



The L-31E Surface Water Rediversion Pilot Project: Phase 1 Final Report

Submitted by:

Southeast Environmental Research Program

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Section A: Project Objectives, Research Design, and Treatment History

A-1. *Project objectives.*

The Biscayne Bay SWIM Plan (SFWMD 1994) recommended restoring wetland habitats and mitigating the effects of canal point discharges of freshwater. To that end the South Florida Water Management District, in collaboration with the National Park Service and Florida International University, initiated a demonstration project in 1993 to redistribute some freshwater flow from the South Dade canal system to mangrove wetlands fringing the southwestern Bay. The ultimate objective of the project was to provide guidance that would allow managers to recreate a more natural sheet flow to Biscayne National Park, thereby reducing the effects of sudden releases that now occur through existing coastal structures. A secondary objective was to restore, to the degree feasible, the original coastal ecosystems that prevailed prior to construction of the canal system.

Throughout the southern Biscayne Bay watershed, existing coastal wetland communities have been cut off from sheet flow for decades. Mangrove species are quite capable of functioning in intermediate salinities, but the effects of increasing quantities of fresh water, nutrients, and possibly contaminants on ecosystem function is unknown. Better understanding of ecosystem responses to small-scale releases is necessary before extending diversions of canal water to other South Dade coastal wetlands. This knowledge should include an understanding of the effects of diversion on: (a) swamp and nearshore water chemistry and hydrology; (b) soil development and microbiological processes; (c) wetland vegetation composition, structure, and production; (d) abundance of epiphytic and epibenthic invertebrates; (e) zonation, production, and phenology of primary producers in the nearshore environment; and (f) exchanges between nearshore and mangrove ecosystems. These responses should be understood in a historical context, and within the context of the current South Dade landscape as a whole. It is also recognized that the character of treatment responses is frequently complicated by ecosystem responses to periodic natural disturbances.

A team from Florida International University, Southeast Environmental Research Program (SERP) was chosen to carry out the monitoring activities associated with the demonstration project. The area chosen for the project was the coastal mangrove ecosystem bounded by the Mowry, L-31E, and Military Canals on the south, west, and north, respectively, as well as adjacent portions of Biscayne Bay (**Figure A-1-1**). Before establishing the monitoring design, FIU (a) reviewed all ground and surface water quality data available from the general area (described in **Section B-1**), (b) developed maps representing current and historical vegetation of the study area and adjacent lands, based on existing aerial photos (**Section C-1**), (c) sampled and described wetland soils, emphasizing a coastal gradient that prevailed throughout the area (**Section E**), and (d) determined the suitability of alternative study units by completing detailed topographic surveys (**Section A-2**).

A-3. *The redirection treatment.*

The hydrologic purposes of the redirection treatment are to restore a more natural temporal and spatial distribution of fresh water to Biscayne Bay, to decrease urban and agricultural nutrient load while increasing export of coastal wetland detritus and nutrients, and to reestablish an estuarine zone with more natural salinity levels and seasonality. If the pilot project is successful, the potential exists to extend the redistribution of fresh water from other southern Biscayne Bay canals through coastal wetlands. For the project to meet its goals, the delivery system must be flexible and easily controlled, must deliver enough water to significantly reduce canal discharge (when projected to the entire area), and must prevent salt water intrusion.

The Mowry Canal is the ultimate source of freshwater available for redistribution to the Treatment Dwarf basin (**Figure A-1-1**). A considerable amount of water is available for redistribution; the mean annual discharge from the Mowry Canal is approximately 91,330 acre ft. However, the availability of water varies seasonally from a mean May discharge of 2,835 to a mean July discharge of 12,568 acre feet. A three-foot diameter culvert connects the Mowry Canal to the L-31E (Delivery) Canal approximately 60 m west of Water Control Structure S-20F. During periods of rapid dropping or rising water level in the Mowry Canal, this culvert impedes free flow into or out of the L-31E Canal. The delivery system from the L-31E Canal into the Treatment area consists of two three-foot diameter tubular culverts which deliver water into spreader canals at the eastern foot of the L-31E levee. The two culverts were located on either side of a modest topographic ridge which divides the Treatment block. The much smaller northern segment of the Treatment Dwarf basin drains into a boundary drainage ditch or small tidal creek to the north. The southern segment --- the site of all monitoring activities in the Treatment block --- drains through a single tidal creek to Biscayne Bay. Each culvert has an end flap gate on the downstream side to minimize salt-water intrusion. In addition, each culvert delivers into its own spreader canal. The two spreader canals are separated by 10 m of undisturbed marsh at the topographic ridge and are approximately 2 m wide and 40 cm deep.

Design criteria were based on an average head difference between the marsh surface at the foot of the levee and the Mowry Canal of 0.8 ft (SFWMD 1994). Based upon this head difference and a design conveyance of two 36" diameter culverts, a discharge of 29 fps per culvert was expected about 80% of the time, or approximately 33,500 acre-feet annually. Several factors arose that caused actual discharges to be considerably less than predicted discharges. First, the tubular culverts were fitted with a weir structure with an adjustable crest. The weir structure permitted flow only through a maximum of about 33 % of the cross sectional area of the circular tube. In addition, it was necessary to set the weir crest at least two inches above consistent high water stage on the downstream side in order to maintain a proper nap for using discharge tables. Another problem was in the assumptions underlying estimation of average head difference. The Mowry Canal stage is managed by season unless untimely rainfall events require additional emergency release. Between May 15 and Oct 15 the target stage is 2'1" and between Oct 16 and May the target stage is 1'5". During the rainy season when water in the Mowry Canal rises above 2'1" the S-20F gates open and water is discharged into Biscayne Bay. When the stage drops to 1'7" the gates close. The weir crest height was initially set at 1'7". After two days of

Section B: Biogeochemistry and Nutrient Dynamics

B-1. *Historical water and sediment quality.*

Twenty-one water and sediment quality studies with direct relevance to the L-31E demonstration project were reviewed and submitted to the SFWMD as a special report in May 1994. The conclusions from the report are summarized below.

1. Southern Biscayne Bay water quality is good, especially when compared to source canal water quality. Sample station arrays from point sources along the western shore eastward into the open Bay reveal that concentrations of pollutants currently decrease rapidly seaward to levels below minimum detection for many deleterious chemicals. Rapid mixing of canal waters with Bay waters, and short Biscayne Bay water renewal time appear to prevent pollutant concentration buildup.

Pollutant levels in Biscayne Bay sediments are higher in some cases than in surface water, but in general they also decrease rapidly offshore. This indicates that pollutants passing through the Bay system do so quickly enough to prevent accumulation in the soils.

2. Each canal system has its "signature" water quality problem. Canal discharges contain different levels of specific pollutants depending on the major land use activities in the respective drainage basins. For example, household and yard chemicals and septic tank pollutants are most abundant in canals that drain predominantly residential areas. Other canals, such as the Princeton Canal, drain agricultural areas primarily, while yet others drain basins with industrial and hydrocarbon pollutants. The L-31E study area lies between the Military and Mowry Canals. The Military Canal has higher levels of many pollutants than does Mowry Canal, although Mowry Canal is nitrate-rich due to agricultural activity. Pollutant differences such as these complicate water treatment planning applied on a Biscayne Bay-wide scale.

3. High levels of certain pollutants are found as chronic problems in some canal systems, while others occur episodically. With a few exceptions, levels are usually below PEL's (when PEL's have been determined). Occasional or seasonal high levels might occur due to concentration processes during the dry season or in association with rainfall runoff events. Fresh source material at area landfills may also cause high variability in levels at one or more stations. Trace metals in particular may vary considerably within the same area: one study near the South Dade Landfill reported mercury levels above EPA guidelines, while a second reported no detectable mercury at all.

4. Comparison of water or sediment quality determined during different seasons, with different methodologies, at different sites, and with different research objectives in mind is necessarily difficult. It is perhaps most useful in such cases to emphasize the range of values rather than the mean or median values from each report.

Coefficients of variation for N+N, SRP, and all four trace metals exceeded 0.5, while other measured variables were relatively constant from day to day. This daily variation should be kept in mind when considering the monthly patterns discussed below.

Measurable levels of iron were present in most water samples, but more often than not, concentrations of copper, lead, and zinc were below detection limits (**Table B-2-2**). Copper was rarely detected in swamp surface or pore water samples, and never in Canal or Bay surface water. The frequency of measurable levels of lead, zinc, and iron were generally in the order: swamp pore waters > swamp surface waters > Canal or Bay surface waters.

Table B-2-2: Percent of surveys in which concentrations of trace metals exceeded detection limits (Cu, 20 $\mu\text{g/L}$; Pb, 5 $\mu\text{g/L}$; Zn, 10 $\mu\text{g/L}$; Fe, 50 $\mu\text{g/L}$), June 1995 - August 1998.

Site	Metal			
	Copper	Lead	Zinc	Iron
Delivery Canal surface water (DC)	0	5	25	70
Biscayne Bay surface water				
Treatment (TBB)	0	20	10	55
Control (CBB)	0	10	5	40
Swamp surface water				
Treatment Dwarf (TD)	5	21	21	74
Control Dwarf (CD)	0	20	40	60
Treatment Fringe (TF)	7	33	27	87
Control Fringe (CF)	13	27	13	80
Swamp pore water				
Treatment Dwarf (TDS)	0	37	53	100
Control Dwarf (CDS)	6	47	41	100
Treatment Fringe (TFS)	5	47	53	89
Control Fringe (CFS)	0	44	31	94

When present in detectable amounts, concentrations of metals were likewise lowest in Canal and Bay waters, and higher in swamp pore waters than surface waters. This pattern is illustrated for iron in **Figure B-2-1**. Iron is an essential micronutrient, but its primary ecological significance may lie in its impact on the availability of phosphorus (e.g., Chambers and Odum 1990). In anaerobic conditions the ferrous ion forms soluble compounds with phosphorus, but in aerobic conditions the ferric ion has a strong capacity to bind P in insoluble forms.

Because of their association with commonly proposed mechanisms regarding controls on productivity in mangrove ecosystems, specific conductivity and concentrations of total nitrogen and phosphorus in ambient waters are of special relevance to the L-31E experimental redirection. Means and standard deviations for these parameters in pore waters of the study area are listed below (Table B-2-4).

Table B-2-4: Mean values of three water quality parameters in swamp porewater at four sites in the L-31E study area. Based on samples collected monthly between July 1995 and August 1997. Standard deviations are in parentheses.

Site	Water Quality Parameter		
	Total N (μM)	Total P (μM)	Conductivity (mS/cm/s)
Treatment Dwarf (TDS)	67.5 (22.3)	0.71 (0.49)	30.0 (9.8)
Control Dwarf (CDS)	180.2 (35.1)	1.29 (1.25)	31.6 (4.6)
Treatment Fringe (TFS)	116.6 (68.3)	5.27 (5.24)	30.9 (9.4)
Control Fringe (CFS)	113.0 (35.1)	6.66 (2.95)	31.5 (7.1)

Pore water specific conductivity did not differ among sites, but differences in both total phosphorus and total nitrogen were observed. Concentrations of nitrogen were lowest in the Treatment Dwarf and highest in the Control Dwarf site, with intermediate concentrations in the two coastal soils. In contrast, total-P was uniformly higher in coastal Fringe sites than interior sites.

The distinct positioning of Delivery Canal and Bay waters in relation to swamp surface and pore waters in **Figure B-2-2** may provide a useful marker of temporal trends associated with the experimental delivery. A physical mixing model predicts that periodic inundation of the swamp by these sources --- due either to tides or to the experimental treatment itself --- will force sample position within the ordination diagram downward and to the left. Deviations from this prediction may be interpreted as signs of (a) limited treatment duration, (b) *in situ* biological or chemical transformations within the mangrove ecosystem itself, or (c) rapid exchange of water with the adjacent marine system. Our data provides a modest indication that surface water quality in the Treatment Dwarf sub-block was affected by the six weeks of delivery in 1997 (**Figure B-2-3, B**), but this brief treatment period did not result in parallel changes in pore water characteristics (**Figure B-2-3, A**). Possibly, much of the added nitrogen was taken up in a narrow zone immediately adjacent to the point of delivery, and therefore did not reach our sampling station in the center of the basin during the early stages of the project. During the next two years, the more prolonged treatment and response period should allow us to get a better handle on the fate of nutrients delivered into the system.

phosphorus are especially interesting, in that leaf element concentrations seem to track pore water concentrations (Table B-2-4) quite closely.

Table B-3-1: Mean composition of several elements in red mangrove leaves, March 1996 - January 1998. Concentrations are in % for N and P, and in mg/kg for the four trace metals.

SITE	ELEMENT					
	Nitrogen	Phosphorus	Copper	Iron	Lead	Zinc
Treatment Dwarf	1.29	0.083	6.12	15.84	0.58	5.25
Treatment Fringe	1.36	0.095	5.07	21.12	0.29	6.66
Control Dwarf	1.44	0.080	3.43	16.63	0.47	6.00
Control Fringe	1.51	0.108	4.34	23.37	0.35	6.53
Site effect ?	$p<0.05$	$p<0.05$	$p<0.05$	$p<0.05$	n.s.	n.s.
Block contrast?	$p<0.05$	n.s.	$p<0.05$	n.s.	-	-
Zone contrast?	n.s.	$p<0.05$	n.s.	$p<0.05$	-	-

B-4. Sediment alkaline phosphatase activity as an index of available phosphorus.

Data presented above demonstrated a strong spatial association between apparent ecosystem productivity, as reflected in existing stand structure (described in detail in Section C), and phosphorus content in (a) pore water (Section B-2), and (b) mangrove leaf tissue (Section B-3). The hypothesis that growth may be limited by phosphorus availability in mangrove ecosystems has been discussed by others (e.g., Feller 1995). We examined the hypothesis further through studies of soil phosphatase activity over a two-year period, at sites distributed throughout the L-31E study area.

Inducible enzyme systems are good candidates for environmental monitoring, since they respond rapidly to changes in the environment. Phosphatases (acid and alkaline) are inducible enzymes in plants, algae, and microfauna which are produced in response to an organism's need for phosphate. They may be induced and released (or retained in the extracellular space) as a single enzyme response, or may be produced and released in association with other systemic responses such as induced senescence in plants (Morita *et al.* 1996). Phosphatases have been found in microfauna as well as algae and marine macrophytes (Lapointe and O'Connell 1989; Hernandez *et al.*, 1992, 1993, 1994). Both microorganisms and plants have the capacity to release phosphatase to the external environment, but it has been suggested (Spier and Ross, 1978) that microorganisms supply most of the soil enzyme activity because of larger biomass, high metabolic rate, and short lifetimes which facilitate release of relatively large amounts of enzyme. However, associated plants, animals and bacteria may also benefit from this extracellular release.

APASE activity throughout the study area exhibited very strong temporal patterns that were largely consistent among sites (**Figure B-4-2** and **B-4-3**). However, neither the peaks observed in May - July of 1996 and November - January 1998, nor the very strong trough of February 1997, are indicative of regular seasonal patterns or the diversion treatment itself. These temporal patterns are expressed most strongly in the Dwarf forest sites furthest from the coast (50 meter location in **Figure B-4-2**, 50 meter location in **Figure B-4-3**). At present, we can not explain these patterns, beyond suggesting that they are a response to some factor or factors (probably hydrologic and/or climatic) that affect all portions of the study area to some degree.

Our APASE monitoring did not extend far enough into the post-diversion period to indicate what effect the treatment will ultimately have on phosphorus availability. Given the low phosphorus concentration (and N:P ratio) of water in the Delivery Canal (**Table B-2-1**), one might predict that phosphorus deficit --- and hence APASE activity --- will increase in the Treatment Dwarf sub-block as the effects of the diversion reverberate more thoroughly through the mangrove ecosystem.

B-5. $\delta^{15}\text{N}$ study of nitrogen use by *R. mangle*.

In order to gain further insight into nitrogen cycling processes within L-31E mangrove ecosystems, we initiated a one-time survey of variation in leaf $\delta^{15}\text{N}$ within the study area. This research is discussed in more detail in a manuscript recently submitted for publication (Fry et al. unpublished manuscript). In the fall of 1994, *R. mangle* leaves were collected along a transect extending from Biscayne Bay to the L-31E levee. Mature leaves were collected from the crowns of 5-10 individual trees at each location, then composited for %N and $\delta^{15}\text{N}$ analysis. Leaf midribs were removed, dried samples were ground and combusted to nitrogen gas in a Carlo-Erba elemental analyzer, and $\delta^{15}\text{N}$ was determined by continuous flow stable isotope mass spectrometry in the FIU-SERP labs. The results are summarized in **Figure B-5-1**. Leaf %N was highest within 100 meters of Biscayne Bay, peaking at 2% about 50 meters from shore. N content subsequently decreased inland as far as the L-31E levee, where it was less than 1.2%. Leaf $\delta^{15}\text{N}$ was about +7 units immediately adjacent to the Bay, increasing to +11 units at ca 50 meters, then decreasing below 0 units before the levee. $\delta^{15}\text{N}$ of plants growing on the banks of the Mowry and L-31E Canals was nearly +13 units. Such a heavy isotopic signal is typical of canal waters that receive high loading from agricultural fertilizers.

The pattern of $\delta^{15}\text{N}$ in the L-31E swamp fits a model in which leaf $\delta^{15}\text{N}$ is affected jointly by the isotope character of source water and the extent of isotopic fractionation occurring during nitrogen uptake. Mangroves near the Biscayne Bay shoreline are bathed in isotopically heavy Bay waters that are strongly affected by Mowry Canal outflow. Moving inland, this source effect may change, with tidal water representing a smaller portion of the total water budget, and direct precipitation and temporary storage in the interior basins becoming more important. Secondly, isotopic discrimination may change, with strongest fractionation (i.e., lightest isotopic signal) in the interior Dwarf forest. According to current models of plant isotopic fractionation, greatest

Section C: Vegetation

The discussion of L-31E vegetation studies from 1992 through the present is divided into four general subject areas: historical and current vegetation of the study area and adjacent areas, plant community responses to natural disturbance, annual variation in plant community structure and population dynamics, and mangrove water relations

C-1. *Historical and current vegetation of the study area.*

Historical vegetation. Historical photointerpretation was based on aerial photo C-JF 6-29 (Soil Conservation Service, July, 1940), in conjunction with recent groundtruthing, and our current understanding of ecological relationships in the coastal zone. These sources suggest that prior to the construction of the L-31E levee and canal, an east-west coastal gradient from intertidal to supratidal to non-tidal freshwater wetlands encompassed the non-agricultural portions of the study area and immediately surrounding lands (**Figure C-1-1**).

The intertidal portion of the gradient consisted of taller and shorter variants of mangrove forest, appearing black and gray, respectively, on the photo. In 1940, the tallest mangrove trees occupied a very narrow (<200 meters) fringe along the Biscayne Bay shoreline. The **Tall Mangrove Forest** also encroached inland along tidal drainages which, with several exceptions, were confined to a coastal zone considerably less than a kilometer in width. In most portions of the study area, the tall coastal fringe was bounded toward the interior by a dense, shorter-statured mangrove forest. This **Low Mangrove Forest** zone was absent immediately north and south of the current Military Canal, an area dissected by a number of short tidal creeks. In contrast, low mangrove forest appears to have constituted a broad band along several major tidal creeks reaching well into supratidal zones.

Throughout the study area, a light-coloured vegetation zone is evident on the 1940 aerial photo immediately interior to the low mangrove forest, and in broad areas between tidal creeks. Working with the same photo series south and west of Turkey Point, Egler (1952) described a zone (Belt IV) appearing as a distinct white band immediately interior to the mangrove belt (Belts V-VII). Egler's sample plots of 1946 indicated that the vegetation of Belt IV was a sparse, relatively even mixture of sawgrass (*Cladium jamaicense*), spikerush (*Eleocharis cellulosa*), and red mangrove (*Rhizophora mangle*). In this context, the white appearance of Egler's Belt IV is probably indicative of reflectance from the white marl substrate, or from a continuous periphyton mat able to form in the wet, open conditions of this graminoid-dominated ecosystem. We suggest that prior to construction of the L-31E levee and canal, a similar **Mixed Graminoid with Red Mangrove** assemblage was also present immediately landward of the mangrove forest zones between Turkey Point and Black Point. However, because of the narrowness of the mangrove belt in this section, the graminoid assemblage may have included a stronger representation of species more typical of supratidal wetlands, e.g., *Distichlis spicata*, *Juncus roemerianus*, and *Spartina spartinae*, than that described by Egler. The relative abundance of these three coastal species in

community were much less than in the taller coastal forests, and resprouting of broken stems has played a more important role in forest recovery. Although the species composition of this assemblage suggests an intertidal position, a wide band of compositionally and structurally typical **low mangrove forest** occurs north of Military Canal and west of the L-31E Canal, outside the influence of daily tides. Finally, **mangrove dwarf**, a structural variant of the modern intertidal landscape that was not identified on the earlier photo, has become well-established throughout the area previously occupied by **mixed graminoid marsh with red mangrove**. **Mangrove dwarf** currently occupies the shallow basins immediately east of the L-31E levee. The dwarf canopy (0.5-2 meters height) is virtually pure red mangrove. White and black mangrove are scattered emergents along the periphery of these basinal communities and in small islands of larger trees which pepper the area, but in the center of the basins they occur primarily as seedlings in subordinate positions amongst the prop roots of the larger red mangroves. Occasional *D. spicata* and *J. roemerianus* clumps are apparently rare relicts of the historical saltmarsh. A photograph from 1965, immediately prior to levee construction but well after drainage of the area, suggests that the beginnings of the transition from open marsh to mangrove scrub predate the L-31E itself, though the levee undoubtedly hastened the process by intercepting freshwater sheetflow entirely.

In direct contrast to the situation described above, the L-31E canal and levee has served as intended to intercept saltwater that had historically reached lands to the west during storms and other high tide events. On this basis, one would predict an increase in the relative abundance of plant species typical of fresh water swamps since 1966. In fact, the prominence of sawgrass or cattails in most of the plant communities west of the L-31E levee (i.e., **Sawgrass marsh, Cattail marsh, Mixed Broadleaf swamp, and Australian Pine swamp**) suggest the fresh water nature of these ecosystems. In the last-mentioned community, tall stands of sawgrass occur beneath an open canopy of *Casuarina spp.*, occasional buttonwood stems, and little else. These communities are largely confined to unplowed lands previously occupied by Buttonwood Woodland. **Mixed Broadleaf swamp** is also a relatively open forest with sawgrass in mixture with buttonwood, wax myrtle, and the mangroves, especially *L. racemosa*. Open **Sawgrass** and **Cattail** (*Typha spp.*) **Marsh** are relatively restricted in area, drainage and the invasion of exotic trees have both contributed to the apparent increase in the overall coverage of woody vegetation. **Brazilian pepper** (*Schinus terebinthifolius*) **forests** are monocultures which occupy abandoned agricultural lands in the western portions of the study area. Despite the preponderance of these three invasive exotics west of the L-31E, their absence in undisturbed mangrove forests east of the levee is significant. However, **Disturbed lands** throughout the area, especially the levees to the omnipresent mosquito ditches, are almost without exception exotic-dominated. Exotic removal which does not return the land to its original contour and hydrologic regime will ultimately be a fruitless exercise.

Table C-2-1: Estimated canopy height (m) along two levee-to-bay transects, prior to and 10-12 months after Hurricane Andrew. Estimates are based on 1993 point-quarter samples of live stem heights, and the maximum lengths of hurricane-killed trees sampled at the same time.

Meters from Levee	Block 2		Block 4		Meters from Levee	Block 2		Block 4	
	1992	1993	1992	1993		1992	1993	1992	1993
0	2.3	2.3	1.1	1.1	380	1.6	1.6	8.0	1.8
20	0.7	0.7	1.1	1.1	400	1.0	1.0	10.5	2.0
40	0.8	0.8	0.9	0.9	420	2.7	1.1	10.2	1.5
60	0.6	0.6	0.6	0.6	440	6.5	2.0	7.8	2.0
80	0.7	0.7	0.6	0.6	460	11.0	2.1	10.0	2.0
100	1.4	1.4	0.8	0.8	480	15.0	1.1	9.2	1.1
120	0.6	0.6	0.8	0.8	500	11.0	0.8	9.4	0.9
140	0.7	0.7	0.8	0.8	520	14.0	0.6	8.0	0.7
160	0.8	0.8	1.3	1.3	540	19.0	1.1	7.6	2.1
180	0.7	0.7	1.1	1.1	560	14.0	0.3	10.1	0.4
200	0.6	0.6	1.2	1.2	580	16.0	0.3	12.5	0.6
220	0.6	0.6	1.5	1.5	600	17.0	0.6	14.5	0.5
240	0.6	0.6	1.6	1.6	620	14.5	1.2	19.3	2.0
260	0.6	0.6	3.0	1.3	640	Biscayne Bay		19.0	1.6
280	0.7	0.7	1.2	1.2	660			18.0	2.0
300	0.7	0.7	1.4	1.4	680			16.0	0.6
320	1.0	1.0	4.1	1.2	700			12.0	1.3
340	0.8	0.8	5.0	1.4	720			Biscayne Bay	
360	0.9	0.9	1.9	1.6					

If mangrove species differ in their sensitivity to cold temperatures, periodic freezes may exert a strong influence on community composition and stand succession. On the basis of their northern range distributions (Little 1977; Little 1978), we predicted that the three major mangrove associates would decrease in cold-hardiness in the order: *A. germinans* > *R. mangle* > *L. racemosa*. To test this prediction, we established 200-meter transects in five areas of extensive freeze damage between the Mowry and Military Canals. We established sampling stations at 10-meter intervals along each transect, and located the closest live or freeze-killed stem taller than 80 cm of each species, to a maximum distance of 5 meters. We then recorded the total pre-freeze height and the height of the highest live leaf of each tree. Sampling was conducted between February 15 and March 15, 1996, 1-2 months after the freeze. Percent mortality and mean pre- and post-freeze heights are presented by species in Table C-3-1.

Table C-3-1: Mortality and dieback of three mangrove species following the freeze of January 1996. Transect numbers correspond to basins illustrated in **Figure A-1-1**.

	SPECIES						
	Transect	<i>A. germinans</i>		<i>R. mangle</i>		<i>L. racemosa</i>	
Percent survival	1-A	86		95		47	
	1-B	30		45		0	
	2	100		100		70	
	3	0		0		0	
	4	100		30		5	
Mean pre- and post-freeze height (cm)	Transect	pre-	post-	pre-	post-	pre-	post-
	1-A	145	141	130	111	133	109
	1-B	102	59	139	119	132	-
	2	129	124	122	118	130	119
	3	104	-	122	-	129	-
	4	102	92	129	102	128	55

The extent of damage varied locally. For instance, all stems > 80 cm were killed in the portion of Block 3 we surveyed, but damage in the area immediately southward (Block 2) was very slight. Of the three species, *L. racemosa* suffered the highest mortality and generally died back furthest, but *R. mangle* and *A. germinans* did not appear to differ greatly in their response. Since *L. racemosa* and *A. germinans* (but not *R. mangle*) are capable of vigorous resprouting after damage, assessment of the longterm effects of freeze events on mangrove species composition must await further analysis of the recovery process.

a good understanding of background behavior of red mangrove leaf populations in the study area. Results reflecting on site differences in leaf production during 1996-97 are presented in **Section C-5**, and a more comprehensive analysis of the demographic study will follow as leaf cohorts which have completed their life span entirely under post-treatment conditions are incorporated.

C-5. Aboveground biomass and production in Fringe and Dwarf forests

Methods - biomass. The two transects used to describe hurricane-related vegetation changes were also used to estimate Fringe forest biomass in 1992 and 1993. Densities of live *R. mangle*, *L. racemosa*, and *A. germinans* in three height classes (0-60 cm, 60-250 cm, and >250 cm), as well as hurricane-killed trees (>5 cm DBH) of each species, were determined as described above. Mean stem and crown diameter for each species-size category at each point were calculated. Aboveground biomass for these mean individuals were estimated from regression equations developed as described below, and multiplied by density to provide point estimates of biomass per unit area. Similarly, a mean DBH of dead stems was calculated for each species at each point. Pre-hurricane biomass of these large stems was estimated from diameter-based regression equations provided by Fromard (pers. comm.) for *R. mangle*, *A. germinans*, and *L. racemosa* in French Guiana (Fromard et al. 1998). Live aboveground biomass in 1993 was calculated as the mean of twelve point estimates, and pre-hurricane (1992) tree biomass from the mean of point estimates based on dead stems plus living stems > 2.5 meters height.

In April-August 1995, permanent plots were established, on the basis of which mangrove biomass was estimated in 1995 - 1998. Seven plots were in Dwarf forest (four in the Treatment block, three in the Control block), and eight were in the Fringe forest type (four in each block). Construction of a network of boardwalks allowed us to approach and work in the rectangular plots without serious disturbance to the fragile mangrove sediments. Plots were 0.5 meter wide, and ranged from 3 to 10 meters in length, with smaller plots in areas of higher stem density. Each mangrove individual rooted within the plots was tagged and given a grid coordinate, and the following measurements were recorded: total height (TOTHT), height to base of crown (CRWNHT), crown length and width (CRWNL and CRWNW), and stem diameter at 30 cm (D_{30}) and 140 cm (D_{140}) (for stems taller than 40 and 150 cm, respectively). The product $[CRWNHT * CRWNL * CRWNW]$ provided an index of crown volume (CRWNV). Measurements were repeated during the periods September - December 1996, November - December 1997, and November - December 1998.

Biomass regressions were developed for *R. mangle*, *L. racemosa*, and *A. germinans* in the summer of 1997. Regressions were based on relationships between the structural measures listed above and biomass components (trunk, branch, proproot, leaf, and total aboveground biomass) of individuals harvested from adjacent areas which resembled the study plots in vegetation structure. The regression data set included 53 *R. mangle*, 44 *L. racemosa*, and 21 *A. germinans* stems. These individuals encompassed a range of sizes from 40 cm to 6.5 meters in height. 5-14 individuals < 40 cm height of each species were also sampled and weighed. All possible regression models of the form

$$\text{Equation 1: } \text{Leaf Prod'n}_{Yr(x-1)} = \Delta \text{ Standing Crop}_{Yr(x-1)} + \text{Leaf Mortality}_{Yr(x-1)}$$

A major assumption of our method was that the production and mortality rates underlying the first and third terms in Equation 1 were similar whether expressed on a density or biomass basis. In solving Equation 1 for biomass, $\Delta \text{ Standing Crop}_{Yr(x-1)}$ was calculated by applying our biomass regression equations to structural data from censuses of successive years in each plot. $\text{Leaf Mortality}_{Yr(x-1)}$ was calculated as follows:

1. Mean leaf turnover rate (in leaf generations per leaf per year) was calculated for three leaf cohorts (i.e., those tagged in June 1996, December 1996, and May 1997), based on the total number of red mangrove leaves tagged and surviving to each subsequent sampling period within each plot. Survivorship beyond the time when 95% of the tagged cohort had fallen was not considered, as these leaves frequently appeared senescent or non-functional. Leaves were assumed to have been initiated or to have died at the midpoint of the interval prior to observation. Mean cohort longevity (in days) was calculated by integrating the area under the survivorship curve, assuming a linear decrease in leaf density between periods. Turnover rate therefore equalled 365 divided by the mean longevity (Dai and Wiegert 1996). Annual turnover rate on a density basis, T_{dens} , was calculated by weighting the turnover rate of each cohort by its relative contribution to the total leaf population during the year beginning July 1, 1996.
2. We calculated $\text{Biom}_{Yr(x-1)}$, the annual mean standing crop of leaves, as the average of crop estimates for July 1, 1996 and June 30, 1997 in each plot, interpolating between the actual census dates.
3. Equating T_{dens} with T_{biom} (turnover rate on a biomass basis), we calculated leaf mortality as:

$$\text{Equation 2: } \text{Leaf Mortality}_{Yr(x-1)} = T_{biom, Yr(x-1)} * \text{Biom}_{Yr(x-1)}$$

and solved for $\text{Leaf Prod'n}_{Yr(x-1)}$ in Equation 1.

Results - Biomass. The regressions equations used to calculate biomass components for the three mangrove species are listed in **Table C-5-1**. For stem biomass, we chose a two-parameter model in which $(D_{30})^2 * \text{TOTHT}$ was the single independent variable. For propoot biomass, total height was the lone independent variable in the selected model. For branch, leaf, and total biomass, regression models which included crown volume and stem diameter as independent variables were selected. Coefficients of determination were highest for stem biomass, and exceeded 0.90 for all species-tissue combinations except *R. mangle* propoot biomass ($R^2=0.74$).

Table C-5-2: Mean biomass (Mg/ha) of L-31E Fringe and Dwarf mangrove forest between 1992-1998. 1992 & 1993 means are based on point-quarter sampling from 13 Fringe locations, and 1995-1998 means are based on 15 rectangular plots, as described in text.

	Community Type			
	Dwarf forest		Fringe forest	
Year	mean	±S. E.	mean	±S. E.
1992	---	---	197.02	50.4
1993	---	---	0.46	0.28
1994	---	---	---	---
1995	14.13	3.74	18.19	4.84
1996	17.04	4.28	38.30	9.82
1997	20.09	4.57	54.84	12.32
1998	20.82	5.35	69.04	16.19

We did not estimate biomass in the Dwarf forest in 1993. Because it is based on point-to-plant distances, the point-quarter method is an effective sampling technique for stands of large trees (e.g., the Fringe forest), but is not optimal for estimating density in the extremely dense Dwarf mangrove community. Plot sampling in subsequent years indicated that the increase in Dwarf forest biomass (14 to 21 Mg in 1995-98) was modest in comparison to that accrued by the rapidly developing Fringe forest (**Table C-5-2**). Compartmentalization of biomass in 1997 March 12, 1999 was similar to the Fringe forest, with slightly higher percentages of total aboveground biomass in leaves (16%) and stems/branches (66%), and slightly lower percentages (18%) in propoots (**Table C-5-3**). As in the Fringe forest, Dwarf plots were dominated by *R. mangle* on a biomass basis.

respectively (Table C-5-4). The comparatively modest response of leaf production reflected the site-insensitivity of leaf turnover rate (Figure C-5-1).

Table C-5-4: Annual aboveground production (Mg/ha/yr) in two South Florida mangrove community types, July 1996 - June 1997.

Biomass Component	Community Type	
	Dwarf Forest	Fringe Forest
Stems	1.2 (0.2)	7.9 (1.0)
Branches	0.5 (0.1)	3.7 (0.6)
Proproots	0.5 (0.1)	4.6 (1.2)
Leaves	6.5 (1.9)	13.6 (3.0)
Total	8.6 (2.1)	29.8 (5.7)

Conclusions. The contrast in productivity between mangrove forests occupying the ocean fringes vs. neighboring communities inhabiting interior basins has been observed and discussed by many researchers (e.g., Davis 1940, Lugo and Snedaker 1974; Twilley 1998). In a sense, these analyses are based on an infrequently tested assumption, i.e., that existing variation in forest structure may be equated to variation in the rate at which biomass is produced. Our results generally confirm the validity of this assumption for the two sites we examined, with both biomass and production elevated by 3-4 times in Fringe forest relative to Dwarf forest. However, based on the trajectories we expect these variables to follow as stand development proceeds, the production rates of these two ecosystem types may not be nearly as different as their respective standing crops at maturity. Moreover, mature Fringe and Dwarf forests are expected to differ more in woody tissue biomass than in leaf biomass, in line with the strong contrast in how production is allocated in the two forest types.

C-6. Variation in mangrove population processes among species, sites, and years.

The sampling design described below was intended to provide spatially-explicit information about the structural development of the L-31E mangrove communities throughout the period of study, as well as the population dynamics of the three major species: *R. mangle*, *L. racemosa* and *A. germinans*. The core data set consists of annual censuses in 15 rectangular (0.5 meter wide) plots in 1995-1998. Four plots were established in each sub-block except the Control Dwarf, which had three. Plot lengths varied from 3 to 10 meters, depending on mangrove stem density at the time of the initial census; Our intention was to monitor about 100 individuals per plot throughout the study, but this number has varied from as few as 35 to nearly 200. Plot establishment and initial measurements were concluded during April - July 1995, while subsequent monitoring was conducted during the early winter months (November-December). Corner stakes

mangle had the higher mortality rate at all sizes beyond 40 cm height. Mortality of both species generally decreased with increasing initial size in the Treatment Dwarf and Fringe forests. This pattern is consistent with the convex survivorship curves of **Figure C-6-3**; it may also be indicative of suppression of understory stems by the larger trees. In the freeze-impacted Control Dwarf forest, larger stems were more prone to death, regardless of species (see also **Figure C-3-1**). This pattern presumably reflected the stratification of temperatures during the freeze event itself, as most of the original cohort mortality in the Control Dwarf forest occurred during 1996 (**Figure C-6-3**).

Table C-6-1: Cumulative % mortality experienced by the 1995 cohort through December 1998. Data from Sub-blocks TF and CF are combined in Fringe Forests group.

1995 Height (cm)	Sub-block or Sub-block Group					
	Treatment Dwarf		Control Dwarf		Fringe Forests	
	<i>L. racemosa</i>	<i>R. mangle</i>	<i>L. racemosa</i>	<i>R. mangle</i>	<i>L. racemosa</i>	<i>R. mangle</i>
0-19.9	74.0	38.5	47.5	33.3	100.0	80.0
20-39.9	23.1	23.9	27.9	48.4	56.4	50.0
40-59.9	0.0	3.8	36.4	50.0	0.0	11.8
60-79.9	0.0	4.5	16.7	50.0	0.0	20.7
80-99.9	0.0	0.0	60.0	87.5	0.0	11.8
100-199.9	-	-	0.0	100.0	0.0	15.3
200-299.9	-	-	-	-	-	19.1
≥ 300	-	-	-	-	0.0	0.0

Mangrove populations may grow or remain stable if mortality is offset by recruitment. *L. racemosa* was by far the most abundant of the three major mangrove species in the seedling community, but recruits of each were encountered in all sites in at least one year (**Figure C-6-4**). *R. mangle* seedling recruitment was highest in the Dwarf forest, especially in the Treatment sub-block, where the density of red mangrove recruits approached that of white mangrove. A single *Conocarpus erecta* seedling was tagged in the Control Dwarf sub-block in 1998. Seedling establishment was highest in 1996, especially in the Control Dwarf sub-block.

Mean cumulative height growth accrued by all individuals of the 1995 cohort that survived through December 1998 are presented by species (red and white mangrove only) and initial size class in **Table C-6-2**. The dramatic difference in growth between Dwarf and Fringe forests has been discussed earlier (**Sections C-2, C-5**). A pattern of more rapid growth with increasing initial height is strong in the Fringe sub-blocks, and very slight in the Treatment Dwarf forest. In the Control Dwarf sub-block, the effects of the 1996 freeze result in a reversal of this pattern; many

C-7. Invasion by non-mangrove species into the Dwarf forest.

The vegetation plots described above for the annual censuses were too limited in extent to effectively monitor invasion of exotic or other plants associated with the diversion treatment or natural disturbances. In September 1997, a 3 x 30 meter plot was established at each of the seven Dwarf forest sites described in **Section C-6**, oriented so that the 0.5 meter-wide census plot was nested in its northwest corner. In September 1998, we added an eighth plot 20 meters from the spreader ditch in the Treatment block. The plots were divided along their long axis into ten 3 x 3 meter sections, and the coordinates for a single 1 x 1 meter quadrat were randomly selected within each section. In September of 1997 and 1998, we estimated the shoot cover of all vascular plant species in each quadrat, using six cover classes. Plot means calculated on the basis of cover class midpoints are presented in Table C-7-1.

Table C-7-1: Mean species cover in eight 3 x 30 meter plots in the Convoy Point Dwarf forest on two dates. Plots CDE, CDI, and CDW are approximately 160, 120, and 60 meters east of the L-31E levee, and Plots TDE, TDI, TDW, TDWW, and TDWWW are about 340, 280, 180, 50, and 20 meters east of the levee, respectively.

PLOT	DATE	SPECIES								
		RHIMAN	LAGRAC	AVIGER	PHIVER	RHABIF	PLU'ODO	DISSPI	CONERE	SESPOR
CDE	9/97	22.6	32	5.5	25.4	1.6	0	0	0	0
CDE	9/98	33.4	21.2	1.0	2.8	0.8	0.2	0	0.1	0.1
CDI	9/97	8.3	11.4	0.9	5.8	0.2	0	0	0	0
CDI	9/98	18.7	26.2	1.4	1.8	0.6	0.2	0	0	0
CDW	9/97	14.4	7.4	0.6	3.3	0	0	0	0	0
CDW	9/98	59.6	10.7	0.4	0.8	0.1	0.1	0	0	0
TDE	9/97	79.6	7.4	1.4	0	0	0	0	0	0
TDE	9/98	83.0	6.2	0.6	0	0	0	0	0	0
TDI	9/97	79.6	6.6	0.1	0	0	0	0.1	0	0
TDI	9/98	83.0	3.8	0	0	0	0	0.1	0	0
TDW	9/97	62.9	4.4	0	0	0	0	0	0	0
TDW	9/98	76.3	1.3	0	0	0	0	0	0	0
TDWW	9/97	27.6	1.4	0.1	0	0	0	0	0	0
TDWW	9/98	31.6	0.7	0	0	0	0	0	0	0
TDWWW	9/98	35.6	0.5	0	0	0	0	0	0	0

1997 and 1998 surveys, 101 such *R. mangle* stems averaged 0.29 cm diameter growth, and 11 *L. racemosa* stems averaged 0.32 cm growth. The strong difference in diameter growth between Fringe forests and interior tree islands may be a result of inherent differences in site productivity, or of differences in the developmental stage of the current forest communities. With regard to the latter possibility, the recently-established Fringe forests are in an exponential growth phase following a major disturbance, while the tree islands are composed almost exclusively of older stems that survived Hurricane Andrew. These trees --- some of which may have been in a suppressed position in the mangrove forest canopy prior to the hurricane --- may take several more years to respond to the light and nutrient resources made available through hurricane mortality.

Mangrove water relations.

While mangroves as a group are defined by their ability to function in saline conditions, their photosynthetic rates are generally maximum in fresh or brackish water (Lin and Sternberg 1992). Mangrove species have developed a range of mechanisms that allow the maintenance of osmotic balance under conditions that cause death in non-adapted species. For instance, *R. mangle* is reportedly efficient at excluding salts at the root-soil interface, while *L. racemosa* and *A. germinans* excrete salts through specialized glands and at the leaf surface, respectively (Tomlinson 1986). One may suppose that the mechanism utilized by *A. germinans* is the most effective, because black mangrove is known to survive at salinities well above those that would be lethal for the other two species (Tomlinson 1986). However, Ball (1988) points out that all mechanisms for dealing with osmotic stress entail costs in terms of growth or other plant functions. Thus, there is good reason to expect that the changes in salinity regime that the L-31E treatment regime should eventually bring about will also affect the competitive balance between these three species. For these reasons, we used several techniques to examine the water relations of the two dominant mangrove species in our study area, *R. mangle* and *L. racemosa*.

C-9. Plant moisture stress as measured by the pressure bomb.

Scholander et al. (1965) introduced the use of the pressure bomb as a tool to assess plant moisture stress in woody plants. The instrument measures the pressure necessary to force water out of xylem vessels exposed at the cut end of a severed twig. The units of measurement are positive atmospheres of pressure, and the readings have been shown to effectively index the moisture potential in the water column prior to severing (Waring and Cleary 1967).

Moisture stress may be induced in several ways, including low matric or osmotic potentials in the soil, or some combination of the two. In the L-31E study, we measured plant moisture stress in mid-April of both 1996 and 1997, i.e., at a time of year when moisture stress in the mangroves should be at a maximum because pore water salinities are approaching their annual maximum. Between 5:30 A.M. and 7:30 A.M., we collected single shoot tips from six *R. mangle* and six *L. racemosa* saplings at each site, and obtained pressure bomb readings within five

Between April 1 and June 6, 1998, we obtained concurrent estimates of transpiration rate for pairs of red and white mangrove trees. To measure transpiration, we attached Dynagage SGA-10 collars to stems 10-11 mm in diameter. The length of all leaves beyond the stem flow collar was recorded, and the total transpirational area was calculated on the basis of a previously established length:area regression (Ross et al. unpublished data); this allowed us to express sap flow in units of g/cm² leaf area/hr. The temperature differential between the upstream and downstream ends of the Dynagage collar --- which is directly proportional to sap flow rate --- was measured in millivolts. Voltage output was stored continuously on a Campbell CR-10 data logger, and averaged over 0.5 hour intervals. In the Treatment block, Fringe and Dwarf forest transpiration were measured during the same period, but estimates for Control Fringe and Control Dwarf trees encompassed separate time intervals because of failure of one of the data loggers. The time course of water movement for each tree was carefully examined, and statistical analyses were based on 150 tree-days of record which exhibited good diurnal patterns (**Table C-10-1**). Typical diurnal transpiration patterns are illustrated in **Figure C-10-1, A & B**, which track the transpiration of individual *R. mangle* and *L. racemosa* stems in the Fringe & Dwarf forests during a four-day period in April 1998.

Table C-10-1: Mangrove transpiration sampling record during 1998.			
Site	Sampling Period	Total number of trees sampled	Number of tree-days with good record
Treatment Dwarf	4/3 - 4/17	8	60
Treatment Fringe	4/1 - 4/15	8	42
Control Dwarf	5/19 - 6/6	6	36
Control Fringe	5/13 - 5/16	3	12

Figure C-10-2 summarizes the mean daily transpiration of 25 individuals over periods ranging from 3-9 days. The graph suggests that transpiration rates for *L. racemosa* exceeded those of *R. mangle* when calculated on a leaf area basis, and that within each species, transpiration was higher in the Fringe than the Dwarf forest. Because it was possible that species or site differences were affected by date, we applied a 2-way Model 1 analysis of variance to a subset of the entire data set --- i.e., the 16 trees sampled in the Treatment block, censored minimally to secure a balanced representation of species and sites over the same time period. Data were log-transformed means of 4 or 6 daily totals for each tree during a common sampling period; the transformation was successful in homogenizing the variance among treatments. The analysis indicated significant effects of site ($F_{(1,12)}=24.77$, $p=.0003$) and species ($F_{(1,12)}=13.19$, $p=.0034$), while the species-by-site interaction was not statistically significant ($F_{(1,12)}=0.07$, $p=.795$).

C-11. Carbon isotope content of red mangrove leaves.

The carbon isotope composition of plants has been shown to reflect the ratio of internal to external carbon dioxide concentration in the leaf (Farquhar et al. 1982a). In most ecological settings, this ratio is controlled by the rate of carbon dioxide uptake through stomates (Farquhar et al. 1982b). Since stomatal conductance is itself controlled by the gradient in moisture potential between leaf and atmosphere, intra-specific variation in carbon isotope composition most often reflects variation in moisture stress (e.g., Guy et al. 1980; Ross et al. 1994).

As described in **Section B-3**, we collected *R. mangle* leaf samples for determination of carbon isotope composition at quarterly intervals beginning in January 1996. Leaves were ground and combusted with cupric oxide. The resulting CO₂ was cryogenically purified and subjected to mass spectrometry analysis in the laboratory of Dr. Leo Sternberg at the University of Miami. Carbon isotope composition was expressed as $\delta^{13}\text{C}$.

$$\delta^{13}\text{C} = 1000 \times (R_{\text{sample}} - R_{\text{standard}})/R_{\text{standard}}$$

where R_{sample} and R_{standard} are the $^{13}\text{C}/^{12}\text{C}$ ratio of the sample and the PDB standard, respectively. Low (large negative) values of $\delta^{13}\text{C}$ were expected to reflect low moisture stress during carbon fixation, and high values to indicate high stress. Statistical analysis of site effects were performed as described in **Section B-3**.

R. mangle leaf $\delta^{13}\text{C}$ did not differ among sampling periods, but site effects were found to be highly significant ($p < 0.001$) in the subsequent one-way analysis of variance (**Figure C-11-1**). Mean $\delta^{13}\text{C}$ values over the two-year period were: Control Fringe, -28.4; Treatment Fringe, -27.6; Control Dwarf, -26.1; and Treatment Dwarf, -25.8. Orthogonal contrasts yielded highly significant differences between forest types and between blocks, indicating lower stress in the Fringe forest and in the Control block. In light of the negative evidence regarding differences in moisture stress (**Section C-9**) among sub-blocks, we hypothesize that the consistent differences in $\delta^{13}\text{C}$ reflect the effects of nutrient limitation --- specifically phosphorus limitation --- on stomatal conductance and photosynthetic rate. We are not aware of other studies supporting a direct connection between nutrient limitation of any sort and carbon isotope composition, but we hope to examine the hypothesis presented here further during the remainder of the study. Since the presence of the nutrient effect hypothesized above does not preclude the well-established effect of moisture stress on leaf $\delta^{13}\text{C}$, diversion of fresh water into the Treatment block may reduce the relative concentration of heavy isotopes in fixed carbon slightly. However, if our hypothesis is true, diversion of water low in phosphorus will not entirely eliminate the site effects illustrated in **Figure C-11-1**.

Table D-1-1a. Hydroperiod summary for four vegetation plots in the Treatment Dwarf sub-block. Period=total time flooded during month; Duration = mean flooding event duration.

Date	TDWW			TDW			TDI			TDE		
	Period (hrs)	Events	Duration (hrs)	Period (hrs)	Events	Duration (hrs)	Period (hrs)	Events	Duration (hrs)	Period (hrs)	Events	Duration (hrs)
Mar-96	194	2	97	337	2	169	337	2	169	36	1	36
Apr-96	-	-	-	-	-	-	-	-	-	-	-	-
May-96	435	2	218	659	2	330	613	2	307	431	2	216
Sep-96	220	1	220	220	1	220	220	1	220	220	1	220
Oct-96	451	1	451	451	1	451	451	1	451	449	1	449
Nov-96	233	1	233	262	1	262	262	1	262	102	1	102
Dec-96	34	1	34	165	1	165	165	1	165	0	0	0
Jan-97	395	1	395	395	1	395	395	1	395	317	2	159
Feb-97	99	1	99	120	1	120	120	1	120	0	0	0
Aug-97	250	1	250	250	1	250	250	1	250	250	1	250
Sep-97	720	1	720	720	1	720	720	1	720	720	1	720
Oct-97	744	1	744	744	1	744	744	1	744	744	1	744
Nov-97	600	2	300	600	2	300	600	2	300	600	2	300
Dec-97	644	2	322	728	2	364	717	2	359	563	2	282
Jan-98	340	3	113	487	4	122	487	4	122	282	3	94
Feb-98	556	2	278	616	2	308	616	2	308	530	2	265
Mar-98	682	2	341	736	2	368	736	2	368	560	4	140
Apr-98	578	3	193	635	3	212	633	3	211	551	3	184
May-98	456	1	456	457	1	457	458	1	458	459	1	459
Jun-98	253	1	253	253	1	253	253	1	253	253	1	253
Aug-98	744	1	744	744	1	744	744	1	744	744	1	744
Sep-98	408	1	408	408	1	408	408	1	408	408	1	408

Table D-1-2a. Hydroperiod summary for four vegetation plots in the Treatment Fringe (TF) sub-block.
 Period = total time flooded during month; Duration = mean flooding event duration.

	TFW			TFI			TFS			TFE		
Date	Period (hrs.)	Events	Duration	Period (hrs.)	Events	Duration	Period (hrs.)	Events	Duration	Period (hrs.)	Events	Duration
May-96	61.6	24	3	71.1	24	3	7.9	5	2	30.8	18	2
Jun-96	-	-	-	-	-	-	-	-	-	-	-	-
Jul-96	-	-	-	-	-	-	-	-	-	-	-	-
Aug-96	-	-	-	-	-	-	-	-	-	-	-	-
Sep-96	-	-	-	-	-	-	-	-	-	-	-	-
Oct-96	-	-	-	-	-	-	-	-	-	-	-	-
Nov-96	-	-	-	-	-	-	-	-	-	-	-	-
Dec-96	-	-	-	-	-	-	-	-	-	-	-	-
Jan-97	-	-	-	-	-	-	-	-	-	-	-	-
Feb-97	-	-	-	-	-	-	-	-	-	-	-	-
Mar-97	22.2	10	2	50.4	20	3	6.1	5	1	10	6	2
Apr-97	-	-	-	-	-	-	-	-	-	-	-	-
May-97	-	-	-	-	-	-	-	-	-	-	-	-
Jun-97	-	-	-	-	-	-	-	-	-	-	-	-
Jul-97	-	-	-	-	-	-	-	-	-	-	-	-
Aug-97	-	-	-	-	-	-	-	-	-	-	-	-
Sep-97	-	-	-	-	-	-	-	-	-	-	-	-
Oct-97	-	-	-	-	-	-	-	-	-	-	-	-
Nov-97	-	-	-	-	-	-	-	-	-	-	-	-
Dec-97	48	29	2	77.1	44	2	4.3	3	1	24.8	20	1
Jan-98	14.6	12	1	38.5	27	1	0	0	0	1.7	2	1
Feb-98	-	-	-	-	-	-	-	-	-	-	-	-
Mar-98	48.5	19	3	58.2	21	3	5.3	5	1	20.3	21	1
Apr-98	177.9	45	4	252.2	52	5	93.5	30	3	129.4	39	3
May-98	212.1	31	7	193.2	34	6	198.5	36	6	139.7	34	4
Jun-98	-	-	-	-	-	-	-	-	-	-	-	-
Jul-98	-	-	-	-	-	-	-	-	-	-	-	-
Aug-98	256	59	4	272	59	5	120	41	3	146	53	3
Sep-98	257	54	5	279	52	5	185	45	4	206	53	4

General conclusions based upon hydroperiod data:

1. Sites with lower elevation in the same part of the watershed have longer hydroperiods. However, in the case of the Treatment Dwarf site, which has the longest hydroperiod, portions of the marsh surface are likely to be dry because of the highly variable microtopography (especially at the TDWW and TDW).

2. Hydroperiods in the Fringe forest range between 8 and 255 hrs / month, with the lowest hydroperiods (0-60 hrs/month) in December through April, with May transitional, and the highest hydroperiods occurring May through September (60 - 255 hrs/month). The number of flooding events for the dry winter months range between 0 and 29 in contrast to the summer months, which range from 39 to 59. The average period for each flooding event at the fringe sites is between 2 and 8 hrs, with a mean of about 4 hrs/event. Because of its slightly lower elevation, both the number of flooding events, and monthly hydroperiods are greater in the TF site than in the CF site.

3. Dwarf forest hydroperiods range between 35 and 744 hrs/month. Much less monthly variation is expressed than in the fringe sites. In part, this is because rainfall events are much more likely to inundate the marsh surface of the scrub site, because of better drainage in the fringe sites. In addition to longer hydroperiods, the scrub sites are also characterized by fewer flooding events than the fringe sites, with approximately two events per month. The Treatment Dwarf sites have longer hydroperiods than the Control Dwarf sites because of topography differences.

The Treatment Dwarf site is unlike the other sites in this study because it is within a topographic basin. Prior to coastal management practices in the first half of this century, the basin was dominated by freshwater sheetflow (based upon calcareous marl soil and freshwater invertebrate remains: see **Section E**). After the construction of the L-31E storm protection levee, sheetflow to the area was entirely interrupted (Meeder et al, 1997), and tidal processes became more important. At present, the water budget is dominated by local rainfall and tidal exchange. The location of our study area immediately north of the Mowry Canal --- which is a primary point source of agricultural runoff water to southern Biscayne Bay with its 93,300 acre-feet discharge per year (Scheidt et al 1983) --- prevents more saline conditions. (**Note:** We stress that whereas canal discharge may lower salinity along the coast for a distance "downstream," thereby decreasing coastal salinities, it does not provide the same ecosystem function as sheet flow across the entire coastal wetland ecosystem. This is true for several reasons: 1) nutrient load and distribution differences, 2) the timing of canal discharge is arranged to meet numerous management objectives (stormwater runoff, agricultural purposes, aquifer recharge, control of salt water intrusion, mosquito control, etc.) which may not replicate the timing required by numerous estuarine-dependent species, and 3) the types of nutrient species and their sources are different). Therefore, when tidal waters enter the Treatment Dwarf basin, they do not totally drain but sometimes remain for weeks before evapotranspiration, infiltration, and seepage cause the basin to become desiccated. Other basins in south Florida that receive tidal waters of marine salinity are hypersaline during the dry season. Like tidal inundations, rainfall events fill the Treatment Dwarf

Table D-2-1. Groundwater discharge calculated by two methods. (From Meeder et al. 1997). Max* and Min** based upon Darcy's calculation of discharge at 50 m using exponential equation;***, at 50 meter well; na=not applicable.

	Darcy's Equation		Well Head		
	Max	Min	Max*	Min**	Mean***
ft ³ /ft ² /day	600	240	600	240	170
ft ³ /0-50m/1m	98,400	39,360	na	na	na
ft ³ /0-100m/1m	196,800	78,720	na	na	na
ft ³ /1-400m by steps	na	na	1720	690	75,200
study area (m ² /day)	38,400,000	15,400,000	335,000	134,000	14,700,000

Well head discharge values approximated the low discharge values derived from the Darcy's equation, and did decrease exponentially with increased distance from shore. Specific discharge was determined by dividing measured discharge by the area of the screen interval (Coletta 1995). Overall discharge was determined by multiplying specific discharge times the depth and width of aquifer (Quinones-Aponte pers. comm.).

Concern that the sediment package impedes groundwater discharge, thereby leading to overestimates by both Darcy's equation and well head discharge calculations, was addressed by the insertion of seepage meters. Seepage meter discharge values were determined by standard technique (Lee 1977). The volume of flow captured (liters/hour) is multiplied by 3.92 (the inverse of 0.255 which is the area of the seepage meter), which results in discharge in liters/hr/m². We then multiplied the mean discharge/m²/day by the area of active groundwater discharge (400 x 2100 m --- the distance offshore discharge was observed by the length of coastline) to estimate the volume of groundwater discharge in the study area. We assumed an average value for seepage discharge to be approximately 1 l/hr/m², which results in 20,160 m³/day. Groundwater discharge based upon seepage meters was less than one order of magnitude less than that calculated by Darcy's equation applied in exponentially decreasing steps offshore.

Discharge measurements based upon the seepage meters were considerably lower than well head discharge measurements, and did not decrease exponentially offshore (**Table D-2-2**). Discharge rates exhibited no distinct patterns in the offshore or longshore directions. We therefore attribute the differences in groundwater discharge rates to site soil thickness and composition and degree of surface cap rock permeability (highly variable because of both dissolution (karst), induration (cementation) or fracturing processes).

for the period 1975 through 1981 (Scheidt et al 1993). **Table D-2-3** compares discharge, nutrient concentrations and loads between the Mowry Canal and groundwater.

Table D-2-3: Summary of nutrient loading to Biscayne Bay study site.									
Nutrient	Mowry Canal			Groundwater (seepage method)			Re-established Sheet Flow		
	Conc. (ppm)	Q (m³/day)	Load (g/day)	Conc. (ppm)	Q (m³/day)	Load (g/day)	Conc. (ppm)	Q (m³/day)	Load (g/day)
TP	0.032	462.400	14.80	0.031	20.160	0.620	0.032	1.085.8	0.035
TN	0.920	462.400	425.4	1.11	20.160	22.40	0.920	1.085.8	0.99
TOC	12.50	462.400	5,780.0	10.80	20.160	217.70	12.50	1.085.8	13.57

The concentration of TP, TN, and TOC vary little in respect to source. However, the magnitude of the delivery volume and therefore load varies significantly between the three sources. The Delivery Canal discharge is only 0.2 % of the Mowry Canal. Ground water load varies little by season or from between sample periods in this two year study. Therefore, the only way to substantially reduce Biscayne Bay nutrient loading is to address means of reducing Mowry Canal discharge.

Groundwater discharge in the development of mangrove tree islands. Mangrove tree islands are clusters of taller mangrove trees in an area without mangroves or in scrub mangrove ecosystems. These tree islands have distinct boundaries which coincide with underlying bedrock depressions, karst features referred to as dolines. Soils underlying tree islands are much thicker than the adjacent soils of the scrub mangroves, more organic (usually peats), and exhibit positive topographic relief in respect to surrounding scrub mangroves (**Figure E-2-1**). In addition, water elevation observations document that: 1) surface water elevation in tree islands is higher than surrounding area, 2) the fluctuation of surface water elevation in tree islands is less than in the adjacent scrub, and 3) fluctuations in the tree island water table is related to the stage of the delivery canal (**Table D-2-4**). When Mowry Canal stage is held at 63.5 cm 100 % of the observations have dome water within 2 cm of the marsh surface. During the winter the dome surface soils are dry to a depth of 5 cm while the adjacent dwarf soils are highly desiccated with developing mud cracks. At a similar site, tree island soil salinities are less than adjacent scrub soil salinities (Meeder unpubl, Turkey Point).

D-3. Tidal Flux.

The balance between tidal ingress and egress in respect to both water volume and nutrient load was calculated for four tidal cycles (two diurnal periods, in December 1996 and April 1997; Meeder et al 1999). Water samples were collected for nutrient analysis, and stage recorded every two hours. In addition, particulate organic material was collected for one hour between water sampling periods. Stage data was transformed to discharge by the application of stage discharge curves prepared for each site, and for both ingressing and egressing tides (Meeder et al 1999). Creek discharge multiplied by the nutrient concentration yields instantaneous load in mg/l/sec. The bihourly instantaneous nutrient and stage data were plotted against time, and intermediate values (hourly) were estimated based upon the curve. By projecting flux for one hour rather than for two hour steps, we produced conservative estimates of flux that minimized the significance of extremely high and low values. Eight tables summarized the hourly load. Negative values were arbitrarily assigned to ingressing tide values, and positive number for egressing tide values for each nutrient species and for discharge.

Hourly loads and discharges are summarized and reported as flux for each tidal cycle and for the diurnal (two tidal cycles, or approximately 24 hrs 20 min) in **Table D-3-1**. In addition, a comparison of POC flux between stations and sample periods, as well as the net influx of marine, and export of terrestrial POC are summarized in **Table D-3-2**. **Table D-3-3** reports the export or import of TOC, TN, and TP in terms of load per unit area of marsh drainage basin.

Tidal cycle flux. In all cases, there was a net movement of water into the coastal mangrove ecosystem over two tidal cycles (**Table D-3-1**). In one case, in December at the Treatment Dwarf site, a single tidal cycle exhibited a net movement of water towards Biscayne Bay. This case is explained by a combination of three factors: 1) the Treatment Dwarf station is in a topographic basin, 2) change in wind direction, and most importantly 3) the preceding tide was very high, moving water well into the basin, which did not all drain out during the following low tide.

Net movement of water landward is indicated by the coastal mangrove surface water and soil salinities. The flux of water landward is explained by several factors: 1) evaporation of surface waters, 2) transpiration (water use by plants), 3) resaturation of soils, and 4) spring tide events.

The major differences between the December and April flux results are attributed to: 1) greater tidal amplitude in December, and 2) December tidal waters had higher concentrations of TP, N+N, and NH₄, while TOC concentrations were higher in April. Considerably more water was exchanged in the Treatment Block than in the Control Block, and nutrient concentrations were usually higher in the Control Block than in the Treatment Block. Finally, the volume of water exchanged was considerably higher at the Fringe sites than the Dwarf sites (**Table D-3-1**).

Interestingly, NH₄ was usually imported into the coastal wetlands from the Bay. The reverse is usually considered the norm. However, in this area nearshore groundwater discharge is high and the NH₄ load is considerable (Meeder et al. 1998), which may help to explain this phenomena.

POC flux. Fringe sites consistently had more POC movement in and out than did the Dwarf sites (**Table D-3-2**). Fringe sites also had one to two orders of magnitude greater marine POC influx than did the Dwarf sites. Likewise, the Fringe sites exported 2 to 10 times more mangrove-derived material than did the Dwarf sites. Nearly 2 to 10 times as much material was transported in December as in April, with the Fringe sites differing more than the Dwarf sites.

Table D-3-2. Particulate organic carbon flux at four stations during two diurnal sampling periods, in grams per day.

	Control Fringe		Control Dwarf		Treatment Fringe		Treatment Dwarf	
	Dec 96	Apr 97	Dec 96	Apr 97	Dec 96	Apr 97	Dec 96	Apr 97
Net In	544.2	259.2	194.5	No data	2453.8	240.0	426.1	162.7
Net Out	2286.2	287.2	192.7	No data	4153.0	182.2	185.4	124.4
Marine Detritus In	331.4	115.7	1.62	No data	1961.9	203.9	1.2	36.9
Mangrove Detritus Out	1258.0	187.2	168.59	No data	1385.5	81.6	165.0	95.0

Nutrient flux at both Fringe sites was considerably greater than flux at the Dwarf sites, usually by an order of magnitude. The only exceptions were the Treatment Fringe and Treatment Dwarf in December, which differed by a factor of two. Also, the Treatment Fringe exhibited a large export of POC in April, probably associated with tidal water lag from previous high tide caused by strong onshore winds. Similar relationships exist with the POC (**Table D-3-2**). Marine POC influx to the Treatment Fringe site was greater than the export of mangrove POC, the

3. Differences between Dwarf and Fringe sites. Fringe sites are exposed to much higher volumes of ingress and egressing tidal waters. Both import and export from the scrub sites (especially the TD site) is highly restricted because of the basin topography. Once a basin is filled it does not drain rapidly.

4. Tide height and flux relationships. In general, the greater the tide height, the greater the flux. This relationship is much stronger for the Fringe sites than the Dwarf sites.

5. Comparisons to other studies. Odum et al (1982) reviewed carbon export studies for coastal mangroves and concluded that mangroves export approximately $1 \text{ gm/m}^2/\text{yr}$. We find nearly the opposite, a net flux into the wetlands from Biscayne Bay. Although this trend is well documented here, the concept was proposed that the coastal wetlands are a sink for marine derived carbon many years ago (Ira Brooks, cited in Odum et al 1982).

D-4. *Water budget*

A water budget of the study area is needed in order to determine the pathways of water movement and to project the effects of delivery canal discharge on the system. Once hydrologic processes and their rates of operation are documented, the nutrient budget will be much easier to calculate in the future when additional data are collected. The major components of the water budget at this site are: a) local rainfall, b) surface evaporation, c) transpiration, d) direct canal discharge, e) indirect canal discharge, f) direct ground water discharge, g) indirect groundwater discharge, h) freshwater sheet flow (zero prior to diversion, equivalent to Delivery Canal release volume thereafter), and i) tidal activities.

a. Local rainfall. Mean annual rainfall for the study region is 155 cm/yr (Homestead, 1951-1980). Rainfall varies seasonally from 4 cm (December through Feb) to between 20-25 cm (June through September) or 1.29 liters/ m^2/day and 8.1 liters/ m^2/day , respectively. Year-to-year variability in monthly rainfall is much less for dry season months than rainy season months. Rainy season variability is largely associated with the number and magnitude of large rainfall events associated with tropical depressions. Duever et al. (1994) provide a more detailed summary of South Florida rainfall data and patterns. Monthly average rainfall from the study site is presented (Table D4-1).

c. **Transpiration.** Transpiration rates are available on a regional basis from several sources. Much less is known about transpiration rates in coastal wetlands than PE. We have estimated dwarf and fringing mangrove community transpiration by measuring sap flow on select trees and using leaf area to estimate transpiration per unit ground area (**Section C-10**). Site estimates are included in **Table D-4-5**. Transpiration rates are calculated for Dwarf ($2.6 \text{ l/m}^2/\text{day}$) and for Fringe sites ($8.6 \text{ l/m}^2/\text{day}$). Evapotranspiration (ET) estimates for South Florida range between 70 and 90% of annual rainfall (Duever et al 1994). Actual community ET in natural ecosystems is greatest during summer wet season when more surface area is inundated and available for both surface evaporation and transpiration.

d. **Direct canal discharge.**

Delivery canal discharge calculations. Construction of the culverts and weirs was initiated in November 1997 but not completed for measurable delivery until 28 Aug 1997. During this period there was minor leakage of canal water into the treatment block, especially during construction. Water stage recorders and staff gauges were installed on both the up and downstream sides of the L-31E levee. The upstream recorder (interval of 15 minutes) and staff gauge data was used to determine both the head (the height of the water over the weir crest) and the length of time the canal water surface was at any particular stage. This data was used to calculate the discharge into the treatment cell once operation began (**Figure D-4-1**). The total discharge for the period 28 Aug to 17 Sept 1997 was $21,720 \text{ m}^3$, or an average of $1,090 \text{ m}^3/\text{day}$ ($0.88 \text{ acre-ft./day}$). Discharge was calculated using Leupold and Stevens (1991) tables for vertically restricted rectangular notch weir. The distribution of head heights by time increment are shown in **Figure D-4-2**. Head varied between 2 and 3 cm nearly 50 % of the time, and was less than 10 cm during 95% of the period.

Basin Capacity. The Dwarf mangrove portion of the Treatment block is located within a topographic basin (**Figure A-2-1**). The basin has a slight ESE to WNW orientation and is ellipsoidal to subcircular in outline, with a long dimension of approximately 300 m and a width of 200 m. The basin averages approximately 20 cm of relief, with the lowest area about 100 m east of the levee. Basin capacity (before surface runoff occurs) is approximately 8320 m^3 or 6.7 acre-ft. Assuming a daily discharge of approximately 2 acre-feet/day, the basin can hold three days of discharge plus the volume of loss by ET processes. A retention time of between 3 to 4 days can therefore be expected, which should be adequate for nutrient assimilation. A few small tidal creeks along the east and northeast sides drain the basin. These tidal creeks merge into a single well-defined tidal creek (at the Treatment Dwarf tidal creek weir site) but do not reach entirely into the basin.

Discharge to the Treatment Dwarf basin. The only direct canal discharge into the study area is by release through the experimental culvert from the Delivery Canal into the Treatment Block (**Figure A-2-1**). Although the volume of freshwater that can be delivered is restricted by engineering constraints (lack of relief, endangerment of saltwater intrusion and flood

Area of

Table D-4-3: Monthly discharge from Mowry Canal at Structure S-20F, 1975-1985.	
Month	Discharge (acre-feet)
January	2,915
February	3,738
March	6,696
April	3,368
May	2,835
June	11,246
July	12,568
August	9,340
September	10,348
October	11,941
November	9,092
December	7,243
Monthly mean	7,611
Annual mean	91,330

Daily discharges for December, April and August are 288,229, and 138,494, and 371,690 m³/day, respectively. That makes our conservative groundwater discharge estimates approximately 10 % of April and 5% of December Mowry Canal discharge.

f. Direct groundwater discharge. Direct groundwater discharge is limited to the small areas over karst features which are not in filled with clay materials and to a narrow zone along the coast. Groundwater movement towards the surface, or the reverse, is highly restricted by the presence of a thick, impervious, freshwater, calcitic marl overlying the bedrock (refer to **Section E**). Therefore, we conclude that groundwater exchange with marsh surface waters in this study area is very low and approaches zero for most calculations

g. Indirect groundwater discharge. Indirect ground water discharge refers to the groundwater that is exchanged with the deep water canals and with nearshore Biscayne Bay which returns via ingressing tide to the coastal wetlands. We have not addressed groundwater-canal water exchange or mixing near the coast in this study, though the process must occur. A

Table D-4-4. Salinity mixing model for nearshore Biscayne Bay. Best estimates for Apr., Aug. and Dec. are in bold.

Month	Biscayne Bay Salinity	Biscayne Bay Volume	Mowry Canal Salinity	Mowry Canal Discharge	G.W. Salinity	G.W. Discharge	Nearshore Salinity
Apr.	31	1	0	6.9	5	1	4
	31	2	0	6.9	5	1	6.8
	31	3	0	6.9	5	1	10.9
	31	10	0	6.9	5	1	17.8
	31	20	0	6.9	5	1	22.4
	31	30	0	6.9	5	1	24.7
	31	40	0	6.9	5	1	26
Aug.	31	1	0	18.4	5	1	1.8
	31	2	0	18.4	5	1	3.1
	31	3	0	18.4	5	1	4.41
	31	10	0	18.4	5	1	10.7
	31	20	0	18.4	5	1	15.9
Dec.	31	2	0	14.3	5	1	3.9
	31	3	0	14.3	5	1	5.4
	31	4	0	14.3	5	1	6.7
	31	5	0	14.3	5	1	7.9
	31	10	0	14.3	5	1	12.5

Numerous combinations of groundwater, canal discharge, and marine waters may produce the same salinity. However, since canal and groundwater discharge and canal, groundwater, and nearshore salinities are known, we can solve for the volume of marine bay water necessary to produce a given salinity (**Table D-4-4**). The first case in the table is the simplest. No freshwater discharge results in 31 ppt nearshore salinities. Though nearshore salinities of 31 ppt are occasionally observed (e.g., once in four years of record), this can not be taken to mean that there was no groundwater discharge during such a period, because offshore winds might push water out. This may cause a rapid turnover of nearshore waters, thereby renewing the more saline waters faster than they can be diluted by groundwater.

h. Freshwater sheet flow. Based upon the distribution of the Perrine Marl, pre-development freshwater sheet flow through the study area was significant enough to have maintained a long hydroperiod freshwater marsh to within 250 m of Biscayne Bay. Additional evidence for considerable freshwater sheet flow include: 1) the abundance of subfossil oysters along mangrove creeks and at their mouths, 2) early fish catch records which indicate estuarine-dependent species

Table D-4-7. Summary of water budget calculations for the four sites and three time periods.

Site	Period	Pct	DCQ	B	Subtotal (+)	PE	T	R	Subtotal (-)	Dif
TD	Dec.	1.45	0	3.49	4.94	2.5	2.81	3.6	8.91	-4
	Apr.	2.00	0	4.1	6.1	5.5	2.81	8	16.31	-10.3
	Aug.-1	6.60	5.9	3.49	15.99	4.9	2.81	0	7.71	8.3
	Aug.-2	6.60	6.5	3.49	16.59	4.9	2.81	0	7.71	13.8
CD	Dec.	1.45	0	1.32	2.77	2.5	2.81	21.3	26.3	-23.5
	Apr.	2.00	0	0.02	2.02	5.5	2.81	38	46.3	-44.3
	Aug.	6.60	0	1.32	7.92	4.9	2.81	9	16.7	-8.8
TF	Dec.	1.45	0	34.3	35.75	2.5	8.56	30	41.1	5.3
	Apr.	2.00	0	24.6	26.6	5.5	8.56	73	87.1	-60.5
	Aug.	6.60	0	34.3	40.9	4.9	8.56	13	26.46	14.4
CF	Dec.	1.45	0	65.5	66.9	2.5	8.56	39	50.1	16.8
	Apr.	2.00	0	33.3	35.3	5.5	8.56	73	87.1	-51.8
	Aug.	6.6	0	65.5	72.1	4.9	8.56	24	37.5	34.6

All sites exhibited seasonal differences in their respective water budgets. The two major variables influencing the water budget were tidal flux and soil resaturation. April always had the greatest water deficit at all stations, with the net deficiency in the Fringe sites exceeding that experienced in the Dwarf sites by a factor of 2-6 times.

Most significantly, Delivery Canal discharge in August at the Treatment Dwarf site produced a surplus of 8.9 l/m²/day, which was exported to the Fringe via surface drainage. The major reasons for this surplus was that Delivery Canal discharge exceeded evapotranspiration during the period, and soils did not require resaturation. In August, Sites TD, TF, and CF all had surplus water, and CD had the lowest deficit of the three sample periods. Most sites require little soil resaturation in August. Of the variables in the water budget calculation, soil resaturation volumes were the most problematic, and estimated water deficits may be due in part to overestimates in this component.

resaturation data for a typical spring tide cycle in December, April, and August, including soil water content, depth of soil resaturation, area of inundation, basin area, and the estimated volume of water needed for complete resaturation, are presented in **Table D-4-5**.

Table D-4-5. Calculation of soil resaturation volumes. To calculate soil resaturation volume (R) for a 1 m² area to a depth of 1 cm, multiply the water volume of the soil (ml/cm³) by 10,000 cm²/m² and divide by 1000 ml/l. The result is in units l/m²/cm soil depth. Restaturation volume per m² per cm of soil depth multiplied by the depth of restaturation results in the maximum water volume required to resaturate the entire dry soil profile (l/m²).

Site	Water volume (ml/cc)	Max saturation depth (cm)	Water volume for restaturation (l/m ² /cm)	Area (m ²)	Percent Area Inundated			Resaturation Depth			R (l/m ² /cm)		
					Dec	Apr	Aug	Dec	Apr	Aug	Dec	Apr	Aug
TD	0.47	5	47	105000	90	2	9	80	5	24	95	0	0
CD	0.58	10	58	92000	85	5	16	75	10	58	90	2	10
TF	0.9	15	90	133000	80	5	45	65	15	135	85	2	18
CF	0.9	15	90	114000	75	7	63	65	15	135	80	4	36

The depth of saturation was estimated by examining water level records for the months Dec, April and August. In general, soils were at or near saturation in August, became drier as December approached, and became relatively desiccated in April and May, requiring the most water for resaturation. Each month tidal inundation height was different, and therefore the area flooded differed. Visual estimates of the percentage of the basin flooded were made during sampling periods. Basin area multiplied by the area inundated multiplied by the restaturation depth multiplied by the restaturation water volume per cm soil depth results in the total volume of water required for basin restaturation on a daily basis.

Section E: Soils

E-1. *Introduction.*

Soil constituents contain numerous environmental signals that permit the reconstruction of prevailing conditions during deposition. In the Southeast Saline Everglades (SESE), which includes the L-31E study area, a comparison of vertical soil sequences with surface soil constituents --- which effectively reflect present plant communities --- indicate that, in general, conditions have become more saline (Meeder et al 1997). Fossil mollusk assemblages proved to be good indicators of salinity regime and are used at the L-31E site. Soils in the study area can be subdivided into two major types: carbonate mud (marl) and fibrous peat. Carbonate muds, composed of calcite, are produced in fresh-to-low salinity areas containing high dissolved calcium carbonate concentrations and abundant filamentous algae (Gleason 1974; Browder et al 1994). Coastal peats are predominately made up of red mangrove root material (Cohen and Spackman 1974; Gleason and Stone 1994). Sawgrass root material is observed in the marl soils, but does not form fibrous peat in this study area.

Aerial photography dating to 1928 documents a fringing mangrove belt that was much thinner than today (approximately 250 m). By the early 1920-30's, ditching for coastal land reclamation, agricultural purposes, and mosquito control effectively drained most of the wetland areas found between the coastal ridge and the fringing mangroves. This reduced or eliminated freshwater sheet flow to Biscayne Bay. By the 1940's, salt water encroachment became a problem in coastal areas, causing the abandonment of many fields (Parker et al. 1955). Parker et. al. (1955), as well as many others, suggested placing salinity barriers in the mouths of major canals to reduce saltwater encroachment. In addition, during the mid 1960's, the L-31E canal and levee were constructed to minimize the effects of storm tidal waters. The construction of the levee resulted in the final reduction of sheet flow to the Bay. Thus, areas east of the levee were transformed in a few decades from coastal wetland ecosystems in which freshwater runoff was a significant part of the hydrologic budget to systems driven entirely by local rainfall and tidal processes. This change resulted in increased coastal basin salinities, which were eventually reflected in changes in both vegetation and soil characteristics.

Sea level has been rising continuously, but at different rates, during the Holocene, leaving an abundant record in the South Florida region dating back approximately 6,000 yrs. (Parkinson 1989). Tidal records from Key West indicate that the rate of sea level rise has accelerated during the last century (Maul 1993). Most researchers believe that the south Florida platform has subsided uniformly, and therefore differences in the rate of salt water encroachment between coastal basins must be a response to changing freshwater discharge (Meeder et al 1997). Given this assumption, the causes of salt water encroachment can be further analyzed by comparing different basins with different histories of hydrologic alteration. A list of regional hydrologic

Root material was plotted against amorphous organic material and carbonate in order to define the relationship between the types of organic material associated with carbonate mud content (**Figure E-2-1c**). Most carbonate marl contains approximately 10% organic material. Most samples plot either as organic rich marl (10 to 20 % organic material) or as marly peat (50 % carbonate or less). This illustrates the absence of peats in the dwarf sites. Most of the organic fraction in the marly peat is comprised of amorphous material.

Table E-2-1. Soil attributes along the east-to-west gradient. **DTB** = Depth to bedrock or total soil depth. **Soil type** = Soil type of the marsh surface (pm = peaty marl, mp = marly peat, p = peat). **Thickness** = thickness of marsh surface soil type. **Age** = basal age of surface soil type (estimated from aerial photography, known time of salinity change, and numerous radiometric dates from the SESE). **S** = Soil accretion rate. **Water content** = mean water content of upper 10 cm of soil. **Saturation volume** = volume of water required to resaturate one square meter of soil to a depth of one cm. **Carbon content** = mean carbon content of upper 10 cm of soil. **Carbon deposition** = carbon deposition rate. **CaCO₃ content** = mean calcium carbonate content of upper 10 cm of soil. **CaCO₃ deposition** = Calcium carbonate deposition rate. *Unlike other locations, carbonate source at 620 m is probably not freshwater marl but marine mud brought in by storm tides from Biscayne Bay.

	50 meters		DOME	200 meters	400 meters	500 meters	620 meters
	High	Low					
DTB (cm)	97	89	235	101	130	180	260
Soil type	MP	PM	P	PM	MP	MP	P
Thickness (cm)	15	7	34	15	19	25	125
Age	34	34	500?	74	100	?	1500?
S (mm/y)	0.44	0.21	0.68	0.2	0.19	-	0.83
Water content (g/cc)	0.58	0.47	0.9	0.45	0.4	0.65	0.9
Saturation volume (l/m ²)	5.8	4.7	9	4.5	4	6.5	9
Carbon content (g/m ² /cm depth)	665	385	475	650	1013	-	1220
Carbon deposition (g/m ² /cm depth)	0.802	0.222	0.885	0.356	0.527	-	2.77
CaCO ₃ content (g/m ² /cm depth)	520	470	20	445	303	-	50*
CaCO ₃ deposition (g/m ² /d)	0.627	0.27	0.037	0.244	0.158	-	0.114*

Table E-3-1. List of molluscan species and their salinity ranges. Salinity rankings used to weigh mollusk species abundances for calculation of salinity index. Rank 1.0 = Freshwater, 1.5= freshwater species with tolerance for low salinity, 2.0 = brackish species, 2.5= brackish species that tolerate marine conditions, 3.0= restricted marine with toleration for lower salinity, 4.0 = marine species with tolerance for low salinity, and 5.0 = marine species. Based on descriptions in Ladd 1957; Tabb and Manning 1961; Moore 1964; Turney and Perkins 1972; Abbott 1974; and Thompson 1984.

Species	Salinity Range	Species	Salinity Range
<i>Bromphalaria havanensis</i>	1	<i>Turbonilla</i> spp.	4.5
<i>Cylindrella</i> spp.	1	<i>Alyama</i> spp.	5
<i>Laevapex peninsulae</i>	1	<i>Anomalocardia moberiana</i>	5
<i>Physella cubensis</i>	1	<i>Bulla striata</i>	5
<i>Planorbella scalaris</i>	1	<i>Caecum pulchellum</i>	5
<i>Polygrya</i> spp.	1	<i>Cardatis</i> spp.	5
<i>Pomacea paludosa</i>	1	<i>Chione cancellata</i>	5
<i>Littoridinops monroensis</i>	1.5	<i>Chione latirata</i>	5
<i>Pyrogophorus platyrachis</i>	2.5	<i>Corbula contracta</i>	5
<i>Creithidea beattyi</i>	3	<i>Lima pellucida</i>	5
<i>Battalaria minima</i>	4	<i>Marginella</i> spp.	5
<i>Brachidontes exustus</i>	4	<i>Meioceras nitidum</i>	5
<i>Cyrenoida floridana</i>	4	<i>Retusa sulcata</i>	5
<i>Littorina angulifera</i>	4	<i>Rissoina catesbyana</i>	5
<i>Melampus coffeus</i>	4	<i>Strigilla carnaria</i>	5
<i>Terebra dislocata</i>	4.5	<i>Tricolia bella</i>	5

Field observations indicate that the more vegetation cover, the less periphyton production. This trend is probably a result of light competition. The implications of this trend for management are significant. Periphyton assimilates nutrients at a much faster rate than do higher plants, which obtain most of their nutrients from the soil. Therefore, as Delivery Canal waters rich in nutrients are delivered to the treatment site, the amount of assimilable nutrients should decrease as a result of increasing tree cover and decreased light. If additional water delivery results in tree growth, the overall impact to Biscayne Bay may be an increased nutrient load due to the lack of periphyton production.

Groundwater influence can increase biomass and productivity of benthic algae and seagrass, and cause benthic zonation (Johannes, 1980; Herrera, 1994). The primary source of nutrients for seagrasses are surface sediments, and sediment nutrient availability may have a direct effect on what seagrass species dominate (Short 1985; Fourqurean et al. 1995; Williams, 1987). The high groundwater nutrient load in the inshore vegetation zone may explain the dominance of *H. wrightii*. Fourqurean et al (1995) found that *H. wrightii* outcompetes *T. testudinum* in Florida Bay when nutrient availability increases. About 30 years ago, Kohout and Kolipinski (1967) documented the dominance of *T. testudinum* within 300 m of shore just north of the L-31E study area. In the current study, *T. testudinum* does not become abundant until 600m offshore. Our groundwater data indicates very little groundwater discharge beyond 400 m (**Section D-2**). The relative lack of *T. testudinum* inshore may be a response to a decline in coastal water quality parameters, which may increase nutrient loads and algal blooms.

A north-south difference was also not apparent in *P. capitatus*. Again, this may be a result of groundwater nutrient sources, because the *P. capitatus* sites were all between 300-425 meters from shore, which is still in the zone where groundwater discharge is present.

Seasonality. Based on the published literature, we expected a seasonal influence on seagrass growth, with highest growth rates in summer (Tomasko and Dawes, 1990; Barber and Behrens, 1985). In our studies, strong seasonality was documented for one of the major seagrass species, but not the second. *T. testudinum* growth was strongly seasonal (**Figure H-2-2a & b**), with productivity rates 58% higher and the biomass 72% higher in July than in February. Our measured productivity rates and biomass were similar to those reported for *T. testudinum* in Florida Bay (Tomasko and Lapointe, 1991). Unlike *T. testudinum*, *H. wrightii* did not show the expected seasonal trend (**Figure H-2-1a & b**). No seasonal difference was present in new leaf growth and the biomass actually decreased in the summer.

H. wrightii is reported to be dominant in areas close to a strong source of phosphorus, and *T. testudinum* is dominant further from the nutrient source (Powell et al. 1991; Fourqurean et al. 1992). *H. wrightii* is less efficient at utilizing nutrients when availability is low and has a much higher demand for sediment nutrients than *T. testudinum* (Fourqurean et al 1992). *H. wrightii* is dominant in the inshore areas where salinities are lower and nutrient concentrations higher than further offshore.

The inshore nutrient load also encourages the growth of filamentous algae. In the summer, when the nutrient load increases, cover of filamentous algae increases to almost 100%, smothering the *H. wrightii* beds, and inhibiting growth (**Figure H-2-3**). This increased algal cover may explain the lack of seasonality in *H. wrightii* biomass and growth rate.

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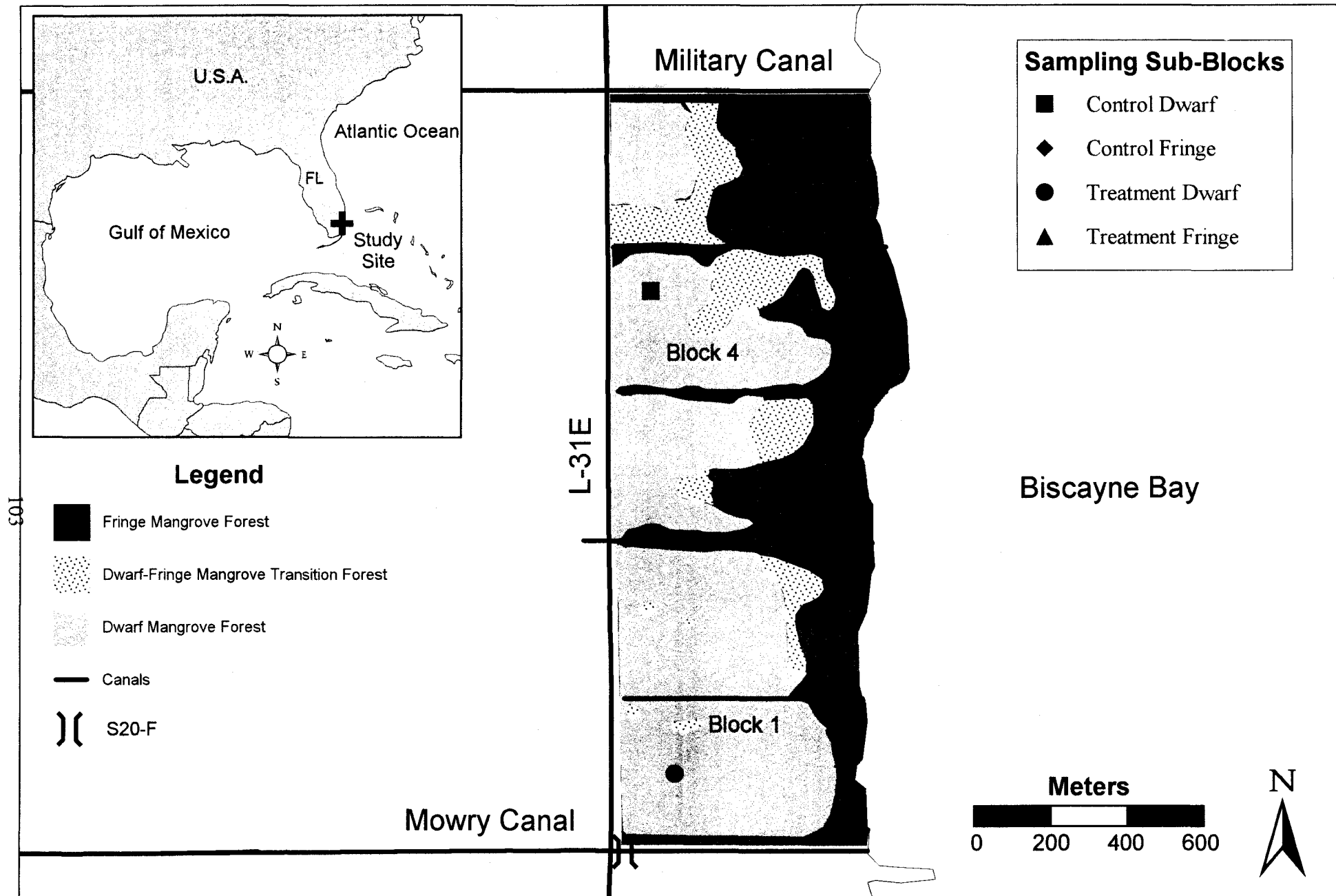
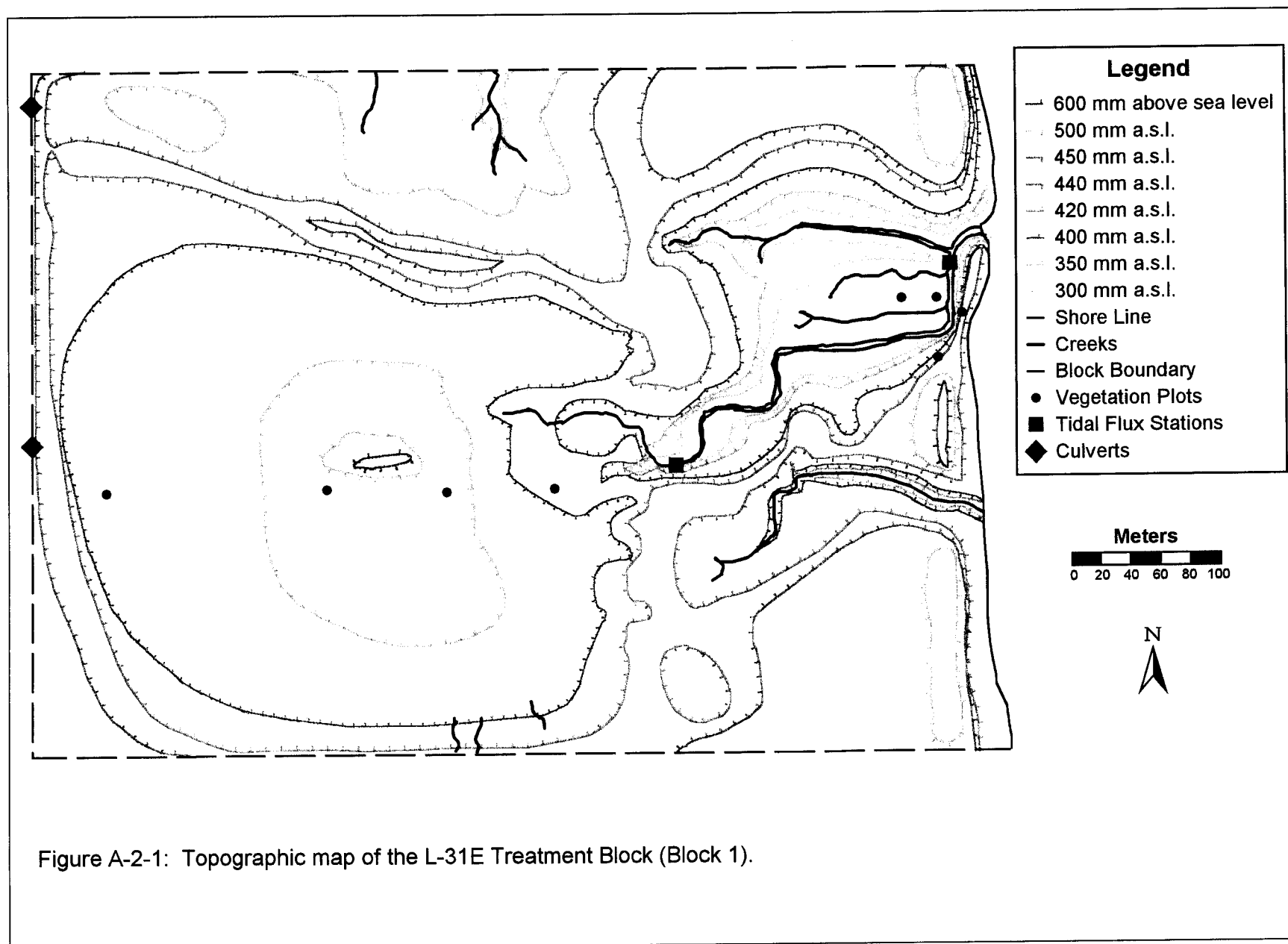


Figure A-1-1: Map of the L-31E study site with sub-block locations. Each sub-block contained four sampling plots oriented from east to west, except for the Control Dwarf, which contained three.



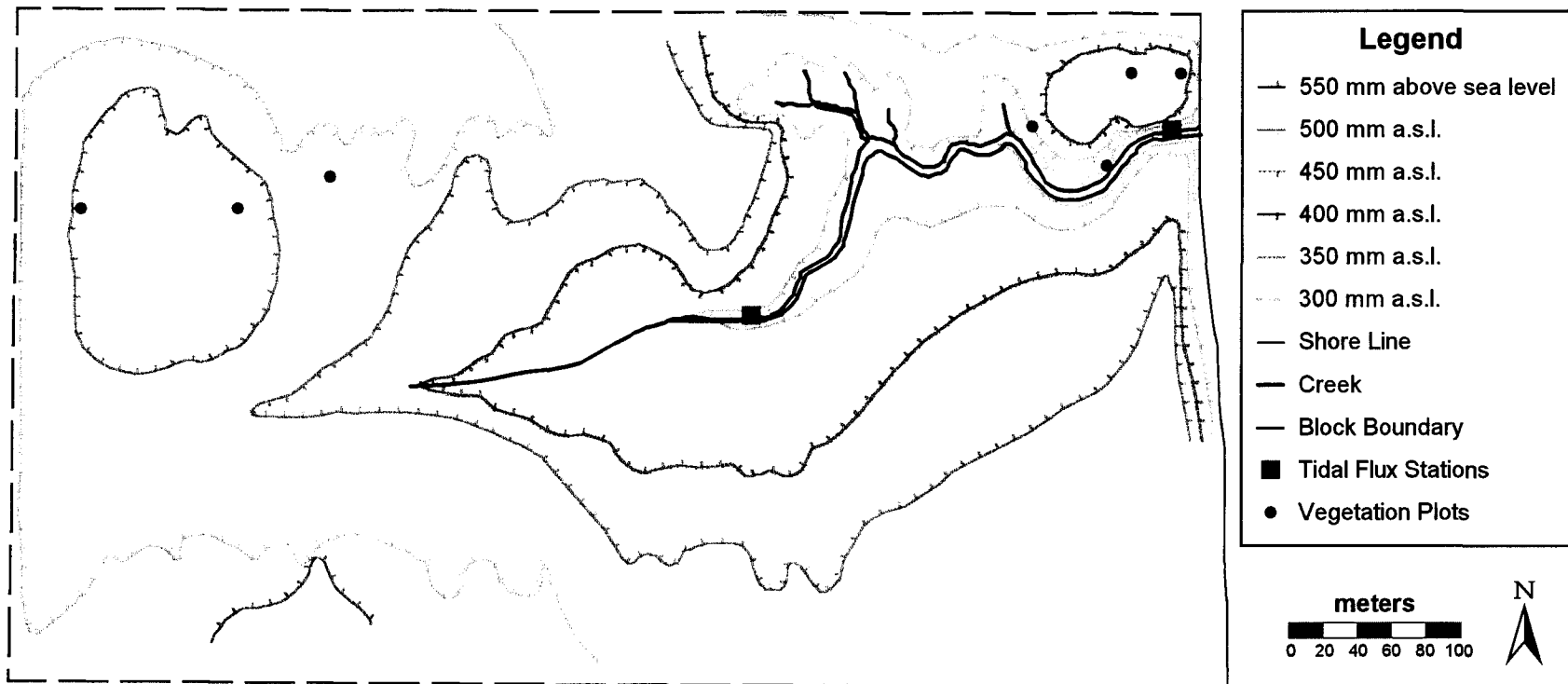


Figure A-2-2: Topographic map of the L-31E Control Block (Block 4).

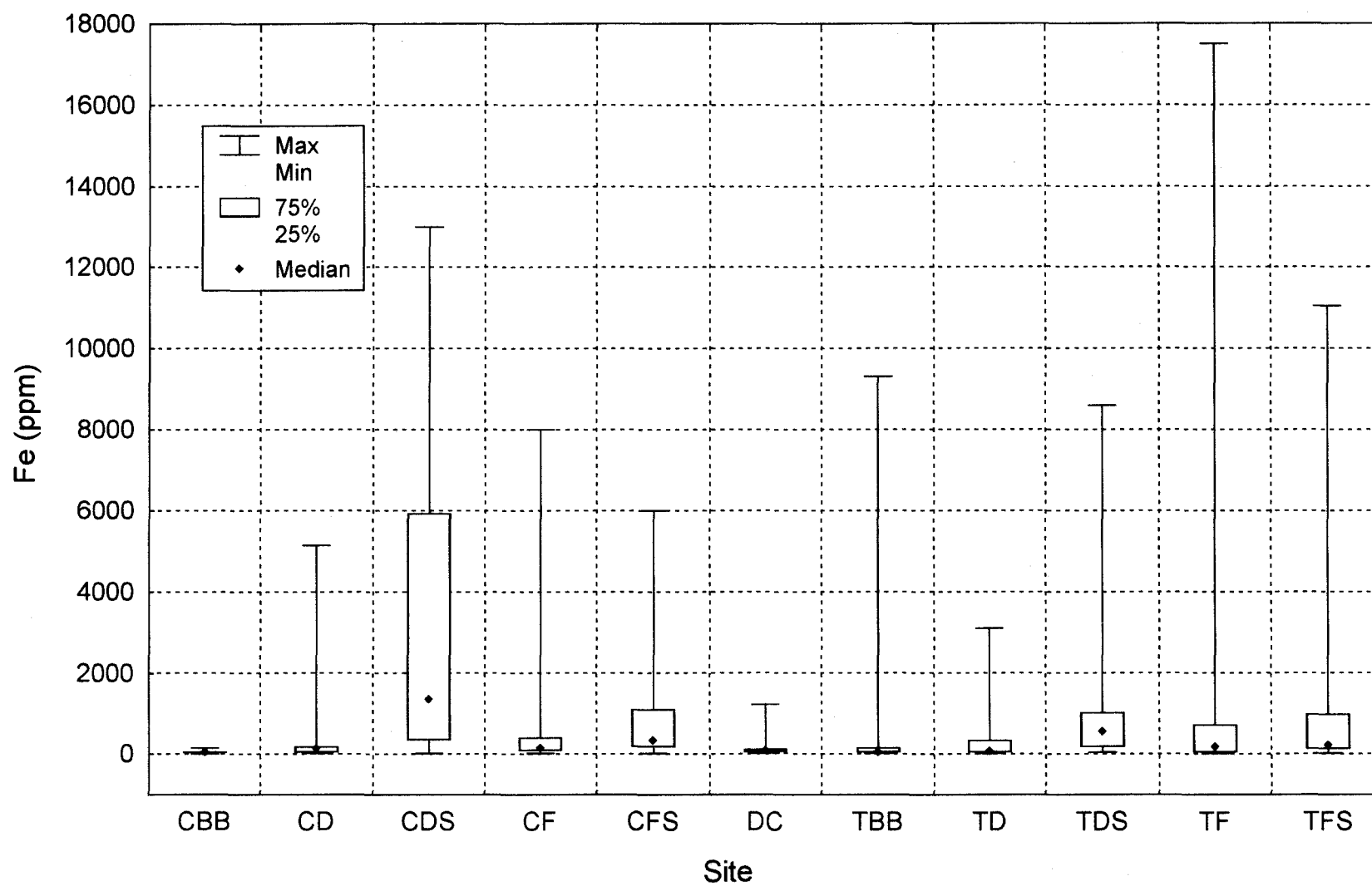


Figure B-2-1: Concentrations of iron in surface and pore water samples in the L-31E study area. Acronyms as in Table B-2-1.

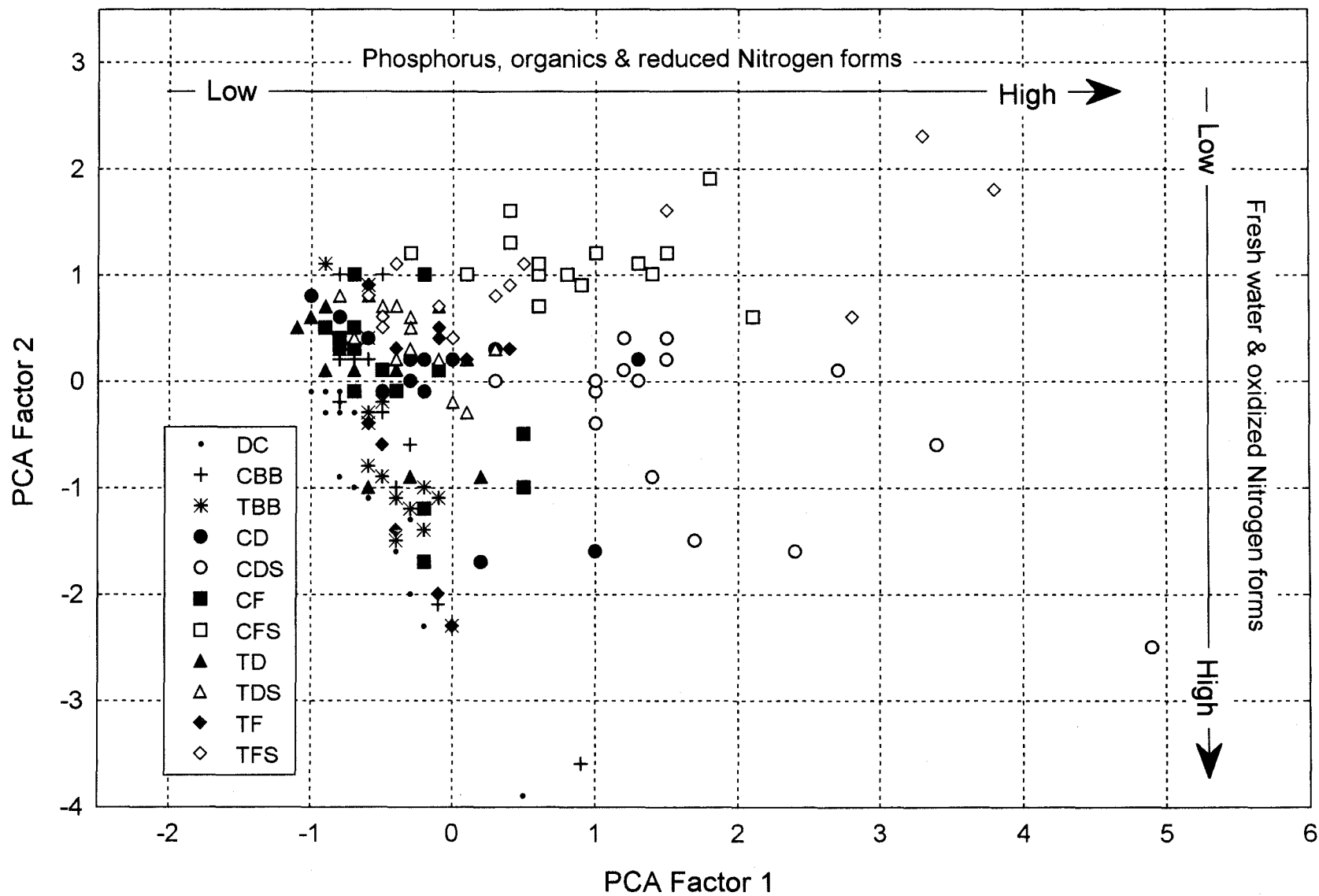


Figure B-2-2: PCA factor scores for seven surface and four pore water sites between July 1995 and February 1998. Acronyms as in Table B-1-2.

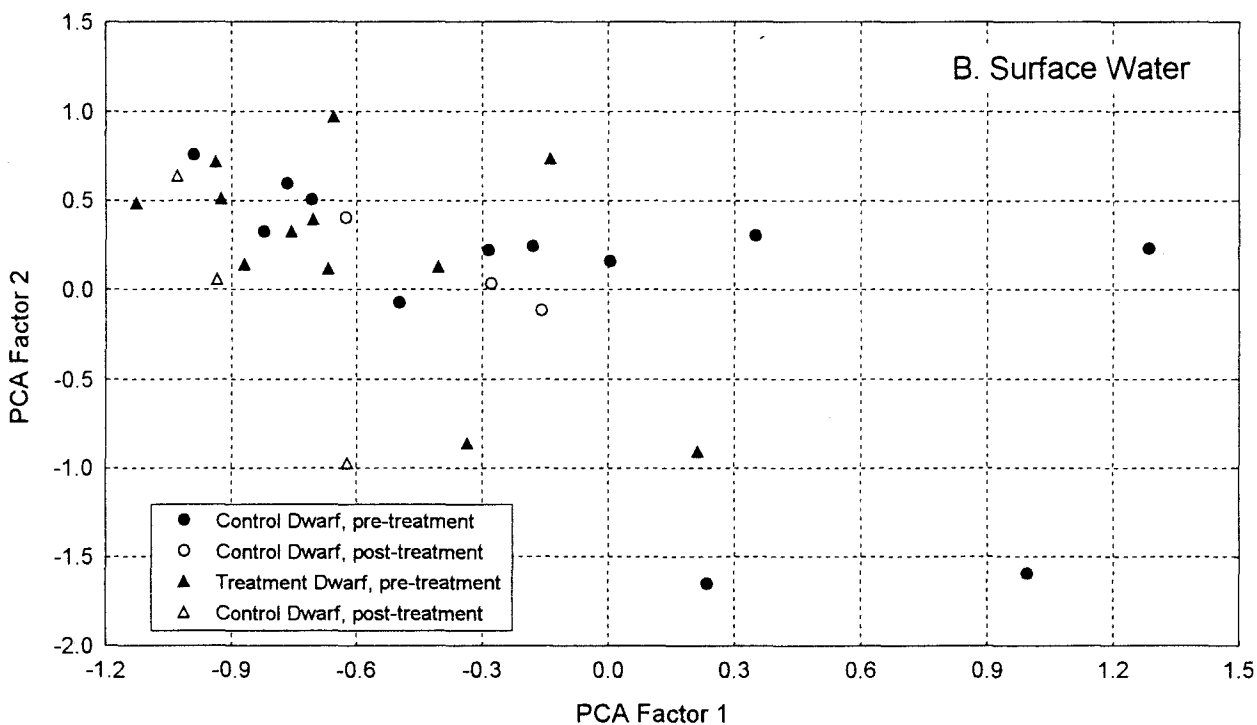
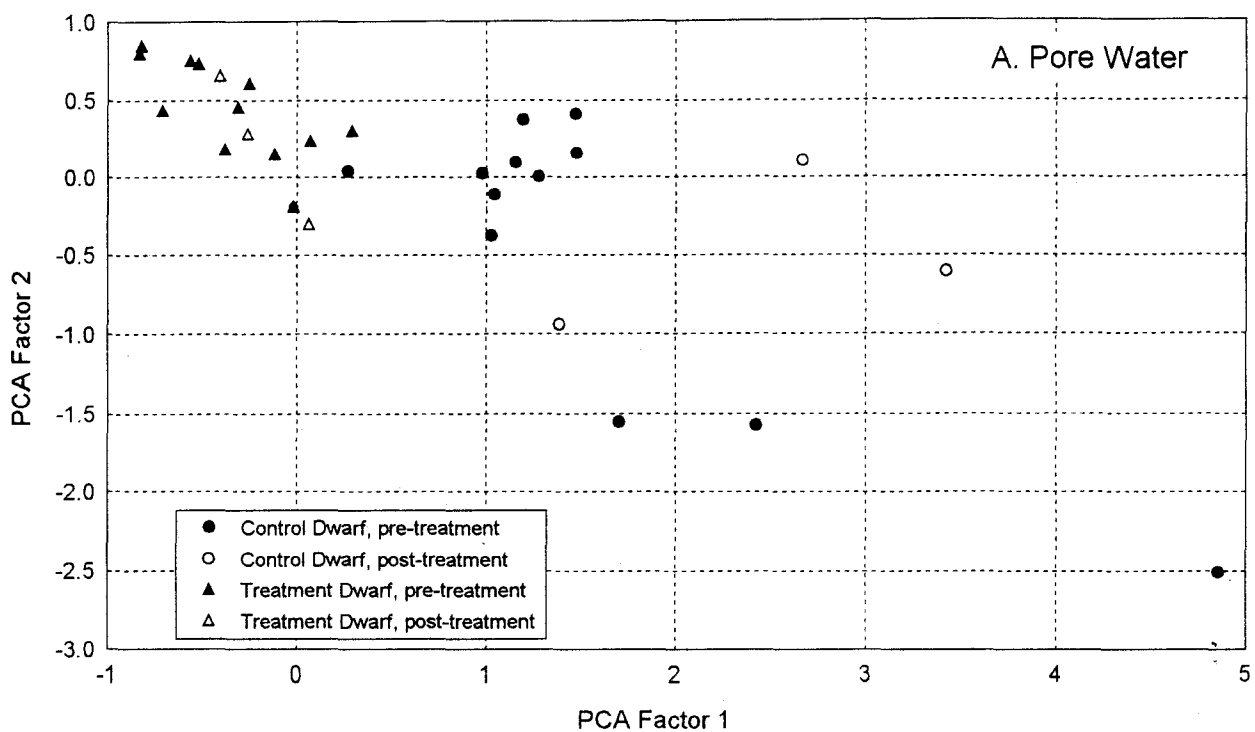


Figure B-2-3: Pre and post treatment PCA factor scores for Treatment Dwarf and Control Dwarf (A) pore water and (B) surface water samples.

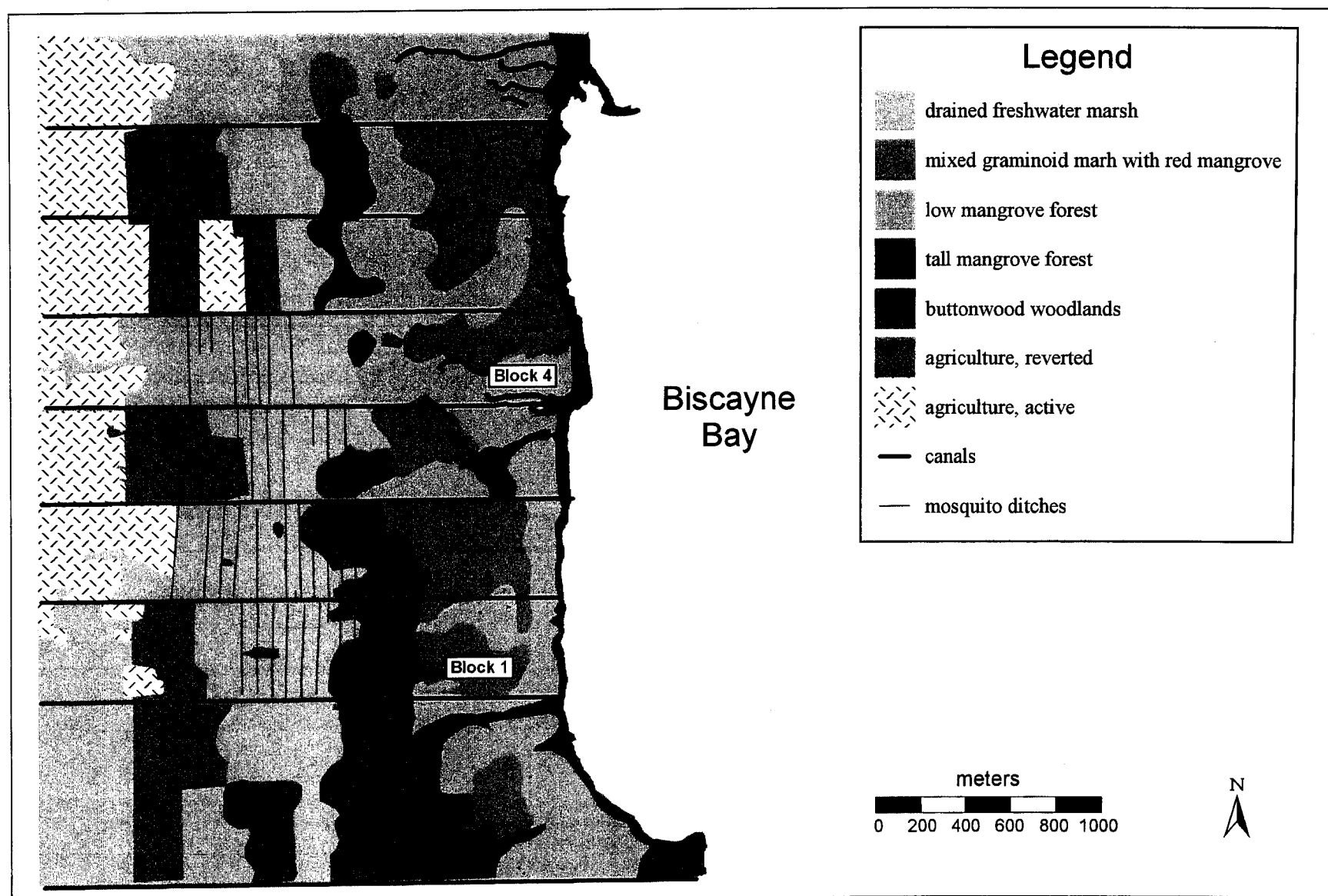
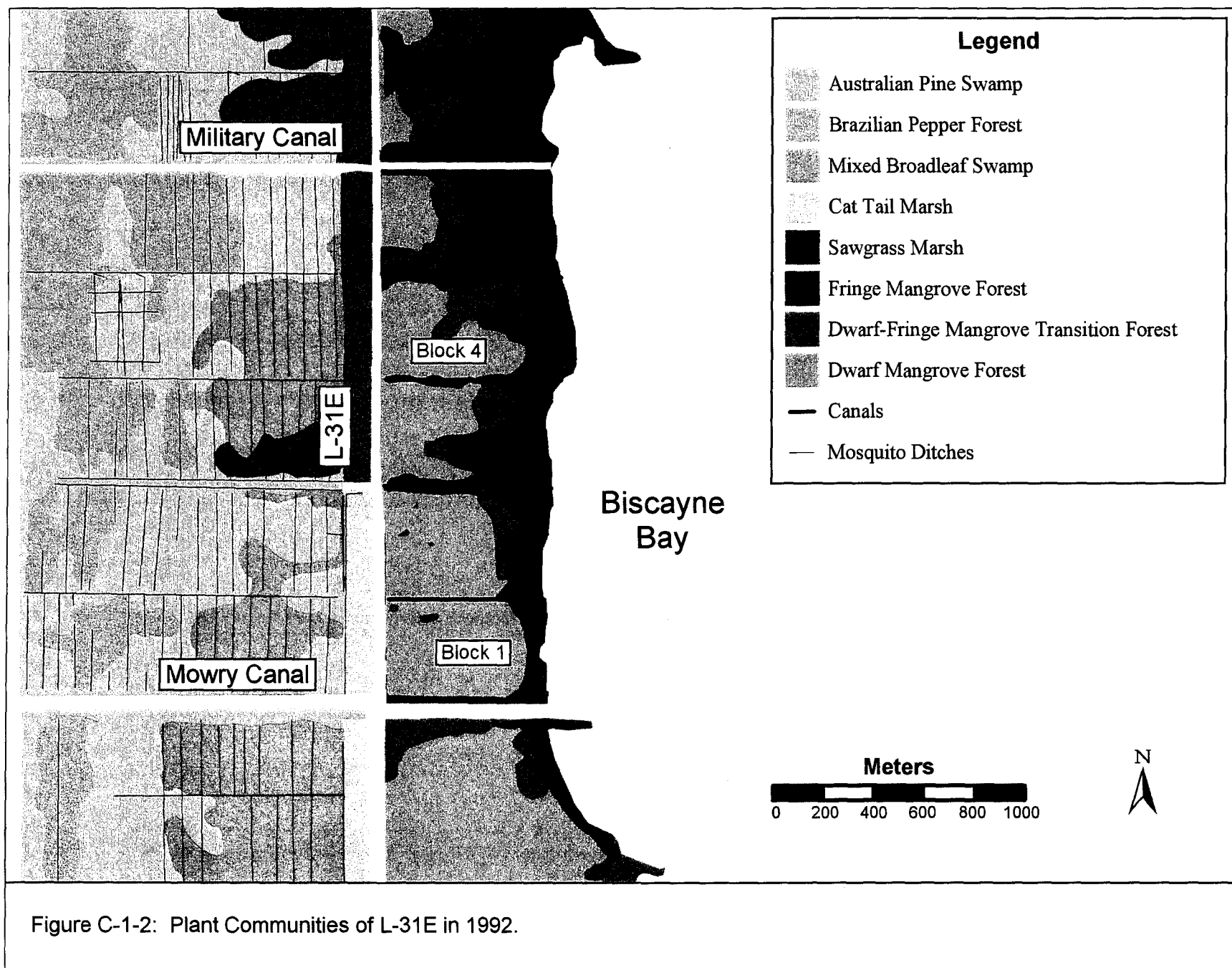


Figure C-1-1: Plant communities of L-31E in 1940.



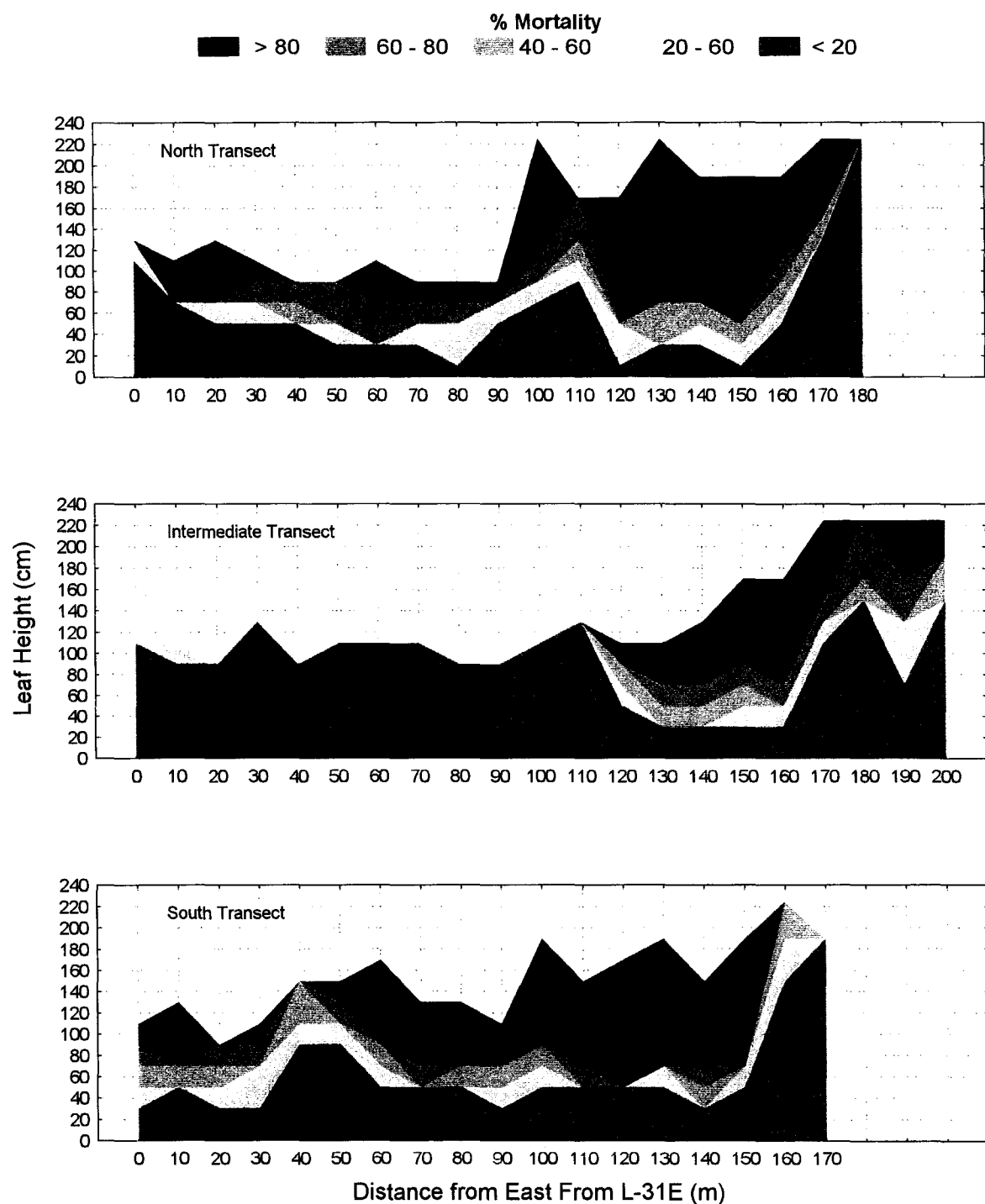


Figure C-3-1: Profile of leaf mortality resulting from freeze of Jan 9-10, 1996 in Control Dwarf sub-block. Transects are 60 m apart running eastward from base of L-31E levee.

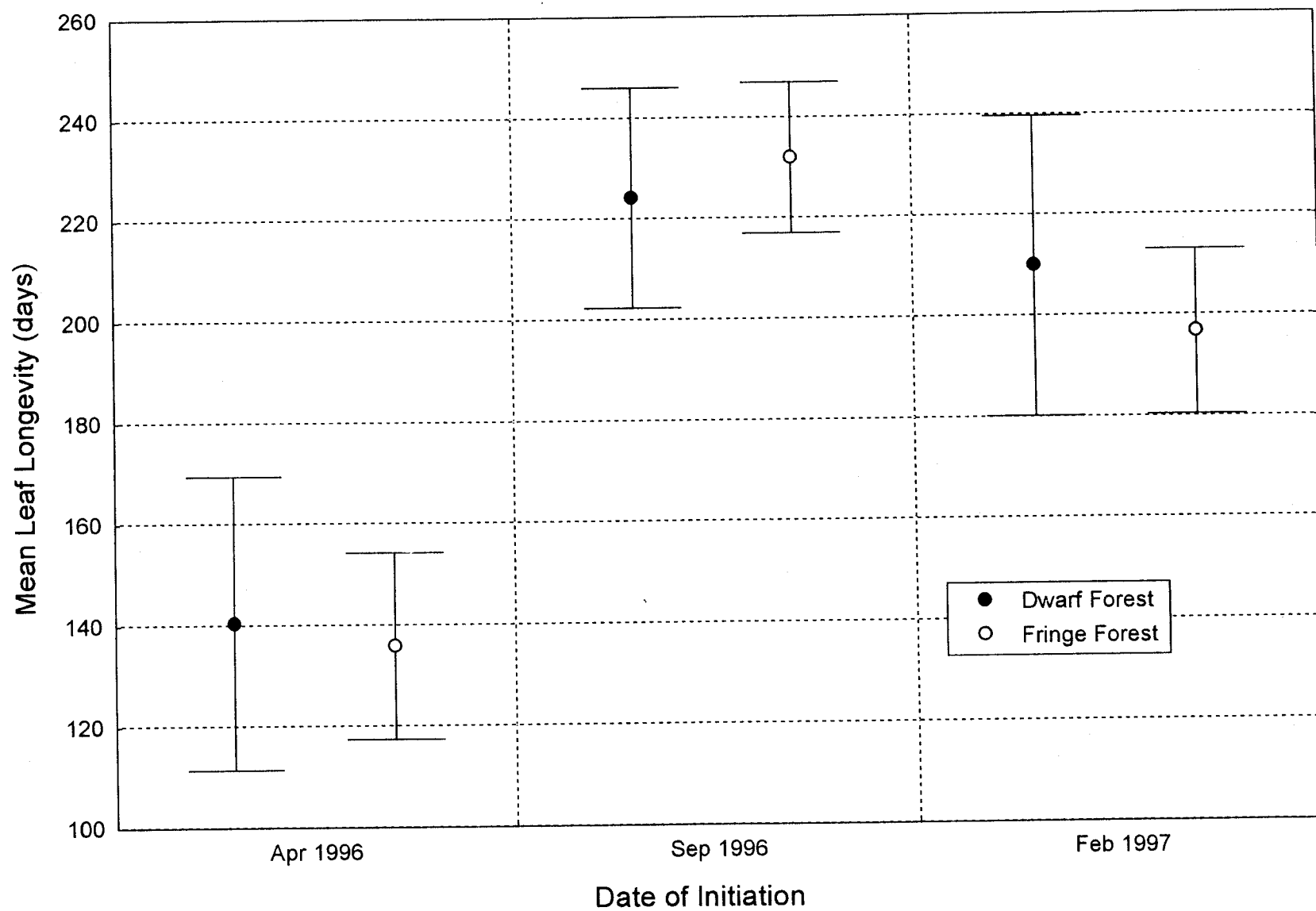


Figure C-5-1: Mean life span (± 1 S.E.) of *R. mangle* leaves in Fringe and Dwarf mangrove forests at L-31E.

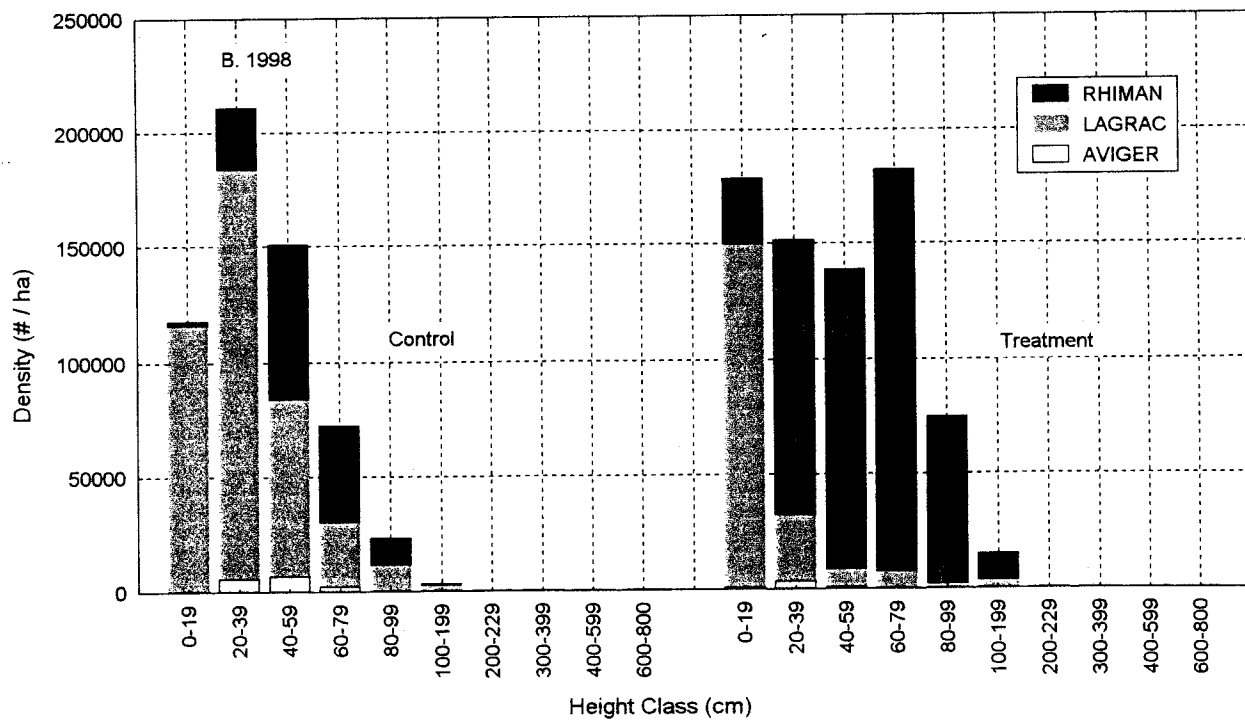
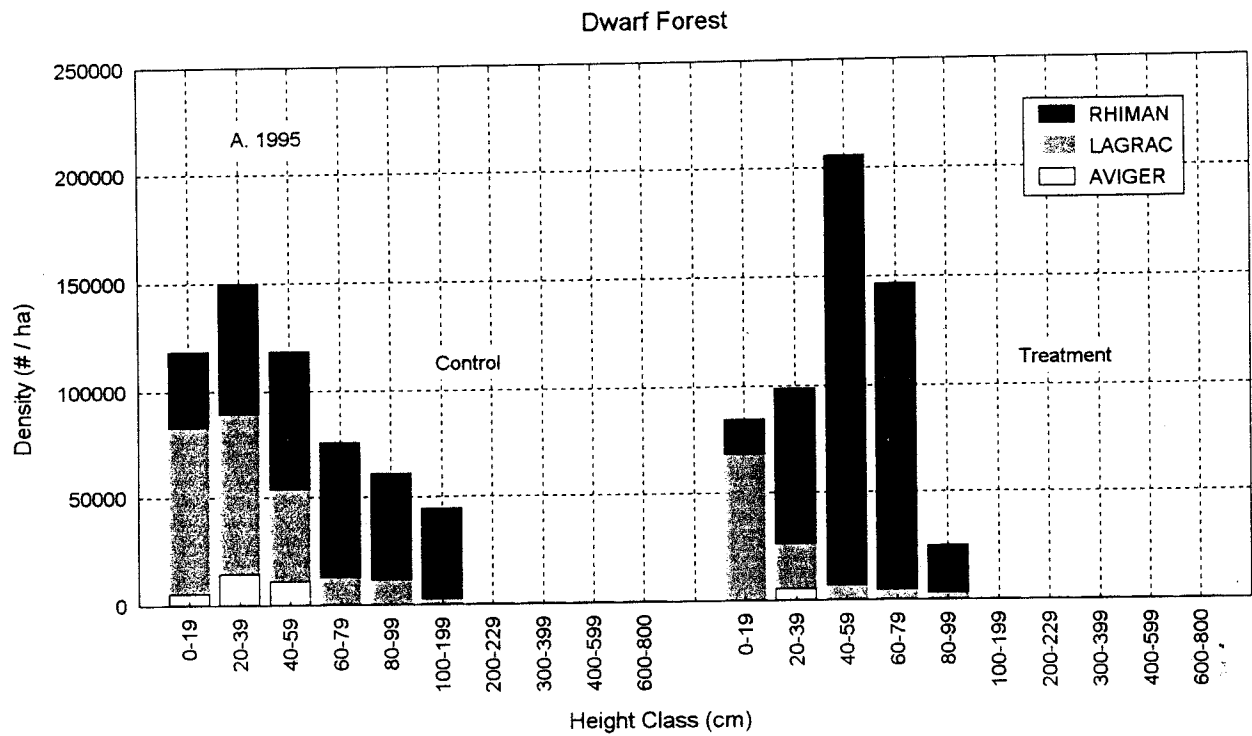


Figure C-6-1: Height distribution of mangrove stems in the L-31E dwarf forest in (A.) 1995 and (B.) 1998.

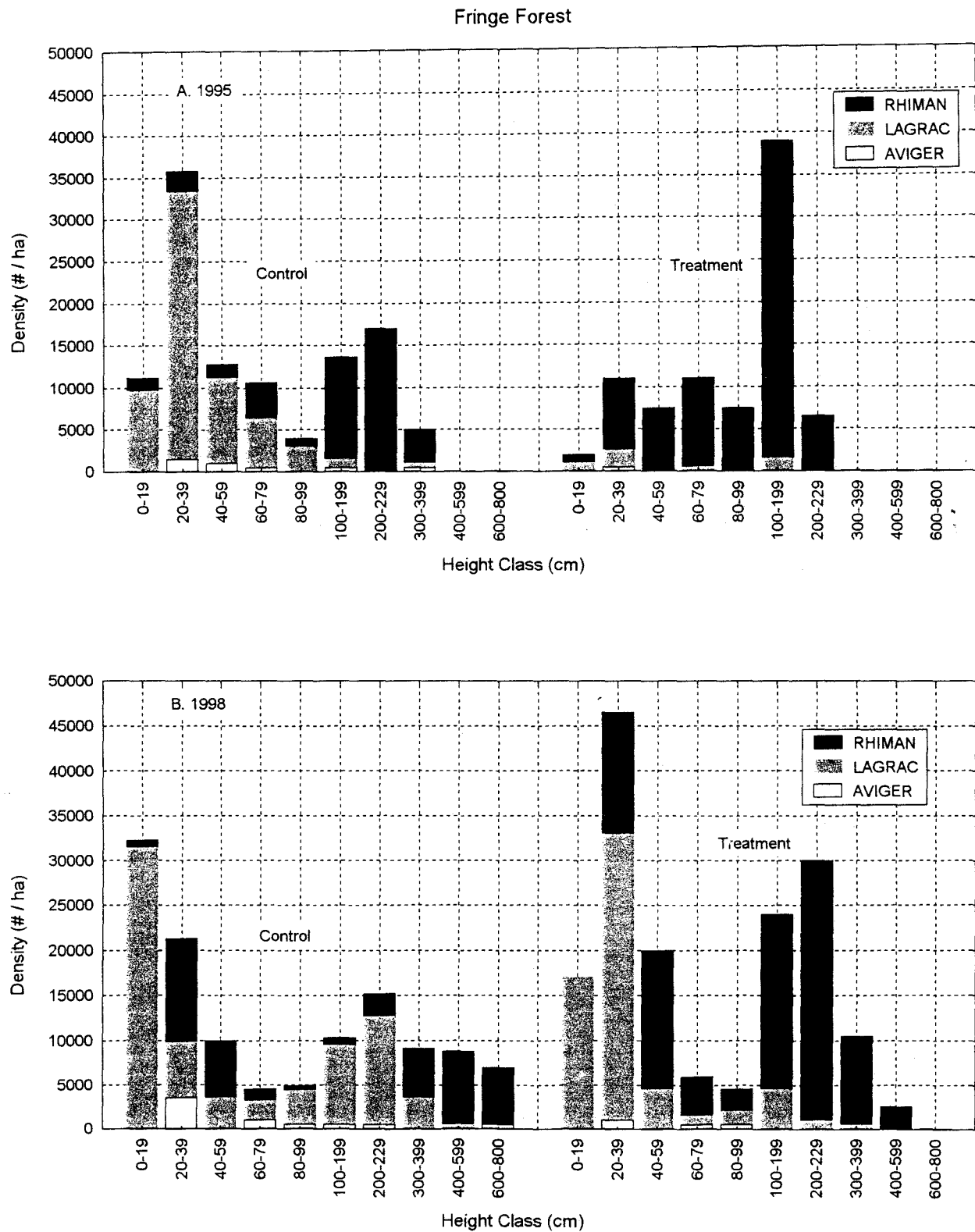


Figure C-6-2: Height distribution of mangrove stems in the L-31E fringe forest in (A.) 1995 and (B.) 1998.

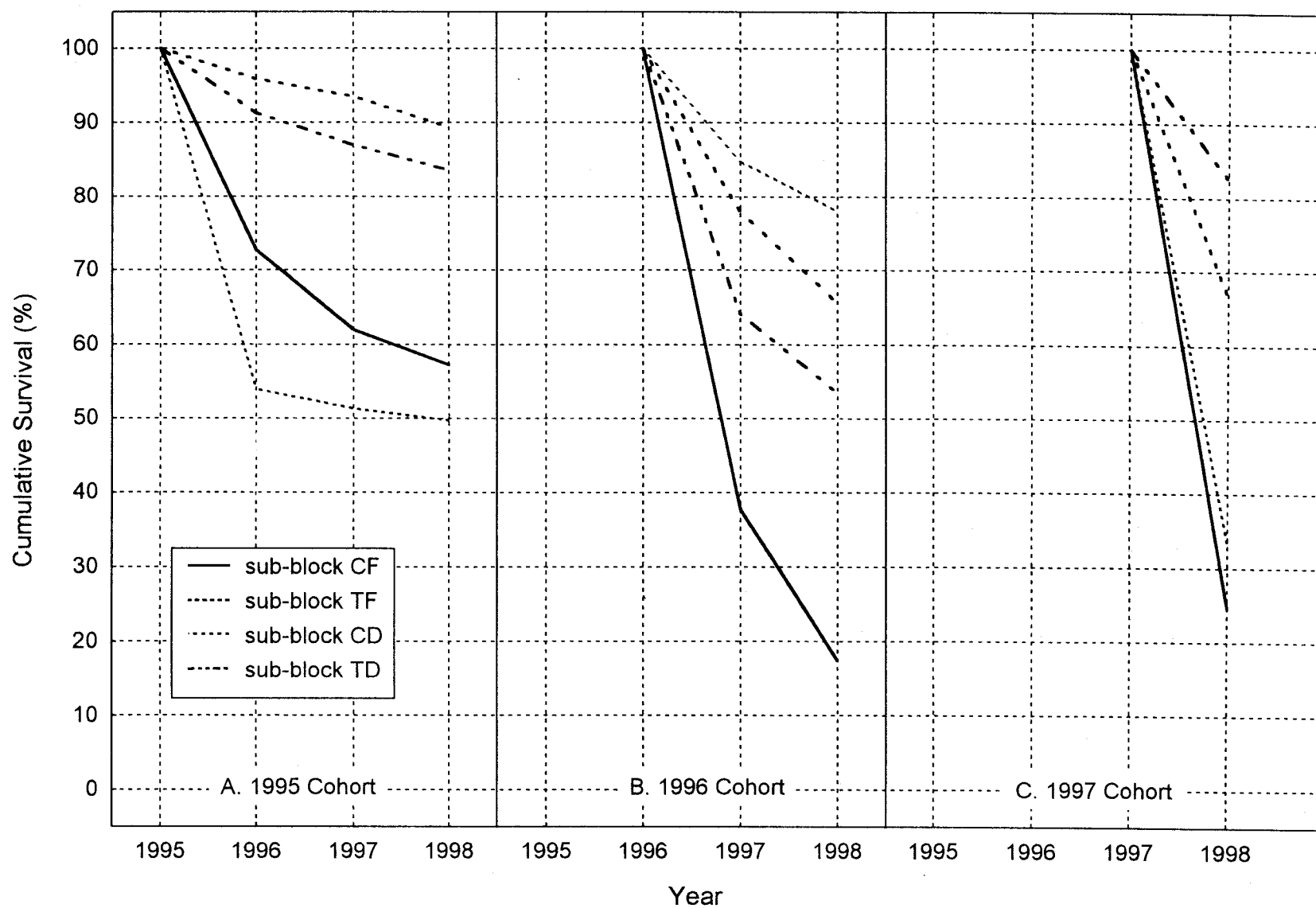


Figure C-6-3: Cumulative survival of mangrove individuals in four L-31E sub-blocks. Data are mixed-species cohorts first tagged in (A.) 1995, (B.) 1996, and (C.) 1997.

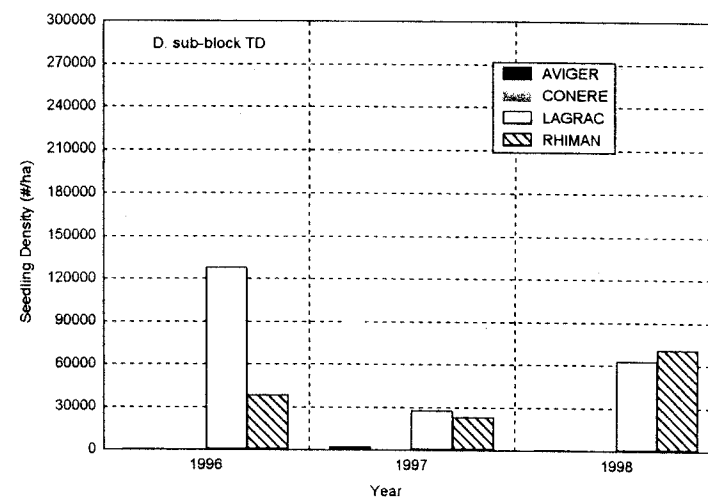
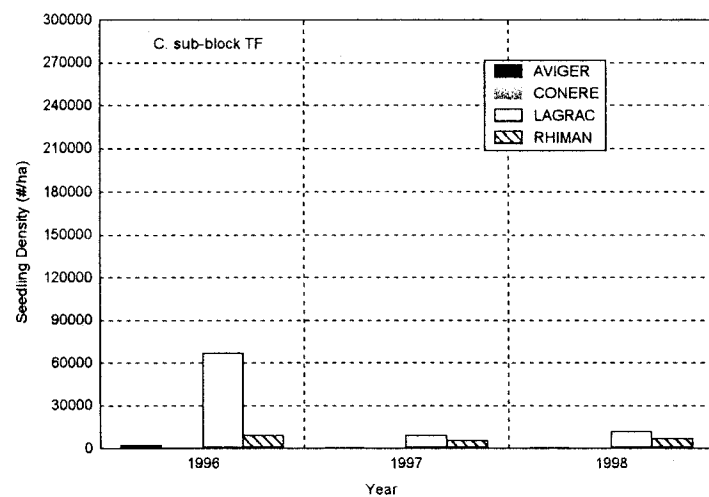
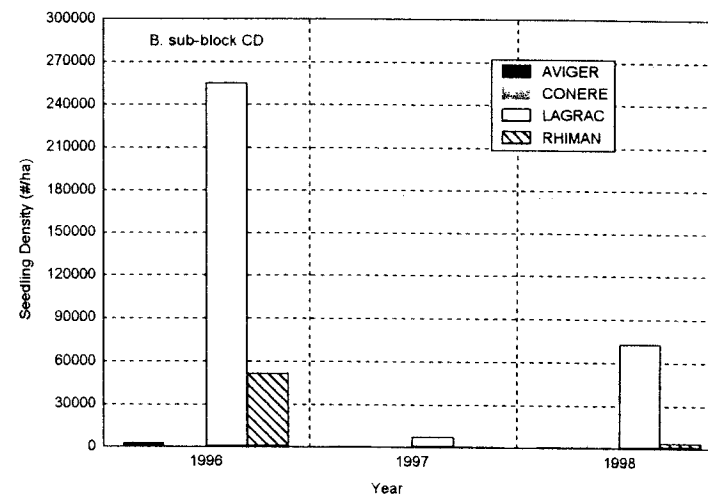
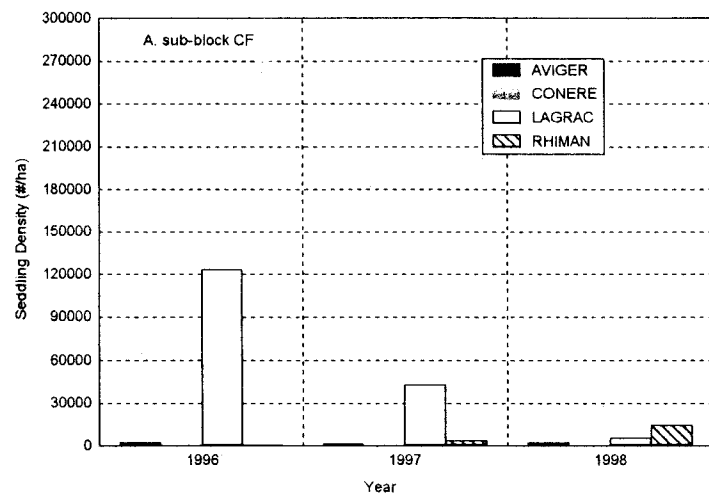


Figure C-6-4: Seedling recruitment of four L-31E mangrove species, 1996-1998 in sub-blocks (A.) CF, (B.) CD, (C.) TF, and (D.) TD.

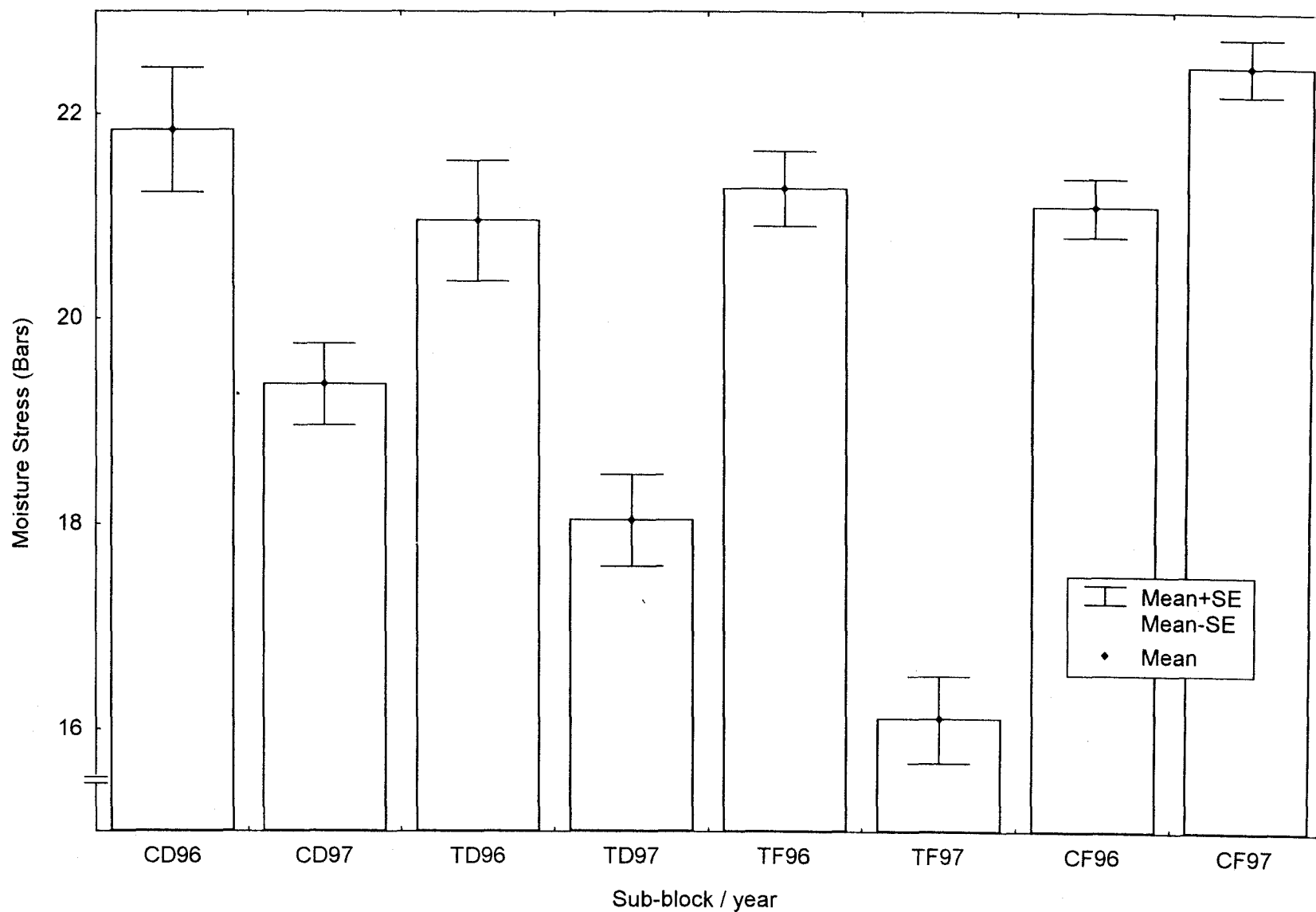


Figure C-9-1: Mean plant moisture stress at L-31E study area (Apr. 96 & Apr. 97). Means are based on pressure bomb readings of 6 *R. mangle* and 6 *L. racemosa* trees per sampling period per site.

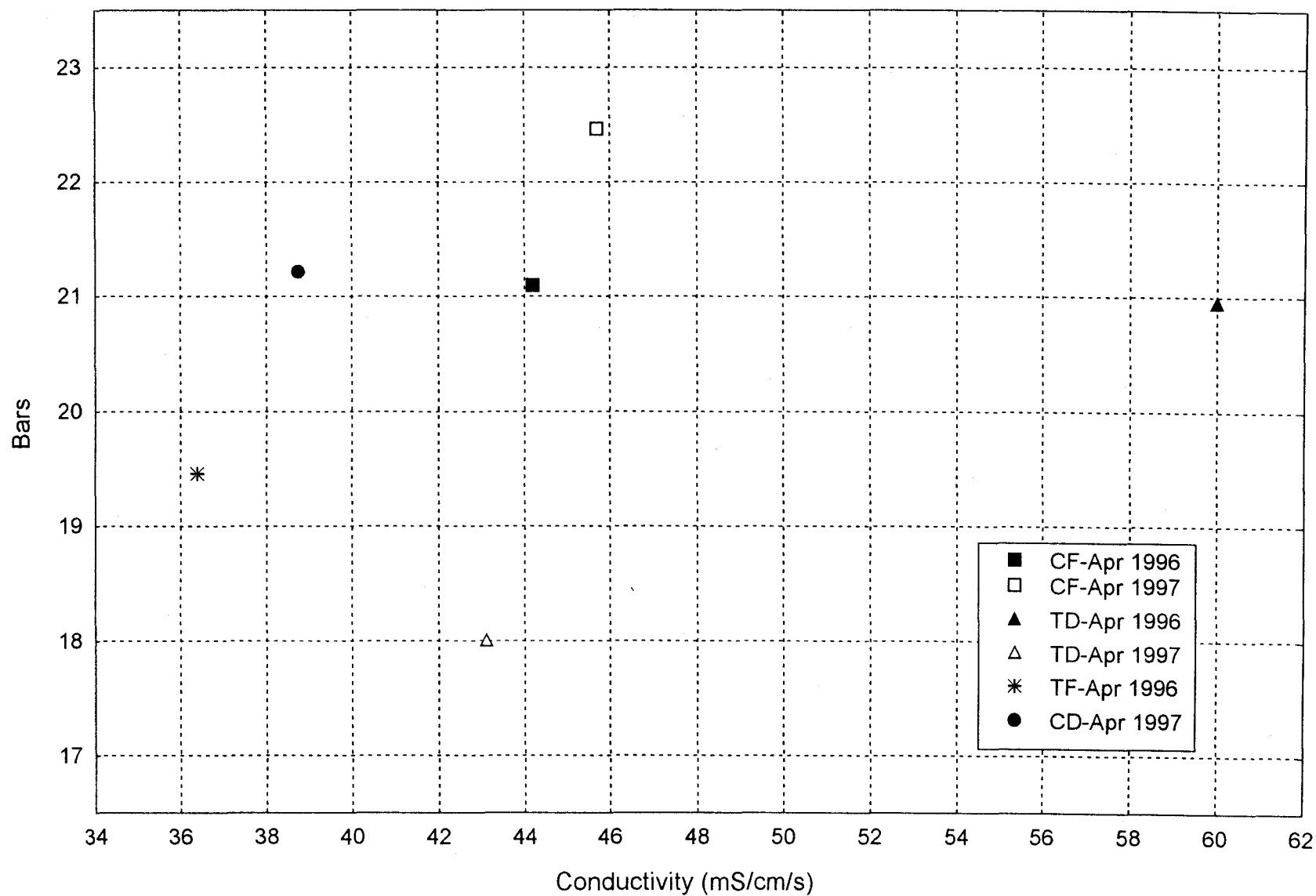


Figure C-9-2: Relationship between pore water conductivity and plant moisture stress during two sampling period in April 1996 and 1997.

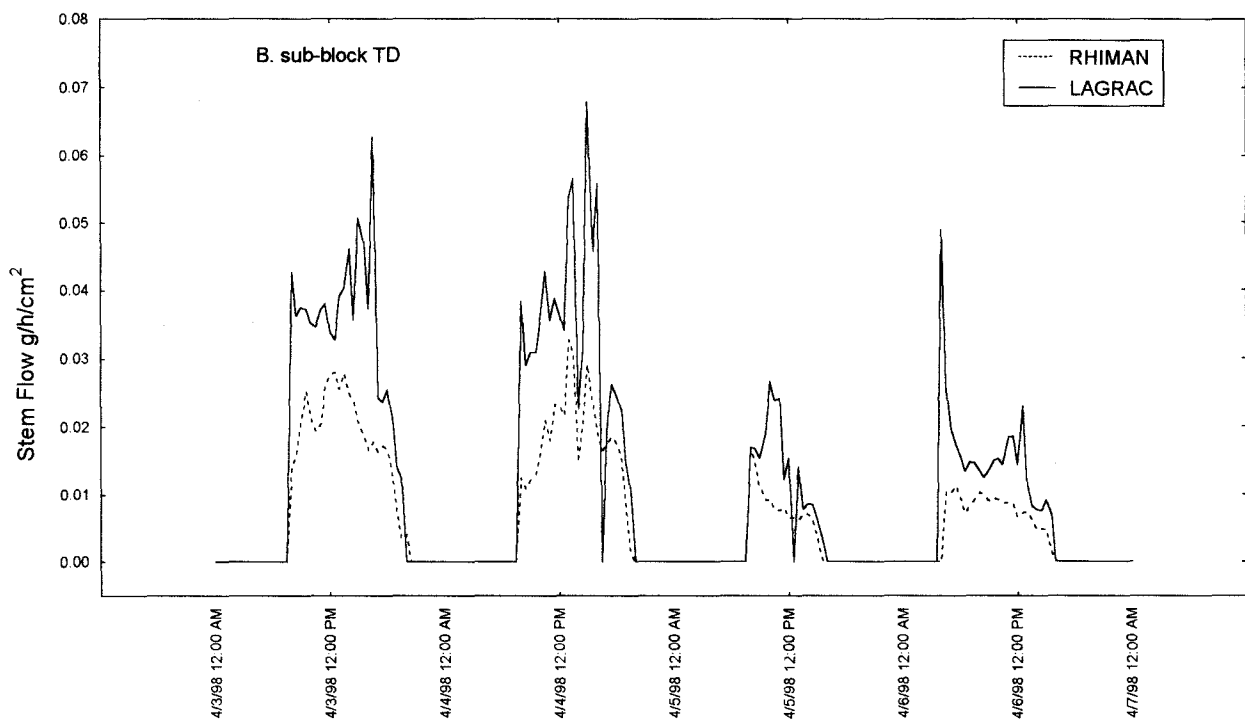
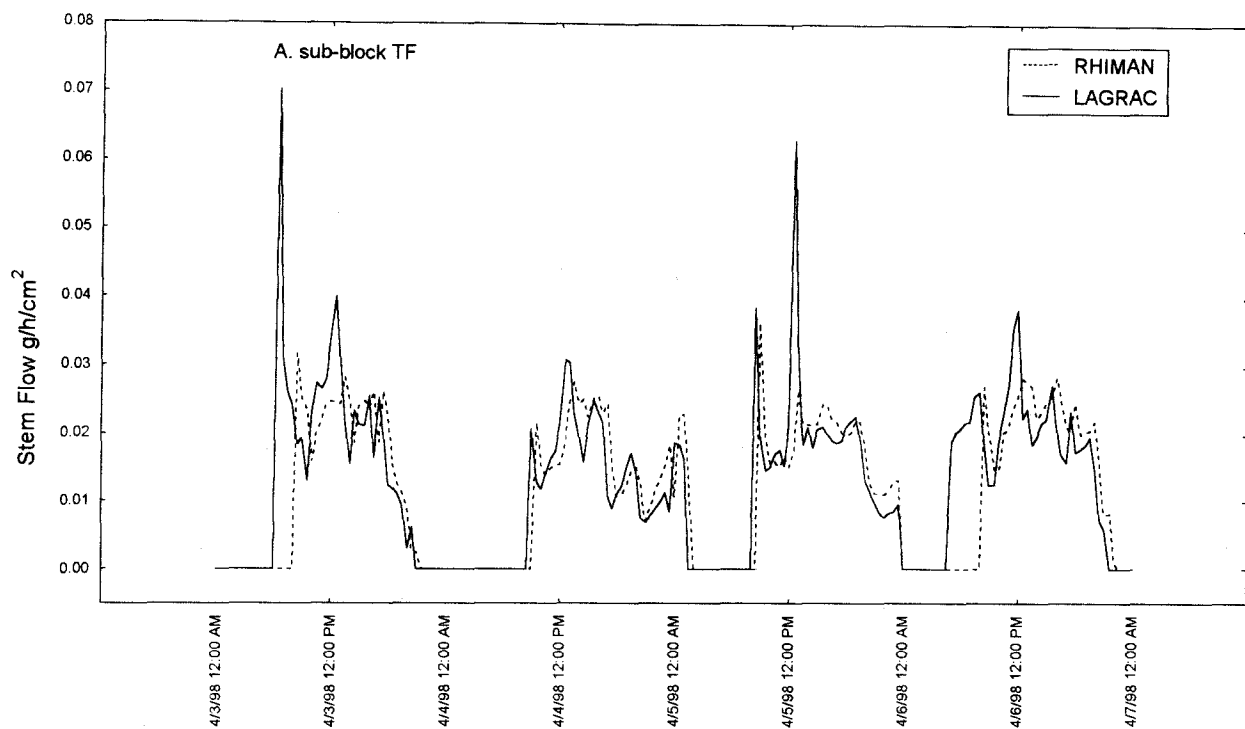


Figure C-10-1: Diurnal tracks of sap flow for two mangrove species in sub-blocks (A.) TF and (B.) TD, L-31E study site.

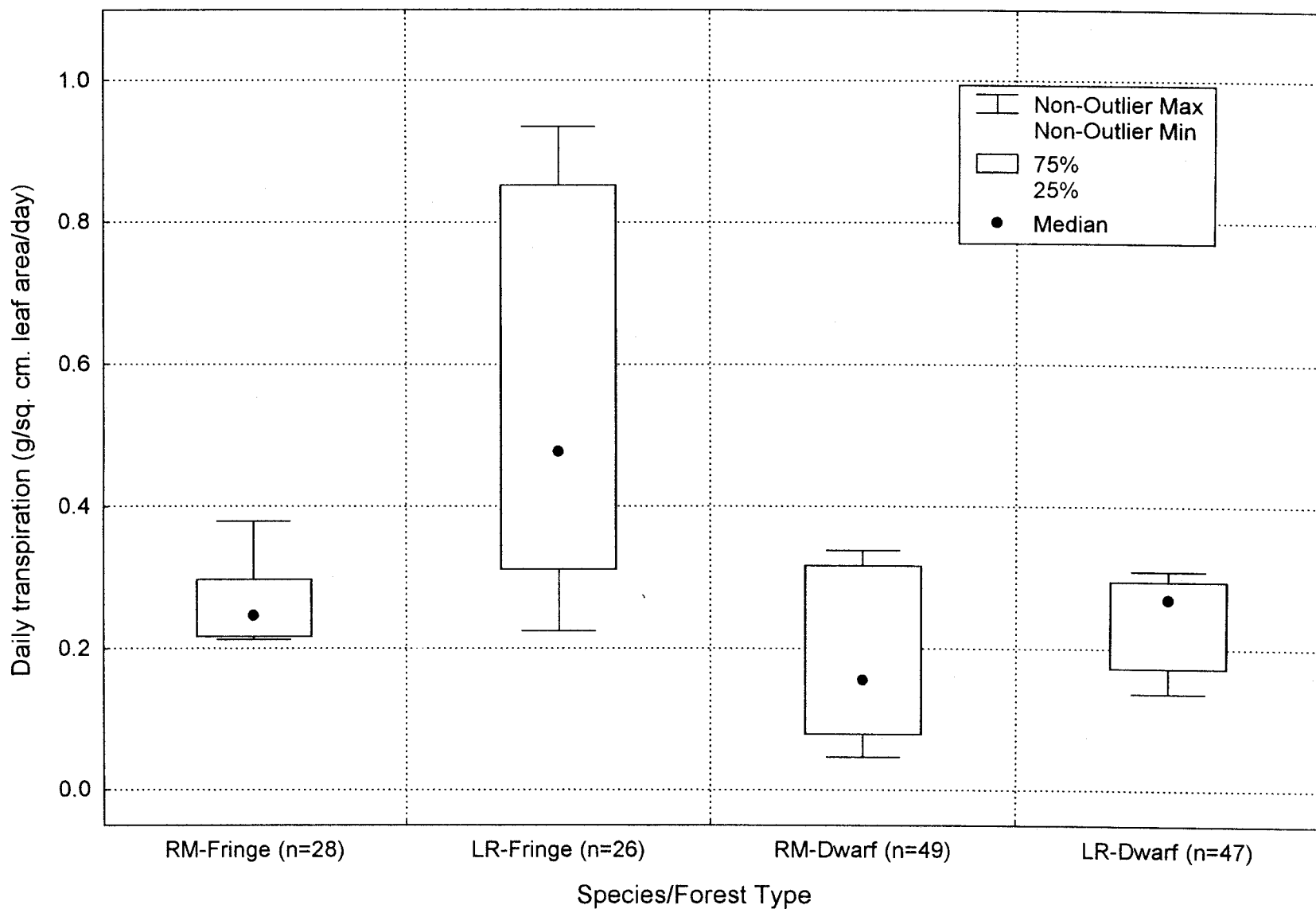


Figure C-10-2: Daily transpiration of two mangrove species in Dwarf and Fringe forest based on 150 complete tree-days of sap-flow monitoring, Apr-Jun 1998.

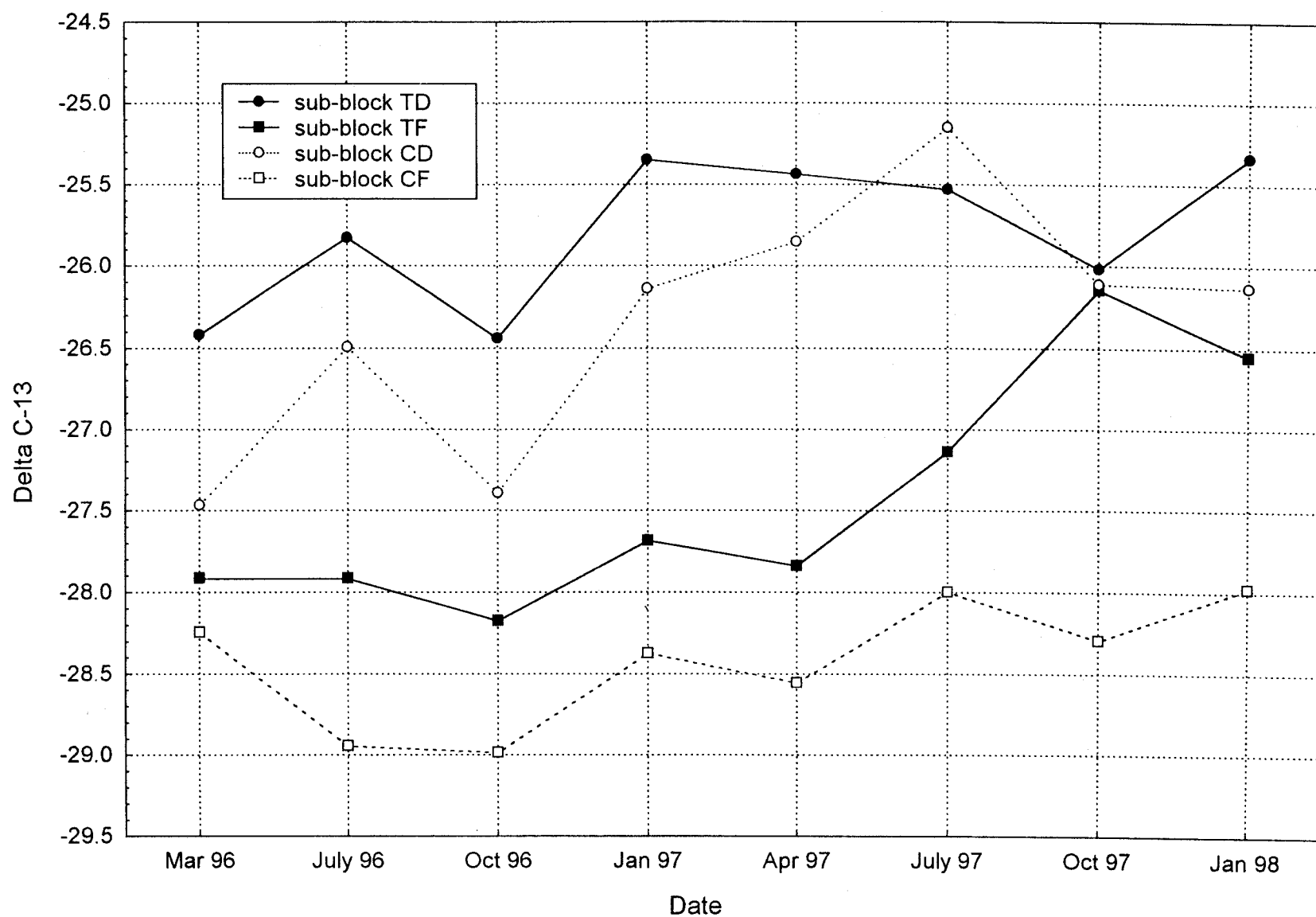


Figure C-11-1: Mean red mangrove leaf delta C-13, March 96 - Jan 98.

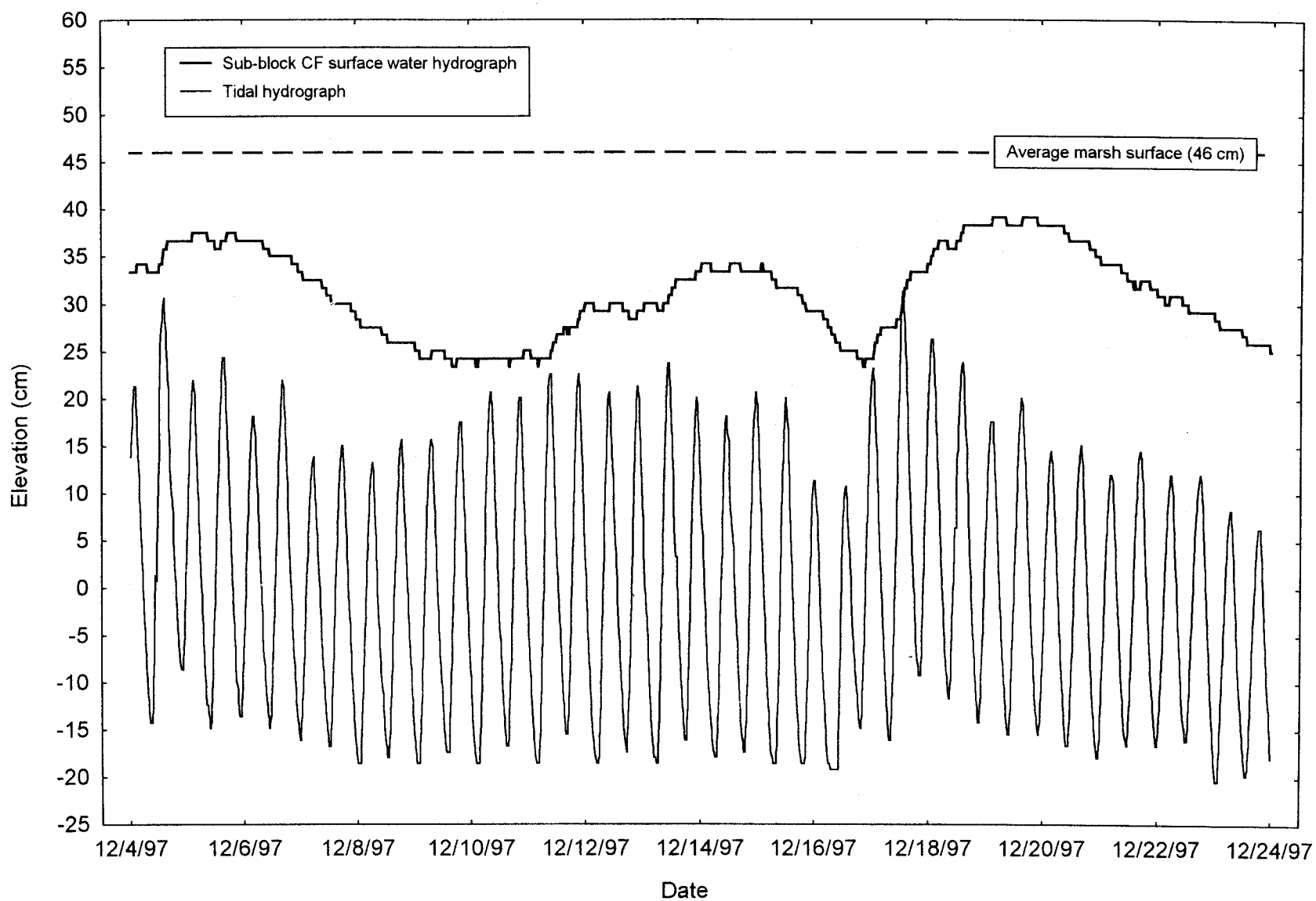


Figure D-2-2: Relationship among tide stage, control fringe marsh surface, and soil water surface for Dec. 4-24, 1997.

L31-E CANAL DISCHARGE INTO SPREADER CANAL

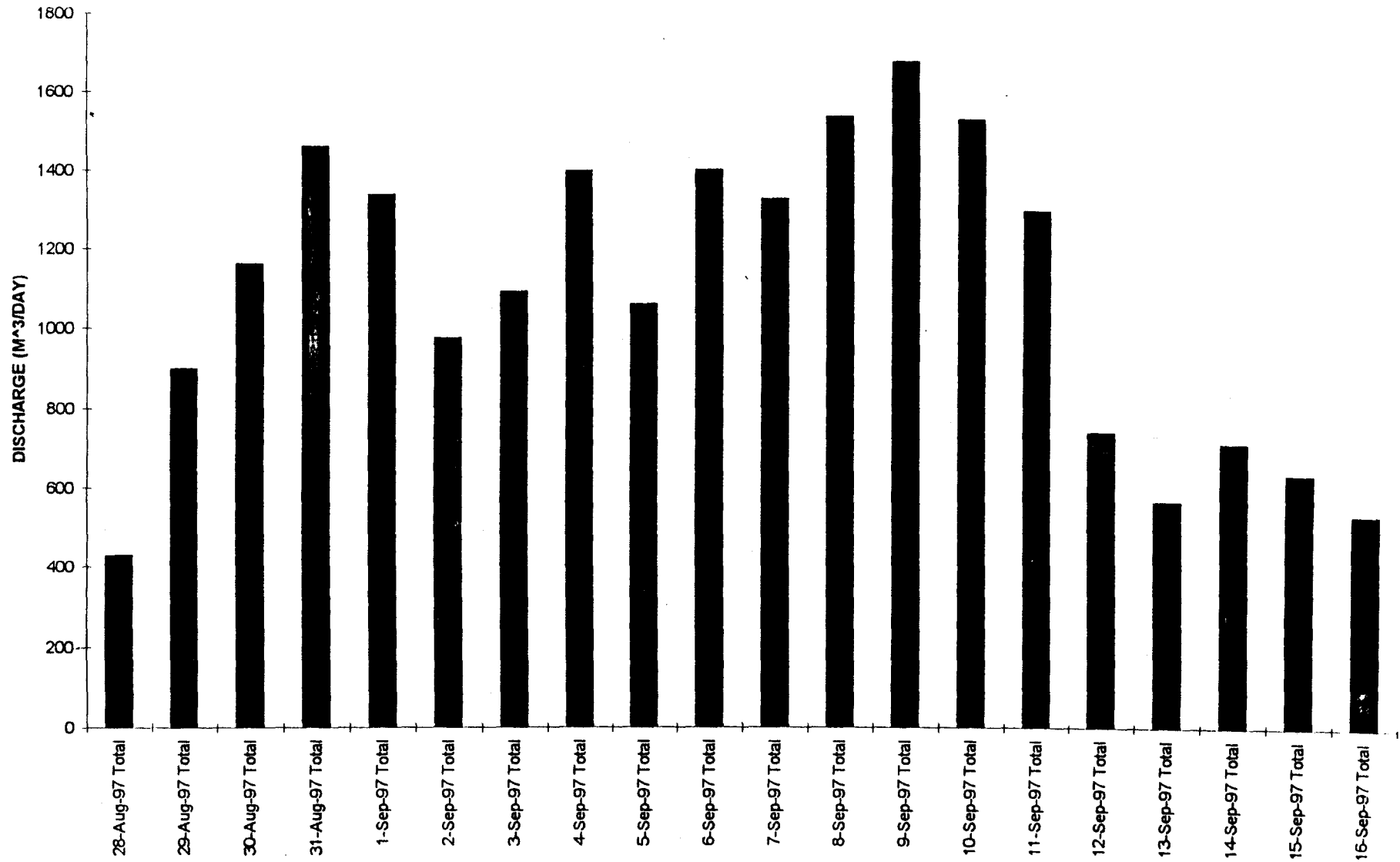


Figure D-4-1: Daily average discharge from L31-E canal to the Spreader Canal in the treatment block.

HEAD_CM (stage.sta)					
	Count	Cumul. Count	Percent	Cumul. Percent	
-1.0000<=x<0.00000	466	466	24.39791	24.3979	
0.00000<=x<1.00000	68	534	3.56021	27.9581	
1.00000<=x<2.00000	236	770	12.35602	40.3141	
2.00000<=x<3.00000	172	942	9.00524	49.3194	
3.00000<=x<4.00000	194	1136	10.15707	59.4764	
4.00000<=x<5.00000	226	1362	11.83246	71.3089	
5.00000<=x<6.00000	177	1539	9.26702	80.5759	
6.00000<=x<7.00000	219	1758	11.46597	92.0419	
7.00000<=x<8.00000	46	1804	2.40838	94.4503	
8.00000<=x<9.00000	47	1851	2.46073	96.9110	
9.00000<=x<10.0000	33	1884	1.72775	98.6387	
10.0000<=x<11.0000	14	1898	.73298	99.3717	
11.0000<=x<12.0000	12	1910	.62827	100.0000	
Missing	0	1910	0.00000	100.0000	

Descriptive Statistics (stage.sta)						
	Mean	Std.Dev	No. cases		N	Missing
HEAD_CM	3.072419	3.078028	Minimum	Maximum	1910	0

Figure D-4-2: Relationship between head (cm above weir crest) and time (as percent and cumulative percent).

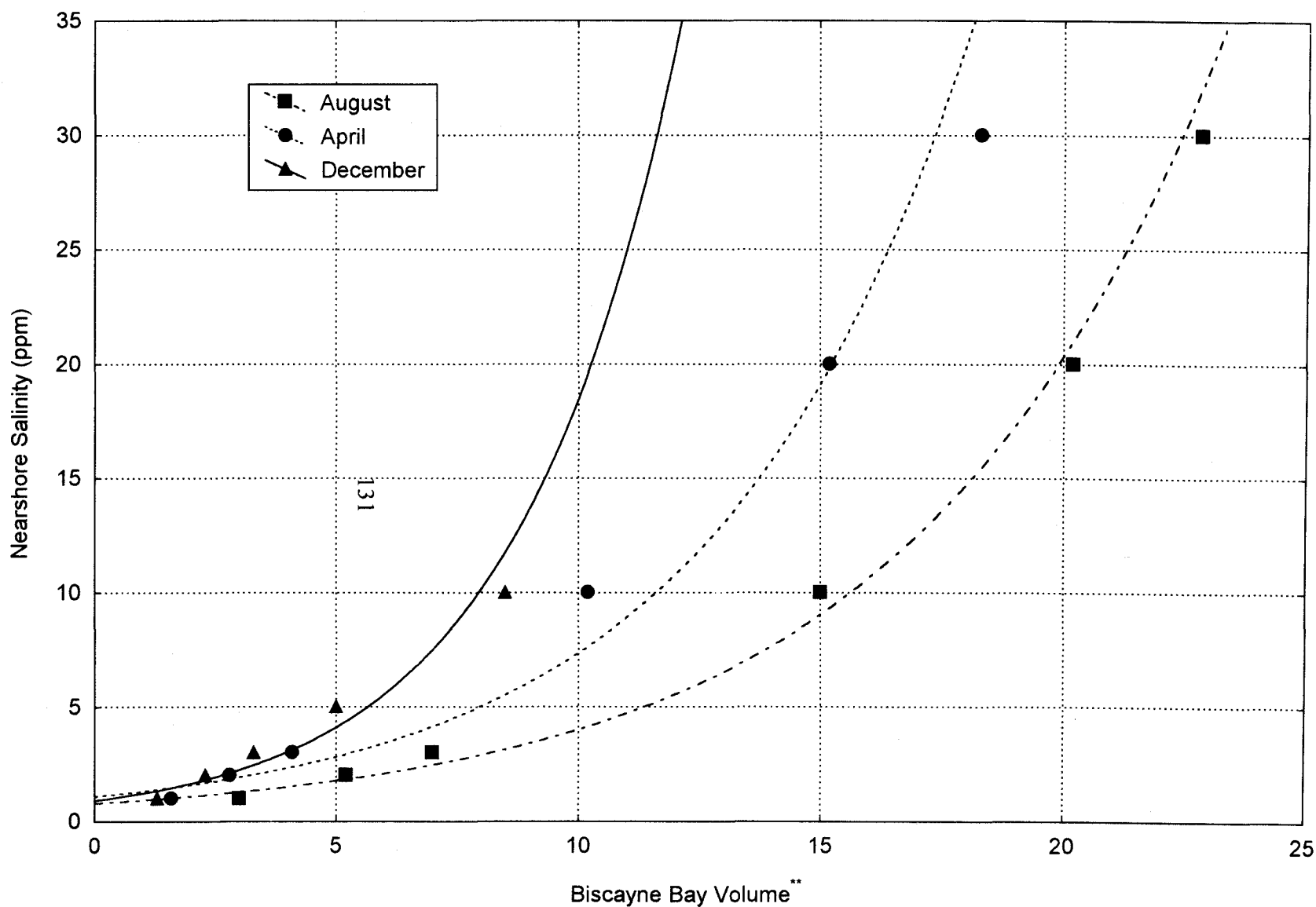


Figure D-4-3: Relationship between Biscayne Bay water volume required to produce nearshore salinities observed for August, April, and December. Ground water and Mowry canal discharges are held the same for data predicted for each month. Data from Table D-4-5.

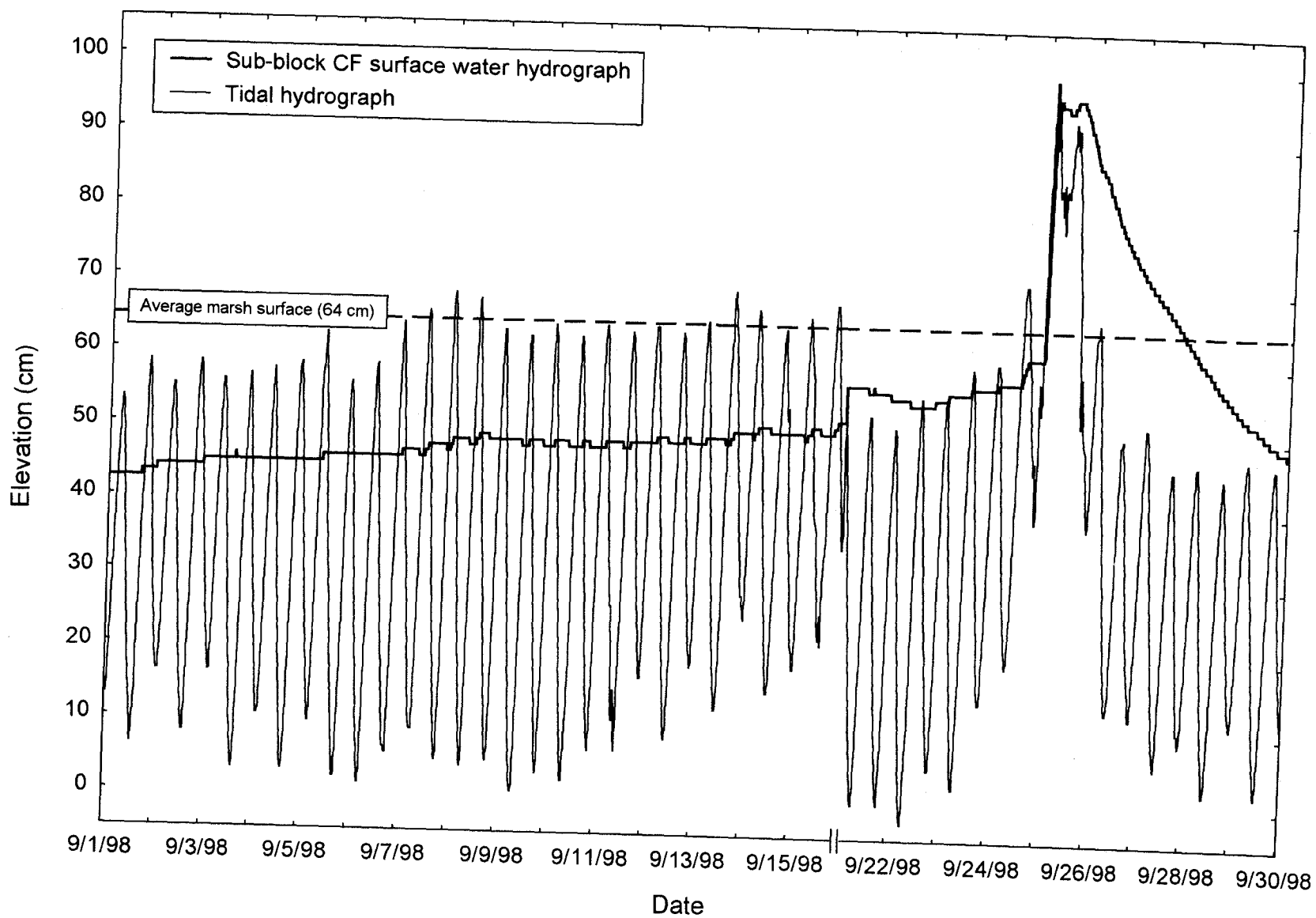


Figure D-4-4: Hydrograph for Sub-block CF during September 1998. No data between Sep. 16-21, 1998.

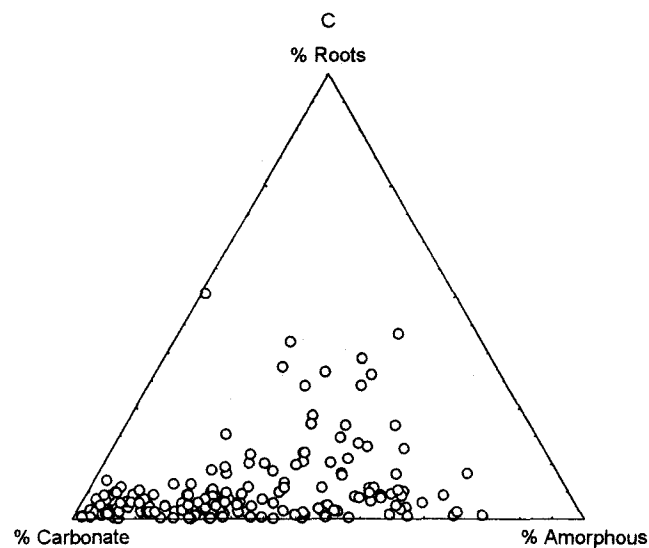
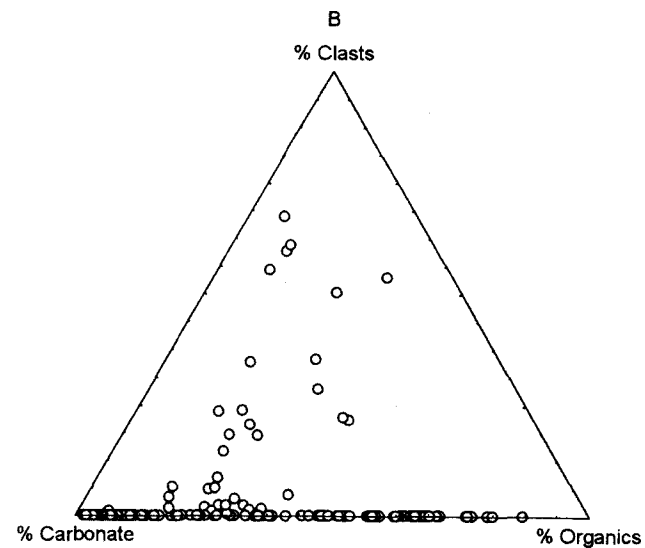
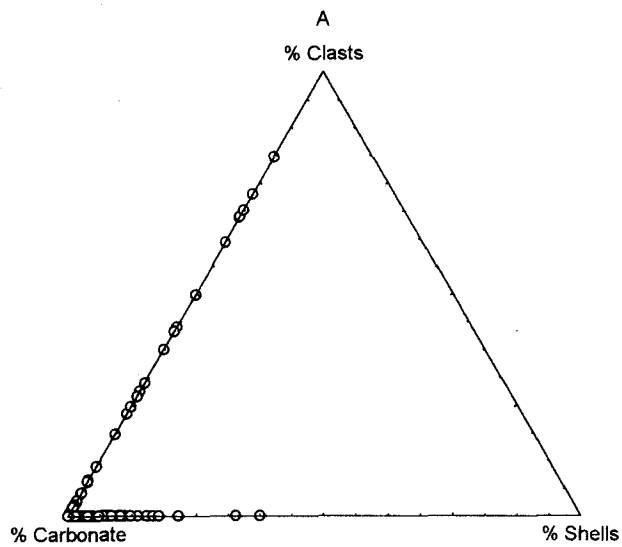


Figure E-2-1: Ternary scatterplots of (A.) clasts vs. shells vs. carbonates, (B.) carbonate content vs. total organics vs. clasts, and (C.) break down of organic material in relation to carbonate content.

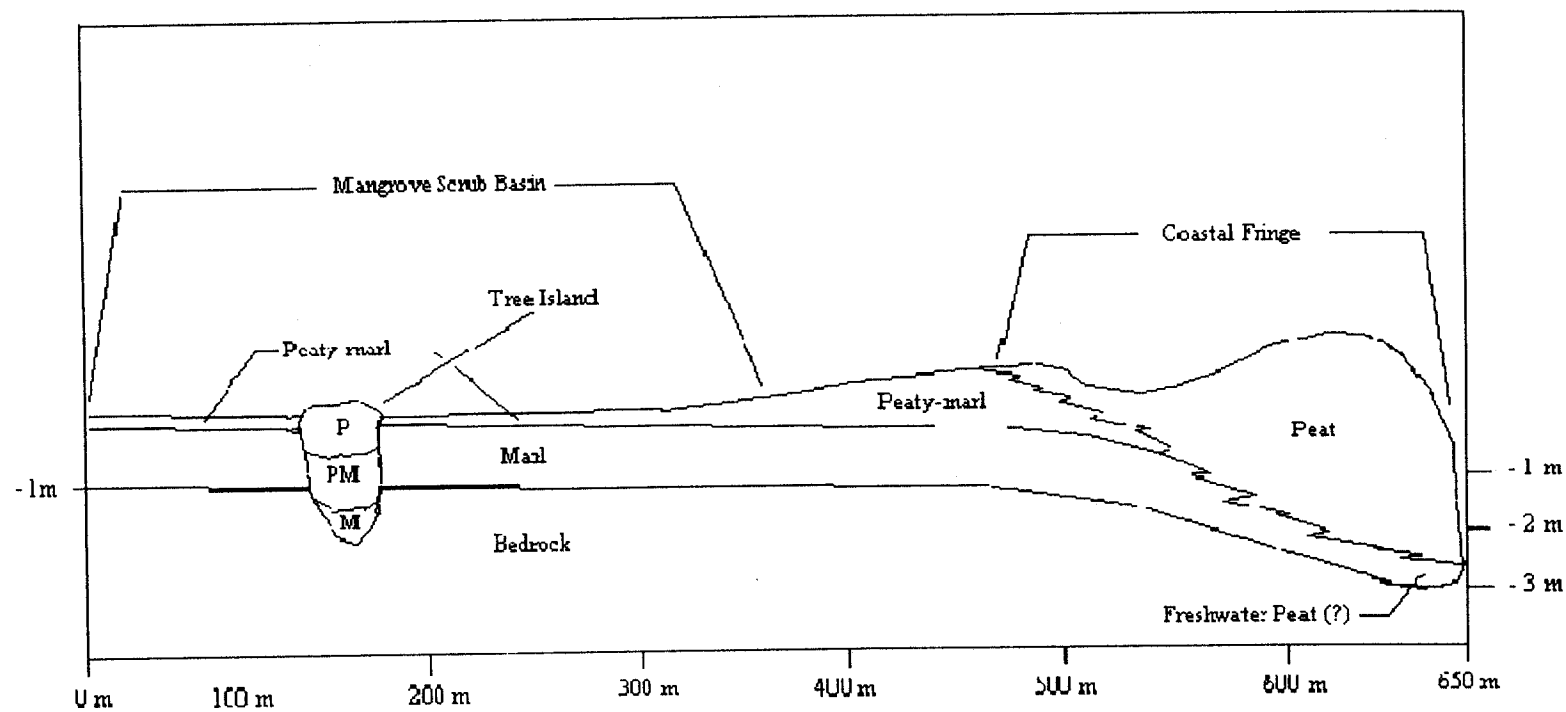


Figure E-4-1: Schematic cross section profile for L31-E levee to Biscayne Bay.

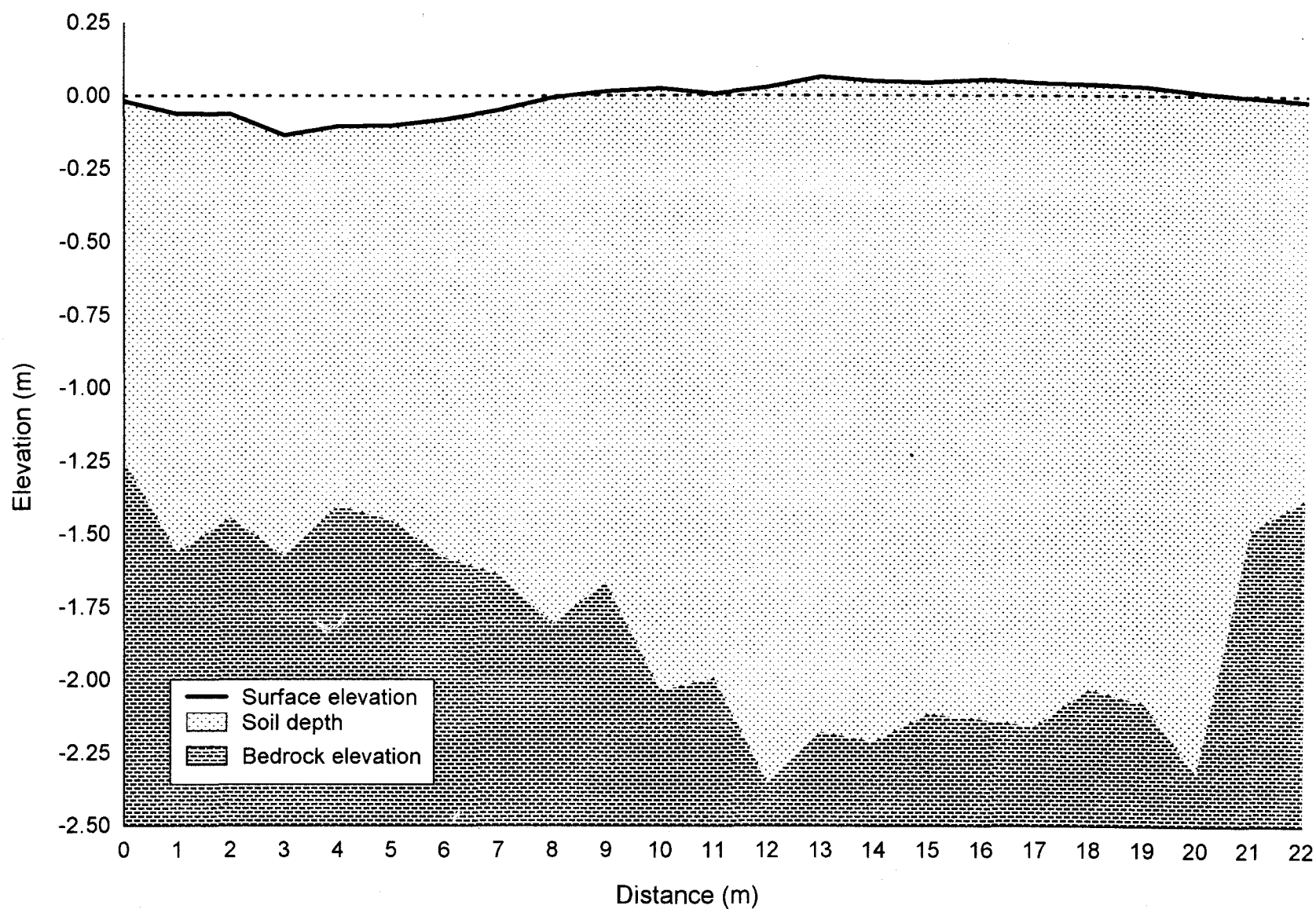


Figure E-4-2: Profile of surface and bedrock topography across a mangrove hammock. Surface water is always within a few cm of the 0 elevation line. The elevation of this line in respect to MSL is 0.410 m, which is higher than the surrounding marsh surface by 50 mm.

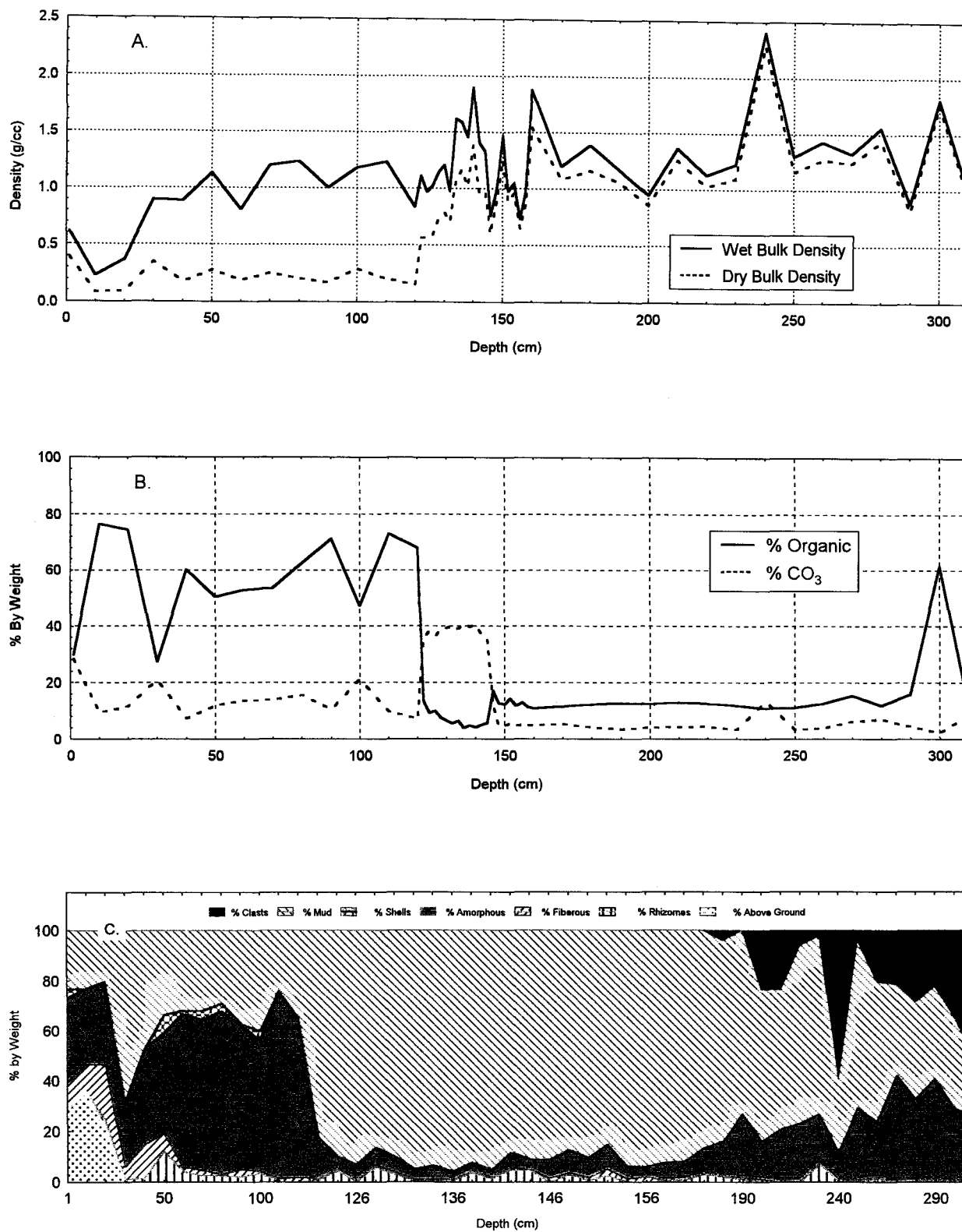


Figure E-5-1: A. Bulk density for TD dome 1 core. B. Bulk analysis of carbonates vs. organics. and C. Percent by weight of organic constituents in TD Dome 1.

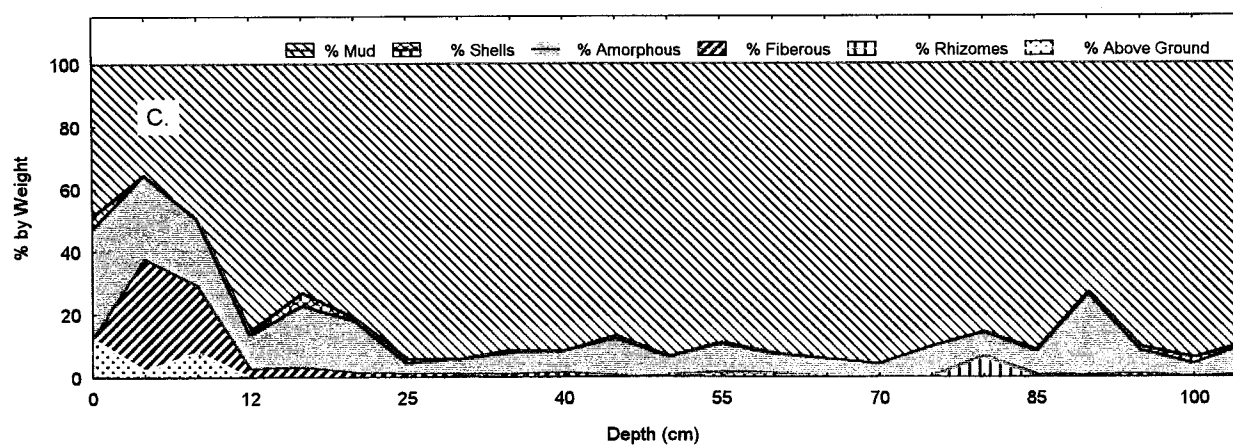
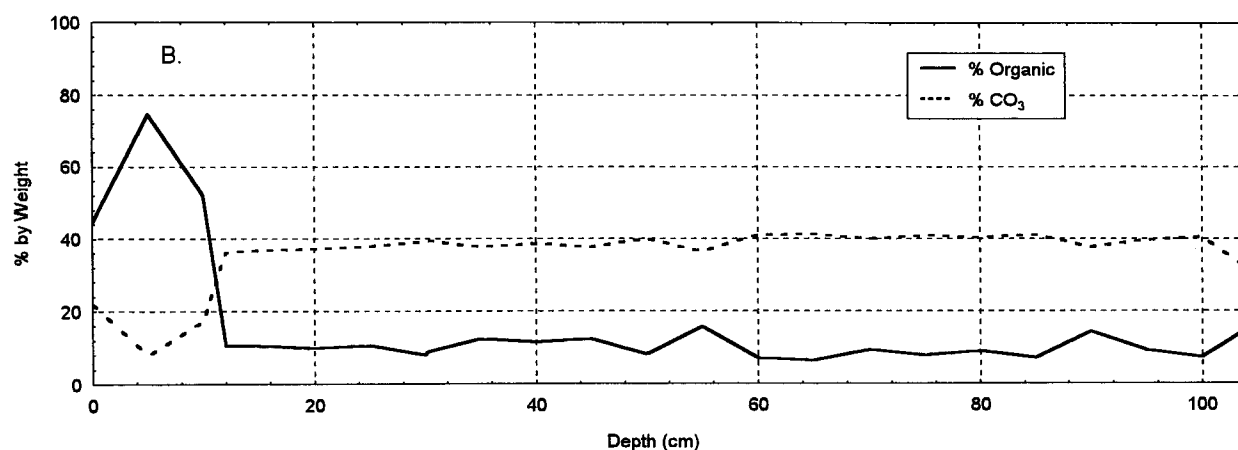
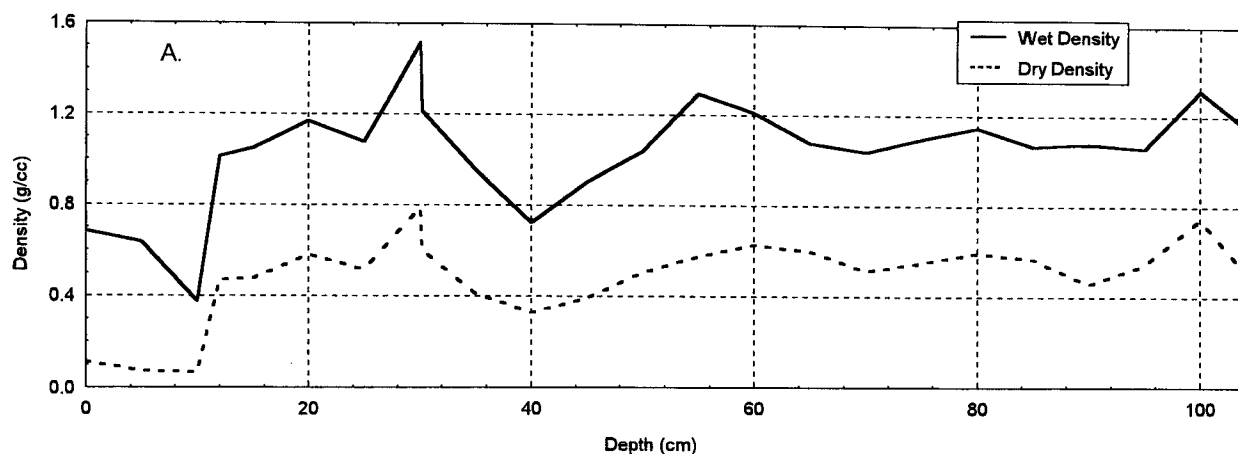


Figure E-5-2: A. Bulk density for TD dome exterior. B. Bulk analysis of carbonates vs. organics. C. Percent by weight organic constituents in TD dome exterior.

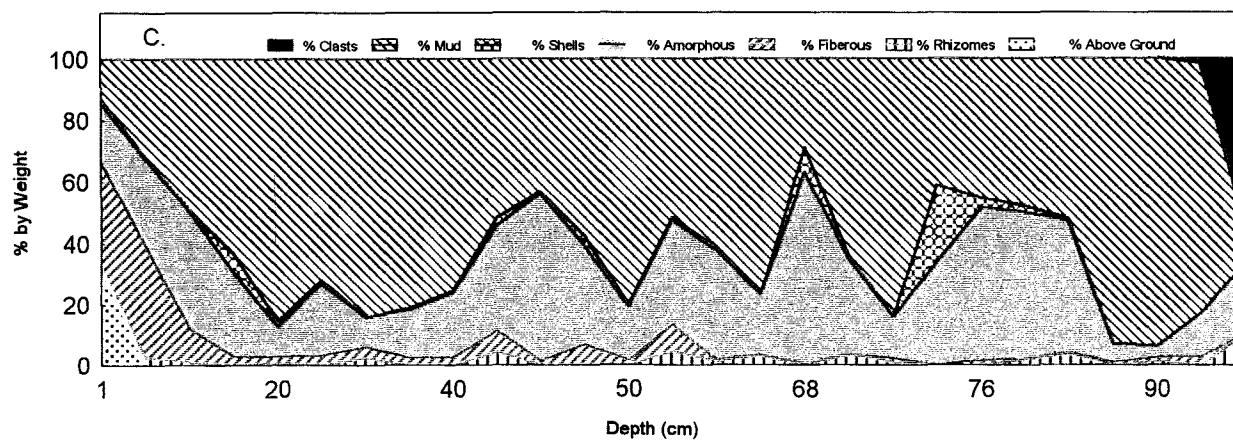
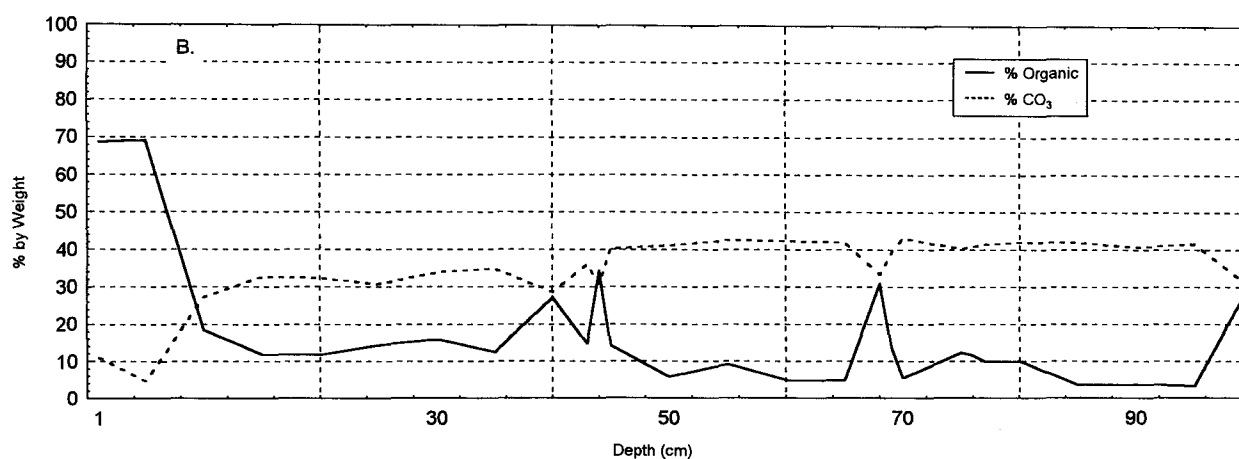
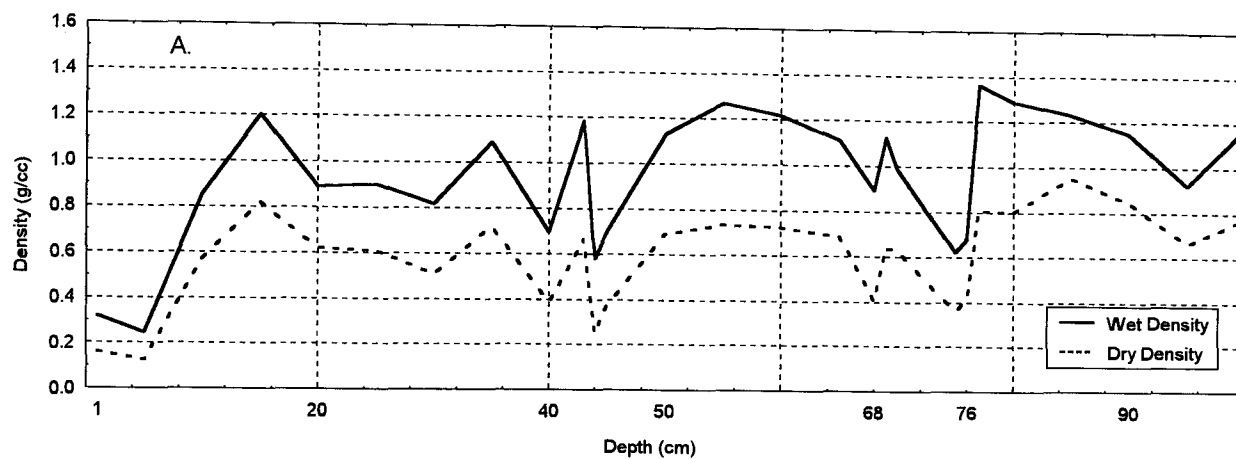


Figure E-5-3: A. Bulk density for TD 400. B. Bulk analysis of carbonates vs. organics. And C. Percent by weight organic constituents in TD 400.

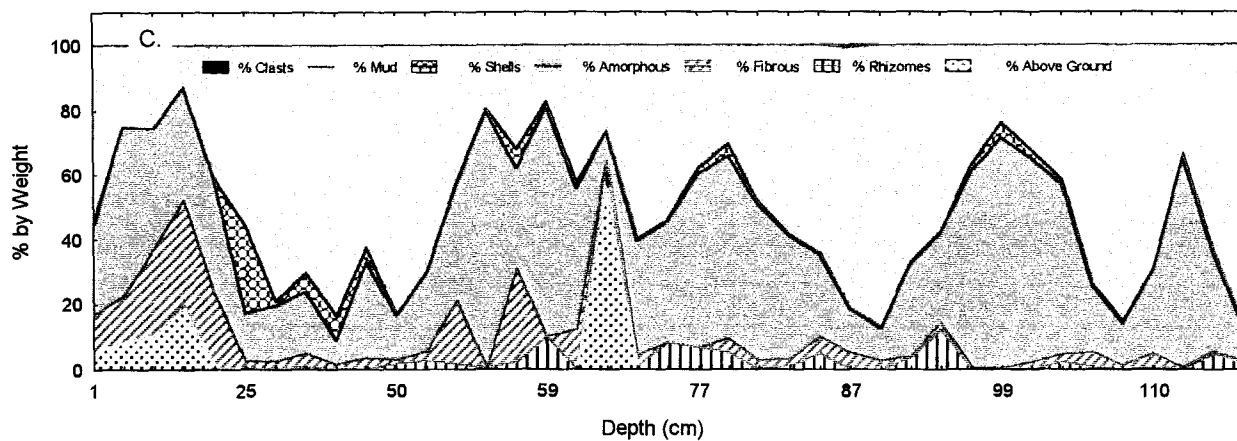
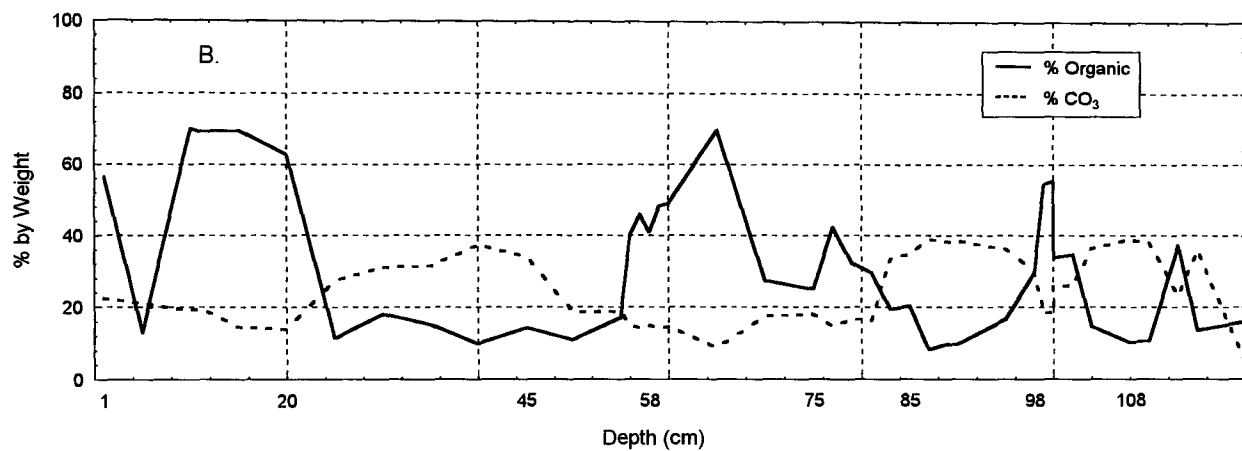
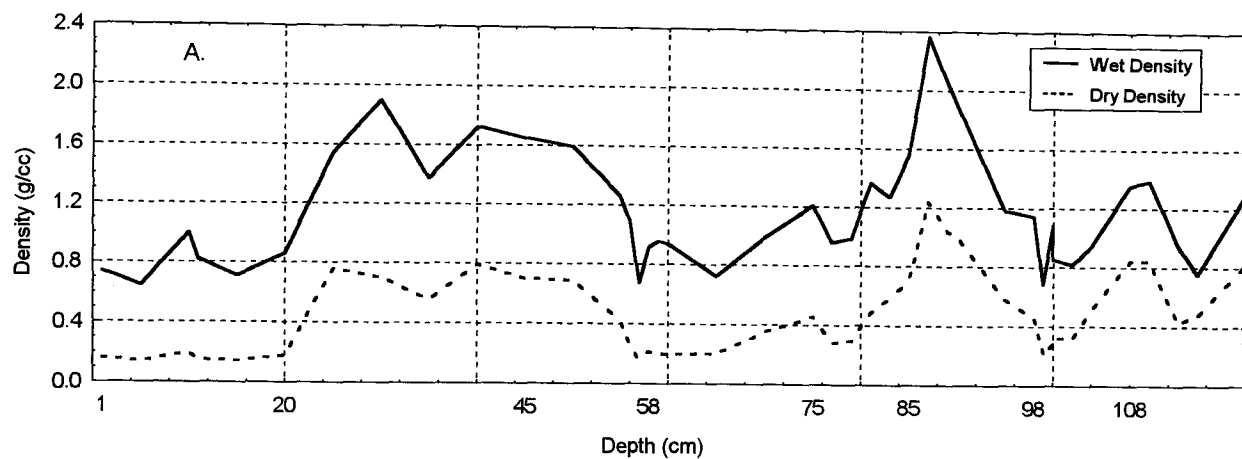


Figure E-5-4: A. Bulk density for TD-500. B. Bulk analysis of carbonates vs. organics. And C. Percent by weight organic constituents is TD 500.

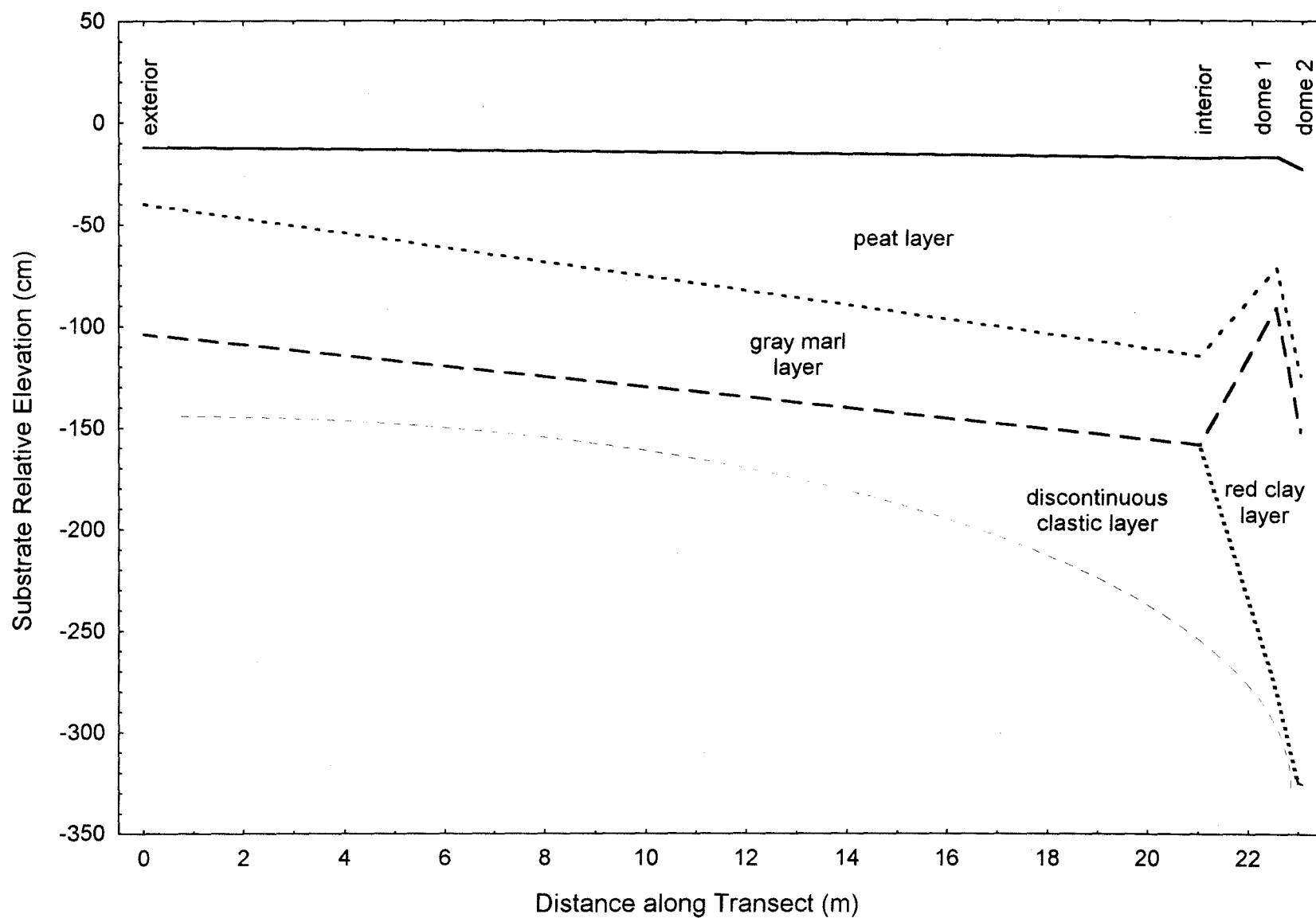


Figure E-5-5: Profile of sediment type along a transect extending from TD exterior to TD dome 2.

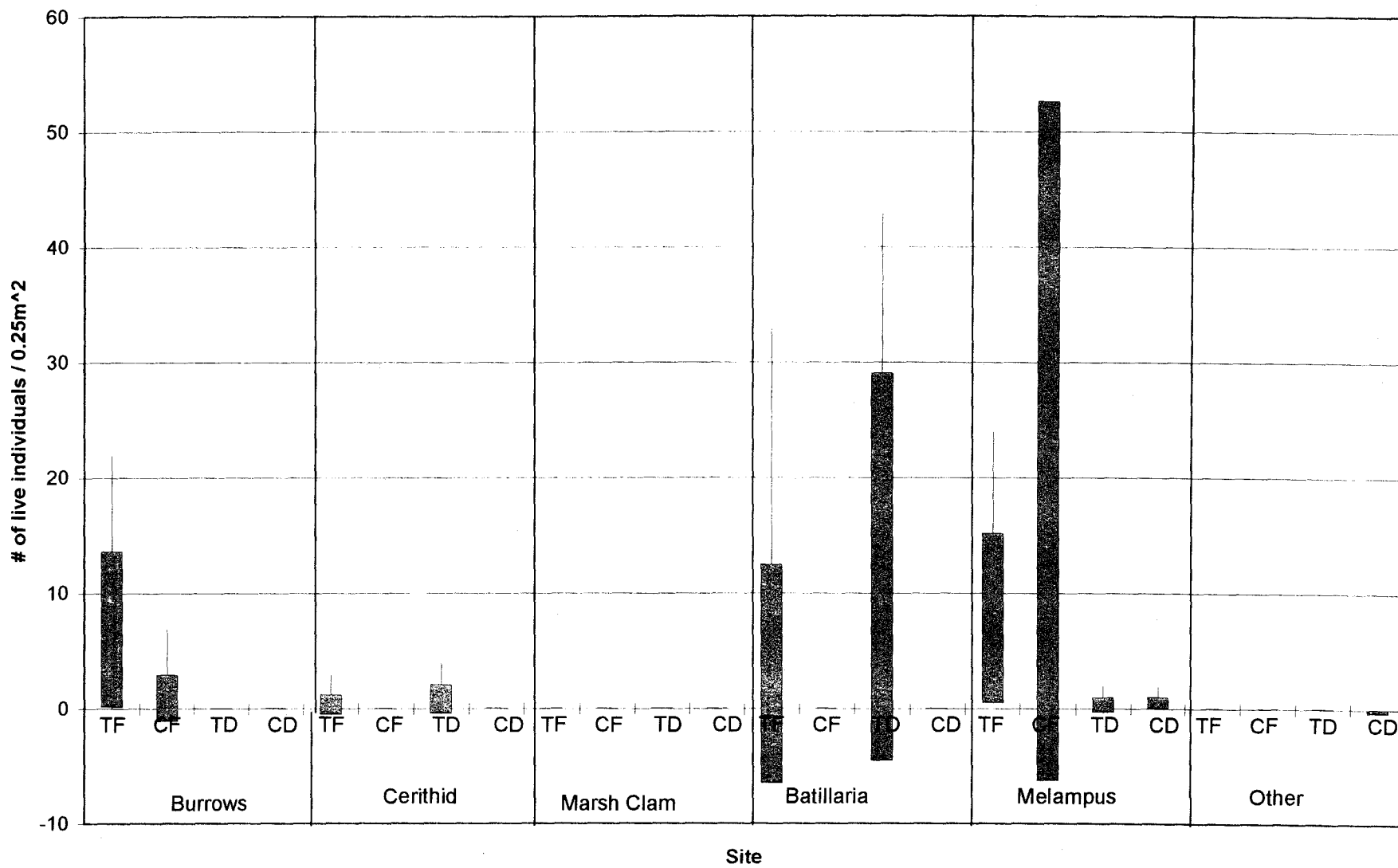


Figure F-1-1: Comparison between the number of invertebrates by station for January 1996.

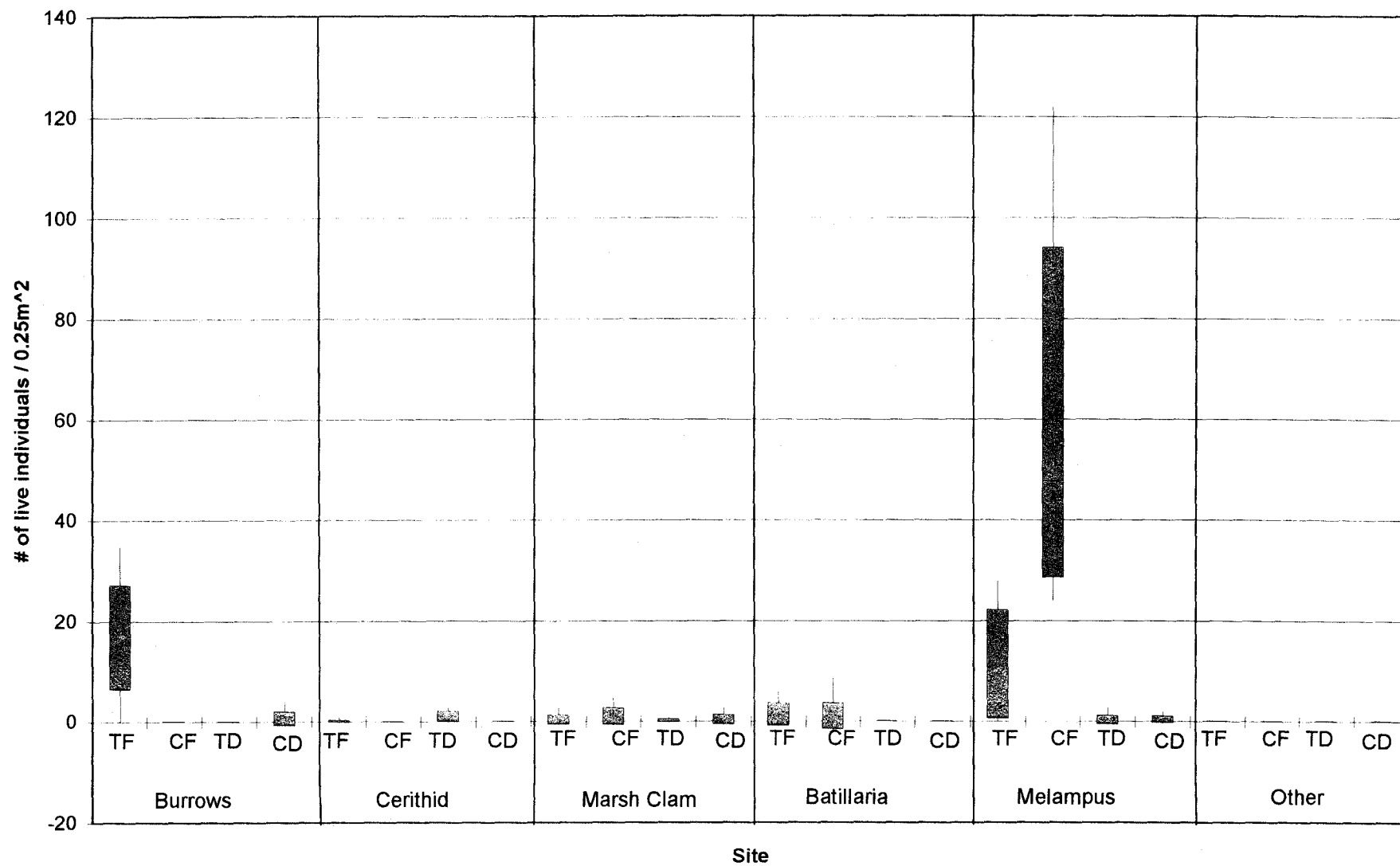


Figure F-1-2: Comparison between the number of invertebrates by station for July/ August 1997.

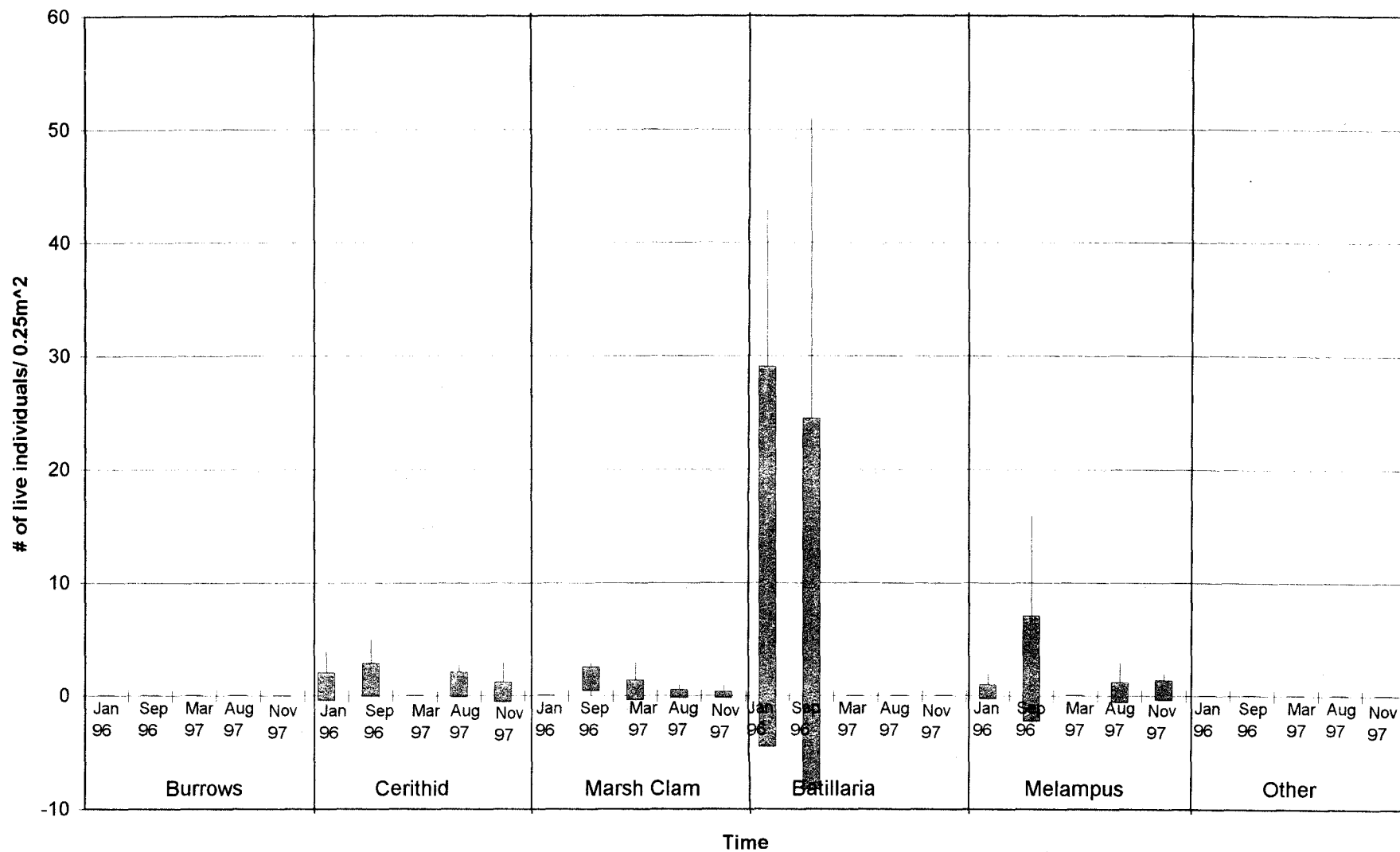


Figure F-1-3: Comparison between the number of invertebrates for different sampling dates at Treatment Dwarf.

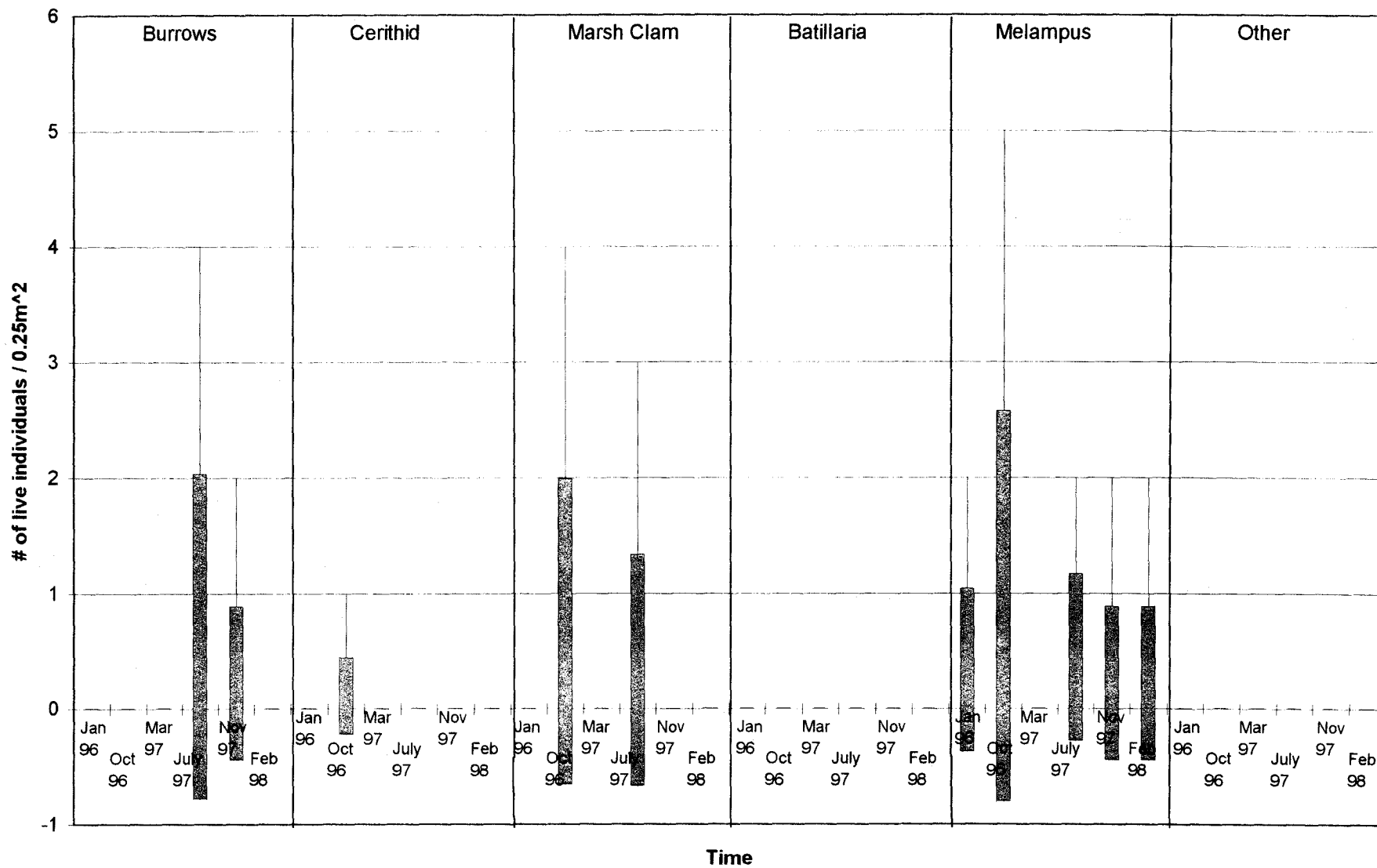


Figure F-1-4: Comparison between the number of invertebrates for different sampling dates at Control Dwarf.

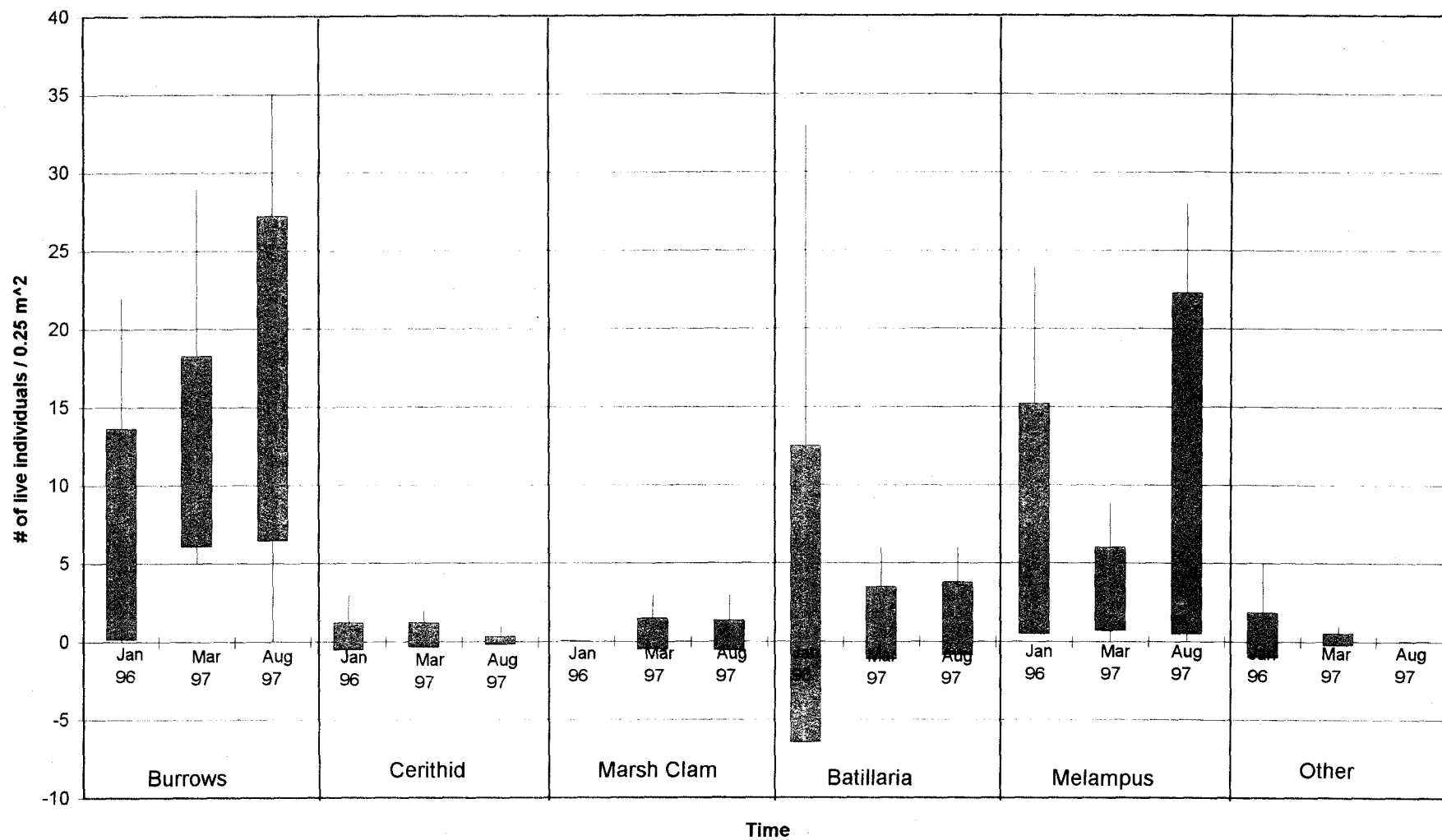


Figure F-1-5: Comparison between the number of invertebrates for different sampling dates at Treatment Fringe.

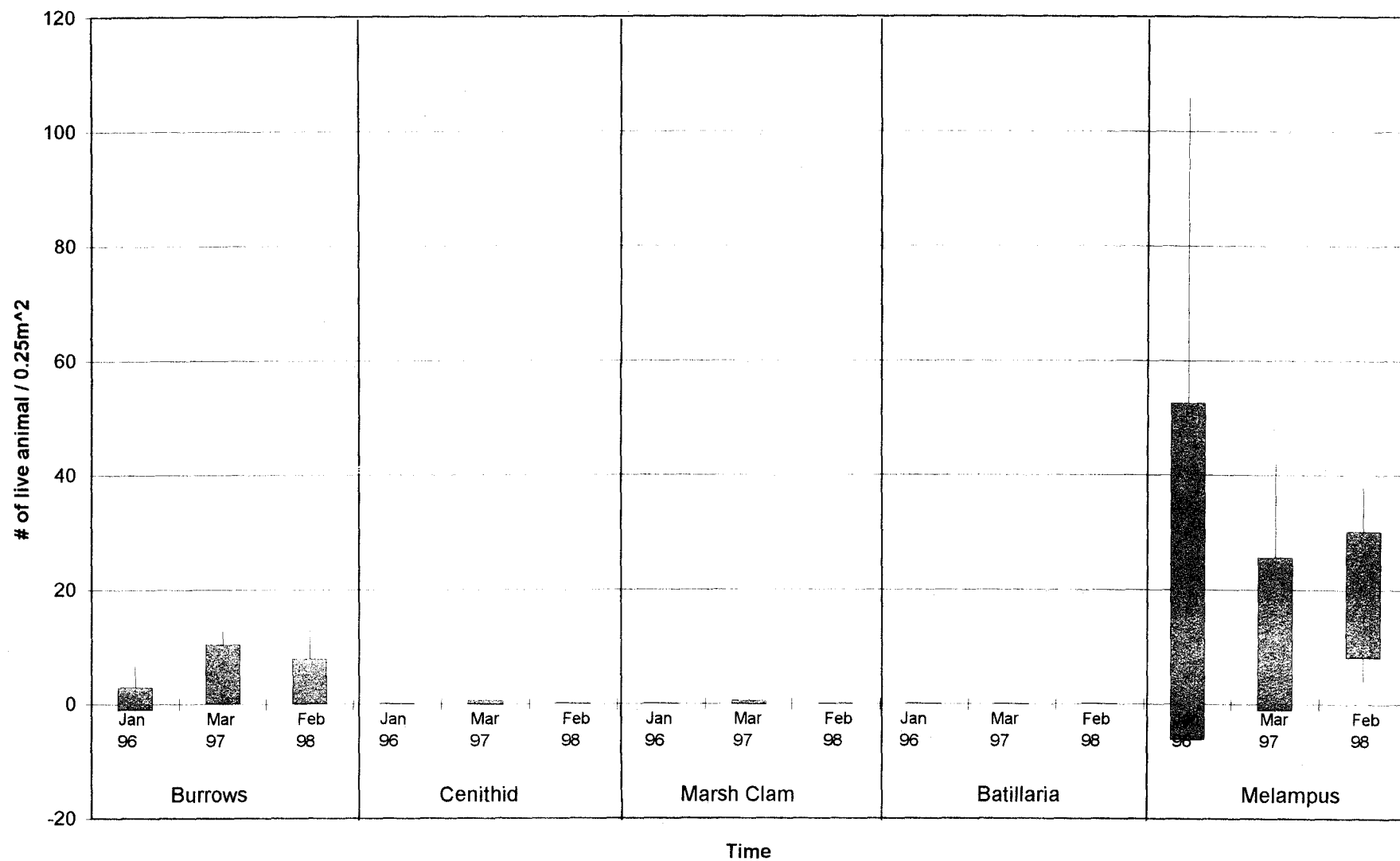


Figure F-1-6: Comparison between the number of invertebrates for different sampling dates at Control Fringe.

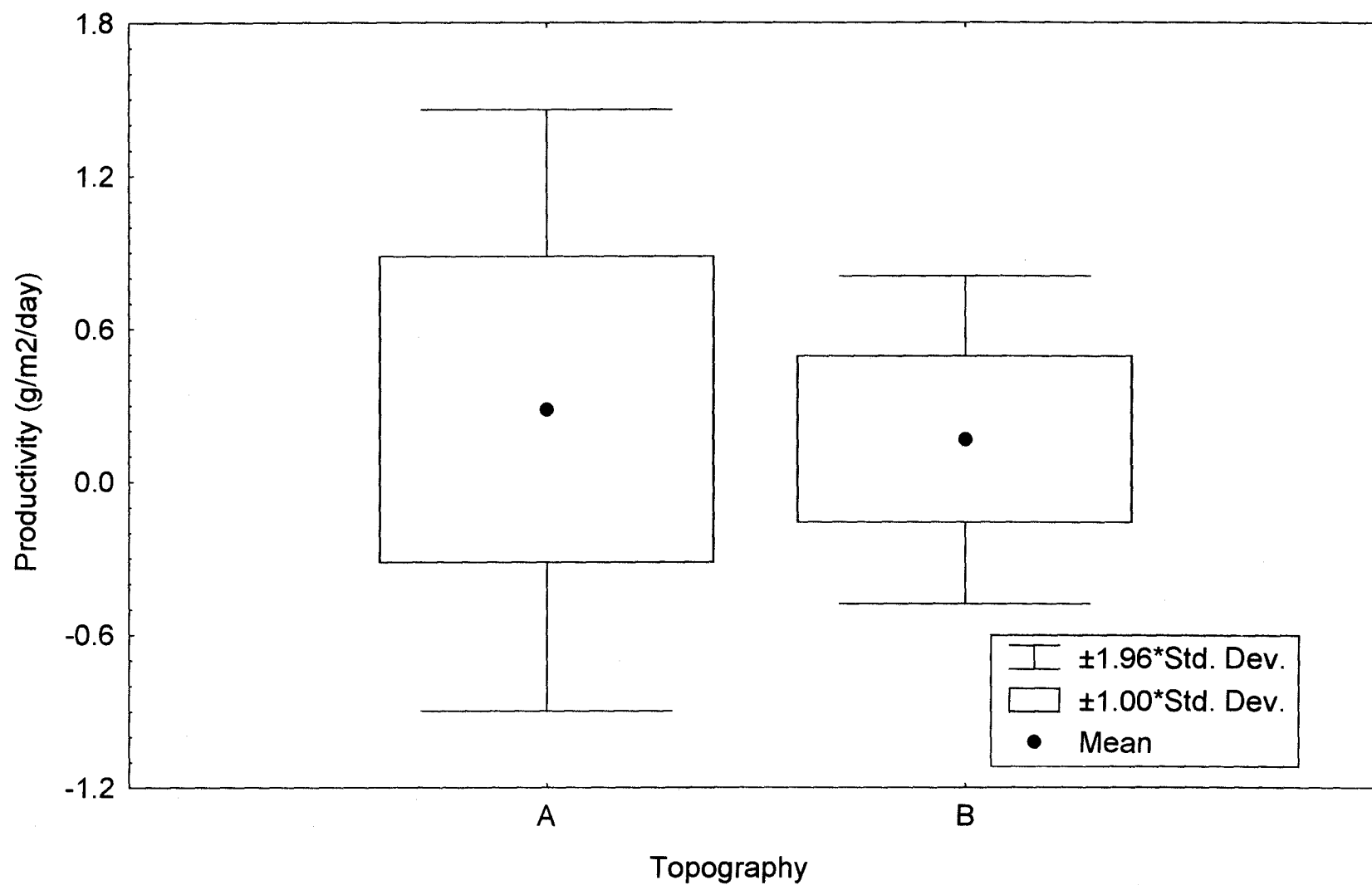


Figure G-1-1: Periphyton productivity at different microtopographic relief at Treatment Dwarf (A=microtopographic high, B=microtopographic low).

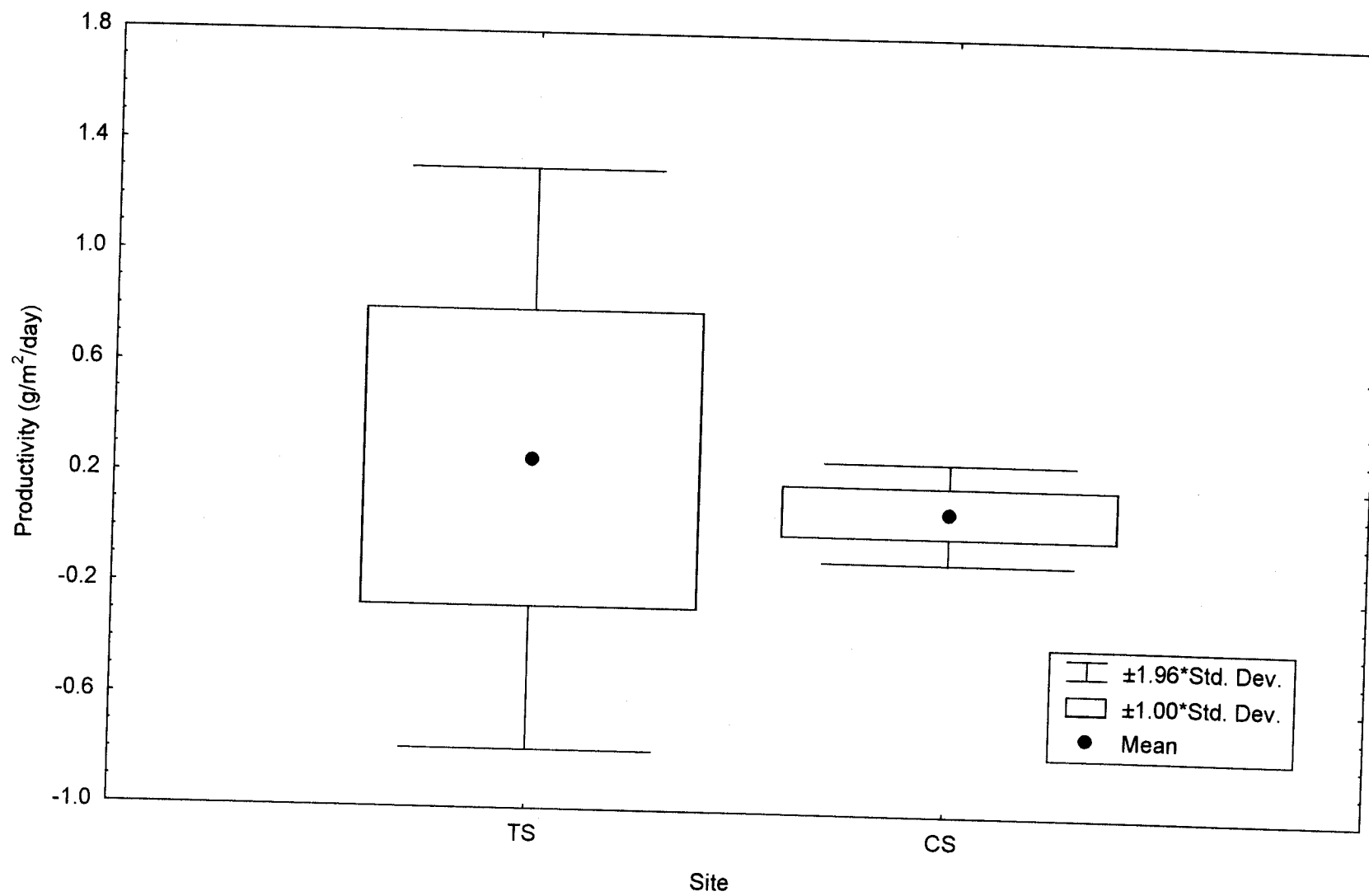


Figure G-1-2: Periphyton productivity at Treatment Dwarf and Control Dwarf.

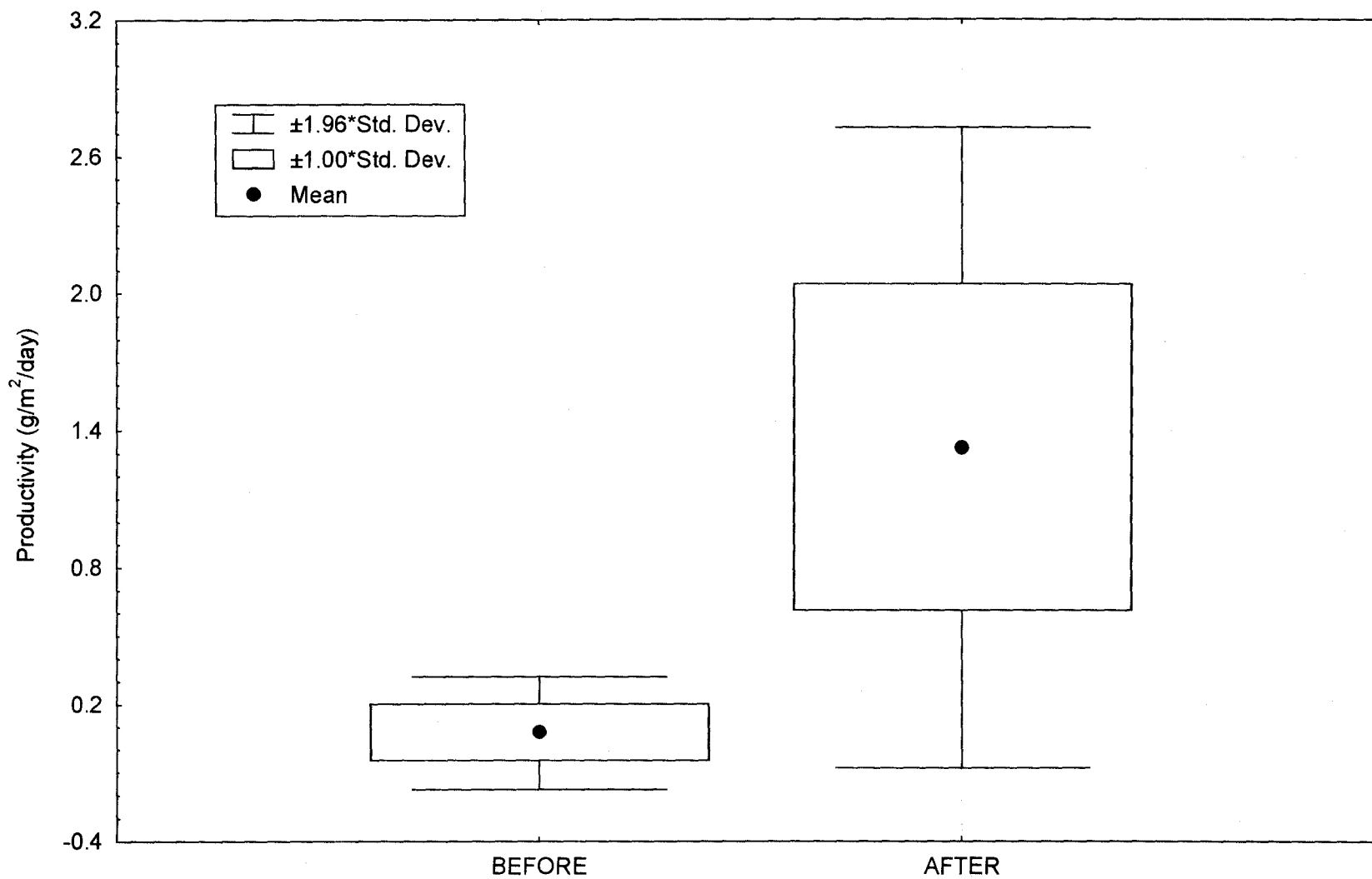
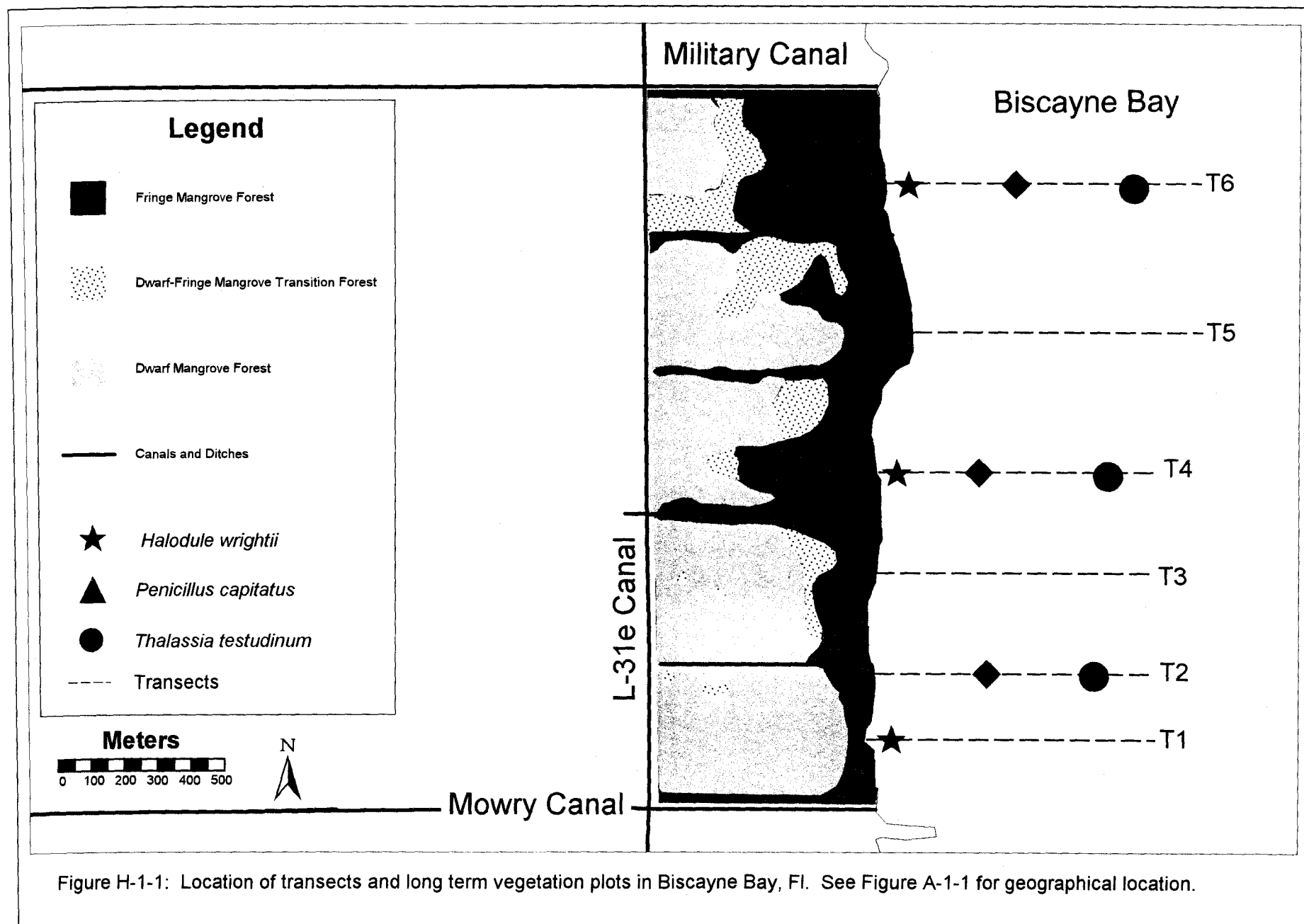


Figure G-1-3: Periphyton productivity in the Treatment Dwarf plot (Before = before freshwater rediversion, After = After freshwater rediversion)



New Short Shoot Growth for *Halodule wrightii*

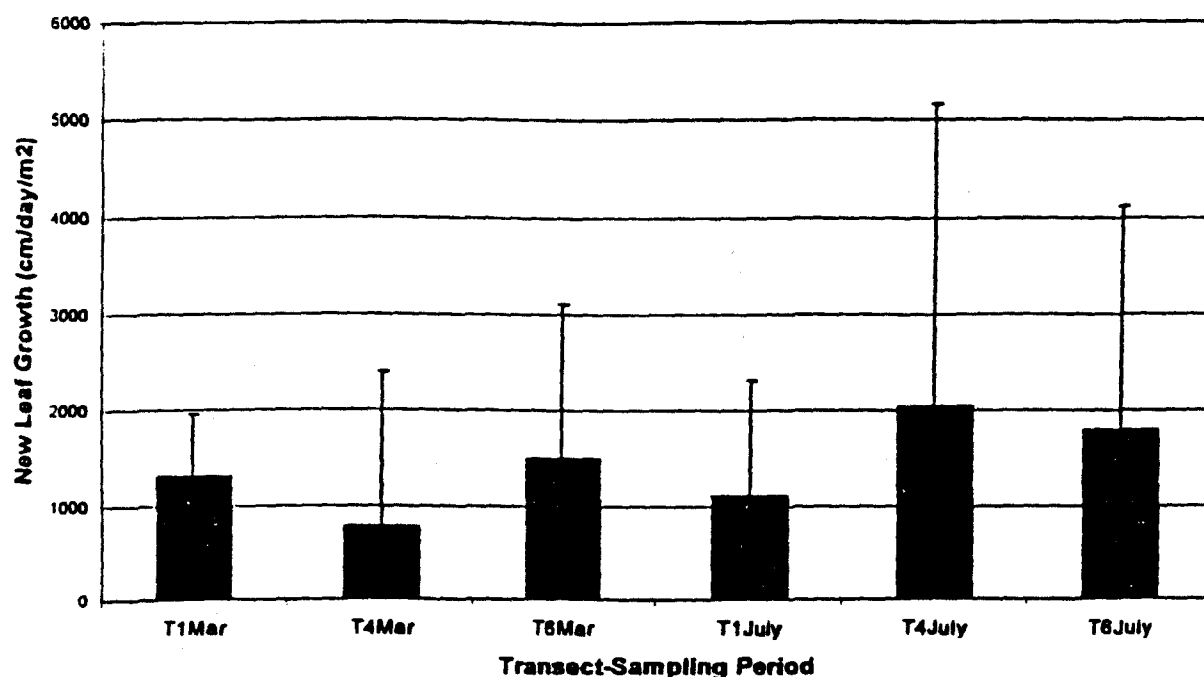


Figure H-2-1a: The relationship between new short shoot growth of *H. wrightii* with regards to location and sampling period. Five 112.5 cm² plots were sampled at each transect.

Biomass for *Halodule wrightii*

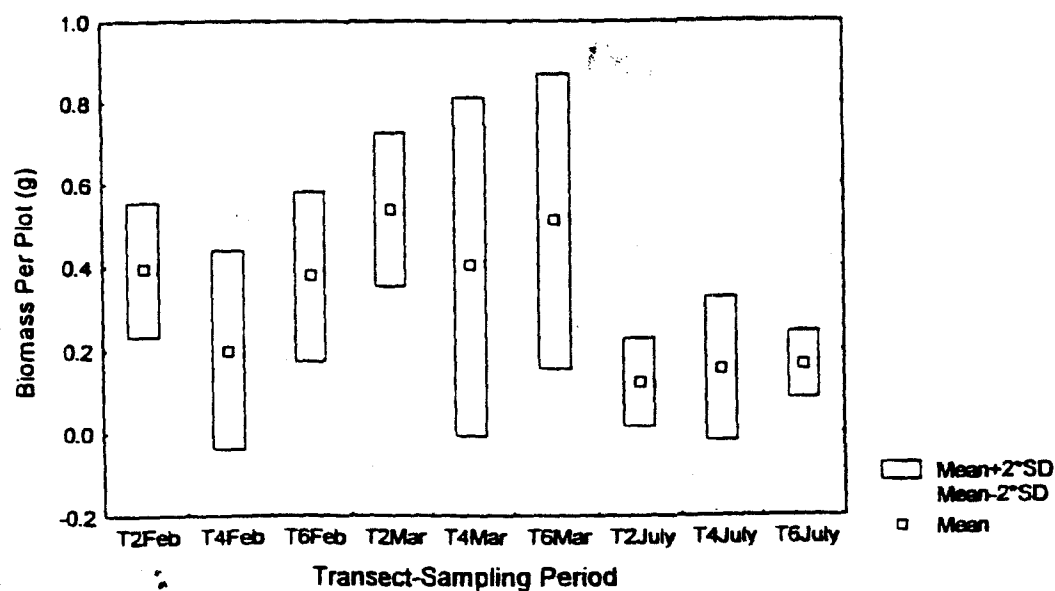


Figure H-2-1b: The relationship between *H. wrightii* mean biomass with regards to location and sampling period. Five 112.5 cm² plots were sampled at each transect.

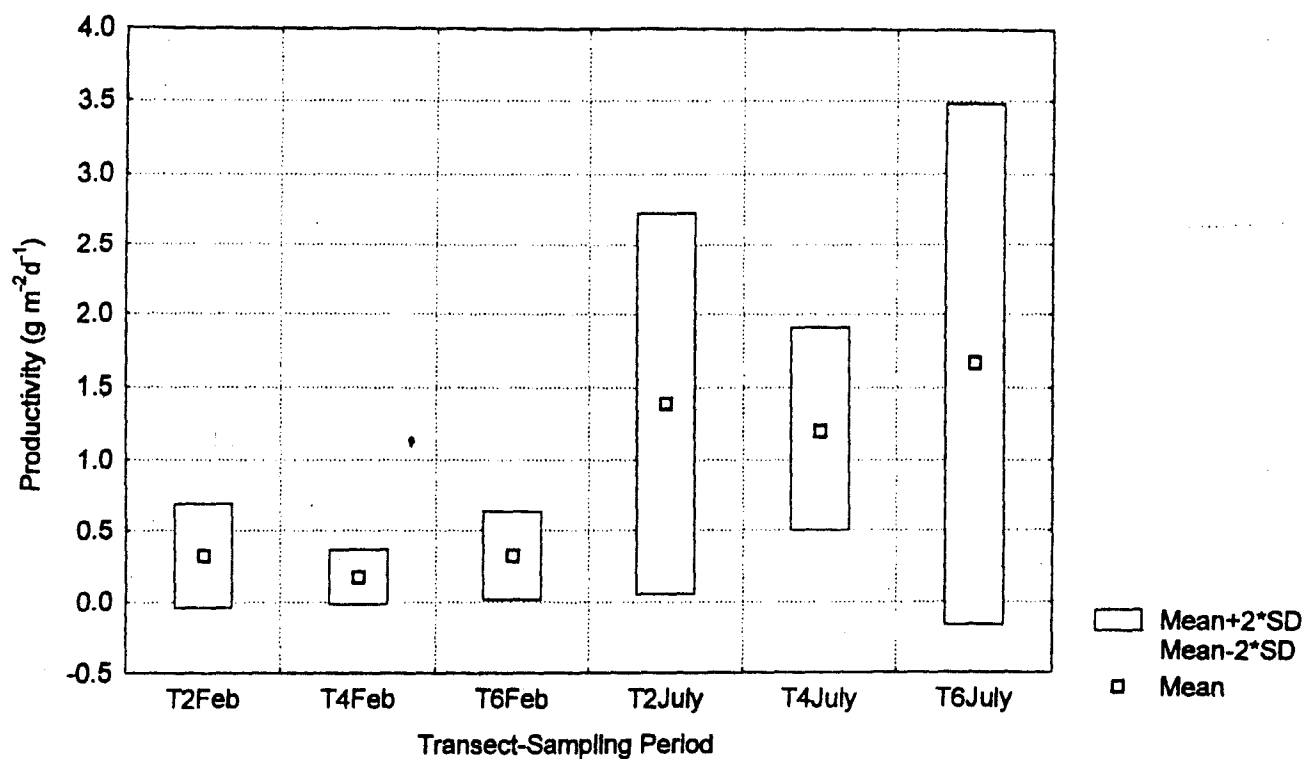


Figure H-2-2a: The relationship between new short shoot growth of *T. testudinum* with regards to location and sampling period. Five 112.5 cm² plots were sampled at each transect.

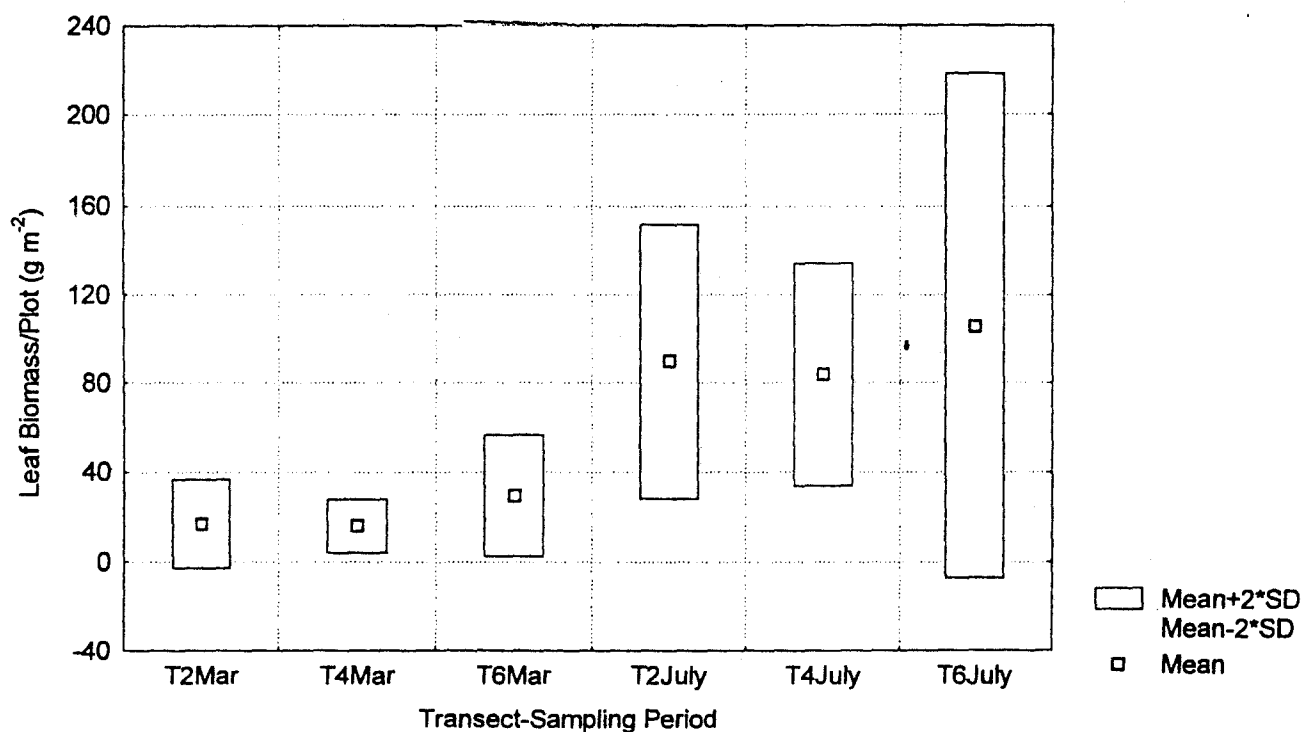


Figure H-2-2b: The relationship between *T. testudinum* mean biomass with regards to location and sampling period. Five 112.5 cm² plots were sampled at each transect.

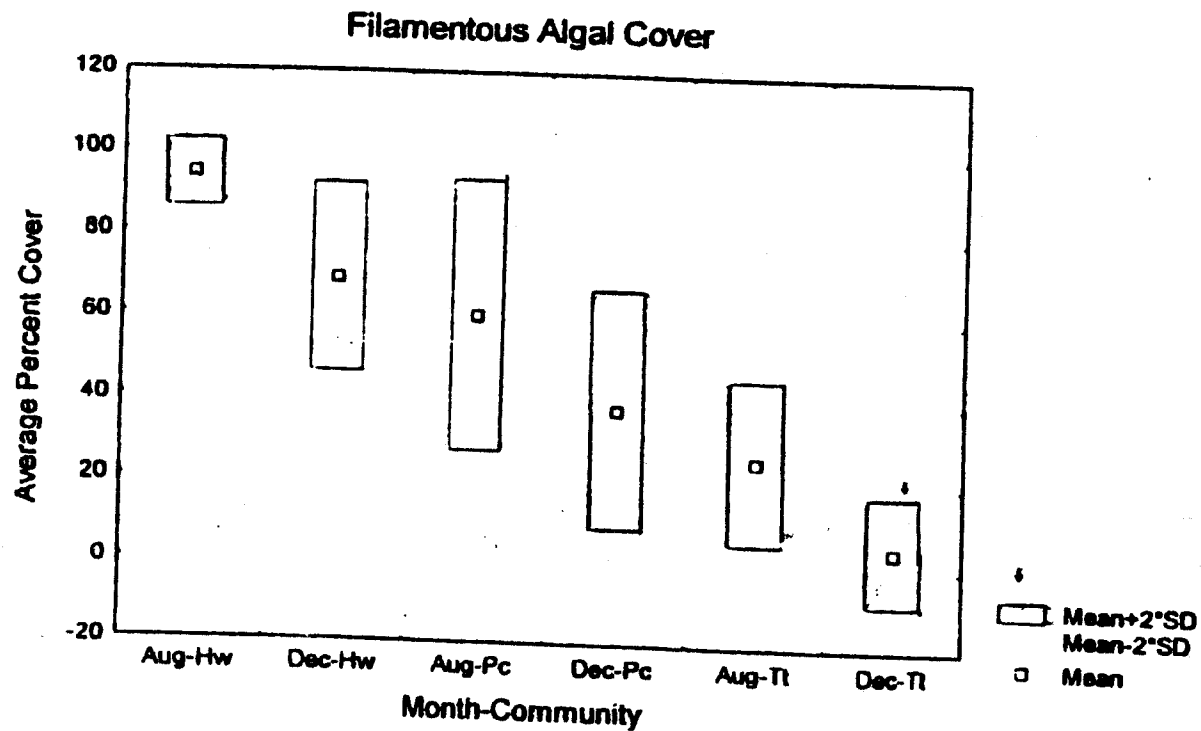


Figure H-2-3: Relationship between filamentous algal cover with regards to location & sampling time. Hw community (0-200 m); Pc community (30-600 m); Tt community (600-800 m).

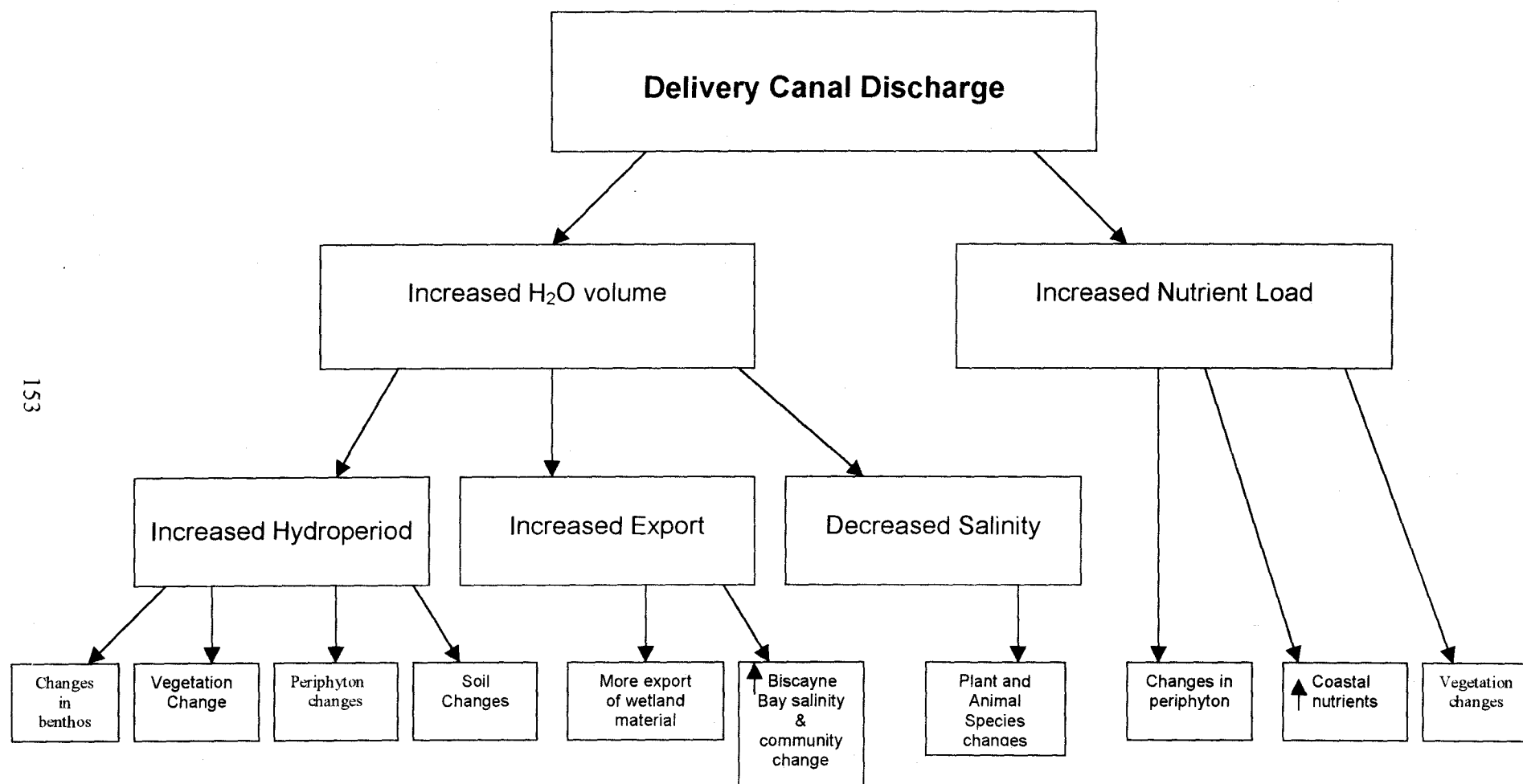


Figure I-1-1: Conceptual model of treatment effects on coastal ecosystems.