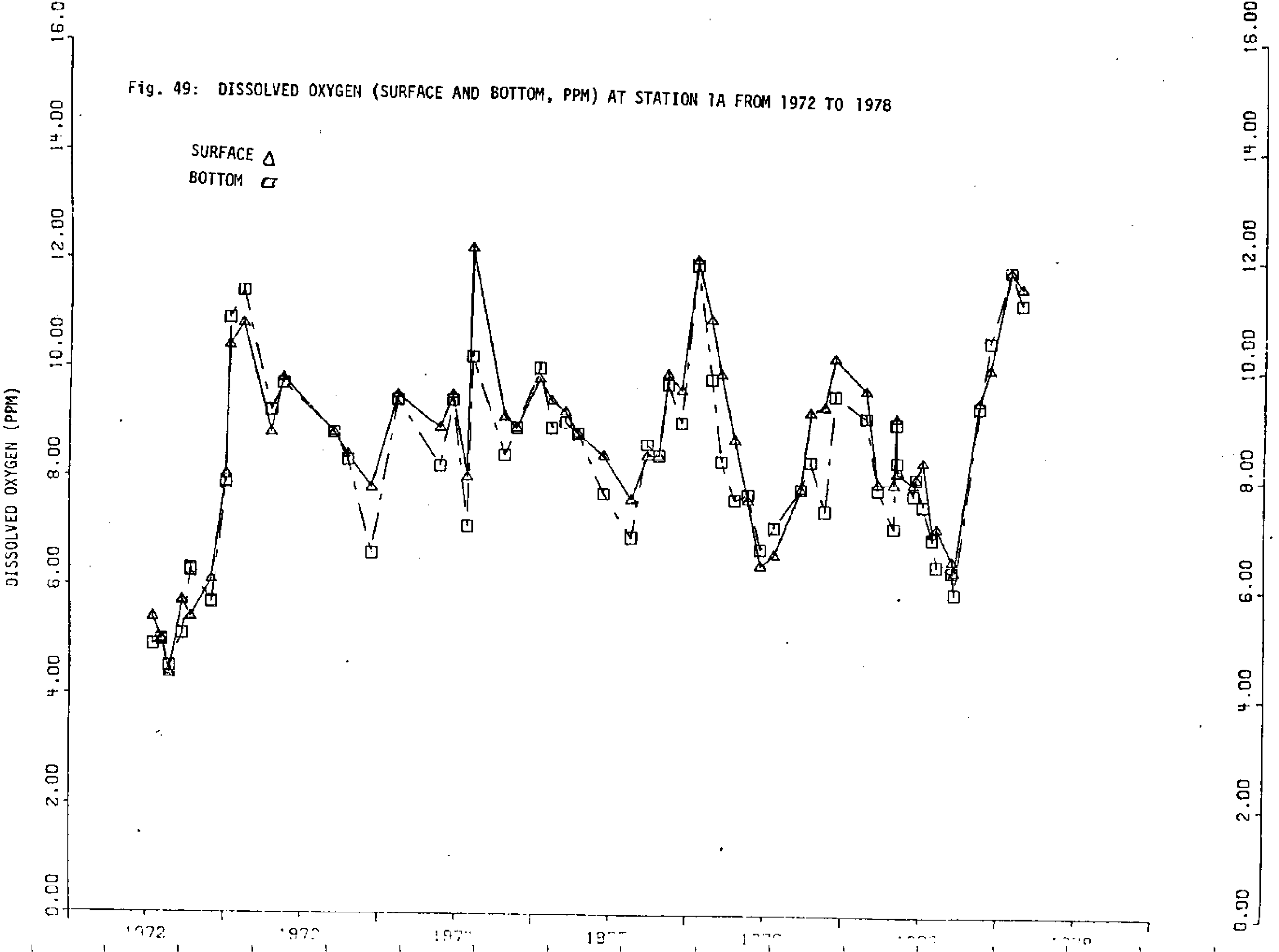


Fig. 49: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 1A FROM 1972 TO 1978

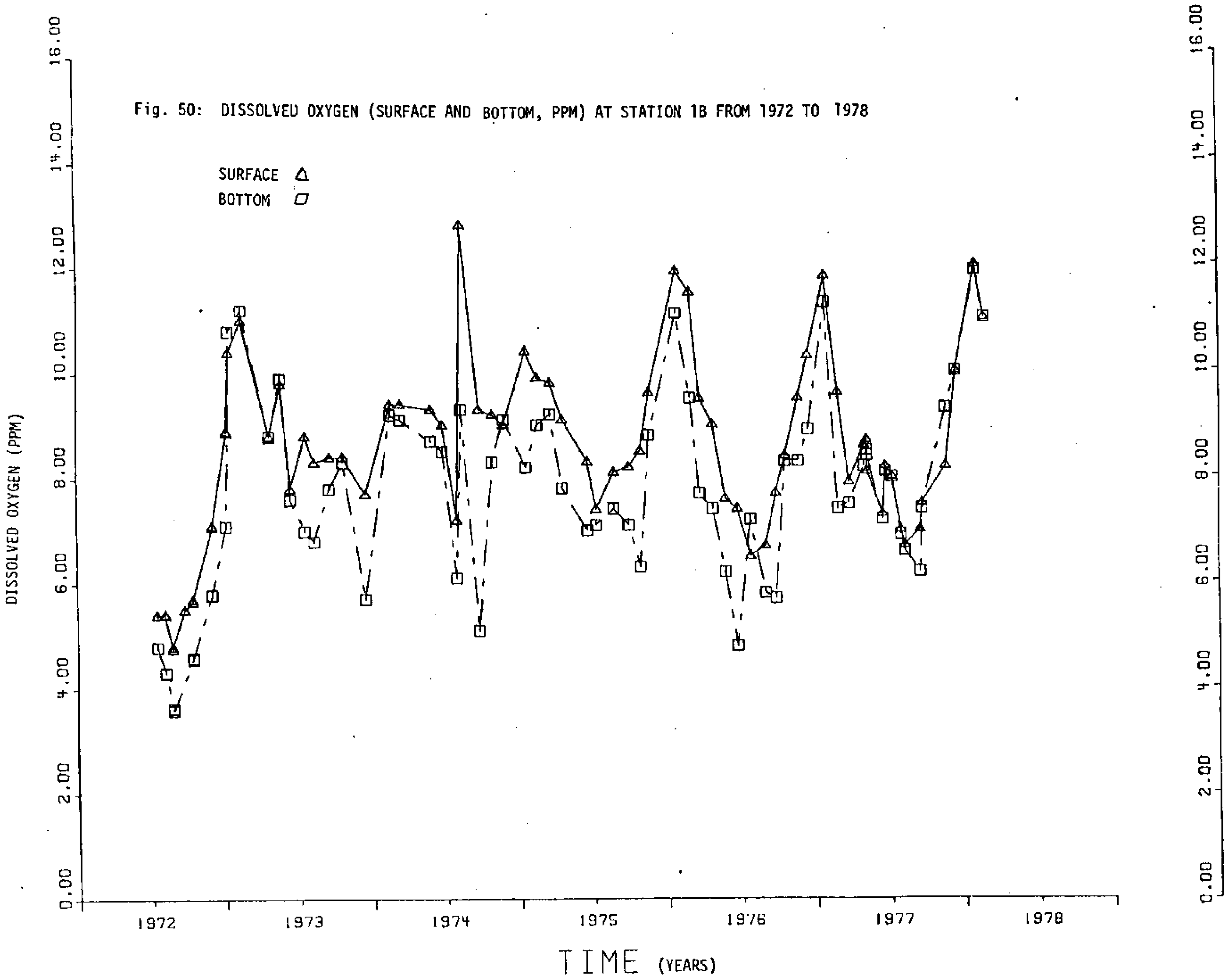


Fig. 50: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 1B FROM 1972 TO 1978

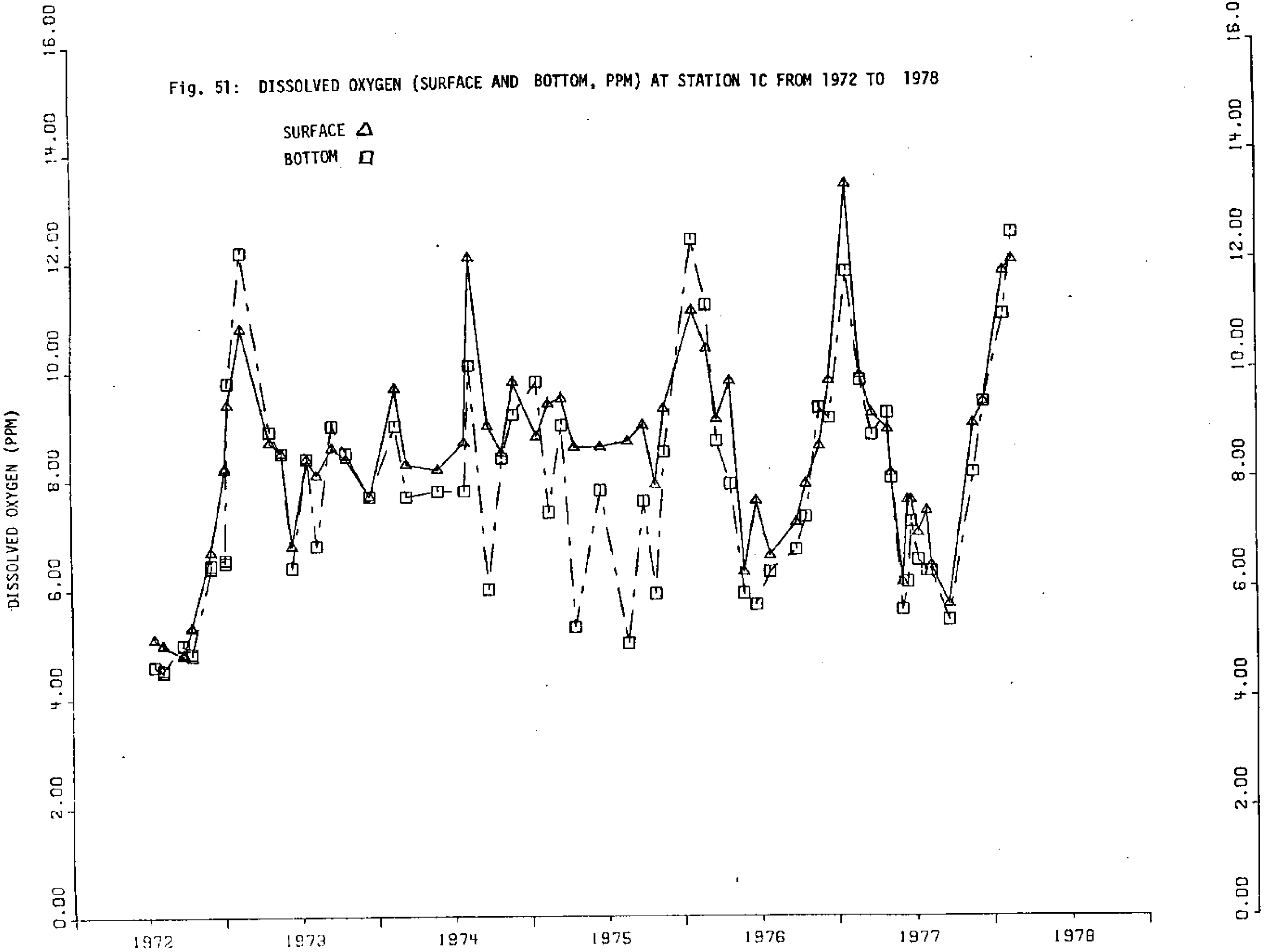
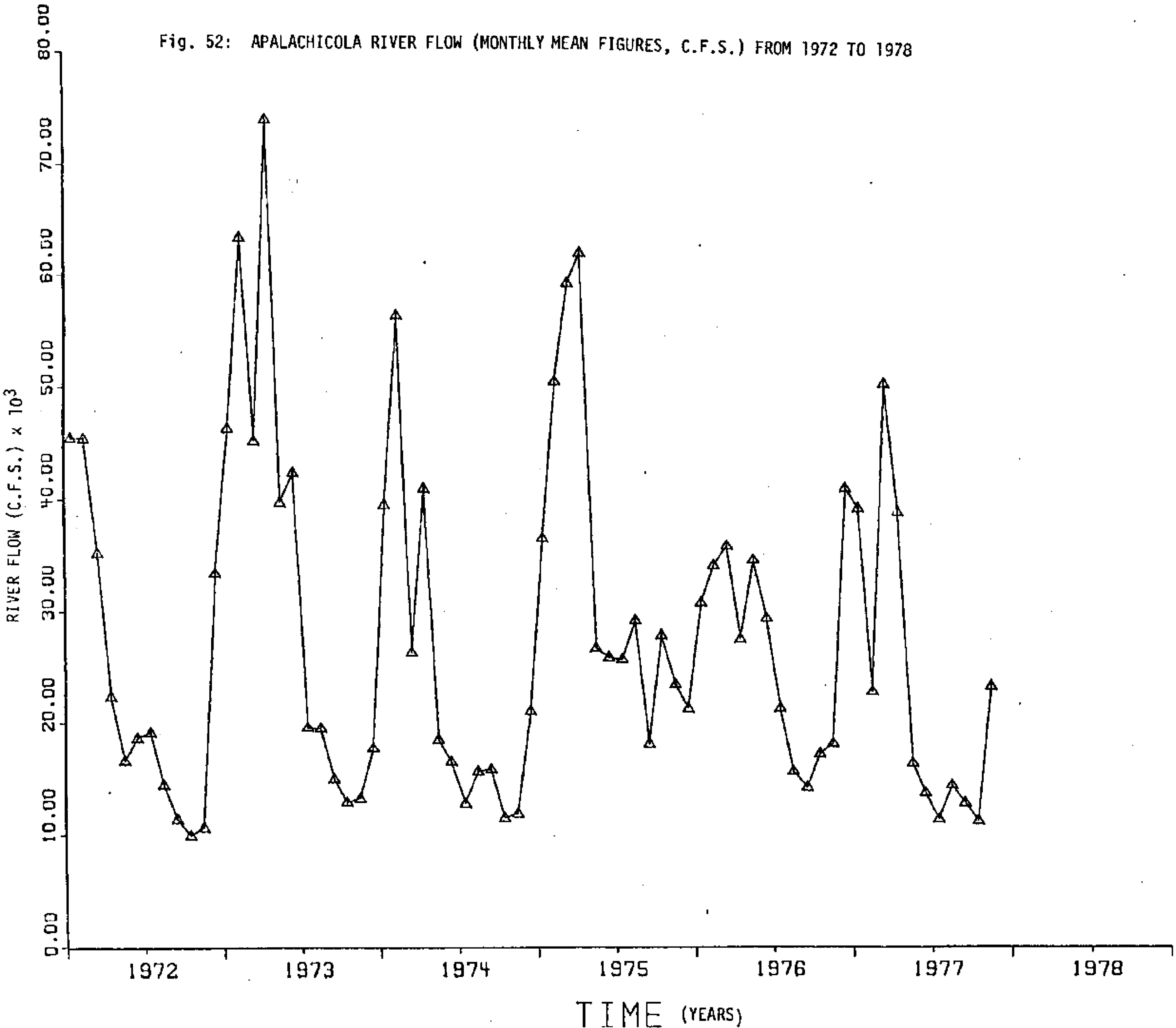


Fig. 51: DISSOLVED OXYGEN (SURFACE AND BOTTOM, PPM) AT STATION 1C FROM 1972 TO 1978

Fig. 52: APALACHICOLA RIVER FLOW (MONTHLY MEAN FIGURES, C.F.S.) FROM 1972 TO 1978



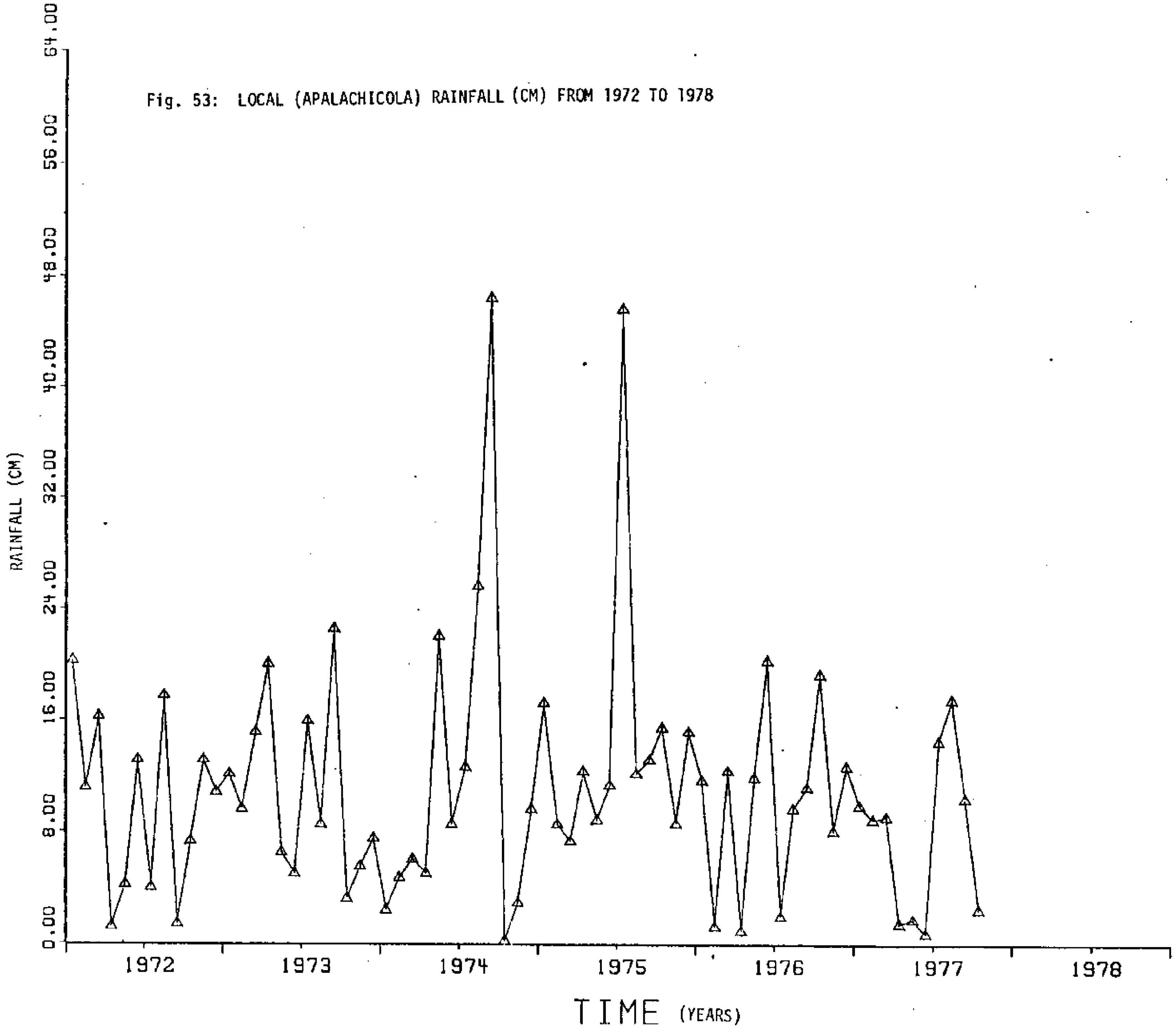
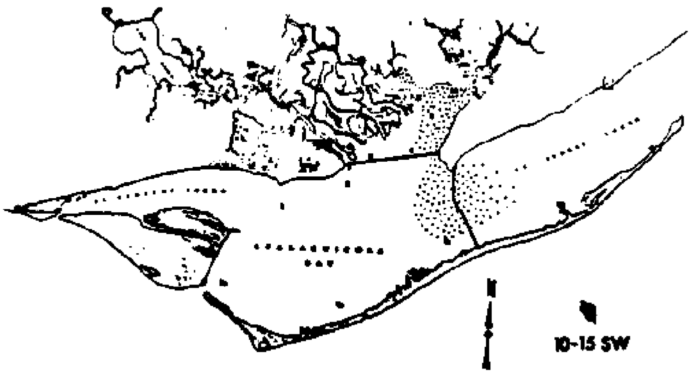


Fig. 53: LOCAL (APALACHICOLA) RAINFALL (CM) FROM 1972 TO 1978

Figure 54: Visual observations of the distribution of colored water in East Bay at various times during the sampling period.



8/74

9/74

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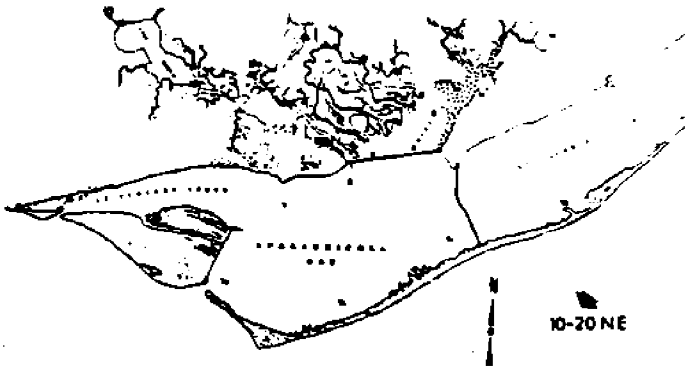
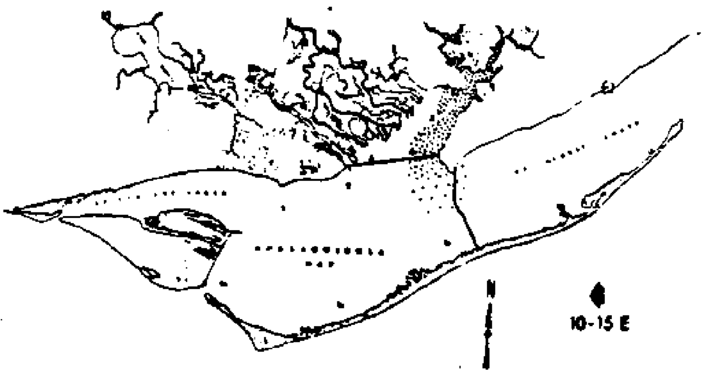


4/75



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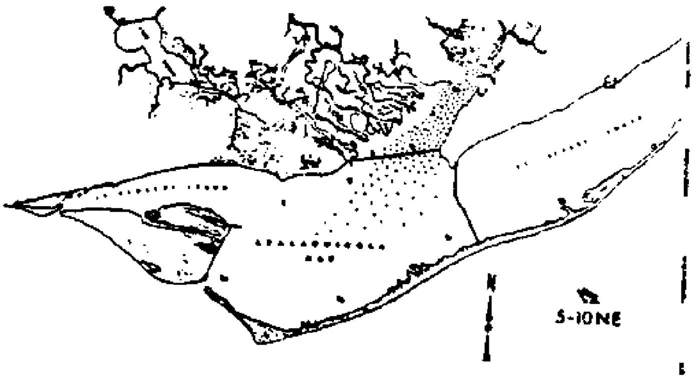

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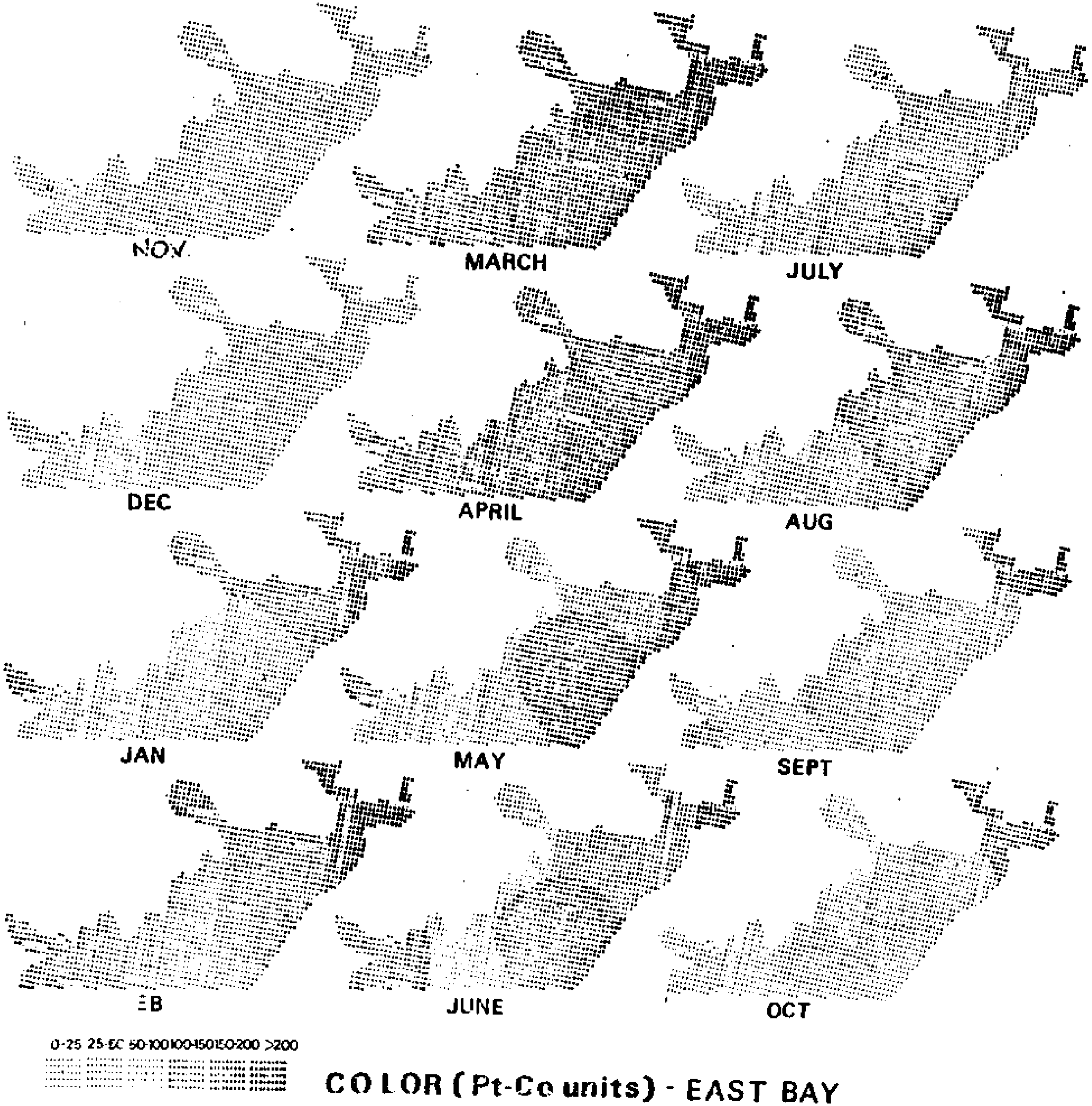
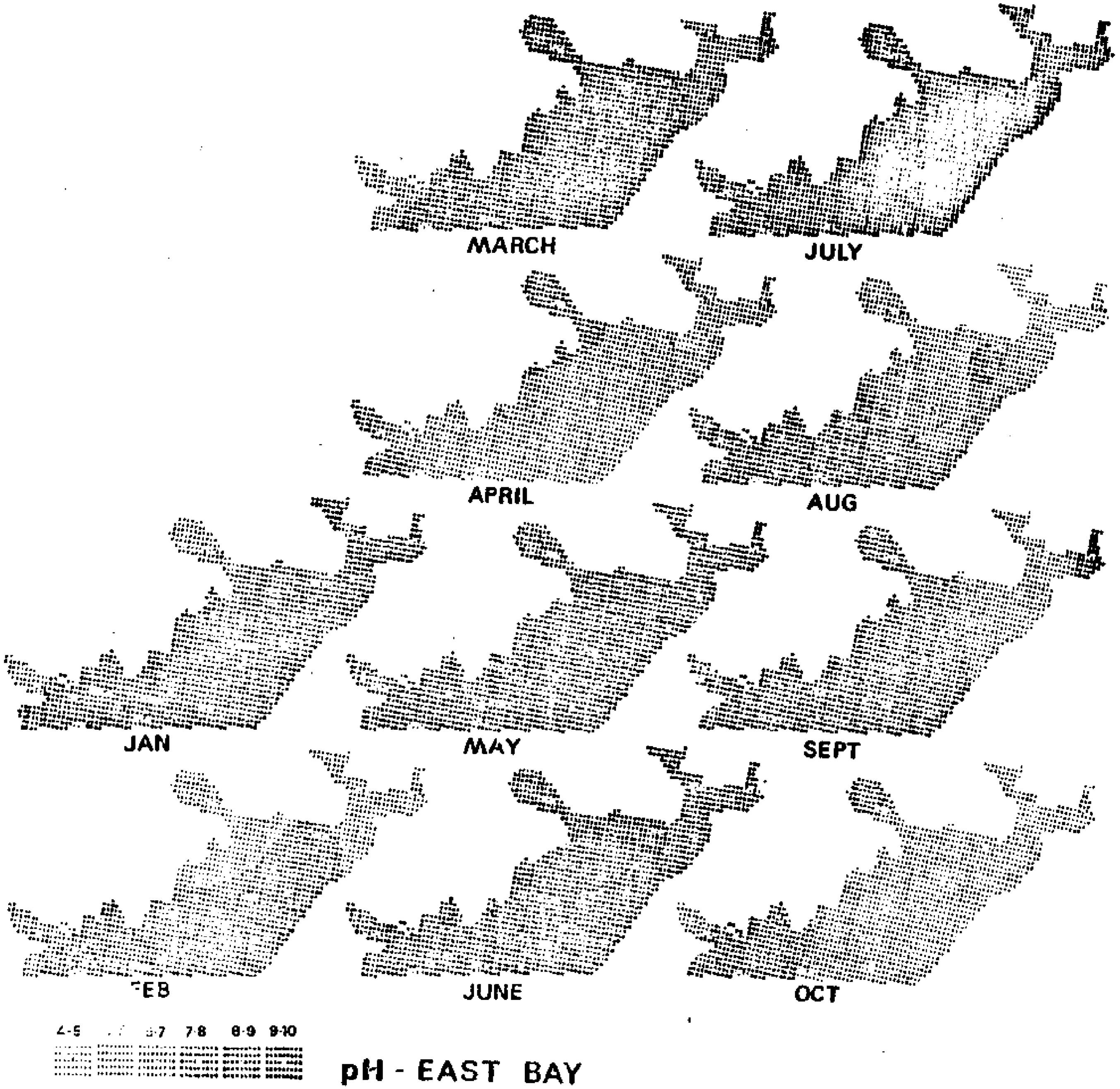
Fig. 55: Distribution of water color (Pt-Co units) in East Bay mapped by month from November, 1974 through October, 1975.



COLOR (Pt-Co units) - EAST BAY

Fig. 56: Distribution of pH in East Bay mapped by month from November, 1974 through October, 1975.



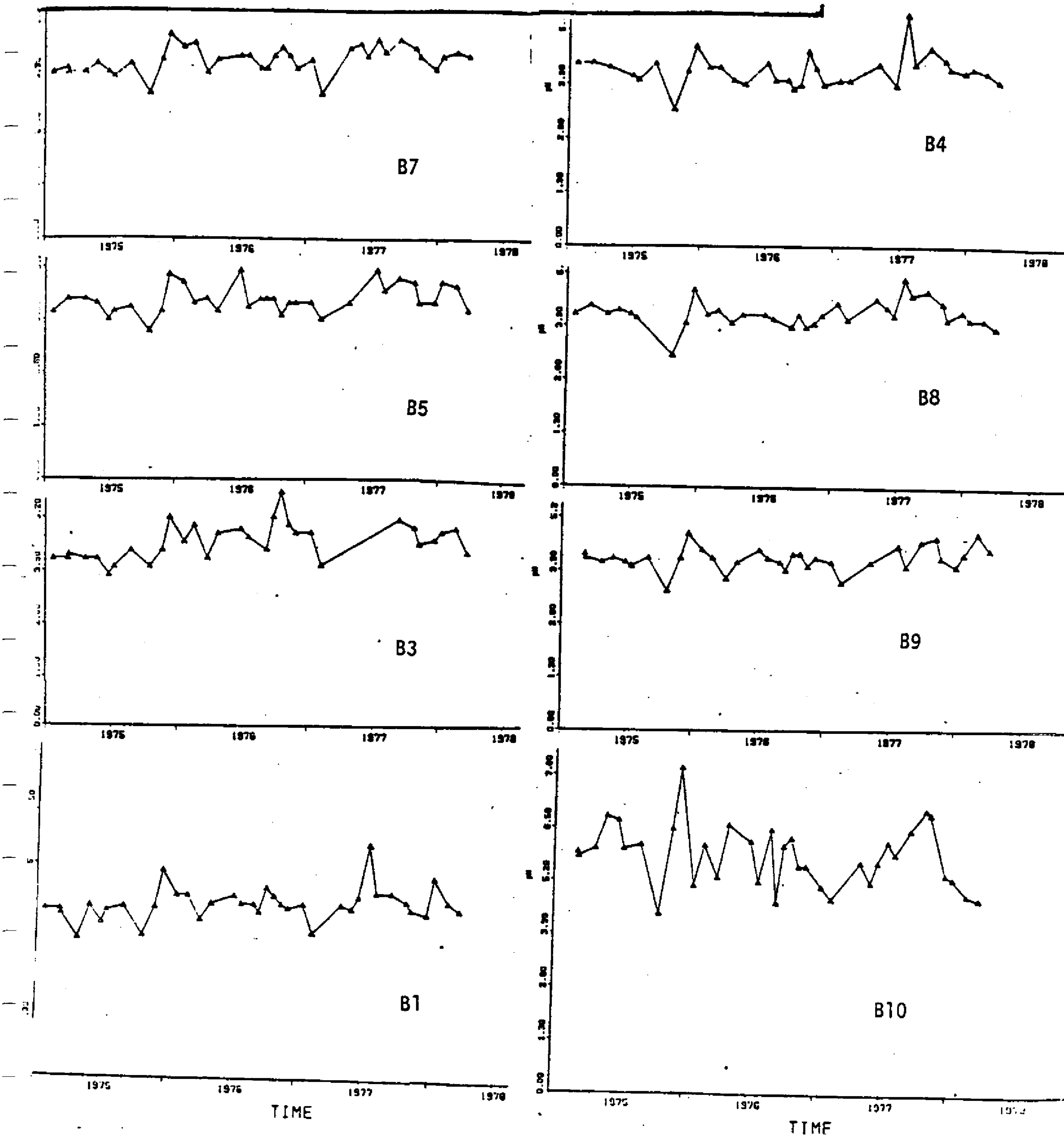
pH - EAST BAY

Fig. 57: Aerial photograph of Round Bay (background) and cleared portions of the Tate's Hell Swamp during the summer of 1976, and drainage ditch leading to the mouth of Round Bay.

Fig. 58: Forestry operations in the Tate's Hell Swamp drainage leading to Round Bay (summer, 1976).

Fig. 59: Forestry operations in the Round Bay drainage system during the summer of 1976.

area and drainage leading
to West Bayou (1975-1978



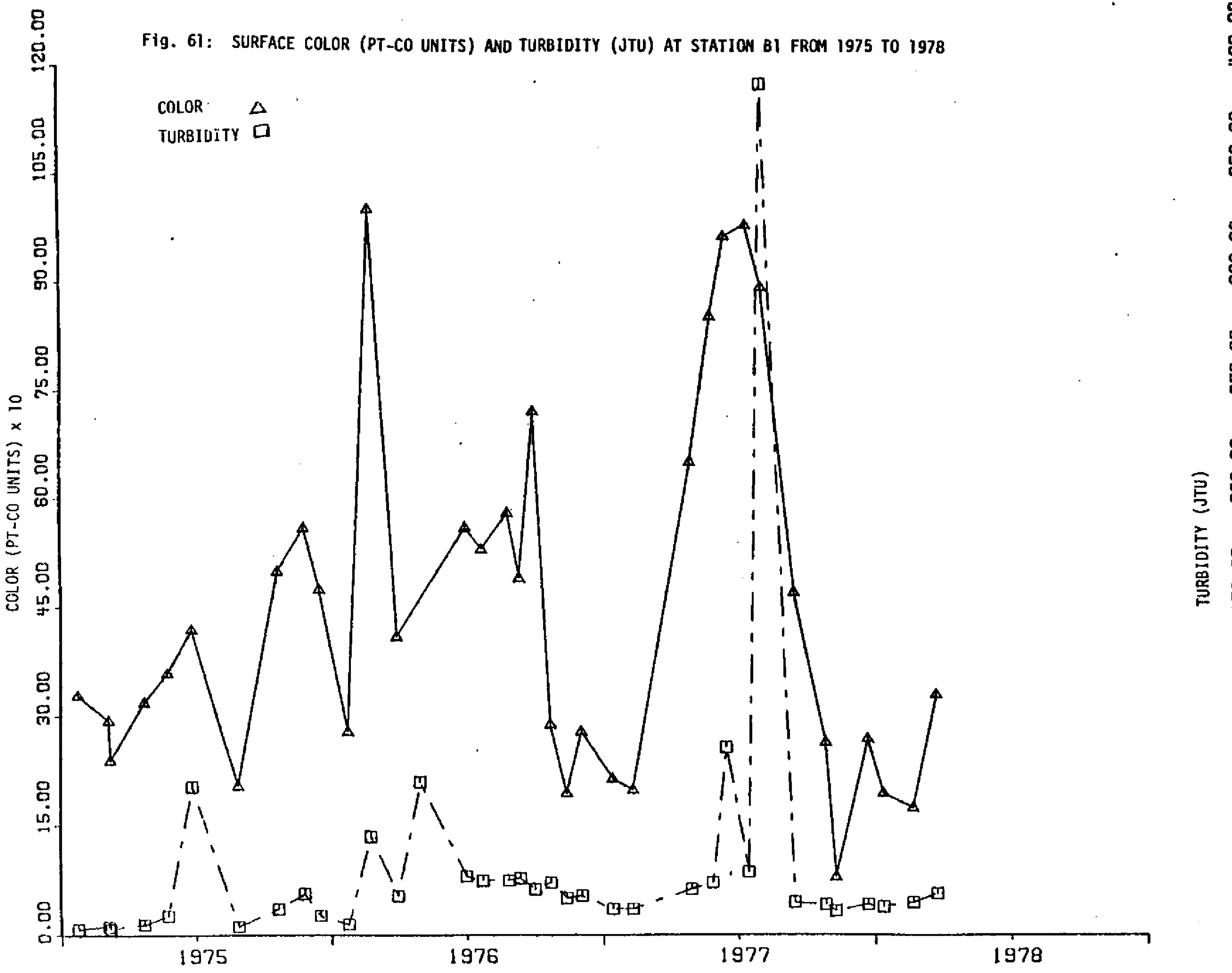


Fig. 61: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B1 FROM 1975 TO 1978

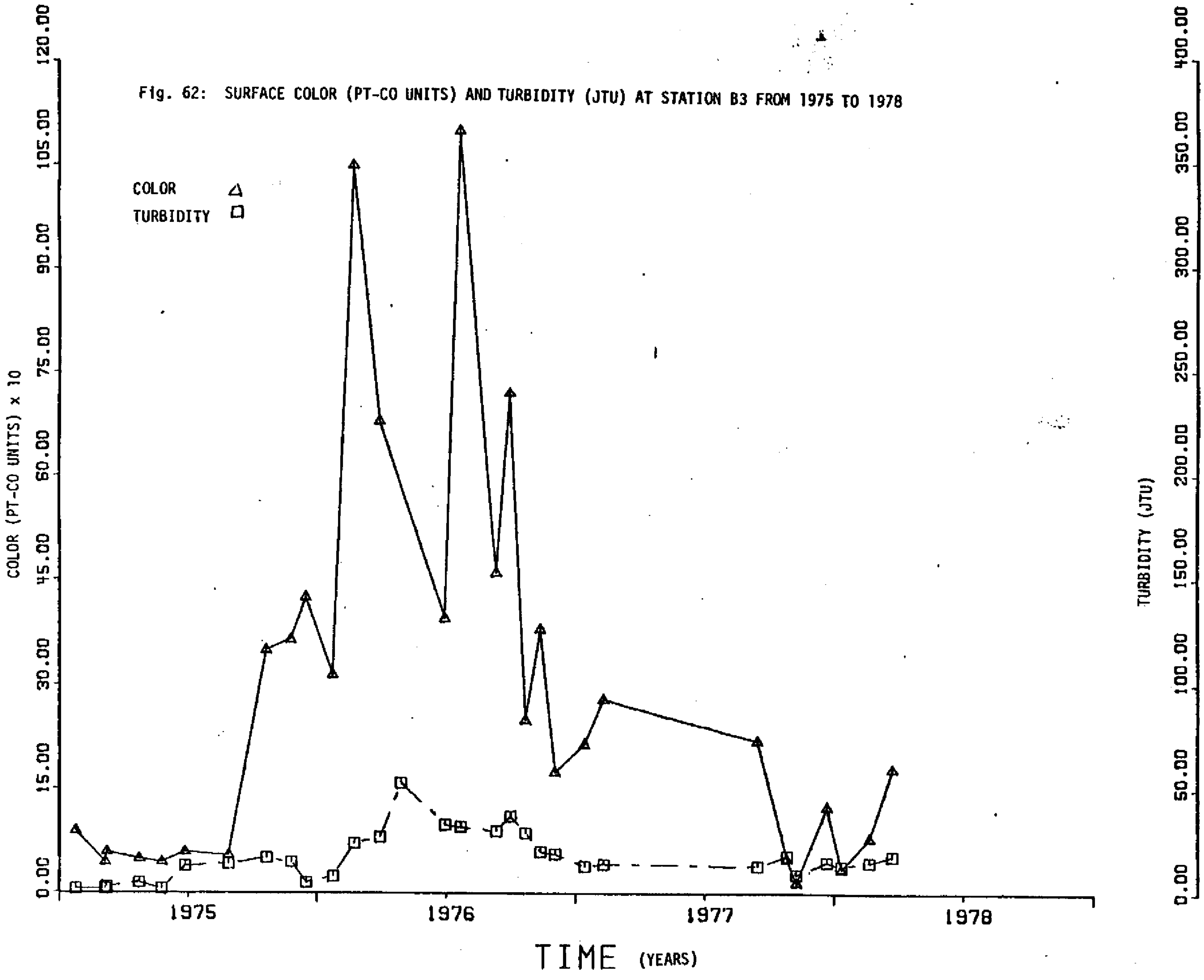


Fig. 62: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B3 FROM 1975 TO 1978

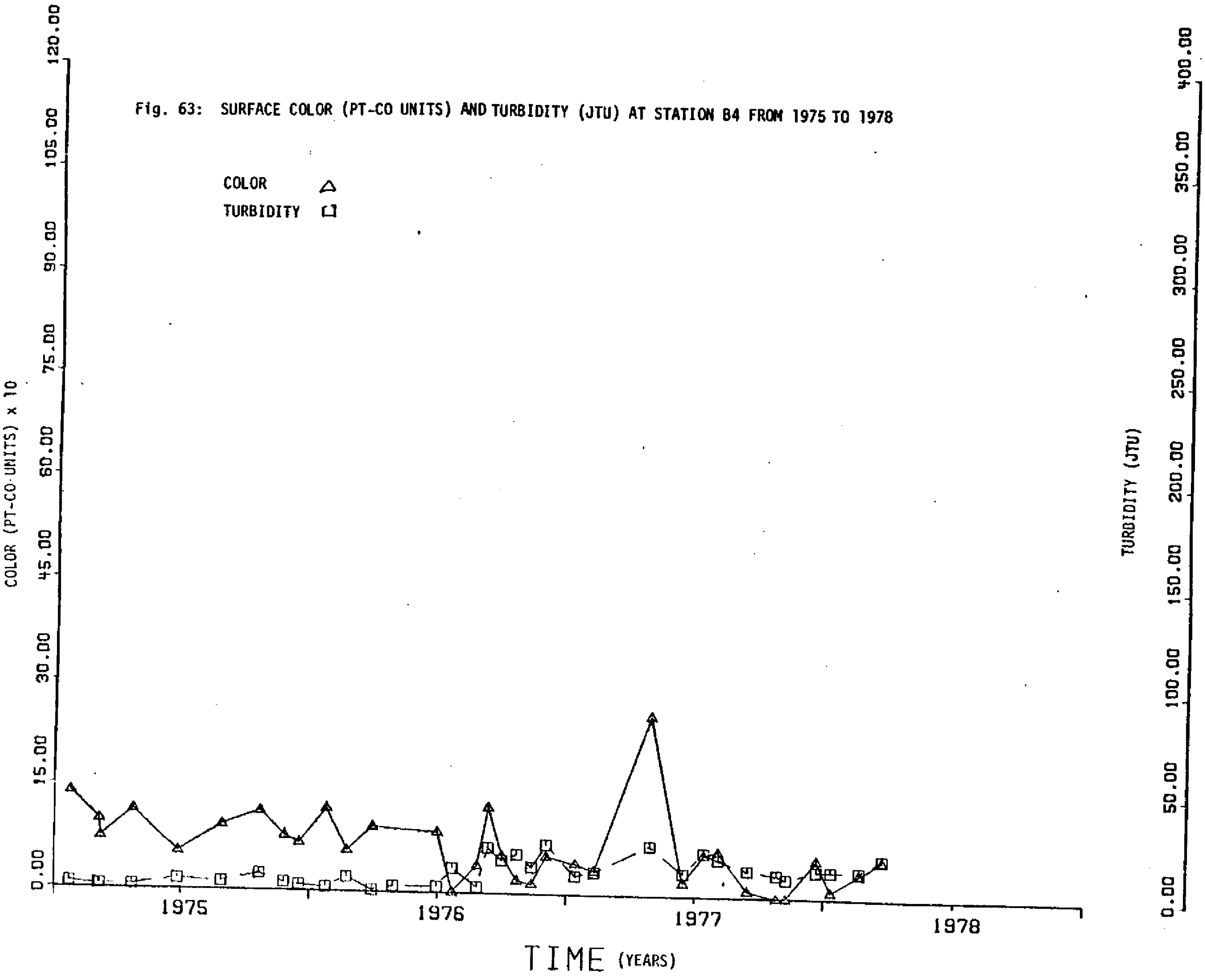


Fig. 63: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B4 FROM 1975 TO 1978

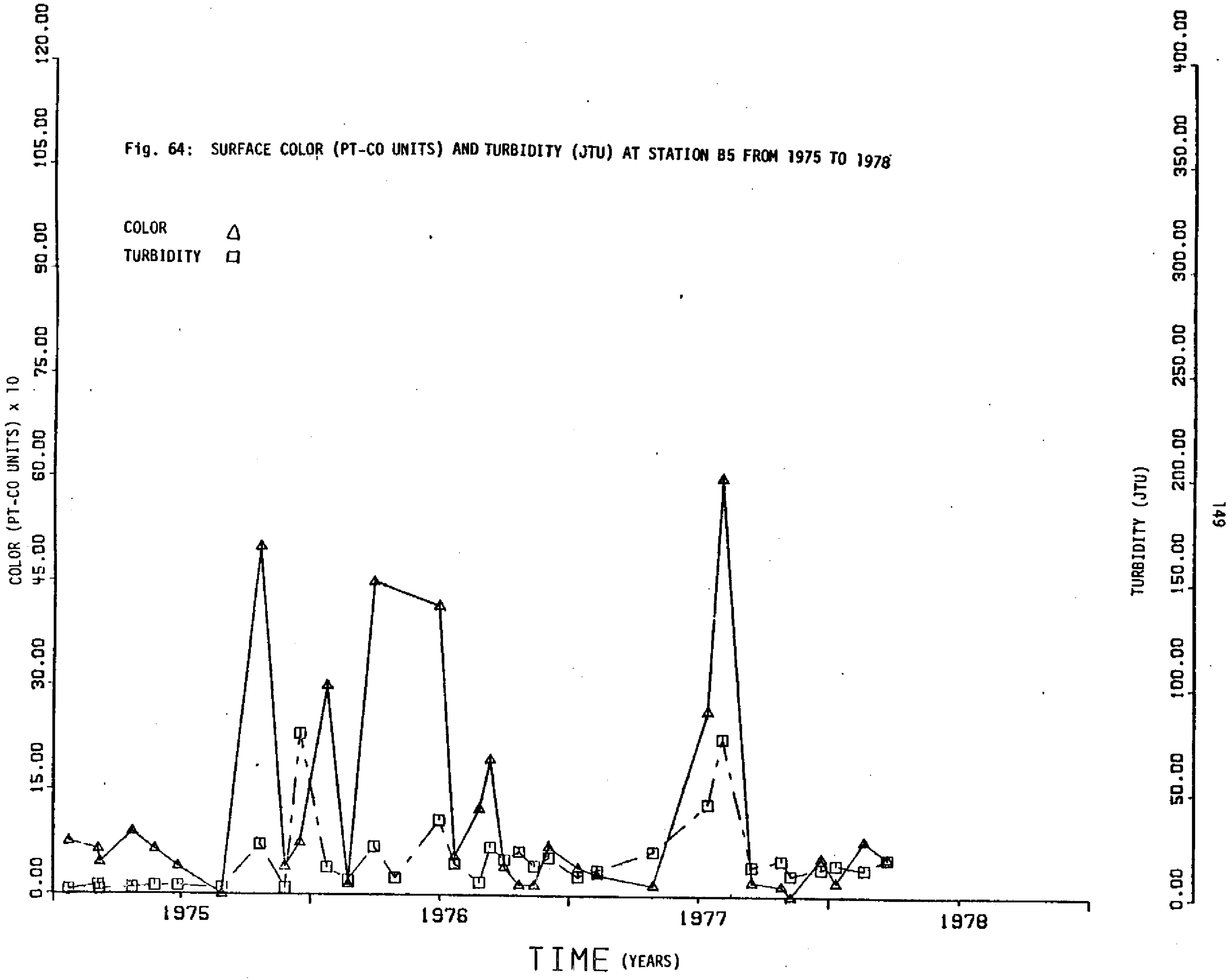


Fig. 64: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B5 FROM 1975 TO 1978

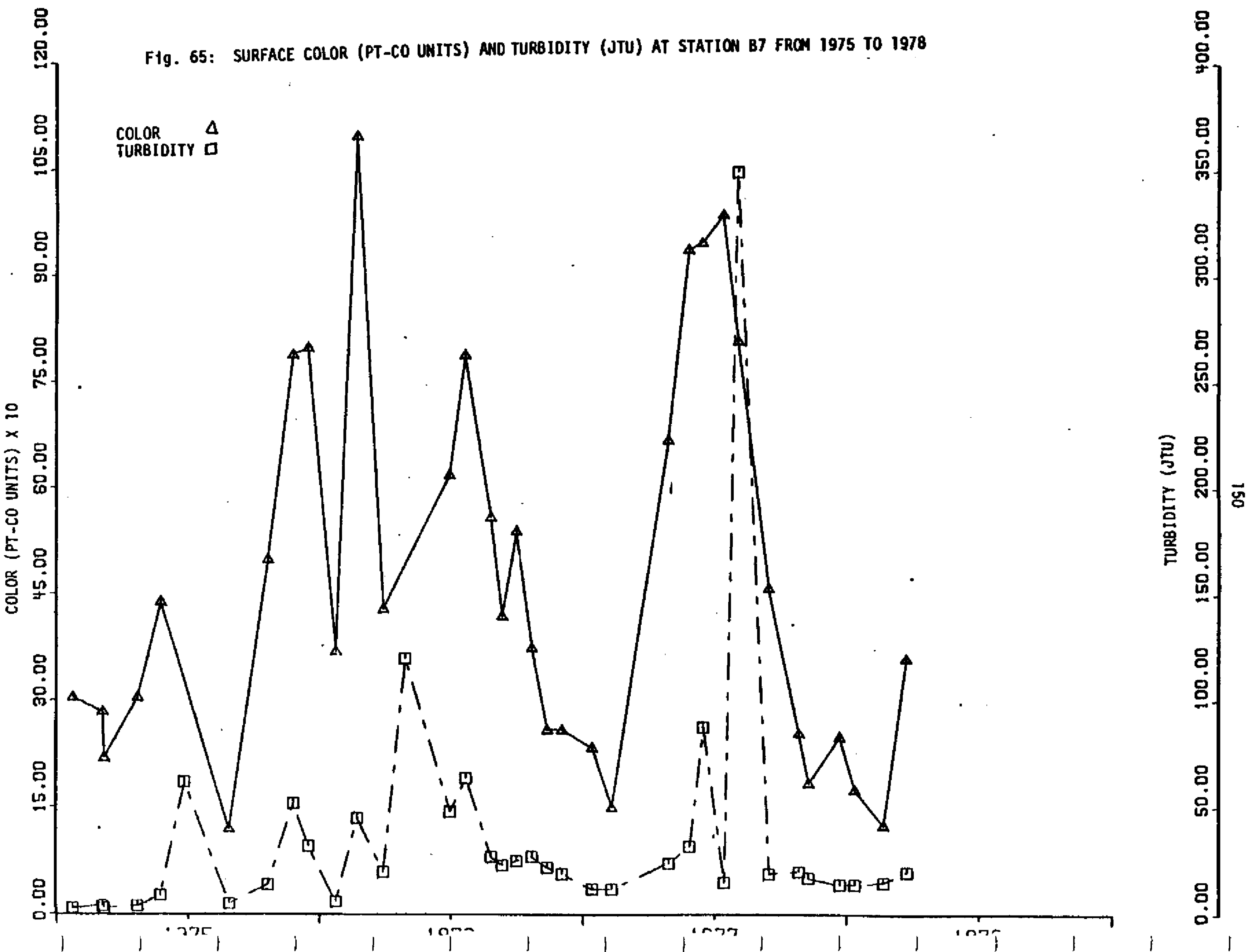


Fig. 65: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B7 FROM 1975 TO 1978

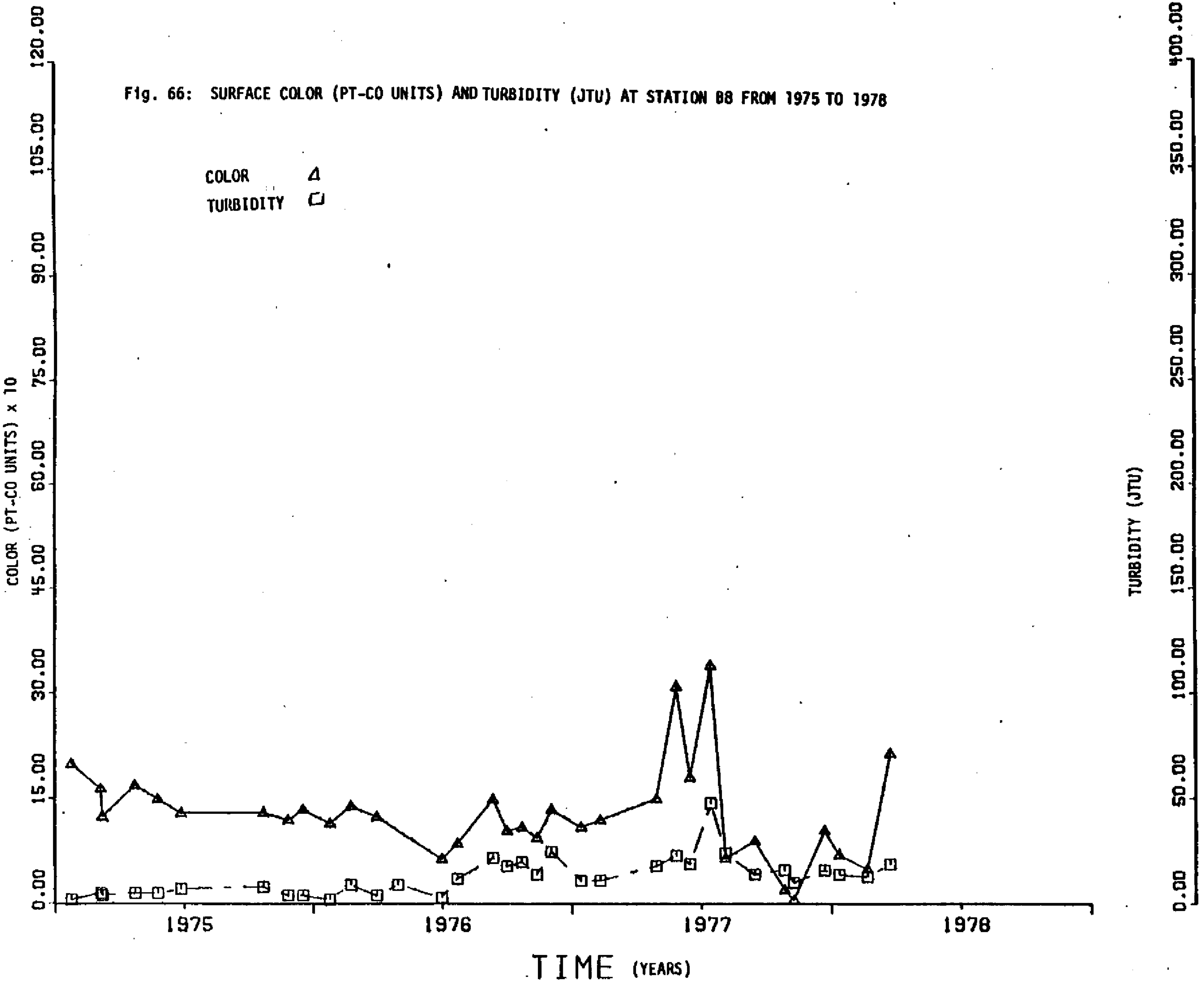


Fig. 66: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B8 FROM 1975 TO 1978

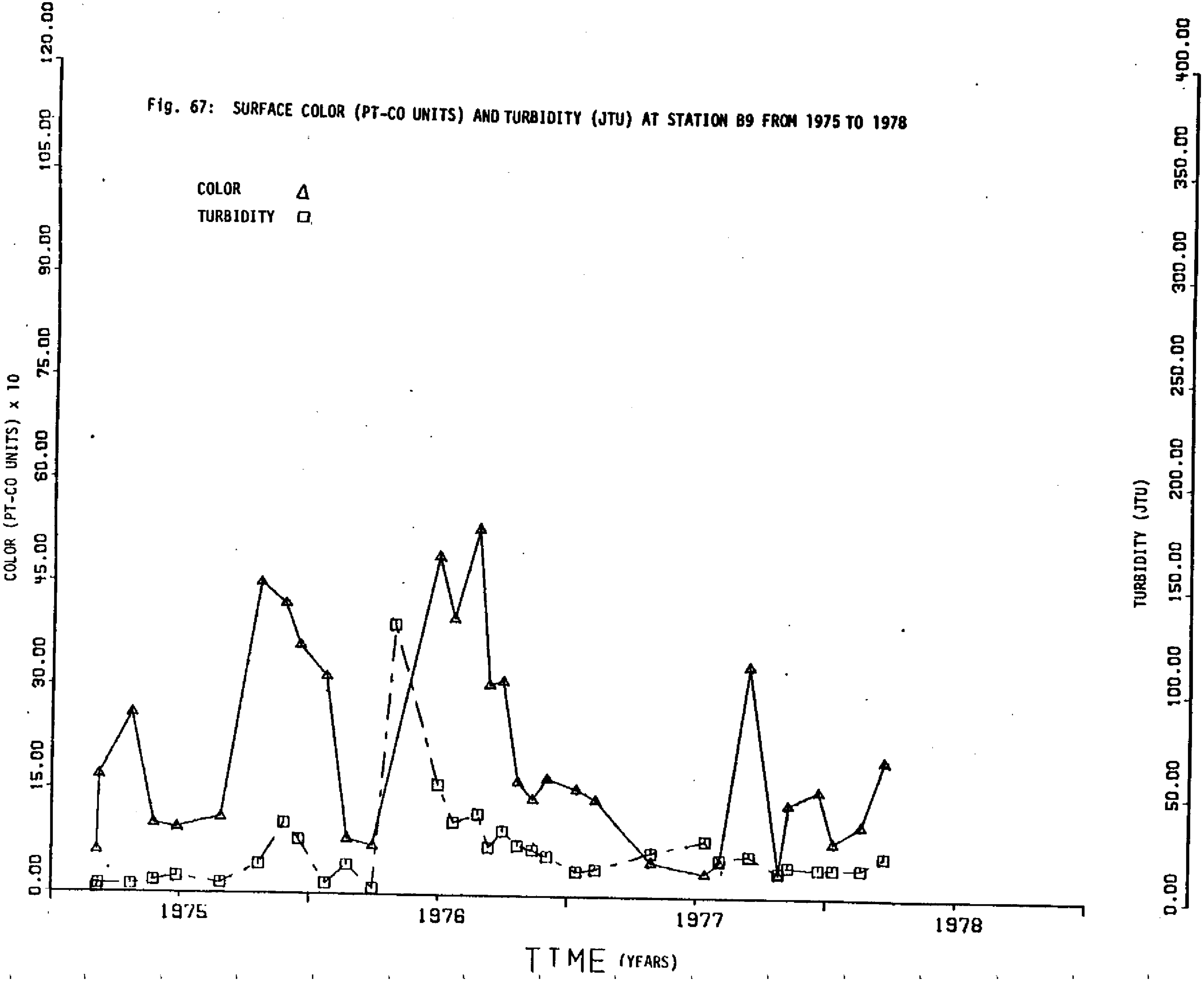


Fig. 67: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION B9 FROM 1975 TO 1978

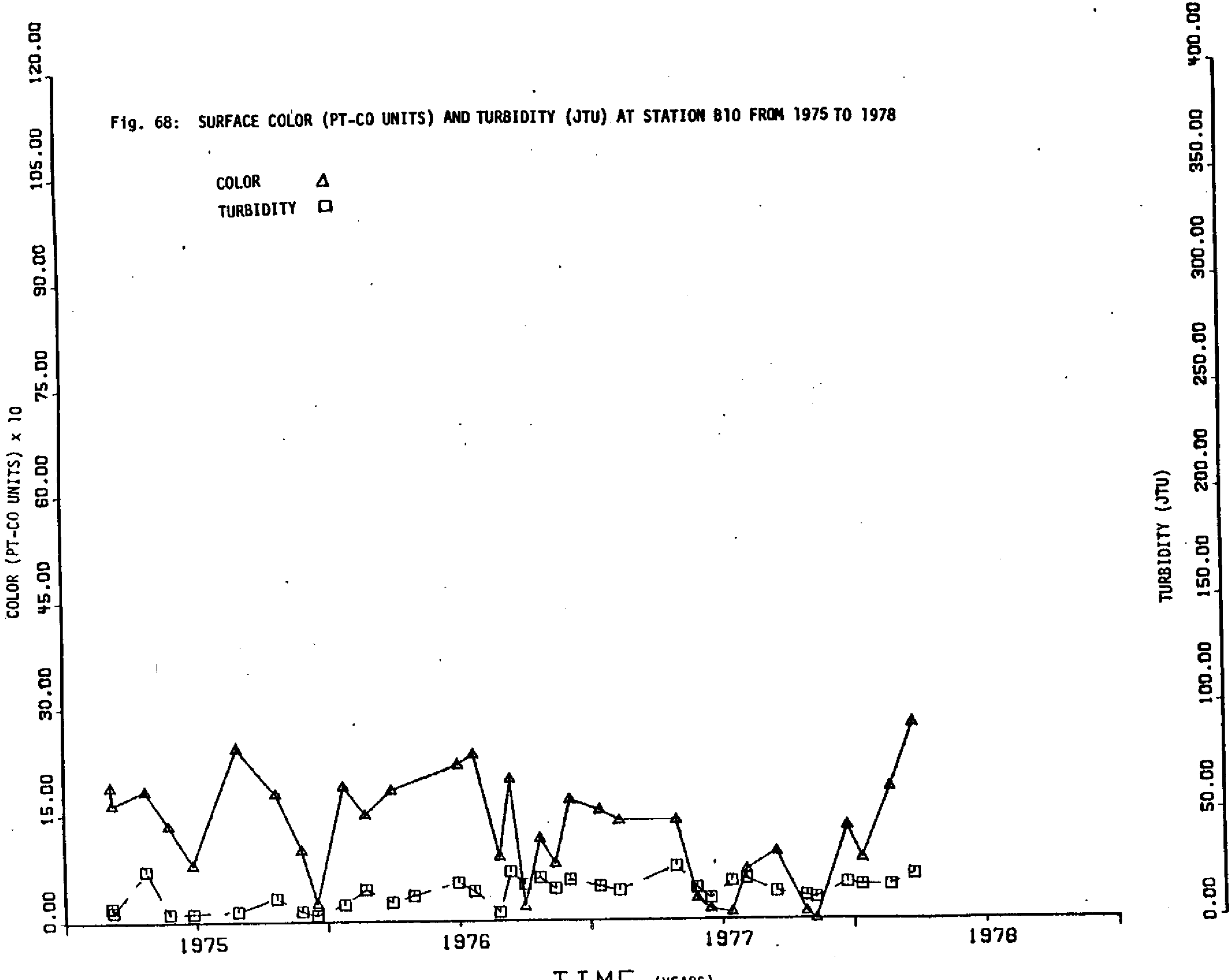
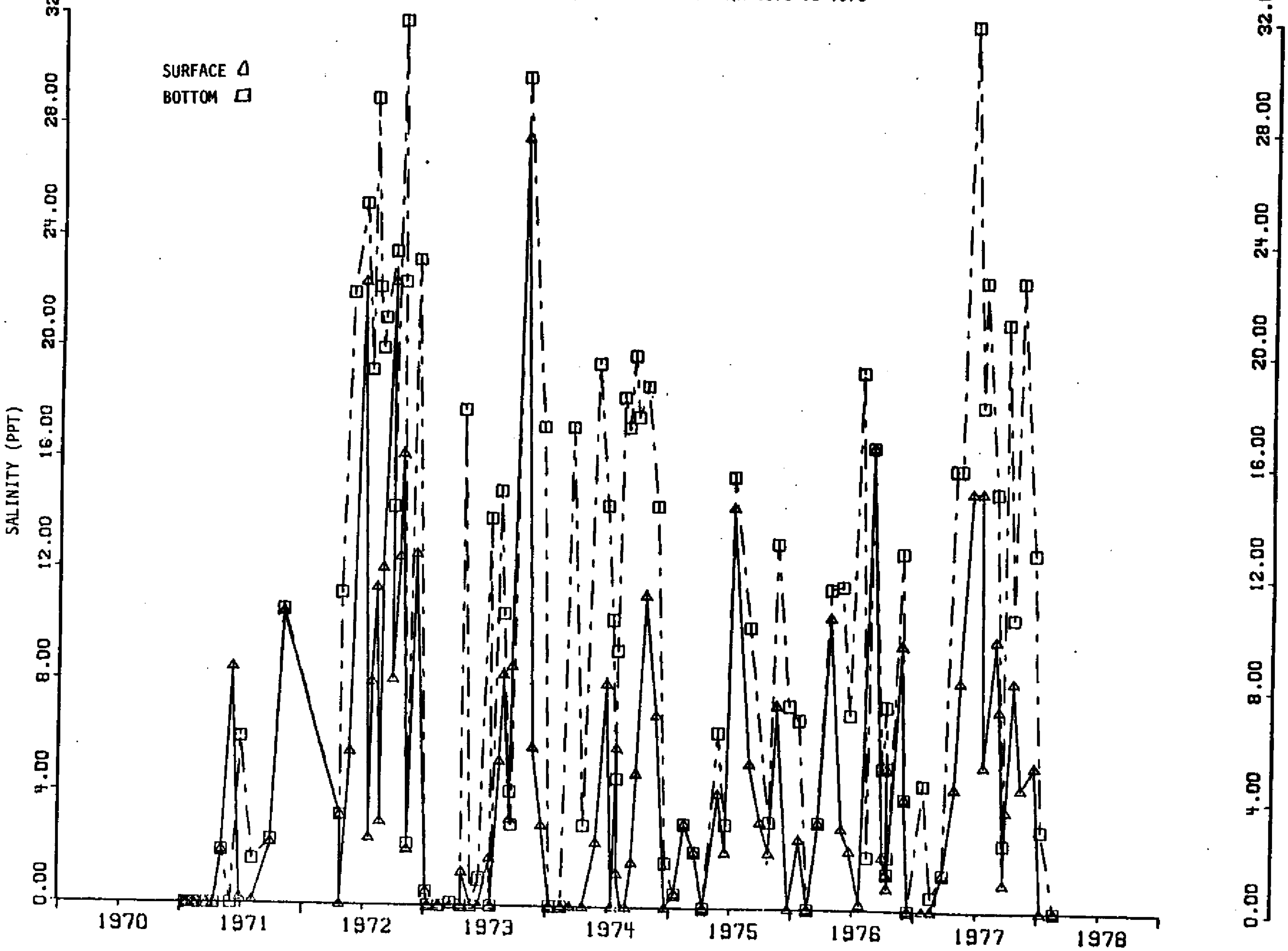


Fig. 68: SURFACE COLOR (PT-CO UNITS) AND TURBIDITY (JTU) AT STATION 810 FROM 1975 TO 1978

Fig. 69: SURFACE AND BOTTOM SALINITY (PPT) AT STATION 4 FROM 1970 TO 1978



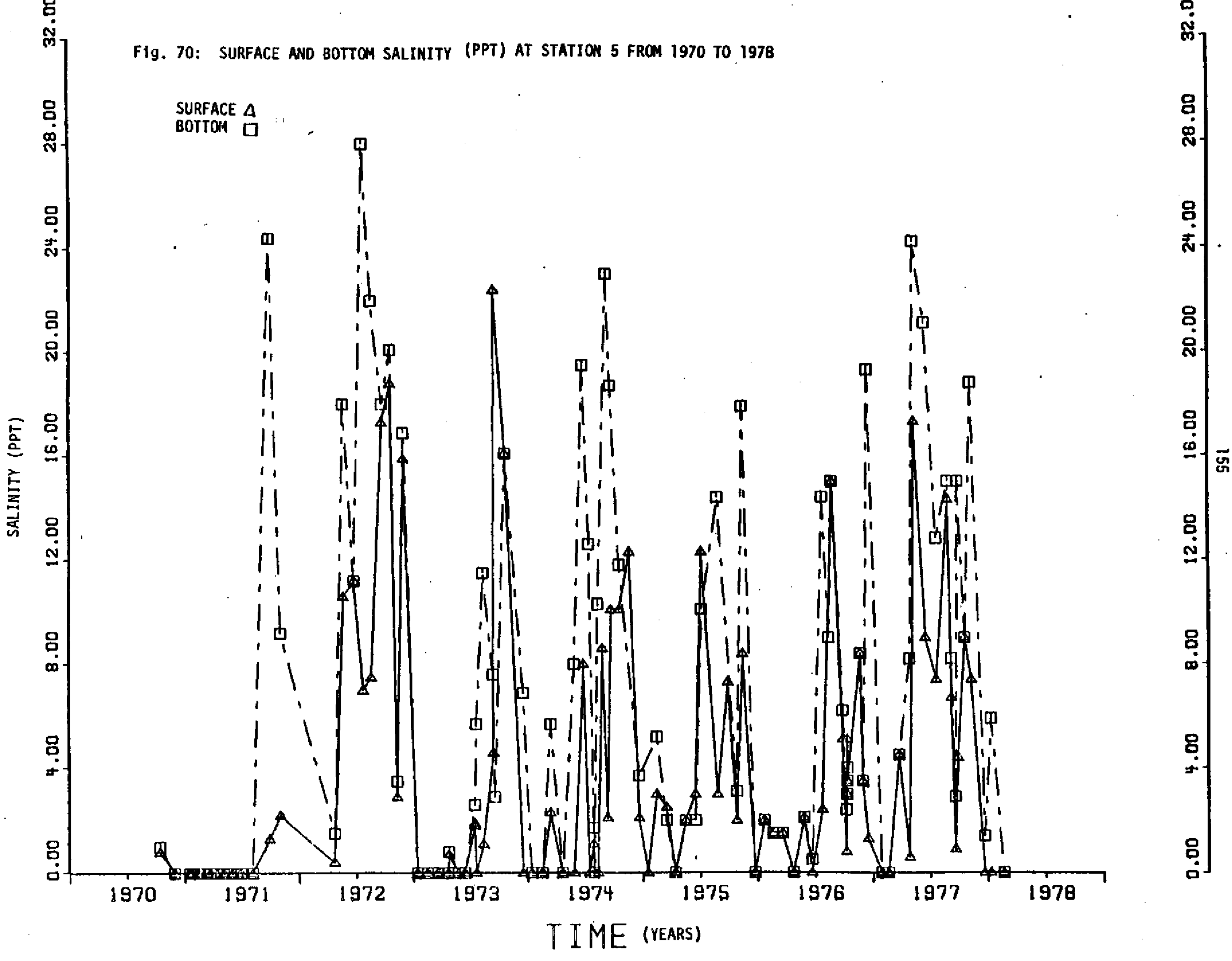


Fig. 70: SURFACE AND BOTTOM SALINITY (PPT) AT STATION 5 FROM 1970 TO 1978

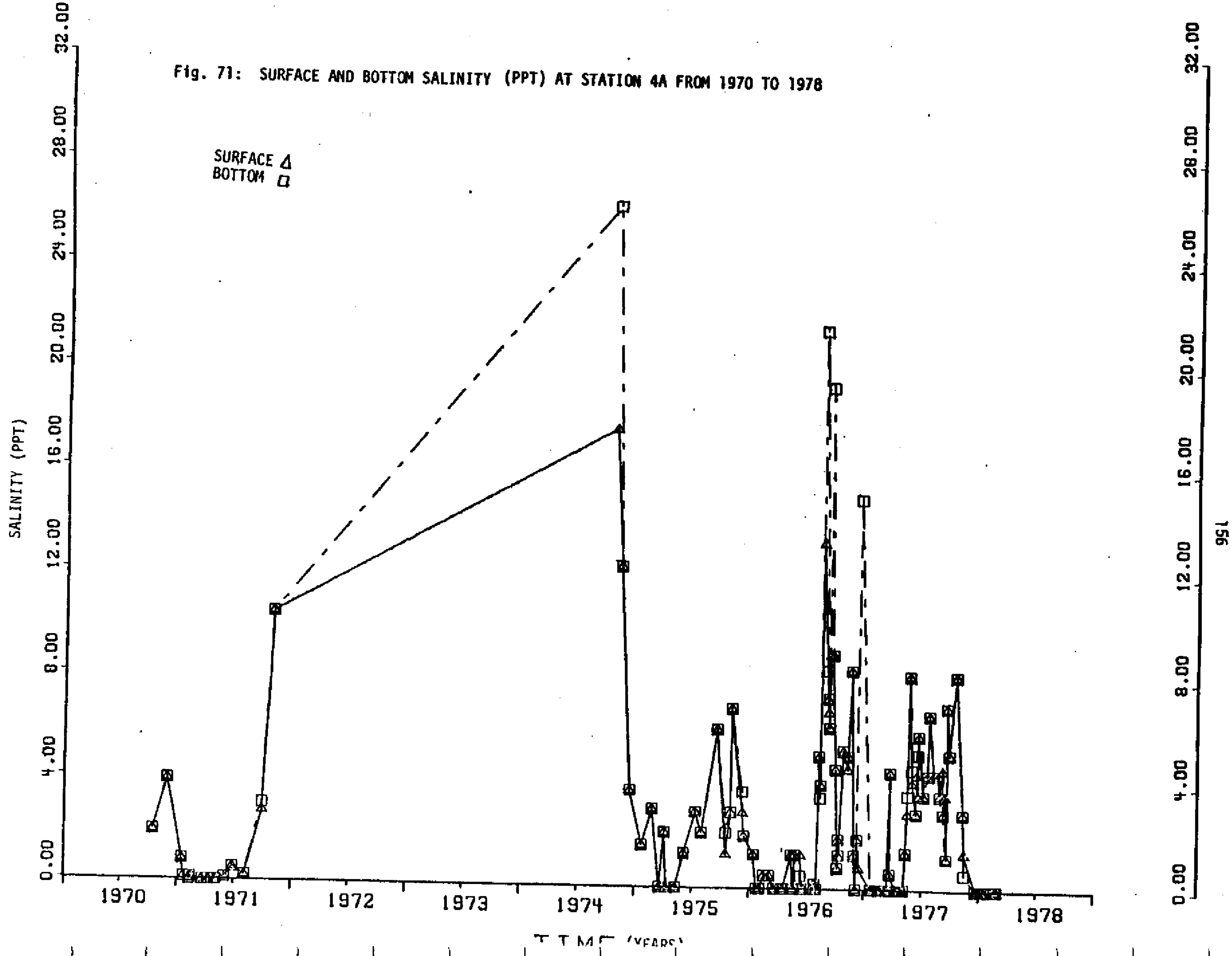


Fig. 71: SURFACE AND BOTTOM SALINITY (PPT) AT STATION 4A FROM 1970 TO 1978

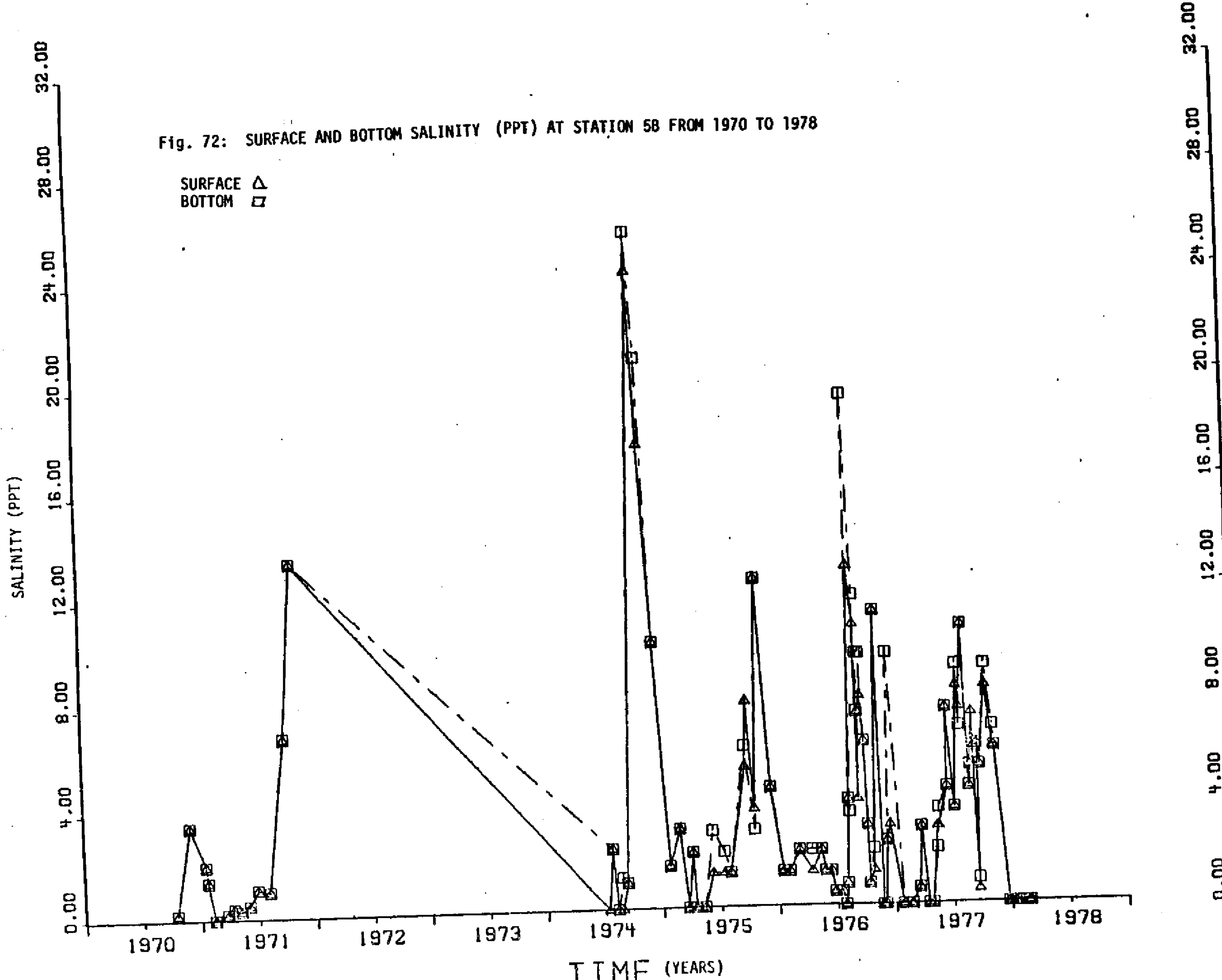


Fig. 72: SURFACE AND BOTTOM SALINITY (PPT) AT STATION 5B FROM 1970 TO 1978

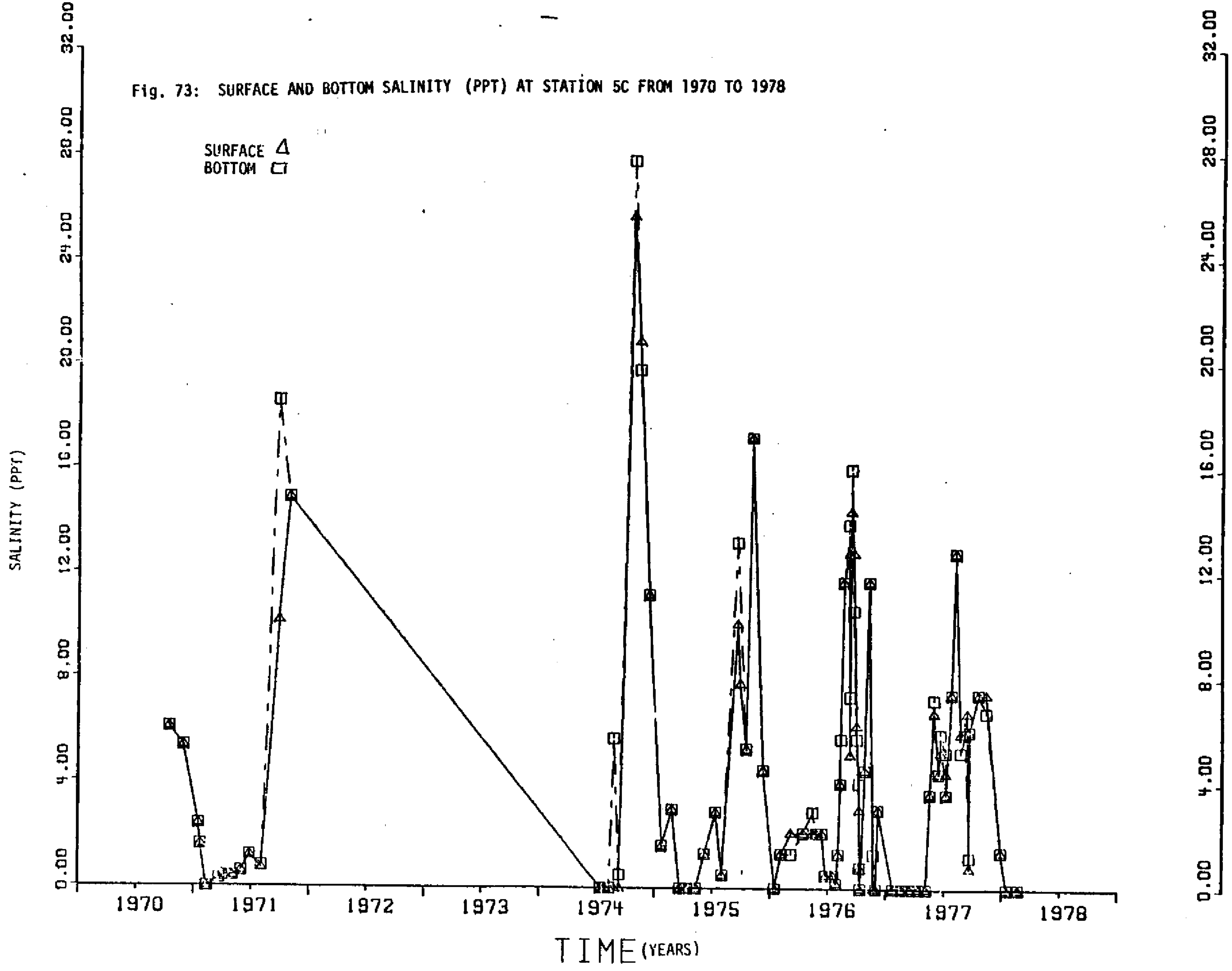


Fig. 73: SURFACE AND BOTTOM SALINITY (PPT) AT STATION 5C FROM 1970 TO 1978

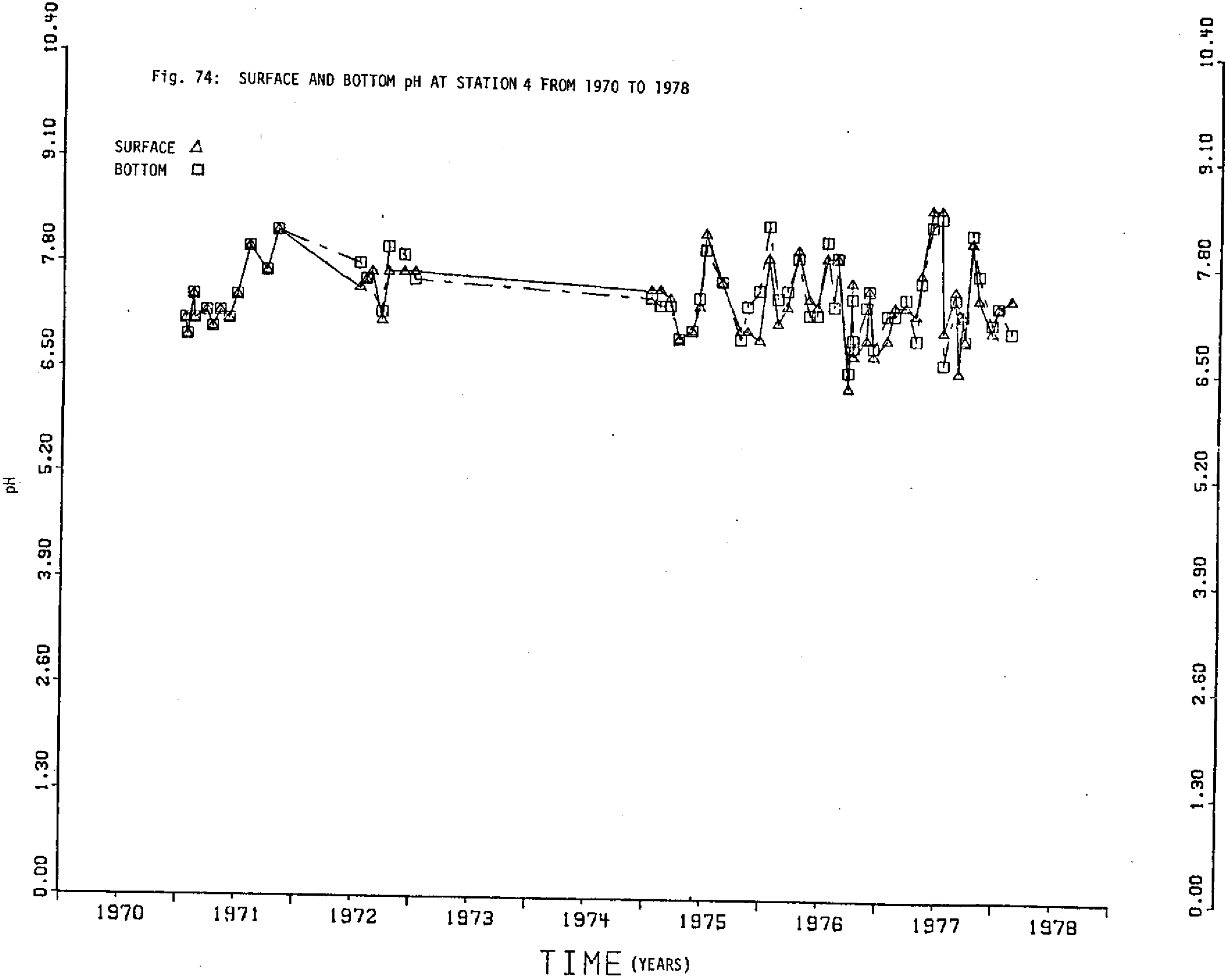


Fig. 74: SURFACE AND BOTTOM pH AT STATION 4 FROM 1970 TO 1978

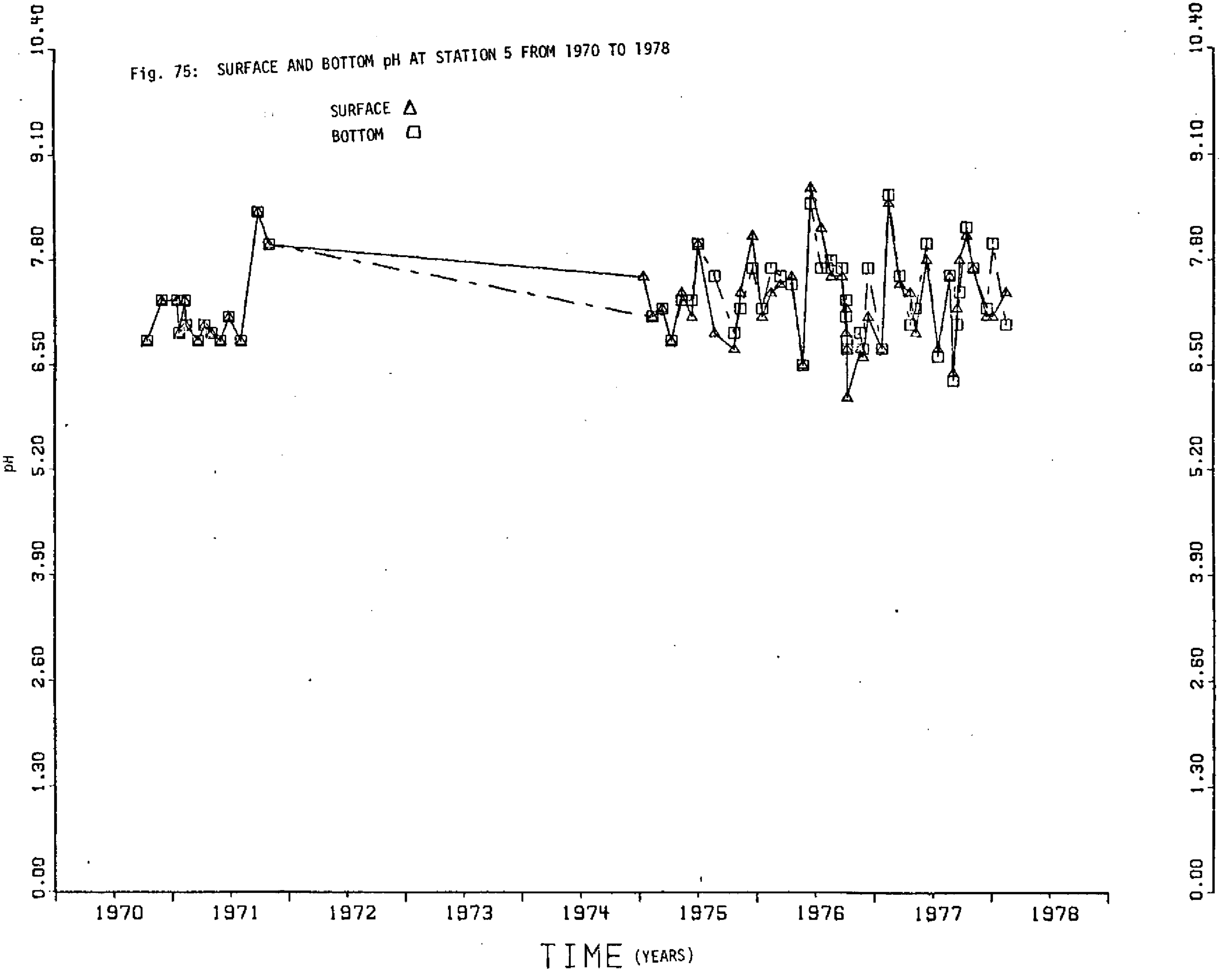


Fig. 75: SURFACE AND BOTTOM pH AT STATION 5 FROM 1970 TO 1978

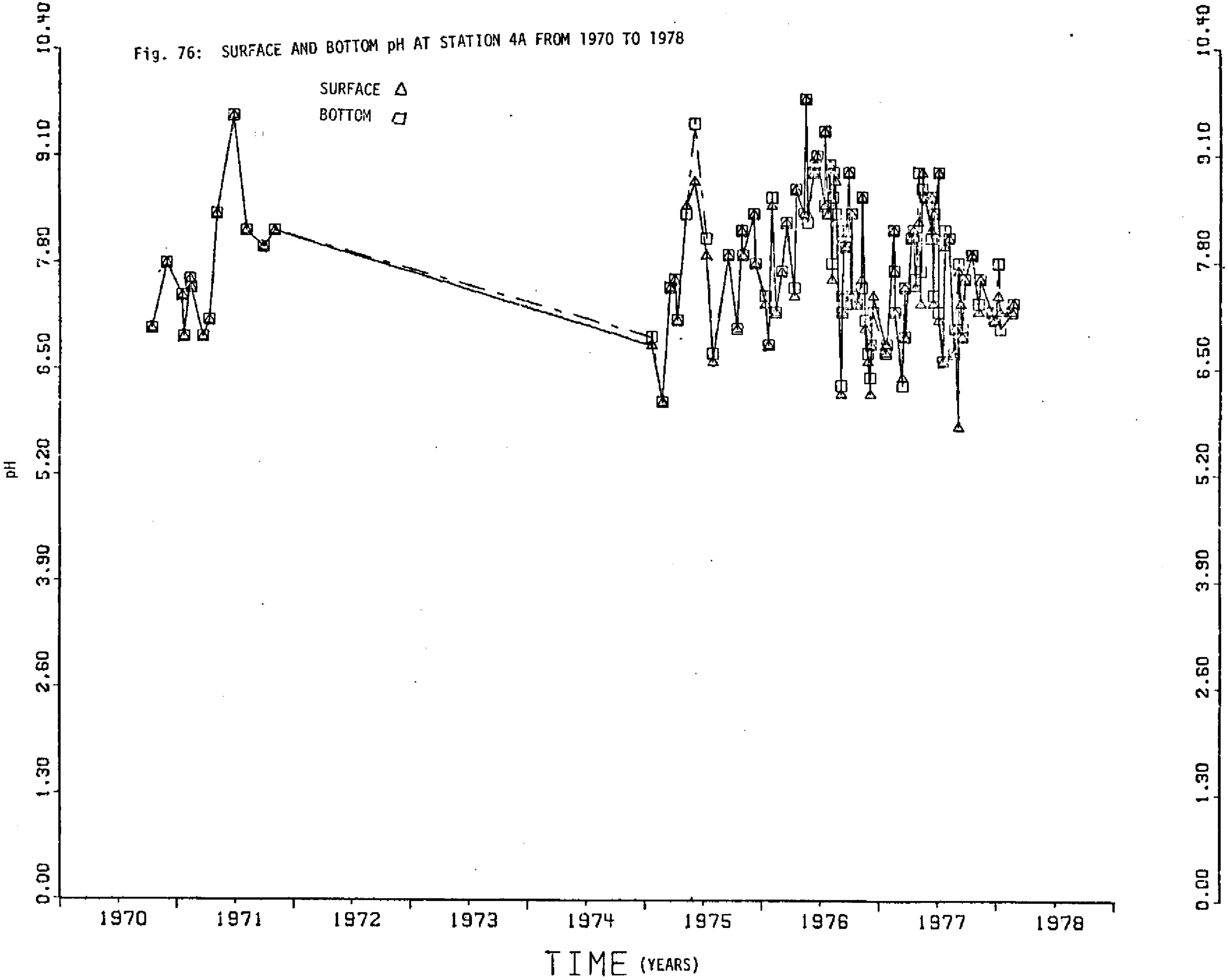


Fig. 76: SURFACE AND BOTTOM pH AT STATION 4A FROM 1970 TO 1978

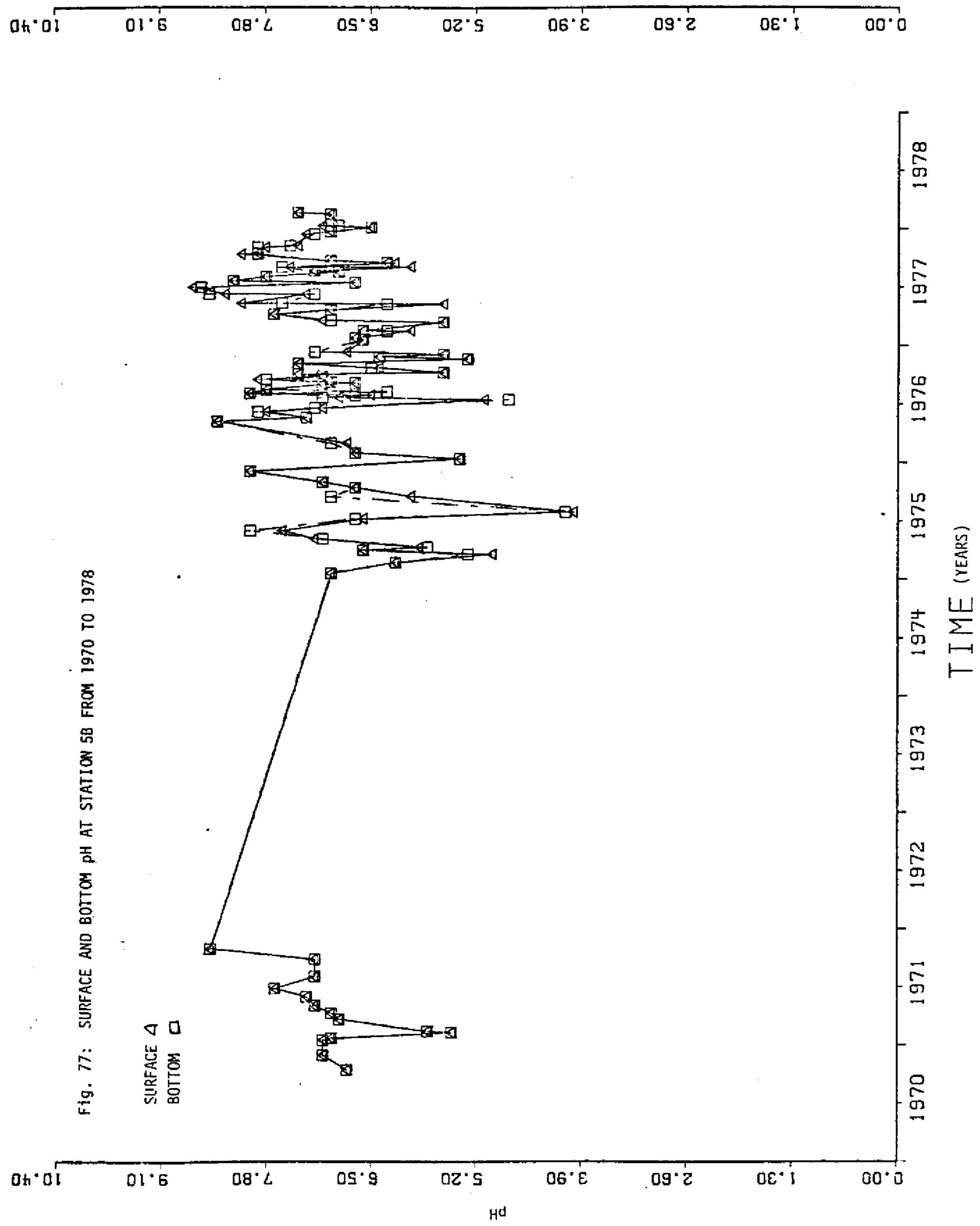


Fig. 77: SURFACE AND BOTTOM pH AT STATION 5B FROM 1970 TO 1978

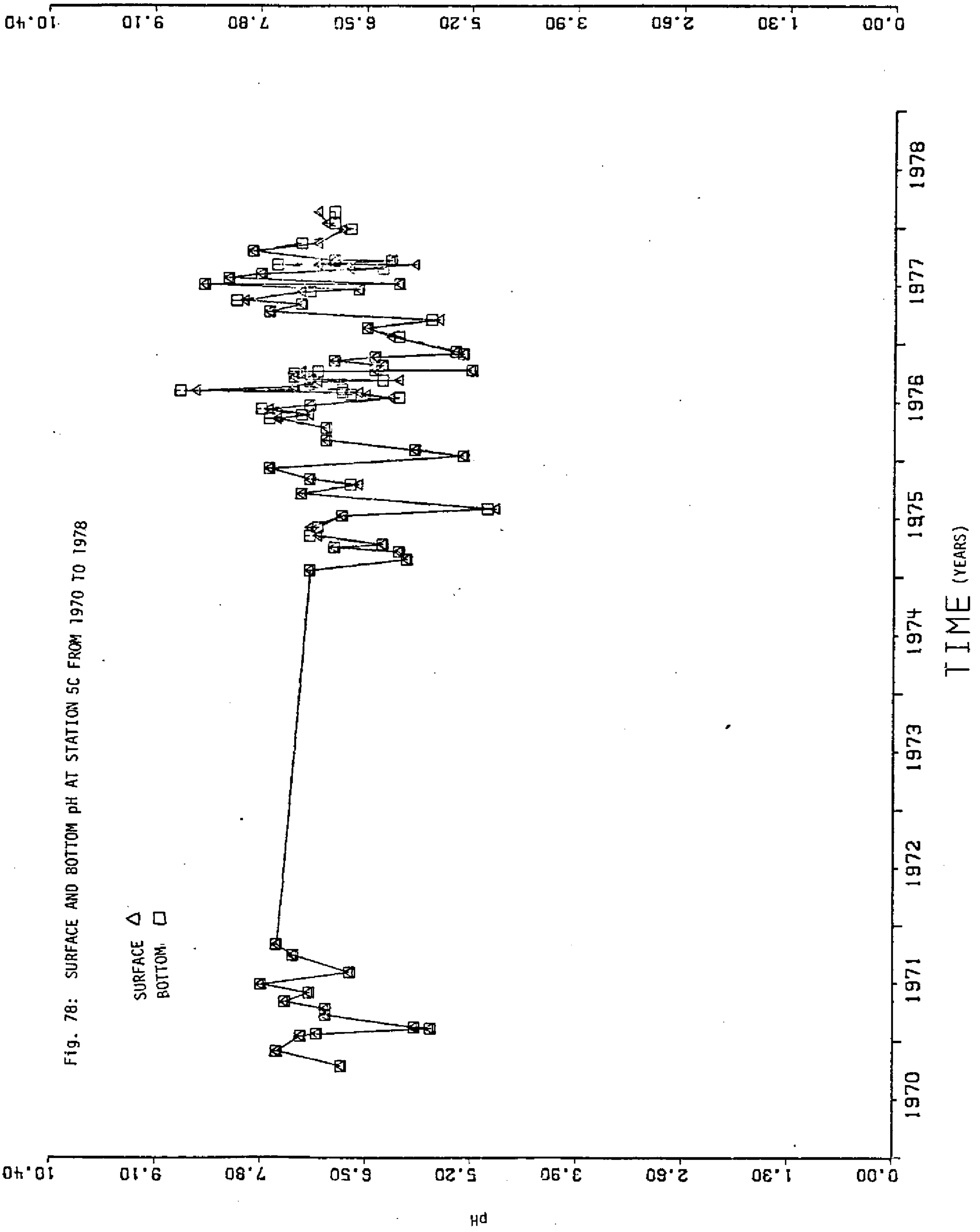


Fig. 78: SURFACE AND BOTTOM pH AT STATION 5C FROM 1970 TO 1978

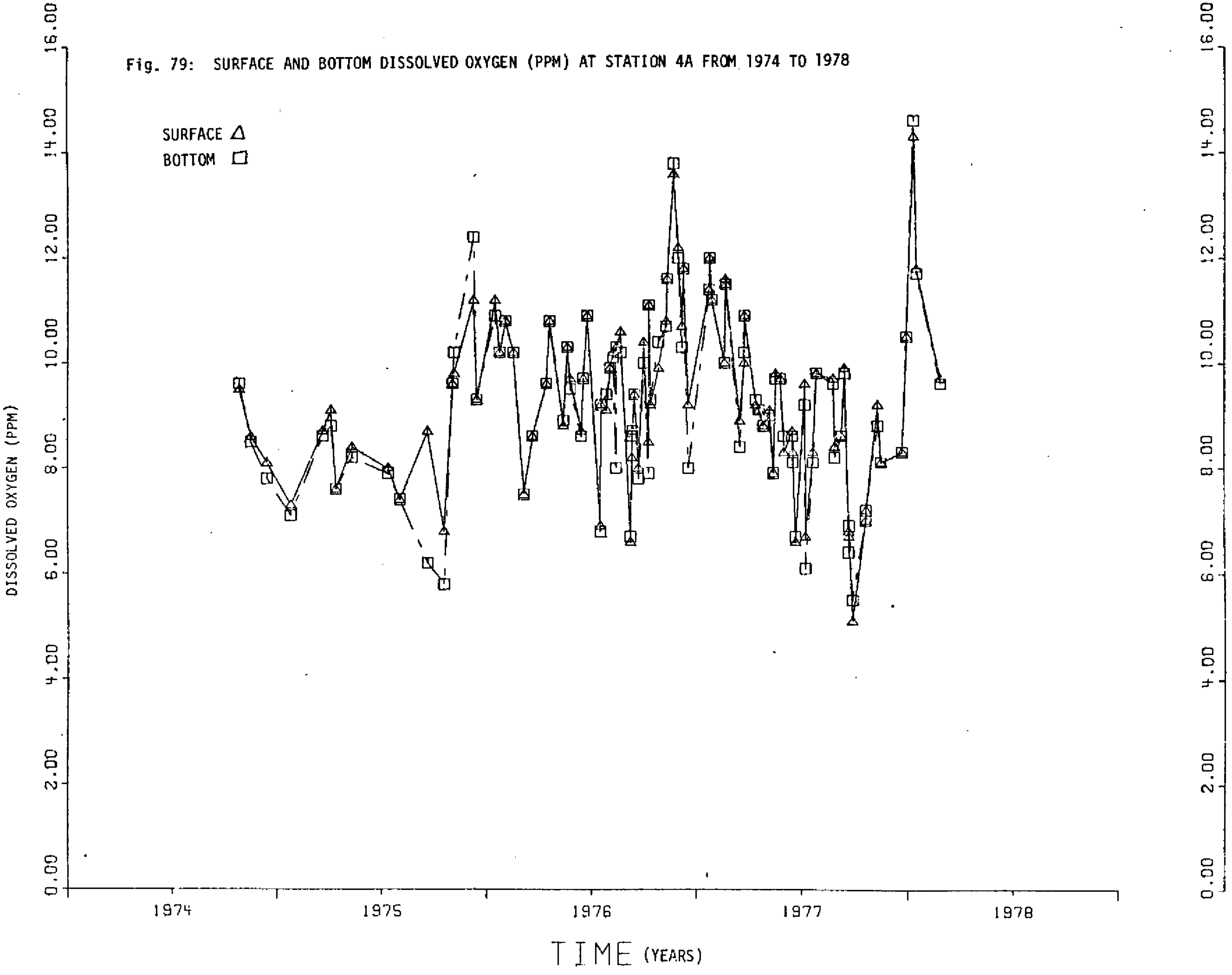


Fig. 79: SURFACE AND BOTTOM DISSOLVED OXYGEN (PPM) AT STATION 4A FROM 1974 TO 1978

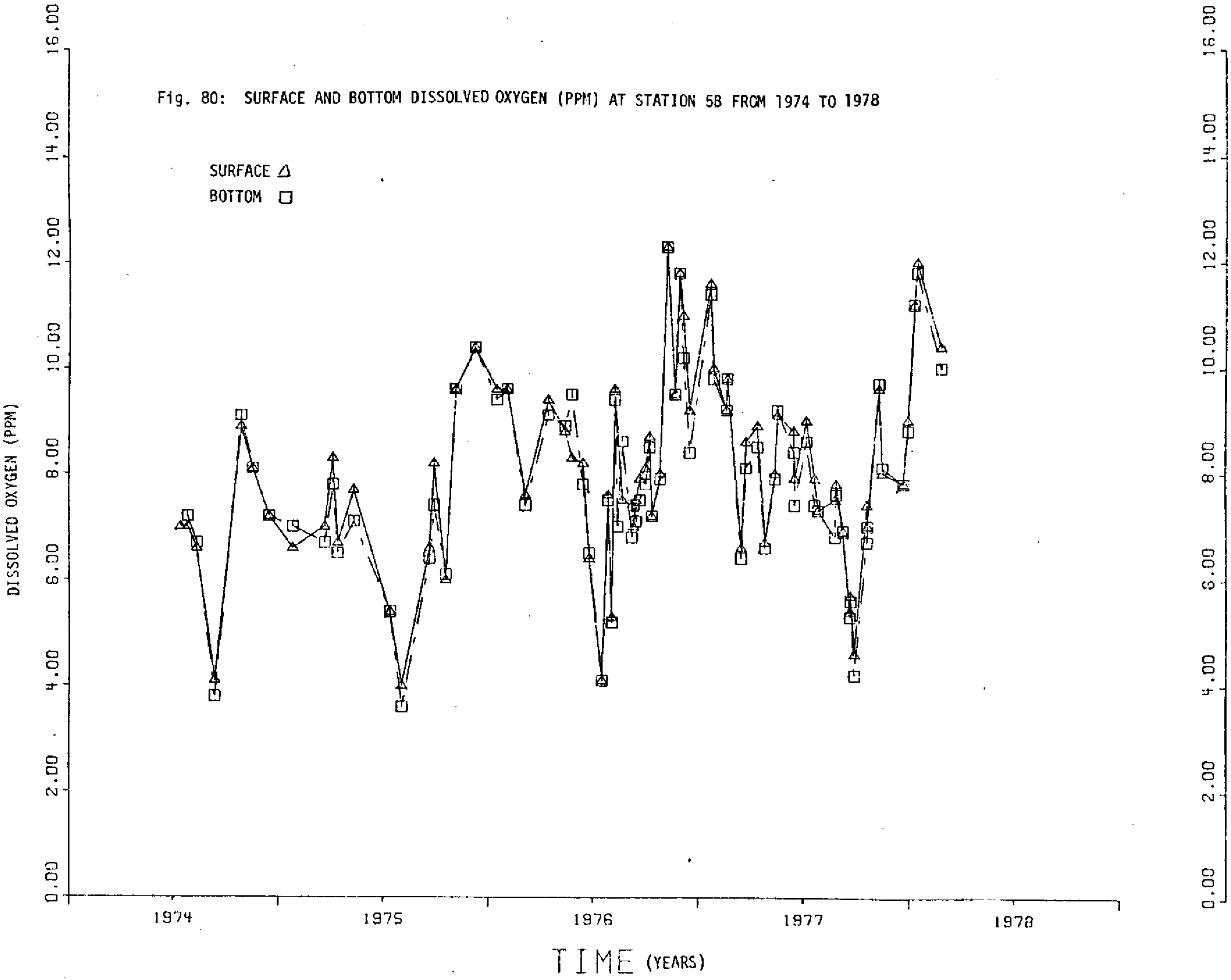


Fig. 80: SURFACE AND BOTTOM DISSOLVED OXYGEN (PPM) AT STATION 5B FROM 1974 TO 1978

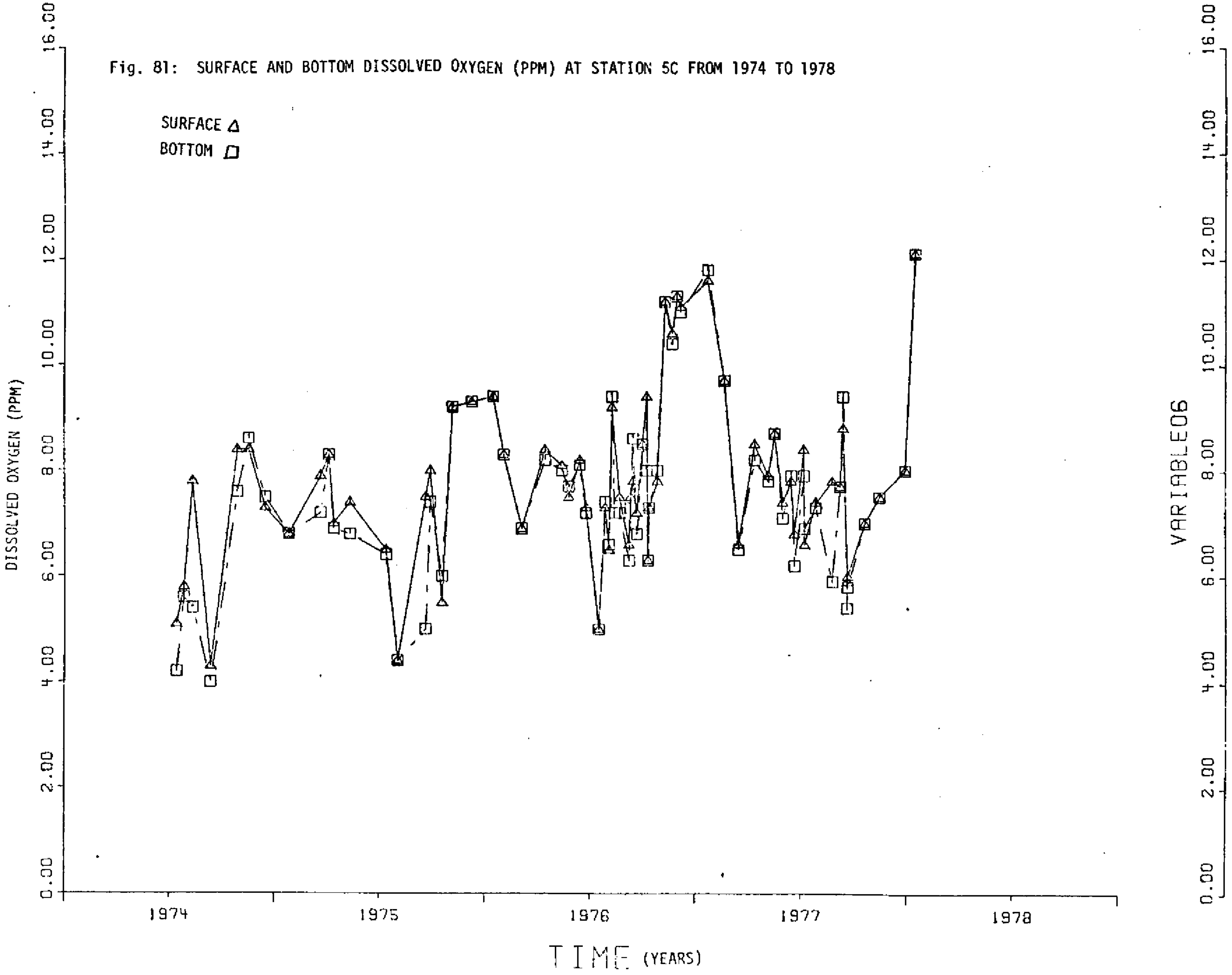


Fig. 81: SURFACE AND BOTTOM DISSOLVED OXYGEN (PPM) AT STATION 5C FROM 1974 TO 1978

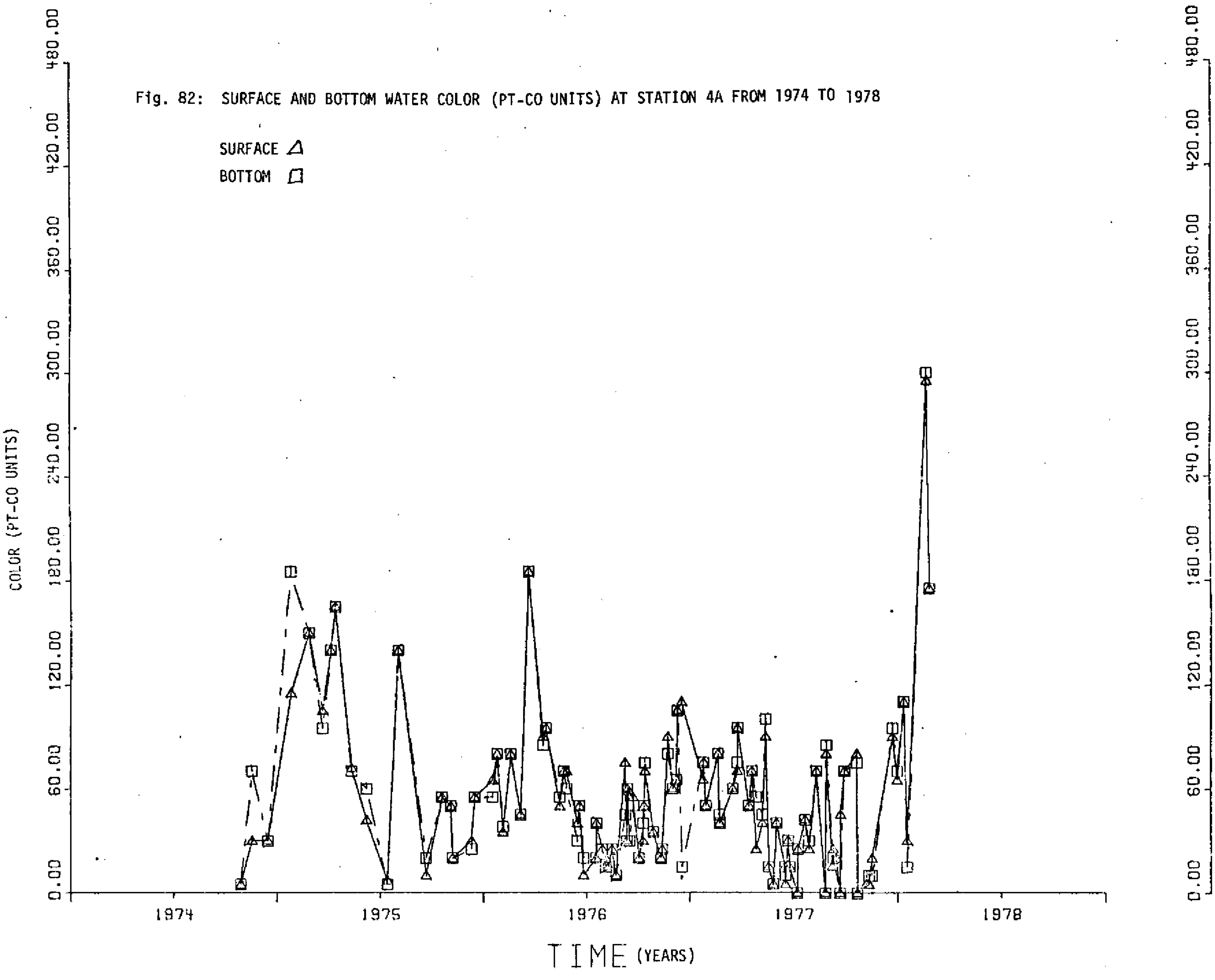


Fig. 82: SURFACE AND BOTTOM WATER COLOR (PT-CO UNITS) AT STATION 4A FROM 1974 TO 1978

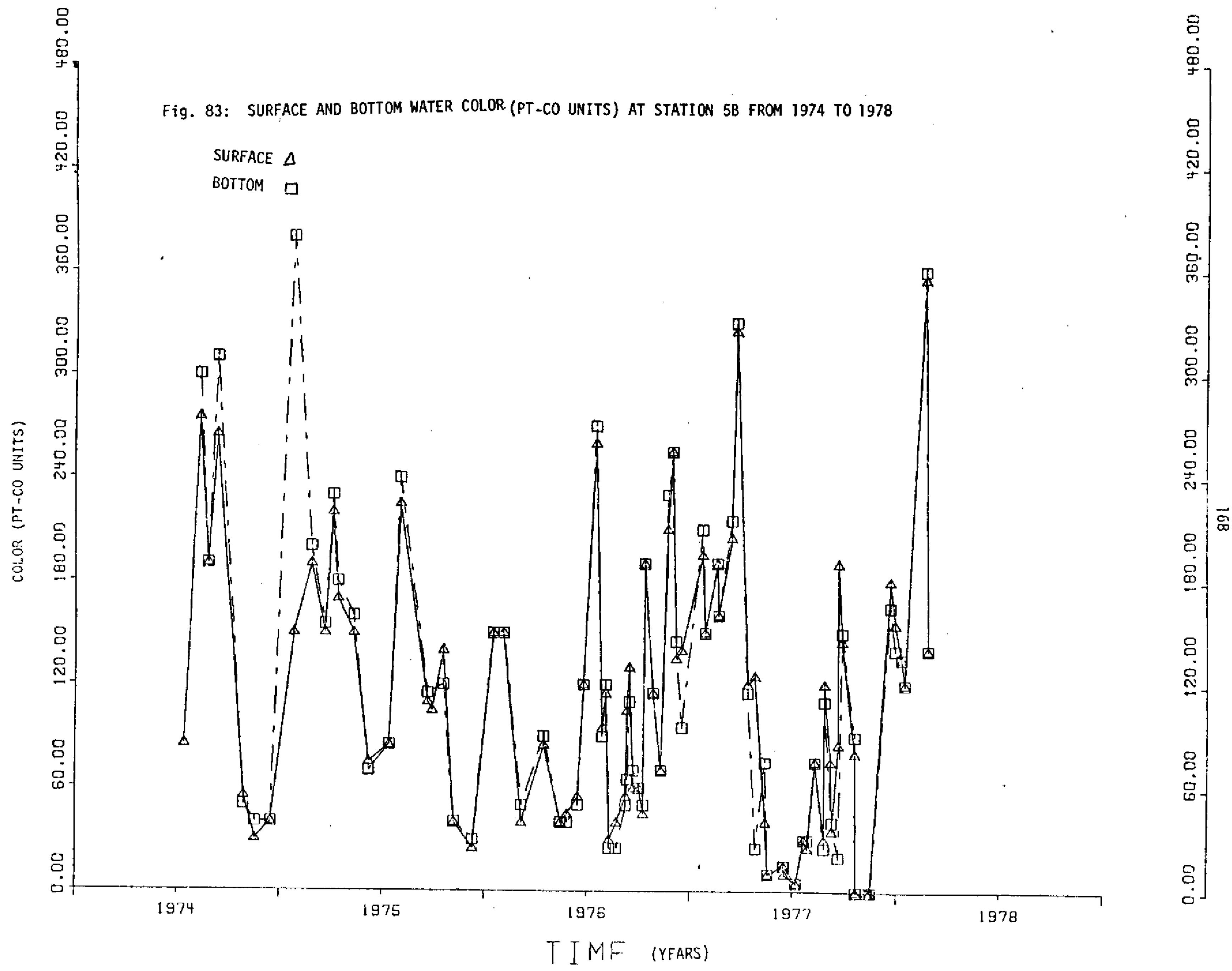
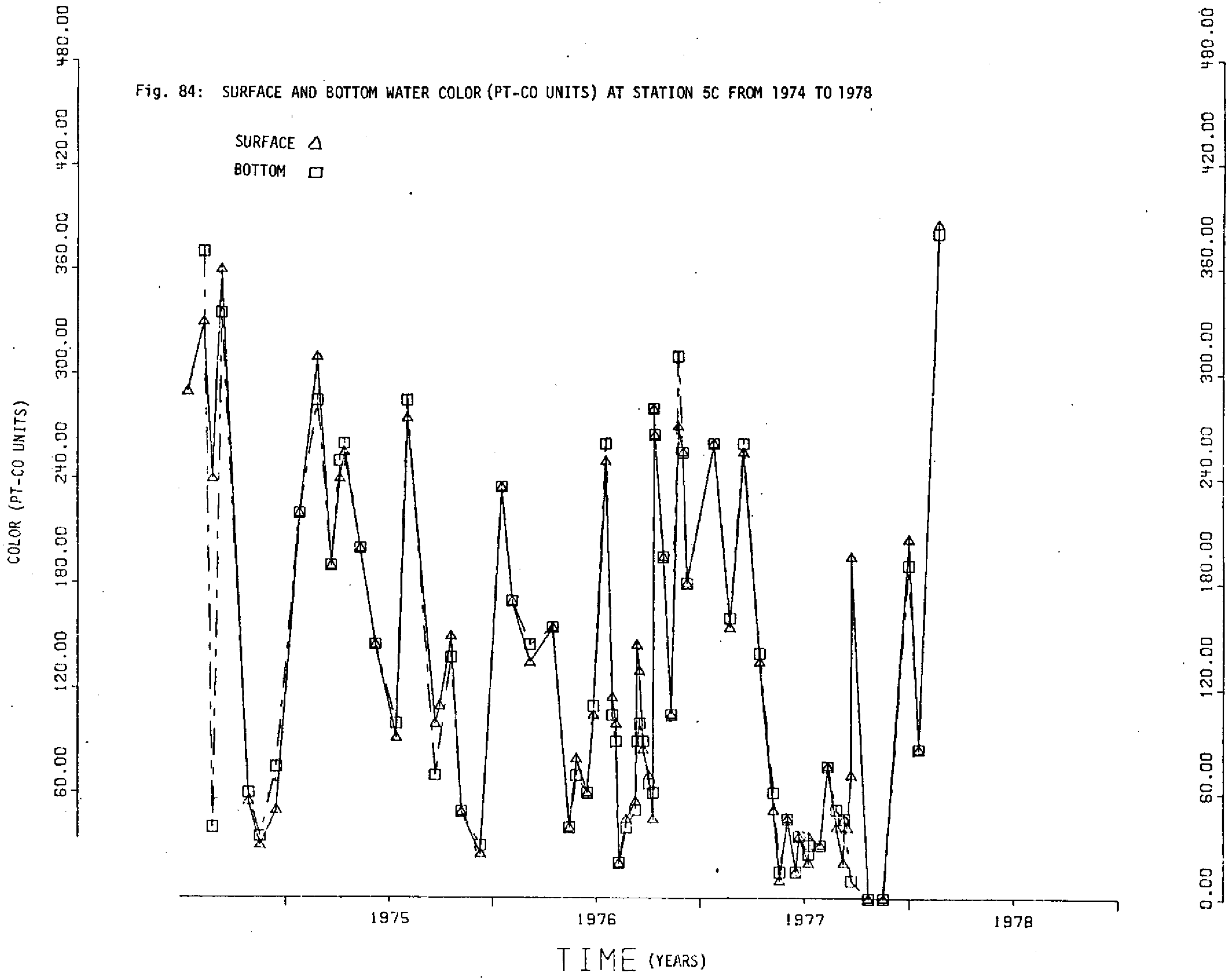


Fig. 83: SURFACE AND BOTTOM WATER COLOR (PT-CO UNITS) AT STATION 5B FROM 1974 TO 1978

Fig. 84: SURFACE AND BOTTOM WATER COLOR (PT-CO UNITS) AT STATION 5C FROM 1974 TO 1978



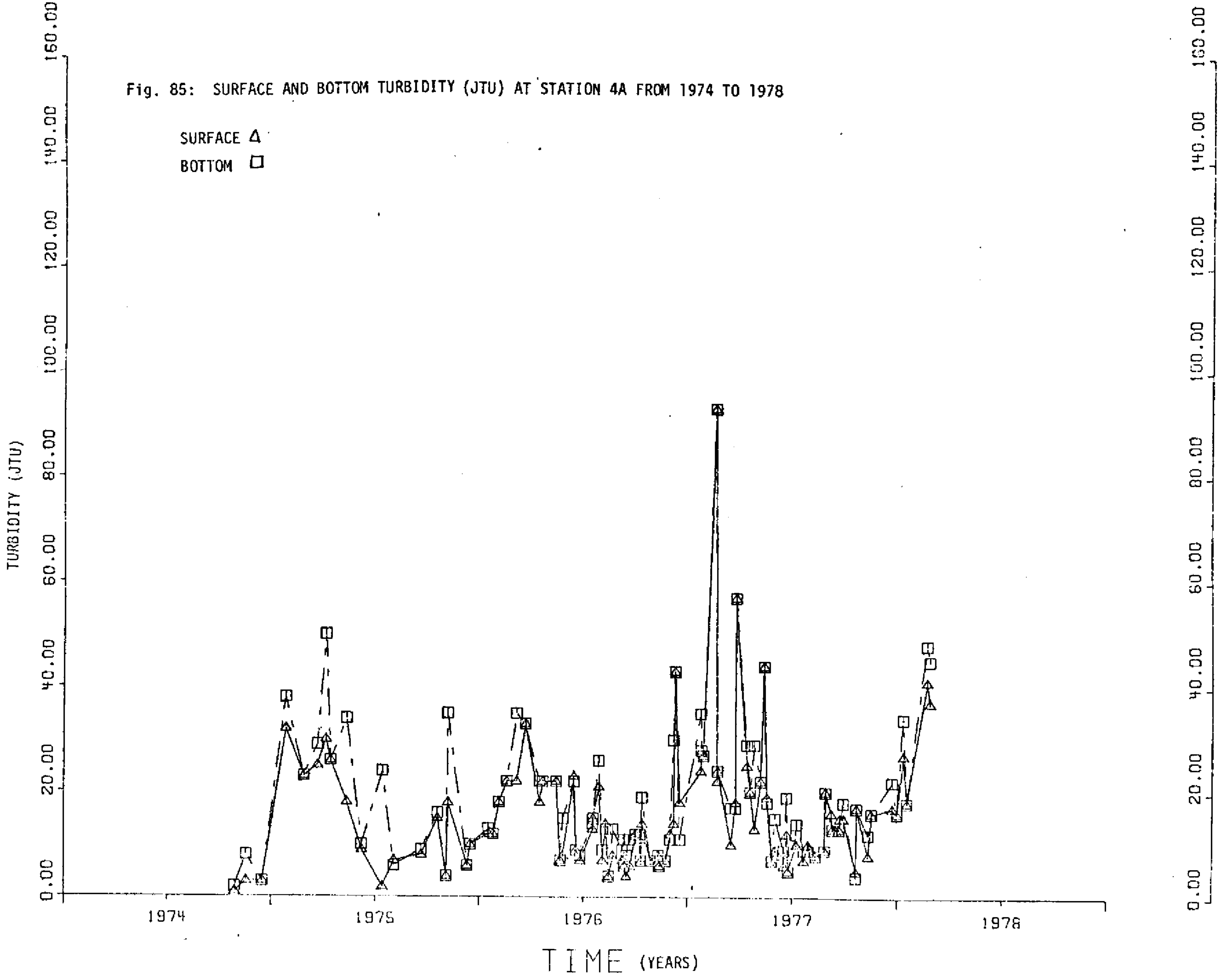


Fig. 85: SURFACE AND BOTTOM TURBIDITY (JTU) AT STATION 4A FROM 1974 TO 1978

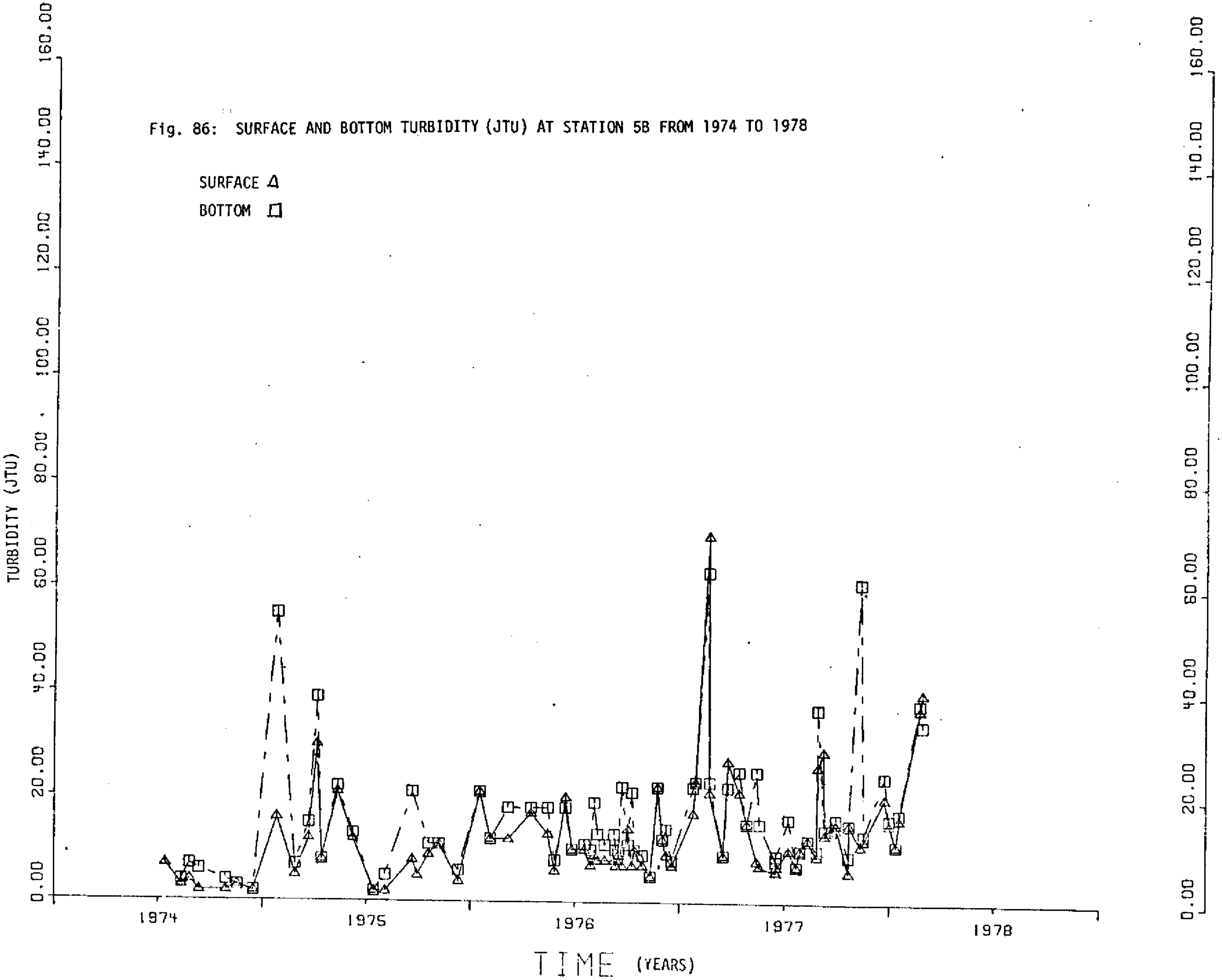


Fig. 86: SURFACE AND BOTTOM TURBIDITY (JTU) AT STATION 5B FROM 1974 TO 1978

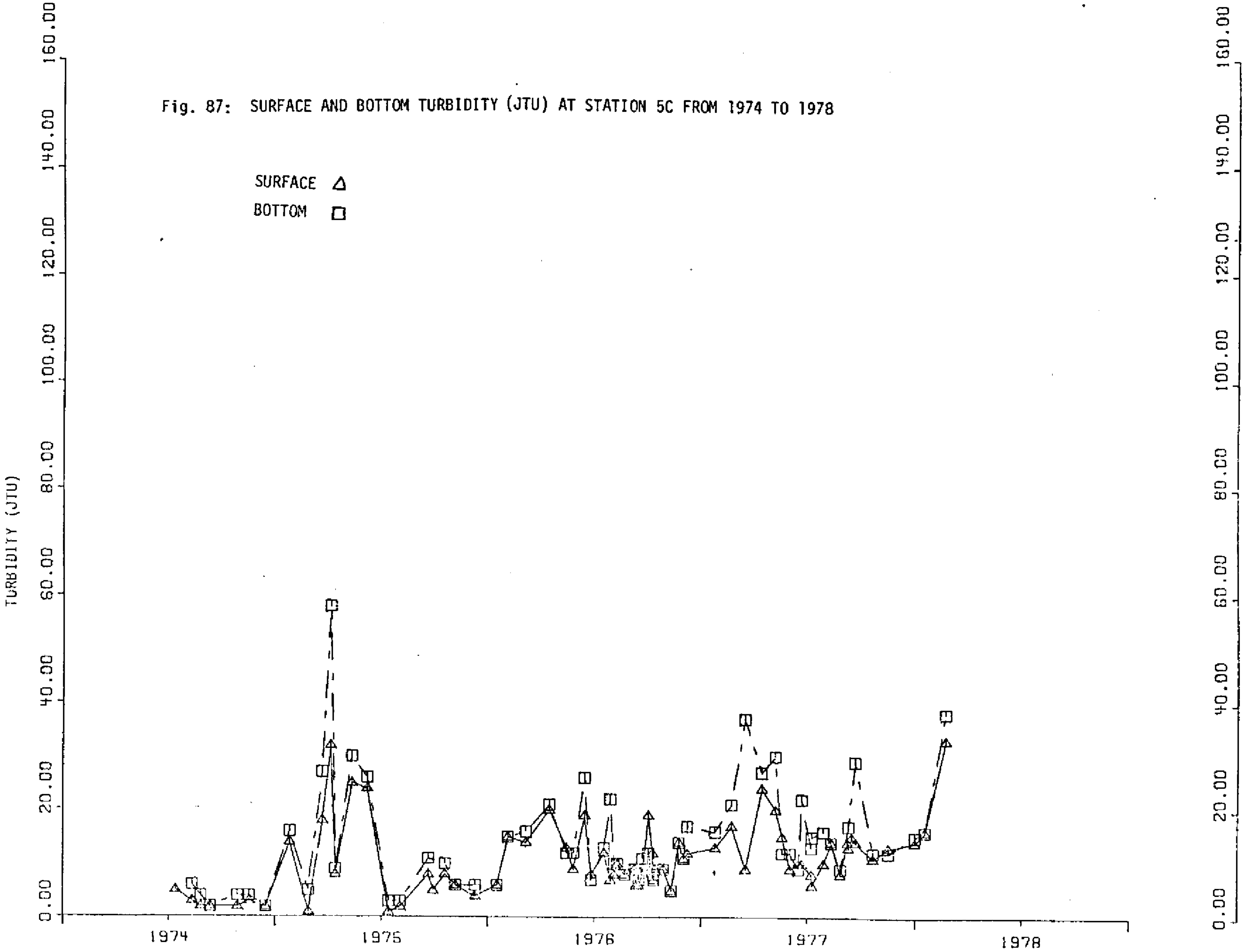


Fig. 87: SURFACE AND BOTTOM TURBIDITY (JTU) AT STATION 5C FROM 1974 TO 1978

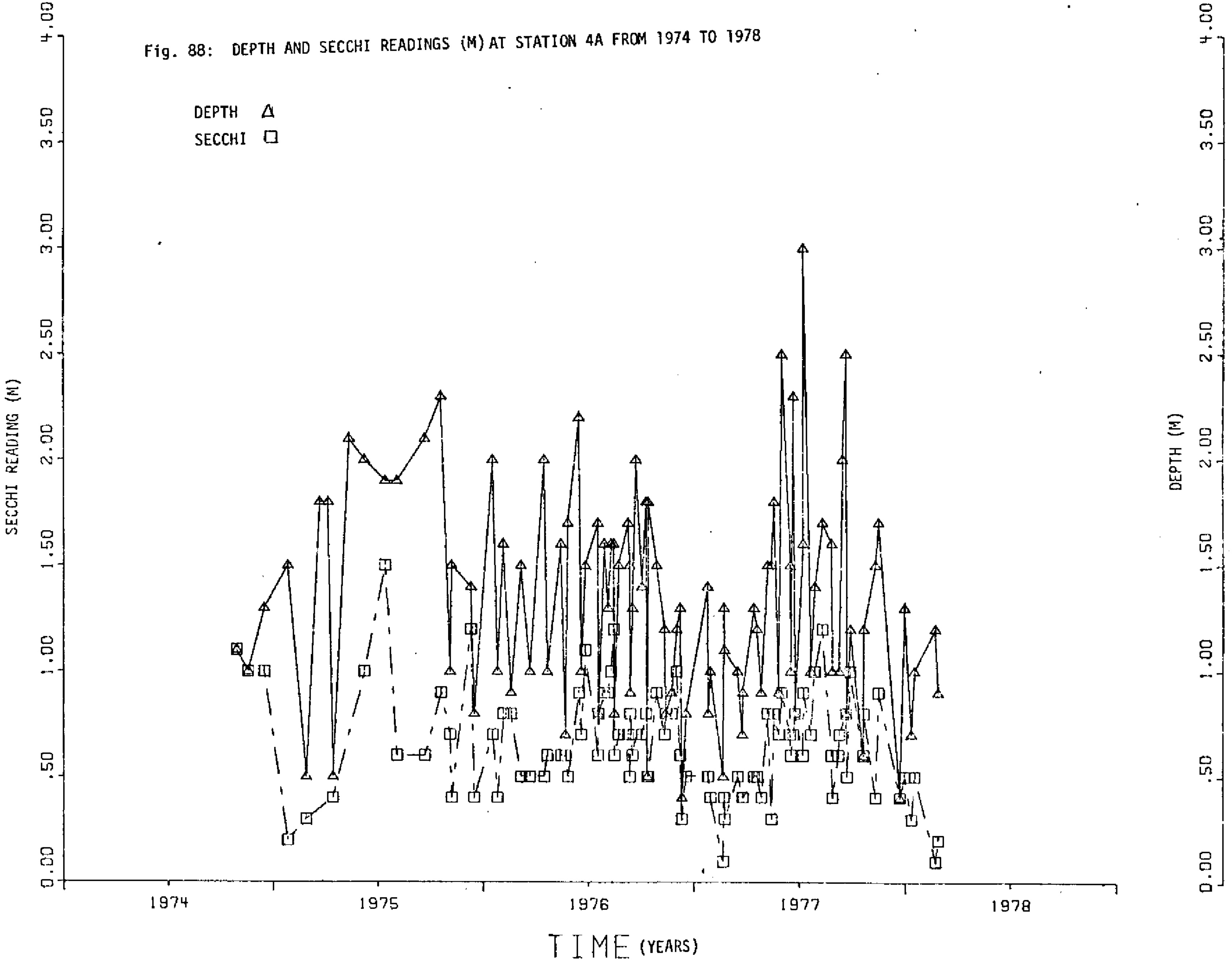
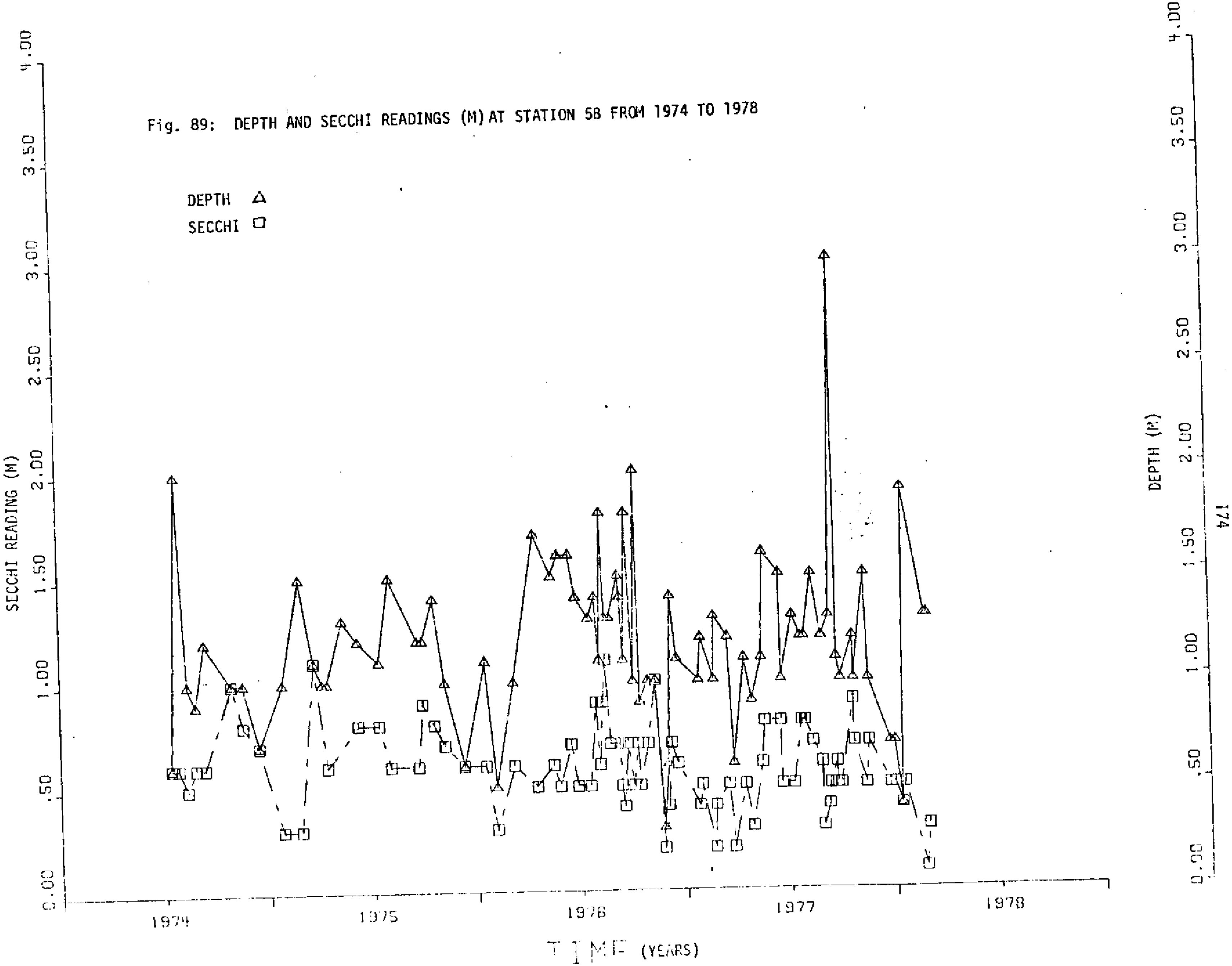


Fig. 88: DEPTH AND SECCHI READINGS (M) AT STATION 4A FROM 1974 TO 1978

Fig. 89: DEPTH AND SECCHI READINGS (M) AT STATION 5B FROM 1974 TO 1978



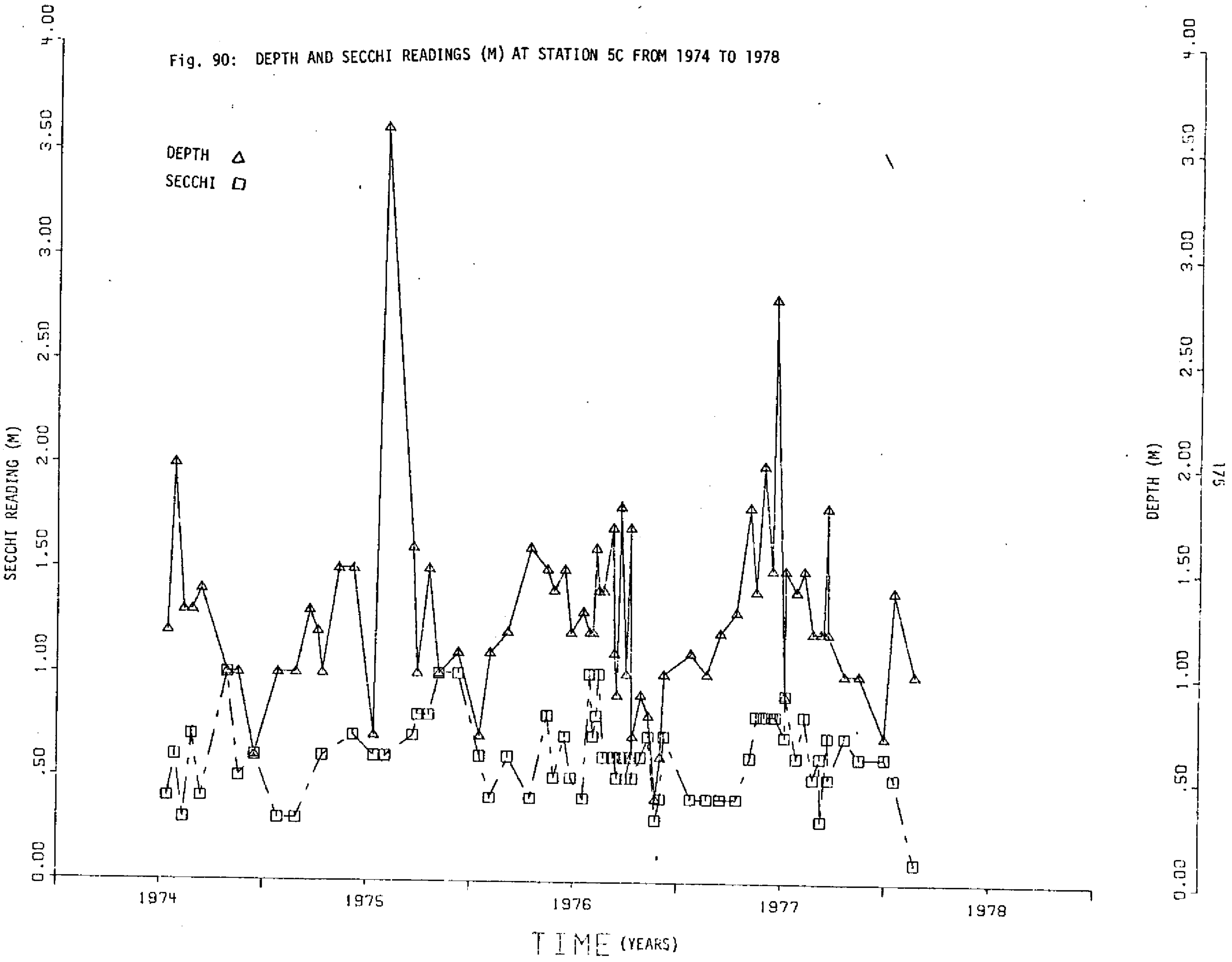


Fig. 90: DEPTH AND SECCHI READINGS (M) AT STATION 5C FROM 1974 TO 1978

IV. Laboratory Bioassays: Blue Crab Avoidance Of Storm-Water Runoff

INTRODUCTION

Recent studies have been made concerning the adverse effects of timbering activities on aquatic systems (Likens et. al. 1970; Livingston et. al. 1974; Livingston, 1975). The results indicate that the methods of land development employed by the paper mill interests (clearing, plowing, ditching) together with the uncontrolled drainage of the developed upland areas into freshwater and marine coastal systems can cause multiple alterations of habitat structure and water quality. However, the adverse effects of reduced pH, typical of such drainage, on the estuarine biota have not been well documented even though laboratory (Loosanoff and Tommers, 1947; Calabrese and Davis, 1966; Kwain, 1975) and recent field evidence (Almer et. al., 1974, Beamish et. al., 1975) have shown potential damage to biological systems.

Little work has been done on the ability of aquatic organisms to detect and avoid water with low pH although avoidance behavior has been used successfully as an indicator of the sublethal effects of other forms of pollution (Jones, 1947; Bishai, 1962; Ishio, 1965; Hansen, 1969). The use of such behavioral data, however, should be carefully reviewed since there are complications associated with various forms of apparatus and techniques, varied reactions of the test animals due to confinement in the avoidance trough (schooling, agonistic responses, etc.) (Sprague, 1964; Kleerekoper et. al., 1972; Livingston et. al., 1976), and seasonal variations in the behavioral repertoire of such animals (Sprague, 1964; Sprague et. al., 1965).

Others have pointed out that although avoidance experiments are instructive, they do not necessarily involve the reactions of organisms in actual environmental situations where a complex interaction of biological and physico-chemical factors dominates the final behavior of such animals (Sprague, 1964; Doudoroff, 1965; Livingston et al. 1976). Unfortunately, there are few quantitative comparisons of field and laboratory avoidance studies.

The present investigation involves an integrated field and laboratory approach to the problem of avoidance responses of blue crabs (Callinectes sapidus) to storm water runoff from recently clearcut upland areas (Livingston, 1976). This includes a preliminary determination of the principle water quality parameters involved in the avoidance behavior.

RESULTS

The results of the avoidance experiments with the small blue crabs (20-60 mm, wide) are shown in Table 1 (A-B). These crabs strongly avoided ($p.<0.001$) the acidic (pH 5.2 and 5.8) and highly colored (water color 245 and 300) runoff taken from the field (Table 1-A). In the control experiments designed to test pH, there was a significant avoidance ($p < 0.001$) to induced pH reductions below 6.0 (Table 1-B).

Net avoidance (mean Avoidance Index) from the control tests was inversely and linearly related to pH ($r = 0.99$) (Fig. 1). The threshold pH level was estimated, from the regression equation, at 6.48 with a 95% confidence interval falling within the range 6.02 - 6.92. The AI obtained at pH 8.5 (Table 1-B) was not included in the regression analysis due to the insignificant responses already found at pH 6.5 and 7.0. Although reactions to water color were not tested, a comparison of the avoidance indices of Table 1 shows a slightly higher response to the runoff with pH 5.8 than to control water with the pH reduced to 5.8.

The results of the tests with the larger crabs (60-140 mm wide) are summarized in Table 2. The runoff from the field (pH 4.6, color 445) elicited a strong avoidance response in which the crabs (N = 10, independently tested) spent 92.9% of the test time in the control water. In the control experiments, the crabs showed avoidance reactions inversely related to pH with maximal responses found at pH 4.6 and 5.5 (mean time of 96% and 92% respectively) and minimal values at pH 6.5 and 7.0 (60.7% and 61% respectively). When these avoidance data were analyzed by comparing, by means of ratios, the mean times spent in a visit and the mean number of visits to the control water versus those to the experimental water, therewas a similar pH-directed avoidance response (Fig. 2). This was evidenced by ratios greater than 2 at pH levels below 6.5 and between 1 and 2 at pH levels 6.5 and 7.0. An inverse relationship between the time-responses and the standard deviation of the mean response was found,showing the increasingly mixed reactions of the large crabs to pH levels between 6.0 - 7.0. Such avoidance responses were sex-independent.

The experiments with buffered runoff designed to test the effect of water color indicate that color _per se_ had little influence on the avoidance responses of the large crabs. This was evidenced by mean time-response of 56.1% (Table 2) and ratio values approaching 1. Thus, the results showed that pH was the primary factor involved in the avoidance reactions of blue crabs to runoff.

Field study

The physico-chemical data (03/75-02/76) showed that temperature, salinity, dissolved oxygen, turbidity and transparency (Secchi disk readings) were quite comparable at the three stations (4A, 5B, 5C). However, consistently lower pH and higher water color were found only at the stations receiving runoff (5B, 5C) (Fig. 5). Station 4A had a mean pH of 7.85 (SD = 0.78) whereas 5B and 5C had means of 6.5 (SD = 1.04) and 6.65 (SD = 0.9) respectively.

Similarly, station 4A had a mean water color of 61 (SD = 40) whereas 5B and 5C had means of 138 (SD = 63.4) and 160 (SD = 80) respectively. Water color was inversely related to pH (r = 0.8). Student t-tests showed that the mean pH and color at station 4A were statistically different from the values taken at 5B and 5C. The most striking differences in pH between the test and control areas occurred in March, August and January when the pH in the former (5B, 5C) dropped to levels close to 4.0 (August) (Fig. 5) and color peaked to levels above 250. These extreme values were usually preceeded by moderate to heavy local rainfall.

When the field pH levels of this study were compared to values taken prior to the major clearcutting activities of 1973-75 (John Taylor, personnal communication) there was a significant ($p<0.05$; t-test) change only in the areas receiving upland runoff from adjacent clearcut areas (5B, 5C). This was particularly pronounced in early spring and late summer.

The field results are shown in Fig. 3 (A, B) (03/75-02/76). During periods of low pH, the areas receiving runoff (5B, 5C) usually had as many or more small crabs than the control area (4A) (March, August, January, February). The largest difference in the abundance of these crabs between the control and experimental areas, however, occurred in July when these areas had pH values greater than 6.5 (Fig. 3). The larger crabs in the field were at least three times more abundant in the control area than in the areas receiving the overland runoff. Such differences, evidenced by high ratio values (No. crabs in control station/No. crabs in test area), usually occurred at times when there was a reduction in the pH of the experimental areas to levels below 6.0, usually in March, August, January and February. This indicates there was avoidance by the large crabs of the affected areas in contrast to the behavior of the smaller crabs.

DISCUSSION

Although no comparable avoidance studies have been made with invertebrates, these results can be compared to those with various species of fishes. The levels of avoidance to reduced pH fell within a range found in the literature. Jones (1947) found that the fish Gasterosteus strongly avoided laboratory pH levels below 5.4. No behavioral reaction to pH levels between 6.0 - 7.0 was found. Bishai (1962) found that older stages of salmon avoided water with pH lower than 6.5, which was above the lower incipient limiting level (6.0 - 6.2). Other studies (toxicity, etc.) have shown similar results (Bishai, 1960; Johansson and Kihlstrom, 1973). Mount (1973), for example, found that the fathead minnow exhibited hyperactivity, abnormal behavior, and body deformities at pH levels below 6.0. The avoidance reactions of aquatic organisms to water color are largely unknown. Although the results of this study showed that color had no additive effect in the avoidance responses of the large crabs to runoff, questions remain concerning such effect on the reactions of the younger ones.

Few comparative field and laboratory avoidance studies are available in the literature. One study involved the avoidance by Atlantic salmon to mine wastes (zinc and copper) in a Canadian river (Sprague, 1964; Sprague et. al., 1965; Saunders and Sprague, 1967). The levels of the waste in the river that caused confused movements of the salmon were about 18 times higher than laboratory thresholds. The authors reasoned that seasonal behavioral patterns of the fish such as territoriality, maternal behavior, etc., and the interaction of biological or physico-chemical factors may lead to salmon tolerating higher concentrations of the pollutant in the field.

Similarly, the present field study showed divergent results when compared to the laboratory findings. Although the field distribution of the large crabs did show an avoidance of the acidic areas, the distribution of the younger ones did not. Increased numbers of these crabs were actually abundant in these areas at times when the pH dropped to levels openly avoided in the laboratory (4.5-6.0) (August, January, February). However, the fact that the largest difference in small blue crab abundance between the affected and control areas occurred in July, when the pH in the former was relatively high (6.5) suggests that other ecological factors are operating in East Bay and determining both the spatial and temporal distribution of the crabs. Livingston et al. (1976) showed that small blue crabs (C. sapidus) moved into Apalachicola Bay mainly in early winter (January, February), and in mid-to-late summer (July, August), coinciding with the times of pH drops in East Bay. Darnell (1959) and Tagatz (1969) found that young blue crabs seek the shallowest and freshest areas of Lake Pontchartrain, Louisiana, and of the St. John River (Bay), respectively, during their early life stages. This was followed by growth and movement of the larger crabs into the deeper areas of both regions. These results plus the fact that in East Bay the affected areas are about 1.3 m (4.3 ft) shallower than the control area suggest that the distribution of blue crabs in East Bay may be related both to the life cycle of these organisms and to the relative structure (depth, etc.) of their selected habitats. In this case, however, the relationship should be of such magnitude as to override any behavioral reactions to pH levels as low as 4.0. Also, striking differences in the food resources between the control and the affected areas may be relevant to this problem. McLane (1977), for example, found a more diverse and abundant infauna (polychaetes, amphipods, etc.) in the control station (4A) compared to the others. Also, these areas are richer in organic debris than the control station. The possible size-related trophic response however, should be viewed together, and not confounded with patterns of intraspecific competition and predation (Darnell, 1959; Tagatz, 1969).

This complex interrelationship of factors that dominate final behavior and distribution of animals in the field complicates the extrapolation of laboratory results to such areas. These extrapolations are further complicated by the fact that field gradients of chemicals (pH) are usually ephemeral, extend for hundreds of meters and undergo short and long term changes which are difficult to correlate with abundance of organisms at a certain point in time. Laboratory gradients, in contrast, are steep and confined to the avoidance apparatus. Thus, laboratory results without associated field evidence may lead to inconclusive results if they are applied for the determination of impact criteria.

The inability by the small crabs in the field to avoid water with the pH reduced to levels below 6.0 is critical. Long exposure to this acid water can seriously complicate the biology of these organisms. Laboratory and field toxicity studies have shown that such complications in fishes may involve growth retardation (Beamish, 1975; Johansson and Kihlstrom, 1975), egg mortality (Calabrese and Davis, 1967; Kwain, 1975), and ultimately the disappearance of entire populations from acid waters (Almer, 1974; Beamish, 1975).

Results showed that the recent major clearcutting operations above East Bay were followed by the acidification of adjacent aquatic habitats. This was attributed to the efficient and rapid channelization of the acid runoff following moderate to heavy local rainfall. The strong avoidance reactions by the blue crabs indicated the sublethal effects of such runoff. This could have serious implications for other sensitive faunal associations in such areas (Laughlin et al., 1978).

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Table 1 A-B: Avoidance responses by small juvenile blue crabs (20-60 mm wide) to (A) naturally acidic and highly colored water and to control water with the pH experimentally adjusted (B). Crab counts were taken immediately before transposing the water sources entering the trough arms and at the end of each experiment (under the reversed conditions). Crabs in the holding area of trough were not included in the percentages or statistical tests. Avoidance index (%) compares the proportions of crabs in the control arm before a switch (P_1) to the proportion of crabs in the same arm after the switch (P_2). Color is given in APHA-Platinum-Cobalt standard units (APHA, 1971); the higher the value the darker the water. In Table A the pH of the control water was set at 7.75 and color at 85 units. In B the pH of the control water ranged from 7.9 to 8.5.

TABLE 1-A

Variables of Exp. Water (Runoff)			AVOIDANCE RESPONSES													
			BEFORE SWITCH					AFTER SWITCH								
		No. Crabs Tested	Control Arm (Control Water)		Exp. Arm (Runoff Water)		Holding Area (Mixed Water)	Control Arm (Control Water)		Exp. Arm (Runoff Water)		Holding Area (Mixed Water)	"Z" Value	Avoidance Index (%)	Crab Size Carapace Width,mm	Sex Ratio
pH	Color		No. Crabs	(%)	No. Crabs	(%)	No. Crabs	No. Crabs	(%)	No. Crabs	(%)	No. Crabs			$\overline{X}$ ± SD	M:F
5.2	245	10	7	(100)	0	(0)	3	9	(90)	1	(10)	0	9.48**	90	(28.1 ± 4.67)	5:5
5.8	300	10	3	(60)	2	(40)	5	6	(100)	0	(0)	4	2.73**	100	(44.3 ± 7.48)	8:2
5.8	300	10	6	(85.7)	1	(14.3)	3	9	(100)	0	(0)	1	6.48**	100	(42.7 ± 7.95)	9:1

** Significance with $p<0.001$

TABLE 1-B

		AVOIDANCE RESPONSES														
		BEFORE SWITCH					AFTER SWITCH									
		Control Arm		Exp. Arm		Holding Area	Control Arm		Exp. Arm		Holding Area			Crab Size Carapace width,mm	Sex Ratio
pH Exp. Water	No. Crabs Tested	No. Crabs	(%)	No. Crabs	(%)	No. Crabs	No. Crabs	(%)	No. Crabs	(%)	No. Crabs	Z Value	Avoidance Index (%)	$\overline{X}$ ± SD	M:F
5.0	10	6	(75)	2	(25)	2	9	(90)	1	(10)	0	3.96**	86	(31.5 ± 2.11)	8:2
	9	6	(75)	2	(25)	2	6	(100)	1	(0)	4	4.90**	100	(40.4 ± 2.71)	7:2
5.8	10	4	(80)	1	(20)	5	6	(75)	2	(25)	2	2.33**	69	(4.25 ± 5.79)	7:3
	10	7	(100)	0	(0)	3	5	(71.4)	2	(28.6)	3	4.18**	71	(35.9 ± 2.42)	6:4
6.0	10	6	(100)	0	(0)	4	4	(80)	1	(20)	5	4.47**	80	(45.1 ± 3.07)	8:2
	10	2	(28.6)	5	(71.4)	3	7	(87.5)	1	(12.5)	2	0.77	55	(42.7 ± 3.23)	9:1
6.5	10	2	(33.3)	4	(663)	4	6	(85.7)	1	(14.3)	3	0.82	48.6	(39.3 ± 3.53)	5:5
8.5	10	5	(100)	0	(0)	5	0	(0)	7	(100)	3	0.00	0	(48.8 ± 1.75)	6:4

** Significance with $p<0.001$

Table 2: Results of the avoidance experiments with large blue crabs. Summary table includes mean avoidance values with runoff water from the upland creek (*), control water with the pH experimentally reduced (**) (to test the pH variable), and field runoff water with the pH experimentally increased (***) (aimed to test the color variable). The pH of the control water varied from 8.1 to 8.2 for all tests. The color of the control water in the controlled tests (**) had same values as the experimental water. Color of control water in (*) had a value of 60 and in (***) had a value of 20.

Variables of Exp. water			AVOIDANCE VALUES									
			Avoidance as percent of time spent in control water from total time spent in both arms of test trough.	95% confidence interval for response	Mean time (Sec) spent in a visit to:		Mean No. of visits to		Size (carapace width)			Sex Ratio
pH	color	No. crabs tested	%	%	Control	Exp.	Control	Exp.	$\overline{X}$	±	SD	M:F
4.6	445*	10	92.9	(85.5, 100)	---	---	---	---	(85.7	±	6.7)	7;3
4.6	40**	7	96.0	(93.8, 98.1)	174	13	78	11	(87.0	±	12.5)	5:2
5.5	50	7	92.0	(85.6, 98.4)	168	28	130	25	(94.3	±	8.2)	5:2
6.0	30	7	81.5	(74.3, 88.7)	83	33	81	33	(87.0	±	6.9)	4:3
6.5	20	7	60.7	(33.1, 88.5)	251	246	26	20	(82	±	6.4)	4:3
7.0	30	7	61.0	(34.8, 40.5)	87	87	49	32	(89	±	10.7)	6:1
8.2	340***	7	56.6	(47.1, 65.1)	144	181	23	16	(112	+	13.9)	5.2

Fig. 1: Relationship between pH and mean Avoidance Index of the small blue crabs ($<$ 60 mm; carapace width) (from Table 1-B). The threshold pH level with its 95% confidence interval is also shown. Data from experiments with upland runoff were excluded. C. I.= Confidence Interval.

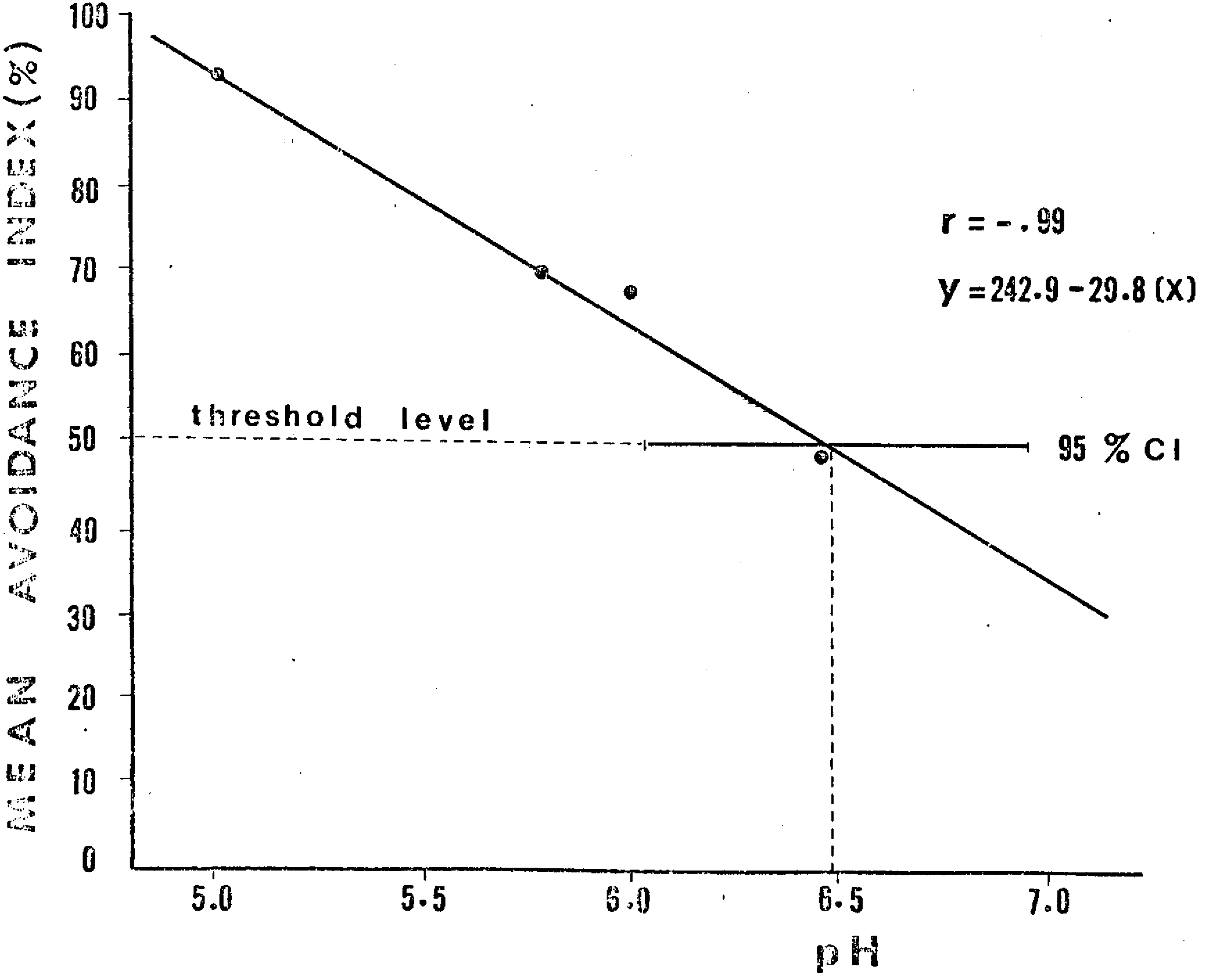

MEAN AVOIDANCE INDEX (%)
100
90
80
70
60
50
40
30
20
10
0
5.0
5.5
6.0
6.5
7.0
pH
r = -.99
y = 242.9 - 29.8 (X)
threshold level
95 % CI

Fig. 2: Relationship of two types of avoidance reactions and pH. The ratios of mean time spent in a visit to control water to mean time spent in a visit to experimental water (with adjusted pH) by large crabs is given by the solid line. The ratios of mean number of visits to control water over experimental are illustrated by the broken line.

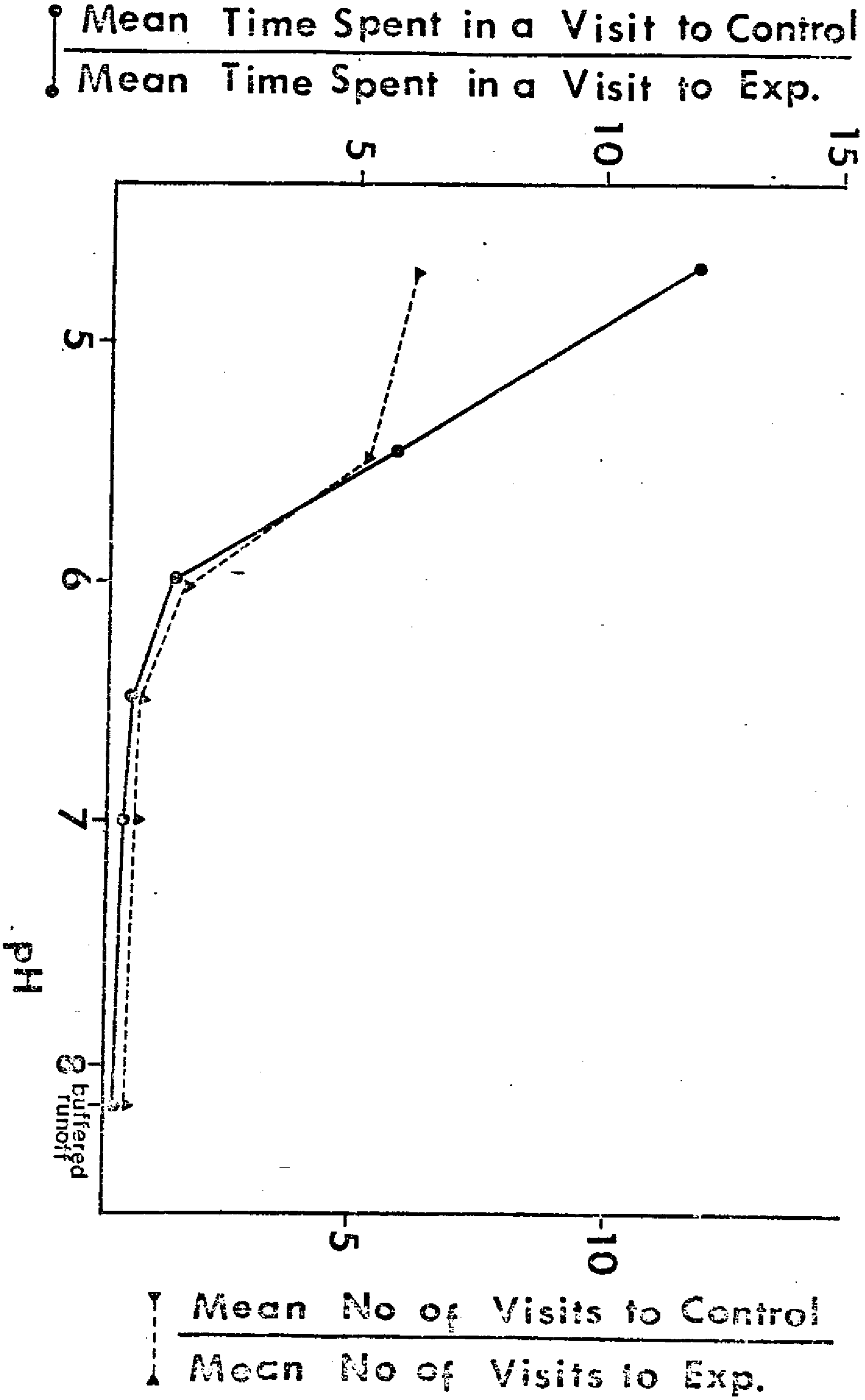
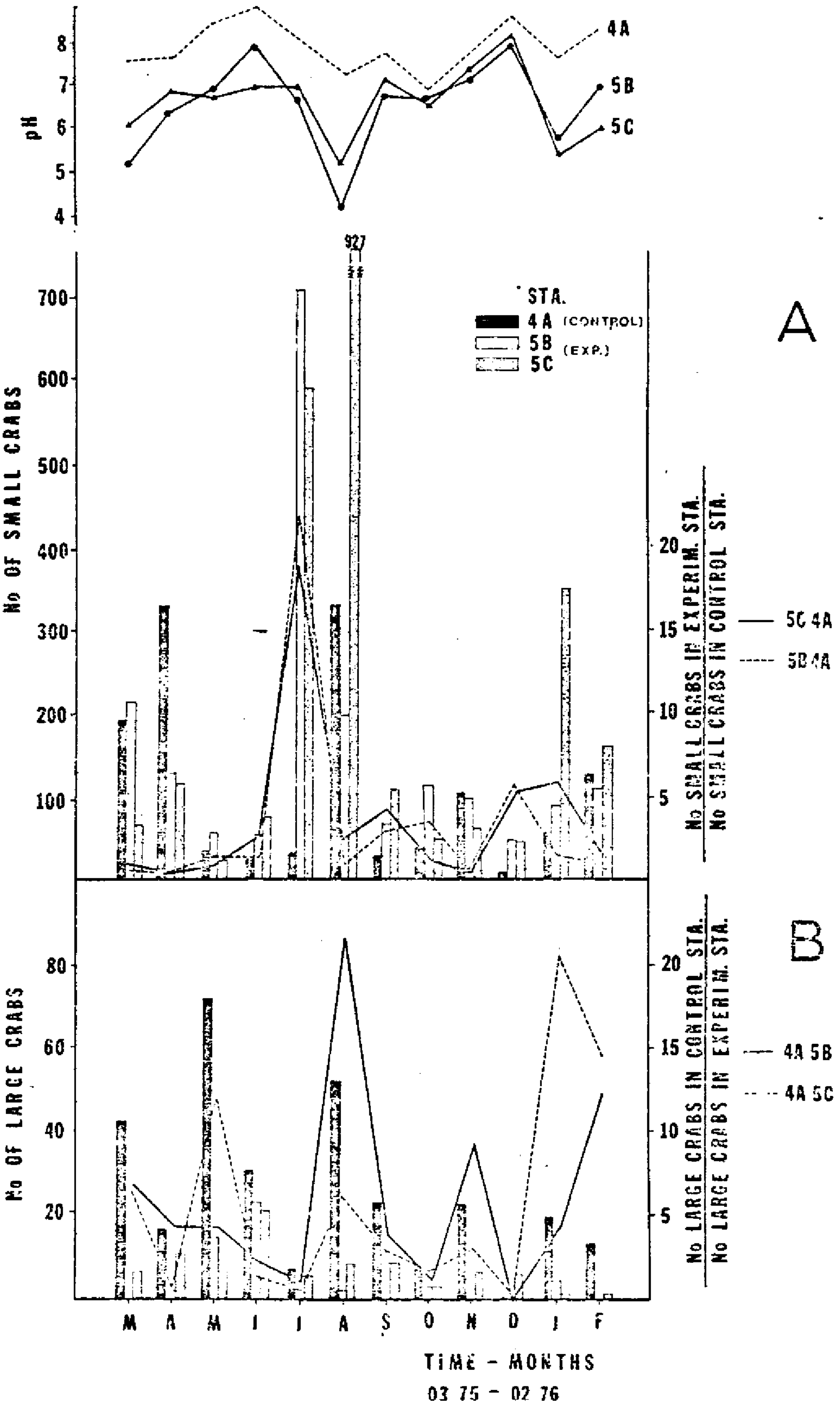

Mean Time Spent in a Visit to Control
Mean Time Spent in a Visit to Exp.
5
10
15
5
6
7
8
buffered runoff
pH
5
10
Mean No of Visits to Control
Mean No of Visits to Exp.

Fig. 3: Field collections of small (0-60 mm wide) crabs (A) and large (60-140 mm wide) crabs (B) taken at stations 4A (control), 5B and 5C (experimental) from March 1975 to February 1976. The pH fluctuations in the control and experimental (mean of 5B and 5C) stations are shown at the top of the figure. Also given are the relationships (ratios) between the number of small crabs in 5B, 5C and those in 4A, (Figure 5A) and the relationship between the number of large crabs found in 4A and those in 5B and 5C (broken and solid lines).


pH
8
7
6
5
4
4A
5B
5C
927
STA.
4 A (CONTROL)
5 B (EXP.)
5 C
A
No OF SMALL CRABS
700
600
500
400
300
200
100
No SMALL CRABS IN EXPERIM. STA.
No SMALL CRABS IN CONTROL STA.
20
15
10
5
5C 4A
5B 4A
B
No OF LARGE CRABS
80
60
40
20
No LARGE CRABS IN CONTROL STA.
No LARGE CRABS IN EXPERIM. STA.
20
15
10
5
4A 5B
4A 5C
M
A
M
J
J
A
S
O
N
D
J
F
TIME - MONTHS
03 75 - 02 76

V. Benthic Infauna

Introduction

The benthic infauna of the East Bay portion of the Apalachicola estuary were studied to determine spatial and temporal biotic variation relative to key forcing functions of the system such as river flow, rainfall, runoff, and water quality. The first 20 months of data (February, 1975-September, 1976) were part of an unfinished masters thesis started by Mr. Bradford G. McLane. Basic questions were asked regarding a comparison of areas dominated by Apalachicola River flow (station 4) with upland portions of East Bay either within (stations 5A, 5B) or outside of (stations 4A, 6) the drainage associated with cleared portions of the Tate's Hell Swamp. Sampling at stations 1, 1X, 3 and 6 was dropped after the first 12 months. The basic hypothesis in the long-term data collection involved temporal changes in benthic infaunal assemblages in stressed areas (stations 5A, 5B) relative to control stations (4, 4A). Complications involving water quality at station 4A have been described elsewhere in this report and will not be analyzed here.

Physico-chemical functions

In addition to that on the basic background status of the physical environment of the bay (presented elsewhere in this report), an analysis was carried out of key physico-chemical changes at the 4 stations of primary interest (4, 4A, 5A, 5B) from February, 1975 to the present.

Monthly river flow and rainfall are shown in Figs. 1 and 1A. Daily rainfall in the Tate's Hell Swamp (East Bay tower) is shown in Fig. 2. During 1975 (winter-summer) there were major influxes of river water and local runoff into the Apalachicola estuary. Over the 3-year period of study, however, there was a general decline in river flow and local rainfall. From late 1975 to the winter of 1978 there was little precipitation, with limited episodes of (local) rainfall occurring during the early fall of 1976. The trends in the salinity regimes at the respective stations are given in Figs. 3-6; there was a general salinity increase at station 4 over the 3-year period with relatively high salinity at stations 4A and 5B during the latter months of 1976. In each instance, the lowest salinities occurred during winter-spring periods with late summer-fall decreases generally correlated with periods of local rainfall. The highest salinities occurred during the second half of the year.

Trends of pH changes (Figs. 7-10) indicate relatively little variation in time at station 4. At station 4A, there was a general increase in pH through mid-1976; during the latter half of 1976 and early 1977 there were repeated decreases in pH to levels below 6.0. During 1977, there was, again, a brief increase in pH followed by a single decrease in the fall. This trend was evident at station 5A except that the winter peaks and fall troughs were less extreme than at Round Bay. At station 5B, there was a sharp decline to a relatively low pH during the summer rains of 1975. After a decrease in pH during the late summer of 1976, there was a general increase in pH at this station with time. Such changes were generally correlated with local (East Bay) rainfall distribution and clearcutting activities in the respective drainage areas. As described, the general patterns of pH changes at stations 4A and 5B differed, with definite recovery (i.e., increased pH) at station 5B with time.

Biological analysis: bay-wide trends

There was a general decrease in total biomass, numbers of individuals, and numbers of species of infauna over the 3-year period of study (Table 1). A list of all species taken during the study period (Table 2) indicates that most infaunal species fall into four major categories: crustaceans, polychaetes, mollusks, and miscellaneous. Figs. 11-13 show the monthly data for numbers of individuals, species richness, and biomass. Biomass usually peaked during winter and fall periods. The river fluctuations appeared to be relatively important to such fluctuations in terms of seasonal and annual trends of infaunal biomass. Total numbers of individuals show a similar trend although the cold winter of 1976 appears to have had a dampening effect on this function. The numbers of species followed a two-peak seasonal trend with a long-term downward trend until the winter of 1977-78. These data are still under study, and detailed analysis will be made at a later time.

Based on a detailed study of the first 20 months of data, the following trends were apparent:

An amphipod, Grandidierella bonnieroides, was the most abundant species taken in the Apalachicola system. This species accounted for 18% of all individuals collected. Eleven other species of amphipods were collected, as well as four species of isopods. Of the eleven species of amphipods, Cerapus sp. was numerically important, composing 4.4% of the total number of individuals.

The polychaete fauna was composed of 18 species, four of which were included in the top 9 species (numerically) in the bay. *Mediomastus californiensis* was the most numerous polychaete and composed 15% of the total number of individuals. Three other numerically dominant polychaetes were *Amphicteis gunneri*, *Streblospio benedicti*, and *Laeonereis culveri*. These, in turn, composed 7%, 6%, and 4% of the total number of individuals.

The mollusca constituted ten species, two of which were numerically important. *Mactra fragilis*, a bivalve, was the most important species (8% of the total number of individuals) while *Littoridina sphinctostoma* made up another 6%.

In order, the nine most important species of the total infauna were the following: *Grandidierella bonnieroides*, *Dicrontendipes* sp., *Mediomastus californiensis*, *Mactra fragilis*, *Littoridina sphinctostoma*, *Amphicteis gunneri*. *Streblospio benedicti*, *Cerapus* sp., and *Laeonereis culveri*. Eleven species (*Amphicteis gunneri*, *Mediomastus californiensis*, *Streblospio benedicti*, *Grandidierella bonnieroides*, *Mactra fragilis*, *Macoma mitchelli*, *Dicrontendipes* sp., *Capitella capitata*, *Loandalia americana*, *Polydora ligni*, and *Cyathura polita*) were collected at all the stations, while all others were collected at four stations or less. Eighteen species were collected only once. Others such as *Fabricia* sp., *Glycinde solitaria*, *Haploscoloplos fragilis*, *Polydontes lupina*, *Sigambra bassi*, *Paracaprella tenuis*, *Leptochelia rapax*, *Ampelisca vadorum*, *Ogyrides limicola* and *Pseudocyrena floridana*, were collected only at station 4. This was thought to be a result of the relatively high salinity in this area.

An apparent migration of several species occurred during the sampling period. *Cerapus* sp. were collected at station 4 only during the first three months of sampling. They appeared at station 6 during the sixth month of sampling in very large numbers. They were collected there for three months, after which time very few were found. This species was found at 5A in significant quantities only during the eighth and ninth months. *Littoridina* was not collected during the winter at station 5A, but was abundant there during the rest of the year.

Station 4

The six most abundant species in this area, in descending order, were the following: *Mediomastus californiensis*, *Streblospio benedicti*, *Amphicteis gunneri*, *Loandalia americana*, *Capitella capitata*, *Corophium louisianum*. Peak numbers were found from February through June the first year. This peak was not repeated the second year, but numbers increased slightly in January of 1976 and fluctuated little throughout the remainder of the year.

Station 5A

The top six species at this station were as follows: *Littoridina sphinctostoma*, *Mactra fragilis*, *Grandidierella bonnieroides*, *Mediomastus californiensis*, *Dicontendipes* sp., *Macoma mitchelli*. The number of individuals collected at this station ranged from 1,029/m^2 to 7,534/m^2, with a mean of 4,139/m^2. Peak numbers were found during spring months (April-June). Numbers of species fluctuated continuously with no apparent trend.

Station 4A

The six most abundant species at this station were: Grandidierella bonnieroides, Dicrontendipes sp., Laeonereis culveri, a nematode, Mediomastus californiensis, Amphicteis gunneri. Peak numbers were recorded during winter months (November-January) with low numbers during summer months (April-August). Monthly totals ranged from 679/m^2 to 5,825/m^2 with a mean of 3,219/m^2. The monthly number of species fluctuated relatively little, with no apparent trend.

Station 6

In this area, the six most abundant species (in descending order) were: Grandidierella bonnieroides, Cerapus sp., Dicrontendipes sp., Amphicteis gunneri, Mediomastus californiensis, Laeonereis culveri. Peak numbers were recorded in summer and early fall months (July-September) and low values during spring (April-May) and fall periods (October-December). Total numbers of individuals ranged from 1,380/m^2 to 10,512/m^2 with a mean of 4,358/m^2. The number of species fluctuated with no apparent seasonal trend.

Station 5B

The top six species were as follows: Dicrontendipes sp., Streblospio benedicti, Mediomastus californiensis, Mactra fragilis, Amphicteis gunneri, Grandidierella bonnieroides. Peak numbers were found during winter-spring months (November-March) with low levels in summer months (May-July). One exception to this trend occurred during August of 1976. The total number of individuals ranged from 591/m^2 to 7,796/m^2 with a monthly average of 2,228/m^2. The number of species collected was considerably reduced during specific summer months (August, June, September) while remaining relatively constant during the rest of the year.

Biomass figures for the first year of study are shown in Table 3. By far, the most productive was station 1X, followed by 5A and 3. Peak values of biomass occurred during winter and early spring months at stations 1X, 2, 3, 4, 5A, and 5B. Biomass at grassbed-dominated stations in East Bay (4A, 6) tended to peak during fall months. This was probably related to associated die-offs of benthic macrophytes in these areas. With the exception of station 5A, upland stations (4, 4A, 5B, 6) had the lowest mean biomass. The lowest monthly figures occurred at station 5B during August and September (1975) and June, July, and September of 1976. This coincided generally with periods of increased local rainfall, increased color, and reduced pH and dissolved oxygen. Dominance tended to peak during summer and fall months (Dc = 98-100% during August, 1975, and June-September, 1976). This is indicative of stress at this station during these periods.

Impact analysis: long-term trends

The analysis of impact was complicated by the fact that natural upland runoff tends to stress upland portions of the bay regardless of proximity to runoff from the areas of Tate's Hell Swamp disturbed by forestry operations (Table 3). Such upland areas are continually subjected to rapid drops in salinity and short-term changes in various key environmental factors. This tends to stress specific populations and reduce biomass, species richness, and various diversity indices.

As noted previously, four stations were used for th analysis. Station 4 is a river-dominated area in East 4A (Round Bay) is out of the direct East/West Bayou dra until the ditching and clearing of 1976, this area was free of impact associated with upland forestry operatio 5B (West Bayou) is in the drainage system associated wit operations, while 5A is located farther out in the bay, the West Bayou drainage pattern. This area is closely as upland Vallisneria beds. As noted above, these stations reflect seasonal trends of winter/spring river flow and f of benthic macrophytes.

Three-year trends of numbers of individuals and bioma various East Bay stations are shown in Figs. 14-21. At st there were somewhat contradictory patterns with large numbe individuals (due primarily to two species: Mediomastus cal and Streblospio benedicti) during the first winter of sampl relatively low numbers of individuals in the following mont Biomass tended to follow a double peak (spring/fall) with in often coinciding with increased river flow. At station 4A, and biomass generally peaked during summer/fall periods. The was a generally low level of infaunal production during the p from 1976 to mid-1977. This coincided with the relatively lo levels during the early fall of 1976 and the cold winter of 1 and was probably the result of a combination of these factors. was a return to previously high levels during the following fa Station 5A showed a general decline in both numbers and biomass winter peaks of biomass generally decreasing with time. Station 5B showed generally low biomass and numbers of individuals with the winter peaks suppressed during the cold winter of 1976-77. Peak numbers of individuals occurred during 1976; this could reflect salinity changes and other water quality trends noted above

The 3-year changes of species richness and diversity are shown in Figs. 22-25. There was a general decline in the number of species (species richness) with time at station 4. Species richness at station 4A tended to reach a low level during 1976 with subsequent increases during 1977. At station 5A, species richness tended to decrease with time. However, at station 5B, there was a sustained increase in the species richness during the last year of sampling. Low levels occurred during summer periods of low water quality. Species richness at this station was generally lower than at the other 3 study sites. With some exceptions, the species diversity tended to follow the trends of species richness. Again, there was extremely low diversity during summer periods of high rainfall during the first two years of sampling. There seemed to be a recovery of diversity during 1977. This would indicate that the benthic infauna in upper East Bay reflect the water quality trends noted at stations 4A and 5B. During periods of sustained rainfall in the Tate's Hell Swamp, there were definite short-term reactions of the infauna to changes in associated water quality parameters and such changes also occurred at the control station (4A) in such a way as to indicate that forestry operations in the Round Bay drainage contributed to the stress of the biological (estuarine) system. These relationships are corroborated by graphical comparisons of different indices in Figs. 25-28.

The data indicate that there are short- and long-term trends in the infaunal assemblages of East Bay which are closely aligned with water quality variations associated with rainfall patterns and runoff conditions. [illegible] with stress (increased runoff and resultant water quality changes due to forestry operations) were characterized by infaunal assemblages with reduced species richness and diversity indices. Increased runoff following the clearing of an upland area and increases in local rainfall led to immediate adverse impact. The extent of such impact appeared to depend on the extent of land manipulation and the rainfall conditions in the Tate's Hell Swamp. Such fluctuations are superimposed over long-term trends which tend to follow river flow phenomena and long-term (annual) fluctuations of local rainfall activity. The changes in the control station (4A) and the area of primary impact (station 5B) indicate that local forestry operations do have an adverse impact on the infauna, but that it is of short duration (one season), and is relevant only within the context of the long-term trends in the bay. As yet, these trends remain largely undetermined, although there are indications that they are the result of trophic phenomena and water quality changes associated with long-term trends of Apalachicola River flow and local rainfall.

These studies have been continued as one part of an effort to analyze long-term trends and associations of estuarine organisms in the Apalachicola estuary with key physico-chemical forcing functions.

Table 1: Summary totals (annual) for benthic infauna taken in East Bay (stations 4, 4A, 5A, 5B) from 2/75 to 1/78.

	Number of species	Number of individuals	Biomass (ash-free dry weight in grams)
Year 1	56	151,594	113.3
Year 2	49	104,851	102.4
Year 3	51	98,539	58.3

Table 2: List of infauna species found in the Apalachicola estuary (stations 4, 4A, 5A, 5B) from 2/75 to 1/78.

Crustacea

A = amphipod
M = mysid
I = isopod
D = decapod
T = tanaid

A	*Grandidierella bonnieroides*
A	*Corophium louisianum*
A	*Cerapus* sp.
A	*Gammarus* sp.
I	*Cyathura polita*
I	*Edotea montosa*
M	*Taphromysis bowmani*
D	*Ogyrides limicola*
M	*Mysidopsis almyra*
D	*Rhithrcpanopeus harrisii*
M	*Bowmaniella dissimilis*
A	*Gammarus mucronatus*
D	*Callianassa jamaicense*
A	*Ampelisca vadorum*
I	*Cassidinidea ovalis*
A	*Hemigena minuta*
A	*Paracaprella tenuis*
A	*Cymadusa compta*
I	*Erichsonella filiformis*
M	*Mysidopsis bigelowi*
D	*Neopanope texana*
A	*Melita* sp.
A	*Gammarus* sp.
D	*Callinectes sapidus*
	copepods
T	*Leptochelia rapax*
D	*Callianassa atlantica*
D	shrimp larvae
D	*Toxeuma carolinense*
I	*Munna reynoldsi*
A	*Melita appendiculata*
D	*Pinnixia chaetopterana*
M	*Mysidopsis furca*
M	*Mysidopsis bahia*
D	*Hippolyte zostericola*

Mollusca

Mactra fragilis
Littoridina sphinctostoma
Macoma mitchelli
Spisula solidissima
Neritina reclivata
Rangia cuneata
Littorina sp.
Brachiodontes exustus
Gastropod sp. 1
Gastropod sp. 2
Pseudocyrena floridanus

Polychaeta

Mediomastus californiensis
Amphicteis gunneri
Streblospio benedicti
Laeonereis culveri
Loandalia americana
Capitella capitata
Sigambra bassi
Glycinde solitaria
Haploscoloplos fragilis
Polydora ligni
Neanthes succinea
Paraprionospio pinnata
Polydora websteri
Heteromastus filiformis
Fabricia sp.
Syllid sp.
Notomastus hemipodus
Capitellides jonesi

Miscellaneous

insect larvae
nemertean sp.
nematode sp.
oligochaete sp. 1
oligochaete sp. 2
cumacean sp.
anemone sp.
turbellarian sp.
nudibranch
ostracod
hydrozoan

Table 3: Biomass (ash-free dry weight, g/m^2) of benthic infauna in the Apalachicola estuary (excluding *Callinectes sapidus*, *Penaeus* spp., and *Rhithropanopeus*)

				STATIONS				
DATE	1X	1	3	4	4A	5A	5B	6
2/75				0.592	1.205	7.211		1.359
3/75	19.508	0.241	4.753	1.227	1.468	7.781		0.526
4/75	56.378	1.074	5.129	1.074	0.898	5.019		0.613
5/75	13.743	1.644	2.608	1.994	0.460	7.277	0.416	0.153
6/75	15.957	1.512	4.384	0.197	0.065	6.334	0.306	1.293
7/75	4.690	0.635	1.709	0.328	0.767	1.161	0.109	0.569
8/75	7.365	0.854	3.265	2.301	0.504	1.950	0.021	1.008
9/75	7.832	1.490	1.994	1.841	2.082	2.717	0.065	1.205
10/75	9.314	3.068	2.321	0.679	2.520	4.690	2.476	0.152
11/75	7.080	0.635	2.586	0.460	1.446	0.591	0.372	1.249
12/75	9.074	1.337	2.338	0.920	0.876	9.469	3.178	0.328
1/76	13.261	4.932	1.578	1.534	0.723	9.359	2.564	0.964
2/76	27.354	0.197	2.410	0.109	0.766	6.554	2.630	0.613
12-month means								
	15.96	1.45	2.92	1.02	1.06	5.39	1.21	0.72

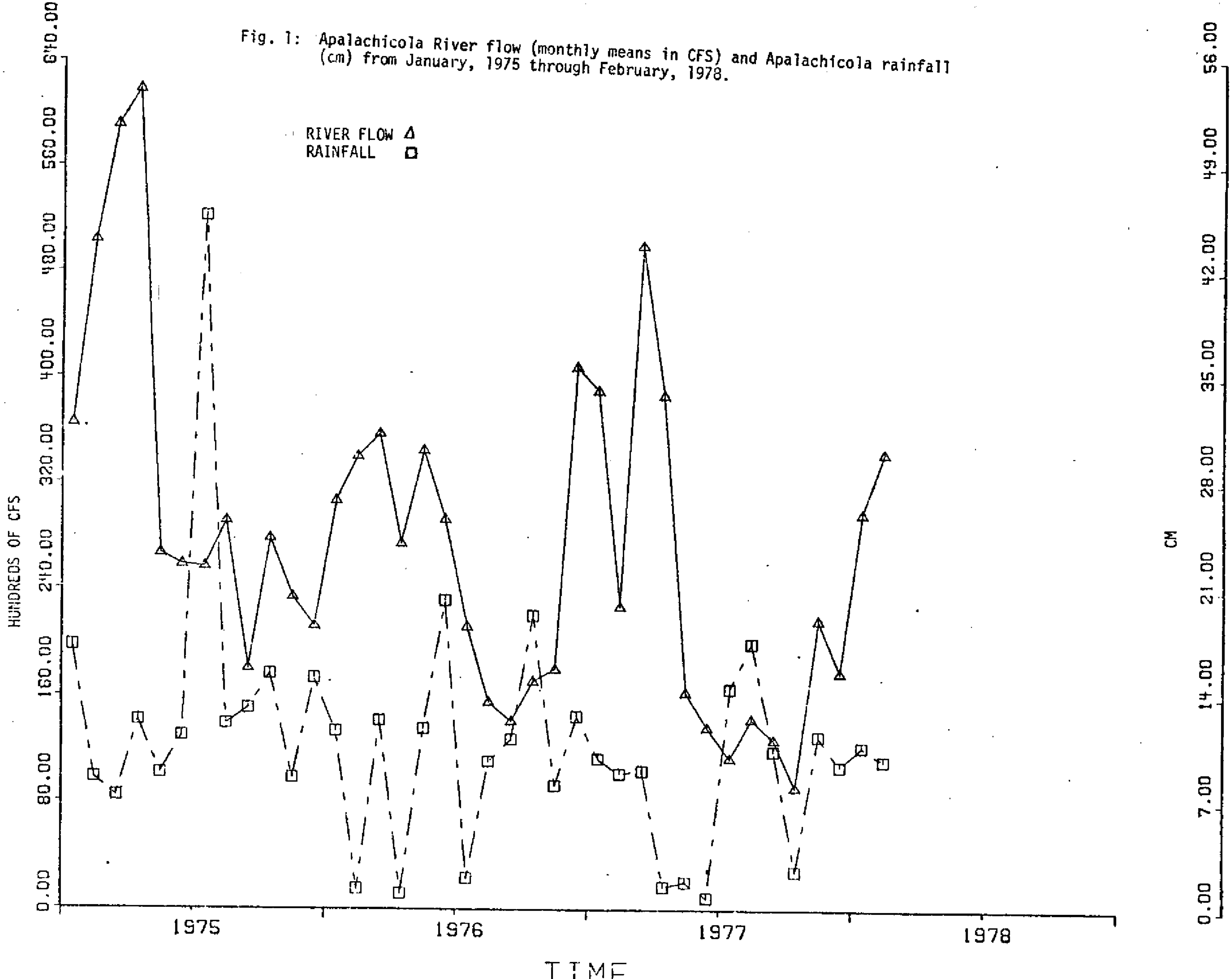
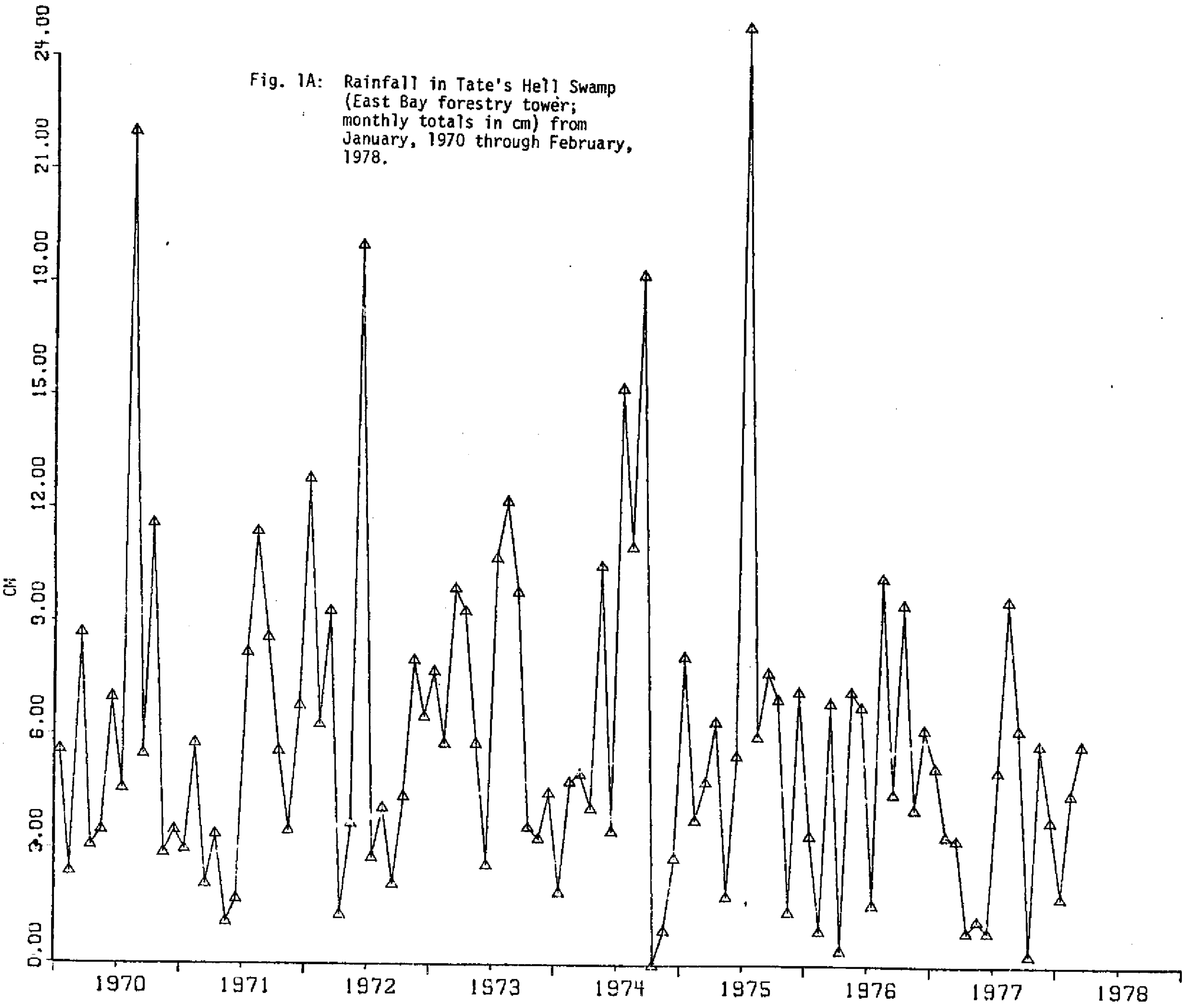


Fig. 1: Apalachicola River flow (monthly means in CFS) and Apalachicola rainfall (cm) from January, 1975 through February, 1978.

Fig. 1A: Rainfall in Tate's Hell Swamp (East Bay forestry tower; monthly totals in cm) from January, 1970 through February, 1978.



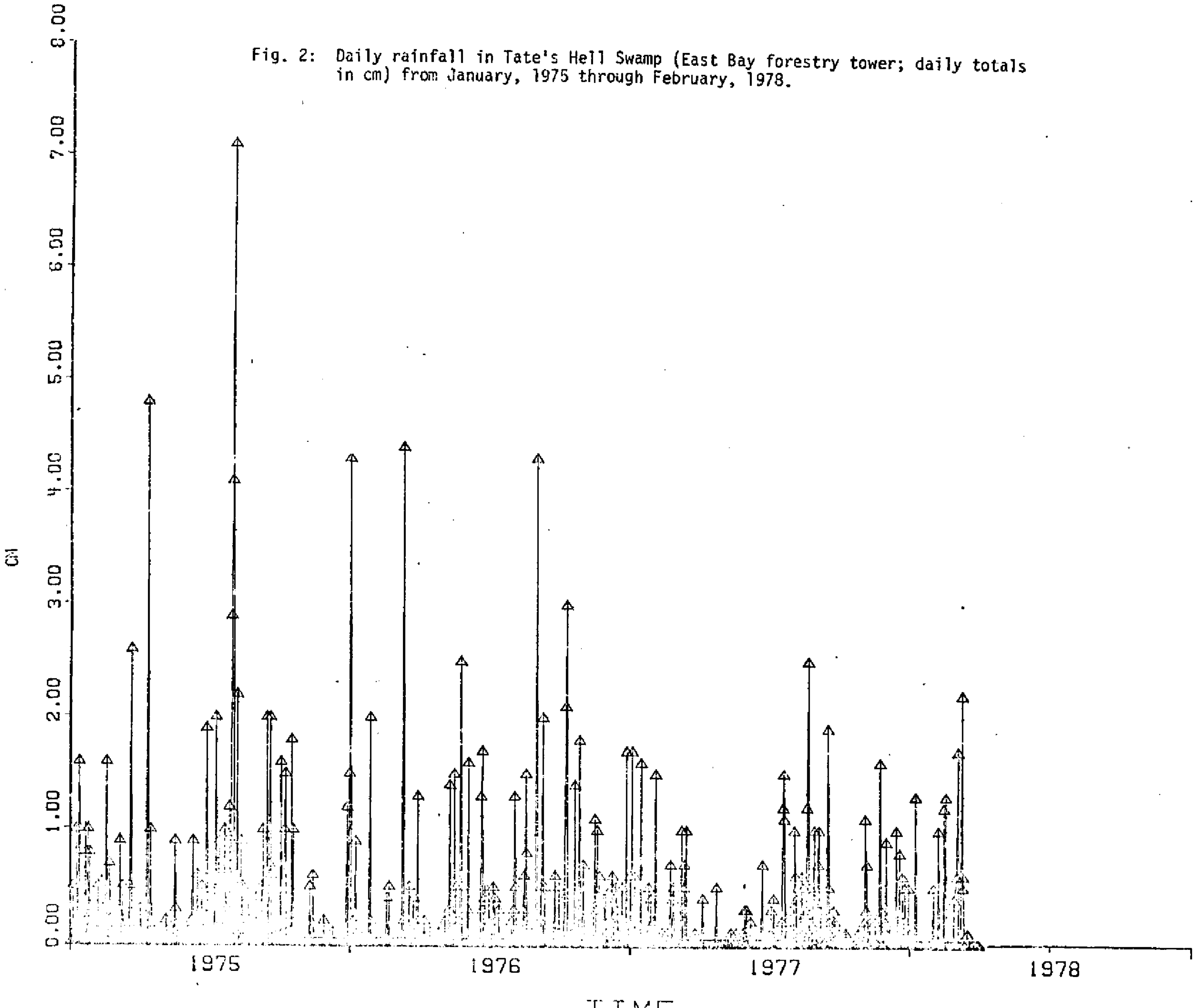


Fig. 2: Daily rainfall in Tate's Hell Swamp (East Bay forestry tower; daily totals in cm) from January, 1975 through February, 1978.

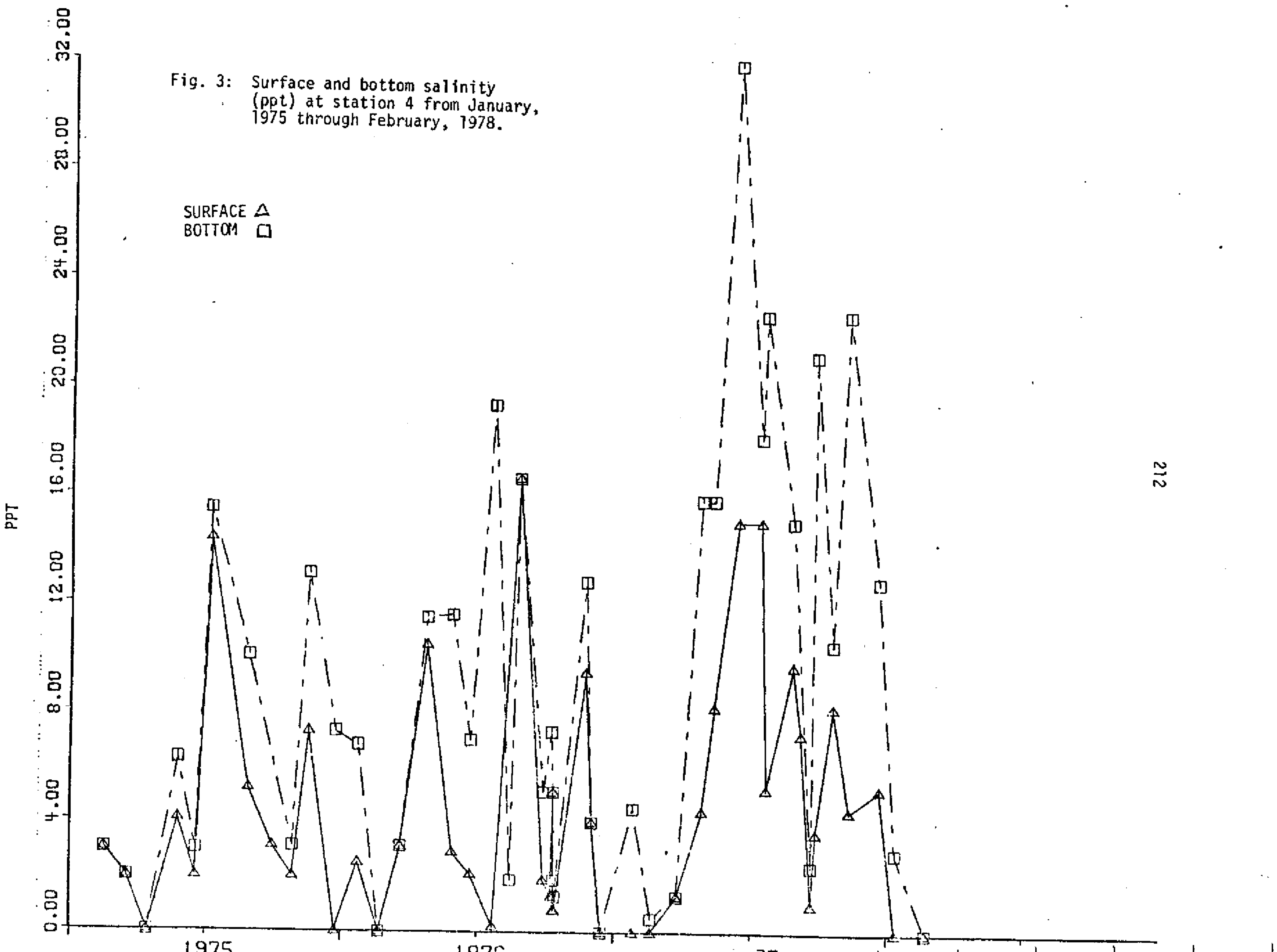


Fig. 3: Surface and bottom salinity (ppt) at station 4 from January, 1975 through February, 1978.

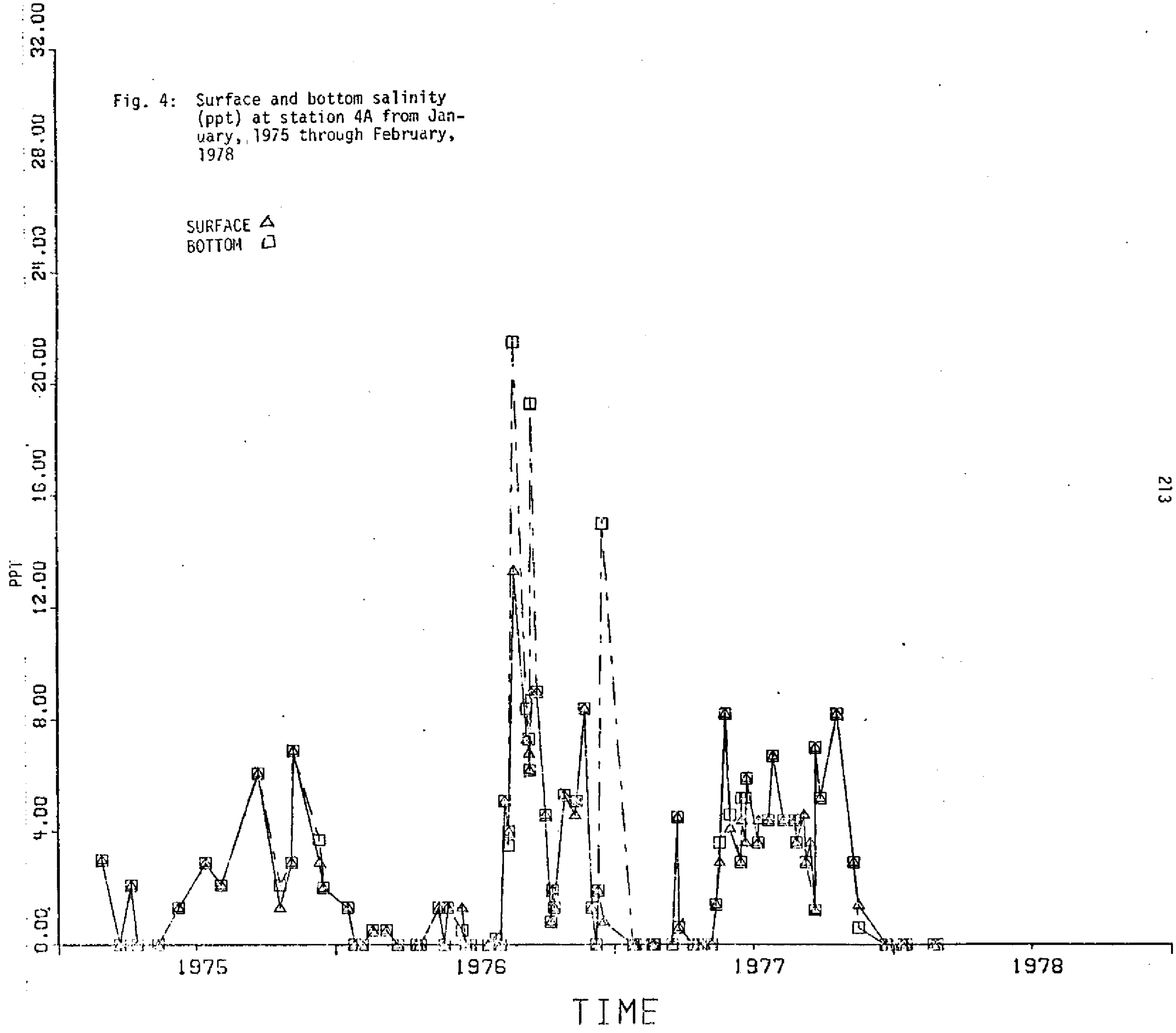


Fig. 4: Surface and bottom salinity (ppt) at station 4A from January, 1975 through February, 1978

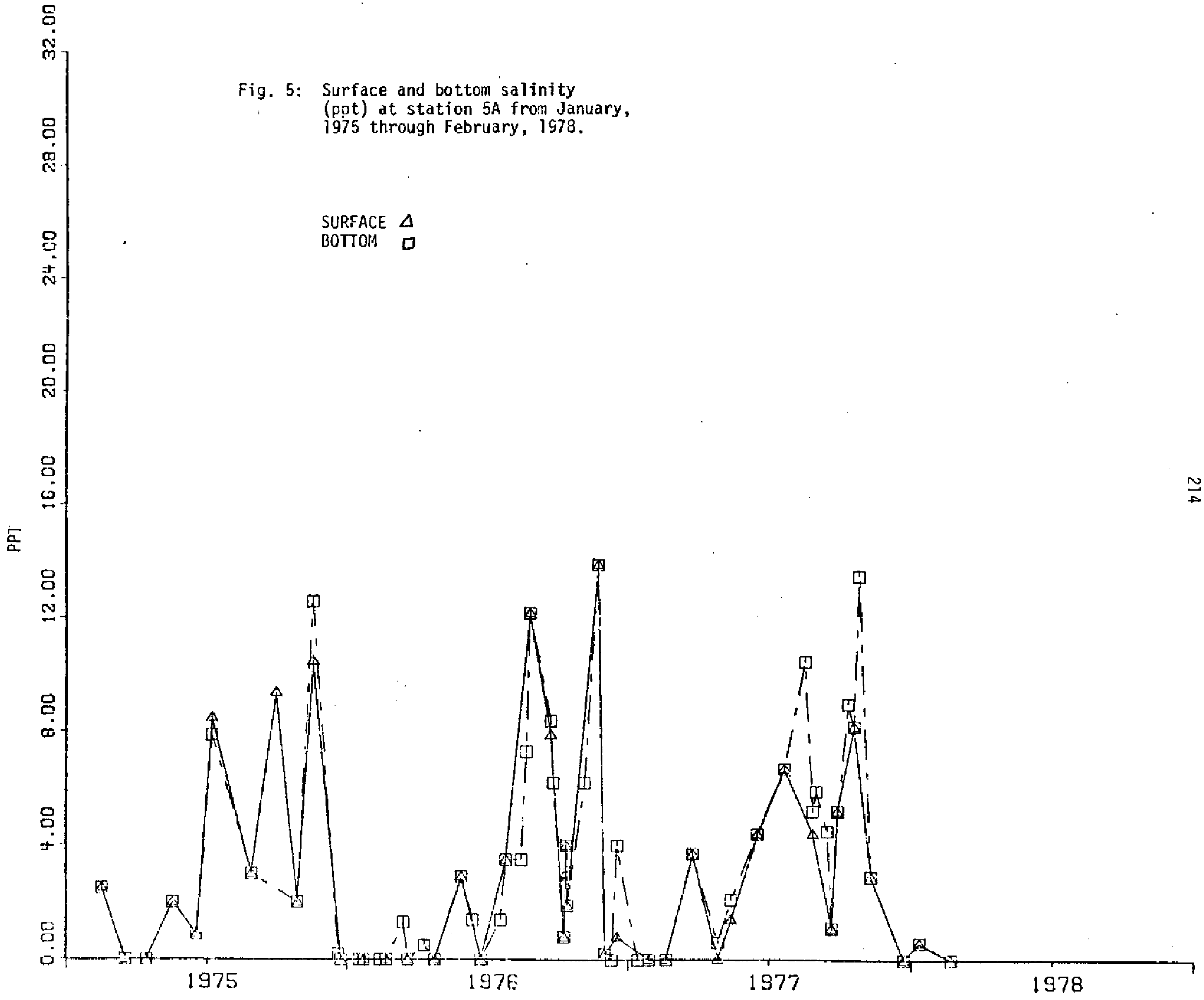


Fig. 5: Surface and bottom salinity (ppt) at station 5A from January, 1975 through February, 1978.

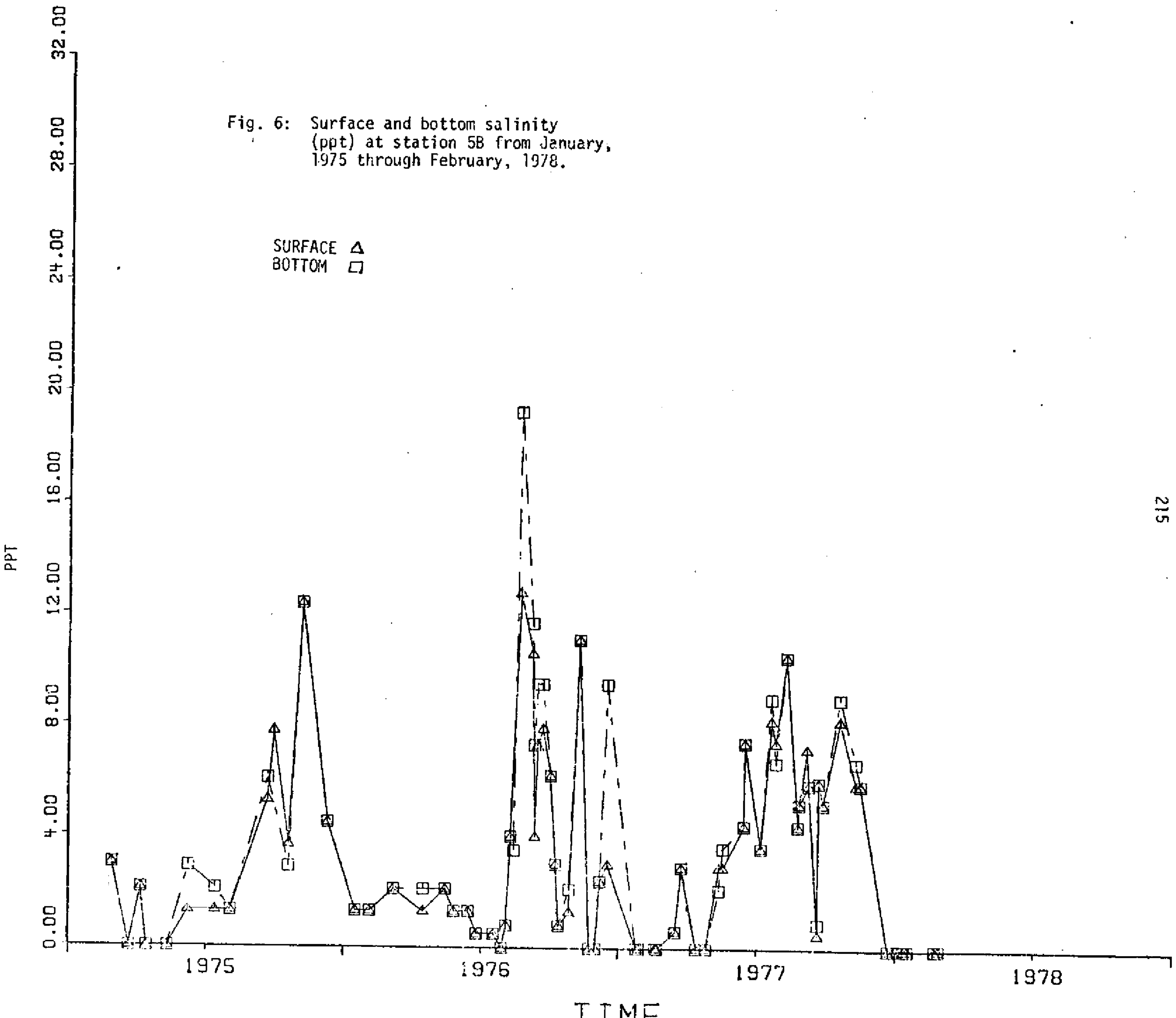


Fig. 6: Surface and bottom salinity (ppt) at station 5B from January, 1975 through February, 1978.

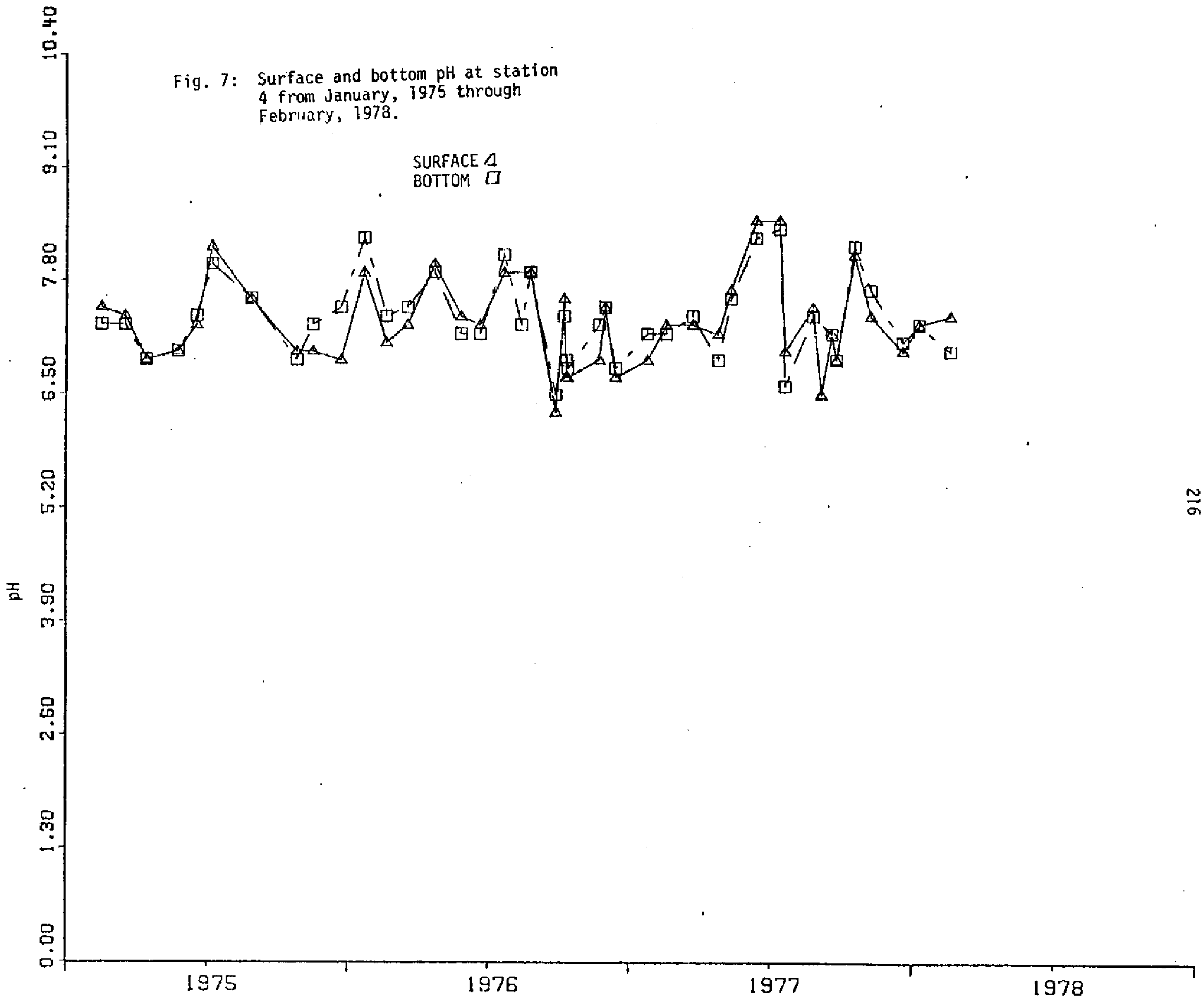


Fig. 7: Surface and bottom pH at station 4 from January, 1975 through February, 1978.

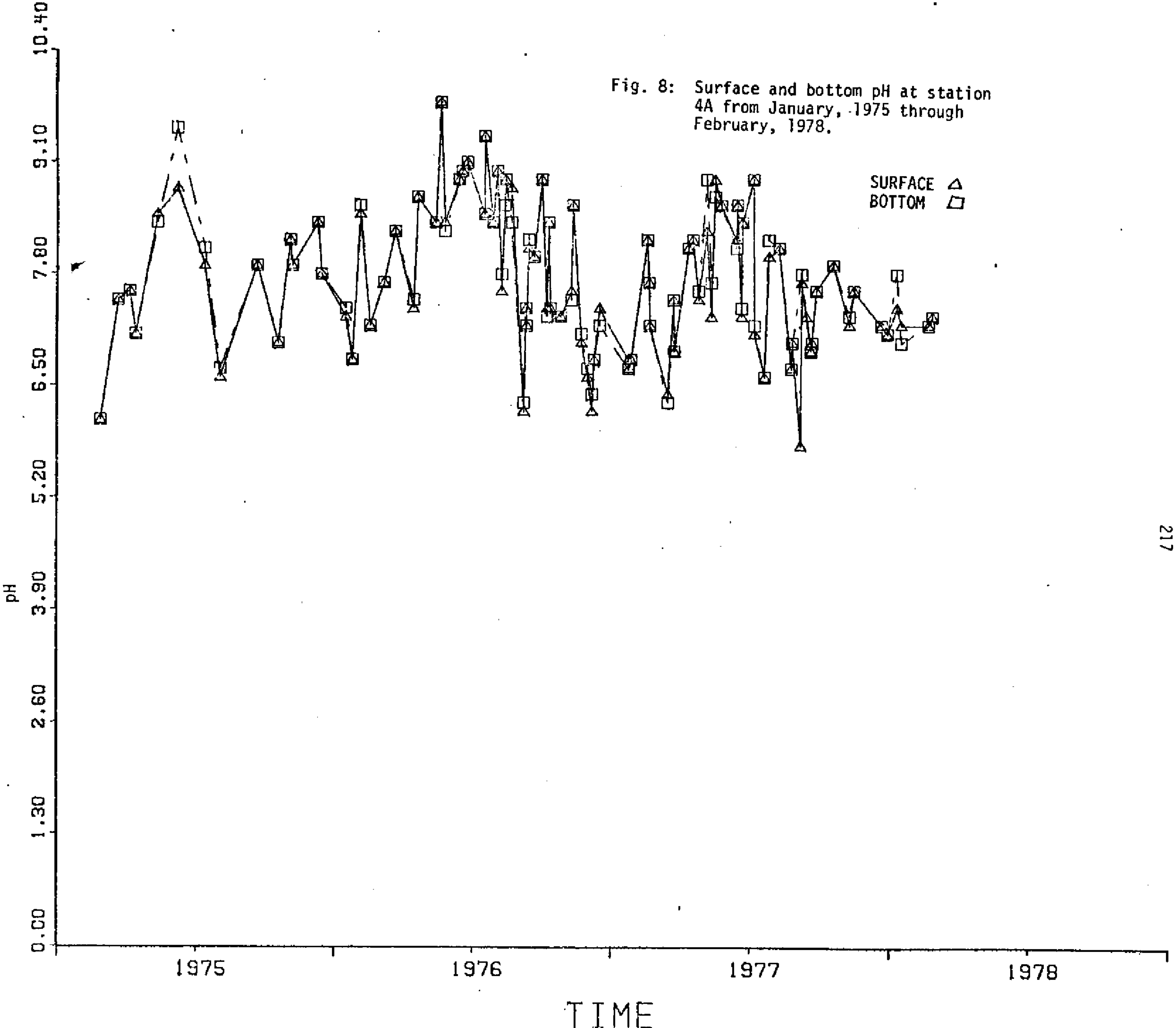


Fig. 8: Surface and bottom pH at station 4A from January, 1975 through February, 1978.

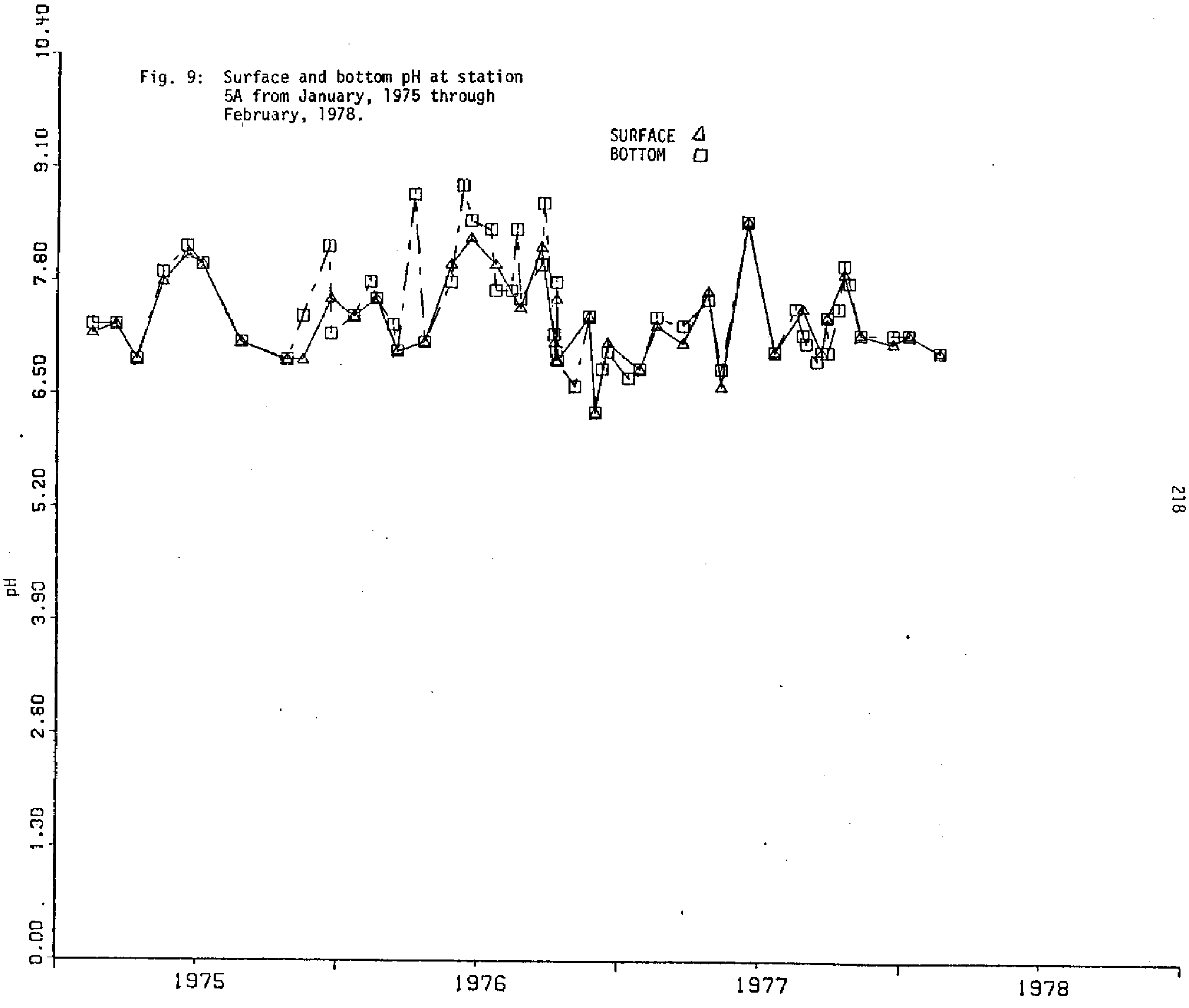


Fig. 9: Surface and bottom pH at station 5A from January, 1975 through February, 1978.

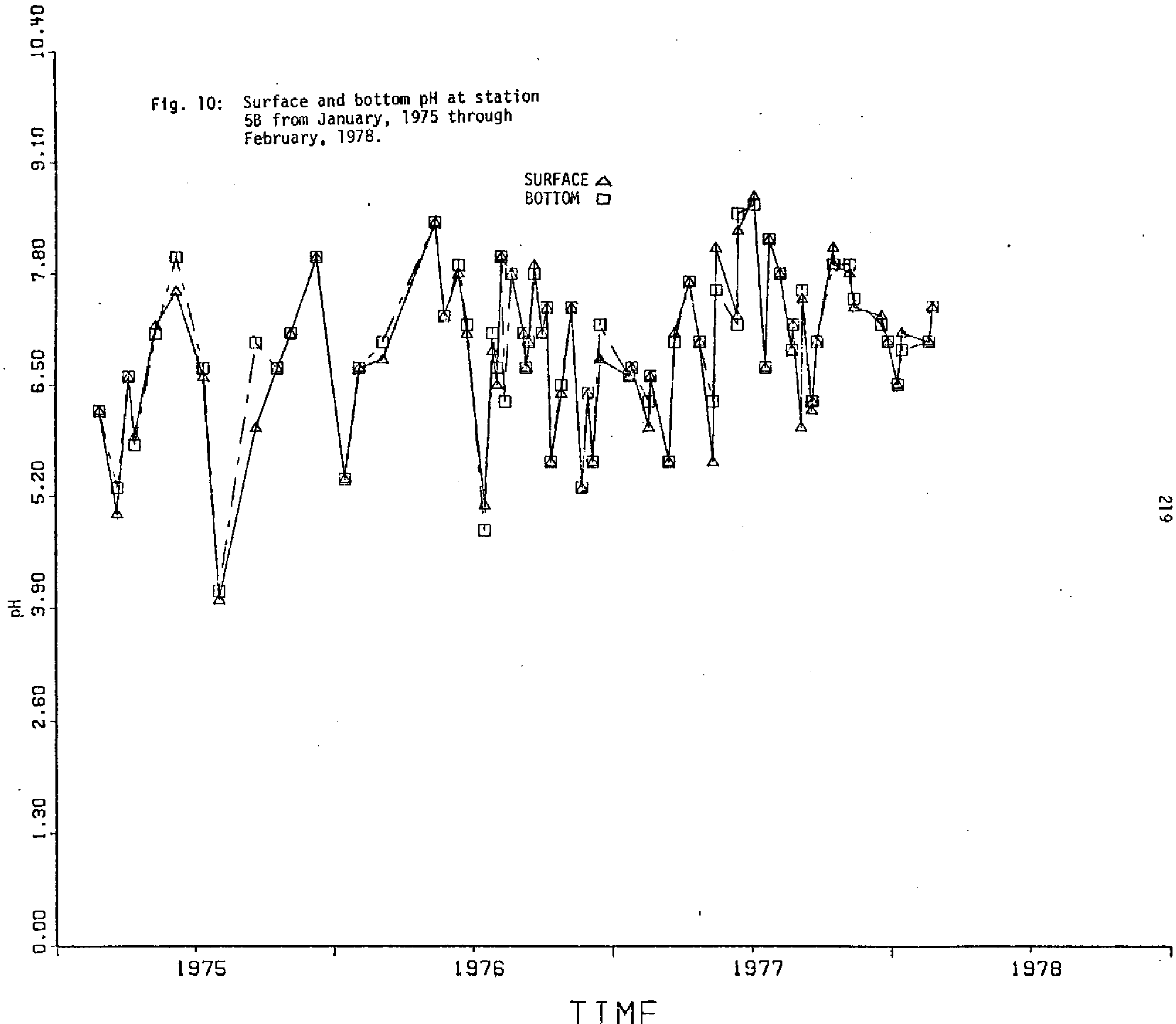


Fig. 10: Surface and bottom pH at station 5B from January, 1975 through February, 1978.

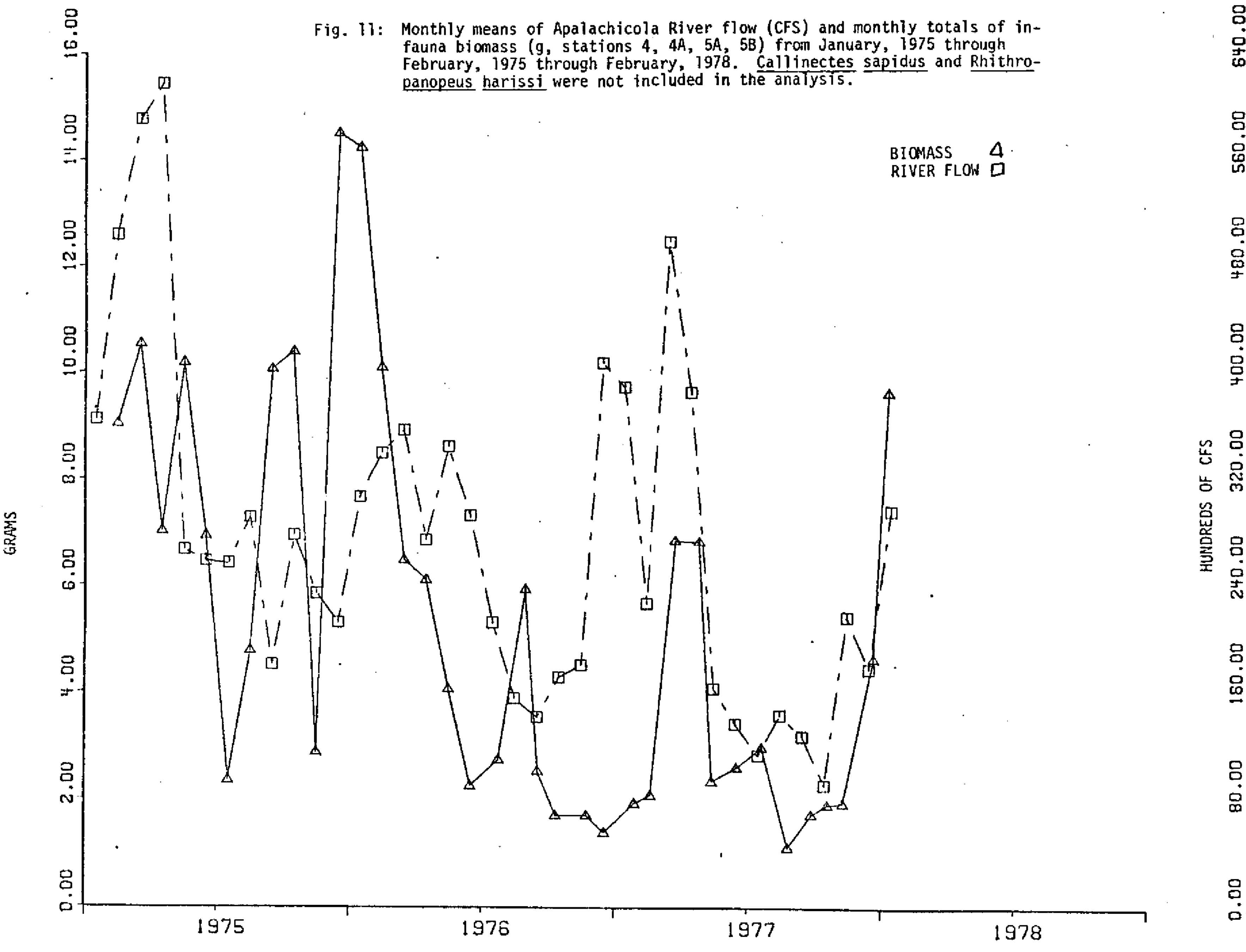


Fig. 11: Monthly means of Apalachicola River flow (CFS) and monthly totals of infauna biomass (g, stations 4, 4A, 5A, 5B) from January, 1975 through February, 1975 through February, 1978. *Callinectes sapidus* and *Rhithropanopeus harissi* were not included in the analysis.

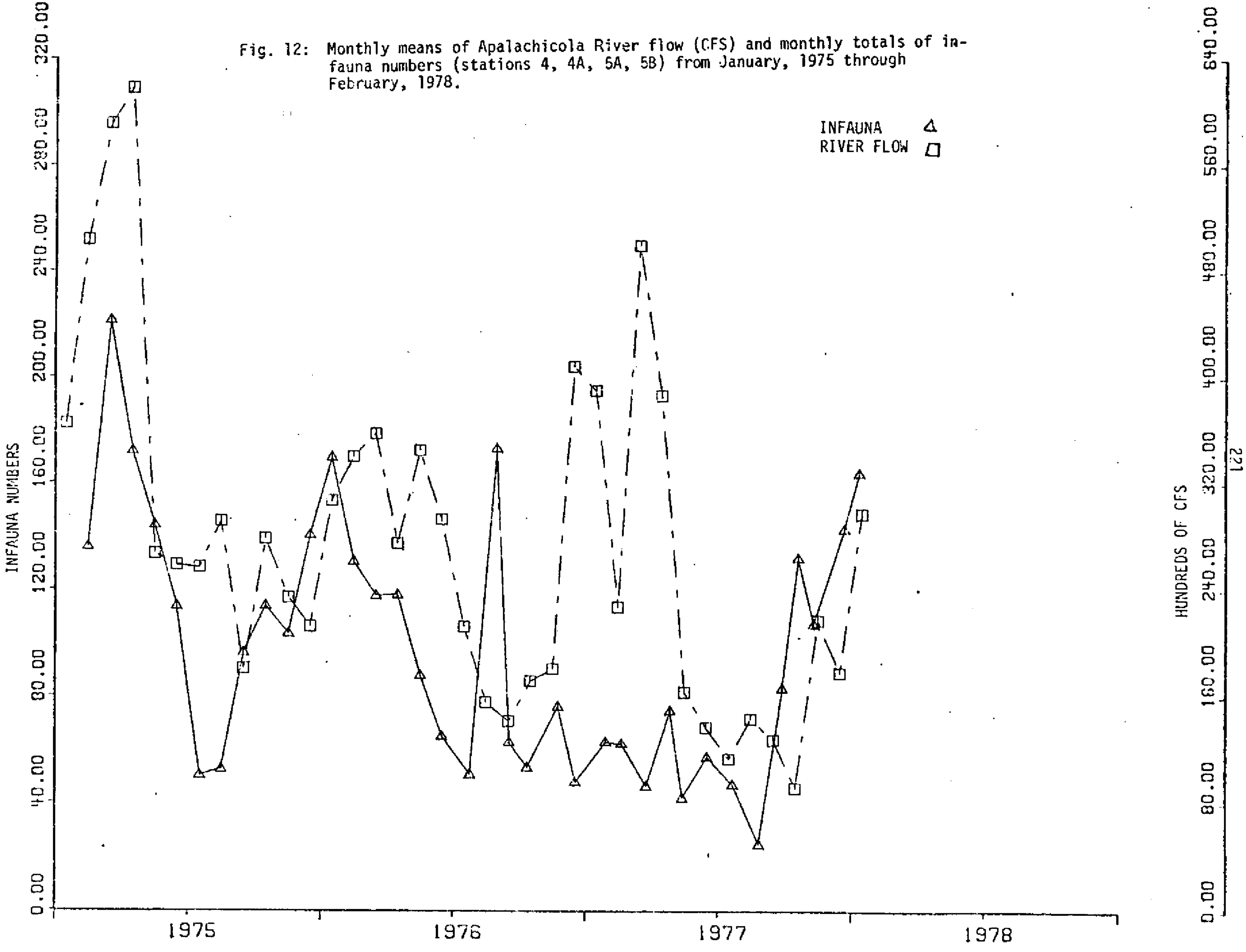


Fig. 12: Monthly means of Apalachicola River flow (CFS) and monthly totals of infauna numbers (stations 4, 4A, 5A, 5B) from January, 1975 through February, 1978.

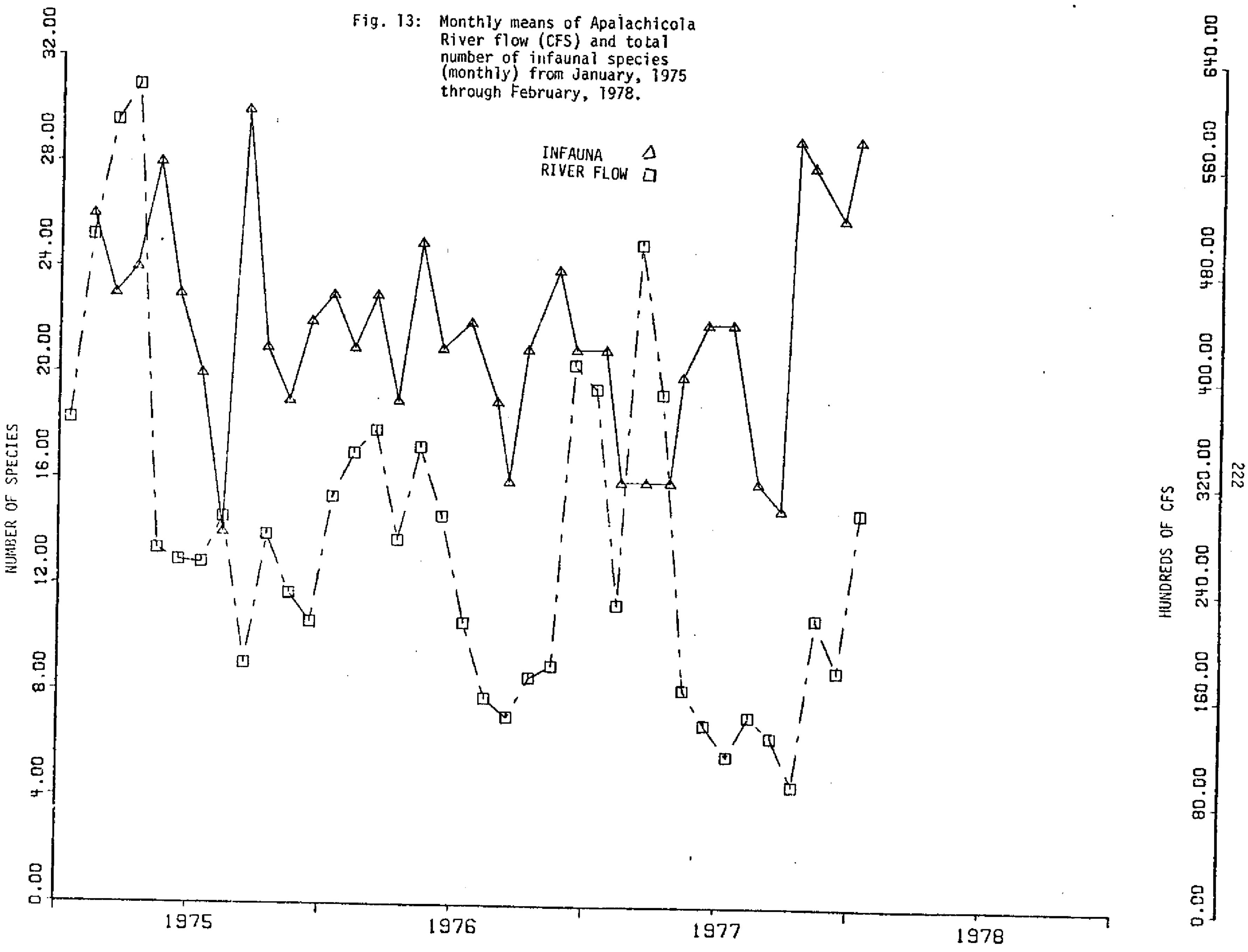


Fig. 13: Monthly means of Apalachicola River flow (CFS) and total number of infaunal species (monthly) from January, 1975 through February, 1978.

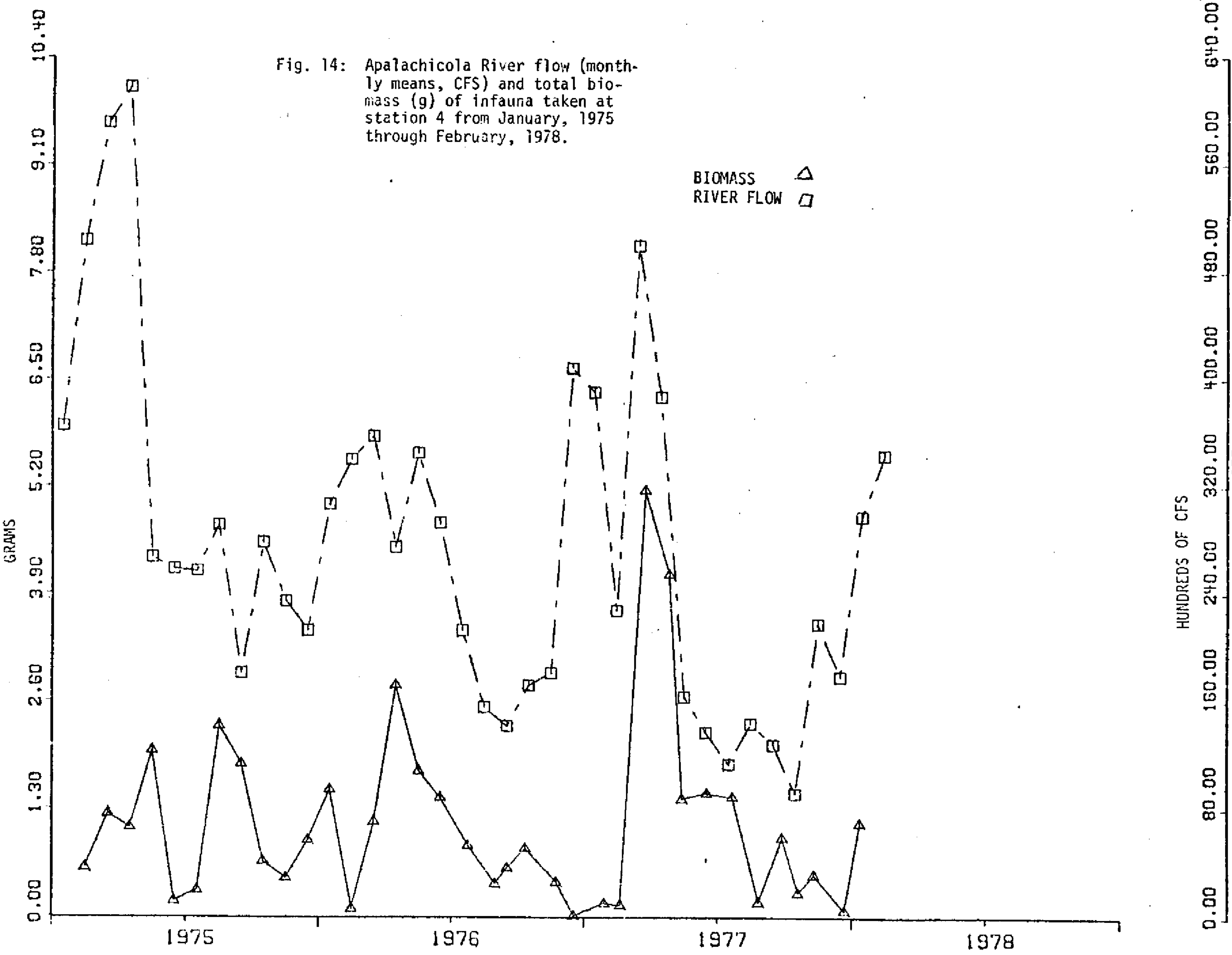


Fig. 14: Apalachicola River flow (monthly means, CFS) and total biomass (g) of infauna taken at station 4 from January, 1975 through February, 1978.

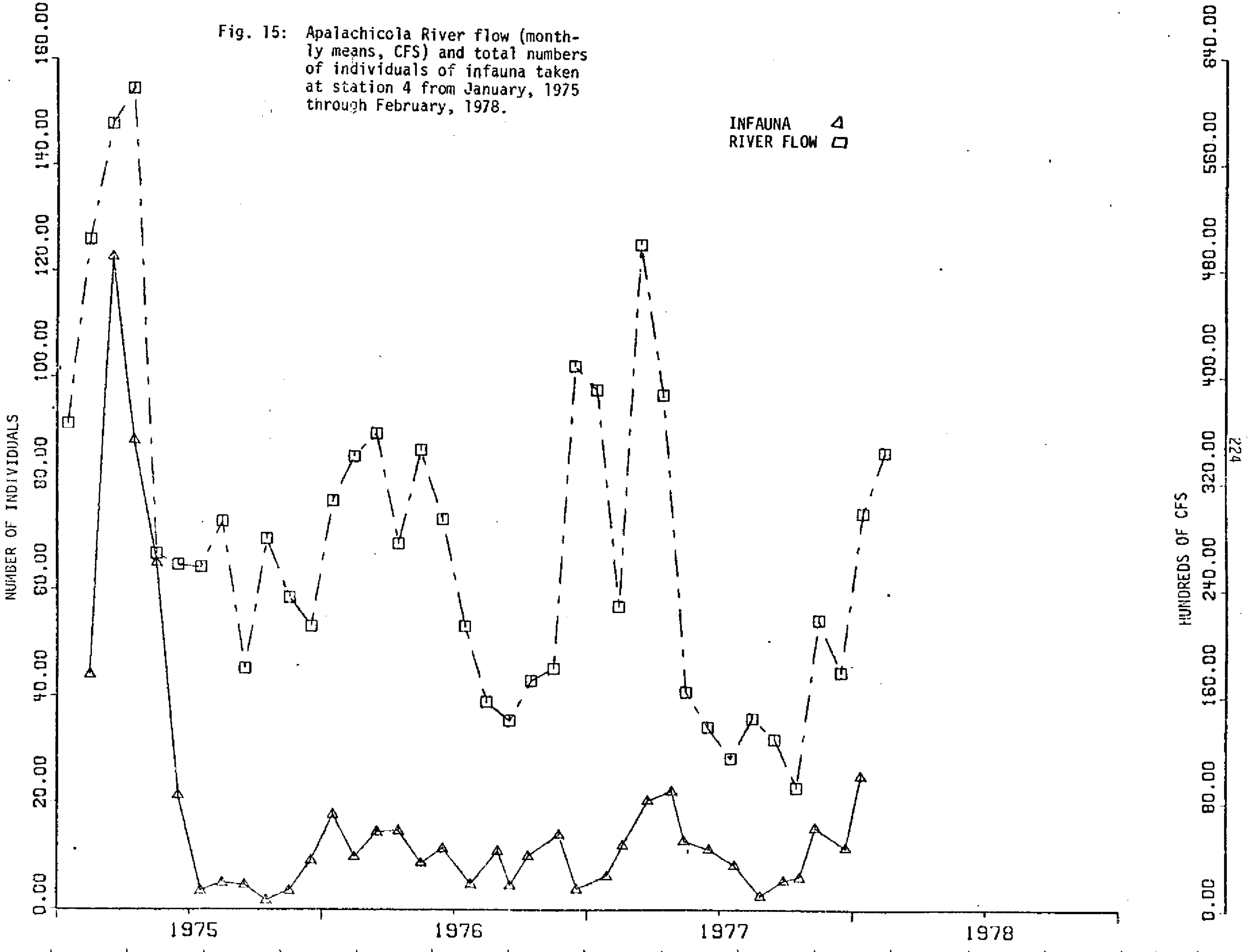


Fig. 15: Apalachicola River flow (monthly means, CFS) and total numbers of individuals of infauna taken at station 4 from January, 1975 through February, 1978.

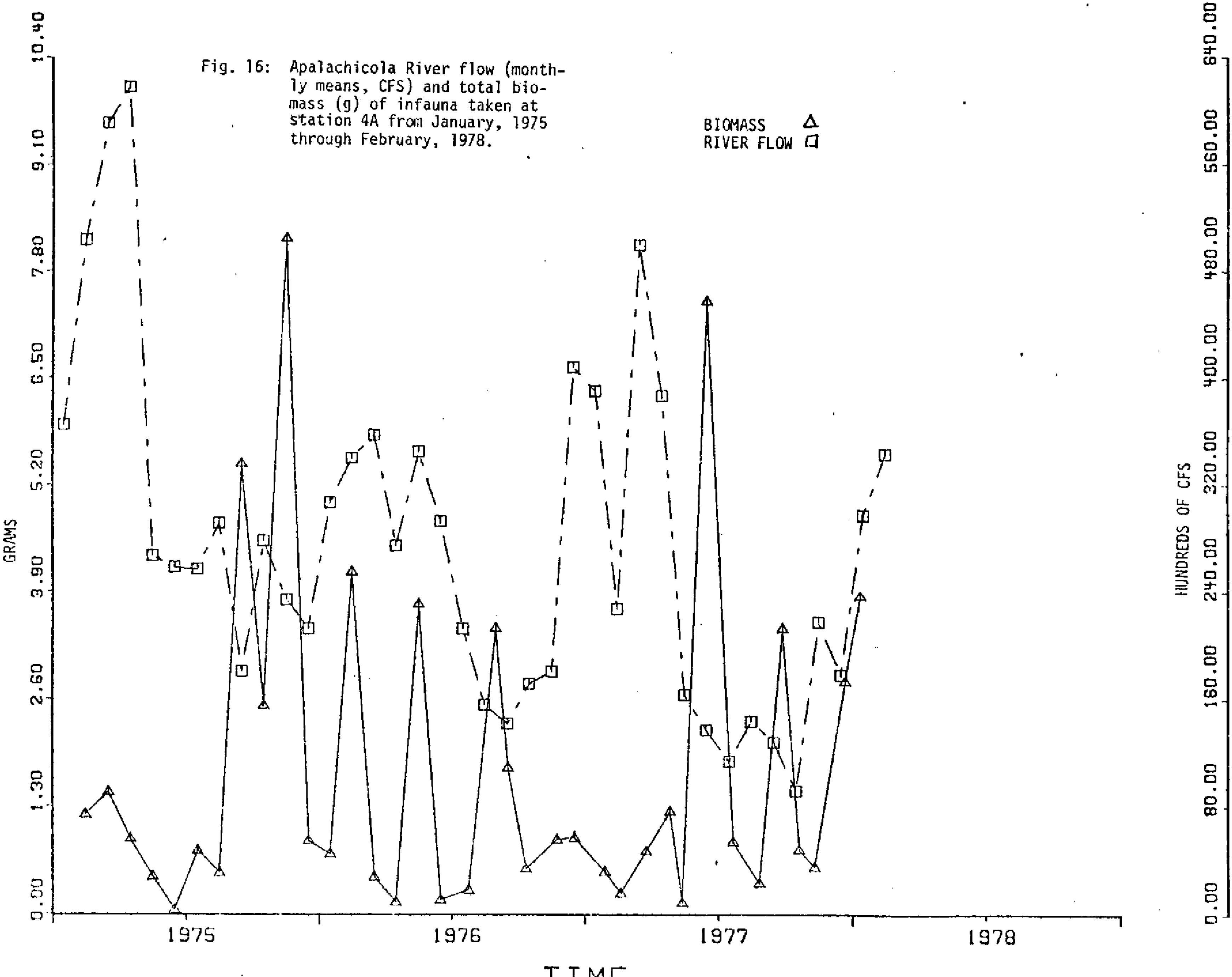


Fig. 16: Apalachicola River flow (monthly means, CFS) and total biomass (g) of infauna taken at station 4A from January, 1975 through February, 1978.

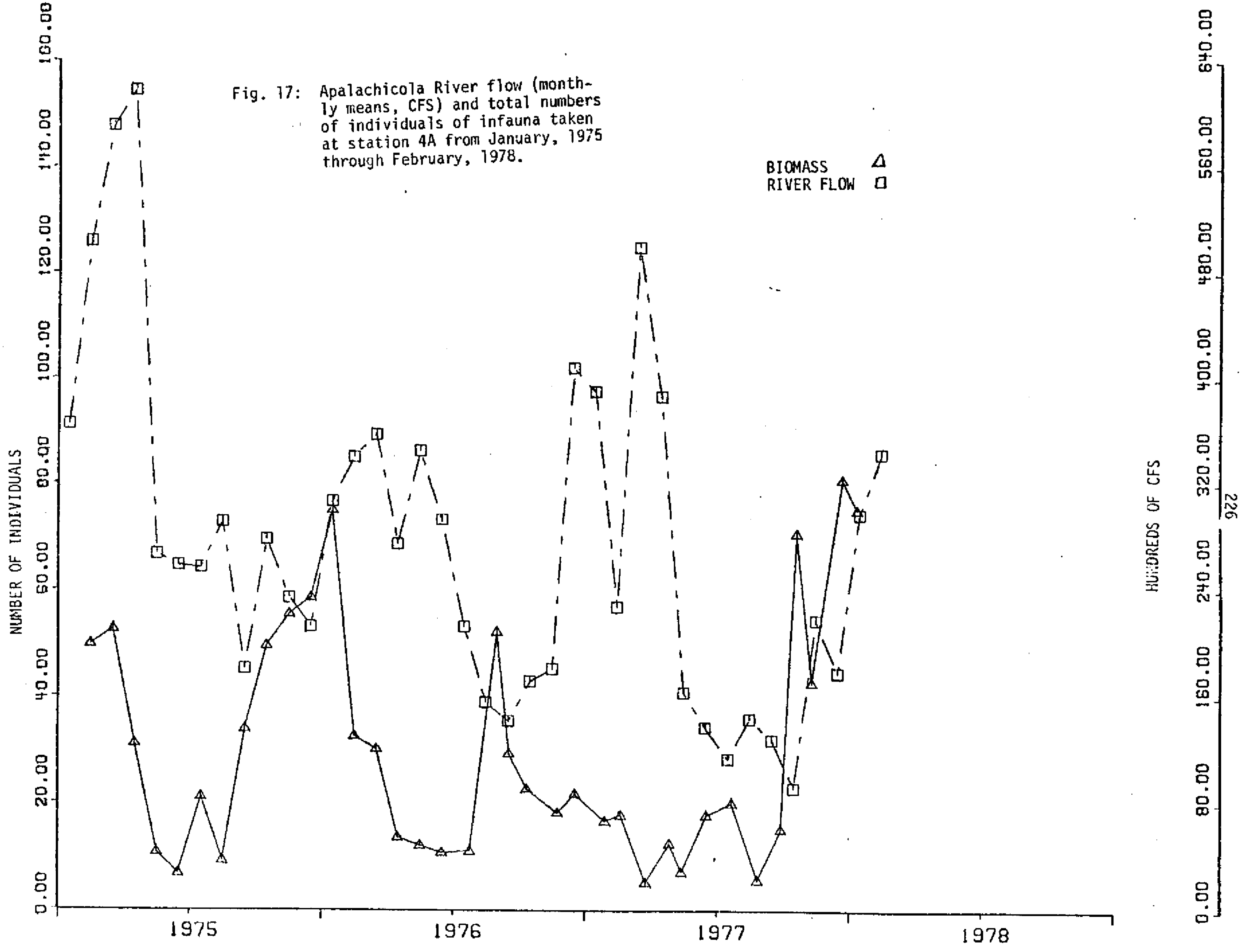


Fig. 17: Apalachicola River flow (monthly means, CFS) and total numbers of individuals of infauna taken at station 4A from January, 1975 through February, 1978.

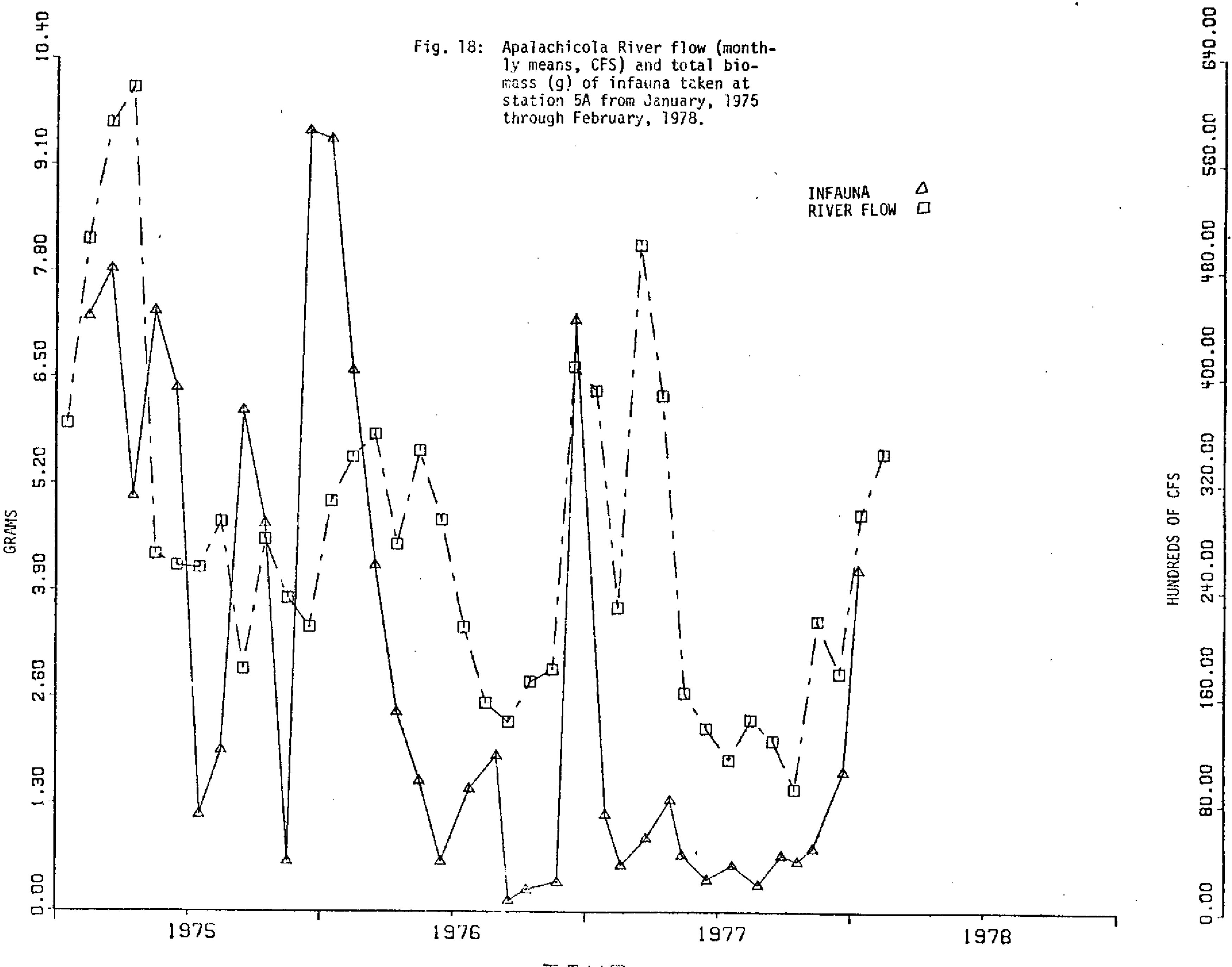


Fig. 18: Apalachicola River flow (monthly means, CFS) and total biomass (g) of infauna taken at station 5A from January, 1975 through February, 1978.

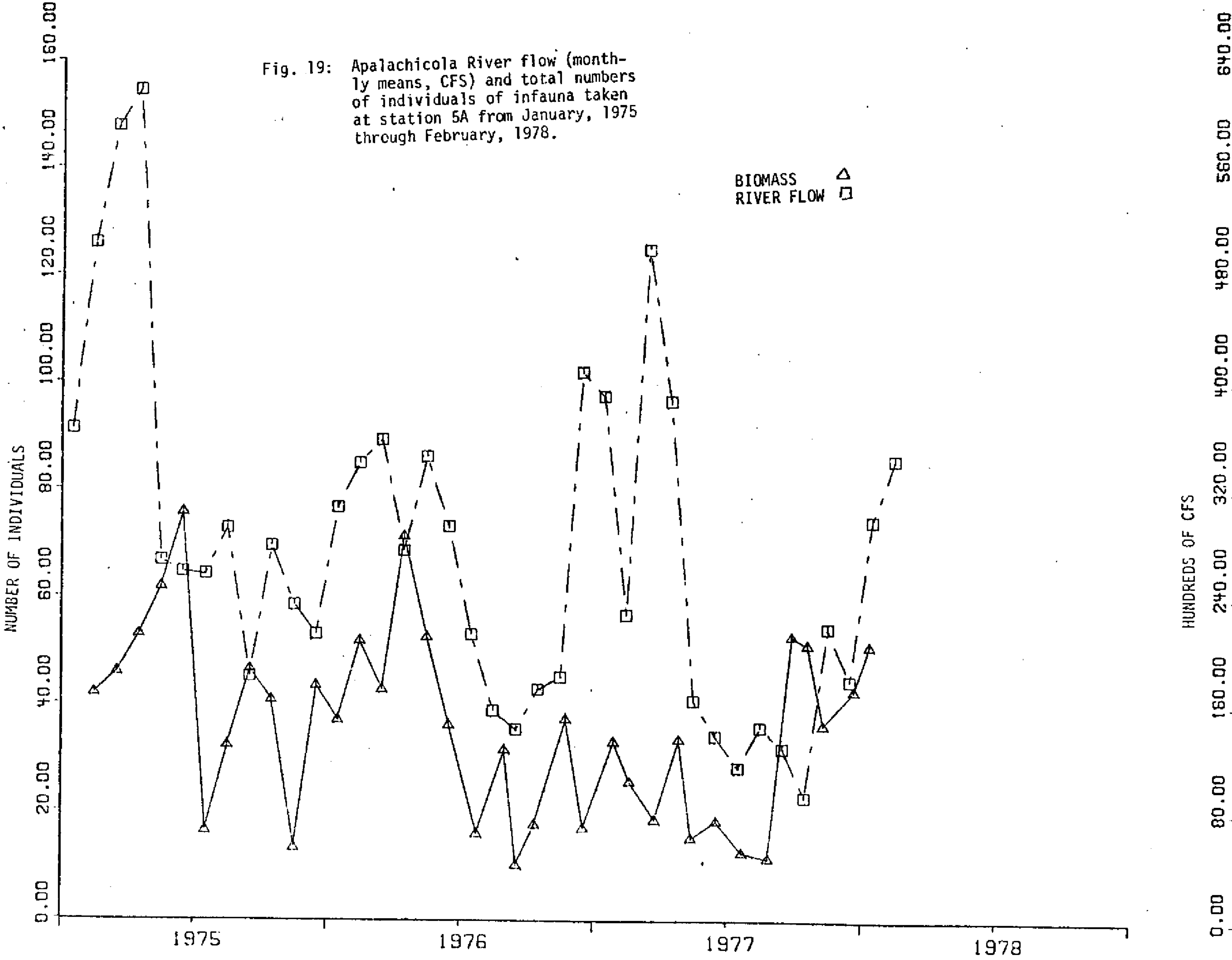


Fig. 19: Apalachicola River flow (monthly means, CFS) and total numbers of individuals of infauna taken at station 5A from January, 1975 through February, 1978.

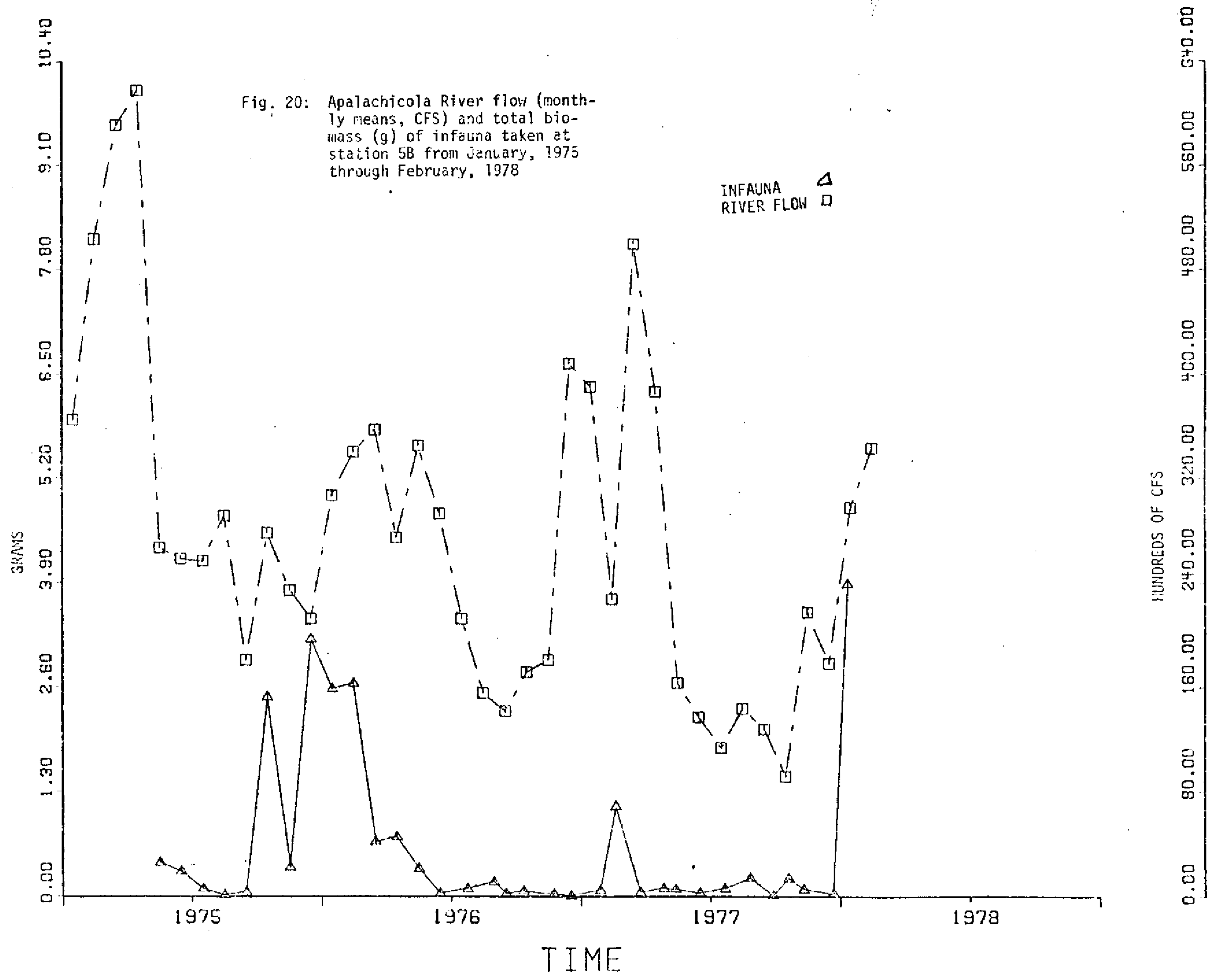


Fig. 20: Apalachicola River flow (monthly means, CFS) and total biomass (g) of infauna taken at station 5B from January, 1975 through February, 1978

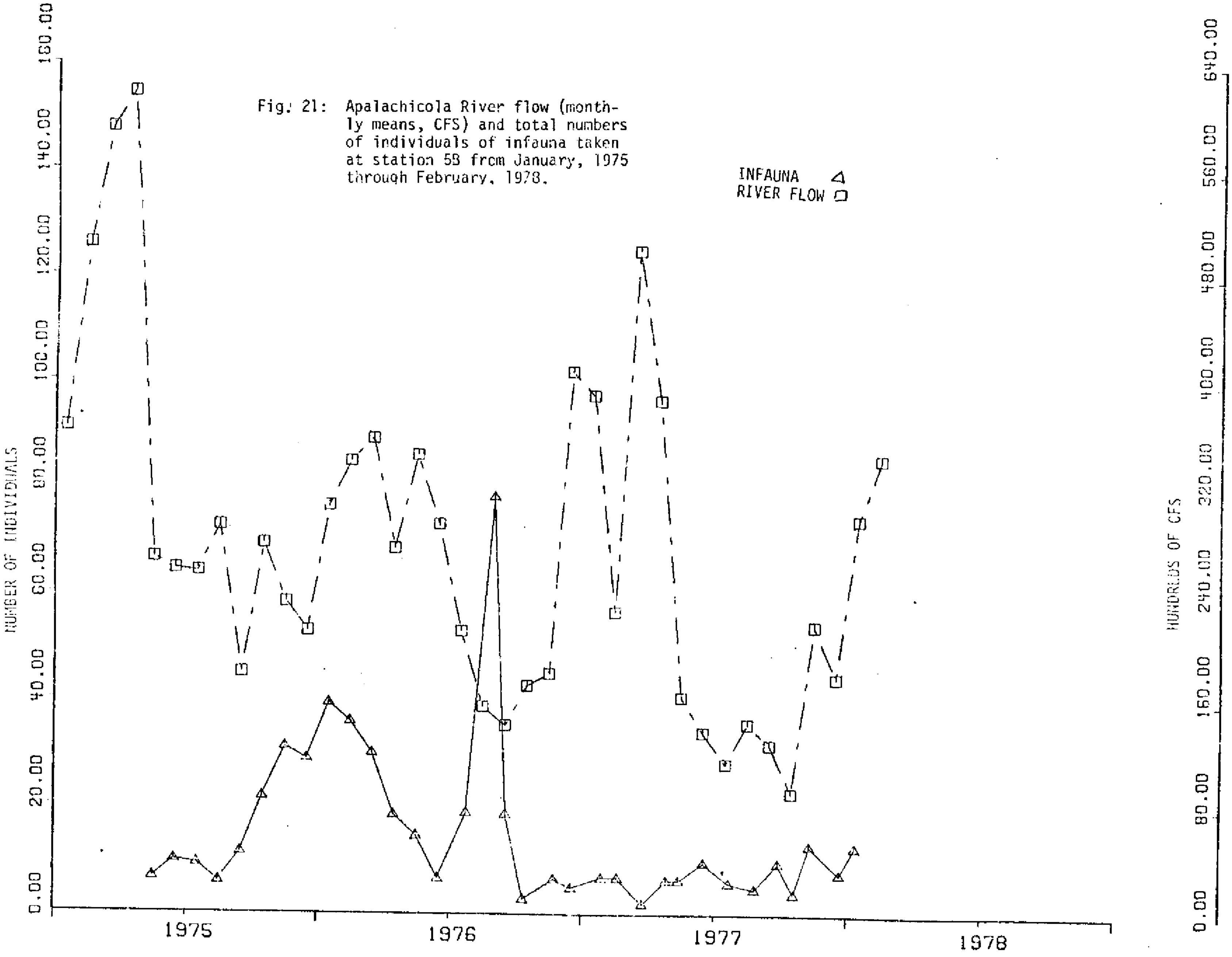


Fig. 21: Apalachicola River flow (monthly means, CFS) and total numbers of individuals of infauna taken at station 5B from January, 1975 through February, 1978.

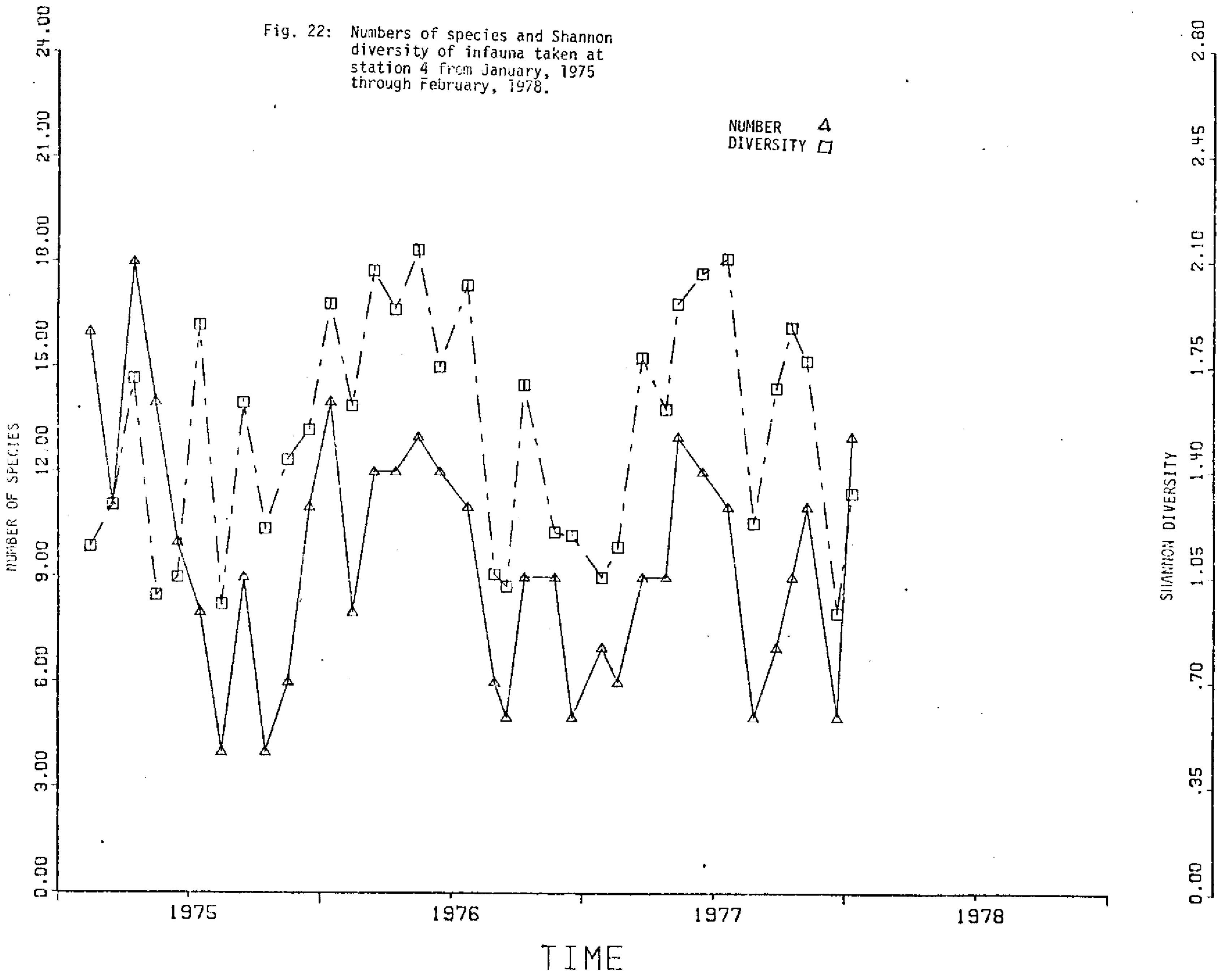


Fig. 22: Numbers of species and Shannon diversity of infauna taken at station 4 from January, 1975 through February, 1978.

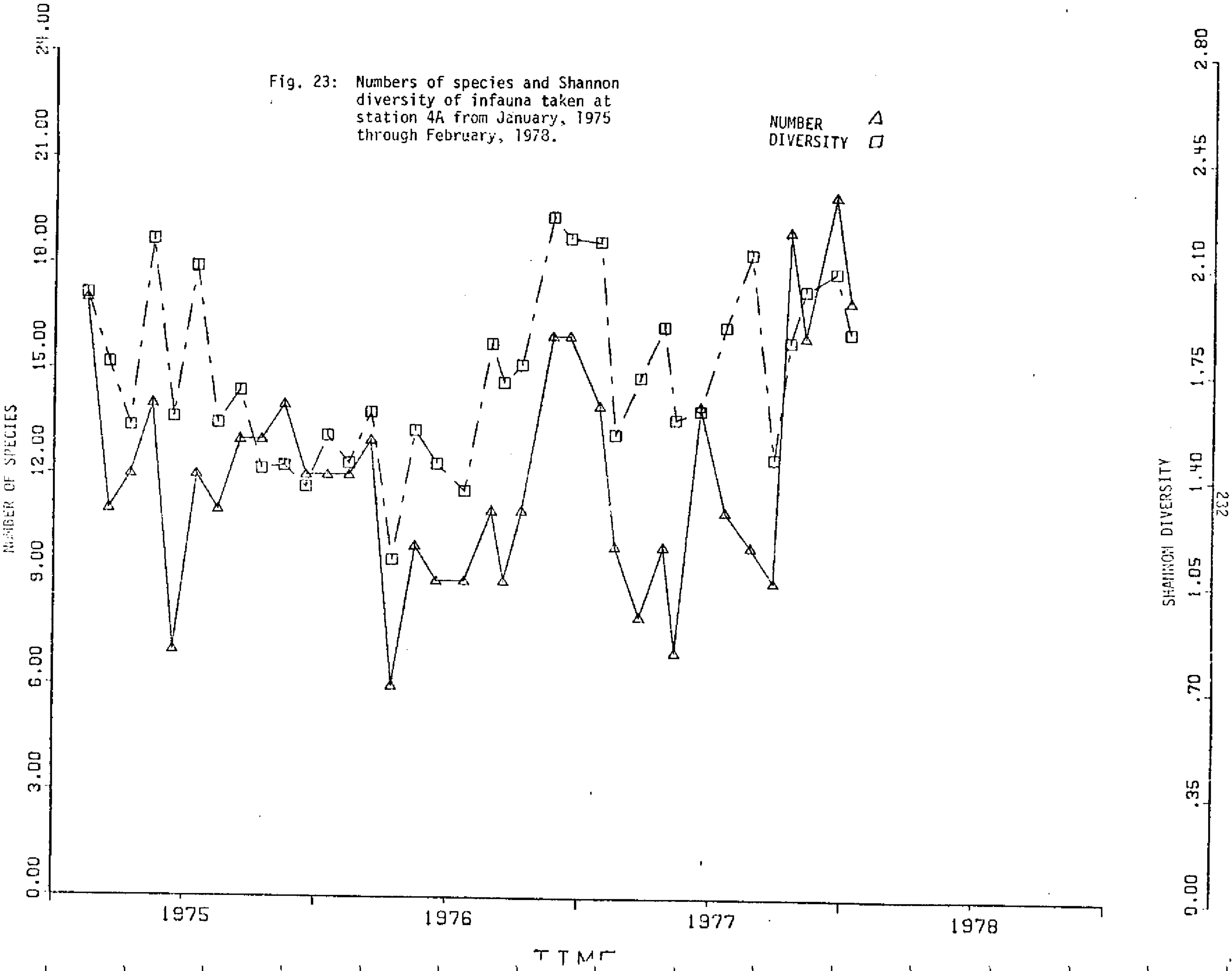


Fig. 23: Numbers of species and Shannon diversity of infauna taken at station 4A from January, 1975 through February, 1978.

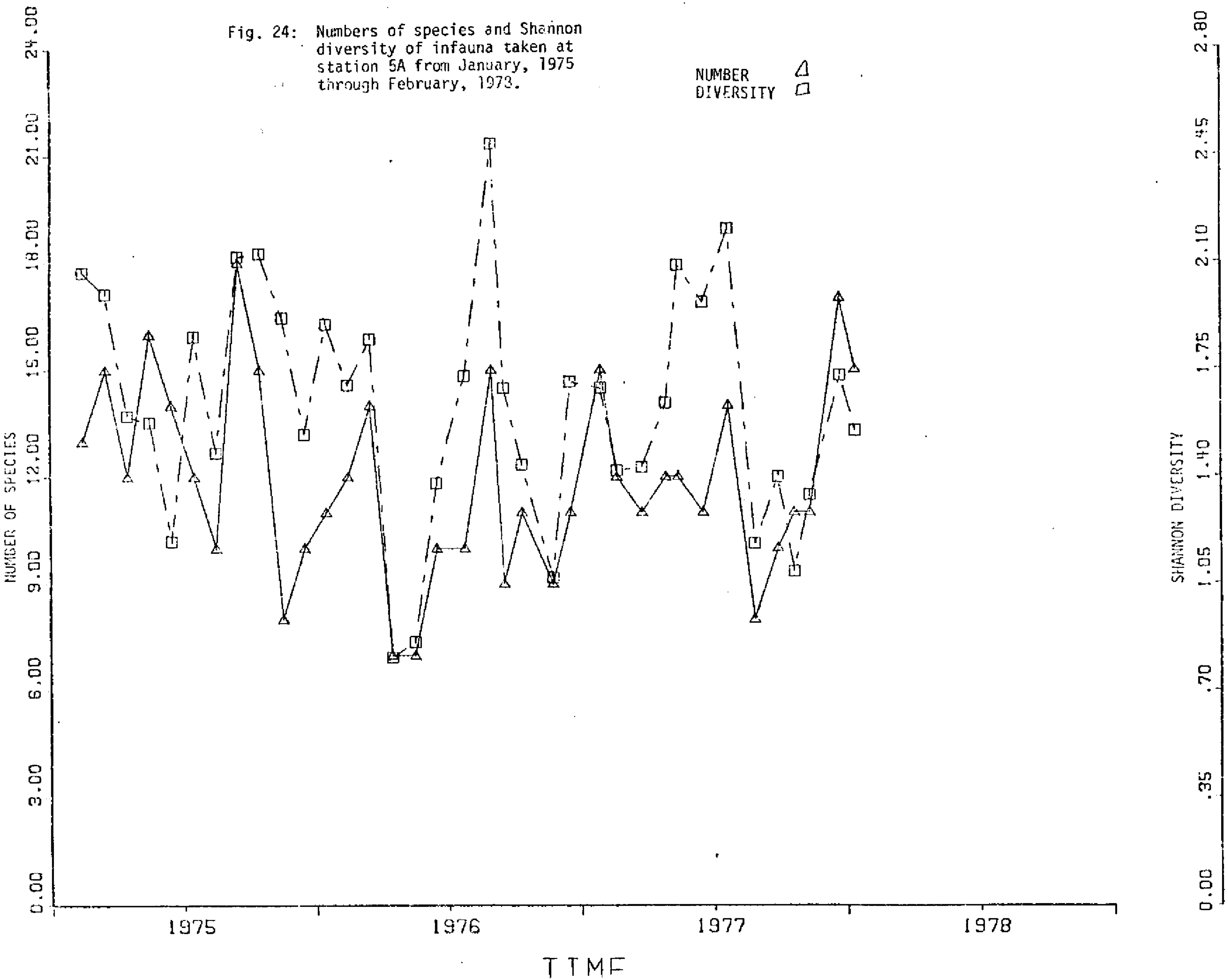


Fig. 24: Numbers of species and Shannon diversity of infauna taken at station 5A from January, 1975 through February, 1973.

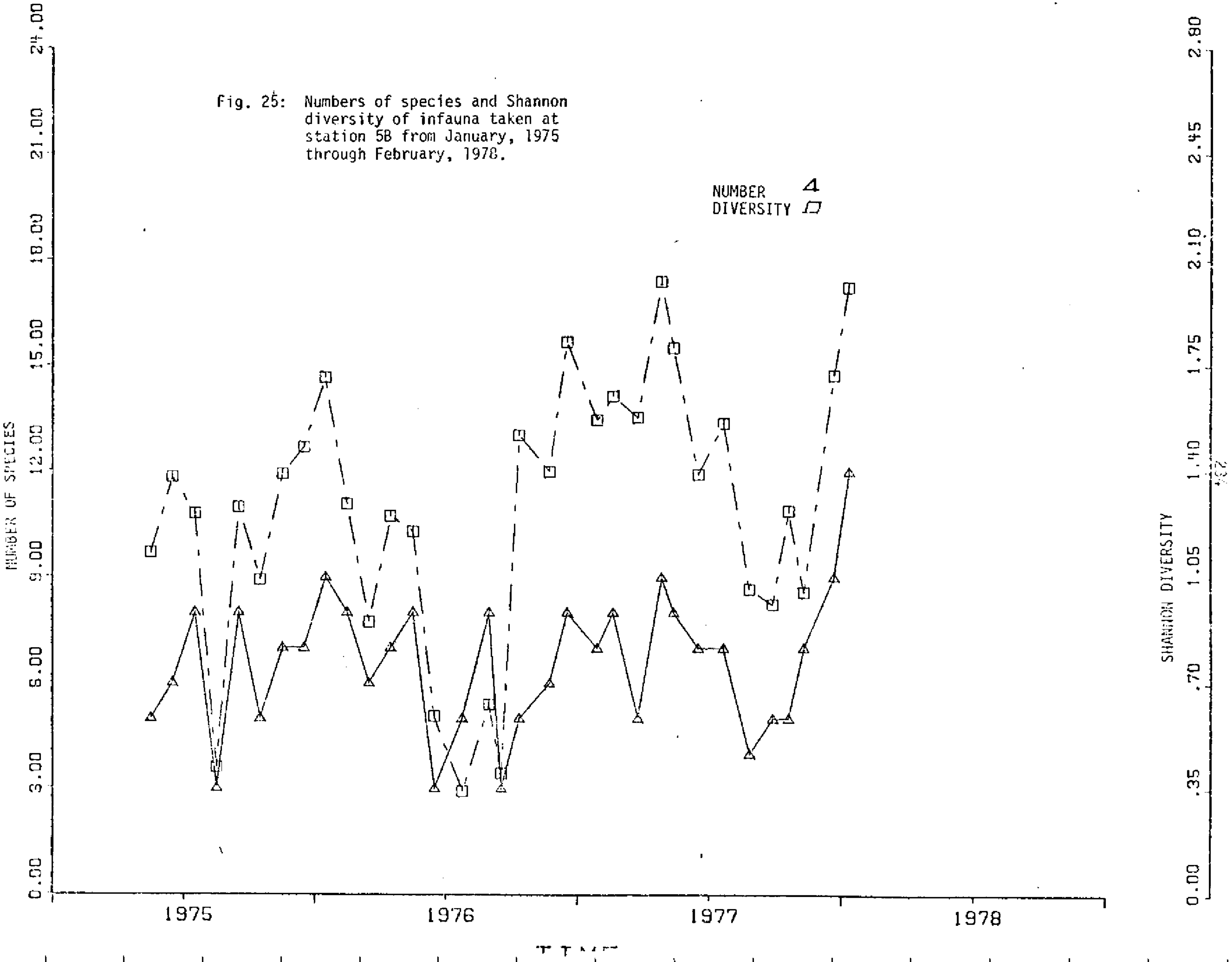


Fig. 25: Numbers of species and Shannon diversity of infauna taken at station 5B from January, 1975 through February, 1978.

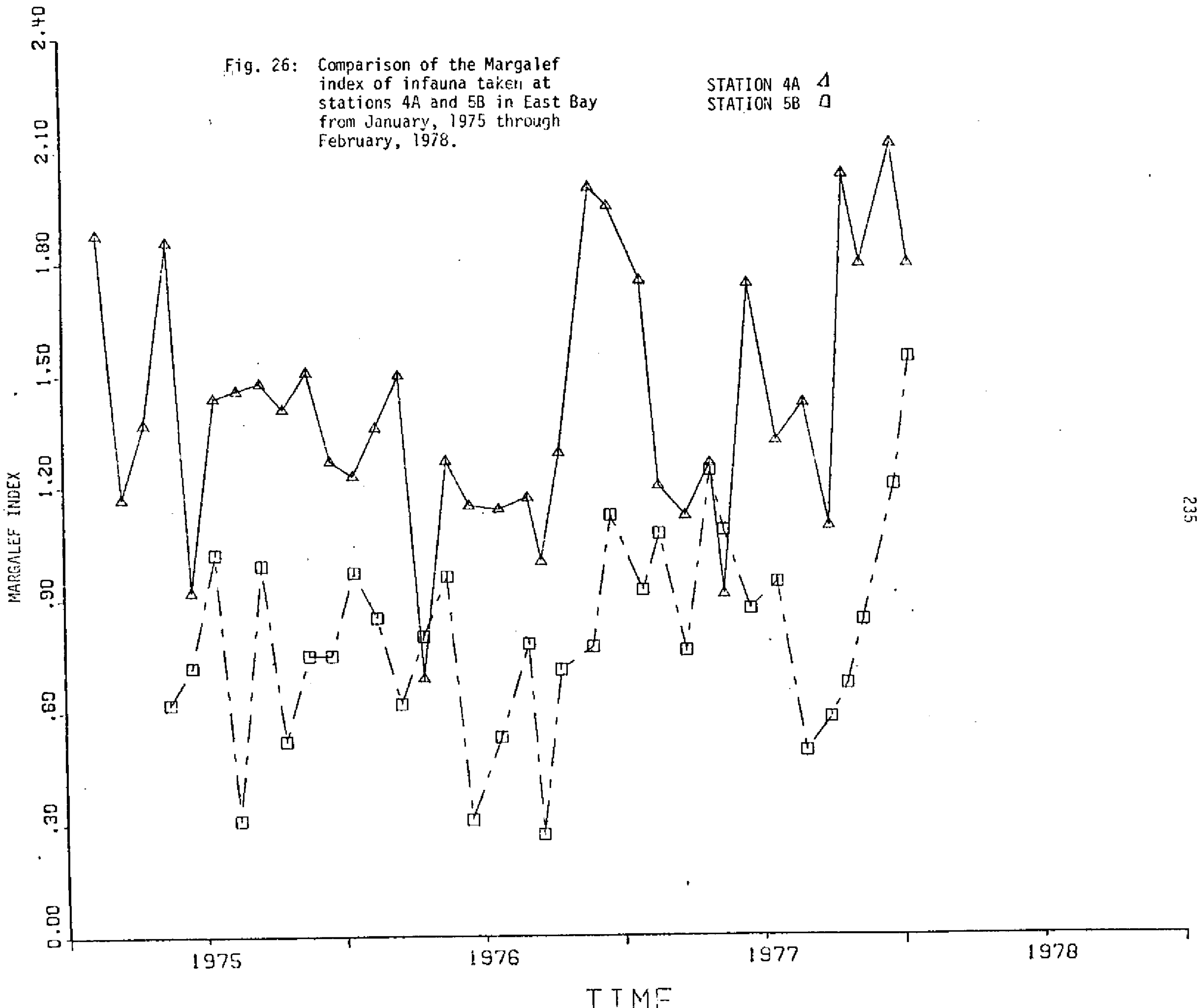


Fig. 26: Comparison of the Margalef index of infauna taken at stations 4A and 5B in East Bay from January, 1975 through February, 1978.

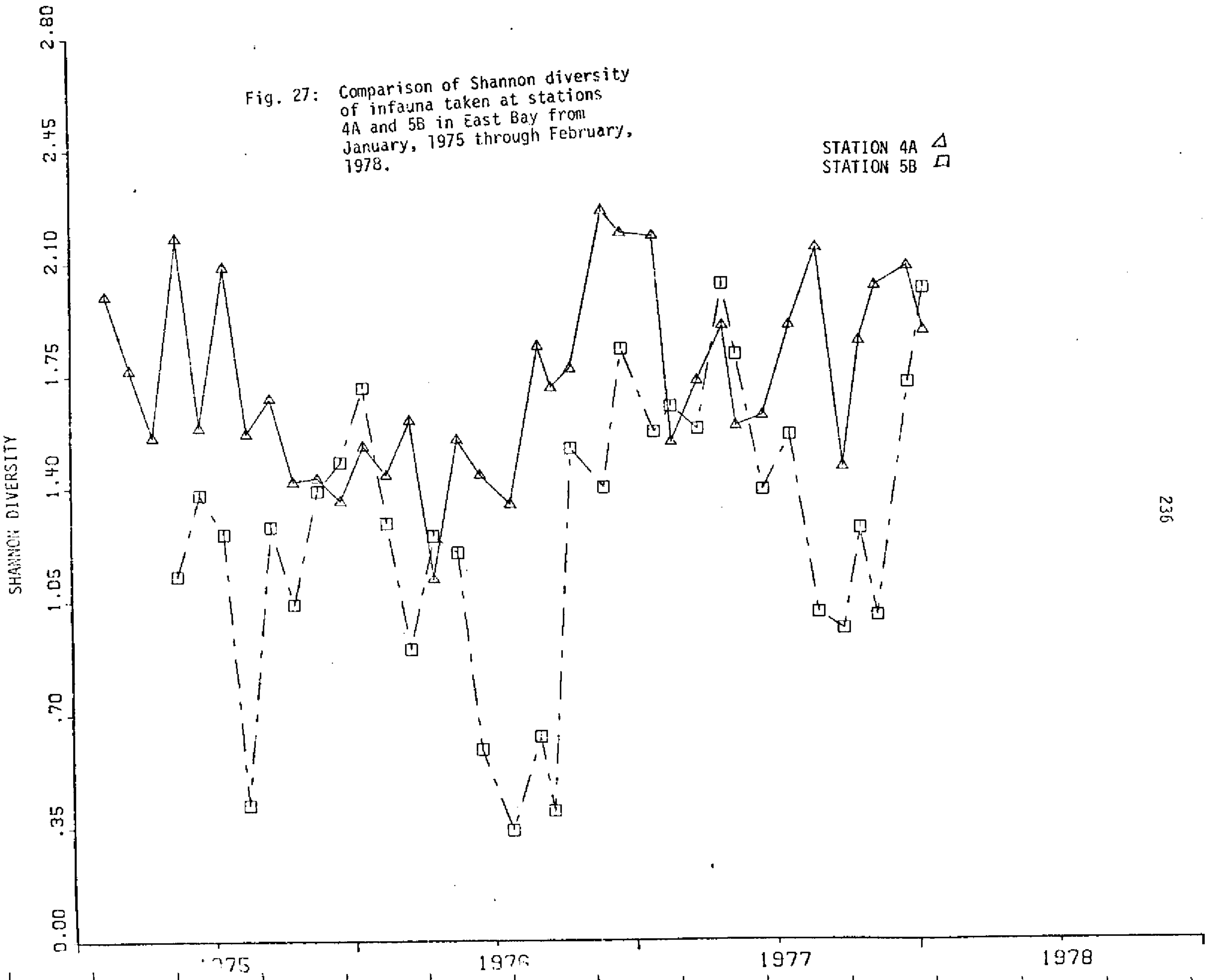


Fig. 27: Comparison of Shannon diversity of infauna taken at stations 4A and 5B in East Bay from January, 1975 through February, 1978.

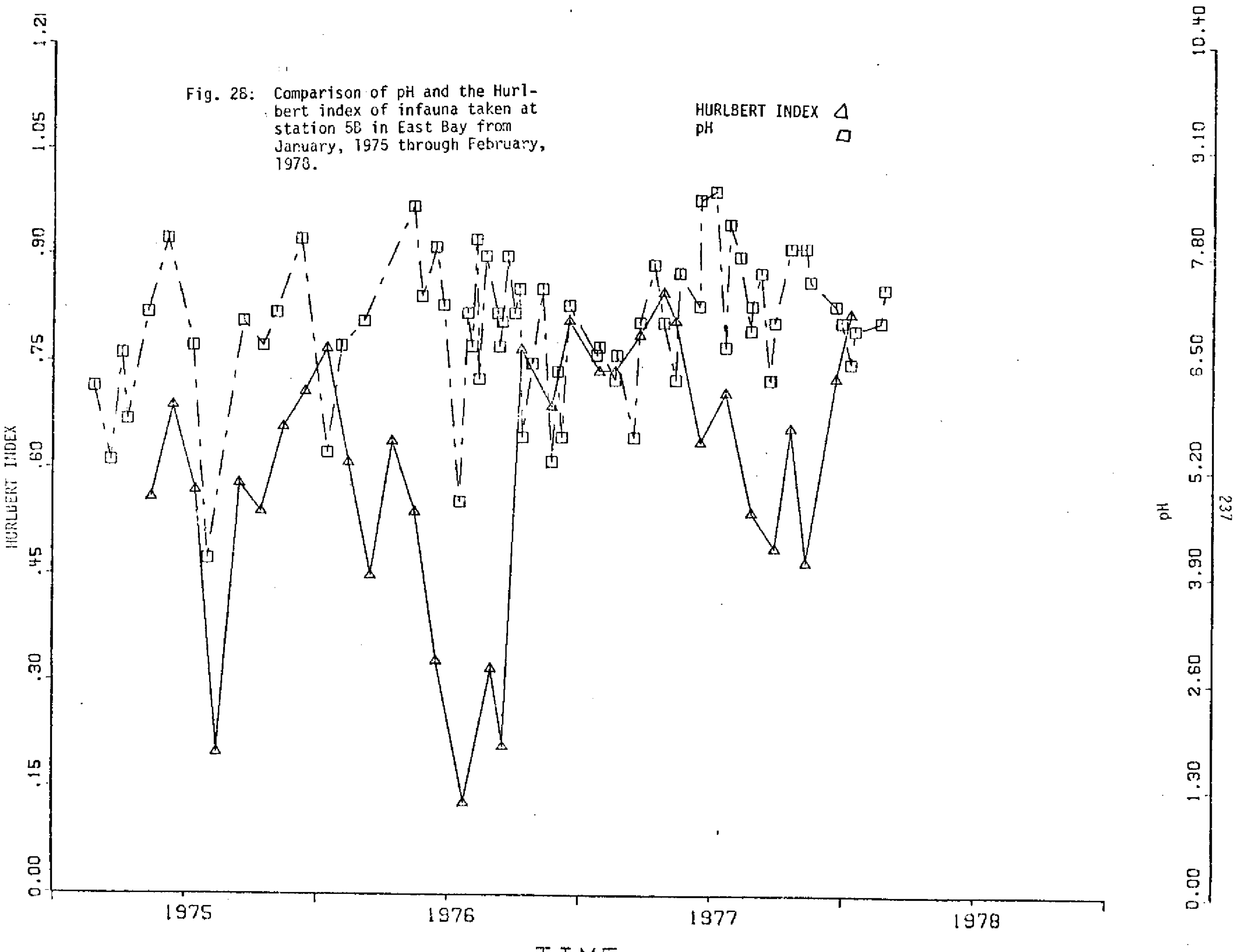


Fig. 28: Comparison of pH and the Hurlbert index of infauna taken at station 5B in East Bay from January, 1975 through February, 1978.

VI. Leaf-litter Assemblages

Introduction

From January, 1974 to December, 1974, a study was carried out at three stations (1X, 3, 5A) in the Apalachicola estuary to determine the species composition and general structure of the community of organisms associated with leaf litter which was commonly found in the bay system. In addition to the usual physico-chemical measurements, 4 baskets containing pre-weighed leaves and 2 controls (no leaves) were dropped in the bay and recovered later (at weekly intervals for 4-6 weeks) to determine the composition of the litter assemblages. Three separate experiments were conducted at intervals through 1974 and the results were analyzed as weekly accumulations and total numbers per sampling period.

Physico-chemical functions

A one-year profile of various physical conditions in the three primary study sites is presented in Fig. 1. Water temperature varied little from one station to the next; peaks occurred during late summer months with periods of transition in the spring and fall. Biological sampling took place during periods of increasing, peak, and decreasing water temperature levels. River flow peaked during late winter and spring months of 1974. Elevated turbidity paralleled river flow at stations 3 and 5A. Farther out in the bay, station 1X was characterized by constant low turbidity levels, being less influenced (directly) by river flow. Local rainfall, out of phase with river flow, peaked during late summer and early fall. Such rainfall was correlated with increased levels of color in water at inshore stations such as 5A and, to a lesser degree, station 3. Color appeared to be more variable than turbidity and often had a direct (inverse) relationship with salinity. Color was uniformly low at station 1X; this was in direct contrast to the situation at station 5A, where the temporal patterns of color followed the local rainfall distribution. Salinity tended to follow these lines. Station 1X was clearly influenced by river flow and rainfall to a lesser degree than were stations 3 and 5A. Station 5A appeared to be primarily affected by local rainfall, having a low mean salinity over the study period with considerable seasonal variation. During winter and spring months, salinity was generally very low in East Bay. Increases in salinity occurred during summer and fall periods in this area with significant variation due to local rainfall and runoff conditions. Station 3, with somewhat higher mean salinities, reflected this pattern to a certain degree, although variation was somewhat less extreme and there were generally higher salinity levels during the winter-spring period than in upland portions of the estuary. During the summer, salinities were comparable at stations 3 and 5A. The comparatively small influence of contiguous land areas on station 3 was also reflected in these data. Thus, the physical parameters at the three primary collection stations were primarily based on their location, temporal variations of river flow, meteorological phenomena (local rainfall), and land runoff conditions. These data are consistent with long-term studies of the Apalachicola Bay system as a whole.

Biological associations

The results of the leaf basket experiments are shown in Table 1 and Fig. 2. In every instance, there was a significant difference (in terms of numbers of species and individuals) between empty baskets and those containing leaf litter. The presence of organisms in empty sampling devices indicates that the enclosures may have a certain shelter function. There was usually a consistent within-station similarity in species composition. Although there was station-to-station variation in species associations, increased salinity was associated with increased between-station similarity. Such changes were often characterized by increased dominance of species such as Gammarus mucronatus, Melita sp., Ericthonius brasiliensis, and Gitanopsis sp. Since such associations were most prevalent during the fall it is probable that factors other than salinity are involved in the determination of species composition of the leaf litter associations. Another important seasonal function in this case would be water temperature, although changes in functions such as stormwater runoff (local rainfall) should also be taken into consideration.

In terms of numbers of individuals (N), the lowest totals occurred at station 5A during the late summer period. This coincided with heavy rainfall in the East Bay area (reflected in the water color data). Such a pattern reflects stress at this time although recovery was evidently rapid since the fall samples at 5A were characterized by relatively high numbers of individuals. During the summer sampling at station 5A there was a decrease in numbers of individuals, numbers of species, Margalef richness, and Shannon diversity with time. Overall, numbers of species (S), Margalef richness (Ma), and Shannon-Weaver diversity (H') increased at all three stations with time. Such indices usually peaked during the fall. Associated with this, there was a general decrease in relative dominance with time. Overall, the patterns of changes tended to reflect stressful conditions in upper East Bay during the late summer of 1974.

Apparently, salinity had an important effect on the distribution of litter-associated organisms in the bay. This is shown in Fig. 3. High positive correlations were found between S and two parameters (Margalef richness and log N). There were also significant correlations of salinity with S, log N, and species richness. It appears that salinity is a primary determinant of leaf litter assemblages. Regression analysis confirms some of these results (F = 30.4; R^2 = 0.26 for salinity and $\log_e$ N). The numbers of species and individuals taken at a given time varied directly with salinity. General salinity increases in the fall coincided with increased similarity coefficients so that even qualitative changes in leaf litter fauna were not unrelated to salinity. The data would thus indicate that salinity is an important parameter concerning the leaf litter assemblages in the Apalachicola Bay system. However, compared to the other 2 stations, the numbers of individuals and species taken at 5A were unevenly distributed. There were consistently lower N and S values at the same salinities indicating the possibility of inhibitory factors other than salinity. These data thus tend to indicate stress in portions of East Bay which could be associated with rainfall, upland runoff, and decreases in water quality in eastern portions of East Bay

Table 5. Total numbers of leaf-litter associated organisms taken at three stations in Apalachicola Bay during the spring, summer, and fall of 1975.

SPRING (April-May) | SUMMER (August-September) | FALL (October-November)

Totals: 4 baskets with 2 controls (no leaves) - samples weekly, 4-6 weeks

STATION 5A

SPRING (April-May)

	with leaves	without leaves
Invertebrates		
Neritina reclivata	802	152
Palaemonetes pugio	253	73
Corophium louisianum	242	5
Gammarus sp. (2)	214	10
Gammarus mucronatus	130	
Grandidierella bonnieroides	107	
Rhithropanopeus harrisii	99	8
Callinectes sapidus	54	11
Melita sp.	37	
Cerapus sp.	15	3
Munna reynoldsi	7	
Cassidinidea ovalis	3	
Leptochelia rapax	3	
13 species		
Fishes		
Bairdiella chrysoura	21	6
Brevoortia patronus	1	
Syngnathus scovelli		1
3 species		
	2,081	269

SUMMER (August-September)

	with leaves	without leaves
Invertebrates		
Corophium louisianum	601	45
Taphromysis louisianae	339	2
Grandidierella bonnieroides	246	2
Neritina reclivata	166	30
Melita sp.	71	
Gammarus mucronatus	29	1
Callinectes sapidus	22	2
Munna reynoldsi	20	1
Penaeus setiferus	18	3
Gammarus sp. (2)	16	
Palaemonetes pugio	15	1
Rhithropanopeus harrisii	7	
Cerapus sp.	7	1
Gitanopsis sp.	7	
Edotea montosa	5	
Cassidinidea ovalis	4	
Sphaeroma terebrans	1	
17 species		
Fishes		
Gobiosoma bosci	42	9
Bairdiella chrysoura	17	2
Syngnathus floridae	2	
3 species		
	1,655	101

FALL (October-November)

	with leaves	without leaves
Invertebrates		
Erichthonius brasiliensis	2,257	235
Gammarus mucronatus	1,373	67
Munna reynoldsi	1,261	46
Gitanopsis sp.	1,128	59
Polydora websteri	948	263
Melita sp.	919	25
Corophium louisianum	303	11
Cassidinidea ovalis	100	6
Neanthes succinea	78	5
Callinectes sapidus	39	4
Neritina reclivata	39	8
Palaemonetes vulgaris	24	5
Grandidierella bonnieroides	14	
Edotea montosa	13	
Cerapus sp.	12	
Rhithropanopeus harrisii	11	1
Melita fresnelii	5	
Cymadusa filosa	2	
Ampelisca abdita	2	1
Nemerteans	2	
Sphaeroma quadridentatum	1	
Penaeus duorarum	1	
Parametopella cypris	1	
Paranaitis speciosa	1	
Mysidopsis bigelowi	1	3
25 species		
Fishes		
Gobiosoma bosci	12	5
1 species		
	8,678	749

STATION 3

SPRING (April-May)

	with leaves	without leaves
Invertebrates		
Melita sp.	2,068	82
Munna reynoldsi	1,400	126
Gammarus sp. (2)	783	89
Cerapus sp.	392	113
Grandidierella bonnieroides	335	81
Palaemonetes pugio	306	91
Rhithropanopeus harrisii	295	14
Corophium louisianum	292	90
Cassidinidea ovalis	99	11
Xanthid juveniles	70	
Neritina reclivata	56	6
Palaemonetid juveniles	33	6
Gammarus mucronatus	23	
Edotea montosa	10	1
Callinectes sapidus	10	
Sphaeroma terebrans	2	
16 species		
Fishes		
Gobiosoma bosci	45	10
Ictalurus catus	4	
Lagodon rhomboides	1	
Bathygobius soporator	1	
4 species		
	6,220	725

SUMMER (August-September)

	with leaves	without leaves
Invertebrates		
Munna reynoldsi	3,137	23
Grandidierella bonnieroides	1,725	26
Melita sp.	1,471	15
Gitanopsis sp.	1,460	4
Corophium louisianum	632	10
Cassidinidea ovalis	389	7
Gammarus mucronatus	290	18
Palaemonetes pugio	130	
Palaemonetes vulgaris	90	49
Gammarus sp. (2)	84	2
Callinectes sapidus	40	6
Edotea montosa	24	
Neritina reclivata	18	3
Penaeus setiferus	8	
Cerapus sp.	7	
Rhithropanopeus harrisii	4	
Sphaeroma terebrans	2	
Xanthid juveniles	2	6
Penaeus aztecus	1	
Argulus sp.	1	
20 species		
Fishes		
Bairdiella chrysoura	55	
Gobiosoma bosci	29	4
Gobiosoma robustum	4	
Orthopristis chrysoptera	1	
4 species		
	9,595	173

FALL (October-November)

	with leaves	without leaves
Invertebrates		
Gammarus mucronatus	1,897	160
Gitanopsis sp.	1,346	51
Corophium louisianum	812	11
Erichthonius brasiliensis	529	23
Munna reynoldsi	375	18
Cassidinidea ovalis	253	11
Melita sp.	173	2
Bittium varium	112	5
Neanthes succinea	34	2
Palaemonetes vulgaris	23	1
Callinectes sapidus	16	2
Cerapus sp.	18	
Neritina reclivata	14	
Leptochelia rapax	6	
Polydora websteri	2	
Edotea montosa	2	
Rhithropanopeus harrisii	1	
Amphicteis gunneri	1	
Nemerteans	1	
Palaemonetes pugio		1
20 species		
Fishes		
Gobiosoma bosci	3	
Lutjanus griseus	1	
2 species		
	5,621	287

STATION 1X

SPRING (April-May)

	with leaves	without leaves
Invertebrates		
Gammarus mucronatus	9,902	92
Melita sp.	2,504	21
Erichthonius brasiliensis	1,951	79
Corophium louisianum	605	62
Cassidinidea ovalis	308	5
Xanthid juveniles	224	3
Leptochelia rapax	143	105
Palaemonetes vulgaris	125	
Gitanopsis sp.	104	2
Nemerteans	41	
Edotea montosa	38	4
Erichsonella filiformis	37	
Ampelisca abdita	37	5
Neanthes succinea	35	4
Cymadusa sp.	15	3
Neopanope texana	10	
Sphaeroma quadridentatum	8	
Callinectes sapidus	5	
Haploscoloplos fragilis	3	2
Penaeus duorarum	1	
Alpheus heterochaelis	1	
Rhithropanopeus harrisii	1	
Cerapus sp.	1	
Taphromysis bowmani		1
24 species		
Fishes		
[illegible]	1	
Lagodon rhomboides	1	
Strongylura marina	1	1
3 species		
	16,[illegible]	[illegible]

SUMMER (August-September)

	with leaves	without leaves
Invertebrates		
Melita sp.	6,017	27
Erichthonius brasiliensis	2,560	149
Gitanopsis sp.	424	1
Cassidinidea ovalis	314	7
Cymadusa sp.	219	17
Nemerteans	101	12
Xanthid juveniles	93	
Gammarus mucronatus	92	19
Palaemonetes vulgaris	82	7
Edotea montosa	22	3
Callinectes sapidus	17	4
Palaemonetid juveniles	16	1
Leptochelia rapax	14	4
Taphromysis bowmani	13	3
Neopanope texana	12	1
Corophium louisianum	5	1
Erichsonella filiformis	3	
Penaeus duorarum	3	
Palaemonetes pugio	2	1
Clibanarius vittatus	1	2
Neritina reclivata	1	1
Nudibranch sp.	1	
22 species		
Fishes		
Gobiosoma bosci	7	
Fundulus grandis	3	
Opsanus beta	2	
Lutjanus griseus		1
4 species		
	10,033	[illegible]

FALL (October-November)

	with leaves	without leaves
Invertebrates		
Melita sp.	1,662	32
Gammarus mucronatus	904	15
Cymadusa sp.	605	40
Erichthonius brasiliensis	417	36
Cassidinidea ovalis	255	14
Neanthes succinea	158	25
Gitanopsis sp.	152	
Podarke sp.	151	
Palaemonetes vulgaris	114	15
Nemerteans	59	
Crepidula plana	43	
Leptochelia rapax	38	9
Bittium varium	27	5
Corophium louisianum	19	1
Haploscoloplos fragilis	18	2
Erichsonella filiformis	13	
Fabricia sp.	8	
Anachis avara	7	2
Neopanope texana	7	
Edotea montosa	6	
Neritina reclivata	5	
Polydora websteri	5	7
Taphromysis bowmani	5	5
Clymenella sp.	4	
Paranaitis speciosa	2	
Penaeus duorarum	2	
Callinectes sapidus	2	1
[illegible]	1	
Penaeus aztecus	1	
Streblospio benedicti		1
30 species		
	4,7[illegible]	[illegible]

Figure 1: Apalachicola River flow, local rainfall, and physical parameters (temperature, salinity, color, turbidity) at primary sampling stations from January to December (1974).

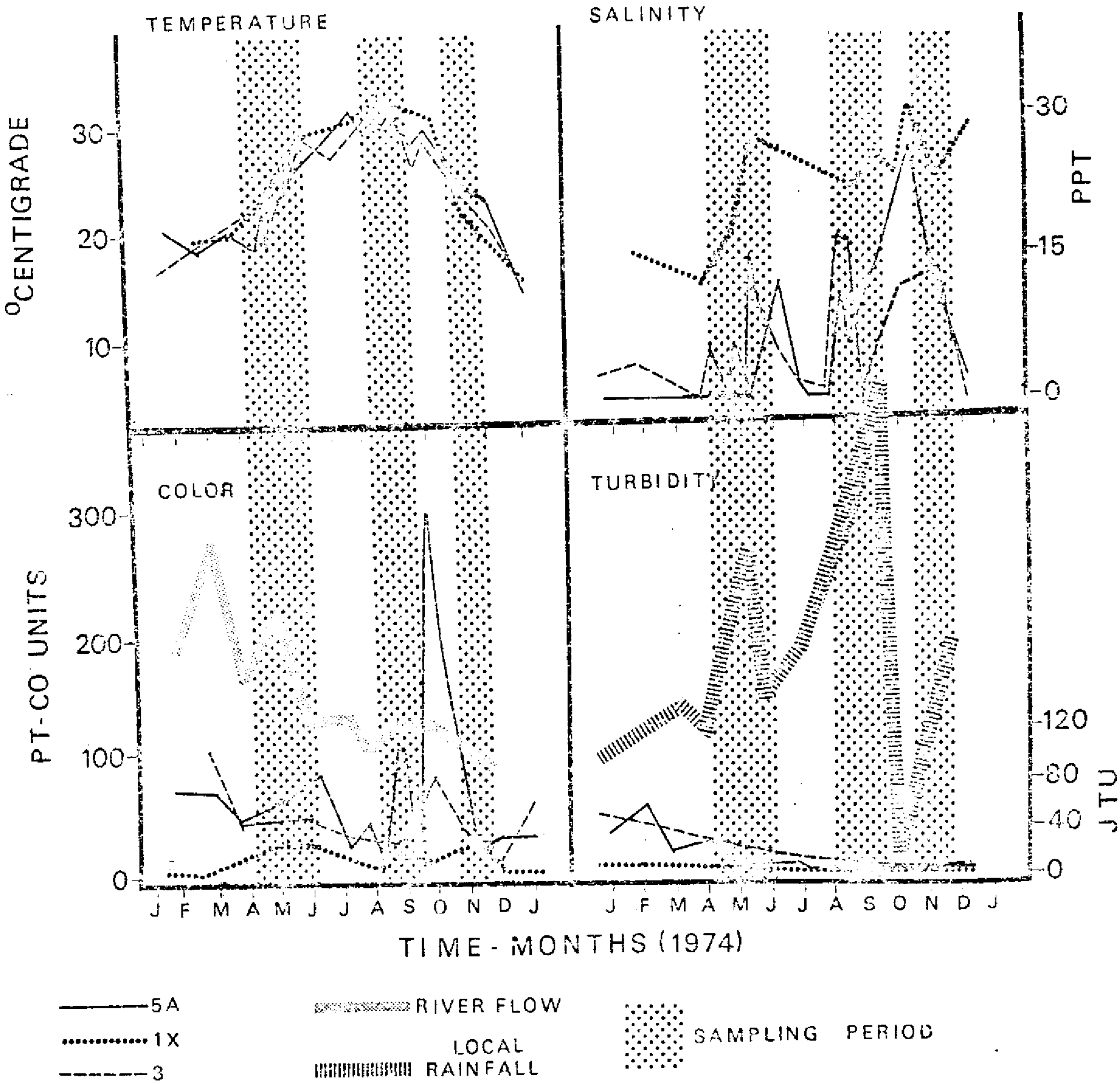

TEMPERATURE
SALINITY
COLOR
TURBIDITY
°CENTIGRADE
30
20
10
PPT
30
15
0
PT-CO UNITS
300
200
100
0
JTU
120
80
40
0
J F M A M J J A S O N D J
J F M A M J J A S O N D J
TIME - MONTHS (1974)
5A
1X
3
RIVER FLOW
LOCAL RAINFALL
SAMPLING PERIOD

Figure 2: Total number of individuals (N), species (S), relative dominance (D) Shannon-Weaver diversity (H') and Margalef richness of leaf litter invertebrates at 3 sampling sites in Apalachicola Bay during 1974.

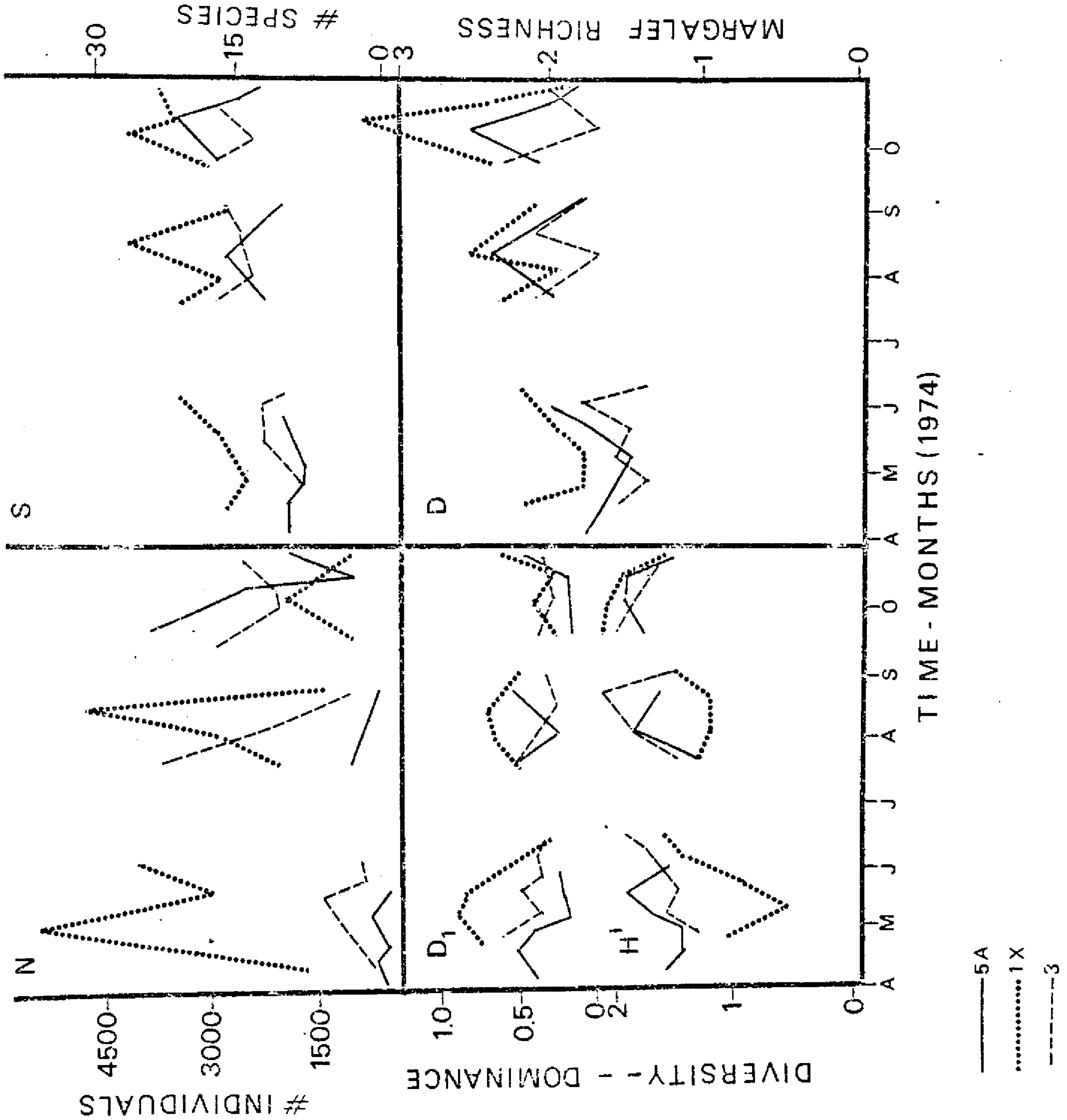
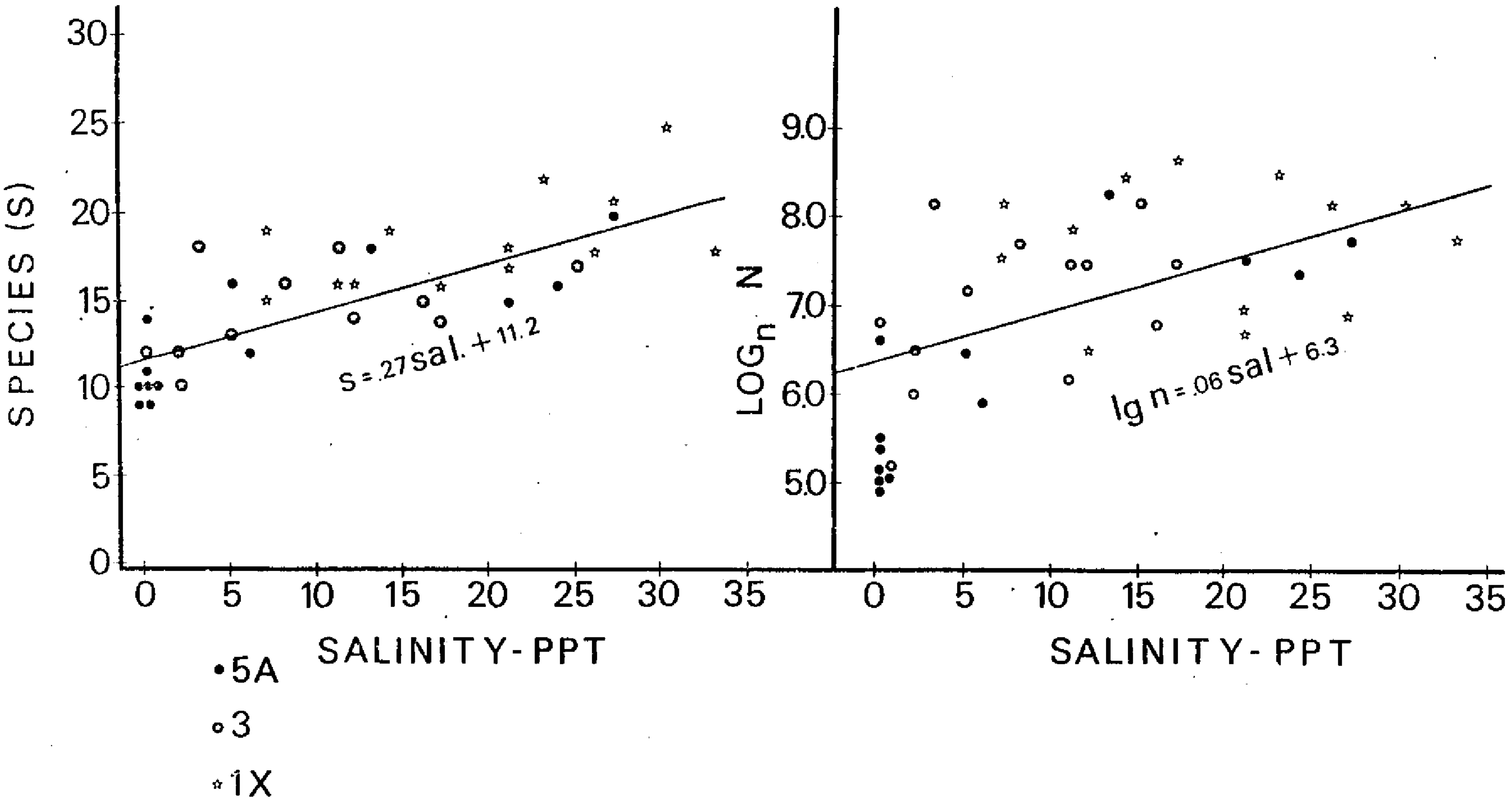

N
S
D₁
H'
D
INDIVIDUALS
4500
3000
1500
SPECIES
30
15
0
DIVERSITY - - DOMINANCE
1.0
0.5
0
2
1
0
MARGALEF RICHNESS
3
2
1
0
A
M
J
J
A
S
O
TIME - MONTHS (1974)
5A
1X
3

Figure 3: Regressions of numbers of species (S) and number of individuals ($\log_e N$) of leaf litter invertebrates with salinity for data taken at 3 stations in Apalachicola Bay.


SPECIES (S)
30
25
20
15
10
5
0
0
5
10
15
20
25
30
35
SALINITY-PPT
S = .27 sal. + 11.2
•5A
○3
☆1X
LOG$_n$ N
9.0
8.0
7.0
6.0
5.0
0
5
10
15
20
25
30
35
SALINITY-PPT
lg n = .06 sal + 6.3

VII. Upper East Bay Grassbeds (*Vallisneria americana*)

Introduction

Details concerning this portion of the study can be found in a Masters thesis by Bruce Howard Purcell (1977). Other comprehensive studies have been made on epibenthic assemblages inhabiting seagrasses in the shallow coastal and estuarine areas of the Gulf of Mexico (Reid, 1954; Simmons, 1957; Darnell, 1958, 1961; Tabb and Manning, 1961; Hoese and Jones, 1963; Dragovitch and Kelley, 1964; Parker, 1969; Eidemiller, 1972; Weaver and Holloway, 1974; Heck, 1976; Hooks et al., 1976; Livingston et al., 1976). Studies of communities associated with mixed assemblages of limnic benthic macrophytes are also available (Nevin and Tounes, 1935; Krecker, 1939; Harrod, 1964; Soszka, 1975), including Wisconsin macrophyte beds containing *Vallisneria americana* (Muttkowski, 1918; Andrews and Hasler, 1943; Andrews, 1946). The only studies involving *V. americana* beds in estuarine areas were conducted in the upper Caloosahatchee Estuary, Florida (Phillips and Springer, 1960; Gunter and Hall, 1965). Such studies concentrated on species composition, size distribution, and biotic reaction to temperature and salinity variations.

Vallisneria located in eastern portions of East Bay received highly colored, acidic runoff during periods of heavy rainfall (Livingston, 1977; Livingston et al., 1977b). This study of the epibenthic fauna associated with *V. americana* beds in East Bay has been undertaken to determine the basic productivity and ecological relationships of such areas and to provide a baseline study of these beds with regard to functions related to episodic physico-chemical changes associated with upland runoff.

Physico-chemical and biological data were collected monthly from November, 1975, to October, 1976, from two stations located in *Vallisneria americana* beds of upper East Bay, Franklin County, Florida. Sampling stations were located in the mouth of West Bayou and in the central, southern portion of Round Bay. The West Bayou station was located in an area subjected to periodic runoff from upland clearcutting. The Round Bay station served as a control because, until August, 1976, it did not receive any such runoff.

Results

Benthic macrophytes. The benthic macrophyte community at West Bayou consisted of the fresh-water monocots *V. americana* and *Potamogeton pusillus* and the brackish and marine monocot *Ruppia maritima*. In addition to these species, the green alga *Cladophora* sp. and one specimen of the marine macrophyte *Halophila engelmanni* were collected at Round Bay. The plant biomass in both areas was dominated by *V. americana*, which comprised over 99.7% of the total yearly dry-weight figures.

A perennial with annual regrowth, *V. americana* entered a dormant stage in the winter (December through February), which consisted of rootstocks to which were attached short, decayed stubs that had once been the bases of leaves. These rootstocks were slightly broader and less numerous at Round Bay than at West Bayou.

Once growth began, it was rapid. Spring growth began for most of the *Vallisneria* plants in Round Bay prior to March 20 and in West Bayou prior to April 20. By April most of the leaves and female flowers (especially at Round Bay) were easily reaching the surface from a depth of one meter. From their maximum of hundreds per 100 m^2 in April, the female flowers gradually decreased in number until their disappearance in October. By May the Vallisneria had turned darker green and had reached its maximum length of 1.2 meters, which it maintained through July. The water surface was covered with numerous, free-floating male flowers in May and June, which had been released from capsules attached to the base of the plants. These flowers decreased considerably in July and disappeared in August. The growth and to some extent biomass of the V. americana (Fig. 1) coincided with water temperature but not with any other physico-chemical parameters. Temperature appeared to be limiting to this species in the vicinity of 20°C. Below this level the plants were dormant; above it they grew.

Rapid spring growth and productivity was followed by gradual decay through the summer and rapid regrowth in the fall. Browning at the end of the leaves was noticeable in both areas as early as May. By August the leaf ends had degenerated and only 70-85 cm of the leaves remained. Of this remainder, holes and brown areas were extensive in the center of the blades, and the terminal 10-15 cm were frayed and completely brown. This condition had worsened by September and October until the plants were only about 40 cm long, largely brown, full of holes, and decayed at the ends. Although there had been a small amount of new plant growth evident even since the initial spring bloom had ended, there was a surprisingly large surge of new growth in September and October coincident with the older leaves dying away. In September about half of the plants had initiated growth from old rootstocks. In October at Round Bay, almost all of the rootstocks had begun new growth, and an equal number of entirely new shoots had arisen from rhizomes. In October at West Bayou, new growth occurred on about half of the old rootstocks and from an equal number of rhizomes. The maximum cumulative net production for the year was estimated as 320 g/m^2 for Round Bay and 350 g/m^2 for West Bayou.

In July and to a lesser degree in June, Cladophora sp. floated on the surface of Round Bay in large mats over the benthic macrophytes. It submerged in August along with the ends of the Vallisneria to which it was loosely attached. It was not observed in West Bayou or at any other time of the year.

Epibenthic fishes and invertebrates. Table 1 gives a complete list of the species collected by the dredge and surface nets, and Table 2 lists the species collected by seining. Ninety-one species of invertebrates and forty species of fishes were collected. The larvae and juveniles of some groups, such as the insects, penaeids, and centrarchids, posed major obstacles to identification. For example, the penaeid shrimp were not identifiable if they were ungrooved and 12-18 mm long or grooved and 12-25 mm long (Williams, 1959, 1965; Farfante, 1970). The size and degree of identification of the centrarchids followed a temporal succession. Centrarchid spp. ($<$8 mm) first occurred and peaked in April. In the following month, there was a peak in Lepomis sp. (9-15 mm) and, in June and July, there was a small peak in L. microlophus ($>$15 mm) and a single L. punctatus ($>$15 mm). A second peak in centrarchid spp. in July and August, however, was followed by a peak in Enneacanthus gloriosus ($>$8 mm) in August and September. Thus a few species may be listed under more than one non-specific name, and a non-specific name may encompass more than one species. The fauna collected by seining represent some of the larger individuals that occurred in the *V. americana* beds. The fish species *Cynoscion nebulosus*, *Lepisosteus osseus*, and *Lutjanus griseus* were collected only by seining. Most species seined were found at both stations; nevertheless, four fish species were seined only in Round Bay and three fish species and two penaeid species were seined only in West Bayou.

Monthly variation in the numbers of fishes and invertebrates, the fish and invertebrate biomass, the number of fish and invertebrate species, and the species diversity indices did not coincide with the monthly changes in the various physico-chemical parameters. Excluding the species diversity indices, they also did not coincide with one another. These common dissimilarities had two exceptions. First, the abundance and biomass of the invertebrates and Round Bay fishes dropped in January and February corresponding to the low temperature and lack of benthic macrophytes. Second, for the invertebrate biomass and the Hurlbert index of the whole community, the monthly fluctuation of the day values corresponded fairly well with that of the night values. All these parameters, however, were usually higher at night and at the bottom. With few exceptions the species present in the daytime at the surface were also at the bottom. At night, however, numerous fish and invertebrate species were only found either at the surface or at the bottom.

Twenty-five hundred fish and 342,894 invertebrates were collected. The total invertebrate biomass was much greater than the fish biomass. Unlike the number of species, the abundance and biomass were more nearly alike between the bottom and the surface during the time of the year (April-July) when the Vallisneria extended to the water's surface. Although the abundance was higher at Round Bay than at West Bayou, the opposite was true for the biomass. The monthly biomass of the invertebrates closely followed the biomass of *Neritina reclivata* because of the species' extremely high biomass dominance. Since the biomass of this snail, however, was based on its average size, which changed throughout the year, the biomass of the community was overestimated for certain times of the year and underestimated for other times. The invertebrate biomass corresponded fairly well with the growth of *V. americana*, but the fish biomass did not.

The Shannon index, Hurlbert's Δ_1, and the Simpson index coincided throughout the year so only the monthly values for the Hurlbert index are listed. Owing to the fishes being a numerically small fraction of the whole community, the values of these indices for the invertebrates alone were essentially identical to those for the whole community; consequently, only the whole community and fishes are discussed. Fluctuations in the top dominant (*N. reclivata*) accounted for a number of the values observed in the indices. For example, the peak in the indices during the winter at West Bayou was largely a result of the reduced dominance, which occurred when the abundance of *N. reclivata* dropped considerably. The Shannon index from both areas for the invertebrates ranged from 0.12 to 2.15 (mean 0.84) in the day and 0.66 to 2.16 (mean 1.31) at night. For the fishes it ranged from 0.43 to 1.43 (mean 0.97) in the day and 0.42 to 1.90 (mean 1.43) at night. The Margalef index of species richness did not correspond with the above indices. The mean of each diversity index was similar at the two stations although the fish diversity was somewhat higher at West Bayou and the whole community diversity was usually higher at Round Bay.

The annual dominance was higher during the day than at night and among the whole community (i.e. the invertebrates) than among the fishes. Monthly dominance did not coincide with any of the physico-chemical parameters. The same dominant species were usually present day and night although they shifted rank each month. Single-species dominance (*Neritina reclivata*) was higher at West Bayou than at Round Bay. The dominance of the top four species, however, was higher at Round Bay at night and with combined day-night figures. This shift was due to massive populations of *Odostomia* sp., *Gammarus macromucronatus*, and *Taphromysis bowmani* at Round Bay. The increased abundance of these three populations at Round Bay more than equaled the considerable difference in the total abundance between the two areas.

For the fishes, the dominance was almost always higher at Round Bay, and the dominant species were usually different. The higher abundance and dominance at Round Bay was due largely to the rainwater killifish (*Lucania parva*) which composed 50.7% of the total abundance (compared to only 7.6% at West Bayou). More than twice the number of pipefish (*Syngnathus scovelli*) were collected in Round Bay. Other groups which contributed to this difference in species composition were the centrarchids and *Heterandria formosa*, which were much more prevalent at Round Bay, and *Leiostomus xanthurus*, *Lagodon rhomboides*, *Anchoa mitchelli*, and *Micropogon undulatus*, which were much more abundant at West Bayou.

The species *Neritina reclivata* underwent a population crash in the early winter followed by a rapid recovery two months later. This recovery immediately preceded the initial growth of *Vallisneria*. These snails were not found in abundance on the upper portion of the *V. americana* leaves until May or June when the epiphytes became more abundant. More were collected at night from late spring into fall. The gastropod *Odostomia* sp. was collected more frequently at night than during the day. In the summer it was present throughout the water column as was *N. reclivata*. With rare exceptions *Taphromysis bowmani* and *Mysidopsis bahia*, as with all the mysid species that were collected, were more abundant at night. Rarely was *T. bowmani* collected at the surface and only once was it collected there in the daytime. Regardless of the leaves' height, however, *M. bahia* was often found at night near the surface. The amphipods *Grandidierella bonnieroides* and *Gammarus macromucronatus* were "out" much more at night than during the day and showed no definite preference for the bottom or for the surface. The number of *G. macromucronatus* corresponded with higher salinity. Since the insect larva *Zygoptera* spp. clings to macrophyte leaves, it was rarely found at the surface other than when the leaves extended to the surface. It was collected more often at night. The commercially important blue crab *Callinectes sapidus* and the caridean shrimp *Palaemonetes pugio* were more numerous at night and on the bottom. The shrimp species was abundant throughout the year. The brackish-water cyprinodont *Lucania parva* was usually more prevalent at night; with one exception, it was not abundant at the surface. Collection of the pipefish *Syngnathus scovelli* was not dependent on the time of day but this species was much more prevalent at the bottom than at the surface. Its presence coincided with higher salinity and river flow. The other six dominant fishes were very seasonal with most having occurred abundantly for only a short time during the year. With the exception of *Menidia beryllina*, they were found in greater numbers at the bottom than at the surface. The species *Menidia beryllina*, *Brevoortia patronus* and *Microgobius gulosus* were more abundant at night, but *Lagodon rhomboides*, *Leiostomus xanthurus*, and centrarchid spp. were not more prevalent at any particular time of day.

Among the thirty-seven fish species, there were twenty-four marine, ten fresh-water, and three brackish-water forms. Similarly, of the thirty-three most abundant invertebrate species, there were fourteen marine, fourteen fresh-water, and five brackish-water forms. Eleven of the fresh-water invertebrate species were insect larvae. The other salinity classifications were composed of mixed groups.

Measurements of biomass in the East Bay grassbeds are shown in Table 3 and Fig. 1. Differences in the spatial distribution of such macrophytes were responsible for some month to month variability as a result of the sampling methods used. It was estimated that some *Vallisneria* leaves had died by September and the generally high levels of biomass at this time were considered an artifact of the sampling procedures. Consequently, biomass maxima for estimates of productivity were taken from the June data. Losses due to grazing were considered negligible and there was no observable leaf loss prior to August. There was some loss of female flowering parts prior to the summer maxima, which could have made the productivity estimates somewhat conservative.

This was probably counterbalanced by the presence of unremoved epiphytes, although few calcified epiphytes were observed throughout the period of sampling. The grassbeds at station 4A showed higher biomass than those at station 4B although the seasonal patterns were generally similar with low biomass occurring during winter and early spring months (December-April) and high biomass during the summer and early fall (June-October). Transition periods occurred in November and May with the first new growth noted in March. Leaves reached the surface by April, and leaf death was first sighted in August. Productivity figures (Table 4) were comparable in the two study areas.

The top 10 species in terms of biomass are given in Table 5. As shown, the gastropod <u>Neritina reclivata</u> was a strong dominant in the area of study. Monthly biomass figures for the study areas are shown in Table 6. The figures at stations 4A and 4B are comparable with generally higher figures at depth except during the period from May to July when peak values in surface collections were taken. Peak biomass figures were evident in both areas during spring (March-May) and late fall (November -December) periods. These periods coincided roughly with periods of transition in grassbed areas (i.e., growth and death) and probably reflected changes in habitat associated with shelter-seeking and feeding functions of the individual species.

The physico-chemical differences in the two areas could explain some of the biomass fluctuations. During the months of November and December, 1975, salinity at station 4B approximated 6-11 ppt whereas at station 4A it never exceeded 3.8 ppt. This difference was associated with the biomass increases at 4B. From January to July (1976), there were uniformly low salinity conditions at both stations with moderate increases (0-6 ppt) from August to October at both stations. Low pH, high color, and reduced dissolved oxygen (indicating local runoff) occurred at station 4B (relative to 4A) during March, July, September, and October. These periods coincided with decreases in biomass at station 4B relative to 4A. Analysis of biomass ratios (Table 4) indicates that with the exception of January (when biomass was low at both stations because of extremely low winter temperatures), the biomass levels at 4B were relatively lower during those months of increased runoff and reduced water quality. Such changes were not as clear with respect to species richness and diversity parameters, individual population changes, etc. Thus, preliminary observations indicate stress in grassbed areas associated with the West Bayou drainage patterns when compared to biomass fluctuations where community analysis did not reveal overt variations associated with such runoff.

A comparison of grass-bed and mud-flat conditions in given in Figs. 2-5. In contrast with the findings of previous researchers of the mud-flat areas of East Bay (Laughlin, 1976; Duncan, 1977; Livingston, 1977; Livingston et al., 1977a, McLane, 1977), the period of the darkest, most turbid and acidic water occurred in the spring after little or no rainfall and coincided instead with the highest flow of the Apalachicola River. Some water-quality parameters also coincided, however, with the period of heavy rainfall (August-October). Color increased and pH decreased at West Bayou in September and October; pH dropped at Round Bay in September. The pH of the water over the *V. americana* did not appear to become limiting, and it had no noticeable effect on the epibenthic community. Since photosynthesis tends to increase pH and since pH was higher in the grassbeds than in the surrounding areas, the Vallisneria could have buffered the associated animals from the full effect of the acidic runoff. This may be the reason for the ambiguous biological results.

There were three conditions associated with the study areas which may have contributed to the high dominance in the biological assemblages. First, since comparable diversity values have been observed for the fish and invertebrate communities of adjacent Apalachicola Bay (Livingston, 1976), which is relatively unpolluted (Livingston et al., 1974; Livingston, 1976), it is probable that the diversity of this bay system was affected by the natural, environmental stress typical of estuaries. Second, the low diversity may also have been caused by the low salinity of upper East Bay, which tended to restrict the marine species not having osmoregulatory adaptability. Low salinity bay heads were found to have comparatively meager fauna by Ladd (1951), Zenkevitch (1963), Parker (1969), Storrs et al. (1969), and Harrel et al. (1976). Third, in terms of animals living on the plant surface, Vallisneria has been found to be less productive than other benthic macrophytes.

Although most of the dominant invertebrate species were present throughout the year, nearly all were much more prevalent in some months than in others. High spatial and temporal variability in species abundance and composition has also been a common observance in other macrophyte beds (Andrews and Hasler, 1943; Andrews, 1946; Rosine, 1955; Harrod, 1964; Macan, 1965; Marsh, 1973; Pieczynski, 1973; Soszka, 1975;

Heck, 1976). All of the fishes except resident species (Syngnathus scovelli and Lucania parva) were characterized by sharp peaks in abundance with scarcity or absence during the remaining months. These sharp peaks were usually attributable to juveniles, who used the Vallisneria beds as a nursery ground on a seasonal basis in conjunction with their reproductive cycle. The dominant species peaked at the same time in Round Bay and West Bayou with the exception of Odostomia sp., Palaemonetes pugio, and Gammarus macromucronatus.

Investigators have found that specific dominants were staggered in time while the actual community structure remained relatively stable throughout the annual cycle (Haedrich and Haedrich, 1974; Livingston, 1976; Livingston et al., 1977c). Likewise, the peaks in numerical abundance of the dominant fishes and invertebrates at Round Bay were found to be evenly staggered throughout the year (Tables 4 and 6). At West Bayou, however, most of the dominant fish species (Leiostomus xanthurus, Lagodon rhomboides, Brevoortia patronus, and Gobionellus boleosoma) were prevalent in the Vallisneria only when the water was extremely dark and turbid. This condition may have helped to protect these young fishes from predators.

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TABLE 1: Complete list of species collected by trawling from both stations with their abundance and biomass for the year. Biomass 0.0005 g indicated by " * ".

INVERTEBRATES

Species Name	Abundance	Biomass (g)
Neritina reclivata	179915	5073
Odostomia sp.	59484	5.95
Gammarus macromucronatus	36316	9.81
Taphromysis bowmani	33059	12.56
Mysidopsis bahia	5936	1.37
Zygoptera spp.	5107	21.96
Grandidierella bonnieroides	4910	.540
Palaemonetes pugio	3871	38.73
Dicrontendipes sp.	2686	.215
Callibaetis sp.	2116	.635
Callinectes sapidus	1493	41.77
Mysidopsis almyra	1423	.626
Mysidopsis bigelowi	986	.197
Gammarus mucronatus	566	.062
Corixidae spp.	559	.274
Littoridina sphinctostoma	526	.147
Melita intermedia	474	.128
Chironomidae sp. 2	464	.260
Hydracarina sp. 1	422	.089
Mactra fragilis	383	.375

TABLE 1 -- continued

Species Name	Abundance	Biomass (g)
Gastropod sp. 7	348	2.78
Caenis sp.	242	.024
Nymphula sp.	236	.097
Amphictes gunneri floridus	198	.028
Penaeus sp.	175	2.31
Palaemonetes intermedia	77	.868
Penaeus duorarum	77	1.88
Shrimp sp. 6	75	.023
Insect larvae sp. 31	72	.021
Insect larvae sp. 26	66	.007
Rhithropanopeus harrisi	52	1.26
Palaemonetes vulgaris	50	.668
Insect larvae sp. 24	49	.012
Cassidinidea ovalis	45	.024
Penaeus setiferus	42	.325
Corophium louisianum	35	.052
Oligochaete sp. 4	24	.001
Hydracarina sp. 2	24	.005
Polydora ligni	22	.001
Leech sp. 1	21	.037
Ceratopogonidae spp.	19	.003
Hippolyte zostericola	17	.037
Gammarus New sp.	17	.009

TABLE 1 -- continued

Species Name	Abundance	Biomass (g)
Laeonereis culveri	17	.017
Cerapus sp.	17	.001
Argulus sp.	13	*
Edotea montosa	13	.001
Anisoptera sp. 1	13	.009
Bittium varium	12	.002
Macoma mitchelli	12	.120
Insect larvae sp. 23	10	.008
Barnacle sp.	8	.002
Chironomidae sp. 3	8	.001
Congeria leucophaeta	6	.048
Anisoptera sp. 2	5	.004
Insect larvae sp. 25	5	.001
Turbellarian sp.	5	*
Nematode sp.	4	*
Hydracarina sp. 3	4	.001
Insect larvae sp. 34	4	.040
Brachidontes exustus	4	.025
Cymadusa sp.	3	*
Rangia cuneata	3	2.58
Taphromysis louisianae	3	.001
Nemertean sp.	3	*
Mysidopsis sp. 2	3	.001

TABLE 1 -- continued

Species Name	Abundance	Biomass (g)
Insect larvae sp. 29	3	.003
Insect larvae sp. 30	3	.001
Mitrella lunata	2	*
Neopanope texana	2	.004
Oligochaete sp. 2	2	*
Crepidula sp.	2	*
Leptochelia rapax	2	*
Cyathura polita	2	.002
Uca minax	2	.211
Insect larvae sp. 32	2	.002
Insect larvae sp. 33	2	.001
Gitanopsis sp.	2	*
Insect larvae sp. 27	2	.001
Neopanope sp.	1	.002
Periclimenes longicaudaus	1	.004
Mysidopsis sp. 1	1	*
Insect larvae sp. 19	1	.013
Insect larvae sp. 35	1	*
Insect larvae sp. 36	1	.001
Streblospio benedicti	1	*
Mysidopsis sp. 3	1	*
Oligochaete sp. 3	1	*
Orchestia uhleri	1	*

TABLE 1 -- continued

Species Name	Abundance	Biomass (g)
Hermit crab	1	*
Pleustidae spp.	1	*
FISHES		
Lucania parva	945	13.81
Syngnathus scovelli	526	22.56
Menidia beryllina	146	23.04
Microgobius gulosus	109	.85
Brevoortia patronus	108	1.84
Centrachid spp.	94	.11
Lagodon rhomboides	93	3.53
Leiostomus xanthurus	86	3.92
Gobionellus boleosoma	70	.63
Gobiosoma bosci	59	2.00
Anchoa mitchelli	48	.77
Micropogon undulatus	42	.37
Lepomis sp.	30	.36
Heterandria formosa	30	.82
Bairdiella chrysura	22	.70
Enneacanthus gloriosus	21	1.46
Eucinostomus sp.	16	.07
Lepomis microlophus	9	1.09
Fish sp.	8	.004

TABLE 1 -- continued

Species Name	Abundance	Biomass (g)
Strongylura marina	6	.17
Micropterus salmoides	6	.85
Mugil cephalus	4	.28
Myrophis punctatus	4	5.47
Trinectes maculatus	3	.35
Lepomis punctatus	2	2.20
Gobiosoma robustum	2	.05
Microgobius thalassinus	1	.001
Gobiesox strumosus	1	.02
Paralichthys lethostigma	1	.02
Anguilla rostrata	1	.05
Elops saurus	1	.09
Notemigonus crysoleucas	1	.04
Archosarus probatocephalus	1	.04
Poecilia latipinna	1	.21
Fundulus chrysotus	1	.30
Syngnathus louisianae	1	.43
Symphurus plagiusa	1	.08

TABLE 2: Complete list of species collected by seining throughout the year at Round Bay and West Bayou.

Species Name	Round Bay	West Bayou
Anchoa mitchelli	X	X
Bairdiella chrysura	X	X
Brevoortia patronus	X	X
Callinectes sapidus	X	X
Cynoscion nebulosus		X
Enneacanthus gloriosus	X	X
Gobiosoma bosci		X
Lagodon rhomboides	X	X
Leiostomus xanthurus	X	X
Lepomis microlophus	X	X
Lepisosteus osseus	X	
Lucania parva	X	X
Lutjanus griseus		X
Menidia beryllina	X	X
Microgobius gulosus	X	X
Micropterus salmoides	X	X
Myrophis punctatus	X	
Notemigonus crysoleucas	X	
Palaemonetes pugio	X	X
Paralichthys lethostigma	X	X
Penaeus setiferus		X
Penaeus duorarum		X

TABLE 2 -- continued

Species Name	Round Bay	West Bayou
Penaeus sp.		X
Rangia cuneata	X	X
Rhithropanopeus harrisii	X	X
Syngnathus scovelli	X	X
Trinectes maculatus	X	

Table 3: Biomass (g/m^2) of macrophytes taken in East Bay. Values include root stocks and uncalcified epiphytes.

DATE	4A	4B
11/02/75	455.7	334.4
12/14/75	200.6	213.0
1/17/76	287.2	167.8
2/18/76	206.8	263.2
3/20/76	269.2	196.6
4/20/76	220.8	138.6
5/20/76	316.9	268.1
6/18/76	563.0	354.9
7/17/76	358.4	568.4
8/14/76	538.8	365.6
9/11/76	585.1	489.8
10/10/76	486.9	438.7
Total	4,489.4	3,799.1
Mean/month	374.1	316.6

Table 4: Estimated productivity of *Vallisneria* beds in East Bay. Data were taken from November, 1975 through October, 1976.

	Round Bay (4a)	West Bayou (4b)
Max. summer biomass:	563 g/m^2 (June) 95% Confidence interval: ± 122	568 g/m^2 (July) ± 121
Mean winter biomass:	241 g/m^2 (Dec. - Mar.) 95% Confidence interval: ± 122	215 g/m^2 (Dec. - Feb.) ± 54 new growth occurred in Mar @ 4B
Change in biomass or Max. cumulative net production:	322 g/m^2	353 g/m^2
Productivity:	322 $g/m^2/yr.$	353 $g/m^2/yr.$

Table 5: Top dominants (fishes and invertebrates) at Stations 4a and 4b (East Bay) in terms of biomass (dry weight) taken over the 12 month sampling period (November, 1975 - October, 1976)

Species	Percentage of total
Neritina reclivata	95.67
Callinectes sapidus	0.79
Palaemonetes pugio	0.73
Menidia beryllina	0.43
Syngnathus scovelli	0.43
Zygoptera sp.	0.41
Lucania parva	0.26
Taphromysis bowmanni	0.24
Gammarus macromucronatus	0.18
Odostromia sp.	0.11

Table 6: Biomass (g, dry weight) of fishes and invertebrates taken in Vallisneria beds in East Bay at night from November, 1975 to October, 1976:

A. Total Biomass/15 m^3

	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.
4A(Bottom)	197.99	103.79	4.75	5.88	171.11	109.96	143.63	100.58	67.27	84.28	102.95	112.23
4A(Surface)	----	0.010	0.43	0.71	1.23	12.18	120.97	96.07	46.46	1.98	4.40	1.79
4B(Bottom)	373.56	133.79	3.57	7.14	141.46	201.51	195.16	83.97	35.68	124.43	95.40	47.43
4B(Surface)	----	.003	0.14	1.23	0.89	7.26	25.61	75.28	44.05	30.09	5.37	0.60

B. Biomass/m^2

	Nov.	Dec.	Jan.	Feb.	Mar.	Apr.	May	June	July	Aug.	Sept.	Oct.
4A(Bottom)	2.20	1.15	.0528	.0653	1.90	1.22	1.60	1.22	.747	.936	1.14	1.25
4B(Bottom)	4.15	1.49	.0397	.0793	1.57	2.44	2.17	0.933	0.396	1.38	1.06	0.527
Ratio $\left(\frac{4A}{4B}\right)$	0.53	0.77	1.33	0.82	1.2.	0.50	0.74	0.65	1.89	0.68	1.08	2.37

Fig. 1: Monthly (1975-1976) biomass/m^2 of <u>Vallisneria americana</u> at Round Bay and West Bayou taken from the mean of eight random 1/16 m^2 quadrats.

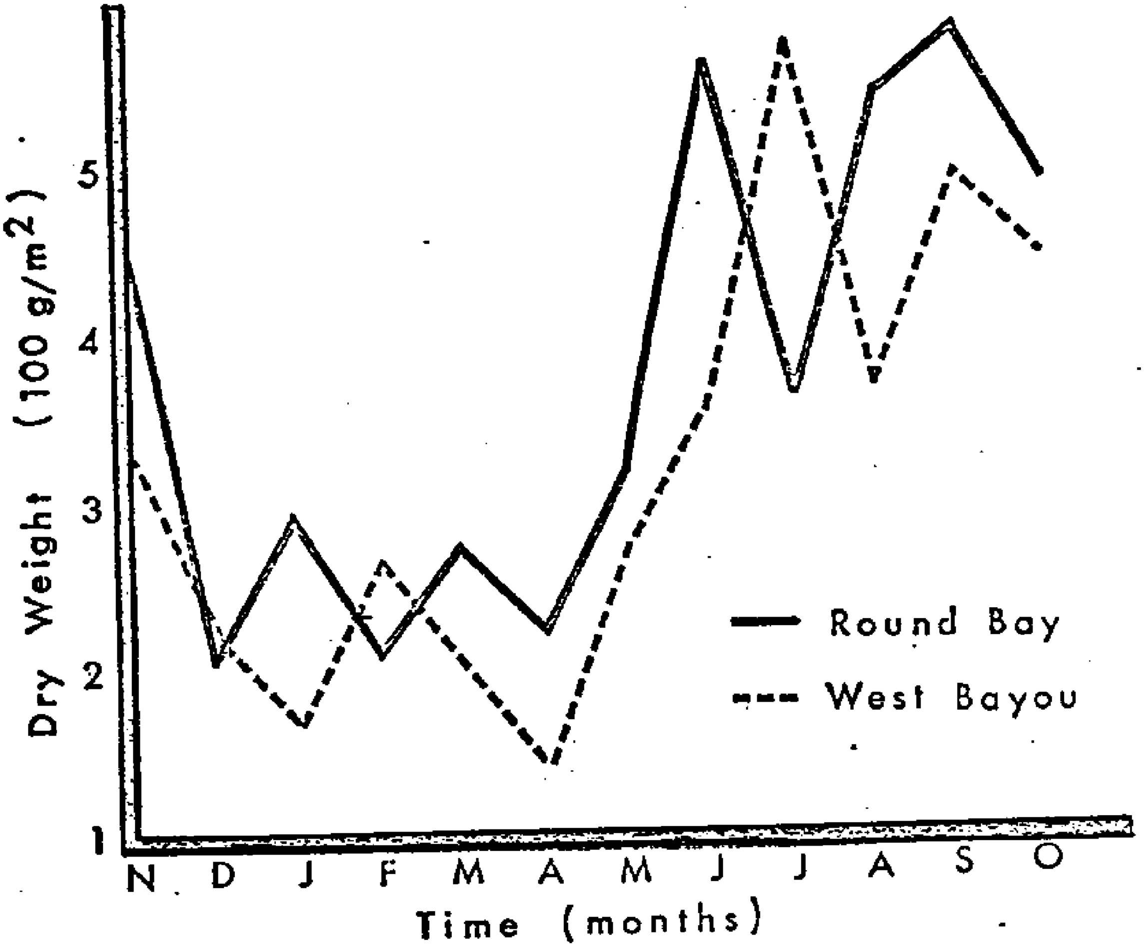

Dry Weight (100 g/m^2)
1
2
3
4
5
N
D
J
F
M
A
M
J
J
A
S
O
Time (months)
Round Bay
West Bayou

Fig. 2: Comparison between the monthly (1975-1976) dissolved oxygen in the Vallisneria bed and in the nearby mud flat in Round Bay.

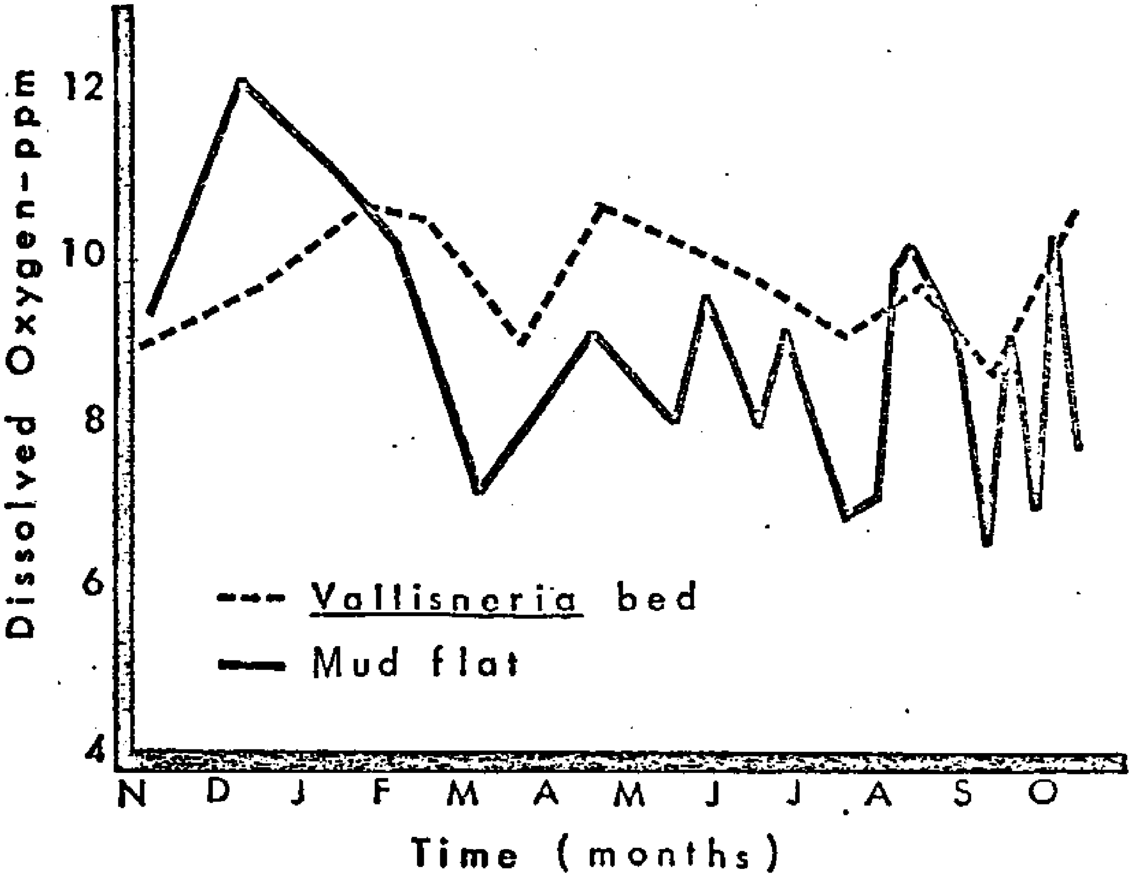

Dissolved Oxygen-ppm
12
10
8
6
4
N D J F M A M J J A S O
Time (months)
Vallisneria bed
Mud flat

Fig. 3: Comparison between the monthly (1975-1976) dissolved oxygen in the *Vallisneria* bed and in the nearby mud flat of West Bayou.

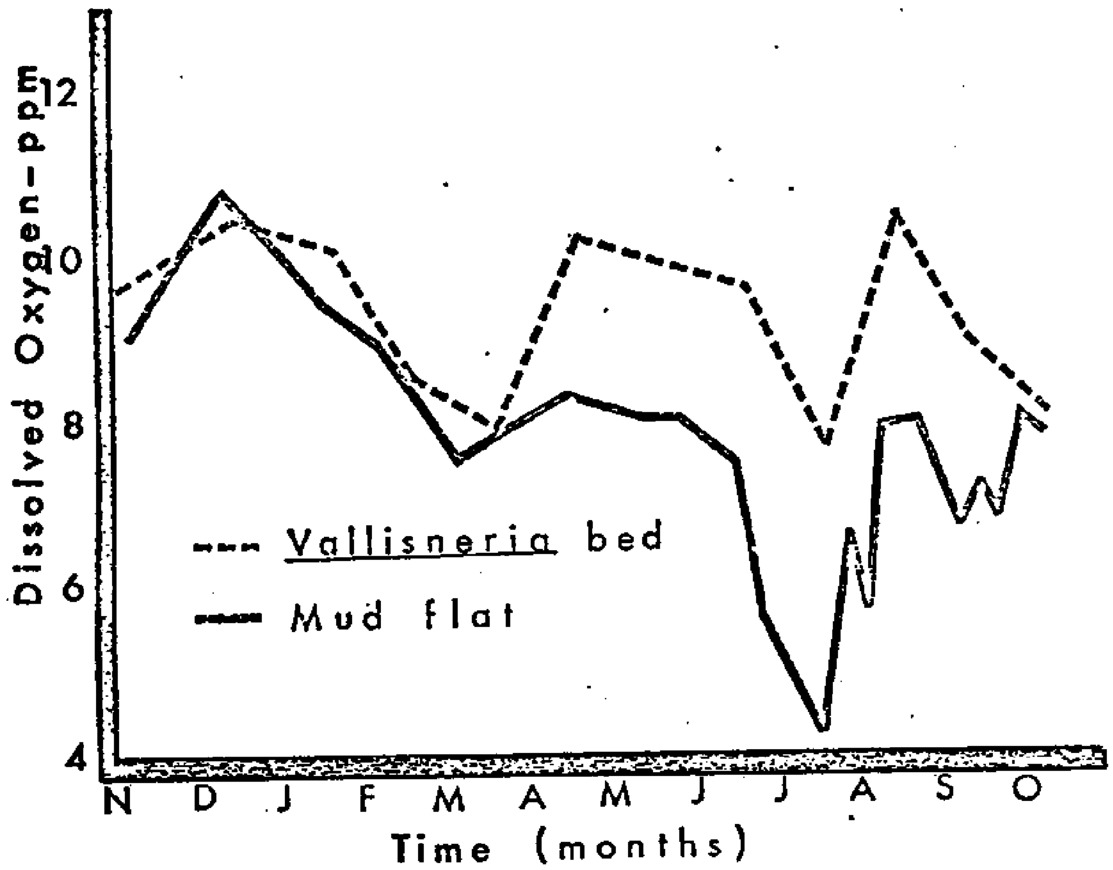

Dissolved Oxygen-ppm
12
10
8
6
4
Vallisneria bed
Mud flat
N
D
J
F
M
A
M
J
J
A
S
O
Time (months)

Fig. 4: Comparison between the monthly (1975-1976) pH in the *Vallisneria* bed and the nearby mud flat of Round Bay.

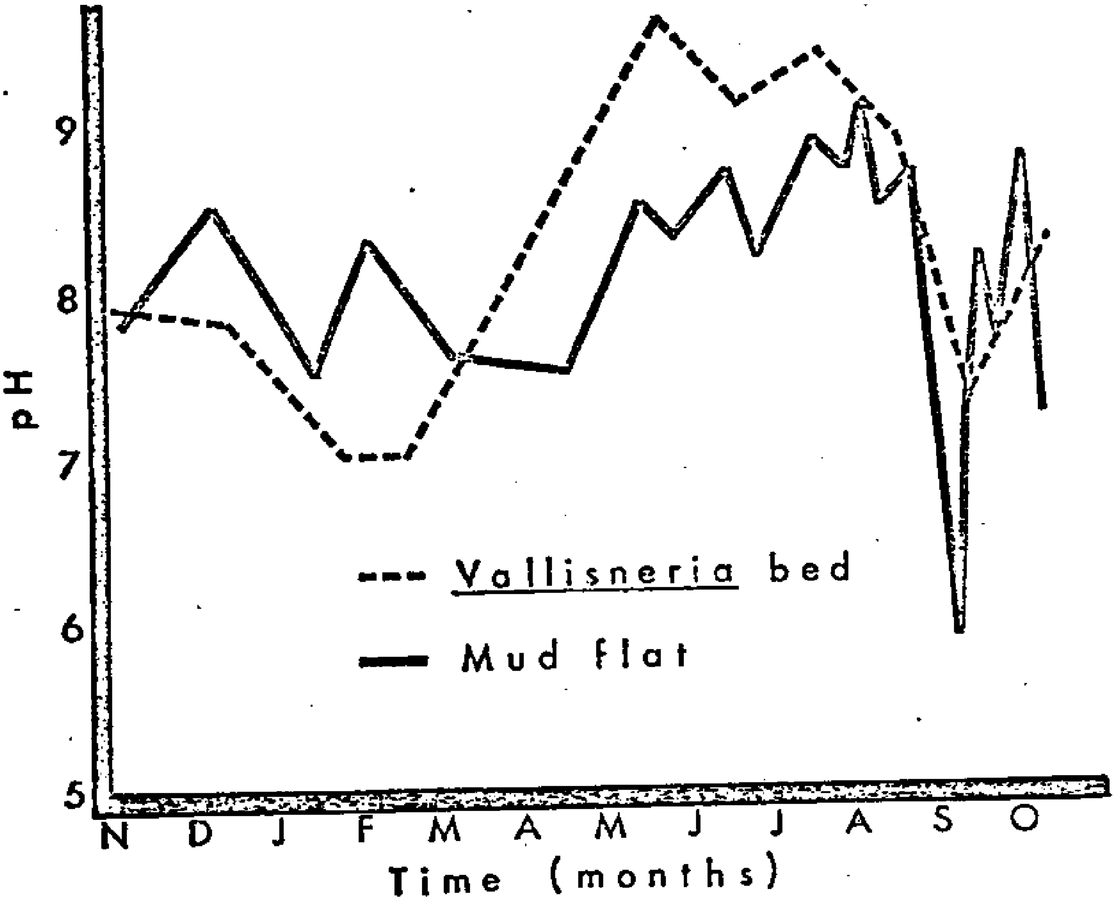
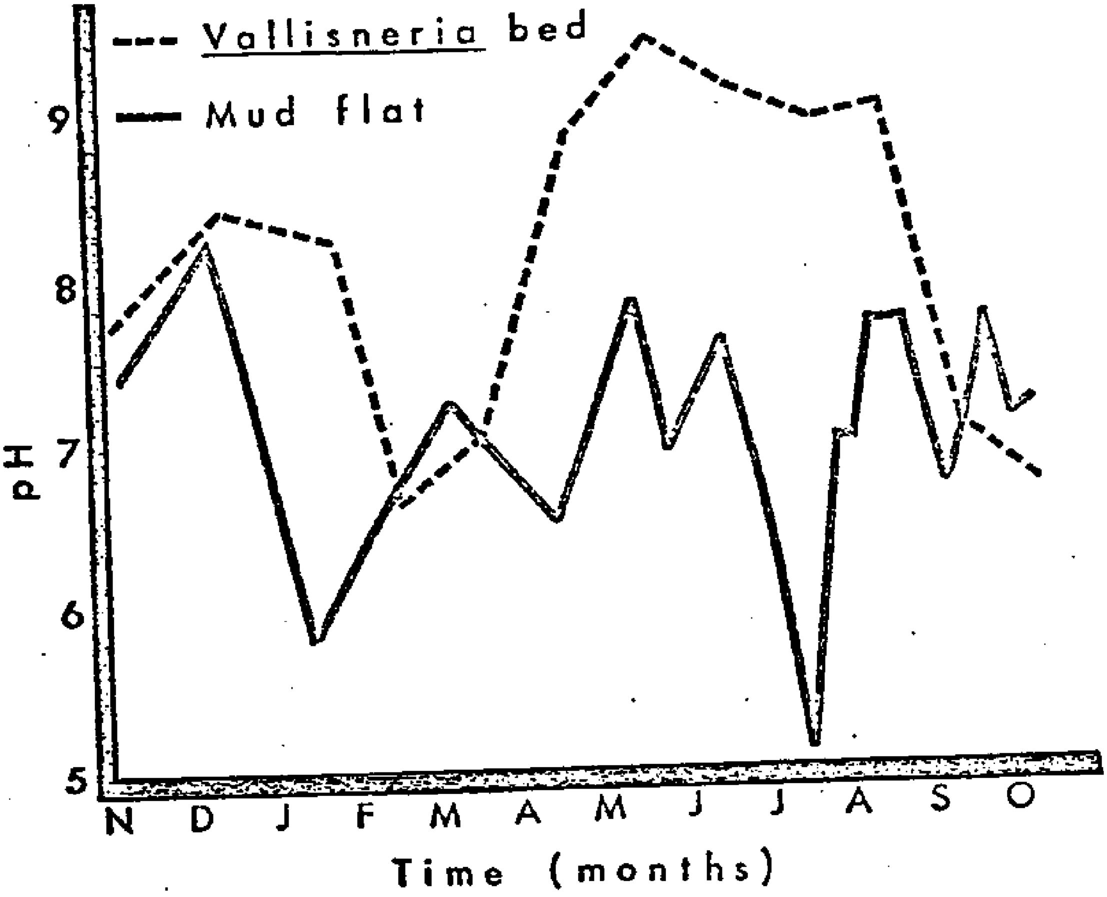

pH
9
8
7
6
5
N
D
J
F
M
A
M
J
J
A
S
O
Time (months)
Vallisneria bed
Mud flat

Fig. 5: Comparison between the monthly (1975-1976) pH in the *Vallisneria* bed and the nearby mud flat of West Bayou.


--- Vallisneria bed
— Mud flat
9
8
7
6
5
pH
N D J F M A M J J A S O
Time (months)

VIII. Immediate (Short-term) Impact of Upland Runoff on Assemblages of Epibenthic Fishes and Invertebrates

Introduction

An intensive sampling effort was maintained during the summer and fall of 1976 and 1977 to determine the nature of short-term fluctuations of organisms in East Bay. Such changes were then compared to meteorological and water quality functions associated with forestry activity. During the first seven months of this program (1976), sampling was carried out and data analyzed by James L. Duncan, and parts of the introduction of this chapter and associated data analysis come from Mr. Duncan's M.S. thesis (Duncan, 1977).

The effects of salinity and temperature fluctuation on estuaries have been extensively studied (Gunter and Hildebrand, 1954; Gunter, 1961; Copeland and Bechtel, 1974; Cherry et al., 1975; Livingston, 1976). However, because of the low natural variability of pH in unpolluted areas (approximately 6.7 - 8.6; Ellis, 1937), there is a paucity of field research concerning the effects of short-term pulses of acidic runoff upon estuarine communities.

Various laboratory studies have shown responses of aquatic organisms to water characterized by low pH. Mount (1973) reported stress (surface swimming and hyperactivity) in fathead minnows (*Pimephales promelas*) after reduction of pH to 4.2. Adult and juvenile blue crabs

(Callinectes sapidus) were found to avoid to a significant degree acidic runoff from clearcut areas in a laboratory environment (Livingston et al., 1976a; Laughlin et al., 1977; also, see other sections of this report). This avoidance appeared to be caused primarily by the reduced pH rather than by the increased color. However, when studied in a field situation, adults appeared to follow this pattern of avoidance while juveniles did not, reflecting the difficulty of extrapolating laboratory results to the field. Natural seasonal migrations of juveniles into these areas during the collection period may be responsible for this observation. Lethal pH values have been reported for many species. A lethal value of 5.8-6.2 was found for young atlantic salmon (Salmo salar) and for sea trout and brown trout (Salmo trutta) in two-day tests (Bishai, 1960). Dahl (1927) reported 80% mortality of yolk-sac trout in 20 days at pH values 4.7-5.4 and 10% mortality in the range 5.1-5.7 using water acidified with peat. Yolk-sac salmon were found to have a median lethal pH value of 4.5 at 12 days in similar water.

Juvenile populations within an estuary are sensitive to pulses of low quality water. Kwain (1975) and Lloyd and Jordan (1964) showed a positive correlation between size and resistance to acid water in rainbow trout (Salmo gairdneri). However, recognition and avoidance ability have been shown to vary with size classes of alevins (Bishai, 1962). Newly-hatched to 4-week-old alevins developed respiratory stress in acidic water (pH < 6.0) but were unable to avoid it effectively. Older stages (5-25 weeks) showed immediate recognition and avoidance of pH 6.5 before respiratory distress occurred. Active migratory fish have also been shown to be highly sensitive to pH variations (Bishai, 1960)

Although fish have been found at pH values between 4.0 and 10.0 in field situations, the safe range has been reported to be 5.0 to 9.0 and for maximum productivity the pH value should lie between 6.5 and 8.5 (ORSANCO, 1955). The European Inland Fisheries Advisory Commission (1969) reported that, below a pH of 5.0, the productivity of an aquatic ecosystem may be considerably reduced. Conditioned response experiments have shown that marine teleosts can discriminate between 0.04 and 0.06 unit reductions in pH (Bull, 1940).

In a physically dynamic estuarine system such as the Apalachicola estuary, many parameters may be shifted because of large episodic inputs of storm water runoff from channelized areas of clearcutting. Dissolved oxygen may be reduced because of acidic release of CO_2 into the water column, which increases biotic respiration, reduction of the euphotic zone by high color (Hassler et al., 1951), and the direct addition of the runoff water which is itself low in dissolved oxygen. Salinity decreases may also occur because of the large fresh-water input. Reduced pH has been shown to act synergistically with other water parameters such as free carbon dioxide concentration (Lloyd and Jordan, 1964; E.I.F.A.C., 1969) and temperature (Kwain, 1975), acting to increase water toxicity.

Biological community indices have previously been used to reflect the effects of impaired water quality conditions on estuarine and coastal assemblages (Copeland and Bechtel, 1971; Boesch, 1972; Holland et al., 1973; McErlean et al., 1973; Livingston, 1975) and have been seriously criticised (Hurlbert, 1971; Livingston, 1975). No single index appears to be sufficient to assess water quality impact upon estuarine communities because of the complex interaction of many parameters, such as faunal density, dominance, species richness, affinity characteristics, and changes in spacial and temporal population dynamics (Livingston, 1975).

This section of the report will involve potential short-term effects of episodic influxes of highly colored and acidic runoff from clearcut areas of Tate's Hell Swamp. Epibenthic community structure changes were examined in the East Bay portion of Apalachicola Bay. Particular emphasis was placed on a comparative analysis before, during, and after heavy local rainfall. In addition to the regular day-night otter trawling, a comparison of the trammel net data (Round Bay vs. West Bayou) was made.

Methods and Materials

In addition to the routine monthly sampling effort, 2 seven-month sampling periods (May-November) were chosen to coincide with maximum local rainfall and periods of peak productivity in the upper East Bay system. Three representative sampling stations were chosen in the East Bay portion of the Apalachicola system because of their proximity to clearcutting activities. Two experimental stations (East and West Bayous, stations 5C and 5B, respectively) receive direct drainage from upland clearcut regions. Both stations are relatively shallow inlets (~ 1.2 m) bordered by small fringing beds of the benthic macrophyte *Vallisneria americana*. Unspoiled marsh and timberland surround these areas with no known anthropogenic impact except upland forestry activities. The sediments at the two stations are similar, with a mean grain size of 4.22 ø units and a mean annual organic content of 11.23% at station 5B.

The control area (station 4A, Round Bay) is located approximately 4 kilometers to the west of the experimental stations. The mean depth is slightly greater (~ 1.5 m) and the mud-silt substrate is of a larger average grain size (3.98 ø units). All sampling was done over the mud-silt substrate which runs through the center of the inlet surrounded by shallower Vallisneria americana grass beds. Mean annual organic content of the sediment is reported to be 8.60%. Prior to 1976, this area was in a relatively natural state with no major drainage from any clearcut regions.

Sampling procedures

Day and night collections were taken following periods of intensive rainfall using a 5-meter (16 foot) otter trawl (1.91 cm wing mesh with a 0.64 cm mesh liner) at speeds of 2-3 knots. Day collections were taken at approximately 1400, 1530, and 1700 hours at stations 5C, 5B, and 4A, respectively. After sunset, sampling was performed in the same sequence of stations at approximately 1930, 2100, and 2230 hours. During dry periods, collections were made at least once every two weeks.

Seven repetitive 2-minute trawl-tows were taken at each station. This trawling technique has been proven adequate for representative sampling (Livingston, 1976). Fishes and invertebrates were preserved in 10% formalin solution for later identification and measurement. Fish specimens were measured for standard length, penaeid shrimp for total length (tip of the rostrum to the end of the telson), and crabs for carapace width. Other shrimp, snails and bivalves were counted.

In addition, night trammel net collections were made at stations 4A and 5B. This was done at monthly intervals from March, 1975 to the present.

Pearson product-moment correlations were calculated to measure the strength of relationship between pairs of physical and biological variables. This strength of relationship reveals both the goodness of fit of a linear regression line to the data and the proportion of variance in one variable explained by another. Statistical use of this correlation coefficient is described by Nie et al. (1970). Significance levels were reported for each correlation coefficient and were derived by the use of the student's t-test with N-2 degrees of freedom for the computed quantity: $r = \left[\frac{N-2}{1-r^2}\right]^{1/2}$

Physico-chemical data

Trends in rainfall over the three-year study period are shown in Fig. 1. Rainfall, together with monthly mean river flow, is shown in Fig. 2. Temperature and salinity trends at the 3 stations in question are shown in Figs. 3-5; pH changes are given in Figs. 6-8. Since these data have already been analyzed elsewhere in this report, no exhaustive review will be made here. However, certain points deserve attention. Despite low (and declining) rainfall during 1976-77, pH tended to drop during this period at station 4A, whereas the opposite was true at stations 5B and 5C. In East and West Bayou, the association of low pH with increased local rainfall tended to weaken with time. In these areas, major pH decreases occurred in late July and August of 1975 with lesser decreases during the preceding spring (March-April) and the months of January; July, and September-October of 1976. A single pH decrease (to 5.5) occurred on September 7, 1976, at control station 4A; however, the pH generally remained well above 7.0. The reason for runoff conditions at the control station was believed to be prior heavy rainfall causing an overflow of a nearby water retention pond. Major input of storm water runoff was not observed at station 4A at any other time. Photographs indicate that recent clearcutting and ditching in the Round Bay drainage area seriously qualifies the use of this station as a control.

During the 1976 sampling period, color and dissolved oxygen were significantly correlated with pH while poor correlations were found between pH and other physical variables (Table 1). Baseline color ranged from 10 to 100 Pt-Co units. This increased to levels above 300 during periods of runoff at stations 5B and 5C. At station 4A the color never exceeded 80 Pt-Co units. For stations 5C, 5B and 4A, the mean diurnal color levels were 120, 94, and 38 respectively. Dissolved oxygen was highest at station 4A ($\overline{X}$ = 9.6 ppm) ranging from 7 to over 13 ppm. Lower dissolved concentrations were found at stations 5B and 5C ($\overline{X}$ = 7.9, 7.8 ppm, respectively), and decreases were observed at these stations during periods of increased storm water runoff when dissolved oxygen was as low as 4.1 ppm. Higher dissolved oxygen levels (maximum 12.3 ppm) at the end of the study were ascribed to seasonal water temperature decreases. Dissolved oxygen and temperature were inversely correlated at all stations (Table 1).

River flow dominated general salinity levels in the East Bay system. Salinity and river flow were significantly correlated at all stations (Table 1). The influence of river flow on various other water parameters in this area has been described (Livingston et al., 1977a). Salinity ranged from near zero during spring months to over 20 °/₀₀ during peak productivity months later in the summer and fall. It should also be noted that during periods of higher salinity, heavy local rainfall and associated storm water runoff significantly reduced this water parameter. It is interesting that, despite the fact that salinity increased in various other areas of the Apalachicola Bay system during the past 3 years, the salinity at stations 5B and 5C tended to decrease over this period of time. This could have been due to increased river flow in 1977, but this deserves more study.

Turbidity and Secchi measurements were found to be highly variable and relatively similar between stations in upper East Bay. These parameters are controlled by both river and local rainfall (Livingston et al., 1977a), although no distinctive correlations were found (Table 1).

Biological trends (1976)

Dominant fishes:

Four species (*Anchoa mitchilli*, *Cynoscion arenarius*, *Leiostomus xanthurus*, and *Micropogon undulatus*) dominated the collections during the intensive survey of 1976. These species comprised 89.7% of the total biomass. The bay anchovy (*Anchoa mitchilli*) was the most abundant species throughout the study period (49.2% of total) and was associated primarily with higher salinity during the fall. The sand sea trout (*Cynoscion arenarius*) was second in abundance (17.0%) with a similar spatial and temporal (late summer-early fall) distribution. The spot (*Leiostomus xanthurus*) and Atlantic croaker (*Micropogon undulatus*) were third (12.3%) and fourth (11.2%), respectively, in dominance occurring early in the study period (spring) during periods of low salinity.

Peak diurnal abundance of *Anchoa mitchilli* occurred from early September through November. Decreases in total abundance simultaneously occurred during an episodic water quality decline at stations 5B and 5C (October 8). However, a decrease was also found at control station 4A while the pH was still relatively high (~7.3). At this time, a significant decrease in salinity occurred at all three stations. Mean river flow (2 weeks prior to sampling date) was still relativey low and, therefore, salinity decrease appeared to be associated with heavy local rainfall patterns. This would explain the acidic conditions at stations 5B and 5C, although, because of the decrease in numbers at station 4A, salinity alteration appears to be the primary physical forcing function with possible contributions from a temperature decline.

Seasonal appearance of Anchoa mitchilli occurred earlier (August) in night collections. Decreases in abundance were observed in early October as with the daytime collections under similar water quality conditions. However, because of the earlier appearance, the September 7 input of storm water runoff elicited a response. Significant decreases in abundance occurred at stations 5B and 5C following water quality deterioration (decreased pH, increased color). Reduction was also seen at station 4A with an immediate, rapid recovery when water quality parameters improved. Salinity decrease (believed due to local rainfall) occurred at this time as well as a slight temperature reduction. Biomass values revealed the same patterns discussed above.

Decreases in Cynoscion arenarius (biomass and abundance) were found to occur on July 16, September 7, and October 8 at both experimental stations (5B and 5C). All cases correlated with the input of acidic runoff into the area. Decline was also observed at control station 4A on September 7 when a pH and salinity decrease occurred. However, no decrease in abundance of this species was observed at station 4A on July 16, when no decrease in water quality occurred. Because of the absence of salinity fluctuations at this time, the decrease in numbers and biomass at stations 5B and 5C appears to be associated primarily with the runoff conditions and a possible high temperature limitation factor. A decrease in abundance was found at all 3 stations in early August with no apparent correlation with physical forcing functions. A slight temperature drop was noted; however, this might have been a natural population fluctuation during transient periods of salinity increase.

Large numbers of Leiostomus xanthurus were found primarily during the early portion of the study (May). Numbers and biomass were reduced during the summer months at all 3 stations, except for an increase at station 5C in late August. An increase of individuals and biomass occurred at station 4A in mid-July while water and quality numbers were low at 5B and 5C. From August throughout the remainder of the study period, considerable variability occurred within this population. Biomass increased in the later portion of the study period while numbers remained relatively low, reflecting the maturing of juveniles which were present in May. In September and October, when water quality decreased, numbers and biomass of this species declined concurrently. The temporal distribution of Micropogon undulatus was similar to that of Leiostomus. After a decrease in abundance in June, no recovery was noted until the recuitment of juveniles in November.

A total of 53 species of fishes were collected during the summer and fall of 1976, with no significant differences in the overall total number of species among the 3 study sites. Transient decreases in numbers of species paralleled water quality reductions in July, September and October at the experimental stations. A decrease occurred at station 4A in early September, although the number of species remained relatively high during July and October. The September decrease coincided with the appearance of dark water at station 4A as previously discussed. Fewer species were observed at all 3 stations in early August. High variability was noted over the entire study period with greater numbers of species generally found at night. Numbers of species decreased at the end of the study period, possibly associated with reduced water temperature.

Total numbers and biomass of fishes declined following peak runoff conditions. The general trend was high numbers and biomass at the beginning of the study (primarily due to the presence of adult Leiostomus xanthurus and Micropogon undulatus) followed by a decrease lasting until the proliferation of Cynoscion arenarius and Anchoa mitchilli beginning in August. The sizeable increase in numbers at stations 4A and 5B in September and October, respectively, was caused by the presence of large schools of anchovies. Numbers and biomass depletion at the end of the study were believed to be associated with declining water temperature. Abundance and biomass at station 5B reflected water quality variations in the form of episodic changes in July, September and October. Immediate biological recovery was noted after water quality improvement.

East Bay was characterized by low diversity due to the few species which dominate the system at a given time. All diversity indices were highly correlated. Differences were noted between day and night as the community structure changed. Dispite high variability due to temporal fluctuations of numbers and species, diversity appeared to change concurrently with water quality shifts. Decline in diversity in July was due to the establishment of a less even distribution of populations within the community. This occurred because the total number of individuals decreased in this period of low water quality, while *Anchoa mitchilli* increased as a resul of seasonal recruitment. Previous data (Livingston, 1977) indicate that the number of species is low during this period of reduced salinity. This corroborates the observed relationships of dominance and diversity. Reduced diversity was observed in September at all stations. The opposite was noted during periods of increased salinity when *A. mitchilli* totally dominated the system. Diversity therefore increased following any decrease in *A. mitchilli* because of the greater number of species and reduced dominance. Margalef's species richness did not follow the same pattern as the diversity indices, increasing significantly during the period of higher salinity. Significant decreases were noted in July during periods of low water quality at station 5B. Although the total number of species decreased, increased species richness occurred during water degradation in September and October because of the large decrease in the total number of individuals. Generally, the high variability of these community indices, due to the rapidly changing system, both physical and short-term biological, reduced their value as indicators of water quality.

Dominant invertebrates:

The dominant invertebrate species during the 1976 intensive study period was the white shrimp, *Penaeus setiferus*, comprising 76% of the total number of invertebrates collected and 52.8% of the total biomass. During the peak abundance of this species (August-October), water quality degradation at all three stations in early September significantly reduced numbers of individuals and biomass. Significant decreases in pH, salinity, and dissolved oxygen as well as a minor drop in temperature occurred at this time. Decreases in numbers and biomass observed in early October reflect these same changes in the system. Low biomass (relative to number of individuals) reveals the recruitment of juveniles into the area in late July and August. Relative biomass rapidly increased until late October. The mean individual size in August was 43 mm, but increased to 100 mm by October. Decreases in abundance in November were attributed to low temperature, which acts as a stimulus for the mature penaeid shrimp to move out of the estuarine areas (Barrett and Gillespie, 1973; Trent et al., 1976). The white shrimp revealed a primarily nocturnal distribution and was most prevalent at the control station (4A).

The blue crab, Callinectes sapidus, was second in invertebrate dominance (17% of total collection; 41.5% of total biomass) with greatest abundance at experimental stations 5B and 5C. Larger individuals were found at station 4A throughout the study period (mean carapace width = 58.4 mm), with the highest number of individuals during the first two months. A similar diurnal distribution was observed at stations 5B and 5C; however, large nocturnal recruitment of juveniles occurred at these stations beginning in August. In May and June the mean carapace width in the experimental areas was 42.2 mm while it decreased to 17.0 mm in August. The spatial preference of juveniles for the East and West Bayous was shown by the reduced average individual size in these areas. Over the entire study period, the mean carapace widths at stations 5B and 5C were 31.1 and 32.6 mm respectively. At both experimental stations, reductions in numbers were observed in September and October. This occurred at station

4A in September. Such changes followed the appearance of storm water runoff.

All 16 species of invertebrates were collected at station 5B, and 5 of the rare species were found only at this location. Mid-range and rare mollusks (*Rangia cuneata* and *Mactra fragilis*) were also associated with East and West Bayous. Species generally associated with grass beds (*Palaemonetes* spp. and *Neritina reclivata*) were found primarily at station 4A. The pink shrimp, *Penaeus duorarum*, also was more abundant at this location. Greatest numbers of individuals were found at station 4A because of the domination of *Penaeus setiferus*.

Both numbers of individuals and total biomass declined during periods of low water quality. Total numbers of species were relatively low and a high degree of variability occurred because of the appearance of rare species. Numbers of individuals were also generally low except in August and early September during juvenile *Penaeus setiferus* and *Callinectes sapidus* immigration. Numbers of individuals and of species increased during months of peak productivity (August-October) while salinity was high. The numbers of species were generally higher at night while little diurnal difference was noted in total abundance (except during August and September). Large variations between day and night diversity indices resulted from this situation. Because *P. setiferus* and *C. sapidus* contributed 93% of the total invertebrate collection, diversity indices were primarily a reflection of changes in these two species in view of changes in total number of species.

Three-year trends

Fishes

Fish data for the combined (day-night) trawling efforts (14 2-minute trawl tows) are shown in Figs. 9-20. In terms of total numbers of fishes, there were major increases during winter periods at all 3 stations; there was also an increase in numbers during the winter months over the entire 3-year period of study. This was due largely to increasing numbers of *Leiostomus xanthurus*, which was dominant during the last 2 years of sampling. The general patterns of abundance were similar in all 3 areas with secondary increases usually occurring during late summer-fall periods. There was also a general tendancy for successive increases in numbers of fishes caught during summer-fall periods over the 3 years of study at stations 5B and 5C; such increases were not as apparent in Round Bay.

At station 4A, the number of fish species and Margalef richness tended to follow similar patterns in time; the Shannon diversity, with certain exceptions, tended to follow these trends. All three indices usually increased during summer-fall periods; this seemed to follow general increases in salinity at these times. Species numbers and Margalef richness in Round Bay tended to go down with the onset of the summer rains but the extremely high rains in late summer of 1975 did not affect these indices to any degree in Round Bay (both indices were increasing as rainfall increased and the pH fell). The general level remained low, however; this could be ascribed to low salinity conditions. The following year (1976), however, the reduced pH (co-occurring with the September rainfall) was associated with precipitous declines in both indices with no such decreases apparent during the succeeding year (1977).

This pattern is even more accentuated when Shannon diversity is considered. The lowest levels of this index occur during the moderate rainfall of September, 1976.

Overall, the seasonal trends in the community structure of fish assemblages in Round Bay can be explained in terms of seasonal cycles of temperature, river flow (salinity), rainfall and associated water quality parameters, and the recent clearing and draining of the upland system by forestry interests. When analyzed from the perspective of the long-term changes in the system, the destabilization of the physico-chemical and biological systems during peak periods of production tends to be short-term, with recovery occurring in terms of weeks or months. However, this form of impact does occur at a particularly sensitive period with regard to the bay cycles of fish productivity.

In addition to long-term increases in numbers of fishes at station 5B, there was a general increase in the species richness and Margalef richness over the 3-year period of study. Shannon diversity appeared to be temporally unstable, and, along with the other indices, reflected the cold winter of 1976-77 to a considerable degree. During the heavy rainfall in the summer of 1975, no fish were taken in daylight hours at station 5B in August; this is a most unusual situation and has never been experienced at any other time or station in the entire bay system. While the general seasonal patterns (increased species richness and diversity during years of higher summer salinity; declines in these indices during winter periods, etc.) are similar, the differences in the summer-fall patterns of fish distribution between stations 4A and 5B are worthy of mention. Richness at 4A was actually increasing during the heavy rains of the summer of 1975 while such numbers were higher at West Bayou than in Round Bay during the summer and fall of 1976. At this time, upland portions of the Round Bay area were being cleared. With the exception of the summer of 1975 at 5B, the species diversity also showed this pattern of differences (little impact during the September, 1976 rainfall at 5B; maximal decline at 4A). By 1977, with the exception of a decrease in Shannon diversity at 5B in late summer, there was little difference between the 2 areas in terms of the major indicators of fish distribution.

In addition to the long-term trends of recovery in numbers, the East Bayou fish data reflected the various seasonal trends (salinity, temperature) discussed above. In terms of numbers of species, Margalef richness, and Shannon diversity, the patterns at station 5C resembled those at 5B; likewise, these data did reflect major differences with biotic trends in the Round Bay area in 1975.

In all, the fish assemblages in upper portions of East Bay reacted to various physico-chemical forcing functions and changes in productivity and were directly related to seasonal variations of temperature (with the cold winter of 1976-77 having a pronounced effect), salinity (summer-fall increases in 1976-77), local rainfall patterns (heavy summer rainfall during 1975, moderate summer-fall rainfall during 1976, light rainfall during 1977), and the spatial distribution and timing of forestry activities in the different drainage basins of the Tate's Hell Swamp-East Bay system.

Invertebrates

Invertebrate data for combined (day-night) trawling efforts (14 2-minute trawl tows) are shown in Figs. 21-32. During the summer-fall periods, the upper East Bay system is a major nursery area for blue crabs and penaeid shrimp, as shown in the distribution of numbers of individuals in system-wide sampling efforts. In August, 1975, blue crabs predominated, especially in East Bayou. After the heavy rainfall, only 8 invertebrates were taken at station 5B during the day. This indicated particular stress due, in large part, to low salinity and pH. The major September peaks at all 3 stations were composed primarily of white shrimp (*Penaeus setiferus*) and pink shrimp (*P. duorarum*). The differences in numbers tended to reflect the water quality features of the 3 respective study areas during the summer-fall of 1975. Numbers of individuals were generally low at all 3 stations the following summer (1976), with penaeid shrimp predominating at this time. The sharp decline in numbers of invertebrates at station 4A in September was closely associated with the pattern of rainfall and water quality at the time. The following year (1977), there was a general recovery of penaeid shrimp in Round Bay during July and August; however, after the September rains and local decreases in water quality, the numbers of invertebrates dropped precipitously in this area. Subsequent recovery occurred only in East and West Bayous during late fall periods. The low numbers of invertebrates in these areas during 1976 remains unexplained. With this exception, the invertebrate numbers appear to follow patterns that follow the seasonal cycles of water quality as a response to the local rainfall and runoff described elsewhere in this report.

The species richness and Margalef richness indices reflect increased sensitivity to runoff in Round Bay with time over the 3-year period even though rainfall tended to decrease substantially during this time. In the summer of 1975, there were actually richness peaks in Round Bay during periods of high rainfall whereas July rainfall in 1976 and August-September rainfall in 1977 were associated with sharp declines in invertebrate species richness functions. Decreases in invertebrate diversity indices followed closely the local rainfall patterns with particularly extended decreases (and high variability) during the summer of 1976. The overall temporal patterns of the invertebrate biota tended to follow the physical functions (temperature, salinity, water quality) described elsewhere in this report.

In East Bayou and, especially, West Bayou, the invertebrate richness and diversity indices tended to reflect the heavy rainfall (and associated water quality changes) of the summer of 1975. In these areas, there was recovery during the drought of 1976. It is possible that the general declines in these indices during 1977 reflected the effects of the particularly cold winter of 1976-77.

In general, when compared to the fish data, the invertebrate assemblages in upper East Bay areas appeared to respond in a similar (though not identical) fashion to short-term and seasonal changes in the physico-chemical environment. There is evidence that the nursery areas of East Bay are vulnerable to short-term water quality changes associated with forestry activities in the respective drainage basins of the Tate's Hell Swamp-East Bay system.

Trammel net data

The results of the trammel net effort in Round Bay and West Bayou are given in Table 2 and Figs. 33-34. During the first year of sampling, the heavy local rainfall was associated with relatively low numbers of organisms in West Bayou relative to Round Bay. In 1976, increased late summer rainfall was associated with declines in numbers in both areas, with partial recovery apparent in West Bayou and full recovery in Round Bay during the following fall. During the summer-fall period of 1977, there was a pronounced recovery of the numbers of organisms in both areas, with the modest fall rainfall associated with decreased numbers of individuals in Round Bay and increased numbers in West Bayou. These changes tended to follow general trends in salinity and specific changes in water quality parameters such as pH. With minor variation, the long-term changes in numbers of species showed similar relationships to the rainfall and water quality data. Since the trammel net data are largely made up of various fish species, it is not surprising that the cold winter of 1976-77 did not appear to have a long-term impact on these biotic indices. Thus, the trammel data indicate recovery of numbers of individuals and species in Round Bay and West Bayou during 1977 with station-specific trends of local rainfall, forestry operations, and water quality indicators that parallel those shown for the otter trawl data over the 3-year period of study.

Discussion

Various independent environmental factors tend to interact to determine temporal and spatial distribution within an estuarine system. These factors include biological functions (predator-prey relationships, trophic competition, migratory patterns, reproductive cycles), geographical functions (depth, substrate), and water quality parameters.

Seasonal occurrence of dominant East Bay fishes agrees, in general, with previous reports from other Gulf and south Atlantic estuaries (Welsh and Breder, 1923; Hildebrand and Schroeder, 1928; Gunter, 1938, 1945, 1950; Roelofs, 1951; Townsend, 1956; Reid, 1957; Dawson, 1958; Roithmayr, 1965; Parker, 1971; Perret, 1971; Perret and Caillouet, 1974; Subrahmanyam and Drake, 1975, Livingston et al., 1977b). The primary dominant, *Anchoa mitchilli*, was most abundant in the fall while *Cynoscion arenarius* was prevalent in late summer-early fall. Reported spawning periods of May-August and March-May, respectively, corroborate this finding (Hildebrand and Cable, 1930; Reid, 1954). The spot (*Leiostomus xanthurus*) and the Atlantic croaker (*Micropogon undulatus*) are predominant in late winter and spring with low numbers during the study period. Spawning occurs from October to January for both species (Hildebrand and Schroeder, 1928; Pearson, 1929; Gunter, 1945; Suttkus, 1954). The staggered reproductive cycles produce this temporal partitioning of the bay, ensuring limited direct competition (Livingston et al., 1977a). Temporal distribution of major invertebrates agreed with previous findings in other estuarine areas (Lindner, 1936; Anderson et al., 1949; Daugherty, 1952; Ingle, 1957; Darnell, 1959; Loesch, 1965; Joyce, 1965; Baxter and Renfro, 1967; Temple and Fisher, 1967).

Trophic relationships may be of critical importance in the spatial partitioning of estuarine organisms (Livingston et al., 1976b). The apparent preference of juvenile blue crabs for the experimental stations may be caused by a trophic attraction to the increased detritus flushed from clearcut areas. Juvenile blue crabs do consume detrital materials while larger individuals (20-100 mm) are omnivorous, feeding upon mollusks (primarily clams and mussels), fishes, crustaceans (primarily amphipods and other crabs) and polychaetes (Van Engel, 1958; Tagatz, 1969; Odum and Heald, 1972). Also, intraspecific (cannibalistic) feeding patterns as a determinant of relative distribution of adult and juvenile blue crabs cannot be ruled out as a possible determinant of blue crab distribution. Juvenile and adult white shrimp are also benthic omnivores, devouring plant and animal material such as algae, small mollusks, polychaetes and small crustaceans (Williams, 1965). Large adults may even feed upon small fishes and other shrimp (Moffett, 1970). Benthic infauna numbers and diversity have been reported to be greater at station 4A, possibly because of larger substrate grain size and benthic macrophyte assemblages. The total epibenthic invertebrate population was also found to be greatest at station 4A. Therefore, the white shrimp preference for station 4A is believed due to a trophic phenomenon and related substrate conditions. Prior studies have shown a definite preference in white shrimp for a muddy substrate consisting of primarily terrigenous silt material such as that at station 4A (Springer and Bullis, 1954; Klima, 1974).

The trophic relationships of dominant Apalachicola Bay fishes are described by Sheridan (1977). Anchoa mitchilli is reported to be a generalized planktivore at the predominant size class collected (35-50 mm) (Darnell, 1958; Odum and Heald, 1972; Carr and Adams, 1973). Cynoscion arenarius (40-99 mm) is reported to consume zooplankton such as mysids, shrimp and larval or juvenile fishes (Darnell, 1958; Springer and Woodburn, 1960) while Micropogon undulatus (10-50 mm) and Leiostomus xanthurus ($<$ 40 mm) feed on smaller zooplankton such as copepods and amphipods (Roelofs, 1954; Fontenot and Rogillio, 1970). Larger spot become benthic omnivores (Springer and Woodburn, 1960). Thus some of the primarily planktivorous fishes of East Bay are less dependent upon substrate, benthic macrophyte assemblages, and allochthonous forms of detritus than the infaunal and detritus feeding fishes and invertebrates.

Habitat structure alteration by water quality degradation may stress the stability of this temporal and spatial succession of organisms in East Bay. During periods of storm water runoff in summer-fall periods of episodic local rainfall, significant decreases were noted in almost all biological indices in receiving areas of the bay. Decreases in total abundance and biomass of fish at these times may also be associated with transient seasonal turnover of dominants.

During runoff conditions, many water quality parameters changed, with various interactions tending to impede biological processes. Discharge of acidic runoff results in the formation of free carbon dioxide, depending upon the hardness of the receiving waters (E.I.F.A.C., 1969). Lloyd and Jordan (1964) reported that low levels of free carbon dioxide can reduce fish survival times in waters with reduced pH. Associated increased biotic respiration will also cause depletion of dissolved oxygen, increasing water toxicity (Jones, 1952). Hydrogen ions have a toxic effect, causing suffocation from the precipitation of mucus on the gill epithelium or by the precipitation of proteins within the epithelial cells (Ellis, 1937; Westfall, 1945). Chronic exposure to low pH may cause hypersensitivity to bacteria (Neess, 1949) and the proliferation of various fish parasites (Bauer, 1959). Increased water color indicates high levels of constituents such as tannins, lignins, humic and fulvic acid complexes, etc.; all known inhibitors of various biological processes. High color may also affect trophic relationships because of visability limitations. During periods of reduced river flow, salinity alteration by the increased flushing of large quantities of fresh water into the system may also cause local water to be unsuitable for many species (Gunter, 1950; Gunter and Hildebrand, 1954). Increased rainfall would thus have considerable potential for impact on water quality conditions. Identification of the long-term changes in such functions is also important to an evaluation of temporary (transient) changes of such functions at the system level.

Diversity indices have previously been used as indicators of water quality (Copeland and Bechtel, 1971). MacArthur (1955) suggested that areas with high diversity have greater stability because of their more numerous energy pathways. Diversity criteria have been used to differentiate polluted areas (diversity < 1) from unpolluted (diversity $>$ 3) (Wilhm and Dorris, 1968). In a system characterized by high dominance such as East Bay, species diversity is naturally low and therefore may be misleading as an indicator of pollution. Diversity at all stations was quite often below one yet never found above two. Water quality stress may disrupt the periodic use of the system by dominants without substantially altering the species diversity (Livingston, 1975; Livingston et al., 1976b). Thus, as was found in this study, analysis of individual dominants, relative numbers of individuals and species, and biomass fluctuations may be better indicators of stress in a high dominance system.

This portion of the study was directed at the short-term response of the epibenthic estuarine community to episodic influxes of storm water runoff. The recent clearing and channelization of portions of Tate's Hell Swamp have significantly increased storm water drainage into areas of East Bay following local rainfall. This increased runoff reduces water quality in these regions. At such times, pH levels are reduced below desirable levels for aquatic biota (ORSANCO, 1955; E.I.F.A.C., 1969). Long-term trends of reduced numbers of larger fishes and blue crabs were noted in the West Bayou area, particularly during the summer-fall runoff periods. Changed environmental conditions which could adversely affect the productivity of any estuary may significantly reduce the local commercial fishery (Sykes and Finucane, 1966). Important commercial finfish and invertebrate species of the Apalachicola Bay area (_Penaeus_ spp., _Cynoscion arenarius_, large blue crabs, etc.) react to the water quality with short-term declines in numbers during periods of runoff.

The data thus indicate that the various assemblages of organisms in East Bay are dependent on seasonal and annual fluctuations of temperature, river flow (and associated water quality functions such as salinity, color, turbidity), local rainfall (and associated water quality functions such as salinity, pH, color), and species-specific biological relationships. Water quality alterations associated with forestry activity occur at the exact time of the year (and during the particular years) of maximal biological productivity in the bay. This includes the occurrence and distribution of key assemblages of developing stages of finfishes and invertebrates. Partly because of regrowth of vegetation in cleared areas and long-term decreases of rainfall in the area, the impact of forestry operations appears to be temporary, lasting anywhere from a few weeks to 24 months. However, since the resulting habitat destruction occurs at the time of maximal productivity of the system, such changes, if widespread, could have serious detrimental effects on the Apalachicola estuary. This is especially true if the clearing activities are continuous from one year to the next in (drainage) areas contiguous to the upper East Bay nurseries.

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Table 1: Pearson correlation coefficients for physico-chemical variables measured at control (4A) and experimental stations (5B and 5C) in East Bay (Apalachicola Bay, Florida). R^2 values are shown with the significance level underneath. Significant correlations are underlined.

STATION 4a night\day

	pH	Salinity	Color	Temperature	Turbidity	Dissolved Oxygen	Secchi	River Flow	Rainfall
pH		-.1832 .233	-.3142 .053	.5475 .009	.2565 .152	.3654 .068	.2894 .122	.4739 .023	-.4097 .046
Salinity	-.0721 .388		-.2130 .197	-.0614 .374	-.3893 .066	.0822 .373	-.2470 .162	-.4032 .049	.1458 .282
Color	-.4933 .019	-.2145 .196		-.3667 .074	.2940 .118	-.3350 .087	-.5179 .011	.1173 .321	.4535 .029
Temperature	.4646 .026	.0287 .455	-.4441 .032		.3223 .096	-.1752 .243	.3258 .094	.3821 .059	-.2207 .189
Turbidity	.1383 .292	-.1864 .227	.2250 .185	-.2018 .211		-.1629 .250	-.0946 .354	.4358 .035	-.2250 .185
Dissolved Oxygen	.1971 .217	.0035 .494	-.2195 .191	-.4481 .020	-.2885 .122		.3794 .060	-.0880 .354	.0758 .382
Secchi								.0114 .482	-.2146 .196
River Flow	.2768 .133	-.4393 .034	-.0486 .424	.3153 .101	.0615 .404	-.1391 .298			-.1441 .284
Rainfall	-.4531 .020	.0528 .418	.4378 .035	-.2270 .183	-.5184 .014	.1384 .292		-.1441 .284	

STATION 5b night\day

	pH	Salinity	Color	Temperature	Turbidity	Dissolved Oxygen	Secchi	River Flow	Rainfall
pH		.3111 .304	-.9060 <.001	.0373 .442	.3519 .076	.6082 .004	.4096 .046	.0752 .323	-.1123 .329
Salinity	.4785 .022		-.4172 .042	-.2138 .197	-.1227 .314	.3233 .095	.1818 .235	-.4351 .035	.0641 .400
Color	-.7549 <.001	-.5051 .016		.0542 .415	-.2016 .211	-.6549 .002	-.3858 .057	.0329 .450	-.0100 .484
Temperature	-.3084 .107	-.2613 .147	.2542 .152		.2867 .124	-.5211 .006	-.1942 .220	.3941 .053	-.2394 .169
Turbidity	.2838 .122	-.2218 .188	.0046 .493	.0848 .359		-.2549 .154	.0091 .486	-.0123 .481	-.2110 .200
Dissolved Oxygen	.5336 .011	.3299 .091	-.4853 .021	-.8643 <.001	.0007 .499		.5893 .007	-.0378 .438	-.0198 .469
Secchi								-.1884 .232	-.4032 .049
River Flow	-.3078 .107	-.5078 .016	.2379 .171	.3848 .057	-.0477 .425	-.3377 .086			-.1441 .284
Rainfall	-.3247 .094	.0207 .467	.0031 .495	-.2060 .206	-.5217 .006	.0371 .442		-.1441 .284	

STATION 5c night\day

	pH	Salinity	Color	Temperature	Turbidity	Dissolved Oxygen	Secchi	River Flow	Rainfall
pH		.3418 .287	-.7570 <.001	.2342 .175	.1346 .297	.4073 .047	.3257 .094	.1664 .255	-.0633 .402
Salinity	.3365 .086		-.3426 .082	-.3046 .110	-.4421 .032	.2632 .146	-.1403 .289	-.4573 .028	.0537 .416
Color	-.6034 .004	-.5104 .015		-.2434 .165	-.0497 .392	-.3483 .078	-.4307 .037	-.0293 .454	.0879 .361
Temperature	.0442 .431	-.1803 .237	.0533 .417		.4955 .018	-.5700 .007	.1961 .218	.3910 .064	-.2424 .166
Turbidity	.1068 .337	-.2247 .165	.0859 .367	.0367 .443		-.1799 .242	.4404 .034	.5728 .006	-.5461 .010
Dissolved Oxygen	.3547 .074	-.0016 .498	-.3363 .086	-.7758 <.001	-.0942 .355		.3186 .099	-.1538 .271	-.1255 .310
Secchi								-.0899 .391	-.3627 .070
River Flow	-.0143 .478	-.4666 .025	.1736 .245	.3969 .061	-.1063 .334	-.2508 .158			-.1441 .284
Rainfall	-.2962 .116	.0708 .390	.1409 .229	-.1805 .237	-.2320 .177	-.1509 .275		-.1441 .284	

Table 2: Larger organisms taken by trammel nets at night at stations 4A (Round Bay) and 5B (West Bayou) in the Apalachicola estuary from March, 1975 through February, 1978.

ROUND BAY (TRAMMEL NETS: STATION 4A)

	3-75	4-75	5-75	6-75	7-75	8-75	9-75	10-75	11-75	12-75	1-76	2-76	3-76	4-76	5-76	6-76	7-76	8-76	9-76	10-76	11-76	12-76	1-77	2-77	3-77	4-77	5-77	6-77	7-77	8-77	9-77	10-77	11-77	12-77	1-78	2-78	TOTAL
Bagre marinus	7	1	3	7		22	8	7	11					3	5	6		3	5	3						6	20	10	15	7	6						155
Arius felis			1	3		15		2		6	1			2	6	16	1	1	3	6	6				2	4	8	41	9	11	5		4				153
Dorosoma cepedianum												1		1		2				29					1		6	3	4	1	1						44
Dasyatis sabina	13	4	3			1	1	4					3		6	1			3	6	2	1				6			1				1				56
Mugil cephalus				3						8						5		2		8					1	1			3	4	3		1		2	1	34
Lepisosteus osseus	1	3	8	1				1	1				2			2			1	3					7	8			1	2	2					3	34
Paralichthys lethostigma	3			1		1								3		1	1	2	1						4		2	4	1						1	1	26
Micropogon undulatus				2		3	1		1			1	1	1	1	1		1								3		2		3	2						22
Ictalurus catus	2											5	1											1	3					1			2		1		16
Brevoortia patronus						6															6												4				16
Cynoscion nebulosus											5	1									6									1	2						15
Morone saxatilis																												6	8								14
Carcharhinus leucas			5	1					2							2													2								13
Leiostomus xanthurus		5					2			1						1			3							1			1								9
Sciaenops ocellata																1				1					1				1				2				6
Ictalurus nebulosus																																				4	4
Elops saurus				1												1																					2
Alosa chrysochloris																								1												1	2
Cyprinus carpio						1																															1
Dorosoma petenense																				1																	1
Amia calva																									1												1
Lepomis microlophus																									1												1
Lepisosteus oculatus																													1								1
Micropterus salmoides																													1								1
Pogonias cromis																																	1				1
Morone chrysops																																				1	1
TOTAL	26	8	12	19		40	11	14	16	9	8	8	12	10	22	42	2	9	16	45	17	1		3	22	28	40	63	46	29	16		13		4	9	454
Callinectes sapidus (M)	9*	2*	20*	60*		12	6	2	7		8	6*	8	3	14	46	12	20	6		6	4	1	4		6	33	13	32	30	8		2		5	8	331
Callinectes sapidus (F)							1								4	5	4	2	2		3	1	1			1	14	3	11	4	3		1		4		71
WEST BAYOU (TRAMMEL NETS: STATION 5B)																																					
Bagre marinus		2	1	6			7	2	7				1	2		14	2	1	3	1						1	4	5	11	8	1						79
Lepisosteus osseus	3	1				1	3	1			1		2		1	1	3		2	3					1		6	20	1	21	3						73
Arius felis		1	2			1	3	1	3	2					1	2	1		2	1							4	25	7	2	8		5				67
Dorosoma cepedianum							6			2	1		3			2			2	1	2				1	11	10		1								41
Dasyatis sabina			1	1				1	1						1	3			4		5			1			2	1	1	1	2						32
Paralichthys lethostigma	9	1		2			1																				1	1						2			17
Mugil cephalus			1					1		8	2																										12
Micropogon undulatus							1								1				3							2	1			1	3						11
Cynoscion nebulosus									1	1			1											1									2	3	2		11
Ictalurus catus	6												1											1												1	8
Carcharhinus leucas							1									2		1									1										6
Sciaenops ocellata			1					1												2																	4
Amia calva	2										1																										4
Brevoortia patronus										1																								2			3
Micropterus salmoides																											2										2
Cyprinus carpio			1																																		1
Alosa chrysochloris																					1																1
Ictalurus nebulosus																									1												1
Pogonias cromis																											1										1
TOTAL	19	4	7	8		3	21	7	13	14	10		6	2	4	24	6	2	15	7	8			6	8	10	33	47	21	34	16		7	7	2	3	348

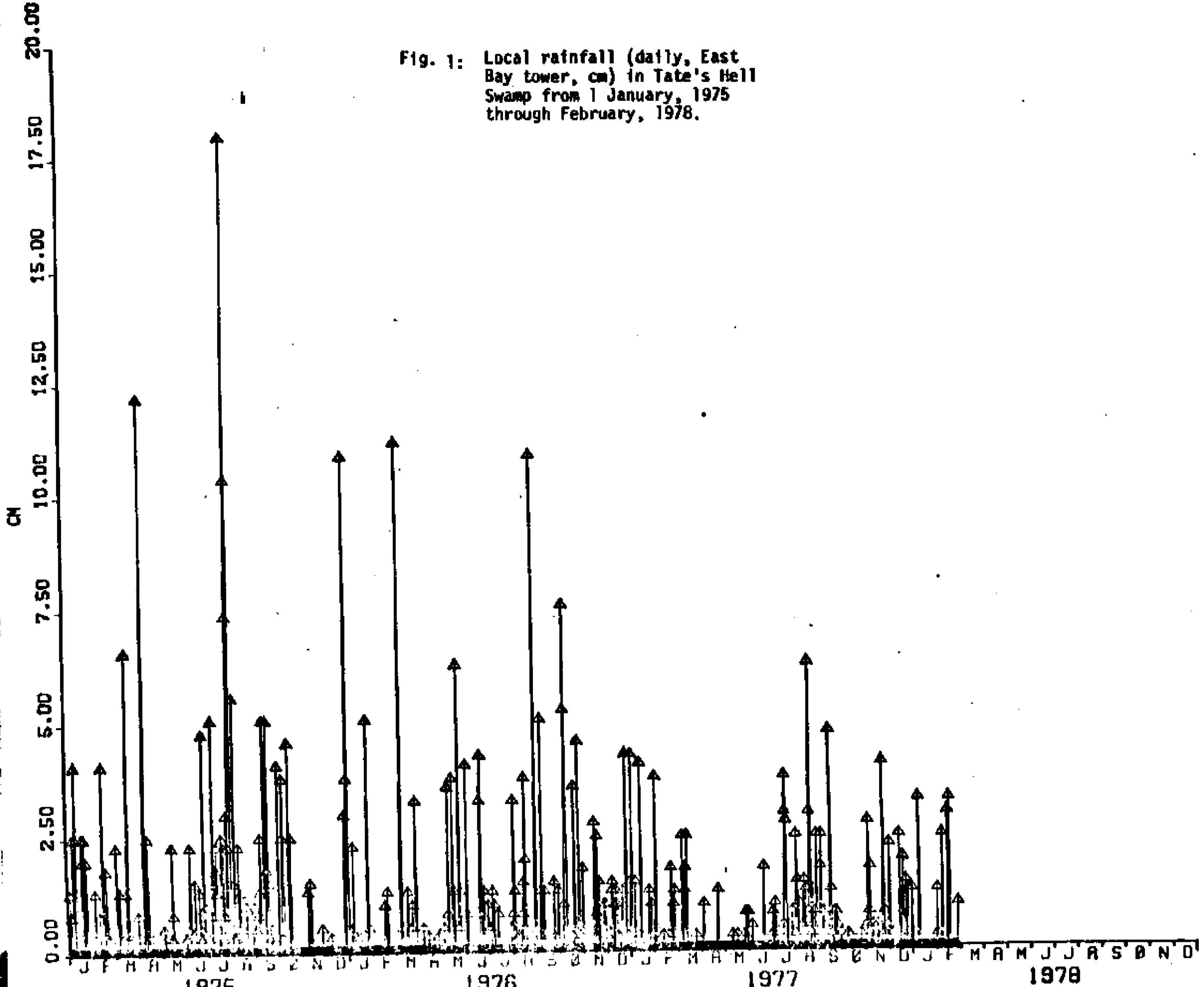


Fig. 1: Local rainfall (daily, East Bay tower, cm) in Tate's Hell Swamp from 1 January, 1975 through February, 1978.

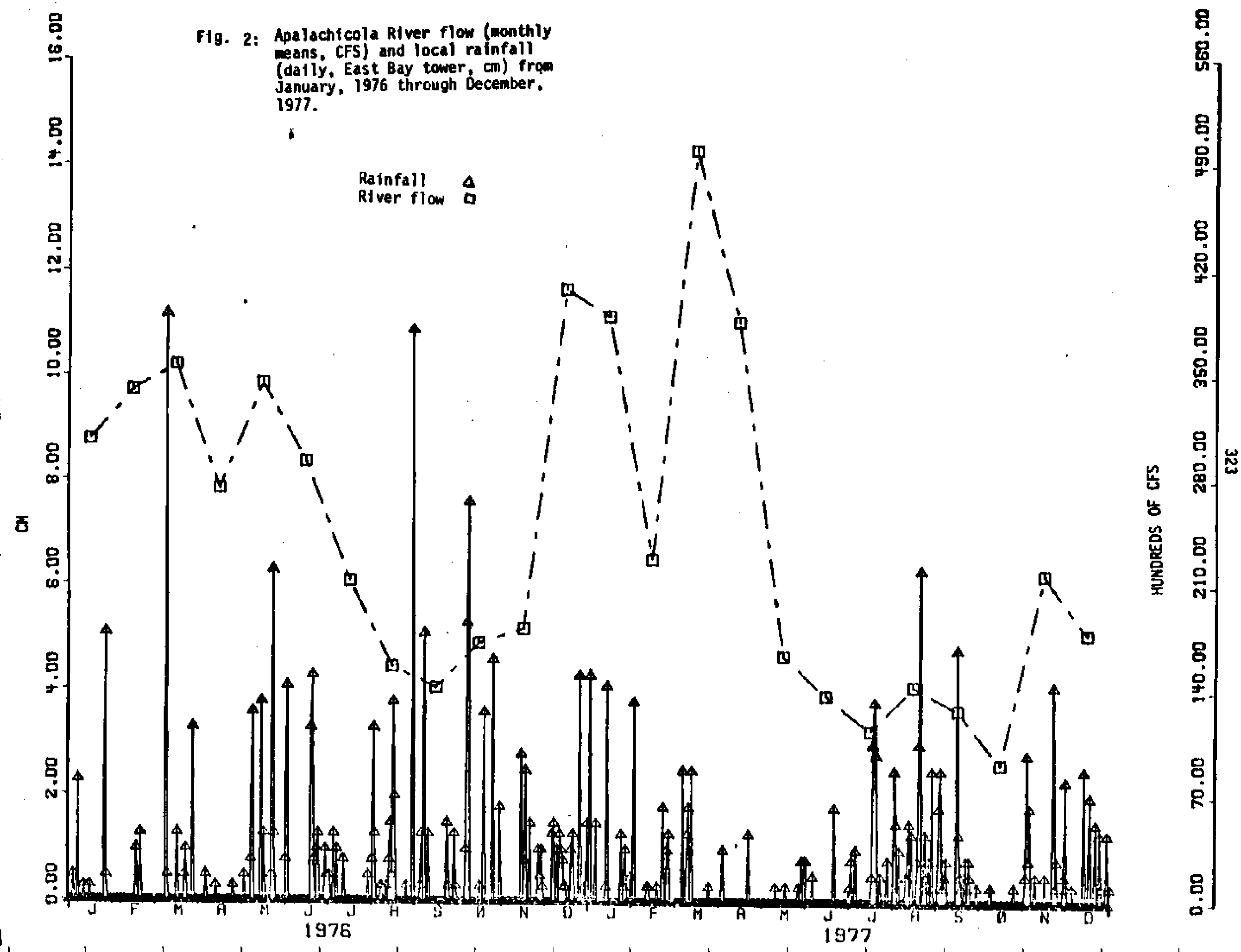


Fig. 2: Apalachicola River flow (monthly means, CFS) and local rainfall (daily, East Bay tower, cm) from January, 1976 through December, 1977.

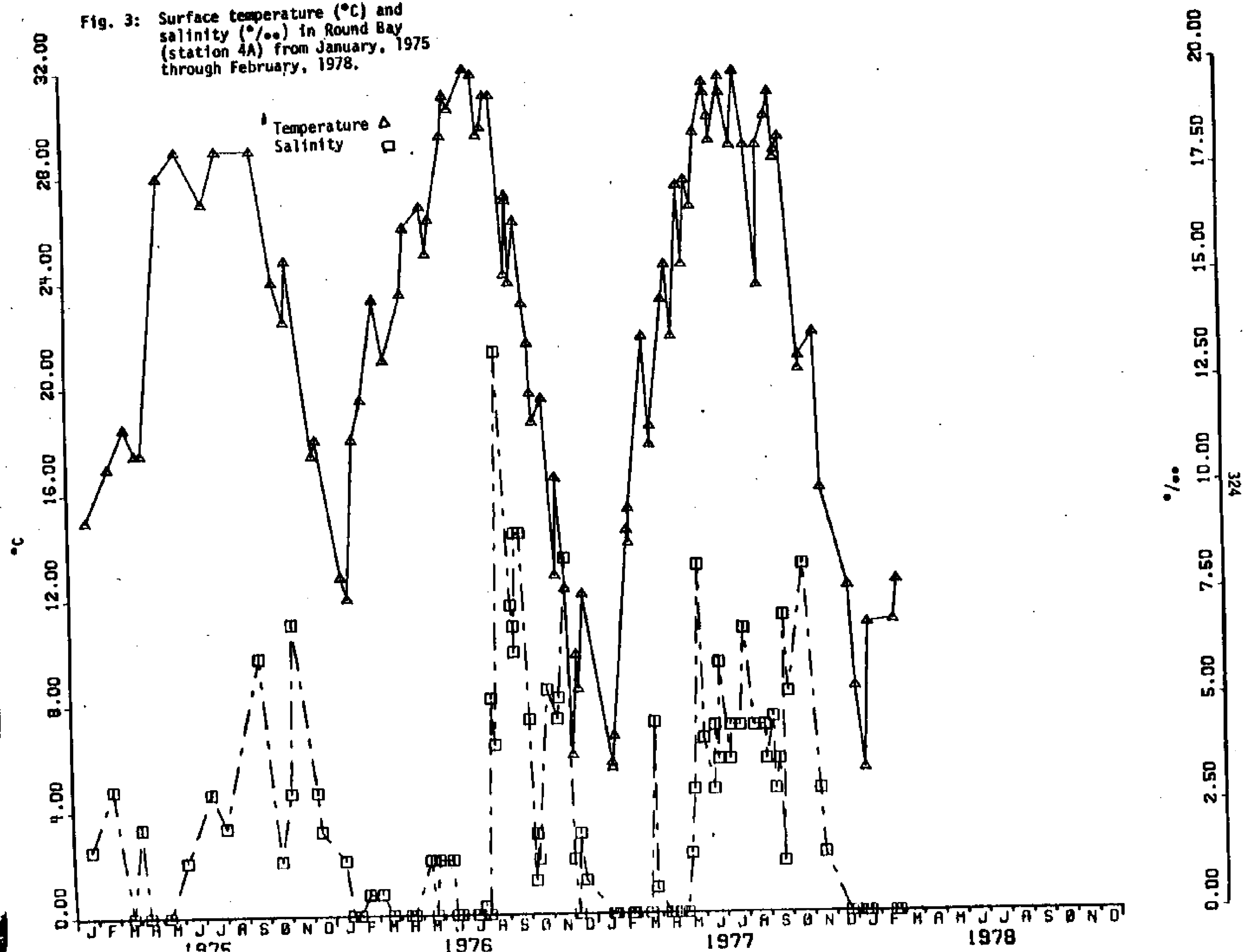


Fig. 3: Surface temperature (°C) and salinity (°/₀₀) in Round Bay (station 4A) from January, 1975 through February, 1978.

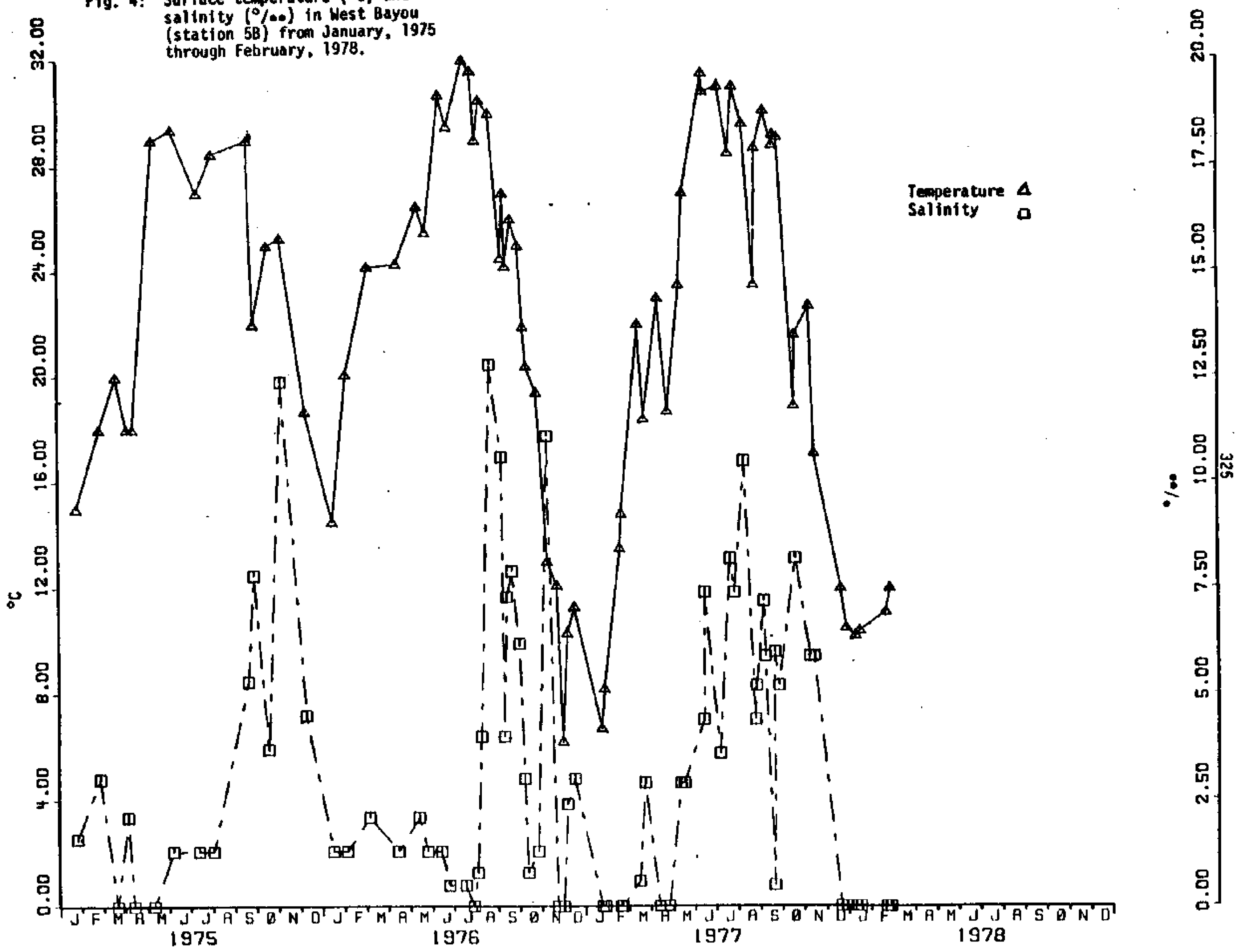


Fig. 4: Surface temperature (°C) and salinity (°/••) in West Bayou (station 5B) from January, 1975 through February, 1978.

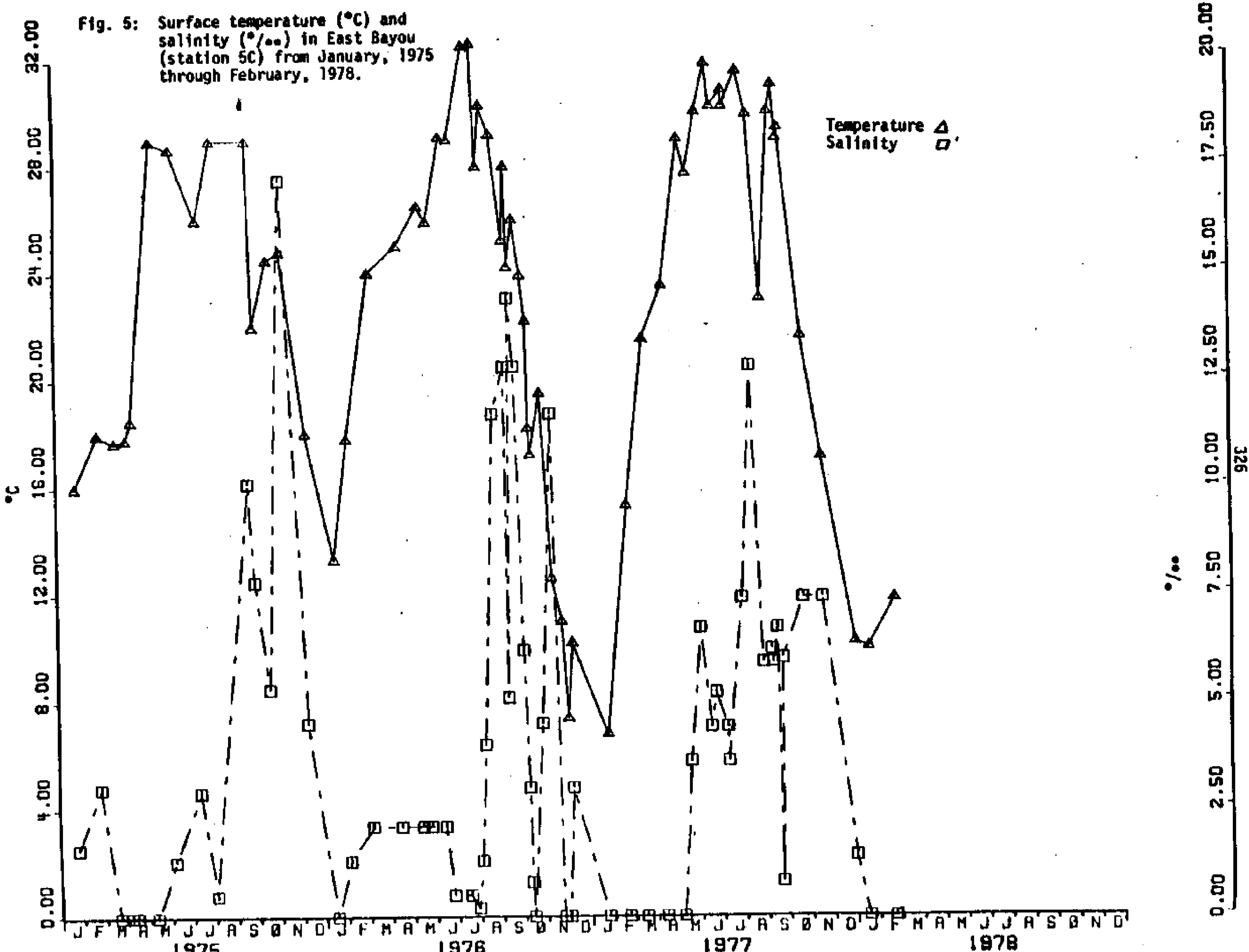


Fig. 5: Surface temperature (°C) and salinity (°/₀₀) in East Bayou (station 5C) from January, 1975 through February, 1978.

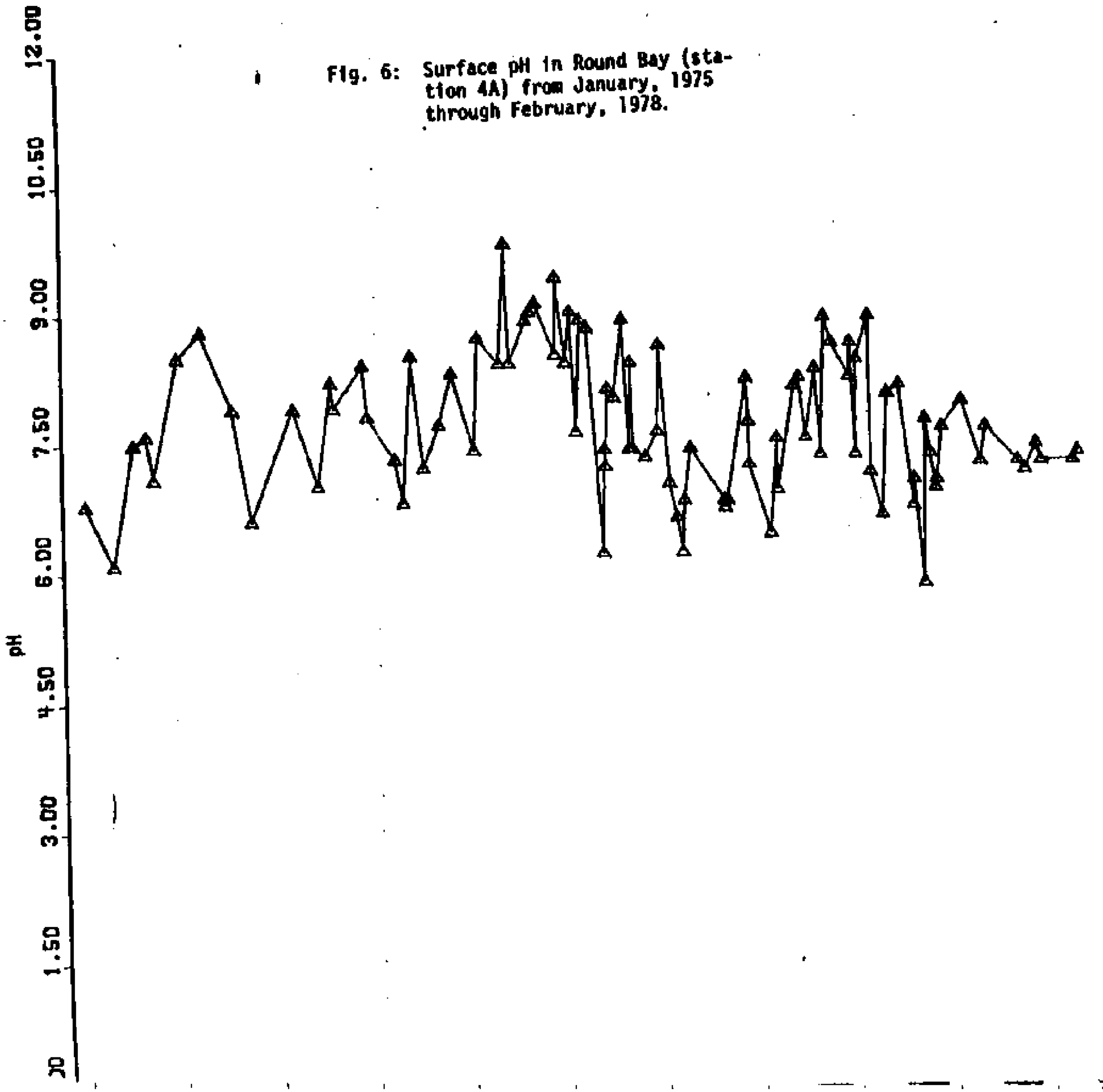


Fig. 6: Surface pH in Round Bay (station 4A) from January, 1975 through February, 1978.

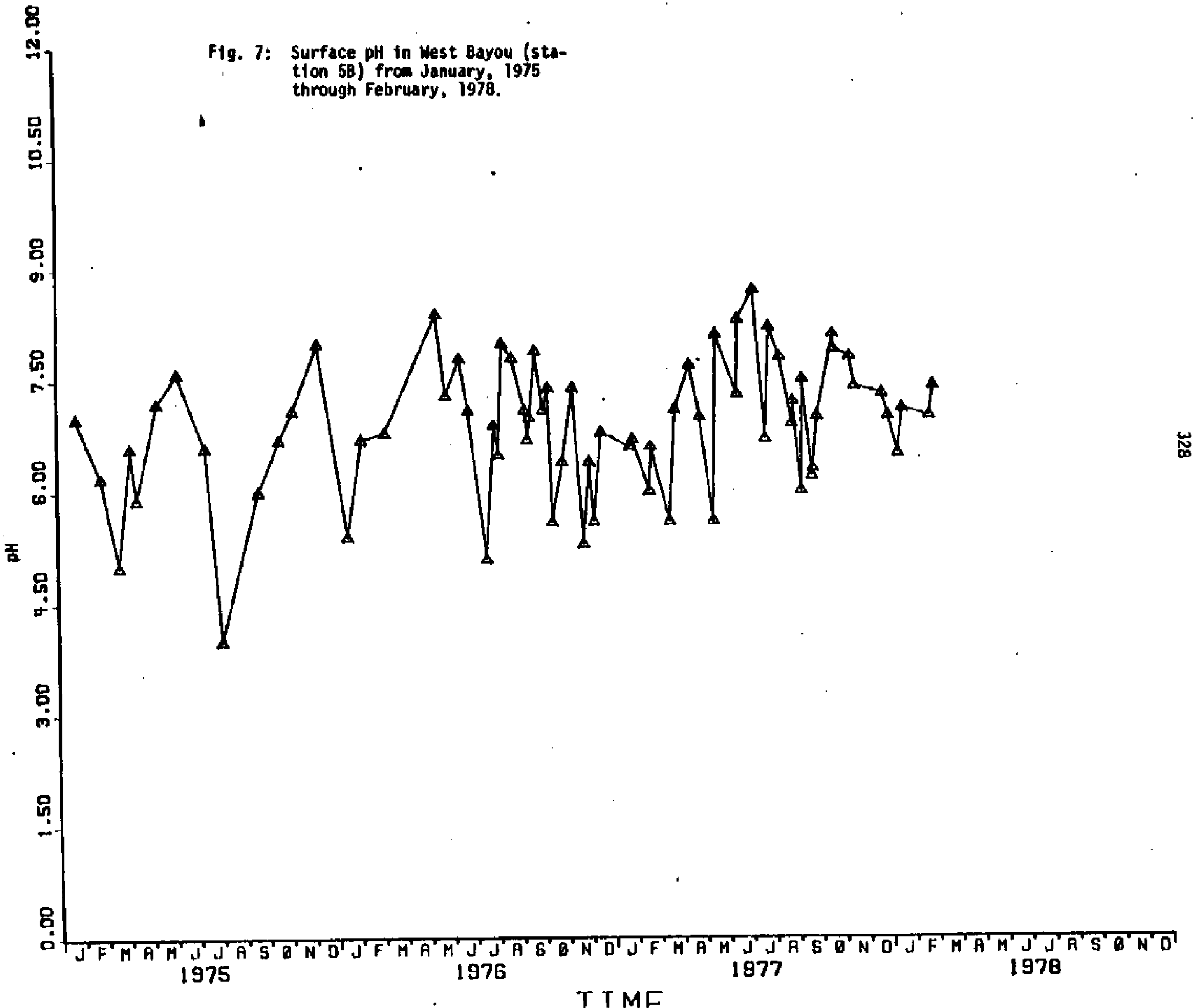


Fig. 7: Surface pH in West Bayou (station 5B) from January, 1975 through February, 1978.

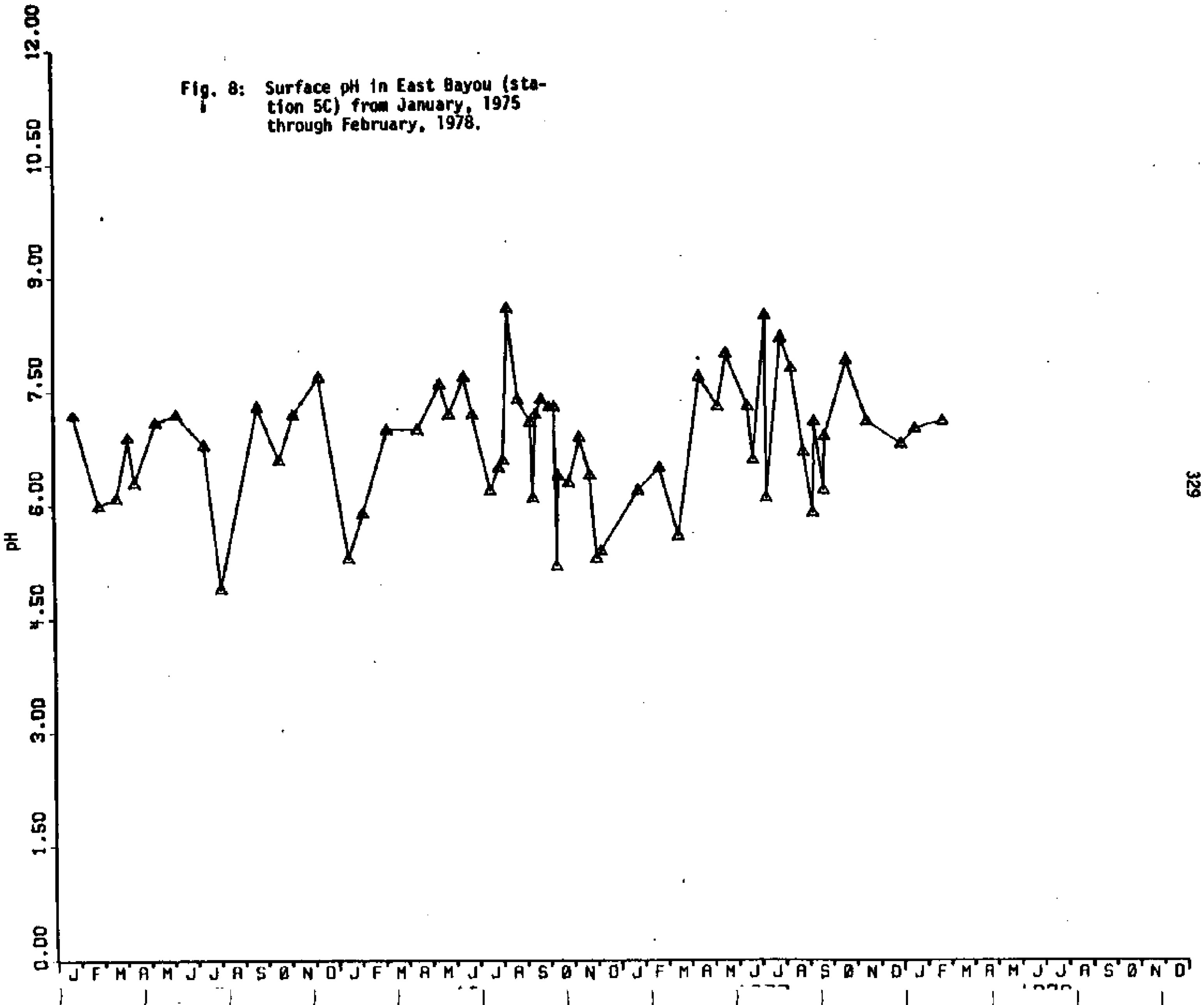
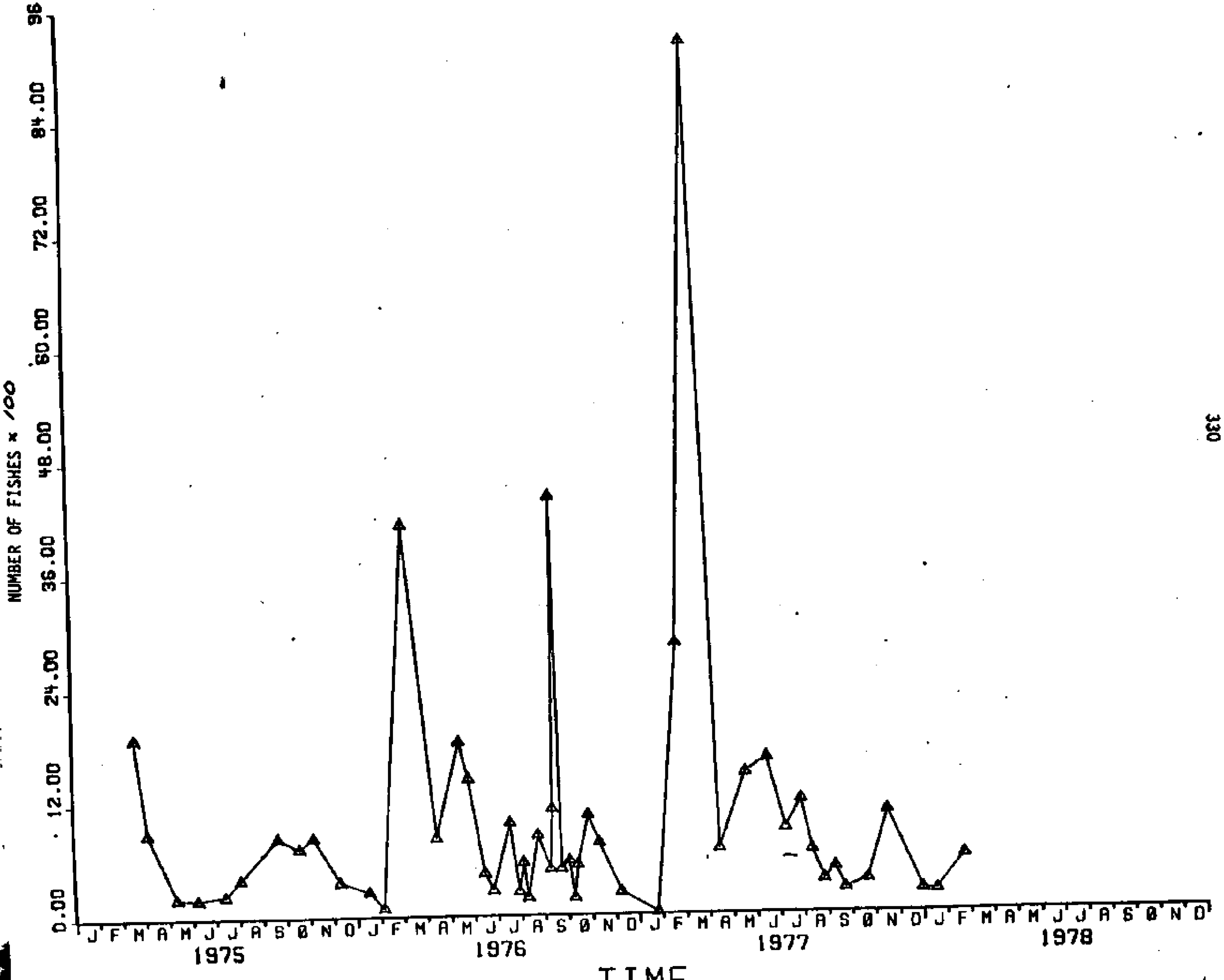


Fig. 8: Surface pH in East Bayou (station 5C) from January, 1975 through February, 1978.

Fig. 9: Total number of fishes taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



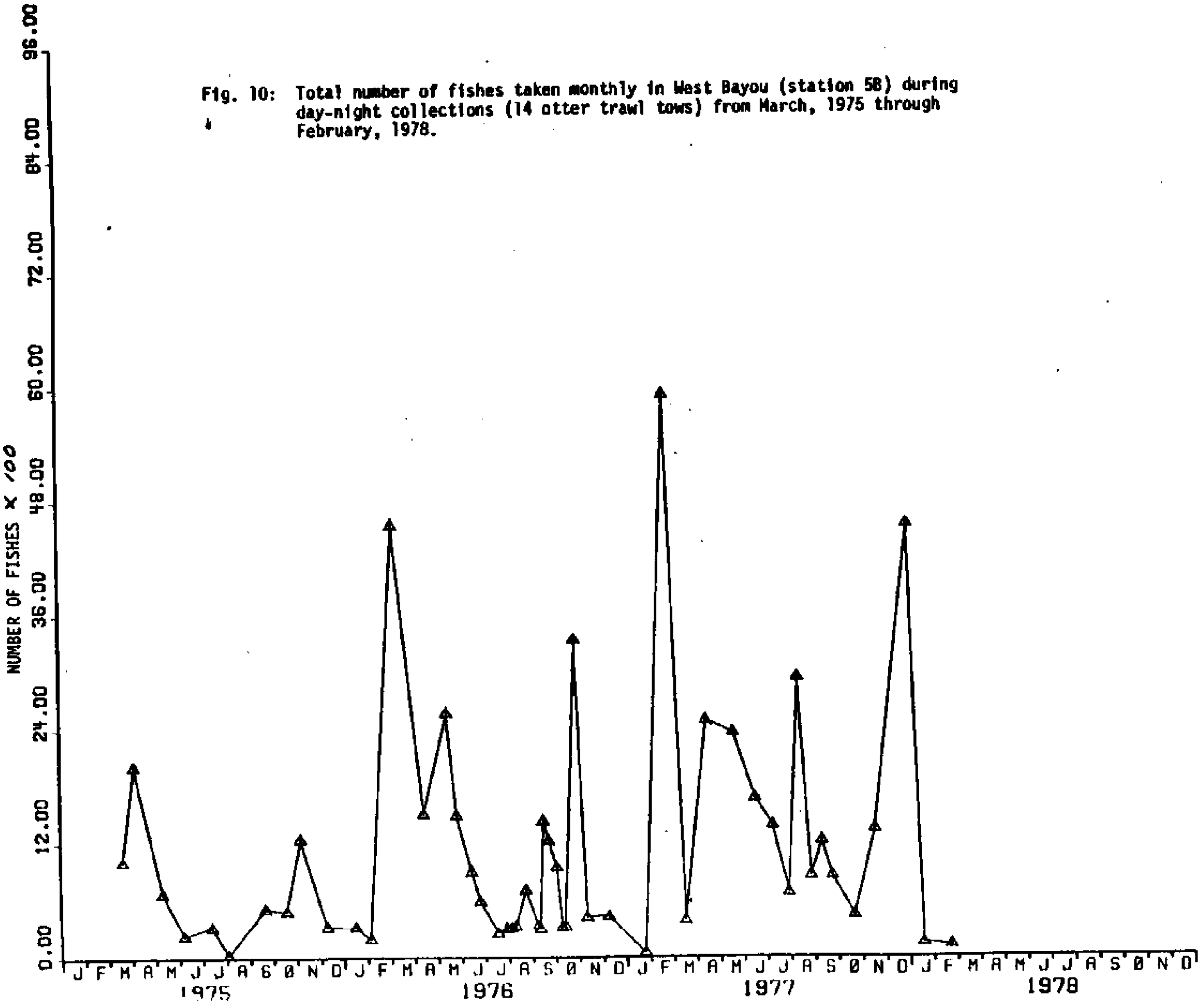


Fig. 10: Total number of fishes taken monthly in West Bayou (station 5B) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

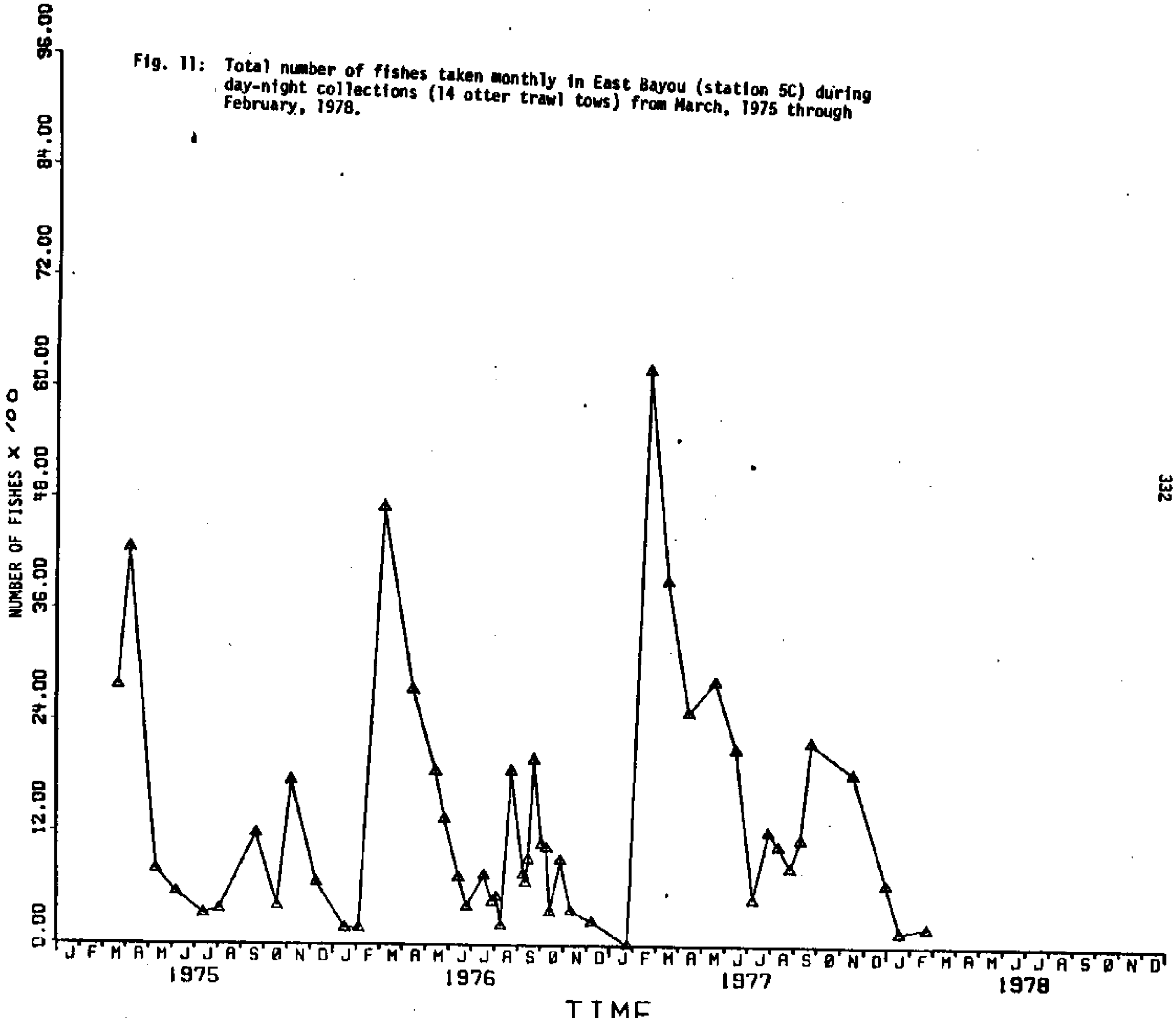


Fig. 11: Total number of fishes taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

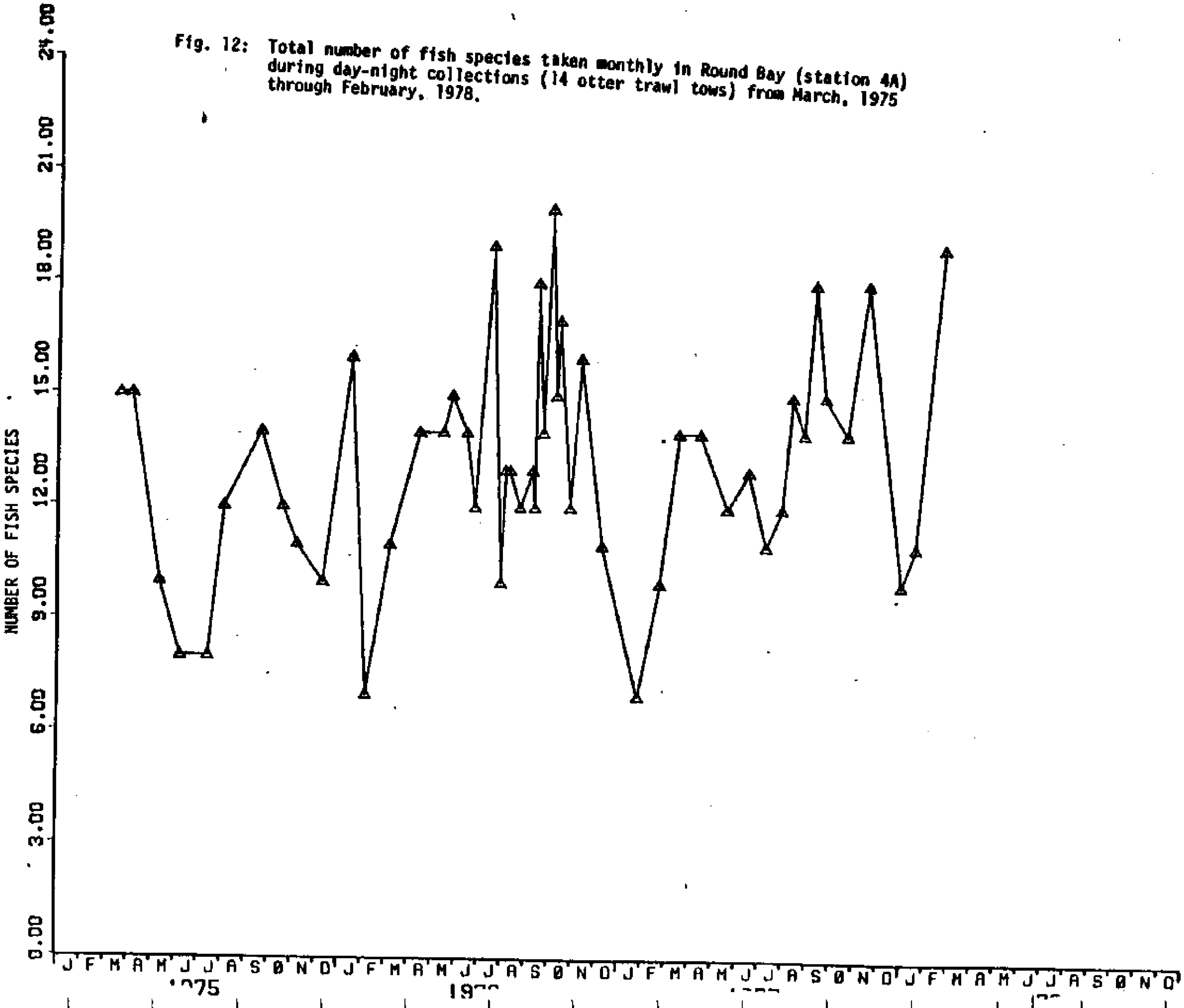
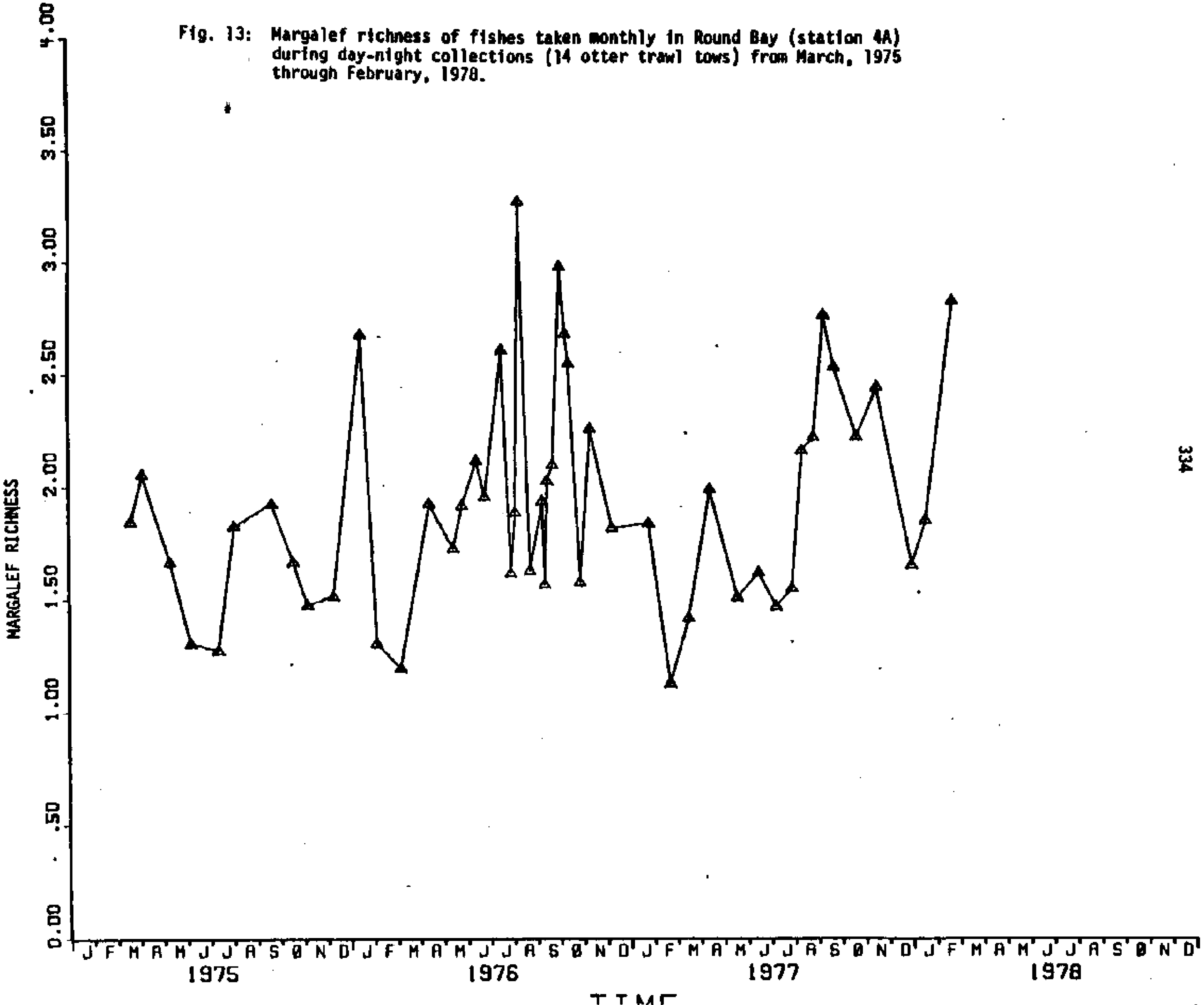


Fig. 12: Total number of fish species taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

Fig. 13: Margalef richness of fishes taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



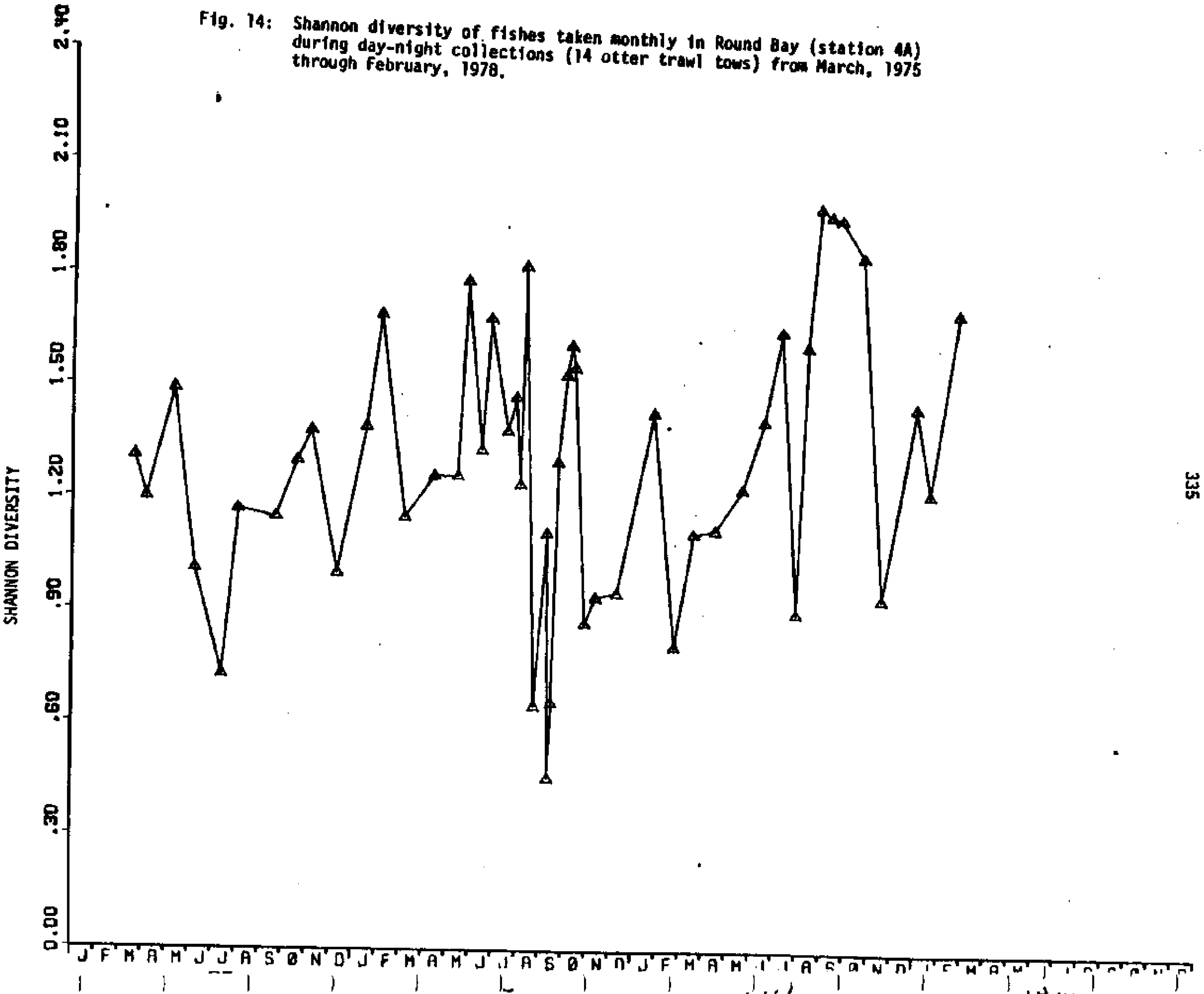
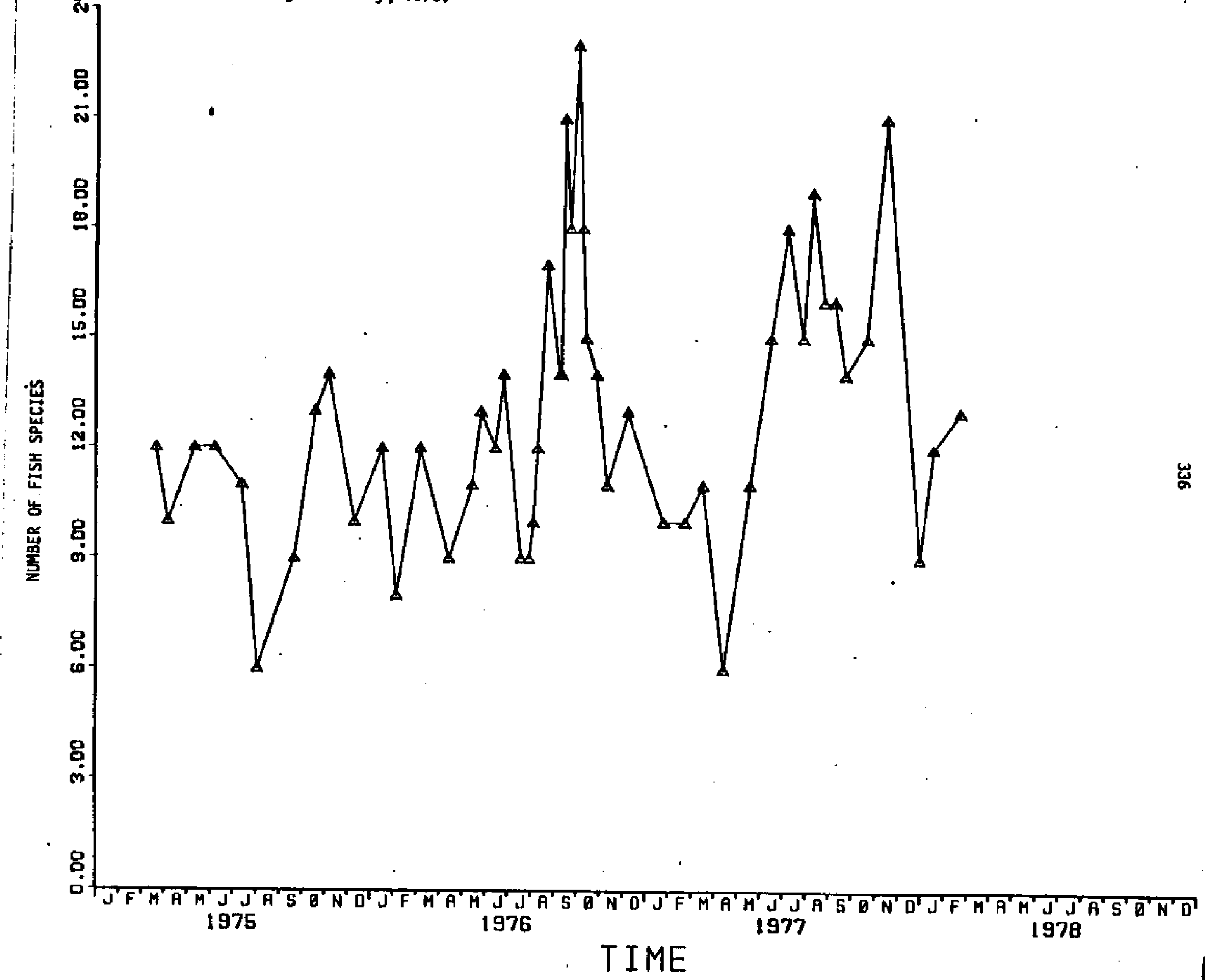


Fig. 14: Shannon diversity of fishes taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

Fig. 15: Total number of fish species taken monthly in West Bayou (station 5B) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



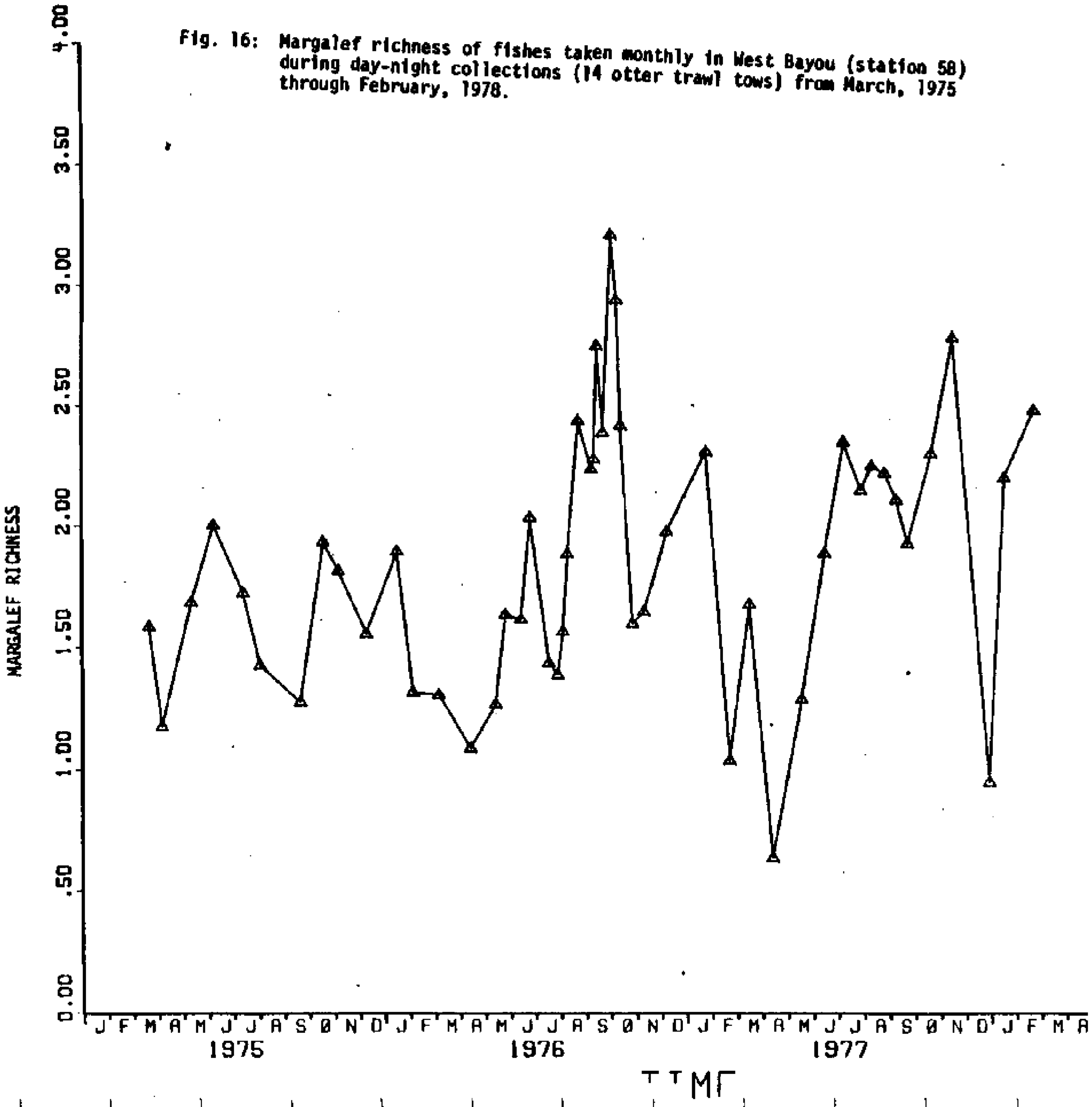


Fig. 16: Margalef richness of fishes taken monthly in West Bayou (station 58) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

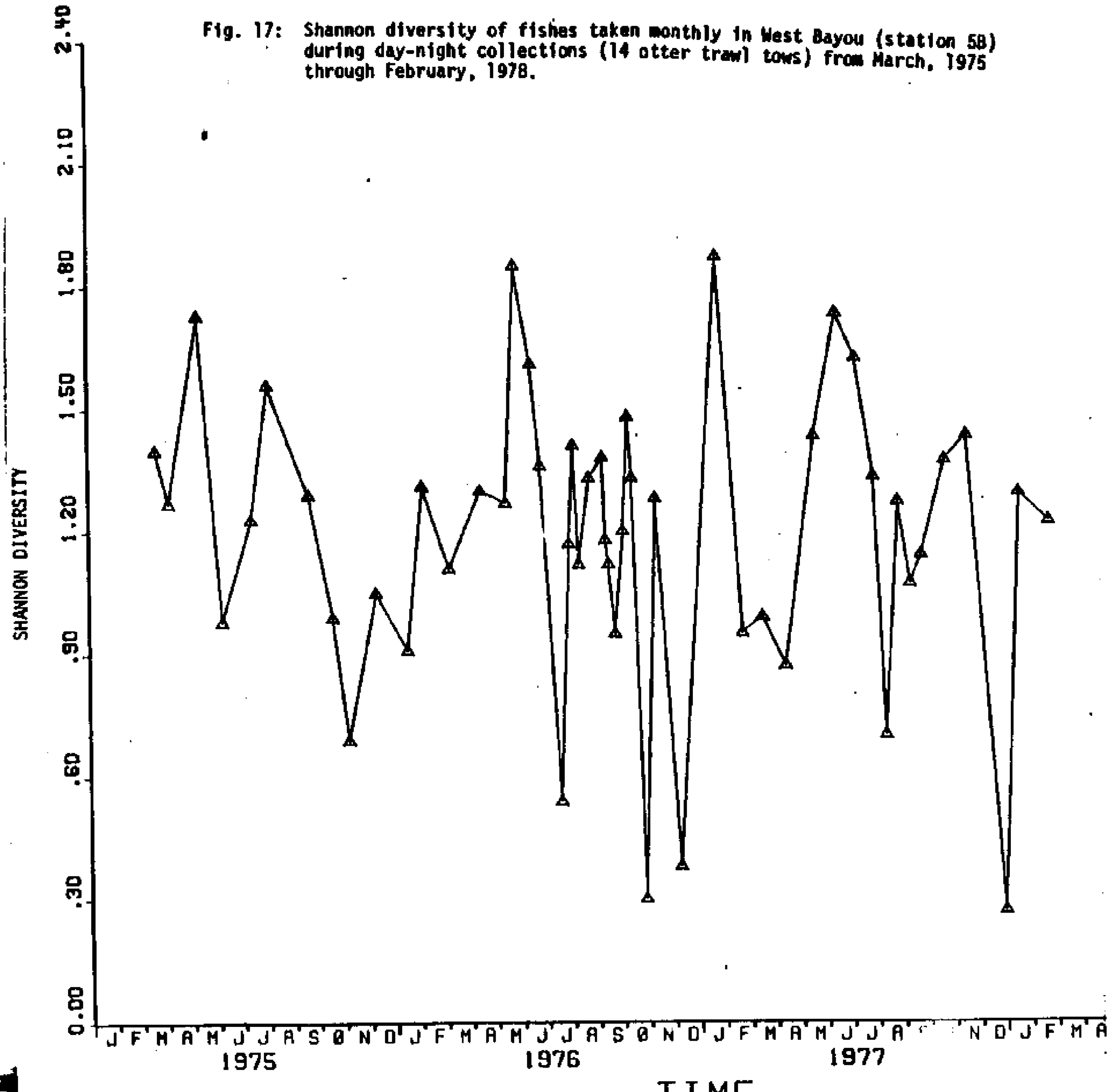


Fig. 17: Shannon diversity of fishes taken monthly in West Bayou (station 5B) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

Fig. 18: Total number of fish species taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

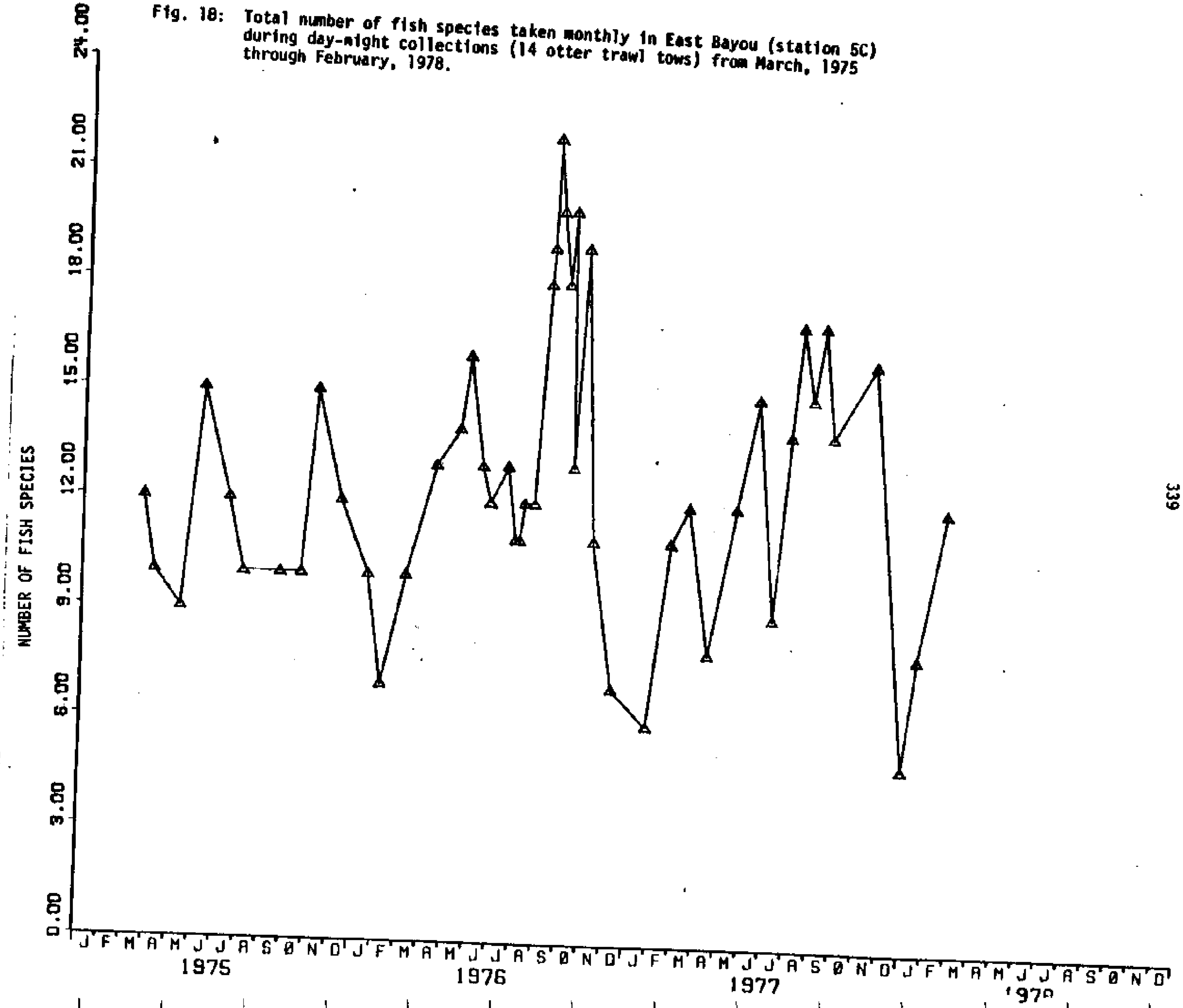
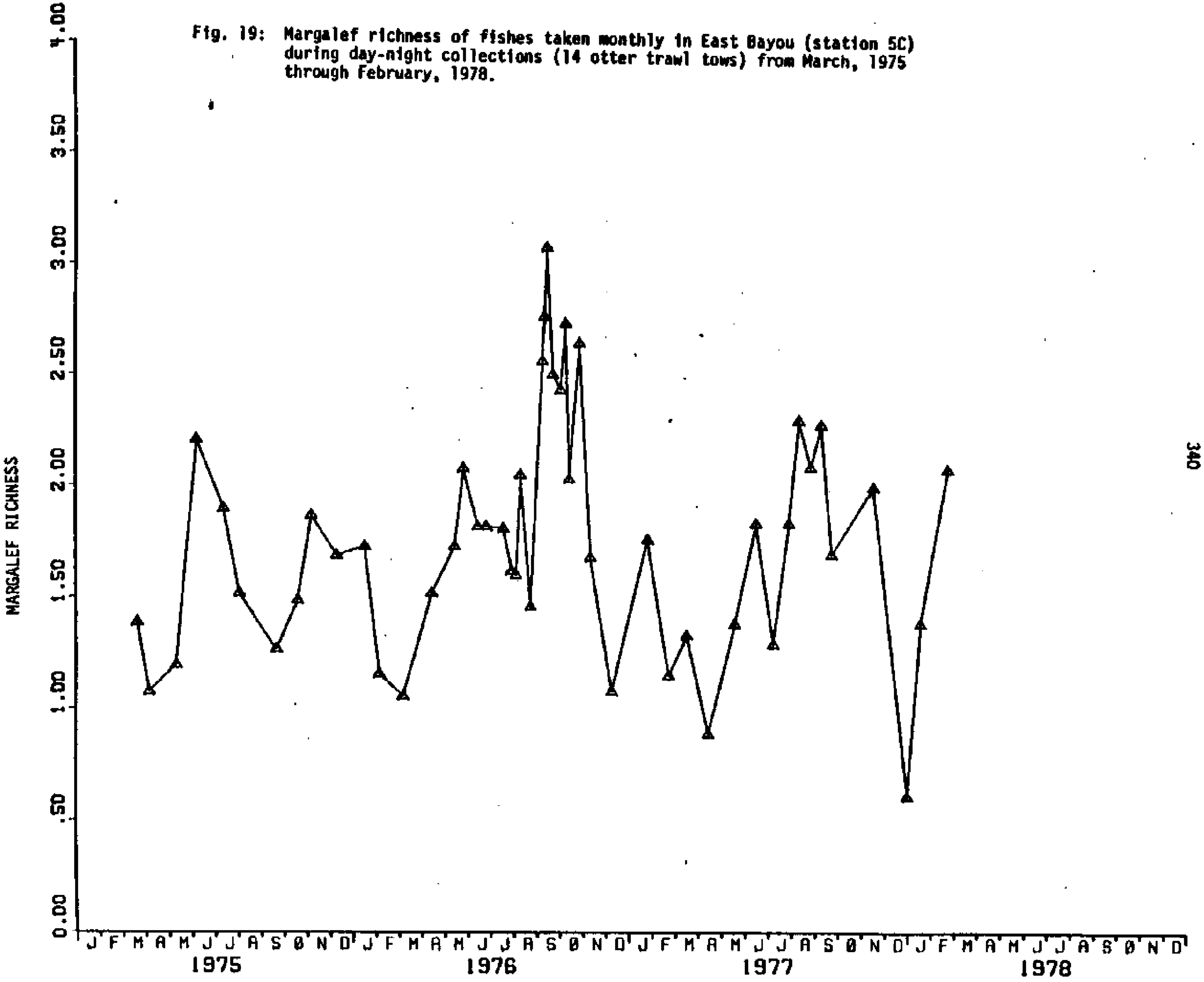


Fig. 19: Margalef richness of fishes taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



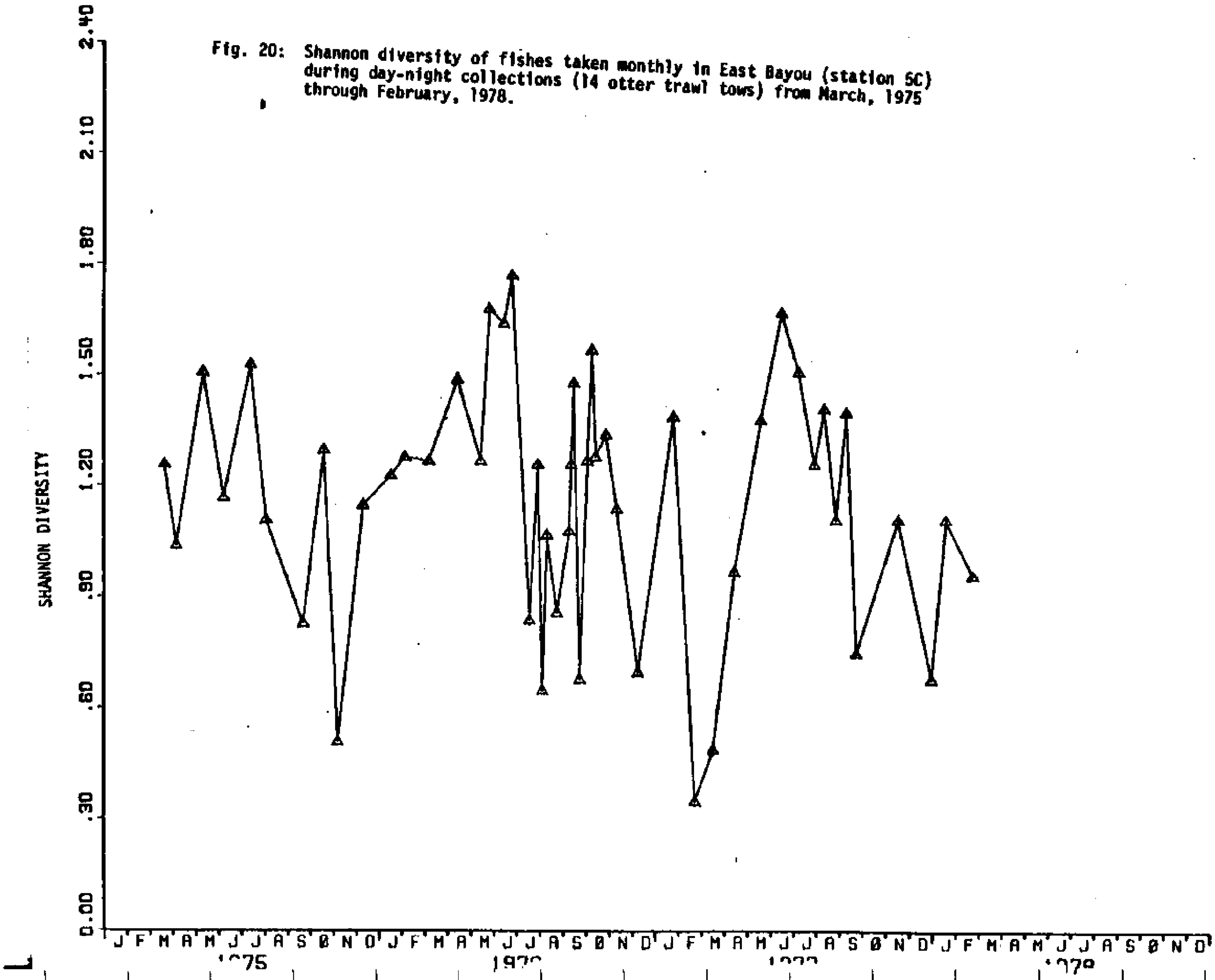


Fig. 20: Shannon diversity of fishes taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

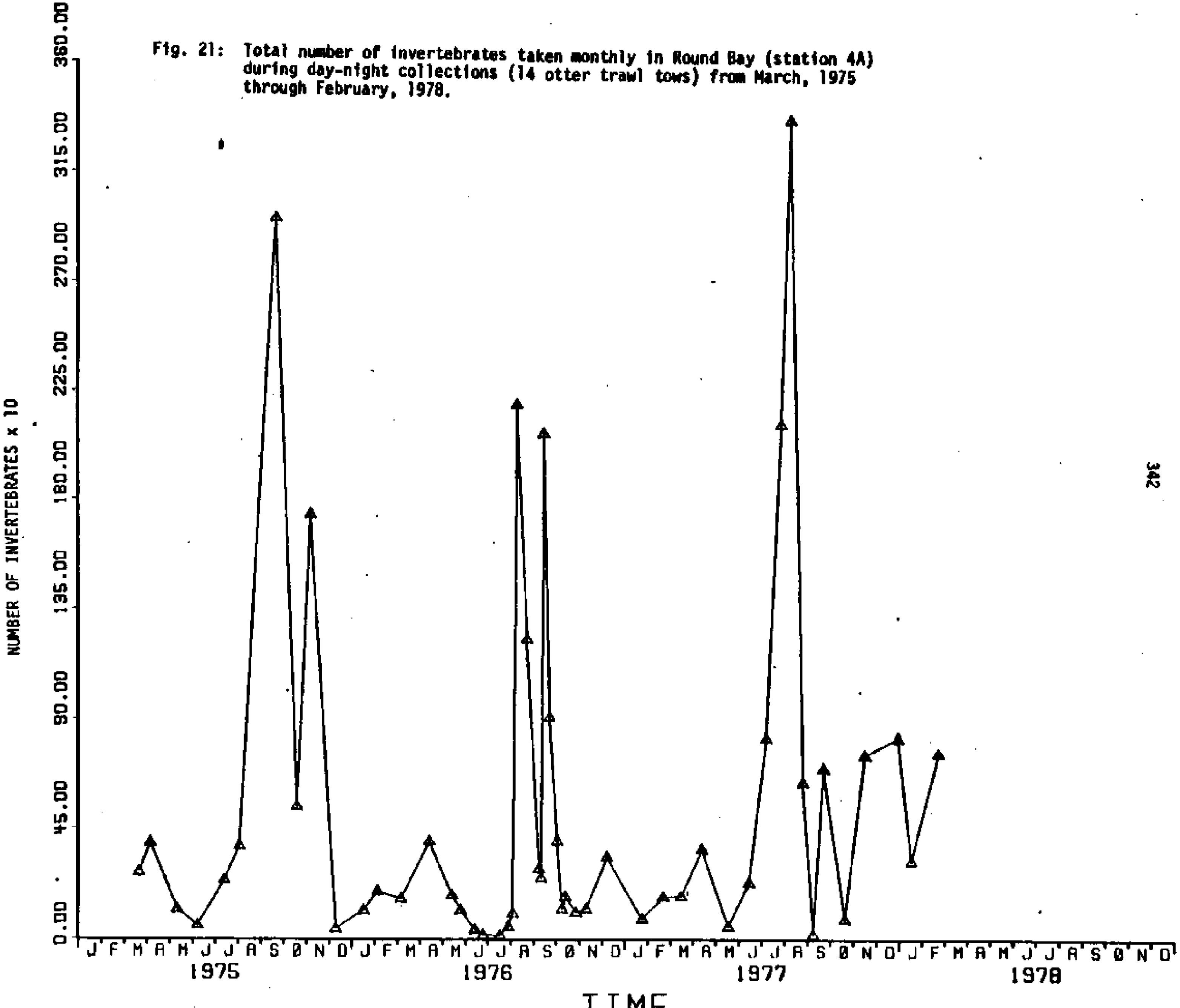


Fig. 21: Total number of invertebrates taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

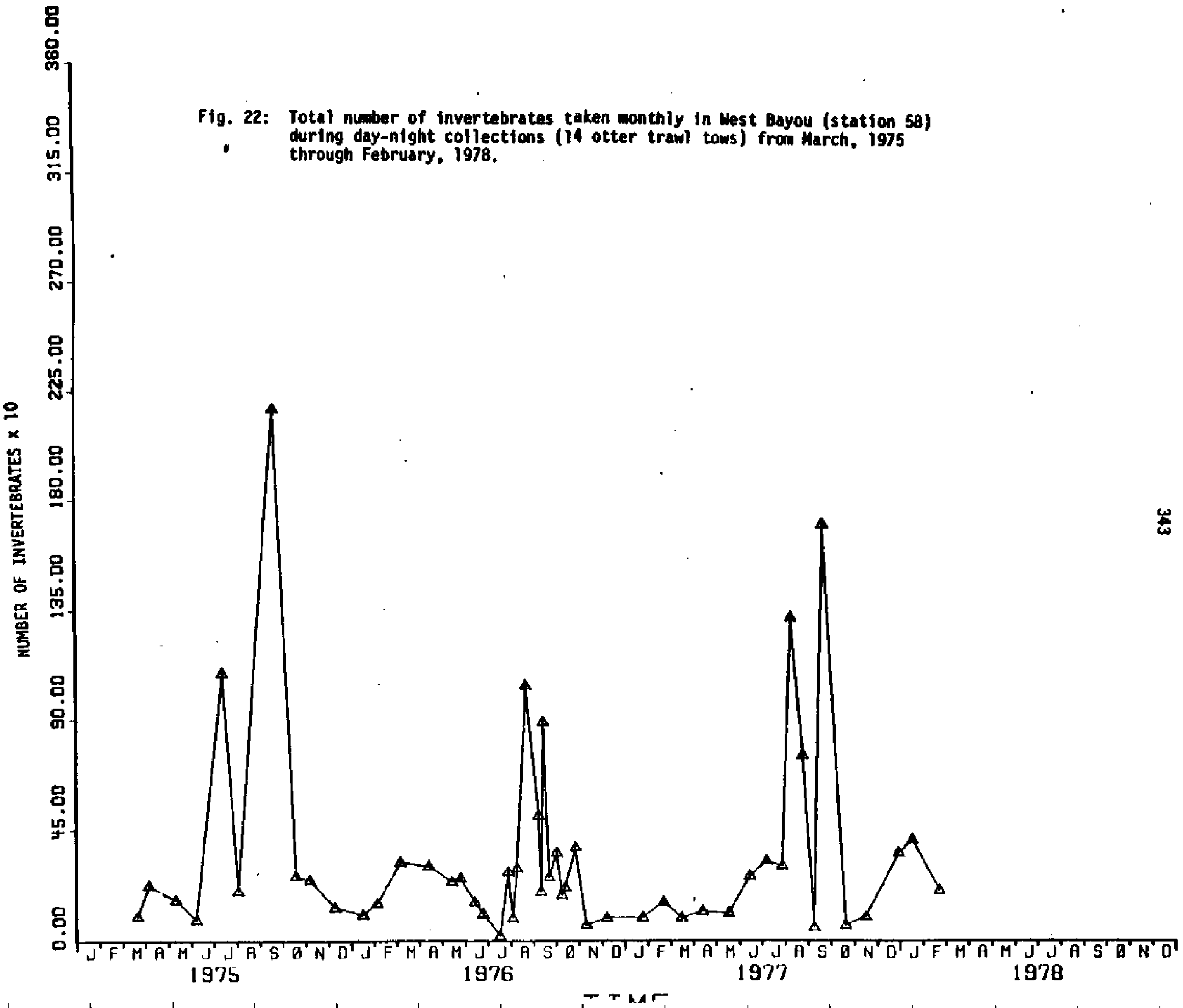


Fig. 22: Total number of invertebrates taken monthly in West Bayou (station 58) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

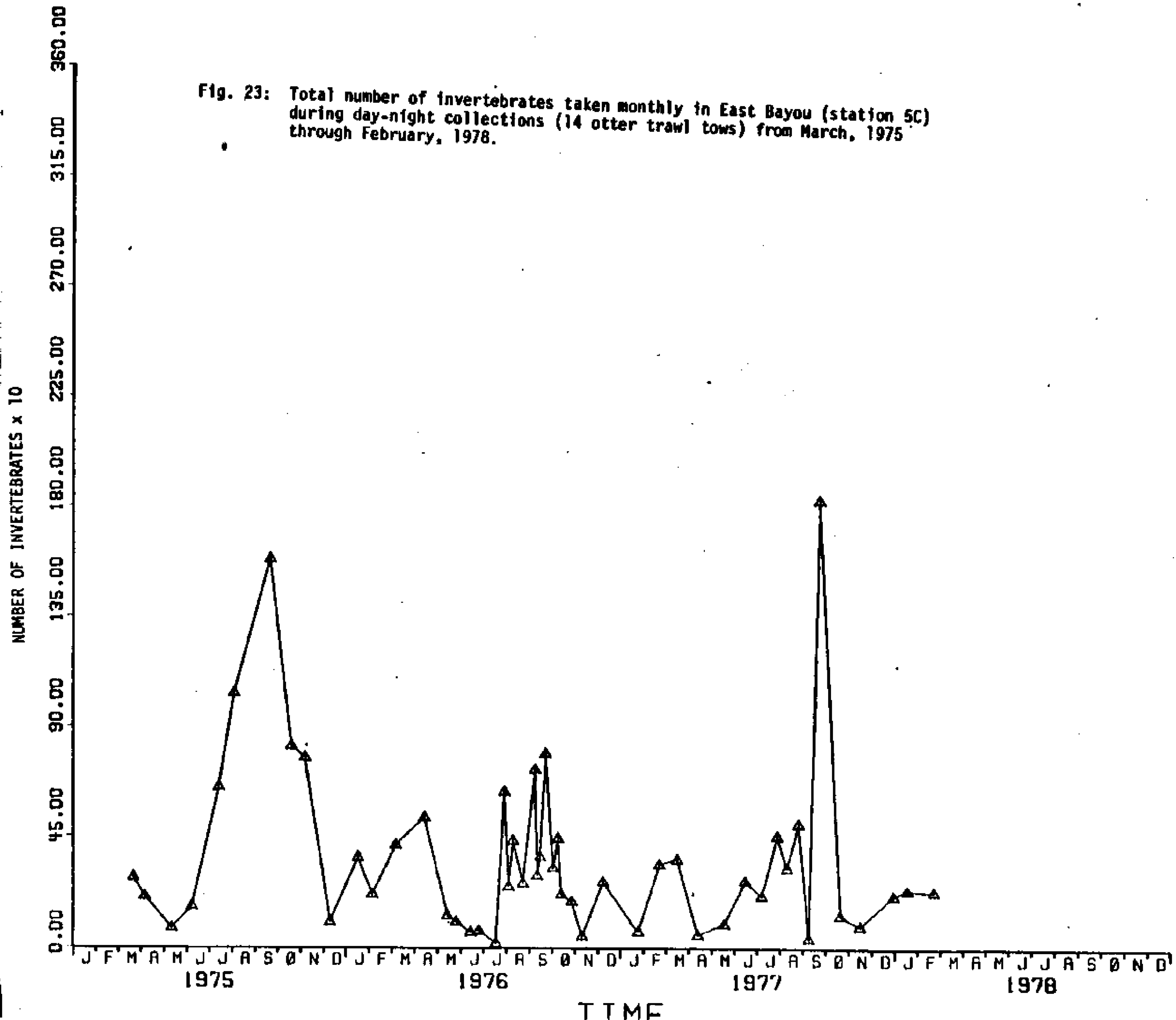


Fig. 23: Total number of invertebrates taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

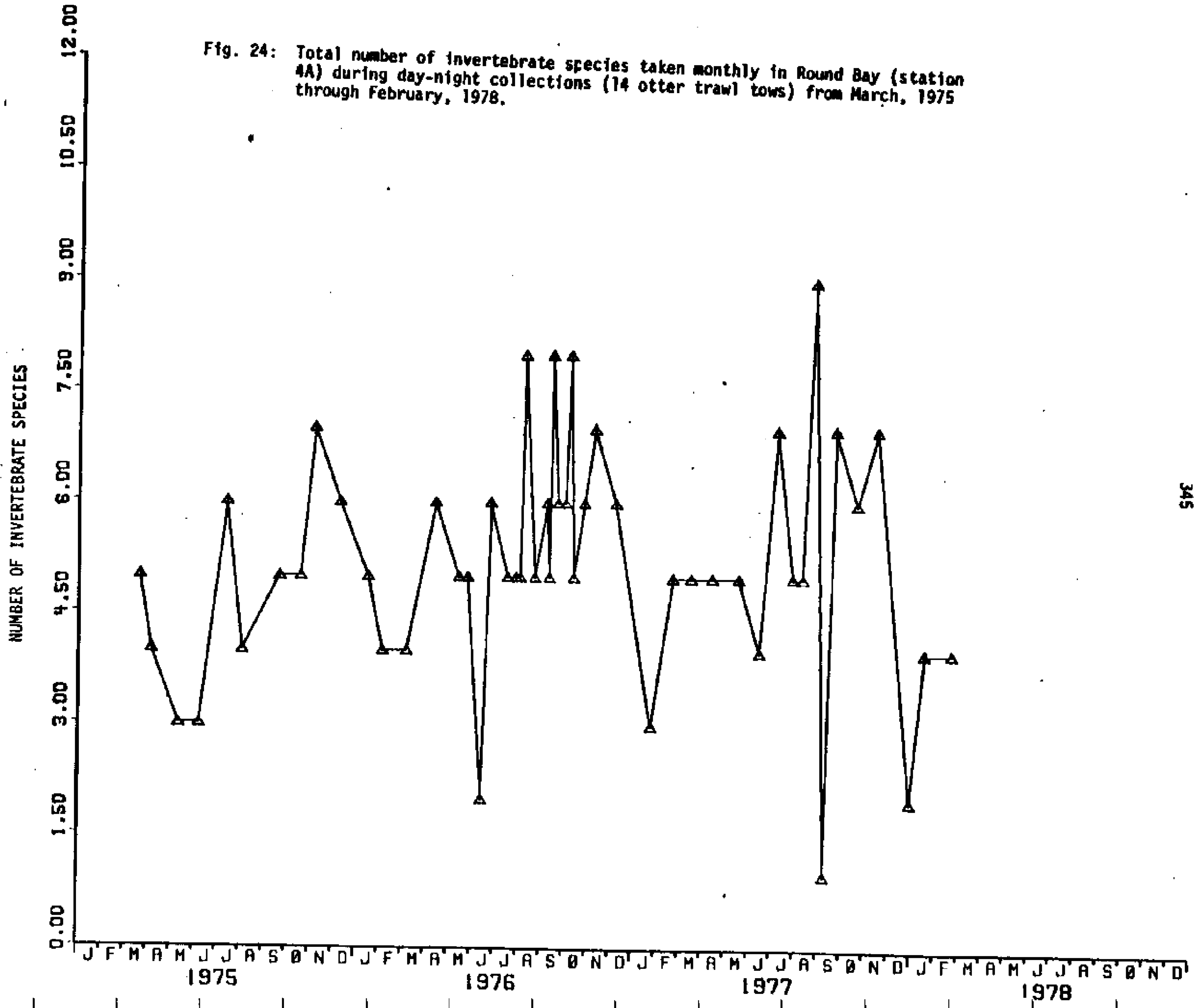
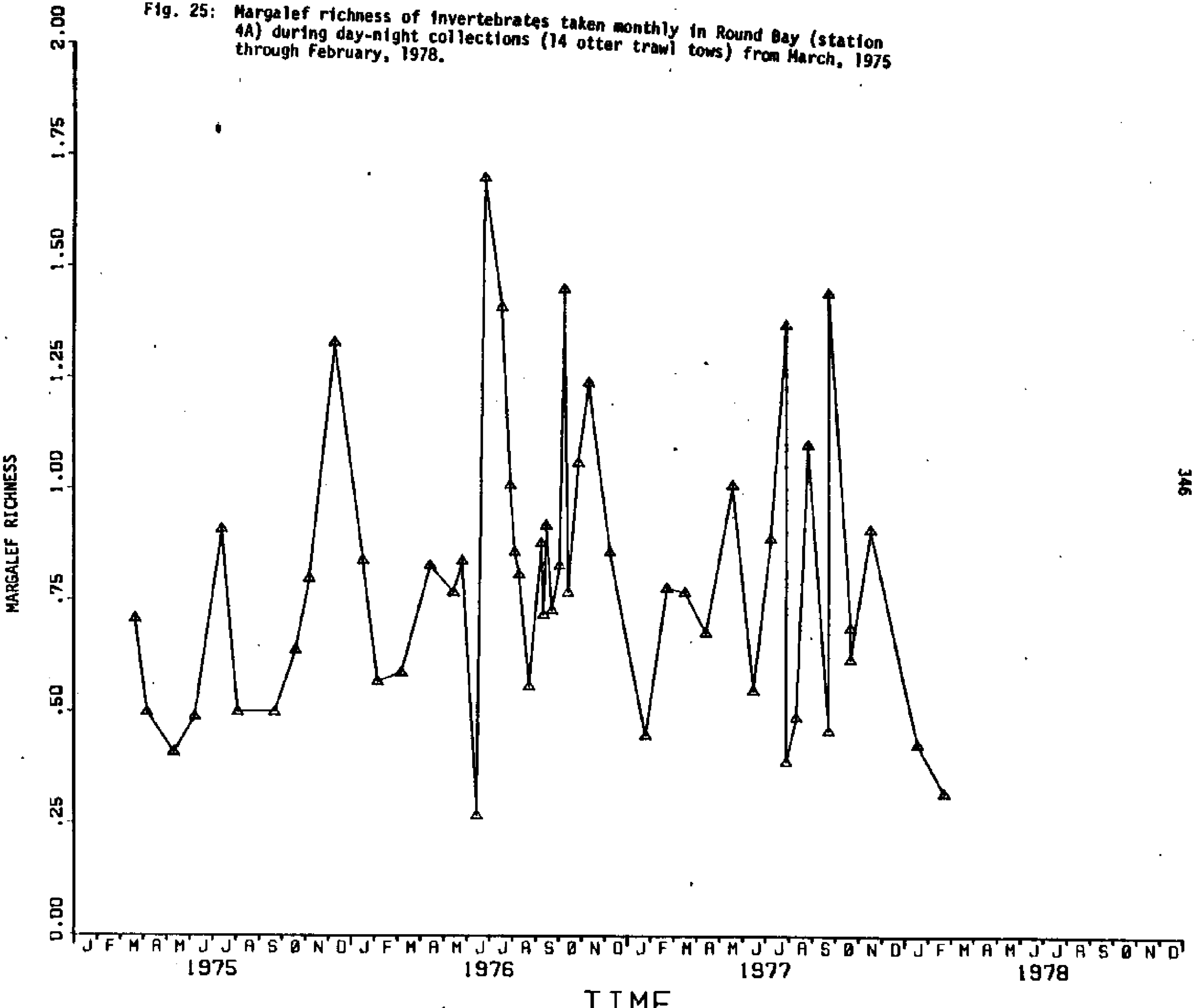


Fig. 24: Total number of invertebrate species taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

Fig. 25: Margalef richness of invertebrates taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



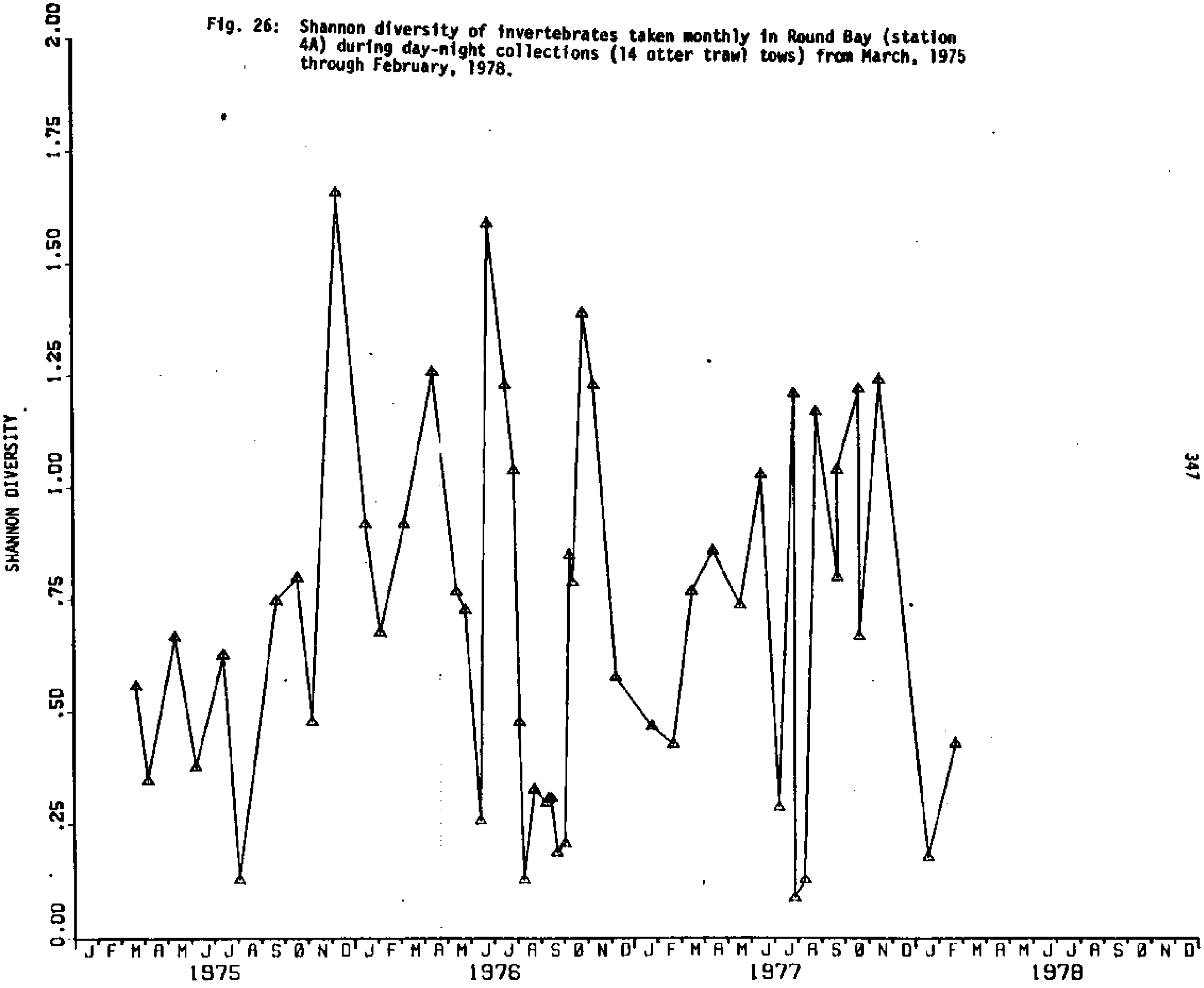


Fig. 26: Shannon diversity of invertebrates taken monthly in Round Bay (station 4A) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

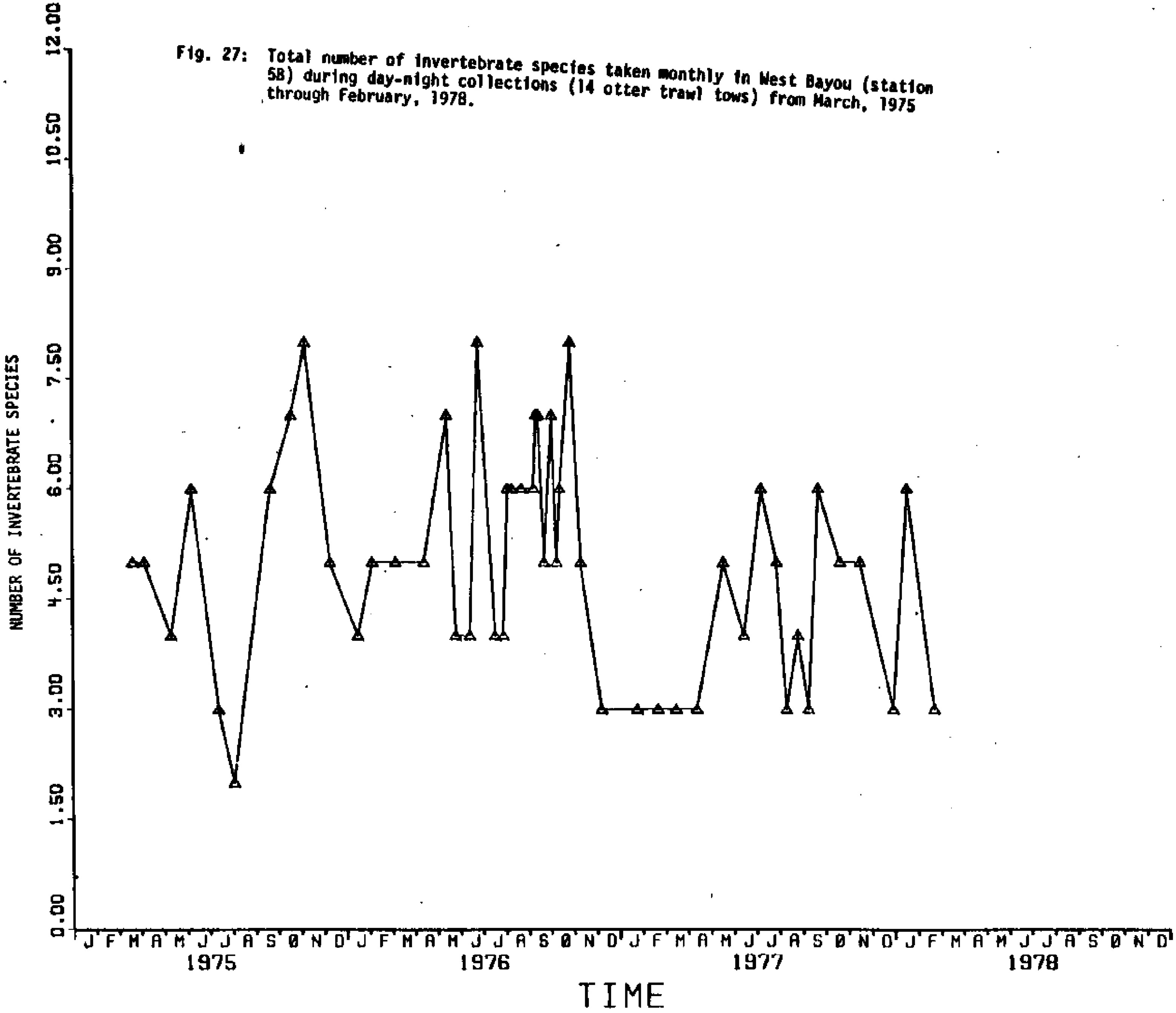


Fig. 27: Total number of invertebrate species taken monthly in West Bayou (station 5B) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

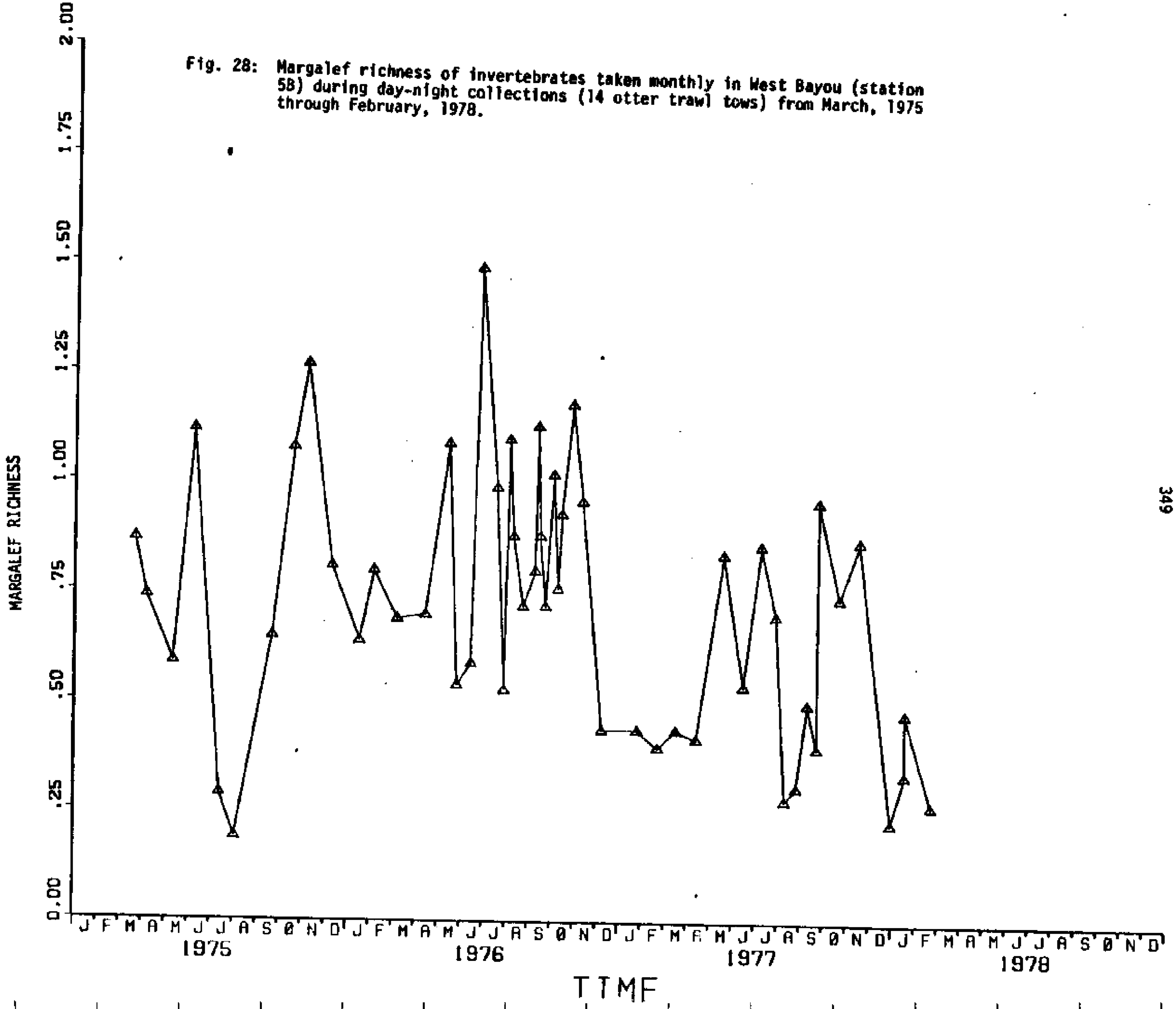
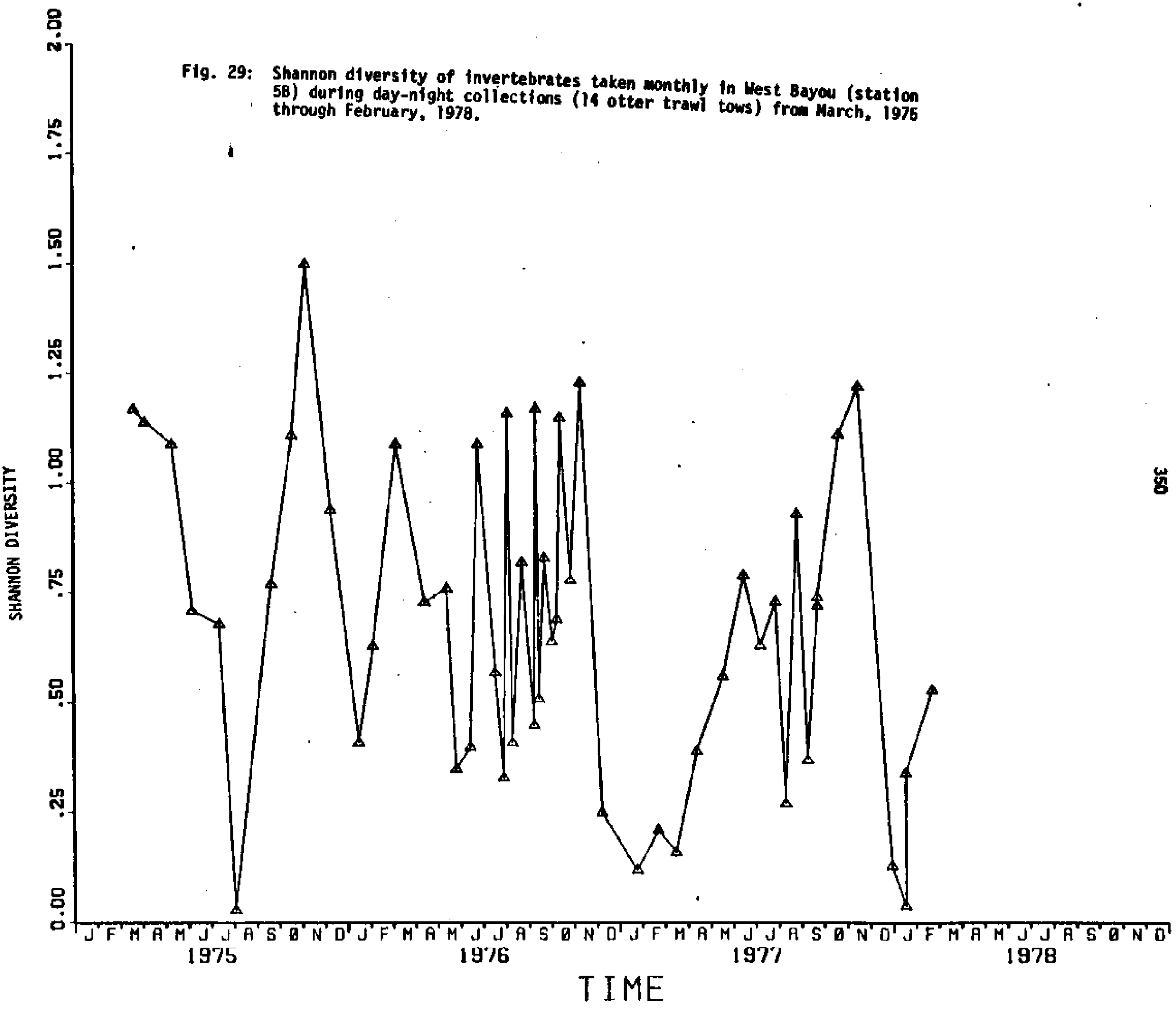


Fig. 28: Margalef richness of invertebrates taken monthly in West Bayou (station 58) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

Fig. 29: Shannon diversity of invertebrates taken monthly in West Bayou (station 5B) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.



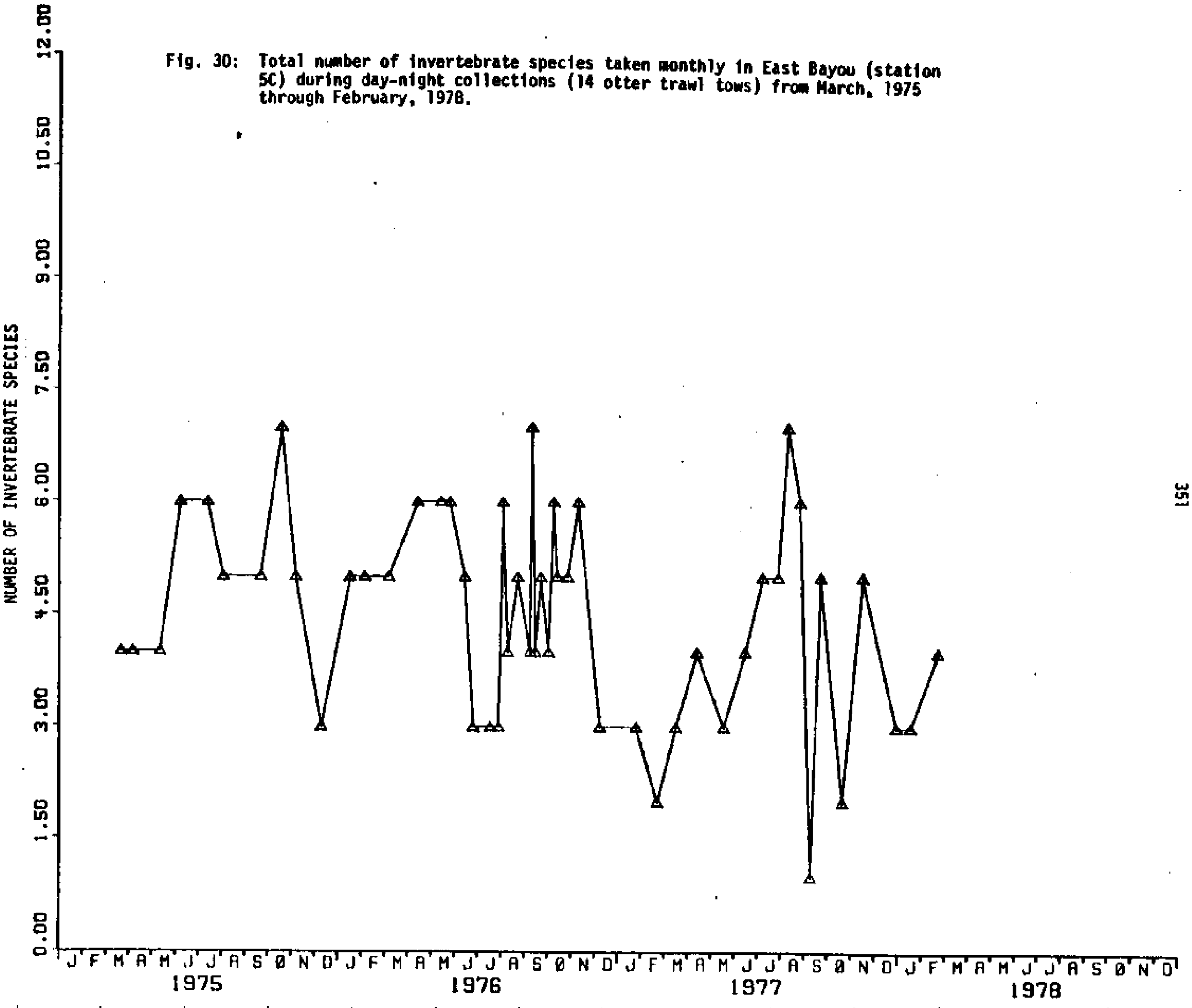


Fig. 30: Total number of invertebrate species taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

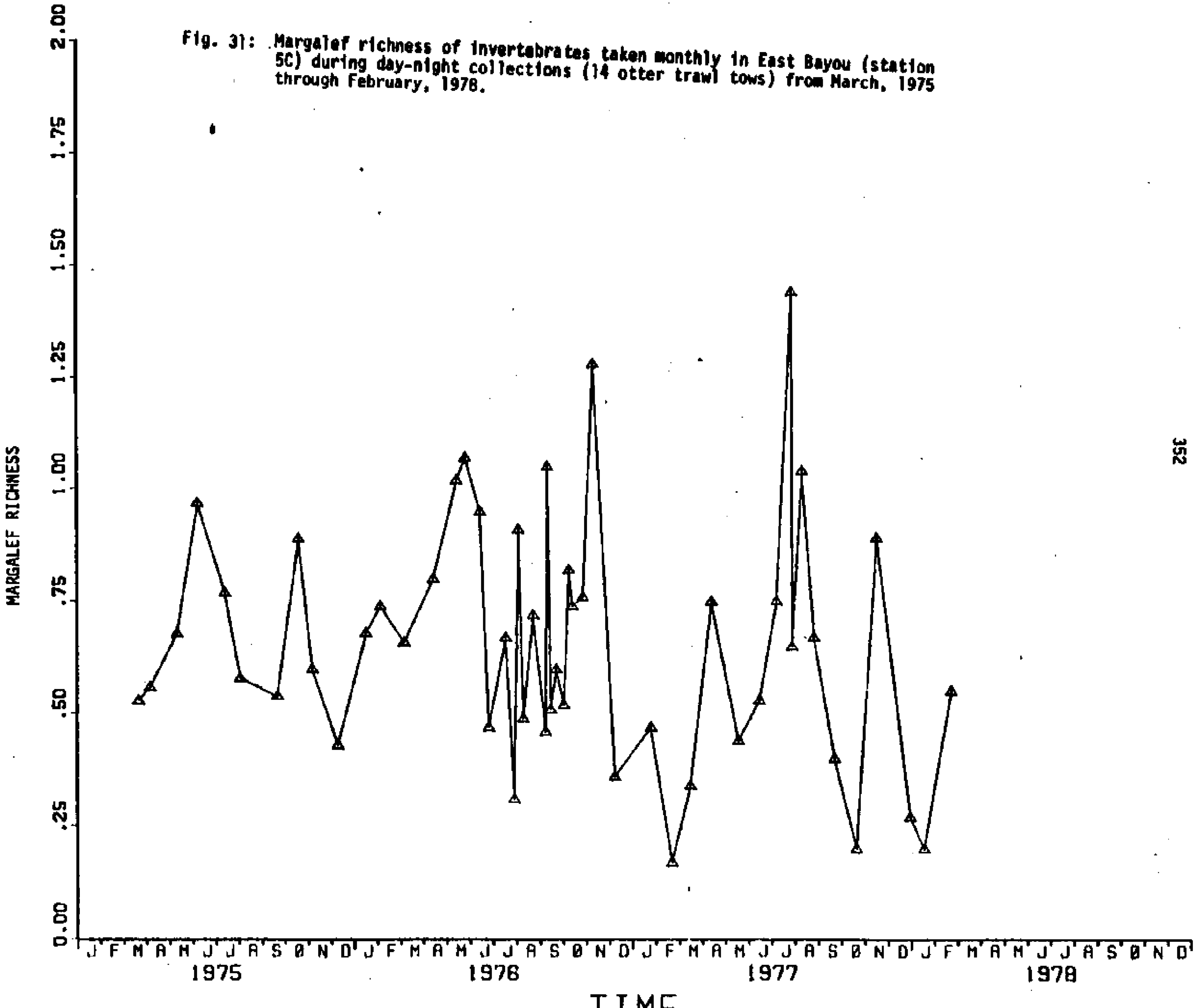


Fig. 31: Margalef richness of invertebrates taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978.

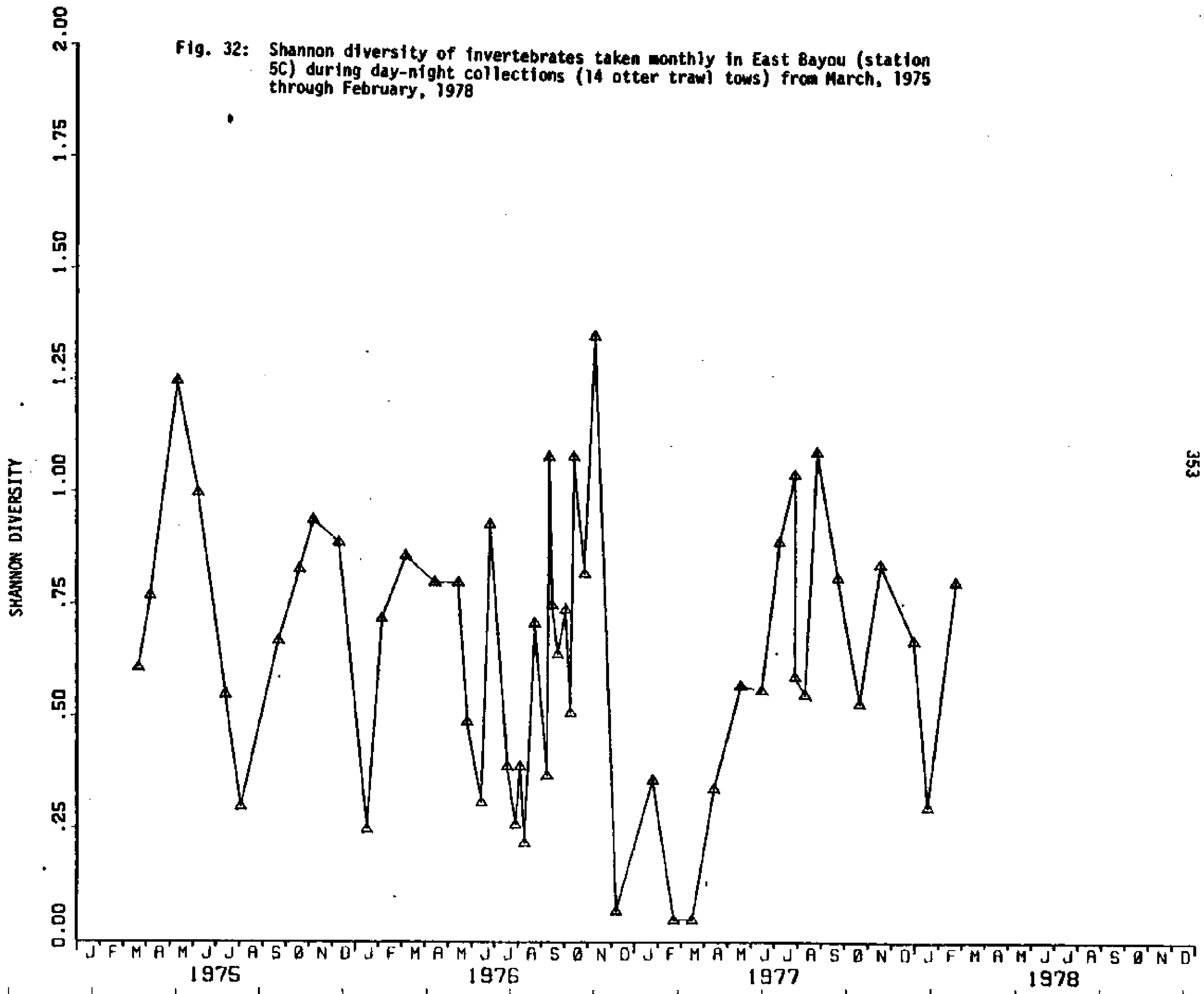
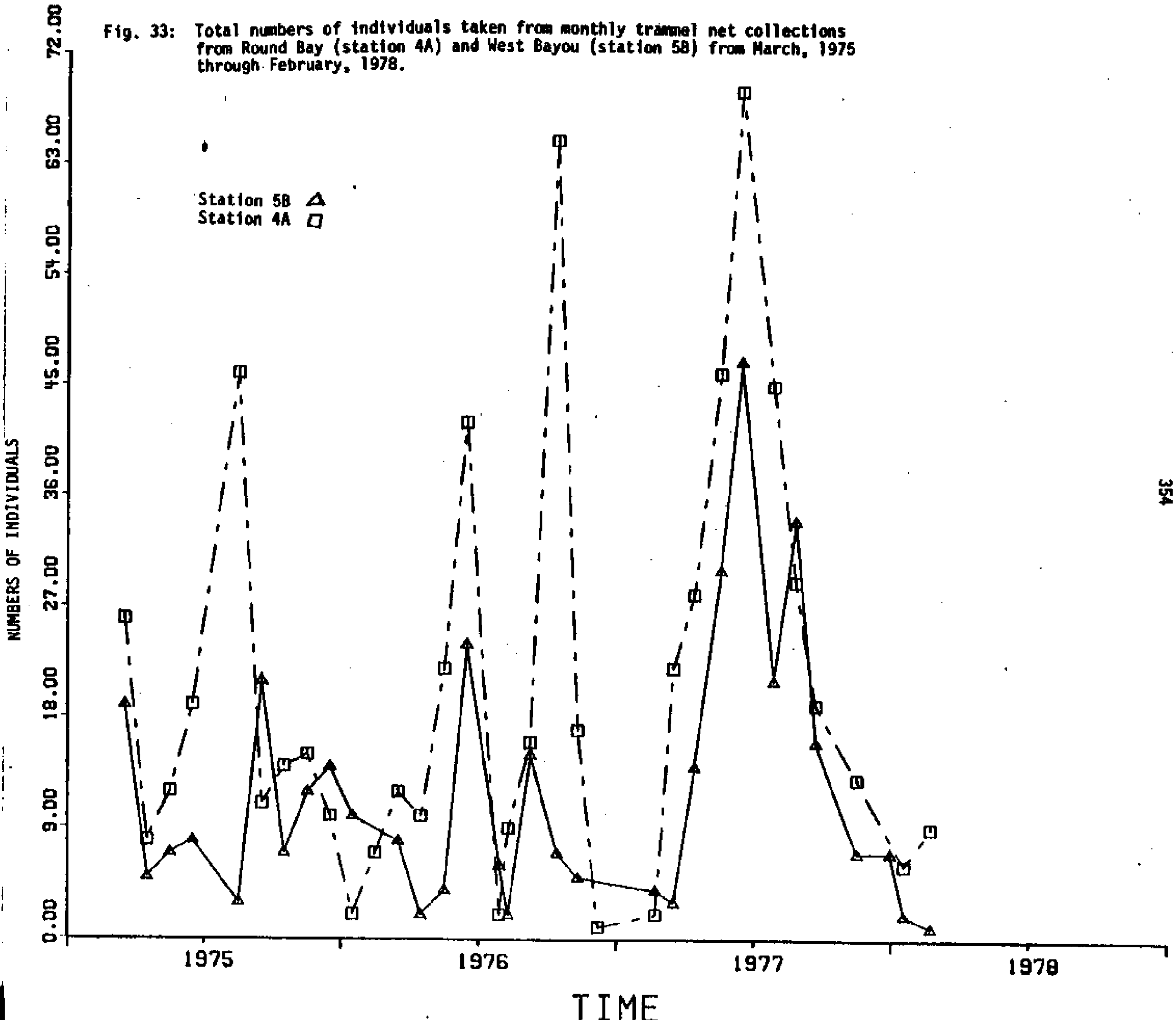


Fig. 32: Shannon diversity of invertebrates taken monthly in East Bayou (station 5C) during day-night collections (14 otter trawl tows) from March, 1975 through February, 1978

Fig. 33: Total numbers of individuals taken from monthly trammel net collections from Round Bay (station 4A) and West Bayou (station 5B) from March, 1975 through February, 1978.



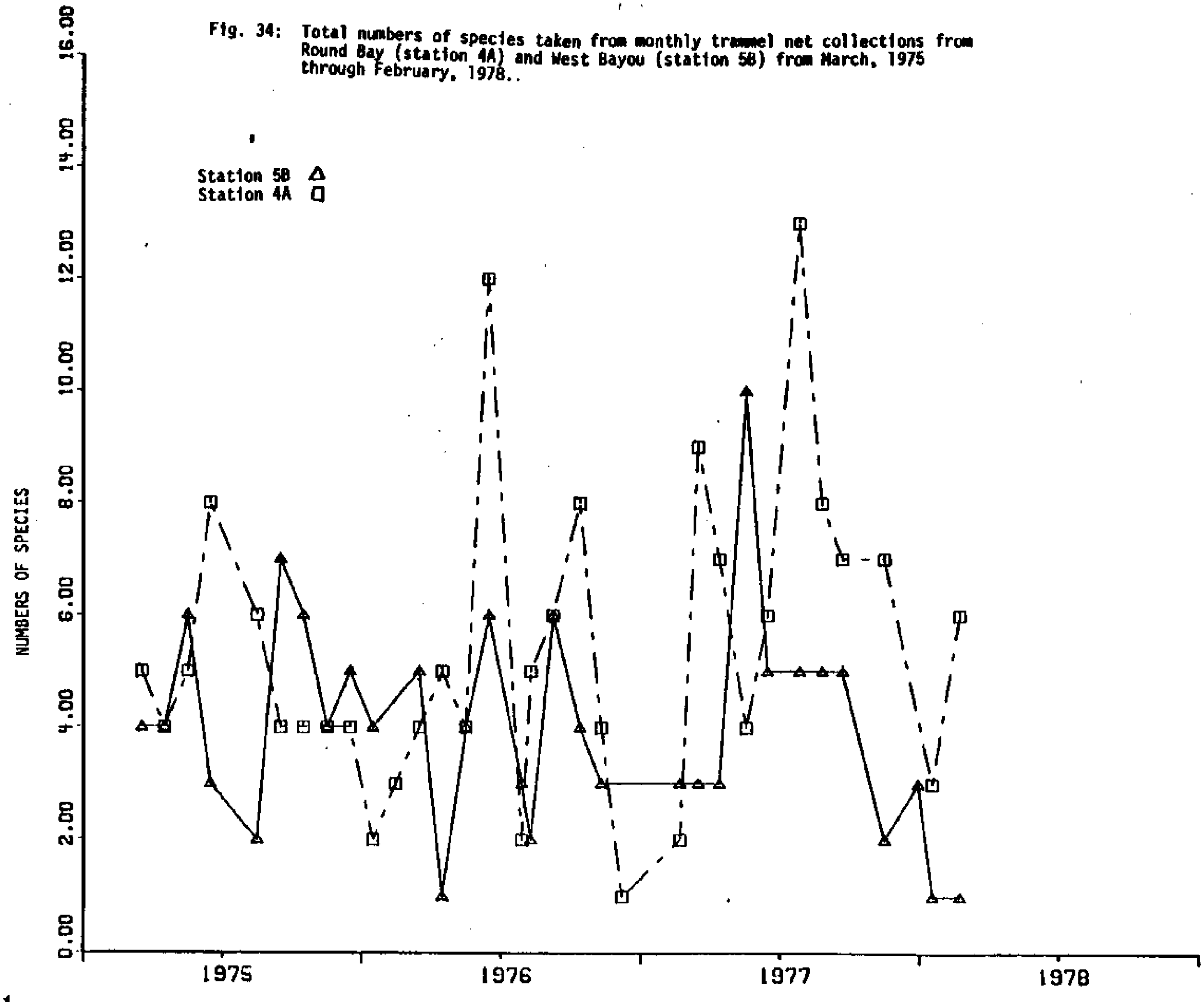


Fig. 34: Total numbers of species taken from monthly trammel net collections from Round Bay (station 4A) and West Bayou (station 5B) from March, 1975 through February, 1978..

IX. Long-term Changes in Epibenthic Fishes and Invertebrates

Introduction

The long-term physico-chemical trends include peak river flow and rainfall from 1973 to 1975. From 1976 to 1978, there were sharp decreases in both functions with drought conditions prevailing during the last year of sampling (1977-78). This was viewed as part of a long-term trend of 5-8 year cycles of river flow and local rainfall. Water quality parameters which reflect upland runoff (salinity, pH, color, turbidity) would be expected to reflect such meteorological trends and this indeed is the case (see Chapter III: Physico-chemical Relationships). East Bay showed reduced salinity from 1973 through 1976, with color peaking in 1975, and considerable decreases in pH in the upper East Bay drainage receiving effluents from cleared areas during the summer of 1975 (no data were available from 1972 to 1974). As a result, changes in the biota in eastern portions of East Bay should be analyzed to determine if there was any change in the patterns (relative to other portions of East Bay and Apalachicola Bay) during the period from 1974 to 1975. It should be noted that, according to forestry records, the period of maximal forestry operations in Tate's Hell Swamp in areas contiguous with the upper East Bay area occurred during 1972-75.

The data for this section of the report include bay-wide collections of epibenthic fishes and invertebrates taken with 5-m otter trawls. Various statistical relationships have been delineated between the biological components of the Apalachicola estuary and key physico-chemical controlling functions (Livingston et al., 1978; Meeter and Livingston, 1978). Using various multifactorial statistical tests, river flow has been directly associated with various ecological factors such as turbidity, color, salinity, light transmisibility, nutrients, and detritus. Also associated with these factors are parameters such as phytoplankton productivity, fish and invertebrate distribution, and temporal sucessions (seasonal) of distinct biotic assemblages (Livingston et al., 1977). Various community indices such as relative dominance, species richness, and species diversity have been analyzed in addition to trophodynamic functions. Many such biological parameters appear to have direct or indirect associations with river flow fluctuations although the long-term implications of river-driven ecological parameters are still under study. Numbers of individuals and numbers of species of fishes generally peak during October-November periods while richness indices peak during periods of low river flow. There is efficient partitioning of food resources in the Apalachicola estuary, with dominant species participating in a trophic spectrum which is relatively well-ordered in space and time. Such biotic components are linked to seasonal successions of energy inputs which, in turn, are related to river fluctuations, rainfall patterns, phytoplankton blooms, benthic macrophyte die-offs, and the periodic movement of detritus through the system.

Long-term biotic changes

Summed otter trawl data for the entire bay system were used as a control for the hypothesized impact area in eastern portions of East Bay.

As shown in Fig. 1, river flow peaks (1973-75) coincided with general increases in various indices of the fish assemblages (species richness and diversity, biomass). Subsequent to 1975, there was a general decrease in such indices and an overall decrease in biomass which seemed to be related to the long-term trends in river flow. Although the peak temperatures tended to be relatively stable from year to year, the winter lows showed a decreasing trend from 1974 to the very cold winter of 1976-77. The effects of such temperature changes can be important to various biological functions, and the relatively high winter temperatures during 1974-75 could have influenced the fish indices to a certain degree. However, the general trends of the fish data did not follow the temperature trends in a direct manner. Biomass fell off during the period of high winter temperatures. Other indices showed downward trends prior to the winter lows of 1976-77; the relatively warmer winter of 1977-78 was not followed by upward trends of species richness and diversity. Obviously, more data are needed to make direct associations, and they are under analysis at this time. However, there is a growing indication that the fish assemblages are strongly affected by annual trends of river flow. This could be be a trophic phenomenon (Fig. 2); the amount of organic matter in the system appears to be directly affected by seasonal and annual variability of the river flow fluctuation. There has been a general decline in the particulate matter delivered to the bay by the river during the past 3 years as a direct function of river flow trends. This connection also requires further analysis, but it appears to be real and could be an important causative agent for the observed trends in the fish assemblages.

The invertebrate data show somewhat different trends (Fig. 3). The number of species and Margalef richness showed a steady increase through 1975 until stabilization (1975-76). Diversity tended to peak at this time while the numbers of individuals peaked during 1974 and tended to decline thereafter. The peak river flow (1972-73) and very low winter temperatures (1976-77) had very obvious and relatively long-lasting effects on various invertebrate indices, which tended to be depressed at such times. While recovery was apparent after the cold winter of 1976-77, a general decrease in these biological indices from 1976 to 1978 may also be in progress which would tend to parallel the fish data in this regard. However, the invertebrate data taken during daytime hours are not considered as reliable as the fish information (Livingston, 1976).

Long-term trends of Apalachicola River flow (annual totals), commercial catches of oysters (_Crassostrea_ _virginica_), shrimp (_Penaeus_ spp.), and blue crabs (_Callinectes_ _sapidus_) are shown in Fig. 4 (all catches are shown in 1000's of pounds: data from the Florida Department of Natural Resources). The various problems associated with the use of commercial fishery data are acknowledged (though not enumerated) and it is within the context of such limitations that these data are used. The oyster catches showed an inverse relationship with river flow; peaks of such productivity usually occurred during low river discharge, probably in response to salinity as well as to nutrient/detritus input from the river. Blue crabs and shrimp usually peaked during periods of peak river flow, reflecting a lesser sensitivity to low salinity and a possible trophic response to detritus, which is fed upon directly by such species at various periods of their development. These data reflect the different strategies of the species in question and show that the long-term (annual) cycles of river flow are important to such species. It follows that sustained high or low flow (without peaking) would disrupt the long-term functions of such species. The significance of these cycles, with application to the long-term data base in the Apalachicola estuary, is now under analysis. The impact analysis should be carried out within the context of this long-term data base.

A comparison of the 6-year trends of the populations of dominant fishes and invertebrates is given in Figs. 5-12. While there are interesting inter-population trends (e.g. Leiostomus xanthurus - Micropogon undulatus), these data are given here largely as background material and are presently under intensive review for publication in a series of papers. It is noteworthy that 1974 represented a peak year for white shrimp (Penaeus setiferus) while 1974 was an important year for blue crabs (Callinectes sapidus) and pink shrimp (Penaeus duorarum). Both 1974 and 1975 were peak years for Palaemonetes pugio.

An analysis was made of the otter trawl data from various portions of the Apalachicola estuary. Summed data for fishes and invertebrates in the outer bay (Apalachicola Bay) are given in Figs. 13-18. Fish indices showed trends similar to those of the bay-wide data with the possible exception of relative increases in species richness indicators during the summer and fall of 1977. The invertebrate data in the outer bay clearly reflected the impact of the river flooding of 1973 and the cold winter of 1976-77. The general trends of these data are similar to those of the whole-bay analysis. East Bay data west of the postulated impact area (stations 3, 4, 6) are shown in Figs. 19-24. Once again, the trends were similar to those in the whole-bay analysis. The fish species richness and diversity tended to increase and level off from 1972 to 1974-75 with general decreases during the last 2 to 3 years of sampling. The invertebrates showed a similar pattern with relatively high species richness values during the first year of sampling. In both groups, there was evidence of impact due to the 1972-73 river flooding and the cold winter of 1976. Thus, peaks for various fish and diversity indices occurred during the period from 1974 to 1975 in Apalachicola Bay and western portions of East Bay. Data for fishes and invertebrates in eastern portions of East Bay (station 5, 5A) are shown in Figs. 25-30. The number of fish species during 1975 tended to be low relative to the bay-wide trends, with the lowest monthly species numbers and Margalef richness occurring during the fall of 1975 (during the peak rainfall). Episodic fluctuations of river flow and low winter temperatures also had a considerable effect on the fish richness and diversity indices. The general patterns in the eastern portion of East Bay differ considerably from those in other portions of the bay and from the bay-wide data in that the 1974-75 period was not a time of peaking indices. The invertebrate richness and diversity indices, unlike the summed bay-wide long-term trends, showed relatively low levels during 1974-75 with no real peaks during this period. The shape of the species richness curve was different from that of the summed data or from western portions of East Bay. These differences are apparent in a comparison of stations 5 and 6 (Figs. 31-41) and seem to reflect the runoff characteristics of the different portions of the bay. The heavy rainfall, together with changes in related water quality in the East Bay drainage area during 1974-75, were probably responsible for increased stress on the biological systems in the receiving portions of the bay system during this period. The differences in the long-term patterns in different sections of the bay are presently being analyzed using various forms of time-series analysis. The resulting models will further test the hypothesis that eastern portions of East Bay have biological trends that differ from those of other portions of the bay.

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Fig. 1: Six-year trends of Apalachicola River flow, bay water temperature, and various fish indices (number of species, Margalef richness, Shannon diversity, and total biomass) taken from bay-wide summed otter trawl data.

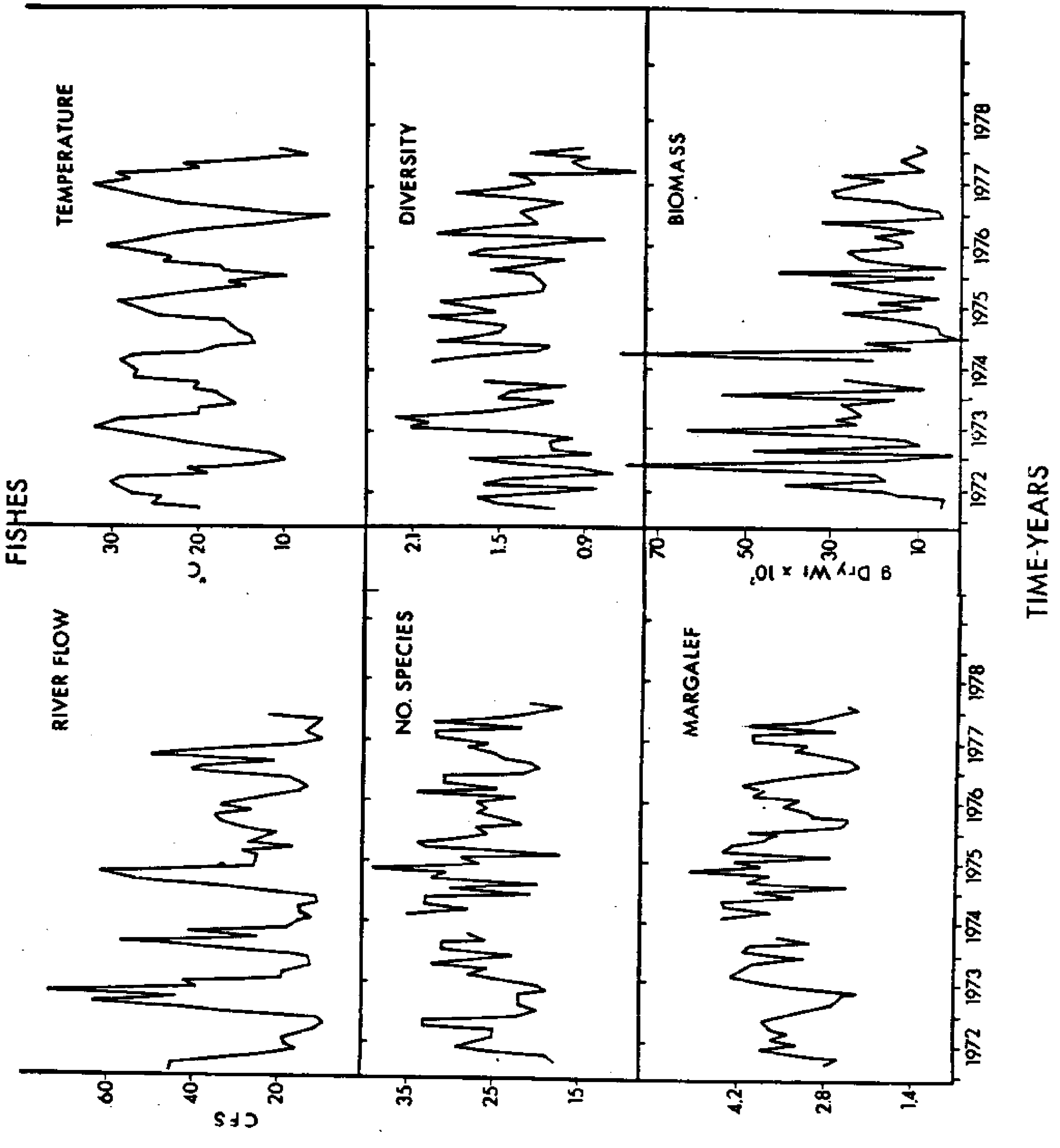

FISHES
RIVER FLOW
CFS
60
40
20
TEMPERATURE
°C
30
20
10
NO. SPECIES
35
25
15
DIVERSITY
2.1
1.5
0.9
MARGALEF
4.2
2.8
1.4
BIOMASS
g Dry Wt x 10²
70
50
30
10
1972
1973
1974
1975
1976
1977
1978
TIME-YEARS

Fig. 2: Long-term (1975-78) trends of Apalachicola River flow (mean monthly flows in CFS), macroparticulates (leaves, detritus, etc. taken from otter trawls in the bay) and microparticulates (particulate matter from 37μ to 2 mm in diameter taken from filtered river water).

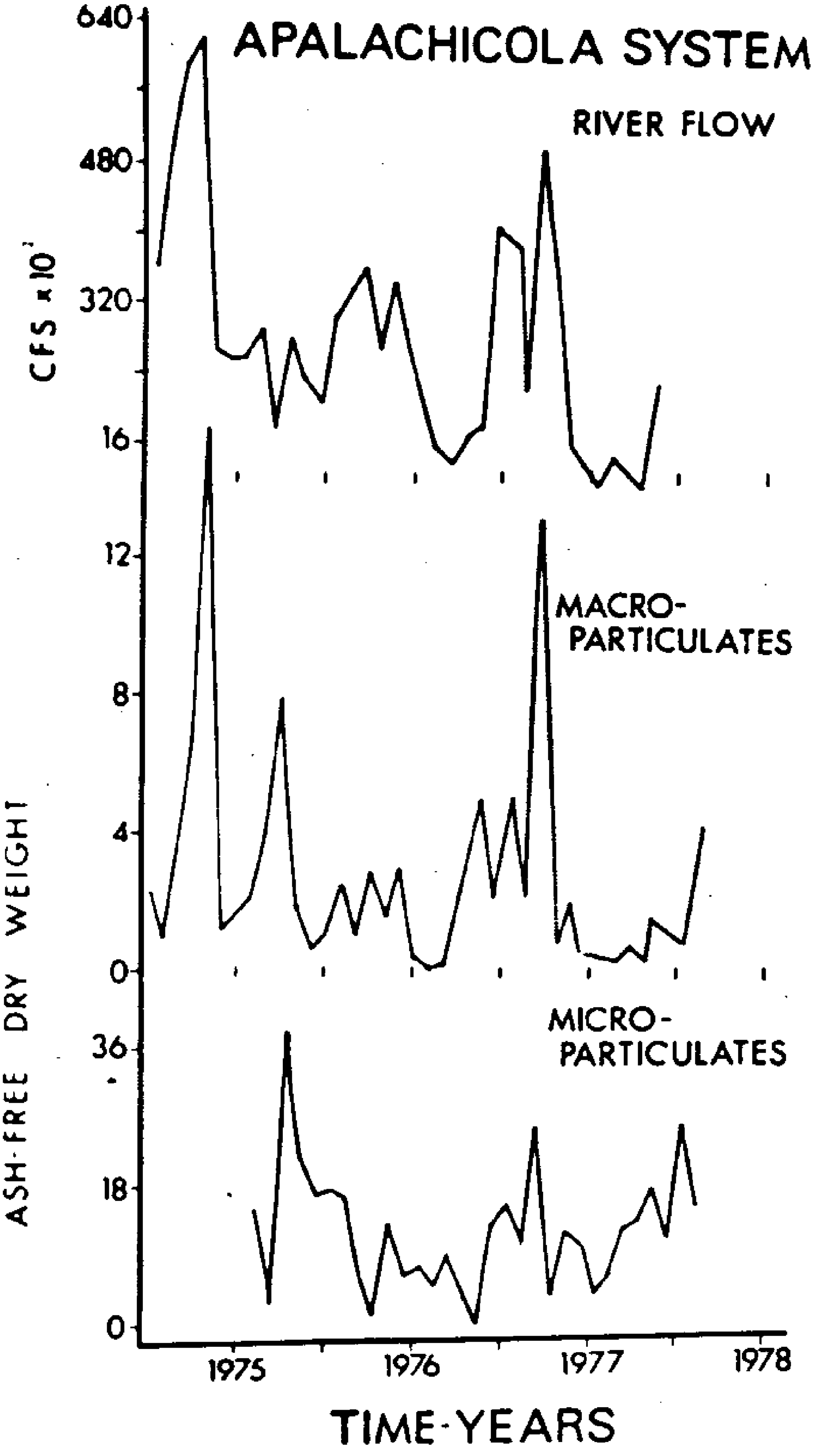

APALACHICOLA SYSTEM
RIVER FLOW
CFS x 10³
640
480
320
16
MACRO-PARTICULATES
12
8
4
0
MICRO-PARTICULATES
36
18
0
ASH-FREE DRY WEIGHT
1975
1976
1977
1978
TIME-YEARS

Fig. 3: Six-year trends of Apalachicola rainfall (total monthly, cm), water color (surface and bottom, stations 1 and 5, Pt-Co units), salinity (surface and bottom, stations 1 and 5, ppt), and various invertebrate indices (number of species, Margalef richness, Shannon diversity, and total number of individuals) taken from bay-wide, summed otter trawl data.

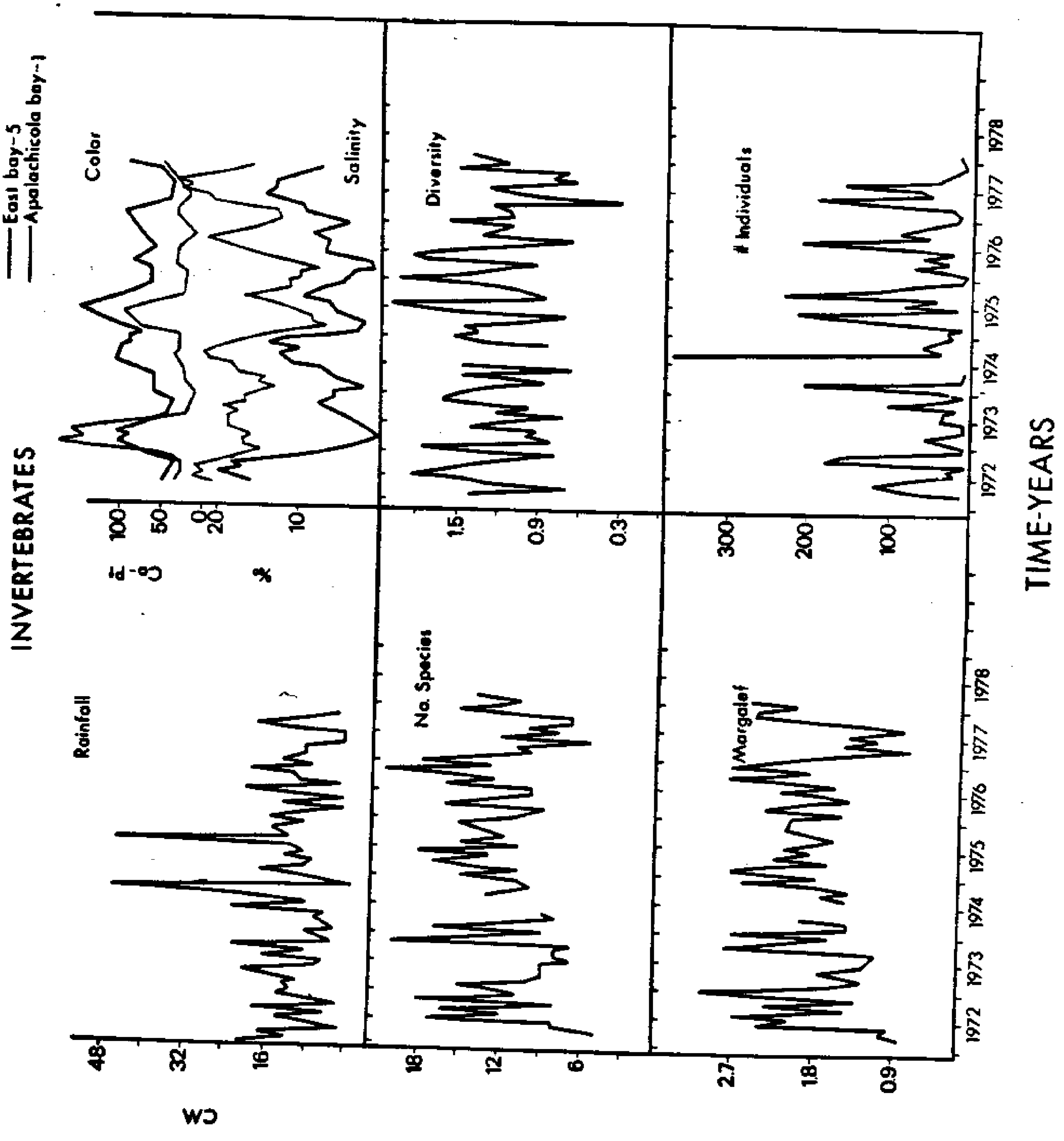

INVERTEBRATES
East bay-5
Apalachicola bay-1
CM
48
32
16
Rainfall
18
12
6
No. Species
2.7
1.8
0.9
Margalef
Co-Pt
%
100
50
0
20
10
Color
Salinity
1.5
0.9
0.3
Diversity
300
200
100
Individuals
1972
1973
1974
1975
1976
1977
1978
TIME-YEARS

Fig. 4: Twenty-year trends of Apalachicola River flow (annual totals, C. F. C.) and Franklin County landings (in thousands of pounds) for oysters (Crassostrea virginica), shrimp (Penaeus spp.), and blue crabs (Callinectes sapidus). Commercial catch data represent annual totals reported by the Florida Department of Natural Resources (Summaries of Florida Commerical Marine Fish Landings) as summarized by Cato and Prochaska (1977). Data taken prior to 1957 were not used because of the abolition of the severance tax at that time (Whitfield and Beaumariage, 1977).

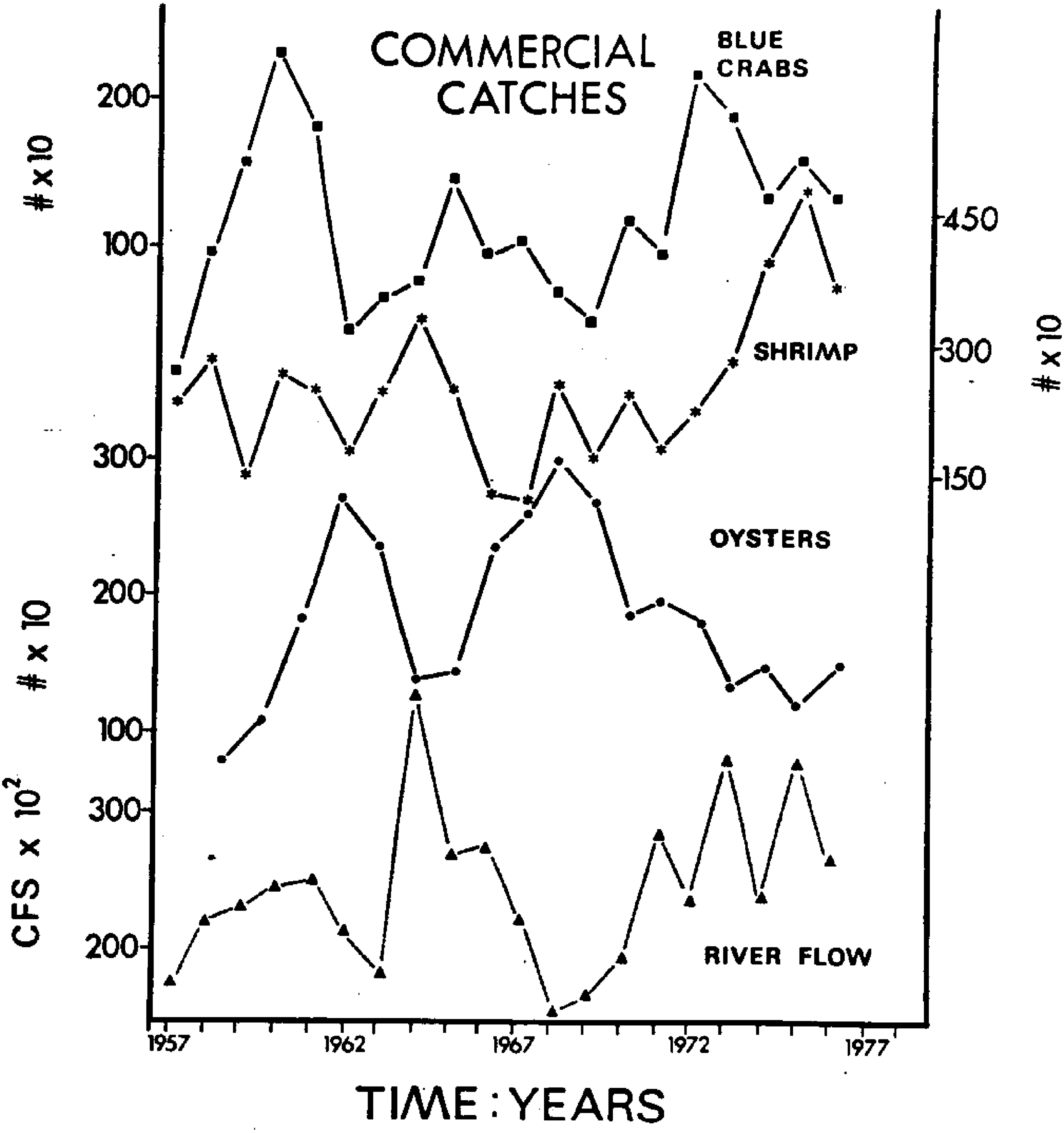

COMMERCIAL CATCHES
BLUE CRABS
SHRIMP
OYSTERS
RIVER FLOW
x 10
200
100
300
200
100
x 10
CFS x 10²
300
200
450
300
150
x 10
1957
1962
1967
1972
1977
TIME : YEARS

Fig. 5: Numbers of individuals of Anchoa mitchilli taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978.

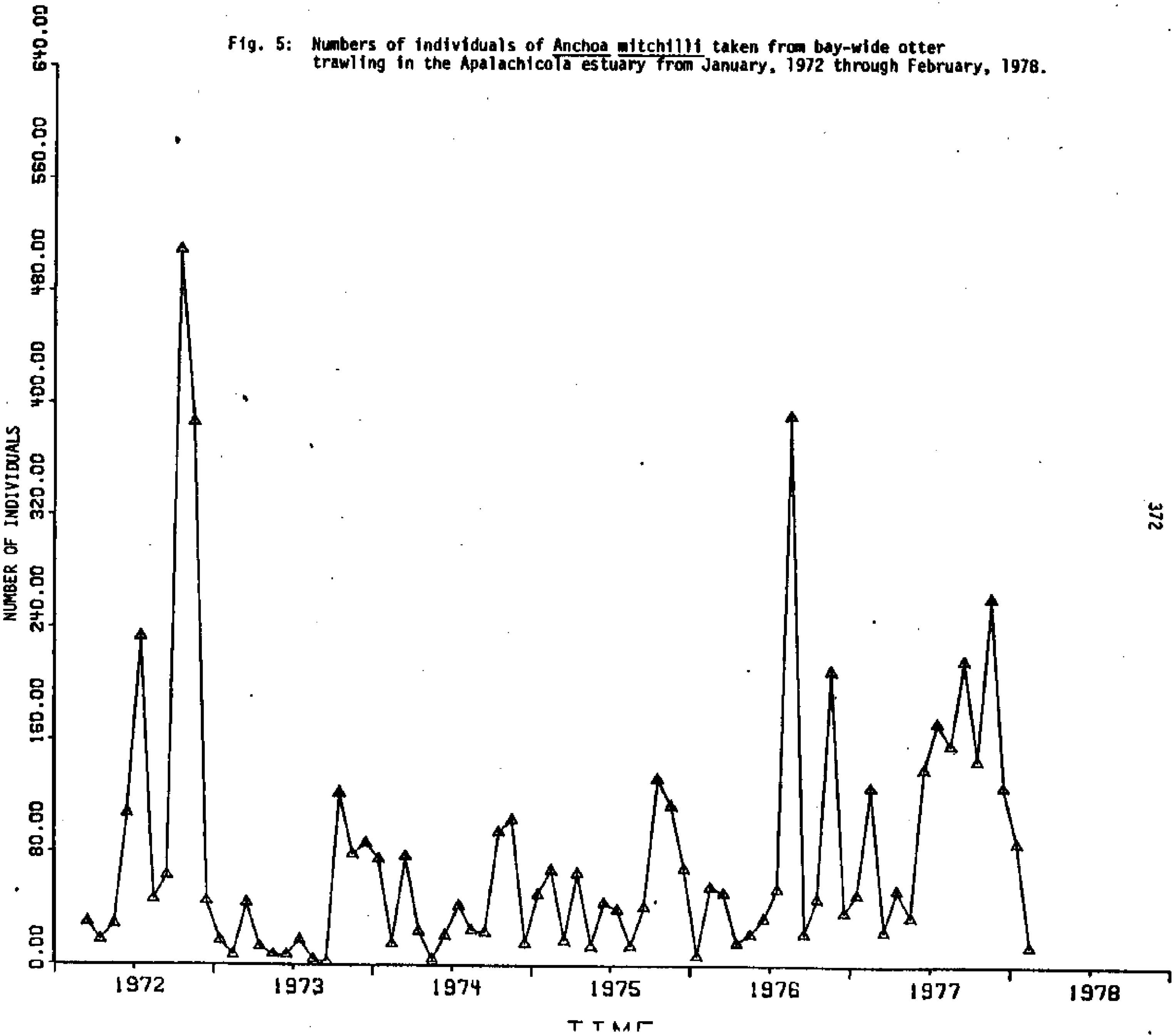
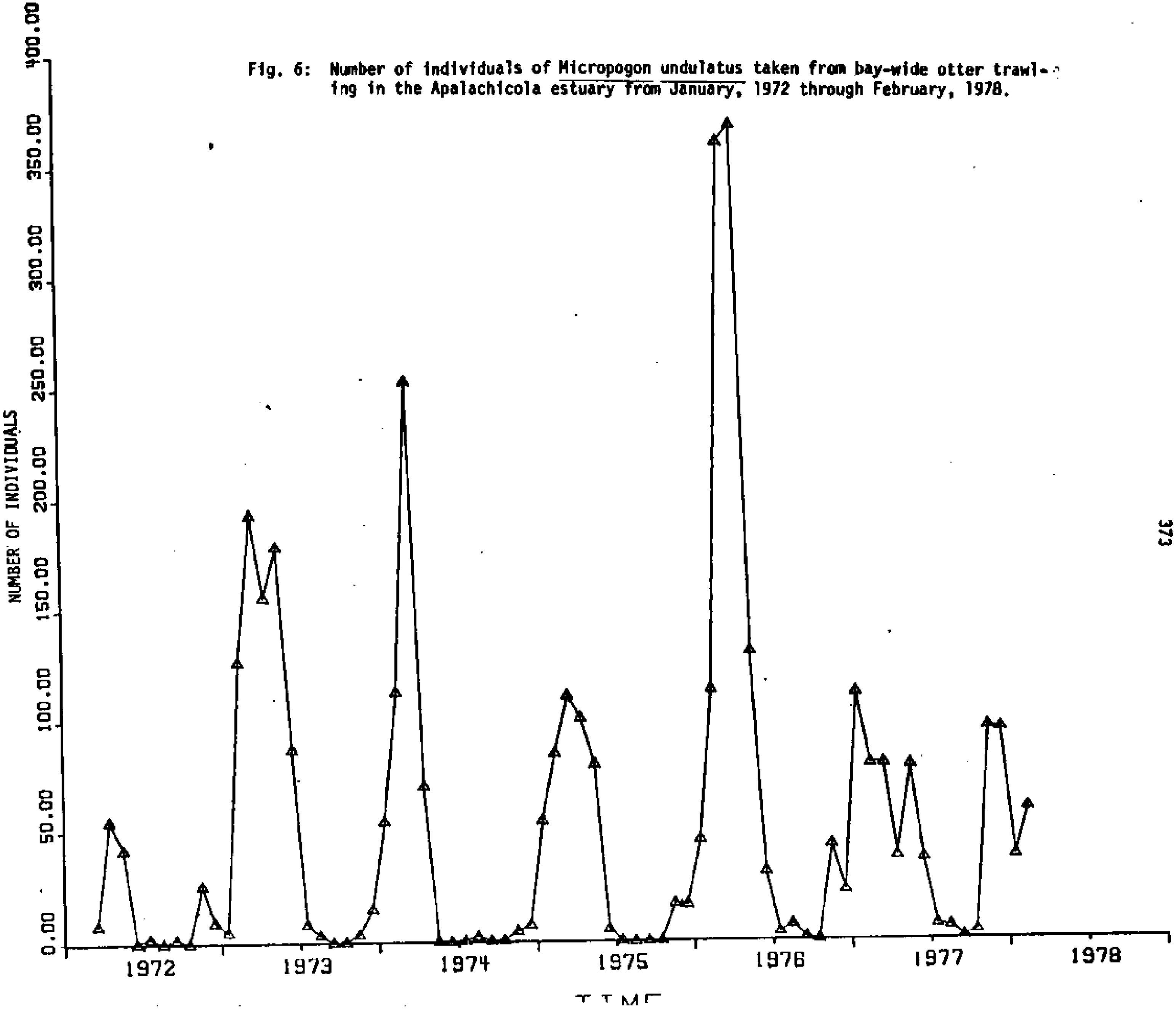


Fig. 6: Number of individuals of Micropogon undulatus taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978.



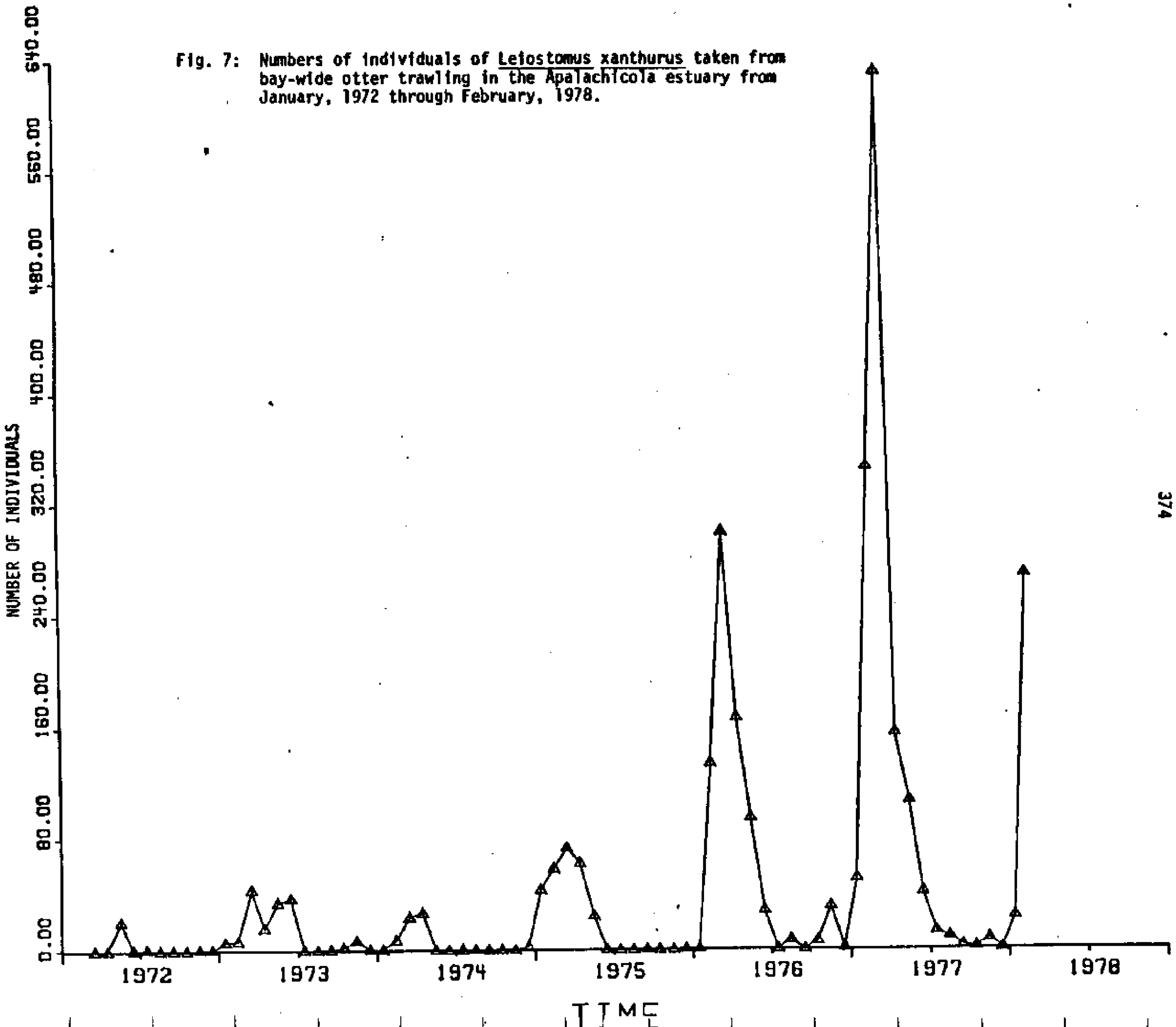


Fig. 7: Numbers of individuals of Leiostomus xanthurus taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978.

Fig. 9: Numbers of individuals of Callinectes sapidus taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978.

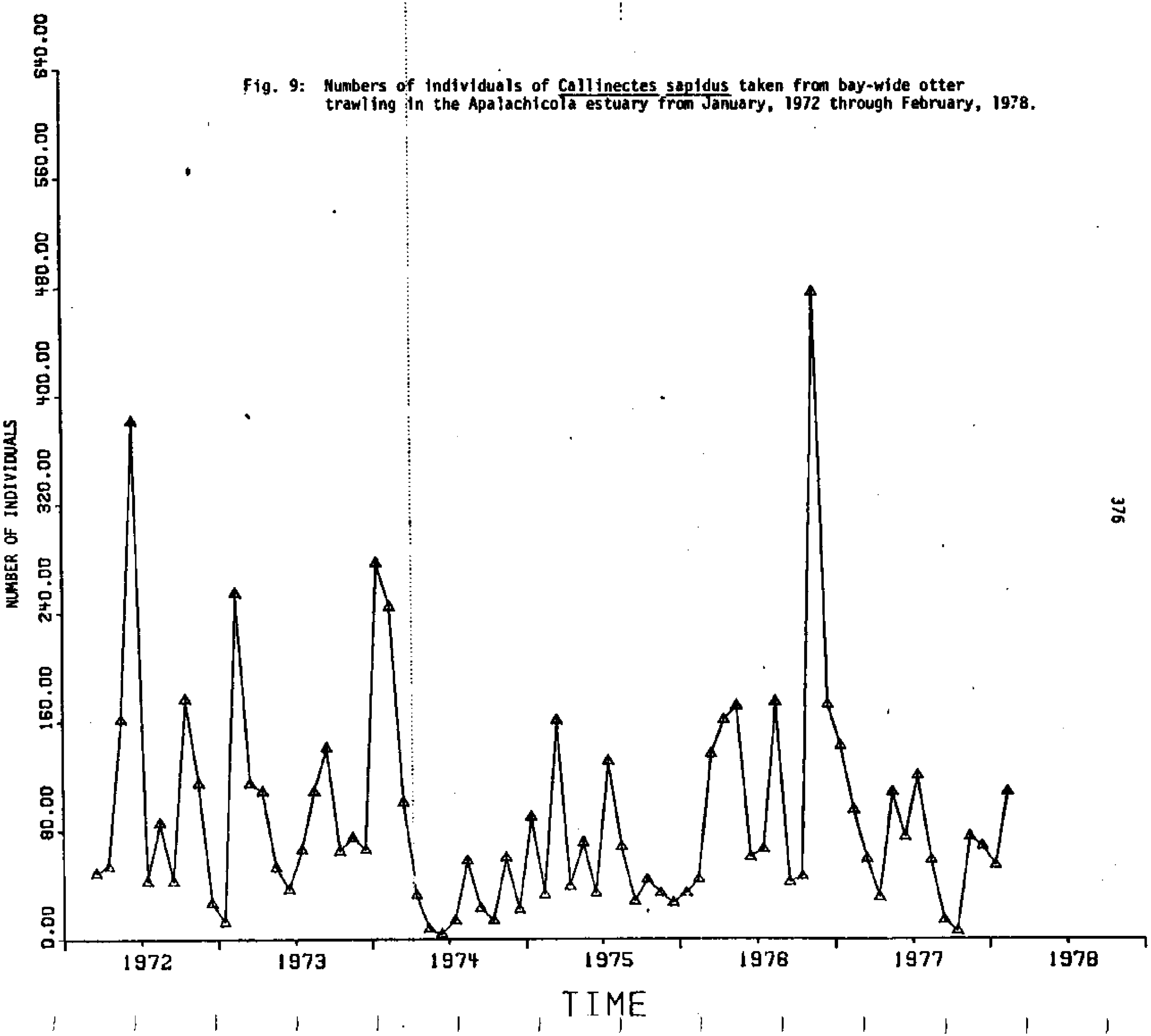


Fig. 10: Numbers of individuals of *Penaeus setiferus* taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978

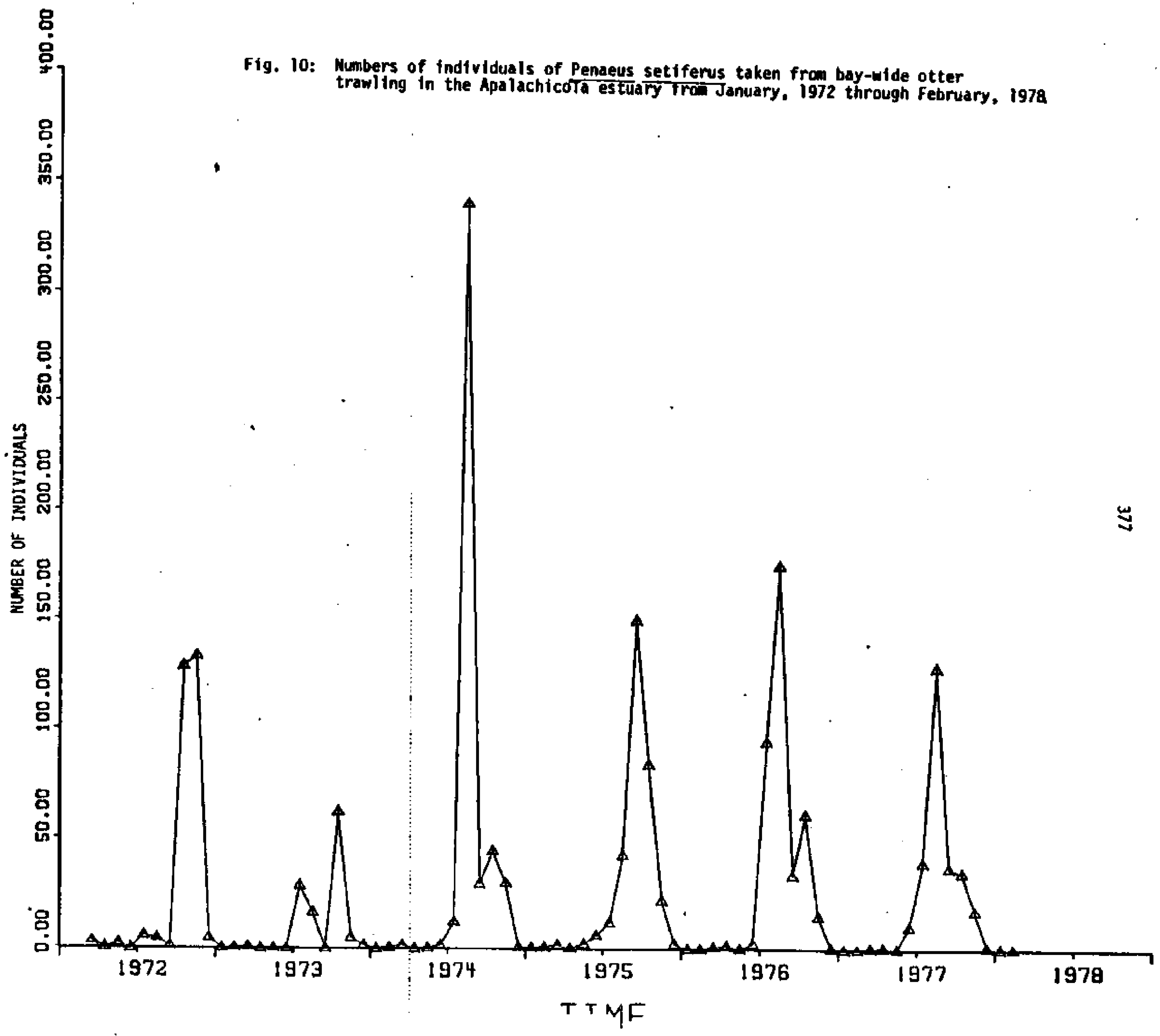


Fig. 11: Numbers of individuals of *Penaeus duorarum* taken from bay-wide otter trawling in the Apalachciola estuary from January, 1972 through February, 1978.

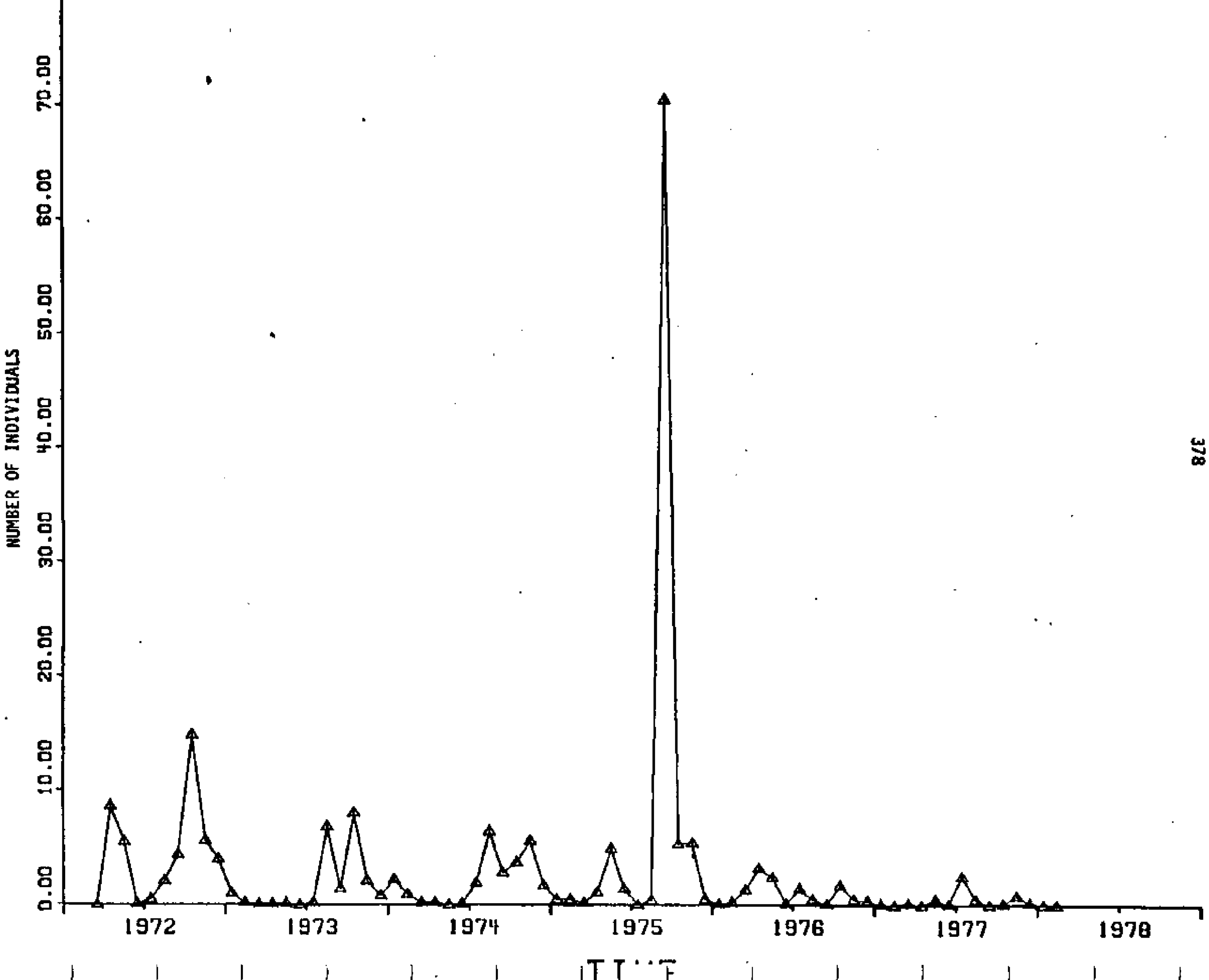


Fig. 12: Numbers of individuals of Palaemonetes pugio taken from bay-wide otter trawling in the Apalachicola estuary from January, 1972 through February, 1978.

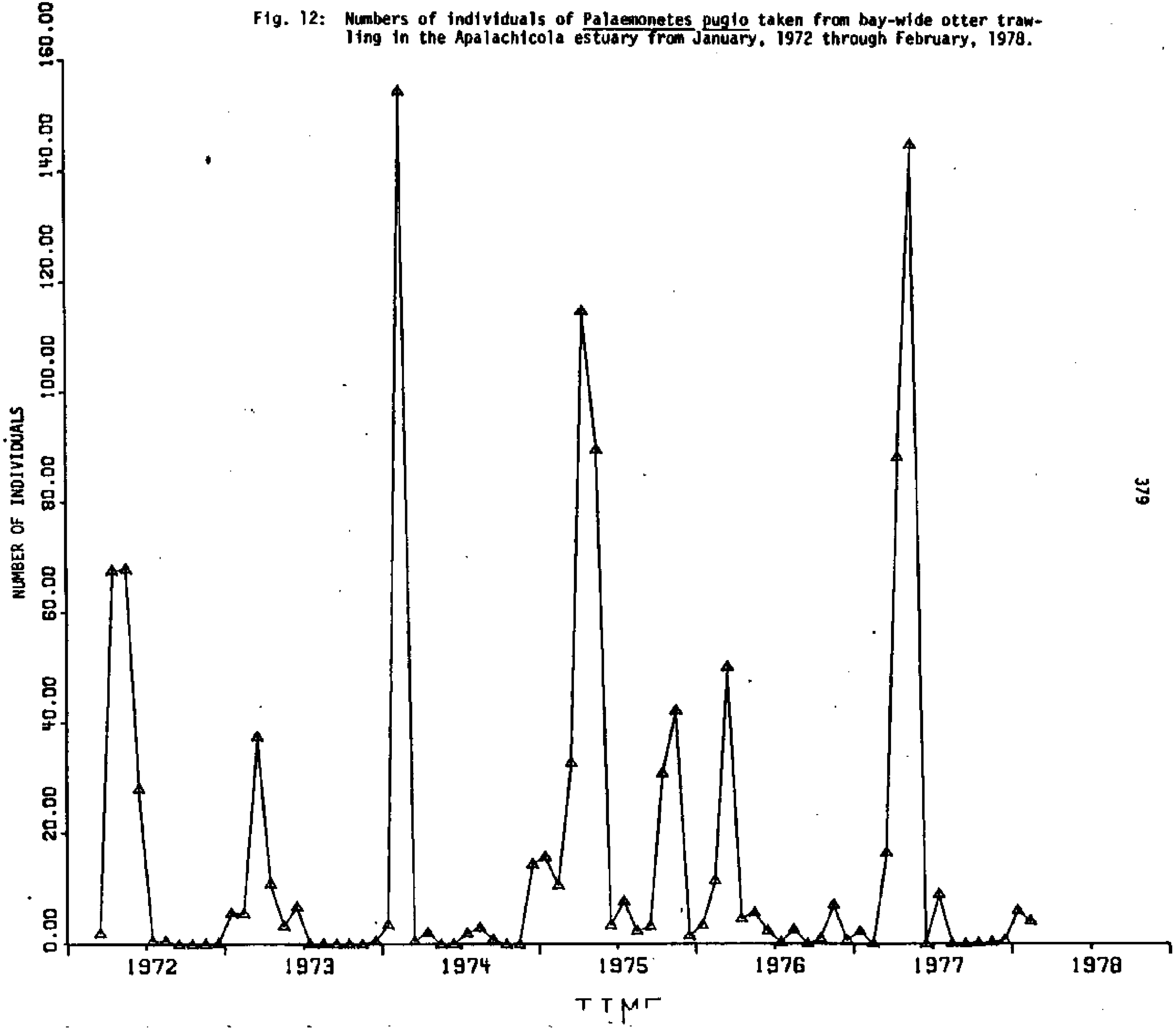
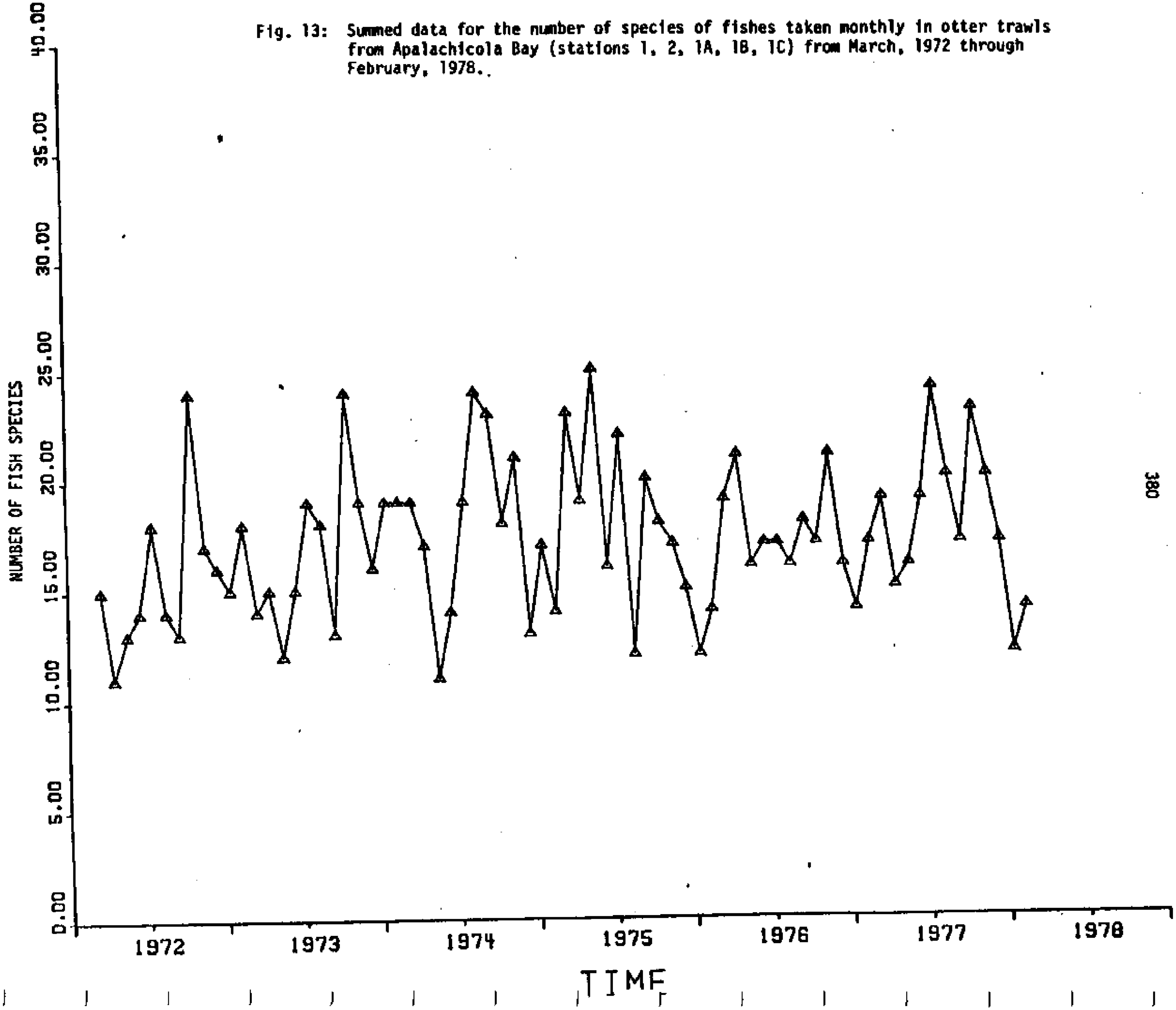


Fig. 13: Summed data for the number of species of fishes taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.



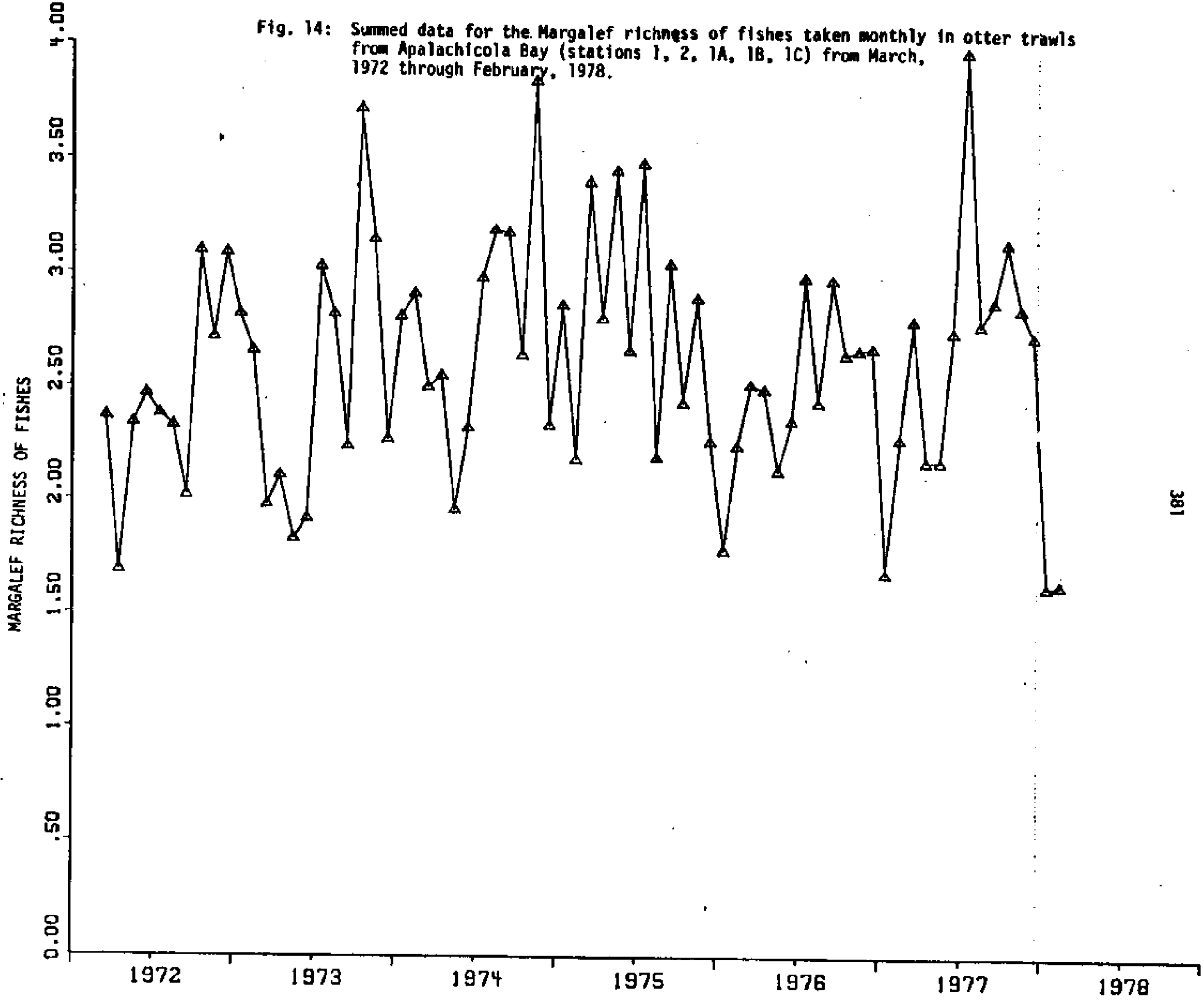


Fig. 14: Summed data for the Margalef richness of fishes taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.

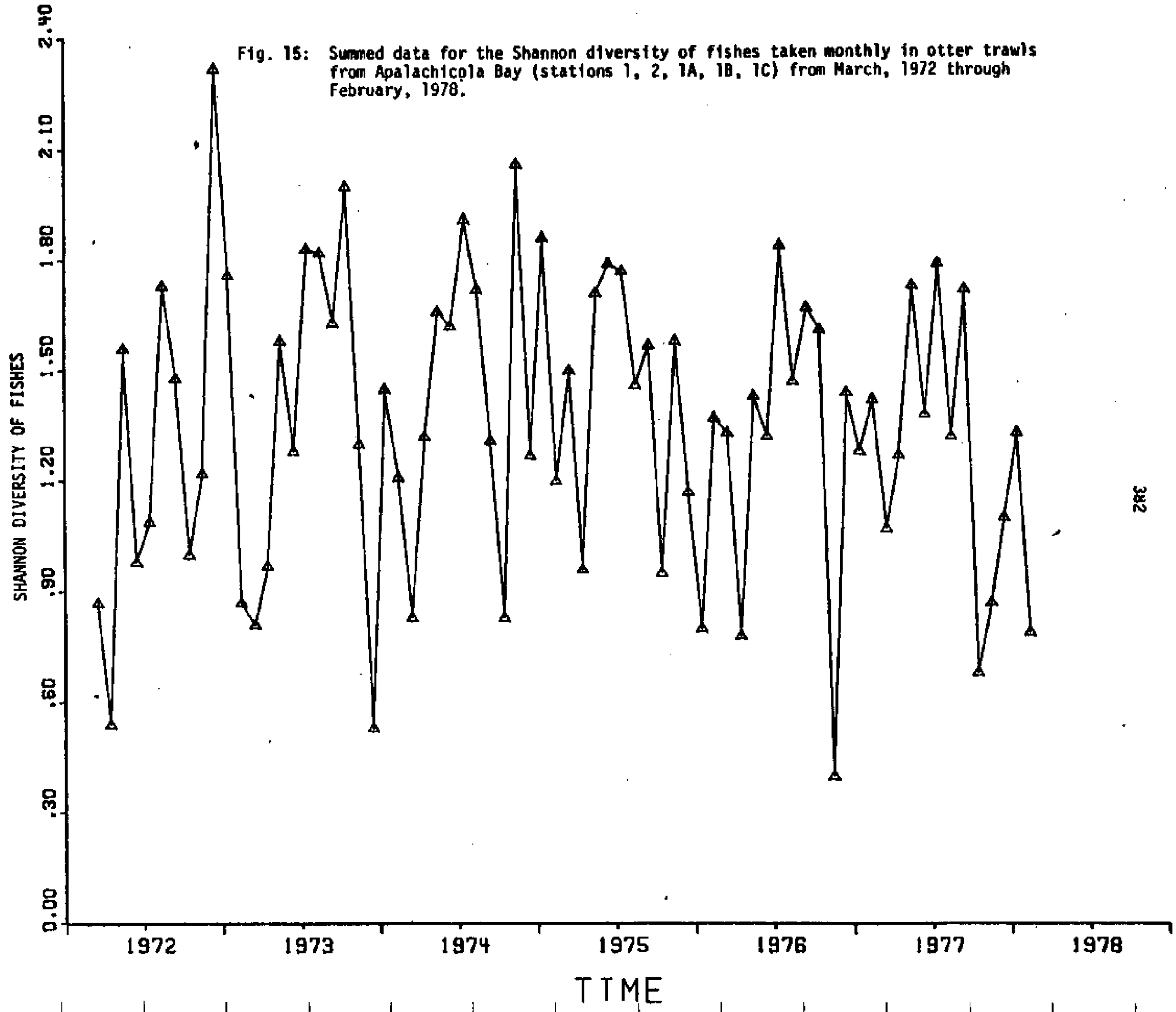


Fig. 15: Summed data for the Shannon diversity of fishes taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.

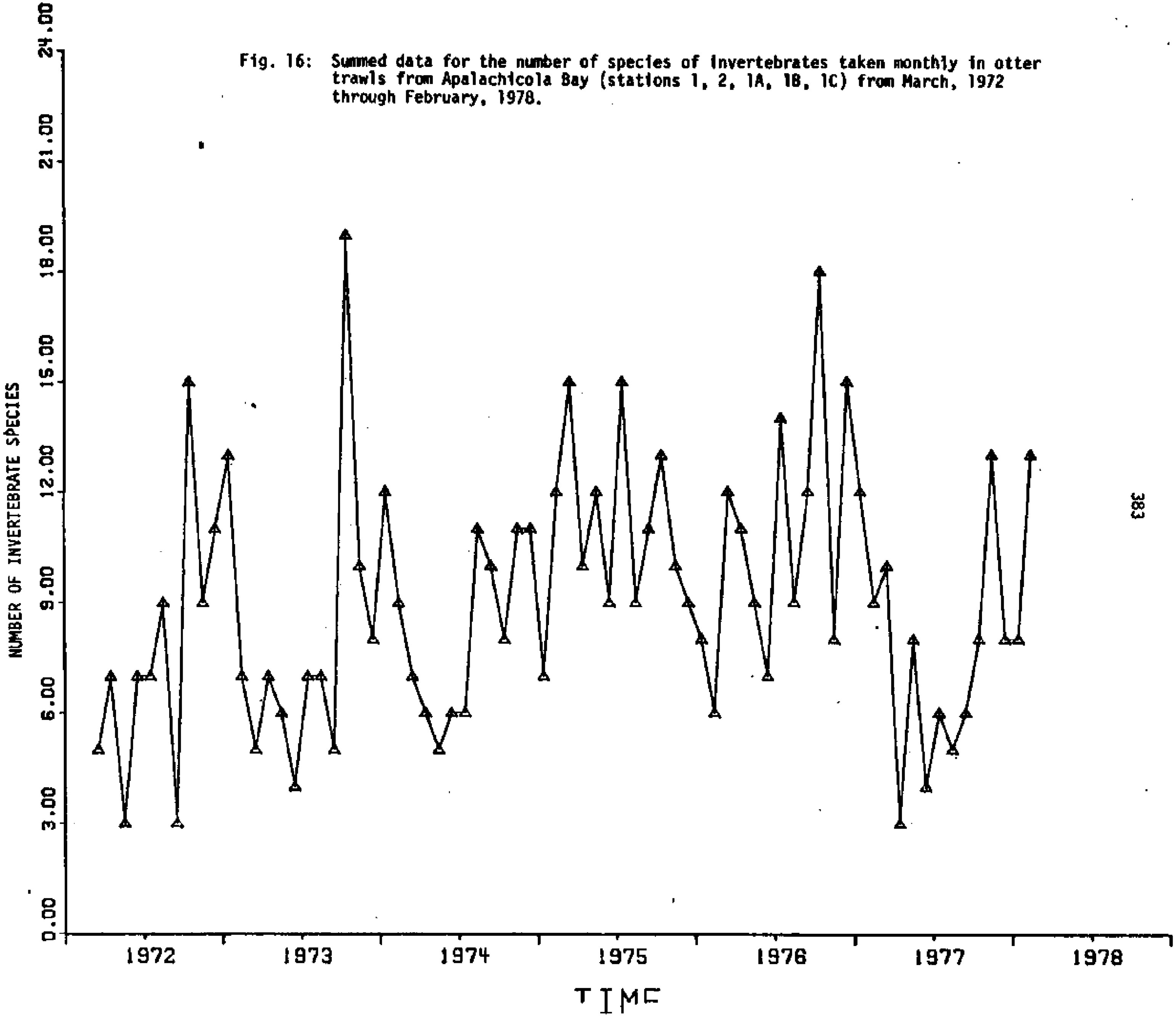


Fig. 16: Summed data for the number of species of invertebrates taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.

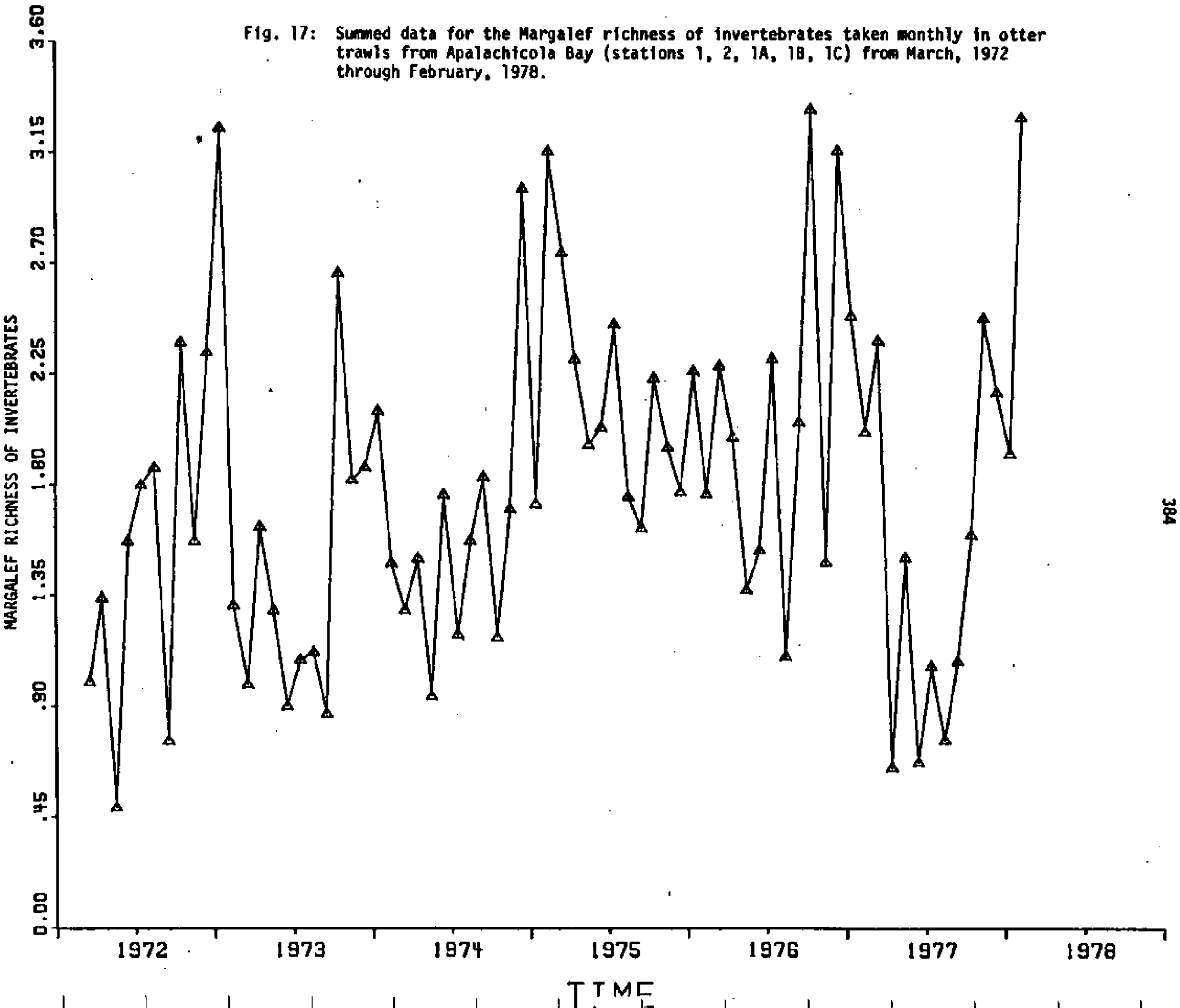
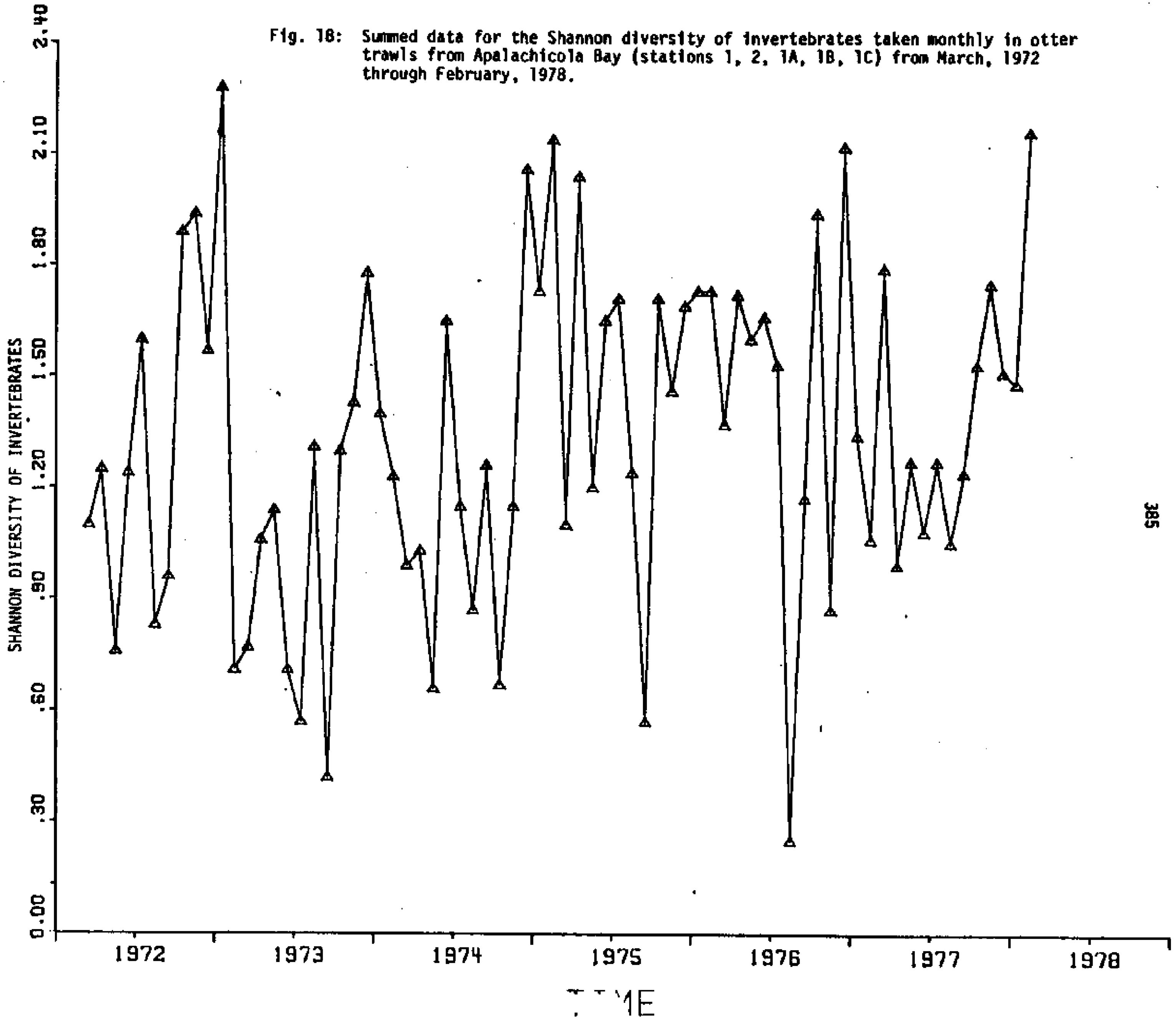


Fig. 17: Summed data for the Margalef richness of invertebrates taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.

Fig. 18: Summed data for the Shannon diversity of invertebrates taken monthly in otter trawls from Apalachicola Bay (stations 1, 2, 1A, 1B, 1C) from March, 1972 through February, 1978.



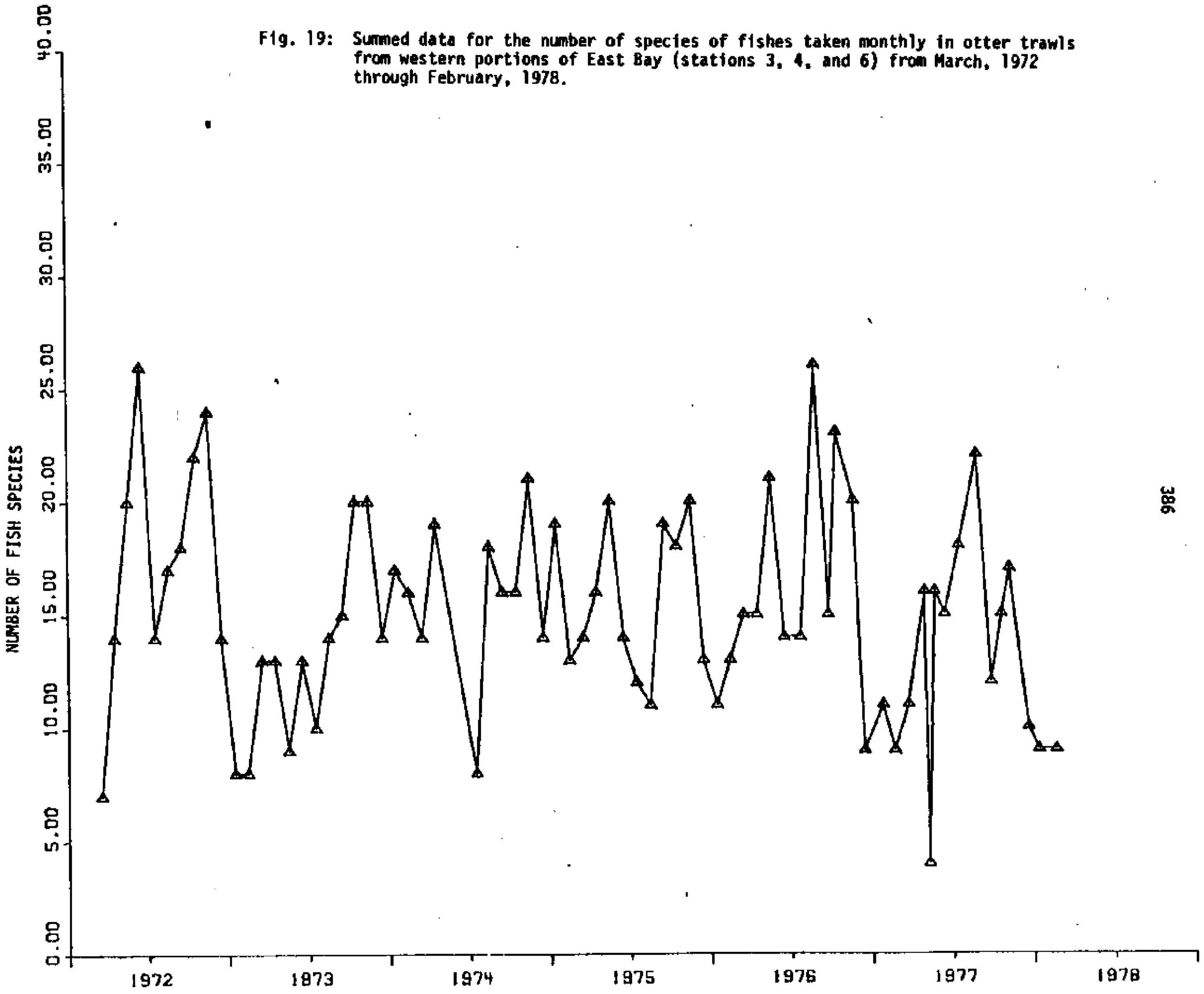


Fig. 19: Summed data for the number of species of fishes taken monthly in otter trawls from western portions of East Bay (stations 3, 4, and 6) from March, 1972 through February, 1978.

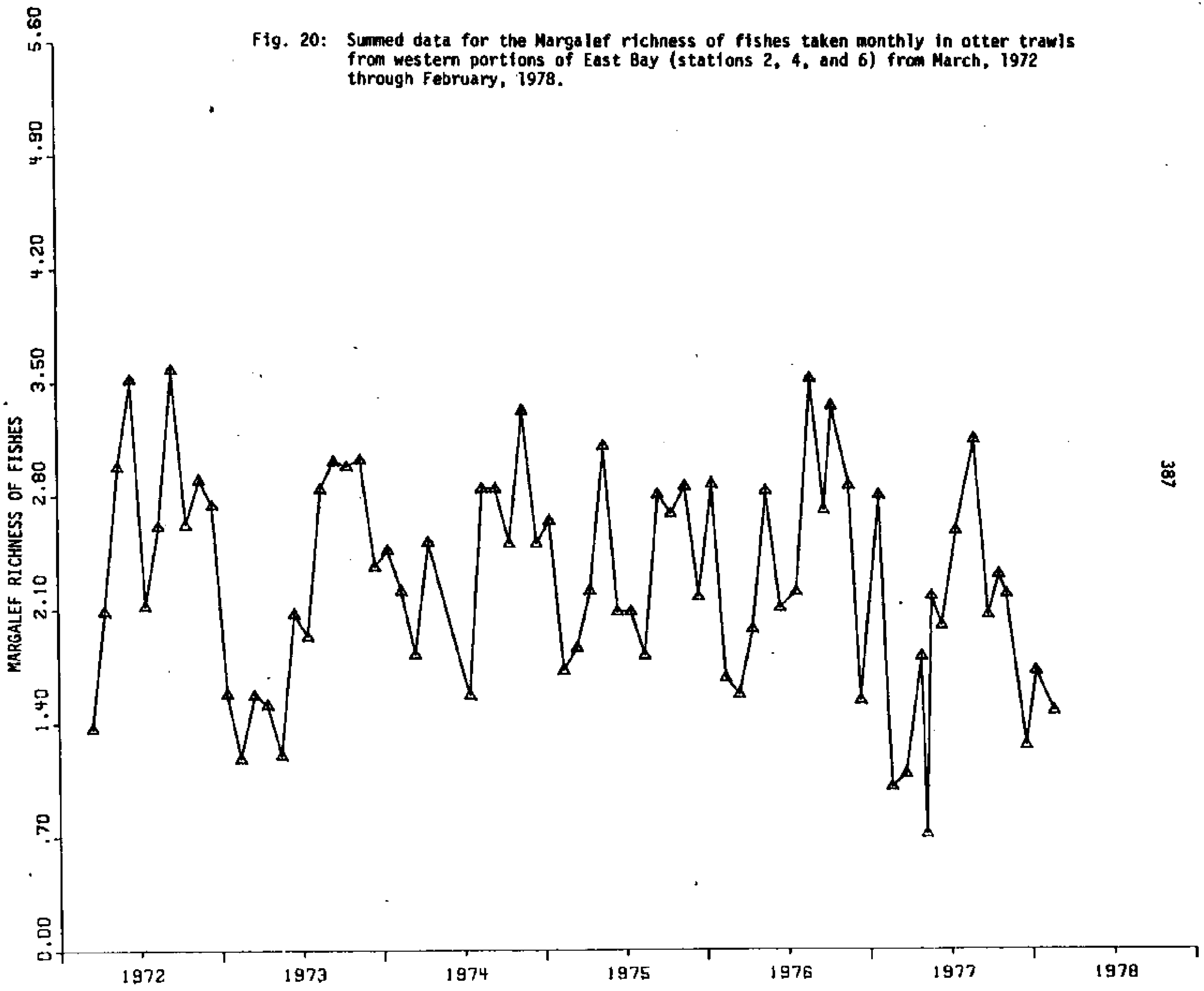


Fig. 20: Summed data for the Margalef richness of fishes taken monthly in otter trawls from western portions of East Bay (stations 2, 4, and 6) from March, 1972 through February, 1978.

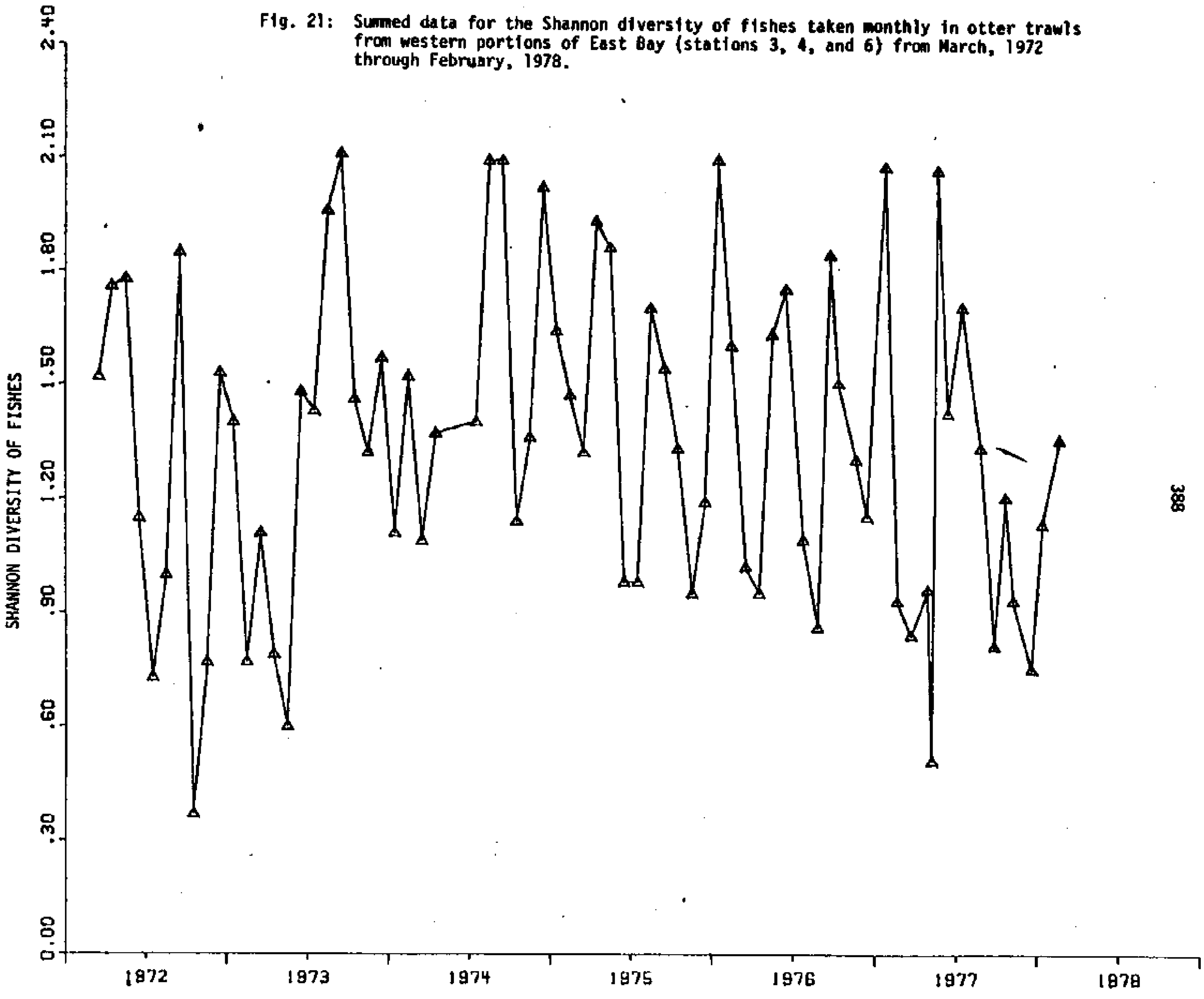


Fig. 21: Summed data for the Shannon diversity of fishes taken monthly in otter trawls from western portions of East Bay (stations 3, 4, and 6) from March, 1972 through February, 1978.

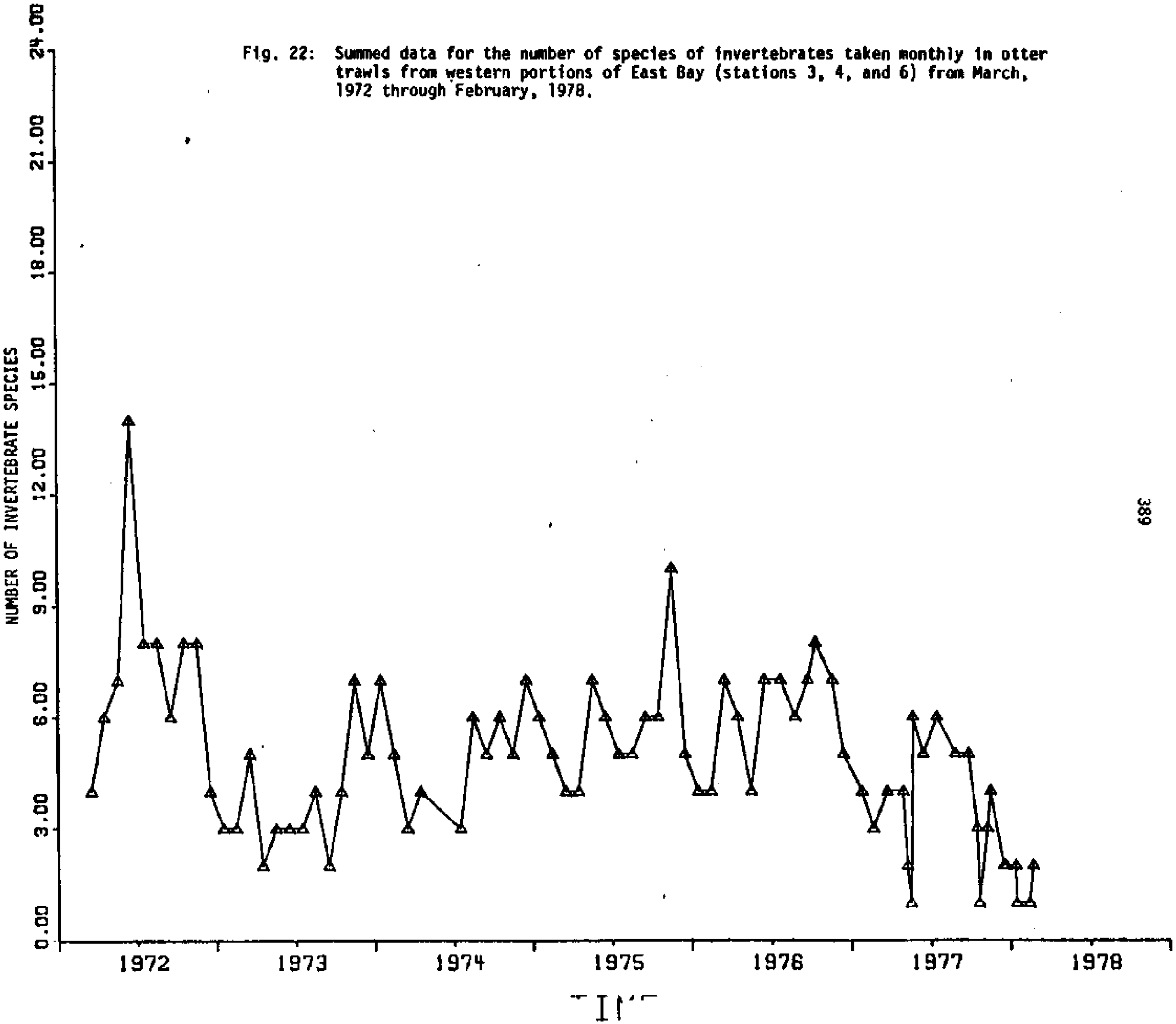


Fig. 22: Summed data for the number of species of invertebrates taken monthly in otter trawls from western portions of East Bay (stations 3, 4, and 6) from March, 1972 through February, 1978.

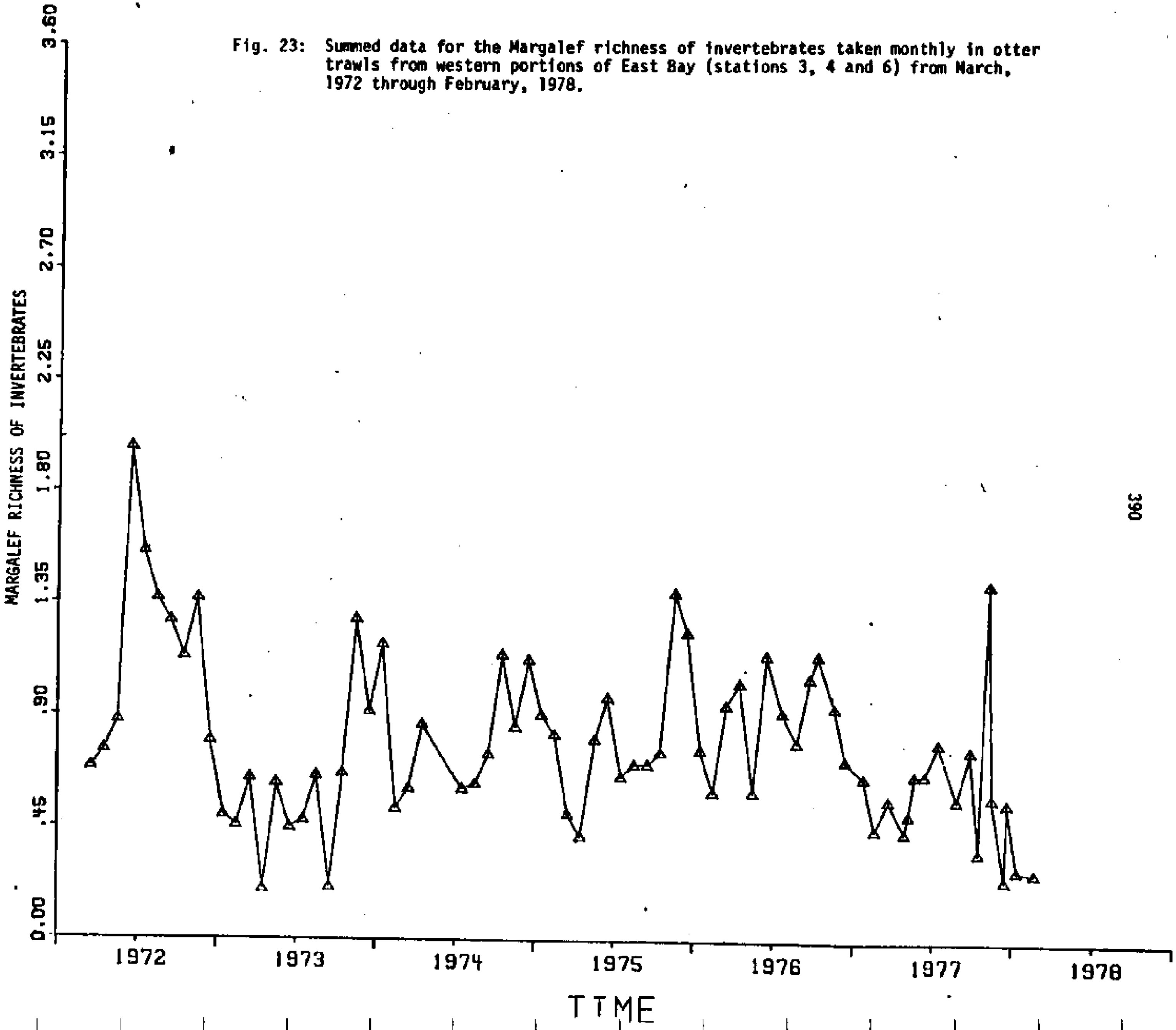


Fig. 23: Summed data for the Margalef richness of invertebrates taken monthly in otter trawls from western portions of East Bay (stations 3, 4 and 6) from March, 1972 through February, 1978.

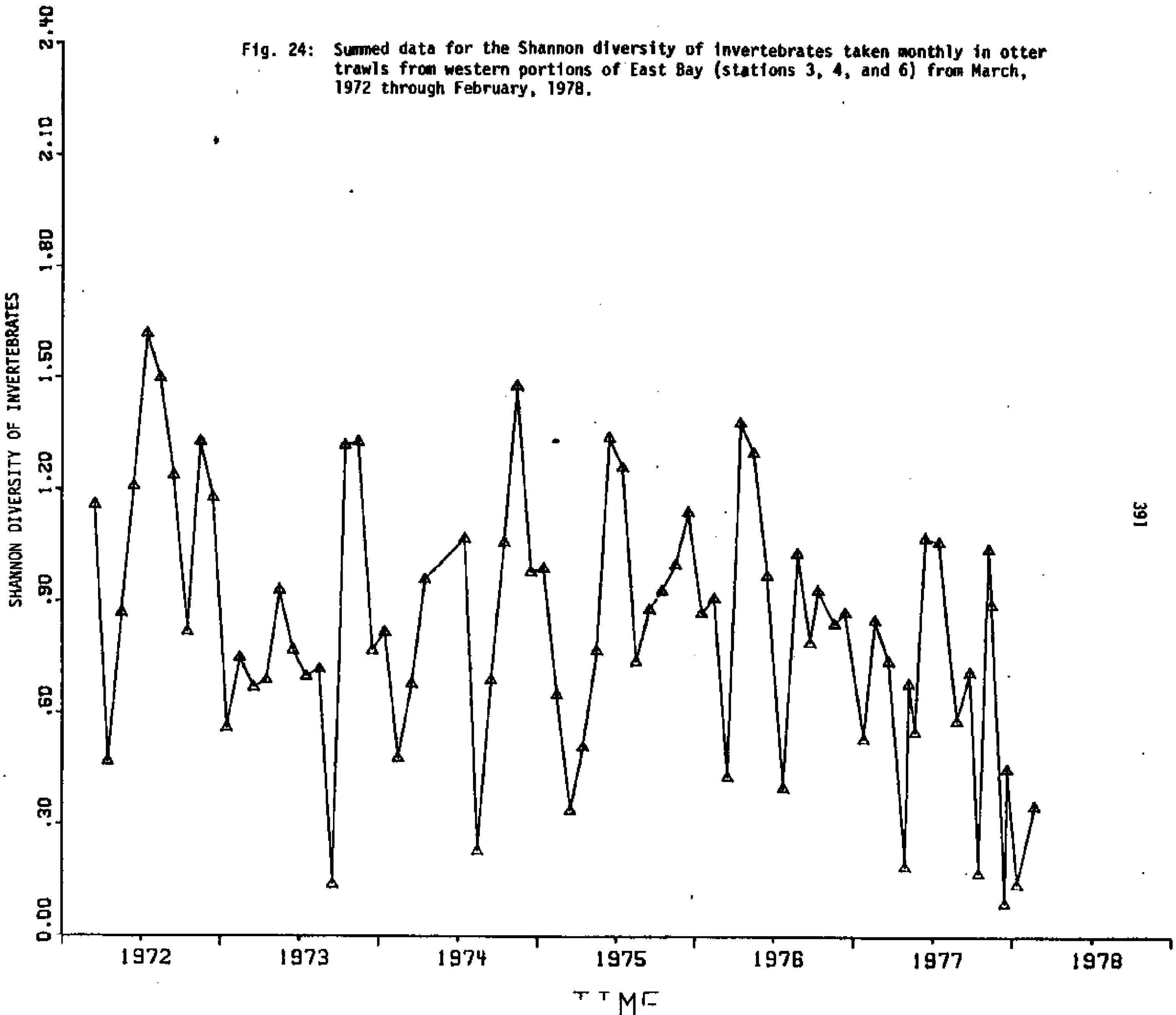


Fig. 24: Summed data for the Shannon diversity of invertebrates taken monthly in otter trawls from western portions of East Bay (stations 3, 4, and 6) from March, 1972 through February, 1978.

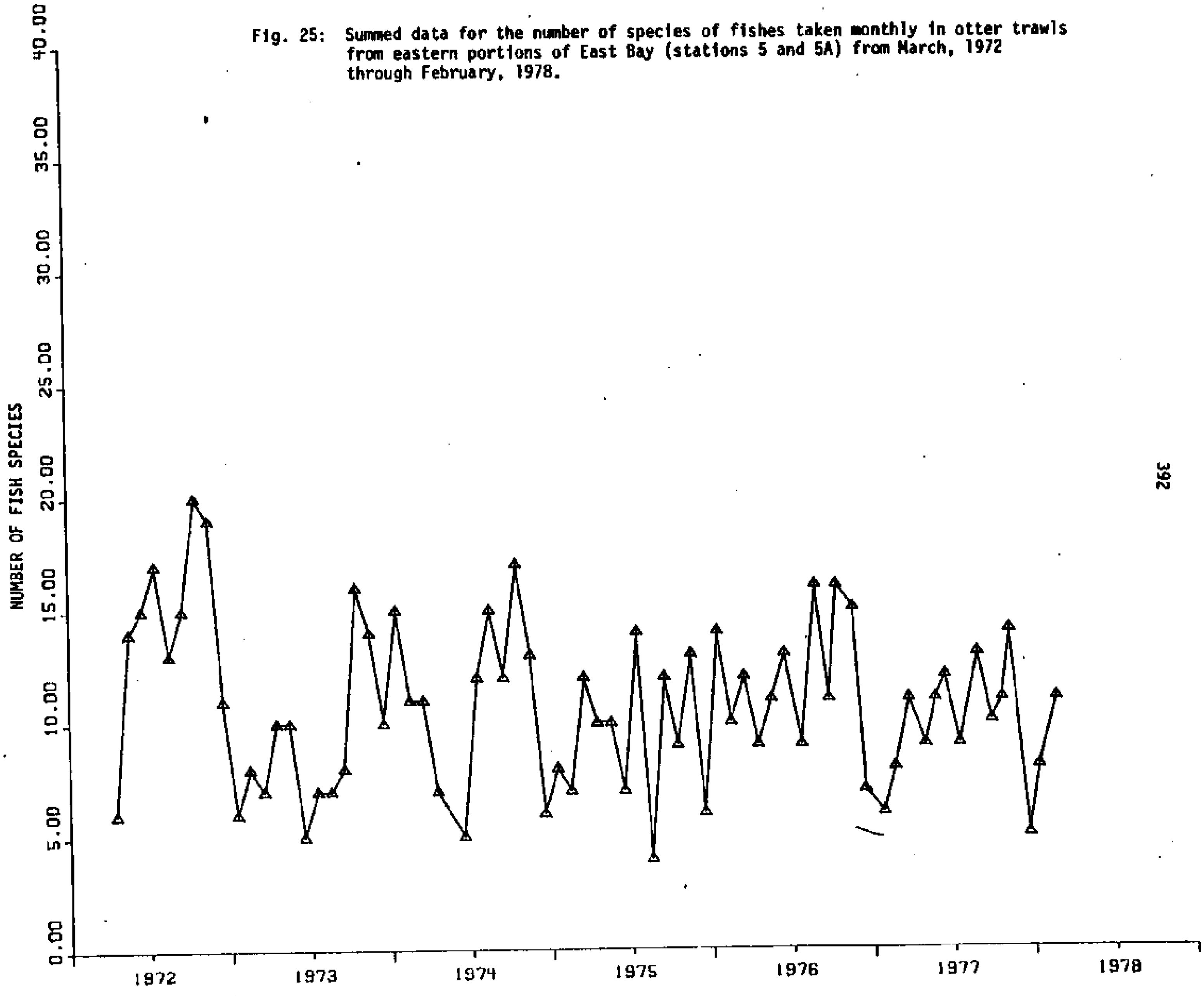


Fig. 25: Summed data for the number of species of fishes taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

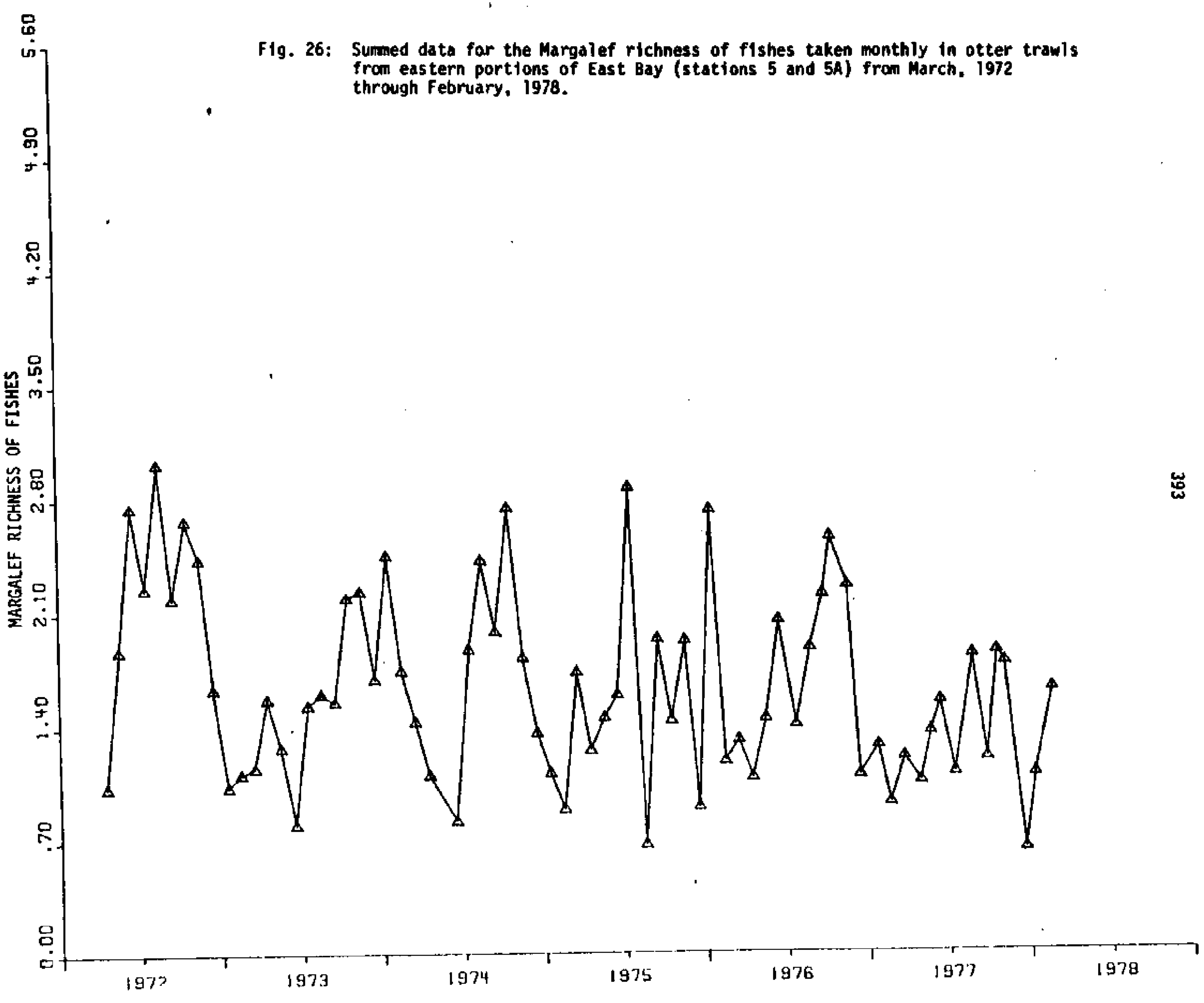


Fig. 26: Summed data for the Margalef richness of fishes taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

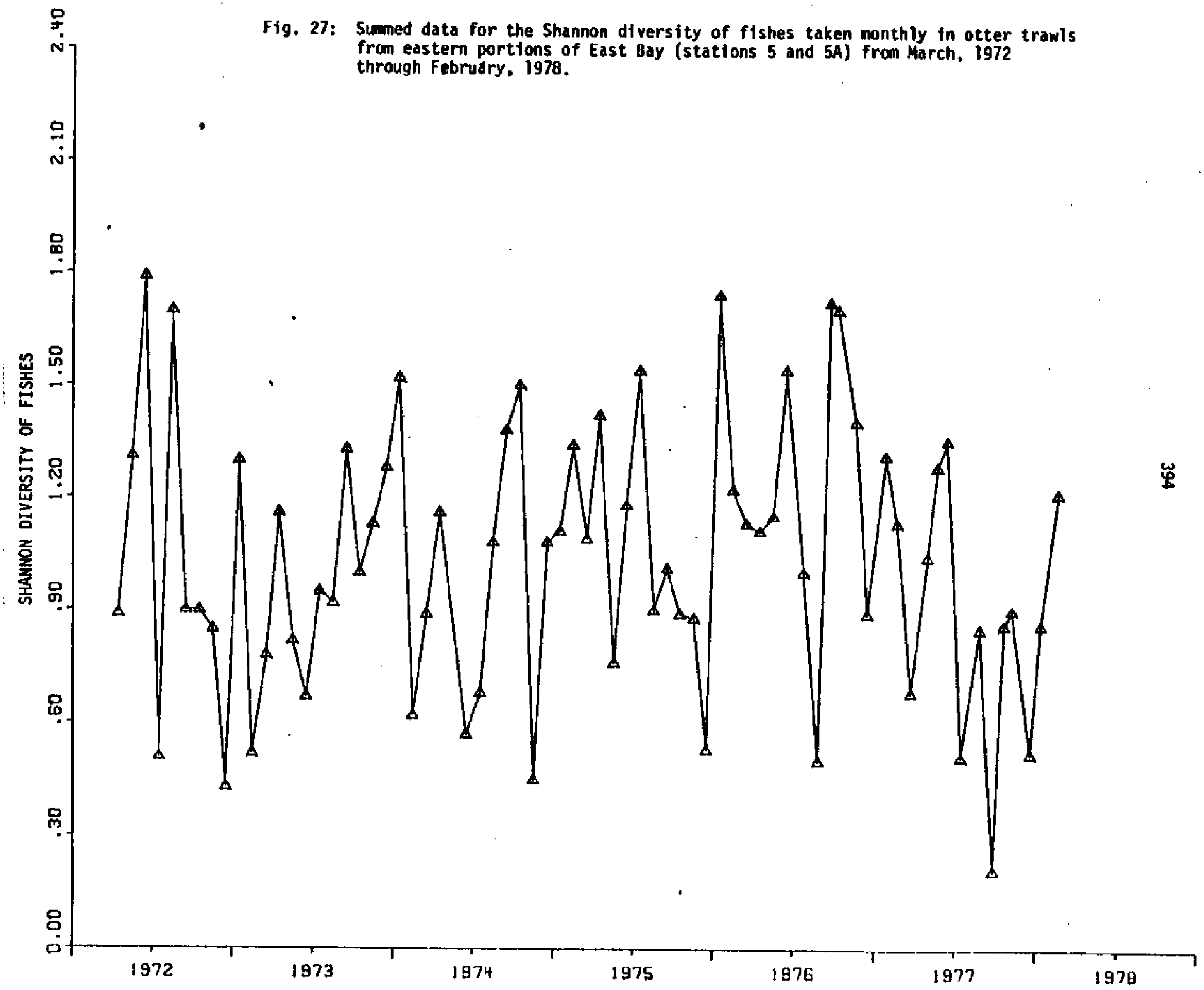


Fig. 27: Summed data for the Shannon diversity of fishes taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

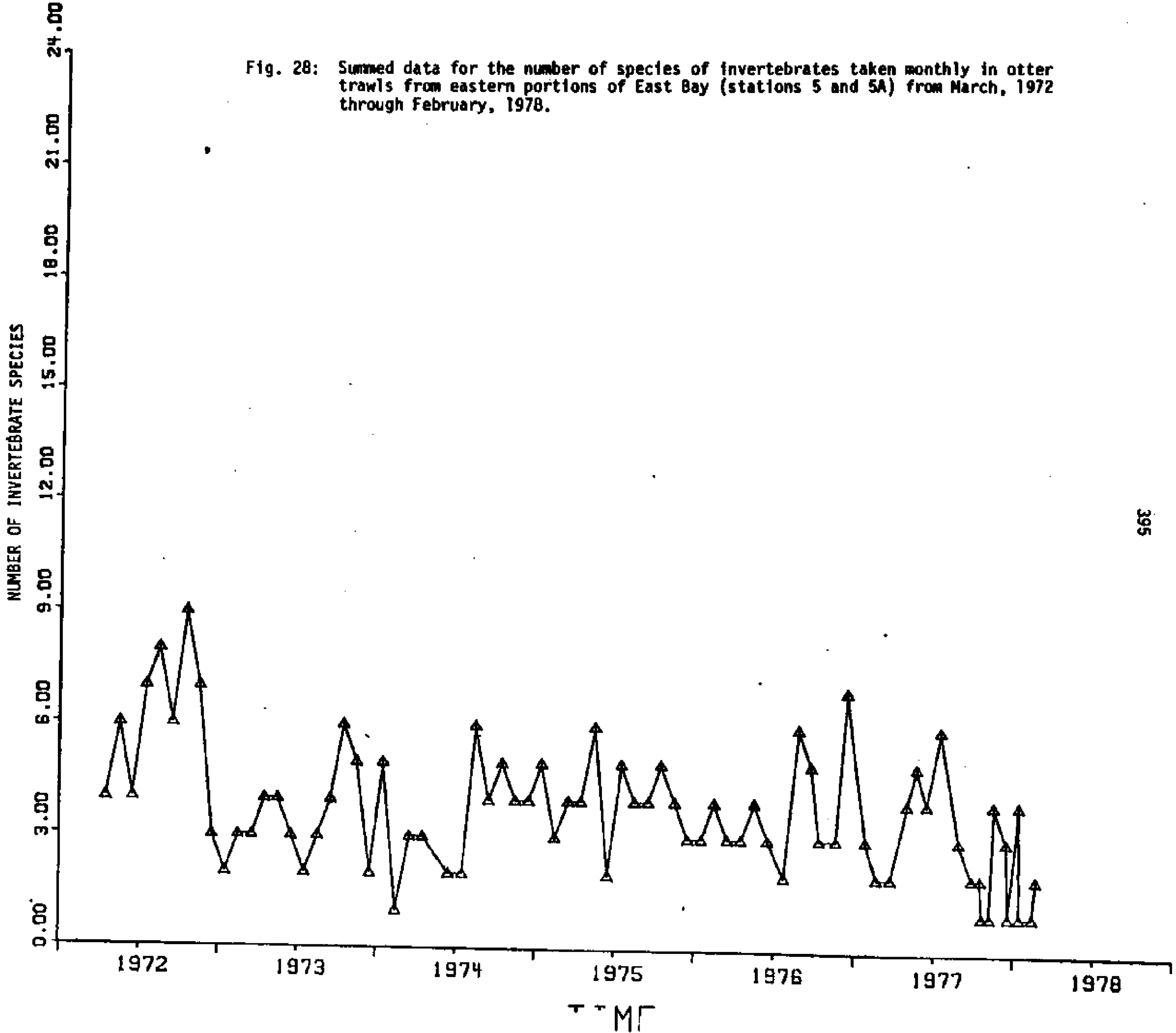


Fig. 28: Summed data for the number of species of invertebrates taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

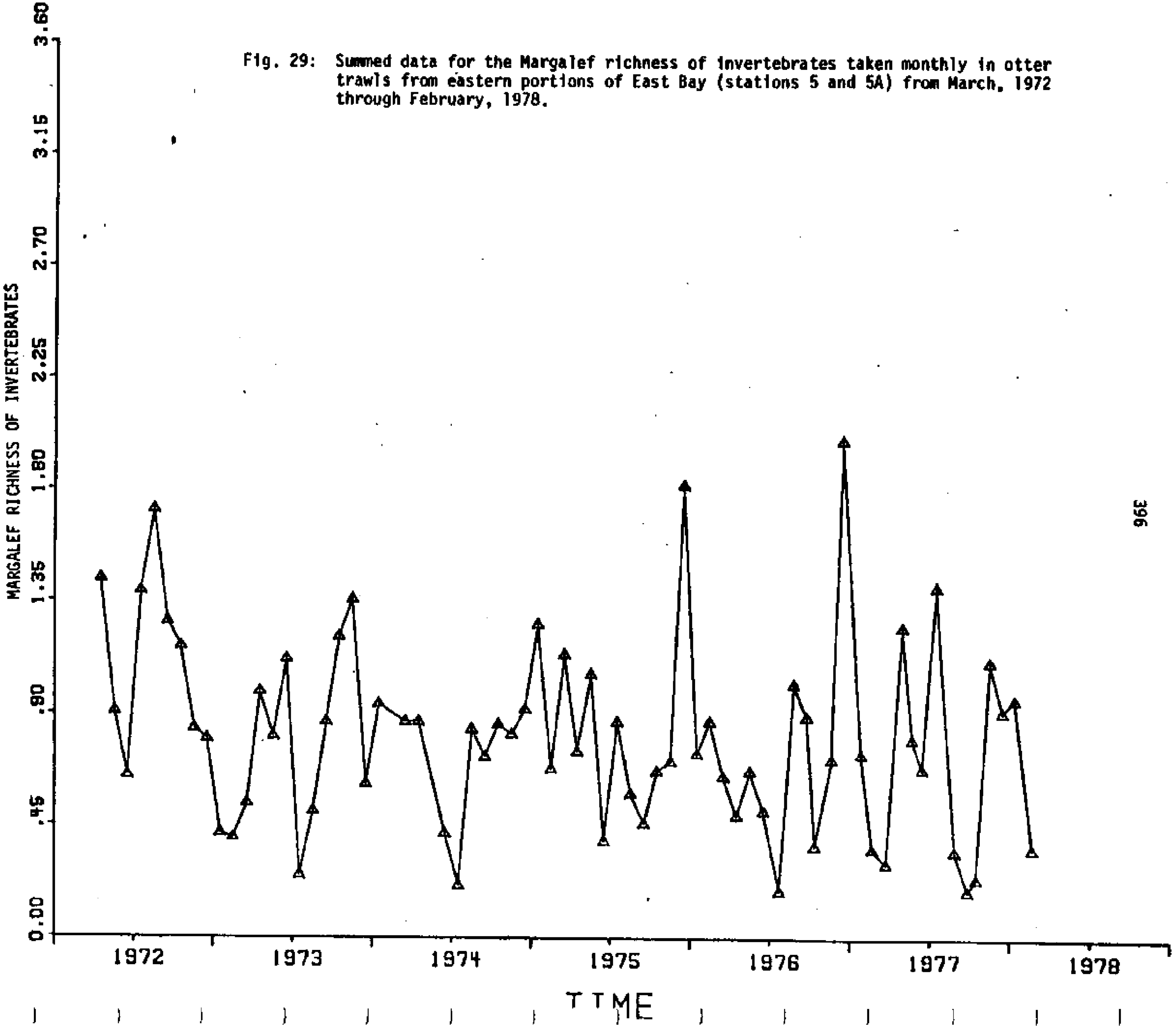


Fig. 29: Summed data for the Margalef richness of invertebrates taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

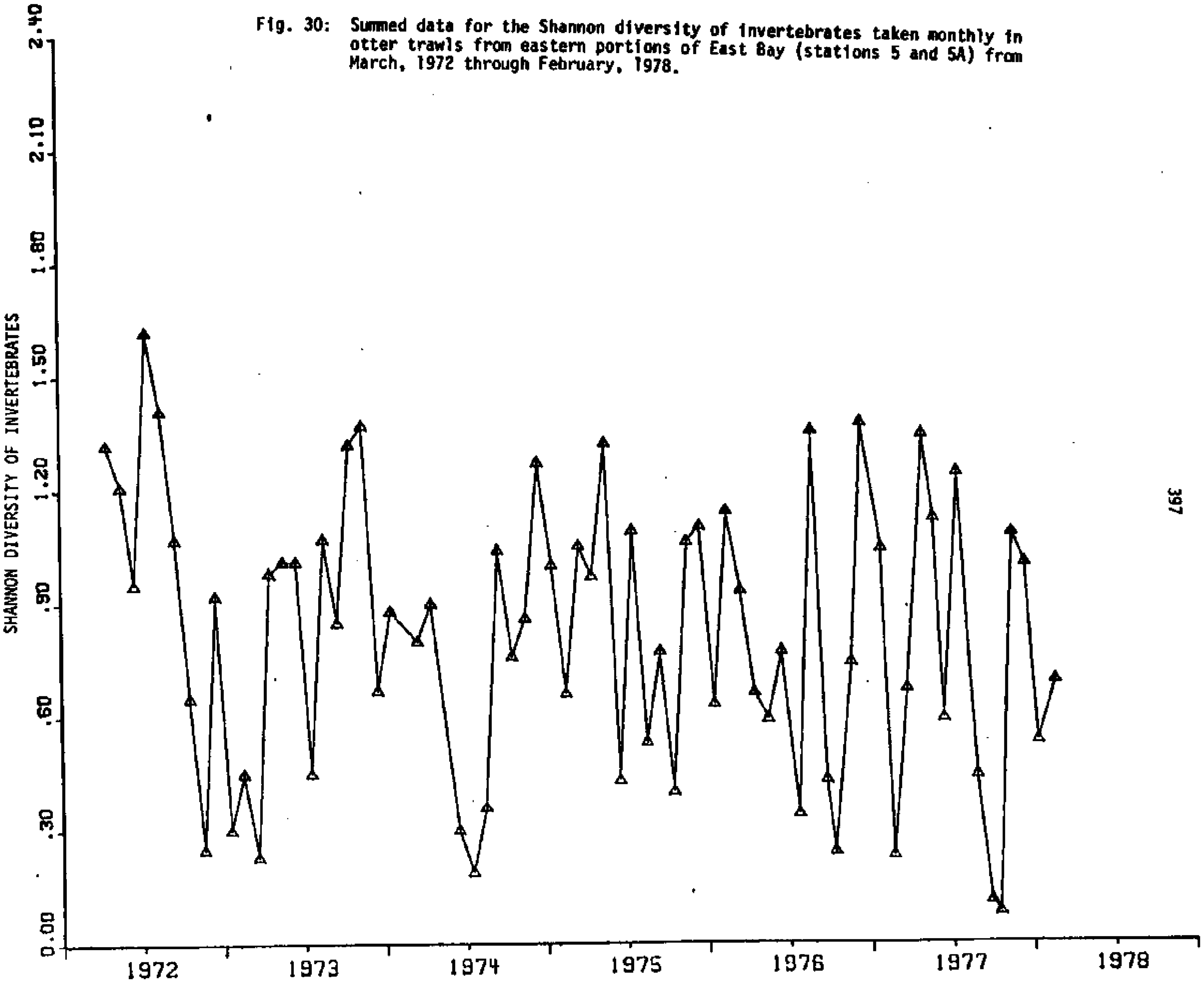
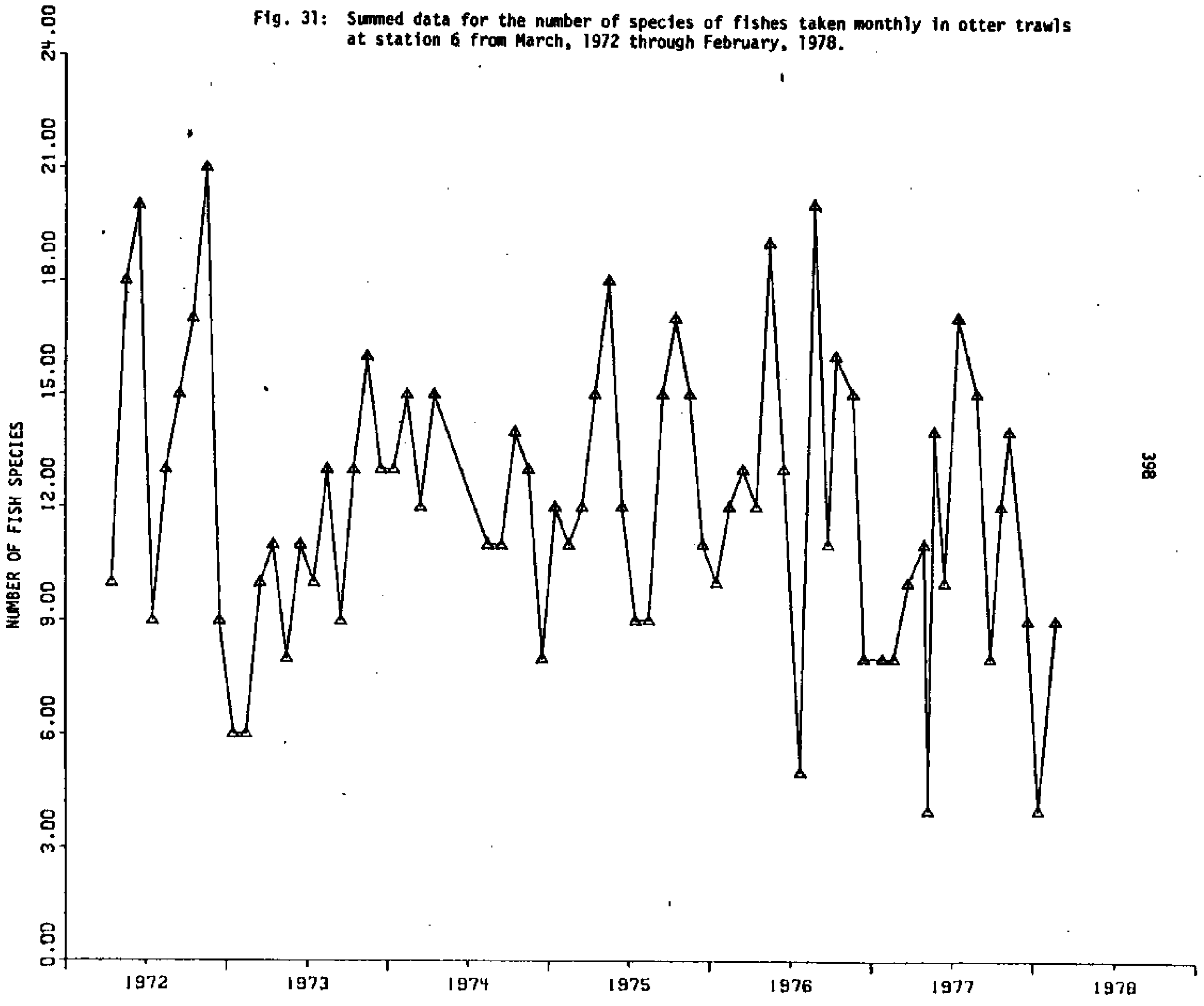


Fig. 30: Summed data for the Shannon diversity of invertebrates taken monthly in otter trawls from eastern portions of East Bay (stations 5 and 5A) from March, 1972 through February, 1978.

Fig. 31: Summed data for the number of species of fishes taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.



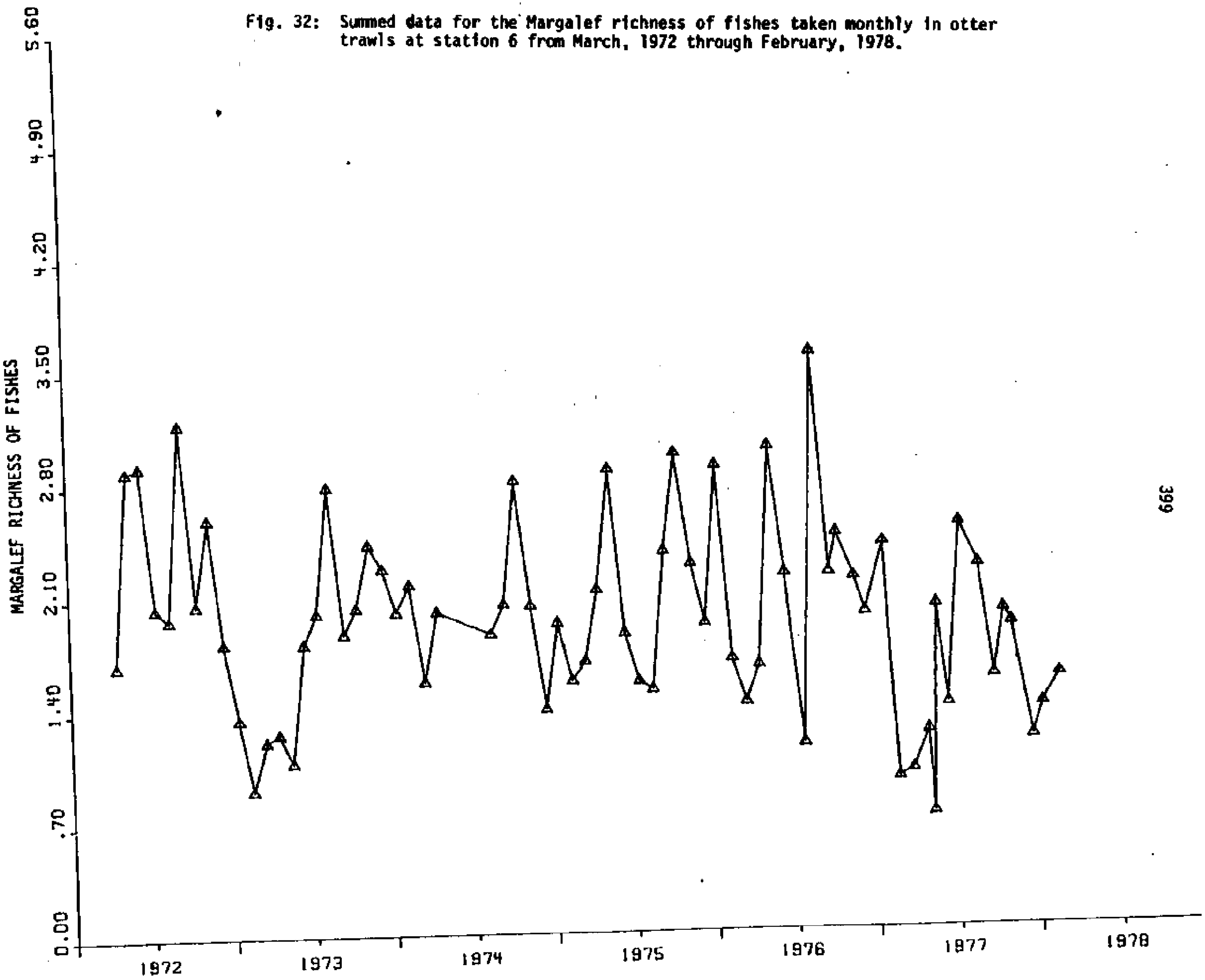


Fig. 32: Summed data for the Margalef richness of fishes taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.

Fig. 33: Summed data for the Shannon diversity of fishes taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.

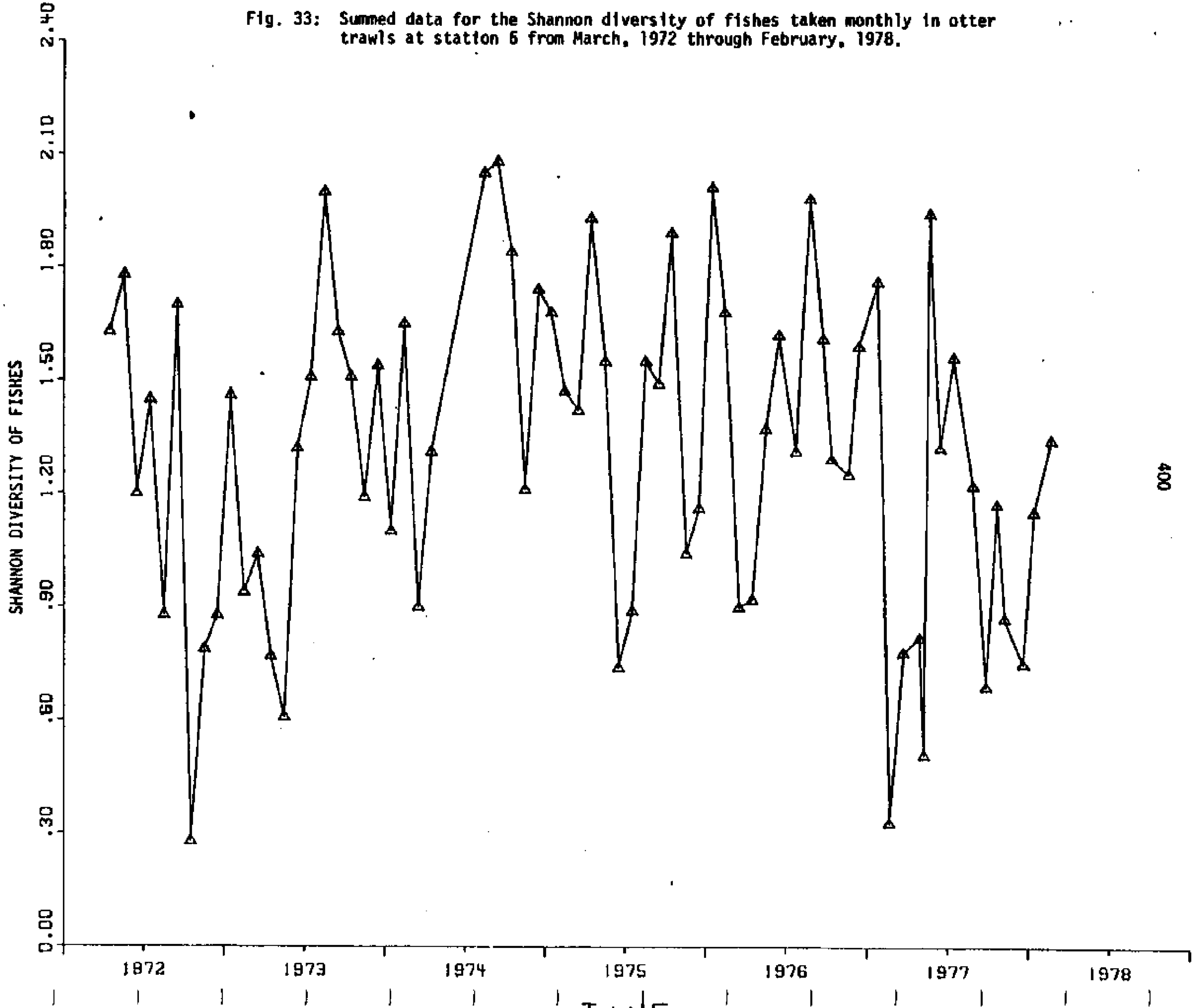


Fig. 34: Summed data for the number of species of fishes taken monthly in otter trawls at station 5 from March, 1972 through February, 1978.

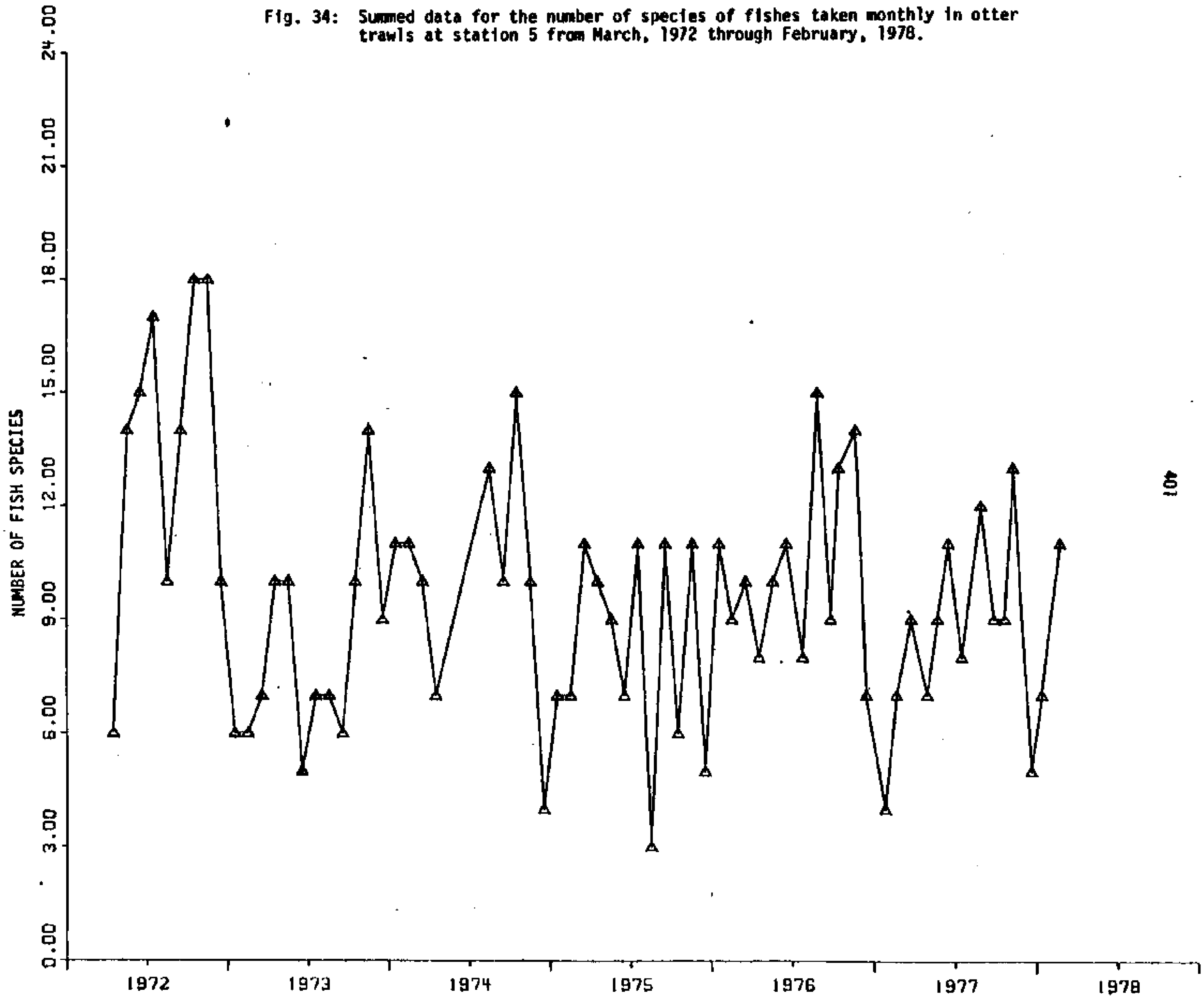
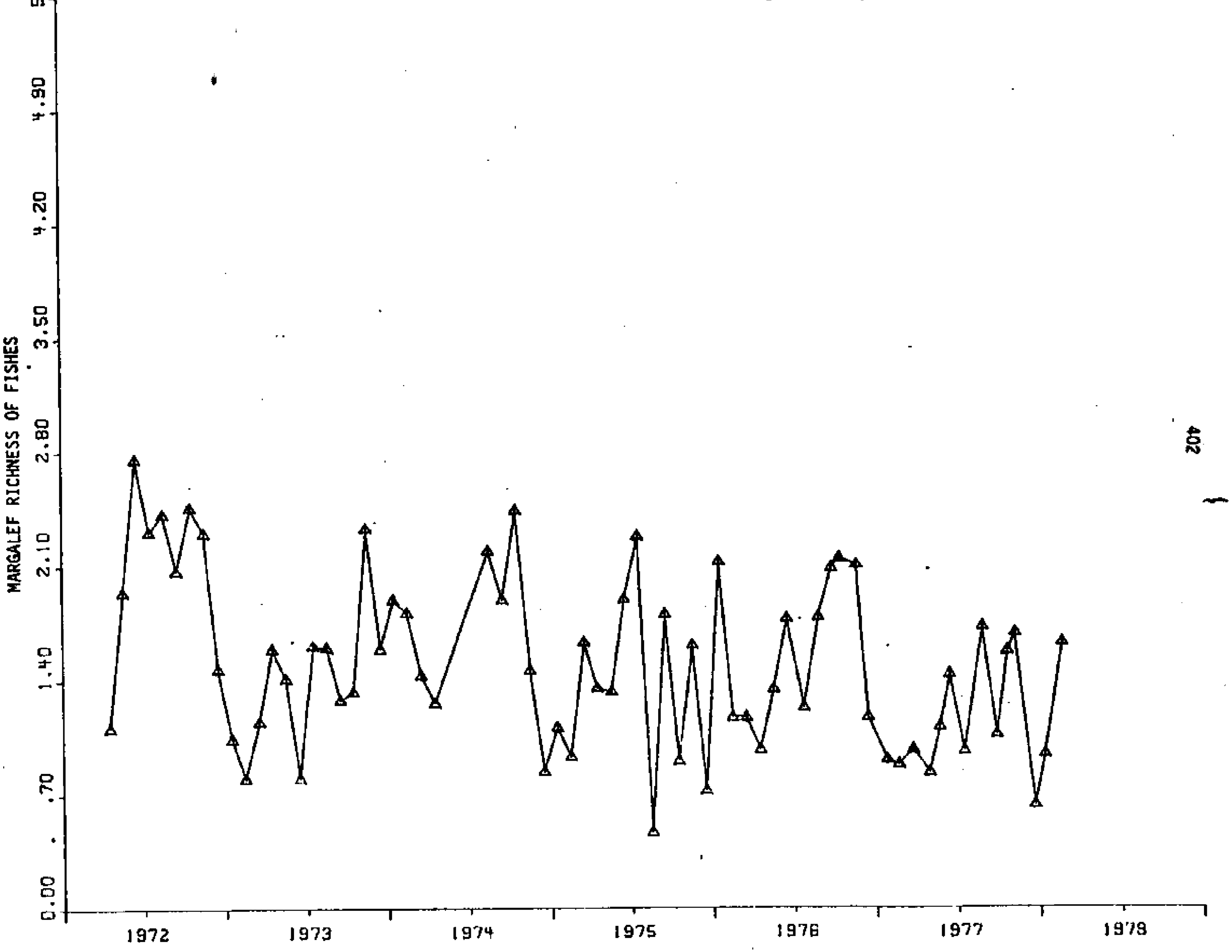


Fig. 35: Summed data for the Margalef richness of fishes taken monthly in otter trawls at station 5 from March, 1972, through February, 1978.



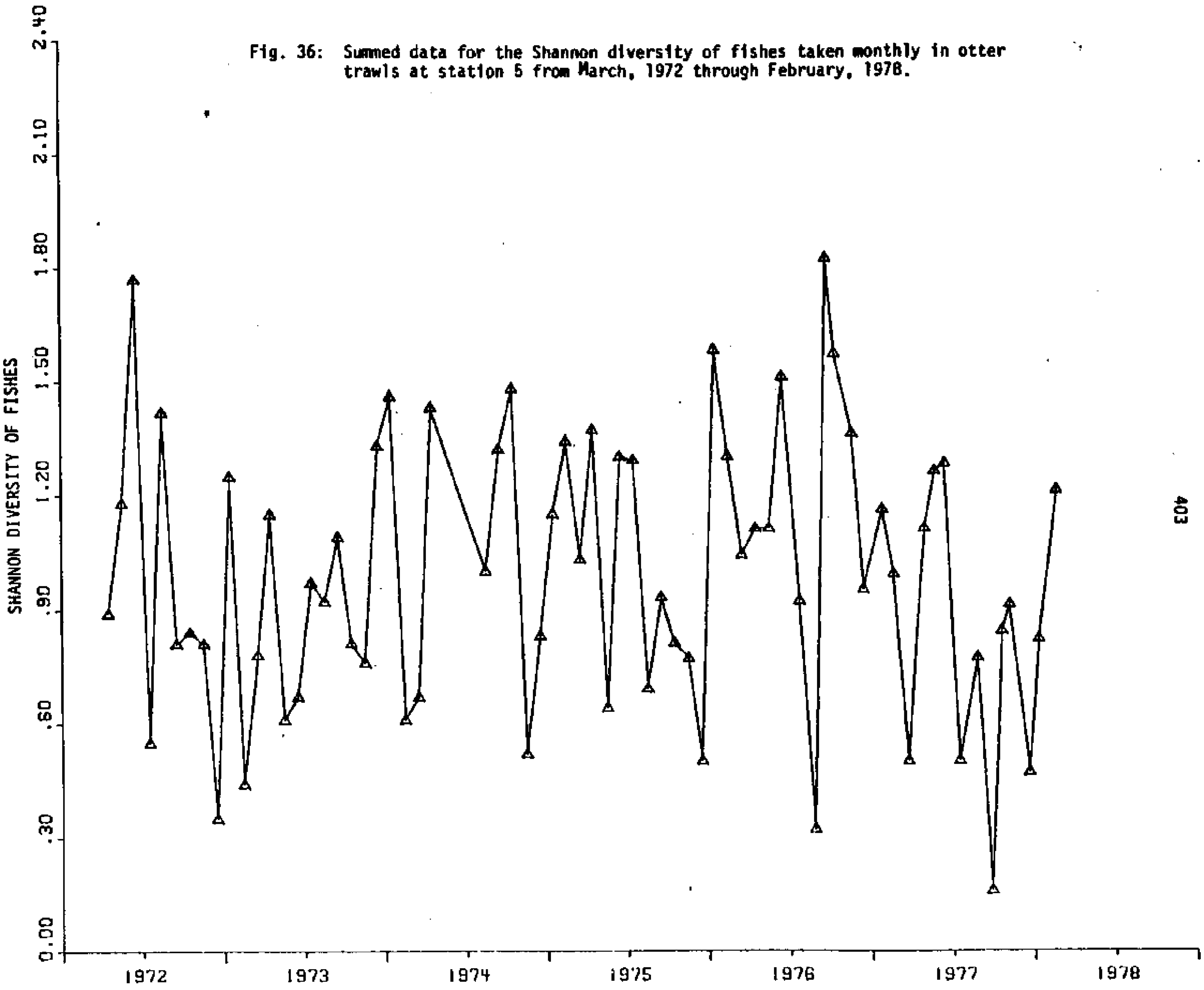


Fig. 36: Summed data for the Shannon diversity of fishes taken monthly in otter trawls at station 5 from March, 1972 through February, 1978.

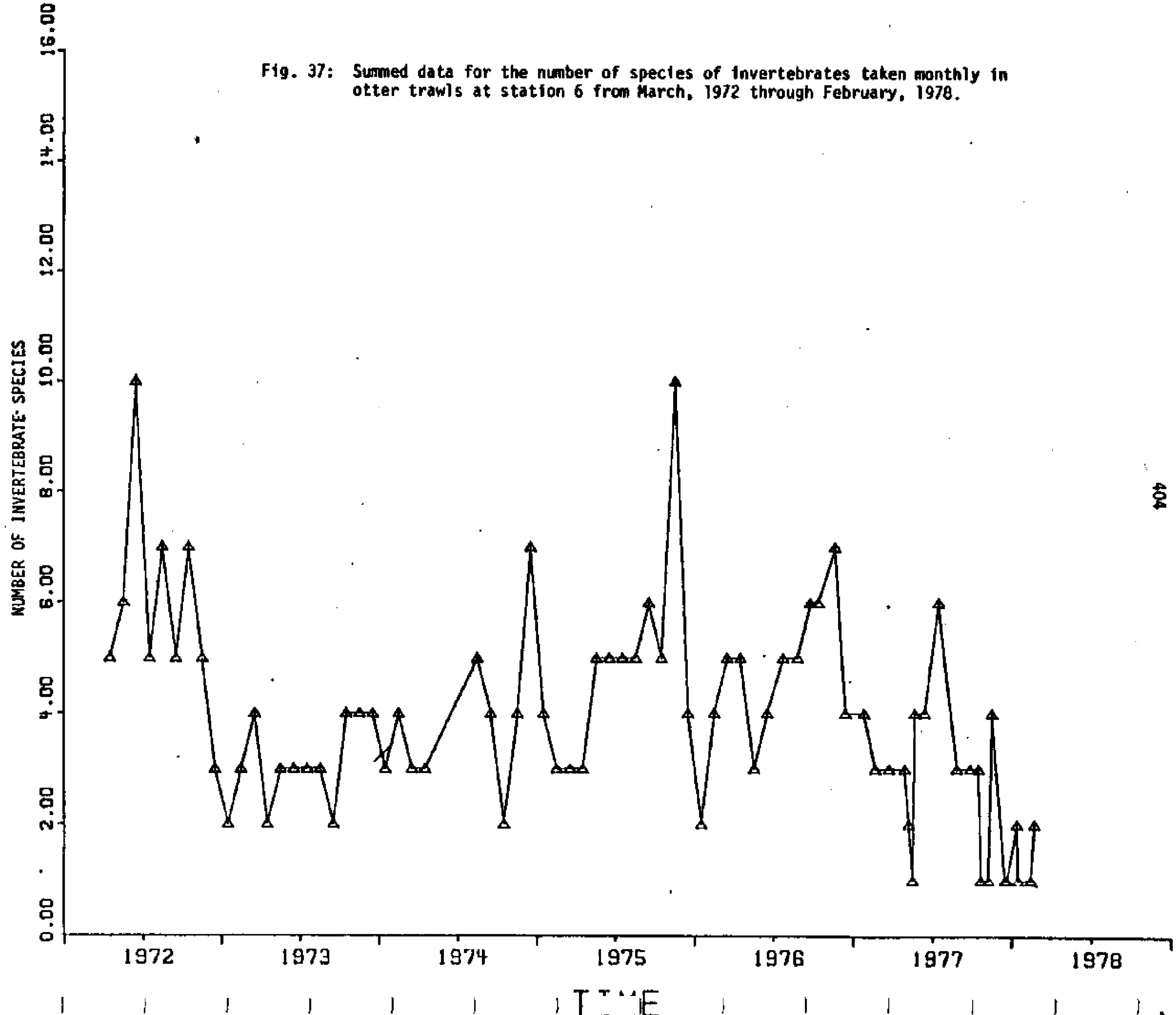


Fig. 37: Summed data for the number of species of invertebrates taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.

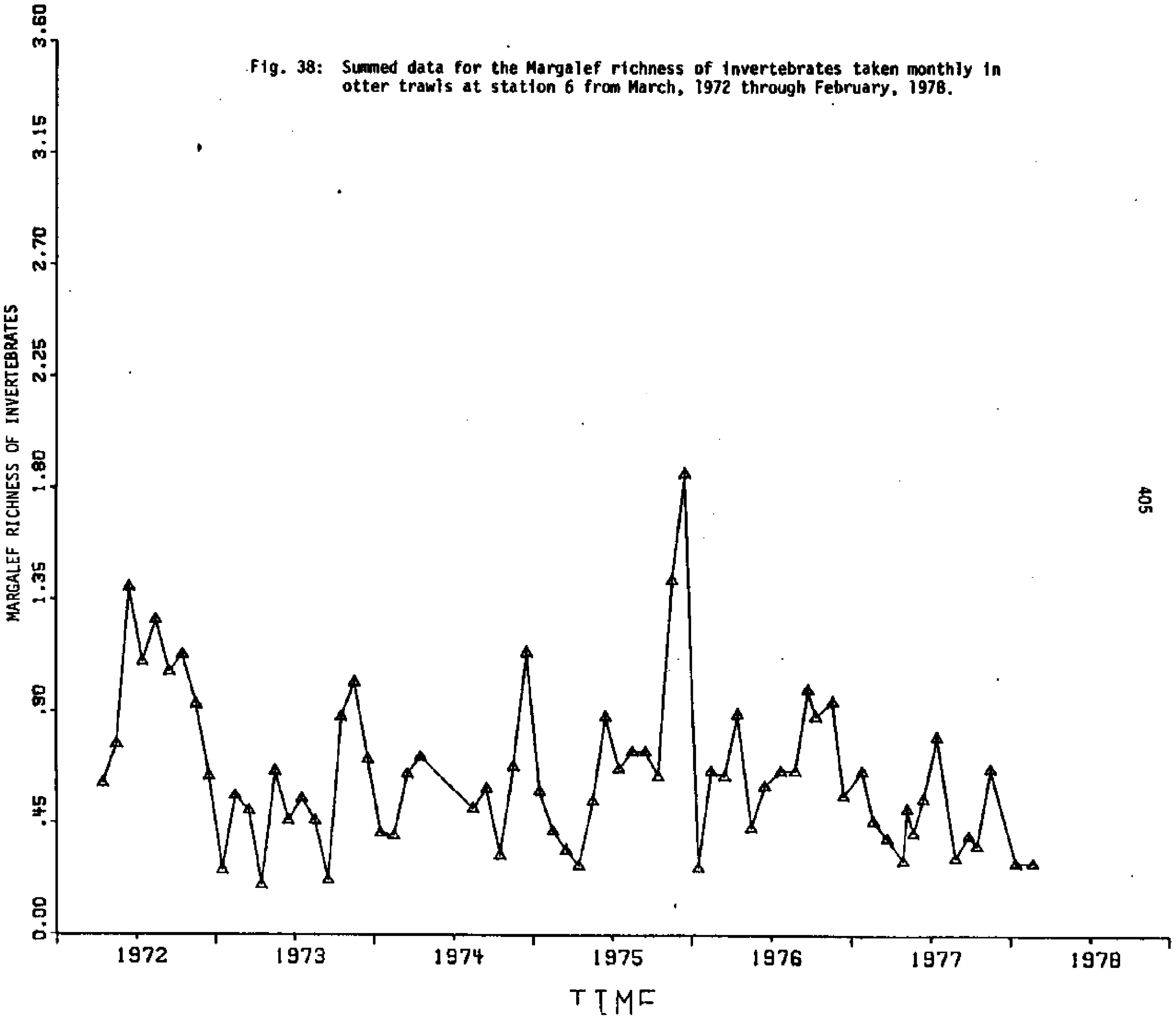


Fig. 38: Summed data for the Margalef richness of invertebrates taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.

Fig. 39: Summed data for the Shannon diversity of invertebrates taken monthly in otter trawls at station 6 from March, 1972 through February, 1978.

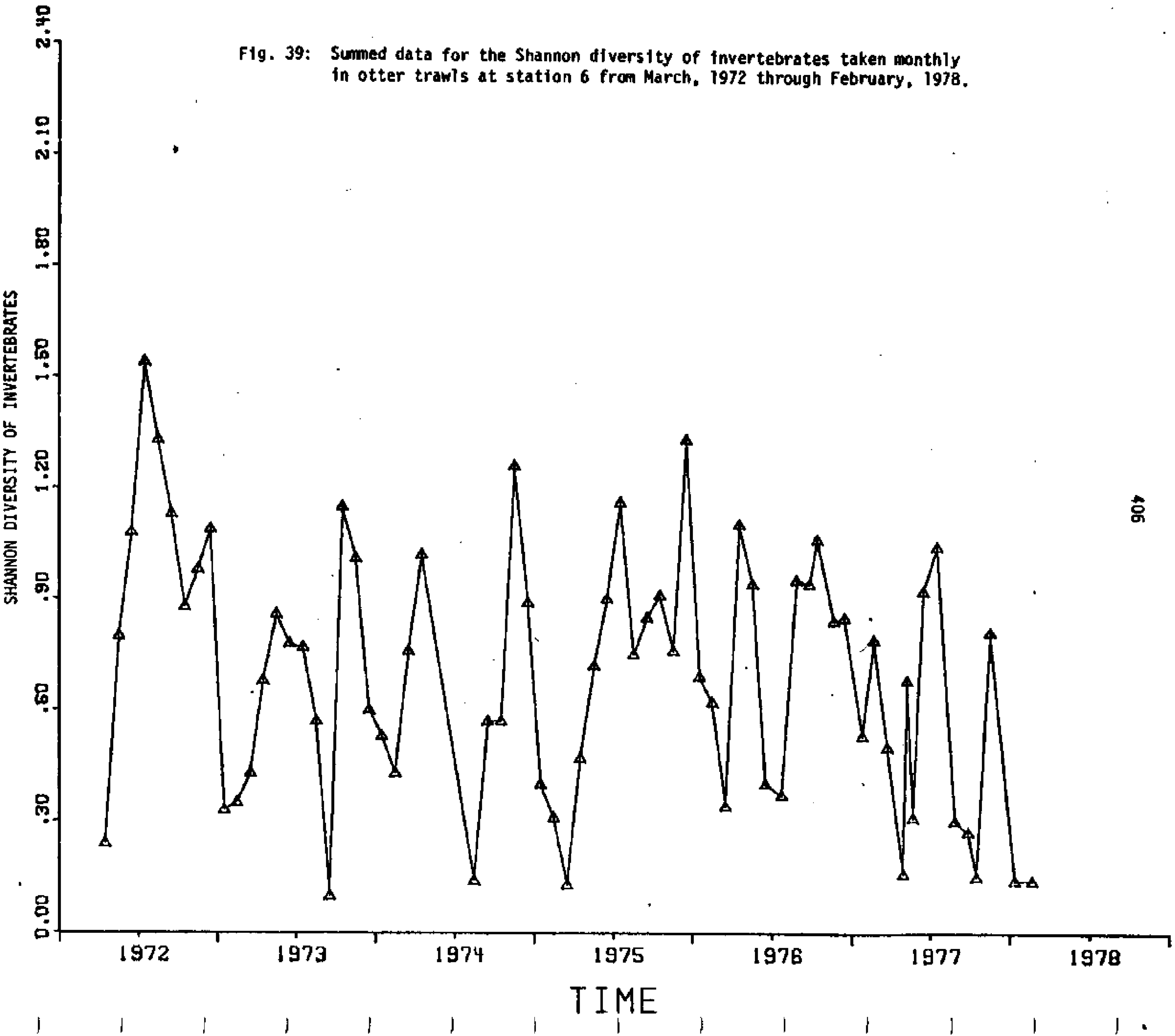
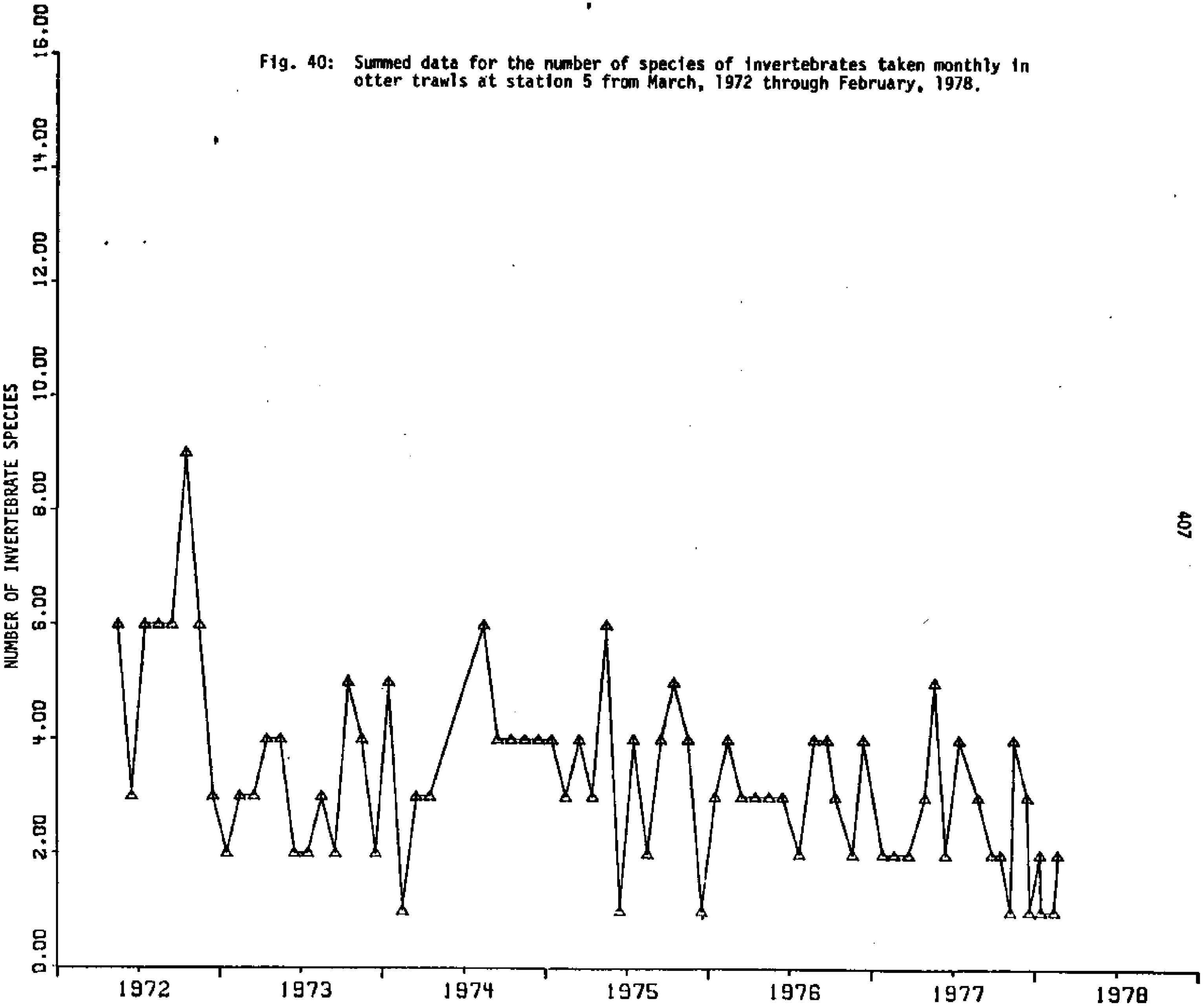


Fig. 40: Summed data for the number of species of invertebrates taken monthly in otter trawls at station 5 from March, 1972 through February, 1978.



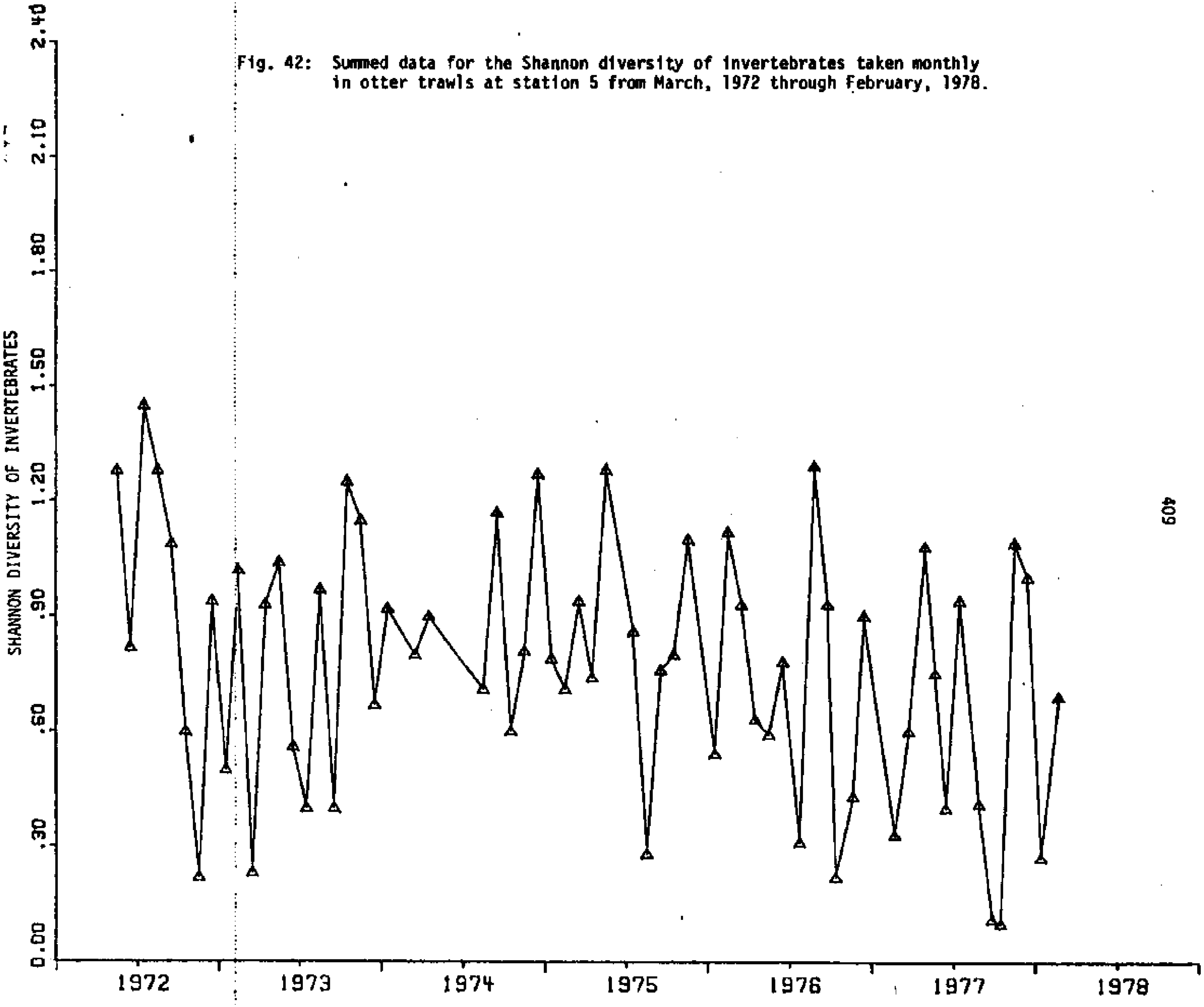


Fig. 42: Summed data for the Shannon diversity of invertebrates taken monthly in otter trawls at station 5 from March, 1972 through February, 1978.