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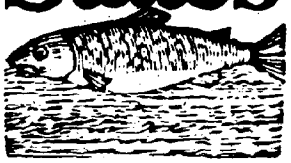

III

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Preface

Coastal Ecological Systems of the United States was originally prepared for the Federal Water Pollution Control Administration as part of the National Estuarine Pollution Survey conducted in 1968 and 1969. It was the product of a group of scientists led by staff members of the University of North Carolina's Institute of Marine Sciences. Its four volumes include a comprehensive survey of scientific information through 1969, as well as a new system for the classification of coastal ecosystems. The manuscript was submitted to the U.S. Environmental Protection Agency (which absorbed the FWPCA in 1970), but was not published.

The Conservation Foundation is now able to publish this work because of an assistance grant (Grant No. 043-158-68) provided by the National Oceanic and Atmospheric Administration's Office of Coastal Environment, which is responsible for implementing the Coastal Zone Management Act of 1972. The purpose of the grant was to assist the Conservation Foundation in preparing an amended version of this comprehensive work for NOAA's program use. Upon completion of that activity, the Conservation Foundation agreed to make this material available to a wider audience by reproducing the amended manuscript in the most inexpensive way possible.

Those whose personal efforts merit recognition are Robert W. Knecht, director, and Edward T. LaRoe, coastal ecologist, of the Office of Coastal Environment, who foresaw the relevance of this work to the practical needs of coastal zone management; Eugene T. Jensen and A. L. Wastler, of the U.S. Environmental Protection Agency, who arranged for the original study; and John Clark and Laura O'Sullivan of the Conservation Foundation, who, respectively, persuaded their organization to publish this massive work and attended to the myriad details of bringing it into print.

The Editors:

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Chapter D-1

GLACIAL FIORDS

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INTRODUCTION

A glacial fiord is a narrow, steepsided coastal inlet fed by a glacier at its upper end. Fiords where glaciers terminate on land and send their meltwater into the tide water via a freshwater river system are considered in a following chapter, "Turbid Outwash Fiords".

EXAMPLES

Examples of glacial fiords are listed in Table 1. See maps in Figures 1 and 2.

All the major indentions of the Southeast Alaska coastline are fiord-type estuaries. The name refers to inlets which owe their distinctive physiography to the action of glacial ice on mountainous coastal regions. Thus, such inlets are usually narrow and deep and relatively straight. Commonly, fiords contain an entrance sill which constricts the free exchange of waters within the inlet. However, the presence of such a sill is not essential to the designation "fiord" for the purposes of this discussion, although it should be noted that Pritchard (1952) has restricted the definition to those inlets which do contain such a sill.

Rosenberg (1966) has classified various inlets and waterways (i.e., estuaries) to give five basic patterns of which two concern us here, namely, the "sill-glacier" and the "glacier" systems. The "sill glacier" type possesses one or more sills and adjoining basins. Both receive their major freshwater runoffs from active glacial sources and are otherwise identical. Discussion within this section will be limited to those fiords with active glaciers in the intertidal zone. In such cases, most of the glacial melt water (i.e., the major estuarine freshwater source) passes directly into the marine environment.

The "glacial fiords" thus defined constitute dynamic oceanographic systems inasmuch as all glaciers are constantly in motion. The glacial system of Southeast Alaska is currently in a period of overall retreat, as is well illustrated by the recent geography of Glacier Bay National Monument. Individual glaciers within the total system, however, may be advancing. The resulting rapid (relative to non-glacial estuarine systems) and irregular changes in the

Table 1. Principal physiographic features of representative glaciated fiords in Southeast Alaska.

Fiord	Seaward Connection	Length (n.m.)*	Mean Width (n.m.)*	Mean Mid-Inlet Depth (m)	Max Depth (m)	Outer Sill Depth (m)
Tarr Inlet	Glacier Bay	60	3.0	220	510	40
Muir Inlet	Glacier Bay	19	1.0	215	315	58
Geiki Inlet	Glacier Bay	7	1.3			
Johns Hopkins Inlet	Glacier Bay	10	0.8			
Taku Inlet	Stephens Passage	72	7.0	295	460	185
Tracy Arm	Stephens Passage	23	1.0	270	370	22
Endicott Arm	Stephens Passage	24	1.8	260	370	33

*nautical miles

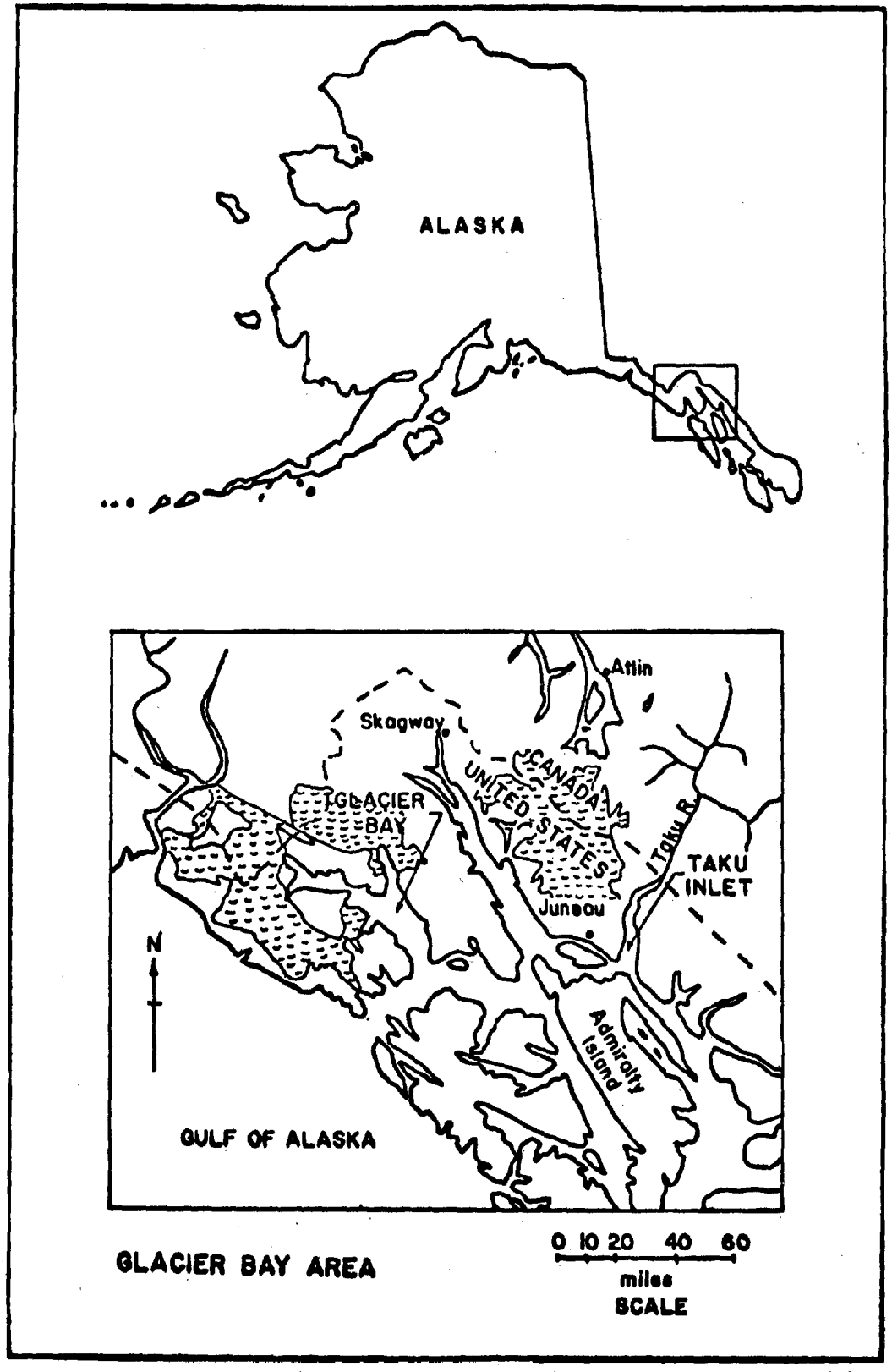

Fig. 1. Glacial fiords in the Glacial Bay area (Hoskin and Burrell, 1968).

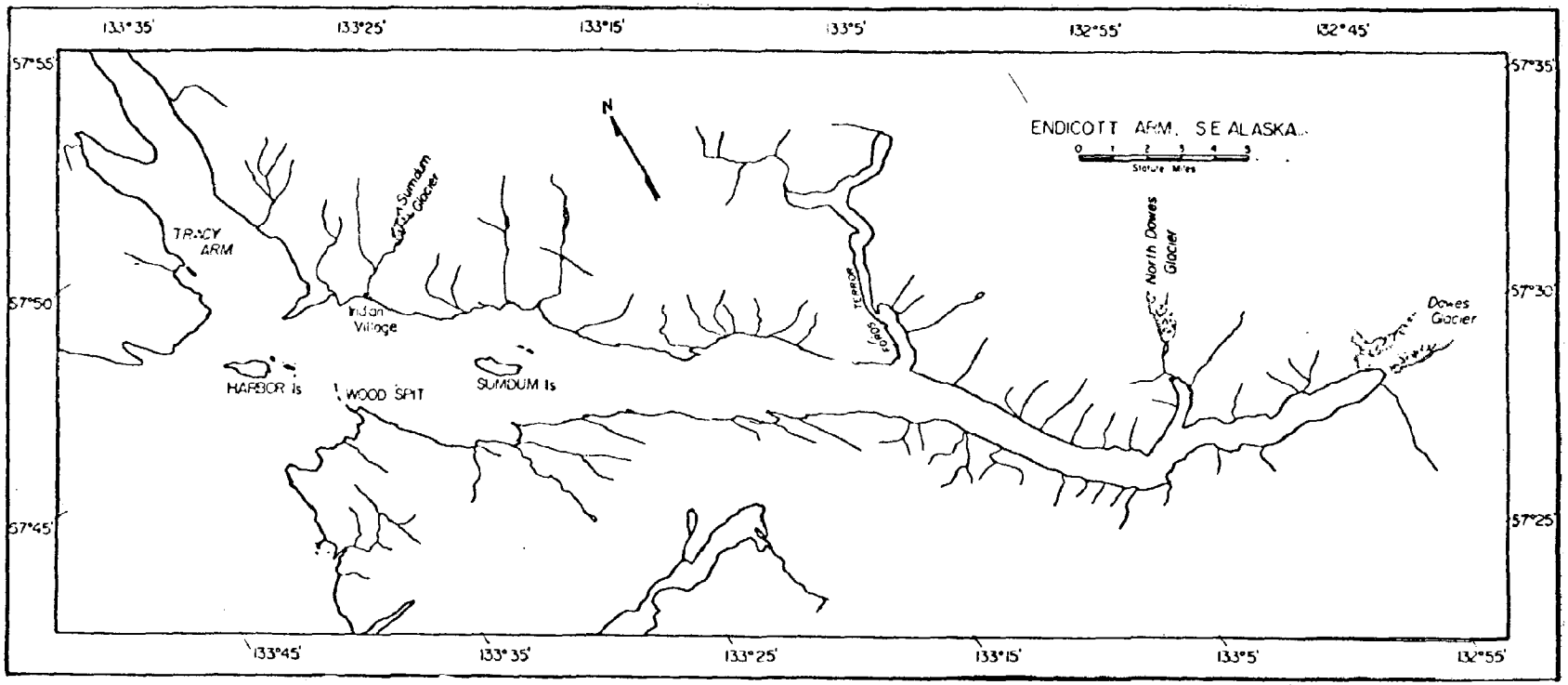

Fig. 2. Endicott Arm, a glacial fiord (Buckley and Loder, 1968).

physiographic configuration of individual inlets is an additional factor which relates directly to other determined physical, chemical, and biological parameters.

The general physical oceanographic characteristics of the fiords of British Columbia have been noted by Pickard and co-workers (e.g., Pickard, 1953, 1956, 1961, and 1963; Pickard and Rodgers, 1958; Tabata and Pickard, 1956). These latter inlets are closely related to those occurring further north in the Southeast Alaska area. However, there are no fiords containing tidal glaciers in British Columbia. Pickard (1967) has also presented some summer data on Alaska fiords, notedly salinity and temperature parameters for Glacier Bay, Muir Inlet, Tracy Arm, and Endicott Arm. Rosenberg and Hall (1968) have presented extensive periodic hydrographic data for all seasons for Taku Inlet, Tracy Arm, Endicott Arm, and the principal inlets within Glacier Bay. All these are fiords with tidal glaciers. The principal features of representative Alaskan glaciated fiords are given in Table 1.

Circulation Patterns

In the fiord estuaries which contain entrance sills, density distributions are often characterized by approximations to the idealized horizontal layering patterns of Stommel and Farmer (1952). Unrestricted exchange of water between a fiord and its seaward connection can be expected to occur to sill depth only. In the general case the salinity of the surface waters increases from the head to the mouth, while the deeper waters remain uniform. Denser water enters the fiords beneath the outward flowing surface water. Mixing of salt between these two waters occurs predominantly upwards with minimal downward mixing of the freshwater. Rosenberg (1966) has noted that such an entrainment of salt water from below may inflate the original volume of outward flowing surface water by an order of magnitude or more. The extent to which the above generalized circulation pattern is developed depends upon many factors but principally on sill depth, the seasonal volume distribution of freshwater runoff, and the density distribution of the marine water outside the fiordal basin.

The majority of the fiords of Southeast Alaska (and British Columbia) have entrance sill depths which allow continuous contact with the exterior source waters such that a slow circulation prevents the basins from stagnating. Where shallow sills penetrate the low salinity outflowing upper layer, exchange of the deep basin water is inhibited and stagnation is theoretically possible. Stagnation as a consequence of restricted circulation would also be expected in areas of limited freshwater inflow. In fact, no fiords studied within the Southeast Alaska system show the formation of such waters to any marked degree, unlike areas in British Columbia where oxygen depletion of stagnant areas has given well developed anoxic basins.

The freshwater inflow into the Southeast Alaska fiord system is strongly seasonal with a pronounced primary maximum during the months of May through July. The sources for most of the rivers and streams which yield this peak runoff are the coastal snow fields and glaciers. Figure 3 (after Pickard, 1961) illustrates seasonal trends in land drainage for the essentially equivalent British Columbia system. A secondary peak occurs during the period September to December which constitutes the period of maximum precipitation.

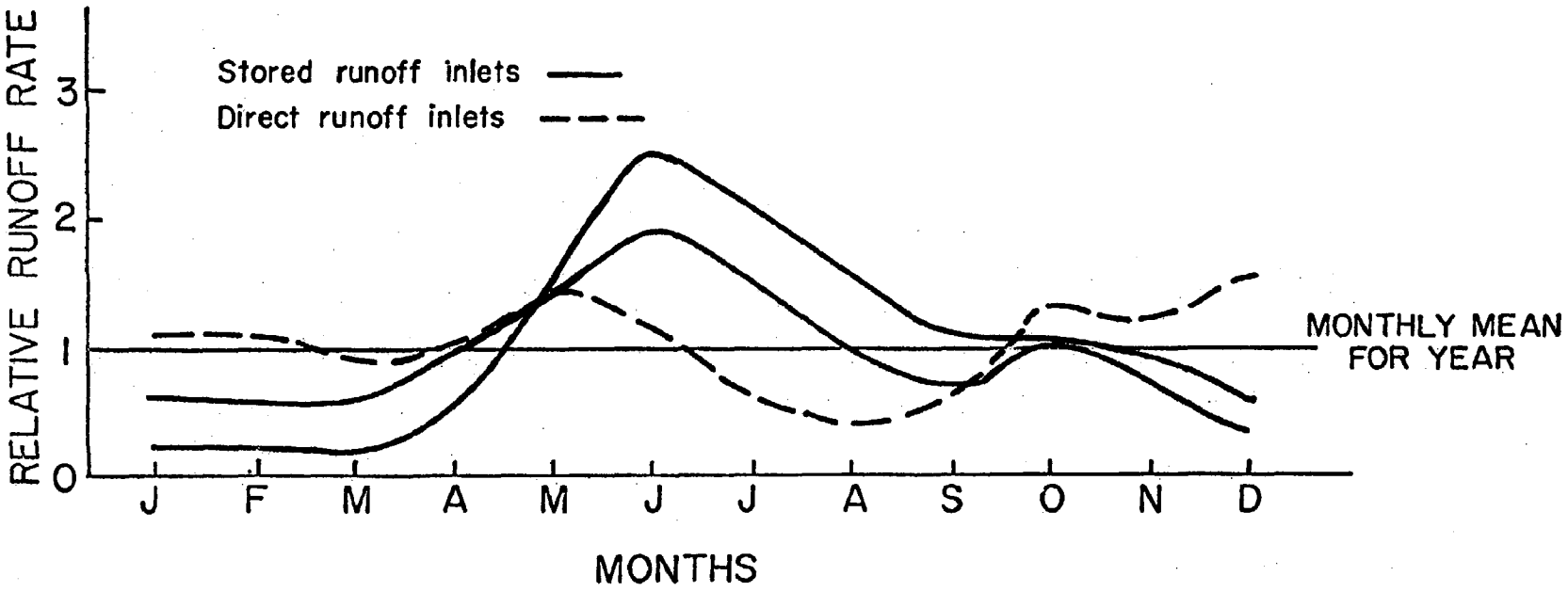

Fig. 3. Monthly runoff patterns for British Columbian fjords, typical of stored runoff (glaciers and snowfields) and of direct runoff (precipitation) expressed as a multiple of a monthly mean for the entire year (after Pickard, 1961).

During the summer months, solar radiation further increases the stability of the fiordal water structure by differentially warming the surface layers. Such a stable structuring in general persists through the fall months. Thus, Figs. 4 and 5 from Wallen and Hood (1968) demonstrate salinity and temperature distributions within Endicott Arm during October. This stability, which would give rise to stagnation, does not persist through the winter months; however, Matthews and Rosenberg (1968) have noted that continuous exchange occurs between the basin and source water of Endicott Arm during this season. The mechanics of the winter and spring water mass formation is not fully understood at present. Mixing processes at the glacier face have not been studied in any detail. However, Hartley and Dunbar (1938), working on non-Alaskan systems, reported extensive upwelling phenomena at this interface.

The major energy source within the fiords appears to be the total freshwater inflow and the effects of tides, with the former predominating. No definitive data are available on the effects of winds on the system. The presence of the glacier leads to the formation of cold, heavy air masses which may form down-slope winds over the fiords, frequently of gale-force intensity. Kilday (1966) has reported on the conditions which give rise to such Föhn winds in Taku Inlet. Such winds are, of course, essentially a winter phenomenon, at which time the upper reaches of the fiords adjacent to the glaciers are usually covered with sheet ice of varying thickness. During the summer months the glaciers continually calve icebergs into the systems. This ice is subsequently distributed with the surface water layer flow. Little work has been done to date on the effects of the presence and movement of this ice within the fiords but it obviously represents a considerable stress on the local ecosystem.

Nutrients

The overall nutrient chemical structure of typical fiords has been considered in some detail by Wallen and Hood (1968). Nutrient materials are apparently continuously introduced from the exterior source waters, from direct precipitation, and from surface drainage. The nutrient parameters may be considered to be conservative except in those seasons in which biological populations peak. During these months, which often correspond to periods of maximum water column stability as noted above, the characteristic dense phytoplankton blooms have a pronounced influence upon the chemical dynamics of the surface layers, which varies markedly from fiord to fiord. Figure 6 (from Wallen and Hood, 1968) illustrates isopleths of phosphate at a depth of 5 meters which well illustrate the seasonal variations. Wallen and Hood (op cit.) have also studied in detail the nutrient chemical regime of the surface layers of selected fiords to a depth of 1 meter. This work is particularly concerned with the chemical gradients developed across the freshwater lens-marine water interface.

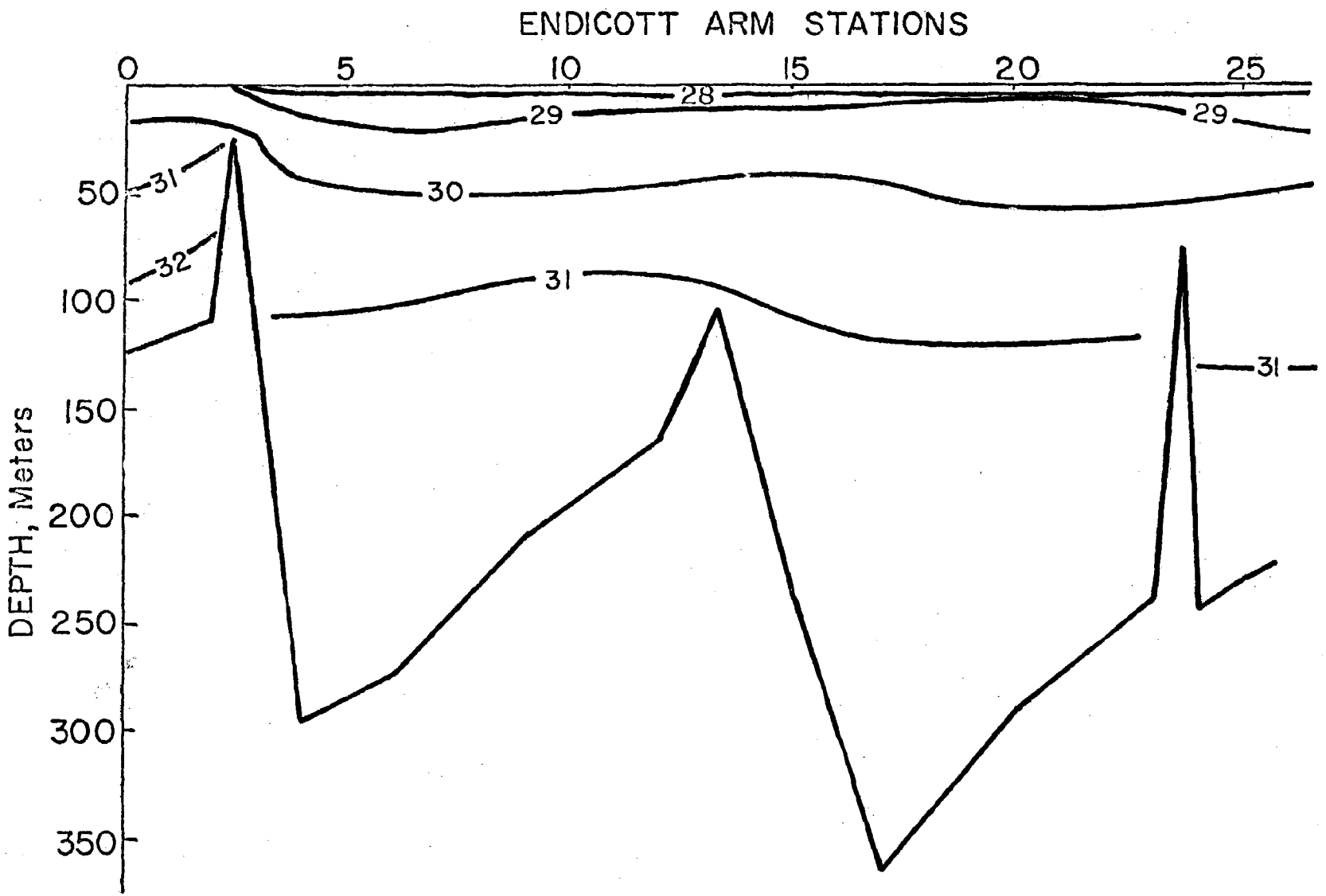

Fig. 4. Distribution of salinity in Endicott Arm, Southeast Alaska, October 1967, 1.0 °/oo intervals (after Wallén and Hood, 1968).

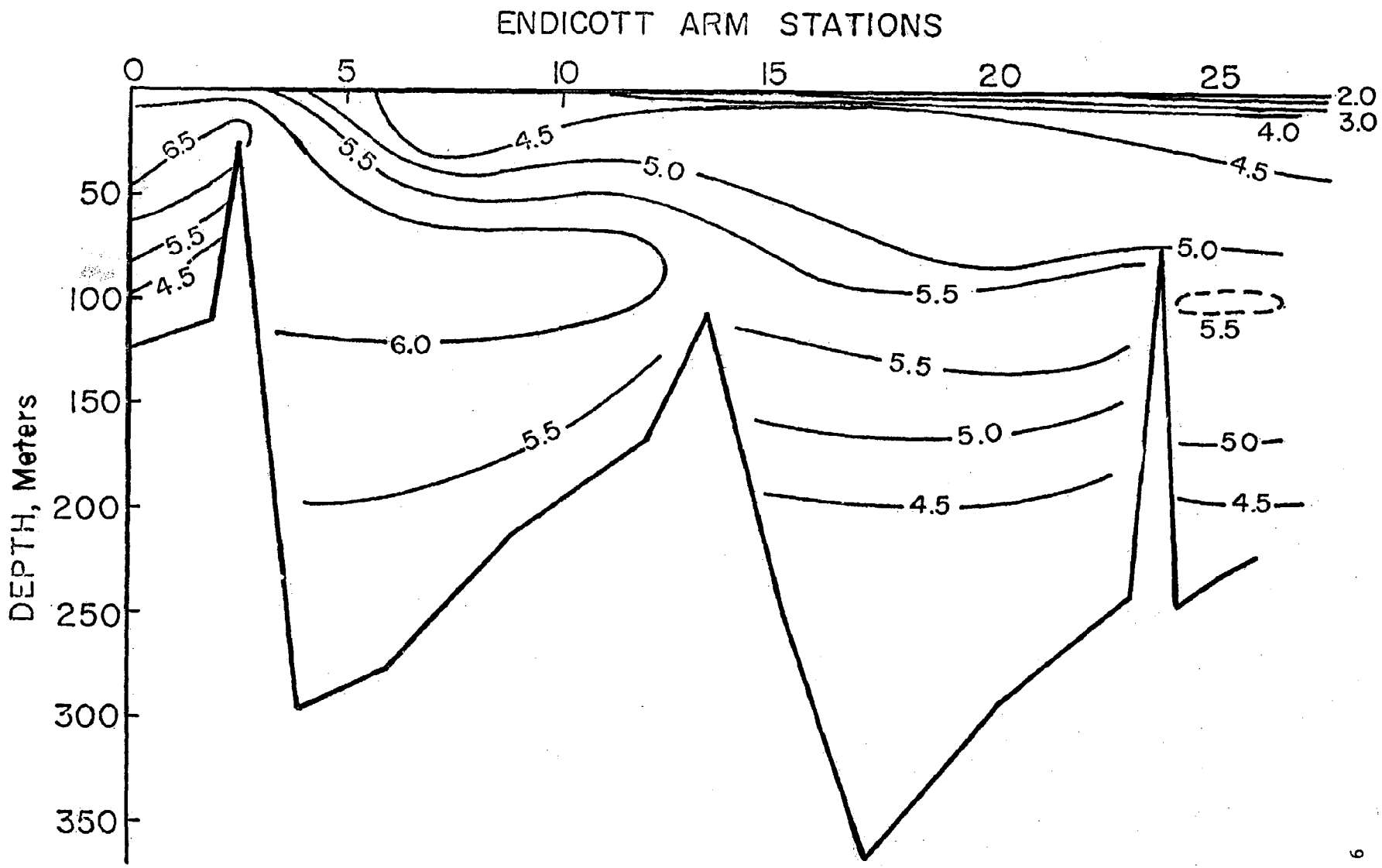

Fig. 5. Distribution of temperature in Endicott Arm, Southeast Alaska, October 1967, 0.5°C intervals (after Wallen and Hood, 1968).

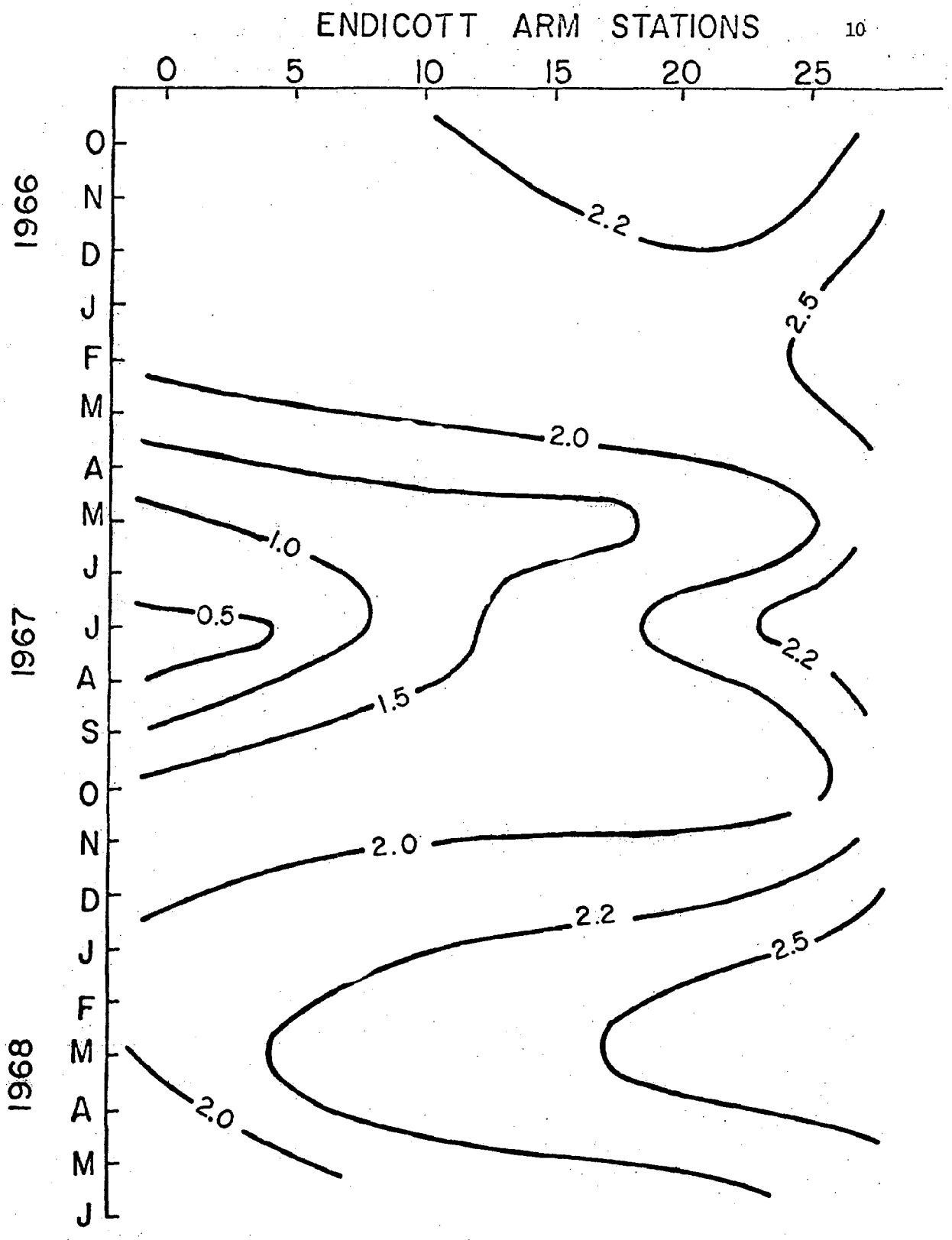

Fig. 6. Seasonal distribution of phosphate at 5 m depth in Endicott Arm, Southeast Alaska; 0.5 µgA/l intervals (after Wallen and Hood, 1968).

Chapter D-2

TURBID OUTWASH FIORDS

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INTRODUCTION

Turbid outwash fiords are fiords that do not have glaciers encroaching on the intertidal area. In contrast to the clear-water environment of the glacial fiord, large quantities of glacially ground sediment are transported into the inlets by glacial melt water from the sediment deposits between glacier and fiord. The generalized hydrographic environment of these fiords is otherwise essentially as outlined in Chapter D-1 and is largely dependent on the presence or absence and form of the sills present in each individual estuary. The reason for separate consideration lies in the characteristic and considerably complex geologic regime developed in these estuarine types. The discharge, transportation, and ultimate disposal of these large particulate sediment loads must presumably represent a great stress on the local ecological environment. However, to date, no definitive studies of the biota developed under such conditions have been attempted. Because of the many factors which must be considered, it is not yet possible to categorize the turbid outwash fiord types of Southeast Alaska. Each inlet thus designated must be considered individually.

EXAMPLES

Taku glacier, near Juneau in Southeast Alaska, is currently advancing at a relatively rapid rate (see Chapter D-1, Figure 1). The glacial front is, in fact, geographically within Taku Inlet. However, the natural development of an extensive terminal moraine has provided the necessary source of glacial sediment for transportation of freshwater runoff into and down the inlet. Kunze et al. (1967) have studied the mineralogy of this particulate material and have found the clay sized material to be most distinctive. Apparently under these erosional conditions in a subarctic environment, degradation of primary minerals is greatly inhibited. Much of the glacial sediment is transported only short distances from the sub-aerial moraine. Jordan (1962) has given some estimation of the high sedimentation rates in waters adjacent to the glacier. Tidal action results in a complex cycle sedimentation sequence whereby mud-flat deposited material is continuously reworked.

Cook Inlet, although not part of the Southeast Alaska system, is most worthy of note here because of the diverse oceanographic work recently accomplished. An extensive river system carries vast quantities of glacial sediment into the upper inlet during the summer months. Love (1955) has estimated the annual load to be of the order of 13-19 million tons. Most of this material is, of necessity, transported during the summer months only, since

the upper inlet is ice covered for at least four months of the year. The absence of any restricting physical barriers and the overall shallowness of the upper inlet ensures that the incoming freshwater and contained sediment load is well mixed vertically with the inlet waters. Such summer hydrographic conditions have been described in detail by Rosenberg and Hood (1967). The sediment load adjacent to the outflow of the principal glacial-fed rivers has been found to exceed 1 g/l during July (Rosenberg et al., 1967). The turbidity diminishes in an approximately regular fashion down the well mixed upper inlet. The subsequent depositional environment of the middle and lower reaches is considerably more complex, as noted by Sharma and Burrell (1968) (Fig. 1).

Several considerably smaller and less complex examples of the "turbid" type of fiord have been studied over several years within Glacier Bay National Monument, north of Juneau in Southeast Alaska (see Chapter D-1, Fig. 1). Queen Inlet is a small (approximately 1 mile x 6 mile) one-glacier system (Fig. 2). Approximately 1 mile of intertidal mud-flat and terminal moraine separates the fiord-proper from the glacier. The sedimentary regime here has been studied over several years and is one of considerable complexity. Active transportation and deposition occurs, as in other such areas, only during the summer months. Deposition is thus strongly seasonal. Bottom sediment core samples exhibit graded bedding and vertically regular oxidized layers. If, as may be presumed, these latter marker horizons represent annual winter stagnations then deposition rates at the head of the inlet may exceed 1 meter per year in some areas. Deposition is far from uniform, however. The intertidal mud-flats are cyclically eroded and redeposited under tidal action and there is evidence for extensive turbidity currents originating on the mud-flat slopes and flowing down the inlet (Hoskin and Burrell, 1968).

The glacial melt water flowing into Queen Inlet, carrying sediment in suspension, flows over the in situ marine water to form the characteristic glacial sediment "plume." This freshwater lens--only a few centimeters in thickness--maintains its identity at low water for over 3 miles down the inlet before mixing with the marine waters. Figure 3 (after Burrell, 1967) illustrates the systematic variations in sediment load, salinity, and temperature parameters of the surface layers of Queen Inlet. The sediment is presumed to flocculate on contact with the saline water but the deposition sequence is far from simple. It is known that the depositing sediments occupy discrete layers within the water column, no doubt due to the exceedingly complex sub-surface current structure. Also, Burrell and Gatto (unpublished data) have found no distinctive differences in the size distribution of suspended sedimentary material from the surface downward through the vertical water columns.

Geochemical studies of sediments within Muir Inlet, Glacier Bay (Burrell and O'Brien, 1968) have essentially confirmed the mineralogical data presented by Kunze et al. for other fiords in Southeast Alaska as noted above. Retardation of weathering processes was further graphically demonstrated by the sharp grain margins as shown in many electron micrographs. The extreme mineralogical uniformity of clay sized material down the length of the 20-mile inlet was particularly noteworthy. The particulate sedimentary material was found

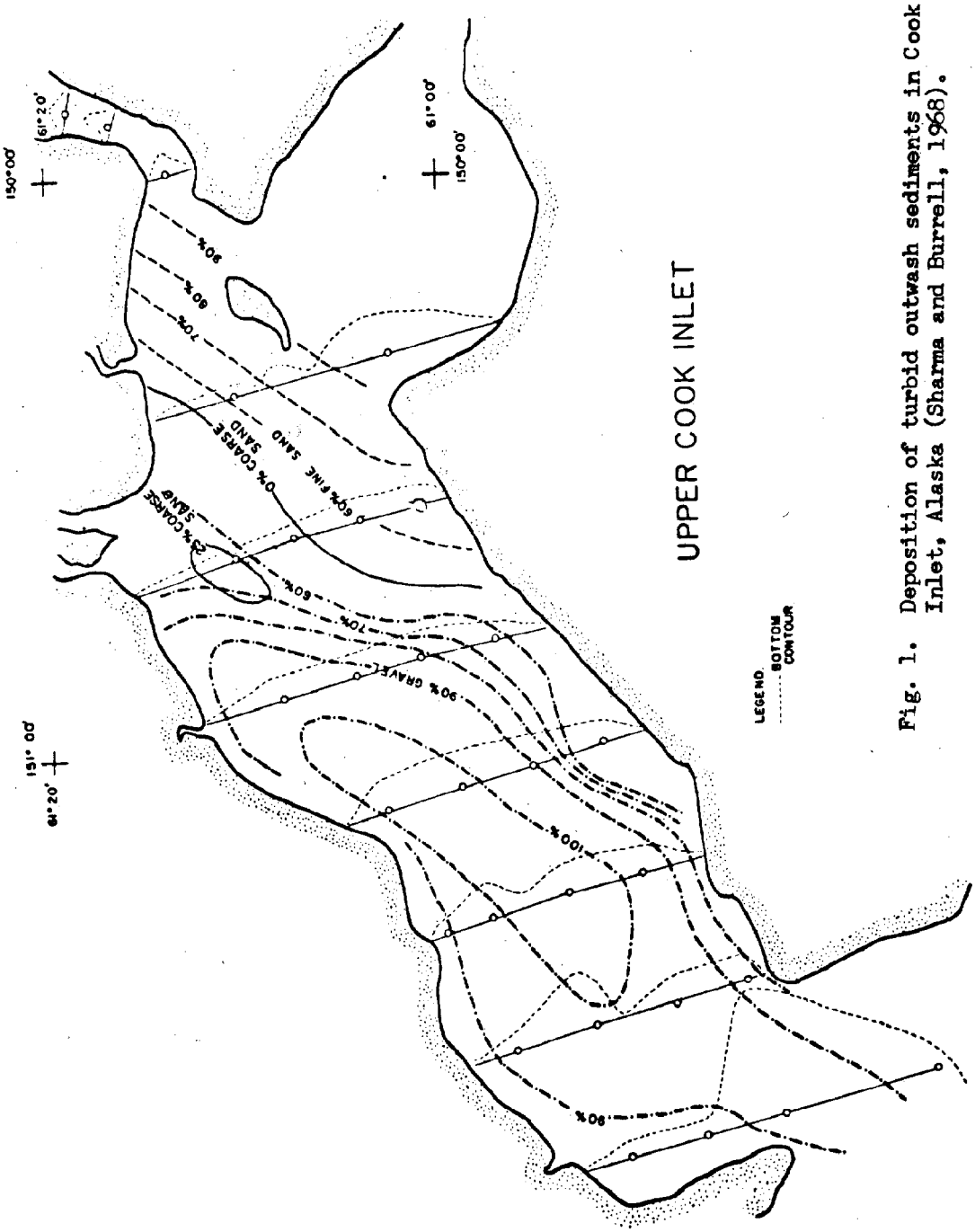

Fig. 1. Deposition of turbid outwash sediments in Cook Inlet, Alaska (Sharma and Burrell, 1968).

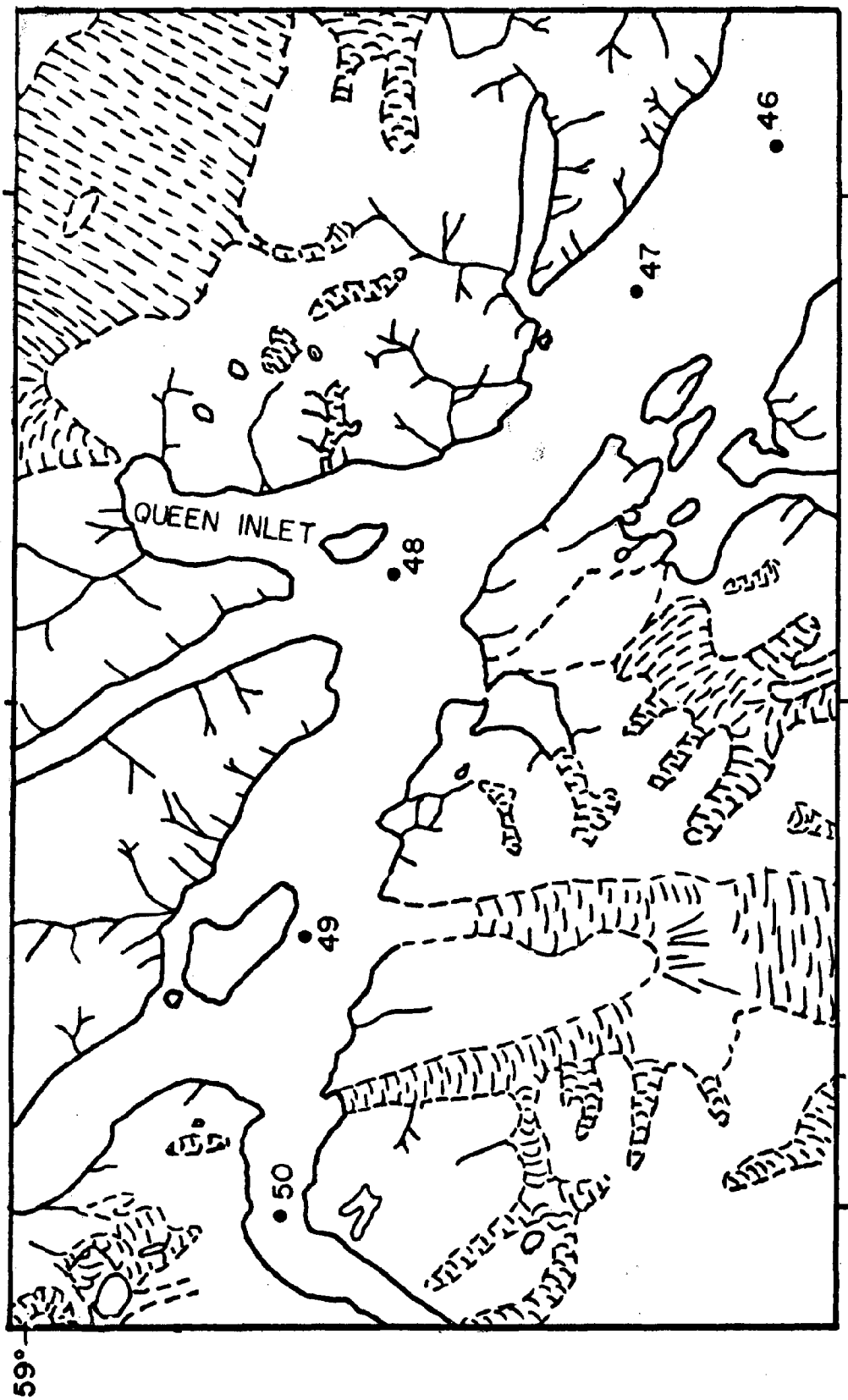

Fig. 2. Glacier Bay, Alaska, showing Queen Inlet, a turbid outwash fiord (Hoskin and Burrell, 1968).

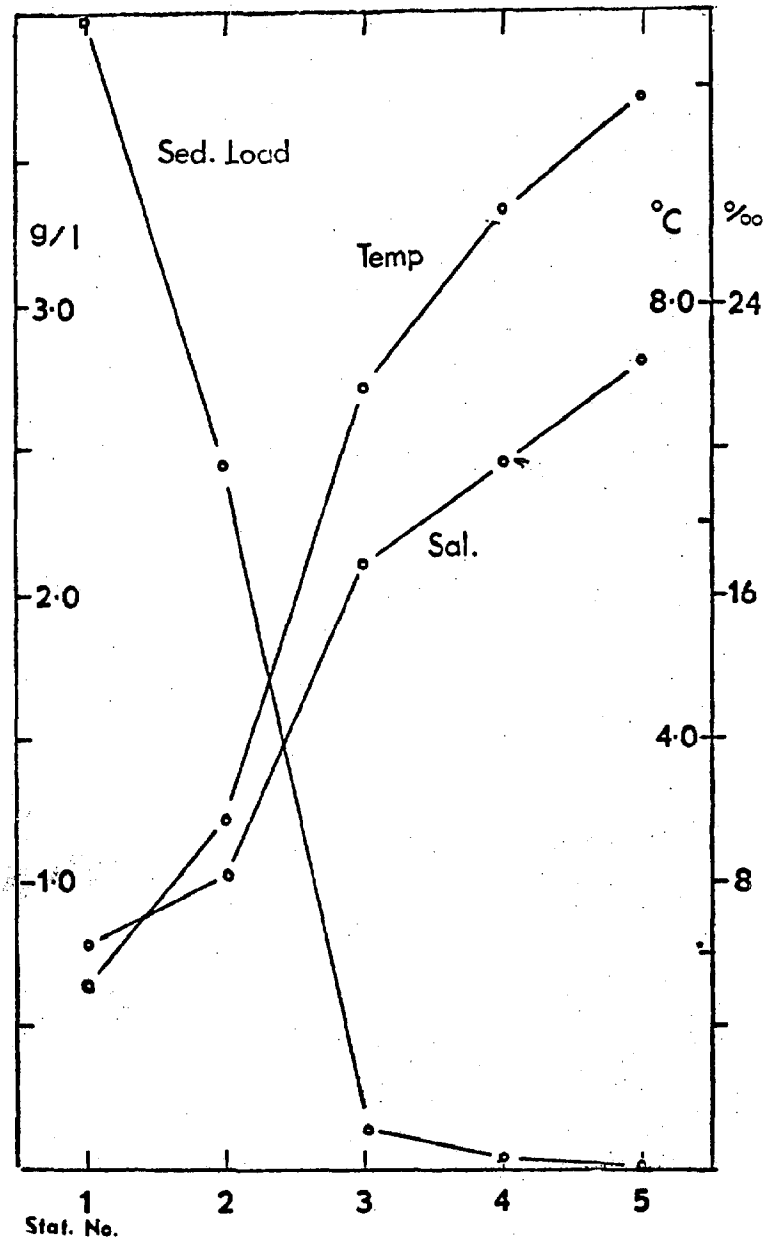

Fig. 3. Surface water parameters for a longitudinal section down Queen Inlet, Glacier Bay, Southeast Alaska; sediment load g/l; temperature °C; salinity °/oo (After Hood and Burrell, 1967).

to equilibrate rapidly with the marine environment with respect to the major ions adsorbed on exchange sites. This has been observed for all other fiords similarly studied (e.g., Sharma and Burrell, 1968). The role of trace elements within turbid-type fiords is, however, considerably more complex. Iron and manganese introduced via the freshwater inflow are rapidly reduced in solution in the marine environment, as would be expected. Manganese appears to be remobilized under the existing redox environment of the surface sediments within Muir Inlet and it is of ecological interest to note that manganese was greatly concentrated and by far the most abundant trace metal in a series of intertidal molluscs analyzed by Burrell and Gross (1967) in Taku Inlet. Initial detailed data on the zinc budget of a typical turbid outwash fiord (Ali and Burrell, unpublished data) have shown that this element is adsorbed on the glacial floor in concentrations up to 20 times that in the coexisting marine environment.

Chapter D-3A

ICE STRESSED COASTS

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Ice stressed coasts are an ecosystem type characterized by ice, ice formed by the freezing of the arctic seas. Ice along a shore has profound effects on the fauna and flora of the coast in that it eliminates most organisms from the littoral zones of the sea. On the arctic coast the pressure on sea ice from wind stress and currents is translated to the fast ice on shore and causes scouring.

EXAMPLES

The ice stressed littoral system reaches a maximum intensity on the northernmost coast of Alaska, from Point Barrow east, where ice is present from September through July and in extreme years may be on and offshore all summer. The effects of ice on coastal systems diminish with the decreasing latitude along the Chukchi and Bering Sea coasts of Alaska. The southern extent of the ice stressed system varies with the intensity of winter; it can extend as far south as Izembek Lagoon near the tip of the Alaska Peninsula (55°N; Fig. 1). In these lower latitudes the ice effects are much less than in the Arctic. Ice cover on the open Bering Sea never extends as far south as the Alaska Peninsula.

Although the stress of ice influences long portions of the shoreline of Alaska (approximately 2800 miles), most studies have been limited to the region near the Naval Arctic Research Laboratory at Point Barrow. For other regions only inferences can be made, based on the Point Barrow work. The best known ice stressed coasts are those of the arctic Soviet Union.

PHYSICAL SYSTEMS AFFECTING ICE SHORES

The ice stressed coast of Alaska, from the tip of the Alaska Peninsula to Point Barrow and beyond, is generally flat and low. Exceptions are the three mountain ranges that intersect the coast: the Kilbuk Mountains on the north shore of Bristol Bay, the mountains of the Seward Peninsula, and the western extremity of the Brooks Range (Wahrhaftig, 1965). Included in this coast are the several island groups of the Bering Sea shelf: the Pribilof Islands (57°N), Nunivak Island (60°N), St. Matthew Island (61°N), and a few

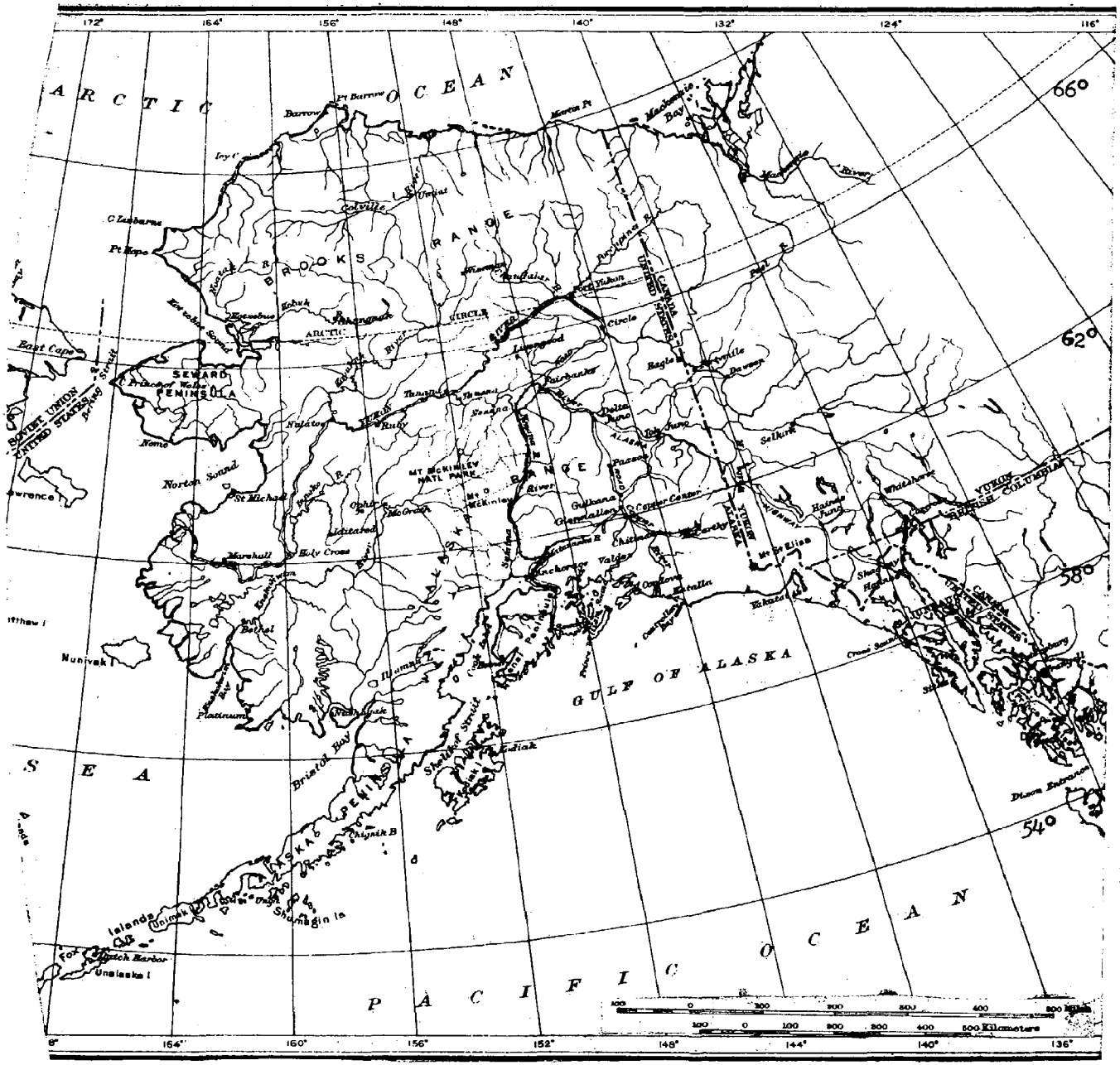


Fig. 1. Map of Alaska (U.S. Federal Water Pollution Control Administration, 1967).

other small islands. Two major rivers, the Yukon and the Kuskokwim, empty into the eastern Bering Sea, forming a large marshy delta with many distributaries. Farther north a few smaller rivers enter the Chukchi Sea; some of these are the Kukpuk, Kobuk, and the Noatak. On the arctic coast east of Point Barrow, the Colville is the largest Alaskan river. Flow of these rivers varies greatly with season (Table 1).

Adjacent to this coast in the Bering and Chukchi seas is a broad, flat, continental shelf (Fig. 2). The gradient on this shelf is only about 0.13 m/mile; depth averages 45 to 55 m (Dietz et al., 1964). Circulation in the eastern Bering Sea is northerly, although intensity may vary seasonally. Pacific water enters through the Aleutian passes and flows north along the Alaska coast into the Chukchi Sea. Velocities are generally low, except in Bering Strait where they reach 1 knot (Fleming and Haggerty, 1966). Fleming and Haggerty also studied summer temperature and salinity for part of this area (Figs. 3 and 4). Ekman (1953) compiled a comparison of summer maximum temperatures for this region (Table 2).

Water continues its northerly flow on the Alaska coast of the Chukchi. Off Point Hope, the flow diverges and one arm continues north along the coast toward Point Barrow (Zenkevitch, 1963). The northerly Chukchi flow and the anticyclonic gyre of the Beaufort Sea converge at Point Barrow. The northerly flow of relatively warm Pacific water from the Aleutians into the Arctic permits range extensions into the Arctic of many boreal (north temperate) species of marine animals (Ekman, 1953).

Tides along the ice stressed coast diminish in the Bering Sea with the increasing latitude (Dietz et al., 1964). In the Bristol Bay region tidal amplitude may reach 6 m. In the Bering Strait region this is about 0.5 m and at Point Barrow it is only 0.1 to 0.2 m (Fig. 5). An additional non-tidal variation in sea level at Point Barrow ranges from 0.27 to 0.65 m, being lowest in March, and is caused by changes in atmospheric pressure (Beal, 1968). North of Bering Strait tides have a minor influence on the scouring movements of sea ice on the beach. South of the sea ice edge, in Bristol Bay and on the Alaska Peninsula, tidal energy imparted to coastal ice causes the major scouring action. This scouring mechanism is confined to the intertidal zone, whereas that on the arctic coast reaches much deeper (Rex, 1955). The beaches of the ice stressed coast are primarily composed of sand or gravel with varying mixtures of silt and clay (Rex, 1964; Carsola, 1954, Creager and McManus, 1966; Moore, 1966). One exception to this is Fairway Rock in Bering Strait (Schumway et al., 1964), and others exist around some of the Bering Sea island groups (Preble and McAtee, 1923). The absence of hard substrates north of Bering Strait is an important factor determining the types of biological communities on this coast.

Table 1. Monthly mean discharge (ft^3/sec) for two northern streams in Alaska (1966). Data from surface water records of Alaska, 1967, Part I, Surface water records. U. S. Dept. of Interior Geological Survey, Water Resources Division

	Jan	Feb	Mar	April	May	June	July	Aug	Sept	Oct	Nov	Dec
Snake River near Nome	36	28	22	20	55	1,480	380	280	149	227	85	50
Kobuk River at Ambler	1,300	900	800	950	15,140	58,730	30,980	23,980	12,360	9,406	3,000	2,000

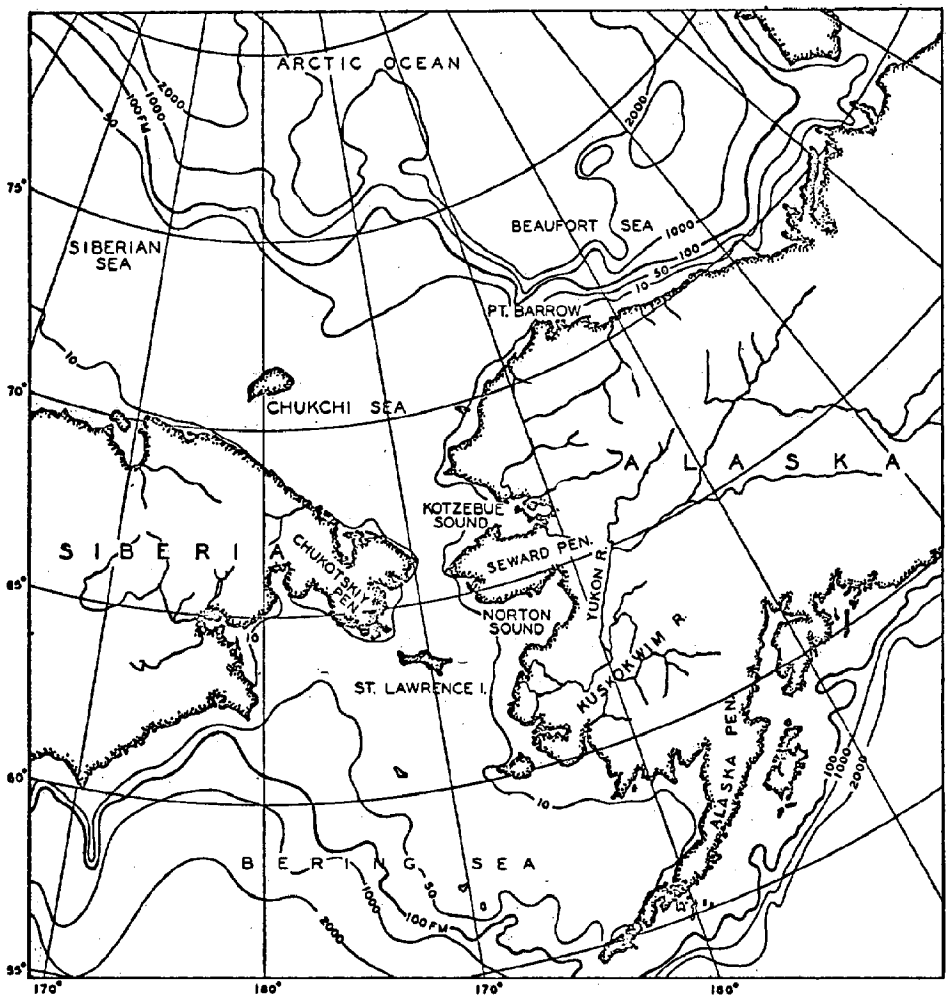


Fig. 2. The Bering - Chukchi - East Siberian continental shelf. Contours in fathoms (from Dietz et al., 1964).

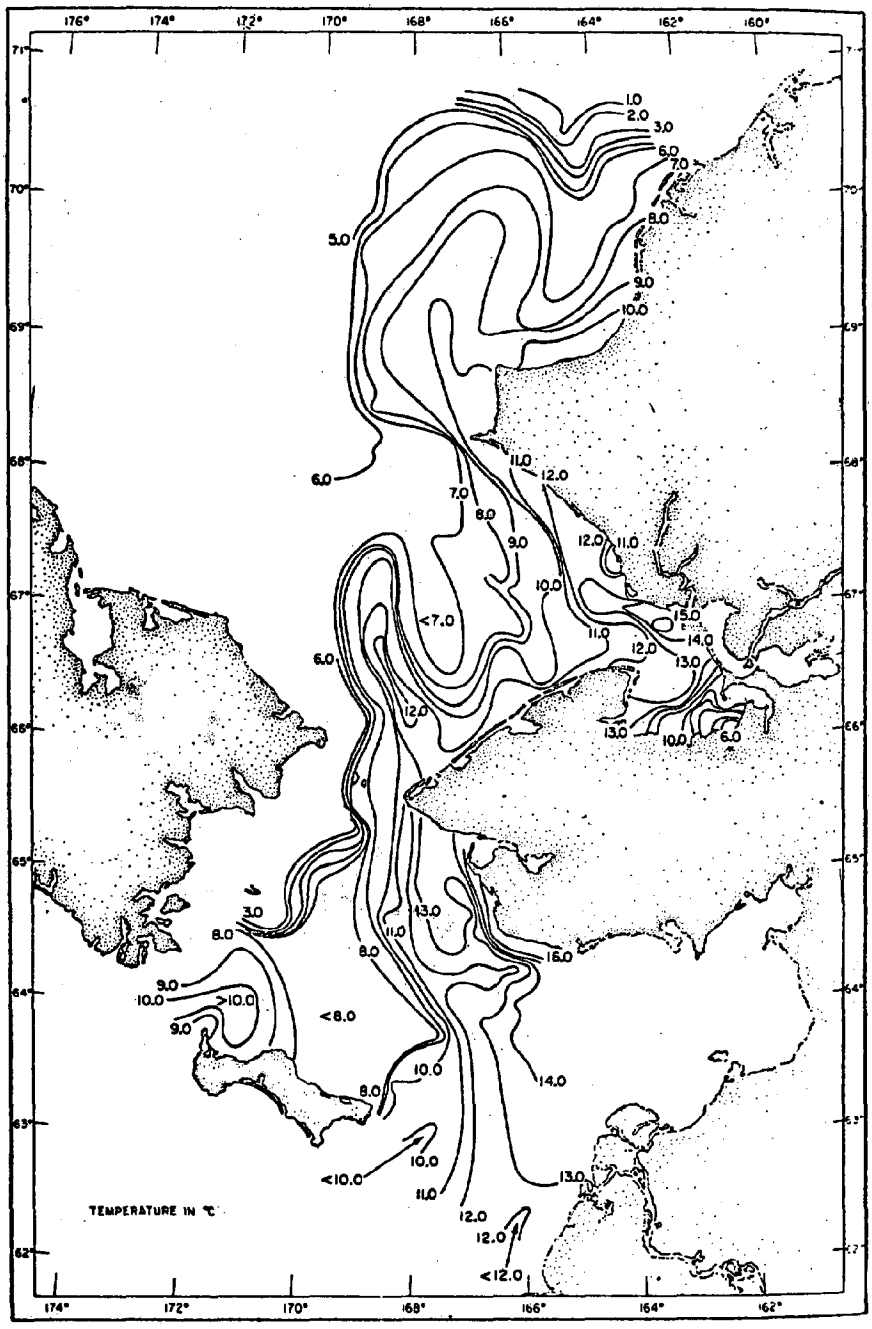


Fig. 3. Surface temperatures in summer in the Bering and Chukchi seas (from Fleming and Haggerty, 1966).

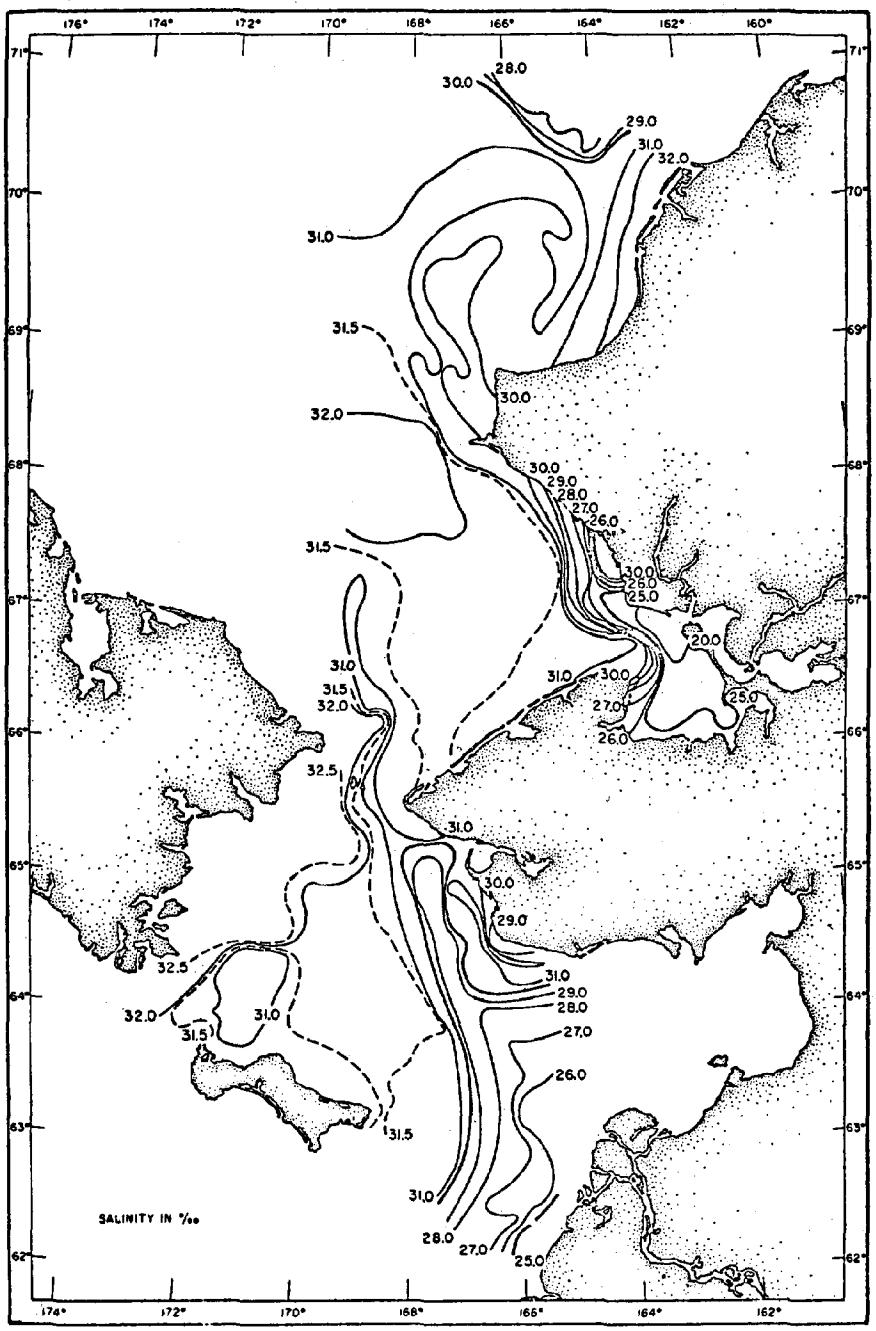


Fig. 4. Surface salinities in summer in the Bering and Chukchi seas (from Fleming and Haggerty, 1966).

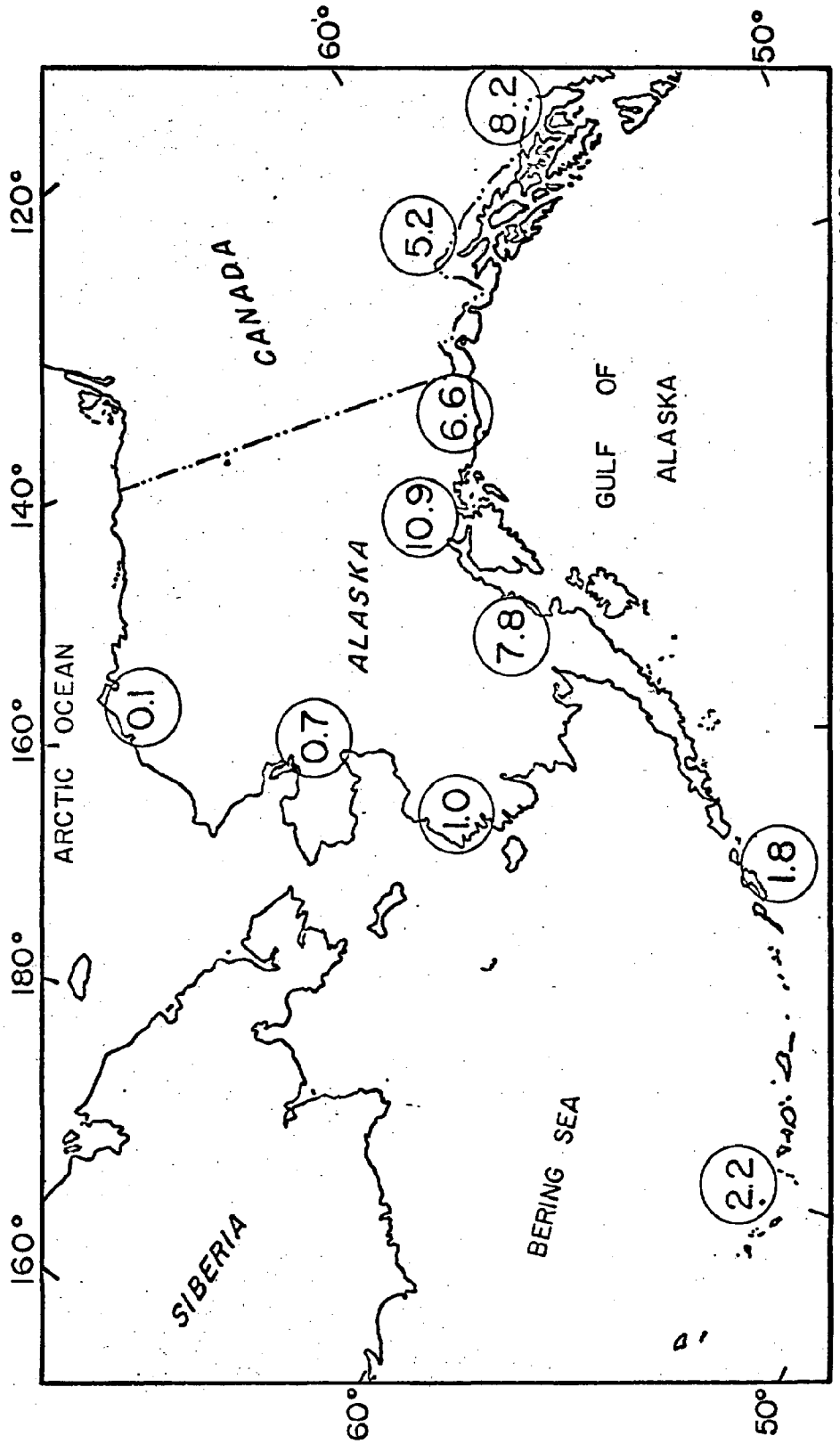


Fig. 5. Tidal differences in meters at selected locations on the coast of Alaska (from US Coast and Geodetic Survey, 1966).

Table 2

THE HIGHEST SUMMER TEMPERATURES OFF THE ALASKAN COAST OF THE BERING SEA AND THE POLAR SEA. A: AT UNALASKA IN THE ALEUTIAN CHAIN; B: BETWEEN UNALASKA AND THE PRIBILOF ISLANDS; C: OFF PRIBILOF ISLAND; D: OFF NUNIVAK ISLAND; E: NORTON SOUND; F: BERING STRAIT; G: NORTH-EAST OF BERING STRAIT; H: SOUTH-WEST OF POINT BARROW (Ekman, 1953).

Depth	Bering Sea					Bering Strait	Polar Sea	
	A 53° 57′ N.	B 55° 30′ N.	C about 57° N.	D about 60° N.	E 63° 30′–64° 30′ N.	F 65° 20′–66° N.	G 66°–68° N.	H 70°–71° N.
0 m.	7·55° C.	7·4–10·2° C.	6·52–7·91° C.	6·29–9·43° C.	10–13·42° C.	7·27–12·29° C.	7·7–10·78° C.	0·09–6·83° C.
10	7·20	6·9–9·3	7·10–7·26	6·16–7·89	10·72–11·50	3·91–11·39	8·3–10·29	−0·11–5·31
20–25	6·67	6·5–8·6	2·96–7·47	6·11–7·71	6·90–6·92	2·44–4·92	6·13–8·12	−0·51–5·11
50	6·33	4·6–7	2·84–6·21				4·65–5·28	
100	5·25	3·24–5·2	3·8					
200		3–4·1						

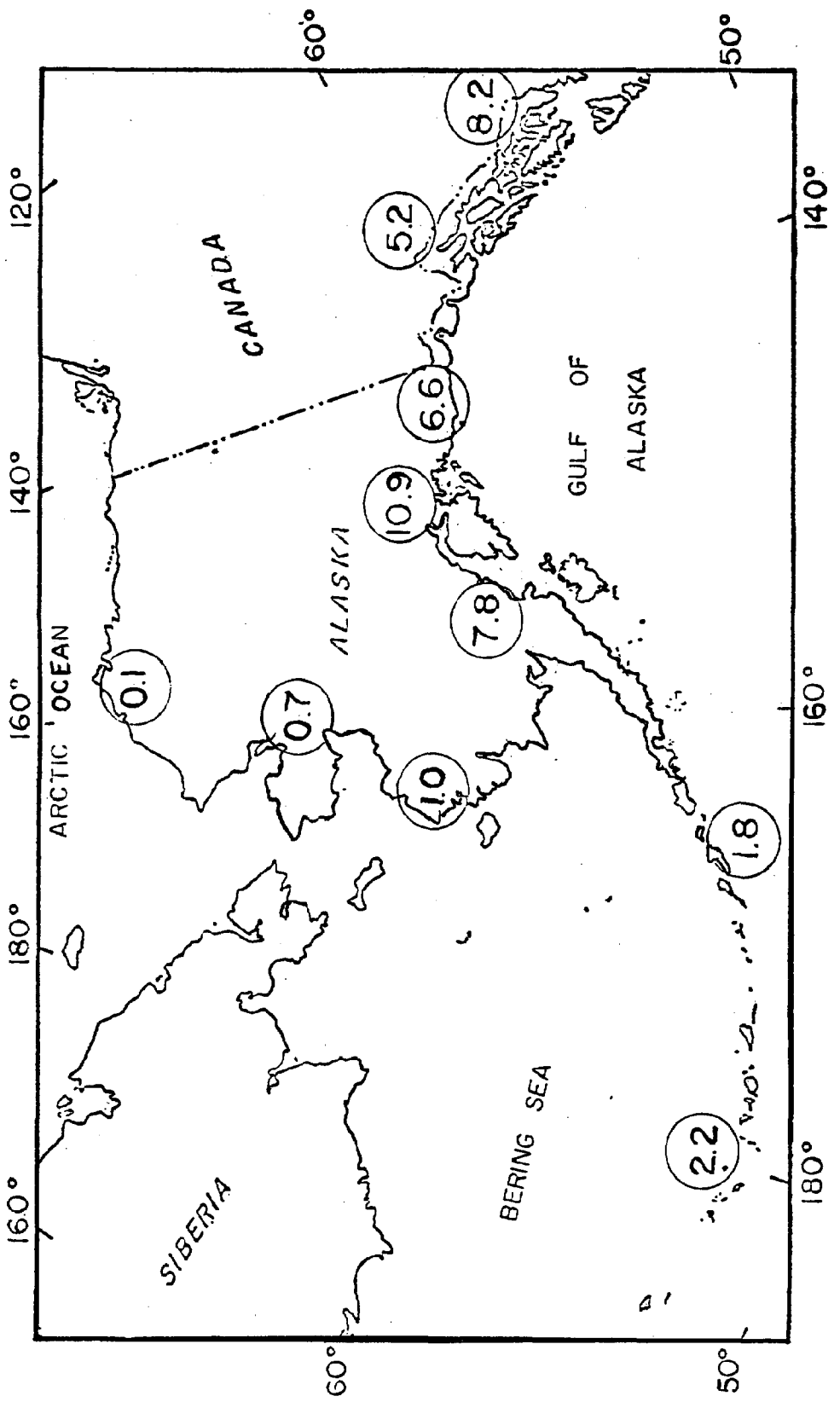


Figure 5. Tidal differences in meters at selected locations on the coast of Alaska (from U.S. Coast and Geodetic Survey, 1966).

A characteristic feature of the entire length of the ice stressed coast is a series of shallow lagoons separated from the sea by barrier islands, beaches, and spits. These lagoons grade from marine, where several passes connect to the sea, to freshwater, where the basin is entirely closed. They vary in size and conditions from the very arctic Nuwuk Pond (228 m long) on Point Barrow (Mohr, 1953) to the more temperate Izembek Lagoon (50 miles, or 80450 m, long) on the Alaska Peninsula (McRoy, 1966). The lagoon system experiences different types and intensities of stress than do the adjacent ocean beaches.

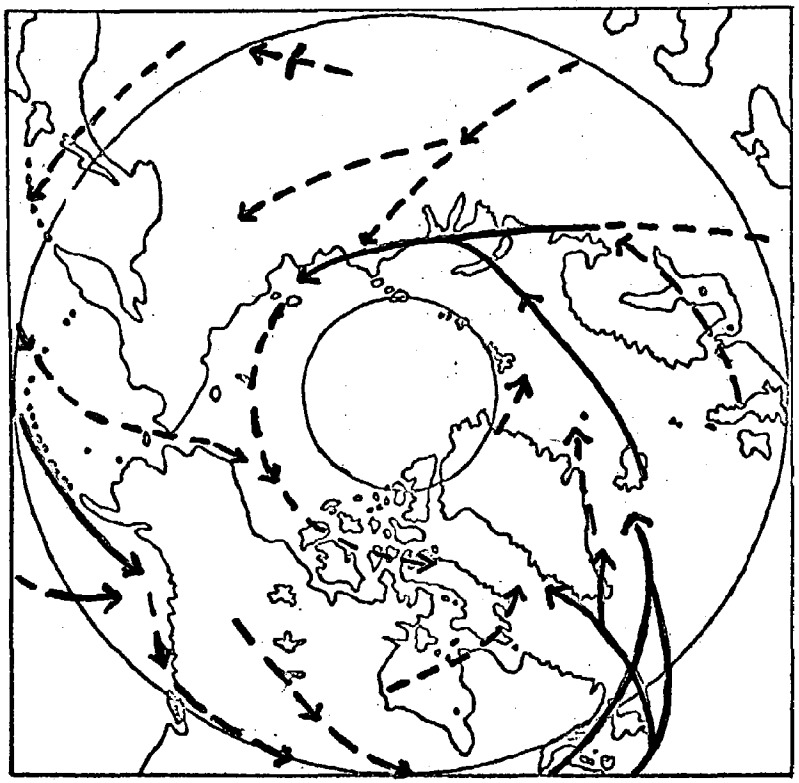
The human population of the ice stressed coast is sparse and scattered. Major towns are Nome, Kotzebue and Barrow; not one of these exceeds 2500 people. There are scattered villages of a few hundred or less, mostly less, along this coast. Outside of the three larger towns there are no pollution problems posed by these communities. The people are Eskimos. Perhaps unlike all other systems associated with man in the United States, these people use their skill and talents within the structure of the natural ecosystems (Saario and Kessel, 1966).

The climate along this coast has many similar features. The lower latitudes are, of course, warmer than the arctic region but the entire coast in summer has a cloudy maritime climate. In winter this becomes a more continental type since the sea is covered by ice and snow (Wilson, 1963). The transition from a cold continental climate to a maritime climate is abrupt. The major winter storm tracks are a sort of giant spiral beginning at Japan and ending north of eastern Siberia; in summer the tracks tend to be zoned following the north coast (Fig. 6.). The intensity of atmospheric circulation is closely related to the extent of the arctic ice pack. The studies of the heat balance of the Arctic have revealed a startling integration of global climate and ice (Fletcher, 1965).

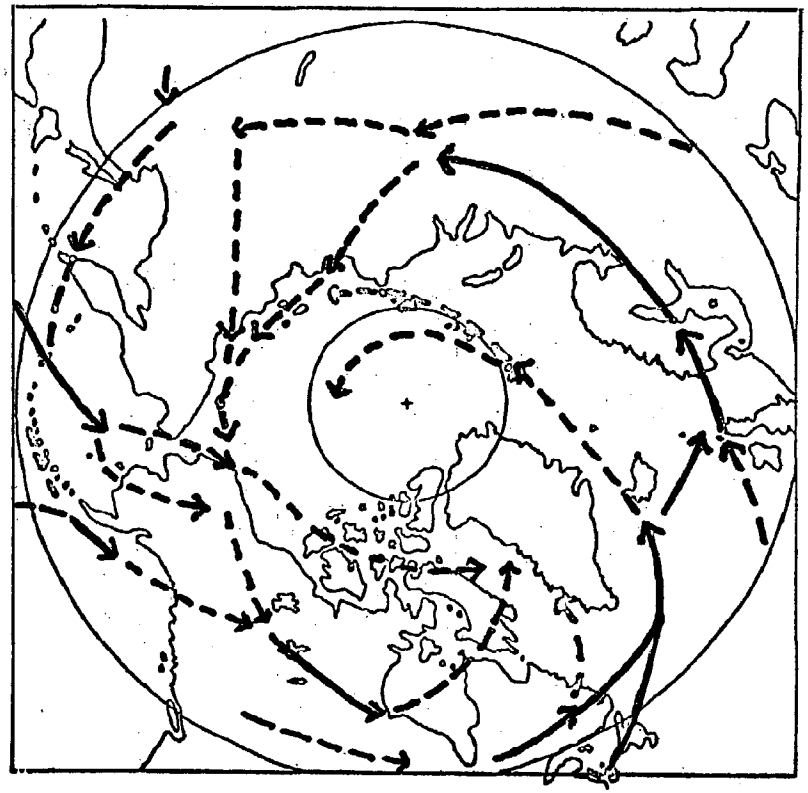
BIOLOGICAL SYSTEMS

An arctic beach is a rigorous environment. The intertidal fauna and flora characteristic of the sea in almost all other parts of the world do not exist in Arctic Alaska (MacGinitie, 1955). Ice combined with the low tide range and unconsolidated beach sediments provides conditions too rigorous for the development of a permanent fauna and flora. The mesopsammon, the animals smaller than sand grains, however, has not been examined, and an assemblage could exist, at least in the ice-free season. In areas of Arctic Canada, where the tide range is greater than the ice thickness and the shore is not directly exposed to the open sea, organisms can occupy the lower levels of the littoral (Ellis and Wilce, 1961).

Although the intertidal is barren the sublittoral contains many species (Table 3). The diversity of species here is much less than on lower latitude beaches but the total number of individuals is large. Quantitative studies of the benthos on the Siberian (Soviet) coast of the Chukchi Sea indicate a low biomass of 25 to 30 g/m^2 or less at 7 or 8 m deep (Zenkevitch, 1963). The mud-bottom population in depths of 30 to 50 m in this same region can be much higher (Table 4).



Principal tracks of lows in January.



Principal tracks of lows in July.

Fig. 6. Principal storm tracks in January and July in the Northern Hemisphere (from Wilson, 1963).

Table 3. Sublittoral invertebrates in order of relative abundance on soft bottoms near Point Barrow (from MacGinitie, 1955).

I. Infauna of Near Shore Mud Zone:
Echiurus echiurus alaskanus (echiuroid worm)
Arenicola glacialis (polychaete)
Myriotrochus rinki (sea cucumber)
Antinoe sarsi (polychaete)
Melaenis lovenis (polychaete)
Harmothoë imbricata (polychaete)
Hamingia arctica (echiuroid worm)
Halicryptus spinulosus (priapulid worm)
Burrowing anemones (2 species)
Burrowing tunicates (2 species)
Mya truncata (clam)
Mya japonica (clam)

II. Infauna of Outer Shore Mud Zone:
Cerianthus sp. (burrowing anemone)
Macoma calcarea (clam)
Astarte montaqui (clam)
Myriotrochus rinki (burrowing sea cucumber)
Nucula tenuis (clam)

III. Epifauna of the Mud Zone:
Foraminifera
Dulichia porrecta (amphipod)
Pagurus splendescens (hermit crab)
Pagurus trigonocheirus (hermit crab)
Hyas coarctatus alutaceus (crab)
Serripes groenlandicus (clam)
Liocyma fluctosa (clam)

Table 4. Biomass of sublittoral benthic invertebrate groups on the Siberian coast of the Chukchi Sea (from Zenkevitch, 1963).

Invertebrate Group	Biomass g/m^2
Polychaetes	35.2
Crustaceans	31.8
Echinoderms	16.2
Gastropods	2.0
Lamellibranchs	114.6
Others	14.4
Total	214.2

MacGinitie frequently walked the beaches around Point Barrow collecting marine animals. In the summer of 1948, his total collection consisted of one polychaete, one empty snail shell, and fragments of bryozoan colonies. This was an extreme year but it indicates the paucity of the intertidal life. MacGinitie (1955) took no quantitative samples of the benthos but was impressed with the abundance of life along the Point Barrow coast. He estimated the maximum biomass to be about 260 g/m^2.

Apparently the effects of ice diminish rapidly below about 10 m and the benthic community attains a biomass comparable to other shallow seas. Zenkevitch (1963) says that the mean biomass of all shallow Siberian arctic seas is similar (about 200 g/m^2). Ice affects this zone, as geological evidence indicates, but the effects must be irregular, permitting a well-developed community.

South of Bering Strait, especially along Bristol Bay and the Alaska Peninsula, littoral organisms are not completely removed by ice. These organisms are frequently exposed to freezing. Typically these organisms are adapted to survive freezing (Kanwisher, 1955 and 1957). Fucus sp. for example, can survive temperatures down to -40°C and photosynthesizes immediately on thawing (Kanwisher, 1957). Scholander et al., (1957) found that fishes in shallow arctic water altered the osmotic concentration of their body fluids to prevent freezing and some deep water arctic fishes actually exist in a super-cooled condition during winter. Freezing adds additional stresses to the systems of arctic and subarctic organisms. Low temperatures, in general, also physiologically stress metabolism. The arctic and subarctic littoral biota have apparently adapted to compensate for these conditions (Dunbar, 1968).

In the lagoons along the ice stressed coast the patterns and intensity of stresses can be quite different, resulting in improved conditions for marine life. Elson Lagoon, adjacent to Point Barrow, apparently receives heavy ice scouring similar to the nearby open beaches, for the fauna collected by Mac Ginitie did not differ much from that from other areas. The more southern lagoons can be quite different. The marine angiosperm eelgrass (Zostera marina) occurs sparsely in the lagoons on the Seward Peninsula north of Bering Strait and forms dense populations in most lagoons from Port Clarence to Izembek (McRoy, 1968). Eelgrass in the lagoons on the Seward Peninsula survives the winter under 1 to 2 m of ice in near darkness. These lagoons apparently receive little scouring below the intertidal zone and a well developed community exists. Biomass (live weight) ranges from 1.3 to 5.7 kg/m^2 in summer.

In Izembek Lagoon, the southernmost of this lagoon series, the eelgrass beds are very shallow and as a result of the relatively large tidal differences (compared to the arctic coast) slabs of sea ice scour long furrows through the eelgrass. These scour marks are visible for several years. The amount of scouring varies with the intensity of the winter; generally only a small portion of the standing stock is removed (McRoy, 1966). Here summer biomass is an average of 1.46 kg/m^2. A somewhat restricted intertidal fauna develops in this lagoon.

The arctic coast has few macroscopic algae. Collections of the Canadian Arctic Expedition (Collins, 1927), the U. S. Coast and Geodetic Survey (Wilimovsky, 1953), and the Naval Arctic Research Laboratory (Wilimovsky, 1954; MacGinitie, 1955) have produced laminaroids and other macrophytes at only 12 of more than 150 stations that stretch from Wainwright to Barter Island (Fig. 7). Mohr et al., (1957) list the algae that have been found at the one known kelp bed and other places along the Alaska arctic coast (Table 5).

The lack of marine macrophytes and invertebrates along this coast is attributed to several factors. The physical scouring by ice will of course prevent any organism as fragile as a marine plant from establishing itself in the upper 5 to 8 m. Ice normally pushes the beaches of the arctic coast into mounds and ridges that sometimes reach 7 to 10 m (Hume and Schalk, 1964). This action, which is an irregular but usually annual feature of northern beaches, may move from 1 to 10% of the beach sediment.

Pressure ridge ice frequently scours the bottom down to 30 m off Point Barrow, creating ridges and trough features (Rex, 1955). This deep grounding of pressure ridge ice plus the usual shallower scouring of unstressed sea ice mixes the upper 1 to 2 m of sediments. The sediments are thus oxygenated but the life of the benthic infauna must be disrupted. Perhaps the animals burrow deeper into the bottom in the Arctic than they do in other areas.

MacCarthy (1953) studied the shoreline in the vicinity of Point Barrow and found it to be rapidly eroding as much as 3 to 5 m per year. The erosion is not attributed to the action of sea ice but to erosion by waves and currents. The ice actually prevents erosion by protecting the beach from waves and currents. Even in the summer a large portion of wave energy is spent on the scattered offshore ice floes. Ice wedges formed by permafrost action in the bluffs and beaches however promote beach erosion and make the shore much more susceptible to erosion by the sea. Continued erosion exposes more and more of the permafrost to thawing, which in turn promotes erosion. MacGinitie (1955) considered the tundra plants that were frequently dumped into the sea by this erosion to be the major detritus food web source for benthic invertebrates of the region. He says that enough plant material to supply detritus for several years will wash into the ocean in one month and will drift over a great area. An estimate of this quantity is possible from the rate of erosion and the measure of biomass of the tundra. The calculation indicates that shoreline erosion may annually add from 0.3 to 1.5 metric tons of plant material (dry) per mile of beach to the surrounding ocean. This is probably greater than the annual production of phytoplankton in the near shore waters of the Point Barrow region.

Sediment carried by ice can also affect benthic fauna and flora. Ice rafted material could cause high rates of accumulation of sediments preventing sedimentary epifauna and slow-growing plants and restricting the character of the benthic community to the most active infauna and epifauna.

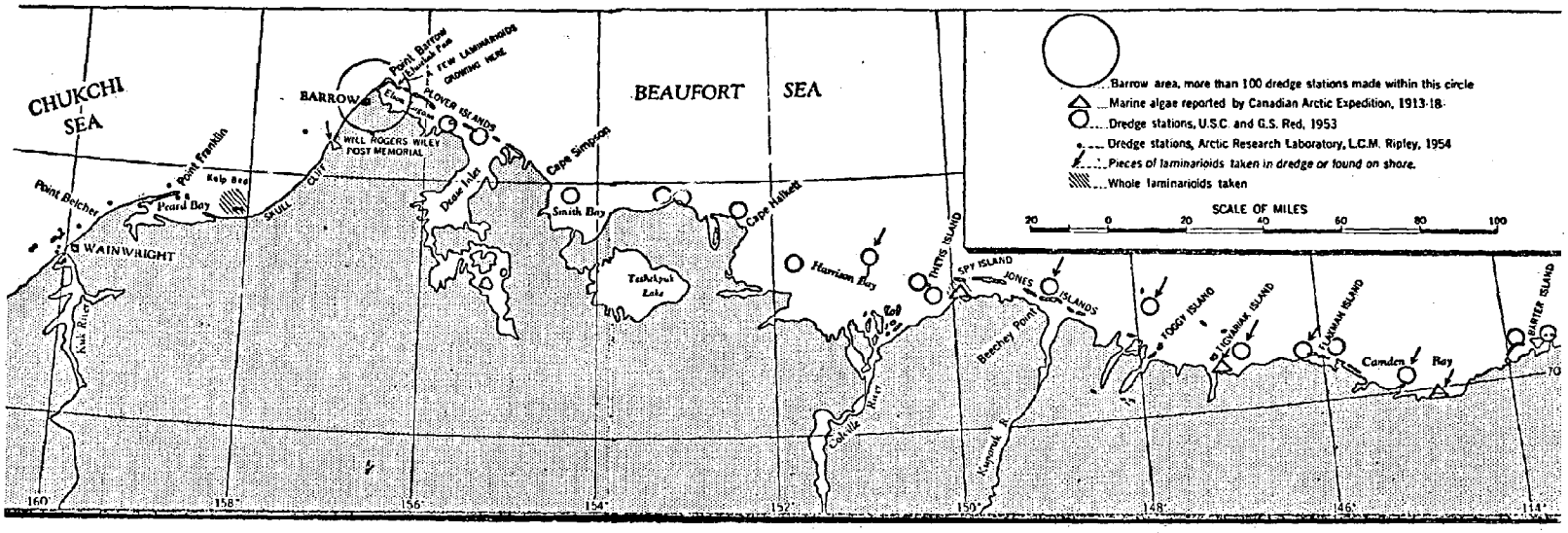


Fig. 7. Locations of biological collections along the arctic coast showing known kelp beds (From Mohr et al., 1957).

Table 5. Macrophyte algae collected from kelp bed on arctic coast of Alaska (from Mohr et al., 1957).

Brown Algae

Phyllaria dermatodea
Laminaria saccharina
Desmarestia viridis

Red Algae

Turnerella pennyi
Phyllophora interrupta
Antithamnion americanum
Phycodrys sinuosa
Polysiphonia arctica
Odonthalia dentata
Rhodomela lycopodioides f. flagellaris

The lack of a hard substrate for attachment is undoubtedly the most crucial factor restricting the distribution of macrophytes in Arctic Alaska. Macrophytes occur on the rocky Siberian coast of the Chukchi Sea (Zenkevitch, 1963) and Fairway Rock in Bering Strait has a kelp bed below about 5 m (Fig. 8). More rocks at suitable shallow depths should increase the abundance of macrophytes on the coast near Point Barrow.

Light is also an important factor for marine plants but it does not appear to determine presence or absence. Light does influence rates of production and the species composition of the flora. Even the autotrophic higher plant, eelgrass, survives the long periods of darkness under the ice in the lagoons of the Seward Peninsula. Through adaptation marine macrophytes in the Arctic have solved the problems of survival.

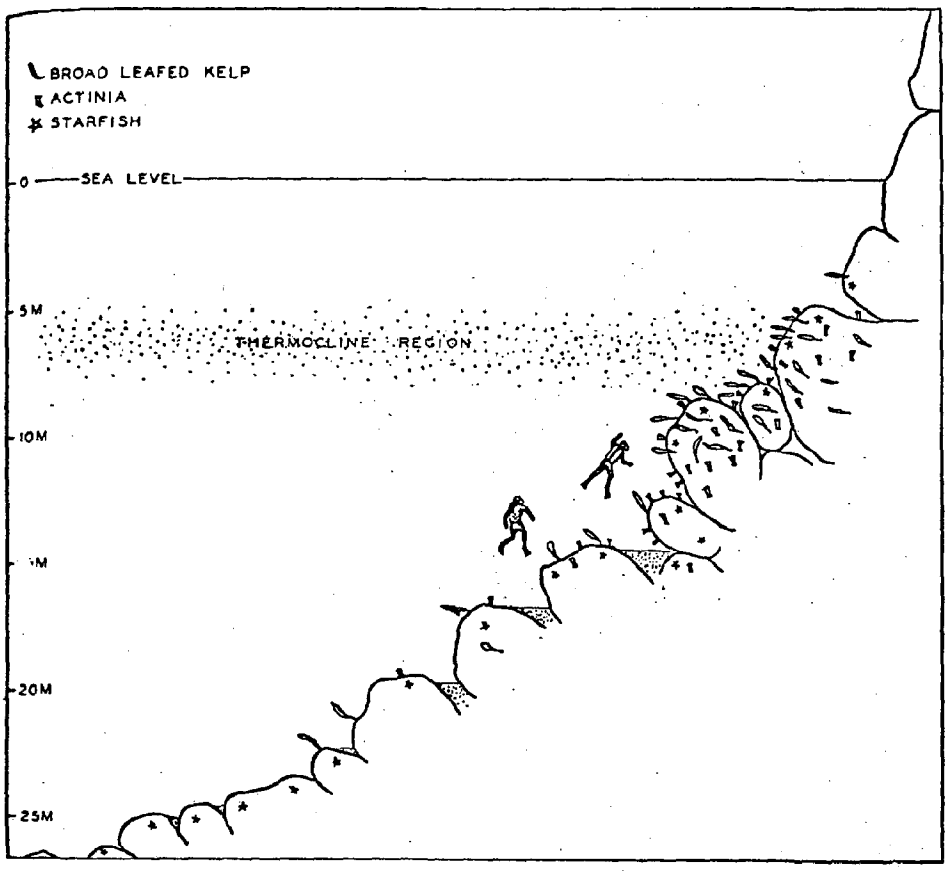


Fig. 8. Biological zonation on Fairway Rock in Bering Strait (from Shumway et al., 1964).

Chapter D- 3B

INSHORE ARCTIC ECOSYSTEMS WITH ICE STRESS

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INTRODUCTION, ECOSYSTEMS OF POINT BARROW, ALASKA

Natural arctic ecosystems with ice phenomena dominating their regimes, are found in northern Alaska, specifically characterized by the Barrow area of the Arctic coastal plain. Estuarine systems are located on both the northwestern and northeastern sides of the Barrow peninsula. None has been extensively studied at any one locality, but an attempt will be made to integrate the available data into a meaningful classification of estuarine types.

Point Barrow, the northernmost point of mainland North America, exists as a peninsula of land at the confluence of the northeasterly trending Chukchi Sea currents and the westerly trending Beaufort Sea currents (Fig. 1).

The tidal range at Point Barrow is only about 6 inches. Variation in barometric pressure and onshore winds have a greater influence than the tides on the local sea level. This influence is further modified by the presence of the floating ice pack which tends to dampen any sea level oscillations.

The Barrow quadrangle (north of 71°) contains a total land area of 1731 km^2, of which 90 km^2 represents area covered by estuaries (Brown, et al., 1968). This area of the Arctic coastal plain is underlain by the Gubik Formation of Pleistocene age. The sediments consist of ice-cemented sands, silts, and clays deposited under nearshore, estuarine, and lagoonal conditions, much like those presently existing in the area today.

The climate at Point Barrow is truly arctic. Permafrost (permanently frozen ground) extends to a depth of 1300' (Lachenbruch, 1957). The winters are long and cold with temperatures as low as -35°C (-60°F). Summers are short and cool with temperatures seldom exceeding 15°C (70°F). Precipitation is less than 25 cms (10 inches) annually. Low clouds and ground fog which limit solar radiation are characteristically found during the summer months. These are accompanied by 10 to 30 mph winds and temperatures between 5.4°C (35°F) and 23.4°C (45°F). Freeze-up usually occurs in September or early October and lasts until mid-June. Vegetation is characteristic of cold northern tundras, consisting of various grasses, mosses, and dwarf willows.

Shoreline Processes

Studies indicate there is a great deal of coastal erosion and the area may be considered to be unstable. The factors responsible are climatic and depend generally upon the temperature and the winds which blow almost constantly NE-SW. The particular form of erosion of importance to estuarine studies

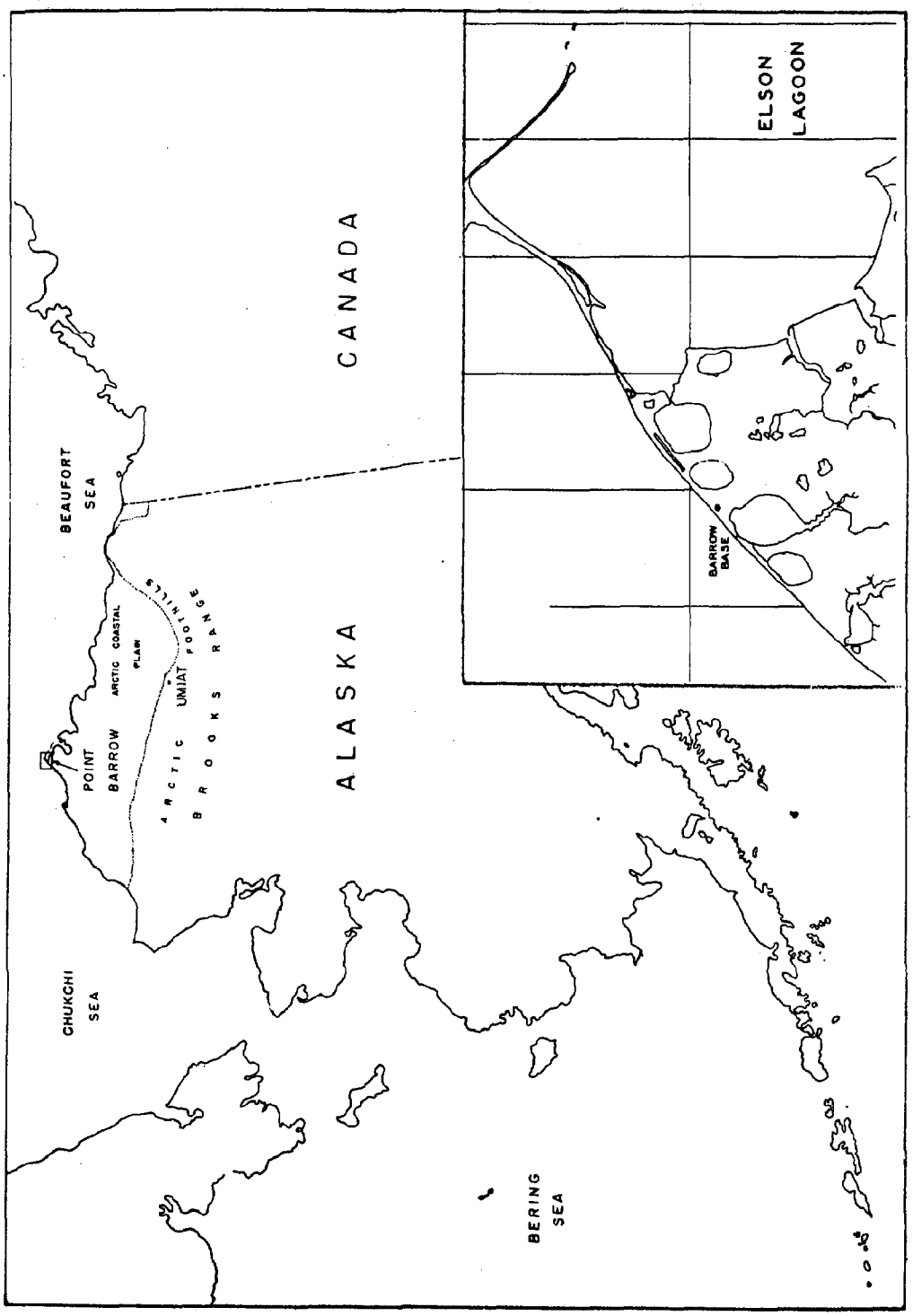


Fig. 1. Map of Alaska showing the Arctic coastal plain and Point Barrow.

is known as "thermal niching" (Walker and Arnborg, 1963), a uniquely arctic phenomenon consisting of the thermal undercutting of ice-cemented sediments by waves or stream water. The frozen soil melts on contact with water and releases the sediment particles, which are then redistributed by the action of the currents.

A second phenomenon, saturated soil flow, occurs during the summer when the active soil layer is melted naturally and the water is unable to drain downward through the permafrost. The water laden sediment flows down to a lower level, often onto the beach, where it becomes available for redistribution within the estuary.

Numerous workers have observed the effects of erosion on the higher areas exposed to the sea. (Black and Barksdale, 1949; MacCarthy, 1953; Rex, 1955, 1964; Rex and Taylor, 1952-53; Carson and Hussey, 1960; Lewellen, 1965; Hume, 1965; Hume and Schalk, 1964, 1967).

Estimates of erosion rates along the southern shore of Elson Lagoon were reported by Lewellen (1965), based on the analysis of aerial photographs taken in 1949 and again in 1964 by USA CRREL (Fig. 2). He found removal rates varying between 0.2 meters and 2.0 meters per year at various places along the Elson Lagoon shoreline, from the Barrow spit to Dease Inlet. This is attributed to the effect of wind-driven waves which undercut the banks by thermal niching and remove the slumped material deposited on the strand line by saturated soil flow.

On the extreme end of the Barrow peninsula, Hume and Schalk (1967) report shoreline changes occurring since 1948 at the tip of Eluitkak Spit and in an area southwest of Point Barrow. The latter area grew about 250 feet between 1948 and 1960 from swash-line deposition and the northeasterly flowing longshore currents (Fig. 3). They estimate the total net yearly transport to be about 10,000 cubic yards of beach material. The net yearly growth of Eluitkak Spit is estimated to be 4,500 cubic meters.

Rex (1964) found that the coarse sands and gravels of the spit extending southeastward from Point Barrow directly overlie the gray muds of Elson Lagoon with no transition zone between sediment types. This would indicate that the spit has been migrating over the lagoonal sediments in response to changes in sediment supply, currents, and sea levels, although Lachenbruch considers the Barrow area to have been relatively stable for the last few thousand years.

Brown and Sellman (1966) show that the Barrow Spit may not have been in its present location earlier than 700 years B.C. Radiocarbon dating of freshwater peat exposed in a breached area of the spit suggest the existence there of a shallow freshwater lake from 2600 to 700 years B.C. Retreat of the shoreline destroyed the lake, probably forming an estuary for a short duration, and finally covered the shallow basin. These stages of landscape development can all be seen along the northwestern shore of the Barrow peninsula today.

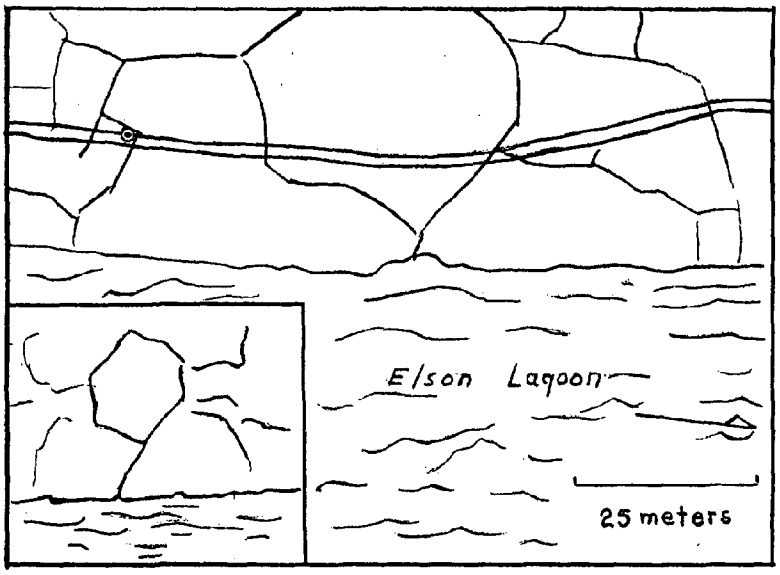


Fig. 2. Line drawing of aerial oblique photograph of southern shore of Elson Lagoon Taken in 1964. Inset shows same view taken in 1949. (U. S. Army Terrestrial Sciences Center)

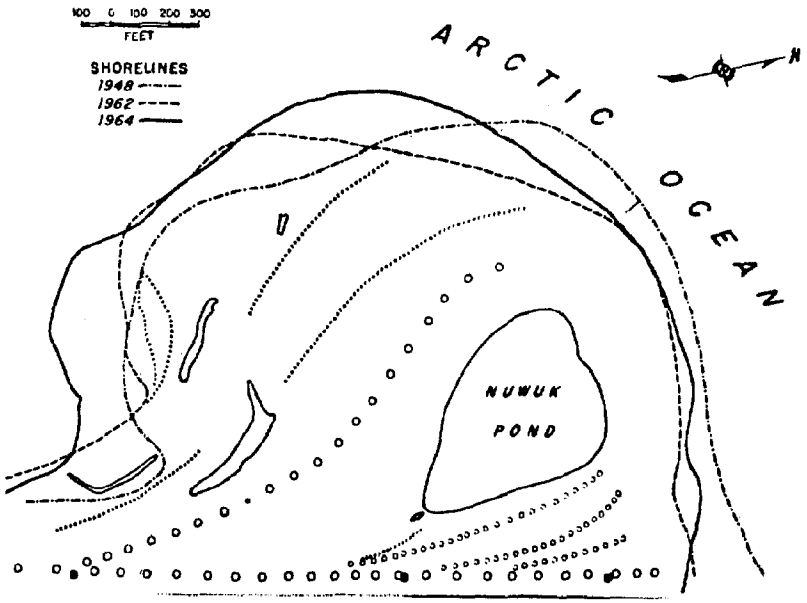


Fig. 3. Map of western part of Point Barrow, Alaska. Shorelines for 1948, 1962, and 1964 are shown (after Hume and Schalk, 1967).

The effects of the October 1963 storm, the worst in 200 years to hit the Barrow area, were thoroughly investigated by Hume and Schalk (1967). In addition to causing major property damage, the high waters, estimated to be about 10 feet higher than normal, flooded the low-lying areas, including the entire Barrow Spit, the estuaries and freshwater lakes.

The most obvious change which resulted from the storm is that Point Barrow temporarily became an island. Three breaks occurred in the spit, allowing the Arctic Ocean to flow directly into Elson Lagoon. Resurvey of the area southwest of Point Barrow showed that approximately 200,000 cubic yards of material had been added, the equivalent of twenty years of normal transport. Eluitkak Spit was extended southeastward by 45 feet, 22,000 cubic yards of material having been added to its tip , while elevations on the spit were increased from 4 to 7 feet above sea level.

Changes also occurred in the beaches. Northeast of the break in the Barrow Spit, the beaches retreated about 30 feet; while to the south of Barrow Village, the sea cliffs, composed of ice-cemented sands and silts of the Gubik Formation, retreated about ten feet. Records of similar storms occurring in the past may be preserved in the lagoonal sediments (Faas, 1966).

Hume (pers. comm.) indicates that the point will soon become complete again, the gaps requiring only about 50 more feet of growth to close. It is expected that the point will again be rebuilt by spring 1969. However, by the end of the summer 1969 field season it still had not completely closed.

Circulation Patterns

Horizontal circulation patterns are quite pronounced in estuaries on both sides of the Barrow peninsula. In each case, during the spring thaw, all tundra pools and estuaries are filled to overflowing. Runoff is particularly effective at this time as percolation into the tundra cannot occur due to the impermeable nature of the then frozen soil and permafrost.

Arnborg, Walker, and Peippo (1967), in their work on the Colville River delta, showed that the river, although flowing over 4 months of the year, did the greatest amount of sediment transportation during the first few weeks of the summer season. In 1962, 43% of the total annual discharge (16×10^9 m^3) and 73% of the total suspended inorganic load (5.8×10^6 tons) were transported to sea in the 3 week period immediately before, during, and after the spring breakup. Breaching of the gravel bars across the estuary mouths on the NW side of the peninsula and flushing out of the hypersaline waters may take place during this period. Hypersalinity occurs during the winter ice-covered period as a result of expulsion of salt at the freezing front of the ice, enriching the bottom unfrozen waters with salt as the ice cover thickens. Generally, uninterrupted interchange of waters between the ocean and the fresh tundra melt waters continues for a period of a week or more. Finally, northeastward longshore drift of sediment closes the bar across the mouth of the estuary, the water becomes brackish, and the normal cycle continues. Periodic storms may top the bar and add saline water to the estuary and, in some cases, may totally disrupt the continuity of the bar for short periods during the summer. When the ice freezes again in the fall, all circulation ceases and winter stagnation begins. Water bodies less than 6 ft. deep in the Barrow area freeze completely to the bottom during the normal winter. This includes a great many of the tundra lakes and those estuaries created from the truncation of lake basins by an eroding shoreline.

On the northeastern side of the peninsula, runoff conditions are much the same. All small estuaries channel drainage from the tundra into Elson Lagoon. In addition, the Mead, Inaru, Topagoruk and the Chip rivers flow into Admiralty Bay and out into Dease Inlet and Elson Lagoon. East of Elson Lagoon is Smith Bay, another large estuary into which flows the Ikpikpuk River. MacGinitie (1955) indicates that this large outflow of fresh water, flowing through Elson Lagoon exercises a greater effect on the ocean water near Barrow than any other freshwater source. The tundra runoff is only about 50% of the measured annual precipitation (Brown, et al., 1968). In addition to increasing sedimentation rates by transporting large quantities of mud, this outflow of fresh water carries large amounts of fresh vegetation and peats from the eroded shore of the lagoon and river watersheds, and deposits this material on the ocean bottom for a distance of 25 miles offshore.

Vertical circulation patterns are quite uncomplicated if, indeed, they exist at all. Due to the shallowness of Elson Lagoon (3.5 m deep), winds are effective at mixing and, for the most part, the waters are isohaline and isothermal throughout. Holmquist (1963) identified the existence of a two-layer system in Elson Lagoon when measurements on July 21, 1961, showed a temperature of 4.1°C and salinity of 23.2‰ at the surface and 3.6°C and 29.6‰ at 3.0 meters depth, indicating an inflow of fresh water. This was of short duration and on July 26, 1961, a windy day, 6°C and 29‰ were found at all depths.

Productivity

Soil productivity at Barrow is limited by low phosphate and nitrate contents. Nutrient recycling was thought related to periods of lemming highs, during which phosphates and nitrates would have been concentrated through greatly increased animal feces. Livingstone (1963a) comments on the more luxuriant vegetation around snowy owl perches and human settlements where such accumulation normally occur. Pieper (1963) indicates that increased phosphorus, nitrogen, and potassium were found in the tundra vegetation in the early part of the lemming high of 1960. Schulz (1962) believed that depth of thaw of the active layer and release of frozen nutrient material may be accelerated during a lemming high due to removal of insulating plant material through increased grazing. This same material may become incorporated into the estuarine sediments during spring runoff, providing a cyclic increase in food supply for detritus feeders. A suggestion of such organic matter accumulation is seen by the alternating light and dark laminae in cores taken from the estuarine sediments (Fig. 4).

While not strictly estuarine, a study of primary production in tundra ponds in the Barrow area by Kalff (1965) may be useful, inasmuch as the smaller estuaries, when sealed off at their mouths, behave in a similar fashion as the ponds - as catchment basins for tundra runoff. Kalff observed great variation in the timing and size of phytoplankton blooms in the adjacent ponds within a few hundreds of meters of each other. It was obvious that the ponds, and presumably the estuaries, were deficient in nutrients and sometimes in other growth factors. Phosphates appeared to be most deficient, followed by nitrates. However, he also showed that light was of greater importance in determining algal production rates than increased phosphorus in the ponds during a lemming cycle. The amount of carbon fixed by the phytoplankton was dependent on light intensities, with diurnal differences affecting the rate of photosynthesis. Maximum photosynthesis occurred between 0.09 and 0.17 langleys per minute during the diurnal period of maximum intensity. Values greater than these caused inhibition of photosynthesis, and at lower values total carbon fixation decreased. He considered the ponds to be nutrient traps, contributing little to the productivity of the tundra. Obviously, much more work needs to be done concerning the relationship between the organisms of the tundra and nutrient transfer in and through the estuaries.

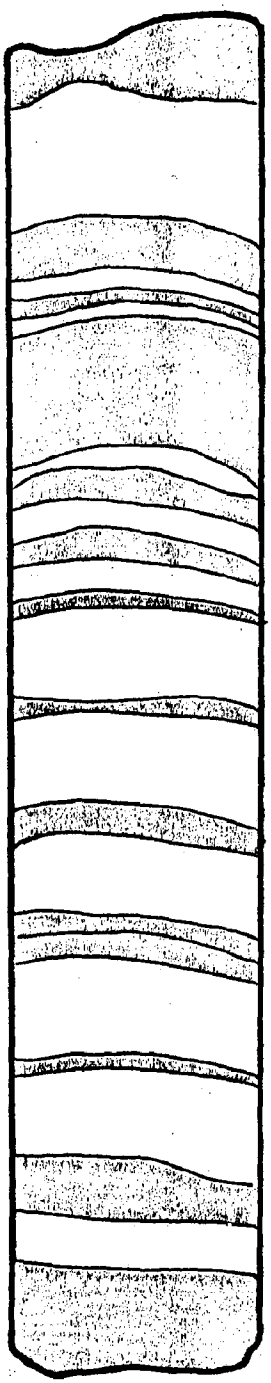


Fig. 4. Line drawing of a core taken from sediments underlying Esatkuat Lagoon, showing distinct alternations of organic matter (from photo taken by author, 1962).

Tundra ponds and lakes may be more productive than the estuaries, particularly those which are self-flushing. Wohlschlag (1953) indicates that fish taken from Elson Lagoon are thinner and more undernourished than most taken from the inland lakes. Very little material seems to be available concerning northern Alaska estuarine fish populations

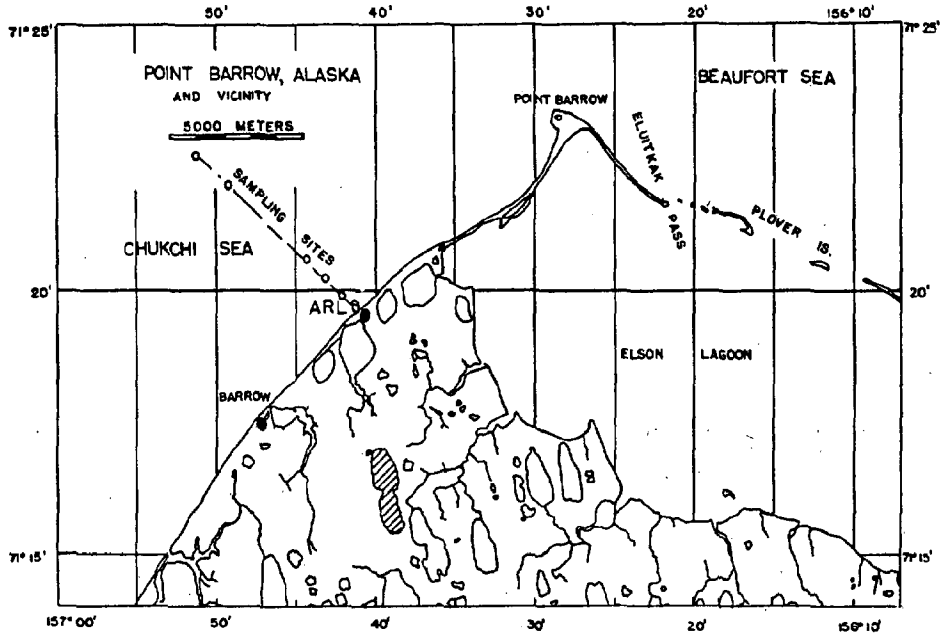
Stress Factors

Certainly the most significant stress factor that has emerged from all the preceding statements has been the effect of the low temperatures of the arctic environment. All other factors adding to the degree of stress in any given environment may be directly or indirectly related to the cold. The increased salinity found in the unfrozen bottom waters of estuaries and lagoons is a result of both downward brine migration and salt expulsion ahead of the downward moving ice front during the process of ice formation. (Adams, et al., 1960). The drastic salinity shocks, from hypersaline to fresh to brackish, results from the influx of fresh meltwater from the tundra due to release of previously frozen water contained in the snow cover. Oxygen deficiencies found in some isolated water bodies develop during the period of ice and snow cover when photosynthesis of algae ceases. The paucity of littoral forms of life is due to the grinding and crushing action of the sea ice on the bottoms during the long cold winters. The littoral fauna of lakes, lagoons, and estuaries is similarly endangered when, during the spring thaw, the central ice mass covering the water body lifts from where it has been frozen fast and bounces from shore to shore, depending upon the prevailing winds.

As might be expected in high stress situations, animal diversity is not large, nor are populations overly abundant. For those organisms which do exist under these conditions, adaptation to the rigorous environment has surely taken place. Holmquist (1963) cites the amphipods, mysids and other organisms of estuarine systems which do not hibernate as resting stages but apparently prefer to remain in the hypersaline unfrozen bottom waters. Their mechanisms of survival require further study. Mysis relicta has been found in waters varying in salinity from 29 mg $Cl^-/l.$ to 37,600 mg $Cl^-/l.$ These euryhaline organisms must be able to adjust their osmoregulation when migrating through these different layers. Mysis can also live in very low temperatures (-2^oC) as well as a constant 17 to 18^oC (Holmquist, 1959). It has been suggested that the metabolic rates of the mysids are very low during the winter and the maturation of the animals is retarded. Young mysids from Elson Lagoon do not escape from the brood pouch until late July. In other environments they were found in continuous propagation.

Mohr and Tibbs (1963), in discussing the amphipods which live directly below the ice in hypersaline conditions, note that during the spring runoff and period of under-ice incursion of freshwater, the amphipods do not appear to avoid this water. Rather, they remain in it until it is incorporated into the normal sea water, usually in a few days. Such drastic salinity shocks would seem to require that the animal make rapid osmotic adjustments. Boyd and Boyd (1963) measured very low salinities under the offshore Chukchi Sea ice (Fig. 5) in early spring of 1957 (5.96 ppt). This increased rapidly to 26.20 ppt a week later. Waters deeper than 4 meters showed variation of only 2 to 3 ppt (Table 1).

As usual, one can observe how selection by a stressful environment has served to develop a unique relationship between ecologic niches and groups of organisms. Bursa (1963) attempted to define six specifically arctic ecologic phytoplankton niches, all of which depended to a greater or lesser extent upon the presence of arctic ice. They were:



Sketch map of the Point Barrow area showing the various sampling stations.

Fig. 5. Bacterial sampling in northern Alaska (Boyd and Boyd, 1963)

Table 1. *Bacteriology and salinity of the ice-covered Chukchi Sea 0.8 km off Point Barrow*

Date (1957)	Depth (cm)		Temperature, C		Sampling depth (m)[2]	Bacteria per ml[3]		Salinity (‰)
	Ice	Snow	Air	Water[1]		Fresh	Saline[4]	
7 March	115	40–60	–20	–1.5	6 (bottom)	2	1	–
15 March	125	40	–27	–1.5	6	10	3	–
22 March	175	35–55	–29	–1.6	6	11	15	28.43
29 March	135	55	–22	–1.6	6	11	12	28.38
12 April	125	50–55	+ 2	–1.4	6	2	8	29.35
19 April	115	50–60	– 8	–1.6	6	4	11	29.73
26 April	135	55	–12	–1.4	6	11	10	29.55
18 May	115	50	– 2	–1.3	6	10	35	30.05
24 May	130	40–45	– 0.6	–1.4	6	18	8	29.40
31 May	120	0–30	+ 4	–1.2	6	69	77	29.35
					5	91	88	–
					4	290	260	–
					3	500	520	29.20
					2	460	470	–
					under ice	–	–	29.20
7 June	100	15–20	+ 3	+0.5	6	370	240	29.39
					5	500	410	–
	(15–20-cm snow in spots; otherwise, 5–10-cm snow meltwater covered ice)				4	590	510	29.40
					3	460	450	–
					2	36	28	–
					under ice	–	–	5.96
14 June	130	0	+ 3	–0.2	6	500	470	28.31
					5	550	520	–
	(approx. 10-cm water on top of ice; difficult to measure ice thickness)				4	620	590	–
					3	2,000	2,000	27.00
					2	1,600	1,300	–
					under ice	–	–	26.20

[1] Water temperature measured at the surface.
[2] Depth measured from top of ice.
[3] Bacteria cultured on ZoBell's agar medium.
[4] Aged seawater (28.40‰ salinity) used to make saline medium.

(Boyd and Boyd, 1963)

1. The phytoplankton cycle below the ice.
2. The microflora of the melt water pools on the ice.
3. The phytoplankton cycle in coastal and offshore leads.
4. The phytoplankton cycle in lagoons and lakes along the shore.
5. The phytoplankton cycle of the inshore waters.
6. The benthic microflora of sandy and muddy sites.

The arctic ice acts as a stress factor for 9 to 11 months due to its inhibition of photosynthetic activities by these organisms. However, when this stress is finally withdrawn after spring breakup, the ice masses form new and uniquely arctic ecologic niches.

EXAMPLES OF ICE STRESSED COASTAL SYSTEMS

Two water bodies, Esatkuat Lagoon and Elson Lagoon, represent the coastal systems in the Barrow area. They are both unique arctic types of ice-stressed ecosystems, yet differ enough from each other to warrant independent classification. In addition to the obvious size differences existing between the two, each undergoes different stress conditions. In some cases these differ only in intensity of stress, in other cases they differ in total stress environment.

I have decided to call Esatkuat Lagoon an arctic "multiple-stressed" system in view of its being affected by the normal environmental stresses and, in addition, from stresses imposed by man. Elson Lagoon, a barrier island-lagoon complex, is an example of an "ice-stressed-intertidal" ecosystem. This classification must be explained inasmuch as the low astronomical tidal range in the Barrow area makes the term "intertidal" slightly incorrect. However, the effect of winds on these shallow basins is sufficient to cause singular water level changes. Rex (1964) indicates changes up to 4 feet in Elson Lagoon are associated with longshore currents impinging on shore-fast ice, generating local pressure gradients. Hume (pers. comm.) reports changes of 3 feet in the lagoon at Pt. Lay to the southwest related to a strong offshore wind, and Brown (pers. comm.) has observed the Elson Lagoon bottom exposed a distance of 30 feet into the lagoon as a result of a similar wind. In all cases, intertidal areas were exposed to drastic environmental stresses, but not at regular, predictable intervals.

Esatkuat Lagoon - A Multiple Stressed Estuary

The water body immediately adjacent to the village of Barrow has been studied for a number of years in connection with a program to provide a continuous supply of fresh water to the facilities of the Bureau of Indian Affairs and the U. S. Public Health Hospital. It is a body of water covering 937,750 m^2 (1,550 m by 605 m), composed of three separate basins; a central oblong trough with a maximum depth of 10 feet in the center; a smaller western basin which drops quite steeply to a depth of 8 feet; and an eastern basin, possessing a sinuous channel with a nearly constant depth of 11 feet extending back some distance into the tundra (Fig. 6).

An investigation in Esatkuat Lagoon in 1962 (Faas, 1964, 1966) was concerned with a stratigraphic study of the Pleistocene sediments underlying the

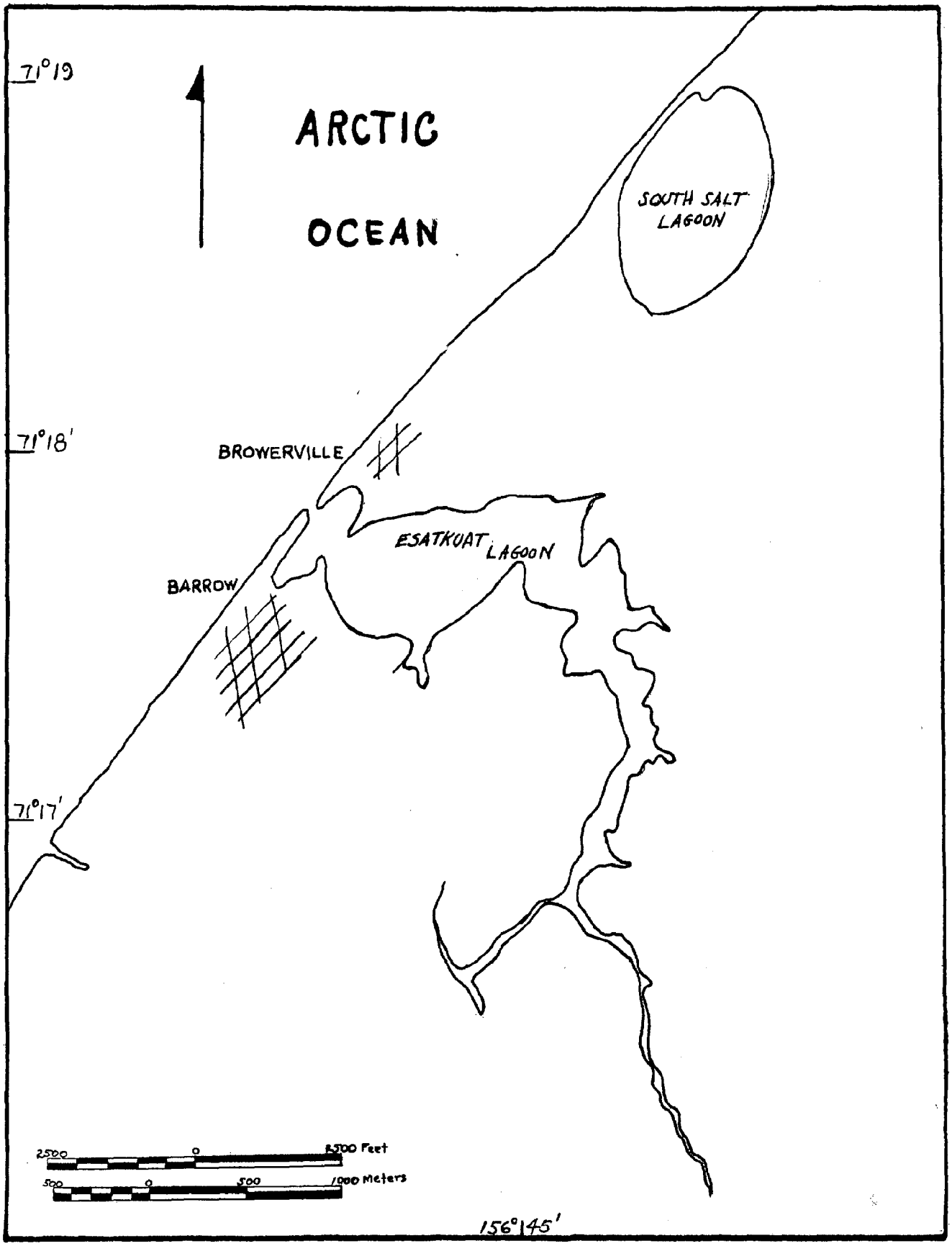


Fig. 6. Map of Esatkuat Lagoon. Note the three part subdivision, i.e., the small westernmost basin, the elongate central basin, and the easternmost basin.

modern lagoonal sediments. The most readily recognizable sediments of the modern lagoon bottom are black, highly organic silts and clays. A few lenticular units of gravel exist near the coastal side of the lagoon and apparently represent washovers of the barrier bar during severe storms or slight adjustments of sea level.

The sediments exhibit higher organic matter contents than usually found in near-shore sediments. Samples from the black clay units in Esatkuat Lagoon average 8.6% organic matter, ranging from 7.3% to 9.5%. This may be attributed to deposition within a small restricted basin and incorporation into the sediments prior to a major breakthrough of the estuary-mouth bar. Normal runoff from the tundra, bringing particulate organic matter and humic acids, plus continual erosion of the lagoon banks, is the source of the organic material (Fig 7).

Stress factors affecting this estuary are 1) oxygen deficiency, and 2) violent salinity fluctuations. These factors are the direct result of, and are dependent on, the existence of the thick ice covering (often in excess of 6 feet) and the normal desalting process which occurs over the winter period. Since smaller water bodies in the Barrow area are characteristically undersaturated with respect to dissolved oxygen due to high organic matter contents (Kälff, pers. comm., 1965), oxygen deficiency is believed to occur during the winter months when ice prevents exchange between the water and the atmosphere. This is assisted by the normal metabolic activities of the estuarine organisms, and possible bacterial decomposition of the organic matter in the bottom muds which occurs after the ice-cover is complete (Brewer, 1958). The author has observed the bottom sediments to be extremely black, sticky, and foul-smelling. Foraminifera taken from these sediments are characteristically pitted with many missing chambers. Acidic muds resulting from anaerobic conditions would most probably cause such solution features.

The modern lagoonal sediments contain a shallow water foraminiferal fauna composed of eurythermal and euryhaline species. Elphidium incertum (Williamson), E. orbiculare (H. B. Brady), E. bartletti (Cushman) and Nonionella auricula (Heron-Allen and Earland) are the dominant species. Representatives of these genera have been found in marginal environments worldwide. A noticeable number of aberrant forms were found in these bottom muds. In many cases the terminal chamber was observed to be offset from the normal symmetry, leading to a grotesque bulge on the organism test.

Complete and disarticulated ostracod carapaces also occur. Many of these are heavy and coarse-shelled and belong to the genus Normanocythere. The remaining genera exhibit varied morphologies and were not identified. A few scolecodonts and numerous chitinous insect parts, i.e., heads and mandibles, were also found. Heavy concentrations of plant organic matter, i.e., large woody chunks, stems, and fibers, are also locally concentrated on the modern lagoon bottom.

Foraminiferal distribution is unpredictable, being locally concentrated in more favorable environments. This "patchiness" in animal distribution is not unique to the arctic and has been commented on by Mohr (1959) as probably a response to localized food concentrations.

Elson Lagoon - An Ice-Stressed, Non-tidal Barrier bar-lagoon Complex

Elson Lagoon is a shallow water body, 55 km long and 10 km wide in its

western part, located directly southeast of Point Barrow Figure 1). It is bounded on the northwest by the peninsula itself, on the northeast by the Plover and Tapkaluk Islands. To the southeast Elson Lagoon turns sharply southwest and becomes Dease Inlet ending in Admiralty Bay, a significant estuary extending far inland. Some wide openings connect Elson Lagoon with the Arctic Ocean, chiefly Eluitkak Pass at the northwestern end, through which a strong current travels, sometimes exceeding four knots (Rex, 1955). However, most currents in Elson Lagoon are weak and wind driven, generally to the west. Some reversing tidal currents through Eluitkak Pass were observed by Rex (1964) where inflow was occurring in the face of an opposing wind of 15 knots. Other openings exist between Deadman's Island and the Tapkaluk Islands. Most of these openings are quite shallow. However, Eluitkak Pass is 13 meters deep. The maximum depth of Elson Lagoon does not exceed 3.5 meters.

Oxygen content seems to be no problem in the larger water bodies. Elson Lagoon is well-aerated due to its shallowness. Bursa (1963) indicates the high oxygen content could also be produced photosynthetically by the autotrophic plankton, benthic algae, and microflora that blanket the bottom of the lagoon.

Brown, et al., (1968) have recently completed a study of hydrologic factors affecting a 1.6 km^2 drainage basin at Barrow. The data were obtained from 1963 through 1966 and included a great range of summer climate, from cold/extremely wet, to cold/extremely dry. Water gauging stations monitored the runoff into Elson Lagoon through Central Marsh Slough, a small drowned river valley which is presently an estuary. Several periods of peak runoff occurred throughout the four year study period and it is seen (Fig. 7) that specific conductance varies inversely with discharge. Minimum conductance values for the maximum runoff ranged between 230 to 290 millimhos. Peak conductance values are usually associated with low runoff as seepage through the marshy tundra.

No analyses are available for total ion content of Elson Lagoon waters. However, Brown also determined Na^+, K^+, Ca^{++}, and Mg^{++} for the runoff into Central Marsh Slough for 1964 and 1965 (Table 2). Greatest variation of Na^+ (0.77 meq/l to 2.9 meq/l) was seen in 1965. In both cases a general increase in total ion content is shown over a summer, associated with a decrease in volume of discharge through the watershed.

Very little data are available concerning the chemistry of the Elson Lagoon waters. As stated earlier, Holmquist found a maximum salinity of 29.9% in late July, and indicates that salinity conditions are probably variable in the spring, becoming more stable in the summer, and reaching maximum stability under winter conditions. Temperatures may rise as high as 6°C in summer and be below freezing in the winter.

Several studies concerning the phytoplankton and the brine shrimp (Mysis relicta) of the Barrow area have been completed recently (Holmquist, 1963; Bursa, 1963). Much of the work done included observations on the biota of Elson Lagoon.

In general, the fauna of Elson Lagoon is quite marine. Holmquist lists hydroids, a sea anemone, bryozoans, the polychaetes Pectinaria and Lepidonotus, priapulid worms Halicryptus spinulosus and Priapulus caudatus, bivalves, snails and amphipods Gammarus and Gammaracanthus, the isopod Mesidotea entomen, decapod

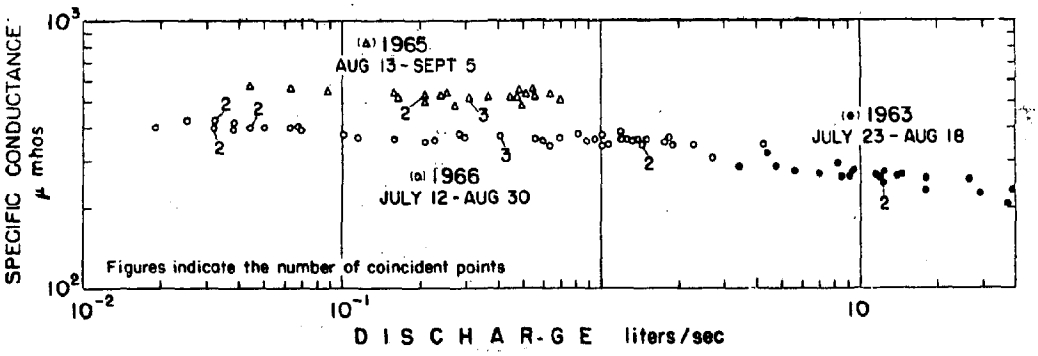


Fig. 7. Composite plot of Specific Conductance versus Discharge from Central Marsh Slough for 1963, 1965, and 1966 (After Brown et al, 1968).

Table 2. **Cation composition of runoff water (meq/liter), Barrow watershed.***

Sample date	*Na+*	*K+*	*Ca++*	*Mg++*	*Sum*	*Cs (μmhos)*
18 Jun 64	2.1	0.03	0.11	0.72	2.96	357
14 Jul 64	2.7	0.01	0.33	1.1	4.14	480
3 Aug 64	3.3	0.01	0.43	1.4	5.14	570
7 Aug 64	3.3	0.01	0.37	1.4	5.08	565
14 Aug 64	3.3	0.01	0.41	0.91	4.63	630
27 Jun 65	0.77	0.08	0.06	0.19	1.10	135
2 Jul 65	1.1	0.05	0.12	0.34	1.61	198
14 Jul 65	1.7	0.01	0.17	0.68	2.56	315
27 Aug 65	2.9	0.01	0.28	1.0	4.19	500
2 Sep 65	2.1	<0.01	0.21	0.77	3.08	362

* Analyses by Walter Grube and William Webster.

(from Brown, et al., 1968)

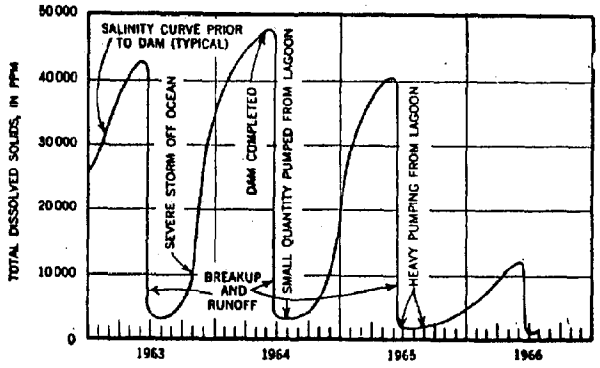


Fig.8 . Plot of yearly variation of total dissolved solids for Nerravak (Esatkuat) Lagoon, 1963 to 1966. After Buchanan, et al., 1966.

larvae, a few euphausiids (Thysonossa raschi), some medusae and the chaetognath Sagitta. Three mysid shrimp, Mysis relicta, M. litoralis and M. oculata were found, the former two species most abundant. Sampling offshore, Menzies and Mohr (1962) found Mysis relicta restricted to the coastal areas with variable salinity. They also include the isopod Idotoega sabini sabini forma barentsi Gurjanova in their collection from Elson Lagoon. Bursa (1963) found the benthic algae, Ulva sp., Enteromorpha sp., Cladophora sp., Sphacelaria sp., and Lithoderma sp., attached to the bottom gravels. Strictly estuarine dinoflagellates were Goniaulax catenata, Chaetoceros wighami and Ch. sociales. A phytoplankton collection from the lagoon in mid-August showed 8,930 diatoms, 1,280 dinoflagellates, 10,900 unidentified flagellates, dominated by Polytomella, and 400 ciliates per liter. Two weeks later Polytomella showed an increase to 82,400 cells per liter, 5,380 diatoms, 1,340 dinoflagellates, and 160 ciliates per liter. It was believed that the maximum had occurred earlier in July and these figures reflected a decrease in numbers.

Lieski (1964) investigated the role of solar radiation in causing the breakup of the ice in Elson Lagoon during the winter of 1962-63. Warming and deterioration of shore-fast sea ice commenced in late March by thermal conduction from the overlying snow. Although the greater warming period occurs from May 1 to 25th, ambient air temperatures averaged -4°C, and midnight temperatures of the snow at the 10 cm level averaged -3°C, demonstrating that the snow warms the ice by conduction and is itself warmed by radiation, not by turbulent exchange with the air.

The conduction rate and sign are controlled chiefly by the net total radiation exchange at the snow surface, and internal absorption of short wave radiant energy by the ice is negligible until the depth of the snow is less than 25 cms.

Measurements of total radiation for the months of January to June, 1963 gave the following: -3,250, -1,600, -2,550, -1,200, 1,250 and 4,200 calories/cm^2. By June 1, the entire ice mass was near its freezing point, making the total amount of energy (4200 calories/cm^2) available for phase transformation.

Holmquist (1963) indicated that the bottom sediments of Elson Lagoon are composed of a yellowish-brown to gray clay or mud in the center, becoming blacker closer to shore, to be finally replaced by coarser material at the shoreline. This distribution is normal and expectable for such low energy, enclosed basins.

No work on foraminiferal distribution is yet available for Elson Lagoon.

MAN AND THE ARCTIC ESTUARIES

The severity of the arctic environment poses hardships for all inhabitants

of the tundra, including man himself. The basic problems are two-fold, 1) acquisition of sufficient water to supply a growing population, and 2) disposing of the waste products of that population.

Several publications (Boyd and Boyd, 1959, 1965; Boyd, 1959) have summarized these problems in general for the Arctic. However, these particular problems are being solved at the village of Barrow in a unique fashion. The lagoon lying just northeast of the village is being used both as a water supply and as a sewage outfall. This appears to be a strange combination but has proved effective.

Initially, with the construction of the new school of the Bureau of Indian Affairs and the U. S. Public Health hospital, it became obvious that the traditional method of obtaining water, by cutting ice and hauling it to the village over the tundra, was no longer feasible. Several alternatives were considered but rejected in favor of developing Esatkuat Lagoon as a water supply. The central part of the lagoon is about 0.7 miles long and 0.3 miles wide. It is approximately 10 feet deep at its deepest part giving a total water capacity of approximately 280,000,000 gallons. However, during the winter the ice freezes to a thickness of 6 feet, leaving only 30,000,000 gallons available for use. A dam across the lower end of the central lagoon would allow the water level to be raised three feet, increasing the amount of usable water to 78,000,000 gallons. This quantity is suitable for projected use for many years. The construction of the dam across Nerravak (Esatkuat) Lagoon presented a number of unique problems (Buchanan, Dudley, and Peoples, 1966). Use of available construction materials required borrowing gravel from the beach and using as "riprap" many 55 gallon fuel oil drums, filled partially with concrete and covered with steel landing mats. As expected, freshening of the normally brackish lagoonal waters began almost immediately, hastened by a 4500 gpm pump which pumped the denser saline lower water layer from under the ice and out of the lagoon (Fig. 8). The dam has withstood the greatest storm in 200 years and is operating efficiently today.

Waste disposal in the Arctic is also a difficult problem. Boyd and Boyd (1965) indicate that the town of Inuvik in the Canadian Arctic, a settlement of 2,000 people dumps its raw sewage directly into a shallow freshwater (oxbow) cutoff which opens into the Mackenzie River. Oxidation occurs during the summer. After the first freeze, an opening from the cutoff into the river is effected which dilutes and disposes of sewage for the year.

At Barrow, a complete sewage disposal plant has been designed which grinds all solid material into a slurry that is treated and aerated. The sewage flows to a final tank, is diluted further, and disposed of in the lower basin of Esatkuat Lagoon. It is well-suited to receive wastes and is separated from the water intake by the ice-cored dam. Periodic storms and annual tundra runoff flush the lower lagoon clean, probably also assisted by percolation through permeable beach gravels.

We see a multiple use of a small estuary which has made possible both a water supply and a sewage disposal area for the village of Barrow, Alaska. Without a doubt, similar usage could be made of other small estuaries, particularly if they have higher tidal ranges and are self-flushing.

A note of caution must be sounded at this point. Measurements made recently (Hume, pers. comm.) indicate the Barrow beaches have retreated over 30 feet since 1964. Construction of a new airport facility in Barrow has made extensive use of gravels from beaches lying southwest of the village. Since the longshore current flow northeasterly, sediment will be carried away from the beach at Barrow and nothing will be available to replace the lost material. Further erosion of the area between Barrow and Browerville will affect the sewage outfall at the lower end of Esatkuat Lagoon. Removal of the gravel bar will create greater fetch and a sufficiently large storm could seriously damage the Bureau of Indian Affairs dam which impounds the remainder of Esatkuat Lagoon.

Outlook

At the present time, the influence of man on most of the estuaries in the Barrow area is relatively small, with the exception of Esatkuat Lagoon. Certainly Elson Lagoon will retain its identity as an "ice-stressed, inter-tidal" estuarine system for a long time. However, as is seen by the example of Esatkuat Lagoon, an entirely new system can be created by the input of relatively minor amounts of energy to a "multiple-stressed" arctic estuary, already existing as a small, isolated unstable system.

The recent oil strike in northern Alaska is quite likely to produce new emerging estuarine systems which will be imposed on natural arctic systems. An arch or flexure, which includes the Prudhoe Bay anticline, and which is characteristic of many oil producing basins, extends westward through the Beaufort Sea and beyond Point Barrow. It can be anticipated that with further exploration and production from these coastal areas, emerging systems associated with petroleum shores, sewage, and possibly petro-chemicals, may develop within the arctic bays and estuaries. Microbial activity, being reduced under arctic temperatures, cannot be expected to effectively decompose hydrocarbon pollutants. One might predict that research under arctic conditions and understanding of the stressed arctic ecosystem will be slow in coming due to the inhospitable nature of the environment, and that remedial measures will again be too little and too late. However, advantage should be taken of the simplicity of these arctic systems which will enable theoretical studies to be made in a programmed fashion. Understanding of the energy basis for estuarine classification may make possible proper management of these systems and serve as a stepping stone for increased efficiency in the utilization and management of more complex systems.

Chapter D-4

SEA ICE AND UNDER ICE PLANKTON

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Floating ice and ice covered sea water are an important type of ecological system of the Arctic seas of Alaska. The upper and lower surfaces of the ice form a substrate for life and affect the marine life in the waters below.

Within the United States sea ice is a coastal system unique to Alaska. The ice itself is a type of beach with associated fauna and flora. Seals and walrus breed on the ice; diatoms and other algae grow on its undersurface; Eskimos depend on it for food and travel. The ice is a complex system very important in Alaska. A characteristic plankton develops in waters below.

EXAMPLES

Ice is the major feature of the Arctic Ocean and northern Bering Sea in winter. The southern boundary of the sea ice varies from year to year. This limit is frequently near the Pribilof Islands (59°N). The summer boundary of the polar ice is between 10 and 100 miles off the arctic coast. During August and September the Arctic Sea adjacent to Alaska has the least ice. Advancement of the sea ice begins again in late September and October but the north flowing current through Bering Strait tends to keep the southern Chukchi Sea open longer. Ice closes Bering Strait by the end of October. In late October and November Norton Sound freezes and the sea ice progresses south to its maximum in midwinter. Breakup begins in mid-April. Open water does not extend into the Chukchi Sea until June.

PHYSICAL CHARACTERISTICS OF SEA ICE

Ice on the sea is not one continuous mass, nor is it flat and uniform. Winds, currents, and other stresses produce openings, hummocks, and ridges in the ice. The surface topography generally reflects the undersurface topography. Polynyas and leads, a result of stresses acting on the ice, are present at all seasons.

Sea water in the Arctic Ocean and Bering Sea freezes to an average thickness of 2 to 3 m. This thickness varies locally with the severity of the winter. Losses due to surface melting are replenished by accumulation of new ice on the undersurface.

The boundary of ice-water may take a variety of forms depending upon the given freezing and melting conditions. In the open sea, only sea ice formed by the freezing of seawater is important. However, near the coast, and in particular near the river mouths, floating river ice is introduced into the oceans.

Defant (1961) presents an excellent brief discussion of the formation, terminology, and physical properties of sea ice.

Salinity

The salinity of sea ice is defined as the amount of solid matter remaining after evaporation of 1000 g of melted sea ice. The salinity of sea ice (usually 3-8°/oo) is not conservative. Beneath the ice surface a part of the salt solution remains enclosed between the ice crystals and determines the salinity. The amount depends on 1) the salinity of seawater from which ice is formed, 2) the speed of freezing, and 3) the age of the ice. The faster the ice is formed the less brine can escape, which results in higher salinity. Also, as ice ages the salinity decreases due to leakage of brine.

A selective separation of dissolved constituents occurs during freezing of seawater (Table 1). This begins at -8.2°C. The first substance which begins to separate is sodium sulphate; chloride is retained since it separates at -23°C. Thus, sulfate is steadily withdrawn by the freezing process from the water, which then becomes enriched in chloride (Table 2). A detailed breakdown of the chemical composition of ice brine is given in Table 3.

Density

The density of sea ice increases with salinity but varies due to air bubbles enclosed in the ice. More air is trapped in the ice during rapid freezing, resulting in a lower density. A second source of air-bubble formation is penetration of air during melting. In the upper part of a piece of ice which begins to melt on the inside the rise in temperature first widens the small intermediate spaces containing the salt solution. As the ice particles melt their volume decreases and air fills the spaces; these spaces finally become so enlarged that the melt water and the salt solution flow out and finally are replaced entirely by air, drastically reducing the density of the ice.

Thermal Properties

Extreme temperature gradients exist in sea ice. Temperatures within the ice vary seasonally. Surface ice temperatures are close to air temperatures and can be -30°C or lower. The lowest layers of the ice are

Table 1. Precipitates from sea ice and temperature of their appearance.

Precipitate	Temperature (°C)
Sodium sulfate, decahydrate	-8.2
Sodium chloride, dihydrate	-22.9
Potasssium chloride	-36.0
Calcium chloride	-54.0

Table 2. Differences in cation - chloride ratio of sea ice and sea water.

Cation Na/Cl	Difference +0.085
K/Cl	+0.008
Ca/Cl	+0.003
Mg/Cl	+0.0055
SO_4/Cl	+0.097

Table 3. Chemical analyses of selected arctic ice and water samples; all values in parts per thousand.

	IONS					
Sample Source	Na^+	K^+	Ca^{++}	Mg^{++}	Cl^-	SO_4^-
Snow surface of level ice, 4 cm	2.40	0.079	0.091	0.276	3.75	1.256
Fresh snow	0.039	0.001	----	0.008	0.078	0.016
Sea water 32.893 Salinity	10.27	0.37	0.394	1.309	18.47	2.55
Sea water 33.154 Salinity	10.38	0.38	0.398	1.274	18.71	2.58

near sea temperature, which is no colder than about -2°C.

Mechanical Properties

Freshwater ice is much stronger than sea ice. The strength of ice decreases with increasing salinity and temperature. Large ice surfaces do not remain unchanged for long periods. The combination of wind, waves, and periodic tidal currents breaks large chunks of ice into many separate floes. With the aid of strong winds these separate floes are piled up by the horizontal pressures and pushed one above the other. The resultant mass, when covered with snow and cemented together by freezing, is pack ice.

The coefficient of thermal conductivity is low in the surface layer of sea ice, due to air bubbles, and increases with depth into the ice. This thermal conductivity is greater for sea ice than for seawater. During most of the year the water under the ice is warmer than the air above, and consequently there is a continuous upward flux of heat. Malmgren (1927) estimates that an average of 6800 cal/cm^2 passes through the ice in a year.

Seawater under the ice has a lower limit of about -2°C. The relatively warm water under sea ice in the Arctic is important to birds and mammals. Irving (1964) and others have examined the physiology of marine animals in relation to temperature.

BIOLOGICAL SYSTEMS OF THE SEA ICE

Several interrelated systems are associated with sea ice. As in other oceans, there exists a pelagic and benthic fauna in the water under the ice; there is also a phytoplankton community that develops when conditions are suitable (English, 1961). Sea ice also has a unique community of micro-algae that develop on and in the undersurface of the ice (Apollonio, 1961). There is also a group of higher animals that is intimately associated with the sea ice (Wilimovsky, 1963). These subsystems will be treated separately here for convenience.

Algae of the Sea Ice

A community of several species of diatoms actively lives and multiplies within the diffuse lower layers of first-year sea ice. Apollonio (1961) reported that brownish diatom pigment was apparent in the lower 3.5 cm of arctic ice which was 2 m thick and Bunt (1963) found diatoms in the lower 30 cm of antarctic ice which was 4-5 m thick. Within the diffuse layers more than twenty species of diatoms are present in both arctic and

antarctic ice (Table 4). It is noteworthy that these are not the same species found in other marine habitats of the area. In the ice the species pursue their individual growth habits, the free-floating or motile species remaining in the interstitial water while the sessile species attach to the large ice crystals (Meguro et al., 1967).

The algae live enclosed in a uniquely structured habitat consisting of large plate-like crystals of ice aggregated into a relatively loose matrix with a considerable proportion of unfrozen water. The boundary between the matrix of ice crystals and the main water column is clear and distinct, with somewhat uniform projecting ice crystals (Bunt, 1963).

The temperature at any point in the ice column can be derived if one assumes the heat conductivity to be vertically constant. In the Arctic, where the under ice water temperature is about -1.7°C, assuming an average atmospheric temperature of -20°C, the temperature between 5 cm and 30 cm from the bottom of a 2-m ice column would lie between -1.7°C and -4.5°C (Meguro et al., 1967).

The light available for photosynthetic activity has been the concern of most studies on sea ice algae. Bunt (1963) made readings while diving under 4-5 meters of ice and found a mean level of illumination in the loose layer of about 8 footcandles. Apollonio (1965) froze a photometer in the ice at a depth of 170 cm which was close to, but not in, the diffuse layer and recorded values from 4 to 240 footcandles from March 21 until June 12. He concluded from his readings that probably less than 20 footcandles reached the algae during the peak period of chlorophyll production, and in the previous year, when the snow cover was 23 cm, probably only 10 to 15 footcandles reached the algae.

The interstitial water was found by Bunt (1963) to be slightly less saline and of slightly lower pH than the water below the ice. Meguro et al., (1967), in a more detailed study, found the salinity to vary from 18°/oo at 5 cm from the bottom of the ice to 44°/oo at 30 cm from ice bottom. Furthermore they found nutrient elements to be present in sufficient abundance for growth throughout the winter, but to be insufficient to support the bloom as soon as growth begins in the spring.

In the Bering Sea the growth of ice algae begins in late February or early March (McRoy, 1968). In the Arctic it begins about the end of March and reaches it's peak near the end of May then declines abruptly as the snow begins to melt and disappears before ice breakup in mid-June (Apollonio, 1961). Bunt (1963) reports a similar cycle in the antarctic community, which increases rapidly to a peak in early January then disappears suddenly when water movements raise the water temperature slightly.

The factor preventing growth earlier in the winter seems to be an adverse osmotic pressure created by low temperature. Meguro et al., (1967) found that a change of ice temperature from -1.75°C to -4.45°C caused an

Table 4. Diatoms in the sea ice off Barrow (from Meguro et al., 1967).

Amphiprora kryophila
Gomphonema exiguum v. arctica
Navicula algida
N. crucigeroides
N. directa
N. gracilis v. inaequalis
N. kjellmanii
N. obtusa
N. transitans
N. transitans v. derasa
N. transitans v. erosa
N. trigoncephla
N. valida
Nitzschia lavuensis
Pinnularia quadratarea
P. quadratarea v. biconstracta
P. quadratarea v. capitata
P. quadratarea v. constricta
P. quadratarea v. stuxbergii
P. semiinflata
P. semiinflata v. decipiens
Pleurosigma stuxbergii
P. stuxbergii v. rhomboides
Stenoneis inconspiqua

increase in osmotic pressure of 380 mm of mercury. Once the temperature rises sufficiently to allow a reduction in osmotic pressure below some critical point, active growth may begin. But once growth and multiplication get underway nutrients soon become deficient in the essentially closed ice habitat.

Meguro et al (1967) suggest three possible sources of nutrient supply. First, there is the possibility of bacterial conversion of organic compounds contained in sea ice; second, nutrients, could be replaced through penetration of seawater under the ice; and third, nutrients could be made available by the process of desaltation. This last physical process may well be the key in rendering the lower ice layers habitable.

Desaltation is described by Ringer (Sverdup et al., 1942):

> "In such rapidly frozen ice the cells containing the brine are large or numerous. If the temperature rises, the ice surrounding the cells melts and the separated salt crystals are dissolved, but before complete solution has taken place the brine cells may join, permitting the brine to trickle through the ice. If, on the other hand, the temperature of the ice is raised to 0°C, all the salts dissolve, the cells grow so large that the ice becomes porous, and all the brine trickles down from the portions of the ice above the sea surface, and the exposed old ice becomes fresh and can be used as a source of potable water."

Additionally, Meguro et al.(1967) observed in early April that sea ice formed near Point Barrow, Alaska, showed a gradual decrease in salinity from top to bottom with an average chlorinity of 2.4°/oo. A subsequent observation of chlorinity in the summer showed just the opposite distribution with a low chlorinity at the top and a graded increase downward. The average chlorinity had dropped to 0.9°/oo. Malmgren (1927) states that the amount of brine in the ice is related to the rate of ice formation or the temperature of freezing before desaltation has taken place in the spring.

This gradual release of brine from the upper layers would progress upwards from the colored layer due to the light absorption and resultant heating of that layer (Meguro et al., 1967). This is true early in the spring and probably throughout most of the growth cycle, but the graded chlorinity increase from top to bottom in summer ice probably results from atmospheric heating.

The most convincing evidence for nutrient supply by desaltation is given by calculations of the theoretical thickness of the algal layer which could be supported by a given total ice thickness. From the difference in chlorinity of spring and summer ice, the amount of nutrient salt supplied can be calculated. Then from the known nutrient requirements of the diatoms, the maximum thickness of the colored layer can be calculated as a function

of the thickness of the solid layer above. Calculations made on this basis gave results roughly comparable to those observed in nature (Meguro et al., 1967).

Besides the limits implied by nutrient supply, the algae must deal with severe temperatures and extremely low quantities of light. The problems of extreme cold include damage due to ice crystal formation within the cell, structural alterations, and accumulation of toxic products due to decreased solubility of compounds in the surrounding medium or due to differential slowing of metabolic reactions with consequent build-up of intermediates. Many organisms adapt directly to these hazards by enduring them (Allen, 1960). Many cryophils, however, show various amounts of resistance to freezing. Generally increased resistance can be associated with an increase in osmotic concentration of the cell contents, a decrease of free water in the cell, an increase in bound water in the cell, and a decrease in the permeability of the cell membrane (Allen, 1960). Another general adaptation made by plankton to low temperatures is to increase the concentration of all enzymes. This helps make up for the thermal slowing of metabolism and slows the cell multiplication rate due to the greater amount of organic material that must be synthesized (Eyring et al., 1960) Probably most, if not all, of these adaptations contribute to the success of the ice algae, and further study may indicate others.

Low light intensities also impose limitations upon the algae. Ryther and Yentsch (1957) give 100 footcandles as the limit at which photosynthetic production of energy just balances the energy requirement on the phytoplankton population in temperate regions. The less than 20 footcandles received by the ice algae must make some stringent demands on them. To meet these demands two major adaptations have been made. The first is the general adaptation made by most "shade-loving" plants and involves using a higher proportion of the cell material for photosynthetic enzymes (Eyring et al., 1960). The second involves more efficient use of available light. This is accomplished by the use of chlorophyll c in addition to chlorophyll a, which is only one half as abundant as chlorophyll c in ice flora. Chlorophyll c was shown by Strickland (1960) to be at least six times more efficient than chlorophyll a in absorbing blue light, which Thomas (1963) showed to be more easily transmitted by snow than the shorter wavelengths.

No one has determined the compensation point of the ice algae. Therefore, it cannot be stated to what degree light limits the population. However, the fact that from April 20 until August 20 there is continuous sunlight at Barrow and the weather is normally clear in May and June (Apollonio, 1965) may be important.

The important factors in destruction of the ice algae population appear to be light and temperature. Apollonio (1961) noted a drastic reduction in chlorophyll concentration in the arctic ice following artificial removal of

the snow to allow greater light penetration. Similar results occurred during natural snow melt. In the Antarctic, Bunt (1963) noted that a rise in temperature from -1.7°C to -1.4°C brought about by water movements caused the disappearance of the community in six days. The abrupt decline in both cases may be the consequence of a breakdown of the physical structure of the ice habitat. However, in the Arctic it has been found that in the spring during natural snow melt the chlorophyll concentration falls long before the physical structure of ice changes. This latter observation would indicate that some optimal light intensity has been exceeded, possibly resulting in photo-oxidation of the photosynthetic pigments (Apollonio, 1961).

The very abruptness of the increase of available light with snow removal or melt in the Arctic or the sudden temperature increase in the Antarctic may well explain the demise of the algal population. It is generally known that phytoplankton can adapt to considerable variations in light, temperature, and salinity, but these adaptations do not occur rapidly. Sudden changes often are fatal (Eyring et al., 1960). Furthermore, Marré (1962) points out that algae adapted to severe conditions of heat or cold are highly specialized in that a slight change above or below optimum may impair biological activity.

The algal population under the ice contains a chlorophyll a concentration many times higher than in the surrounding water (Table 5). This high chlorophyll a concentration could indicate a high rate of production, but actual rate determinations are difficult and not yet available.

Under Ice Plankton

The production of organic matter under the permanent arctic ice pack is limited to the dilute arctic upper water (0-50 m), which is freely convective and characterized by physical and chemical homogeneity. When ice melts, water of lower salinity and higher temperature is added at the sea surface. The maximum change in the arctic upper water is only several tenth of one degree centigrade and several tenths of one part per mille salinity (English, 1961).

One of the more striking seasonal environmental changes in the North Polar Sea is the amount of incident solar radiation delivered to the sea surface. The daily insolation, even as far north as 70°N, is negligible from early November until early February. From early February thru May it increases rather uniformly and reaches a maximum about mid June, then decreases through summer and autumn. Thus the radiation available for photosynthesis is limited and is increased by melting of the snow cover, a decrease in ice floe thickness, the formation of ponds on the floes, any decrease in albedo, and the formation of leads. Maximal radiation in the water lags behind surface radiation, occurring after melting of the snow cover and the formation of ponds. An average ice floe appears to reduce the depth of the euphotic zone to less than one-half of the depth of the freely convective layer (English, 1961).

Table 5. Average content of chlorophyll-a and nutrient elements in various sections of sea ice and sea water collected off Barrow in July and August 1964. (from Meguro et al., 1967).

	Chlorophyll-a μg/l.	*Chloride* ‰	*Phosphorous* P μg/l.		*Silicon* Si mg./l.	
			Phosphate-P	**Diatom-P*	*Dissolved silicate-Si*	**Diatom-Si*
Pool water at the surface of sea ice.	0.3	0.79	2.0	0.5	0.12	0.006–0.009:
Middle section of sea ice.	trace	0.92	9.3	—	0.07	—
Plankton-coloured layer at the bottom of sea ice.	120.3	1.13	27.0	204.0	0.44	2.4–3.7
Sea water around sea ice just after melting.	3.1	4.71	10.0	5.3	0.05	0.06–0.09
Sea water in open lead.	2.1	13.3	17.0	3.6	0.14	0.04–0.06

*Diatom-P and Diatom-Si are the elements of diatoms. The values are calculated from chlorophyll-*a* content of the samples according to Parsons *et al.* (1961). P/ch-a = 1.7 and Si/ch-a = 9.4 in *Coscinodiscus* sp.; P/ch-a = 1.7 and Si/ch-a = 14.3 in *Skeltnema Costatum*.

English (1961) studied the seasonal variations in primary production in the under ice water at IGY drift station Alpha. Chlorophyll *a* was used to obtain an estimate of the time of maximum phytoplankton populations and as a relative measure of production and photosynthetic potential. The ^{14}C technique was also used to study variations in photosynthesis with light intensity, chlorophyll *a* concentration, and sampling depth. Rates of carbon fixation were 5 - 6 mg C/m^2 per day in July and August. The standing stock at this time was 30 to 66 mg chlorophyll a/m^2. Maxima in chlorophyll *a* was abundant in the water between floes. A fine net retained less than 10 percent of the chlorophyll *a* retained by a membrane filter (.45 μ pore size), suggesting that most photosynthetic organisms were small enough to pass through the meshes.

The ^{14}C results indicated that the phytoplankton population is apparently not adapted for low light intensities. English also found an increasing proportion of photosynthetically inactive chlorophyll *a* with depth, and the ratios of ^{14}C uptake to chlorophyll *a* concentration were lower than those reported for lower latitudes.

Birds and Mammals of the Sea Ice

Impressive populations of birds and mammals are associated with the sea ice system. Many of the species of birds that form large nesting colonies along the coasts of the Bering and Chukchi Seas winter in the southern margin of the ice-covered Bering Sea. The ice edge also attracts large numbers of mammals. There are 13 species of birds and 10 species of mammals known to occur in the seasonal ice and permanent polar ice (Table 6).

The birds all nest on land, but for most of the mammals the young are born on the ice. The walrus, beluga, ringed seal, and bearded seal stay with the ice as it annually advances and recedes. The other marine mammals become pelagic when the ice is gone from the Bering and Chukchi Seas. The narwal and harp seal remain with the polar pack ice (Wilimovsky, 1963).

The populations of these animals can be immense. Only a few of the bird populations have been estimated. The most studied of the birds, the murres, are considered the most abundant birds in the Northern Hemisphere (Tuck, 1960). Tuck estimates that in the Polar Basin there are about 15 million thick-billed murres and 1 million common murres; in the North Atlantic there are 15 million thick-billed murres and 6 million common murres; and in the North Pacific (largely the Bering Sea) there are 20 million of both species. The total population of the Northern Hemisphere is then 50 to 100 million. Not all of these birds winter in the sea ice; a portion of the common murre population seeks warmer waters. The marine mammals attain equally impressive numbers (Table 7).

Table 6. Birds and mammals associated with sea ice in the Bering and Arctic seas.

I. Birds (from Irving et al., in press)

Old Squaw (*Clangula hyemalis*)
King Eider (*Somateria spectabilis*)
Fulmar (*Fulmarus glacialis*)
Gaucous Gull (*Larus hyperboreus*)
Ivory Gull (*Pagophila eburnea*)
Ross Gull (*Rhodostethia rosea*)
Pacific Kittiwakes (*Rissa tridactyla*)
Thick Billed Murre (*Uria lomvia*)
Common Murre (*Uria aalge*)
Pigeon Guillemot (*Cepphus columba*)
Black Guillemot (*Cepphus grylle*)
Snowy Owl (*Nyctea scandiaca*)
Snow Bunting (*Plectrophenax nivalis*)

II. Mammals (from Burns and Kenyon, 1968; Wilimovsky, 1963)

Spotted Seal (*Phoca vitulina*)
Harp Seal (*Phoca grochlandica*)
Ringed Seal (*Pusa hispida*)
Ribbon Seal (*Histriophoca fasciata*)
Bearded Seal (*Erignathus barbatus*)
Walrus (*Odobenus rosmarus*)
Narwhal (*Monodon monoceros*)
Beluga (*Delphinapterus levucas*)
Polar Bear (*Thalarctos maritimus*)
Arctic Fox (*Alopex lagopus*)

Table 7. Population estimates of marine mammals associated with sea ice (from Burns and Kenyon, 1968).

Species	Estimated Number of Individuals	Biomass metric tons
Ringed Seal	150,000 to 250,000	17,000 to 19,000
Bearded Seal	200,000 to 300,000	60,000 to 90,000
Spotted Seal	100,000 to 150,000	13,000 to 17,000
Ribbon Seal	75,000 to 150,000	10,000 to 17,000
Walrus	73,000 to 113,000	73,000 to 113,000
TOTAL.	598,000 to 963,000	173,000 to 256,000

The biomass estimates of these populations are accordingly large. The murres alone constitute 100 to 200 million kilograms, and the marine mammals more than 256 million kilograms (for the species in Table 7). These animals become quantitatively important in the food webs of the sea ice. They cycle large quantities of nutrients that are transported and concentrated in waters around the ice edge. Wilimovsky (1963) put together the generalized trophic cycle of life for the permanent ice pack region (Fig. 1). In the regions of seasonal sea ice this would include additional species resulting in a more diversified food web (Fig. 2).

Man is intimately involved in these food webs of the sea ice system. The coastal Eskimos of Alaska depend on the marine mammals for food and materials. The Alaska Eskimos annually harvest 19,000 to 26,000 seals principally ringed, bearded, spotted, and ribbon, in the Bering Sea (Burns and Kenyon, 1968). The Soviets harvest an additional 37,000 to 48,000 of these animals. The annual walrus harvest by Soviets and Americans is about 4,200 animals (Burns and Kenyon, 1968). Burns considers these populations to be in jeopardy and strongly recommends cooperative international plans for the study and rational exploitation of walrus and seals in the Bering and Chukchi Seas.

The polar bear is another mammal sought by man. About 300 are taken by Alaskan hunters (U. S. Department of the Interior and University of Alaska, 1966). The polar bear roams over a large region of the ice-covered sea and international cooperation is now in progress to develop circumpolar management principles. The birds, murres and eider, are also harvested for meat and eggs by Alaskan Eskimos. At one time the Eskimos also used the bird skins for parkas. Birds are not hunted on the ice but taken from nesting colonies.

Eskimos also depend on beluga, other whales, and fishes. No estimates are available for their annual take of these species.

Sea ice is a complex system that seasonally extends the coast and influence of man to large areas of the once open sea. The top trophic levels of this system are currently in demand by people of America and other nations. The sea ice system requires international cooperation for the basic research essential to management.

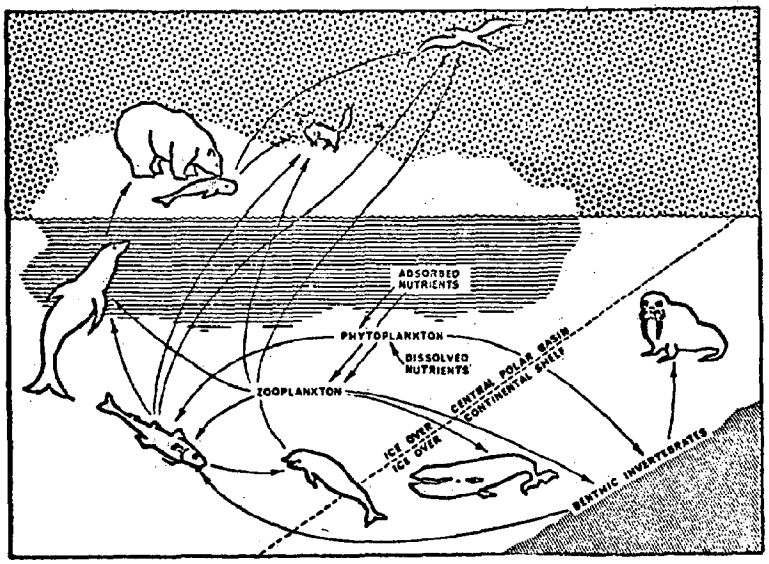

Fig. 1. Generalized food web of permanent North Polar sea ice (from Wilimovsky, 1963).

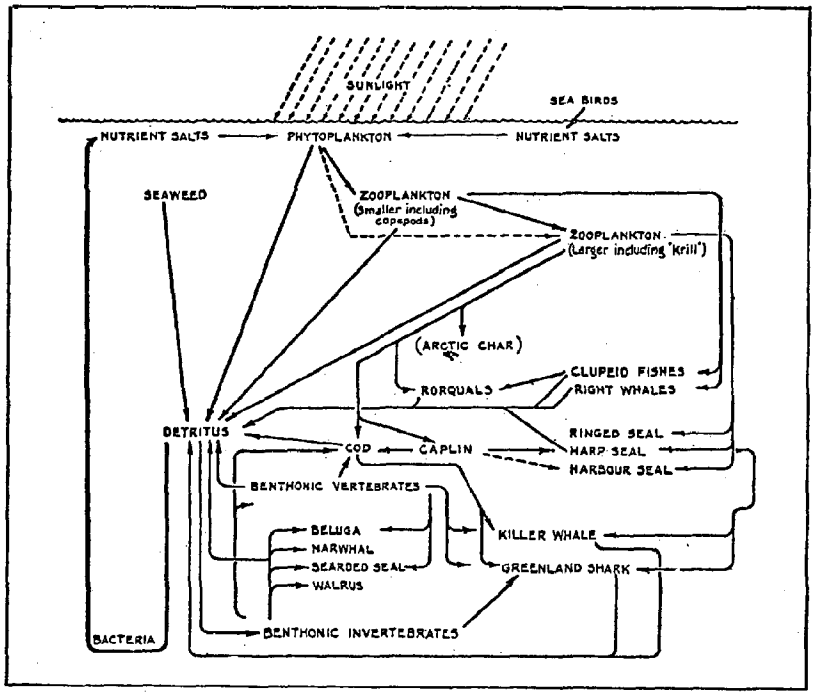

Fig. 2. Food web of arctic seas (from Dunbar, 1954).

Chapter E-1

ESTUARINE ECOSYSTEMS THAT RECEIVE SEWAGE WASTES

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INTRODUCTION

Municipal wastewater or sewage is a mixture of materials in suspension and solution. Since these materials were originally plant and animal tissues they contain the elements N, P and K which are essential to plant growth (Table 1). The application of sewage directly to land as a plant fertilizing procedure has been a method of wastewater disposal. While effective as a technique for handling of limited quantities of wastewater, the fertilization of the land by application of sewage has generally not been too successful. However, when wastewaters are discharged to an aquatic environment, either fresh or marine, the direct utilization of available nitrogen and phosphorus by the plant life at the base of the food pyramid enhances development of dense associations of microscopic plant cells. A sewage waste system discharging into the estuary encourages development of photosynthesizing microscopic plants and in sequence the larger consumer microorganisms, protozoa, crustaceans, rotifers until fish at the top of the food pyramid become the ultimate benefactor of the fertilization. The precise temporal pattern relating to the inflow of waste and the developing sequences of organisms will depend on the degree of dilution and the nature of the indigenous and available species of organisms.

Characteristic of the sewage wastewater system discharging into the marine environment, are abundant coliform bacteria, those organisms associated with human fecal waste and which are discharged from the intestines of all people. These bacteria disperse outward from the point of waste discharge reducing in numbers due to die off and consumption by planktonic animals, by filtering of water by shell fish and by antagonistic effects of metabolic excretions of other organisms until they reach numbers which are normally not associated with high wastewater concentrations. Because of their regularity of die off and disappearance, as well as possible association with discharges from a person who was ill or a carrier of a communicable enteric disease, the coliform organisms and their numbers are generally used to define the boundaries of sewage stress systems and their numbers are used for the classification of such waters. Virus in New Hampshire waters is illustrated in Table 1b.

EXAMPLES

Experimental Tank Microcosms With Sewage Waste

Copeland and Wohlschlag (1968) examined such a stress on a laboratory microcosm. These laboratory simulations were filled with water, vegetation and mud from Red Fish Bay, Texas, and held at a constant 25°C with a 12 hour

Table 1a

Average Composition of Domestic Waste Water at an Average Volume of 100 gpcd, ppm (Fair, Geyer, and Okun, 1968)

State of Solids	Solids, mg/l			
	Mineral	Organic*	Total	Five-day 20° C BOD
Suspended	65	170	235	110
Settleable	40	100	140	50
Non-Settleable	25	70	95	60
Dissolved	210	210	420	30
Total	275	380	655	140

*Organic Composition

40% Nitrogenous substances

50% Carbohydrates

10% Fats

	Daily Per Capita Excretion	
	Urine	Feces
Cl	7.0 gr	---
N	7.0 gr	0.9 gr
P_2O_4	2.0 gr	1.0 gr
K_2O	2.3 gr	0.2

Table 1b. Virus in New Hampshire estuary (Source unknown)

Indicator Bacteria and Enteroviruses in Oysters and Seawater Overlaying Oyster Beds

	Seawater Numbers per 100 ml			Oysters Numbers per 100 gm		
Samples	Percentile	Coliforms (MPN)	Enterovirus Isolations	Samples	Fecal Coliforms (EC-MPN)	Enterovirus Isolations
17	10	4.8	1/19	12	34	2/12
	Median	49.0				
	90	94.0				
30	10	4.0	5/8	10	15	1/10
	Median	47.5				
	90	220.0				

light-dark cycle. Artificial sewage was added every 60 days in concentrations constituting 0.01, 0.1, 1.0 and 10.0 per cent of the total microcosm volume. As shown in Figure 1(A), the stirred microcosm responded at the lowest sewage concentrations with an increase in production of CO_2 whereas photosynthetic rates showed increases up to the 1.0 per cent sewage concentration and then declined at 10 per cent sewage. The 0.01 per cent sewage concentration had the same effect as that of the control. The effect of strong wind action characteristic of the Gulf coast was simulated by stirring. Under quiescent conditions, as shown in Figure 1(B), the rate of photosynthesis increased with increased sewage concentration up to and including the maximum added 10 per cent.

In all cases the ratio of photosynthesis to respiration remained near unity as increases in photosynthetic rates were accompanied by an increase in respiration rate. This was the result of the production of labile organic material by photosynthesis which enhanced the respiration rate of the microcosm. Thus, the introduction of sewage into an estuarine ecosystem resulted in eutrophication and in increase in basic productivity. This increased productivity may not be beneficial to man because the number of carnivores at the top of the food chain may be reduced due to increased toxicity or changes in energy flow pathways. The net effect would have to be characterized for each estuarine system on an individual basis.

Comparing the results of this simple experiment which examines sewage stress on a laboratory microcosm with the estuarine environment, it is apparent that the system becomes infinitesimally more complex in estuaries since there exists the superimposition not only of the energy stress, but of differences in temperature, circulation, and distribution patterns as well as differences in the plant and animal communities into which the stress is being introduced.

Sewage Wastes in Shallow Lagoon
Great South Bay - Moriches Bay, N.Y.

Shallow embayments are commonly found behind barrier beaches along the Atlantic Coast. These embayments typically have low tidal flushing rates which allow for the accumulation of nutrient materials within the system. Such an estuarine environment is the Great South Bay - Moriches Bay - Forge River complex. This system has been studied in detail by Barlow et al., 1963, Ryther, 1954, Lackey, n.d., Galtsoff, 1956, and others with regard to the polluting influence of the wastes from the many duck farms located along the shorelines of these waters. Since the resulting effect of introducing either duck farm wastes or sewage wastes into an estuary may be expected to be the same, namely the addition of plant nutrients (primarily N and P) to the system, a description of the Great South Bay area is illustrative of the effect of nutrient build up in this type of estuary.

The Great South Bay is about 25 miles in length and varies in width from 1.5 to 5 miles (Fig. 2). Its area is 90 sq. miles and its average depth 4.25 ft. The deeper parts of the bay do not exceed 13 ft. in depth except in the narrower inlet channels where depths up to 25 ft. occur.

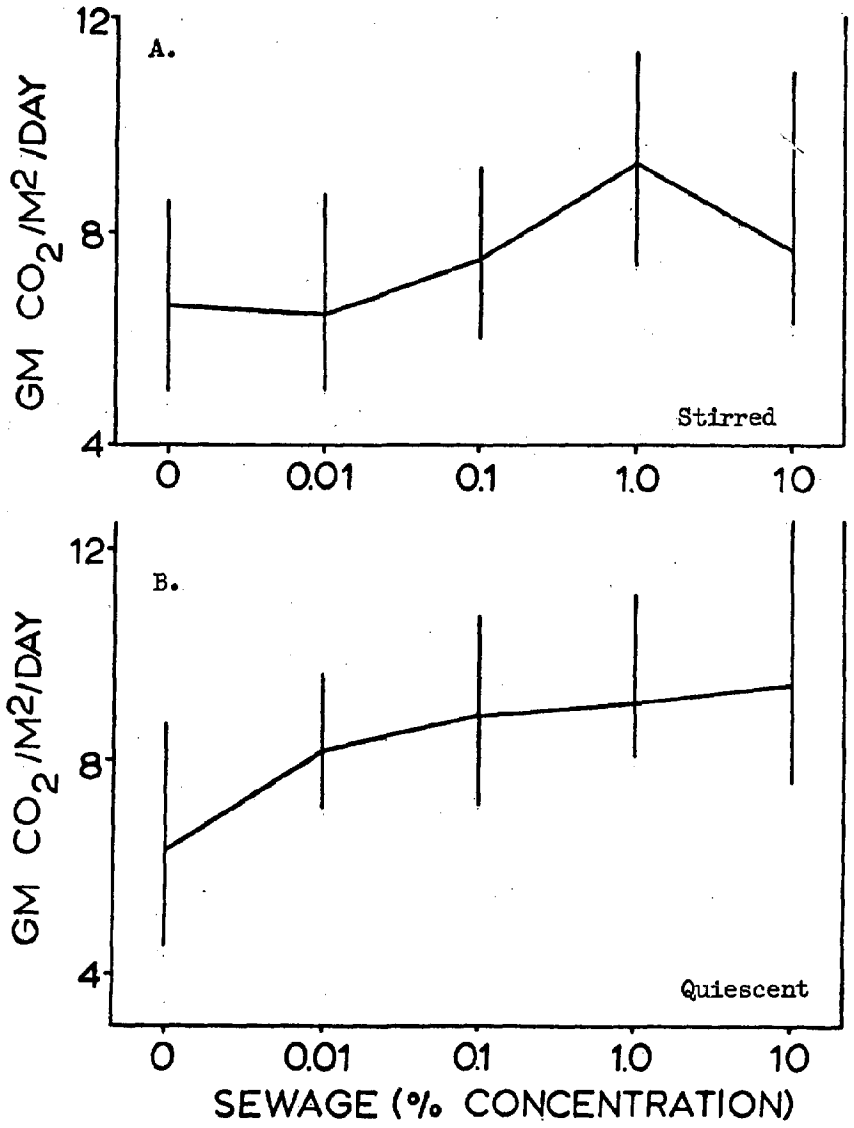


Fig. 1. Photosynthetic rate of simulated estuarine systems as affected by various concentrations of sewage. Percent concentration is by volume. Artificial sewage following the recipe given in Table 19 was used in all cases. The vertical line at each point indicates the range; the horizontal line is drawn through means of 10 or more values (From Copeland and Wohlschlag, 1968; Fig. 22).

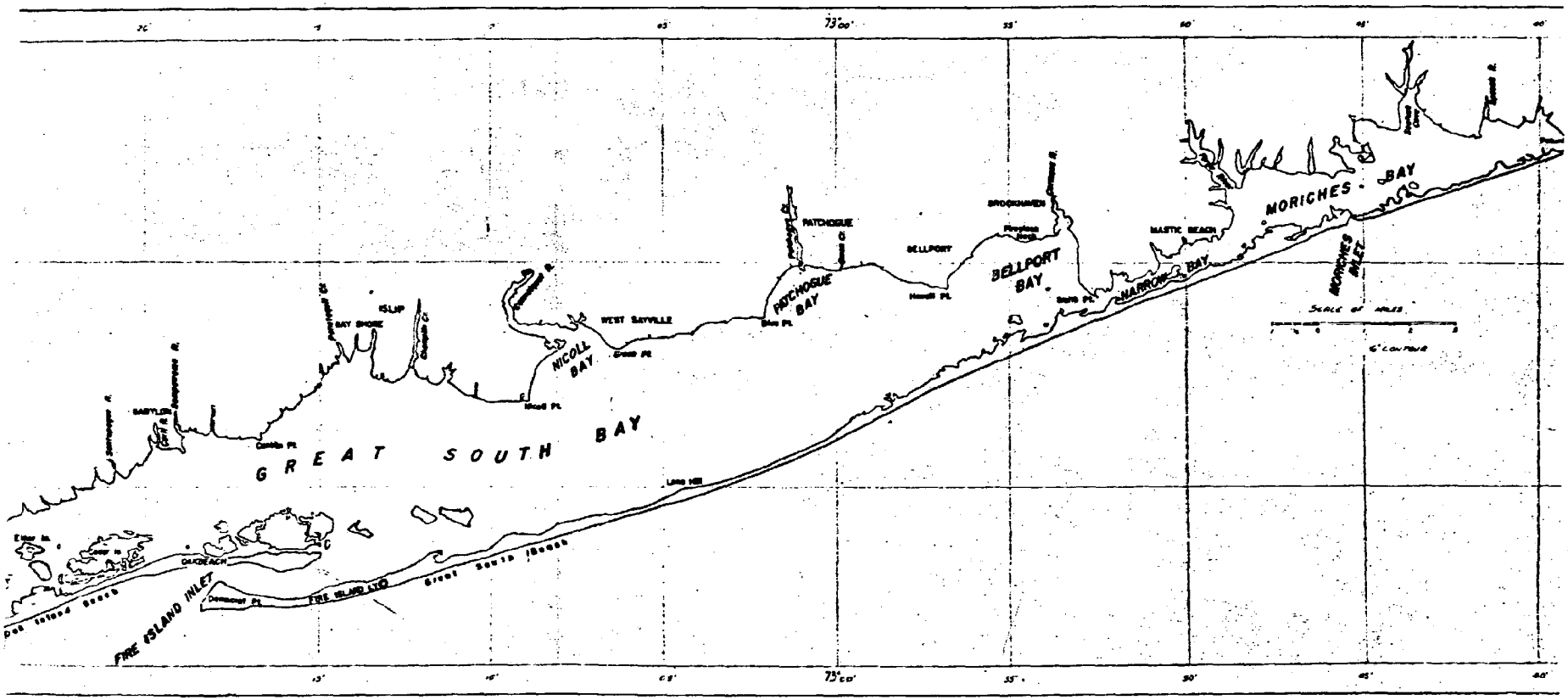


Fig. 2A. Orientation chart of Great South Bay and environs (From Woods Hole Oceanographic Institution, 1951; Fig. 4).

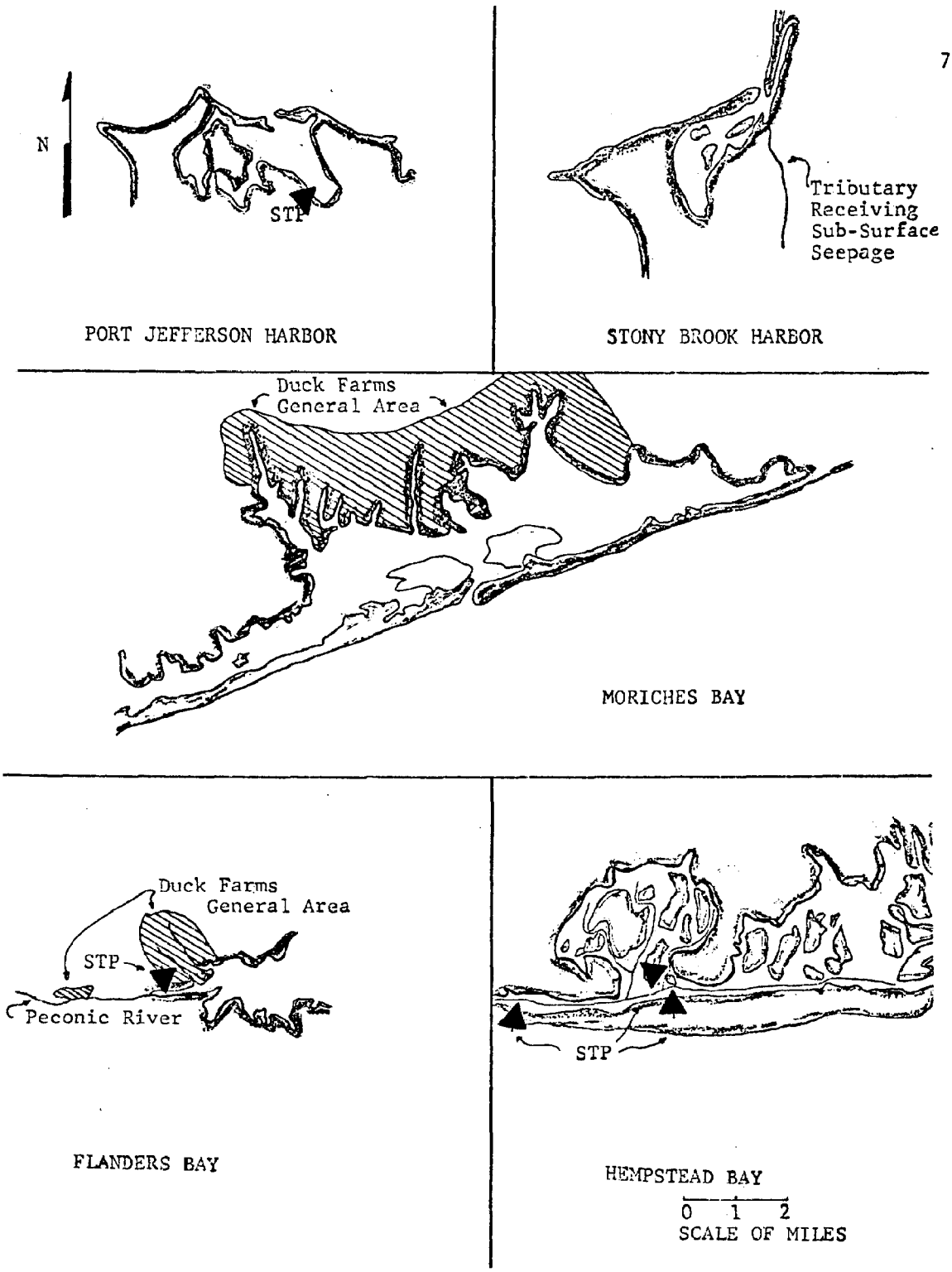


Fig. 2B. Duck farm waste areas on Long Island, New York (Strobel, 1967).

Since Moriches Bay, 9 miles long and 1 - 2 miles wide, is smaller than Great South Bay, the tide rises earlier at the eastern end of Narrow Bay than in Bellport Bay. The result is that the current flows westward from Moriches Bay into Bellport Bay. The east winds also contribute to the westward flow of the water. Low tidal flushing and long retention times cause little net outward water movement from the eastern end of Great South Bay. It is estimated that the retention time in Great South Bay is one month. The long retention times allow the development of high water temperatures - up to 30°C during the summer months - and a dilution of the salinities by the inflowing fresh water to 50% of that of sea water in the bays and to less than 10% in the estuaries.

Since 1940, there has been a decline in the oyster and fish production of Great South Bay. These conditions have coincided with the build up of the duck industry in the areas surrounding Moriches Bay. It is estimated that there are 6 million ducks on 40 farms on the tributaries and coves of Moriches Bay. The wastes from the duck farms effectively fertilized these waters but with a low ratio of nitrogen to phosphorus.

As a result of increased nutrients, especially phosphorus, the waters of Great South Bay have exhibited increased algal populations. Heavy growths of algae developed in the early spring and persisted through summer and fall. At its peak, the concentration of algal cells exceeded 10 million/ml. The dominant bloom alga was a small, unicellular species often termed "small form." This alga differed greatly from the flora typical of bays and estuaries in the same region and its persistence over long periods of time eliminated the typical seasonal succession of forms in the bay.

The decline of the oyster industry was directly correlated with the increase in the "small form." This was due to the fact that the optimum conditions for oyster growth include a mixed algal population. Although oysters do feed on the "small forms," these algae are an inadequate nutrient source. Serpulid worms which are capable of effectively utilizing the "small forms" for food have overrun the oyster beds periodically and thereby adversely affected oyster production by competitive exclusion.

The growth of the "small form" algae, Nanochloris and Stichococcus, in relation to the temperature, salinity and nutrient conditions of the bay waters was studied by Ryther (1954). Both species exhibited good growth in all forms of nitrogen. The common neritic diatom, Nitzschia closterium, also included in the study, grew well in nitrate and nitrite, poorly in ammonia, and slowly or not at all in the organic N compounds. Lowering the N:P ratio had little effect on the diatom but approximately doubled the growth rate of Nannochloris and Stichococcus. It appeared, therefore, that the increased phosphorus concentration in the duck from wastes stimulated the growth of the problem algae.

The duck wastes which enter the bays have been found to contain uric acid and ammonia with nitrite and nitrate present only in traces. The tributary waters were found to contain no uric acid and nitrogen but did contain ammonia. No inorganic N was found in any form in either Moriches Bay or Great South Bay. Phosphorus, however, was found to be present in large amounts

in both tributary and bay waters. Filtered water samples from the bay supported growth of Stichococcus only after the addition of ammonia-N indicating that the latter was the principal limiting factor for the growth of the bloom algae.

The bloom algae normally occurred in Moriches Bay and its polluted tributaries. The pattern of distribution in Great South Bay suggested that their presence in these waters was largely the result of the seaward flushing of Moriches Bay water and that growth was confined to an area close to the source of the nutrient rich duck farm pollutants.

In the original duck wastes, nitrogen occurs as excreted uric acid and amino compounds. Investigations have found that bacteria present in the duck farm effluents are capable of decomposing uric acid very rapidly. This decomposition is accompanied by an increase in ammonia. In the sea, the decomposition of organic nitrogen to ammonia is followed by the nitrification of the ammonia to nitrite and nitrate. This phase of the cycle appears never to occur in the bay area since the nitrogen, as the limiting factor, is utilized as quickly as it becomes available. Therefore, the organisms which are able to utilize nitrogen in its earliest stage of decomposition have the advantage. The laboratory results indicate that the bloom algae have such an advantage over the marine diatom. The high concentrations of phosphorus (lower N:P ratio) in the bay waters also favor the growth of the bloom algae as shown by Figure 3.

The three species of algae were grown in a series of media of different salinities. The optimum salinity for Nitzchia, the typical marine diatom, was near that of sea water. The bloom algae, however, grew well within a wide range of salinities. This is significant in that a wide range of salinities occurs in Great South Bay and Moriches Bay and their tributaries. The bloom algae are thus able to continue to grow as they are carried out to sea until their source of nutrients becomes depleted. At no time does salinity limit their growth.

Studies of the effect of temperature on the growth of the three algal species revealed that the water temperatures in Moriches Bay from May to September (13° - 30°C) favor the growth of the "small form" algae over that of the marine diatom. Since the heaviest enrichment of the bay waters also occurred during this period, both temperature and nutrient levels favored the growth of the bloom species.

Barlow et al. (1963) studied the Forge River, the largest estuarine inlet on Moriches Bay. It is 4000 yds long, 200-500 yds wide and has maximum depths of approximately 6 ft. Most of the allocthonous nutrients and organic matter in the estuary are from the river. The drainage from oxidation ponds treating duck wastes (note: probably installed since Ryther's study) also adds nutrients to the estuary.

The phytoplankton populations of this estuary were very large due to the heavy enrichment and low tidal flushing. The algae were maintained under a state of continually renewed nutrients and the limiting metabolic products

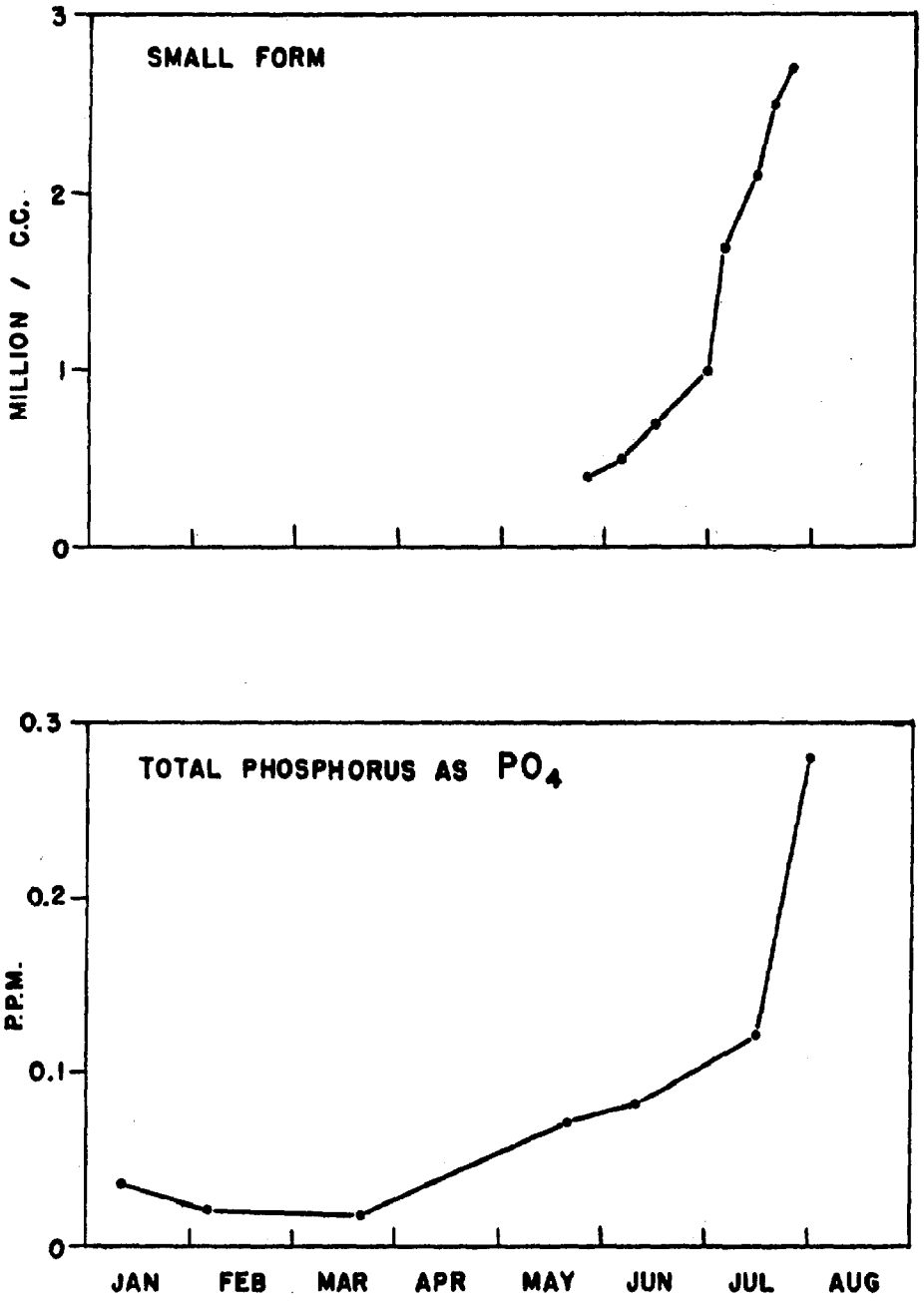


Fig. 3. Monthly concentration of small form and total phosphorus in bay water off West Sayville in 1950 (From Woods Hole Oceanographic Institution, 1951; Fig. 22).

were transported away and diluted by the circulation. This situation was conducive to the active growth of the algae. The average photosynthesis: respiration ratio of the estuary was 1.1 - 1.2, which led to a rate of cell increase sufficient to maintain the population.

One effect of the large algal population in the estuary was to cause significant diurnal fluctuations in the DO. In the early morning, the DO often dropped to extremely low levels. Since the light penetration was reduced, there was little or no photosynthesis by the algae in the deeper water and anaerobic conditions often developed for extended periods in the bottom mud. The benthic forms must therefore be adapted to such conditions.

The circulation patterns described above in which the net flow was from Moriches Bay to Great South Bay occurred only when Moriches Inlet was closed. When this inlet was open, the net flow was reversed. Consequently, in an effort to reduce pollution of Great South Bay, this inlet was dredged open in 1954. Pollution was substantially reduced. The inlet silted in, however, after a period of time and polluted conditions increased again. Another development is that increased septic tank wastes in the area may have augmented the duck farm wastes to a substantial degree. Also, as a result of bacterial contamination, many oyster beds have been closed.

Sewage Wastes in a Deeper Estuary - Raritan Bay

As described in Chapter 1, the medium-salinity plankton based estuary is the "typical" estuary. It is characterized by a high level of productivity and generally supports a diverse biota. Such an estuary, the Raritan Bay, is located between the states of New Jersey and New York. Many industrial and municipal wastes enter directly into the bay. Its three main tributaries, the Raritan River, Arthur Kill, and the Narrows, are also heavily polluted and contribute their waste loadings to the total amount received by the bay. This example illustrates sewage waste system complicated by other wastes.

Raritan Bay and Arthur Kill receive directly more than 480 mgd of wastes which represent a BOD loading of 430,000 lbs/day. When discharges to the Upper Bay and Raritan River are included, the bay receives 1,500 mgd of wastes and 1,300,000 lbs/day BOD. Approximately 75% of the total waste volume is from industry, but sewage effects, such as bacteriological contamination and fertilization, are a major problem.

From 1962 to 1966 various studies were conducted on Raritan Bay (US-FWPCA, 1967 i). Figure 4 shows the sampling stations utilized. The mean water temperatures were found to be 15 to 16 °C and were uniform throughout the bay. The mean chloride concentrations averaged from 13,000 to 14,000 mg/l throughout the bay as against 20,000 mg/l for ocean water. Mean BOD values ranged from an average of 3 to 4 mg/l in the western end of the bay to less than 2 mg/l at the seaward end. The highest BOD value observed was 11 mg/l at Station 56.

The average DO concentrations ranged from 6 mg/l at the mouth of Arthur Kill to 9 mg/l in the center of the bay to 6 mg/l close to the Narrows. Annual, tidal and diurnal cycles affected the DO values. Autumn appeared to

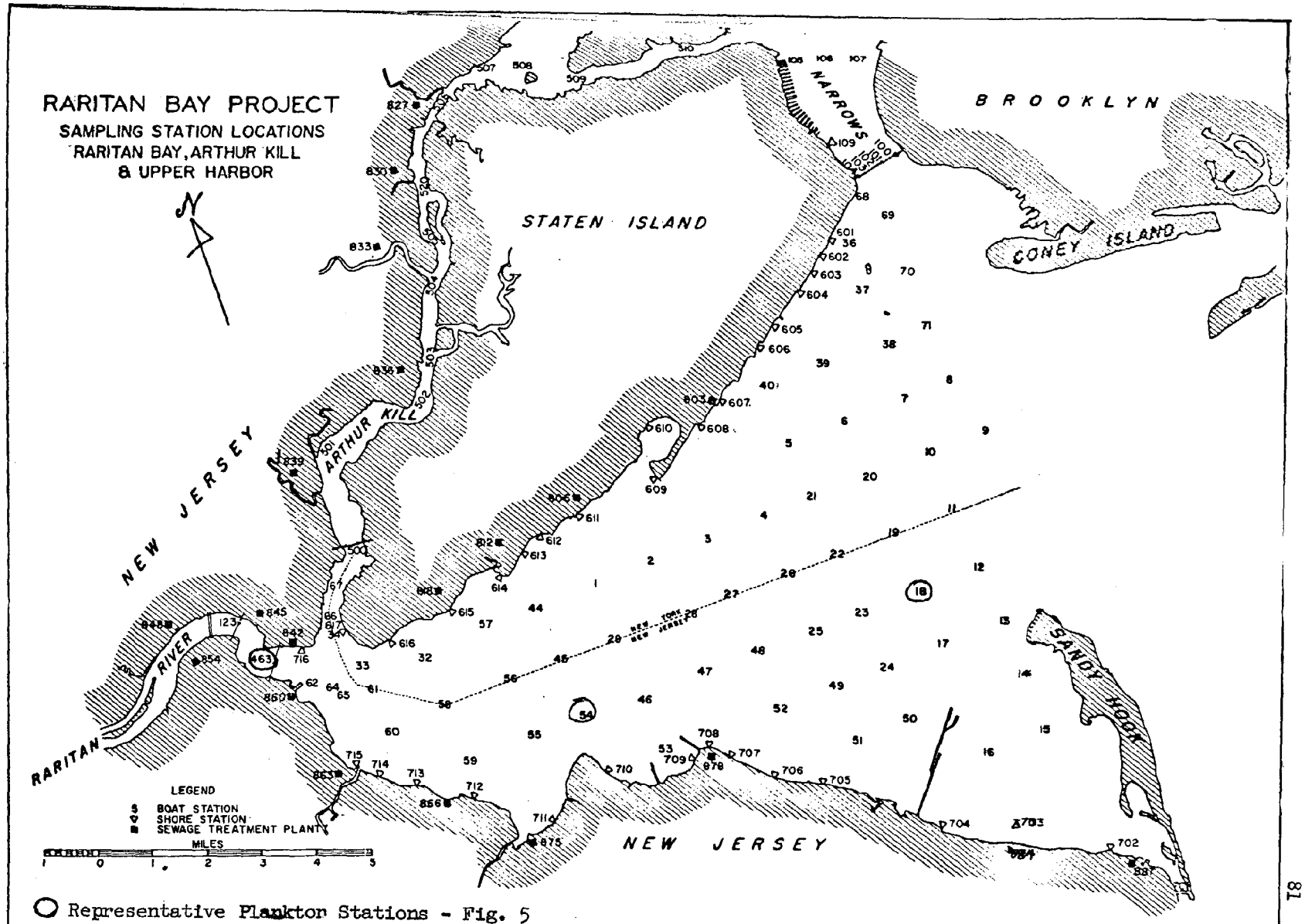
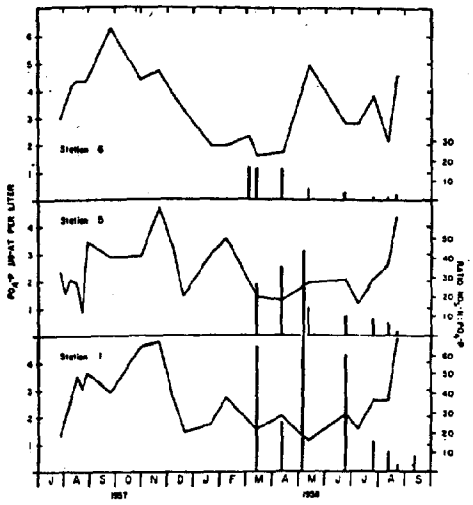
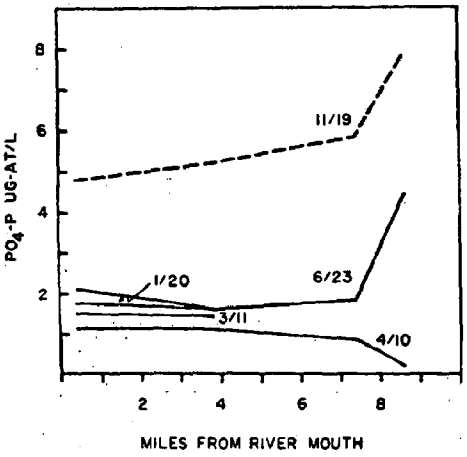


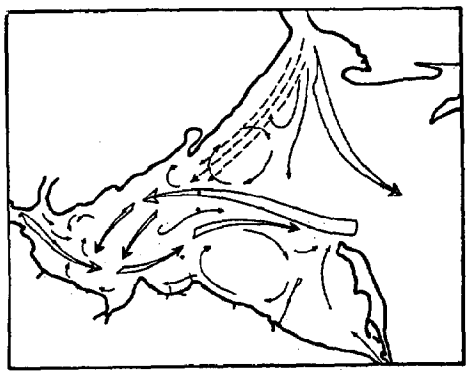
Fig. 4A. Map of Raritan Bay and environs (From US-FWPCA, 1967i; Figure 3).



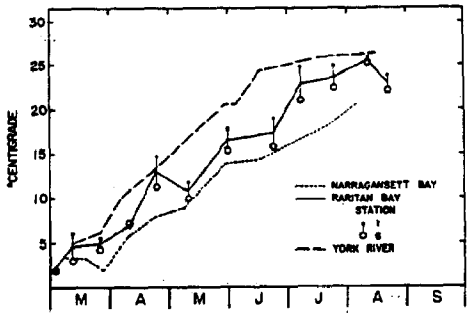
Surface inorganic phosphate-phosphorus and nitrate to phosphate ratio (bar graphs) at stations 1, 5, and 6.



Spatial distribution of surface phosphate-phosphorus in the Raritan River before (broken line) and after (solid lines) operation of the trunk sewer.

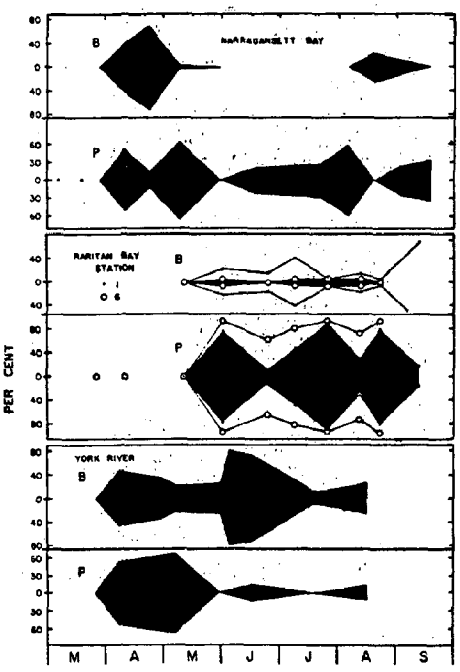


Schematic representation of net currents in Raritan and Lower Bays.

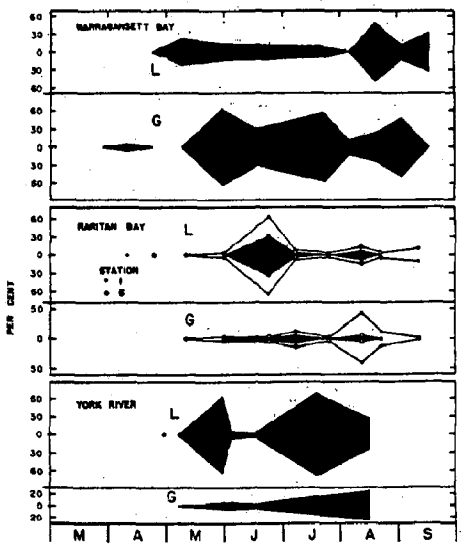


Seasonal distribution of surface temperature in three estuaries, 1958.

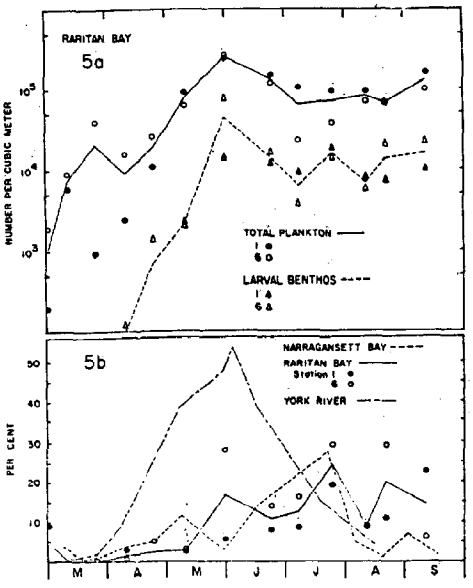
Fig. 48. Characteristics of Raritan Bay (Jeffries, 1962c, 1964)



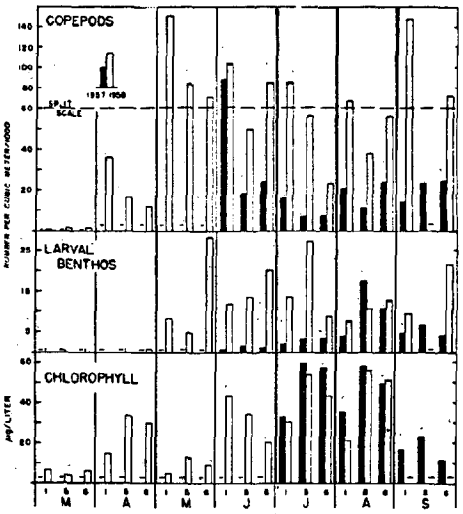
Seasonal distribution of *Balanus* nauplii (B) and polychaete larvae (P) in per cent of the total larvae of benthic invertebrates. Symmetrical plot around time axis. Raritan Bay, 1958.



Seasonal distribution of lamellibranch (L) and gastropod (G) larvae in per cent of total larvae of benthic invertebrates. Raritan Bay, 1958.



Total numbers of zooplankters and the concentration of temporary components in Raritan Bay (5a). Relative density of larval benthos in three estuaries as a per cent of the total zooplankton (5b). 1958.



Comparisons of the plankton components in Raritan Bay before (1957) and after (1958) operation of the trunk sewer. Stations 1, 5, and 6 monitor the quality of water and its entrained populations. Dashes indicate no observations.

Fig. 4c. Characteristics of Raritan Bay, N.J. (Jeffries, 1964).

be the most critical period throughout the bay although the critical period near the Narrows, Raritan River and Arthur Kill was during the summer. Photosynthetic activity appeared to be a major factor in maintaining the DO levels in the bay. Increases in net plankton concentrations were accompanied by increases in DO levels.

The mean MPN confirmed coliform counts were higher at the conjunction of Arthur Kill and Raritan River and at the Narrows than in mid-bay. The densities also decreased from the Narrows through the bay into Sandy Hook Bay. The bacteriological densities along the Staten Island shoreline were higher at Arthur Kill and Narrows than at mid-island. Along the New Jersey shoreline, the bacteriological densities decreased from high values in the vicinity of the Raritan River to lower values in the Sandy Hook Bay. The relative magnitudes of indicator organisms along both shorelines suggested that human waste discharges were the cause of the contamination in these areas.

Stations 463, 54B and 18 were chosen for plankton analysis as most representative of the environmental differences within the estuary. Station 463 was influenced by the Raritan River and had the widest range of salinity. Station 54B was located in the path of outgoing water movement and thus the pollution from the western end of the bay, and Station 18 is farthest removed from the pollution sources.

Figure 5 shows the seasonal variations and abundance of phytoplankton at these three stations. At all stations the nanoplankton, which comprised 94% or more of the total phytoplankton population, was high during the summer and low in the winter. Phytoplankton populations were dominated by two algal species, Nanochloris atomus, a green alga, and Skeletoneuria costatum, a diatom. Generally, fewer species were found towards the western end of the bay than at the seaward end.

These results agree with those of Patten (1962b) who found that the mean diversity levels of net phytoplankton in the estuary increased down bay in association with diminishing pollution and that the spatial pattern was similar to the general patterns of water mass circulation (Figure 6). The diversity levels decreased towards the Raritan River mouth indicating the effect of the pollution in this river.

Zooplankton density, as found by the FWPCA project, generally decreased from Station 18 to Station 463. Zooplankton densities exhibited seasonal variations paralleling both water temperature and plankton blooms.

Jeffries (1964) compared the zooplankton populations during 1957 with those in 1958 after the installation in January, 1958, of a trunk sewer which discharged wastes into the bay. He found that the densities of holoplanktonic copepods that were produced in the summer of 1958 generally exceeded the 1957 crop by a factor of more than two.

Two unidentified rotifers were found throughout Jeffries' investigation. The distribution and abundance of these holoplankton appeared to be related to amounts of suspended organic matter in that: 1) maximum rotifer densities

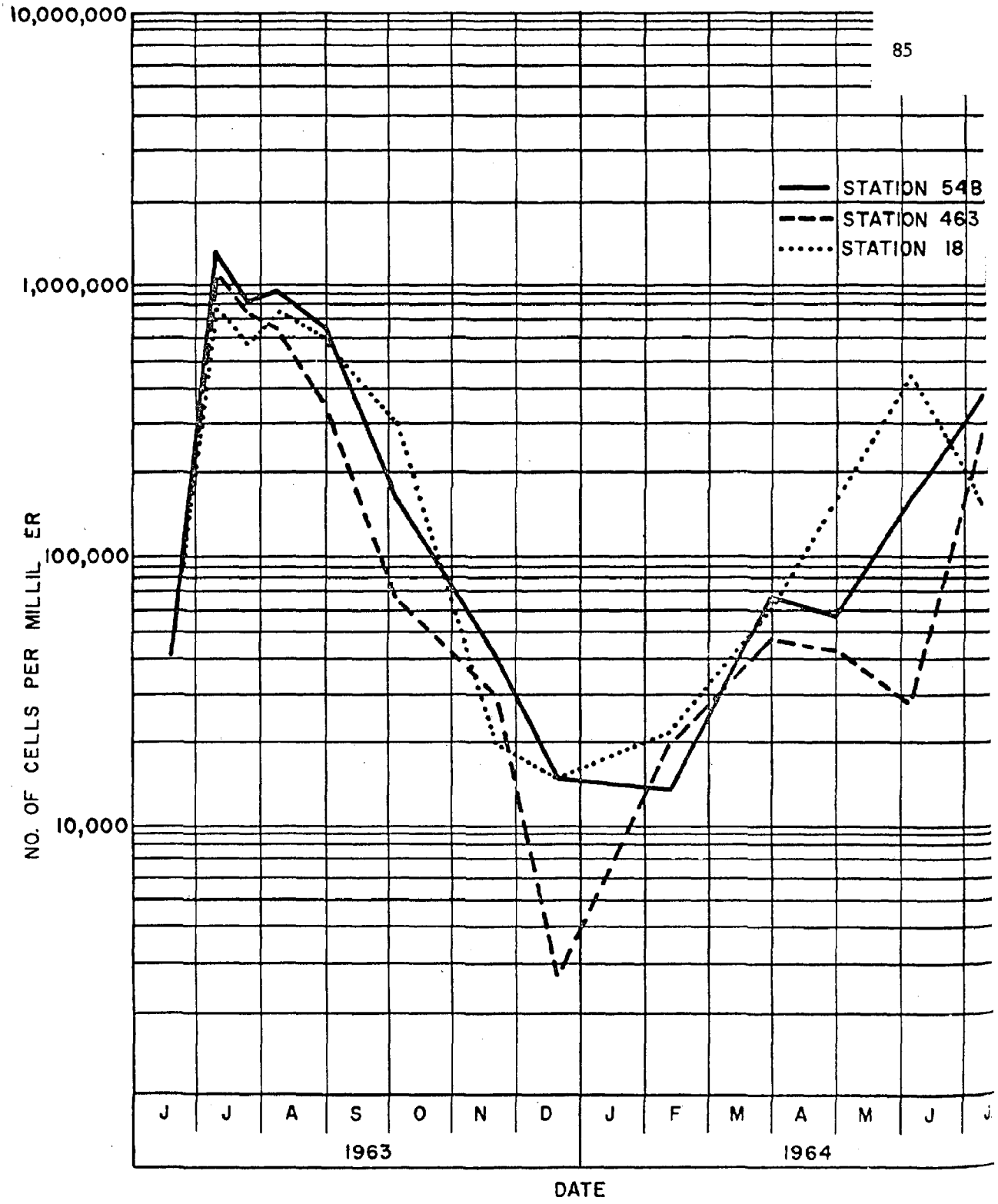


Fig. 5. Total phytoplankton variation and abundance, 3 stations - Raritan Bay (From US-FWPCA, 1967i; Figure 13).

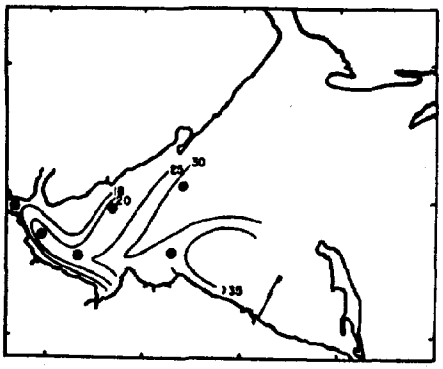


Fig. 6. Species diversity profiles in Raritan Bay (From Patten 1962b; Figure 4).

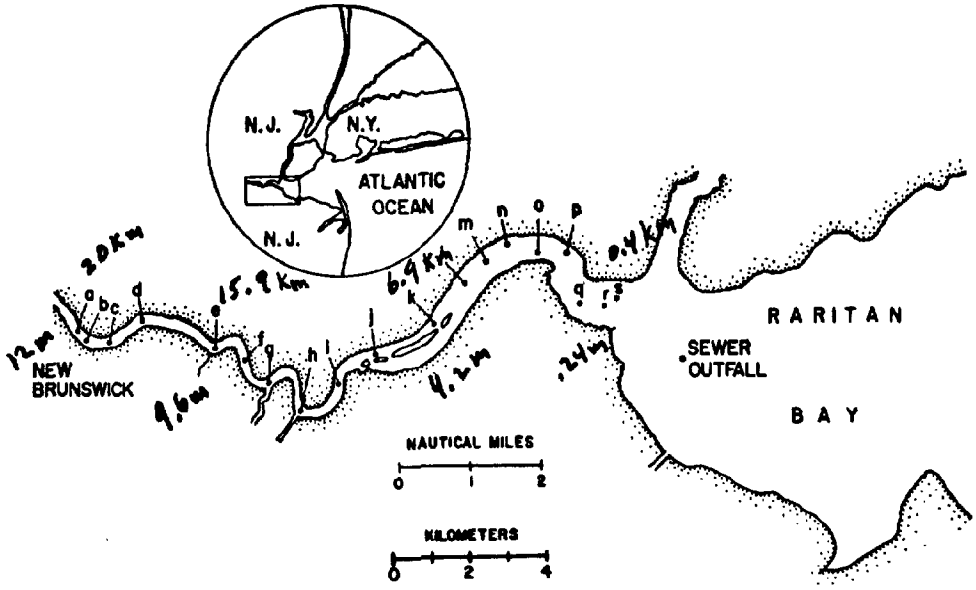


Fig. 7. Chart of Raritan River showing location of Stations a to s. (From Dean and Haskin, 1964; Fig. 1).

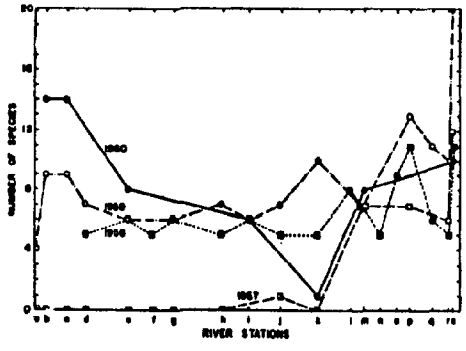


Fig. 8. Abundance of species in the Raritan River, 1957-1960. Number of species includes all species found regardless of sampling method. (From Dean and Haskin , 1964; Fig. 5)

coincided with the height of the spring diatom blooms, and 2) one species was limited primarily to the Raritan River during 1957 but spread seaward when the sewer discharged quantities of treated wastes into the head of the bay.

Nutrient levels (N and P compounds) as found by the FWPCA project, were generally highest at Station 463 and lowest at Station 18. High concentrations of phosphate were found at Station 54B, while Station 463 showed nitrite and nitrate concentrations higher than the other two stations.

Jeffries (1962c) sampled the Raritan Bay from June 1957 through August 1958 in an attempt to relate the distribution of physical and chemical factors to the patterns of circulation. He found that the nutrient levels in the Raritan River decreased after installation of the trunk sewer line. Nitrates were generally minimal at the most seaward sampling station indicating lesser contributions from the sea than from the river.

The two major phosphate sources were the lower bay water entering the bay along the northern shore and discharged sewage and runoff. After the introduction of primary treatment and installation of the trunk line, the concentrations of phosphate in the river were lower than that in the bay. There was a rapid mixing of sewer effluent and bay water.

Jeffries described the phosphate mechanism of the bay as follows: 1) waters high in phosphate enter the bay with the net landward tidal flow along the northern shore; 2) absorption by phytoplankton reduces the concentration progressing into the head of the bay where contributions from the river and trunk sewer are received; and 3) the utilization of phosphate continues in the net ebb current directed along the southern shore with the result that water leaving the bay is relatively low in phosphate.

The FWPCA project collected benthic samples in 1963 and 1964. Those stations with sediment composed of the smallest size particles had fewer animals than those areas with the larger grain size. For all median grain size categories the number of individuals and the number of species increased progressively with distance from the mouth of the Raritan River.

The polychaetes, or segmented worms, and arthropod crustaceans were the dominant benthic organism. Tube dwelling worms, regarded as pollution tolerant organisms, were more numerous towards stations adjacent to the Raritan River mouth, indicating a greater degree of pollution in that area.

Jeffries (1964) identified larval stages of two bivalves, Mya arenaria and Mercenaria mercenaria in plankton samples. Mya increased in numbers from 1957 to 1958 and Mya young were found in previously barren areas signifying the success of spawning and survival in 1958. Mercenaria exhibited a 50 fold increase in 1958 over 1957 in the Raritan River mouth. This was possibly caused by the initiation of the trunk sewer which improved conditions in the river.

Dean and Haskin (1964) studied the repopulation of benthic organisms in the Raritan River before and after the installation of the trunk sewer line. The Raritan River is tidal to Km 33 (mile 30). Saline water extends

to about Km 19 (mile 11). Although the wastes entering the river were not exclusively sewage, the organic loading in the river was estimated to be equivalent to the raw sewage of 750,000 people before treatment.

In 1957, no DO was found at miles 3.8 and 7.4 in the river (Fig. 7). After installation of the trunk line, the DO at these points ranged from 20-60% saturation depending on the time of year.

Bottom sediments changed generally from 1957 to 1960 from coarse cobble to sand and silt with an increased abundance of decaying plant material.

In 1957 no fresh water species of benthic organisms were observed and 17 marine species were found. In 1958, 6 fresh water and 21 marine species were observed. In 1959, 13 fresh water and 25 marine species were found. In 1960, fresh water forms increased to 17 but marine species declined to 21.

The changes in total number of species found might not be as important as changes in the population densities. These densities increased yearly from 1957 to 1960 at most stations. There was a marked increase in the population density of benthic organisms between Km 10 and 20. This reach had no organisms in 1957 and 1958 but increased to 100-1000/m^2 in 1960.

For estuaries in general, under unpolluted conditions, the total number of species upward from the mouth decreases to a low point in the upper estuary and then increases as fresh water is entered. In the Raritan River in 1957 under polluted conditions, the number of species decreased upward from the mouth then failed to increase in fresh water. After pollution abatement, the number of fresh water species increased each year, and by 1960, the classic V-shaped curve of species distribution cited above was established (Fig. 8).

The next two estuarine systems are also medium-salinity types but are sub-systems of the complex Chesapeake Bay estuary. The nature and location of the waste discharge relative to the head of tide and morphology of the river channel plays an important role in establishing the net effect of sewage stress.

In contrast to the Raritan Bay system in which most of the wastes are introduced directly into the bay and the surrounding saline waters, the James River estuary in Virginia receives most of its waste loading indirectly from the fresh water portion of the river. The potential stress of the wastes to the estuary is therefore moderated by passage through the upper river.

Sewage Wastes in River Estuaries

James River

The James River, studied by Brehmer (1967) is the most southerly major tributary of Chesapeake Bay. It is 400 miles in length, 5 miles wide at its mouth and 0.1 mile wide in its upper reaches. Depths of up to 90 ft. have been reported. The James River is tidal from its mouth to Richmond, a distance of 90 nautical miles (Fig. 9). Salt water intrusion extends 35 miles upstream from the mouth. At high run-off, fresh water extends to mile 20 above the mouth and, during drought conditions, saline water extends to mile 64 above the mouth. During the study period, salinities ranged from 20 to 24°/00 at mile 4.

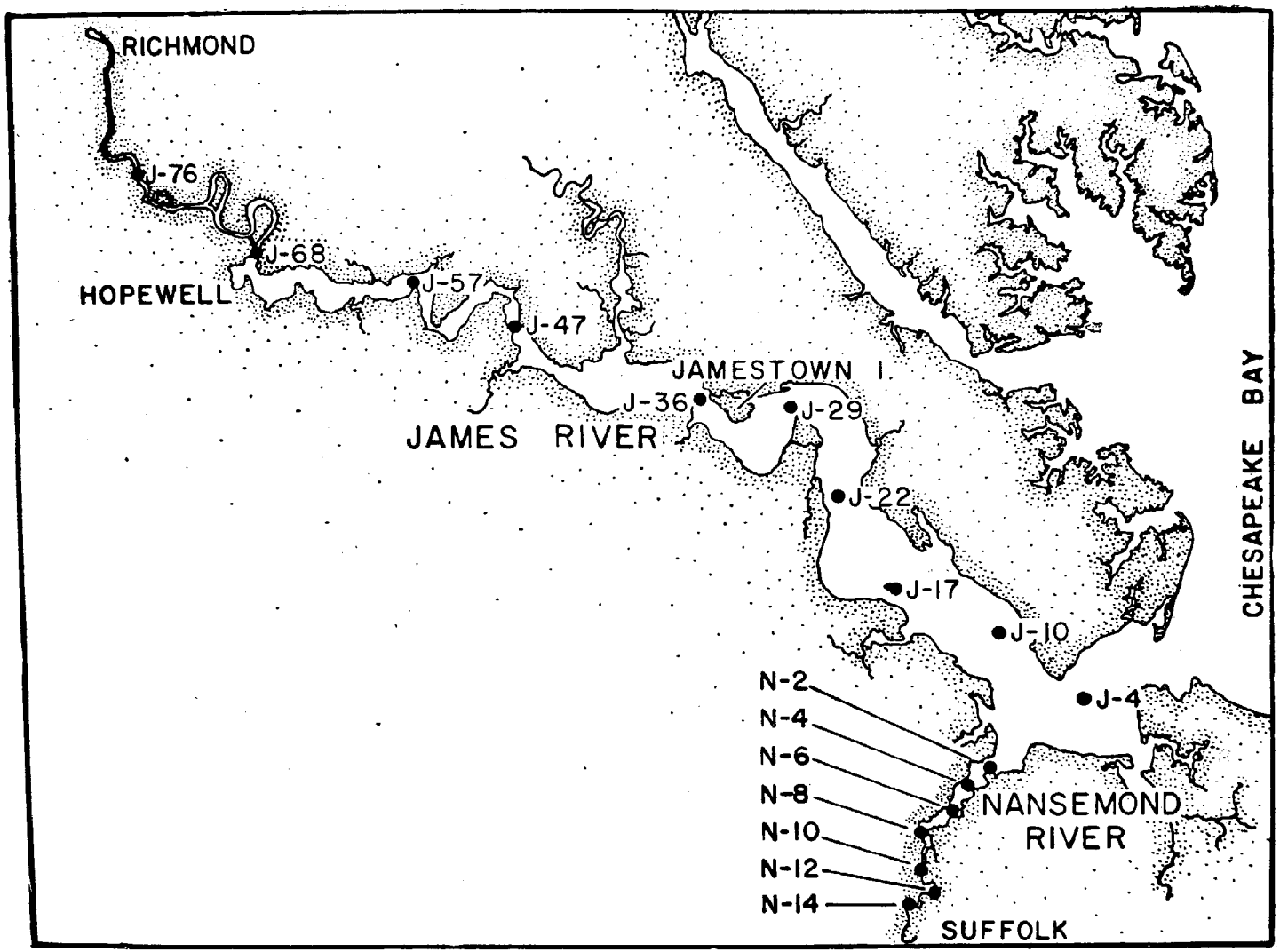


Fig. 9. James and Nansemond Rivers. Sampling Stations designated according to miles from mouth (From Brehmer, 1967; Fig. 1).

The Nansemond River is 16 miles long and empties into the James River at mile 8 above the mouth. The salinity at mile 2 of the Nansemond River ranges between 11 and 20°/00. At mile 15, the salinity ranges from 0.4 to 12°/00.

There were three main sources of enrichment on the James River. Richmond, at mile 90 has a population of 500,000 and 33% of its wastes entered the James River untreated. The BOD of Richmond's wastes exceeded the oxygen capacity of the James River during low flow. Each day 2.7 metric tons of phosphorous and 8.2 metric tons of nitrogen are discharged into the river from Richmond. Hopewell, at mile 60, and Hampton Road, at mile 8, also contribute nutrients to the James River.

The Nansemond River receives the sewage treatment plant effluent from Suffolk. It is estimated that approximately 1.5×10^{-2} metric tons of phosphorous and 4.5×10^{-2} metric tons of nitrogen enter the river from this source.

The phytoplankton populations of the James River were studied over a one year period since over-enrichment produced atypical photoplankton populations which degraded the environment. In the upper, fresh water portion of the river, phytoplankton biomass increased as the water temperature increased. Dense blooms, surface scum and shoreline depositions were observed. At the two stations sampled in the estuarine section of the river, a slight increase in phytoplankton was observed in the fall. Degradation in the fresh water portion of the James River was produced by the blue-green alga Anacystis cyanea. The estuarine pulses were usually produced by the diatom Thalassiosira sp.

In September, phytoplankton densities were high in the fresh water James River at mile 76. Nitrogen was low. At mile 57, below the Hopewell outfalls, the nitrite and nitrate levels increased.

The salt water - fresh water interface was between miles 47 and 57. The phytoplankton population decreased through this zone indicating that the blue-green alga responsible for the nuisance conditions in the fresh water region could not tolerate the increased salinity. The mortality and decomposition of this algae should lead to the regeneration of nutrients below mile 47. This did not occur in the estuarine areas nor did increases in phytoplankton density.

In December, the nutrients introduced into the James River at Richmond and Hopewell were not utilized by fresh water algal forms. Nitrogen and phosphorous were present far in excess of the minimum required for phytoplankton growth. Phytoplankton densities increased toward the mouth in contrast to the situation in September.

Since the nutrients were not regenerated in the estuarine section of the James River, it was postulated that these nutrients were incorporated into the sediments. Studies of the nutrient levels of the sediments indicated only a twofold increase in nutrients whereas a 90% reduction in phytoplankton had taken place. Phosphorous levels did not indicate an increase in benthic phosphorous due to phytoplankton deposition.

Phytoplankton populations in the Nansemond River did not produce aquatic

nuisance conditions. The highest populations were observed during the winter months when water temperatures were minimal. An exception was a bloom in June. In September, virtually uniform conditions existed in the Nansemond River for phytoplankton, nitrogen and phosphorous. In December, nitrogen levels increased in the upper estuary and a bloom of Thalassiosira sp. occurred. In the lower estuary, no such growth was observed.

In the Nansemond River, the normal reduction in total nitrogen in the water from the head to the mouth was not accompanied by an increase in sediment nitrogen. Sediment phosphorous decreased from head to mouth. This distribution was probably due to the net upstream current in the bottom water layer.

The James River exhibited a difference in assimilative capacity between its fresh water and saline sub-systems. In spite of the fact that tidal action retarded the net downstream movement, an action which might be thought to decrease the assimilative capacity by allowing more time for phytoplankton uptake of nutrients and growth with resulting nuisance conditions, few adverse conditions developed in the brackish portion of the river. This may have been due to the lack of phytoplankton forms capable of utilizing available nutrients. Nutrients introduced directly into the Nansemond River having saline water did indeed produce an increase in the standing crop of phytoplankton although the environment was not degraded.

It is felt that the James River estuarine system is just beginning to feel the effects of nutrient addition. Adverse developments have only occurred under extreme conditions, such as low flow or high temperature. Extended periods of nutrient addition as well as increased levels may well temper the ability of the system to compensate for the brief flare ups of phytoplankton, for example, and gradually the nuisance conditions will begin to remain after the extreme environmental conditions have subsided.

Rappahannock River

Of the many rivers flowing into the composite estuary of the Chesapeake Bay system, the Rappahannock, because of its configuration, isolation, one point of introduction of organic wastes, and the canalized channel provides an unique illustration of the stress effect of the tidal oscillation on the stream biota. The Rappahannock is located in the northeastern part of Virginia, bounded on the north and west by the drainage basin of the Potomac River and on the south by the James and York River systems. East of Fredericksburg, the Rappahannock estuary has a controlling depth of 10 feet and a tidal range increasing from 1.8 feet at Tappahannock to 2.8 feet at Fredericksburg, the head of tide. Whereas considerable fluctuations in river flow have occurred in the vicinity of Fredericksburg, the frequency distribution shows that the flow there is less than 250 second-feet for 36 days of each year and greater than 10,000 second-feet for five days of each year.

Investigations carried out by the Virginia Fisheries Laboratory (Massman, et al, 1952) sought to establish the net effect of organic waste discharged by the city of Fredericksburg (raw sewage) and industrial wastes from the American Viscose Company at Fredericksburg totaling for the two the BOD from

a population equivalent of 76,000 people. At the time of the study, neither waste received any treatment prior to discharge. As part of the overall investigation, the U. S. Public Health Service, Division of Water Pollution Control, North Atlantic Drainage Basins (Weiss, 1952) established the distribution and abundance of bottom organisms in the Rappahannock and adjacent rivers. The distribution of benthos illustrated in Figure 10 highlights the relationship between species composition and total numbers of organism as compressed into a relatively short distance by the tidal oscillation of this river. Not only was the organism distribution, both quantity as well as quality, related to tidal movement but the nature and quality of the bottom reflected sedimentation of particulate material with a seasonal movement of the sediments in a net downstream direction resulting from the annual rainfall and runoff cycle.

The organic enrichment of the Rappahannock River by the waste discharge from the city of Fredericksburg and the industrial operation appeared to have stimulated the productivity of the tolerant organisms by several orders of magnitude. At one station, the freshwater clam (Musculium transversum) was found to number 2,800 per square foot of bottom. Another species of freshwater clam (Pisidium sp.), also present in considerable numbers, reached a sampled peak concentration of 880 per square foot. In comparison, other high productivity levels for similar sewage stress situations, such as on the Illinois River and the Chicago Drainage Canal have shown a maximum production of M. transversum of 388 per square foot. The Pamunkey River, the stream immediately to the south of the Rappahannock, and which received, at that time of the study, no pollution load, had as its dominant bottom organisms another species of freshwater clam, Elliptio complanata. Comparing the productivity of the two streams on a numerical basis, the Rappahannock would appear to be the more productive river. However, on a weight basis, the average wet weight of M. transversum being .0937 grams and 404.2 grams for E. complanata, the productivity of the Pamunkey based on the maximum clam sample was calculated to be over 12,000 grams per square foot whereas the Rappahannock yielded only 262 grams per square foot.

Although another species of clam, Pisidium sp. was also present on the Rappahannock, their dimensions and weight was such that they contributed little to the total productivity on a wet basis. While the contribution of nutrients to the Rappahannock River from the organic pollution resulted in a rise in productivity, it was limited to a short stretch of river and to pollution tolerant species. On a wet weight basis per unit area of bottom, it did not compare as well as with the clam productivity of the non-polluted Pamunkey River.

The effect of tidal oscillation can be summarized in terms of both physical characteristics as well as the distribution and numbers of bottom organisms in the Rappahannock River. The zone of degradation and decomposition appeared to extend for a distance of approximately 4.1 miles (nautical) with a zone of recovery continuing for a distance of 0.8 miles. The remainder of the river that was examined for bottom organisms showed no material effect of upstream pollution. This comparatively short distance is, of course, a result of tidal action which tended to limit downstream movement of polluting materials and compressed the zones of pollution effects. While the enrichment of the Rappahannock

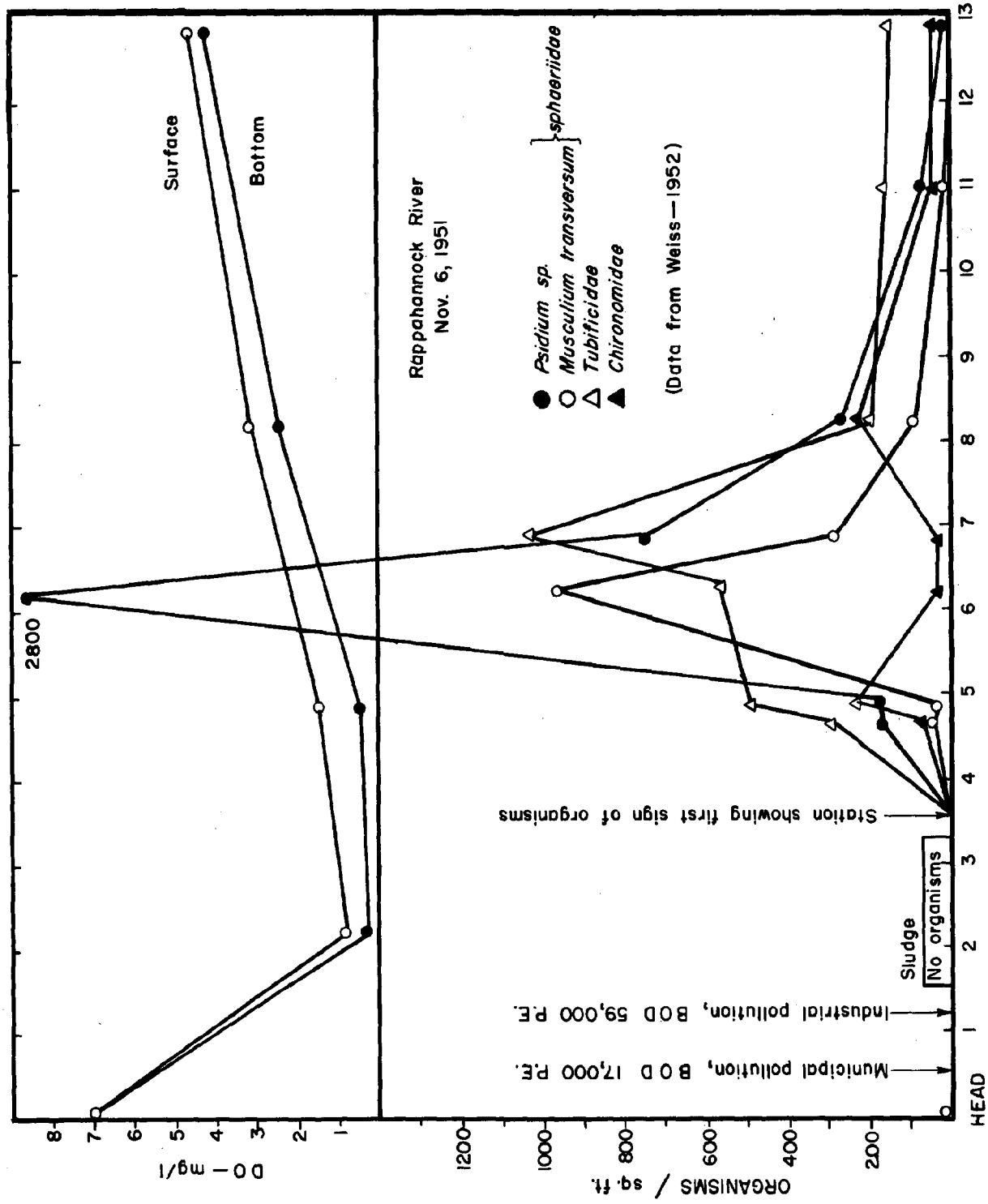


Fig. 10. Mass of bottom animals in the Rappahannock Estuary.

River by organic pollution produced the development of enormous numbers of individuals of the pill clam, M. transversum, the enrichment did not surpass the total clam productivity on a weight basis of the Pamunkey River. It should be noted, however, that these clams belong to the family Sphaeriidae and serve as food for many species of fish. The Rappahannock with the fish food chain supplemented by enormous numbers of M. transversum may thus sustain a greater fish population than the Pamunkey along comparable stretches of river.

A result of the general fisheries survey associated with the bottom organism investigation indicated that the pollution on the upper Rappahannock even though part of the tidal estuary did not appear to have any direct effect on spawning of those fish species which entered the estuary. Although the low oxygen conditions in the immediate vicinity of the organic waste discharge could serve to block upstream movement of young shad and other commercial species, the relatively short distance of stream involved did not seem to materially affect the total productivity of the overall estuary.

It would thus appear that in an estuarine river, such as the Rappahannock, in which pollution is physically restricted by tidal movement, adverse conditions are minimized since a smaller area is directly affected.

Sewage Wastes in a Sub Tropical Lagoon

Many of the bays on the Florida coast are systems in which the shallow depth allows for light penetration sufficient for the growth of submerged vegetation and algae. Among the stresses caused by a sewage waste input to these systems, an important one is the limitation of light penetration and the resulting effect on normal benthic associations. Such a system, the Biscayne Bay in Florida, was studied by McNulty et al., 1960, McNulty, 1961, and Lynn and Yang, 1960.

Biscayne Bay is located between Miami and Miami Beach. It is non-uniform in width (2 to 4 nautical miles) and is approximately 6 nautical miles in length. The Miami River enters the southwest portion of the bay.

There are numerous dredged channels in the bay for ship navigation purposes. The flushing time of the main ship channel is 7 to 9 days. The flushing time of the other parts of the bay where flushing occurs at all is 8 to 11 days. The flood tide surface currents are shown in Figure 11. The currents with the maximum velocities are seen to occur in the ship channels and northward along the Miami shoreline.

The average salinities in the bay ranged from 25 to 32.5°/00 except at the mouth of the Miami River where a minimum of 17.5°/00 salinity was observed. The DO of the surface water of the bay was above 40% saturation with the exception of the Miami River mouth where septic conditions existed.

The total number of sewage outfalls entering Biscayne Bay was 70. The Miami River, carrying the sewage from 29 outfalls, was the major pollutant source. It is estimated that 30 to 50 mgd raw sewage flows into the bay.

In Biscayne Bay, as well as in other estuaries, there exists a rapid

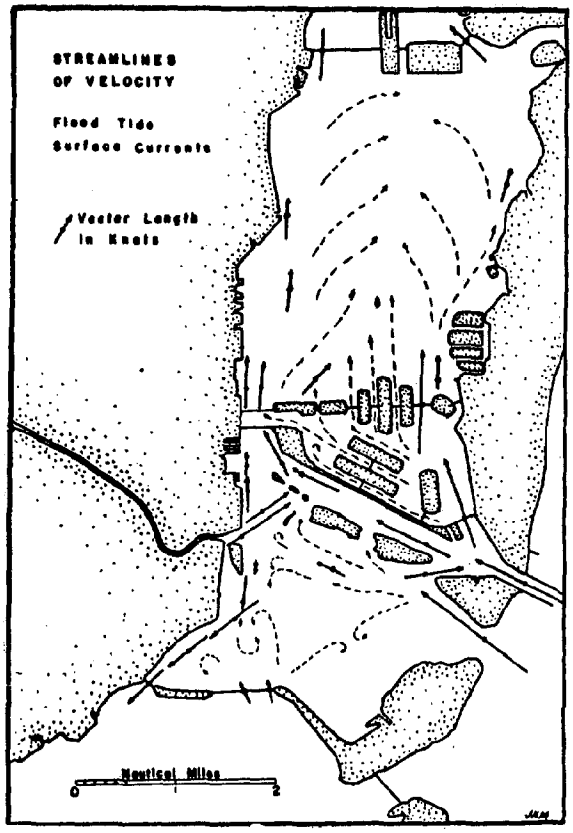
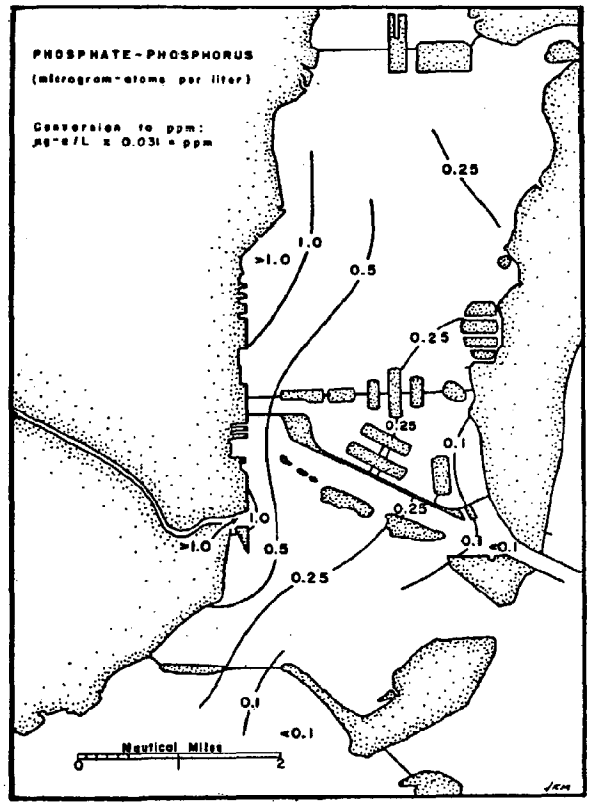
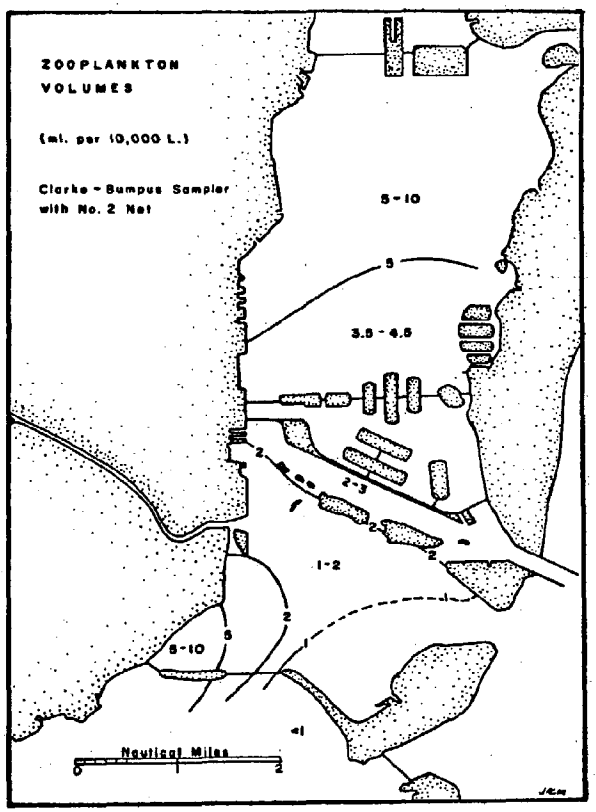


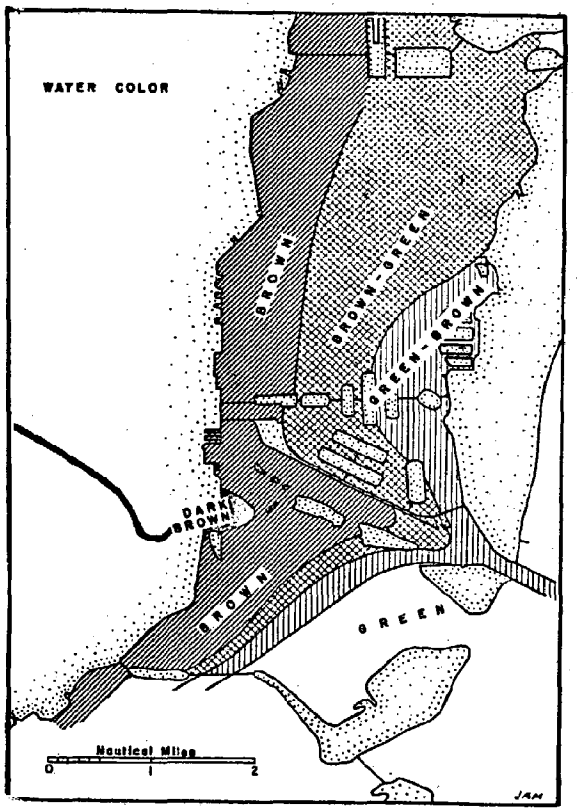
Fig. 11a. Streamlines of velocity, average flood tide surface currents (From McNulty et al., 1960; Figure 2).



Horizontal distribution of average phosphate-phosphorus.



Horizontal distribution of average zooplankton volumes.



Horizontal distribution of average water color from visual field observations.

Fig. 11b. Properties of waters in Biscayne Bay, Miami, Florida (McNulty, Reynolds, and Miller, 1960).

decrease of coliform bacteria seaward from the sewage sources. Application of Ketchum's (Ketchum et al. 1952) mathematical model of coliform distribution produced an average MPN reduction of almost 98% from the mouth of the Miami River to the outer bay areas 1 to 2 nautical miles distant. However, the final predicted MPN values were up to 4 times higher than the observed values. This difference was attributed to adsorption and sedimentation; predation by plankton which escaped the net so were not accounted for; predation by other organisms; and competition for the limited food supply.

The phosphate-phosphorus concentration decreased from greater than 0.031 ppm at the Miami River mouth to less than 0.0031 ppm one to two miles distant in the outer bay. This same decrease in concentration was also observed for nitrite- and nitrate-nitrogen. Both phosphorus and nitrogen concentrations were high along the Miami shore where other outfalls existed and where high current velocities moved water northward from the Miami River mouth.

Zooplankton was low in terms of volume at the Miami River mouth as in areas subject to tidal action. Zooplankton volumes were higher in areas where little tidal flushing occurred.

Biscayne Bay was characterized according to the bottom communities present. The bay was divided into two major zones according to the fixed vegetation: 1) red algal zone; and 2) spermatophyte zone. Zone 1 was found along the Miami shoreline and Zone 2 covered most of the surrounding bay. Two other zones existed: 3) a zone of essentially no life found at 3 stations above the Miami River mouth in areas of degredation; and 4) a zone in mid-bay containing no fixed vegetation but characterized by the presence of the Ophiurian O. limnicola.

Fouling organisms were found, in terms of volume, primarily in areas characterized by 1) rich nutrients and zooplankton; 2) little tidal flushing; 3) an abundance of pilings and sea walls; and 4) brown colored, turbid water high in DO and never below 2°/00 salinity. Tubiculous amphipods were the dominent fouling organism. Their appearance in abundance was limited to areas of high pollution (MPN greater than 10,000).

The softest sediments were found along the Miami shoreline just north of the Miami River mouth. Soft sediments also occurred in mid-bay with harder sediments along the shores of Miami and Miami Beach where the currents are stronger. The sorting of the sediments was most efficient along the shorelines where currents existed. Less efficient sorting occurred in mid-bay where current velocities were lower.

The oxygen uptake of the sediments was highest just north of the Miami River mouth, in the northwestern portion of the bay, and in the deep water south of the Miami River mouth. These zones were relatively deep, had poor bottom circulation and were zones of major deposition of organic-rich material. In shallower water adjacent to these areas where the currents were stronger, lower oxygen uptake values were generally found. The oxygen demand in the major navigation cut was higher than in surrounding areas. The mid-bay area, where tidal action was less, also had high oxygen uptake values indicating deposition of organic materials. This is consistant with the finding of high MPN values in the mid-bay area.

The highest oxygen uptake values occurred in the softest sediments, generally in response to increased MPN. At larger grain sizes, low oxygen uptake values occurred regardless of MPN. This indicates that even though the pollution was higher in these areas of large grain size, as indicated by high MPN values, current conditions prevented the deposition of an organic load.

The relation between the major macro-organism zones and the other parameters studied is given in Table 2.

Some species distribution may be related to sediment particle size, sediment oxygen uptake and MPN. For example, Figure 12 shows the abundance of three organisms, all of which show preference for medium size particles within the particle size studied, sediments of low oxygen uptake, and medium to high log MPN values. Other species exhibited dissimilar preferences.

Both harmful and fertilizing effects were therefore observed in Biscayne Bay. The harmful effects were indicated by the absence of life. These areas were within 200 yards of sewage outfalls, were greater than average depth and had soft, sticky mud with high amounts of oxidizable organic matter. The fertilizing effects were most pronounced in areas 200 - 600 yards from outfalls in shallow water with good tidal circulation in firm sandy mud. Species associations within definite communities were found to be indicative of both the harmful and fertilizing effects.

The Biscayne Bay system is illustrative of the fact that in benthic grass bottom systems, reactions to sewage stress such as eutrophication, low DO and bacterial contamination may be accompanied by changes in the composition of the benthic communities.

Sewage Wastes with Strong Tide - Duwamish Estuary

High Velocity Channel

Strong tidal currents, high flushing rates and deep narrow channels are characteristics of this estuary type. The Duwamish estuary in Washington was studied by Welch (1968) and Gibbs and Isaac (1968). This river-estuary is very narrow (1/8 - 1/4 miles) and, due to dredging, extremely deep (500 ft.) in the lower portion. The lower part of the river was investigated in reference to the installation of a sewage treatment plant at mile 13.1 above the mouth of Elliott Bay (Fig. 13). This plant began operation in June, 1965, and provides secondary treatment for domestic and industrial wastes before they are discharged into the river.

At low tide, saline water extends to about mile 6 above the river mouth except during periods of very high fresh water inflow. At high tide, the river is saline to miles 8 - 9. The mean tidal range is 7.5 ft. and the diel inequality in tides, i.e., a difference in height of successive high and low tides, characteristic of Pacific Coast tides, is evident. The fresh water discharge varies from 200 - 12,000 cfs with a mean flow of 1,000 cfs. The relatively great fresh water discharge and tidal flushing results in unstable conditions existing in the euphotic zone of the estuary during most of the year.

TABLE 2

Relation of Environmental Factor to Major Vegetation Zones (From McNulty, 1961)

Environmental Factor	Red Algal Zone	Spermatophyte Zone
Sediment Size	Very fine to medium sand	Silt to coarse sand
Oxygen uptake of sediments	Very low to high uptake sediments	Very low to medium uptake sediments
Pollution (MPN)	Medium to very high 2.4-5.3 log MPN	Low to fairly high 1.8-4.2 log MPN

The indicative value of the organisms in the designated zones may be inferred by the relation of the organisms to the MPN values as follows:

Zone 1 - Red Algal Zone
Certain organisms within this community association were found to preferentially occur in regions of specific MPN values. Some species were found in narrow MPN ranges while others occurred in wide MPN ranges.

Zone 2 - Spermatophyte Zone
No clear orientation of these species to pollution. However, this community was not found in areas with greater than 4.3 log MPN.

Zone 3 - Zone of Little Life
This zone was definitely indicative of pollution and occurred only where log MPN was 4.4 or greater.

Zone 4 - O. limnicola Zone
This zone occurred only in the range of 3.3 - 3.7 log MPN

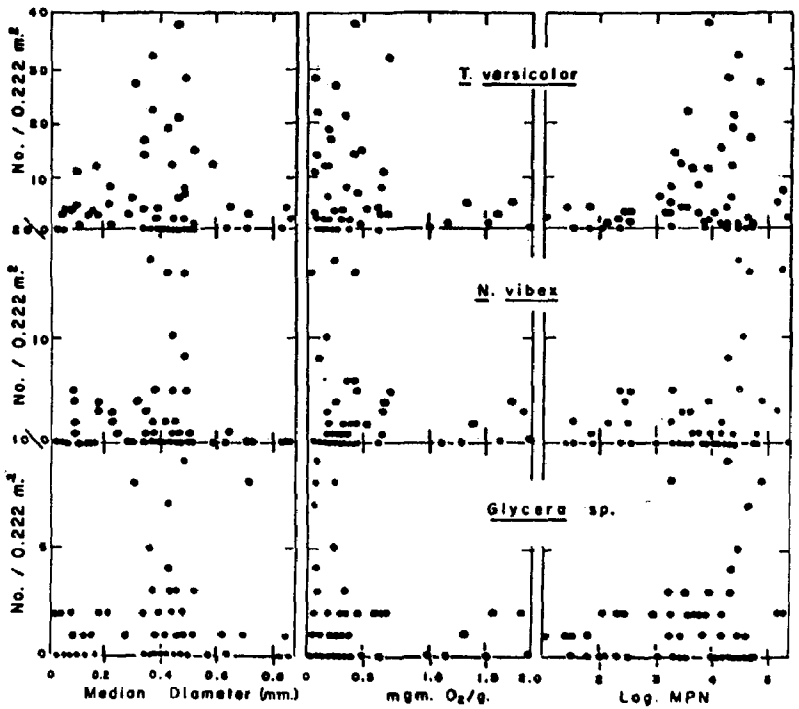


Fig. 12a. Scatter diagrams of the abundance of 3rd order characterizing species of the red algal association against median grain diameter, oxygen uptake of sediments, and log. MPN. (From McNulty, 1961; Figure 11).

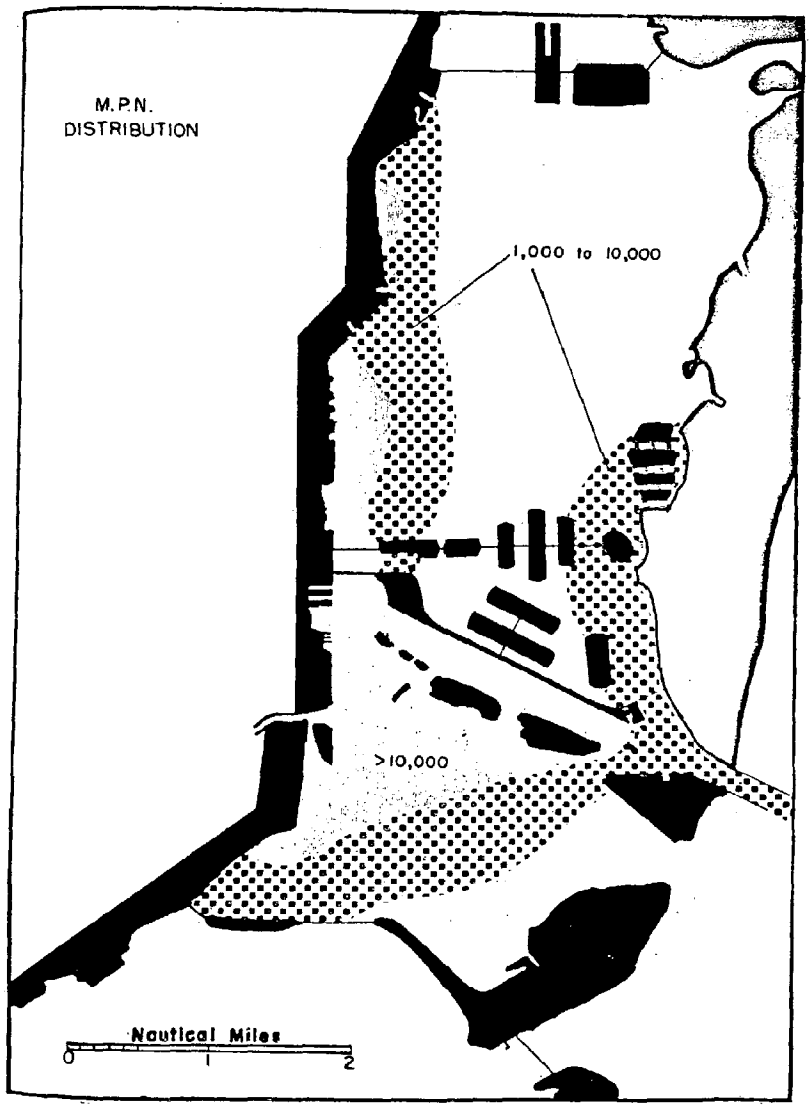


Fig. 12b. Horizontal distribution of MPN (From McNulty 1961; Figure 2, after Minkin).

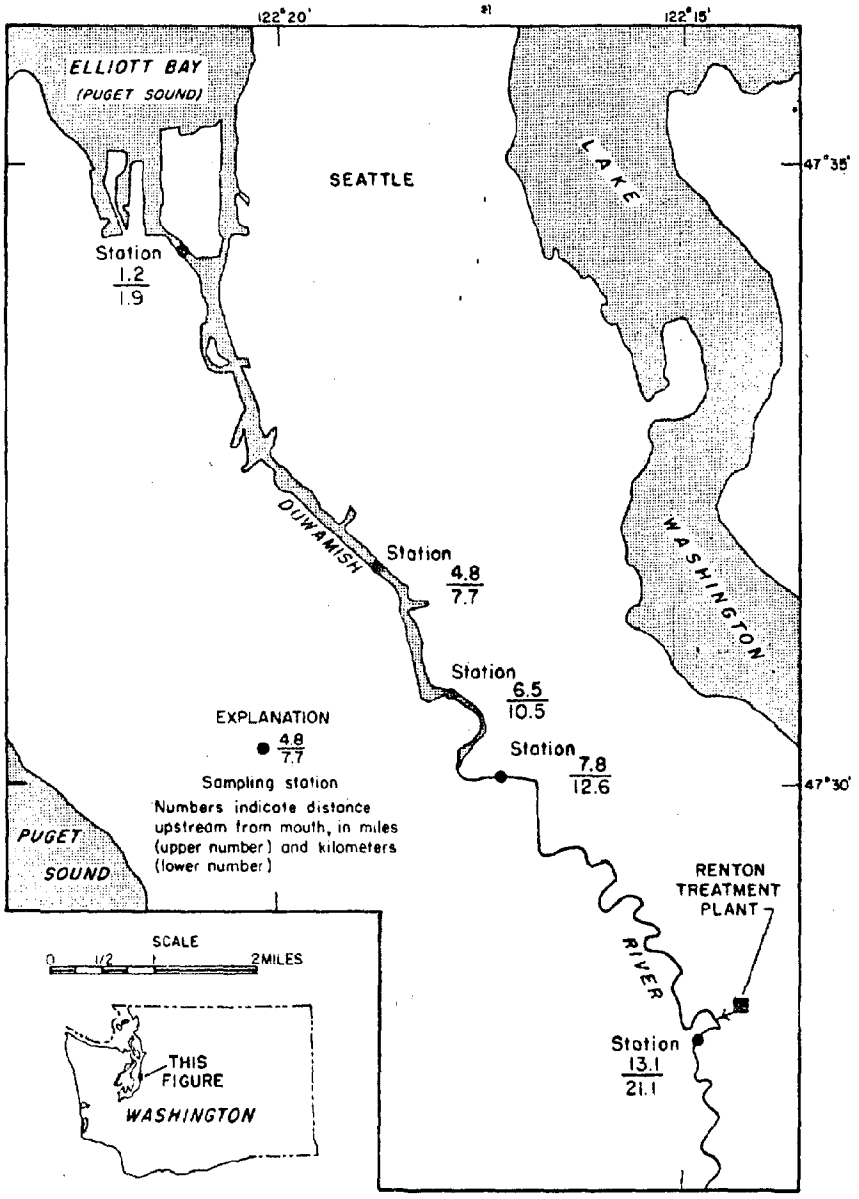


Fig. 13: This map of the Duwamish River and vicinity shows the principal sampling stations and the Renton Treatment Plant. (Mile x 1.6 = km.) (From Welch, 1968; Figure 1).

The stability increases during the low flow conditions of late summer.

Welch studied the effect of nutrient additives from the new sewage treatment plant on the productivity of the Duwamish River. He hypothesized that because the nutrient concentrations in the river were already high, the discharge of additional nutrients might increase the maximum biomass produced during algal blooms. Welch found that although the ammonia, soluble phosphate and total phosphate concentrations were increased after the sewage treatment plant began operation, the algal blooms which occurred could not be attributed entirely to this cause. He found that the blooms resulted more from the hydrographic characteristics of the estuary than nutrient addition since in two of the three years studied, peak phytoplankton activity occurred during periods when the retention time and system stability were at a maximum due to low fresh water inflow and tidal flushing. Thus, it is apparent that even when nutrient concentrations sufficient to support algal blooms are present in an estuary, suitable hydrographic conditions must be present for such blooms to occur.

Arctic Ecosystem

The naturally occurring stresses of the Arctic estuarine systems include ice cover, short photosynthetic energy inputs as well as silt loads from turbid rivers. One such system receiving sewage wastes is the Knik Arm at Anchorage, Alaska. Knik Arm is a long, narrow extension of Cook Inlet (Fig. 14). Because tributaries to Knik Arm carry heavy glacial silt loads, silt deposition causes mud and silt flats to be formed adjacent to Anchorage.

The existence of the broad mud flats allows the establishment of a shallow tidal prism. The prism is warmer and lower in salinity than offshore waters and therefore tends to remain on the surface. Little vertical mixing occurs and the tidal prism floats on the surface of the incoming and outgoing tidal water.

Although little data are available concerning the effects of sewage wastes on the chemical and biological parameters of Knik Arm, the potential harmful influence which sewage wastes may exert on this system type will be apparent from a description of the Knik Arm estuary.

There are three main waste sources in the Anchorage area (Alaska, Dept. H. and W. 1964). Elmendorf Air Force Base has two sewers. The Bluff Road sewer discharges untreated sewage near low-tide level but not below low-low tide level. The second sewer, from Elmendorf Hospital, discharges sewage 1,500 ft. from shore on the mud flats below low-low tide level. The second major waste source is the city of Anchorage from which two sewers enter Knik Arm. The Chester Creek sewer empties into Knik Arm 1,500 ft. out on the mud flats below the mean low tide level but well above the low-low tide. The Turnagain sewer discharges 700 ft. from shore on the mud flats well above mean low tide. This sewer therefore discharges on the exposed beach much of the time. The third source of wastes entering Knik Arm is from Fort Richardson. This discharge empties into the Eagle River which enters the Knik Arm north of Anchorage.

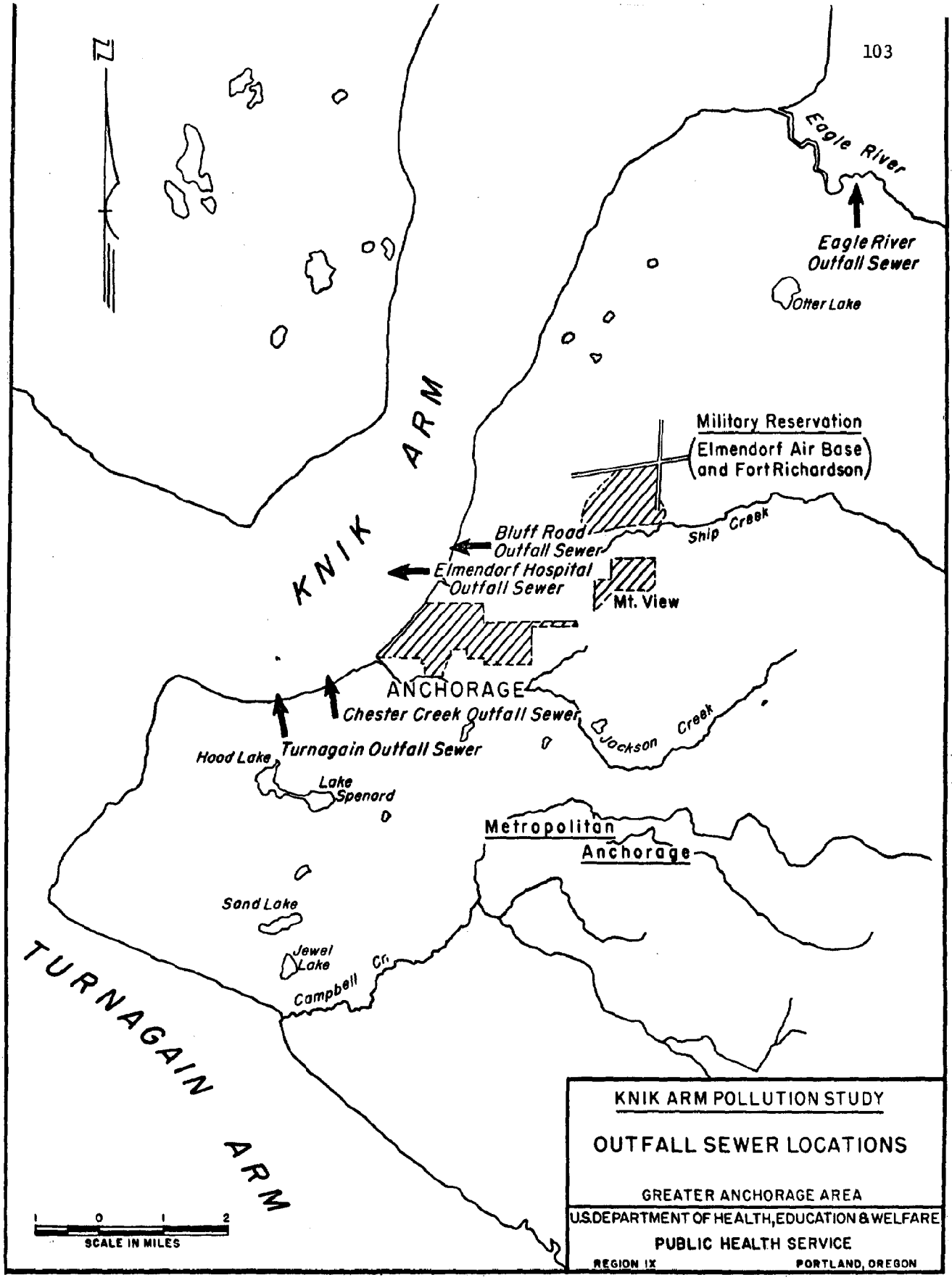


Fig. 14. Outfall Sewer locations at Knik Arm, Alaska (From Alaska, Dept. HEW, 1964; Figure 1).

The combined total discharge of raw sewage into Knik Arm is 8.95 - 9.65 mgd. Since Knik Arm is subject to poor flushing action due to the shallow prism, problem situations have arisen along the Anchorage waterfront. Because of the shallow tidal prism, the untreated wastes which are discharged on the tidal flats enter and become part of the pool of surface water floating on top of the deeper water. Mixing and dispersion of the wastes do not occur, therefore, and the wastes are concentrated into a small body of water which continually sweeps back and forth along the shore of the Anchorage waterfront. The tidal fluctuation of up to 27 ft. does not cause sufficient flushing action to occur.

Two main problems have arisen from this situation. The first is the presence of floating solids which degrade the water and are deposited on the beaches. The second problem is one of bacteriological contamination. Based on the maximum allowable coliform concentration of 240 MPN/100 ml, some locations along the beach are at times up to 280 times greater than the allowable limit. Since these waters and beaches are used for recreation, this situation is extremely dangerous.

The Anchorage, Alaska, situation illustrates how improper outfall placement coupled with a shallow tidal prism can cause a stress on the estuary. Even though ample water is probably available for dilution, shallow sewer outfalls do not allow the entry of the wastes into the water of maximum movement and flushing. Instead, the wastes are introduced into the surface water which, due to the hydrographical characteristics of Knik Arm, undergoes minimal mixing and exchange with the deeper water.

Sewage Outfalls in California

The direct discharge of sewage waste into the marine environment by extended marine outfalls rather than into the intervening estuarine waters defines still another marine sewage stressed system. It is characterized by a long pipe line serving as a sewage outfall placed on the bed of the ocean and extending for some distance seaward. It is generally located, relative to current systems to usual water movement away from shore. It is generally also provided with terminal diffusi n g elements to facilitate and improve the mixing of the relatively fresh water sewage with full sea water. The use of marine outfalls is generally found along coastlines which permit major communities to develop on or adjacent to the shore and where there is no intervening river or estuary for more convenient location of waste discharge points.

The California coast, due to the concentration of population and industry in coastal communities represents within the United States probably the most highly developed situation of ocean outfalls for municipal waste waters. It has been estimated that approximately two-thirds of the municipal and industrial water-borne wastes of this state are discharged to saline waters with a total of 125 communities disposing of their waste water effluents, following varying degrees of treatment, through submarine outfalls. It is expected that as municipalities increase with development and spread of suburban areas, the extension of trunk sewers inland will be reflected in an increase in marine waste disposal since it permits large volumes to be handled with

relatively low degrees of prior treatment. Complexing the problem in California is the high intensity of the usage of the coastal waters by various recreational activities as well as providing an environment for both sport and commercial fishing and the harvesting of kelp.

As shown on the studies on the Rappahanock River, the distribution of bottom organisms relative to the inflow of sewage waste and their movement down an estuary, establishes ecological niches into which specific organisms are capable of entering and multiplying. The specific organisms, their numbers as well as distribution, will depend upon the particular estuarine location relative to the proportion of fresh and sea water as well as their geographical location along the coasts of the United States.

The characterization by the nature of the benthic organisms in the full marine environment into which wastewaters are discharged has been extensively examined by the studies of Reish (1960). In the waters of Southern California, where he carried out most of his investigations in the small bays and harbors of this coast, the Polychaetes constitute approximately 60% of the total number of species and 25 to 50% of the number of specimens in the shallow offshore waters.

The data from benthic studies are best summarized in the form of communities; that is, assemblages of similar species that are grouped together with the community being defined after the generic name of the one or two dominant species. In the marine environment, not only did Reish find differences in species composition due to pollution but also changes in food cycles which would in turn change the characteristic of the community. He noted that a natural marine environment is composed of plants and animals of all feeding types. Among the bottom dwelling animals are the herbivores, carnivores, scavengers, filter feeders and detritus feeders. When pollution is introduced, the ecology of the area is altered. The plant life composed largely of algae is killed and cannot repopulate because of the toxic effect of pollution. Substrates may be eliminated reducing attachment capability as well as the decrease in the availability of sunlight due to an increase in water turbidity. The herbivores are thus eliminated and as waste discharges increase, the carnivores and scavengers then go followed by the filter feeders and finally the detritus feeders remaining as the only organism left of the community.

The sludge worm Tubifex tubifex is characteristically the organism found in sludges near sewer outfalls in freshwater streams. In the marine environment, this similar role seems to be taken by the polychaete Capitella capitata. With C. capitata characterizing polluted bottom areas, other organisms have been observed to be characteristic of increasing healthiness or reduction in pollution. Two distinct communities were found off the Southern California coast, a semi-healthy bottom area in which Cirriformia luxuriosa predominated. Another semi-healthy bottom showed a community with Polydora paucibranchiata and Dorvillea articulata predominating. The healthy bottom area was characterized by large numbers of polychaete species with the predominant forms being Taryx parvus, Cossura candida and Nereis procera predominating. A summary of these results is shown in Table 3.

Table 3. Summary of the biological, chemical, and physical characteristics of the five ecologic areas of Los Angeles -- Long Beach Harbors (From Reish, D. J. 1960; Table 2).

Characteristic	Healthy Bottom	Semi-Healthy Bottom I	Semi-Healthy Bottom II	Polluted Bottom	Very Polluted Bottom
Dominant species of polychaete	*Tharyx parvus*, *Cossura candida*, *Nereis procera*	*Polydora paucibranchiata*, *Dorvillea articulata*	*Cirriformia luxuriosa*	*Capitella capitata*	No animals
Number of Animal species (ave.)					
Polychaetes	7	5	5	3	0
Non-polychaetes	3	2	2	2	0
Dissolved oxygen (p.p.m.) (median)					
Surface	6.0	2.5	2.5	3.5	1.6
20-foot depth	6.0	3.2	3.2	3.5	2.2
pH (median)					
Surface	7.8	7.3	7.4	7.6	7.5
20-foot depth	7.8	7.4	7.6	7.6	7.5
Substrate	7.2	7.2	7.2	7.3	7.1
Nature of substrate (in order of importance)	gray mud, black mud, black sulfide mud	black sulfide mud, gray clay, sand, and mud, black mud	black sulfide mud, gray clay, black mud	black sulfide mud	black sulfide mud
Organic carbon of substrate (%) (median)	2.5	2.0	2.7	2.7	3.4

The Hyperion outfalls in Santa Monica Bay consisting of a 1-mile effluent outfall in operation since 1949, a 7-mile digested sludge outfall in operation since 1957 and a new 5-mile effluent outfall in operation since 1960, have been studied in detail as to their effects on the marine environment (Hume, et al., 1962). Biological effects at the end of the 1-mile outfall are illustrated in Table 4 and Figure 15. At the end of the 7-mile sludge outfall large populations of polychaetes developed as illustrated in Figure 16. Worm populations reaching 200,000/m^2 have been observed.

The impact of the sewage and sludge discharges on higher levels in the food chains is seen in the systematic fishing statistics of Table 5 and Table 6. Over a period of eight years, no indication of significant change in the standing crop of phytoplankton during the spring blooms was observed off the Hyperion outfall. The seasonal minima in succeeding winters dropped to nominal values (Allan Hancock Foundation, 1964). This is what might be expected along an open coast line where coastal currents complete a replacement of water at least once each year and thus the oceanic situation is not comparable to closed systems such as found in lakes where rates of eutrophication exceed those of turnover and exchange. While this is but one of the many differences that exist between a waste discharge into the marine environment and that found in a stream or lake or an estuary, each marine environment will have its own unique feature. Whereas the Pacific is generally rich in phosphorus and poor in nitrogen, other coastal areas, to take an extreme example, the Mediterranean coasts, are poor in phosphorus and rich in nitrogen. In general any effect of sewage appears to be directly related to the dilution factor at the point of discharge and any effect in the larger marine environment is limited to the immediate area of the terminus of the marine outfall itself.

MANAGEMENT OF STRESSED SYSTEMS

For the foreseeable future, the estuary, the interface between fresh and sea water, may continue to be the ultimate sump of waste materials that have been hydraulically transported to this point. Whereas the degree of unstable organic material may be markedly changed by higher levels of waste treatment prior to discharge, the degraded organics and the oxidized forms of nitrogen and phosphorus will still be entering the coastal environment and imposing an additional stress on this particular ecological system. Since the input of nutrients, that serve either as a substrate for microbiological activity or for phytoplankton growth will continue, it is imperative that within the energy stresses established at each particular location by tide and temperature limits, the nutrients be so managed that the net effect optimizes productivity of the marine environment rather than allowing exaggerated or undesirable conditions to develop or that the nutrients be rendered ineffective by too high a degree of dilution. In this manner, the marine environment can be managed to recycle as rapidly as feasible, caloric energy.

ADDITIONAL RESEARCH REQUIRED

1. Since local conditions will play an extremely significant part in

Table 5. Trawling results in Santa Monica Bay (1957-1959) (From Hume et al, 1962).

Item	1957	1958	1959	3-Yr Period
(a) Fish				
Total number	3,970	16,115	19,175	39,250
Net hauls	16	99	148	263
Each haul (avg)	248	163	130	149
Max no. of species/complete survey	40	49	56	—
Total no. species	—	—	—	93
(b) Invertebrates				
Total number	6,916	99,195	75,243	181,354
Net hauls	16	99	112	227
Each haul (avg)	432	1,000	672	800
Max no. of species/complete survey	50	81	109	—
Total no. species	—	—	—	217
No. trawling days	4	16	20	40

Table 4. 1956 biological effects of 1-mile outfall discharge (From Hume et al, 1962)

Distance from Outfall (miles)	Faunal Zone in Benthos	Average Number of Bottom Fish (Flatfish/10-min trawl)
0–½	Limited pollution fauna	12.5
½–2	Pollution tolerant fauna	16.0
2–4	Limited enriched fauna	21.9
4–6	Unlimited diminished fauna	—
6	Normal fauna	12.4

Table 6. Stomach contents of catches in Santa Monica Bay (From Hume et al., 1962).

Common Name	Food Habits
Pacific Sanddab	Polychaetes, amphipods, euphausiids, brittle stars, nudibranchs, shrimp, midshipmen, and gobies.
Dover Sole	Polychaetes, small bivalves, and brittle stars.
Spotfin Sculpin	Polychaetes, small bivalves, shrimp.
English Sole	Polychaetes, brittle stars, clam siphons.
Calico Rockfish	Amphipods, *Solenaya johnsoni*, crabs.
Yellow Chin Sculpin	Small crustacea, fish eggs.
Blue Spotted Poacher	Polychaetes, amphipods.
White Croaker	Polychaetes, crago shrimp, fish.
Speckled Sanddabs	Polychaetes, amphipods, nudibranchs.
Fringed Sculpin	Shrimp, rock crab.
Rex Sole	Polychaetes, amphipods.
Shortspine Combfish	Polychaetes, amphipods, shrimp.
Longspine Combfish	Polychaetes.

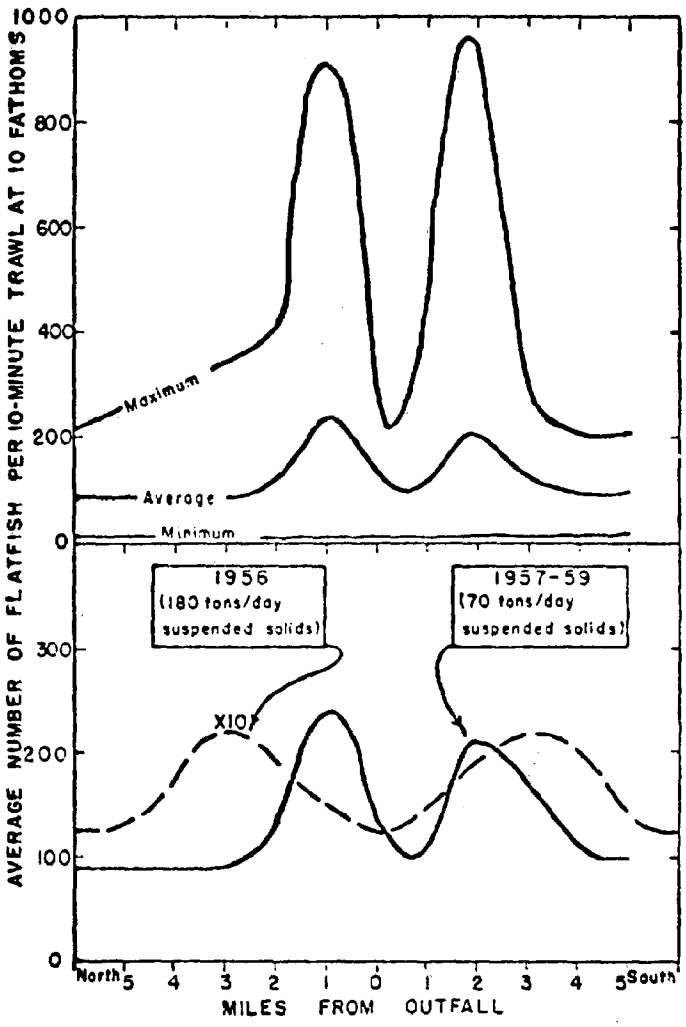


Fig. 15. Effect of 1-mile outfall on bottom fish (From Hume et al., 1962)

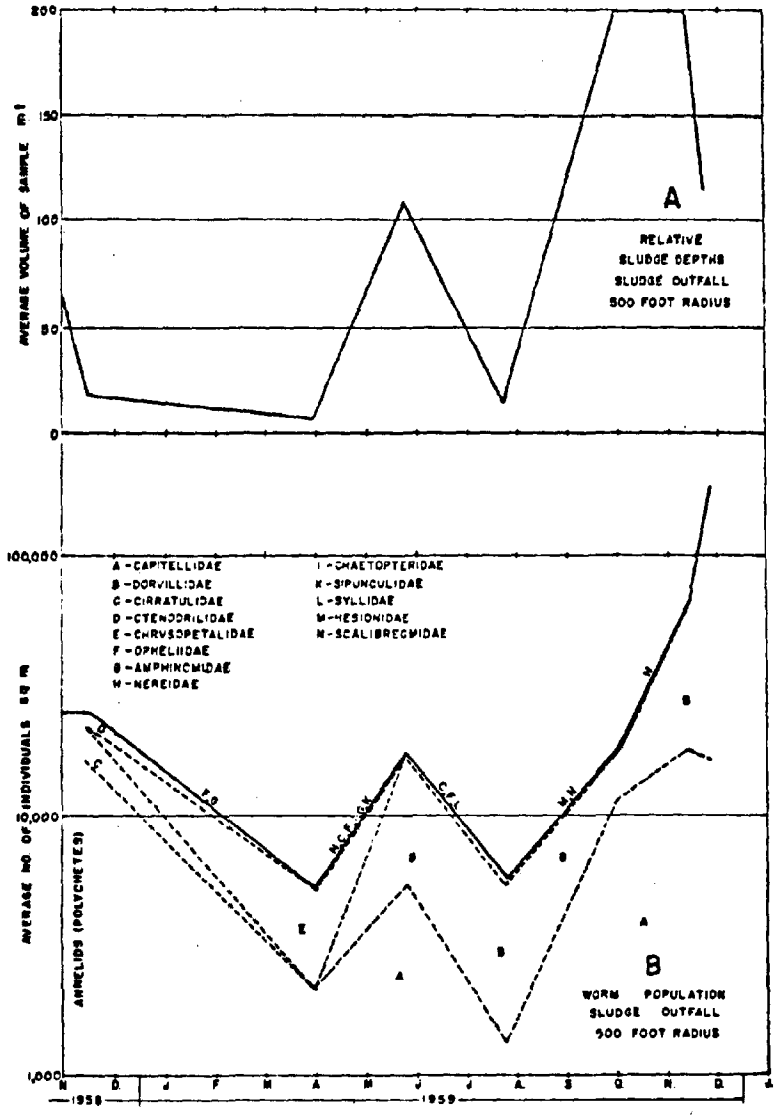


Fig. 16. Bottom deposits and worms in vicinity of 7-mile outfall. (From Hume et al., 1962).

ascertaining the specific response of an estuarine ecosystem to the introduction of sewage wastes, the knowledge of the indigenous marine plants and animals is imperative at each location.

a. What are the effects of various N/P ratios on specific phytoplankton?

b. What higher organisms in the fish food chain are preferentially increased when raw sewage is discharged?

2. Controlled experiments will be needed to determine what level of input nutrients produces particular patterns with respect to organism selectivity and rates of growth.

3. Depending on the dilution factors that might be available, it should be determined what degree of pretreatment is optimum for maximum rates in the utilization of the nutritional input without reaching undesirable conditions.

4. What is the effect of tidal energy on the ability of the estuary to accept and mineralize sewage without anaerobic conditions?

5. The behavior of enteric bacteria and viruses in sea water following various types of sewage treatment and their removal by physical and biological mechanisms is still a question that requires further resolution.

CONCLUSIONS

The introduction of sewage wastes into estuarine waters places a third stress, a nutritional one, on the existing tidal and temperature stresses that vary from locale to locale. Since the rate of interaction of the sewage waste in the estuary becomes a function of its initial strength, its rate of dilution which in turn is controlled by the tidal movement as well as temperature regime of the particular location, the net effect will vary not only monthly and seasonally but no two locations can be said to have a precisely similar set of interacting values.

The management of the sewage waste stress system in the estuary offers the possibility of taking positive action in utilizing the nutritional values inherent in this material when handled on a properly controlled basis. However, this will not be feasible until many questions concerning the effects of these wastes on marine organisms is more clearly understood.

Chapter E-2

SEAFOOD WASTE ECOSYSTEMS

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INTRODUCTION

The dumping of waste fish or parts of fish into inshore waters, in the vicinity of a seafood packing plant for instance, introduces an additional source of energy to an ecosystem. Although qualitatively this nutrient source represents a return of materials previously removed, its release may be irregular and in a concentrated form often high in protein and fat. Such extraneous sources of nitrogen and phosphorus, as well as vitamin complexes and soluble growth-promoting factors, often lead to eutrophication. This is especially true in a semi-enclosed body of water which has a low rate of exchange with adjacent water masses.

EXAMPLES

Texas Harbors Receiving Shrimp and Crab Wastes

At Aransas Pass and Brownsville, Texas, shrimp processing wastes are released to a small harbor which has little tide or wind circulation. Eutrophication, high respiration, and blooms of phytoplankton develop, the waters being turbid with microbial activity.

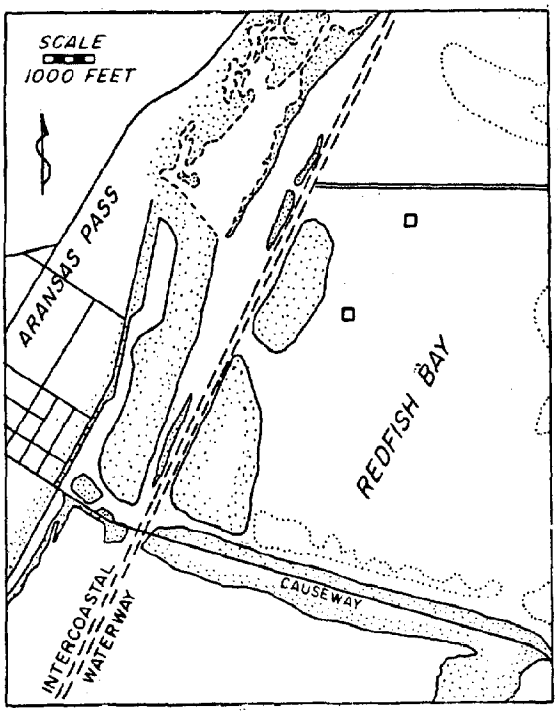
An investigation into the effects of seafood wastes on the harbor ecosystem was conducted at Aransas Pass, Texas by Odum, Cuzon et al. (1963)(See Figs. 1 - 3).

Waste and refuse from fifty to 400 shrimp boats and associated packing plants is dumped into Conn Brown Harbor (Fig. 1). The water, which is three meters deep, was found to be mixed fairly well at times owing to the continual passage of shrimp boats. Water exchange, however, is limited and occurs through a channel entrance at the south end by means of wind- and tide-induced water level changes of small magnitude.

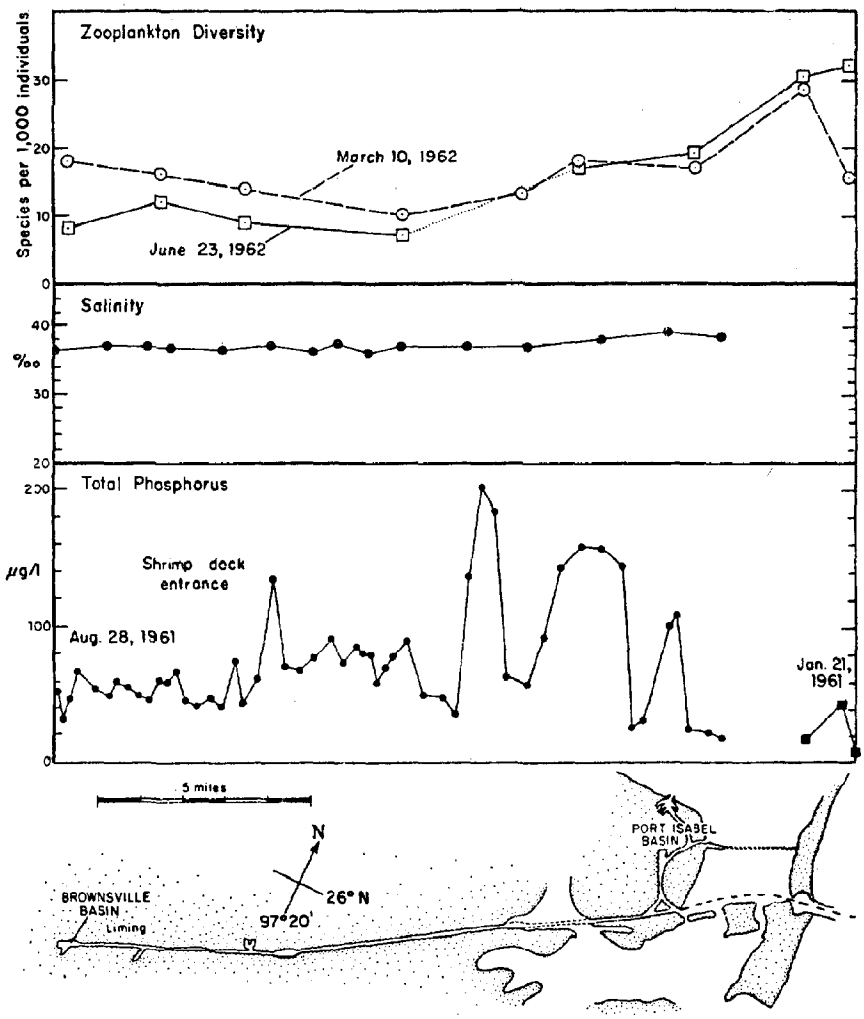
Measurements of oxygen and carbon dioxide concentration over short distances showed partial thermal stratification, and patchy distribution of areas of high and low metabolism during the daytime (Fig. 3). At night, with cooling at the surface or in winds, there was better mixing of basin water (Fig. 3b). The dominant role of metabolism in causing patchy variation in properties was indicated by the correlation of pH and oxygen.

High phosphorus values (averaging 143 ppb) were found in the harbor as compared to the outside waters (49 ppb).

The zooplankton species diversity of the outside bay waters averaged fourteen species per thousand individuals, decreasing to about six species per thousand in the north end of the harbor. Other trophic groups exhibited a similar pattern (Copeland, 1967a).

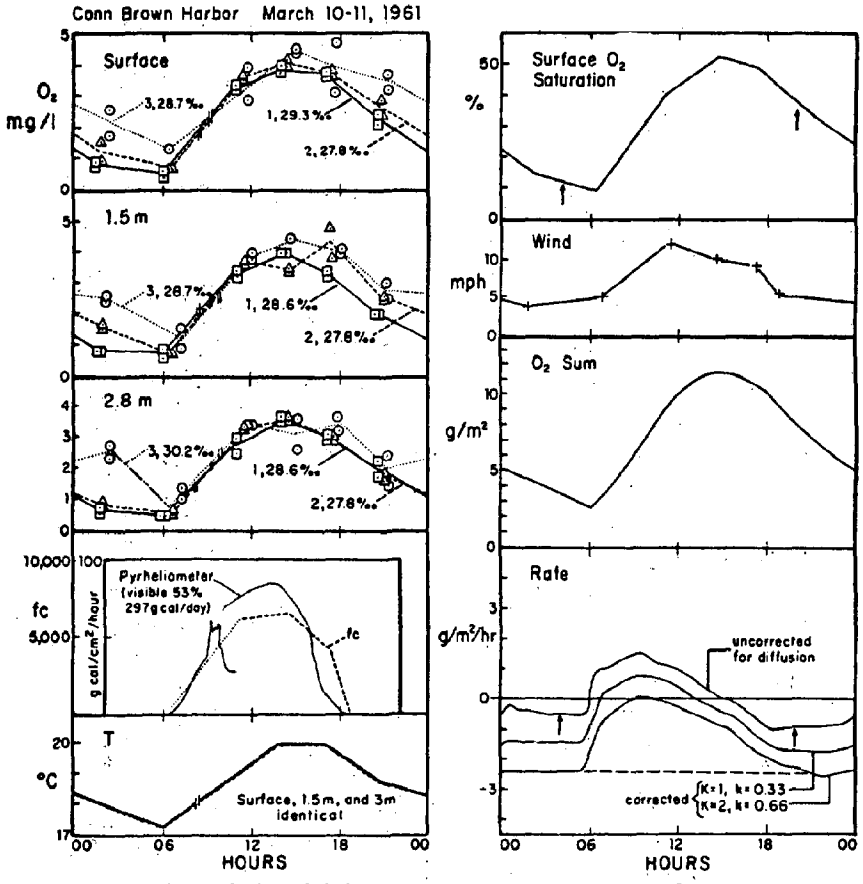


A. Conn Brown Harbor Aransas Pass, Texas (Odum, Cuzon, Beyers, and Allbaugh, 1963).

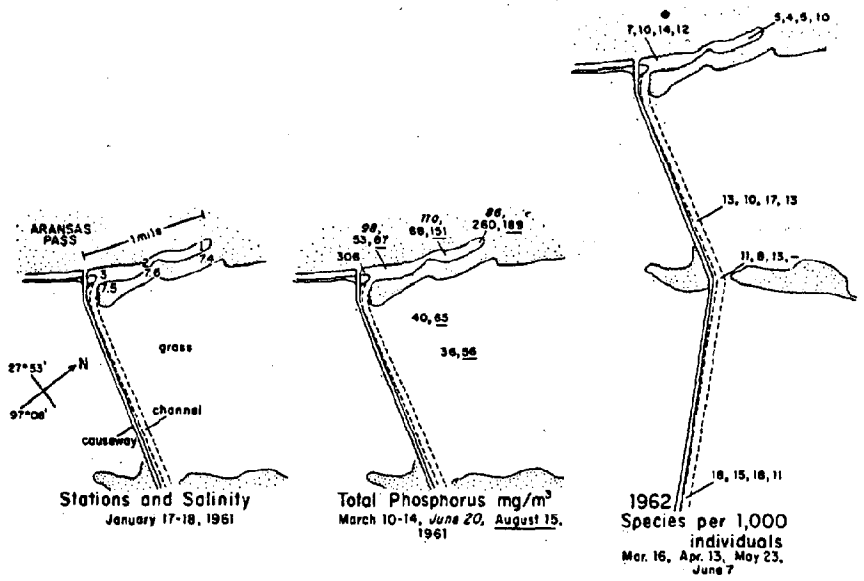


B. Distribution of variables in the Brownsville Ship Channel. Graphs and map drawn to horizontal scale.

Fig. 1. Small boat harbors receiving waste from seafood processing.



Diurnal record of wind, light, temperature and oxygen, in Conn Brown Harbor March 10–11, 1961. The sum of the 9 oxygen curves is plotted and used to determine the rate of change. Diffusion corrections for reaeration are made according to two assumptions using the percent saturation curve of the 3 surface curves.



Distribution of variables at stations in and near Conn Brown Harbor, Aransas Pass, Texas. Diagram on the right depicts channel through Aransas Pass jetties.

Fig. 2. Data from Conn Brown Harbor with sea food wastes (Odum, Cuzon, Beyers, and Allbaugh, 1963)

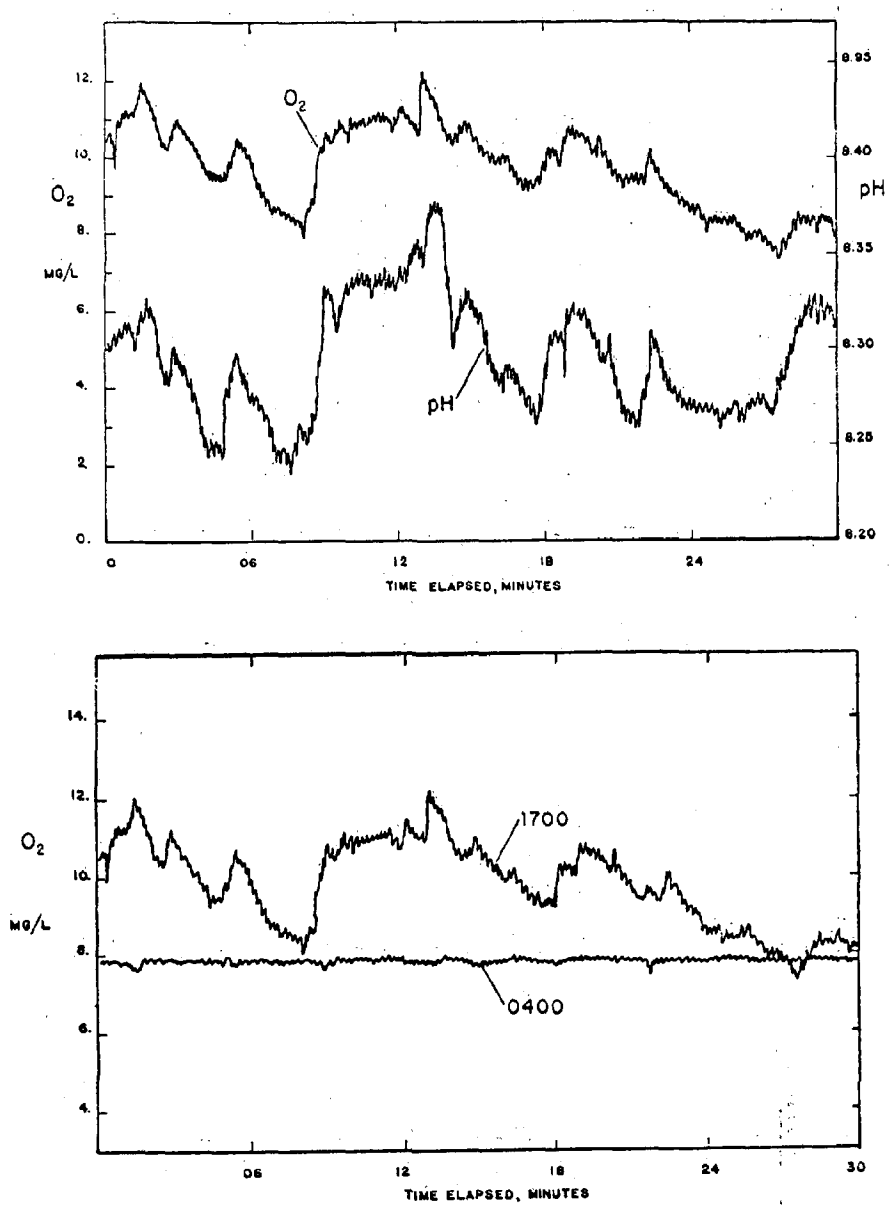


Fig. 3. Patchiness in oxygen and pH in a harbor with seafood waste. a) Dual c channel recorder trace of a Mancy-Okun electrode for oxygen and a Beckman model W pH meter, both operating on a pumped stream of water in amoving small boat in Conn Brown Harbor during the afternoon, Dec. 3, 1961. b) Comparison of recorder traces of oxygen with the Mancy-Okun electrode during the afternoon and later at night after vertical surface cooling had produced mixing (From Odum, Cuzon et al.1963; Figs. 11 and 12.)

A similar situation was found in the Brownsville shrimpboat channel which also receives waste from seafood processing. Here the waste is dispersed over a much longer channel than at Aransas Pass.

California Harbors Receiving Fish Cannery Waste

Reish (1957b) in Los Angeles Harbor (Fig. 4), in Alamitos Bay, in the lower San Gabriel River, and in Newport Bay, California found that very polluted bottoms were without animals but were surrounded by a zone dominated by a red annelid worm, Capitella capitata. The latter were in numbers as high as $6000/m^2$ in sediments with little else except fish scales (See shaded zones in Fig. 4). Oxygen concentration of 3.5 ppm was mentioned for this zone. Within a short distance of the worm zones, animal diversity reached 15 species per station. The author discusses the possibility of higher fish populations (characteristic plankton feeders) around a sewage outfall in the harbor, but his data suggest that the station near the cannery outfall may have even more fishes, of many types.

Tropical Harbor Receiving Cannery Wastes

Tuna cannery wastes discharge into Mayaguez Harbor on the western end of Puerto Rico along with some sewage wastes. Dredging of a channel has produced fine silts. Fig. 5 (Lowman, Phelps, Ting, Martin, Swift, and Escalera, 1967) shows circulation and relationship of islands and rivers. Ramsey (1967) compared Mayaguez Bay stations, in his study of fishes caught in 40 ft. otter trawls, with stations in Boqueron Bay further south, which he regarded as a control (Table 1). His trawl station 5 was a 30 minute drag in 30 to 35 feet depths along the harbor's northern shore over silt bottom possibly related to the ecosystem there receiving cannery wastes. Waters were clearer than further south near the sewage outfall (Station 4). In Table 1 the trawl station near the cannery outfalls had greatest number of fish and variety per trawl. Per fish caught, species variety was better in Boqueron Bay, a clear-water control area.

Cannery Discharge in Alaska

An operation in which organic enrichment is a potential (if not actual) problem is the Pacific salmon canning industry, particularly in Alaska, with sites in Juneau, Kenai, Sitka, Ketchikan and Kodiak. Salmon cannery waste constitutes about one-third of the fish's weight, and in Alaska amounts to more than 100 million pounds annually. Some is reduced to fish meal, a little is used as mink food, but a considerable proportion is dumped into the sea.

Clark and Groff (1967) evaluate seafood wastes from plants. Kodiak has a peak daily volume up to a half million pounds. Total wastes for Unalaska and Dutch Harbor are 20 million pounds. Very high tides in these regions tend to spread wastes rapidly. In localities with strong tidal flow the unwanted heads, tails, fins and viscera are dumped directly into the water, together with fish blood emanating from the butchering operations. Indiscriminate dumping is prohibited in many areas not blessed with good tidal flushing, so the refuse is carried to deeper water by barges. Much of the waste from the canneries is promptly taken by seagulls.

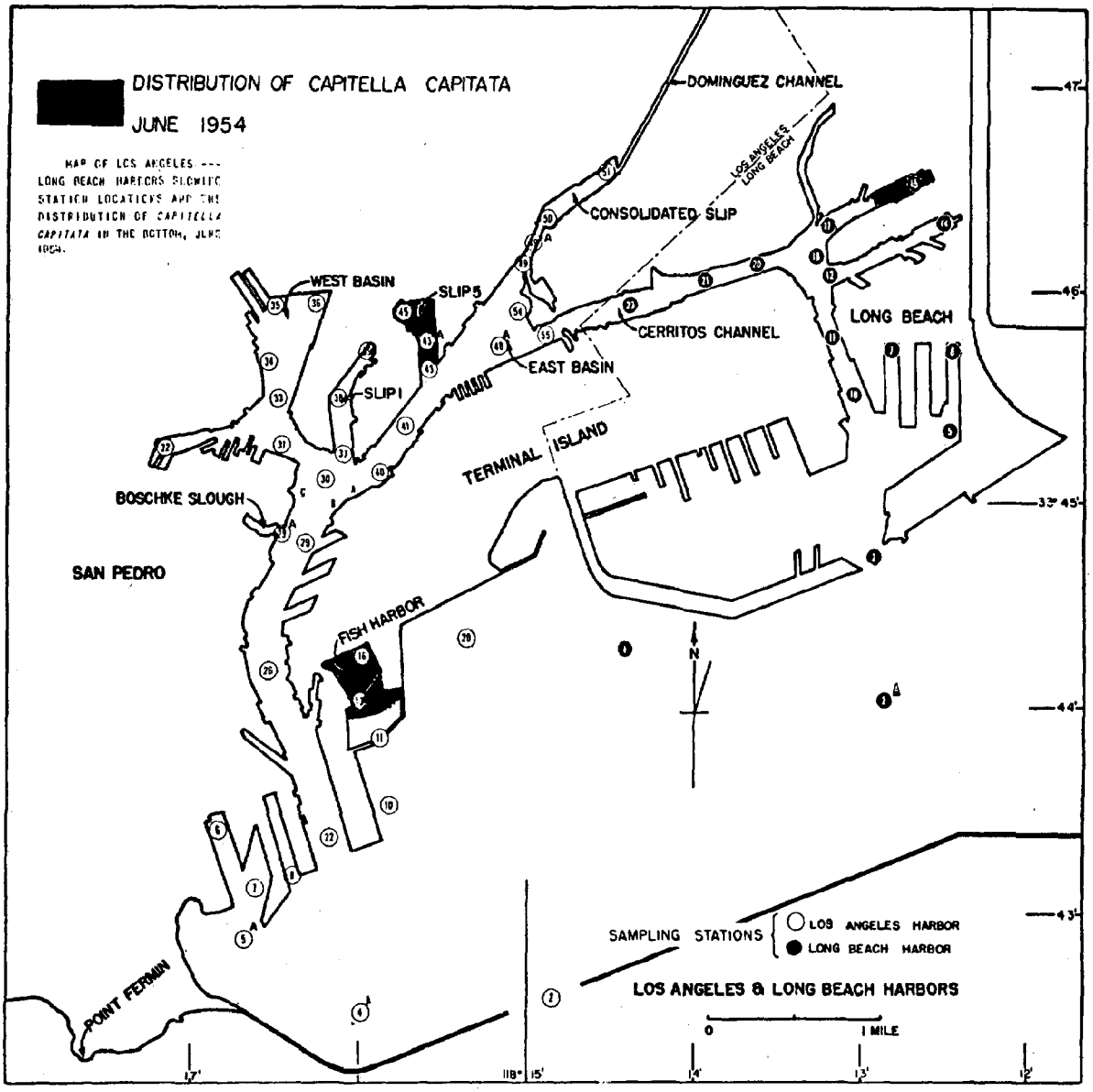
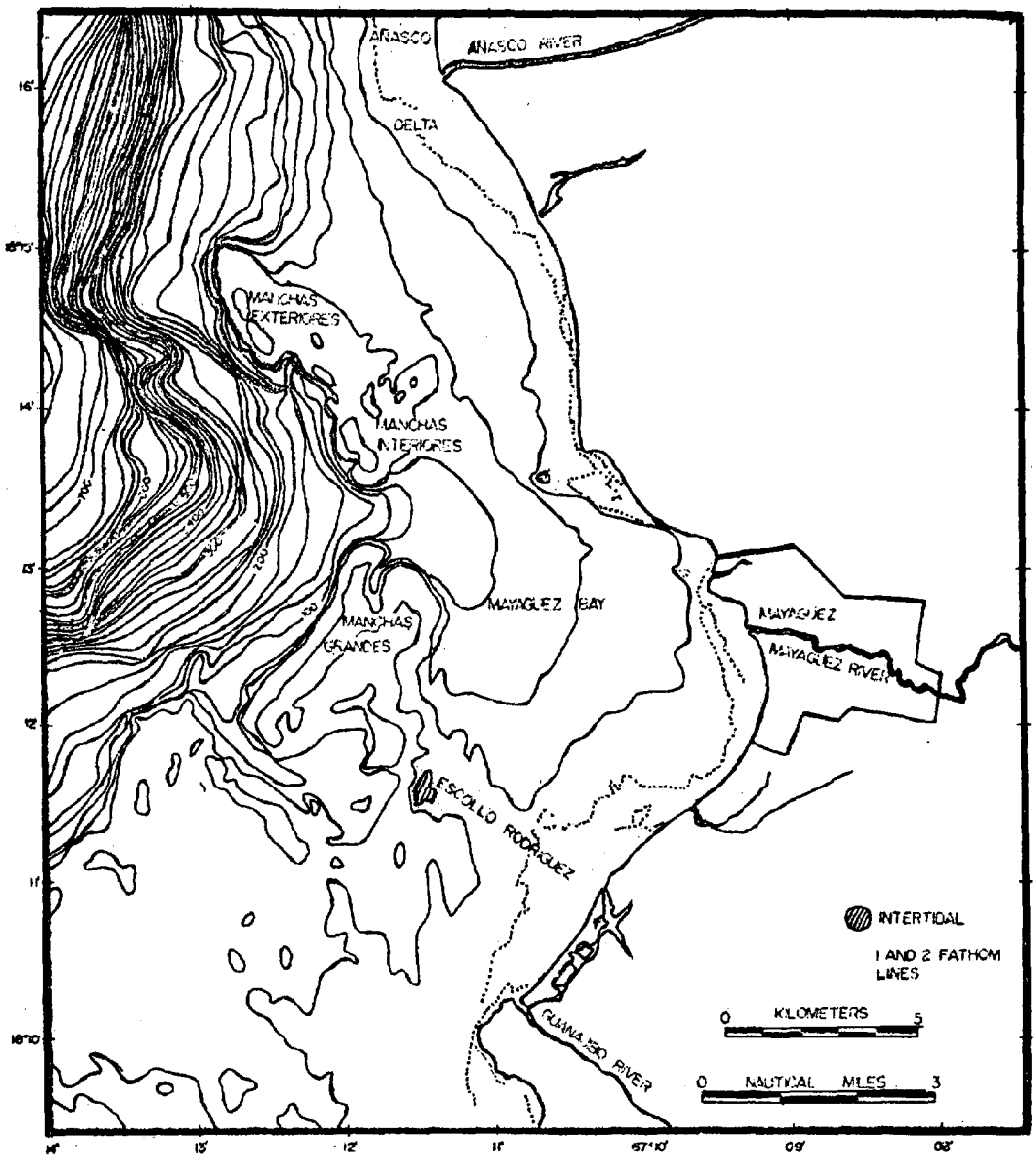


Fig. 4. Pacific harbor receiving seafood waste (shaded areas) and developing capitellid worm bottoms (From Reish 1957b; Fig. 1).



Bathymetry of Mayaguez Bay. From Coast and Geodetic Survey Chart 931.

Fig. 5. Mayaguez Harbor on the western end of Puerto Rico (Lowman, Phelps, Ting, Martin, Swift, and Escalera, 1967).

Table 1: Fishes in 30 minute trawls with 40 ft. otter trawl near Mayaquez, Puerto Rico (adapted from Ramsey, 1967)

Station	Situation	Mean Number of Species per Collection	Number of Fish per Collection
1	Boqueron Bay, clear water, silt, and Halophila	4.2	9.9
2	Boqueron Bay, like Sta. 1	3.3	6.4
3	Mayaguez Bay, south shore, coral rubble, red algae, Halophila ; clarity varying	4.6	10.4
4	Mayaguez Bay, within 100 feet of raw sewage outfall; turbid	3.2	14.9
5	Mayaguez Bay, north shore 100 yds. off docks, silt bottom, clarity varying (near tuna canneries)	4.0	16.8

Clark and Groff discuss a report by Charles L. Jensen reviewing volumes, characteristics, and possible uses of seafood plant wastes. He gives BOD values of 30,500 ppm (including considerable water) from one step in a Menhaden reduction process. Clark and Groff estimate that a probable BOD for whole fish wastes, without diluting waters, would be 150,000 to 200,000 ppm, or 0.15 to 0.20 pounds of oxygen per pound of waste.

EUTROPHICATION EFFECTS

In response to high nutrient levels the photosynthetic rate of the community will increase provided that sufficient light is available, and a phytoplankton bloom will result. The bloom of phytoplankton produces turbid conditions, resulting in decreased light penetration, and consequently is to a certain extent self-governing.

The abundance of nutrients available under eutrophic conditions favors organisms which expend less metabolic energy, and also confers an advantage on forms which are able to assimilate organic breakdown products at early stages in the decomposition cycle. For instance, an autotroph which can utilize dissolved ammonia as a nitrogen source will be able to compete more successfully than an organism which requires the stepwise oxidation of ammonia to nitrate. Consequently, eutrophic conditions are often characterized by low species diversity. Dominant species under such conditions are those which are able to utilize urea compounds or ammonia as a nitrogen source, or those in which total metabolism involves low expenditure of energy.

Indicators of Pollution by Seafood Wastes

Most of the indications given here are typical of eutrophic conditions irrespective of the primary cause of eutrophication.

1. Blooms of phytoplankton, often nannoplankton composed of small green flagellates, blue-green algae, and small diatoms.

2. Slight excess of respiration over photosynthesis since input of inorganic nutrients is usually accompanied by large quantities of organic matter.

3. Large amplitude of diurnal curves of oxygen saturation. Since both photosynthetic and respiratory rates are augmented by input of organic and inorganic material, normal diurnal fluctuations in oxygen content will be amplified.

4. Patchy water. Localized transient concentrations of nutrient-rich water cause high metabolic rates and possibly lowered oxygen content.

5. High concentrations of phosphorus. Derived initially from input of waste and maintained by rapid recycling through a system which has a high community metabolism.

6. Reduced species diversity in comparison with otherwise similar areas not receiving waste materials.

7. Stratification under certain conditions.

8. Some populations adapted to higher concentrations feeding on refuse.

Occurrence of Pollution from Seafood Wastes

Although the potential for pollution is apparent wherever fish are landed, such pollution rarely occurs because in most instances the fish waste is used for a variety of purposes and little is discarded. With one or two exceptions the major fisheries of the United States are concentrated at relatively few ports. Consequently, supplies of fish waste and unwanted fish are available in sufficiently large quantities to permit economic processing of this material as fertilizer or as a livestock food supplement.

In general it would appear that pollution may often be a problem when a local fishing industry is too small to support a reduction plant, when the unwanted parts of the fish are not suitable for reduction to fish meal, or when large quantities of blood and scraps of viscera are dumped into the water during washing processes in a large fishing port.

Almost all waste material and unclassified trash fish from the vast New England demersal fisheries is used by fish meal plants situated in most major ports. Similarly, waste from the halibut and associated fisheries of Washington and Oregon is reduced to meal or is used as mink food. Waste materials such as heads and viscera from sardine canning houses in California are also reduced to fish meal.

Species such as menhaden, Brevoortia spp., and alewife, Alosa sapidissima, are taken almost exclusively for reduction to fish meal. In common with most reduction plants, these fish meal factories convert the wet fish protein into a dry meal of low oil content. Fish oil is extracted and sold separately. The waste water (stickwater) from the cooking and extraction processes was formerly run off into the nearest harbor or bay and was a serious source of pollution. However, since this water has been found to be a rich source of B complex vitamins and other growth-promoting factors, it is now further processed by evaporation and is no longer released into the environment.

Pollution can still occur if water used to pump out the holds of vessels is returned to the bay, but it is almost always recirculated and its content of fish solubles and oils extracted by evaporation and centrifugation (Lee, 1963).

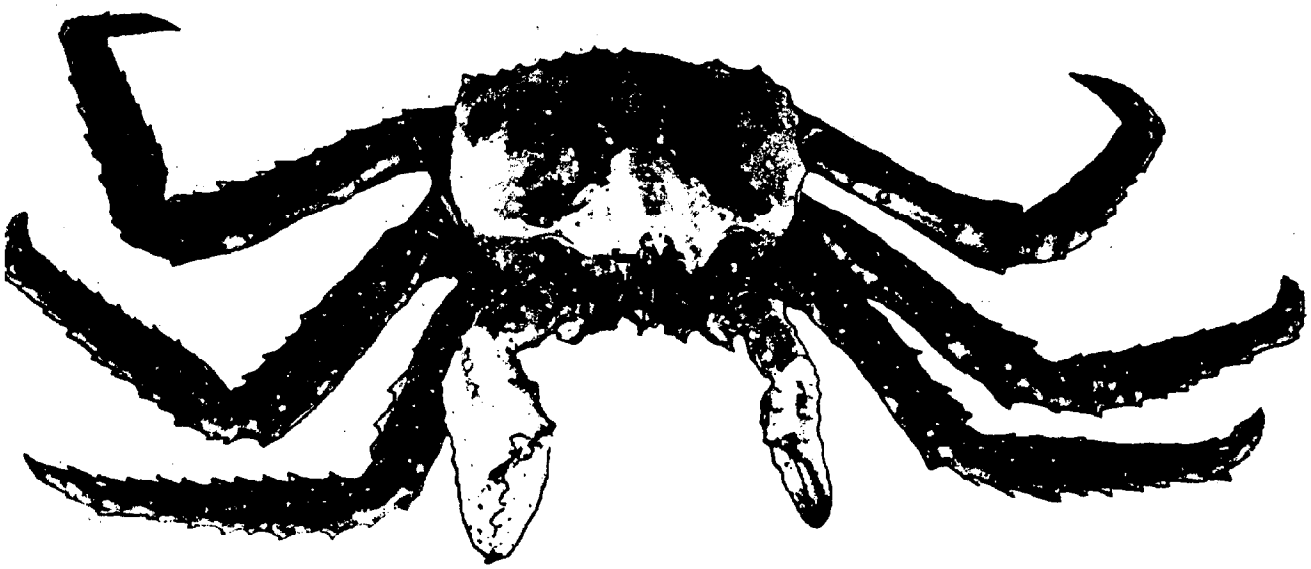
There is at present little utilization of wastes from certain invertebrates of commercial importance. Among these are materials such as scallop, clam, and oyster viscera, shells and viscera of crabs, and the head and thorax of shrimp. Some unusable materials from clam and scallop processing plants are used as a base for pet foods, but much is discarded. Over eighty percent of the fresh weight of the commercially important blue crab, Callinectes sapidus, is waste, and thirty to fifty percent of this is dried to produce a low protein flour. The remainder is discarded.

As shrimp are often 'headed' on board ship during actual fishing operations, pollution from shrimp packaging plants may not be a serious problem in most localities.

SUMMARY

The pollution of inshore waters by seafood waste may be relatively restricted in comparison with other forms of pollution. A detailed survey may indicate that intense eutrophication occurs only in areas where fish meal plants are absent. Fish populations may be attracted. Reduction plants are located at most major fishing ports unless the unwanted portion of the catch is unsuitable for reduction, or unless landings are scattered over numerous small ports.

Because of this relationship between seafood plants and the fish meal industry, the dumping of waste fish may increase when market demand for fish meal falls. Conversely an increased demand for fish meal may lead to a reduction in the amount of waste discarded.



Alaskan King Crab which contributes to sea food wastes
(Clark and Groff, 1967)

Chapter E-3A

SYSTEMS WITH PESTICIDE

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The indiscriminate use of insecticides in marshes, fields, and other places where the residues drain directly or indirectly into estuaries is a serious menace to the bottom fauna, and to the animals that feed upon the bottom fauna. In fact, insecticides affect all trophic levels. The report of Boudreaux, Strawn, and Gallas (1959) shows clearly the potential dangers and resulting diasters that can ensue from spraying from the air. Heptachlor sprayed on fields to control fire ants, reached adjacent irrigation canals and streams, killing many species of fish, either directly by contact with the insecticide, or through the destruction of crustaceans and insects used as food by the fishes. Cattle were prevented from coming in contact with water so deadly to fish, but there was a heavy toll of birds and other mammals. Two months later young fishes, abundant prior to the poisoning, were not found. There was 100% kill of fishes in shallow depressions in rice fields, and very severe effects in drainage ditches. The report deals primarily with the effect of heptachlor on fishes and points out that care must be used to keep it from streams. Only casual mention is made of the great mortality of birds, and of the insects that are the food of insectivorous birds.

Harrington and Bidlingmayer (1958) provide a study of an estuarine dieldrin kill. Some of their results are given in Tables 1-3. In 1958 Dr. Robert Lunz reported grave dangers to fish and shrimp and other wildlife of marshes and adjacent bodies of water arising from the program of the U. S. Department of Agriculture to control fire ants in southeastern states. He indicated a total loss of shrimp in experimental ponds after several low passes of a crop-dusting plane using benzine hexachloride, one of the milder insecticides. Benzine hexachloride when diluted one part to one million parts of water will kill fish and shrimp. Spraying for fire ants continued in coastal areas in the mid 1960's, without regard to the effect of the insecticide on animals other than ants. There appears to have been no survey of the fauna either before or after the spraying. Visits of biologists not involved in the spraying to one marsh in three consecutive years revealed practically 100% mortality of ribbed mussels in the marsh and of mud-fidder crabs along the banks of the drainage canals, only very few of the normally abundant salt marsh dragonflies, and a pronounced scarcity of other insects and birds. It is not known what happened to fishes and crabs in the streams and drainage ditches. See Table 4 for additional data by Chin on toxic effects of heptachlor and dieldrin.

A vivid example of what can happen when insecticides are liberated into waterways occurred in mid-July, 1968, in North Carolina. The pesticide Endrin, poured into a storm sewer, found its way through tributaries into the Northeast Cape Fear River. Thousands of fish in the river over a distance of 40 miles were killed (Durham Herald, July 19, 22, 26, 1968). At least twelve species, including several (eel, gizzard shad, striped bass) that are migrants upstream from the sea, were involved in the disaster. The company responsible

TABLE 1.—FISH IN 200 FEET OF THE EASTERN MARGINAL DITCH. KILLED BY THE MAY 4 APPLICATION OF DIELDRIN PELLETS

Species	Number	Pounds
Ten pounder or bigeye herring (*Elops saurus*)	1	—
Sea catfish (*Galeichthys felis*)	4	—
Sailfin molly (*Mollienesia latipinna*)	2	—
Striped mullet (*Mugil cephalus*)	86	17¾
Snook (*Centropomus undecimalis*)	7	1⅝
Irish pompano (*Diapterus olisthostomus*)	237	15⅛
Channel bass or red drum (*Sciaenops ocellata*)	1	—
Spot (*Leiostomus xanthurus*)	12	—
Sheepshead (*Archosargus probatocephalus*)	7	1⅞
Totals (9 species)	357	37

TABLE 2.—FISH IN WESTERNMOST 1,100 FEET OF A 5,600-FOOT DITCH. KILLED BY THE MAY 10 APPLICATION OF DIELDRIN PELLETS

Species	Number	Pounds
Sea catfish (*Galeichthys felis*)	1	—
Rainwater killifish (*Lucania parva*)	1	—
Marsh killifish (*Fundulus c. confluentus*)	1	—
Gulf killifish (*Fundulus g. grandis*)	635	—
Longnose killifish (*Fundulus similis*)	1	—
Broad killifish or variegated minnow (*Cyprinodon v. variegatus*)	214	—
Eastern mosquitofish or gambusia (*Gambusia affinis holbrooki*)	18	—
Sailfin molly (*Mollienesia latipinna*)	148	—
Striped mullet (*Mugil cephalus*)	635	104¾
White mullet (*Mugil curema*)	37	—
Irish pompano (*Diapterus olisthostomus*)	17	—
Spot (*Leiostomus xanthurus*)	26	—
Atlantic croaker (*Micropogon undulatus*)	8	—
Pinfish (*Lagodon rhomboides*)	13	—
Fat sleeper (*Dormitator maculatus*)	9	—
Emerald goby (*Erotelis smaragdus*)	6	—
Mapo (*Bathygobius soporator*)	6	—
Lyre goby (*Evorthodus lyricus*)	25	—
Esmeralda (*Gobionellus smaragdus*)	7	—
Spotfinned whiff (*Citharichthys spilopterus*)	8	—
Southern hogchoker (*Trinectes maculatus fasciatus*)	4	—
Toadfish (*Opsanus tau*)	4	—
Totals (22 species)	1,824	115¼

TABLE 3.—BIWEEKLY CENSUS ALONG 5,600 FEET OF MARSH DITCH AFTER A CATASTROPHIC MARSH-WIDE FISH KILL BY DIELDRIN

Weeks after Treatment	2	4		6		8	10		13	17
Species	Obs.	Obs.	Coll.[1]	Obs.	Coll.[2]	Obs.	Obs.	Coll.[2] Coll.[3]	Coll.[3]	Coll.
Tarpon atlanticus[4]	—	—	—	—	—	—	—	— —	—	17
Galeichthys felis	—	—	—	—	—	1	—	— —	—	—
Stongylura marina[5]	2	8	—	9	—	4	10	— —	—	—
Fundulus g. grandis	—	50	7	100–200	61	common	abundant	63 30	3	32
Fundulus c. confluentus	—	—	14	—	7	—	—	2 182	6	337
Fundulus similis	—	—	—	—	2	—	—	— —	1	—
Cyprinodon v. variegatus	13	800–1000	10	1000–2000	495	abundant	tens of thousands	15 9	2	47
Gambusia a. holbrooki	—	—	3	—	17	—	—	— 2	16	63
Mollienesia latipinna	—	—	—	—	24	—	—	3 —	64	—
Mugil cephalus	28	33	—	53	—	—	18	— —	—	—
Mugil curema	—	—	—	—	—	—	—	— —	1	—
Diapterus olisthostomus	3	7	—	—	—	—	—	— —	—	—
Sciaenops ocellata	—	—	—	1	—	—	—	— —	—	—
Dormitator maculatus	—	—	—	—	2	—	—	— —	—	—
Evorthodus lyricus	—	—	—	—	4	—	—	1 —	—	—
Sphoeroides spengleri[6]	1	4	—	7	—	26	—	— —	—	—
Fiddler Crabs	10	64	—	25	—	20	12	— —	—	—
Blue Crabs	1	none	—	none	—	none	none	— —	—	—
Shrimps	—	—	few	—	—	—	—	— —	—	—
Amphipods	—	—	few	—	—	—	—	— —	—	—
Nudibranchs and Snails	—	common	abundant	abundant	—	abundant	abundant	— —	—	—

[1] West end of ditch in algae. [2] West end of ditch. [3] East end of ditch. [4] Tarpon. [5] Atlantic needlefish. [6] Puffer.

(From Harrington and Bidlingmayer 1958; Tables 1, 2, and 3)

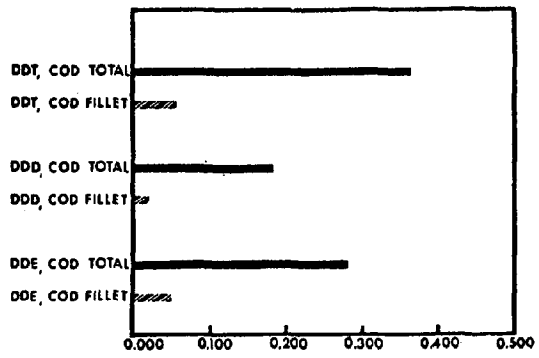
Table 4. Heptachlor and dieldrin concentrations toxic for a variety of shrimp and fish species (From Chin 1960).

Species	Heptachlor		Dieldrin	
	Avg. length	24-hr. TL_m	Avg. length	24-hr. TL_m
Brown shrimp	14 mm.	11 p.p.b.	13 mm.	15 p.p.b.
Brown shrimp	50	20	46	10
White shrimp	86	43	96	60
Spot croaker	82	25-50		
Sailfin molly	42	50-100	43	10-25
Broad killifish	42	143	35	<50
Diamond killifish	25 (approx.)	150		
Longnose killifish	77	250		
Striped mullet	117			20-50

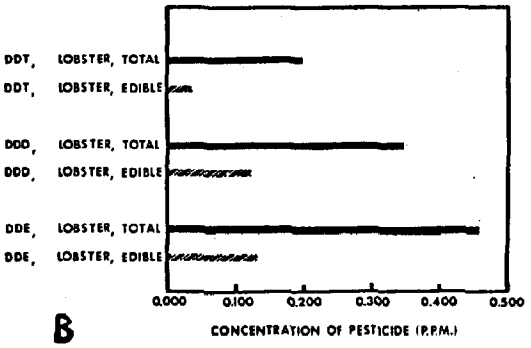
for pouring the Endrin in the storm sewer agreed to restock the river, but restocking is useless until the food for the fish has been restored also. The insecticide that kills fish also kills the numerous bottom invertebrates that serve as fish food. Since the breeding season of many of these invertebrates has passed, it could be many months before there would be an adequate food supply. As one wildlife protector stated, it will take from three to five years for the river's fish populations to return to normal levels. As this was a single event and the insecticide became more and more diluted as it moved down stream, there was probably little or no damage to fauna in the Cape Fear tidal estuary. Had it been a continuing operation, or even an annual event (as in spraying marshes and tidal pools to control ants and mosquitos), permanent destruction of fauna could have resulted. Had it occurred nearer to the tidal estuary, crustaceans, oysters, clams, nursery stocks of many sea-going fishes, as well as the bottom fauna that serves as food for many fishes (and people), could have been destroyed. In newspaper reports of the Endrin episode, only the death of fish was mentioned, but just as important was the destruction of fauna at each trophic level.

Apparently a few fishes can tolerate massive quantities of Endrin. The mosquito fish is one of these. Rosato and Ferguson (1968) found that Endrin-exposed mosquito fish, when fed to 11 species of vertebrates, including other fishes, reptiles, and birds, caused death among the predators in 95% of the cases. Although the mosquito fish are not food fish for humans, they are consumed by fish and other animals that are used as food for man. Ferguson (1967) also found that sunfish, after eating Endrin-exposed mosquito fish, succumbed, although they themselves had not been exposed directly. Large-mouth bass, sensitive to Endrin, are absent in heavily contaminated areas. Analysis of green sunfish that had eaten exposed mosquito fish showed high concentration of Endrin (11 - 26.5 ppm) in edible tissues. Ferguson, Ludke, and Murphy (1966) showed that there are indications of physiological toleration. An Endrin-tolerant fish may survive, but release enough Endrin to kill other more susceptible fish. If insecticides are stored in living fish, the danger to man is apparent. Fig. 1 shows differences in hazard for different tissues of the same organisms.

Another insecticide that has been a source of worry to estuarine biologists is DDT. As stated by Woodwell (1967), it is such pesticides as this that, once released into the environment, may enter biological cycles that in turn distribute them and concentrate them in animals to dangerous levels. DDT sprayed on water may pass through all of the various trophic levels. It may be taken up by plankton and detritus; the detritus-feeders and filter-feeders may absorb and retain the DDT; predatory fishes and birds feed on the detritus- and filter-feeders and absorb and store the DDT in their tissues. Thus the DDT accumulates. It is very stable, breaks down slowly in a natural environment, and soon becomes widely spread. Coastal fisheries are vulnerable to the end effects of insecticides because of their dissemination in drainage systems and final accumulation in estuaries. Cottam and Higgins (1946) summarized evidence that DDT, applied in adjacent marshes for the control of insects, was toxic to amphibians, birds, marine fishes, crustaceans, and shellfish resources. Holland, Coppage, and Butler (1966) have shown that in some species survivors of DDT exposure have an increased sensitivity to further exposure, and succumb to very low concentrations.



--Reduction of chlorinated pesticide residue present in cod (Gadus morhua) following filleting.



--Reduction of chlorinated pesticide residue present in lobster (Homarus americanus), following cleaning.

Fig 1. Chlorinated pesticide residue in cod (A) and lobster (B) and its reduction following filleting or cleaning (From Fish and Wildlife Service Report, Circular 262, Bureau of Commercial Fisheries 1967; Figs. 5 and 6).

Woodwell, Wurster, and Isaacson (1967, See Tables 5 and 6) reported that DDT residues in soil in a Long Island salt marsh (Great South Bay) averaged 13 lbs. per acre, with a maximum accumulation of 32 lbs. per acre, and that DDT concentrations increased with each trophic level from 0.04 ppm in plankton to 75 ppm in gulls. The greatest concentration occurred in scavenger and carnivorous fish and birds. The birds had 10 to 100 times that of fish. DDT has a cumulative effect in bottom feeders: shrimp, mud snails, hard clams, toadfish, and flounders.

Lowe (1965) found that juvenile blue crabs, reared in flowing sea water containing sublethal dosages of DDT, fed, molted, and grew for nine months in water containing DDT at a concentration of 0.25 part per billion, but could survive only a few days in concentrations above 0.5 ppb. Only 22% of all his experimental crabs were able to survive the full nine months of his study. Butler (1966a)showed that shell deposition (growth) in oysters is greatly inhibited by continuous exposure to DDT (10 parts per billion) (Fig. 2). Growth is completely stopped in dilutions as low as 0.1 part per million. DDT is stored in various tissues -- intestinal tract, digestive gland, gonads, and eggs. (See Tables 7--10) Ames (1966) related DDT levels in Ospreys to nesting success (Fig. 3). Holden gave pesticide levels in fresh water and marine fish (Table 11 and 12). Bailey and Hannun (1967) reported high levels in Suisun Bay California and the landlocked Salton Sea (Table13). Pesticide concentration in sediment increased as grain size diminished (Figs. 4 and 5). Johnson (Figs. 6, 7, and 8) showed selective distribution of pesticides in food chains with bacteria and animals. George and Frear (1966) analyzed tissues of the skua from the Antarctic and found evidence of DDT at this remote location (Table 14). Koeman and Van Genderen (1966) related high levels of pesticide in tissues to death of animals in a coastal habitat (Table 15 and16). Kerswill et al (1960) related decreased numbers of young salmon to forest spraying with DDT (Fig. 9). Mortality of fish in a tidal marsh ditch following application of DDT is shown in Fig. 10. Large differences in species sensitivity are suggested by the data in Table 17 . The general widespread distributions of pesticide in the United States is suggested by the river basin maps in Fig. 11. Tables 18-22 show pesticides in estuarine animals in Massachusetts from upstream agricultural use.

Table 5 DDT residues (From Woodwell, Wurster, and Isaacson 1967; Table 1):

DDT residues (DDT + DDE + DDD) (*1*) in Carmans River marsh and in the bottom mud of Great South Bay, N.Y., August 1966. Each sample was a composite of six subsamples, taken to the depths indicated.

Zone	Sample No.	Depth (cm)		Total residues Lb/acre	Total residues Kg/ha
Spartina mat	1	0–20		2.69	3.01
	2	0–40		9.23	10.3
	3	0–20		7.86	8.81
	4	0–40		32.6	36.5
			Mean	13.1	14.7
Drainage ditch	1	0–20		4.63	5.19
	2	0–40		1.10	1.23
			Mean	2.87	3.21
Bay bottom (submerged)		0–40		0.28	0.31

Table 6 DDT residues in whole organisms (From Woodwell, Wurster, and Isaacson 1967; Table 2).

DDT residues (DDT+DDE+DDD) (*1*) in samples from Carmans River estuary and vicinity, Long Island, N.Y., in parts per million wet weight of the whole organism, with the proportions of DDT, DDE, and DDD expressed as a percentage of the total. Letters in parentheses designate replicate samples.

Sample	DDT residues (ppm)	Percent of residue as DDT	DDE	DDD
Water*	0.00005			
Plankton, mostly zooplankton	.040	25	75	Trace
Cladophora gracilis	.083	56	28	16
Shrimp†	.16	16	58	26
Opsanus tau, oyster toadfish (immature)†	.17	None	100	Trace
Menidia menidia, Atlantic silverside†	.23	17	48	35
Crickets†	.23	62	19	19
Nassarius obsoletus, mud snail†	.26	18	39	43
Gasterosteus aculeatus, threespine stickleback†	.26	24	51	25
Anguilla rostrata, American eel (immature)†	.28	29	43	28
Flying insects, mostly Diptera†	.30	16	44	40
Spartina patens, shoots	.33	58	26	16
Mercenaria mercenaria, hard clam†	.42	71	17	12
Cyprinodon variegatus, sheepshead minnow†	.94	12	20	68
Anas rubripes, black duck	1.07	43	46	11
Fundulus heteroclitus, ummichog†	1.24	58	18	24
Paralichthys dentatus, summer flounder‡	1.28	28	44	28
Esox niger, chain pickerel	1.33	34	26	40
Larus argentatus, herring gull, brain (d)	1.48	24	61	15
Strongylura marina, Atlantic needlefish	2.07	21	28	51
Spartina patens, roots	2.80	31	57	12
Sterna hirundo, common tern (a)	3.15	17	67	16
Sterna hirundo, common tern (b)	3.42	21	58	21
Butorides virescens, green heron (a) (immature, found dead)	3.51	20	57	23
Larus argentatus, herring gull (immature) (a)	3.52	18	73	9
Butorides virescens, green heron (b)	3.57	8	70	22
Larus argentatus, herring gull, brain§ (e)	4.56	22	67	11
Sterna albifrons, least tern (a)	4.75	14	71	15
Sterna hirundo, common tern (c)	5.17	17	55	28
Larus argentatus, herring gull (immature) (b)	5.43	18	71	11
Larus argentatus, herring gull (immature) (c)	5.53	25	62	13
Sterna albifrons, least tern (b)	6.40	17	68	15
Sterna hirundo, common tern (five abandoned eggs)	7.13	23	50	27
Larus argentatus, herring gull (d)	7.53	19	70	11
Larus argentatus, herring gull§ (e)	9.60	22	71	7
Pandion haliaetus, osprey (one abandoned egg)‖	13.8	15	64	21
Larus argentatus, herring gull (f)	18.5	30	56	14
Mergus serrator, red-breasted merganser (1964)‡	22.8	28	65	7
Phalacrocorax auritus, double-crested cormorant (immature)	26.4	12	75	13
Larus delawarensis, ring-billed gull (immature)	75.5	15	71	14

* Estimated from Weaver *et al.* (*4*). † Composite sample of more than one individual. ‡ From Captree Island, 20 miles (32 km) WSW of study area. § Found moribund and emaciated, north shore of Long Island. ‖ From Gardiners Island, Long Island.

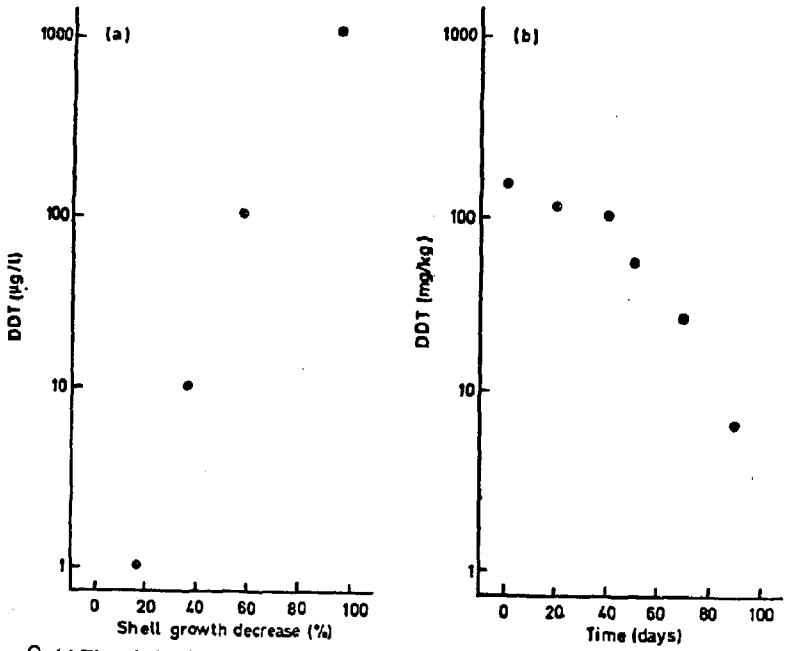
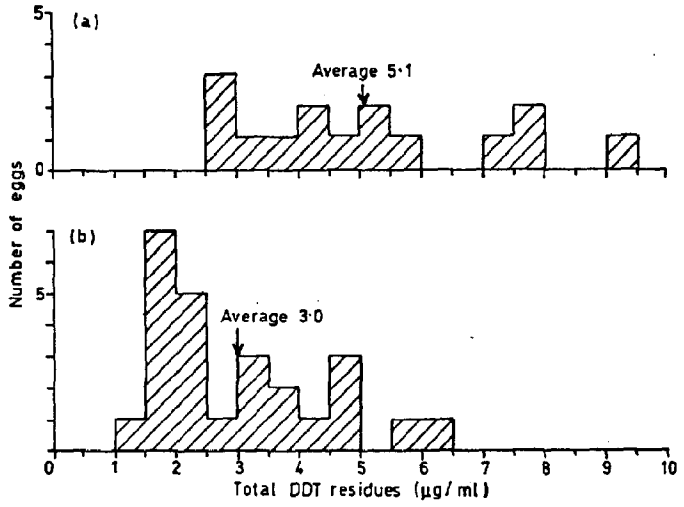


Fig. 2, (a) The relation between the environmental concentration of DDT and the percentage decrease in shell deposition in oysters in 96 h. (b) The relation between an initial body residue in the oyster of 151 ppm and the time required to flush it out in an unpolluted environment. Scales are semilogarithmic.

(From Butler 1966a; Fig 1).



Comparison of residue levels in eggs of Ospreys from (a) Connecticut, and (b) Maryland, in 1963.

Fig. 3. DDT residues in osprey eggs from Connecticut and Maryland (From Ames 1966; Table 1).

Table 7. DDT accumulation by molluscs (From Butler 1966a; Table 1)

Accumulation and retention of DDT by molluscs exposed for 7 days to 1·0 µg/l in flowing sea water

Mollusc	Residue (ppm) After 7 days exposure	After 15 days flushing	After 30 days flushing
Brachidontes recurvus, Hooked Mussel	24	–	–
Crassostrea virginica, Eastern Oyster	26	2·5	1·0
C. gigas, Pacific Oyster	20	16·0	–
Ostrea edulis, European Oyster	15	8·0	4·0
O. equestris, Crested Oyster	23	5·0	–
Mercenaria mercenaria, Northern Quahog	6	0·5	–

Table 8. DDT accumulation in oysters and their gametes (From Butler 1966b; Table 2).

—Accumulation of DDT and its metabolites in oysters and oyster gametes following 12 days exposure to 1.0 part per billion of DDT

Sample	Sum of DDT and metabolites (parts per million)
Experimental	
Whole oysters	
10 males	20
10 females	14
12 individuals, unsexed	5–23 Average 10
Gametes	
eggs—5.1 milliliters	25
sperm—0.3 milliliters	9
Control	
Whole oysters	
15 unsexed	0.06
Gametes	
eggs and sperm—1.4 milliliters	Not detectable

Table 9. Pesticide deposition in oyster shell (From Butler 1966b; Table 4).

—Variations in shell deposition in oysters exposed to common pesticides
[Numbers represent average percent decrease or (+) increase in rate of deposition as compared to controls after 96 hours]

Pesticide	Concentration (parts per million)				
	1.0	0.1	0.01	0.001	0.0001
Aldrin	100	86	43	36	11
Endrin	70	40	20	0	
DDT		100	52	0	
DEF	100	50	0		
Baytex	91	+4	0	+14	
Dibrom	76	12	+15	+20	

Table 10. Comparison of pesticide toxicity (From Butler 1966b; Table 3).

–Relative toxicity of common types of pesticides to estuarine fauna compared to herbicides rated as unity

Pesticide type	Plankton	Shrimp	Crab	Oyster	Fish
Herbicide	1	1	1	1	1
Insecticide					
Organophosphorus compounds	½	1,000	800	1	2
Polychlorinated hydrocarbon compounds	3	300	100	100	500

Table 11. Concentrations of pesticides in fish (From Holden 1966; Table 5)

Table 5. *Pesticide levels in various salmonids*

		Range of concentration (ppm)		
		Gills	Muscle	Liver
Salmon parr* (Highland area)	Dieldrin	–	0·01–0·06	–
	DDE	–	0·01–0·06	–
Brown Trout† (arable area)	Dieldrin	0·02–0·10	0·01–0·03	0·01–0·09
	DDE	0·03–1·05	0·10–0·26	0·11–0·80

* Eighteen fish in sample.
† Nine fish in sample.

Table 12. Concentration of pesticides in sea trout (From Holden 1966; Table 6).

Table 6. *Pesticide levels in Sea Trout* from estuary*

	Range of concentration (ppm)				
	Gills	Muscle	Liver	Spleen	Pyloric caeca
Dieldrin	0·02–0·08	0·01–0·03	0·01–0·05	0·01–0·12	0·01–0·10
DDE	0·02–0·07	0·01–0·09	0·03–0·09	0·02–0·32	0·01–0·13
TDE	0·01–0·04	0·01–0·03	0·01–0·04	0·02–0·35	0·01–0·07
DDT	0·01–0·02	0·01–0·05	0·01–0·10	0·03–0·35	0·01–0·17

* Six fish in sample.

Table 13. Pesticide concentrations in animals from Suisun Bay and the Salton Sea (From Bailey and Hannum 1967; Table 4)

—PESTICIDE CONCENTRATIONS IN AQUATIC ORGANISMS SAMPLED IN SUISUN BAY AND THE SALTON SEA, CALIFORNIA

Sample Organism	Age, in years	Crude Fat, in percentage	Concentration, in micrograms per liter					
			DDT/DDD	DDE	Toxaphene	Lindane	Dieldrin	Total[a]
Net Plankton		0.6	15			10	.5	30
Shrimp	1	2.0	50	5			5	60
Anchovy Fry	1	2.1	135	45	140		10	330
Anchovy	2-3	4.8	300	200	800		40	1,340
Pond Smelt	1-2	8.7	470	100	400	50	25	1,045
Starry Flounder	2	3.4	250	50	400		10	710
American Shad	1	4.5	430	130	300	25	35	920
Striped Bass								
Flesh 1	2	0.9	100	170	200		24	494
Flesh 2	3-4	4.7	360	160	300		210	1,050
Entrail Fat 2	3-4	69	10,300	4,000	8,000		280	22,580
Carp								
Flesh	4-6	7.5	1,600	750	500	50	50	2,950
Eggs		3.9	200	150			14	364
Corvina[b]								
12 inch	1	3.0	18	9	9			36
17.5 inch	2	1.4	100	40			7	147
18 inch	3	1.1	480	220	100		40	840
20 inch	3	2.3	320	140	100		8	568

[a] Total derived from summation of identified compounds.
[b] Salton Sea samples consisted only of Corvina.

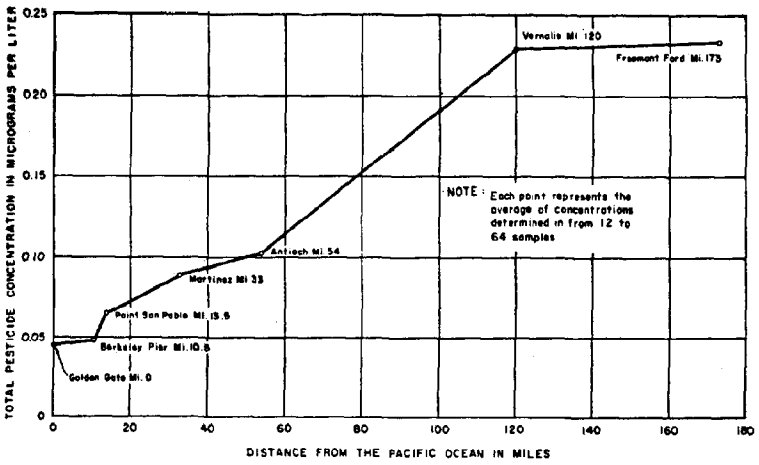
Table 14. Concentrations of pesticide residues in the Skua (from George and Frear 1966; Table 3).

Data on age, sex, date and place of capture, tissue analysed, and residue if any in parts per million (ppm) for the Skua (Catharacta skua maccormicki)

Field collection number	Age	Sex	Date of capture (1965)	Place of capture	Tissue(s) analysed	Wet weight ppm residue DDT	DDE
43	A	F	4 January	Cape Roberts	Pectoral muscle	0·04	0·06
					Viscera	0·03	0·04
44	A	F	4 January	Cape Roberts	Heart	0·07	0·05
					Liver	N.D.	0·05
					Pancreas	N.D.	0·05
					Pectoral muscle	N.D.	0·03
					Kidney	0·02	0·02
					Lungs	0·10	0·02
45	A	M	4 January	Cape Roberts	Pectoral muscle	0·01	0·06
					Heart	0·11	0·05
					Liver	0·02	0·05
					Pancreas	N.D.	0·05
					Lungs	0·13	0·04
					Kidney	0·03	0·02
46	A	F	4 January	Cape Roberts	Heart	N.D.	0·26
					Gut	0·01	0·05
					Brain	N.D.	0·04
					Lung	N.D.	0·01
					Liver	N.D.	N.D.
					Ovary	N.D.	N.D.
					Pectoral muscle	N.D.	N.D.
47	A	F	4 January	Cape Roberts	Lung	N.D.	0·18
					Ovary	N.D.	0·09
					Heart	0·28	0·07
					Pancreas	N.D.	0·07
					Liver	0·06	0·02
					Kidney	N.D.	0·01

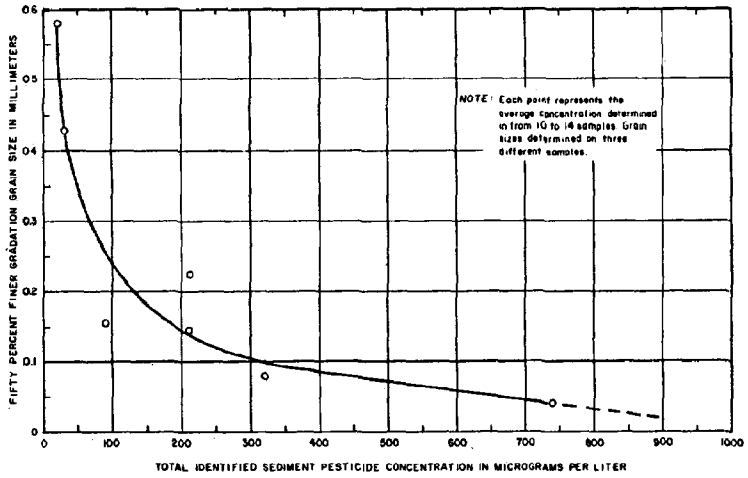
Table 14 (continued). (From George and Frear 1966; Table 3).

Field collection number	Age	Sex	Date of capture (1965)	Place of capture	Tissue(s) analysed	Wet weight ppm residue DDT	DDE
51	C		6 January	Dailey Island	Whole chick	N.D.	0·03
52	A	F	6 January	Dailey Island	Lung	N.D.	0·17
					Liver	N.D.	0·13
					Pancreas	0·37	0·10
					Kidney	0·05	0·04
					Heart	N.D.	0·03
					Pectoral muscle	0·02	0·03
53	A	F	6 January	Dailey Island	Pectoral muscle	0·11	0·08
					Heart	0·27	0·05
					Kidney	0·09	0·04
					Liver	0·03	0·01
					Lung	0·07	0·01
54	A	M	6 January	Dailey Island	Pancreas	0·12	0·11
					Pectoral muscle	0·04	0·08
					Liver	N.D.	0·06
					Kidney	N.D.	0·04
					Heart	0·07	0·03
					Lungs	0·10	0·03
					Testes	N.D.	0·03
55	A	M	6 January	Dailey Island	Gonad	N.D.	0·08
					Lung	0·27	0·05
					Heart	0·16	0·03
					Kidney	N.D.	0·03
					Pectoral muscle	0·07	0·03
					Liver	0·12	0·01
56	A	M	6 January	Dailey Island	Heart	0·68	0·38
					Pancreas	1·1	0·36
					Testes	1·3	0·30
					Kidney	0·12	0·10
					Pectoral muscle	0·07	0·05
					Liver	0·06	0·03
57	A	M	7 January	Hut Point McMurdo	Kidney	N.D.	2·8
					Lung	N.D.	1·5
					Pancreas	N.D.	1·2
					Liver	N.D.	0·73
					Heart	N.D.	0·15
					Testes	0·30	0·10
58	A	M	8 January	Cape Crozier	Kidney	0·13	0·46
					Pectoral muscle	N.D.	0·40
					Heart	N.D.	0·38
					Lungs	0·49	0·31
					Pancreas	N.D.	0·09
					Liver	N.D.	0·07
					Testes	N.D.	0·05
59	A	F	4 January	Cape Roberts	Pancreas	0·31	0·34
					Pectoral muscle	0·15	0·32
					Ovary	N.D.	0·26
					Heart	0·15	0·19
					Kidney	0·07	0·09
					Liver	0·32	0·03
60	A	M	4 January	Cape Roberts	Kidney	0·67	1·2
					Liver	0·15	1·1
					Pancreas	0·25	0·32
					Heart	0·11	0·22



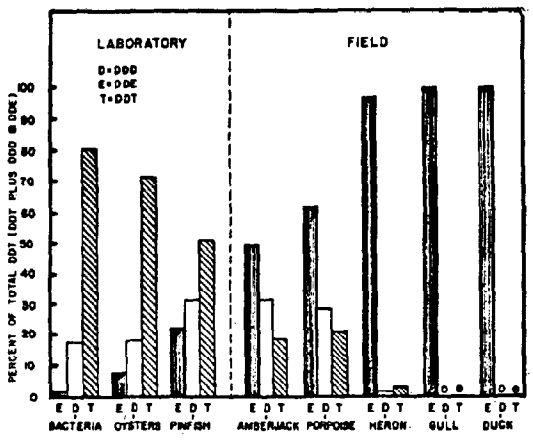
-LONGITUDINAL VARIATION IN PESTICIDE CONCENTRATIONS IN SURFACE WATERS

Fig. 4. Concentrations of pesticides in surface waters in California (From Bailey and Hannum 1967; Fig 2).



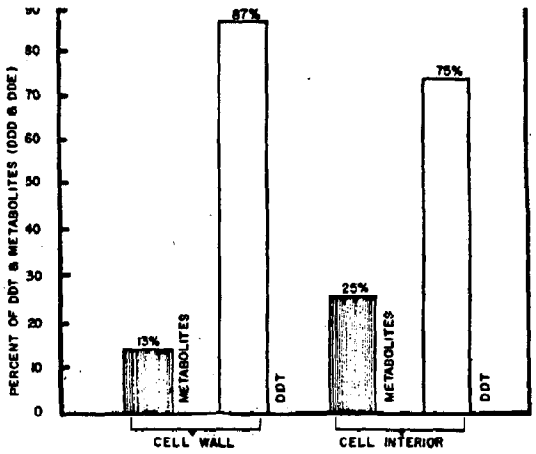
-PESTICIDE CONCENTRATION IN SEDIMENTS OF VARIOUS GRAIN SIZE

Fig 5. Concentration of pesticides in sediments (From Bailey and Hannum 1967; Fig 4).



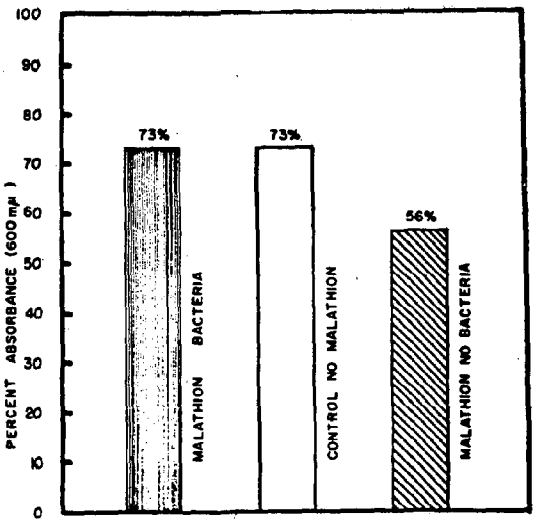
--Distribution of DDT and its metabolites, DDD and DDE, in representative organisms of a food chain.

Fig 6. DDT distribution in a food chain (From Johnson 1967; Fig 9).



Comparison of the distribution of DDT and its metabolites in the cell wall and cell interior of a marine bacterium (Pseudomonas piscicida).

Fig 7. DDT distribution in a marine bacterium (from Johnson 1967; Fig 8).



--Relative growth of algae, illustrating breakdown of malathion by a marine bacterium, Pseudomonas piscicida. Density of algal cells is determined by the amount of absorbance in a spectrophotometer.

Fig 8. Breakdown of malathion by a marine bacterium (From Johnson 1967; Fig. 10).

Table 15. Pesticide residues in birds (From Koeman and Van Genderen 1966; Table 1)*Residues of chlorinated hydrocarbon pesticides in birds found dead or dying in a coastal habitat*

Species	Region	Tissue	Dieldrin	Endrin	*pp*'DDE
			(parts per million)		
Spoonbill* (*Platalea leucorodia*)	Texel	Liver	5·8	2·0	0·1
Spoonbill*	Texel	Liver	6·1	3·0	0·9
		Breast muscle	1·8	0·4	
Spoonbill*	Texel	Liver	6·0	0·6	0·1
		Breast muscle	2·6	0·5	
Oystercatcher* (*Haematopus ostralegus*)	Texel	Liver	9·1	Trace	1·6
Oystercatcher*	Texel	Liver	9·5	0·3	1·6
Sandwich Tern* (*Sterna sandvicensis*)	Texel	Liver	4·6	0·9	0·2
Sandwich Tern*	Texel	Liver	4·2	0·4	0·5
Common Tern* (*Sterna hirundo*)	Texel	Liver	9·1	3·0	6·0
Common Tern	Goeree	Liver	5·8	Present	0·1
		Breast muscle†	2·2	0·9	0·4
		Kidney	4·5	2·6	0·2
		Brain	2·0	1·2	
Common Tern	Texel	Liver	1·8	0·5	Trace
		Breast muscle	5·6	3·9	0·7
		Kidney	5·9	7·9	0·2
		Brain	1·5	1·5	
Black-headed Gull (*Larus ridibundus*)	Den Helder	Liver	Trace		0·5
		Breast muscle			0·2
		Brain			0·3
Black-headed Gull	Den Helder	Liver			0·8
		Breast muscle			0·2
		Brain			0·2
Eiderduck* (*Somateria mollissima*)	Texel	Liver	2·2		0·1
Shelduck (*Tadorna tadorna*)	Texel	Liver	0·7		1·05
		Breast muscle	0·1		0·2

* Specimen submitted to gross pathological and parasitological examination.
† Results of Tunstall Laboratory: Dieldrin, 1·4 ppm; Endrin, 0·4 ppm; *pp*'DDE, 0·2 ppm.

Table 16. Pesticide residues in seals (From Koeman and Van Genderen 1966; (Table 2).

Residues of chlorinated hydrocarbon pesticides in seals found dead or dying in a coastal habitat

Species	Region	Tissue or fat	Dieldrin	*pp*'DDT	*pp*'DDE	*pp*'DDD
			(parts per million)			
Harbour Seal* (*Phoca vitulina*)	Wadden Sea	Subcutaneous fat	2·3	9·8	14·0	3·6
Harbour Seal*	Wadden Sea	Subcutaneous fat	1·4	3·5	5·4	0·7
		Liver	0·07		0·4	
Harbour Seal*	Texel	Subcutaneous fat	0·3	3·7	7·4	1·0
		Liver			0·1	

* Specimen submitted to pathological and parasitological examination.

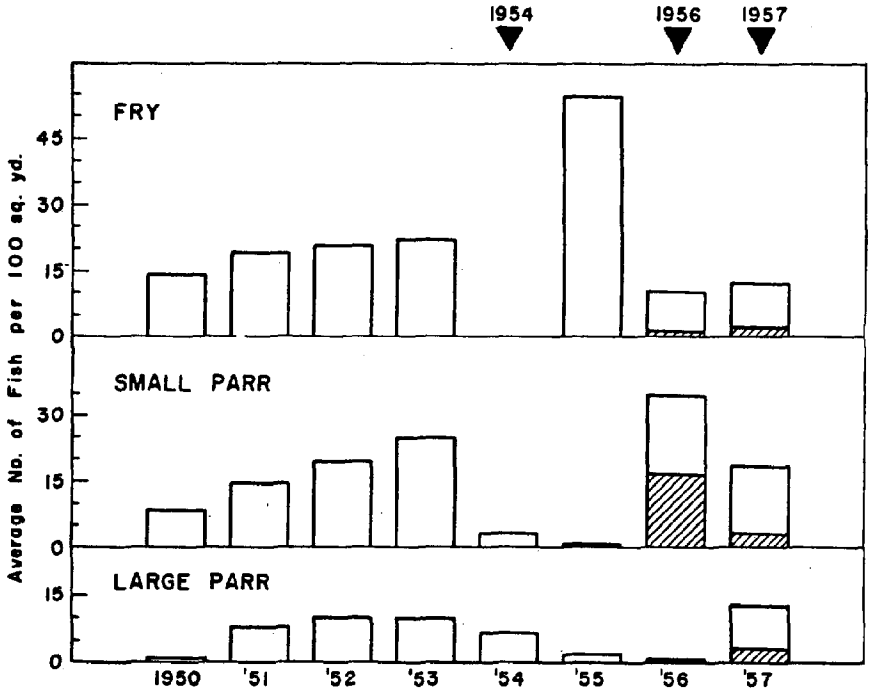
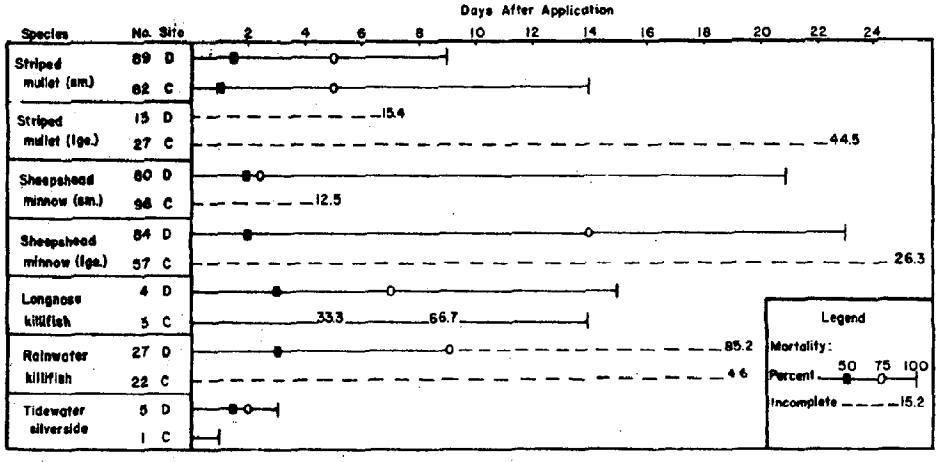


Fig. 9. Mortality of young salmon in years of forest DDT spraying (1954, 1956, 1957) (From Kerswill et al. 1960).



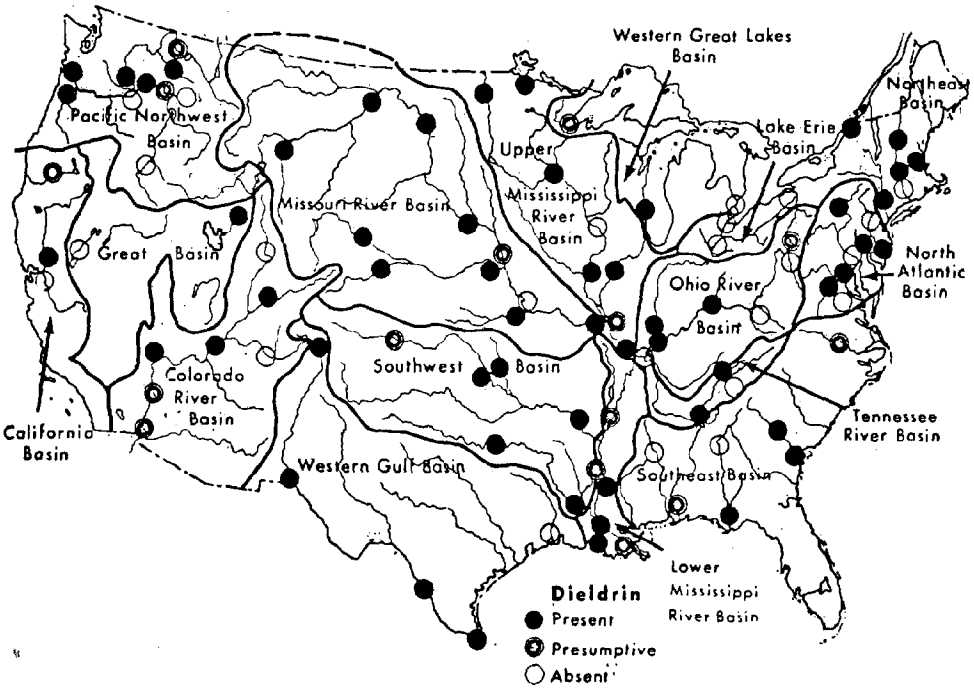
-Cumulative per cent mortality of fish held at holding sites C and D

Fig. 10. Mortality of fish following application of DDT to a tidal marsh ditch (From Croker and Wilson, 1965).

Table 17. Toxicity of organic phosphorus insecticides to minnows houseflies, and white rats (From Henderson and Pickering 1958)

Compound	Fathead minnows 96 hr TL_m ppm.	M. domestica Topical LD_{50} microgram/ gram	White rats Oral LD_{50} milligrams/ kilogram
EPN	0.20	1.9	12-40
Para-oxon	0.33	0.5	3-3.5
Parathion	1.4	0.9	13.0
TEPP	1.7	-	2.0
Chlorothion	3.2	16.5	880.0
Systox	3.6	-	6.2
Methyl parathion	8.3	1.0	14-42
Malathion	12.5	28.0	1375.0
OMPA	121.0	-	10.0
Dipterex	180.0	-	630.0

Occurrence of dieldrin in major river basins, September 1964



Occurence of endrin in major river basins, September 1964

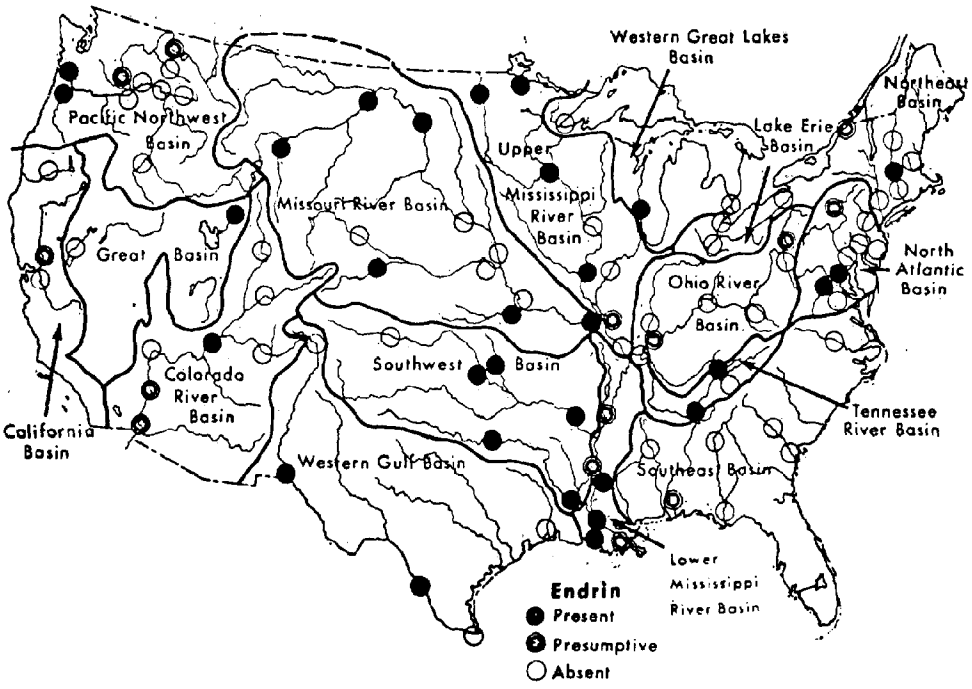


Fig. 11. Occurrence of dieldrin and endrin in major U.S. river basins (From Weaver et al. 1965; Figs. 1 and 2).

Table 18. *Pesticide Analysis of Finfish Samples Taken at Carr's Island, Salisbury, Merrimack River Estuary, 1964.*

Date Collected	Source of Sample	Dieldrin** or DDE (ppm) (LBW)*	DDT (ppm) (LBW)
July 7	Intestine (*Alosa pseudoharengus*)	10.000	10.000
July 7	Body Muscle (*Alosa pseudoharengus*)	5.900	5.800
July 7	Intestine (*Alosa aestivalis*)	0.260	0.800
July 7	Body Muscle (*Alosa aestivalis*)	4.500	3.400

*(LBW): Live Body Weight.

**Dieldrin—DDE: Not separable by method used.

Table 19

Pesticide Analysis of Clam Meat Samples Taken in the City of Quincy, Quincy Bay, 1964.

Date Collected	Station Site	Heptachlor (ppm) (LBW)*	Heptachlor Epoxide (ppm) (LBW)	Dieldrin** or DDE (ppm) (LBW)	DDT (ppm) (LBW)
February 24	Quincy P_1 Quincy P_2 (combined)			0.027	0.046
July 31	Quincy P_1	0.013	0.079	0.020	0.030
July 31	Quincy P_2	0.013	0.044	0.006	0.003
July 31	Quincy P_3	0.088	0.011	0.014	0.013

*(LBW): Live Body Weight.

**Dieldrin—DDE: Not separable by method used.

Table 20

Pesticide Concentrations in Clam Meat Samples Taken at Sites P_1 and P_2, Newburyport and Salisbury, Merrimack River, 1964.

Date Collected	Site	Heptachlor (ppm) (LBW)*	Heptachlor Epoxide (ppm) (LBW)	**Dieldrin or DDE (ppm) (LBW)	DDT (ppm) (LBW)
June 26	P_1	No Test	0.100	0.420	1.180
26	P_2	No Test	1.530	1.500	0.460
July 10	P_1	0.033	No Test	No Test	0.008
10	P_2	0.027	No Test	No Test	0.016
July 20	P_1	0.125	No Test	No Test	0.050
20	P_2	0.108	No Test	No Test	0.012
July 30	P_1	No Test	2.500	None	None
30	P_1	No Test	2.500	None	None
30	P_2	No Test	2.800	None	None
30	P_2	No Test	2.750	None	None
Sept. 8	P_1	0.100	No Test	No Test	0.033
8	P_2	0.100	No Test	No Test	0.025
Dec. 28	P_1	0.070	None	None	None
28	P_2	0.008	None	None	None

*(LBW): Live Body Weight (ppm).

**Dieldrin—DDE: Not separable by method used.

Tables from Jerome, Chesmore, Anderson, and Grice (1965).

Table 21

Pesticide Concentrations in Water Samples Taken at Newburyport and Salisbury, Sites P_1 and P_2, Merrimack River Estuary, 1964.

Date Collected	Sample Site	Heptachlor Epoxide (ppm)	Dieldrin* or DDE (ppm)	DDT (ppm)
June 17	P_1	No Test	0.001	0.001
17	P_2	No Test	0.001	0.001
July 10	P_1	0.002	None	0.003
10	P_2	0.001	None	0.001
July 20	P_1	0.005	No Test	0.002
20	P_2	0.006	No Test	0.002
July 30	P_1	0.001	None	0.002
30	P_2	0.001	None	0.002

*Dieldrin—DDE: Not separable by method used.

Table 22

Pesticide Concentrations in Mud Samples Taken at Sites P_1 and P_2, Newburyport and Salisbury, Merrimack River Estuary, 1964.

Date Collected	Sample Site	Heptachlor Epoxide (ppm) (DW)*	Dieldrin** or DDE (ppm) (DW)	DDT (ppm) (DW)
June 17	P_1	No Test	None	2.400
June 17	P_2	No Test	None	4.100
July 10	P_1	None	None	2.600
July 10	P_2	None	None	5.300

*(DW): Dry Weight.

**Dieldrin—DDE: Not separable by method used.

Chapter E-3B

PESTICIDES IN ESTUARIES

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Organic chemicals have been important as pesticides for only about 20 years but their uses, primarily in control of insects and weeds, have reached a high level. The United States production of all pesticidal chemicals in 1966 was estimated to be 4.5×10^8 kg. Some of this is exported, but large quantities are used in the United States. For example, the estimated domestic use of DDT for the crop year 1965-1966 was 20.9×10^6 kg (Shepard et al., 1967).

Early it was observed that catastrophic damage to aquatic life accompanies heavy application of the synthetic chlorinated hydrocarbon insecticides in estuaries (Springer and Webster, 1951); more recently it was recognized that accumulation of the persistent pesticides in the environment is also a hazard. Meanwhile, evidence was accumulating that some aquatic life is sensitive to very low concentrations of these chemicals (Butler, 1964; 1965a). At present, some ecologists feel that pesticides may be exerting deleterious stresses on the biota of estuarine systems; apprehension of such damage, actual or potential, was expressed in a number of the states visited (introductory remarks). No documented case was found where pesticide stresses alone have produced such changes of an estuary that it must be termed a new system. This statement means little, however, because this question has been given almost no study.

There is some cause for apprehension and a corresponding need for investigation to provide missing answers. Estuaries are required by many invertebrates and marine fishes for residence during part or all of their life cycles. This is especially true of the species supporting the estuarine-dependent seafood resources. Waterfowl, shore birds and fish-eating birds also use estuaries. At the same time, estuaries are traps for nutrients and pollutants. The greatest source of estuarine pollution is the inflowing streams. Suspended colloids and sediments may settle out; if they remain suspended there is a long residence time in the estuary. A major fraction of the human population and two-thirds of the factories making agricultural pesticides are located in the coastal states with drainage into rivers that feed into estuaries; in some areas drainage is directly into estuaries (Pimentel et al., 1965). Fig. 1 illustrates movement of pesticides with respect to an estuary. The suspicion

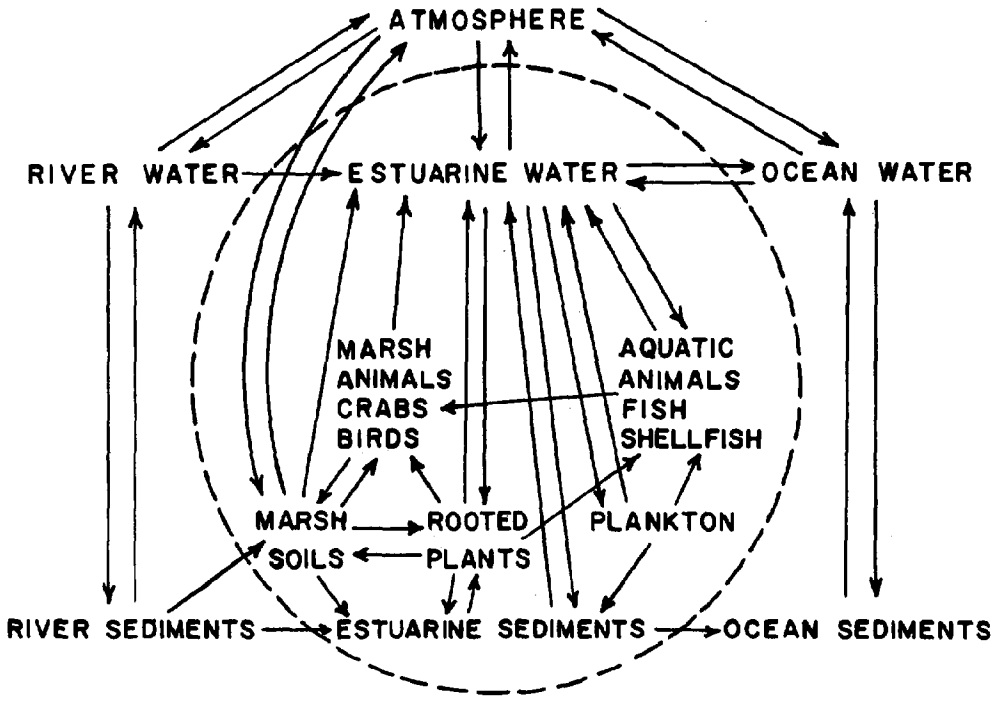

Fig. 1. The more important paths of pesticide movement into, out of, and within an estuary(dashed circle).

that pesticides are potentially a hazard to many valued species in estuaries is supported by the findings that some are toxic at low levels, that DDT is apparently ubiquitous in distribution and that dieldrin and endrin are present in low concentrations in many estuaries.

ANALYTICAL CHEMISTRY AND RESIDUE DETERMINATION

The chemical analysis of residues is a vital part of any quantitative study of pesticides in the environment. Increasingly, this is an area of sophisticated specialization of methods and instruments. The sensitivity of gas chromatography is now at a level where repeatable measurements of some insecticides and herbicides in water are made in the parts-per-trillion range. But as the sensitivity of measurement has increased, so have the requirements for experience and ability of the analytical chemist. Interference problems are acute at such levels. Each new environmental substrate has its problems of analysis. For example, methods developed for terrestrial soils are not always adequate for aquatic sediments which sometimes need additional cleanup stages. Because extremely low concentrations are ordinarily found in water, large volumes must be extracted in order to detect and measure the pesticides. Such extensions of methods require additional concentration and cleanup steps. The determination of residues in the environment is emerging as a separate specialized field.

Methods of pesticide analysis based on gas chromatography have benefited from many developments in electronic engineering during the decade. Advances in this eminently successful technique have been dependent on the development of highly sophisticated electronic systems for detecting and measuring small amounts of certain elements as well as separation by gas chromatography. The electron-capture detectors, utilizing mainly tritium and nickel as sources of electrons, have made possible the measurement of chlorinated hydrocarbon insecticides, chlorinated phenoxy herbicides, and certain other pesticides in the parts-per-trillion range. Halogens, mainly chlorine, are the major electron-capturing constituents of pesticide molecules. Highly sensitive detectors for sulphur, phosphorus, and nitrogen are now coming into general use. One is prone to reflect on what the levels of chlorinated hydrocarbons might have been without these advances in instrumentation. The ability to measure at low levels led to discovery of environmental accumulation of the persistent insecticides and to regulation of their use.

In spite of its success and usefulness, gas chromatography has some limitations. One of the major problems is positive identification of suspected pesticide residues. Often the amount present is too small for usual confirmation procedures such as infrared, ultraviolet absorption, nuclear magnetic resonance, or mass spectra analyses. Confirmation methods now in common use are thin-layer chromatography and p-value determinations (Beroza and Bowman, 1965), both of which include gas chromatography as part of the procedure.

Thus, as these useful methods have been improved, the requirements in support and experience have increased. The cost and availability of trustworthy analytical services have become the factors limiting pursuit of environmental studies. An important investigation should be carried out by a team, with a chemist who can then become expert in the special problems of the study. A further requirement is the careful preliminary analysis of the problem to assure an adequate and appropriate sampling plan.

LEVELS OF PESTICIDES IN ESTUARIES

Levels of pesticides in estuaries may be understood as a balance between inputs and outputs of the estuarine system. Inputs are primarily water-borne materials, some dissolved, but many sorbed on sediments. Other inputs include residues from direct application and drainage from adjacent areas, aerial transport including drift, washout from the atmosphere by rain, direct discharge from industry and sewer outlets, and movement of contaminated animals and plants (Figure 1). Within the estuary, pesticides circulate biologically and physically. Outputs from the system include outwash into the sea of dissolved pesticides and those sorbed on fine sediments and residues transported biologically. Some of the sorbed pesticides may be at least temporarily removed by burial. An unknown but probably small amount is lost to the air from the water surface. The estuarine system loses a pesticide continually through degradation of the molecule by chemical, biological or photochemical processes.

Water transport is the most important mechanism of pesticide movement into the estuarine system. Chlorinated hydrocarbon insecticides are of very low water solubility and exist in solution only at minute concentrations. Rarely do values reported for pesticide content of water refer to the dissolved molecules, since ordinarily the transported sediments are included in the analysis. Residue transport on sediment is important, and sorbed pesticides probably account for most of the recorded values for the persistent chemicals in water. The processes involved in the sorption of pesticides onto soil particles are not well understood, but knowledge is beginning to accumulate for some chemicals. Sorption probably occurs on land, and the pesticide-bearing colloid is then transported by erosion into streams. In the aquatic environment the sediments constitute one of the important storage reservoirs. The sorbed pesticide molecules are probably in equilibrium with the solution phase; thus release from sediment could occur in response to shifts in the solution concentration. A relationship between particle size and pesticide content of bottom sediments has been demonstrated in the San Joaquin River of California, where the higher concentrations were found in the finer sediments (Bailey and Hannum, 1967). The higher concentrations in water (including transported sediments) were found upstream in the section of river studied, with lower pesticide content downstream and in San Francisco Bay. In a study of the pesticide content of the important river systems of the United States, a high fraction of stations with infrequent or no positive records were at or just below impoundments, eliciting the comment, "This suggests that sedimentation and other phenomena associated with impoundments are dominant factors in removing chlorinated hydrocarbons from surface waters" (Green et al., 1967). It seems quite probable, therefore, that the important pesticide transport by river systems may be tied up in the sediments being carried. Most reported values do not allow a direct comparison, but by combining information from several sources, we observe that the concentration in water reported for DDT and its analogs near the mouth of the Mississippi River can be approximately accounted for by the transported silt burden if we can accept as the DDT content of silt that reported for bottom sediments at the same point (Green et al., 1967; Livingstone, 1963; Guilcher, 1963; Hammerstrom et al., 1967).

Important implications result from the probability that pesticides are transported on sediments. Flood stage events must be important in pesticide transport into estuaries. Some of the pesticides sorbed on sediments may become biologically unavailable when they are buried as sedimentation occurs. The very fine clay and organic colloids, which carry an important part of the residues, are spread widely throughout the estuary as well as transported directly into the sea.

Relatively few determinations of pesticide content of bottom sediments have been made for estuaries. By far the most spectacular are those reported for 1966 from a brackish marsh in Great South Bay, Long Island, where cores 20 or 40 cm deep contained an average residue of 13 lb. per acre of DDT and analogs (range 2.7 to 32.6 lb. per acre) in the Spartina mat, 2.9 lb. per acre in a drainage ditch and 0.3 lb. per acre for the submerged bay bottom (Woodwell et al., 1967). In most other studies, only the top layer of sediments has been examined and results have been reported as concentration (wet weight) instead of mass of residue (with terrestrial soils, an approximation often used is that 1 ppm represents 1 lb. per 3-inch acre). For DDT and analogs, samples from San Francisco Bay have been reported (Bailey and Hannum, 1967) as 57-204 micrograms per liter (thus less than 0.06 - 0.20 ppm), while values observed in bottom sediments in the mouth of the Mississippi River and nearby estuaries ranged downward from 0.03 ppm (Hammerstrom et al., 1967).

Most important direct applications of pesticides to estuaries have been for control of insects in salt marshes and on beaches. The high values of DDT and analogs observed in bottom sediments of the Long Island estuary presumably reflected mosquito control operations (Woodwell et al., 1967). In Florida, an unexpected peak in the DDT residues found in oysters was traced to an apparently minor control program where windrows of sea weed on the beaches were sprayed with DDT to kill larvae of a biting dog fly (Butler, 1966a). There are also examples of run-off from local agricultural uses; for example, pesticide contamination after applications in cranberry bogs were reported from Washington and Massachusetts.

Aerial transport carries some pesticides into estuaries, but the relative importance of this pathway is not known. Not only is there drift from neighboring aerial and ground applications, but a more general transport occurs in the atmosphere, both in the vapor phase (Abbott et al., 1966) and as sorbed molecules on dust particles (Risebrough et al., 1968). Atmospherically transported pesticides may enter estuaries as washout by rain (Abbott et al. 1966; Wheatley and Hardman, 1965; Weibel et al., 1966).

Discharge of pesticides directly into estuaries takes place from some pesticide manufacturing plants. Wastes draining from dry-cleaning plants into streams have been suspected as sources of dieldrin residues in rivers. Pesticides are brought in also through sewer discharge from cities; the storm sewers carry the surface discharge with residues from urban insect control programs.

There is biological transport of pesticide into and out of estuarine systems. The relative importance of this mechanism is unknown, though the tendency of the biota to concentrate and store the chlorinated hydrocarbon insecticides in body fat means that biological transport is much more important for these chemicals than the biomass of the animals might indicate.

Most pesticides (other than the chlorinated hydrocarbon insecticides) do not accumulate in animals. Compounds such as organic phosphate and carbamate insecticides and phenoxyalkanoic acid, s-triazine, and substituted urea herbicides are metabolized in plants and animals to innocuous products or are excreted through normal waste elimination.

Within the estuary, the chlorinated hydrocarbon insecticides are circulated throughout the biota. Bottom forms and filter feeders ingest sediments and organic material and remove the sorbed pesticides. The lower pH found in many digestive systems is probably an important factor in biological availability (Johnson et al., 1967). Once within the biota, pesticides may be passed through the food web into every form. Differential storage in body lipids provides the mechanism of biological concentration in food chain passage (Butler, 1966 b; Butler 1966 c; Ferguson et al., 1965). Fish, shellfish, and probably other forms absorb pesticides directly from water containing very low concentrations. Stored pesticides are metabolized and excreted, providing a second mechanism for exchange throughout the biota (Gakstatter and Weiss, 1967). Plankton carry pesticides and provide another point of entry into the food web. Few measurements are available. For DDT and analogs a value of 0.040 ppm (wet weight) was observed for "mostly zooplankton" in Great South Bay, Long Island (Woodwell et al., 1967), and in San Francisco Bay a measurement of 15 micrograms per liter was made for net plankton (Bailey and Hannum, 1967). Monthly samples with a no. 10 plankton net in Santa Rosa Sound, Florida from February through December 1965 ranged from trace to 0.89 ppm (dry weight) of DDT and metabolites; values ranged from minimum to maximum in a 30-day period and although there were not enough data to establish a good correlation with the dog fly spray program, results were suggestive (Butler, unpublished).

The pathways and factors of chemical degradation have been investigated in the laboratory, and some work has been done on the influence of soil characteristics on degradation. But the rates of decomposition under the many varied conditions found throughout the ecosystem are not well known, are subject to speculation, and are confused by the unknown amounts that are transported elsewhere.

We do not know the relative importance of the processes which are increasing or decreasing the stores of pesticides in estuaries. The question has two phases -- changes in stores which are biologically available, and changes in other stores including buried material. Has either phase reached the local condition of an approximately steady state postulated for some agricultural fields (U. S. Dept. Agric., 1966)? What are the biological implications of sediment storage of pesticides, if in fact this is taking place? Will great storms ecologically mobilize these stores?

Presently very little is known about even the order of magnitude of the processes of pesticide transport and storage. Pesticide content for September 1966 was reported for 109 river stations in the United States (Green et al., 1967). Using these synoptic values to represent annual means, the average for the 25 sites near a coast was 14 parts per trillion for DDT and analogs, when all determinations of less than detectable level were assigned a value of zero. The lowest value reported was 1 ppt. The total discharge from rivers of the contiguous United States may be estimated at 14.8×10^{14} kg per year (Livingstone, 1963b). If 14 ppt of total DDT is taken as an average for all river water, 20.5×10^{3} kg per year of total DDT would be transported into estuaries and oceans. This is a small quantity in view of annual use in the

United States which has been estimated as 20.9×10^6 kg for 1965-1966 (Shepard et al., 1967). The river transport value seems even smaller when we consider the high levels of United States use in former years (a peak value of about 35.7×10^6 kg in 1958-1959) and the total use since the late 1940's of approximately 500×10^6 kg (U. S. Department of Agriculture, 1953-1966). The total sediment load discharged by the rivers of the United States has been estimated with different results by different authorities (Twenhofel 1952; Howell, 1959). At an average value of 596×10^9 kg per year, the sediments may be expected to carry an average content of 0.035 ppm (dry weight), assuming that they transport all the DDT (20.5×10^3 kg per year). To compare this value with the few records of sediment residues, it must be adjusted to the wet weight basis (divide by perhaps a factor of 4). Thus there is approximate agreement with some of the values reported for surface sediments.

PRESENT WORK

Much of the work on the effects of pesticides on estuarine biota has been carried out at the Gulf Breeze, Florida, Laboratory of the Bureau of Commercial Fisheries, United States Department of the Interior. Support for this activity began in 1958. The earlier studies emphasized identifications of those compounds that have the most acutely toxic effects on estuarine animals, especially the commercially important species such as shrimp and oysters. Appropriate methods for quantitative bioassay were developed (Butler, 1964; 1965 a; 1966 a; 1966 b; 1966 c) and the features of the local estuarine system were studied. Laboratory tests demonstrated that growth of oysters is inhibited by many pesticides, with a measurable effect at one part per billion of aldrin in water, and a 50 per cent reduction in growth with less than 100 parts per billion for several but not all of the chlorinated hydrocarbon insecticides, for some of the organophosphorus insecticides, and for a few fungicides and herbicides. Bioassay with shrimp and juvenile fish revealed similar patterns of high sensitivity to some pesticidal chemicals, most but not all of which were chlorinated hydrocarbon insecticides. Recently, more subtle effects on estuarine organisms have been investigated. For example, it was found that fish avoided DDT by entering water free of the pesticide but generally could not discriminate between different concentrations. It has been shown that DDT interferes with normal thermal acclimation of fish (Ogilvie and Anderson, 1965). Investigations of the capability of fish to develop populations resistant to DDT showed no significant change after eight generations, although other workers have demonstrated this effect with insecticides in field populations of freshwater fish (Ferguson et al., 1965). Studies have been made of the inhibition of cholinesterase in the fish brain by organophosphate pesticide residues in the environment.

Present emphasis is being placed on the effect of pesticides on the ecology of estuaries and on the dynamic aspects of pesticide pollution. For example, the effect of the herbicide dichlobenil on primary production is being tested on Santa Rosa Island where the ecology of four small ponds has been studied since January, 1968 (Walsh, unpublished). The herbicide was added to one pond at a concentration of 1 ppm in the water. Preliminary results indicate that many of the important rooted aquatic plants were eliminated and that after they died, a phytoplankton bloom occurred. This bloom caused a seven-fold increase in phytoplankton chlorophyll from March to April, an increase not observed in the control ponds. There was also an increase in the diel oxygen pulse with little or no change in the ratio of photosynthesis to respiration. Herbicides labelled with radioactive elements are being used in studies of the rates of movement of herbicides through the water, sediment, and the biota of experimental ecosystems.

Laboratory observations are being made on the rates of accumulation and loss of labelled herbicides, using estuarine organisms that are maintained under different environmental conditions. Special attention is being given to the rate of exchange of herbicides between estuarine water and sediments.

Future studies at Gulf Breeze will emphasize movement of pesticides in estuaries and subtle effects of these chemicals on plants and animals. The research program will be divided into three broad areas; biological and chemical assays, ecological studies, and physiological studies. More work will be directed toward relating levels of residues in organisms to their effect on the organisms. As in the past, many of these studies will provide information for the Pesticides Registration Division of the United States Department of Agriculture to enable them to determine if particular pesticides should be labelled for use in or near the marine environment.

MONITORING PESTICIDES IN ESTUARIES

An important part of present investigations of the Bureau of Commercial Fisheries is the sampling of oysters or clams and sediments in estuarine environments as part of the National Pesticide Monitoring Program (Johnson et al., 1967). This program is designed to measure present levels as a base line for future comparison, but data are now so scanty that the first results from the monitoring program should be highly informative. Technical aspects of the monitoring are built upon previous work at the Gulf Breeze Laboratory. Sampling sites have been established in 24 estuaries along the coast of the contiguous states. Samples of oysters, or clams when oysters are not available, will be examined three times each year, with more frequent measurements where cooperating agencies provide assistance. The uppermost layers of the bottom sediments are to be sampled at the same time. No detailed results have yet been formally published, but a summary (Butler, 1967 a) indicates that the molluscan tissues for the most part carried less than 0.1 ppm, with only one estuary where residues are typically as high as 0.1 to 0.5 ppm. In many others, measurable residues were absent for 6 months or more of the year (measurements were quantified when greater than 10 parts per billion for the identified compounds). The summary concludes that the majority of the estuaries of the United States are contaminated with organochlorine pesticides, but not to an extent that endangers man's food supply.

The present state of knowledge about pesticides in estuaries may be summarized as follows: There is increasing apprehension based upon accumulating evidence of toxicity at low concentrations of chlorinated hydrocarbon and other pesticides and a growing body of facts about disturbance of the physiology at sublethal levels of these chemicals. An ambient level sufficient to promote body storage at measurable levels exists almost generally for DDT. But there are very few known facts about the biologically effective levels in estuaries or about the populational effects of these levels. It is fair to say that we know and understand the problem only well enough to outline the areas where we need to know more.

RESEARCH NEEDS

Research on any subject which increases understanding of the estuarine ecosystem will also add to knowledge of how pesticides operate in this environment. From a more particular view, the following five areas, two general and three specific, are proposed for immediate investigation.

General questions:

1. What are the dynamics of pesticides in estuaries? How do they enter and leave and in what quantities, how are they picked up and stored in the biological and physical compartments of estuaries and how do they move through the food web and at what rates? What is the biological availability in different storage sites? What is the fate of pesticides which enter the estuary? Can a model be set up and quantified sufficiently with real information to help predict the behavior of a given pesticide in an estuarine system?
2. What are the effects of pesticides upon such measurable characteristics of whole ecosystems as energy flow, nutrient cycling, food web complexity, species diversity and system stability?

Specific questions:

3. What is the present status of residues in estuaries; at what level are they stored and where?
4. What are the best sampling designs for estuarine studies? Can methods of residue identification and measurement be improved? Can levels of detection be reduced by a factor of 10 or 20?
5. Can index values be devised that will allow some judgement of the biological effect of pesticides, given certain observed residue levels in organisms, in the water or in the sediments?

Chapter E-4

SYSTEMS RESULTING FROM DREDGING SPOIL

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INTRODUCTION

A prevalent modification of coastal areas is the practice of dredging for navigation channels in the shallow areas and the construction of small-boat harbors. Not only is the environment modified by the increases in depth, but the deposit of spoil, often in the adjacent estuarine areas, creates an artificial sedimentation factor.

Interruption of normal pathways of energy flow can result from dredging operations. The placement of deep channels cause channeling of tidal currents, thus changing normal circulation patterns and usual transport mechanisms for incoming foods and flows. The construction of "spoil banks", usually resulting from disposal of dredging spoil, not only modifies the current mechanisms by physical barriers, but covers large areas with a thick layer of sediment that eliminates the normal bottom component of biological energy cycling.

Characteristic ecological systems result from the dredging modification, governed by the new flow patterns and/or by the abrupt change in bottom characteristics. Although few reports exist explaining the results of dredging operations, enough ecological information exists from scattered studies to construct a picture of a characteristic dredging-effected system.

SYSTEM EXAMPLES

Upper Chesapeake Bay, Maryland

The University of Maryland Chesapeake Biological Laboratory (1967, 1968) investigated the hydrography and biota of an area in upper Chesapeake Bay (near the mouth of the Sassafras River) during a dredging operation. A diagram of the area is included in Figure 1. The dredge was removing sediment from one area (indicated in Figure 1) and depositing it in an adjacent area.

A map of the distribution of turbidity during dredging operations and in relation to different tidal activities is shown in Figure 2. The movement of silt in the water was detected as a function of light transmission, which is plotted in Figure 2. High transmission indicates water with relatively little silt and low transmission indicates water with relatively more silt load. Diagram A in Figure 2 depicts the spoil deposit area after dredging had been interrupted for 8 hours previously, indicating relatively high light transmission of greater than 40% (10 cm path). Diagrams B to D show the same site after dredging operations had begun during a flood tide. Diagrams E to G are during

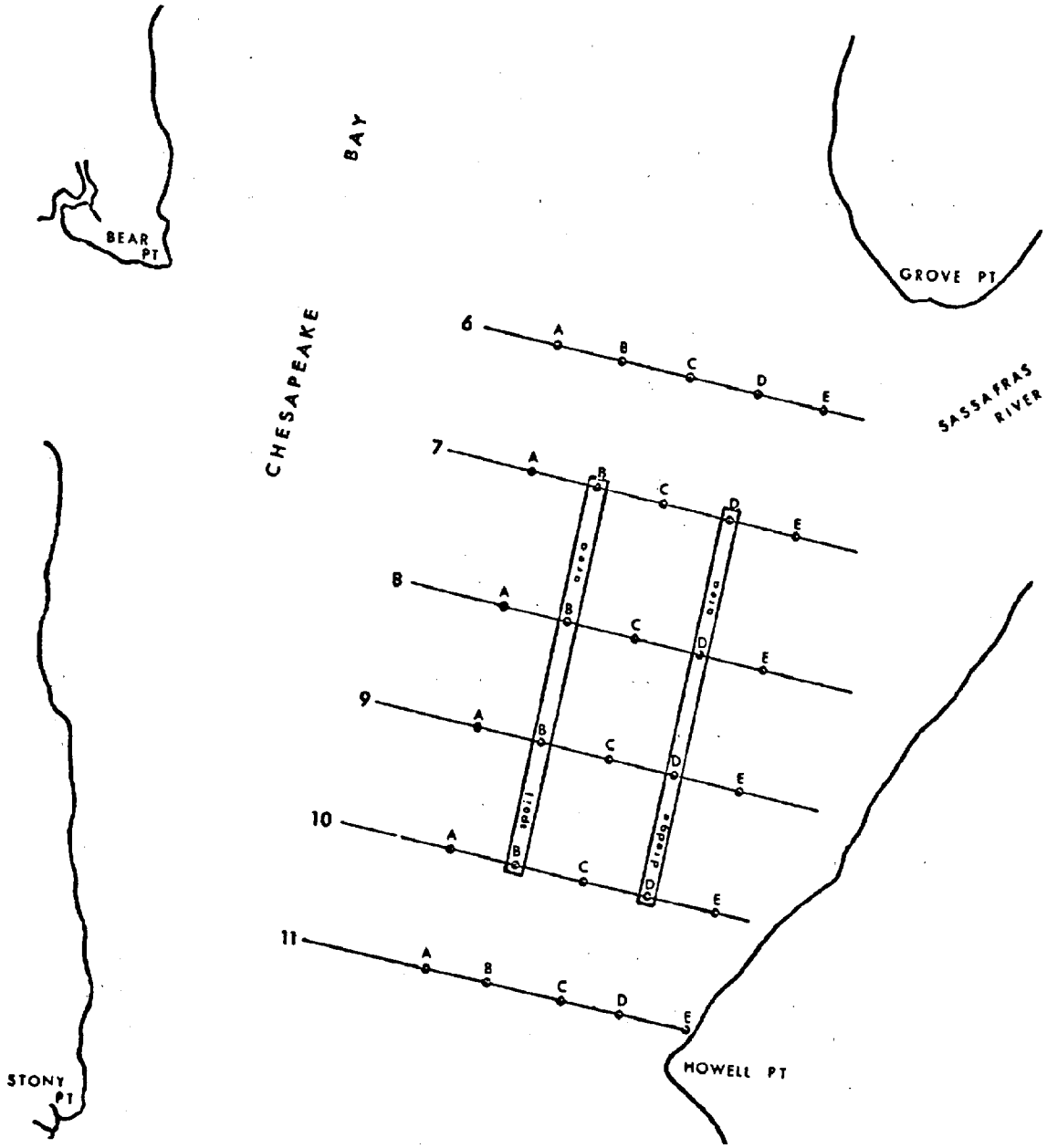

Fig. 1. Diagram of upper Chesapeake Bay, Maryland showing areas of dredging and spoil deposit. (From Maryland University, Chesapeake Biological Laboratory 1967d.)

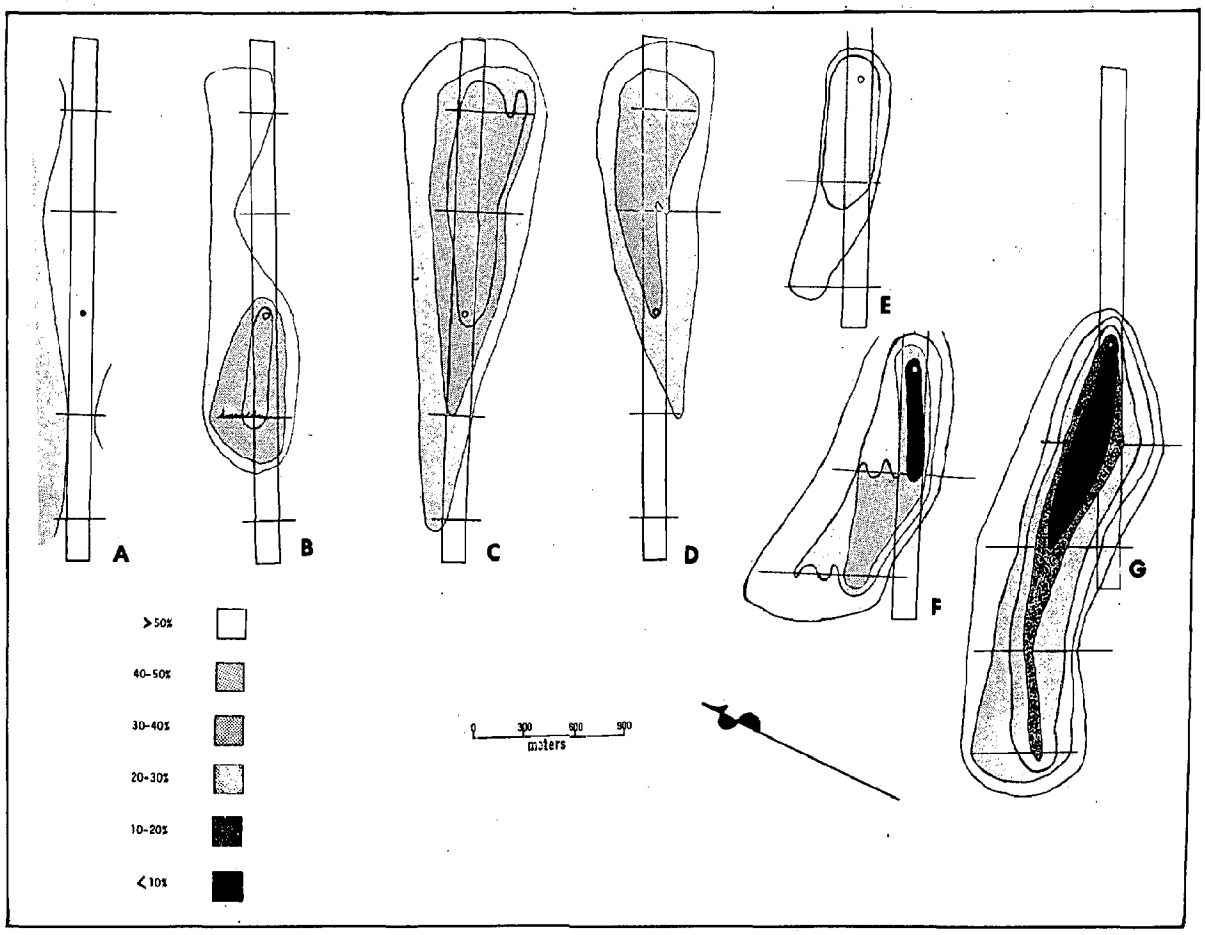

Fig. 2. Diagram of light transmission (10 cm path) in a dredged system, Chesapeake Bay, Maryland. The point of spoil deposit is indicated by the open or black circle. Lines indicate transects. (From University of Maryland Chesapeake Biological Laboratory 1967a.)

A. 24 Oct 1966--1500 hrs.--No dredging.
B. 24 Oct 1966--1600 hrs.--Dredging; Flood Tide.
C. 24 Oct 1966--1800 hrs.--Dredging; Flood Tide.
D. 24 Oct 1966--1900 hrs.--Dredging; Flood Tide.
E. 25 Oct 1966--1000 hrs.--Dredging; Ebb Tide.
F. 25 Oct 1966--1100 hrs.--Dredging; Ebb Tide.
G. 25 Oct 1966--1200 hrs.--Dredging; Ebb Tide.

ebb tide. The suspended sediment apparently moves in the direction of the current (which is affected by the tidal direction), thus dispersing sediment for several hundred meters away from the spoil area. In other words during flood tide there is transport of sediment up bay and during ebb tide there is sediment transport down bay. Similar findings were discussed by Masch and Espey (1967) for dredging operations in Galveston Bay, Texas.

Interruption of normal biota patterns were observed in the dredging and spoil deposit areas. With the sediment transport from the spoil area, considerable bottom area was covered with soft mud. A list of the benthic organisms collected in the area described in Figure 1 is presented in Table 1 in order of their abundance. The same number of organisms were found in both the dredged and spoil deposit areas, but the order of their rank in the system was different. This indicates that some are better able to withstand the stress of one kind than of the other. Macoma balthica and M. phenax are among the dominant organisms in the dredged area, but rank far down the series from the spoil deposit area (Table 1).

It is important to note that the numbers and species of organisms taken in the spoil deposit and dredged areas are quite different than those taken in surrounding areas in upper Chesapeake Bay (University of Maryland Chesapeake Biological Laboratory 1967, 1968).

Redfish Bay, Texas

The effects of spoil deposition from dredging operations in Redfish Bay, Texas were studied by Hellier and Kornicker (1962) and Odum (1963a). The study area is shown in Figure 3, with the sampling quadrats indicated. An intracoastal canal was dredged through the area during 1959. Odum (1963a) described the area and sequence of events as follows:

> "Prior to dredging the water was deepest at Station 1 where Thalassia beds were maximal (about 0.5 m at ordinary tide levels), shoaling gradually to 0.3 m at station 5 with increasing percentages of Diplanthera. The dredge began to affect the area in the last of February 1959 with clouds of drifting silt obscuring the grass on many days during March. The dredging was stopped when rock was reached so that by March 13 the dredge had been removed and the water again was clear. Final dredging began again in November 1959 with the deposition of the spoil island and silt on top of the grass as far as station 2."

Sedimentation was measured by Hellier and Kornicker (1962) before and after the dredging. Odum (1963a) measured the productivity and chlorophyll before and after the dredging.

The sedimentation before dredging and one week and 18 months after dredging is reported in Table 2. A significant amount of sediment was deposited over stations 1 and 2, with very small amounts deposited over the other three stations. These data show that sediment deposition resulted from the dredging operations over a distance of less than 0.5 miles from the dredge site; at greater distances the sedimentation was negligible. Ingle (1952) reported that damage caused by deposition of sediment on the bottom did not extend beyond 0.23 miles from the dredge in a Florida bay, results very similar to those reported for Redfish Bay. Hellier and Kornicker further stated that little sorting of sediments took place during the spread of sediments from the dredge site.

Table 1. Benthic organisms taken from the spoil and dredged areas in upper Chesapeake Bay during dredging operations in rank of their density in the system. (From Maryland University Chesapeake Biological Laboratory 1968b.)

SPOIL AREA	DREDGE AREA
Hydrobia sp.	Hydrobia sp.
Leptocheirus plumulosus	Scolecolepides viridis
Scolecolepides viridis	Macoma balthica
Gammarus fasciatus	Macoma phenax
Procladius sp.	Gammarus fasciatus
Cyathura polita	Leptocheirus plumulosus
Monoculodes sp.	Monoculodes sp.
Rangia cuneata	Rangia cuneata
Chironomus attenuatus	Cyathura polita
Macoma phenax	Tubulanus pellucidus
Macoma balthica	Procladius sp.
Chiridotea almyra	Edotea triloba
Tubulanus pellucidus	Chiridotea almyra
Edotea triloba	Retusa canaliculata
Odostomia sp.	Congeria leucophaeta
Crypotchironomus argus	Hypaniola grayii
Coelotanypus sp.	Nereis succinea
Hypaniola grayii	Rhithropanopeus harrisi
Chaoborus punctipennis	Mogula manhattensis
Retusa canaliculata	Oxytrema virginica
Oxytrema virginica	Crypotchironomus argus
Tubifex sp.	Neomysis americana
Corophium lacustre	Physa sp.
Congeria leucophaeta	Tubifex sp.
Neomysis americana	Odostomia impressa
Physa sp.	Odostomia sp.
Odostomia impressa	Chaoborus punctipennis
Epitonium rupicolum	Acteon punctostriatus
Acteon punctostriatus	Epitonium rupicolum
Amnicola limosa	Gyraulus parus

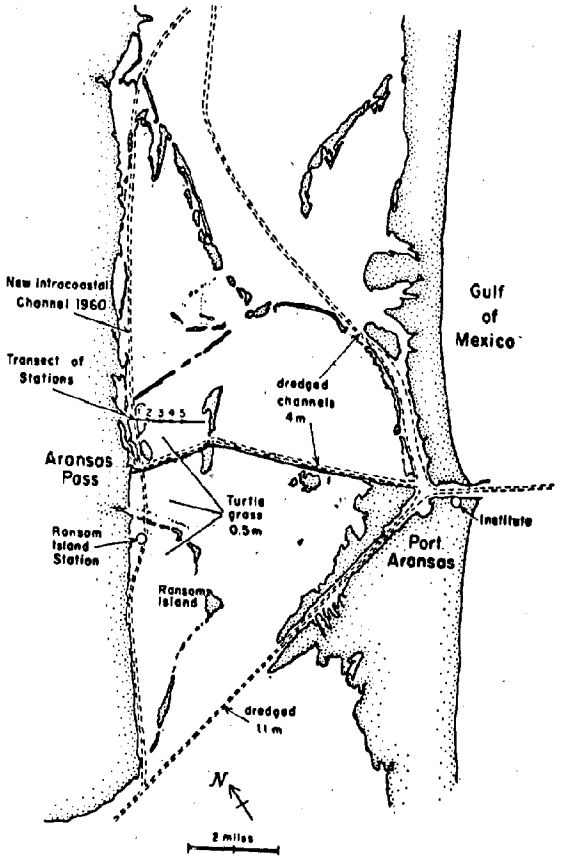

Fig. 3. Map of Redfish Bay near Aransas Pass, Texas, indicating the study area, the 5 stations and the intracoastal channel. (From Odum 1963a; Fig. 1).

Table 2. Amount of sediment deposited at the 5 Redfish Bay stations before and after dredging. (From Hellier and Kornicker 1962; Table 1).

Station quadrat no.	Thickness of sediment deposited in 9 months prior to dredging, cm	Thickness of sediment 1 week after dredging, cm	Thickness of sediment 18 months after dredging, cm
1	negligible	27	32
2	negligible	22	33
3	negligible	½	negligible
4	negligible	½	....
5	negligible	½	3

* Thickness of sediment measured from reference surface marked with red gravel spread in December, 1958.

The effects of the sediment transport on the benthic producing system were reported by Odum (1963a), as indicated by productivity and chlorophyll measurements. The change in chlorophyll a concentration per square meter is indicated in Table 3. Productivity data are presented in Figure 4.

The productivity and chlorophyll data indicate that fertility diminished during 1959 when there was some disturbance due to dredging silts shading the bottom-producing system. Productivity and chlorophyll were less than during the previous year or the year following. Productivity values were extremely low during the main dredging operation late in 1959 and early 1960. During 1960, however, the productivity and chlorophyll values were much higher than for any previous year, indicating enhancement of productivity from the redistribution of dredge spoil. Thus, Ingle's (1952) suggestion that dredging may stimulate by adding nutrients may have merit.

Dredging removed 125 feet of bottom area for the channel and between 0.2 and 0.5 miles of bottom production under and adjacent to the spoil, but did no permanent damage to the area further away. As compared to the findings of Ingle (1952) and Mackin (1961a), which indicated that damage was over a smaller area from the dredged channels in deeper water, the damage covered a larger area. This may be related to the depth of water in which the spoil was deposited. Discharge from a channel of the Redfish Bay dimension into 0.5 m water or less may shoal water depth critically over a much wider zone than where the discharge is in bays 10 feet deep. Furthermore, the type of system was not bottom-producing in the studies reported by Ingle and Mackin; thus the sedimentation did not destroy the major productivity component.

Intra-coastal Canal in South Carolina Marshes

Lunz (1938) studied effects of dredging and spoil deposition from an intra-coastal canal on oysters in South Carolina marshes. Sample results of these data on sedimentation rate, mortalities, and weight of oyster meats are given in Figs. 5 and 6 and in Tables 4-8. Except where spoil was released directly on oyster reefs, little if any direct negative effects on the oysters were found. No study was made of the effects of changed circulation or turbidities on photosynthesis or trophic aspects of the ecosystem. No acid waters developed.

CONSTRUCTION OF DEEP CHANNELS

The construction of channels in shallow coastal areas usually results in the alteration of current structure, which can either lead to deterioration of the system's productivity or enhance it. In most cases, however, the result is a change in the system's energy source and therefore alteration of normal energy cycling. When this occurs, the system structure is altered or replaced by another system or subsystem type. For example, reefs of molluscs have become established in channels because of the sufficient currents in the channel where there were no reefs before due to the lack of currents.

Hargis (1966), in summarizing the effects of the large James River navigation channel and harbor in Virginia, noted that the current structure and sediment transport were modified. One result was the inability to transport oyster spat upstream, whereas before the construction of the channel the current structure was such as to allow the upstream transport of oyster spat. Hargis found little changes in the salinity values, but there was an alteration of the salinity structure with the alteration of currents and freshwater flow.

Table 3. Chlorophyll a in benthic plants along the Redfish Bay transect. Data in gm/m^2. (From Odum 1963a; Table 1).

Station	Distance from new channel	Summer 1959	Summer 1960	Winter 1961
1	0	0.0078*	Station now	
		0.0113*	out of water	
		0.0180*	as spoil island.	
		0.0086*		
		0.029*		
		0.052*		
		0.0015*		
		0.115*		
		0.011†		
2	0.25 mile east	0.0208‡	Beds	
		0.0106‡	covered	
		0.0074‡	with 30	
		0.0052‡	cm of	
		0.0105‡	soft silt;	
		0.0110†	no plants.	
3	0.5 mile east	0.079§	1.14¶	0.021
		0.042§	1.21¶	0.019
		0.087§	1.32¶	0.153
		0.022§	1.46¶	0.103
		0.062†		0.130
				0.012
				0.055
				0.133
4	0.75 mile east	0.087§	0.26¶	
		0.033§	0.42¶	
		0.011§	0.49¶	
		0.048†	0.47¶	
5	1.0 mile east	0.050§	0.20¶	
		0.022§	0.26¶	
		0.0195†	0.28¶	
Mean		0.03379	0.6827	

* May 30, 1959.
† July 15, 1959.
‡ April 27, 1959.
§ June 30, 1959.
¶ July 26, 1960.
Feb. 7, 1961.

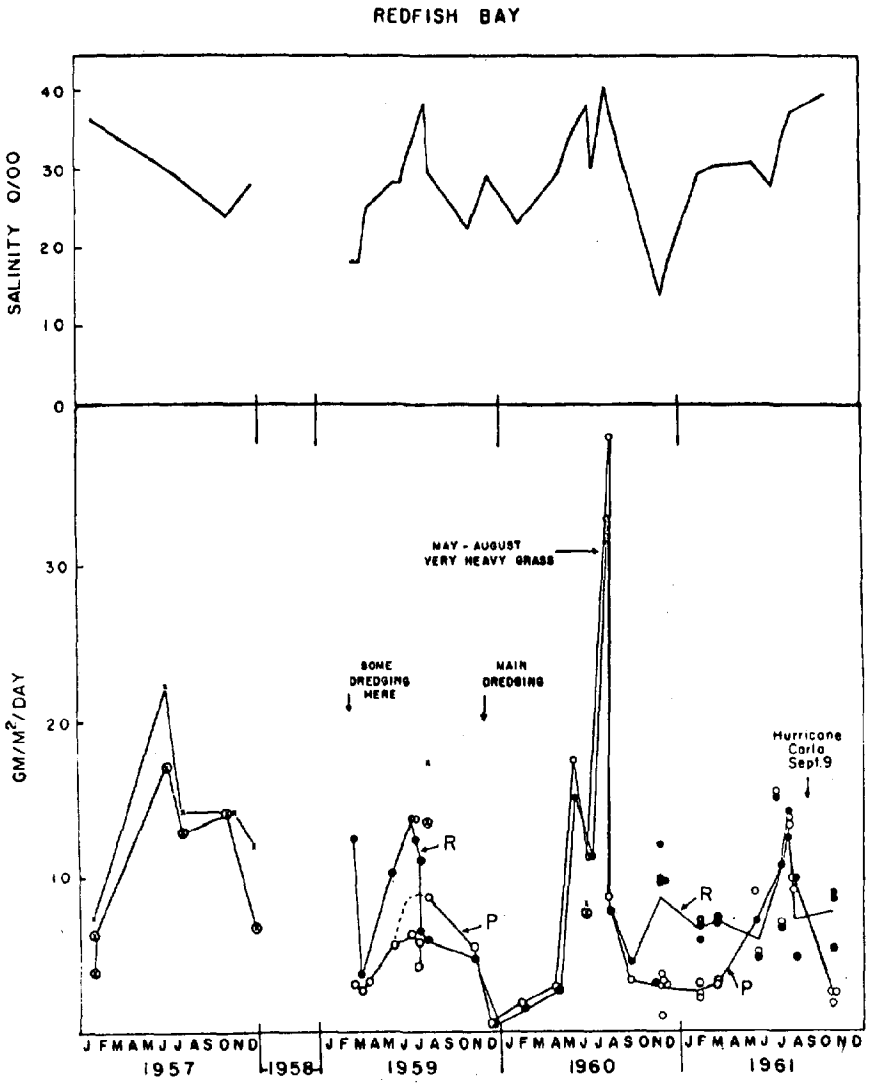

Fig. 4. Record of salinity, gross photosynthesis and total respiration for Redfish Bay, Texas during 1957-1961. Most data taken between stations 2 and 3 on the transect from the dredging operation. (From Odum, 1963a; Fig. 6).

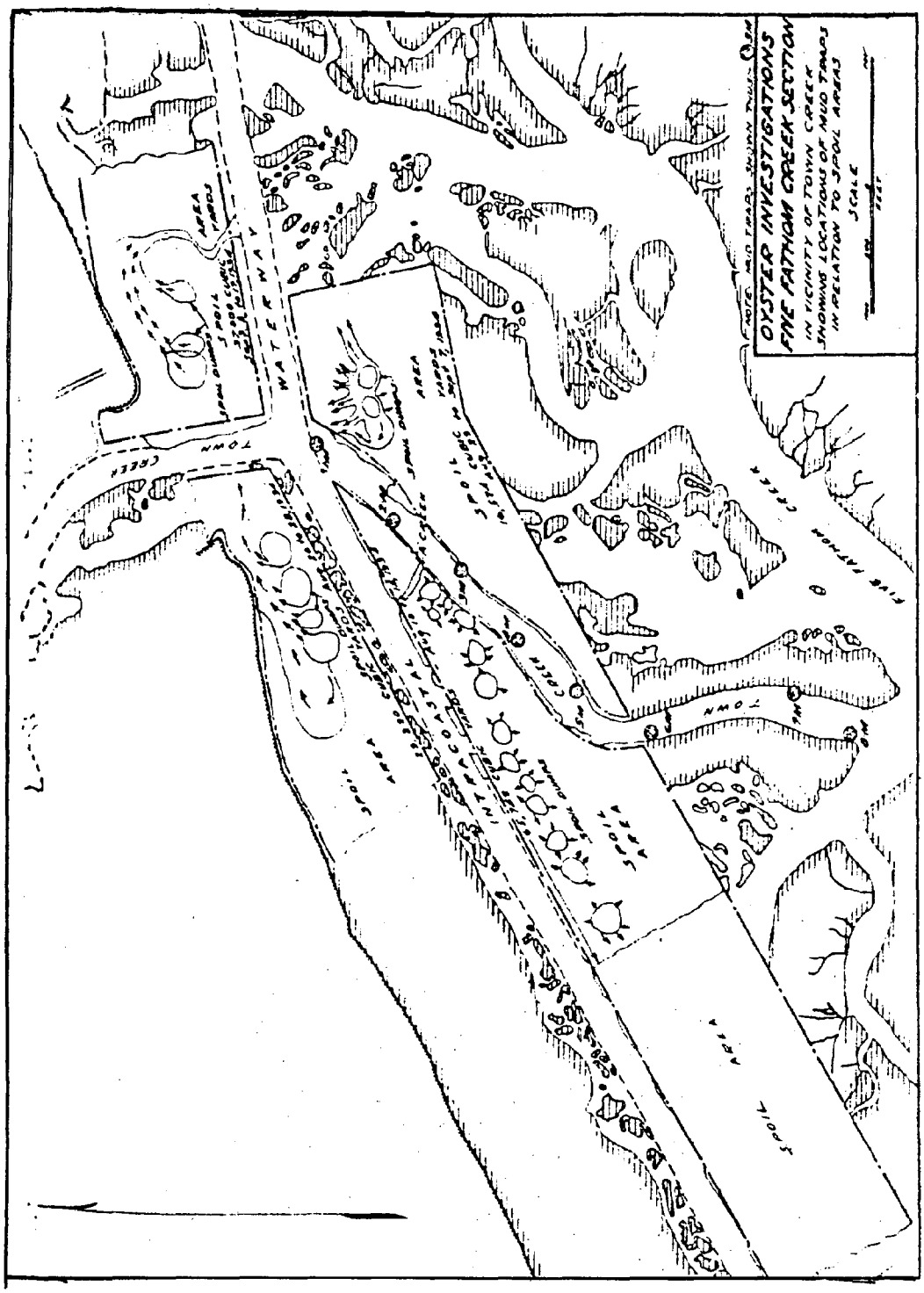

Fig. 5. Section of intra-coastal canal dredging in South Carolina (Lunz 1938; Fig. 18).

Table 4. Rates of mud deposition before and during dredging (Lunz 1938).

A. Amount of mud, in grams, settling on one square yard of bottom in 24 hours - Town Creek - before dredging.

Station	June 4	June 5	June 6	
1m	33.02	1.50	3.86	
2m	--	0.99	4.12	
3m	7.52	6.13	7.76	
4m	11.20	6.12	7.85	
5m	7.49	--	6.75	
6m	56.13	48.72	54.86	
7m	3.73	8.20	5.87	
8m	3.76	4.06	--	
Average	17.55	10.82	13.01	= Ave. 13.79

B. Amount of mud, in grams, settling on one square yard of bottom in 24 hours - Town Creek - during dredging.

Station	Aug. 7	Aug. 11	Aug. 12	
1m	59.85	43.41	39.70	
2m	74.90	288.76	28.87	
3m	3.60	46.92	--	
4m	22.53	99.26	50.53	
5m	23.44	128.13	252.43	
6m	37.98	21.65	21.65	
7m	5.89	14.43	36.81	
8m	37.85	18.04	25.24	
Average	33.26	82.57	65.03	= Ave. 60.29

C. Mud accumulating on oyster beds, as shown by mud boards - Five Fathom Creek Section.

Station	Amount of Mud (in millimeters)
57	0
58	0
59	Missing
60	+20
61	+10
62	+20
63	Missing
64	Missing
65	- 5
66	+25
67	Missing

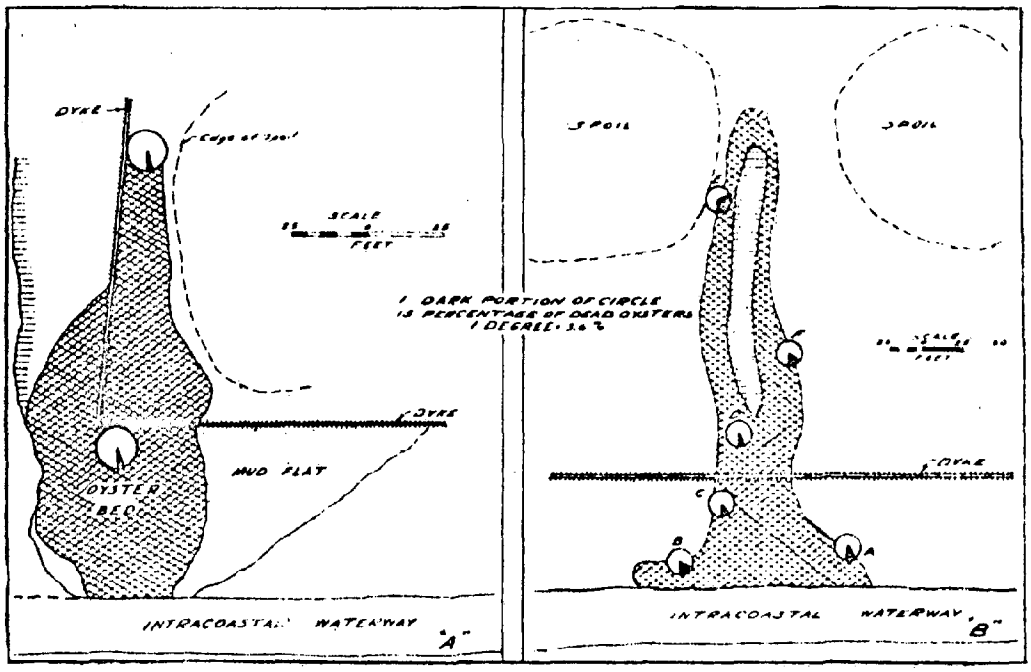

Fig. 6. Oyster mortalities relative to a shielding dyke (Lunz 1938).

Table 5. Percentage of dead oysters in Bullyard Sound (Lunz 1938).

Inside Dike		Outside Dike	
Number	Percentage	Number	Percentage
1	6.82	2	5.26
3	4.62	4	3.64
5	6.12	6	8.06
7	6.84	8	6.25
9	6.35	10	7.55
11	6.66	12	6.12
13	7.55	14	8.11
15	10.26	16	8.82
17	9.37	18	9.75
19	11.90	20	13.33
Average	7.65	Average	7.69

Table 6. Changes in weights of oyster meats (Lunz 1938).

Sample	Change in Weight	Time (in days) between surveys	Distance from Site of dredging
1	+2.53 grams	94 days	10,560 feet
2	+4.02 grams	94 days	10,560 feet
3	-0.06 grams	91 days	10,560 feet
4	+2.70 grams	90 days	7,920 feet
5	-0.92 grams	91 days	6,600 feet
6	+0.78 grams	91 days	6,600 feet
7	-1.10 grams	89 days	6,600 feet
8	-1.45 grams	62 days	5,280 feet
9	+2.81 grams	82 days	2,500 feet
10	-0.46 grams	83 days	2,000 feet
11	+3.14 grams	73 days	100 feet
12	+0.20 grams	81 days	100 feet

Table 7. Number of dead oysters - Five Fathom Creek (Lunz, 1938). May - November, 1936.

Station	Number of dead oysters in baskets of 20				
	May 26	June 15	July 8-9	Aug. 5-7	Oct. 22
57	0	0	0	1	1
58	0	1	1	1	2
59	0	0	1	1	1
60	1	1	1	1	1
61	1	1	1	1	1
62	0	0	2	2	2
63	0	0	1	1	*
64	0	0	1	1	1
65	0	0	0	0	1
66	0	0	0	0	0
67	0	-	6	7	-

* Basket missing.

Water Samples

After dredging operations had begun in the Five Fathom Creek section on July 1, water samples were taken at intervals until the completion of the survey in November. The total results of these observations follow.

Table 8. Salinity, pH and Temperature of Water - Five Fathom Creek Section (Lunz, 1938).

Station	JULY			AUGUST			OCTOBER		
	Salinity	pH	Temp. °F	Salinity	pH	Temp. °F	Salinity	pH	Temp. °F
57	23.71	7.7	86.0	23.87	7.9	86.0	6.56	7.1	64.5
58	25.90	7.7	87.0	26.38	7.9	86.0	10.17	7.3	70.0
59	25.90	7.7	86.5	26.52	7.9	86.5	10.30	7.3	70.0
60	27.33	7.7	87.0	28.46	7.9	87.0	11.20	7.3	64.5
61	25.53	8.0	86.0	28.53	7.9	87.0	11.78	7.5	64.5
62	27.25	8.0	85.5	29.49	7.9	87.0	14.02	7.7	65.0
63	28.22	8.0	85.0	30.56	7.9	86.0	16.21	7.7	64.5
64	31.04	7.9	86.0	32.90	8.1	88.0	19.05	7.8	64.0
65	31.98	7.9	85.5	33.39	8.1	87.0	20.93	7.9	64.0
66	33.04	8.0	86.0	33.21	8.1	86.0	18.86	7.7	64.5
67	34.51	8.0	86.0	33.21	8.1	87.0	20.81	7.8	64.0

Behrens (1966) reported salinity values in the Texas Laguna Madre for a year following the lowest three-year rainfall on record (77 years of record) to be limited to about 88 parts per thousand. In earlier years, before the dredging of the intracoastal canal throughout the length of the Laguna, the salinity had reached values higher than 100 ppt (Rusnak 1960; Collier and Hedgpeth 1950). Behrens attributed the lower salinity values following the construction of the intracoastal canal to the fact that water could be transported into and out of the Laguna Madre through the channel, thus providing an exchange mechanism that did not exist before.

Masch and Espey (1967) demonstrated that a dredged channel was useful for the transportation of silt downstream from accumulations in shallow areas. They pointed out, however, that the channel must be sloped downward and widened away from the shallow area to facilitate the transport of sediment.

In Louisiana marsh systems, the practice of "cutting" channels into the marsh area has resulted in marsh destruction (Louisiana Fish and Game Division Report, Dr. Ted Ford, Personal Communication). The deep channels allow salt-water intrusion, which kills the Spartina alterniflora (the dominant marsh grass). After the destruction of the grass there is nothing to hold the soil structure, thus allowing severe erosion to alter the physical structure of the substrate. Channels in the Louisiana marsh system also allow rapid drainage of the system, which removes the nutrient pool and lowers productivity.

DEPOSITION OF DREDGING SPOIL

The most obvious effect of spoil disposal in the adjacent areas would be the alteration of bottom type, with additional layers of sediments deposited by the dredging operations. However, a subtle effect would be the additional sediments in suspension in the water. During the actual dredging operations, sediments may be suspended in water at a greater concentration than usual (Masch and Espey 1967). Turbidities produced by these suspended sediments, however, are usually of short duration, thus causing only localized and short-time interruptions of light dependent processes such as photosynthesis.

Studies designed to determine the rate of settling of suspended materials in marine waters have indicated that the settling rate is dependent to some extent on the salinity of the water (McCoy and Johnston 1964). The results from Currituck Sound, North Carolina shown in Figure 7 indicate that normal turbidities are achieved within 72 hours after additional sediment suspension if the water has a salinity at least 20% of sea water. These results can be interpreted to mean that water with 20% or more seawater salinity will not maintain a turbidity greater than that already in suspension by natural means. As salinity is decreased, however, turbidity from dredge-caused sediment suspension could become a problem. Masch and Espey (1967) obtained similar results in Galveston Bay, Texas.

Because of the sedimentation rate in marine waters, suspended materials caused by dredging operations are not transported great distances from the dredge site. As shown in Figure 8 Masch and Espey (1967) reported that the suspended materials caused by shell dredging in Galveston Bay, Texas were deposited by the time the material had been transported about 1200 feet from the dredge site. Mackin (1961a) reported similar results for Louisiana dredging operations. The sedimentation rate reported in Figure 8 (Masch and Espey 1967), near the dredge site, was appreciable and prolonged dredging in the area would cause a layer of sediment to be deposited on the bottom that would cover oyster reefs and bottom assemblages.

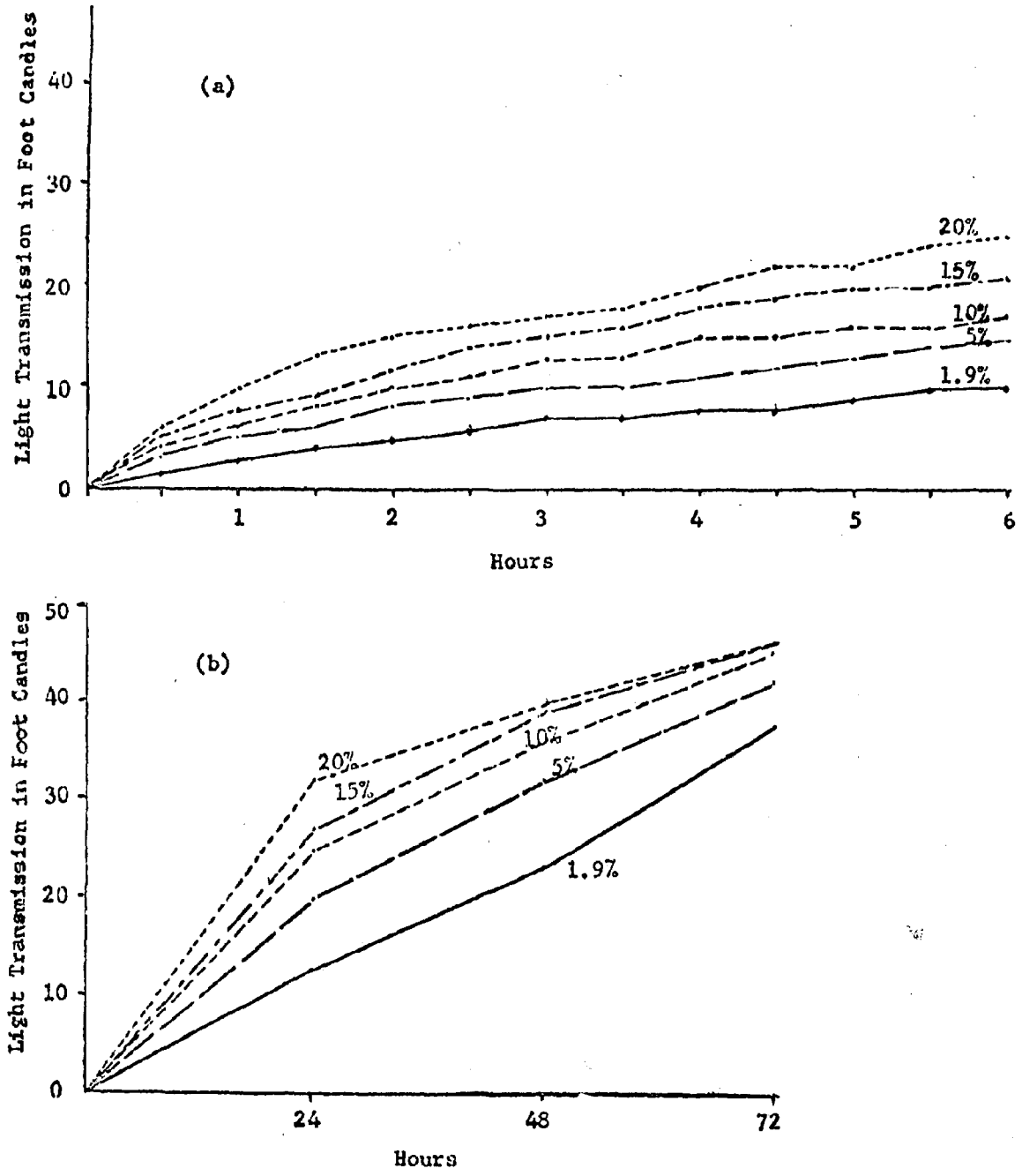

Fig. 7. Light transmission through a water solution of different percentages of seawater strength, each with initial equal turbidity induced by 10 grams of silt (From McCoy et al. 1964; Fig. 1). The foot candle reading was 50 before the extra turbidity was added.

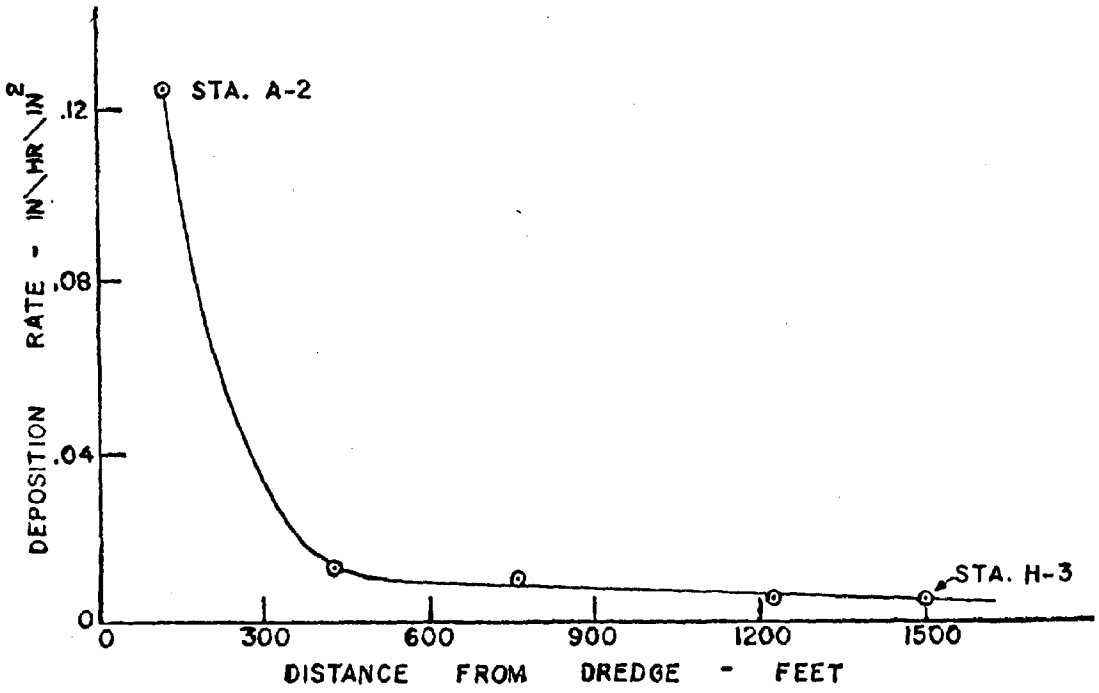

Fig. 8. Distribution of deposition rate from a dredge output (From Masch and Espey 1967; Fig. VI-6).

Table 9. Data from experiments testing the effect on oysters of graded turbidities produced by mud suspensions (20 oysters in each tank). (From Mackin 1961a; Table 7.)

Experiment no.	Turbidity: experimental tank, ppm	Mortality: experimental tank, percent	Turbidity: control tank, ppm	Mortality: control tank, percent	Dates
1	100	0	15	5	2-28-55 to 3-21-55
2	200	15	15	20	4-5-55 to 4-26-55
3	300	10			
4	400	35	10	25	4-27-55 to 5-18-55
5	500	20			
6	590	15	15	15	5-23-55 to 6-12-55
7	710	20			

Mackin (1961a) reported the results of several experiments to determine the effects of increased turbidities on oysters. He found no significant difference in the mortality of oysters exposed to turbidity higher than found in the natural environment (about 300 to 700 ppm) than with controls in low turbidity water (Table 9). No data were reported, however, on the comparative pumping rates or metabolic characteristics. Demoran (1955) reported that fish were not harmed by increased sediment concentrations in water, but the fish did tend to avoid the higher turbidity areas.

Chapter E-5

IMPOUNDMENT SYSTEMS

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INTRODUCTION

When a part of an estuary is isolated from circulation by spoil banks, dredgings, or dams, new ecological systems replace the old ones. This chapter concerns the class of disturbed systems that result. With the rapidly expanding industrialization and population along the waterways of the world, impoundment of rivers to form freshwater reservoirs divert river waters so that the estuarine systems are changed. As summarized by Copeland (1966a), rivers are important contributors to the maintenance of coastal systems. Thus, the practice of impoundment of coastal rivers decreases the inflow of freshwater, nutrients, organic matter and circulation maintenance of open channels.

The practice of bulkheading and filling of marsh and mudflat systems in coastal areas for the construction of building sites for homes and industry has altered the morphology of coastal systems. Since these vulnerable areas are important for nursery grounds of many of the commercial and sports fishing organisms, the removal of them by impoundment considerably lowers the productivity of the coastal systems. These marsh and mudflat systems are also important for nutrient regeneration and cycling of organic matter, thus making them important to primary productivity as well.

In some areas, where hurricanes are a menace, dykes have been constructed to divert hurricane tides for protection of population centers. While these dykes seldom completely isolate the area behind them from the sea, they restrict flows of water through the narrow openings.

In all of the above cases, the transport of energies through coastal systems is severely restricted. The incoming energies of organic fuels from rivers, marshes and mudflats and the energies of circulation necessary for system maintenance are limited through the practice of impoundment.

SYSTEM EXAMPLE - SOUTH BAY, TEXAS

An example of a system resulting from impoundment is South Bay, the southernmost bay in Texas which is located on the south end of the Laguna Madre between the Brownsville Ship Channel and the Rio Grande. The area has been studied by Breuer (1962) and is shown in Figure 1.

The Brownsville Ship Channel was completed to a depth of 28 feet in 1938, and much of the resulting spoil from the east end of the channel was placed in

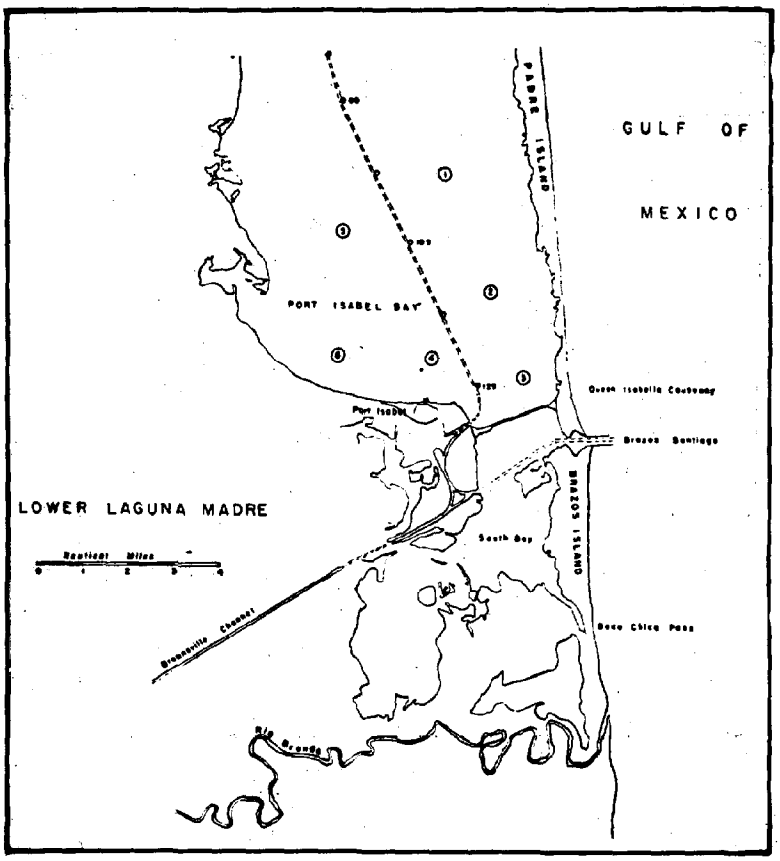


Fig. 1. The South Bay area, Texas. South Bay is adjacent to the lower Laguna Madre of Texas (From Breuer 1962; Fig. 4). Note the placement of the Brownsville Ship Channel.

a line along the north end of the bay. This and subsequent redredgings effectively closed the entrance of the bay with the exception of a very narrow, shallow inlet. The net effect was the prevention of circulation and flushing of South Bay by waters from the Laguna Madre and from the Gulf through Boca Chica Pass and Brazos Santiago (Breuer 1962). The maintenance of Boca Chica Pass was dependent on the strong northers to push large volumes of water from the Laguna Madre, through South Bay, and out through the pass to keep it scoured. As a result of the spoil deposition, Boca Chica Pass filled in rapidly and closed finally in 1945. Since that time, circulation in South Bay has become almost non-existent, and the average water depth has decreased from 1.2 meters to an average of less than 0.4 meters (Breuer 1962).

The South Bay oyster population was destroyed due to the lack of circulation, especially immediately following each new dredging operation (Breuer 1962). With the new bottom of soft mud, the water remains relatively turbid because of wind action. This prevents the establishment of the thin grass bottom that formerly existed over most of South Bay.

Price (1947) and others have discussed the area of bays as a controlling factor of the depth of the bottom. They indicate that in small bays, insufficient wave energy develops to keep the bottoms from filling in with sediment to very shallow levels. This seems to be the case in South Bay after the impoundment operation, as it was apparently reliant upon large wave energies from the Laguna Madre wind-driven currents (northers push large volumes of water southward in the Laguna Madre during more than half the year providing tremendous flushing activities, Copeland et al. 1968).

Odum and Wilson (1962) indicated a lower productivity in the South Bay system because of the lack of circulation and smothering of grass bottoms due to the impoundment. Breuer (1962) found less fish and invertebrates in the system, which he attributed to the sedimentation problem. Apparently, small bays are to some major extent reliant upon circulation energies provided from adjacent, larger systems for circulation and flushing through passes and openings for the maintenance of high productivity.

Other Examples

Copeland (1967b) and Copeland and Jones (1965), in their studies of the Mexican Laguna Madre, reported a decrease in productivity and an increase in sedimentation rates after the Lagoon was impounded in 1961. In addition to the increase in salinity because of evaporation and lack of circulation, the regular exchange between the lagoon and the Gulf of Mexico and the San Fernando River was eliminated.

The impoundment of the Everglades in Florida is described by Tabb et al. (1962b), Idyll (1965b) and Kolipinski and Higer (1966). A schematic drawing of the impoundment area is shown in Figure 2. With the decreased flow through the Everglades, there has been an increased salinity in the receiving bays, and the migration and propagation of shrimp into the Everglade systems has been curtailed.

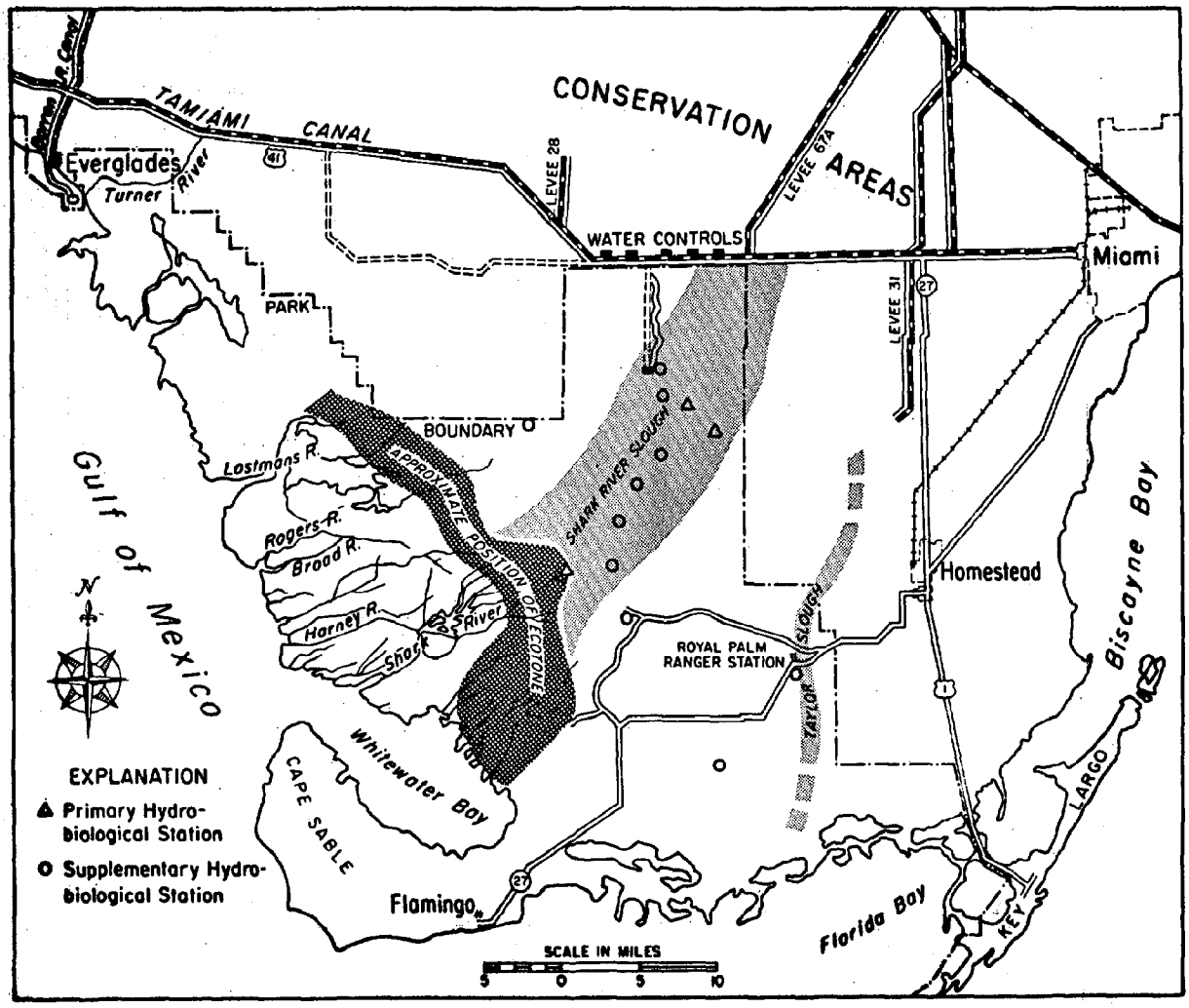


Fig. 2. The lower Florida Everglades, showing the position of water control structures and the approximate limit of the brackish water zone. (From Kolipinski and Higer, 1966).

IMPOUNDMENT OF RIVERS

Hydrography

The most important hydrobiological parameter is salinity. Under normal conditions, salinity is variable in most coastal systems. Collier and Hedgpeth (1950) showed a direct correlation between river flow and estuarine salinity in Corpus Christi Bay, Texas. If river flow is restricted by upstream reservoirs, the salinity level in the receiving coastal system may increase to affect a change in biological communities.

Coastal systems that have proved to be important nursery areas possess a well defined salinity gradient between river mouth and tidal pass, accommodating a large variety of species. River inflow maintains the salinity gradient to a large extent and without it the entire estuary could become hypersaline, as in the Texas Laguna Madre. On the other hand, too much freshwater inflow may cause the entire estuary to become fresh or near-fresh (e.g., Sabine Lake, in east Texas) and destroy the salinity gradient.

Pritchard (1967a), in summarizing the physical parameters controlling stratification and circulation in an estuary, stated that river input and tidal action were the most dominant. He emphasized the ratio of the volume of water flowing up the estuary during flood tide to the volume of fresh water flowing into the estuary during a complete tidal cycle. When the ratio is small the stratification and circulation resemble those of a salt wedge system, but when the ratio is large (on the order of 1000) they resemble those of a vertically homogenous system. Thus, impoundment of rivers could cause a complete change in the stratification and circulation of the receiving system, robbing it of the needed energies.

Closely connected to river flow is the maintenance of natural inlets between barrier islands, connecting coastal systems to the sea. Simmons and Hoese (1959) suggested the necessity of river flow for maintaining discharge through Cedar Bayou, a natural inlet on the Central Texas coast. Lack of river flow during the early 1950's resulted in the closing of Cedar Bayou. Passes can be kept open only at great expense if river flow is reduced below the level required to maintain them. The passes are important passageways for organisms between nursery grounds and breeding areas (Daugherty 1952; Hoese 1958; Copeland 1965b).

Nutrients

The influx of freshwater is one of the principal sources of dissolved nutrients into coastal systems. The relationship of growth in estuarine phytoplankton to terrigenous nutrients is shown by the increase in growth following periods of rainfall and runoff. Organic materials, which are decomposed by bacteria and fungi to release large amounts of organic and inorganic substances that are absorbed by estuarine organisms via the aquatic medium, are brought into the system by river inflow (Nash 1947).

Vitamins, which are essential for the growth and reproduction of animals, are also brought into coastal systems from streams. Burkholder and Burkholder (1956) reported a greater concentration of vitamin B_{12} in the mud and estuarine waters of Georgia than in the adjacent seawater. Maurer and Parker (1968) reported similar results for muds on the Texas coast. Starr and Sanders (1959) postulated that productivity of the near-shore sea was greatly dependent on vitamins adsorbed to suspended solids brought into the coastal waters by river flow. Birke (1968) showed that tremendous amounts of vitamin B_{12} were transported into the San Antonio Bay, Texas via the contributing streams. The significance of vitamin B_{12} contribution is illustrated by the fact that the vitamin had to be added to filtered seawater in order to allow shrimp to develop from larval stage to juveniles in laboratory experiments (thus raising the possibility that larval stages of marine organisms require residency in coastal environments for reasons other than moderate salinities).

Fisheries Products

It is a well known fact that the young of many organisms important to the commercial and sports fishery of this country must rely on coastal systems for at least a portion of their life cycle. Not so well known, however, are the reasons for this dependence. One thing is clear; i.e., the migration into the coastal areas is related to freshwater input and its salinity moderation as well as the tremendous amounts of food energies required by the rapid growth and maintenance during metamorphosis.

Oysters

Oysters grow in a wide range of salinity, and the production of oysters is exclusively estuarine. Since the oyster is sessile, it depends upon the maintenance of current systems to bring food and vitamins to it and to transport its waste products away. Impounding river systems could affect the oyster through the alteration of the current system maintained by the freshwater inflow.

Under normal conditions, occasional flushing with fresh water from rivers helps rid oyster populations of damaging parasites and eliminates species that compete with oysters for food (Galtsoff 1964). Reduction of freshwater input usually results in an increase in salinity of coastal waters, resulting in an increase of parasites and competing species (Hopkins 1956, 1962; Andrews and Hewatt 1957; H. W. Wells 1959 , 1961).

Galtsoff (1964) discussed the decrease in the oyster population of Texas bays during the drought of the 1950's. He suggested that the resulting increase in salinity in coastal water during that time allowed the influx of parasites and diseases, and the higher salinities restricted gonad development. The same thing could happen in coastal systems where the river input is eliminated by reservoirs upstream.

Shrimp

One of the more valuable fishery products, the shrimp are dependent upon the coastal systems for a part of their life cycle. Their coastal

dependence seems related to freshwater runoff as illustrated in Figure 3, which shows a correlation between catch of shrimp and average rainfall for the state of Texas. Correlation coefficients were significant to the one percent level when rainfall of the two previous years was correlated with an annual catch. This method of analysis was considered valid since the reproductive cycle of shrimp is one year or more and most of the rain in Texas falls during the fall after the major shrimp spawning has occurred. An increase or decrease in rainfall is followed by similar fluctuations in shrimp catch, generally after a two-year period. After the drought in Texas during the early 1950's, white shrimp populations never fully recovered. Presumably, the addition of several reservoirs in Texas during this time prevented river output from approaching the level of previous years and the white shrimp are more dependent upon lower salinities than other species of shrimp.

Other fishery products

A discussion of oysters and shrimp is only an indication of the reaction of the system to impoundment effects. Changes in speciation and structure of food chains are also indicative of coastal systems that are affected by impounded streams. Texas coastal systems where fresh water under normal conditions is in short supply, yields the opportunity to study impoundment effects during times of drought. Hoese (1960a) reported changes in the speciation and community structure of a Texas bay when little or no fresh water entered it, as compared to a year later when the bay was flooded with fresh water.

The data available for comparison of fresh water input and commercial fishery output are scant. However, in Texas, where fresh water input is at a critical level in most coastal areas, some startling conclusions can be drawn. The minimum fresh water contribution required to maintain the present commercial fishery is not reached in some years in Matagorda, Aransas and Corpus Christi Bays (Figure 4). In Galveston and San Antonio Bays, where fresh water input is relatively large, larger commercial fishery yields have been harvested during years of intermediate or below average fresh water input.

The data presented in Figure 4 do not complete the whole picture. Perhaps more important than the total commercial fisheries output is the change in species composition that makes up the total fishery harvest. During the year following an above average fresh water input, shrimp and oysters make up a larger percentage of the total. These organisms are known to be more dependent upon fresh water and are more valuable economically than are fish products, which make up a larger percentage of the total during years of below average fresh water input.

IMPOUNDMENT OF MARSH AND MUDFLAT AREAS

The apparent effect of impoundment of marginal areas of coastal embayments is that of complete isolation of the area from the system to which it

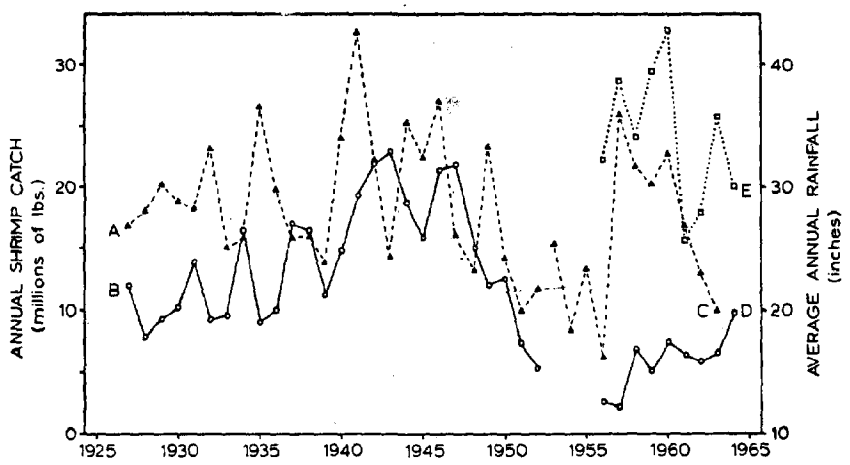


Fig. 3. Annual shrimp harvest and average rainfall for the state of Texas, 1927-1964. (From Copeland 1966a; Fig. 2).

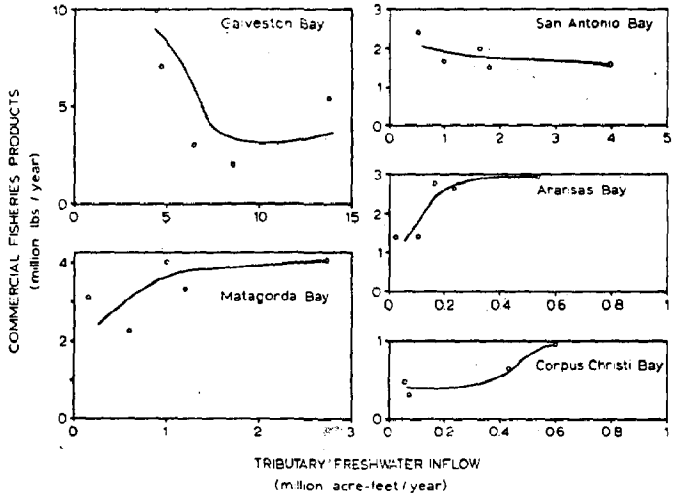


Fig. 4. Commercial fisheries harvest vs. annual tributary freshwater inflow for five Texas bays, 1959-1964. One year's freshwater contribution was plotted against the following year's commercial harvest. (From Copeland 1966a; Fig. 3).

is connected as a subsystem. Little, however, is known about the overall effects the isolation (and thus destruction) of these marginal subsystems exert on the coastal system. A diagram of such a system is in Figure 5.

The construction of a highway through the coastal area of Louisiana and Mississippi has effectively separated the inland areas of the coastal marshes from the outer marsh areas, completely altering the circulation patterns of the entire marsh system. The result has been saltwater intrusion into the outer marsh system (in the absence of the freshwater inflow from inland sources now prevented by the highway), with the subsequent results of soil alteration and eventually alteration of the marsh vegetation. The "dam" has prevented the normal circulation of waters, nutrients and organisms, which has resulted in lower productivity in the coastal systems (Mr. William Turcotte, Mississippi Fish and Game Commission; personal communication).

In New Jersey, where the harvest of salt-hay is profitable, dykes have been constructed in marsh areas. The resulting change in circulation and salinity structure has led to Spartina paetens replacing S. alterniflora as the dominant marsh plant (see New Jersey Division of Fish and Game 1968, for a discussion concerning marsh dyking in New Jersey).

Drainage and diversion by dams has interrupted the normal flow of fresh water from the Okeechobee area south across the Everglade marshes in Florida (Idyll 1965a; Kolipinski and Higer 1966). The resulting higher salinities in the Everglades estuarine systems has caused a decrease in shrimp and fish production (Tabb et al. 1962b; Idyll 1965a). The lower production of these more "desirable" organisms is only a portion of the change that has occurred as a result of the damming activities; there has been a change in the community structure (Tabb et al. 1962b).

The practice of bulkheading and filling shallow area (mudflat subsystems) around the margins of coastal systems for development of real estate is becoming widespread (G. W. Allen, 1964). These areas are important to the ecology of adjacent coastal systems as nursery grounds for the young of many estuarine organisms, nutrient regeneration and production of organic matter (Diener 1964). The loss of these marginal areas lowers the overall productivity of the coastal system.

Price (1968) described the effects of the construction of a spoil bank resulting from the digging of the intracoastal canal in the Laguna Madre of Texas adjacent to a large mudflat area (about 35,000 acres). The spoil bank prevented the normal circulation of water from the Laguna over the mudflat during normal fluctuations of the tides. Only during the extremely high water levels of spring and fall and storm tides was it possible for the water to circulate on the flat. During the remainder of the year the water evaporated leaving a salt layer, which later was transported over pasture land by wind and destroyed several thousand acres of grass.

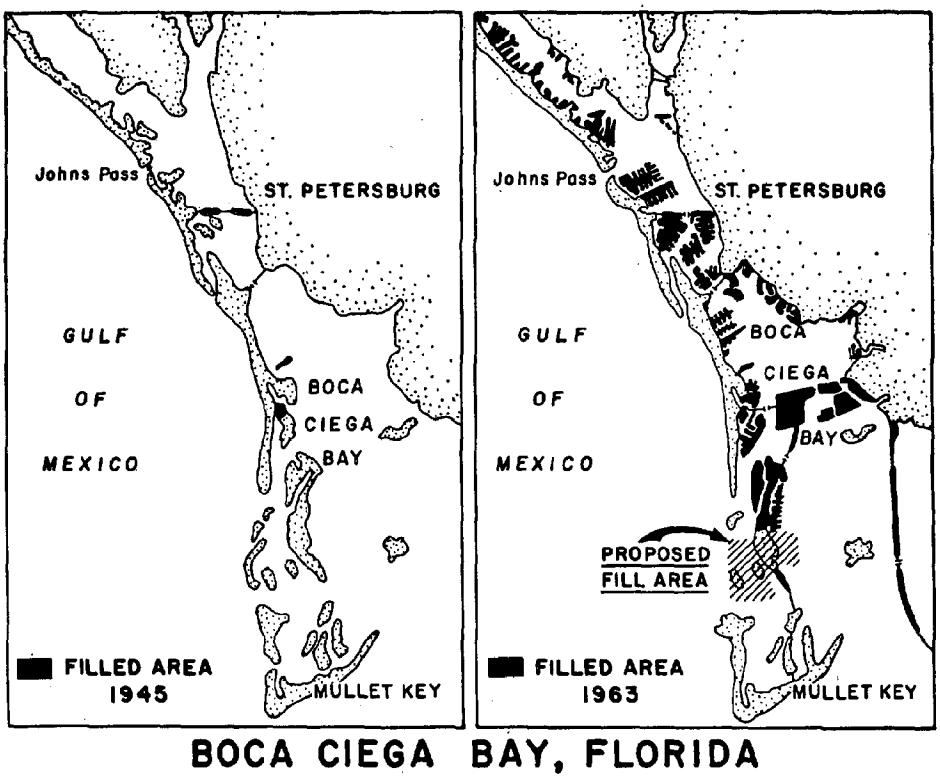
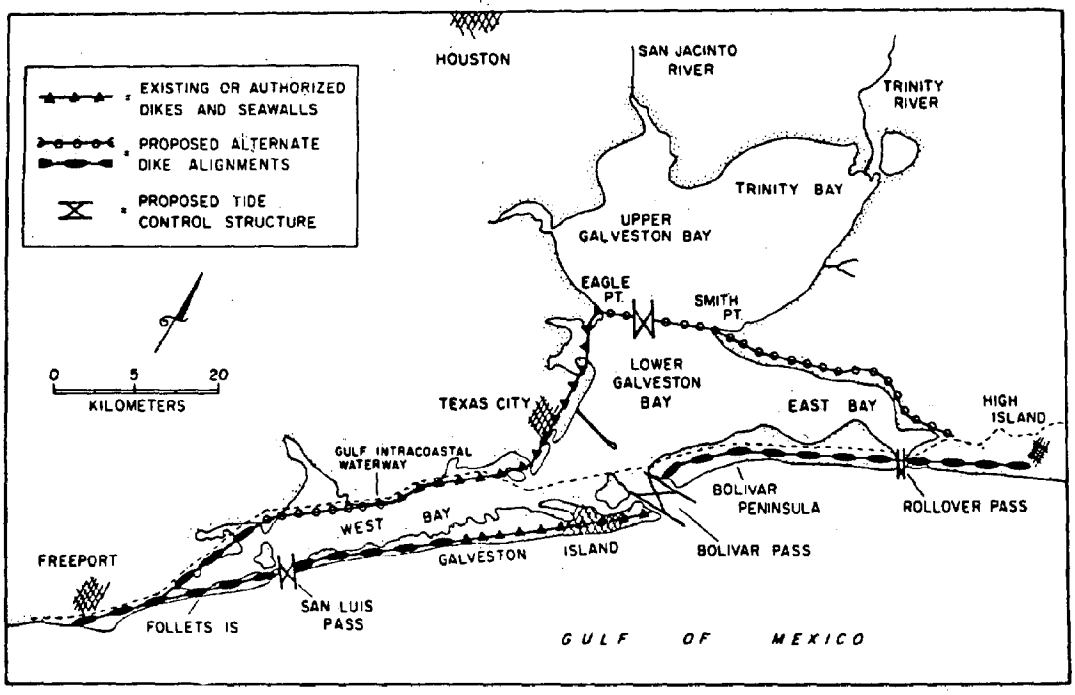


Fig. 5. Diagram of Boca Ciega Bay, Florida showing the effects of bulkheading and filling since 1945. The black in the right hand figure represents loss of bottom from filling (From Smith 1966; Plate 4).

HURRICANE PROTECTION DYKES

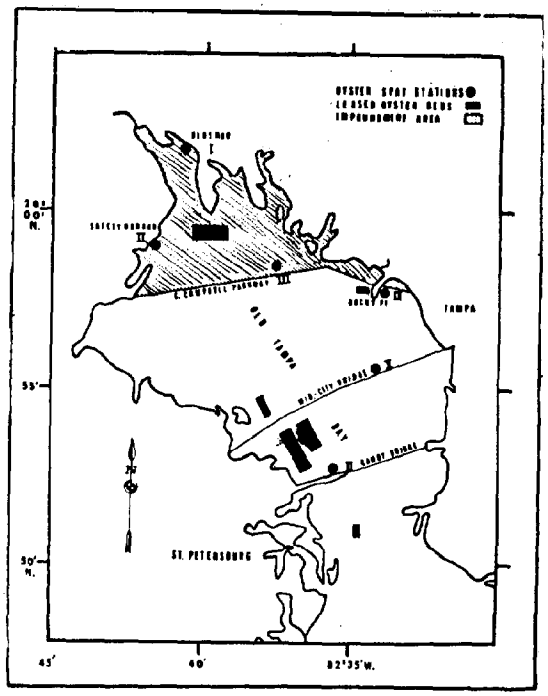
To provide protection of population centers in coastal areas, dykes have either been constructed or are planned across the mouths of several estuaries. These dykes possess narrow openings through which the exchange of fresh and sea water occurs.

Such a dyke exists in Narragansett Bay, Rhode Island and are planned for Galveston Bay, Texas and Mobile Bay, Alabama. C. A. Maquire and Associates (Providence, Rhode Island) studied the physical characteristics that resulted from the construction of the dyke in Narragansett Bay (report not available) and found that current structure and circulation were significantly modified. Hydrographic data indicated that the water above the dyke was fresher than prior to the installation of the structure. Saila (1962) reported the results of a study concerning the migrations of winter flounder in relation to the dyke and found insignificant differences in migration patterns as a result of the dyke being there. More work needs to be done before general conclusions can be drawn concerning the effects of the dykes on the ecology of the coastal systems. Fig. 6 is the proposed impoundment for Galveston Bay. Fig. 7 is a proposed impoundment for Tampa Bay, Florida.



--Proposed alignments for dikes and water-control structures to prevent hurricane flooding in the Galveston Bay area.

Fig. 6. Proposed impoundment for Galveston Bay (Hoogland 1966; Fig. 30).



--Sampling stations, major commercial oyster producing areas, and proposed fresh-water lake impoundment in Old Tampa Bay, Fla.

Fig. 7. Proposed impoundment for Tampa Bay (From Finucane 1966; Fig 9).

Chapter E-6

ECOLOGICAL SYSTEMS RECEIVING HEATED WATER

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The flow of hot water into estuaries may produce new ecological systems which are characteristically simple in species variety. Our best knowledge of the role of high temperatures and unusual temperature fluctuations comes from hot springs, some of which are additionally stressed by relatively high concentrations of dissolved salts. With the proliferation of nuclear fueled steam electric power plants along the coast using tidal waters for cooling, hot water ecological systems may develop in estuaries where the factors of tide, freshwater inflow and normally adapted marine populations introduce new dimensions. Although there is a large body of information on the thermal physiology of individual species, only a few case histories of ecological systems affected by heated water are available.

EXAMPLES

Patuxent River Estuary, Maryland

Heated water is introduced from a nuclear power plant at Chalk Point (Fig. 1). This is an oligohaline environment where the salinity averages 6‰. An example of the heat dissipation and thermal plume from the plant just before high slack water is shown in Figure 2. The plant is located in an area which comprises a spawning, nursery and feeding area for white perch, Roccus Americanus (Fig. 1). A before and after study in the vicinity of the discharge canal shows significantly more fish occupying the warm water during the winter (Fig. 3) but fewer immediately at the discharge site in the summer (Fig. 4). There has also been a reduction in primary production as measured by ^{14}C assimilation in the effluent water (Fig. 5). Studies by Mihursky and Stross (1967) have shown that up to 95% of the plankters are killed on passing through the plant condensers.

San Joaquin River, Central California

The Contra Costa Power Plant is located on the San Francisco Bay delta (Fig. 6) in an oligohaline environment that occasionally reaches a salinity of 2 - 3‰ in the summer. The natural temperature regime ranges between 7°C and 23°C with a weighted mean temperature increase across the condensers of 9°C (Kerr, 1953). Fig. 7 shows a characteristic thermal plume at this location. The resident fish in this oligohaline estuary are principally fresh water species (Table 1), but an important anadromous component of valuable commercial fish either spawn at this location or pass through on their upstream and downstream migrations. Particular attention has been paid to protecting these commercially important species with the expectation that other less important species will be similarly protected (Kerr, 1953). The juveniles of king salmon (Oncorhynchus tschawytscha), a cold water species, is particularly subject to the possible harmful effects

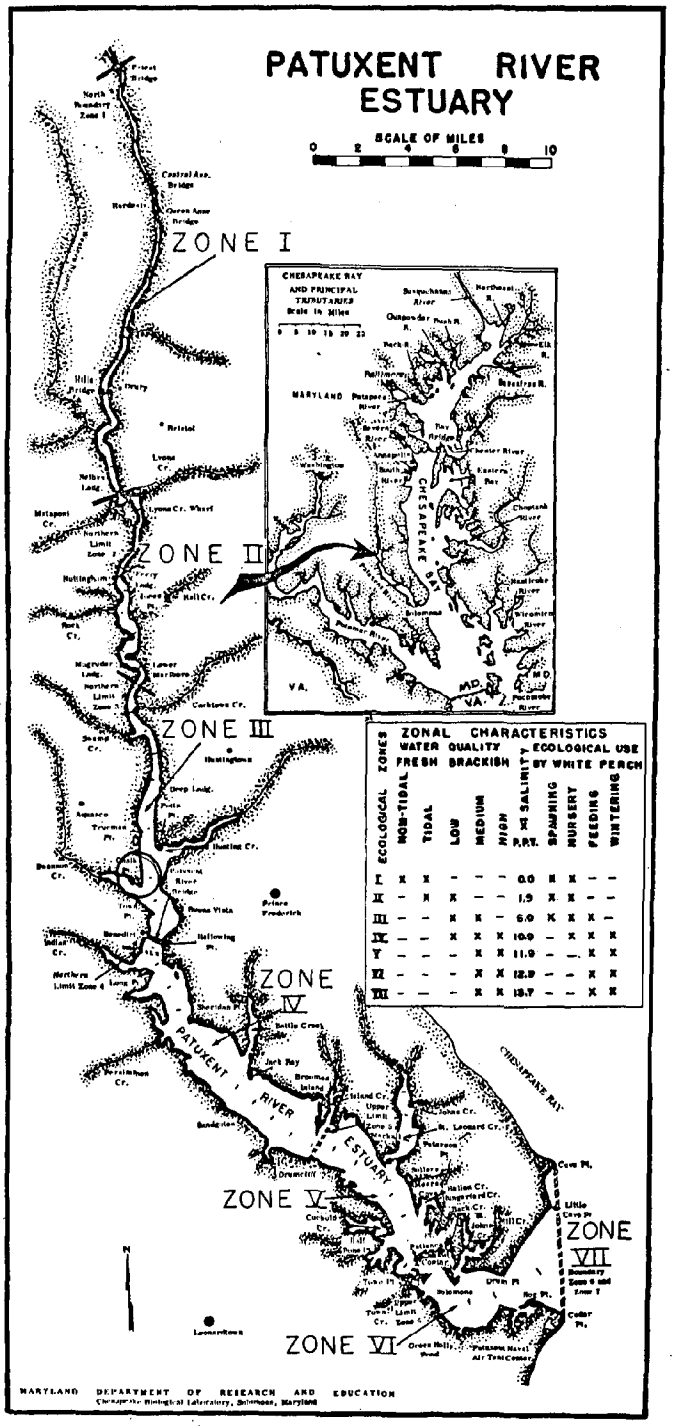


Fig. 1. Patuxent River estuary, Maryland, showing general ecological zones, collecting localities, and locations of tag returns of white perch, Roccus americanus. The inset of the Maryland portion of Chesapeake Bay shows the general location of the Patuxent tributary in relation to the upper part of the Chesapeake system. Relative salinity values in the various zones are as follows: low--1-8 ppt; medium--3-12 ppt; and high--6-15 ppt. Circle at Chalk Point indicates location of Pepko generating plant (adopted from Mansueti, 1961b).

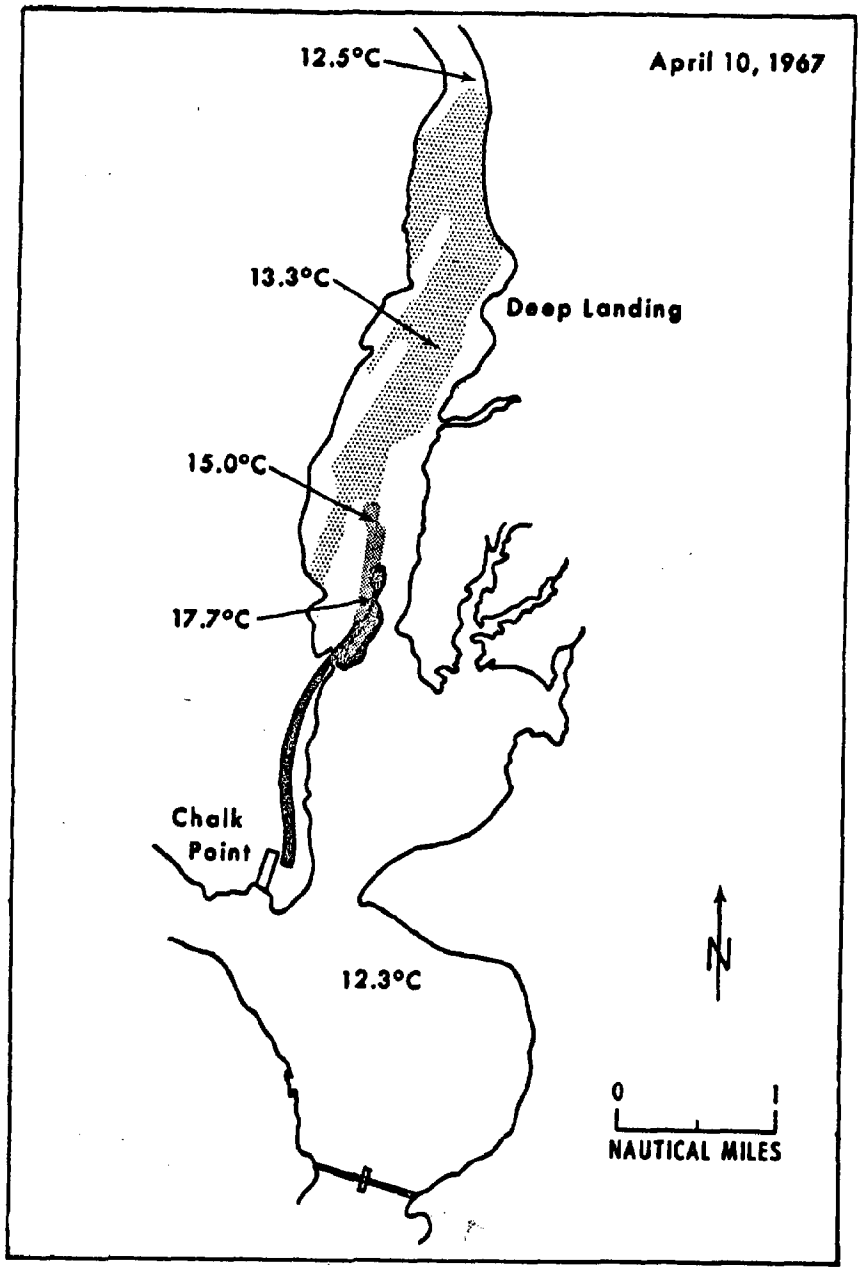


Fig. 2. Diagram of power plant effluent plume taken from N.A.S.A. U.S.G.S. infrared photo just prior to high slack water. Temperatures indicated are from nearby surface temperature recorders and are estimates of true values (From Cory and Nauman, 1968).

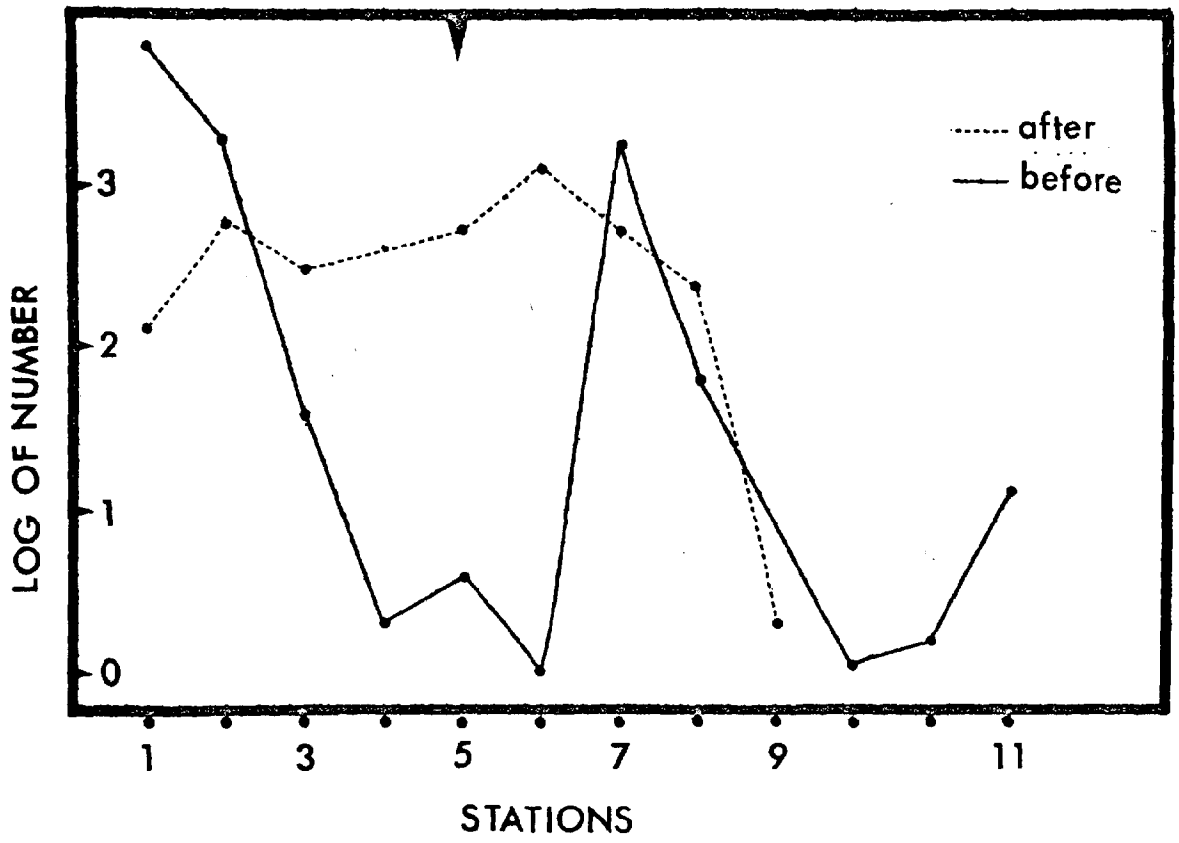


Fig. 3. Deep water trawl catch of white perch during winter in Patuxent River estuary before and after operation of steam electric station at Chalk Point, Maryland (From Mihursky, et al, 1967).

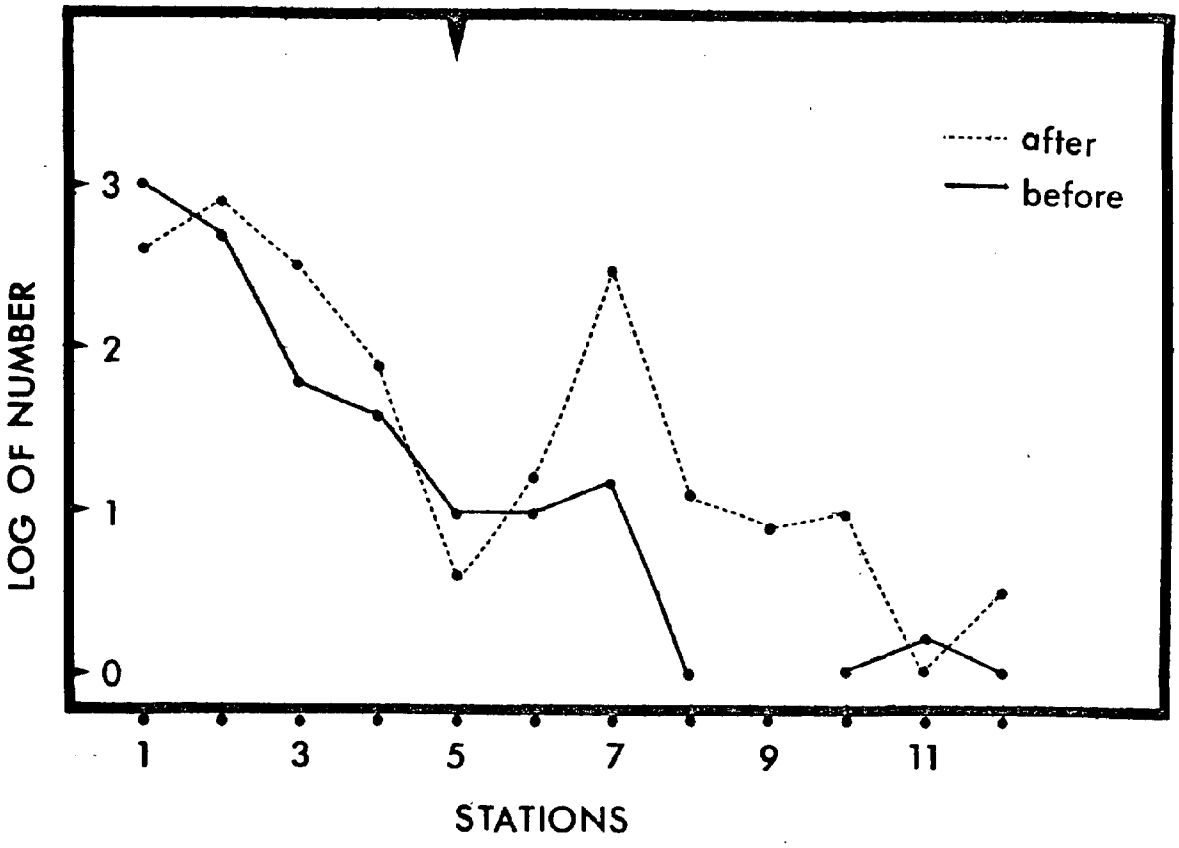


Fig. 4. Deep water trawl catch of white perch during summer in Patuxent River estuary before and after operation of steam electric station at Chalk Point, Maryland (From Mihursky, et al, 1967).

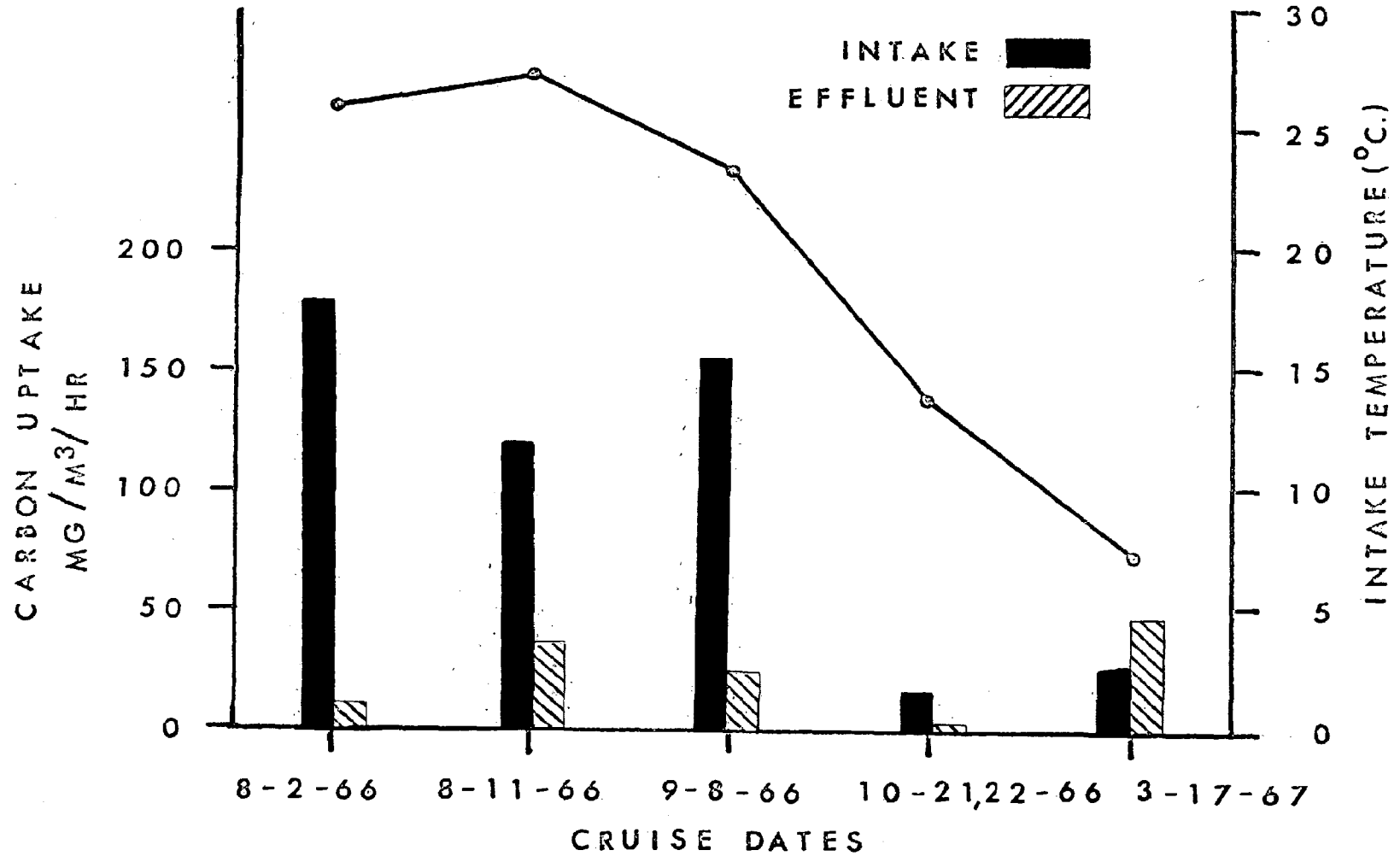


Fig. 5. Carbon uptake in water samples from the intake and effluent canals (histograms) and intake temperatures (connected points). The data shown are 10 A. M. to 12 noon samples from the dates indicated (from Mihursky and Stross, 1967).

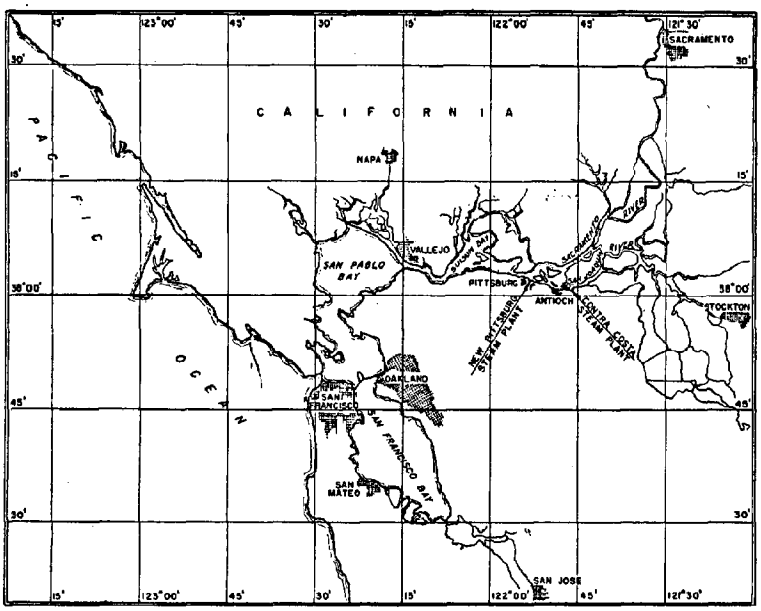
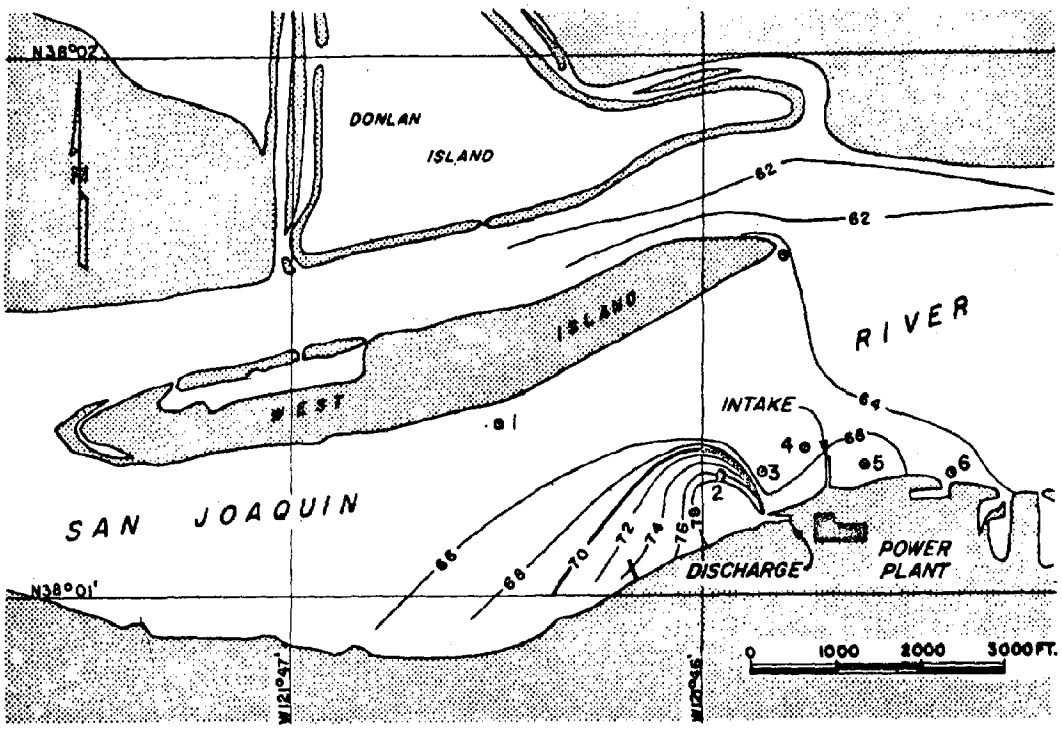


Fig. 6. Vicinity map of the Delta showing the locations of Contra Costa and Pittsburg Steam Plants of the Pacific Gas and Electric Company (From Kerr, 1953).



Run No.	1	2	3	4
Average MW	471	471	471	471

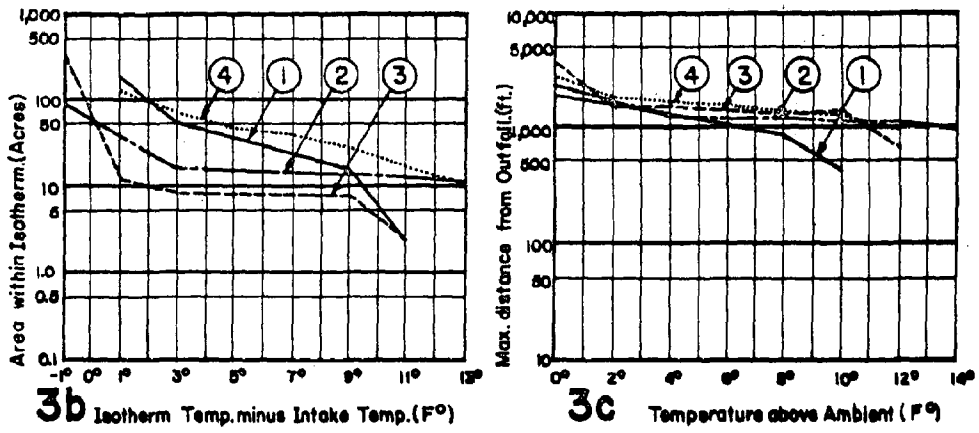


Fig. 7. Isotherms in the San Joaquin River estuary, California from run no. 3 taken May 21, 1964 at the Contra Costa generating plant and offshore thermal characteristics during 4 separate surveys at the same plant site (From Cheney and Richards, 1966).

Table 1. List of common and scientific names of fishes taken in the San Joaquin estuary - site of the Contra Costa generating plant (From Kerr, 1953).

ANADROMOUS FISHES

Pacific lamprey (*Entosphenus tridentatus*)
White sturgeon (*Acipenser transmontanus*)
Green sturgeon (*Acipenser medirostris*)
Shad (*Alosa sapidissima*)
King salmon (*Oncorhynchus tshawytscha*)
Steelhead rainbow trout (*Salmo gairdneri*)
Striped bass (*Roccus saxatilis*)

RESIDENT FISHES

Sacramento smelt (*Spirinchus thaleichthys*)
Freshwater smelt (*Hypomesus olidus*)
Western sucker (*Catostomus occidentalis*)
Carp (*Cyprinus carpio*)
Hardhead (*Mylopharodon conocephalus*)
Hitch (*Lavinia exilicauda*)
Sacramento squawfish (*Ptychocheilus grandis*)
Splittail (*Pogonichthys macrolepidotus*)
Channel catfish (*Ictalurus punctatus*)
White catfish (*Ictalurus catus*)
Brown bullhead (*Ameiurus nebulosus*)
Black bullhead (*Ameiurus melas*)
Starry flounder (*Platichthys stellatus*)
Largemouth black bass (*Micropterus salmoides*)
Warmouth (*Chaenobryttus coronarius*)
Bluegill (*Lepomis macrochirus*)
Black crappie (*Pomoxis nigro-maculatus*)
Freshwater viviparous perch (*Hysterocarpus traski*)
Prickly sculpin (*Cottus asper*)
Staghorn sculpin (*Leptocottus armatus*)

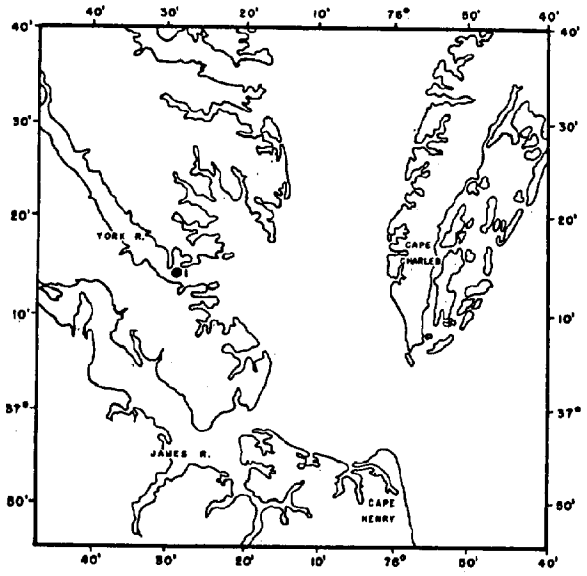


Fig. 8. Lower Chesapeake Bay, Virginia showing the approximate location of Virginia Electric and Power Company's steam electric generating station at Yorktown, Virginia (Adopted from Patten, et al., 1963).

of heated water on their downstream migration during the warmer summer months.

York River Estuary, Virginia

Heated water from a power plant at Yorktown is discharged into the York River estuary near its mouth (Fig. 8). Characteristics of the thermal plume are shown at flood and ebb tides in Fig. 9. Where there is appreciable stability imparted to the water column by temperature or salinity, the thermal discharges tend to be concentrated at the surface as shown in Fig. 9. This environment has an average salinity of 18‰ and is dominated by regular diurnal tides. Oysters are an important natural resource in this area and Warinner and Brehmer (1966) have shown (Fig. 10) that the species diversity of benthic fauna is lowered within several hundred yards by the heated water discharge.

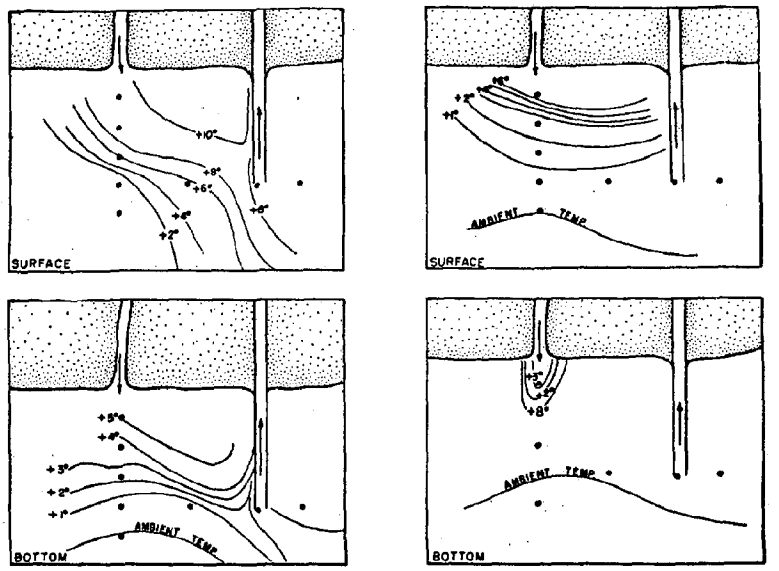
Morro Bay, Southern California

Here heated water is discharged directly into a high saline coastal marine environment (Fig. 12). North (1968a) observed that species diversity was affected by the heated water. In the discharge canal (Fig. 11) macroscopic algae were very rare but a total of 71 species of animals was found. Sea anemones that possess a symbiotic unicellular algae were particularly abundant, perhaps due to competitive exclusion of thermally sensitive species. However, in a similar canal discharging heated water into a marine coastal environment at Humbolt Bay, California, algae were apparently not so severely affected as reported by North (1968a) from personal communication from J. R. Adams. Here, there was an abundance of green algae (Ulva and Eteromorpha) in the canal and red and green algae (Ulva, Iridaea and Odonthalia) immediately outside. In the transitional zone (Fig. 11) algal species were sparse and the density of stands was low. Only seven species were found at transect A (Fig. 11) while 20 were observed at B in the normal region. Some preliminary evidence shown in Figure 13 indicated that the zonation patterns of the two zones was essentially similar, and there was no downward shift caused by the warmer surface waters. Only 27 animal species were observed in the transitional zone and low concentrations were characteristic. Recovery to normal conditions occurred abruptly within a horizontal distance of 10 m. In the normal region there were 33 species of macrophytes and 44 animal species.

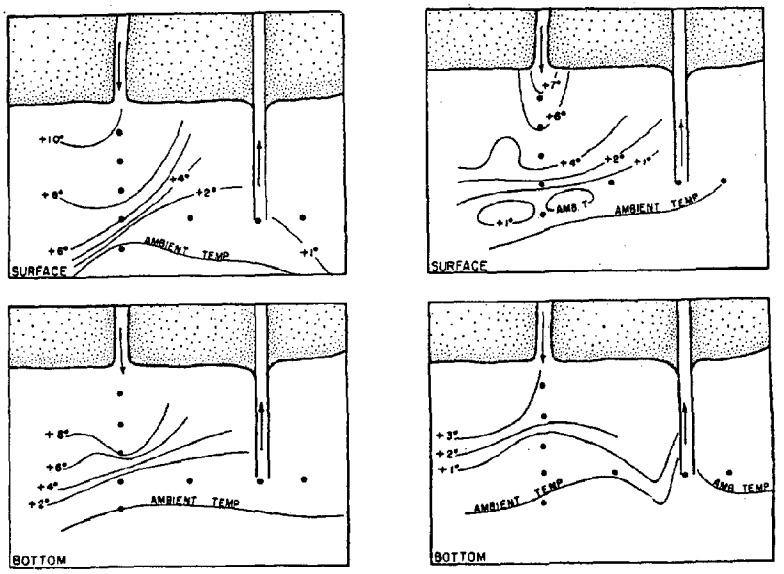
River Blackwater, England

This is a cold temperate high salinity estuary (31 - 33‰) discharging into the North Sea (Fig. 14). Heated water is released into the estuary along a barrier wall. Figs. 15 and 16 show characteristic thermal plumes at high slack water and the concentration of warm water at the surface. There is rapid cooling due to initial mixing of the heated discharge so that the greatest recorded increase in water temperature was 5.6°C within 100 m of the outfall (Fig. 15).

An intensive before and after study of oyster populations, plankton and smaller benthic invertebrates has not shown any changes which could be attributed to the heated water. The zooplankton fauna was characterized by Acartia, several other calanoid species and the harpacticoid copepod, Euterpina acutifrons (Figs. 17 and 18). There was a second brood of Euterpina at the Barrier wall



A



B

Fig. 9. Distribution of thermal effluents at surface and bottom during (A) Maximum flood current (left) and high slack water (right); (B) Maximum ebb current (left) and low slack water (right) (After Warinner and Brehmer, 1966).

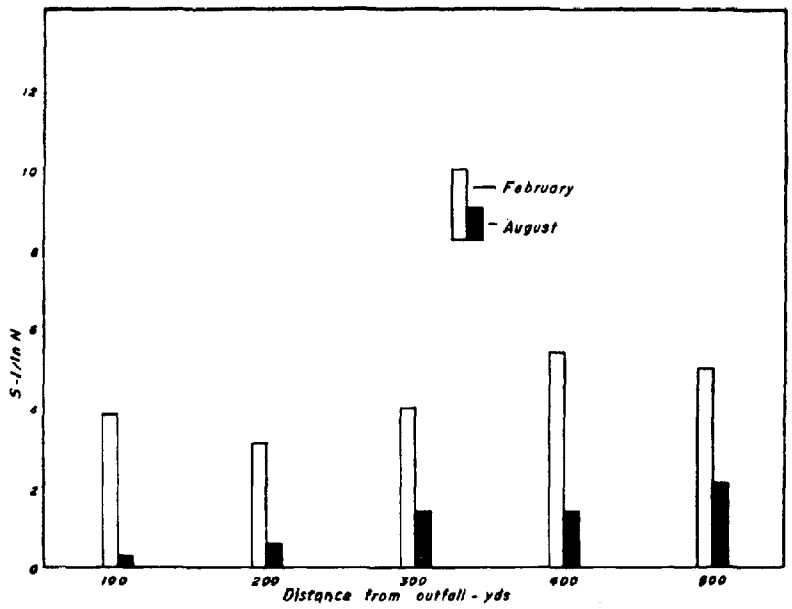


Fig. 10. Species diversity of benthic fauna according to the index S-1/ln N at stations opposite a discharge canal. (From Warinner and Brehmer, 1966).

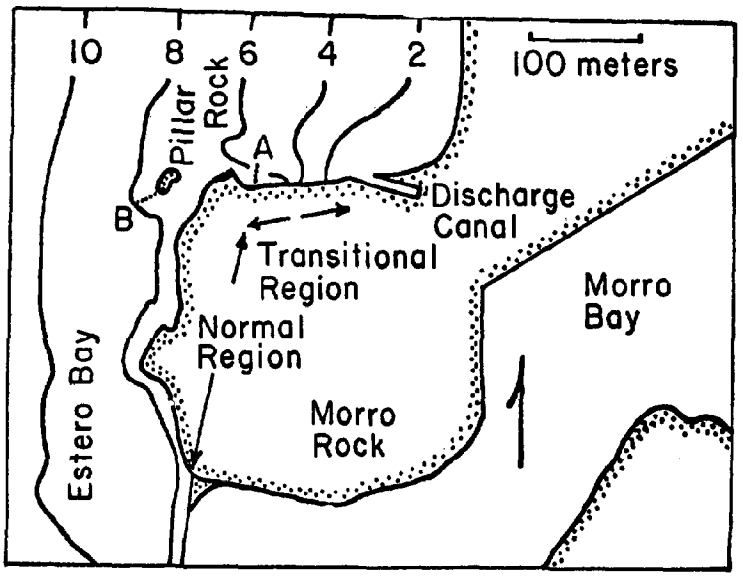
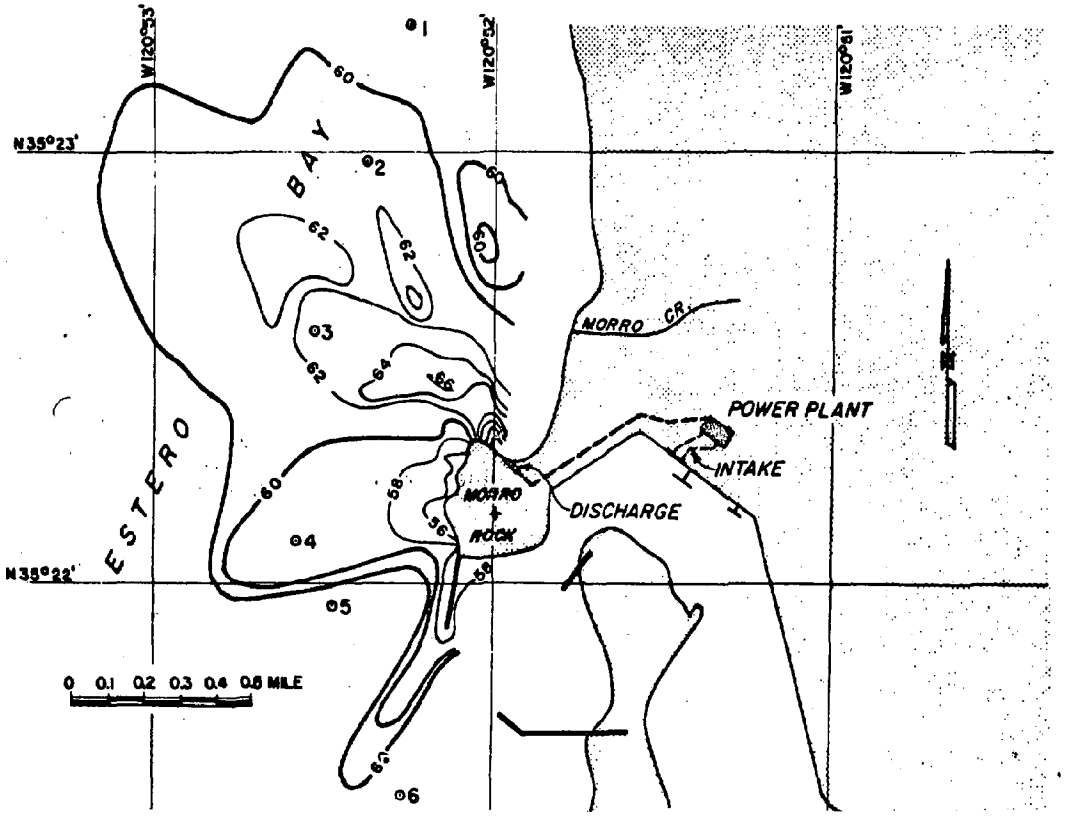


Fig. 11. Locations of the regions surveyed. Transects A and B shown as dotted lines. Depth contours in meters (From North, 1968a).



Run No.	1	2	3	4	5	6	7	8	9
Average MW	758	833	868	570	200	160	660	660	660

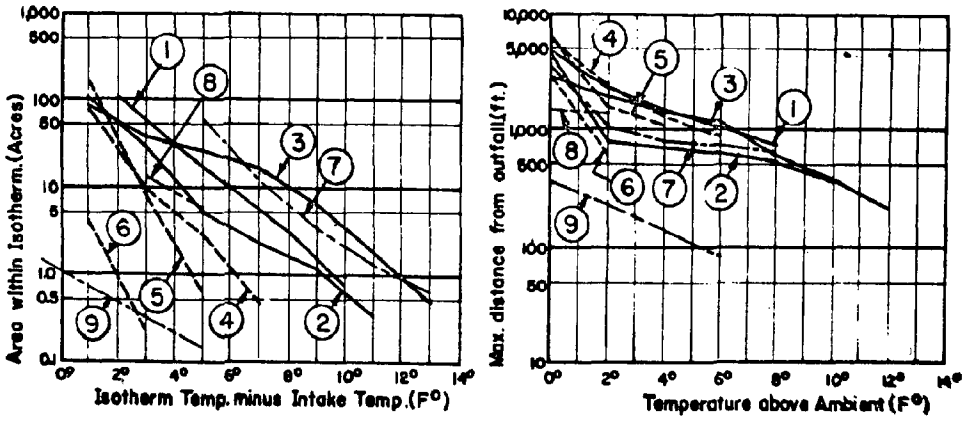


Fig. 12. Isotherms taken at Pacific Gas and Electric's Morro Bay Power Plant on run number 2, September 12, 1963; and offshore temperature characteristics taken on nine occasions at the Morrow Bay Power Plant (From Cheney and Richards, 1966).

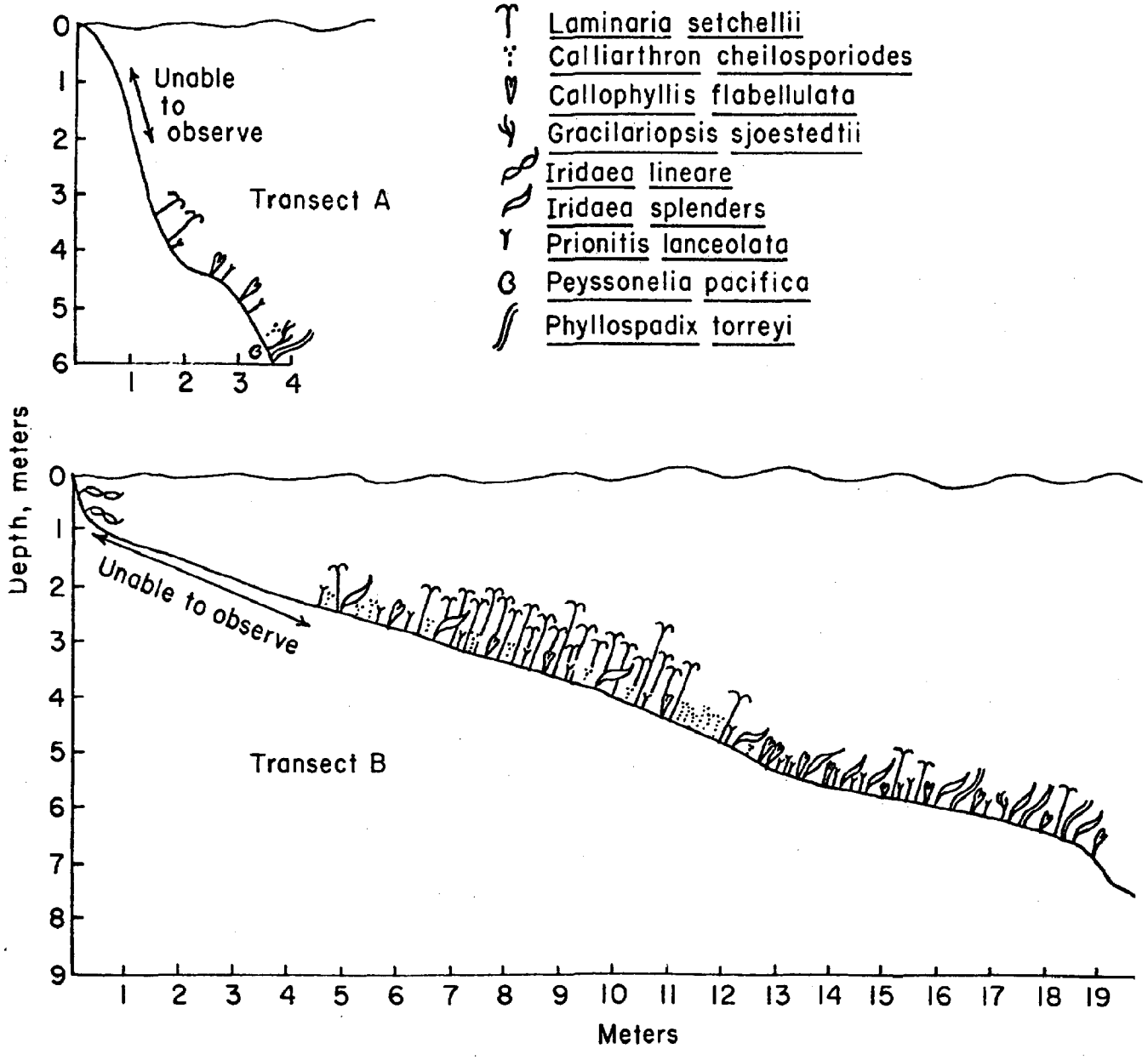


Fig. 13. Distribution of principal seaweed species along Transects A and B at Morro Bay Power Plant (From North, 1968a).

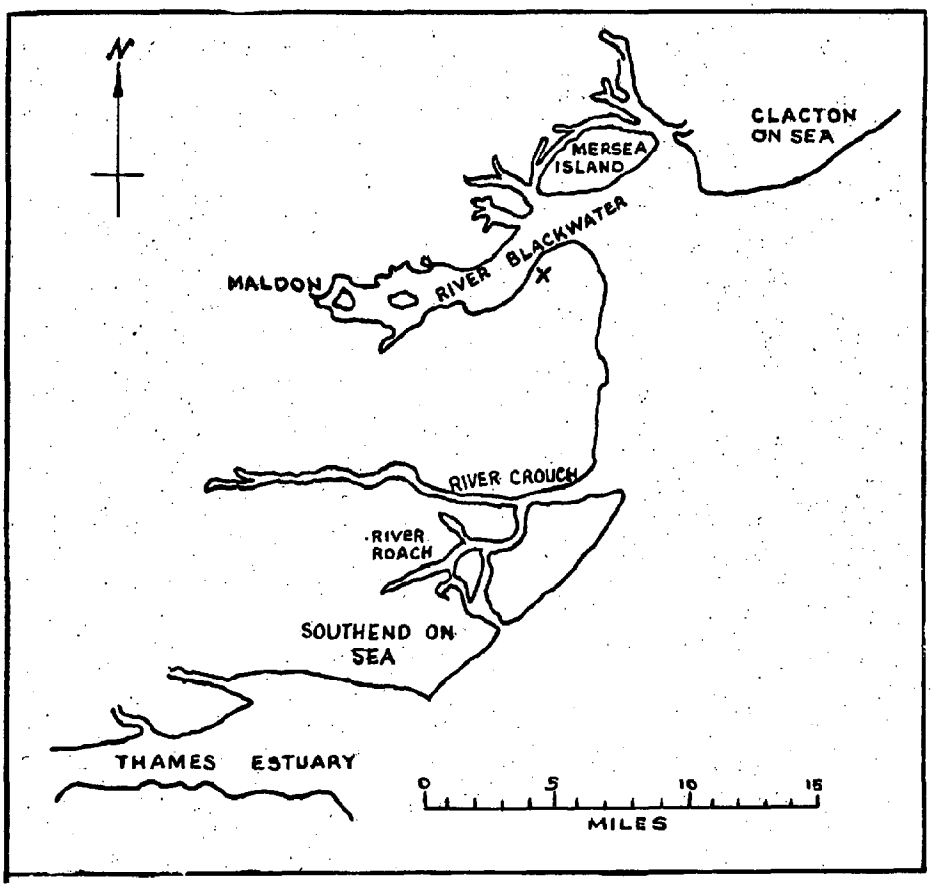


Fig. 14A. Blackwater Estuary with power station (x) in England (Hawes, 1965).

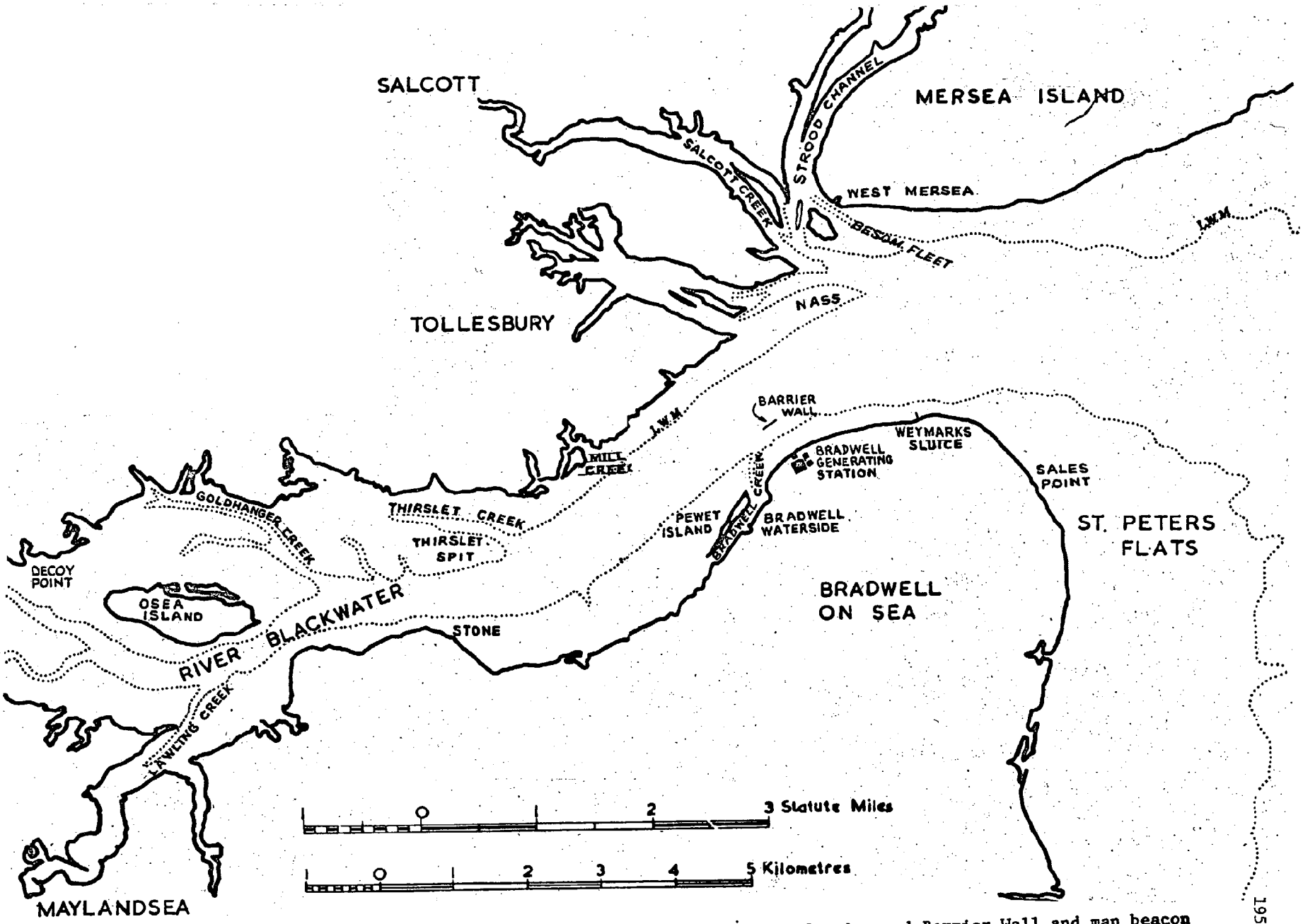


Fig. 14B. River Blackwater estuary indicating position of Bradwell Generating Station and Barrier Wall and man beacon comply areas (from Hawes, 1965).

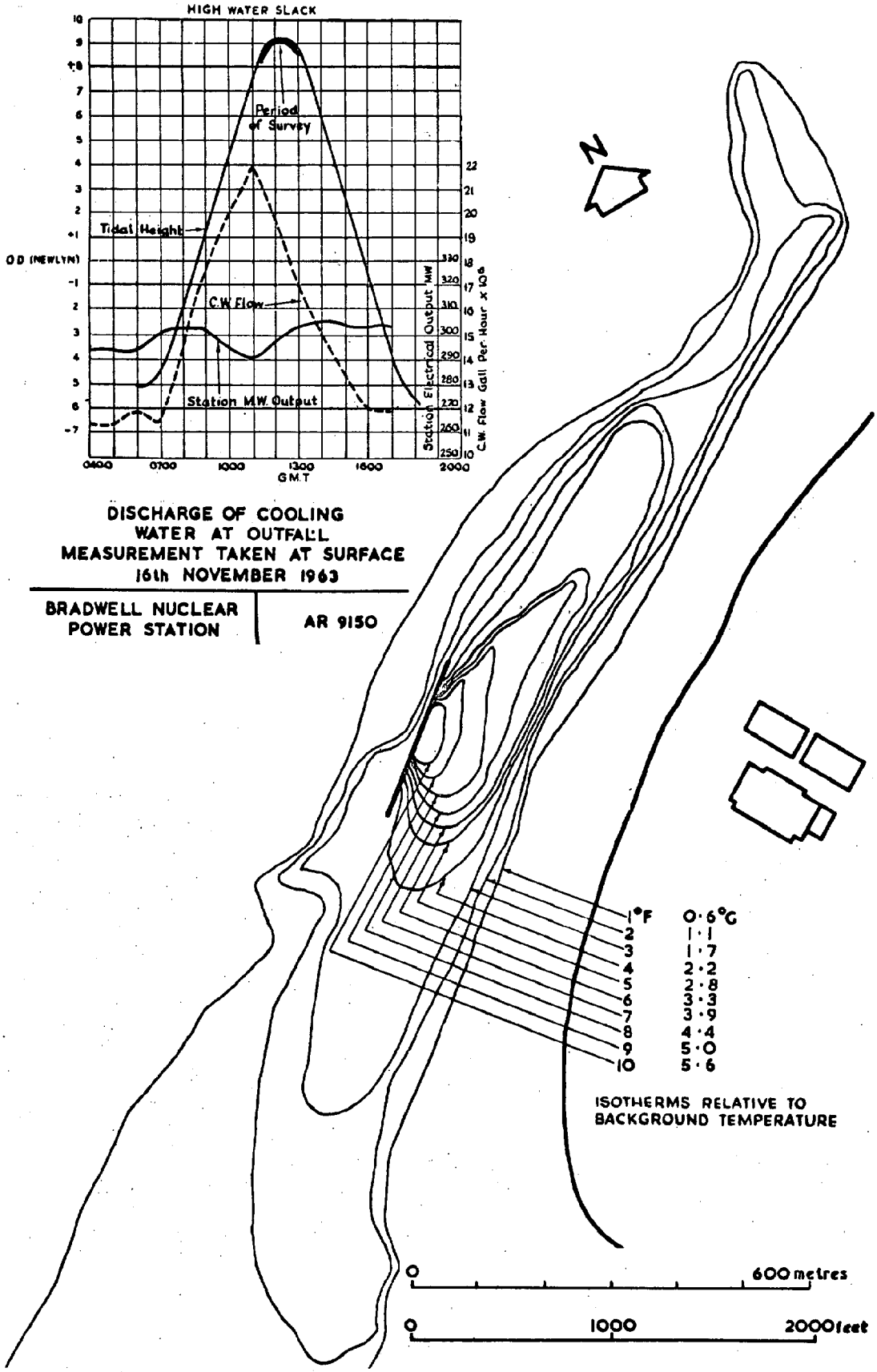


Fig. 15. Isotherms in River Blackwater estuary from Bradwell nuclear generating station at the surface at high slack water (from Hawes, 1965).

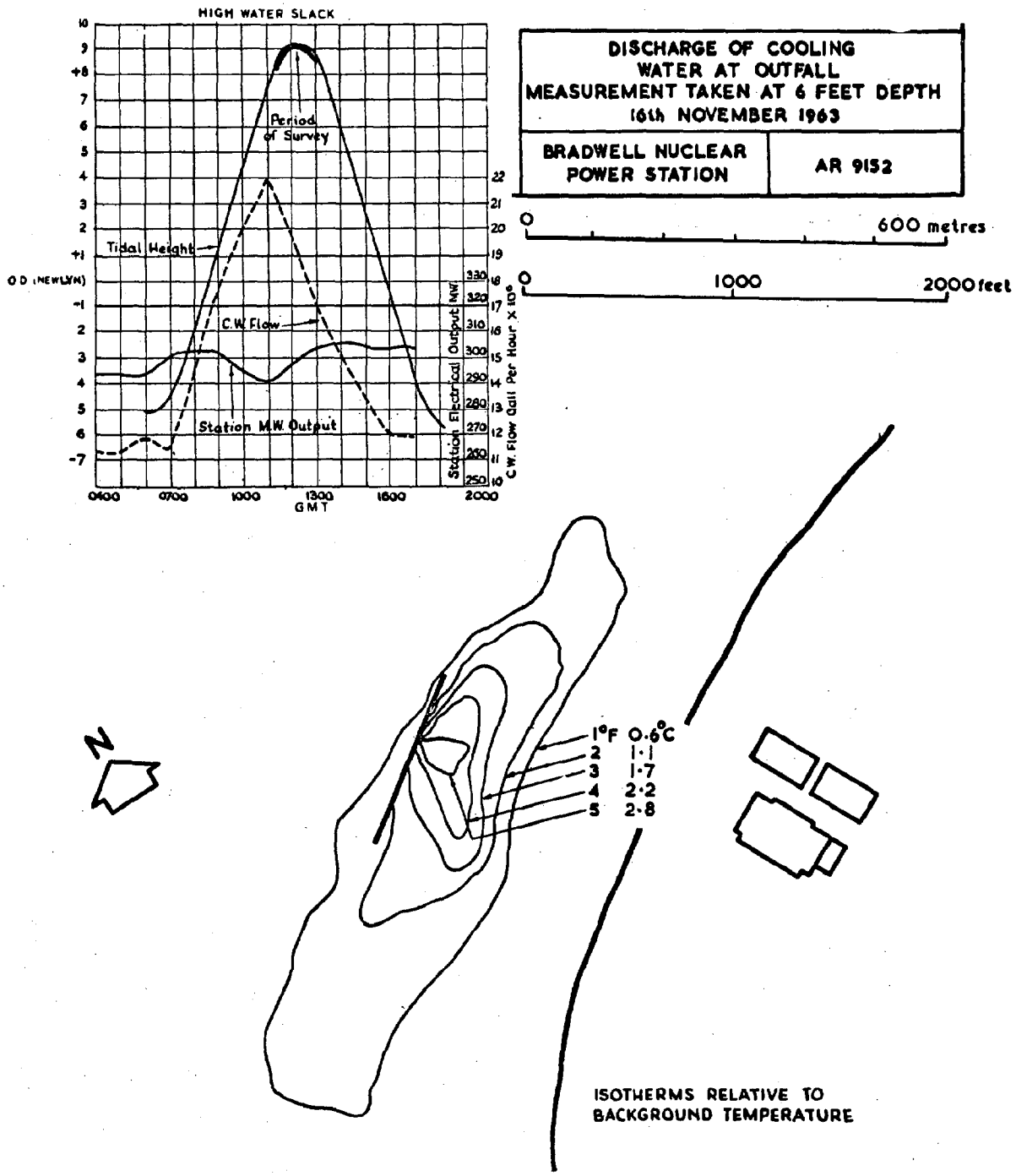


Fig. 16. Isotherms in the River Blackwell estuary at the Bradwell nuclear generating station on the bottom during high slack tide (From Hawes, 1965).

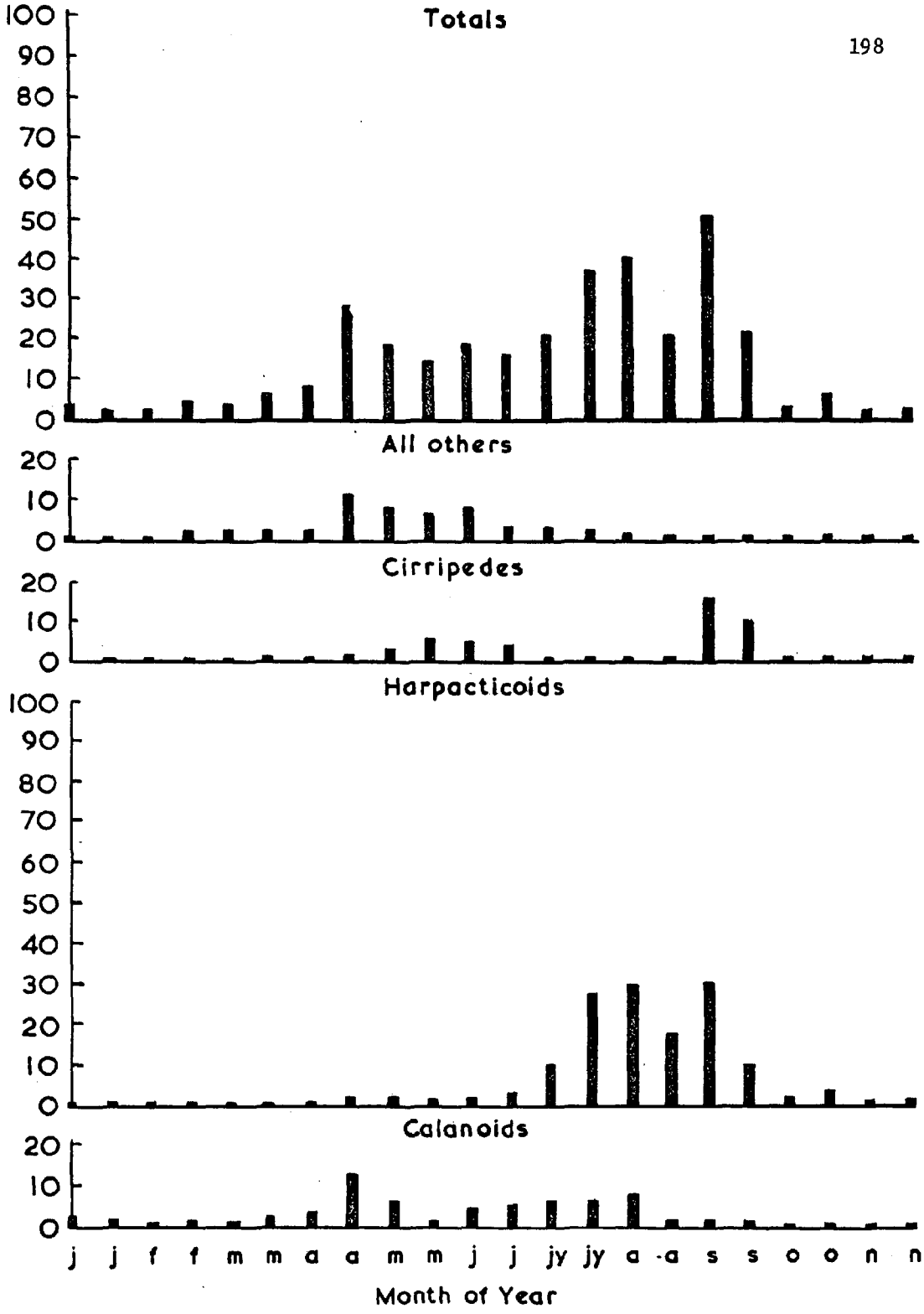


Fig. 17. Number of zooplankton organisms per $m^3 \times 10^3$ collected at Barrier Wall Station, River Blackwater estuary, 1964. (Hawes, 1965).

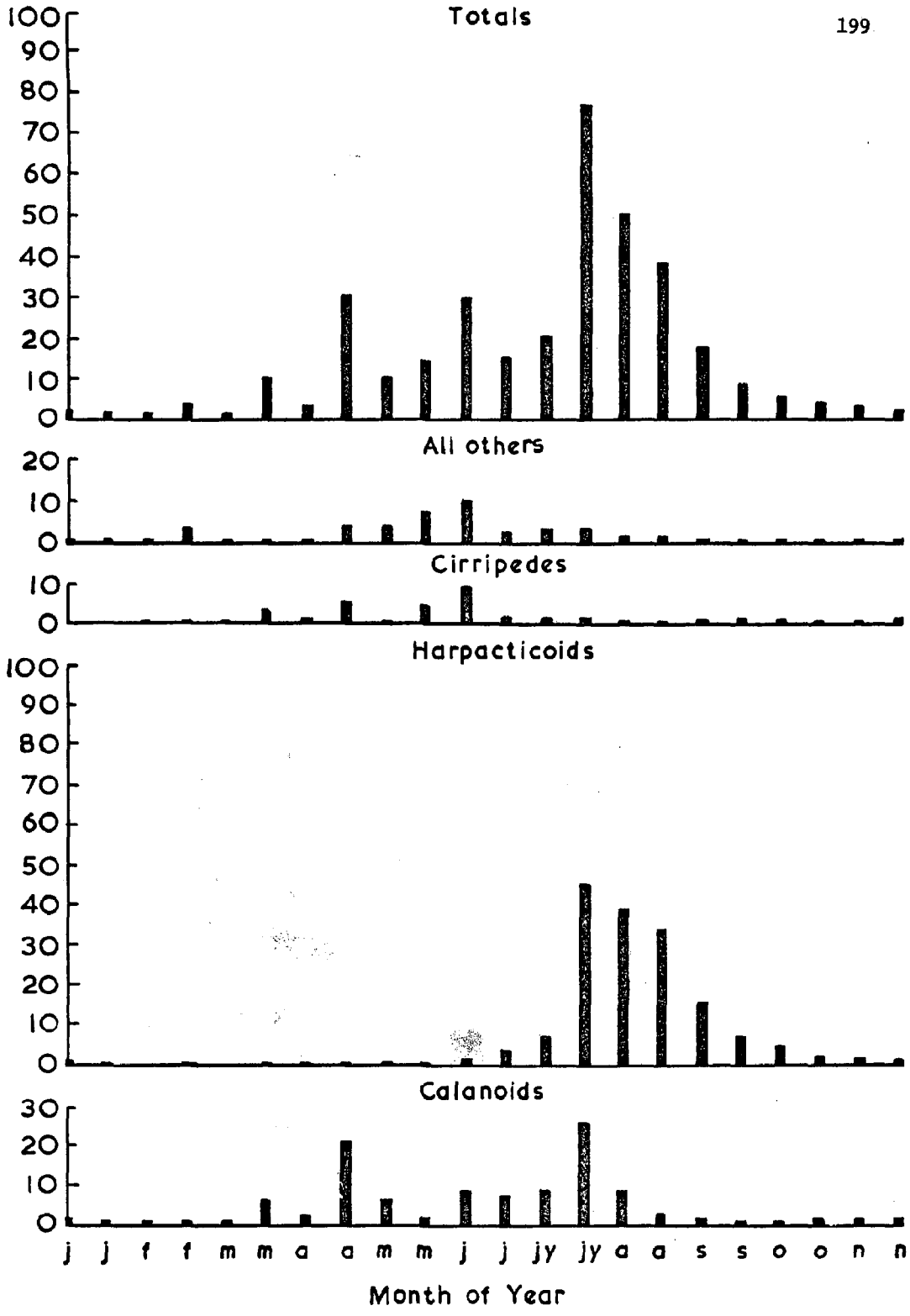


Fig. 18. Number of zooplankton organisms per $m^3 \times 10^3$ collected at Nass Beacon Station, River Blackwater estuary, 1964. (Hawes, 1965).

station (Fig. 17) not recorded at the station outside of the influences of heated water (Fig. 18) nor in the before study. Fig. 19 illustrates that there were considerable year to year fluctuations in the general succession and abundance of planktonic forms, but very little difference between heated water and normal stations. Although no effects were definitely attributed to the heated water discharges, it was emphasized (Hawes, 1965) that longer term studies are required. In some related studies, Crepidula transferred into a heated water trough at the plant site were found to grow faster, reach a greater size and spawn earlier than those placed in a cold water trough.

DISCUSSION

The effect of heated water effluents in estuaries is of only recent concern. One reason is that heat has not been considered a pollutant in the ordinary sense of the word in spite of the fact that heat kills commonly occur in the marine environment (Brongersma-Sanders, 1957). However, all estuarine systems are subjected to daily and seasonal temperature changes which trigger the inception of migration for many species of fish and the release of gametes by benthic invertebrates. Only with the tremendous growth of the power industry and particularly with the adoption of atomic fuel rather than fossil fuel has "thermal pollution" been added to the list of man-induced hazards to the estuarine ecosystem. It is obvious that if a cold water environment is replaced by a warm water regime that profound changes in the estuary will result and a new system will emerge.

In natural systems there are a few instances in which warm water meets cold water to bring about changes, sometimes catastrophic, in naturally adapted populations. In the Cape Cod Canal, significant temperature shocks are manifest on each change of the tide as warm water from Buzzards Bay meets the cold water from Cape Cod Bay. Fish have been observed to be affected and there is a very marked reduction of species diversity within the canal. Similar temperature shocks accompany the meander of the equatorial counter current (El Nino) when it comes close to the coast of Peru and causes tremendous mortalities of fish (Brongersma-Sanders, 1957). Glynn (1968) described heat kill of echinoids in a tropical reef environment and Hedgpeth and Goner (1969) a kill of sea urchins in an intertidal area in a cold-temperate environment on the central Oregon coast. As will be discussed later, frequent fish kills also accompany seasonal changes in temperature, particularly in temperate estuaries where there is a replacement of cold and warm water fish faunas seasonally.

Natural fluctuations in temperature have been correlated to abundance of fishstocks in a number of studies reviewed by Hedgpeth and Goner (1969). They stress that these long-term natural fluctuations in temperature evidently influence the presence or absence of many species and must be taken into consideration in studies on the possible effects of thermal effluents.

However, although these examples may be used to illustrate the potential dangers involving the release of thermal effluents, there is very little evidence that substantial heat kills have accompanied the installation of power plants on estuaries, and the indirect effects of heat and its interaction with other stress factors may be more important. Presently, a large and growing volume of research

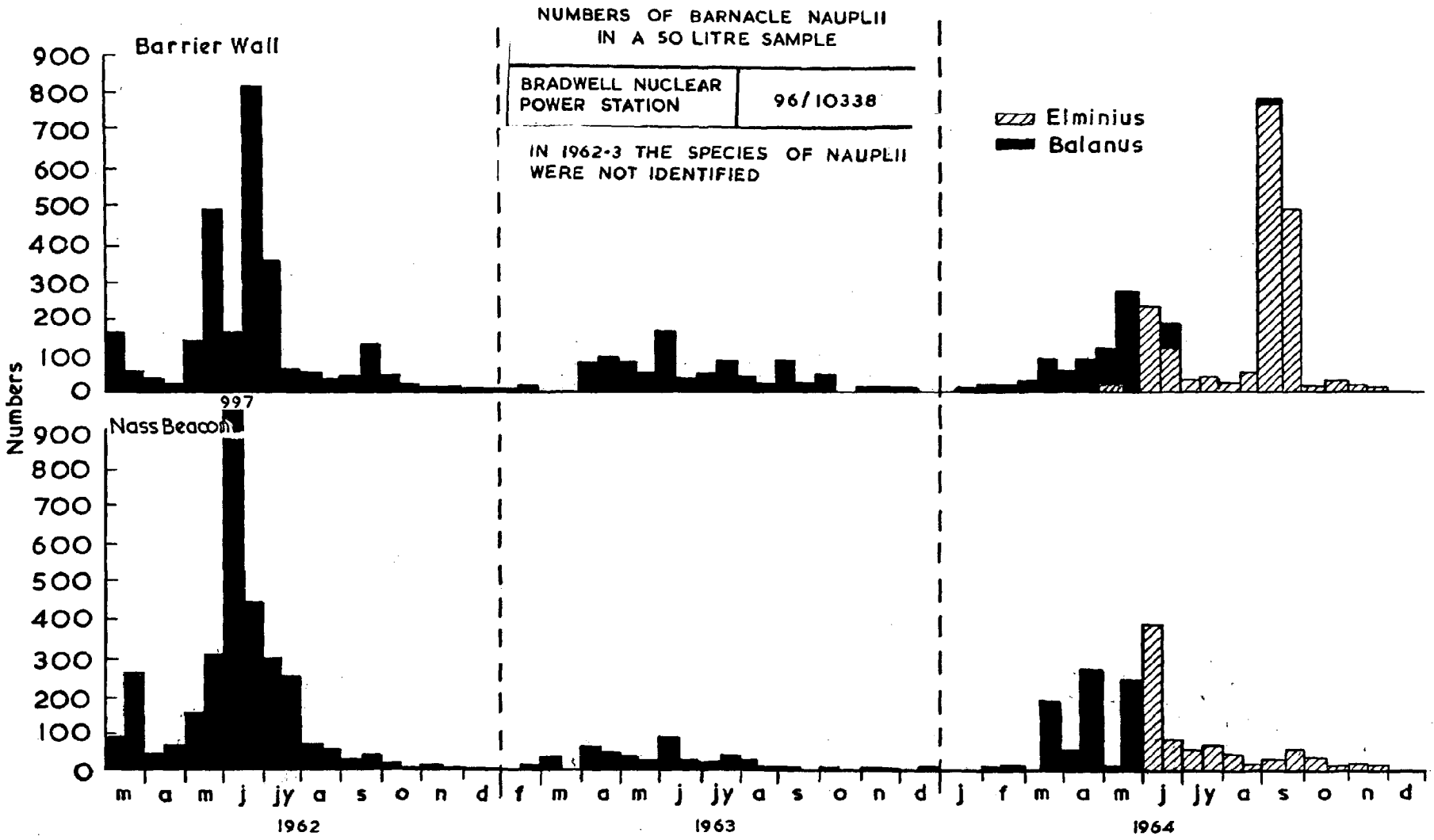


Fig. 19. Abundance of two species of barnacle *nauplii* during each month of 1962-1964 at the barrier wall station and the Nass beacon station in the River Blackwater estuary (from Hawes, 1965).

on the effects of heat effluents on estuarine organisms is underway. However, as yet very little has been published and most of the available information is preliminary in nature and does not involve complete estuarine systems. In this report we will attempt to summarize the important information available and present the possible consequences of thermal loading on representative estuarine types.

Figure 20 presents in diagrammatic position the general problems associated with a power plant heat exchange system. The flow rate figures and discharge plume characteristics are only representative and do not illustrate any specific installation. However, particularly in the case of atomic fueled steam electric generating plants which use about 30 percent more cooling water than fossil fueled plants (Hogerton, 1968), a large percentage of the net estuarine flow may be used. Figures from Squire (1967) and elsewhere indicate that the cooling water is heated about 10°C above ambient water temperature before being discharged. The thermal plume is affected by the flow characteristics of the estuary and can be modeled like any other introduced contaminant (e.g., Pritchard and Carter, 1965). Monitoring of water temperatures often shows that the plume is concentrated along the shore and that heat dispersion is rapid providing the receiving body of water is relatively large. The plume is also expressed vertically with the heated lighter water concentrated at the surface. Tide and wind displace the heated water so that large temperature fluctuations are characteristic except in, and immediately outside of, the discharge canal (Fig. 9).

Comprehensive bibliographies on thermal pollution have been compiled by Kennedy and Mihursky (1967) (1,220 entries), Raney and Menzel (1967) (1,217 entries), Raney and Menzel (1968)(400 entries), Gerber (1967) (863 entries). For additional information, review articles on the ecological effects of temperature on the marine environment (Gunter, 1957b), the physiological effects of temperature on marine organisms (Kinne, 1963a, 1964b; and Prosser, 1967) and the effects of heated effluents on marine organisms by Naylor (1965b) and Mihursky and Kennedy (1967) are particularly valuable.

Fig. 20 illustrates the three major consequences of heated water effluents from power generating stations:

1. Plankton, including fish eggs and larvae, and the larvae of benthic invertebrates can pass through the intake screens of the plant and are subjected to a rapid temperature increase.

2. High temperatures can directly affect organisms in the vicinity of the discharge canal. Depending on latitude and season, the lethal limits of endemic species could be reached in this area.

3. Outside of the zone of maximum temperature increase, a variety of direct and indirect temperature effects may be important. Temperature fluctuations can bring about inopportune gamete release by invertebrates and heat or chill death of fish. The joint effects of temperature, dissolved oxygen and salinity and the indirect effect of temperature on stability of the water column may be more important than any direct effect of a modest temperature increase.

Although maximum water temperatures in estuaries during the summer are

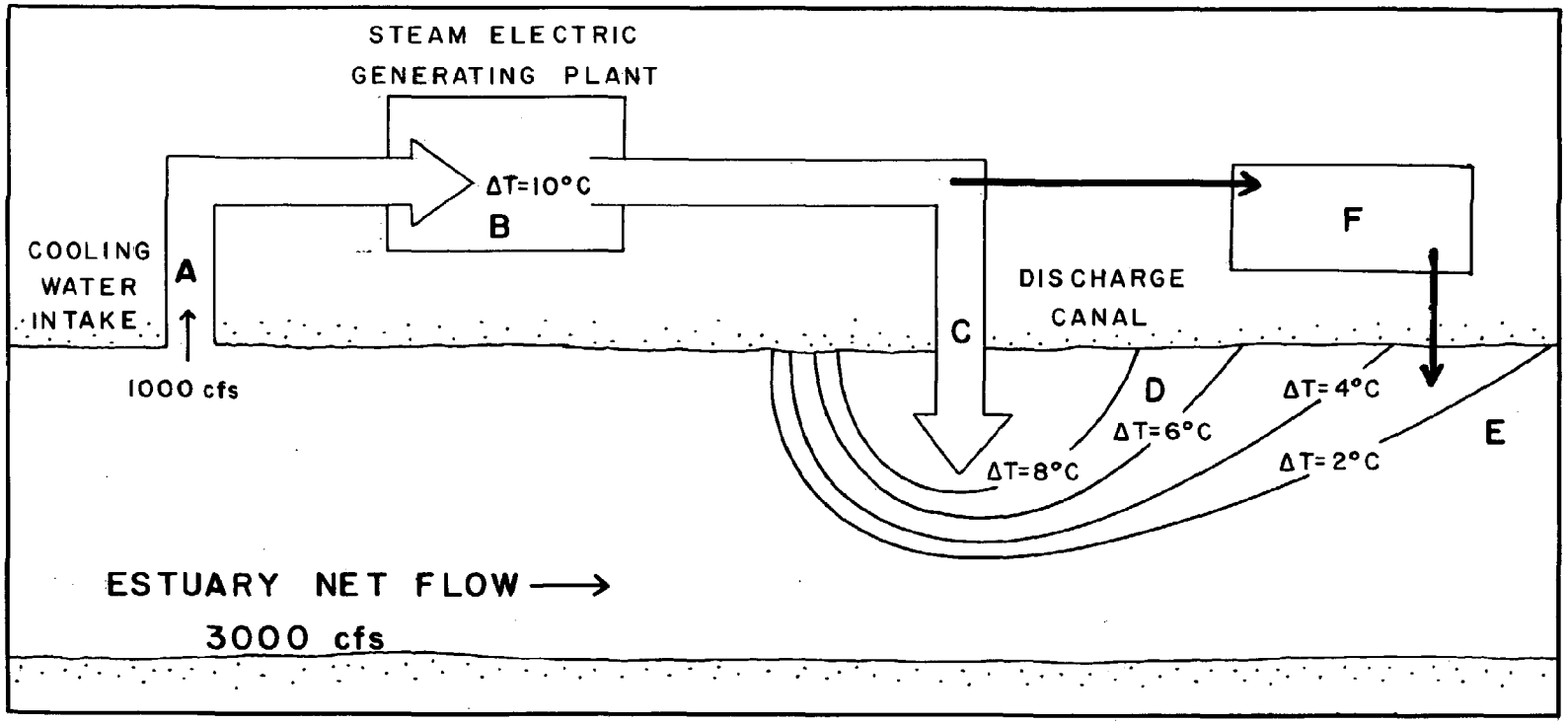


Fig. 20. Diagrammatic representation of a cooling water diversion through a power plant. Locations are shown where important thermal effects on an estuarine ecosystem may occur. (A) A significant proportion of the net estuarine volume flow may be used for cooling. (B) Entrained organisms (plankton and juvenile fish) are subjected to a rapid increase in temperature up to about 10°C above ambient. They are subject to the high temperature for varying periods of time depending on plant design, volume flow, and size of the discharge canal. (C) Fish are attracted to the discharge canal where they can be killed by a combination of high temperature and low dissolved oxygen. Fish can also be trapped in (A) when they clog the intake screens. Species diversity of benthic organisms may be severely affected at (C) and (D). At (E) various long term effects on increased temperature on water column stability, plant and animal respiration, benthic invertebrate feeding and growth rates may result in population shifts and species replacement. Fish migration may be blocked, and egg and larval development affected by meanders of the thermal plume. (F) Controlled system aquiculture could utilize the heat energy to commercial advantage.

quite similar throughout a broad geographical range (28-30°C), in the more northerly areas, surface water temperatures in excess of 28°C are rare. However, to the south, surface water temperatures in shallow estuaries with restricted tidal exchange are commonly greater than 30°C for prolonged periods, and heat related fish kills frequently occur. Additional heat loading in these environments pose particular hazards.

Certain estuarine systems may be particularly endangered by waste heat. An oligohaline system already severely stressed by salinity and low in species diversity may be more affected by thermal discharges than a more stable system which has a greater capacity to absorb thermal shocks. Miltiple stressed systems of all kinds are similarly subject to greater danger.

Plankton Based Oligohaline Systems

Although temperature is of importance to phytoplankton in culture since many species have specific optima for growth, virtually no bioassay of the effect of a sudden exposure to high temperature has been made for any planktonic organism (Allen, 1969). Studies by Mihursky and Stross (1967) have shown that up to 95 percent of phytoplankton and zooplankton are destroyed after passing through a steam electric generating station. In view of the large quantities of water used in atomic-fueled plants, this effect becomes extremely important. If the regeneration rate is not able to keep pace with the rate of plankton destruction, then many species could become severely depleted. This effect may be particularly severe in certain oligohaline river systems where most of the river flow passes through the turbines and tidal exchange is minimal.

The seasonal succession of algal species is regulated in part by seasonal temperature changes. Cairns (1956) and Patrick (1969) show in an example appropriate to oligohaline systems, that diatoms, green algae and blue-green algae may succeed each other (Fig. 21) because of differences in temperature optima. The blue-greens with a temperature optima over 35°C may be favored in regions of greatest heat loading in the summer. However, as Patrick (1969) points out in her summary of temperature effects on fresh water algae, many other interacting ecological factors such as light and nutrient availability also influence the seasonal succession of species. Provasoli (1958), for example, noted that vitamin B_{12} requirements for flagellates were found to increase several hundred-fold with a small rise in temperature near the lethal point.

Bottom Based Systems

A general reduction in species diversity below thermal effluent outfalls has been noted by Trembley (1960, 1965) and Cairns (1969) in freshwater environments, Warinner and Brehmer (1966) in a medium salinity estuary and North (1968a) in a high salinity neutral embayment. However, reduction in species diversity does not always appear to be a consequence of heated water insertions (Crains, in press), and probably occurs only in locations which are subjected to high temperature stress for relatively long periods of time (Patrick, 1969). For example, North (1968a) found a reduction in species diversity of animals and macroscopic algae over a distance of only 200 m from the mouth of a heated water discharge canal with a cooling water capacity of 300 m^3/sec. Recovery to normality occurred within 10 m of the outer terminus of this zone. In the Hunterston Investigations in Great Britain, Barnett and Hardy (1966) have not

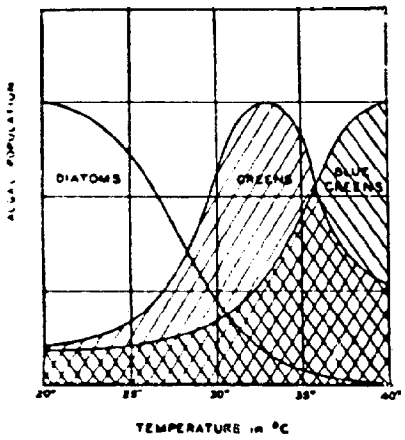


Fig. 21. Algal population shifts with temperature (from Cairns, 1956).

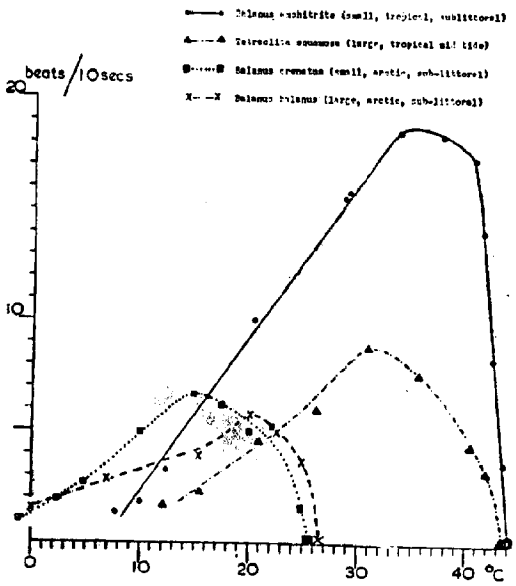


Fig. 22. Cirral activity/temperature curves for pairs of arctic and tropical species of barnacles (from Southward, 1964).

observed any effect of heated water discharge on molluscan fauna during the first two years of plant operation. They report on laboratory investigations with Dosinia exoleta and Tellina tenius and observe that the pelagic phases are shortened at higher temperatures and that this effect would be expected in the vicinity of the effluent outfall. However, the warming did not cause Tellina to spawn earlier. Perhaps the sampling techniques in current use are not sensitive enough to demonstrate changes in benthic invertebrate populations that can be attributed to thermal effluents except in the immediate vicinity of the discharge canals. Subtle population shifts by competitive exclusion may only gradually be manifest and long term studies involving carefully designed sampling programs may be necessary to document changes. At this time we can only infer from laboratory experiences what effects increased temperatures will have on natural communities.

In general, benthic species have specific temperature tolerance limits. Some are stenothermic, others eurythermic, but all can readily adapt to the natural temperature fluctuations of their environment. When species are stressed with temperature fluctuations greater than those experienced in nature, various effects have been noted. There can be a general failure to reproduce, or the larvae will not develop normally. Hedgpeth and Goner (1969) have reviewed the influence of temperature changes and thermal stress on estuarine benthos. They emphasize that temperature is the most important factor in synchronizing reproductive periodicity of benthos with the seasons. Light and other factors are only of secondary importance. Temperature fluctuations can trigger gamete release by invertebrate animals. If spawning occurs when other environmental requirements (e.g., food or temperature levels outside of the thermally stressed area) are not suitable then the larvae will not develop. Southward (1964) in studying cirral activity in barnacles found that although activity was present over a wide range of temperatures, optimal activity was rather narrowly limited, and that different species had different optima (Fig. 22). This implies that a heated water area in a temperate estuary may favor the survival of species at the northernmost edge of their range which, under the new environmental conditions, have a greater filtering efficiency than endemic species.

Controlled Systems

In temperate and sub-arctic estuaries heat stress may result in the replacement by immigration of species ordinarily restricted to warmer temperatures farther south. Naylor (1965b) suggests that deliberate introductions of pre-adapted species could be made to replace endemic species eliminated by pollutional stress. Ansell, et al. (1964) suggest that this phenomenon of species replacement could be used for commercial advantage in growing clams. He describes a study (Ansell, 1962) which utilized the heated water effluents from a power station in England to grow the introduced American quahog, Mercenaria mercenaria. Gaucher (1968a, 1968b) has reviewed the world-wide experience and commercial value potential in the use of heated water effluents to optimize growth of various species in aquicultural practices. The use of waste heat energy from power stations for controlled system culture would help to prevent thermal pollution of naturally adapted systems. Also, multiple pollutants (e.g., heat, sewage wastes and CO_2) can be advantageously incorporated into aquicultural plans (Ansell, 1962).

Doudoroff (1969) states that results from bioenergetic studies on

fish should form the basis for water quality temperature criteria. Inasmuch as fish can exist under a wide range of temperatures, water temperatures should be maintained which optimize the production of valuable fish species (Doudoroff, ibid) and not merely allow for the fish's continued existence. Unfortunately, a complex estuarine system is composed of many species within different parts of their geographical range, having a variety of optimal temperatures for growth and production. Perhaps bioenergetic studies would be of greatest advantage in specifying the optimal temperature regimes in controlled systems, where one or a few species would be managed for maximum production at carefully monitored water temperatures, salinities and feeding rates.

Temperature and Productivity

In nature, the effect of increased temperature on photosynthesis is probably unimportant although very little work has been done on this (Raymont, 1963a). Wimpenny (1958) found that the percentage daily increase of carbon uptake of Rhizosolenia was 55% at 5°C, 122% at 10°C and 168% at 15°C using a standard illumination of 16,000 lux. Phinney and McIntire (1965) found that oxygen evolution of algae in artificial fresh water streams increased at higher temperatures when light intensity was 22,000 lux, but was not affected at 11,000 lux. At the relatively lower light intensities found in turbid estuarine environments, high temperatures may increase algal respiration more than photosynthesis and net productivity would decrease. Mihursky and Stross (1967) have demonstrated substantial differences in carbon assimilation values for phytoplankton between the intake and thermal effluent discharge waters when ambient temperatures were greater than about 24°C (Fig. 5). Some of the differences shown may be due to the destruction of algal cells in the passage of water through the condensers.

The indirect effects of increased temperature on estuarine primary productivity are much more important than any direct effect. Bacteria multiply faster and bacterial respiration increases at higher temperatures. Thus

at high temperatures the mineralization of organic matter to nutrient salts is enhanced. Low oxygen concentrations brought about by bacterial respiration allows the release of nutrients through the reduced mud-water interface. However, Sieburth (1967) presents evidence that bacteria in the water column may "void nutrients or recycle them" in such a way as to prevent sustained phytoplankton growth. His work was done in Narragansett Bay where bacterial populations were high in the summer when phytoplankton populations were at their lowest ebb.

A rise in surface water temperature increases the stability of the water column; a condition that favors phytoplankton growth in the spring of the year in temperate estuaries. Vertical mixing is reduced by the thermally induced stability, and plant cells are allowed to remain relatively longer in the nutrient rich euphotic zone. Later in the season, however, the stable water column resists wind mixing, and nutrients are not renewed to the now impoverished euphotic zone. Although most estuaries are not sufficiently deep or stratified severely enough to prevent occasional nutrient replenishment in the summer months, thermal effluents could increase the stability and the frequency of surface water nutrient depletion in already stratified estuaries. In severely stratified estuaries like Pettaquamscutt River, Rhode Island, heated water effluents at the surface would tend to permanently seal the bottomwaters from the surface. This estuary exhibits anaerobiosis with high concentrations of hydrogen sulfide below sill·

Infrequently during the fall, a reduced thermocline and halocline (in dry seasons) allows brief periods of wind mixing with resultant fish kills in shallow waters. Induced stability from heated water effluents may prevent this from happening.

Temperature and Fish

Heat related fish kills are common in temperate estuaries where sudden changes in temperature, salinity and dissolved oxygen can occur. Fish can acclimate quite rapidly to increased sub-lethal temperatures. Doudoroff (1942) indicates that fish can acclimate to a 10°C rise within 24 hours. However, the rate of acclimation to colder temperatures may be very slow. This latter physiological characteristic of fish poses specific problems in temperate estuaries during the winter months where hot water effluents are present. Small fish passing through the turbines or larger fish in the discharge canal could acclimate to the higher temperatures, but may not be able to acclimate to cold temperatures with subsequent plant shutdown or with displacements of the thermal plume by wind or tide. Even if fish were not killed outright, immobilization by cold shock would render them susceptible to predation. Each species has specific thermal limits (Fig. 23), and these limits are narrower for successful larval development and narrower still for maximum egg hatch (Fig. 8). Salinity interacts with temperature to affect hatching success and larval development of estuarine fish (Fig. 24). Work in progress at the Pamlico Marine Laboratory (Horton and Bridges, unpublished) shows that salinity acclimation has an effect on the resistance to high temperatures of some species of estuarine fish but does not have a significant effect on others. When there is an effect, survival at increased temperatures is enhanced at higher salinities. Spot, Leiostomus xanthurus, have a significantly greater resistance to high temperatures when acclimated at 16 hour photoperiods than fish acclimated at 8 hour periods (Horton and Bridges, unpublished), indicating possible seasonal differences in the ability of fish to withstand higher temperatures. Fig. 25 shows the effect of temperature on oxygen consumption of different fresh water species. Fish metabolize more rapidly and require more oxygen at higher temperatures, but the solubility of oxygen in water decreases with a temperature increase. The multiple effects of high surface water temperature during the summer on oxygen solubility, plant and animal respiration and water column stability, have resulted in significant fish kills (Fig. 26). A kill similar to the one illustrated in Fig. 26 occurred on the North Shore of the Pamlico River estuary in North Carolina in the summer of 1967 following a strong northeasterly wind. Surface water blown offshore by the strong wind was replaced by deeper oxygen-deficient water.

Migratory Sub-Systems

Particular problems are posed in regard to migratory estuarine fishes and heated water effluents. When anadromous fish move into an estuary on their way to upstream areas for spawning, they are liable to be affected in several ways by heated water in the vicinity of the discharge canal. Many species of fish when offered alternative choices of routes in an upstream migration will choose the route with the greatest current velocity. Alabaster (1969) has found that when fish (bream) are presented with a choice of temperatures, the fish will choose according to their prior thermal history (Fig. 27). He also found (Alabaster, 1964) that when fish are in a thermal gradient, they will

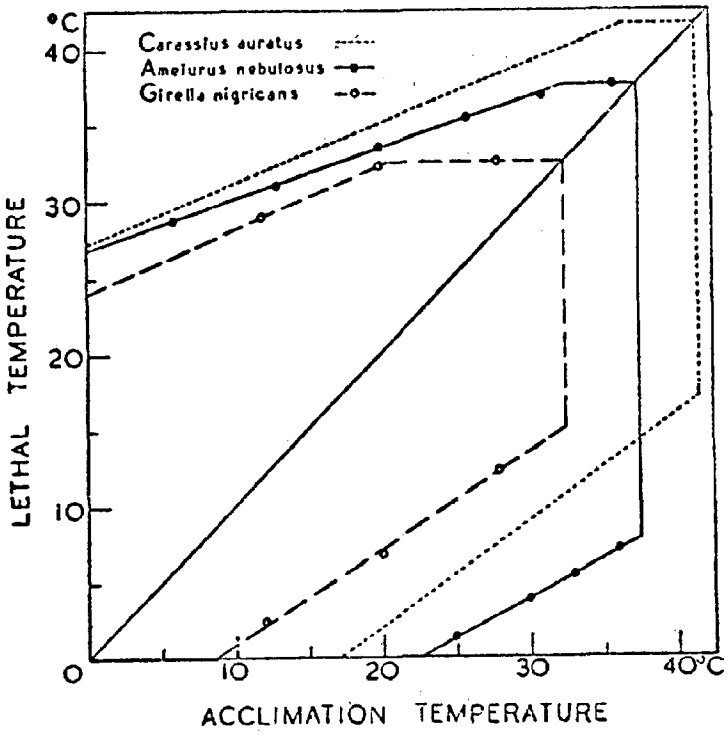


Fig. 23. The relation between the acclimation temperature and the upper and lower lethal temperatures for Carassius auratus, Ameiurus nebulosus, and girella nigricans (from Brett, 1944).

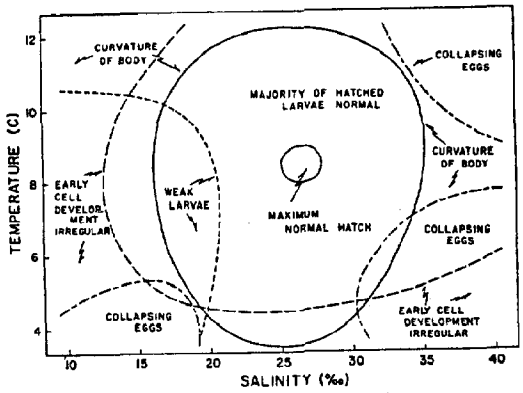


Fig. 24. Approximate areas over the salinity - temperature range examined in which various qualitative differences in development of English sole (Parophrys vetulus) were observed (from Alderdice and Forrester, 1968).

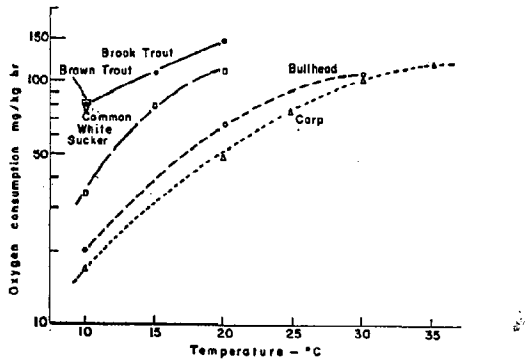


Fig. 25. Standard oxygen consumption in relation to temperature for four freshwater fish species (from Beamish, 1964).

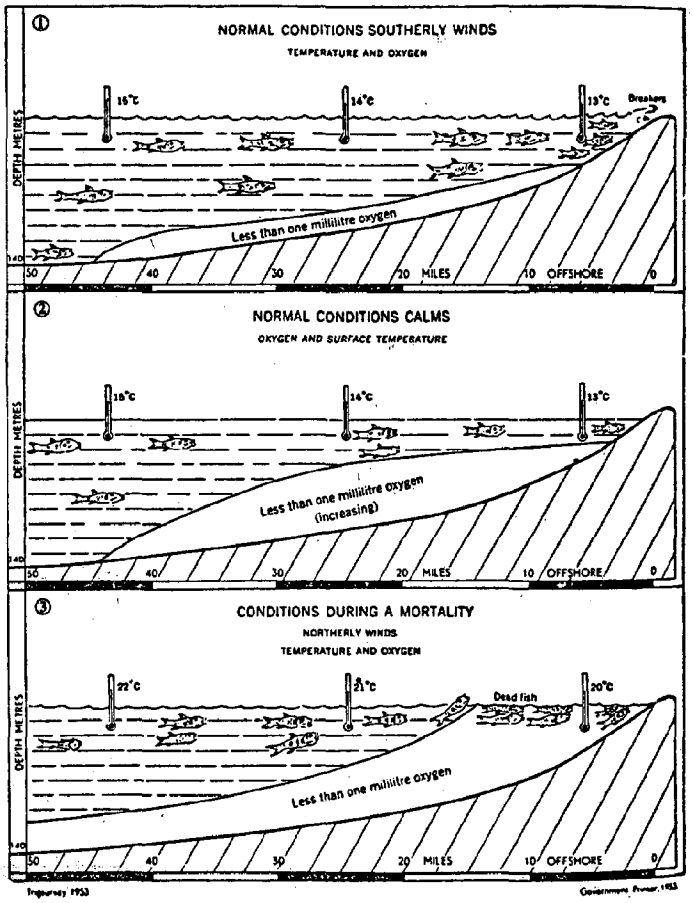


Fig. 26. Temperature and oxygen conditions with mortalities in Walvis Bay (from Copenhagen, 1953).

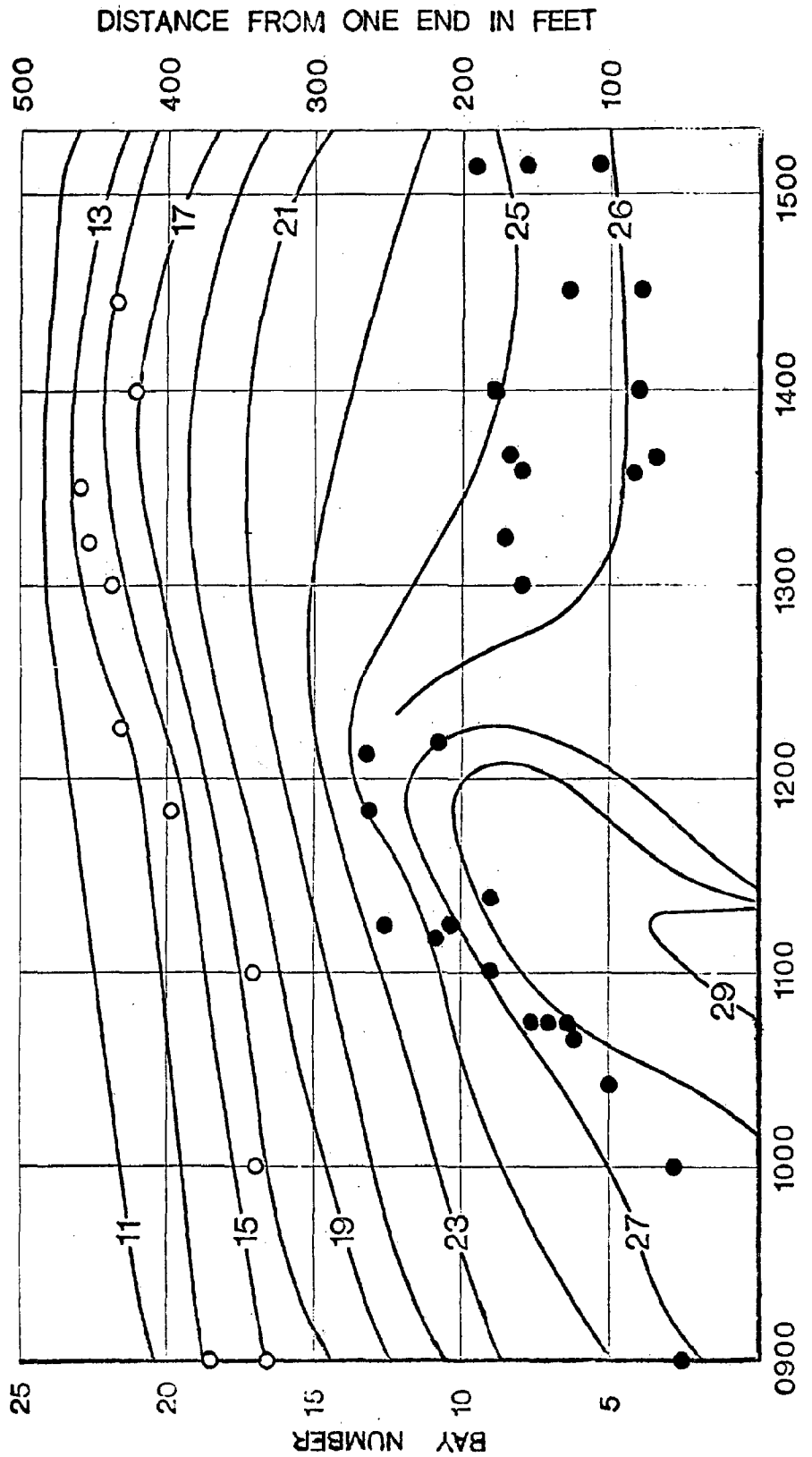


Fig. 27. Position of bream shoals and distribution of temperatures in a winding channel 500 feet long and 4 feet wide. O - Cold adapted fish; ● - Warm adapted fish; isotherms are in degrees centrigrade (From Alabaster, 1969).

gradually become acclimated to and choose higher and higher temperatures. These data suggest that when upstream migrant fish encounter a thermal effluent plume they may gradually become acclimated to it even though they are cold acclimated fish. They may then be attracted to the discharge canal by a combination of warm water preferenda and rheotaxis to the discharge velocity in the canal. Most fish apparently do have the ability to avoid lethal temperatures however. Trembley (1965) found that fish were attracted to the warm waters of a discharge canal during the winter, but avoided the effluent during the summer.

Recent evidence on float marked or electronically tagged migrating fish indicates that some species follow the channel contours in the region of greatest bottom relief (Fig. 28). The work by Nakatani (1969) on salmon and trout and Merriman, et al. (1968) on shad is preliminary and involves a relatively small number of observations. However, studies by Horton (1965) on the small estuarine fish Fundulus heteroclitus indicated that the movement and distribution of these fish was relatively independent of depth and environmental cues other than bottom topography. Fish returned "home" by navigation along near-shore bottom topographical slopes. If this kind of movement-orientation is common to other upstream or intra-estuarine migrating fish, modifications to the estuarine bottom configuration adjacent to the plant site could help divert migrants around dangerous discharge areas. Raney (1969) mentions the necessity of proper engineering of intake screens and discharge canal diversions to aid in allowing fish to escape from areas of potential danger.

After adult fish have successfully migrated past thermal effluent outfalls and spawned in upstream areas, the downstream migrating larvae and juveniles can be drawn into the cooling water intake of the power plants where they are subject to a rapid increase in temperature. Dean and Coutant (1968) exposed juvenile chinook salmon to high temperatures. As each fish lost equilibrium it was returned to the lower acclimation temperature. They found (Fig. 29) that although the initial recovery was 100 percent, a latent mortality of 85 percent was experienced.

Tropical and sub-tropical species of fish commonly enter and inhabit temperate estuaries of the east coast of the U.S. as far north as Cape Cod. It is likely that fish of these species will be attracted to heated water discharges as the water turns cold in the fall. Substantial mortalities could result if the plant is shut down or the effluent plume suddenly diverted by wind or tide. In some instances, warm water discharges have been found to benefit local fisheries. For example, the general temperature increase in the San Gabriel River from the California Edison power plant has permitted the establishment of a "winter haven" for fishes that would normally migrate to warmer waters during the summer months. Cooperative efforts between the power company and the City of Los Angeles have resulted in establishment of a year-round recreational fishery in the vicinity of the outfall (Gaucher, 1968a).

Summary

Although there is very little information on the effects of heated water effluents on various estuarine systems as yet in the published literature, certain known effects of heat and its interaction with other environmental factors indicate that there are significant potential dangers. Oligohaline systems

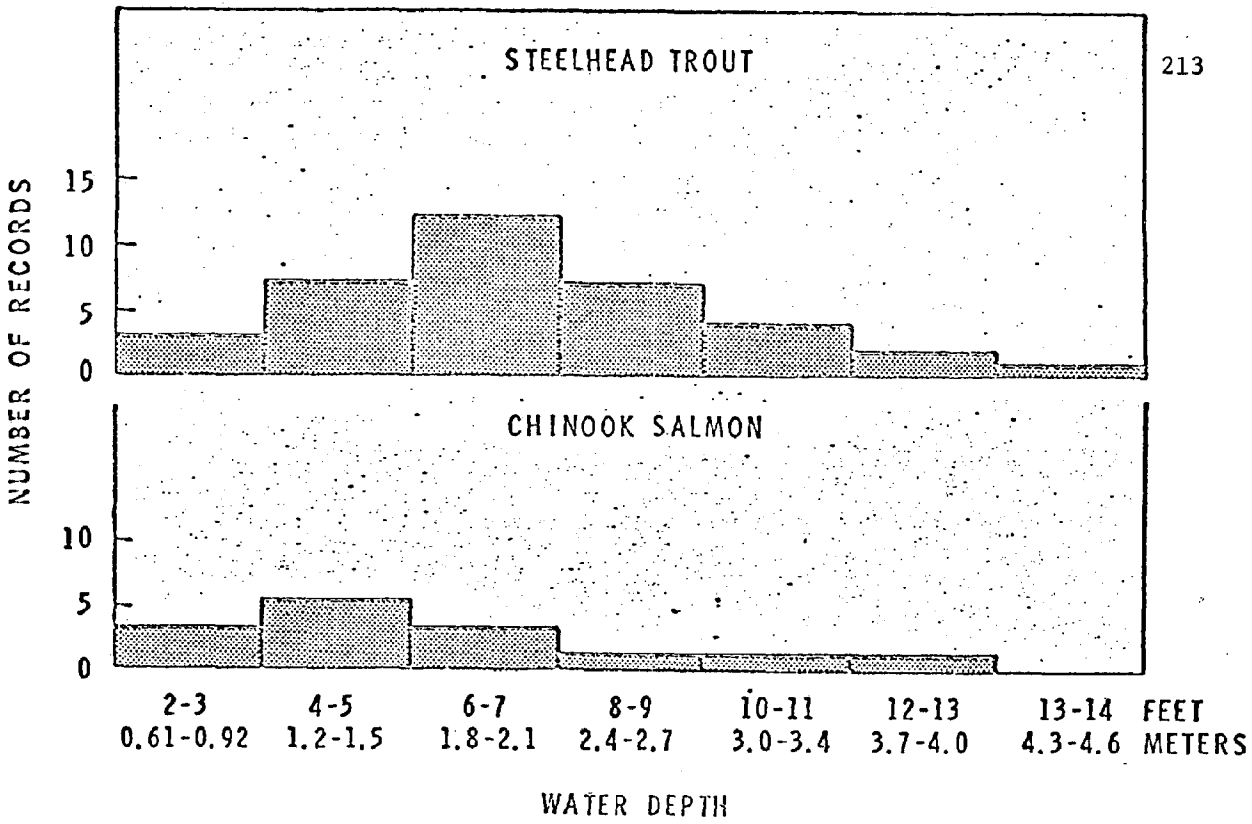


Fig. 28. Frequency of water depths recorded for upstream - migrating chinook salmon and steelhead trout. (Depth data not taken for all fish) (From Nakatani, 1969).

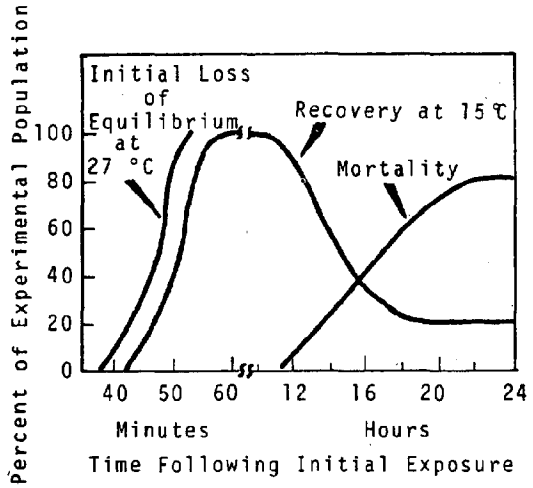


Fig. 29. Time course of relationship between loss of equilibrium and mortality of juvenile chinook salmon (from Dean and Coutant, 1968).

in temperate regions are perhaps most susceptible to the consequences of thermal pollution. These estuaries are already considerably stressed and low in species diversity, but contain high population densities of certain species, some of which have considerable economic importance (e.g. Mya arenaria). Oligohaline estuaries are also the locus of breeding and spawning migrations of many species of valuable fin fishes. Larval and juvenile downstream migrants of these species can easily be drawn into the cooling water condensers of electric generating plants where they may suffer severe mortalities.

The most promising potential solution to waste heat in the aquatic environment would appear to be in commercial aquiculture. The heat could be used in controlled estuarine systems, thus sparing the naturally adapted ones.

Chapter E-7

PULP MILL WASTE SYSTEM

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INTRODUCTION

In the coastal areas of the United States where paper manufacturing and pulp production are conducted, the disposal of the waste products of these operations is a significant problem. As with other industrial wastes, the proper evaluation of the nature of the wastes and the character of the receiving body of water must be undertaken to understand environmental disturbances.

There are two major pulping techniques: the sulfite process and the Kraft (sulphate) method. The sulfite pulping process consists of digesting wood chips under pressure in an aqueous bisulfite solution containing an excess of sulfur dioxide. During this digestion, the cellulose fibers in the wood are freed by dissolving the lignin that binds these fibers together. The cooking liquor remaining after the digestion of the wood chips is termed Spent Sulfite Liquor (SSL) and contains everything that the wood chips contained except the cellulose: lignins solubilized by sulfonation and pentose and hexose sugars hydrolyzed from hemicelluloses. SSL is a clear fluid that looks much like fresh-brewed coffee and has a pH of 2 to 4. The characteristics of SSL vary with the bases used in the cooking process and the botanical species pulped.

In contrast to the sulfite pumping process which utilizes an acid digestion, the sulphate (or Kraft) process employs an alkaline cooking liquor. The pH of the Kraft mill effluent is therefore basic. The Kraft process allows for a greater recovery of the cooking liquor than does the sulfite process and usually produces lower organic waste loadings than SSL. Kraft wastes may, however, be more toxic than SSL as such toxic constituents as hydrogen sulfide, mercaptans, resin acids and soaps are usually present in higher concentrations.

Pump mill wastes, upon entering coastal systems, affect the system both directly and indirectly. One of the more obvious direct effects of SSL and Kraft wastes is the exertion of an immediate oxygen demand on the receiving waters. Both the chemical oxygen demand exerted by SO_2 in the SSL and the BOD exerted by the sugars are immediate in nature and serve to deplete the dissolved oxygen in the water. SSL, containing a higher concentration of organic material, usually exerts a greater BOD than do Kraft wastes. Other direct effects include changes in the pH of the receiving water as a result of the acidity and alkalinity of SSL and Kraft effluent, respectively; turbidity which causes the shading of photosynthetic processes; temperature changes in the water as a result of waste introduction; and the toxicity of the wastes to

the biota. Indirect effects consist of turbidity where the suspended materials may settle out and form sludge banks which not only render the bottom sediments unsuitable for normal benthos but also may exert an immediate oxygen demand on the water; a reduction in surface tension which causes an increase in foaming, thus making the water unattractive aesthetically; and the long-term subtle effects of materials on growth, reproduction and metabolism of the biotic community. In other words, the direct effects of pulp mill effluents are those which cause physical and chemical changes in the water, bottom sediments and the biota on a short term basis, with recovery possible in a relatively short time if the stress were removed. Indirect effects, however, gradually degrade the environment as a result of the build-up in the system of physically harmful and/or slowly oxidizable waste components.

The general effects of pulp mill pollution discussed above may all be manifested in both freshwater and estuarine systems. The differences in the effects between freshwater systems and estuaries are primarily a result of the hydrographical characteristics of estuaries. Tidal flushing, fresh water inflow, circulation patterns and saline stratification are peculiar to estuaries and these factors, in relation to the placement of the outfall, affect the degree to which the wastes exhibit a stressful influence on an estuary. The energy required for combating the stresses imposed by pulp mill wastes leaves less available for normal processing by the system, thus resulting in lower species diversity, productivity and adaptability.

SYSTEM EXAMPLE - SILVER BAY, ALASKA

On August 26, 1965, water samples were collected in Silver Bay, Alaska, to study the effects of the surface discharge of SWL by a pulp mill located on the bay (U.S. FWPCA 1966c). The sampling stations, tidal conditions, SWI distribution and turbidity measurements during the sampling period are shown on Figure 1. Among the other parameters investigated were dissolved oxygen concentration and pH.

Figures 2 and 3 show the vertical distribution of SWL, DO and pH for six representative stations. The maximum SWL concentrations were observed at the surface at each station and decreased rapidly with depth. The horizontal distribution of the surface SWL and secchi-disc readings, as shown in Figure 1, indicated no strong one-way dispersal pattern away from the source. This indicates that the bay is subject to slow flushing action with no strong net outflow of fresher surface water. There seems, however, to be some net dispersal of the wastes against the north shore due to wind effects.

The surface DO concentrations ranged from 40% to 71% saturation then increased with depth. A gradual decrease in DO concentration with depth then occurred below the depth of maximum concentration (Figs. 2 and 3). The depth of the low-oxygen surface layer coincided with the depth of the waste-containing, low-density layer of fresher surface water. The horizontal distribution of DO at a given depth was quite uniform throughout the bay.

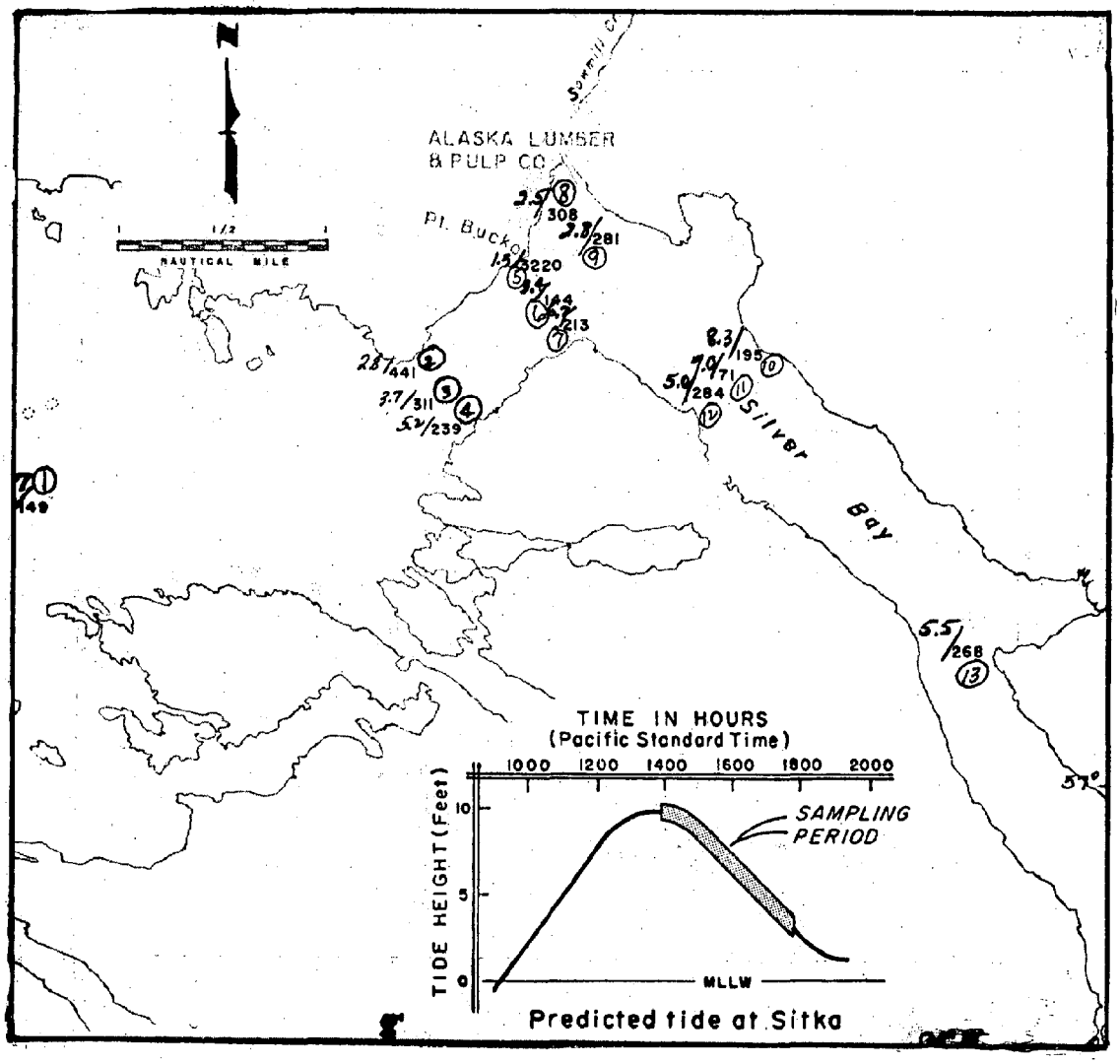


Fig. 1: Map of Silver Bay, Alaska showing station locations (circled numbers), SWL concentration and secchi disk readings in meters on 26 August 1965. No./No. = visibility/SWL in mg/l. Inset shows tide condition and time of sampling. (From U. S. FWPCA 1966c; Fig. 3-6).

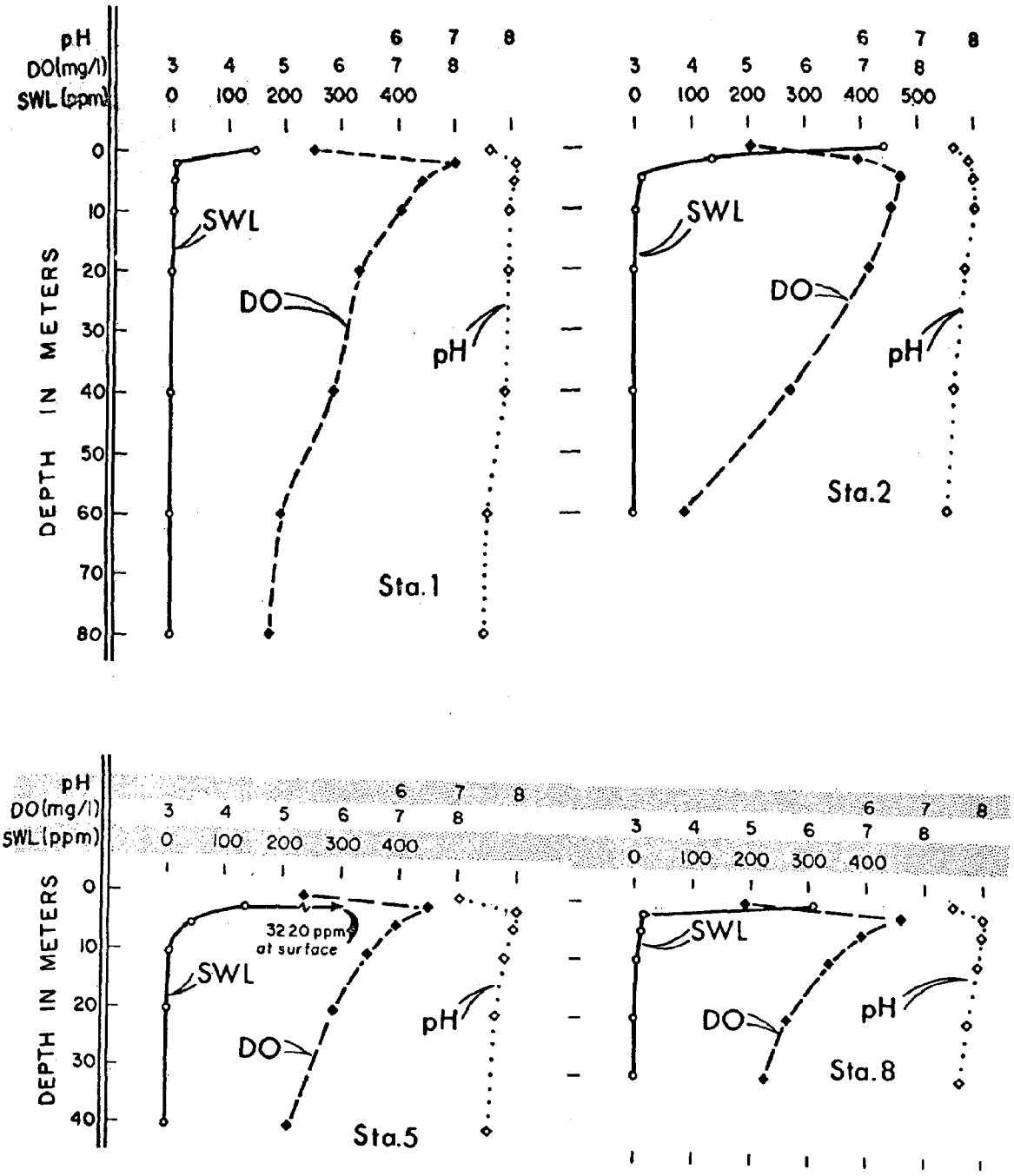


Fig. 2: Vertical distribution of water quality parameters at stations 1, 2, 5 and 8 in Silver Bay, Alaska on 26 August 1965 (From U. S. FWPCA 1966c; Figs. 3-2 and 3-3).

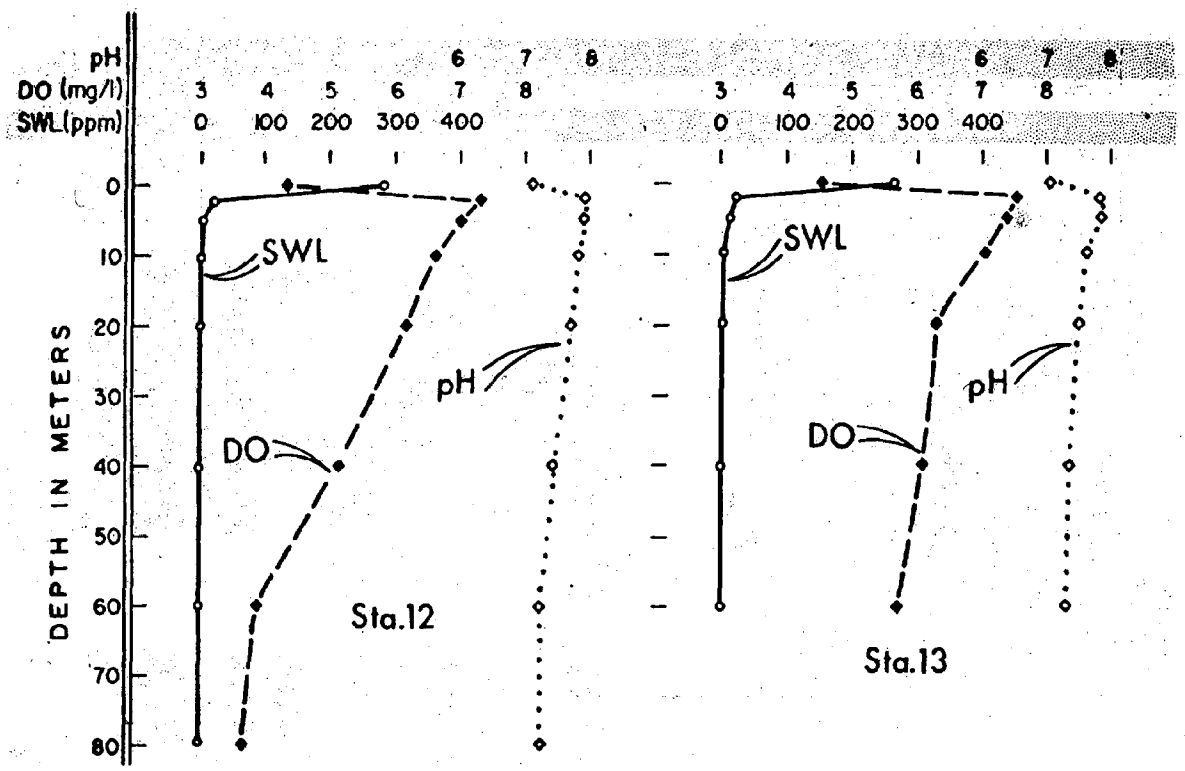


Fig. 3: Vertical distribution of water quality parameters at stations 12 and 13 in Silver Bay, Alaska on 26 August 1965 (From U. S. FWPCA **1966c**; Fig. 3-4).

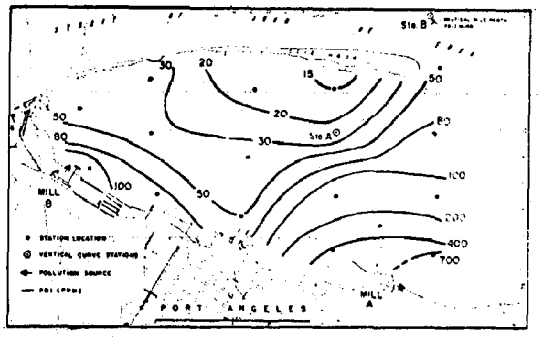


Fig. 4a: Average surface PBI in Port Angeles, Washington (From Bartsch et al. 1967; Fig. 9).

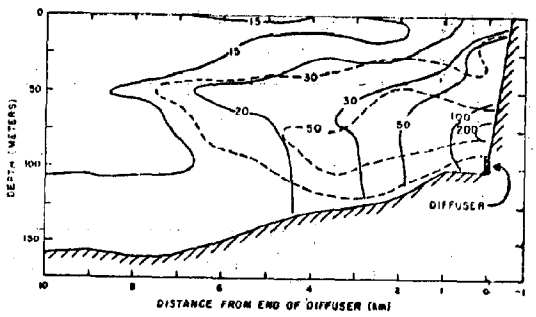


Fig. 5: Vertical section of PBI at Everett, Washington (From Bartsch et al. 1967; Fig. 6).

The vertical distribution of pH at each station was similar to that for DO. Low surface values were observed, followed by a rapid increase with depth to a maximum value at a depth between 2 and 10 meters. A gradual decrease with depth below the depth of maximum pH then occurred.

The uniformity of the SWL concentration due to low tidal flushing indicated that the observed SWL concentrations were of sufficient levels to kill food chain organisms such as copepods, euphausids, mysids and candle fish based on bioassays reported in a pre-pollution study of Silver Bay (Eldridge and Sylvester 1957). Since even higher surface SWL concentrations might be expected to occur during periods of strong southerly or westerly winds, SWL concentrations high enough to cause the death of herring and fingerling salmon might be reached. Laboratory bioassay tests have also indicated that SWL concentrations well below those found during the August 26, 1965, survey of Silver Bay may cause developmental abnormalities and mortalities of oyster larvae and bottom-fish eggs.

The low-oxygen condition of the surface waters of Silver Bay was attributed to both the biochemical oxygen demand of the SWL and to a possible reduction in phytoplankton oxygen production as a result of the inhibiting effects of the pulp waste. The observed surface DO values are borderline to the generally recommended minimum value of 5 mg/l necessary for marine life. Since more critical conditions with regard to DO concentration may be expected to occur, sub-minimum DO values could possibly develop during periods of low water movement.

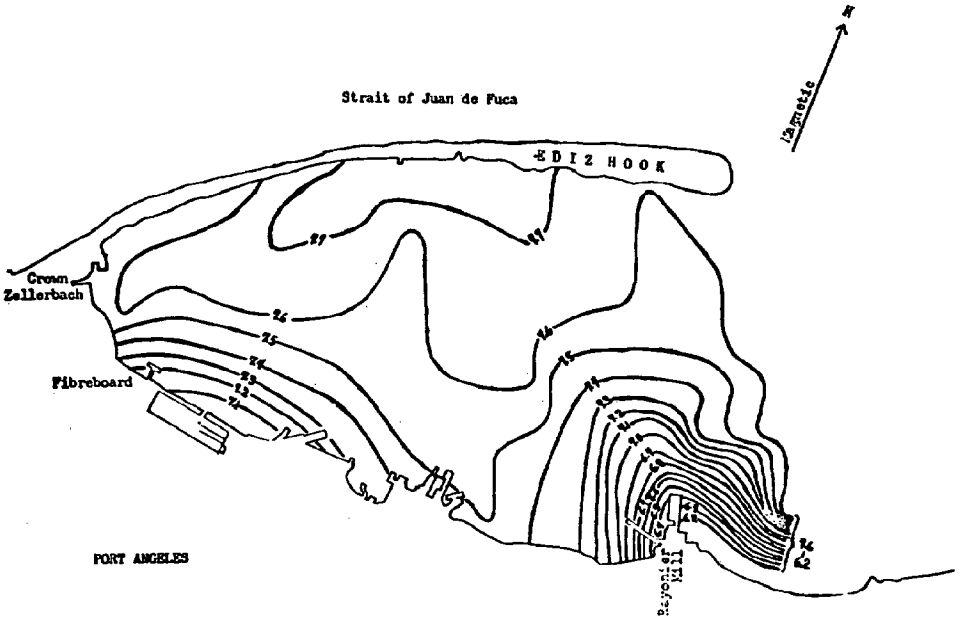
The pH measured in Silver Bay during pre-pollution studies generally increased from a low value at depth to a maximum near the surface. Such a vertical distribution is normal in coastal waters during the summer. The pH measured during the August 26, 1965, study, however, increased gradually from a low value at the bottom to a maximum value at 2-10 meters depth, then rapidly decreased to low surface values. The low pH of the surface waste-confining waters was attributed to the combined effect of waste decomposition, the acid nature of SWL and reduced photosynthetic activity in the surface water.

Effects of System Types on Waste Distribution

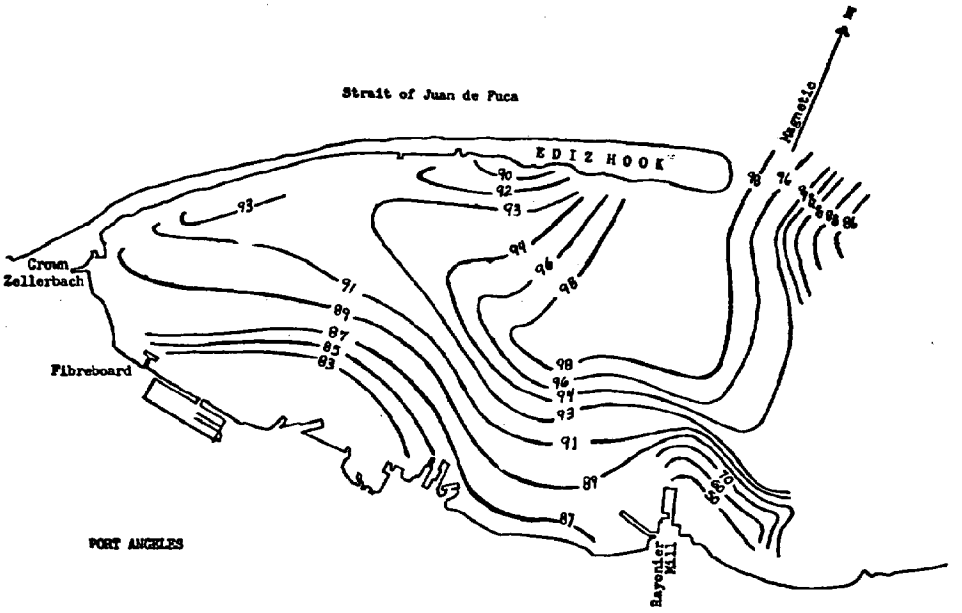
The physical characteristics inherent in various coastal types affect to a large extent the transport and distribution of pulp mill wastes entering the system. The effect will be different, for example, if the wastes enter an estuary via the tributaries or rather through some stratigically located wastes outfall. From Waldichuk (1960), Bartsch et al. (1967), Saville (1966), and others, the effects of different system types on the distribution of pulp mill wastes entering coastal waters are summarized.

High Velocity Channel

In high velocity channels, considerable water movement and flushing are provided by rapid currents resulting from tidal action. Port Angeles, Washington, is located on such a channel, the Strait of Juan de Fuca, where two sulfite pulp mills introduce wastes directly into the harbor (Bartsch et al. 1967). The distribution of the Pearl Benson Index is shown in Figure 4. A comparison of the data for the harbor station and a station in the Strait

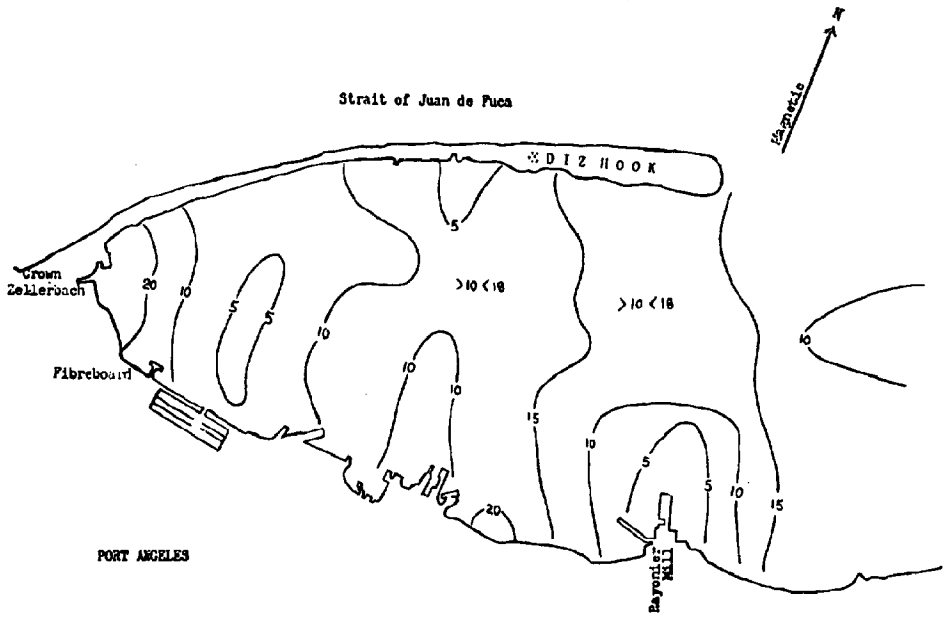


Isogram Showing Variation of Surface Hydrogen Ion (pH) Concentration Within Harbor

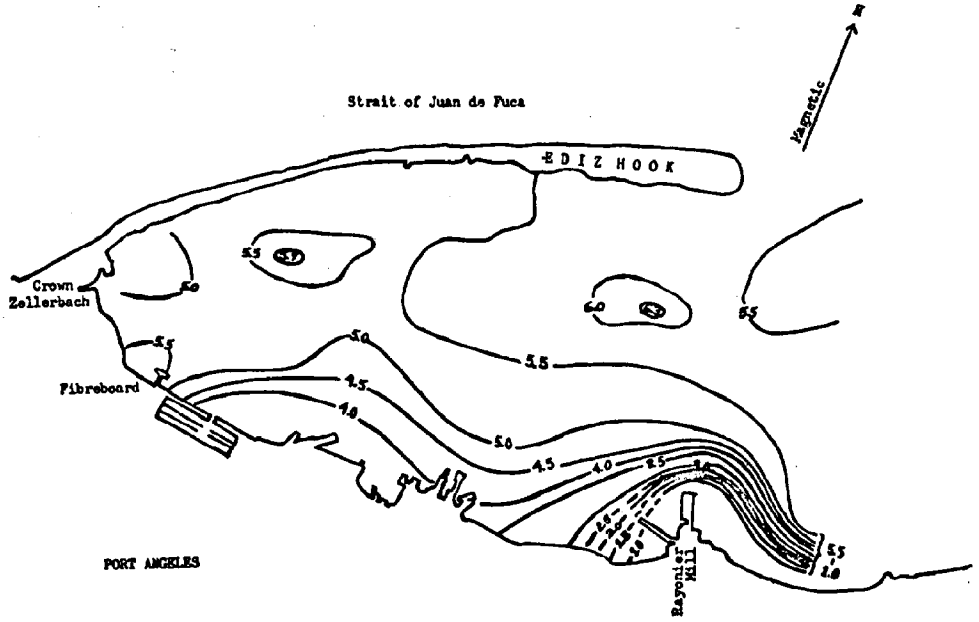


Isogram Showing Percent Transmission of Blue Light Through Surface Water Samples

Fig. 4b. Characteristics of paper mill waste system in Washington (Stein, Denison, and Isaac, 1963).



Isogram Showing Distribution of Spent Sulfite Liquor 20 Feet Below Surface



Isogram Showing Surface Distribution of Dissolved Oxygen Within Harbor

Fig. 4c. Characteristics of paper mill waste system in Washington (Stein, Denison, and Isaac, 1963).

indicates that the water outside the harbor is little affected by the pollution in the harbor (Figs. 4 and 6).

A Kraft mill was built on Osborn Bay, B.C., in 1958. Osborn Bay is a restricted embayment with slow flushing and surface circulation by wind action. Little vertical mixing occurs, especially in the summer when the surface salinity is low and the temperature high. Oyster beds are located just outside the bay. To obtain the maximum dilution of the wastes and to minimize their contact with the oyster beds, the outfall was located in the deep water of Stuart Channel outside the bay in a place where the high flushing rate would transport the wastes away from the beds. A dilution of the wastes at the oyster beds closest to the outfall was found to be 1:280 - 1:550. Although a dilution of 1:2000 is desired, no adverse effects of the Kraft mill effluent on oyster production has been observed (Waldichuk, 1960a).

Sheltered deep estuary

SSL is discharged into Neroutsos Inlet in British Columbia, a deep inlet with low fresh water inflow, a small amount of runoff and poor flushing characteristics (Waldichuk, 1960a). In the summer, the surface water is of low density and high temperature with little vertical mixing. The wastes are discharged into these surface waters with the result that the wastes are confined to the surface layer for long periods of time. Adjacent to the outfall, the concentration of SSL is 600 ppm. At a distance of 10 nautical miles from the mill, the concentration of SSL is still 30 ppm. At depths below 20 feet, the SSL concentrations are below 20 ppm even in water near the outfall. Thus, it is apparent that a large amount of horizontal movement of the SSL occurs and little vertical diffusion. As a result, the surface layer has very low DO values and little or no life forms whereas the bottom waters are still used by fish to migrate to the tributary streams.

An example of a deep outfall is found at Everett, Washington, (Figs. 5 and 9) as reported by Bartsch et al. (1967). Vertical distribution of the PBI indicates diffusion of waste through a mid-section of the system (Fig. 6). The combined waste effluent from two sulfite mills is emptied into the estuary through a deep diffuser. The depth of the upper end of the diffuser is 91m and that of the lower 104m. In this estuary there is a thin freshwater layer on top and little vertical mixing. The net bottom current is north from Possession Sound and decreases with distance from the outfall. At the diffuser, the PBI increases at the lower depths (Fig. 7). The pH and DO decrease at the lower depths. At a station 7.2 miles from the diffuser, the effect of the SSL is still evident though lower in magnitude and depth. Bioassays have shown that at times the water of Port Gardner Bay are lethal to juvenile chum salmon due to low pH and DO (Fig. 9). The concentration of SSL was not at lethal levels.

Medium salinity plankton system

Bellingham Bay in Washington represents this system and the pulp mill effluent enters at the head of the bay (Bartsch et al. 1967). The surface SSL concentration decreases with distance from the waste source (Fig. 10A).

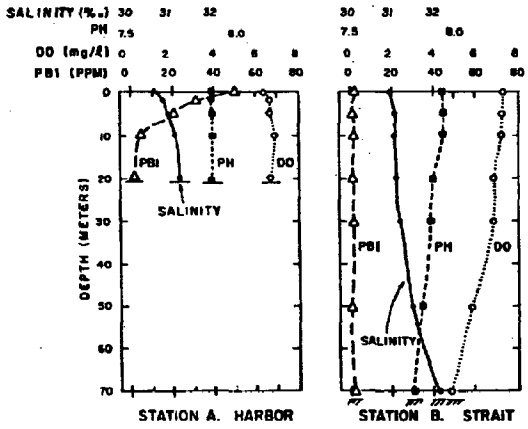


Fig. 6. PBI at a harbor station and a strait station. Port Angeles, Washington (From Bartsch et al. 1967; Fig. 12).

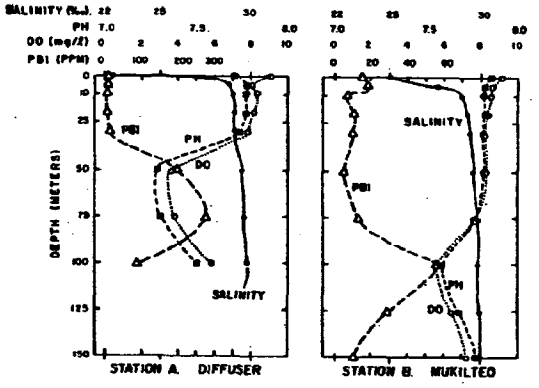


Fig. 7. PBI at the diffuser and Mukilteo. Everett, Washington (From Bartsch et al. 1967; Fig. 7).

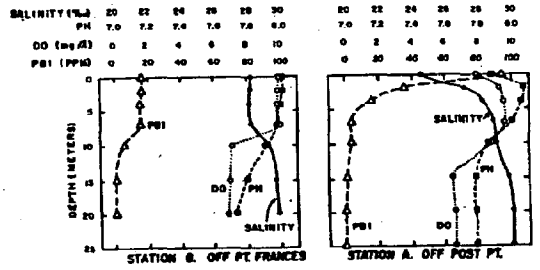


Fig. 8. PBI off Pt. Frances and Post Point. Bellingham, Washington (From Bartsch et al. 1967; Fig. 3).

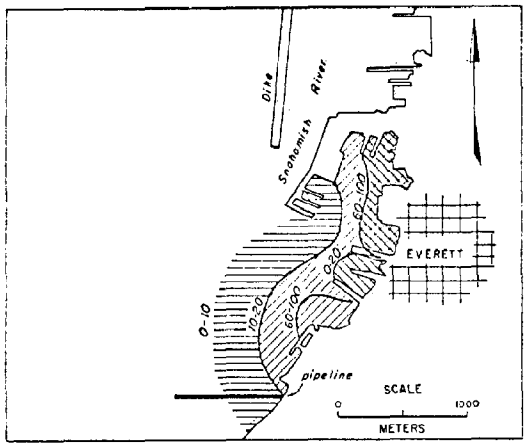


Fig. 9. Zones of toxicity to juvenile salmon as percentage of mortality in 24 hours at Everett, Washington. Composite of six tests (From Bartsch et al. 1967; Fig. 8).

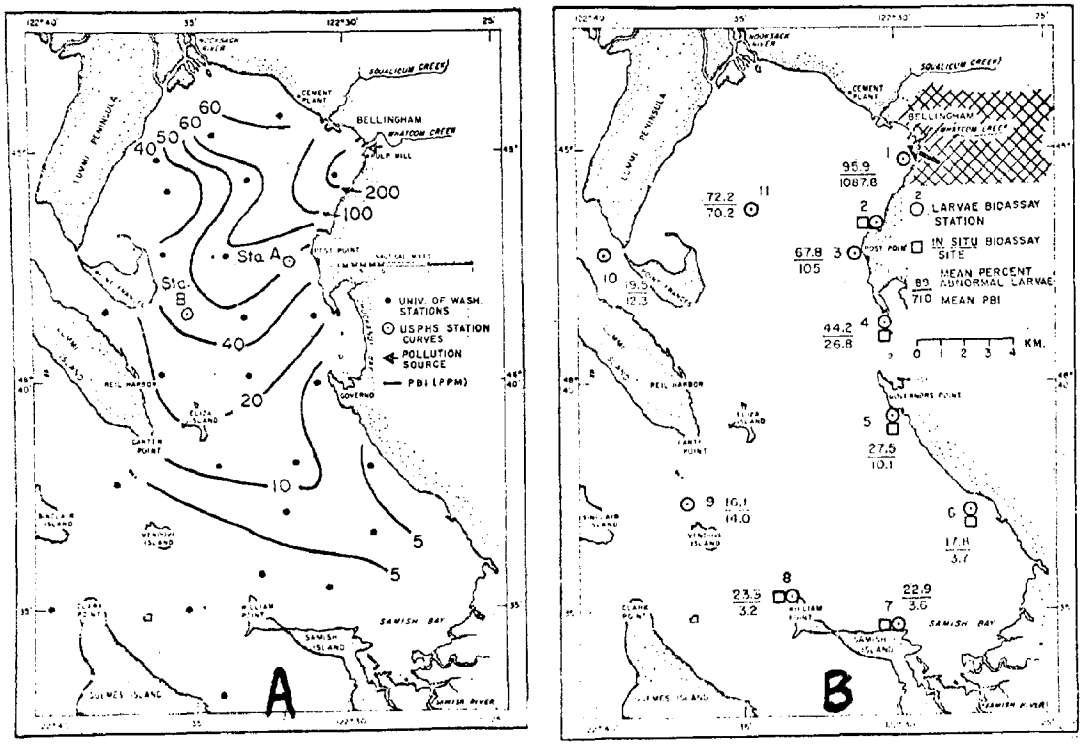


Fig. 10. Bellingham, Washington. A. Average surface Pearl-Benson Index distribution. B. Distribution of abnormal oyster larvae in relation to PBI distribution. (From Bartsch et al. 1967; Fig. 4).

The waters of the system are very turbid and light penetration is therefore low, with secchi disk readings generally less than two meters. The effect of the mill effluent in lowering DO levels is localized to within a 1 - 2 mile radius of the outfall, and oxygen saturation values of 120 - 140 percent often occur in the upper 10m during plankton blooms. As shown on Figure 8, the concentration of SSL (PBI) is greater at Station A nearest the outfall than at Station B approximately three miles further distant. The DO and pH are decreased in the upper 2m at Station A but are unaffected at Station B. The decrease in surface salinity at Station A is due to its relative position to the Nooksack River. The relationship between SWL concentration and percent abnormal oyster larval is shown in Figure 10B. As the PBI decreased the % abnormal larvae decreased (from 96 to 16%).

Effluent from a Kraft process mill enters St. Joseph Bay, Florida, a medium-salinity plankton based system (Copeland 1966b). The gentle mixing of the bay system results in the waste becoming thoroughly mixed with the bay water in a short distance from the waste outfall. As illustrated in Figure 11, the effects of the pulp mill wastes are detectable as far away as one mile. It is notable, however, that the lowering of redox potential is greatest near the surface at the outfall with mixing occurring as distance increased in the mixing bay water.

Oligohaline system

Kraft mill wastes are introduced into Albermi Inlet, British Columbia (Fig. 12). This inlet has a large freshwater inflow and good tidal flushing action. The outfall is located in mid-harbor in the jet-stream of the river. The effect of the pollution therefore varies with the river flow and harbor stagnation is minimized. During the period of study, the DO of the harbor never fell below that of the incoming river water. The BOD of the harbor water ranged from 1 to 5 ppm while that of the Kraft mill effluent ranged from 30 to 150 ppm. There was no detectable loss of fish due to the Kraft mill effluent. In this case, therefore, an advantageous placement of the waste outfall minimized the immediate harmful effects of the wastes. A long term problem may arise in the harbor, however, from the settled particulate matter (Waldichuk, 1960a).

The Fenholloway River and estuary in Florida is an example of a system in which Kraft mill wastes, while not the sole disturbing factor, are the major polluting influence (Saville, 1966). The Kraft mill effluent is not introduced directly into the estuary but, instead, into the freshwater river approximately 23 miles upstream from the mouth on the Gulf of Mexico. During periods of low flow (300 cfs) stratification is at a minimum. The lower portion of the estuary becomes highly stratified in periods of medium flow (400 - 600 cfs), while above Bird Island, approximately two miles above the river mouth, the stream is essentially fresh water. When the flow exceeds 1000 cfs, the salinity intrusion is slight and confined to the mouth. The flushing action of the fresh water input pushes the mixture of waste and river water downstream to the estuary. The effects of the Kraft mill effluent on the condition of the river and estuary are shown on Figure 13.

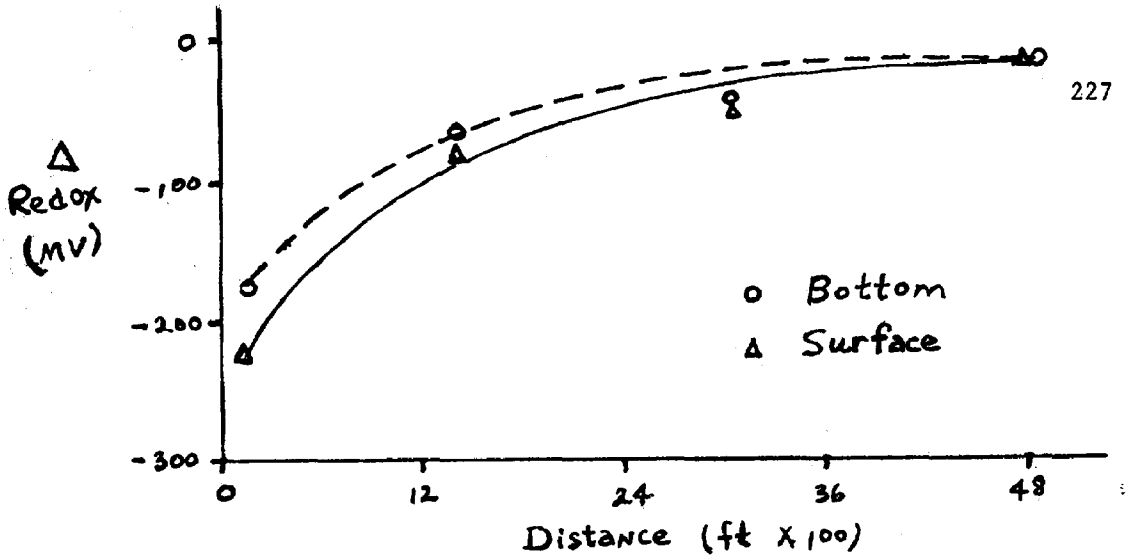


Fig. 11. Depression of redox potential from kraft process waste in St. Joseph Bay, Florida with distance from the source (From Copeland 1966b).

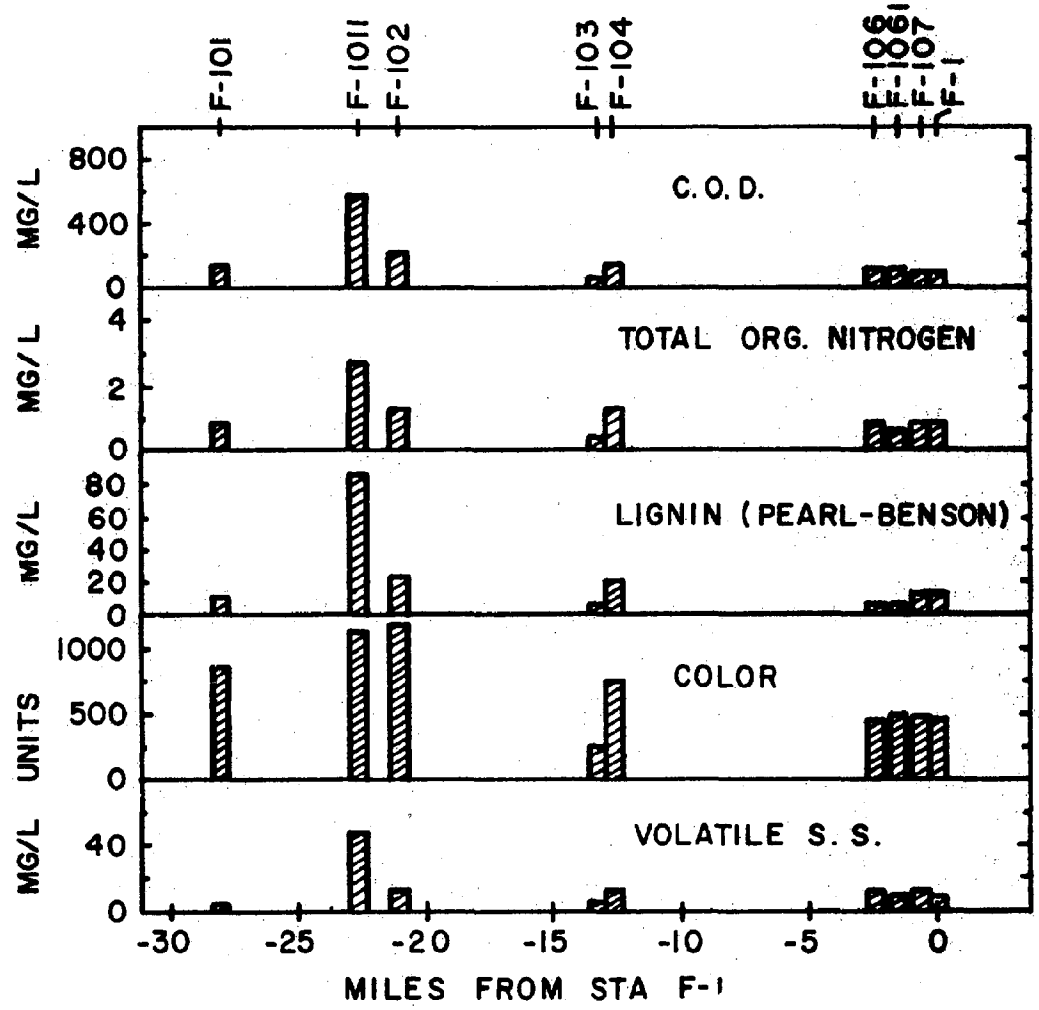


Fig. 13. Longitudinal Profiles of COD, Total Organic Nitrogen, Lignins, Color, and Volatile Suspended Solids, March 22, 1965. (From Saville, 1966)

Influence of the Kraft Mill effluent on the Fenholloway River and estuary, Florida.

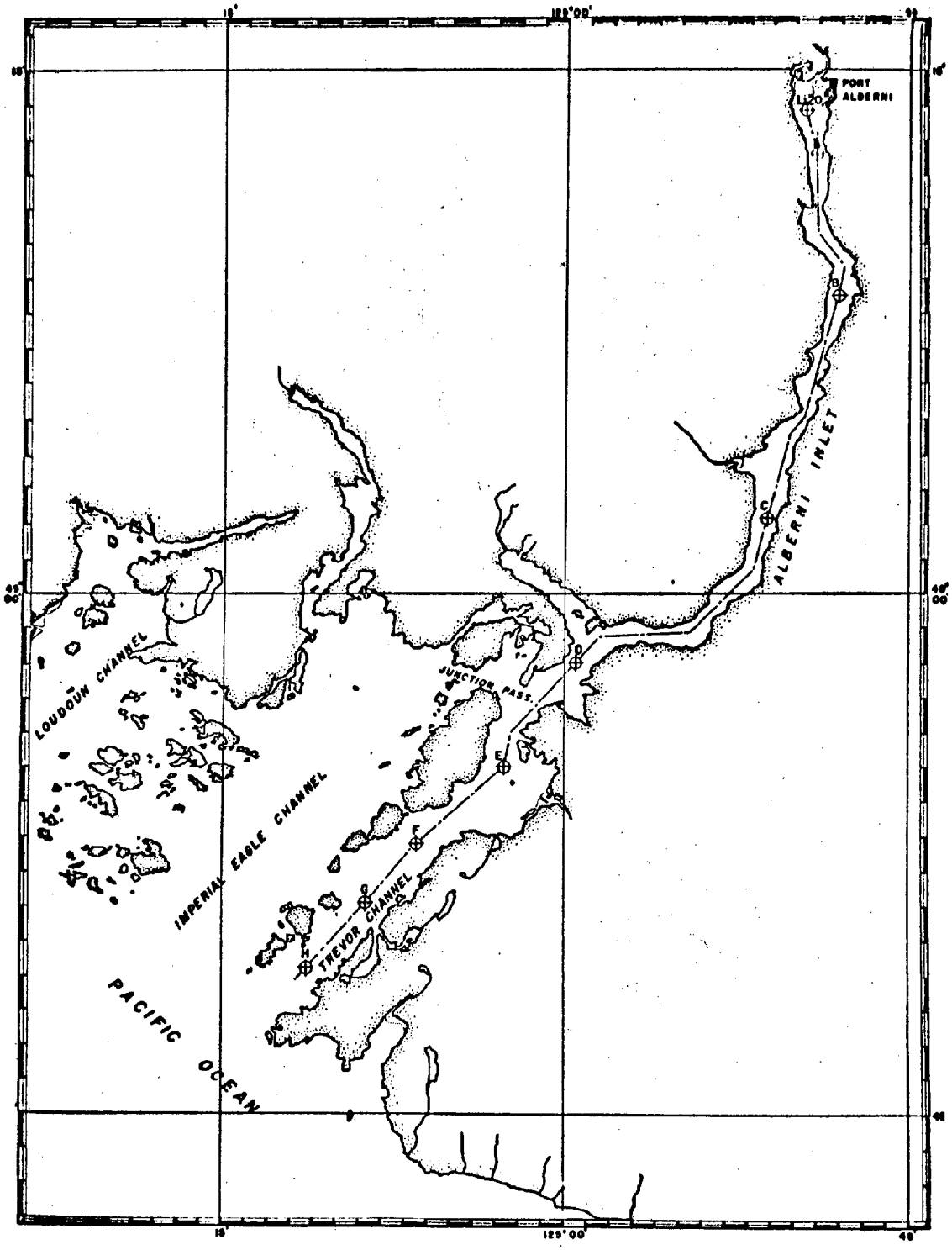


Fig. 12a. Chart of Alberni Inlet in Barkley Sound, B.C., Canada (From Waldichuk 1960a, Fig. 6.

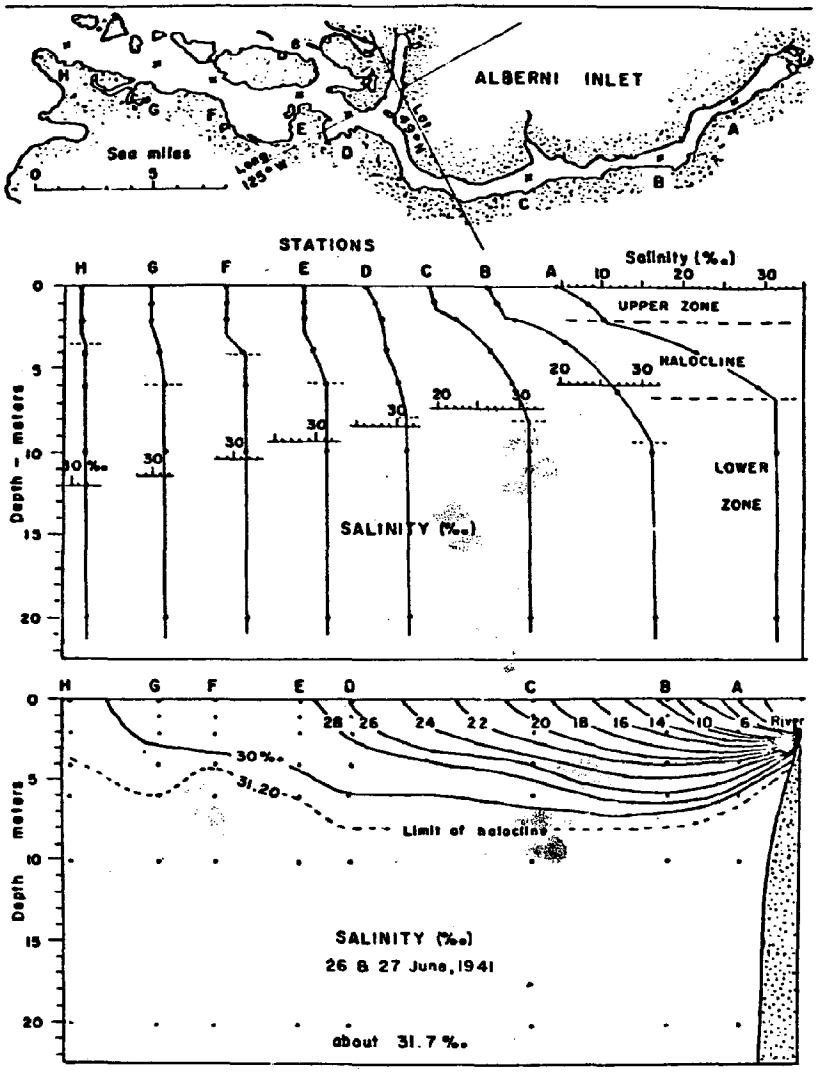


Fig. 1.

Salinity structure in Alberni Inlet, B.C.

Fig. 12b. Stratification in a tidal estuary receiving wastes (Tully, 1960).

EFFECTS ON SYSTEM COMPONENTS

Although the impact of SSL and Kraft wastes is on the estuarine community as a whole, it is the response of the individual system components which makes up the total reaction of the system to the disturbance. Data are not available concerning the effect of pulp mill wastes on every component in the disturbed environment so that an analysis of the effect of this stress on all the individual system components and the interactions between them is not possible. There is information, however, available on the effects of pulp wastes relating to toxicity, reproduction, metabolic response, BOD exertion, productivity and species diversity that enable a summary evaluation of the system effects to be made.

Toxicity

The economic value of oysters found in estuaries has led to studies concerned with the effects of pulp mill wastes on these organisms. Toxicity studies have been conducted, but, for the most part, the investigations have attempted to determine the sub-lethal effects of the wastes. This emphasis has been a result of the fact that, in general, the toxicity of SSL is less of a problem than its BOD exertion and that the effects of the high BOD will be felt before those of a toxic nature.

The toxicity of calcium-base cold blown SSL to adult oysters was reviewed by Gunter and McKee (1962). They reported that after 132 days exposure, oysters exposed to 100 mg/l SSL exhibited 34.4% mortality vs 18.7% for the controls. No effect of the SSL was noted up to 41 days exposure. In other experiments at an exposure time of 575 days, the two controls had a mortality of 52% and 53% and oysters exposed to 13.0, 26.1, and 128.9 mg/l SSL had mortalities of 73%, 84% and 98% respectively. Thus, SSL is definitely toxic to oysters exposed for long periods of time. As previously stated, however, the other effects of SSL at these concentrations may become apparent before the toxic effects.

The most critical life-cycle stages of organisms are the egg and larval forms. Since the organism is more sensitive to toxic compounds at these stages, whether or not the adult can survive is immaterial. Woelke (1960a), summarizing about ten years' efforts and data, reported the mortality rates of Crassostrea virginica (oysters) and Venus mercenaria (quahogs) eggs and larvae in various dilutions of SSL. In contrast to the data reported by Gunter and McKee (1962) for the adult oyster, the eggs and larvae exhibited a very high mortality in relatively low concentrations of SSL (Fig. 14). The eggs of both species of molluscs were very sensitive to the SSL, with only about 10% of C. virginica eggs surviving 10 mg/l SSL. The significance of these data is the fact that unless eggs are produced elsewhere and then, after hatching, transported into areas affected by SSL, the oyster cannot reproduce in systems containing only small concentrations of pulp mill wastes.

Holland et al. (1960) studied the effects of synthetic Kraft mill effluent on salmon. Their results indicated that the older chinook salmon were more susceptible to the waste than were the younger ones. Of the two

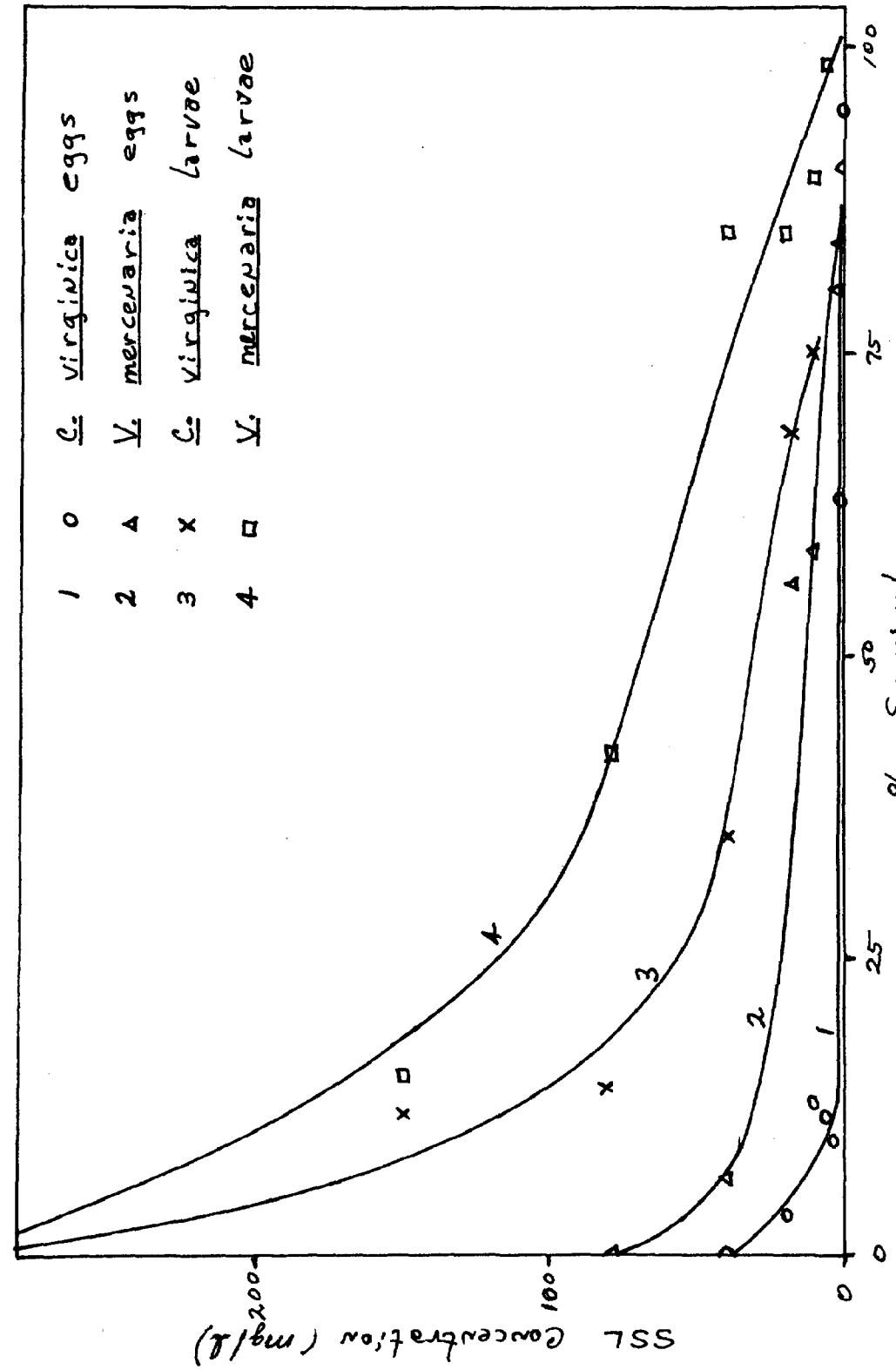


Fig. 14. Percent survival of the eggs and larvae of Crassostrea virginica and Venus mercenaria in various concentrations of SSL. (Summary of data presented by Woelke 1960a).

major components of Kraft mill wastes, the condensate was more toxic than the black liquor in the amounts in which they normally occur in coastal systems. The results of the study indicated that the toxicity threshold index of 20 day-old chum salmon exposed to calcium oxide-base SSL for 30 days in aerated flowing brackish water was 880 mg/l, with little difference in the toxicity of ammonium-base and calcium oxide-base SSL.

Alderdice and Brett (1957) studied the effects of Kraft mill wastes on young Pacific salmon. Their results are summarized in Figure 15, which show that 50% mortality can be affected in a very short time at only 10% concentration.

At low concentrations of Kraft mill wastes, fish can often survive but the condition of the survivors may be deteriorated (Fujiya 1961). Examination of certain tissues of the surviving fish taken from a pulp mill affected estuary in Japan, revealed decreases in glycogen in the liver and RNA in the hepatic and pancreatic cells, and changes in kidney tissue. Other physiological changes were also observed.

Little research has been conducted on the toxicity of SSL and Kraft wastes on plankton organisms even though they are a major food source of such economically valuable animals as oysters. Gunter and McKee (1960), however, discuss some preliminary investigations in this area that have been undertaken. They report that Woelke (1957) exposed the marine flagellate Monas sp., an alga used successfully to culture bivalve larvae, to hot-blown ammonium-base SSL. The SSL was adjusted to 10% solids. An SSL concentration of 10,000 mg/l exhibited a toxic effect after three days exposure. No significant toxic effects were observed for exposures of 13 days to SSL concentrations up to 1,000 mg/l. On the contrary, a fertilizing or stimulating effect was indicated.

Using the same type of SSL, Woelke (1958) studied the effect on three other algal species: Actinocyclus, a green diatom; Chryptomonas, a red flagellate; and Isochrysis, a green flagellate. No significant effects of SSL concentrations up to 304 mg/l were observed on these algae.

Lasater (1953) exposed two species of copepods, Epilabidocera amphitrites and Calanus finmarchicus to SSL. Significant mortalities of E. amphitrites occurred at 157 mg/l in 48 hrs and with C. finmarchicus in 50 mg/l after two weeks. Lasater (1956) did not find, however, that the high concentrations of SSL in the waters of Everett Harbor affected the zooplankton populations.

Other workers (Academy of Natural Sciences of Philadelphia 1957) exposed a mixed culture of Puget Sound diatoms, predominantly Skeletonema costatum and Asterionella japonica, to a mixed calcium- and ammonium-base SSL of 12.6% solids. The median toxic concentration after one week exposure was found to be 3850 mg/l and the "biologically safe concentrations" were calculated to be 1160 mg/l for 12.6% SSL and 1460 mg/l of 10% SSL.

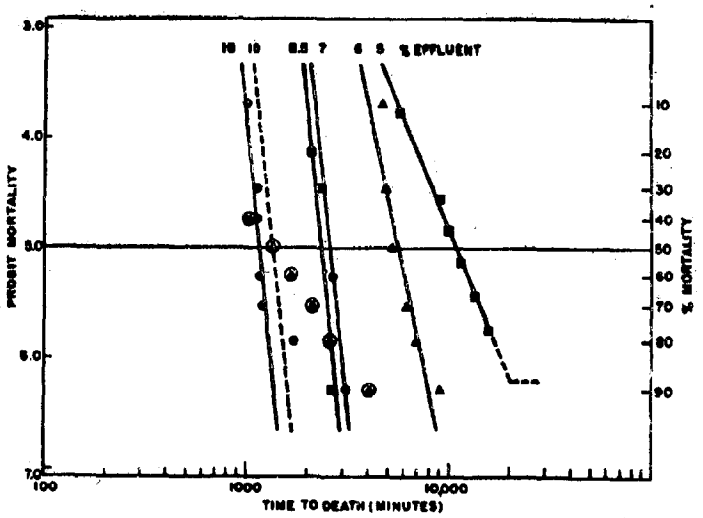


Fig. 15. Time-mortality curves for sockeye salmon underyearlings exposed to various concentrations of kraft mill effluent (Alderdice and Brett 1957)

Table 1. Summary of results of bioassays of sulfite waste liquor with Olympia and Pacific oysters (From Woelke 1962).

Type of bioassay	Type of response measured	P.B.I.* at which adverse effects occurred
Floating live boxes (Olympias)	Mortality and condition index	Average of 7 ppm.
Long range laboratory bioassay (Olympias)	Mortality	About 16 ppm.
Olympia oysters in laboratory	Reproduction	Between 8 and 16 ppm.
Development of Pacific oyster eggs fertilized in sulfite waste liquor	Percent abnormal larvae	3 ppm.
Lagoon bioassays with Olympia and Pacific oysters	Growth and mortality	13 ppm.
Fertilized Pacific oyster egg development in field water	Percent abnormal larvae	7 ppm.

* Pearl-Benson Index

Reproduction

Woelke (1962) studied the effects of SSL on Olympia and Pacific oysters to determine if the SSL concentrations encountered by the oysters in their natural habitat were detrimental to their well being. He utilized *in situ* bioassays, laboratory bioassays on adults and larvae, lagoon studies, and bioassays with fertilized Pacific oyster eggs using field water. The results of these experiments are reported in Table 1. The author concluded that although it was not established that the SSL was the only harmful influence, it was found that the existing ambient SSL concentrations could create a stress on the system. It also appeared that the reproductive stage of the life cycle was the most sensitive life stage.

In another study, Gunter and McKee (1962) reported that "long term" exposures of Olympian oysters to cold and hot blown SSL resulted in fewer larvae being spawned in 17, 55 and 170 mg/l cold blown SSL and 55 and 170 mg/l hot blown SSL than in controls.

While certain developmental stages of organisms respond in various ways to toxicity or other waste parameters, it is important to understand the ability of adults to reproduce in polluted systems. Even though adult oysters are known to "survive" in relatively high concentrations of SSL (Gunter and McKee 1962), the ability to produce eggs and develop them to larvae is severely affected by relatively low concentrations of SSL (Woelke 1960c). The results of an experiment concerning the abortion of eggs by *Ostrea lurida* in various concentrations of SSL are shown in Figure 16 (Woelke 1960c). Above about 20 mg/l of SSL, a significant percentage of the *O. lurida* aborted eggs. The significance of these experiments lies in the inability to reproduce in waters containing relatively low concentrations of SSL, especially for sessile organisms like the oyster which is unable to vacate its environment.

Metabolic Response

Odlaug (1949) studied the effect of SSL in "unpolluted seawater" on the pumping rate of the Olympia oyster, *Ostrea lurida*. He observed a 6% reduction in the number of drops pumped at 400 mg/l SSL in 54 minutes and an 83% reduction after 77 minutes in 5000 mg/l SSL. The feeding activities of the oysters would therefore be affected at these SSL concentrations.

Galtsoff *et al.* (1947) studied some physiological effects of sulphate wastes on oysters growing in the York River, Virginia. Sulphate wastes were introduced into the upper river. Studies of the meat quality, as expressed as glycogen content, showed that the upper York environment was decidedly harmful to the oysters (Fig. 17). The average glycogen content of the oysters in the upper York River was lower than that found in the unpolluted Piankatank River as shown in Figure 18. The glycogen content of oysters transplanted from the James River to the Piankatank and lower York Rivers increased much more rapidly than in oysters transplanted to the upper York. The authors state that apparently sulphate wastes cause an inhibition of carbohydrate metabolism.

Laboratory studies by Galtsoff and his co-workers found that the pulp mill effluent reduced the number of hours that the oysters remained open and

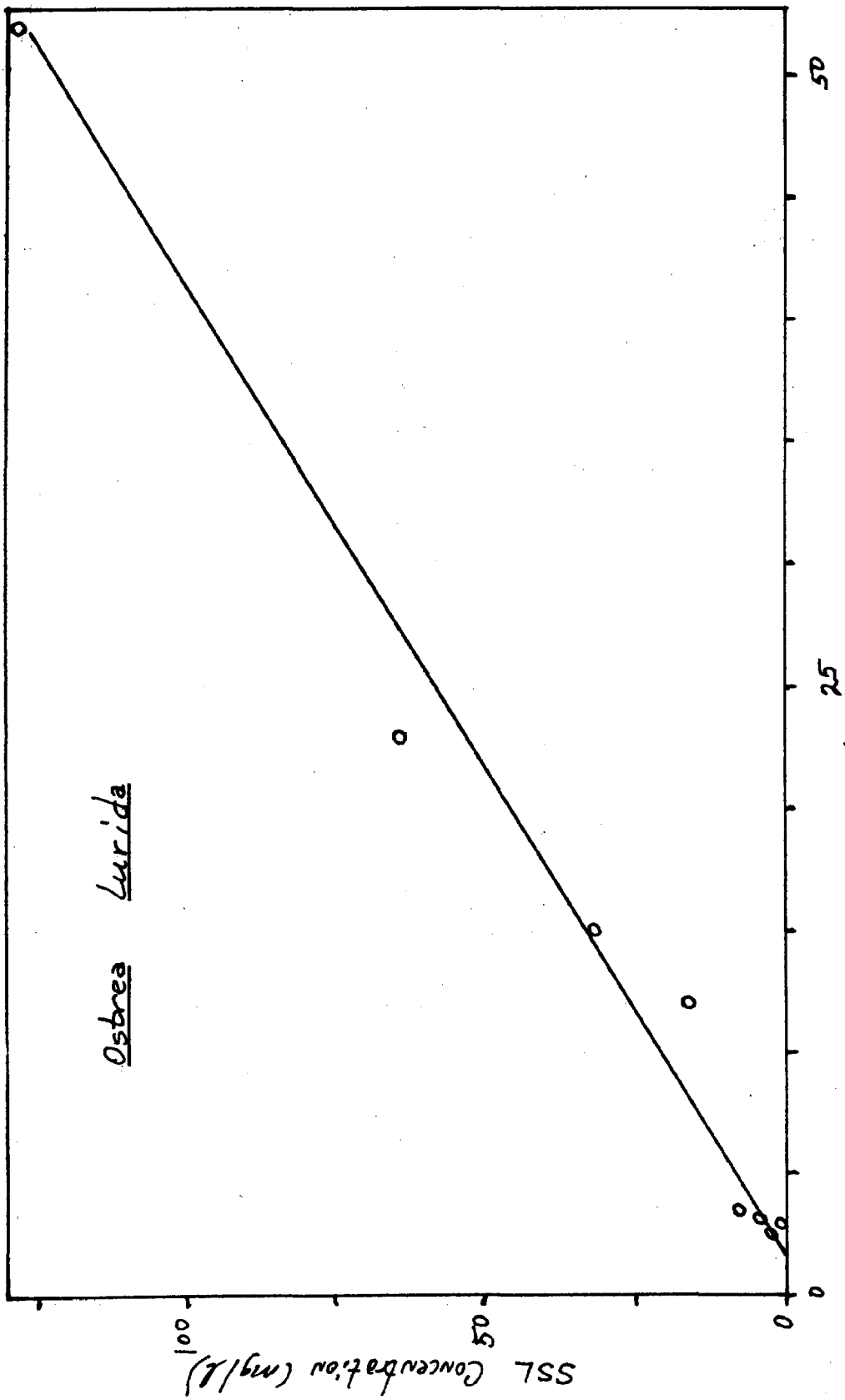


Fig.16. Percent abortion of eggs by Ostrea lurida in various dilutions of SSL concentrations (Summarized from Woelke 1960c).

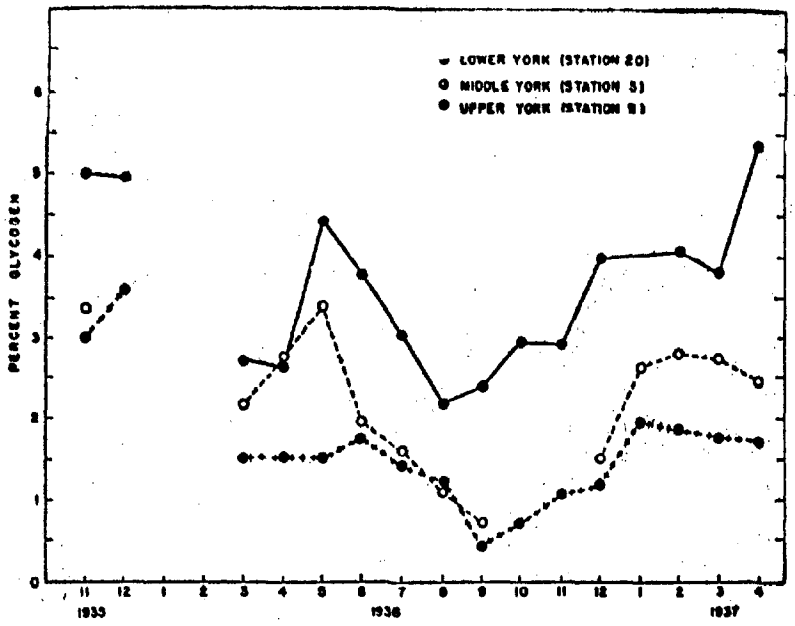


Fig.17 Glycogen content of York River oysters from the lower, middle and upper parts of the river (From Galtsoff et al. 1947; Fig. 33).

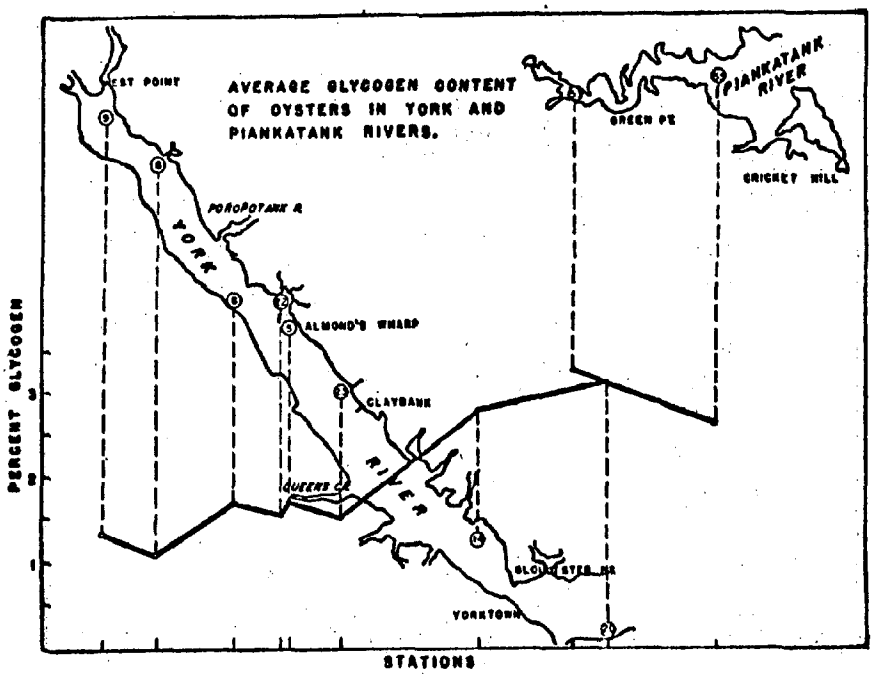


Fig.18 . Comparison of glycogen content of oysters in two river systems, showing the decrease attributed to pulp mill wastes (From Galtsoff et al. 1947; Fig. 35).

therefore decreased the time which could be used for feeding. The pumping rates of the oysters were also reduced by the sulphate wastes. After the oysters were removed from waste contact, the pumping rates increased. It was also noted that the pulp mill effluent depressed the activity of the ciliated epithelium of the gills.

Chipman (1948a) also studied the effect of sulphate wastes on the time that oysters remain open. As the concentration was increased, the time that the shell was open decreased. Chipman also observed that the muscular activity of the oysters was affected in concentrations of 1:1000 or stronger. An irritating effect occurred immediately after the material was added as rapid and frequent shell closures were observed. In very strong concentrations, the oysters lost their ability to maintain their shell closed. The strength of the adductor muscle gradually diminished until, after the shells remained open for long periods of time, death ensued.

BOD Exertion

The exertion of oxygen demand by the pulp mill wastes is manifested through its competition with organisms for dissolved oxygen in the water. The labile materials causing the BOD readily takes oxygen (thus making it unavailable for organisms), resulting in a competitive disadvantage for organisms. An example of the effects of BOD exertion by Kraft mill effluent is given by Alderdice and Brett (1957). Not only are the respiration requirements for salmon increased with increasing concentrations of Kraft mill effluent, but the dissolved oxygen available decreased with increasing concentrations (Fig. 19). The point of intersection of the dissolved oxygen curves with effluent concentration, equivalent to a concentration of 2.5% effluent in the receiving water, provides an estimate of the concentration of effluent at which oxygen availability becomes a limiting factor for fishes (Alderdice and Brett 1957).

Other examples of BOD exertion have been provided by Waldechuk (1960) for Neroutsos Inlet, B.C. and by Bartsch et al. (1960a) for Bellingham and Everett, Washington. In all three cases, the dissolved oxygen of the receiving waters was significantly decreased at the surface due to the input of the effluent.

Productivity

Hopkins et al. (1931) studied Oakland Bay prior to and after the introduction of SSL into the system. Oakland Bay has strong currents as a result of tidal fluctuations which range from 8 to 18 feet. Before the pulp mill was built, the water in the bay was clear, but after the wastes were introduced, the water became brown. The concentration of SSL in the water ranged from 1 part SSL/870 parts water to 1 part SSL/233 parts water. The oysters in the bay exhibited a high death rate after the pulp mill began operation and little spawning occurred. Other members of the community such as clams, barnacles, mussels and hydroids no longer grew on submerged objects as they did under normal conditions. The diatom Melosira borreri which was normally found in small numbers on the oyster beds grew in dense masses in areas of slow current after the introduction of the SSL into the bay water. The community changes

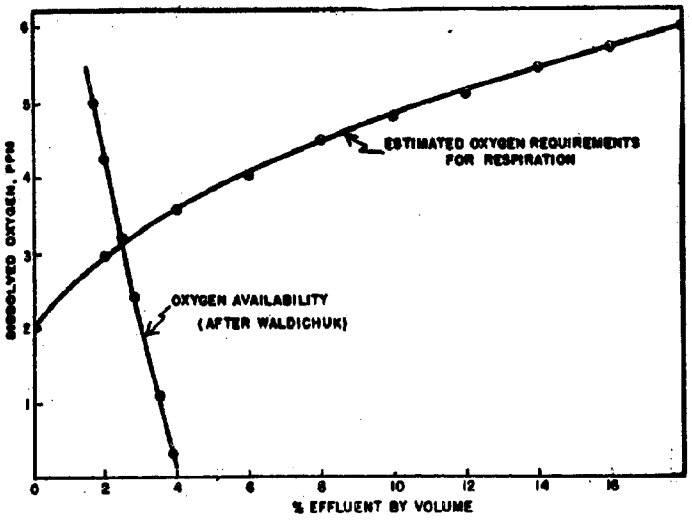


Fig.19 . Comparison of calculated net oxygen availability after effluent oxidation with respiratory requirements of sockeye salmon under-yearlings at various concentrations of kraft mill effluent (From Alderdice and Brett 1957; Fig. 5).

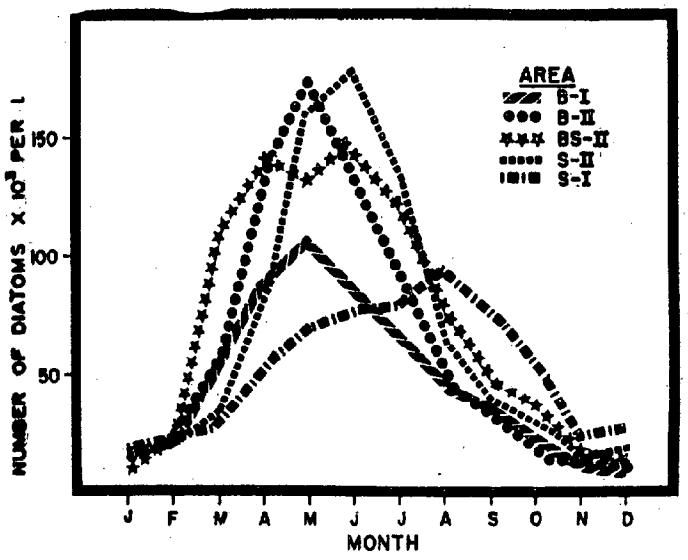


Fig. 20. Total monthly diatom population by geographical area of Bellingham Bay, Washington. The areas are listed in order of their increasing distance from the waste outfall (From Tollefson 1963; Fig. 8).

were not explained by changes in temperature or salinity. The concentration of SSL in the bay was shown by laboratory studies to adversely affect oysters. These data suggest that as a result of the SSL stress on the bay, an unbalanced ecological system developed and the net productivity of the system was reduced.

The general pattern of pulp mill effluent effects on a bay system is one of depression of primary productivity in the area adjacent to the waste outfall due to toxicity, with a general increase in primary productivity as toxicity is decreased and the fertilization effects are utilized at greater distances from the outfall. As demonstrated by Tollefson (1963) for Bellingham Bay, Washington, diatom production adjacent to the pulp mill outfall was less than 1/2 as great as the production at intermediate distances from the outfall (Fig. 20).

Species Diversity

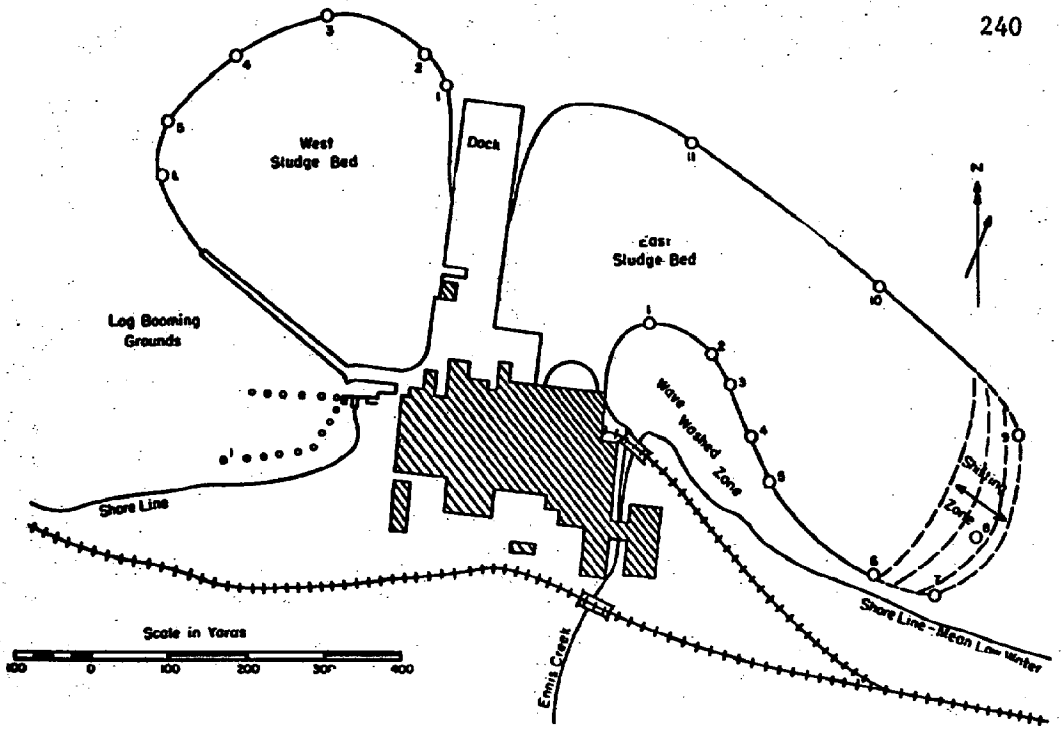
It is an old principle that the number of species or organisms able to survive an environmental stress is decreased as the stress is increased (Gleason 1922). Thus, species diversity as a measure of system organization may be expected to decrease as abnormal, newly-organizing, less-evolved system combinations resulting from pulp mill wastes replace the older more-evolved systems.

Copeland (1967a) reported the zooplankton diversity in St. Joseph Bay, Florida which is affected by Kraft mill effluent. The results of that study (Fig. 21) revealed that the diversity adjacent to the outfall was less than six species of zooplankton per 1000 individuals encountered and increased to 18 about four miles out into the bay. Copeland and Cameron (1968) calculated a correlation of 0.94 between species diversity and distance, indicating a direct effect of the Kraft mill waste lowering the diversity. Oliver and Dorris (1964) demonstrated the same relationship for algae species diversity in a fresh water system affected by Kraft mill wastes.

HORIZONTAL AND VERTICAL DISTRIBUTION OF WASTES

Wastes that are introduced into rivers may be expected to become uniformly mixed within a relatively short distance below the outfall depending on the flow and turbulence characteristics of the river. In estuaries, however, with gradations in salinity and temperature, there may be only periodic periods when uniform conditions are prevalent and in many cases such conditions never exist. Therefore, the extent of horizontal and vertical movement of effluents depends on the location of the waste outfall in relation to the water movement patterns characteristic of the estuary in question.

As discussed by Pritchard (1960b), the horizontal and vertical distribution of the wastes from a point source is accomplished by turbulent diffusion. The oscillatory movement of the tide moves the wastes back and forth in the estuary and supplies the major energy for the turbulent diffusion. A net transport of wastes occurs from regions of high concentration to those of low concentration both horizontally and vertically.



Scale Drawing Showing Exact Location, Shape, and Size of Rayonier Sludge Deposits

Fig. 4d. Characteristics of paper mill waste system in Washington (Stein, Denison, and Isaac, 1963).

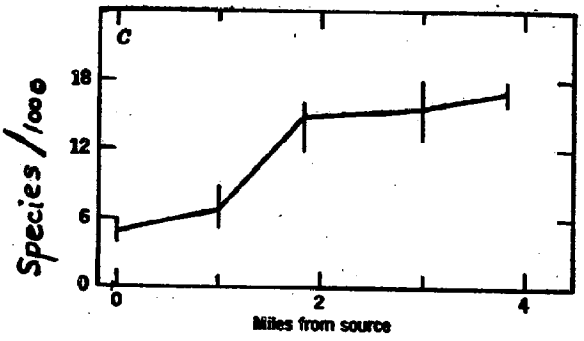


Fig. 21. Zooplankton species diversity per thousand individuals encountered in St. Joseph Bay, Florida as affected by kraft mill wastes (From Copeland 1967a, Fig. 2).

Vertical diffusion occurs most rapidly in layers having vertical homogeneity. If relatively large density differences exist between estuarine layers, vertical diffusion will occur to a large extent only within each layer and interlayer mixing will occur only slowly. The layers in a stratified estuary have varying seaward net-movements depending on the relative amounts of freshwater inflow and tidal flushing. The residence time of pulp mill wastes in estuaries therefore depends upon the net seaward movement of the layer into which the wastes are introduced. This, in turn, determines the residence time of the wastes in the estuary and, ultimately, the exertion of harmful influences.

SUMMARY AND HINTS FOR MANAGEMENT

It has been demonstrated that pulp mill wastes, upon entering the coastal environment, affect various components of the system. Where and under what conditions the effluent enters the coastal system is important to the resultant effect of the waste in the system. Thus, any management scheme would necessarily have to consider the physical arrangement of effluent input.

Different phases of the organismal structure are differentially affected by the effluent. For example, the young and/or reproductive stages of most organisms are more greatly affected than are the adults, resulting in the necessity in many cases of transporting organisms to an affected system from some other reproductive area. This would require energies expended by either man in artificially transporting juvenile organisms to affected systems or by the organism itself in reproducing in one area and swimming to the affected system. There would be no selective advantage to this scheme, thus tending to eliminate the organism in question.

Generally, the effects of the pulp mill wastes on the basic productivity of a system is that of toxicity near the outfall and fertilization at some distance from it. This generality is supported by data from Tollefson (1963) and the diversity index shown by Copeland (1967a). Management, therefore, to take advantage of the greater productivity would involve arrangement of system components in such a manner as to avoid the toxic areas and utilize the fertile areas. For example, the movement of oyster rafts into the increased productive area would result in increased oyster production due to the increase food availability. Care would have to be taken, however, to avoid the subtle effects of tissue damage to the organism by the small concentrations of waste materials as shown by Fujiya (1961).

Even though some scheme could be devised whereby management techniques could be realized for "farming" systems receiving pulp mill effluents, the solid components of the effluent are persistant in the environment (Selleck 1964). The slow decomposition of these materials would exert a BOD upon the oxygen supply in the water as well as provide a blanket over the bottom components of the system. Large deposits of fibers exist in some systems, particularly in the northwest where pulp mill processing has occurred over a long period of time and environmental temperatures are low enough to prevent rapid decay.

Finally, BOD exertion of pulp mill wastes impose an oxygen demand upon the system, thus competing with organisms for the available oxygen. The significance of this problem to the system is that energies must be expended to provide oxygen or to develop mechanisms for the organism to survive low oxygen concentrations (e.g., oxygen shunt systems or special pigmentation for transport and exchange of oxygen in low concentrations). In either case, the end result will be the lowering of diversity and/or the decrease in the energy available for growth and reproduction.

The obvious recommendation, however, is that the wastes be impounded for periods of time to allow for oxidation of toxic and organic components before the waste enters the system. This would eliminate some of the direct effects on system components.

Special Research Needed

Unfortunately, a study of the effects of pulp mill wastes on a whole system is not available. Thus, before the impact of the disposal of pulp mill wastes in coastal systems can be evaluated, we must know the magnitudes of energies involved in combating toxicity, BOD exertion, modification of tissues and interruption of energy pathways on a system scale. We have shown, in this chapter, some aspects of individual responses, but were unable to demonstrate the effects on the whole system. We recommend, therefore, that a study be initiated to analyze a typical system disturbed by pulp mill wastes.

Chapter E-8

SUGARCANE WASTE SYSTEMS

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INTRODUCTION

In the tropical regions of the United States where sugar manufacturing is of economic importance, the shallow coastal systems are receiving wastes from sugarcane processing plants and related activities. Because most estuarine systems of the tropical coasts are of high species diversity and are dependent upon high light penetration and constant environmental conditions, disturbances in these conditions result in emerging new systems characterized by different biota and ecological parameters. Wastes from sugarcane processing plants may contain bagasse (the leftover fiber), large quantities of silt from the bulldozing technique of harvesting, and high concentrations of organic materials (mostly sugars). The main stress, therefore, is the reduction of light penetration by the high turbidity wastes which tends to eliminate the high light dependent organisms such as coral and its symbiont species.

SYSTEM EXAMPLE - HONOKAA SUGARMILL OUTFALL, HAWAII

Located on the north coast of the island of Hawaii (largest of the Hawaiian Islands) is a complex of six sugarcane processing plants (U.S. FWPCA 1967e). These mills are remotely situated along an inaccessible shoreline characterized by steep cliffs one hundred to two hundred feet high. The alongshore currents push the wastes long distances along the shore and then out into the ocean. A diagram of the Hawaii Island coast is shown in Figure 1.

A composite diagram (Fig. 2) has been constructed for a disposal area of the Honokaa sugarcane processing mill using unpublished data obtained from Mr. Robert Burm, Federal Water Pollution Control Administration, Honolulu, Hawaii. The main effects of the sugarcane wastes have been the shading of coral by the highly turbid waters, the occurrence of high phosphorus and coliform concentrations, and the lowering of fish diversity and productivity. The slope of the ocean floor nearshore is steep and great depths are reached in a short distance. Thus, the mixing and dilution capacity of the deep water minimizes the effects within a short distance offshore, with some waste effects drifting along shore with the currents.

Hydrography

With the mixing and current structure of the steeply sloping ocean bottom off Honokaa, the effects of the sugarcane mill wastes on the hydrography of the area is negligible. There is no significant difference in the

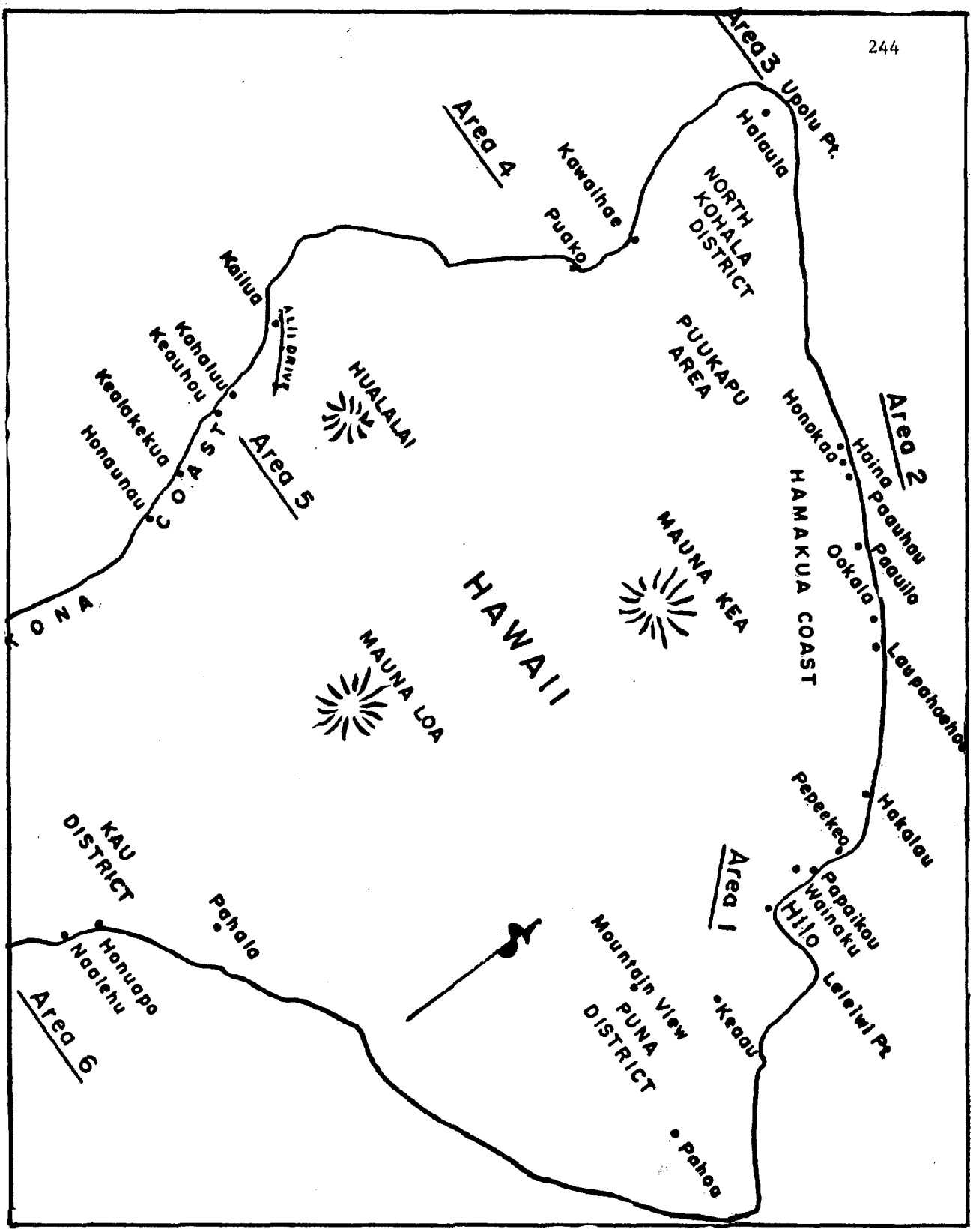

Fig. 1. Diagram of the island of Hawaii, Hawaii, showing location of the Hamakua coast (From U.S. FWPCA, 1967e).

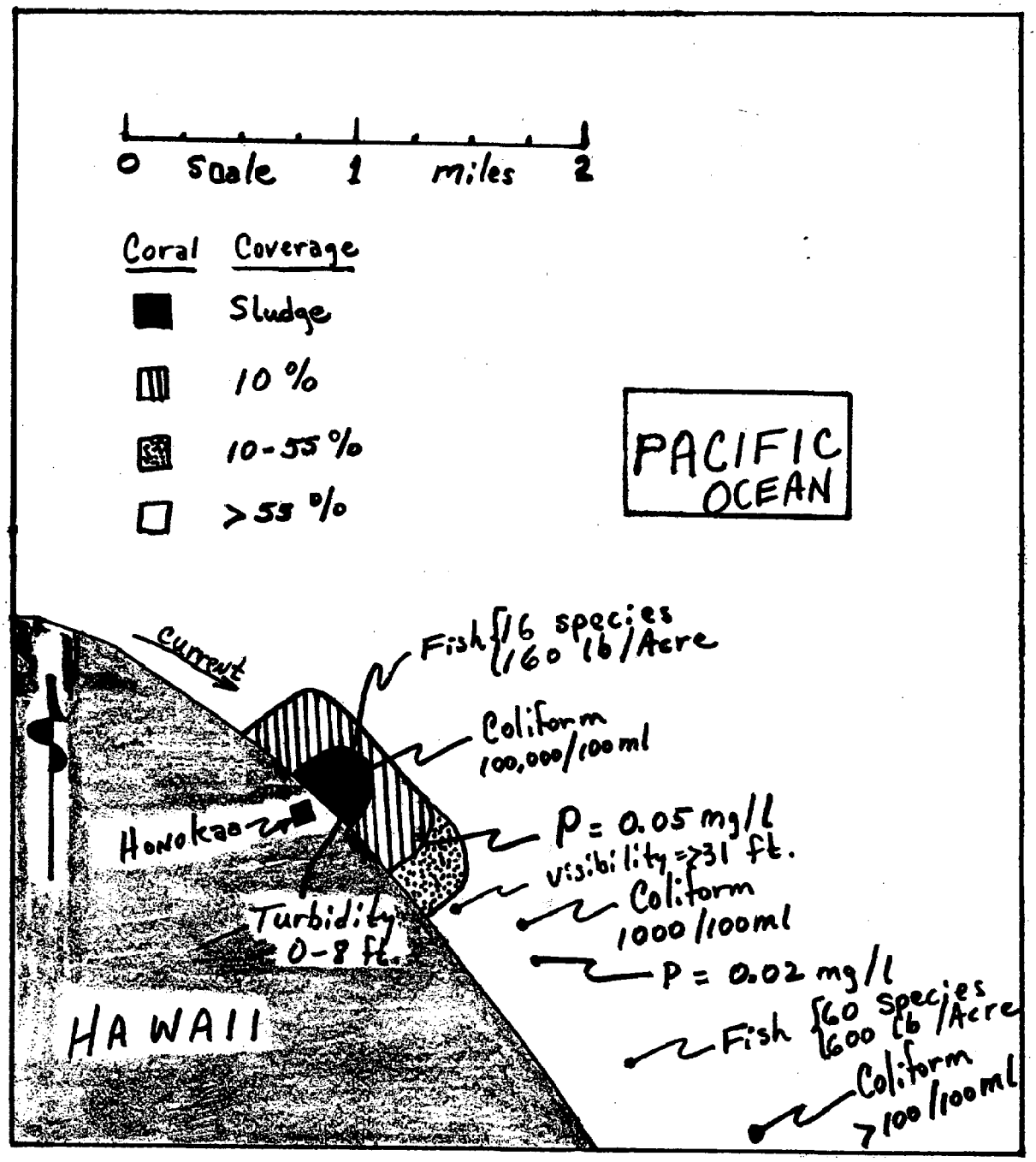

Fig. 2. Composite diagram of the Hamakua coastal area showing the results of sugarcane mill discharge into the Pacific Ocean from the Honokaa sugarcane processing mill. Island of Hawaii. (Data from Robert Burm, Federal Water Pollution Control Administration, Honolulu, Hawaii, Personal Communication).

oxygen concentration, temperature or salinity in the outfall area. The color of the waste from the sugar mill is that of the soil carried with the cane from the fields (the common mode of harvesting sugarcane is with the aid of a bulldozer and considerable soil is scraped up with the cane and hauled to the processing mill). The soil is a bright red-brown color, and this color, plus the turbidity produced by washing the cane before crushing, is discharged into the ocean producing a vivid contrast to the surrounding blue water. The alongshore currents carry this turbidity great distances along the shore instead of allowing it to be diluted further out at sea.

System Components

Coral

One of the more distinguishing characteristics of a tropical coast is the large quantity of coral. In the sugar mill waste disposal area at Honokaa, the coral has been completely covered with sludge (composed mainly of bagasse and settleable solids) within a radius of one-quarter mile from the outfall. For the next one-quarter mile on either side of the sludge deposit, the coral coverage has been reduced to about 10% total coverage. For the third quarter mile down current from the outfall, the coral coverage is between 10 and 55%. The coral coverage on the down current side of the outfall does not reach normal density until about 3/4 mile from the outfall, where coverage is about 55% (which is considered normal for comparable areas). There is little doubt that the reduced coral density is a result of the increased turbidity, since coral is reliant upon light penetration for its formation and maintenance (see chapter on Coral Reef systems).

Bacteria

At many sugarcane mills, the normal procedure is to combine human sewage with the sugarcane wastes (Guzman, 1962). This practice results in very high concentrations of coliform bacteria, as the bacteria in the warm sugar-laden waste multiply rapidly. At the outfall of the Honokaa mill, the coliform count was 100,000 per 100 ml. The coliform concentration was still as high as 1000 per 100 ml at a distance of one mile down current from the outfall. Coliform concentrations of this magnitude would render the Honokaa area unacceptable for bathing and other water sports as well as indicate the possible contamination of animals in the area available for human consumption.

Fish

Many tropical fish are dependent upon the coral reef structure for protection from predators and on the organisms symbiont with coral reefs for food. Since the coral in the Honokaa sugar mill outfall area was destroyed, it is reasonable to expect that the fish population had also deteriorated. As shown in Figure 2, the diversity of fishes in the outfall area decreased to 16, as compared to a normal 60 found two miles away. It should also be noted that the biomass of fish was also reduced near the waste disposal area; 160 lb/acre during grinding season, compared to 600 lb/acre two miles away.

A list of the 26 species of fishes collected near a sugar mill outfall (Iao Creek) on the island of Maui, Hawaii, is given in Table 1. Collections were made by the Hawaii Fish and Game Division during 1952. Although the effects of the sugar mill waste disposal were not great, possibly because the wastes were diluted and ponded for settling before being released, the 26 species found near the area were much less than the 44 species found in a similar area without the sugar mill influence.

Nutrients

Phosphorus concentrations in the Honokaa sugar mill disposal area was two to three times higher than in the outlying waters (Fig. 2). The higher phosphorus concentrations were possibly related to the sewage input in the sugar mill waste.

OTHER EXAMPLES

Hawaii

On the island of Maui, Hawaii. there are three additional examples of sugarmill waste pollution (FWPCA, 1967f). Two of the three main mills on the island are in the Wailuku-Kahului area and the third at Paia (Fig. 3). All three mills utilize a hydroseparator and settling ponds to prepare the mill waste water for irrigation by sediment removal. Problems result when excess effluent not needed for irrigation are introduced into streams which enter the sea. Heavy rainfall may also cause overflows of the holding ponds and flush out the streams thereby carrying an increased loading of waste water to the ocean.

At the Wailuku Mill, the sludge pond supernatant and the hydro-separator effluent carry high color, sediment and a high BOD (500 to 1000 mg/l). The estuary of the receiving stream is red-brown in color from the mud which shades out the coral in the Kahului Bay. The other two mills, with similar effluents, are located at Puunene and Paia and also empty into Kahului Bay (Fig. 3).

Puerto Rico

Sugarcane wastes consisting of bagasse, sludge cake (cachaza), cooling waters, and concentrated wastes containing organic substances, simple sugars and scums discharging in Puerto Rico coastal systems are summarized by Biaggi (1968). A map of plant locations and disposals is shown on Figure 4. Related wastes from rum manufacturing are generated west of San Juan.

One case of utilization of the bagasse from sugarcane mill processing is at Arecibo, where the bagasse is utilized by a paper mill to manufacture low-grade paper.

Louisiana

A highly seasonal sugarcane processing waste disposal activity occurs in Louisiana. Keller and Huckabay (1960) discuss the sugarcane processing problems in Louisiana. A map of the Louisiana sugarcane growing areas is shown on Figure 5.

Table 1. Fish species collected near Iao Creek mouth, Maui Island, Hawaii. (From: Hawaii Division of Fish and Game, 1953b.)

Species	Size	I	II	III	IV	V	Total
Acanthurus sandvicensis		14	1				15
Zebrasoma flavescens	2"				20	1	21
Zebrasoma veliferum					2		2
Acanthurus barienne			5	1	1	1	8
Acanthurus olivaceus	10"					3	3
Acanthurus lineolatus			3	5			8
Ctenochaetus strigosus					2		2
Acanthurus leucoparicus			6	7	3		16
Chaetodon setifer		1					1
Chaetodon fremblii		1					1
Cirrhitus pinnulatus				1		2	3
Red Uhu					2		2
Thalasoma duperrey		55	100	37	13	31	236
Bodianus bilunulutus	4"-10"	4	1	11		1	17
Iniistius sp.		1					1
Anampses cuvier		1					1
Pseudupeneus pleurostigma			2	4	2		8
Pseudupeneus porphyreus				4	2		6
Pseudupeneus bifasciatus		1	1				2
Pomauatrus jenkinsi		6	1	3	7	3	20
Abudefduf abdommalis					1		1
Chielio inermis		1					1
Canthigaster jactator		1					1
Monacanthus spilosoma					1		1
Balictapus aculeatus			1				1
Zanelus cornutus				7			7
				Total			385

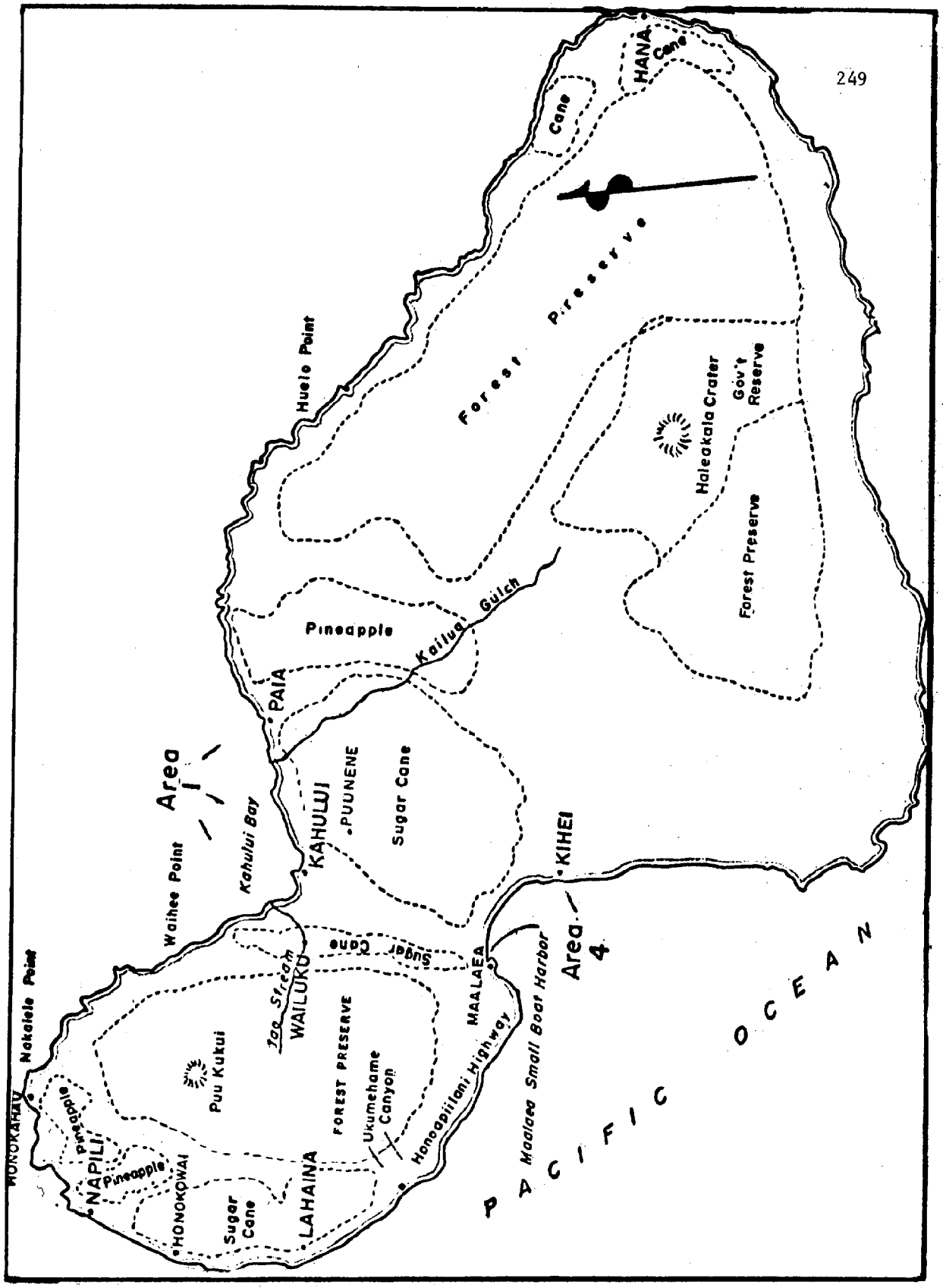

Fig. 3. Diagram of the island of Maui, Hawaii, showing the location of several sugarcane producing areas (From FWPCA 1967f).

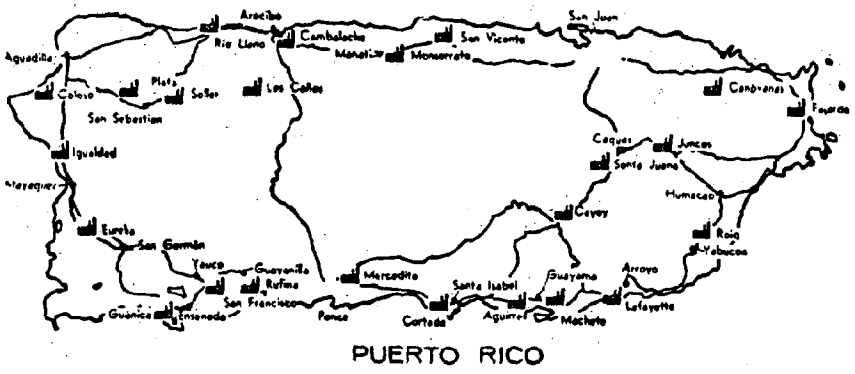

Fig. 4. Map of Puerto Rico showing the locations of sugarcane processing plants and points of ocean disposal of wastes (From Biaggi, 1968; Fig.1).

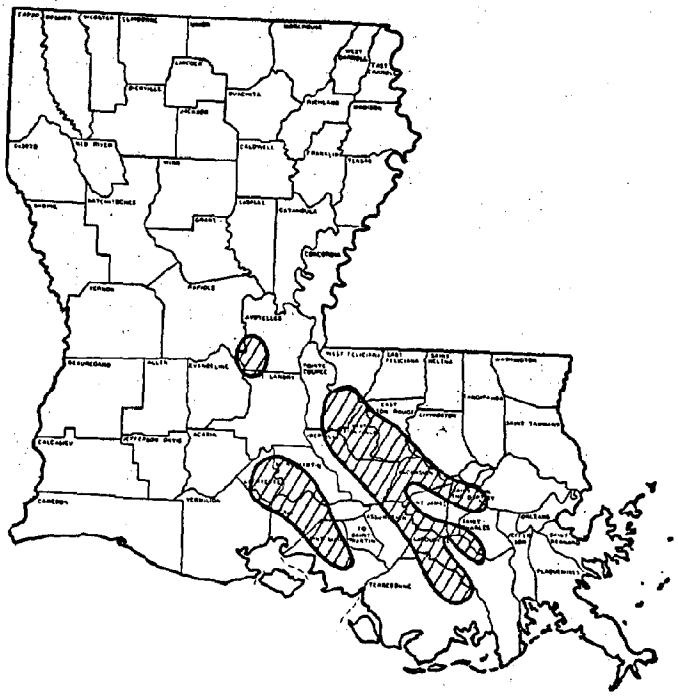

Fig. 5. Sugarcane growing and processing areas of Louisiana. (From Keller and Huckabay, 1960; Fig. 1.)

The Bayou Teche drains the New Iberia sugarcane growing area and receives considerable processing wastes during the grinding season (autumn). The Bayou Teche enters coastal waters in the Terrebonne Parish and the sugarcane waste during autumn causes large fish kills. The Bayou (and at its mouth) contains brown water during the autumn grinding season that possesses an extremely high BOD from the sugar, which results in anaerobic conditions and a subsequent "kill-off" of most organisms.

Chapter E-9

ECOSYSTEMS RECEIVING PHOSPHATE WASTES

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Phosphorus is needed in all living organisms, yet is present in very low concentrations in normal plant and animal environments. In estuaries, phosphorus as dissolved phosphate is found at close to 30 parts per billion (1 µg-atom/liter; 0.03 mg P/l).

Because the concentration is so low, the amount of phosphorus may fall to a point where photosynthesis and growth are reduced; the phosphorus is then considered to be the limiting factor. Most of the high phosphorus levels found in estuaries are a result of sewage pollution, and other nutrients, such as nitrogen and vitamins, are also abundant. Estuarine situations where phosphorus is the sole pollutant are rare and are confined to the phosphate mining regions of Florida and North Carolina. The mining activities also introduce two non-nutrient pollutants, slimes and fluoride, into the estuaries. This section, therefore, deals with effects of these three pollutants on the estuarine systems.

The commercially important deposits are found as phosphate sands from the Pamlico River in North Carolina to Savannah, Georgia, and as phosphate rock in central and northern Florida (Odum, 1953). One company is currently working in North Carolina (with four or five others planning future operations) and close to fifteen operating in Florida. Whether the phosphate is in sands or in rock, the general mining and treatment procedure is the same. The ore is reduced to fine particles followed by several washing and floatation steps to concentrate the phosphate. At this point, the sands, clays, clay-sized particles of rock and unrecovered phosphate, plus the fatty acids and amines used in floatation, are transported to settling basins (slimes ponds). The phosphate concentrate is then treated to make phosphoric acid or fertilizer. Fluoride, from calcium fluoride apatite, is the main pollutant arising from the last treatment, but a number of plant pollutants also escape to the estuaries and rivers due to cleaning of tanks, occasional pipe-line breaks, etc.

The possible effects of these various pollutants are known from laboratory experiments and observations on other aquatic systems. The most important system affected is the primary producers. Any changes in this group of algae and rooted aquatic plants will have tremendous importance for the remainder of the organisms. Although many different ions are needed by the primary producers, the phosphorus and nitrogen compounds are usually the limiting factors because of their low concentrations. Increased amounts of P and N have led to different species becoming important and the increased productivity has led to deep-water anaerobic layers that have killed some benthic species (e.g. the mayflies in western Lake Erie). Phosphorus as a lone pollutant will have little effect unless nitrogen compounds are already present or unless nitrogen fixing blue-green algae are abundant (e.g. Shapiro's (1965) work on the Potomac Estuary). If enough phosphoric acid or other acidic compounds are released, then the pH will be drastically lowered and organisms eventually killed. For example, all of

the invertebrates were killed in Bear Branch (Florida) when the pH dropped below 3 because of phosphate mining effluents (Lanquist, 1953). Low pH also increases the amount of dissolved phosphorus in the water.

The most important potential source of serious pollution is the slime waste. By definition, this is ultrafine material, consisting of clay, silt, and phosphate sands smaller than 200 mesh. The majority of the solid material is between 0.3 and 3 μ. An analysis of the Florida slimes was presented by McElroy (Table 1). He described it as a solution of several substances mixed with a colloid and containing a coarse mechanical suspension of solids. The material can contain up to 1/3 of the phosphorus of the original ore. When the slimes are pumped into the settling basin, most of the particles settle out and the effluent water is quite clear. The settled slimes still contain 60 - 80% water and years of settling produces little change. Cattails will grow on the semi-solid top few inches of old slimes, but the material beneath the surface (which may be 8 meters thick) remains semi-liquid and will move out through any opening dikes. This is a real and continuing hazard and dike breakages have occurred a number of times in Florida, the latest in March, 1967. In rivers, the slimes are kept in suspension by the moving water, but settle in every pool and kill many organisms by clogging their gills. Others are eliminated by the blocking of suitable substrates or by the covering of their food, such as dead leaves.

Some fluoride is also released from the mining operation. This is dissolved from the ore during treatment with sulfuric acid and usually will be contained within the plant. However, because of overflows and seepages, some fluoride is usually present in the plant wastes. Fluoride is poisonous at relatively low concentrations even though it is present in all waters. For example, oysters from the Pamlico River died in 30 mg F/l (the natural level is 0.5 mg/l).

EXAMPLES

Rivers and Tampa Bay

The Florida mining operations discharge wastes into the Peace and the Alafia Rivers which run, respectively, southwest into Charlotte Harbor and west into Tampa Bay (Fig. 1). Table 2 indicates that the rivers have a high phosphorus content, some of which may also arise from the natural rocks of the drainage basin (Odum, 1953). The turbidity and pH were normal during 1948 as less than 100 mg/l turbidity is not harmful to life. The large break that occurred on 11 March, 1967 (Table 2) killed 91% of the fish (Ware, 1967) over 76 miles of river (an estimated 983,000 fish). The flood of turbid water raised river levels by as much as 4 feet and deposited 1 to 3 inches of slimes on the banks and flood plain and up to 21 inches on the river bottom. Each heavy rain will bring these slimes back into suspension and it may take 5 to 10 years before normal habitat is restored. Based on the work of Lanquist (1953), it is likely that the mussels and other clean stream invertebrates will be eliminated and organisms typical of low oxygen, polluted waters will be the first to reenter the stream. This system will occur because the organisms are adapted to obtaining oxygen from air and have no gills to be clogged by the slimes.

In spite of the phosphorus entering into Charlotte Harbor (Odum, 1953),

Table 1. Chemical analysis of typical phosphatic slime. (McElroy, 1967)

Substance	%
P_2O_5	20.0
SiO_2	21.0
CaO	18.0
Al_2O_3	20.0
Fe	4.5
F	1.7
Loss on Ignition	12.0
Miscellaneous	2.8
Total	100.0

Table 2. Data from the Peace and Alafia Rivers, Florida (from Specht and from Ware, 1967).

Location	Date	Phosphorus (ug-at/1)	Turbidity	pH
Peace River	April 5, 1948	280	30	6.8
	April 12, 1948	513	50	7.1
Alafia River	April 2, 1948	17	25	7.3
	April 23, 1948	150	25	7.0
Peace River				
1.5 miles above pollution	March 17, 1967		20	
1 mile below break	March 12, 1967		31,200	
5 miles below break	March 14, 1967		37,250	
28 miles below break	March 14, 1967		40,000	
76 miles below break	March 17, 1967		18,750	

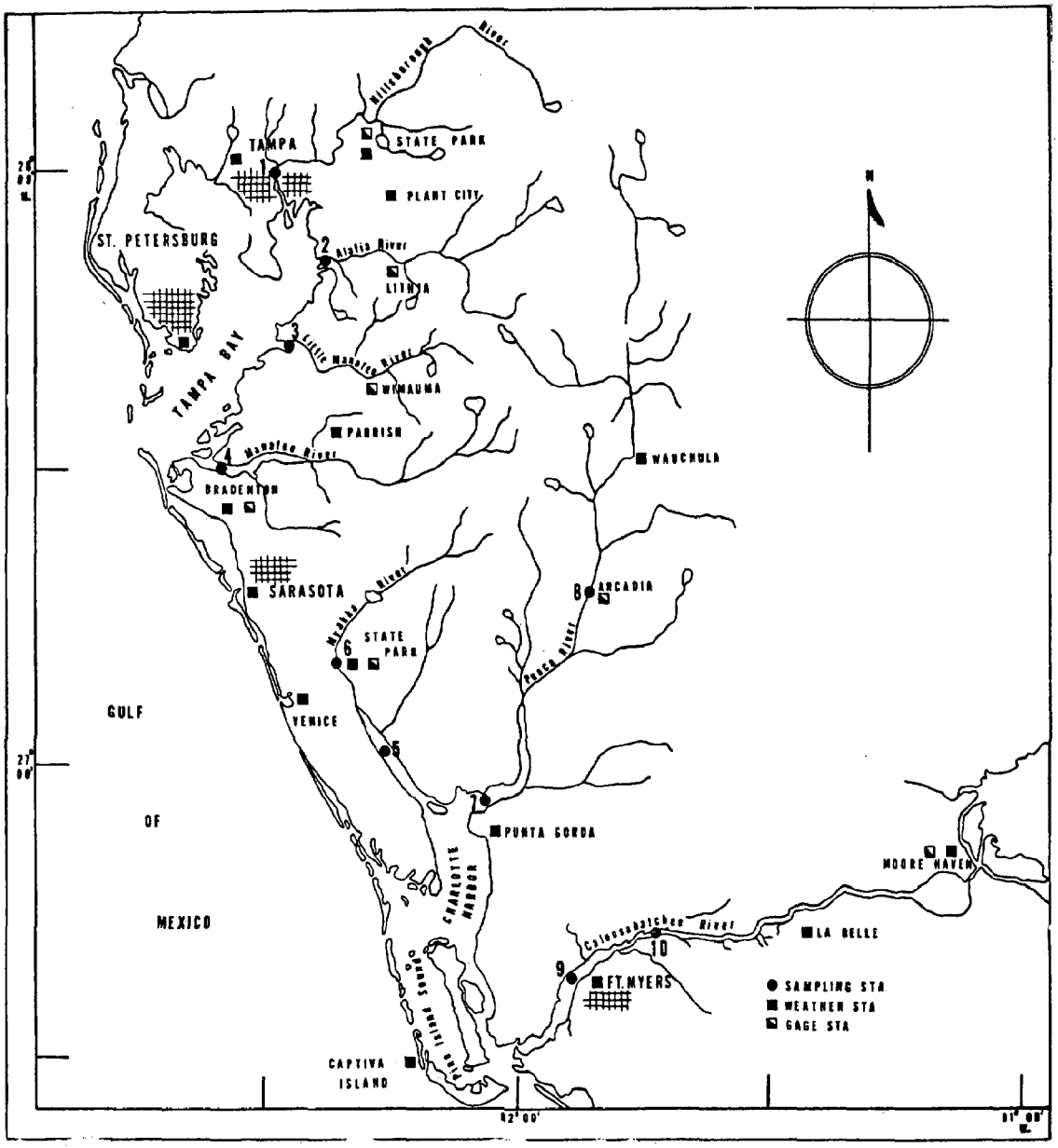
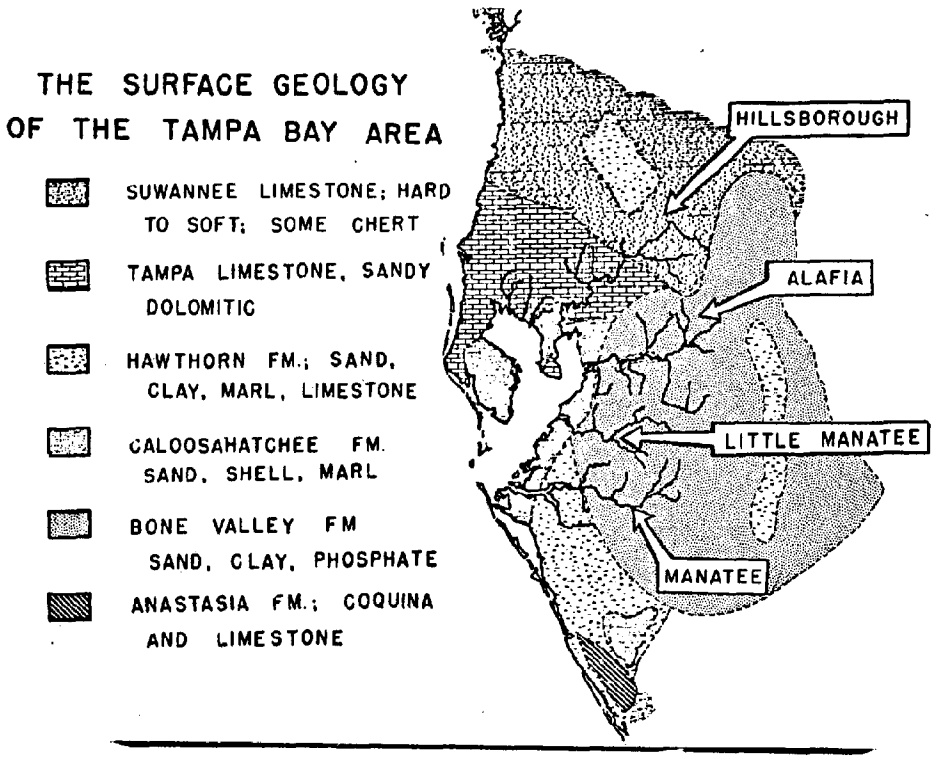
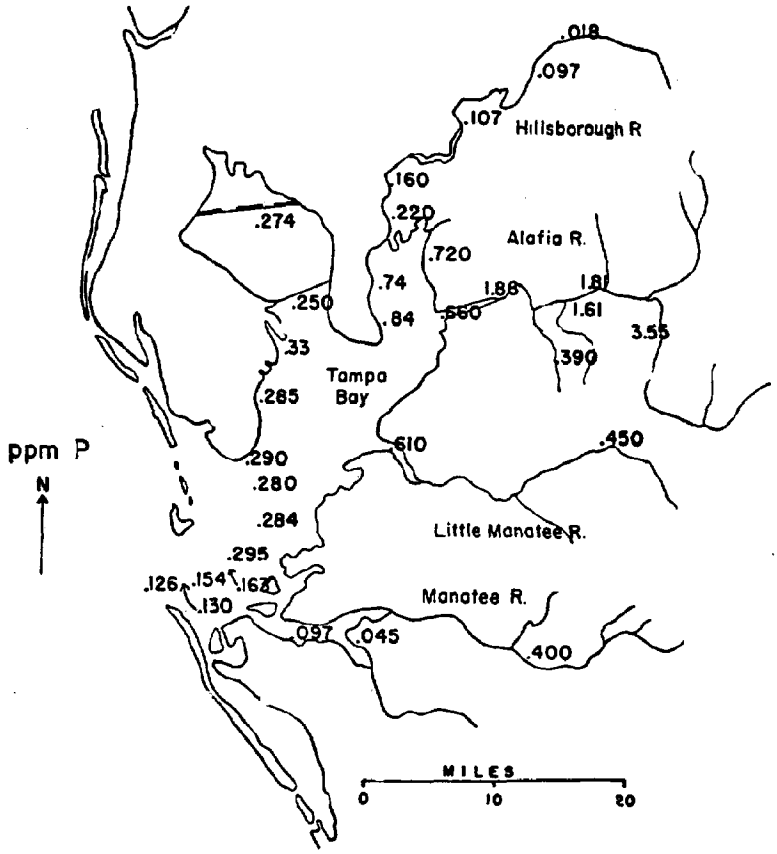


Fig. 1a. Florida west coast showing Tampa Bay and Charlotte Harbor (from Dragovich, Kelly, and Goodell, 1968).



—*Surface geology and stream drainage in the area tributary to the Tampa Bay System.*

Fig. 1b. Phosphate bearing geological formations (Goodell and Gorsline, 1961).



—Dissolved phosphorus in the Tampa Bay region, Florida. Data obtained in cooperation with Dr. Nelson Marshall, Florida State University in the fall, 1952.

Fig. 1c. Nutrient phosphorus (total) in Tampa Bay (Odum, 1953).

there is no evidence that any change in the natural estuarine system has taken place (Dr. Perry Gilbert, personal communication). Evidently another nutrient, probably nitrogen, is limiting algal growth. The sediments, however, have an increased phosphate load (Fig. 2). Table 3 has data on fishes.

The situation is very different in Tampa Bay, where sewage, phosphate wastes, and decaying water hyacinths have created eutrophication seldom equalled in the United States. There is so much nitrogen and phosphorus coming into Hillsborough Bay (the northeastern part of Tampa Bay) that the role of the phosphate wastes are difficult to determine. An F.W.P.C.A. project under the direction of T. Gallagher and J. Hagen (Southeast Water Lab., Athens) has determined that phosphate mining wastes are the major source of phosphorus for Hillsborough Bay. Data presented here are from a preliminary report (final report to FWPCA expected in early 1969).

Hillsborough Bay (Figure 3) is a protected arm of Tampa Bay about 4 miles wide and 8 miles long. The maximum depth is 28 feet and the average is 12 feet. Several 34-foot-deep dredged channels traverse the bay. The tidal effect is slight and the maximum current measured was 0.5 miles per hour. Phosphorus enters the bay chiefly from the Alafia River (Figure 1) and the highest concentrations, 4.9 mg P/l, were found along the eastern shore close to the river. The concentration of organic nitrogen at the mouth of the Alafia River averaged 0.77 mg N/l during June, 1967. Decaying water hyacinths from the Hillsborough River enter the bay at the rate of ten million pounds/year or about 700,000 pounds of N and 56,000 pounds of P. The Tampa sewage contributes an estimated 3500 pounds N/day (calculated from the sewage flow).

In response to the abundant nutrients, the system of primary producers has changed. Instead of a mixed benthic flora now found in other parts of Tampa Bay, Hillsborough Bay has almost a pure stand of Gracillaria, a red alga. This alga is killed by fresh-water at least once a year and tremendous odor problems result. Over one 16 month sampling period, the biomass of Gracillaria fluctuated from 2300 to 400 pounds/acre or a maximum of 260 gm/m^2. The dominant primary producer was the phytoplankton with an average gross photosynthesis of 5.3 gm O_2/m^3/day (net 3.3; respiration 1.9). This was equal to about 3.8 gm dry organic matter /m^3/day or 1380 gm/yr. If the photosynthetic zone is taken as 2 or 3 meters deep, this becomes a very high value comparable only to the most fertile estuaries (Odum, 1959). A Gymnodinium species had a bloom of red-tide proportions in February, 1967.

Information available on the other aquatic organisms of Hillsborough Bay (Figs. 4 and 5) indicates a reduced fish population and lower numbers of lancelets and brachiopods. Area IV in Figure 4 is Hillsborough Bay while Area III is Old Tampa Bay.

In conclusion, it is obvious that the estuarine system that has evolved in Hillsborough Bay is responding to the high level of several nutrients and is not a special phosphate-caused system. The phosphorus levels are 100 times higher than normal estuarine concentrations, but the nitrogen level is only slightly higher than normal. Usually a N:P ratio of 8:1 is expected in natural estuaries, but in Hillsborough Bay the ratio is close to 1:3. There is an excess of phosphorus in this system and nitrogen is probably the limiting factor.

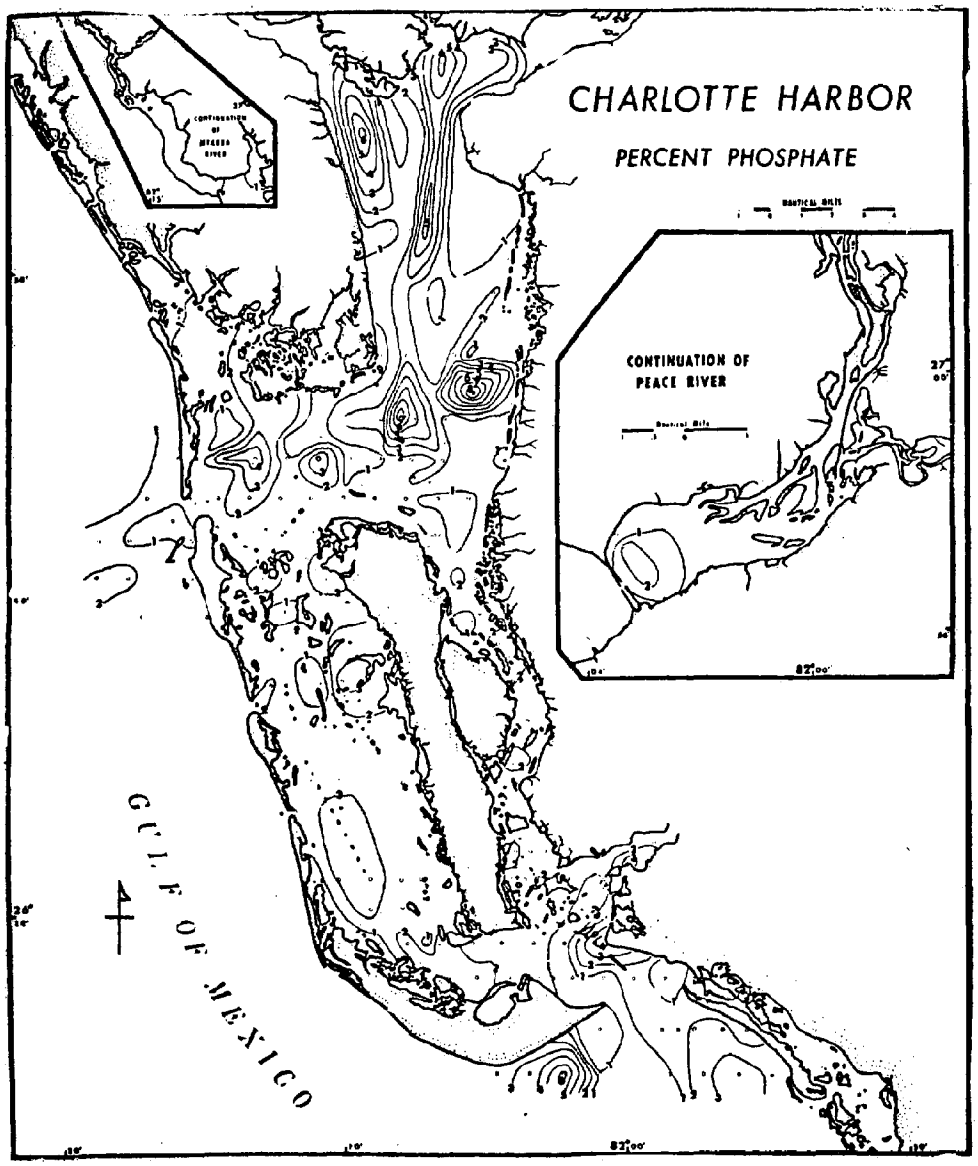


Fig. 2a. Phosphorus distribution in sediments of Charlotte Harbor, Fla. (from Huang and Goodell, 1967).

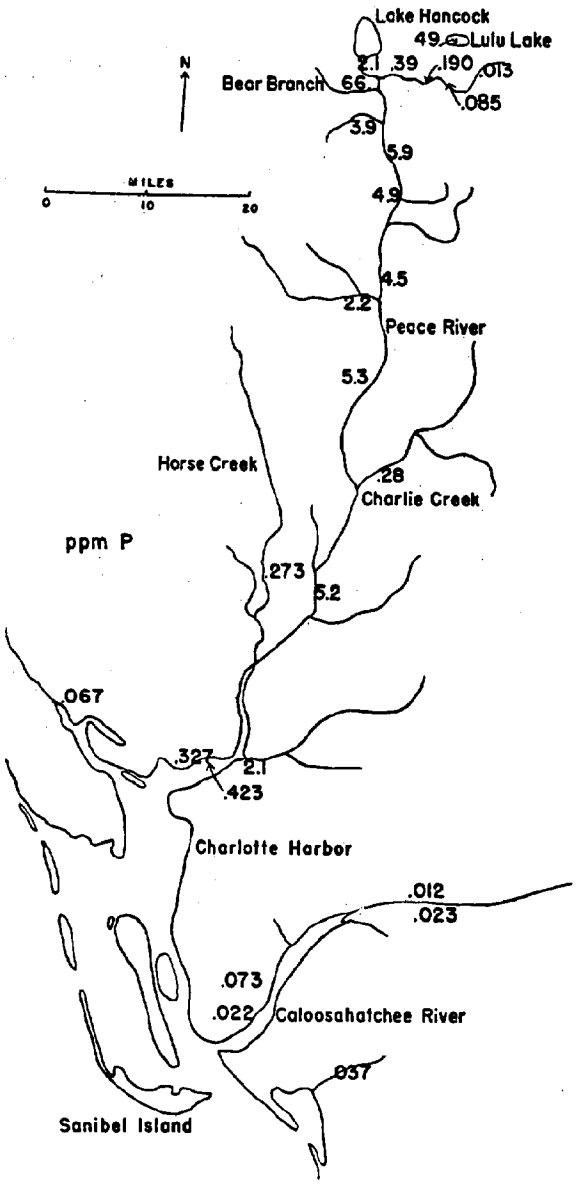


Figure 5.—Dissolved phosphorus in the Peace River system. Data obtained in cooperation with Mr. Ellis Landquist.

Fig. 2b. Total dissolved phosphorus in Peace River and Charlotte Harbor in 1952 (Odum, 1953).

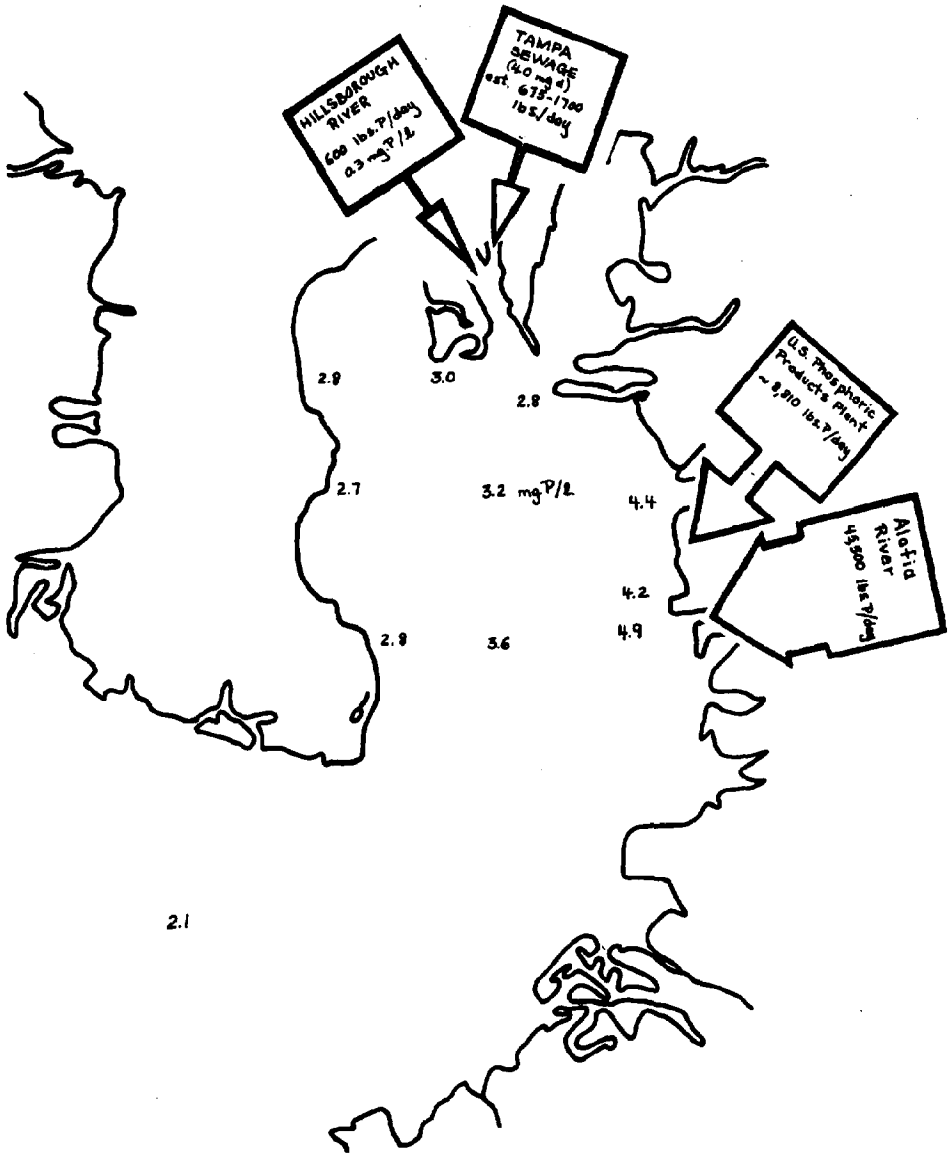


Fig. 3. Hillsborough Bay, Florida, showing dissolved total phosphorus and the amount of phosphorus entering the bay.

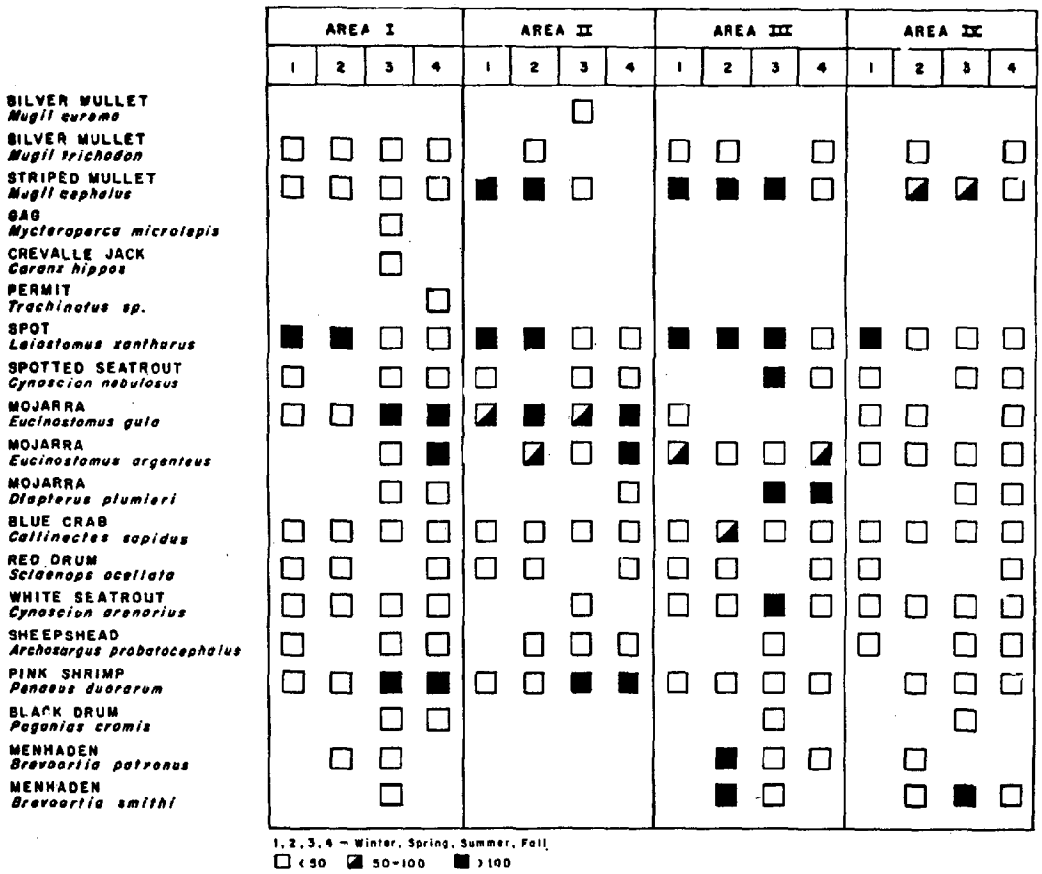


Fig. 4. Occurrence of immature species of commercially important fish and shellfish in Tampa Bay, Fla., by season and area (from Finucane, 1966).

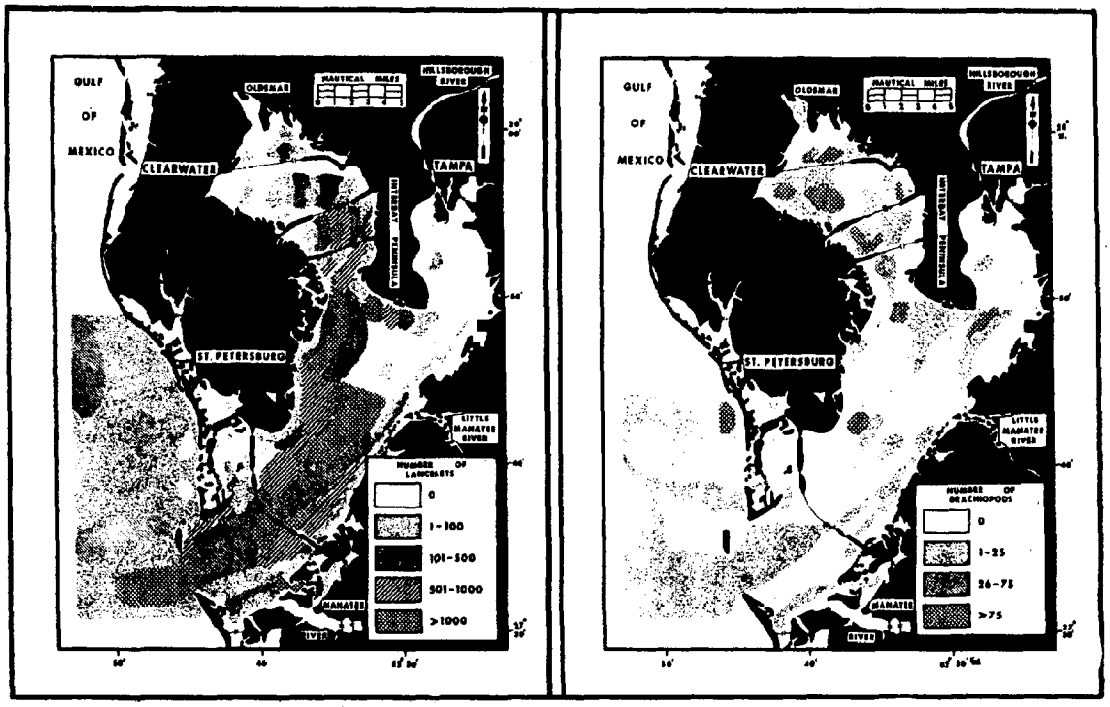


Fig. 5. Quantitative distribution (number per sample of about 300 $cm.^2$ -- 46.5 square inches) of the Lancelet, B. floridae, and the brachiopod G. pyramidata, in Tampa Bay, Fla. (from Taylor and Saloman, 1967).

Pamlico River, North Carolina

In North Carolina, the phosphate mining area is located on the Pamlico River (Fig. 6). The phosphate, in small nodules intermixed with sand, is covered with 60 - 90 feet of sand, shell, and clay. Processing wastes are discharged at the plant (TGS in Fig. 7), and slimes decant water at S.P. The river section shown (15 x 3 miles) has a salinity of 5 to 18‰ and is rarely over 18 feet deep (average 12-14 feet). The tidal amplitude is only 1/2 foot and currents are slow. In general, this is an oligohaline estuary of the type described in another chapter. Shallow bays have Potomogeton and Ruppia growing on the bottom.

The dominant bivalve is Rangia, and dinoflagellates make up most of the phytoplankton. Data reported here have been collected by projects of D. B. Horton and J. E. Hobbie.

The decant water from the slimes pit is clear water containing up to 20 μg at P/l. However, the discharge is small (50 lbs/day) compared with the volume of the river and so little effect on the total phosphorus is noted (Fig. 7). From this figure, the importance of the plant discharge is obvious and the general pattern of circulation can be noted. Water movement is slow enough so that the high phosphate waters could be followed for more than six weeks. Evidently a large amount of phosphate, perhaps as phosphoric acid, had escaped from the plant some days earlier. This was an unusually large discharge that only happened once or twice a year. The pH in the plant area was lowered to 6.5 from the usual 7.5 to 8.

The response of the algal primary producers to the high phosphorus levels can be looked at in terms of biomass or of potential photosynthesis. The plankton biomass was estimated by direct count of sedimented plankton and is given in Fig. 8 as mg wet weight/l. The diameter of each circle is proportional to the cube root of the biomass with a range of 2.3 to 53.0 mg/l shown here. A large bloom of Peridinium triquetrum, Prorocentrum minimum, and Katodinium sp. was responsible for most of the biomass and this bloom lasted for nearly three months.

Potential photosynthesis was estimated from surface samples incubated with carbon-14 under controlled temperature and light conditions. The diameter of circles is proportional to the cube root of the mg C/m^3/hr retained in the algae with a range of 0.8 to 387 mg (Fig. 9). It appears that high productivity occurs in the middle part of this river section, but that it is not necessarily associated with the high phosphorus levels.

Figs. 10, 11, and 12 cover the same parameters for the end of July 1967. Although dinoflagellates were still dominant, the Peridinium triquetrum were absent and the biomass ranged from 0.8 to 4.5 mg. Dominant species were Peridinium trochoideum, Nitzschia closterium, Prorocentrum minimum, a large Gymnodinium sp., and Gyrodinium estuariale. The range of potential photosynthesis was 8 to 178 mg C/m^3hr. At this date, despite the different concentrations of phosphorus found in the river, the biomass and potential photosynthesis were at the same level throughout the measured section.

Another way to examine effects of phosphorus on the Pamlico River is

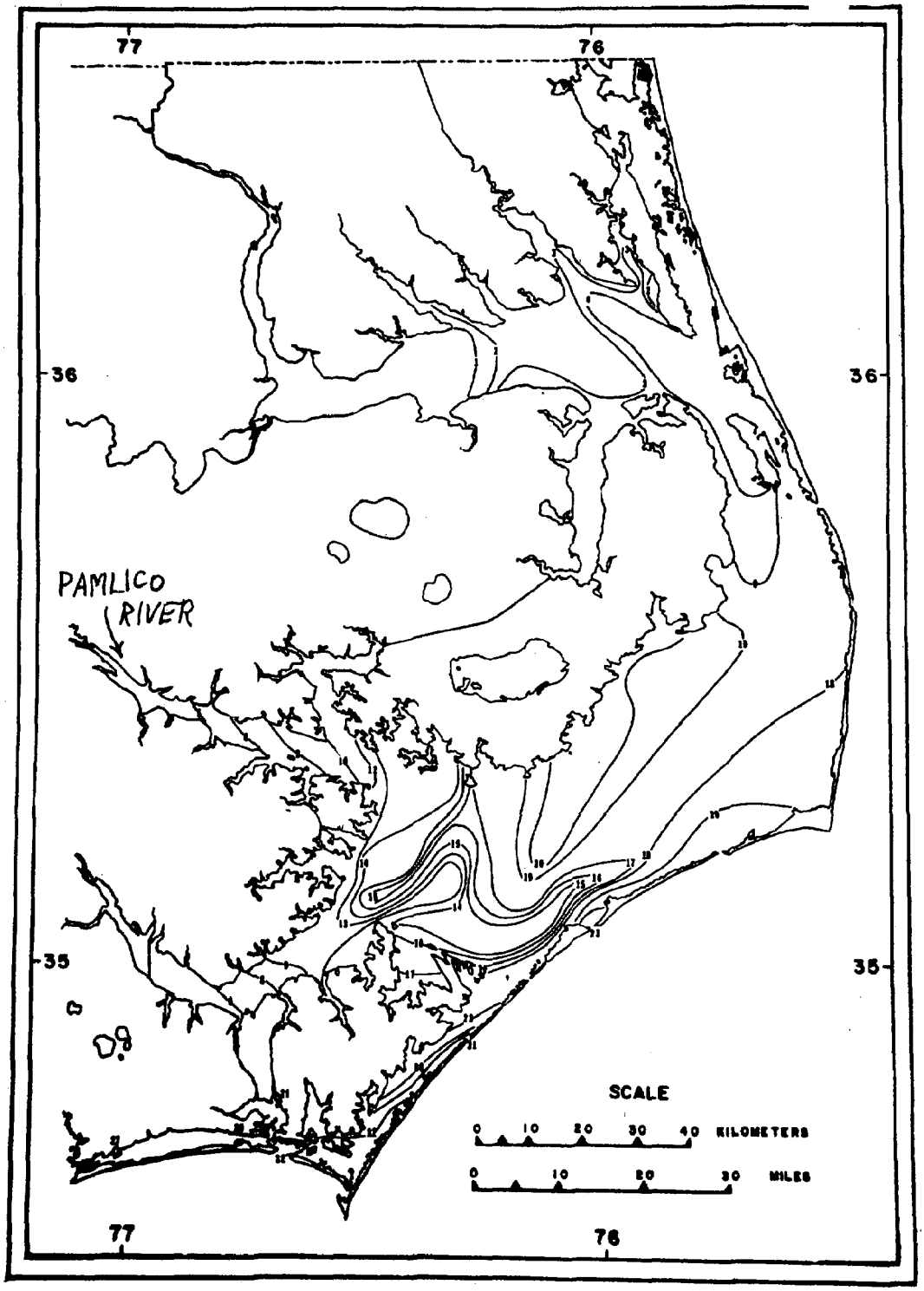


Fig. 6. Location of Pamlico River, N. C. and February surface isohalines for North Caroline estuaries (from Woods, 1967).

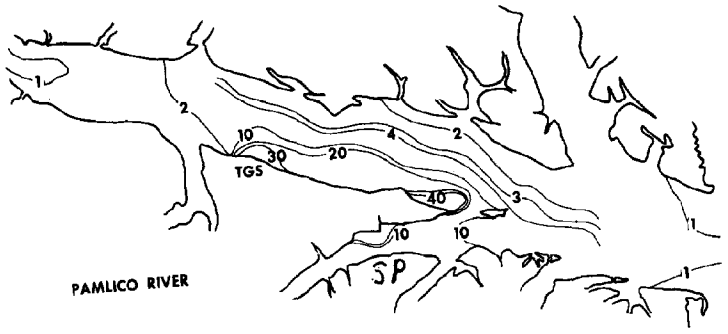


Fig. 7. Total unfiltered phosphorus in the Pamlico River, 8 January, 1968. Concentrations are in µg-at./l.

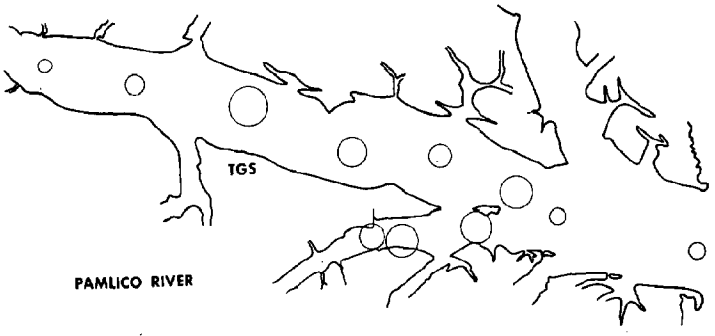


Fig. 8. Biomass (wet weight) of phytoplankton in the Pamlico River on 5 January, 1968. The diameter of the circles is proportional to the cube root of the biomass (mg/l).

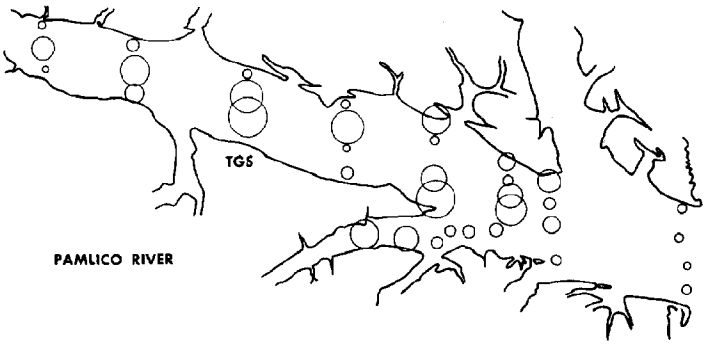


Fig. 9. Carbon-14 uptake (mg $C/m^3/hr$) under controlled light and temperature conditions for samples from the Pamlico River on 8 January, 1968. The diameter of the circles is proportional to the cube root of the mg C fixed.

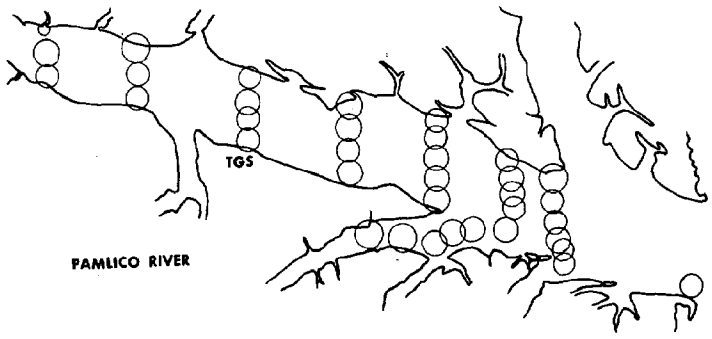


Fig. 10. Carbon-14 uptake (mg C/m^3/hr) under controlled light and temperature conditions for samples from the Pamlico River on 29 July, 1967. The diameter of the circles is proportional to the cube root of the mg C fixed.

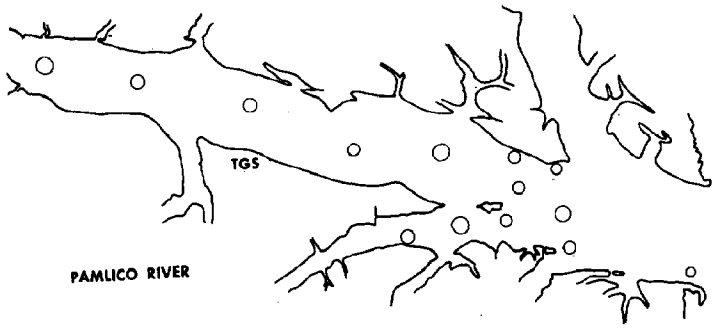


Fig. 11. Biomass (wet weight) of phytoplankton in the Pamlico River on 31 July, 1967. The diameter of the circles is proportional to the cube root of the biomass (mg/l).

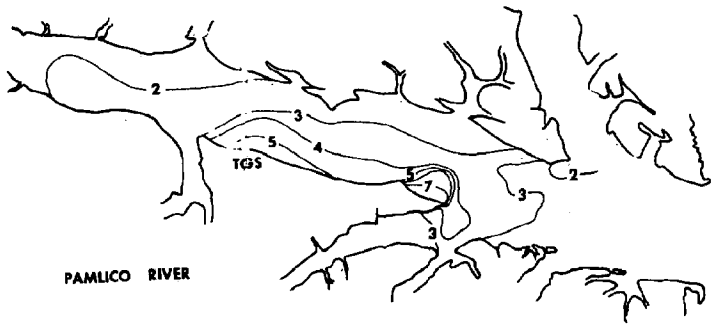


Fig. 12. Total unfiltered phosphorus in the Pamlico River, 31 July, 1967. Concentrations are in µg-at./l.

through nutrient addition studies. In this experiment (Figure 13), water samples were taken inside the slime pit and in the river, 3/4 mile away from the pit. The samples were incubated in the laboratory with carbon-14 plus addition of phosphate, nitrate, vitamins, or other salts needed for algal growth. The sample from inside the slimes pit showed no response over controls, but the river sample gave a good response to nitrate alone or to nitrate plus phosphate. Total nitrogen concentrations in the river are usually close to 1 mg at./l (0.014 mg N/l).

Examination of the distribution of benthic organisms revealed one anomaly; close to the present plant there is an area of river bottom devoid of organisms. This is probably the place where slimes were put into the river during a pilot plant operation before 1965. Evidently the slimes were not washed away in the slow currents of the estuary, and are not suitable for benthic organisms.

There is evidence that some fluoride is reaching the river from the plant drainage. Normal background in the river is 0.5 mg F/l and close to the plant over 10 mg F/l have been measured. This quantity is rapidly diluted (Figs.14, 15) however, and no increase over background is found 1/4 mile from the plant.

In summary, the estuarine system of the Pamlico River has not changed measurably because of the phosphate mining operation. The potential hazard of the slimes has been successfully dealt with so far and fluoride is mostly removed at the plant. With a N:P ratio of 1:4 or 1:40 instead of the 8:1 usually found, it is obvious that nitrogen is the limiting factor. In contrast to Hillsborough Bay, the nitrogen levels in the Pamlico River have remained low. Thus, although nitrogen is limiting in both situations, the increased nitrogen of Hillsborough Bay permitted new types of organisms to come in.

Table 3

--Dominant families and species of fish caught in Charlotte Harbor and Pine Island Sound, January 1964 through January 1965

/Percentage of catch taken from total number of fish caught monthly at five trawl and two beach seine stations in each area/

Families	Catch	Species	Catch
Charlotte Harbor			
	Percent		Percent
Engraulidae.....	47.9	Bay anchovy *Anchoa mitchilli*.	46.2
Sparidae.....	12.3	Pinfish *Lagodon rhomboides*.	12.2
Sciaenidae.....	10.6	Spot *Leiostomus xanthurus*.	6.5
Atherinidae.....	7.6	Silver jenny *Eucinostomus gula*.	5.4
Gerridae.....	6.0	Tidewater silversides *Menidia beryllina*.	5.0
Cyprinodontidae.....	4.1	Menhaden *Brevoortia* spp.	3.9
	88.5		79.2

(Finucane, 1966)

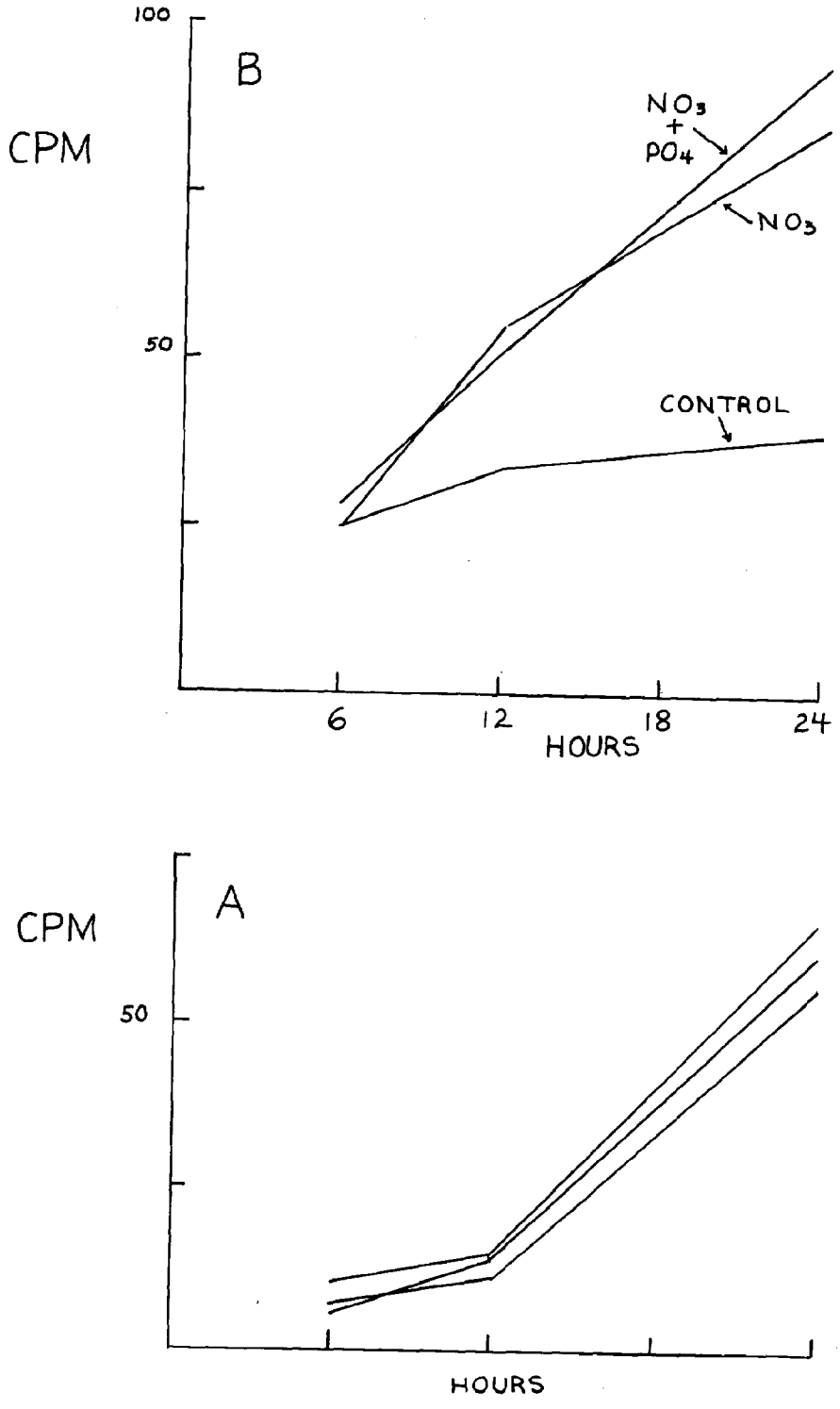


Fig. 13. The effect of the additions of NO_3 and PO_4 plus NO_3 on the uptake of carbon-14. Samples were taken from the slimes pit (A) and from a point 3/4 mile away from the pit effluent pipe (B) in the Pamlico River.

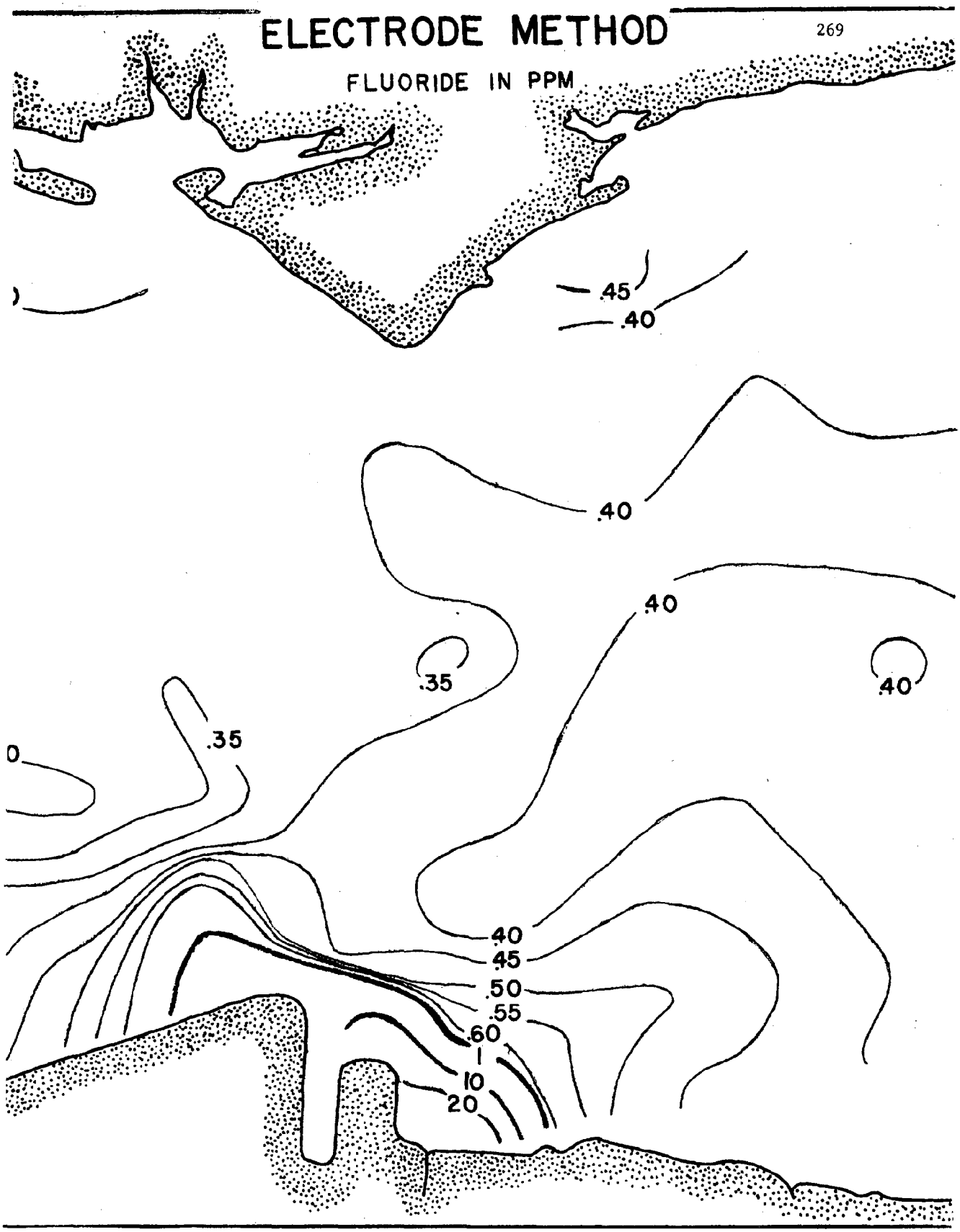


Fig. 14. Distribution of fluoride (mg/l) near TGS.

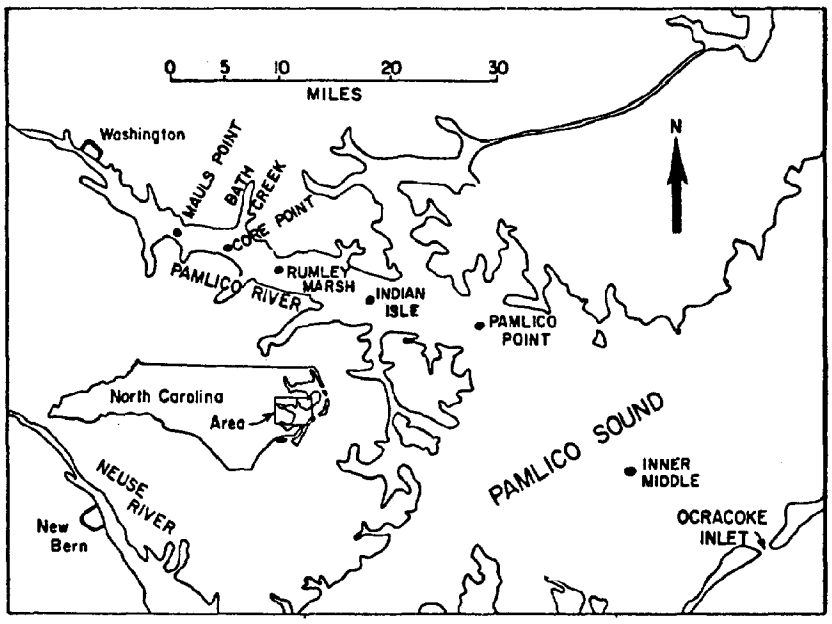


Table 1. Comparison Between Total Fluoride Concentrations and the Concentration Determined from Activity Measured with the Fluoride Electrode

Sample	Total M. F^- x 10^5	Free M. F^- x 10^5	Complexed M. F^- x 10^5	Percent F Complexed
Inner Middle (S)	3.92	1.63	2.29	58
Inner Middle (B)	2.60	1.27	1.33	51
Pamlico Point (S)	3.78	1.62	2.12	56
Pamlico Point (B)	3.84	1.62	2.12	55
Indian Isle (S)	3.10	1.56	1.54	50
Indian Isle (B)	2.16	1.22	.94	44
Rumley Marsh (S)	1.94	1.19	.75	39
Rumley Marsh (B)	3.09	1.42	1.67	54
Core Point (S)	1.56	1.03	.53	34
Core Point (B)	2.53	1.54	.99	39
Mauls Point (S)	1.29	.66	.63	49
Mauls Point (B)	1.86	1.14	.72	39

Table 2. Comparison of Fluoride Concentration Determined from In situ and Laboratory Electrode Measurements

Sample	In situ M. F^- Act. x 10^5	Laboratory M. F^- Act. x 10^5	In situ-Laboratory M. F^- Act. x 10^5	Percent Difference
Inner Middle (S)	1.68	1.14	.54	32
Inner Middle (B)	1.66	.89	.77	46
Pamlico Point (S)	1.69	1.15	.54	32
Pamlico Point (B)	1.59	1.15	.44	28
Indian Isle (S)	1.42	1.12	.30	21
Indian Isle (B)	1.44	.88	.56	39
Rumley Marsh (S)	1.56	.87	.59	38
Rumley Marsh (B)	1.27	1.02	.25	20
Core Point (S)	1.56	.76	.80	51
Core Point (B)	1.20	1.14	.06	5
Mauls Point (S)	1.23	.50	.73	59
Mauls Point (B)	1.01	.84	.17	17

Fig. 15. Fluorides in Pamlico River, North Carolina (Daugherty and Johnson, 1968).

ACID WATERS

When marine waters become very acid, ordinary marine organisms cannot develop ecological systems and are replaced by simple ecosystems with only a few species. The salts in sea water include 125 ppm carbonates and some borates which maintain the solution's acid-base balance in estuarine waters mainly between 7.5 and 8.2 on the pH scale (7 is neutral. A decrease of 1 means a ten fold increase in acid ions). The fluids within animals have similar composition and maintain their own necessary homeostasis in a similar way. Fig. 1 shows the pH range where Calabrese and Davis (1966) found continued development of eggs and larvae of oysters and clams. Because plants must incorporate carbon dioxide in their photosynthetic process, because animals must get rid of the excess carbon dioxide from their respiration, and for other reasons the acid-base condition in the external waters and in their tissues must be held within a small range if their functions are to operate without stress. If acid is added to sea waters in large quantities, the buffering salts are neutralized, the pH values drop, driving out the carbon dioxide in well stirred waters. Below pH 5 there is little carbon dioxide for photosynthesis. The external acid conditions put a stress on other organisms to hold their interior acid-base equilibrium. It has been observed in studies of acid waters, mainly in fresh waters so far, that very characteristic populations replace the ordinary ones, the new organisms apparently adapted as specialists capable of developing an ecological system in the acid conditions. Since many wastes have acid content, the acid water ecosystem occurs with varying degrees of complication due to other wastes.

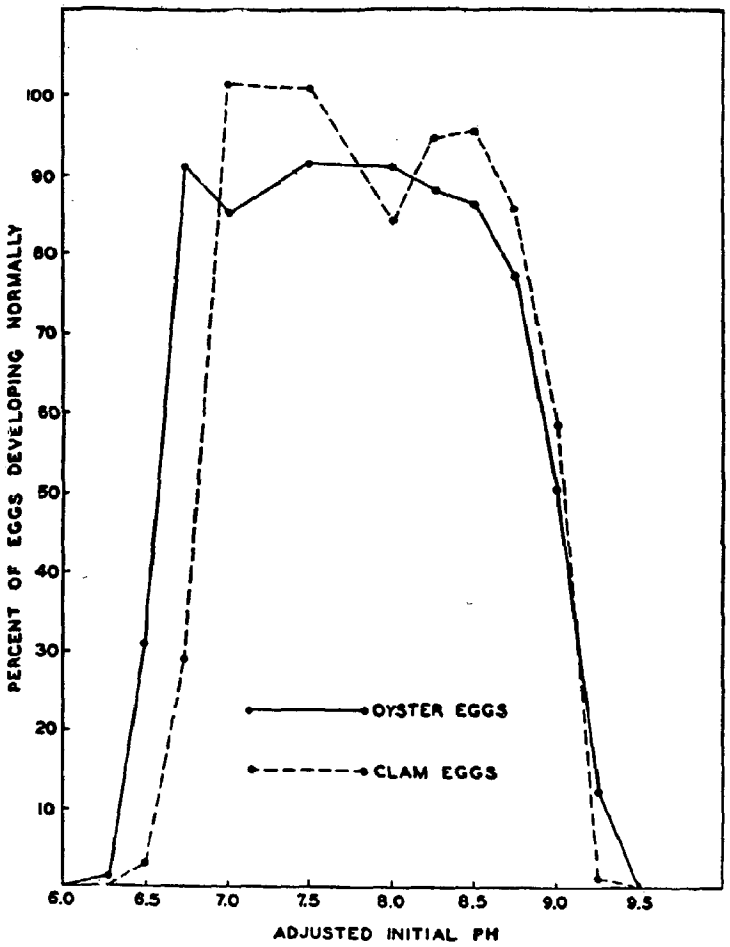
EXAMPLES

Acid Marine Pond from Oxidized Marsh Mud

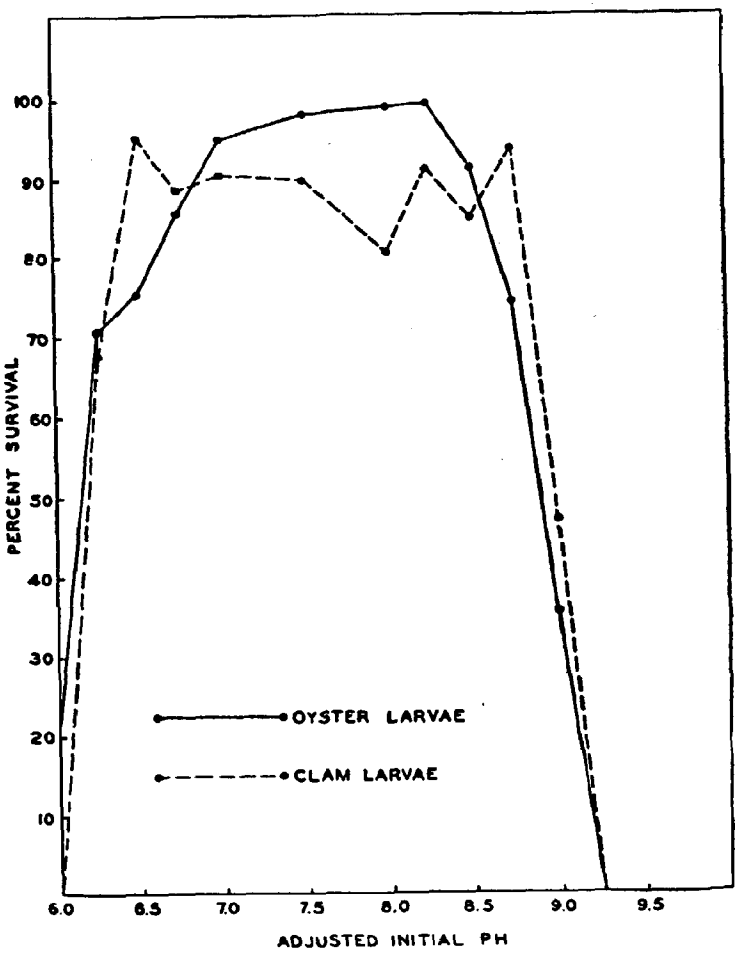
An example of an acid water system developed in some ponds newly dug and lined with marsh mud at Morehead City, N. C. (Fig. 2). Oxidation of sulfides occurred while the mud was exposed to air, developing sulfuric acid which dropped the sea water to pH 3.6 when the pond was refilled. A characteristic phytoplankton developed as compared with control ponds. Many normal organisms which had been added such as grass shrimp dropped out. Photosynthesis as measured with diurnal oxygen methods was much less than in control ponds, or in the same ponds after they were neutralized later by pumping in new waters.

Adams (1962) dried and aerated sulfide-containing marsh muds allowing oxidation. After rewetting, very low pH's resulted as given in Table 1. Wherry (1920), Edelman and Staveren (1958), and Fleming and Alexander (1961) found acid soil important in marshes and a factor in the determination of communities at the margins of marshes.

The acid condition in the two new ponds at Morehead City were studied by several staff of the University of North Carolina. J. D. Johnson and J. Day repeated the drying experiment done by Adams with similar low pH resulting (Progress report to North Carolina Board of Science and Technology, Dec. 31, 1968). They titrated the acid waters finding some residual buffer capacity even though carbonates had been eliminated.



Percentage of clam and oyster eggs that developed into normal straight-hinge larvae at different pH levels, expressed as a percentage of the number developing into normal larvae in control cultures.



Percentage of clam and oyster larvae that survived at different pH levels, as a percentage of survival in control cultures.

Fig. 1. Survival of marine organisms as a function of pH (Calabrese and Davis, 1966).

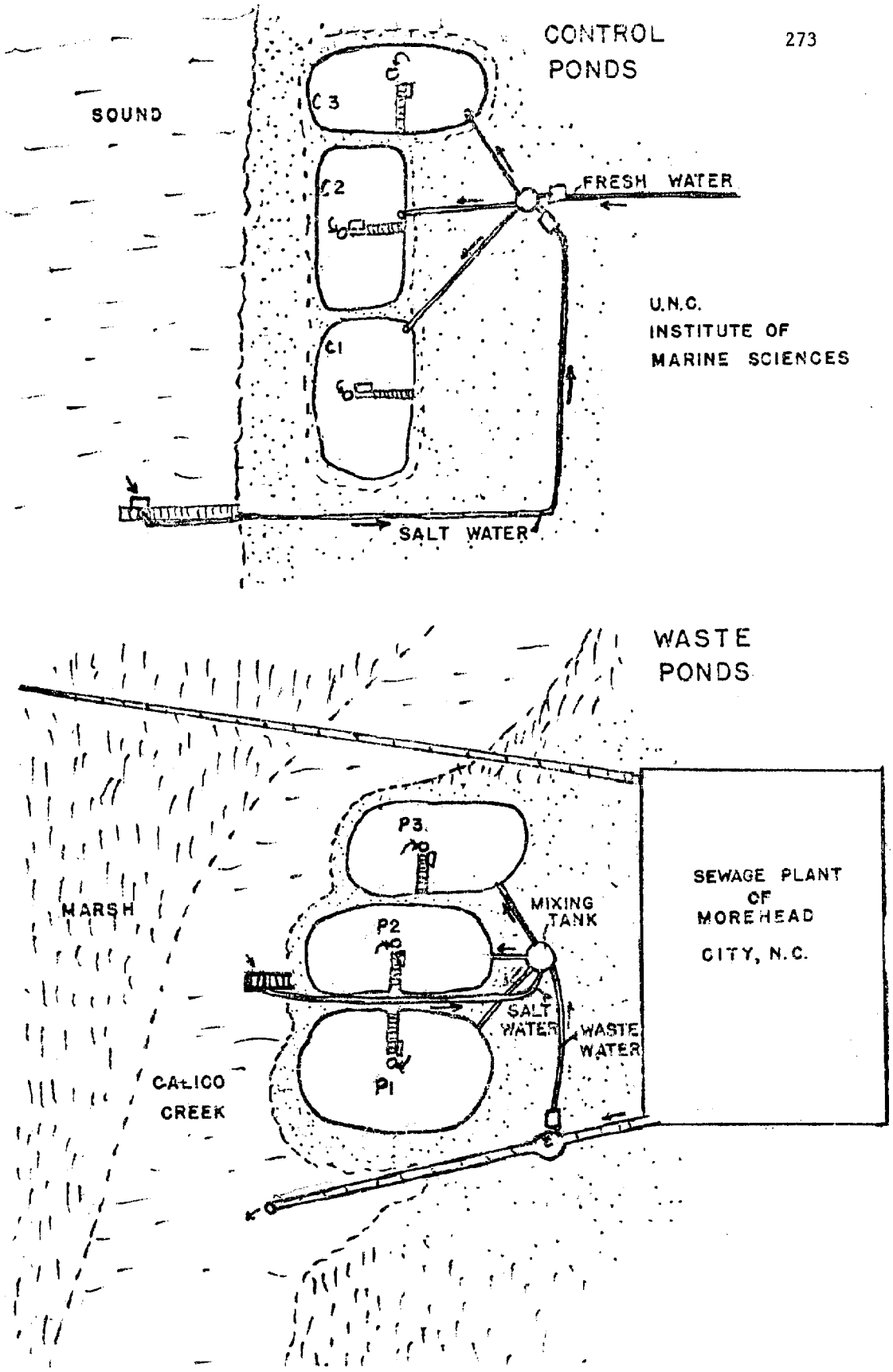


Fig. 2. Ponds at Morehead City, N. C. C-2 and C-3 became acid after filling.

Table 1

Soil pH (wet) versus soil pH (dried and rewet) at Oak Island and Southport, N.C. (from Adams, 1962).

	Oak Island	Southport
n[a]	45	46
Mean pH_{wet} [b]	5.75	7.62
Mean pH_{rewet} [c]	4.19	7.62
Mean pH change	1.56	0.48
pH greater than:	**Percent of Samples**	
1.0	33	6
2.0	28	2
3.0	7	4
Total % with pH change greater than 1.0	68	12

[a]The symbol (n) indicates the number of samples used in computing the respective means.
[b]pH of the original samples (1:1 soil/water slurry).
[c]pH of the rewet air-dry samples (1:2 soil/water slurry).

Table 2.

—Distribution of recognized genera and species of plants and animals occurring at or below pH 3.9

Plants	Number of species	Animals	Number of species
Thallophyta:		Protozoa:	
Fungi	2	Mastigophora:	
Algae:		Euglenidae	7
Myxophyceae	3	Protomastigina	7
Chrysophyceae:		Sarcodina:	
Chrysomonadales	3	Rhizopoda	15
Chrysotrichales	1	Heliozoa	2
Bacillarieae:		Infusoria:	
Pennales	5	Ciliata	19
Chlorophyceae:		Trochelminthes:	
Volvocales	1	Rotatoria	6
Ulotrichales	2	Gastrotricha	1
Chlorococcales	1	Nemathelminthes:	
Zynematales	6	Nematoda	1
Dinophyceae	2	Arthropoda:	
Bryophyta	1	Crustacea:	
Pteridophyta	1	Isopoda	1
Spermatophyta	1	Copepoda	1
Lackey (1967a)		Arachnida:	
		Tardigrada	1
		Insecta	8
		Amphibia	1

E. J. Kuenzler and L. D. Davidson describe phytoplankton bloom of Ochromonas (Fig. 3) and a low diversity of other species whereas the non-acid pond started at the same time had 17 species. Moore and Odum provide a diurnal curve of productivity in the acid pond (Fig. 4) showing a productivity similar to that in the non-acid ponds at the same time even though the acid ponds were very low in carbon dioxide concentrations. A. Williams and R. Outen found high mortality in marine invertebrate larvae in the acid condition.

Marine Waters Receiving Strong Acid Industrial Waste

Strong acids are frequently disposed at sea, often from barges far from land where large volumes of water dilute and neutralize the acid against the bicarbonates. Whenever these wastes are released into restricted and poorly mixing waters, the bicarbonates may be driven out with the result described for the marsh mud oxidation case. A possible example is a large effluent volume of 880 tons per day sulfuric acid proposed for Helgoland on the North Sea in Europe. Kinne and Rosenthal (1967) studied the effect of the acid on the eggs of herring, finding them sensitive in their early stages of fertilization, start of development and development of locomotion with negative effects down to 6 ppm. Where the acid is sulfuric, the sulfate added may be small relative to sulfate already in sea water. For example, only 300 ppm sulfate goes with a pH of 3. Thus effects may be due to the hydrogen ion (acidity) rather than the anion (sulfate). In the impure waste studied for effect on herring of high concentrations, however, iron was present which precipitated as pH was raised by dilution in sea water. Precipitation of iron has been used in laboratory work for removal of living micro-organisms and their normal organic metabolites, which become incorporated in precipitates. As seen in Table 3 for coal wastes, soluble metals tend to accompany the acids. Ketchum (1960) shows acids released from a barge in open sea waters neutralized in 15 minutes (Fig. 5).

Fresh Water Mine Drainages

In instances of stress, the differences between fresh and salt water ecosystems tend to disappear, especially if the population pressures of the sea are removed as in partly isolated lagoons. The patterns of life in the better known acid mine waters were similar to the pond situation observed at Morehead City. Lackey (1967a) provides a list (Table 2) of characteristic organisms of acid mine waste waters. At or below pH 3.9 few species were found, only 11 at pH 2.6. Characteristic species of the plankton are shown in Fig. 3 from Bonnd (1965b) and Lackey (1967a). See patterns in Fig. 6 and 7.

The acid water system is one of many kinds of disturbance ecosystems that are involved with extensive mining where excavations are in reduced or anaerobic sediments. There have been recent discussions of mining below the marshes of the Atlantic coast where acid pool systems are likely to be a result. Proposed copper mining in Puerto Rico and elsewhere may increase the frequency of acid mine drainages reaching the sea. Table 3, from Woodley and Moore (1967) shows some properties of mine drainages from reduced sediment.

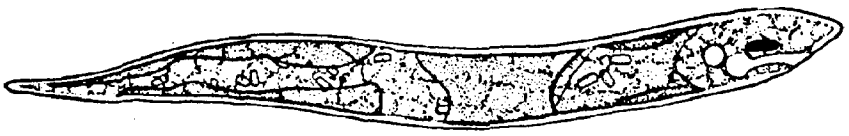
Table 3. Mine drainage wastes

Quality of Drainage from Abandoned Coal Refuse Disposal Site

Quantity Measured	Value on Given Date				
	12/12/62	11/15/63	5/20/64	6/10/64	11/25/64
pH	3.4	3.6	3.1	2.8	3.6
M.O. acidity (mg/l)	1,250	7,200	284	1,660	1,082
P. acidity (mg/l)	12,600	7,900	2,450	10,200	9,400
Sulfates (mg/l)	17,000	7,900	19,000	30,000	12,000
Iron (mg/l)	4,240	3,460	760	3,400	3,500
Manganese (mg/l)	40	—	50	50	40

(Woodley and Moore, 1967)

AQUATIC LIFE IN ACID WATERS

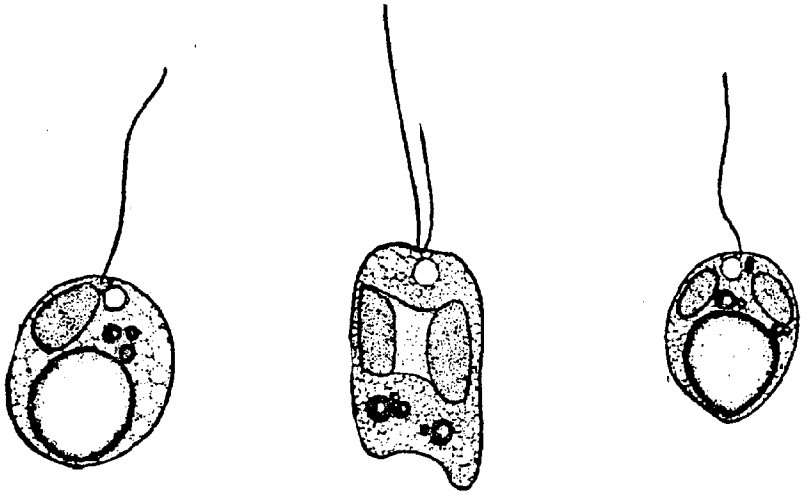


Euglena mutabilis, showing two or three heavy chloroplastids, conspicuous stigma, small rod-like paramylum bodies, and apparent absence of flagellum.

like paramylum bodies, and apparent absence of flagellum.

Ochromonas

(Round, 1965b)



Figures 3 and 4 - Chromulina sp., showing one or two chromatophores, stigma, and large posterior granula.

Fig. 3. Aquatic Life in waters polluted by acid mine waste (Lackey, 1967a).

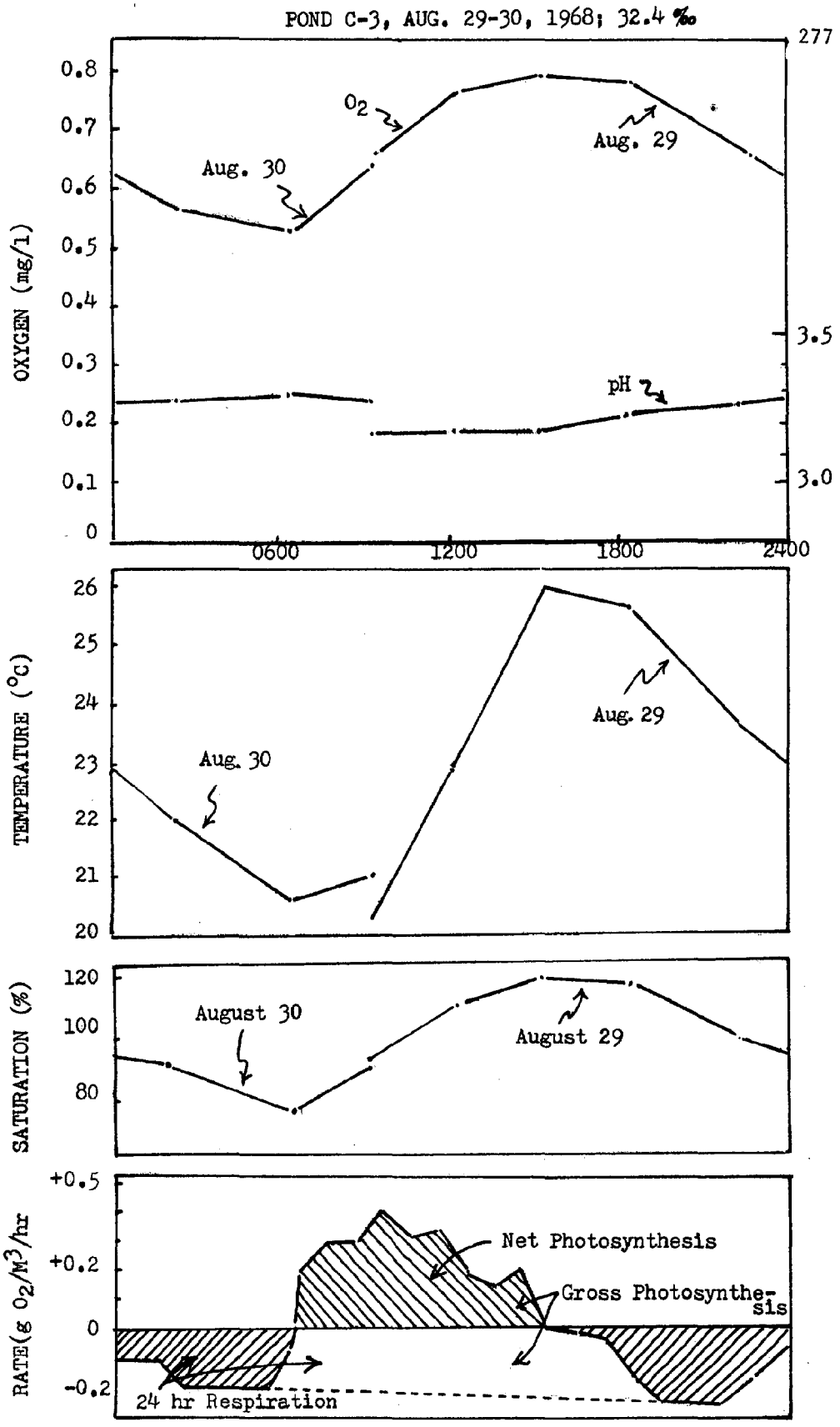
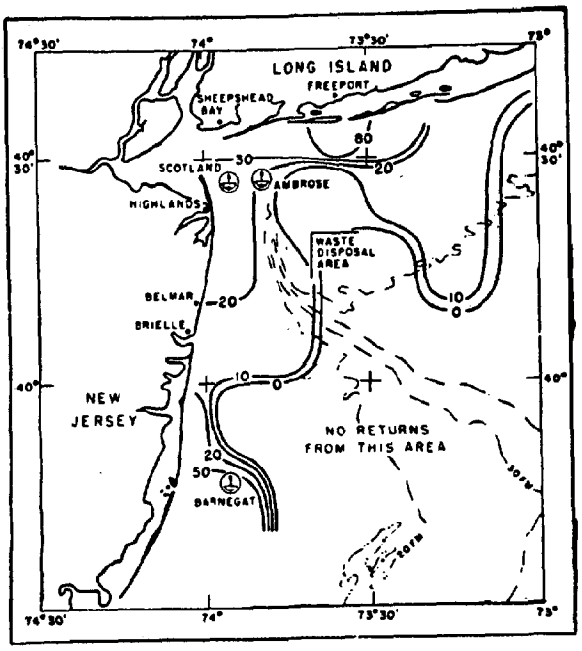
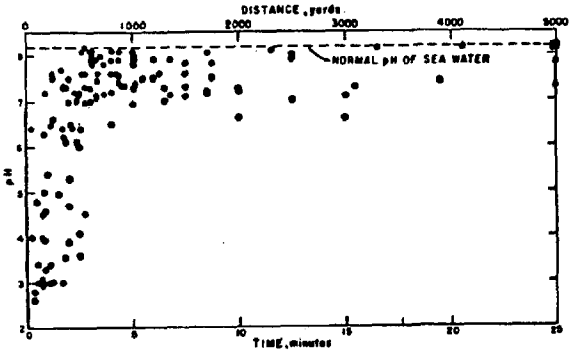


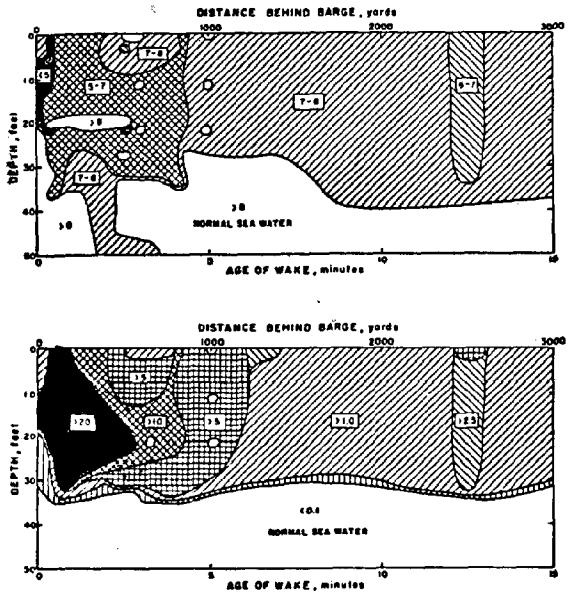
Fig. 4. Diurnal record of oxygen, pH, and photosynthesis in an acid marine pond, August 29-30, 1968. (A. Moore and H.T. Odum).



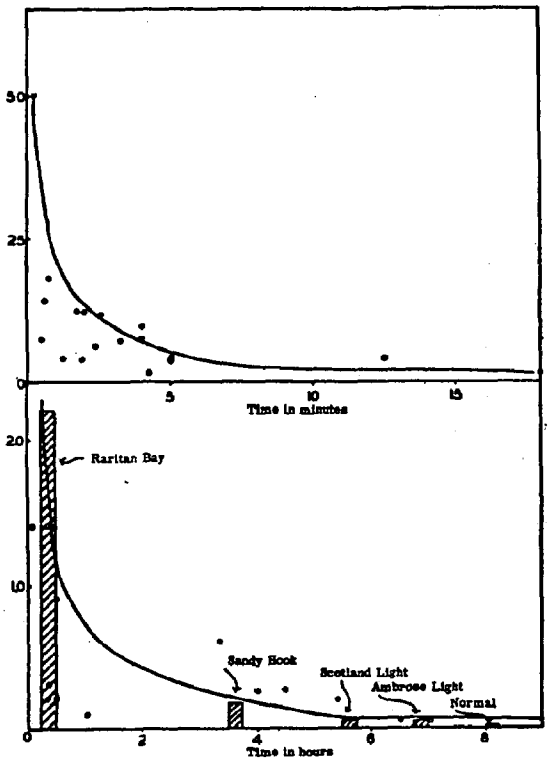
The New York Bight showing the location of an industrial waste disposal area and contours of the percentage return of drift bottles to the coasts of Long Island and New Jersey (Redfield and Walford, 1951).



The surface distribution of pH in the wake of a barge discharging a waste containing ferrous sulphate and sulphuric acid (Redfield and Walford, 1951).

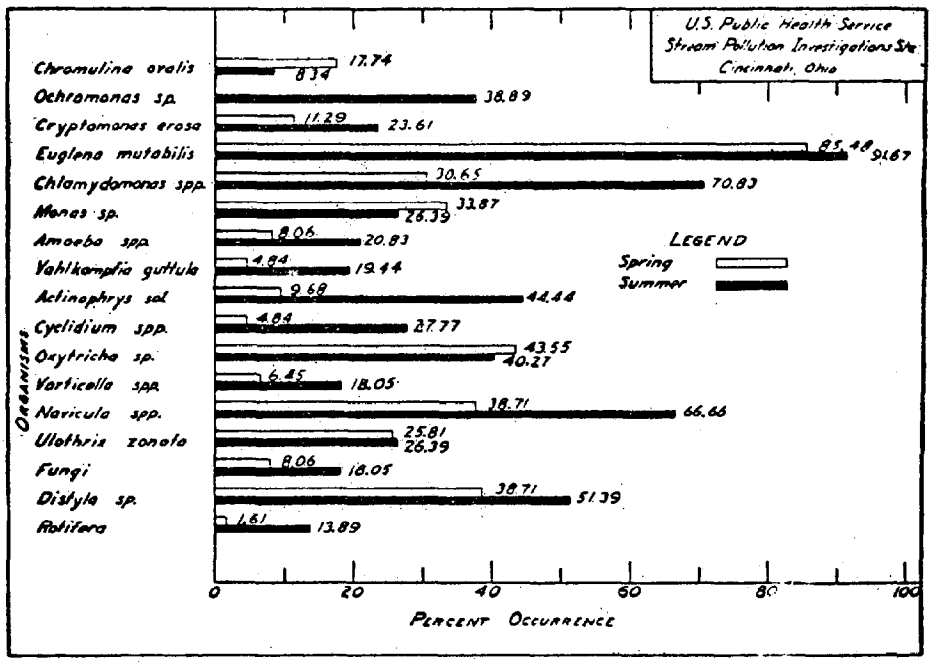


The vertical distribution of iron (lower) and pH (upper) graph in the wake of the barge (Redfield and Walford, 1951).

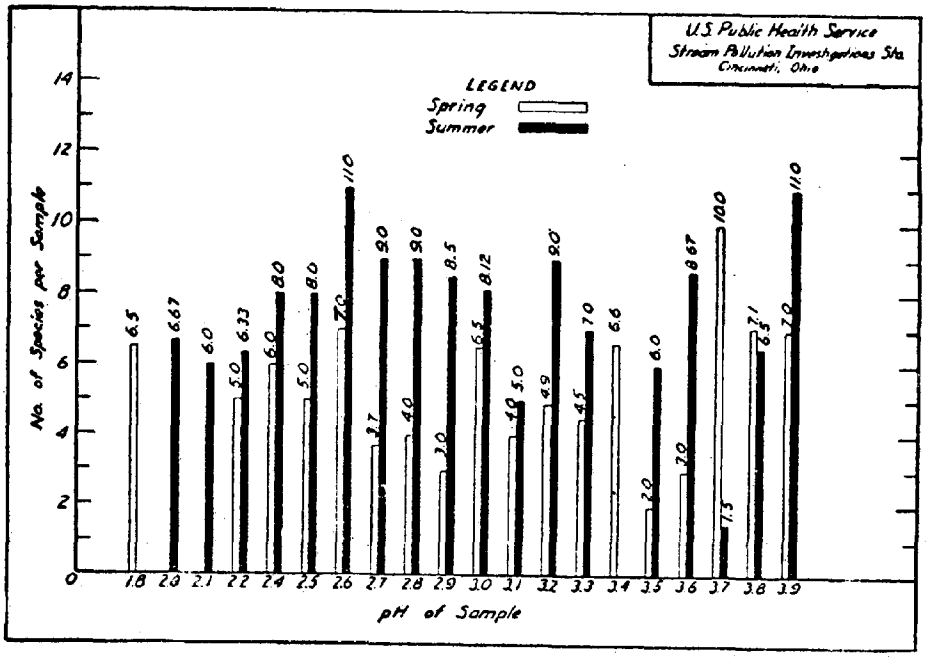


The surface concentration of iron in the wake of the barge at various times (upper portion). Local concentrations in various parts of the New York Bight at some distance from the disposal area are shown by the bars in the lower figure.

Fig. 5. Distributions of pH and iron in the wake of a barge discharging acid wastes (From Ketchum, 1960).



Percentage of occurrence of the 17 most common organisms in all samples.



Average number of species per sample within the pH range 1.8 to 3.9.

Figs. 6 and 7. Organisms in acid mine waters (Lackey, 1967a).

Chapter E-11

OIL SHORES

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INTRODUCTION

When the inflow of oil to a coastal area is large, special new ecological systems begin to develop. Whereas the oil serves as a stress and eliminates many of the larger organisms, it provides an energy source for specialized micro-organisms capable of metabolizing the oil. The interaction of oil and sediment produces characteristic substrates. Although few situations with steady, regular input of oil have been studied, there are a number of case histories of oil spill accidents which have covered relatively long periods, and these suggest the nature of the ecosystems that develop under oil stress.

Previous Reviews

Recent summaries and bibliographies relating to problems of marine oil spills include those by Hawkes (1961), Moss (1963), ZoBell (1964), the Battelle Memorial Institute (1967), Yee (1967), Smith (1968), and Carthy and Arthur (1968).

The most comprehensive general review of literature relating to oil spillage is the Battelle report (1967). It considers not only methods of prevention, control, and restoration, but also summarizes results of field and laboratory studies up to 1967 of biological and ecological effects of oil. The material is organized into sections, each with an extensive bibliography.

The Smith book (1968) is a detailed report by the Plymouth Laboratory of Marine Biological Association of the United Kingdom on the effects of oil released by the wreck of the Torrey Canyon and of detergents used in the mopping up operations. It discusses properties of the oil and detergents, but is devoted mainly to a discussion of their effects on the marine life of British shores.

ZoBell (1964) considers the occurrence, sources, and effects of oil in the sea, as well as the natural means by which it is eliminated: evaporation, dispersion, microbiological oxidation, settling, weathering, and so on. Hawkes (1961) reviews problems associated with oil pollution, especially its effects on marine life. Moss (1963) deals mainly with the complexities and possible solutions to sea pollution by oil as these relate to tanker traffic, and Yee's paper (1967) consists of a bibliography of the literature (1950-1967) on oil pollution in marine waters, arranged by such topics as Sources, Prevention, Detection, Treatment, and Effects on Marine Life. Carthy and Arthur (1968) have edited the proceedings of a symposium held at Pembroke, Wales on oil effects on littoral communities.

Sources of Spills

The use of petroleum and its products as a power source and for petrochemicals has increased steadily during the last century. The consequent increase in amounts used and transported - both crude oil and refined petroleum products - is reflected in the rise in frequency of oil spills. Reports of oil release into marine waters come from every coastal state in the Union. For example, 75 incidents of oil pollution were reported in Cook Inlet, Alaska between June 1966 and December 1967 (Alaska Water Laboratory, 1968). Spills result from accidents in tanker traffic, leaks during pumping and piping --such as the recent (February 1969) oil leak off Santa Barbara, California, cleaning of tanks, pumping of bilge water, and disposal of petroleum wastes from industrial processes (Moss 1963; ZoBell 1964; Maehler and Greenberg 1967; Pipel 1968; US Departments of Interior and Transportation 1968). Some oil is also released through natural submarine seeps off California, Louisiana, and Texas (Rosen et al. 1960, ZoBell 1964).

EXAMPLES OF OIL DISTURBANCE

Laboratory Studies

A relatively large number of laboratory studies have been carried out testing the effect of various concentrations of oil (or of oil-cleansing detergent) on the survival, growth, reproduction, ciliary action, etc. of various marine organisms. (See Battelle Report, 1967, for bibliographies).

Tests by Galtsoff et al.(1935) gave evidence of relatively low toxicity of crude petroleum oil. This was in line with earlier studies by Elmhirst (1922) and Orton (1925). Lund (1957) and Mackin and Hopkins (1962) found no significant differences in ciliary action or mortality between oysters exposed to oil and control oysters, under the conditions of their experiments. Aljarinskaya(1966) reported that Mytilus galloprovincialis was able to stand relatively high concentrations of oil due to the resistance of the ciliated epithelium of their gills and mantle. Tagatz (1961a showed that resistance of American shad to petroleum products varied with the product (crude oil being least and gasoline most toxic) and with dissolved oxygen.

A large number of studies of effects of detergents on marine organisms have been carried out. Hidu (1965) studied effects of surfactants on survival and growth of larvae of clams (Mercenaria mercenaria) and oysters (Crassotrea virginica). Effect varied with kind of surfactant (cationic's being more toxic than anionics and nonionics). Clam larvae were less sensitive than oyster larvae. Corner et al.(1968), using the barnacle Elminius modestus, tested four kinds of detergent for toxicity. All were more toxic than crude oil, and effects on the barnacle varied with stage of development. Bardach et al.(1965) found that very low concentrations of detergent (0.5 ppm) affected sensory reception, swimming, and feeding behavior in lake fishes (Ictalurus natalis), and Eisler (1965) showed that

the greater the salinity of the medium, the more toxic the effect of detergent on juvenile estuarine fishes (Menidia menidia, Fundulus heteroclitus, Mugil cephalus, and others).

Studies of effects of detergents on isolated tissues from marine organisms have been made also. For example, Manwell and Baker (1967b) obtained results indicating that detergents cause activation of certain enzymes, the inhibition of others and increase in the extractability of proteins from cells (See also the Battelle bibliography).

Coastal Ecosystems

Effects of oil disturbance on coastal ecosystems vary with ecosystem, amounts and kinds of oil released, tidal currents, wind patterns, and other factors. Most of the ecosystem types as defined in these chapters have been subjected to varying degrees of oil disturbance at one time or another, but because of the tendency of oil to float and to wash ashore, its effects on beaches and rocky shores are most obvious and most frequently recorded. Effects on mangrove swamps, marshes, and plankton systems have been studied to a lesser degree.

Specific effects of oil on interrelationships of the components of an affected ecosystem depend on latitude, which determines the characteristic organisms present, but similarities in effect can be seen from case studies made at latitudes as diverse as those of Moclips, Washington; Cornwall, England; Guanica, Puerto Rico; and Baja, Mexico.

In most cases in which extensive studies of effects of oil on an ecosystem have been made, the disturbance (usually resulting from a tanker accident) has been relatively short lived. Studies on long term effects on an ecosystem when oil is a dominant factor are very few.

Documented cases of oil disturbance on natural coastal ecosystems include the following: a) a wild oil well in Louisiana in 1956; b) the Tampico Maru oil spill at Baja, California - Mexico in 1957; c) the Guanica oil spill off the south coast of Puerto Rico in 1962; d) the oil spill at the Quinault Indian Reservation in 1964 off Moclips, Washington; e) the Torrey Canyon accident in 1967 off Cornwall, England; and others.

The kind of oil released in most of these cases was crude oil, but in cases (b) and (d) diesel fuel was the disturbing factor. After the Torrey Canyon wreck, detergents were used in cleaning operations, and their effects were also considered in evaluating oil disturbance effects.

Case History 1: Wild Oil Well in Louisiana (Mackin and Sparks, 1962)

On November 17, 1956 an oil well in an oyster-producing area around Lake Grande Ecaille went out of control, caught fire, and spilled crude oil

for two weeks. Presumably all the light fractions burned or evaporated. The exact amount of oil that escaped was impossible to determine, but in regions of greatest concentration it appeared as a heavy coating on the water surface, and shorelines nearest the well were entirely saturated.

In an attempt to determine whether or not such oil might affect survival, growth, or setting of oysters under natural conditions, investigators established a series of 7 stations (4 experimental, 3 control) at varying distances from the well site. Six of them were stocked with trays containing oysters dredged from an area which had been exposed to the oil for more than two months. One of the control stations was stocked from an unaffected area.

Measurements made over the next two years included oyster mortality and condition, surveys of the macroscopic fauna of the trays (in addition to oysters), and analyses of the effects of the oyster disease produced by the fungus Dermocystidium. The four experimental stations were within the oil-affected area, while the three control stations were outside it.

Results showed that in none of the measurements made did the experimental and control communities differ significantly. The investigators concluded that modifications in the environment brought about by the crude oil were not reflected in any change in the normal oyster reef community. Normal survival, reproduction, setting, and growth occurred throughout the study period. An hypothesis that oil might in some way be prerequisite to development of infection with Dermocystidium was disproved.

Case History 2: The Tampico Oil Spill at Baja, California-Mexico (North, Neushul, and Clendenning 1965)

On March 29, 1957 the tanker "SS Tampico Maru" was wrecked in a remote area on the Pacific coast of Baja about 180 kilometers south of the Mexican-California border. The ship blocked about 3/4 of the entrance to a small cove (roughly 100 x 100m) and acted for several months as a breakwater for an environment that normally was subjected to violent water movements. The shore is one that is heavily pounded by storms during late fall, winter, and spring. Extensive upswelling occurs, and water temperatures are typically low (12-16°C).

About one-third of the cargo of 50,000 barrels of dark diesel oil was lost at the time of stranding. The rest was liberated sporadically over the next 8 months. By the end of 1957 the ship had broken up, depositing strewn wreckage on the cove's bottom and exposing the cove once again to the full force of the surf. Several factors, therefore, were directly involved in modifying the ecology of the cove: oil spillage, change in wave exposure, and bottom modifications. The investigators believed, however, that oil effects were a major factor in destruction of organisms in the area, both by killing them directly and by forcing them to release their attachments to the substratum and be washed ashore.

Because of the cove's remoteness, its biota had not been surveyed in detail at the time of the accident, but a few pertinent observations had been

recorded. Systematic surveys were begun about a month after the wreck and included not only the cove itself but also adjacent (presumably unaffected) areas.

The cove was judged to be typical originally of the region in having luxuriant attached vegetation, the dominant algae being Halidrys dioica, Porphyra perforata, and Lithothamnion. Other algae were doubtless also present. A small clump of giant kelp (Macrocystis pyrifera) was noted in the center of the inlet shortly after the accident, a rather surprising finding for such shallow water. (See Chapter C-11 for further discussion of kelp effects).

The normally common animals of the cove were believed to include three kinds of abalone, two kinds of sea urchins, herbivorous fishes, tunicates, scallops, mussels, barnacles, cowries, sea stars, lobsters, anemones, periwinkles, crabs, polychaetes, sea cucumbers, and other small molluscs and fishes.

The first survey was made in April 1957, when oil deposits were evident on beaches, in tide pools and on the water surface. A month later the amount of oil was much reduced, and two weeks after that only small amounts were noted on the beaches. Further spillage from newly-ruptured tanks apparently occurred in November, but on subsequent visits the investigators noted no unusual concentrations of oil. Intertidal organisms were surveyed in a 40 m section of the northern part of the cove, and sub-tidal organisms were surveyed along a 100 m transect line stretching from the middle of the cove shore line out to the center of the inlet.

At first there was a complete disappearance of all but a few animals, a period lasting through the spring of 1957. Then forms capable of migration (e.g. fishes, certain crustaceans) began appearing in the cove, and finally more sessile organisms were represented as recolonization increased. An idea of faunal changes between 1957, the year of the wreck, and 1961 can be gained from Tables 1 and 2, and floral changes from Tables 3 and 4. The species are grouped as intertidal or subtidal and are arranged by the investigators according to the year in which they were first observed alive and apparently healthy. This arrangement puts them in a sequence which is probably "indicative of the severity of the catastrophe upon the ecology of each species."

The most resistant of the animals appeared to be the anemone, Anthopleura xanthogrammica, and the periwinkle, Littorina planaxis, both of which escaped destruction.

A striking change in the cove's vegetation, which in turn influenced greatly the other biota, was a great accumulation of plant growth, chiefly the giant kelp, Macrocystis. Tremendous numbers of juvenile plants appeared whose age (as gauged by blade number) showed that their germination time coincided closely with the date of the accident and implied a correlation. A massive kelp bed eventually developed in the cove (Figure 1), associated mainly with the strewn bottom wreckage, and between 1957 and 1964 passed through at least two complete succession cycles. These influenced profoundly the associated plants and animals in ways not directly related to the ship wreck. Although

Table 1: Common animals observed over a seven year period in the intertidal zone at the site of the Tampico Maru wreck.

Species	Date of Survey														Years first Observed Alive
	9/ 9/ 56	4/ 26/ 57	5/ 26/ 57	6/ 10/ 57	7/ 10/ 57	9/ 19/ 57	5/ 1/ 58	7/ 25/ 58	10/ 10/ 59	10/ 15/ 60	11/ 4/ 61	4/ 28/ 62	1/ 1/ 63	11/ 29/ 63	
Anthopleura elegantissima and/or *A. xanthogrammica*		P	P	P	P	P	P	P	P	P	N	N	N	N	1957
Girella nigricans					P	P	P		P	N	P	N		N	
Littorina planaxis		N	N	N	N	P	P	N	N	N	N	N	N	N	
Mytilus californianus		D	D	D	P,D	P,D	P	P	N		N	P		P	
Pachygrapsus crassipes					P	N	P	P	N	P	N	N		N	
Chthamalus fissus		D				P	P	P	N	N	N	N	N	N	1958
Littorina scutulata								P	P	P	P	P		P	
Lygia occidentalis								P	P						
Acanthina spirata									P			P			1959
Acmaea pelta and/or *scabra*									P	P	N	N		N	
Balanus glandula			D						P		P	P		N	
Haliotis cracherodii	P	D	D						P		P	P			
Mitella polymerus			D						P		P	P		P	
Pisaster ochraceus		D	D						P		P	P		P	
Strongylocentrotus franciscanus		D	D						P			P			
Strongylocentrotus purpuratus		D	D						P		N	P		N	
Tegula funebralis									P	N	P	N		P	
Tetraclita squamosa		D	D						P		P	P		N	
Chama pellucida										P					1960
Ischnochiton sp.										P	P	P		P	
Pagarus spp.										P				P	
Clinocottus analis											P	P			1961
Fissurella volcano											P	P		N	
Lottea gigantea											P	P		N	
Mopalia muscosa											P	P		P	
Thais emarginata											P	P		P	

P, present; N, concentration judged to be about normal for the region; D, dead or moribund specimens found. (From North, Neushul, and Clendenning 1965, Table 1.)

Table 2: Subtidal animals observed over a seven year period at the Tampico Maru wreck site.

Species	Date of Survey 9/ 9/ 56	4/ 26/ 57	6/ 10/ 57	6/ 29/ 57	7/ 10/ 57	9/ 19/ 57	11/ 12/ 57	5/ 1/ 58	10/ 10/ 59	10/ 15/ 60	11/ 5/ 61	4/ 28/ 62	1/ 1/ 63	11/ 29/ 63	Year first Observed Alive
Girella nigricans	P			P		P					P	P		P	1956
Haliotis fulgens	P	D													
Haliotis rufescens	P	D													
Strongylocentrotus franciscanus	P	D							P					P	
Strongylocentrotus purpuratus	P	D	P						P			P	P		
Anisotremus davidsonii					P	P		P		P	P		P	P	1957
Anthopleura artemesia			P								P	P	P	P	
Anthopleura xanthogrammica			P			P	P						P	P	
Aplysia californica					P	P	P					P			
Atherinops affinis						P				P	P				
Balanus nubilus			D												
Balanus tintinabulum			D												
Cancer productus		D	P.D												
Citharichthys stigmaeus						P									
Corynactis californica			P											P	
Embiotoca jacksonii				P	P	P				P	P	P		P	
Harenactis attenuata						P						P			
Hermissenda crassicornis			P									P			
Hyperprosopon argentum				P		P					P				
Membranipora membranacea					P			P			P				
Oxyjulis californica			P			P		P			P	P		P	
Panulirus interruptus		D	D		P					P	P	P	P	P	
Pateria miniata						P					P	P		P	
Pododesmus macrochisma			D												
Pimelometopon pulchrum					P	P					P				
Pisaster giganteus					P							P		P	
Sebastodes sp.			P								P		P	P	
Seriphus politus						P									1957
Zonaria spadicea			D												
Sponges (unidentified)			P								P			P	
Strongylura exilis							P								
Styela montereyensis			P				P				P	P		P	
Urobatis halleri					P										
Zalophus californicus			P			P			P	P	P				
Cyanosciphon nobilis									P						1959
Aletes squamigerous											P				1961
Brachyistius frenatus											P				
Chaetopterus variopedatus											P				
Diopatra sp.											P				
Hypsurus caryi											P				
Mitrella carinata											P				
Phanerodon furcatus											P				
Serpula vermicularis											P	P	P	P	
Stichopus californicus											P	P			
Taeniotoca lateralis											P				
Norrisia norrisii												P		P	1962
Unidentified chiton												P			
Megathura crenulata														P	1963
Porichthys ..otatus													P		
Scorpaenichthys marmoratus		D												P	

P, present; D, dead or moribund. (From North, Neushul and Clendenning, 1965; Table 2.)

Table 3: Intertidal plants observed over a seven year period at the Tampico Maru wreck site.

Species	Date of Survey 9/ 9/ 56	4/ 26/ 57	5/ 26/ 57	6/ 10/ 57	6/ 29/ 57	7/ 11/ 57	9/ 19/ 57	11/ 12/ 57	5/ 2/ 58	7/ 25/ 58	10/ 10/ 59	10/ 15/ 60	11/ 4/ 61	4/ 28/ 62	11/ 29/ 62	Year first Observed
Porphyra perforata	P	P	P	P	P	P	P	P	P	P		P	P	P		1956
Bossiella dichotoma		D					P		P		P	P	P	P	P	
Corallina officinalis							P		P		P	P	P	P	P	1957
Ectocarpus granulosus							P							P		
Egregia sp.						P	P		P	P	P	P	P	P	P	
Gigartina canaliculata			P				P		P		P	P	P	P	P	
Halidrys dioica							P				P	P	P	P	P	
Iridea sp.			P			P			P	P	P		P			
Lithothamniom sp.		D					P		P		P	P	P	P	P	
Macrocystis pyrifera							P				P	P	P	P		
Phyllospadix torreyi		P			P				P		P		P	P	P	
Ulva lactuca		P		P	P	P	P	P	P	P	P	P	P	P	P	
Endocladia muricata									P							1958
Gigartina spinosa									P	P	P			P		
Nemalion lubricum									P							
Smithora naiadum									P							
Codium fragile											P		P			1959
Bryopsis hypnoides													P		P	1961
Calliarthron cheilisporioides													P	P	P	
Cladophora sp.													P			
Laurencia gardneri													P			
Rhodoglossum americanum													P			1961
Sargassum agardhianum													P			
Laminaria andersonii														P		1962
Leptocladia binghamiae															P	1963

P, present; D, dead or moribund. (From North, Neushul, and Clendenning, 1965; Table 3.)

Table 4: Subtidal plants observed over a seven year period at the site of the Tampico Maru wreck.

Species	Date of Survey 9/ 9/ 56	4/ 26/ 57	6/ 10/ 57	6/ 29/ 57	7/ 10/ 57	9/ 19/ 57	11/ 12/ 57	3/ 2/ 58	10/ 15/ 60	11/ 4/ 61	4/ 28/ 62	1/ 1/ 63	11/ 29/ 63	Date First Observed
Cystoseira osmundaceae	P		P		P	P	P	P	P	P	P	P	P	1956
Macrocystis pyrifera	P	P	P	P	P	P	P	P	P	P	P	P	P	
Pterosiphonia dendroidea	P							P						
Bossiella dichotoma					P	P			P	P	P	P	P	1957
Botryocladia pseudodichotoma					P									
Botryoglossum farlowianum					P	P		P		P		P	P	
Desmarestia latifrons					P	P								
Ectocarpus granulosus					P			P						
Egregia laevigata		P		P	P	P		P		P	P	P	P	
Gelidium cartilagineum						P		P					P	
Gigartina californica				P									P	
Gigartina leptorhynchos					P									
Gigartina serrata					P									
Gigartina spinosa			P			P	P	P	P	P	P			
Gigartina volans			P	P						P			P	P
Gracillaria cunninghami					P			P						
Gymnogongus leptophyllus					P									
Laminaria andersonii			P	P	P	P	P	P	P	P	P	P	P	P
Microcladia coulteri							P							
Nienburgia andersoniana					P			P						
Phycodrys setchellii					P									
Pikea pinnata					P									
Plocamium pacificum					P									
Polyneura latissima			P		P									
Polysiphonia mollis						P								
Prionitis cornea					P	P		P				P	P	
Prionitis linearis					P	P		P				P		
Ptilota californica					P			P						
Ralfsia pacifica					P								P	1957
Rhodoglossum americanum					P			P						
Rhodymenia pacifica			P		P			P				P	P	
Smithora naiadum					P			P						
Ulva sp.					P	P		P					P	
Calliarthron cheilisporioides					D			P		P	P	P	P	1958
Corallina officinalis									P	P	P		P	1960
Callophyllis violacea										P		P		1961
Cryptonemia obovata										P				
Leptocladia binghamiae										P	P	P	P	
Phyllophora californica										P				
Prionitis lanceolata										P		P		
Desmarestia tabacoides													P	1963
Farlowia crassa													P	
Lithothamnion giganteum					D								P	

P, present; D, dead or moribund. (From North, Neushul, and Clendenning, 1965; Table 4.)

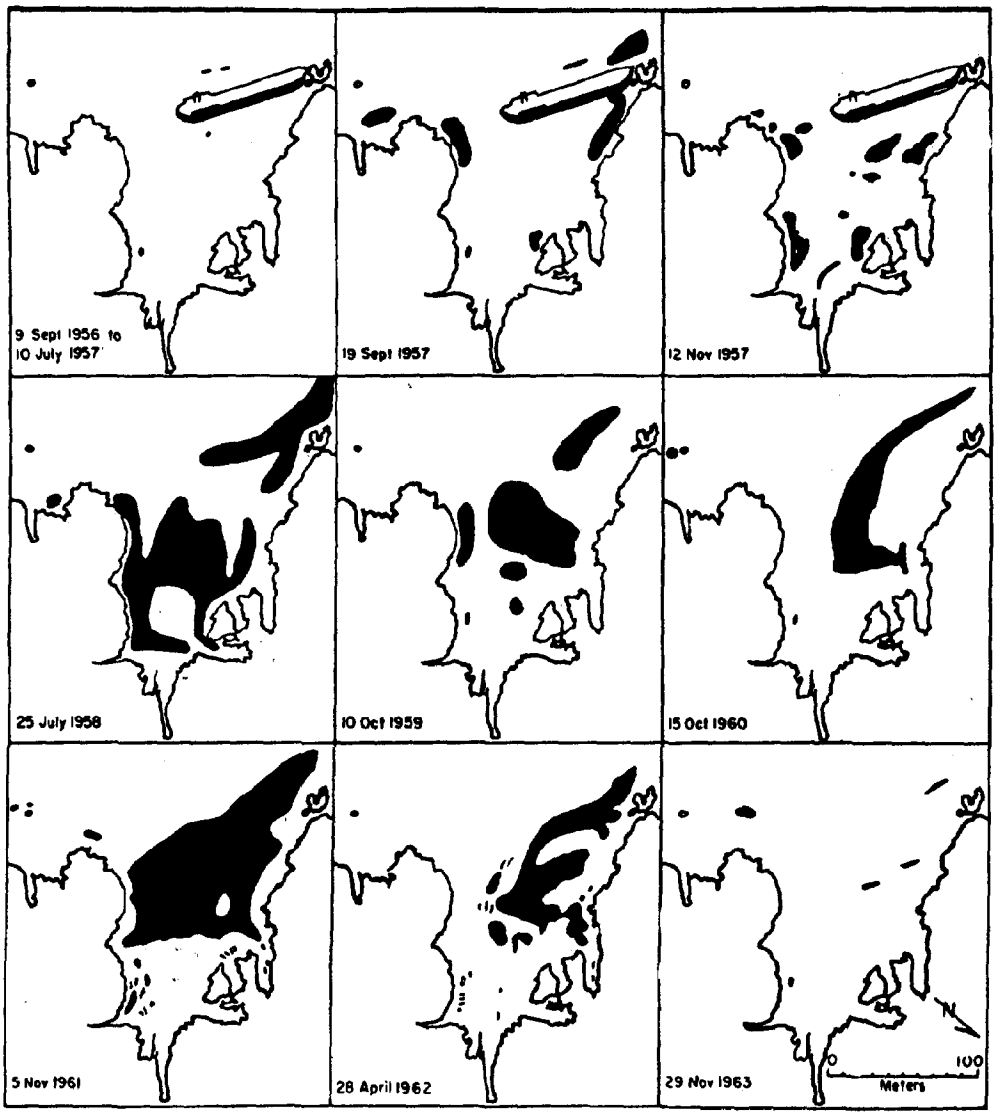

Fig. 1: Charts of the cove, where the Tampico Maru was wrecked, showing fluctuation of the giant kelp canopies (Macrocystis) during the period from July 1957 through November 1963 (From North, Neushul and Clendenning, 1965; Fig. 3).

many organisms had been re-established, the cove in 1961 was not the same as it had been prior to the wreck.

Case History 3: The Guanica Oil Spill, Puerto Rico (Diaz-Piferrer, 1962).

On July 16, 1962 a tanker ran aground on reefs off Guayanilla Harbor on the south coast of Puerto Rico. In refloating the ship, 10,000 tons of crude oil were pumped into the sea. It was blown and washed ashore as a thick blanket on beaches as far west as the bay at Boqueron and onto offshore coral reefs.

The shore area of Guanica, which was normally characterized by a particularly rich and varied marine flora, was first affected. It had been an algae-collecting station for an ecological study for a year prior to the accident, as well as during the years of 1958 and 1959. Extensive damage to the area was reported, including heavy erosion of the oil-soaked beaches, virtual destruction of heavily affected mangrove swamps, and a great mortality of marine organisms (lobsters, crabs, sea urchins, star fishes, sea cucumbers, gastropods, octopuses, squids, fishes, and sea turtles). Sea birds were reported to be absent from the sea after the accident. Plants living in intertidal and sublittoral zones were seriously affected, large rocky areas being completely denuded. Beds of the seaweed Thalassia were badly affected.

Monthly visits to the area were made to check on further changes and on recovery. In November 1962, oil-soaked sand was still evident under several feet of water and Thalassia beds continued to degenerate. From October on, algae began repopulating intertidal rocks and pools. Algal types, however, changed to consist almost entirely of blue-green algae (Myxophyceae).

Case History 4: Moclips Oil Spill at the Quinault Indian Reservation, Washington. (Heckman, 1964).

The stranding of a petroleum barge in heavy seas on the beach opposite Moclips, Washington on March 12, 1964 resulted in the release of considerably more than 500,000 gallons of oil over a period of 7 days. The released oil was reported to be mostly diesel fuel. The area is one in which razor clams are an important source of livelihood.

A measure of the biological effect of the oil spillage was obtained by counting dead and dying razor clams on 16 plots (each 25' x 25'), representing four different stations in the area on March 15 and 16. The plots averaged 9 dead clams each, indicating that the clam loss was of major proportion (estimated at several tons). Subsequent checks showed a heavy kill of razor clams south of the wreck to a distance of 8-10 miles. Serious depletion of age classes of clams that would have supported digging for three years was reported. Dead horse clams (Schizothaerus nuttalli) and dungeness crabs (Cancer magister) were noted in addition to razor clams.

Numerous dead and distressed sea birds were also found. In a one-mile stretch north of Boone Creek, for example (a region south of the wreck) 45 such birds were encountered.

Case History 5: The Torrey Canyon Disaster (Cornwall, England) (O'Sullivan and Richardson, 1967; Bellamy et al., 1967b; Bourne et 1967; Nelson-Smith, 1968; Corner et al., 1968; Smith, 1968; Simpson, A. C. 1968).

On March 18, 1967 the tanker Torrey Canyon ran aground and broke up on the Seven Stones Reef 15 miles west of Lands End, England. About 117,000 tons of Kuwait crude oil were released from her ruptured tanks over a period of several weeks. It spread over large areas of the sea and washed ashore on the coastlines of Cornwall (an estimated 13,000 tons), Brittany, and the Isle of Gurnsey (some 21,000). About 10,000 tons of detergent were used on British coasts to disperse the oil and to clean the beaches and rocky shores.

The Plymouth Laboratory of the Marine Biological Association of the United Kingdom made both field and laboratory studies of the effects of oil and detergent on marine life. Surveys of the plankton and the biota of rocky shores, sandy shores, and river mouths were carried out, as well as laboratory toxicity tests.

Rocky Shores: The rocky coast at Trevone had been studied fairly extensively prior to the Torrey Canyon disaster, so there was a good background of information available for assessing changes in the intertidal biota there. On March 29 and 30 Trevone received a deposit of oil reported to be more than half an inch deep over all the rocks. When the first biotic surveys were made, on April 10, detergent treatment had been under way for four days. Dead animals noted included limpets, top shells, periwinkles, dog-whelks, annelid worms, and crabs. The only apparently unaffected organisms were two species of barnacles (Chthamalus stellatus and Balanus balanoides), mussels, and fucoid algae. On April 23, several days after detergent spraying had ceased, many thousands of limpet scars were noticable on the rocks, indicating sites of their former attachment. Shallow rock pools which, prior to the accident, were known to have contained a variety of marine life (anemones, limpets, periwinkles, various sea weeds, small crabs, prawns, and small blennies), contained after detergent treatment only a single species of anemone and a few small algae. A small patch of intertidal rock (45 x 35 cm) which had been photographed in August 1966 to illustrate the typical fauna of the area was rephotographed on April 23, 1967, after detergent treatment. The numbers and kinds of animals recorded on the two occasions were as follows:

Species	Number in Aug. 1966	Number in April 1967
Actinea equina (anemone)	1	0
Chthamalus stellatus (barnacle)	many	few
Mytilus sp. (mussel)	2 or 3 (small)	1 (small)
Patella vulgata (limpet)	24 medium & small	0
Monodonta lineata (top shell)	11	1
Gibbula umbilicalis (snail)	1	0
Littorina saxatilis (periwinkle)	4	0

This reduction in animal life appeared to be typical of the oil/detergent-affected rocky coasts, with the main effects attributable to detergent action.

Differences in sensitivity were noted in both flora and fauna. At Porthleven reef, for example, surveys showed that damage to algae was extensive, especially to delicate filamentous, membranaceous red algae. Sensitivity varied with species.

A similar variation in sensitivity to oil-detergent treatment was noted in the animal life of the area. The anemones Actinia equina and Tealia felina, appeared to be the most resistant animals on the shore. They were found in intertidal pools that had no other life. Other anemones (Anemonia sulcata, Sagartia elegans, and Cereus pedunculatus) were much less resistant and rarely survived.

Crustaceans were especially sensitive. The lobster Homarus vulgaris and the crabs Porcellana platycheles, Cancer pagurus, Portunus puber, Xantho incisus, Pilumnus hirtellus and Carcinus maenas were completely wiped out in the areas studied. No shrimps, prawns, or gammarids were seen during the surveys, and dead Ligia oceanica and Orchestia sp. littered the rocks at high tide. The only resistant crustaceans were the barnacles, but even they were killed off in intensively sprayed areas.

Among the mollusca the limpets were especially vulnerable. In contrast to the top-shells, for example, they had no operculum for closing out the environment. Oil and detergents first affected limpets by causing them to loosen their attachment to the substratum, after which gills and mantle edge were exposed. Differences in sensitivity even between the three limpet species in the area were noted. Patella vulgata was most sensitive, P. aspersa was most resistant, and P. intermedia lay between the two.

Echinoderms were very sensitive. Only fragments of starfishes (Marthasterias glacialis and Aterina gibbosa) and sea urchins (Psammechinus milaris) were found following detergent treatment.

Polychaete worms (Lepidonotus clava, Eulalia viridis, Nereis pelagica, Perinereis marioni, Eunice harassii, Lysidice ninetta, Marphysa sanguinea, Arabella iricolor, Dodecaceria concharum, Potamilla reniformis, and Dasychone bombyx) were relatively resistant. Their lack of sensitivity was presumed on the basis of laboratory tests to be due partly to a natural physiological re-

sistance and partly to the fact that they lived in crevices or among weeds deep in pools where they could avoid the worst of the detergent-affected water. Crevice-dwelling bivalves (e.g. Hiatella striata) also had better survival records, as did animals living on the undersides of overhanging rocks.

A particularly striking change in the ecosystem following the disturbance was the unprecedented growth of algae on the rocks, especially Enteromorpha, with heaviest growth at low water and mid-tide levels. Color photographs are included in the Smith (1968) book of a region on the Trevone coast near a sewer outfall where diluted detergent washing over the rocks killed limpets and top shells, but not mussels. One photograph, taken in 1955, shows the normal summer appearance of the rocks with ragged patches of algae, while the other photograph shows their condition on July 9, 1967, about $3\frac{1}{2}$ months after the accident. On this date the entire rock surface was an almost unbroken mass of green Enteromorpha, Ulva, and some Porphyra. The contrast between the two photographs is striking. The cause of this extremely luxuriant algal growth was presumed to be the destruction of limpets and other grazing molluscs.

Sandy shores. Oil deposits on sandy beaches were usually hosed with detergent into the water. Some of the oil was carried out to sea as an emulsion but much unemulsified oil and detergent were spread to varying degrees over the beach areas. Some oily layers were buried by wave action through deposition of clean sand on top.

These sandy beaches, being unstable and normally low in organic food content, did not support much animal or plant life before the accident. A small isopod crustacean, Eurydice pulchra, normally common on the beach at Mawgan Porth, was not completely eliminated by the detergent treatment and by August 1957 had repopulated the entire beach. Laboratory experiments showed it to have detergent resistance that was "above average for crustaceans". Other animals typical of clean, sandy shores did not survive. Shortly after the detergent treatment, only remains were found for the sand eel, Ammodytes immaculatus, the small burrowing crab, Pirimela denticulata, the heart urchin, Echinocardium cordatum, the razor clam, Ensis siliqua, and the bivalve, Mactra.

At Perranuthnoe, where surveys of a heavily affected beach were made on May 10, 1957, sieving of sand through a fine net collected very few of the fauna under a millimeter in length that normally are found there. On the other hand, numerous small living oligochaete worms and a few nematodes were collected in the same area, again indicating the greater resistance of these worms.

Plankton. Ten days after the wreck of the Torrey Canyon a plankton survey was made in the western English Channel, both in the Seven Stones area and in presumably uncontaminated water. All samples contained apparently healthy plankton populations representative of these areas at this season, and relatively little damage to planktonic organisms was evident (Smith 1968) Spooner, however, reported finding plankton injury (Battelle 1967).

Effects of oil on seabirds

Oil slicks on water are reported to have a special attraction for water

birds (U.S. Dept. Interior-Dept. Transportation 1968). Contact with oil leads to the matting of plumage, to loss of buoyancy and protection against cold, and to subsequent death by drowning, starvation, exposure, or predation. (See bibliography in Battelle report, 1967). Over 30,000 seabirds were estimated to have been killed as a result of the Torrey Canyon disaster (Bourne, Parrack, and Potts 1967). During the months following the wreck, 1223 wings, each representing a dead bird from SW Britain, were examined and identified. Nearly 98% were wings of the larger auks, guillemots and razor bills with guillemots predominating, while 2% represented 6 other species. Another group of 7,851 birds were sent to cleaning centers on the Cornish coast, and 2,000 to French centers in Britanny. Nearly all the birds at the cleaning centers died.

Other spills

Milford Haven on the S.W. coast of Britain is a major oil port which has experienced several serious oil spills. Studies of the normal marine biology have been made in order that effects of oil pollution might be detected (Moyse and Nelson-Smith 1963, Nelson-Smith 1967).

In July 1960 the tanker "Esso Portsmouth" caught fire and released crude oil into waters of Milford Haven. Detergents were used extensively in cleaning operations. A decrease in populations of winkles (Littorina neritoides) was noted thereafter, and the algae Ascophyllum and Himanthalias disappeared in certain areas (Nelson - Smith 1968).

George (1961) reported that where no detergents were used, the barnacles, limpets, algae and other marine organisms appeared to be relatively unaffected by the oil, which was gradually eliminated by such natural processes as evaporation, weathering, and browsing by limpets.

In March 1962 the "Benjamin Coates" wrecked in the mouth of the Haven releasing oil which was subsequently treated with detergents. Most of the limpet population was eliminated, but many winkles, topshells, and sea anemones survived. Scallops (Pecten maximus) which lived under water on the shallow bottom appeared to be unaffected, as did larger algae. Without limpets to graze them down, heavy growths of algae developed on rocks (as in the Torrey Canyon case) and persisted for three or four years before returning to normal levels (Nelson-Smith 1968).

In January 1967, the "Chrissi P. Goulandris" leaked several hundred tons of crude oil into the Milford Haven area which was treated with detergent. Following the spill, surveys of marine organisms were carried out to assess their distribution and abundance. Fig. 2 shows the data for two shores in the form of kite histograms (Nelson - Smith 1968). In general contrast to the Torrey Canyon case, the topshell Monodonta lineata was especially affected as was the barnacle, Balanus balanoides.

Ceramé - Vivas et al (1968) described effects of crude oil released from the tanker "Ocean Eagle" which was wrecked in the mouth of San Juan Harbor in March, 1968. Nine days after the accident a brief survey of San Juan Bay

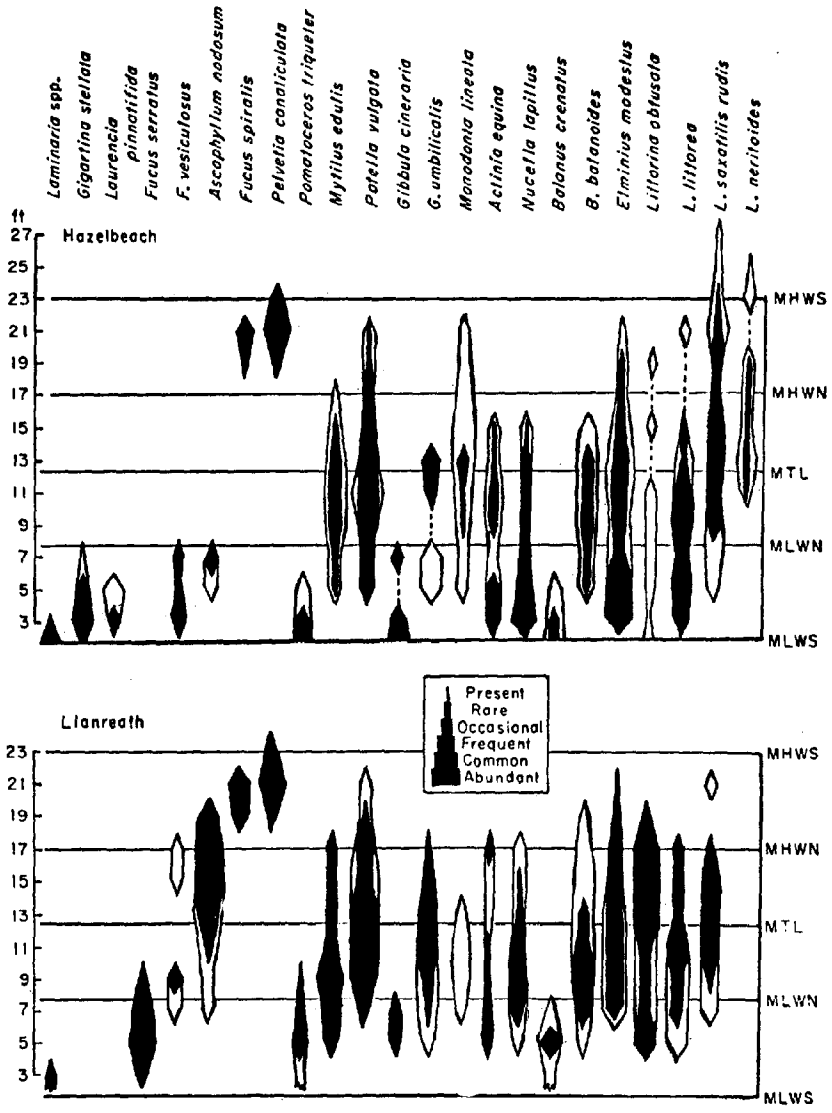

Fig. 2: Kite histograms showing zonation of common plants and animals on two shores in Milford Haven, Wales before (in outline) and after (in solid black) disturbance by fresh crude oil and detergent treatment. Heights are in feet above the lowest water level. Width of histograms (as shown in the key) indicates abundance at that level. MHWS, mean high water at spring tides; MHWN, mean high water at neap tides; MTL, mean tide level; MLWN, mean low water at neap tides; MLWS, mean low water at spring tides (From Nelson-Smith, 1968; Fig. 1).

revealed a number of dead fish representing several species, plus observation of gross lesions on living fish. Pelicans were found with oil-plastered feathers. Nineteen days after the accident incidence of lesions in schools of Opisthonema oglinum (sardina) was estimated at 95%, with accompanying behavioral abnormalities. The causative agent was not certain, but was presumed to be associated with oil/detergent effects. Toxicities of various detergents on sea urchins were also tested in laboratory studies.

Reish (1965) studied marine benthic invertebrates chronically exposed to oil refinery (plus other) wastes in Los Angeles Harbor. The bottom substrate in the areas examined was oily, black, and possessed a sulfite odor. Physical, chemical, and biological surveys were made before and after dredging, a process which exposed uncontaminated substrate to the water. The number of species increased immediately after dredging, but diminished again as the bottom regained its cover of oily wastes. Reish concluded that decrease in number of species was related to waste water volume, high organic carbon, and low dissolved oxygen, and that the oil refinery wastes altered the environment within a short period of time. He could not determine whether death of benthic species was due directly to the oily wastes through toxic reactions or indirectly through oxygen depletion of the water, brought about by the biological oxygen demand of the waste.

Some idea of longer range adaptation of marine communities to steady oil release is given in the case of an oil seep area in NE Venezuela at Lake Guanoco. The volatile compounds are continually evaporating, leaving a large deposit of pure asphalt. Lasser and Vareschi (1959) studied succession in this system, and found that at the border of the asphalt deposit succession begins with floating plants, continues through swamp plants, and leads eventually to mangrove scrub (Pterocarpetum rhizophorosum). Figure 3 illustrates a vertical view of the system.

DISCUSSION

Disturbance of coastal ecosystems by oil/detergent has been a short term effect in most cases so far studied. Nevertheless it has been shown to alter the balance of these systems, especially those dominated by attached algal growths. The mechanism appeared to be the destruction of grazing organisms, such as limpets and sea urchins, leaving certain algae free to expand enormously. This, in turn (as in the massive kelp growth following the wreck of the Tampico Maru, (North et al. 1965)) attracted numerous organisms that might normally not have been found in the area.

Although species varied according to latitude and geographical location of the affected ecosystems, the animals that appeared to be among the more sensitive to oil/detergent disturbance were crustaceans, razor clams, echinoderms, and limpets. A tendency toward resistivity was shown by anemones, periwinkles, mussels, barnacles, nematodes, and certain polychaete worms. The resistant animals possess certain adaptations that permit survival under conditions of temporary oil/detergent stress: physiological resistance, possession of shielding devices such as an operculum, and habits of hiding in crevices or among weeds deep in pools.

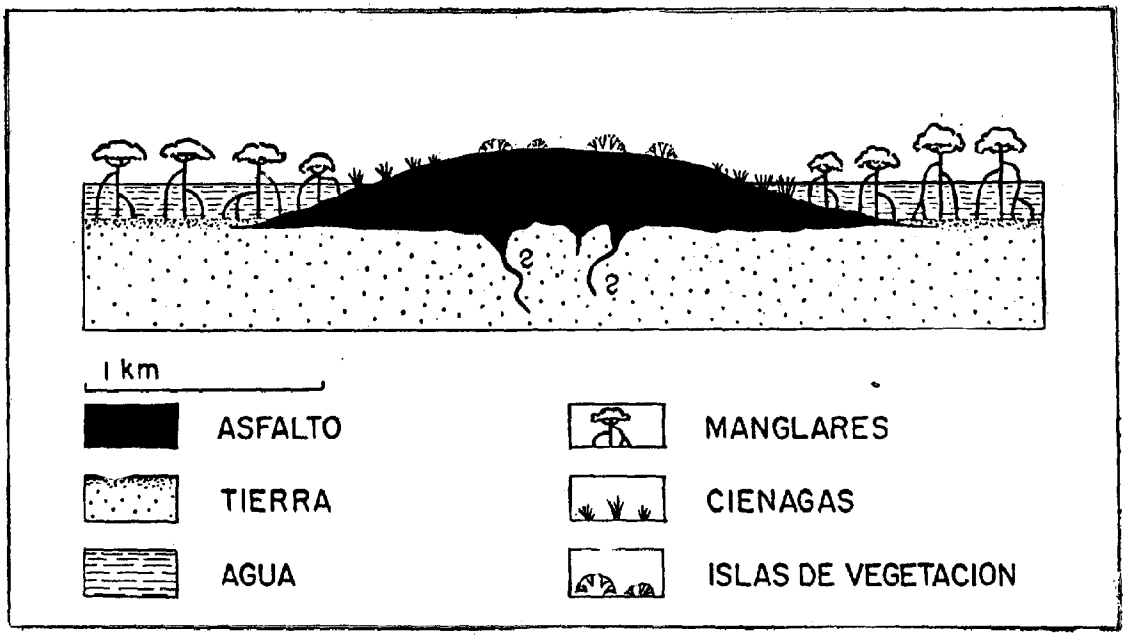

Fig. 3: Vertical view of the asphalt Lake of Guanoco in NE Venezuela. The asphalt, land, and water components are indicated, as are the three zones of vegetation: mangroves, grasses, and islands of vegetation. (From Lasser and Vareschi, 1959; Fig. 2.)

Plant species also showed a wide range in sensitivity to oil/detergent. This was true not only for the algae of rocky coasts but also for other coastal plants. The Battelle report (1967) discusses studies by Mackin on salt marsh plants (Distichlis spica, Salicornia begelovii, Spartina alterniflora, and young mangroves) which indicated that those which grew with their roots and stems in the water were more subject to injury by oil than were oysters. Oil concentrations had to exceed 10 ml/ft^2 before injurious effects were noted, however. Lasser and Vareschi (1959) reported that mangrove scrub (Pterocarpetum rhizophorosum) and other plants grew at the border of natural asphalt deposits. Diaz-Piferrer (1962) reported extensive oil damage to mangroves off southern Puerto Rico, and Clendenning (1959a) found an arrest in photosynthetic capacity of kelp blades when they were exposed to a 0.02 mm film of diesel oil for three days at 20° C.

Disturbance effects have been most marked in littoral ecosystems, especially in intertidal zones. Although intertidal biota have been selected for their resistance to relatively stringent environmental conditions (Smith 1968), they have been the ones most severely damaged by oil/detergent effects because the latter have been greatest in that zone.

Rate and extent of recovery of flora and fauna in areas denuded by oil-detergent stress has depended on the severity of the treatment, on the presence of suitable substratum, on whether recolonization had to depend on emigration of motile forms or on settling of planktonic larvae, and on other factors. North et al. (1965) found that plant species usually tended to be found on denuded areas before animal species. They suspected a difference in toxic thresholds, resulting perhaps from the possession by plants of a protective cell wall. In any case recovery was usually well under way within two or three months after the disturbance. Refaunation was begun to a large extent by the settling and establishment of juveniles of species with planktonic larvae. In most of the field studies made, oil/detergent appeared to have no adverse effect on plankton. Mackin and Sparks (1962) found that oyster spawning and rate of settling were not affected by the presence of crude oil. Wilson (1968 a,b) on the other hand, found in laboratory experiments that the larval form of the worm Sabellaria was inhibited from settling in sand treated with detergent.

Crude oil appeared generally to be less toxic than diesel fuel or detergents (Orton 1925, Mackin and Hopkins 1962, Smith 1968). In fact, except for its lethal effect on water fowl, crude oil under field conditions was found to have low toxicity for marine life. After a few weeks oily patches on rocks are actually eaten away by limpets and other grazers (George 1961, Smith 1968). An exception to this reported low toxicity was Diaz-Piferrer's (1962) report of especially bad effects of crude oil on the biota of the south coast of Puerto Rico. This effect was presumably not one of latitude. Usually the argument is that organisms of cold regions, being highly stressed by the cold climate, are those having the narrowest limitations for tolerance of additional stresses such as oil exposure (Alaska Water Laboratory Report, 1968).

Oil decomposition is carried out through a number of natural processes, so that eventually oil that has been spilt in the sea will disappear. These processes include weathering, sand abrasion, browsing gastropods, evaporation

of volatile continuents into the atomosphere, sinking of oil-coated debris, auto-oxidation, and oxidation by microorganisms. Although a stress for larger organsims, oil serves as an energy source for many microorganisms. In equatorial regions the rate of oxidation of oil by microorganisms may be several hundred grams of oil per m^3 of contaminated sea water per year (Pipel 1968). Thin layers of floating crude oil may be completely colonized by bacteria in two weeks or less and completely decomposed in 2-3 months. Protozoa feed on the bacteria which thus serve an important role in marine food chains (ZoBell 1964). Oil oxidizing bacteria have been shown to exist in Louisiana mud (ZoBell and Prokop 1966) and to be able to destroy large numbers of the compounds of crude petroleum.

Following oil/detergent treatment of the Cornish coast after the Torrey Canyon disaster, sands from various places on the coast were examined for their bacterial content (Smith 1968). Oil-decomposing bacteria were found in high numbers (up to 400,000,000/ml in wet sediment) along with other bacteria (heterotrophic-proteolitic types). Laboratory experiments showed that detergents at concentrations around 10 ppm were capable of killing most oil-degrading bacteria, but some bacteria could survive concentrations of 100 ppm and multiply rapidly. Presumably this happened on the Cornish shores. The high bacterial concentration in the oil/detergent treated sand led to the assumption that biodegradation of oil was proceeding at a fairly high rate.

Normally the sands of Cornish beaches are low in organic matter and do not exhibit gray layers. From May 1957 abnormal and conspicuous development of gray layers was noted. Laboratory experiments showed that graying of sand resulted from the action of bacteria working on oil anaerobically. The anaerobic condition presumably resulted from activities of the main aerobic degraders. The intensity of grayness was related not directly to oil content but to lack of oxygen.

Some investigators have suggested that detergent treatment may actually aid bacterial oxidation of oil by causing it to be spread thinly through a beach (Smith 1968). Others point out that the toxic effect of the emulsifier on microorganisms may reduce the rate of decomposition (Pipel 1968). The great increase in bacteria in sandy beaches of Cornwall may have been due not only to the enrichment of the sand by oil but also to the decrease (through killing) of microfauna that normally live between the sandgrains and feed upon their bacterial films. For such interstitial animals, detergent either in interstitial water or adsorbed on sand grains, is of even more toxic significance than for larger beach animals (See Battelle bibliography, 1967).

NEEDED RESEARCH

Short term effects of oil disturbance have been well documented, but there is still insufficient understanding of what happens to an ecosystem when oil stress is chronic. In many estuarine systems (such as ship channels) the presence of oil is only one of many types of disturbances. Natural ecosystems near subterranean oil seeps as in the Venezuelan example (Lasser and Vareschi, 1959) should provide study areas in which the sustained effect of oil alone can be evaluated.

Knowledge of oil effects on plankton and other marine biota is accumulating from laboratory studies, but more information is needed from studies of natural systems.

It has been suggested (Corner et al. 1968) that detergent used in oil spill clean ups may enter the food chain with possible effects on man eventually. Such possibilities need examination. Since detergent action is often lethal to marine life, means for increasing the rate of elimination of oil by natural means should be further investigated. For example, continued research on oil oxidation by bacteria (See ZoBell, 1964) is important for predicting oxidation rates under different ecological conditions.

Chapter E-12

TREATED PILING SYSTEMS

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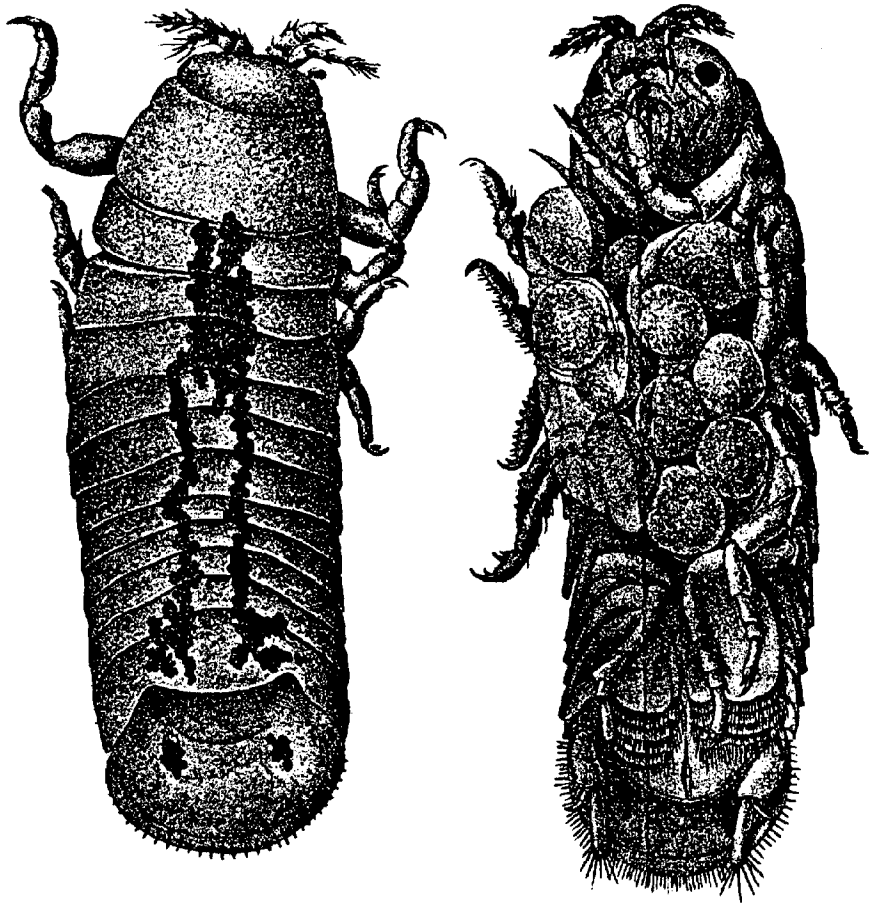
Wood in the marine environment is rapidly covered and riddled with boring and decomposing organisms such as shipworms and fungi. If treated with creosote and other chemical preservatives it resists invasion by most decomposers, and treated wood piling in piers often stand for many years. The chemical substances, usually introduced into the wood spaces under pressure treatment, constitute a stress on the ecological system which may tend to develop in such wood. Treated piling do develop simple communities of boring animals, small crustaceans sometimes called gribbles or by the scientific designation Limnoria. The distribution of gribble ecosystems has been studied intensively in relation to environmental factors and types of wood treatment because of their economic importance in the failure of marine installations. The relationships of the wood boring organisms and the chemical stress are like the adaptations of other ecological systems to chemical stress from waste disposal. As treatment concentration increases, the ecological system diminishes in diversity and may be eliminated altogether, if chemical stress exceeds the energies available to the community.

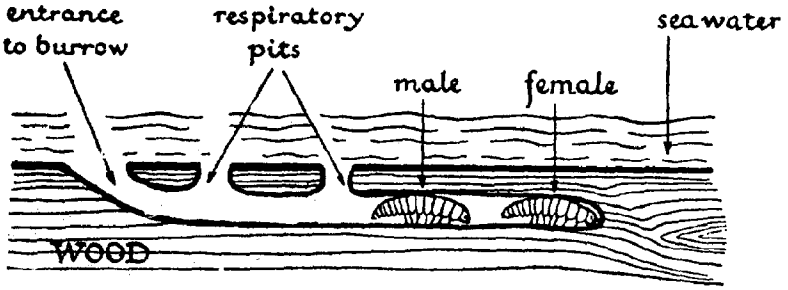
One of the species of Limnoria is pictured in Fig. 1, while Fig. 2 diagrams the burrows in which it lives, consuming wood which it digests as food and deriving oxygen from waters outside the wood. The simple gribble ecosystem contrasts with the more diversified communities of many species in untreated wood such as those shown in Fig. 3. The current creosote-coal tar processing tends to minimize surface fouling. Several reviews (Crisp, 1965; Woods Hole Oceanographic Institute, 1952a; Ricketts and Calvin, 1968) along with other chapters of this report adequately discuss fouling ecology. Treatment will also protect marine piling for many years against boring by Teredo or Bankia shipworms but neither creosote nor creosote-coal tar processing is effective in preventing attack by a species of gribble, Limnoria tripunctata, commonly present in U. S. temperate and tropical estuaries (Colley, 1967; Mochman, 1967). The gribble is an isopod crustacean reaching a maximum length of only 5 mm though one pile may harbor several hundred thousand individuals collectively causing failure to the structure (Fig. 4). Although creosote treatment does markedly retard the attack through a lowering of the realized reproductive ability of the population, L. tripunctata does survive and reproduce in creosoted wood (Beckman, Menzies and Wakeman, 1957) and is the primary living component of this system. In contrast to most other systems of the estuary, the initial treated piling system thus consists of the interaction between the physical and chemical environment with only one dominant species.



1 2

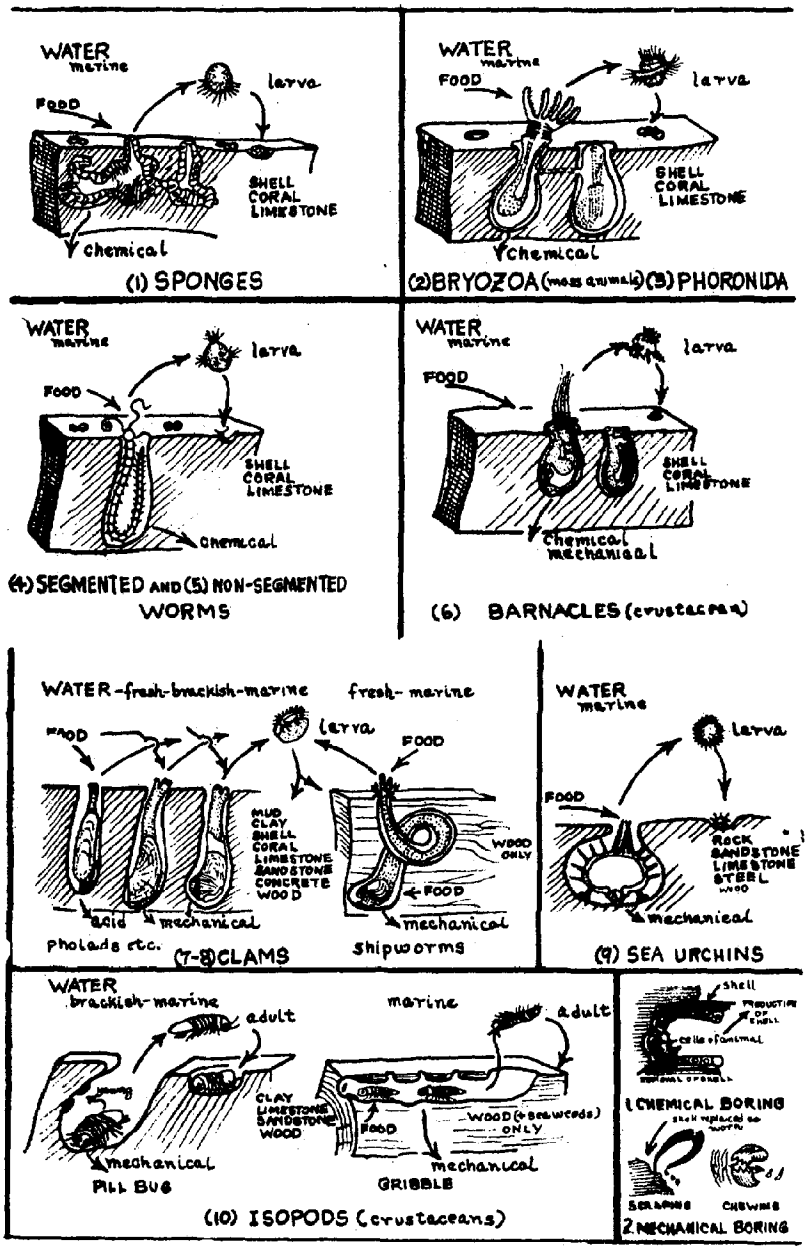
Limnoria lignorum. 1, Dorsal view of male. 2, Ventral view of gravid female. × 28. After Hoek, 1893.

Fig. 1. Limnoria lignorum. 1, Dorsal view of male. 2, Ventral view of gravid female. x 28. (from Kofoid and Miller, 1927).



Gribble, *Limnoria lignorum*, diagrammatic section of burrow with animals in position. (Modified from Russell and Yonge, *The Seas*.)

Fig. 2. Diagrammatic section of piling with Limnoria in burrow (from Yonge, 1949).



Ten Major Categories of Wood Borers.

Fig. 3. Major categories of marine wood borers (from Menzies and Turner, 1957).

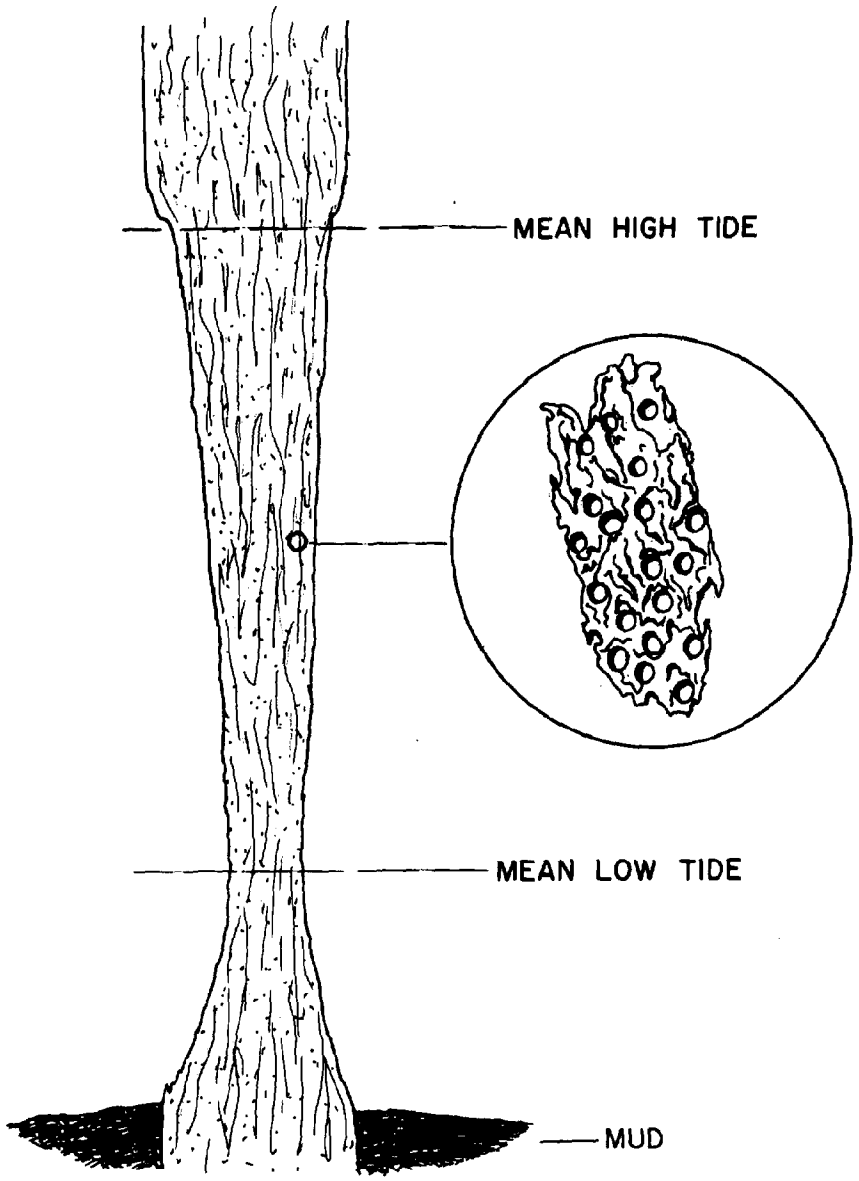


Fig. 4. Diagram of a gribble-infested pile illustrating hour-glass shape resulting from physical erosion. Inset shows wood chip formed as a result of extensive burrowing. Circular burrow holes average about 1.5 mm in diameter.

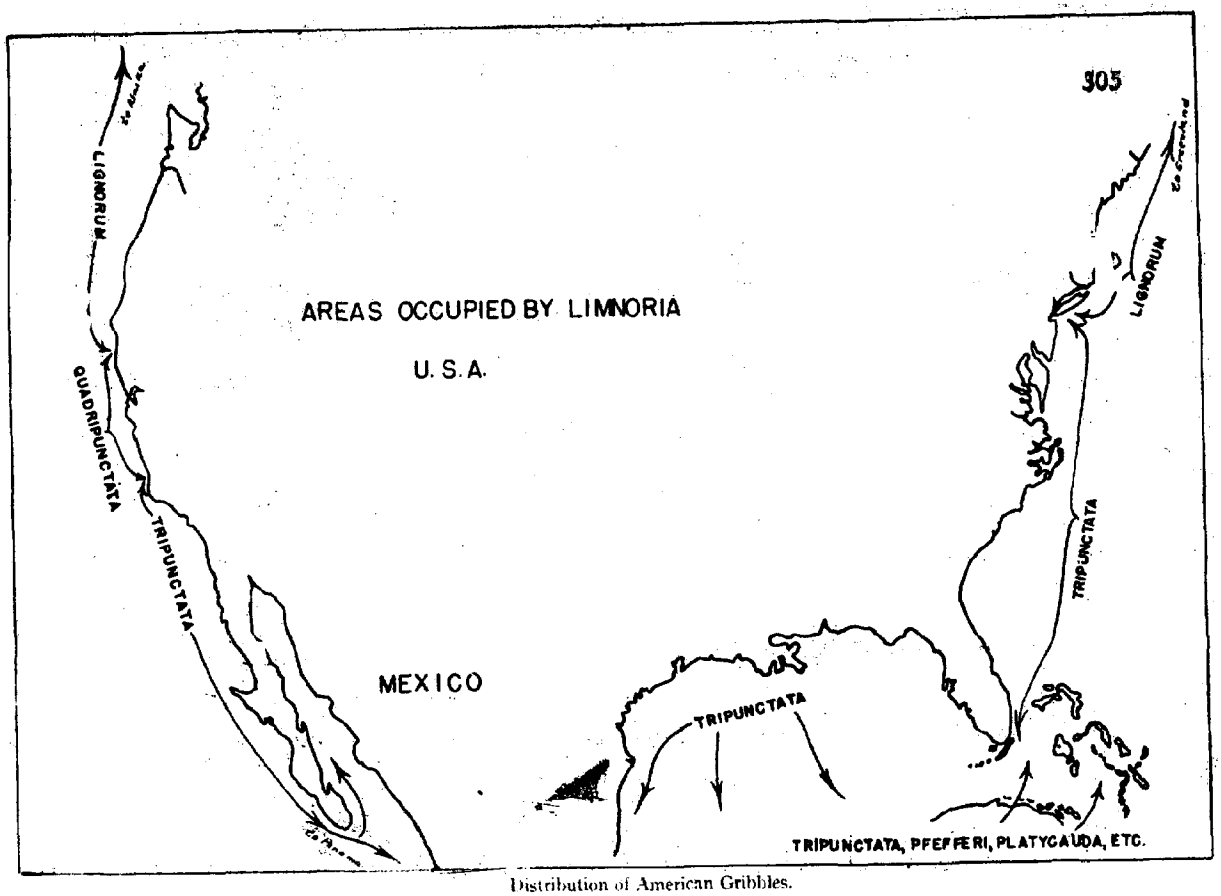
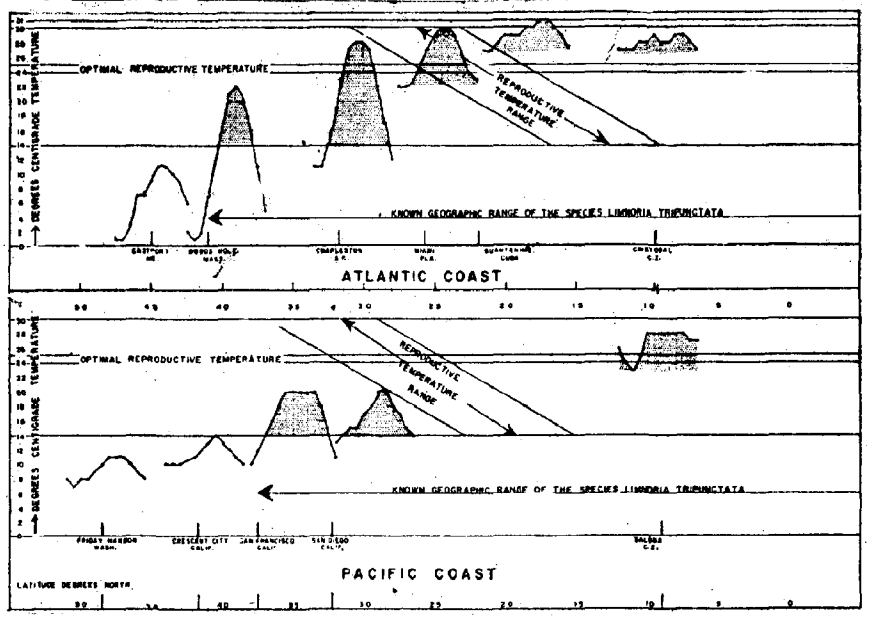


Fig. 5. Distribution of Limnoria in America (from Menzies and Turner, 1957).



Geographical distribution, mean monthly surface sea water temperatures plotted at approximate longitude and reproductive temperature limits (the horizontal lines connected by the sloping block): The shaded areas represent the period of time that the temperature is within the reproductive range of the species. The minimum reproductive temperature, 14° C., is estimated from the average of the three experiments to be at this temperature. At this point one young would be produced for each adult.

Fig. 6. Geographical distribution of Limnoria tripunctata; mean monthly surface sea water temperatures plotted at approximate longitude and estimated reproductive temperature limits. The shaded areas represent the period of time that the temperature is within the estimated reproductive range (14°C) of the species (from Beckman and Menzies, 1960).

GEOGRAPHICAL DISTRIBUTION

The geographical distribution of Limnoria species responsible for boring communities in treated piling is shown in Fig. 5 (Menzies and Turner, 1957). Particularly impressive is the wide range enjoyed by L. tripunctata, the species least affected by creosote-coal tar processing. The adaptation to temperature and light regimes with latitude is accomplished by species substitution. Fig. 6 (Beckman and Menzies, 1960) shows that Limnoria tripunctata is limited in breeding to temperatures above 14° C. Like some other stressed ecosystems, the treated piling system is similar in species composition over a broad latitudinal distribution.

SEASONAL PATTERNS

The treated piling system seems programmed so that migration, boring activity and reproduction in temperate latitudes coincide with the seasonal pulse of energy input. However, tropical systems remain to be studied. Temperature appears to be of prime importance although other environmental factors such as day length may control programming.

The review of Eltringham (1966) notes that temperature is probably the most important factor controlling L. tripunctata migration. It appears to initiate migration but the apparent thermal threshold has considerable geographical variation. Fig. 8 (Johnson and Menzies, 1956) shows a definite seasonal migratory period in San Diego Bay which reaches its peak during the summer season of highest water temperatures. However, more northern observations (Figs. 9 and 10) tend to indicate a spring migration pulse prior to peak water temperatures.

The onset of the breeding season of L. tripunctata is considered by most investigators to be dependent upon temperature (Coker, 1923; Johnson and Menzies, 1956; Eltringham and Hochley, 1961; Hochley , 1966) but field observations of the critical temperatures also vary with the locality. Laboratory experiments (Kampf, 1957; Beckman and Menzies, 1960) under constant temperature conditions still show a seasonal pattern of reproduction. This suggests that seasonal gravidity does not appear to be changed by temperature. Study of a tropical system with small or no seasonal pulses may help explain programming mechanisms.

VERTICAL DISTRIBUTION

Boring occurs over a wide vertical range as shown by Greenfield (1952) in Fig. 7. The vertical distribution of L. tripunctata within the piling system may be correlated with decreasing light intensity. Laboratory experiments (Isham, Smith and Springer, 1951) on Limnoria exposed in a shaded vertical tube indicated that burrows were most numerous at low intensity (Fig. 11) and decrease in number with increasing intensity of light. Bourdillon (1958) found that at light intensities greater than 0.01 lux, the average phototaxis of adults is negative but at lower light intensities the phototaxis is reversed or positive. The ecological role of light during settlement in nature thus results in most Limnoria exhibiting a photonegative behavior. Although attack between high

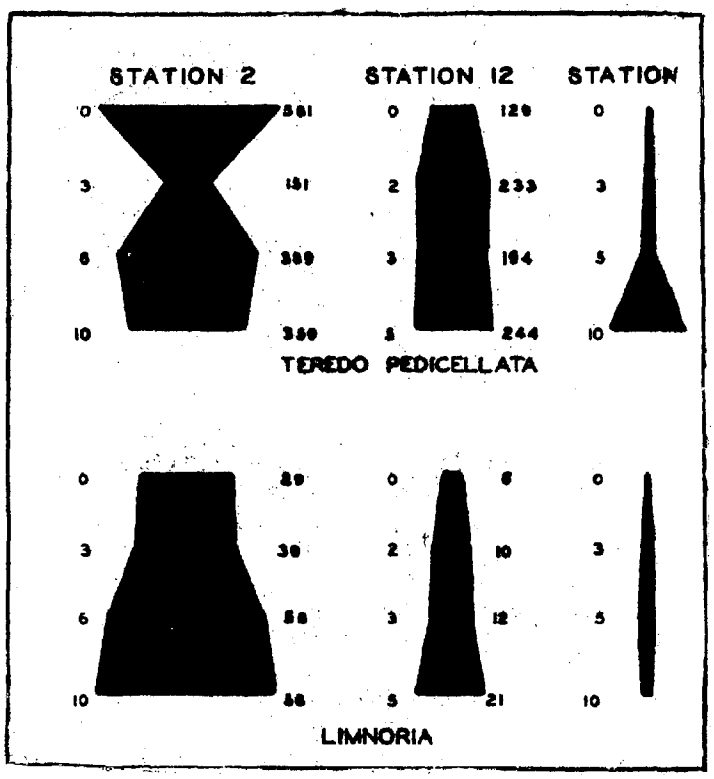
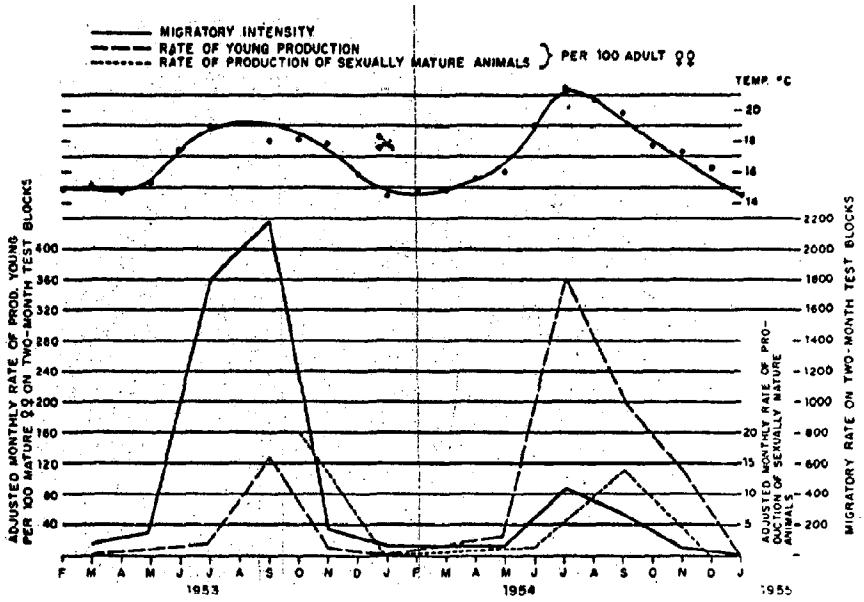
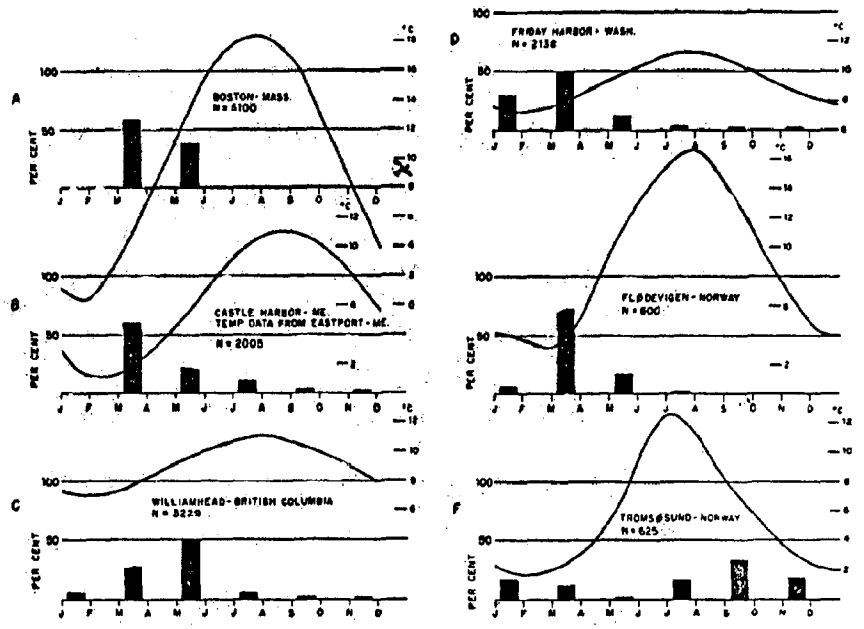


Fig. 7. Vertical distribution of Limnoria under bay and river conditions. Numbers at left of each figure represent the depth of water in feet from surface to bottom. Numbers at the right of each figure represent the average monthly number of borers (from Greenfield, 1952).



Limnoria tripunctata. The relationship between monthly production rates and migration in two-month periods, Station A, San Diego Bay.

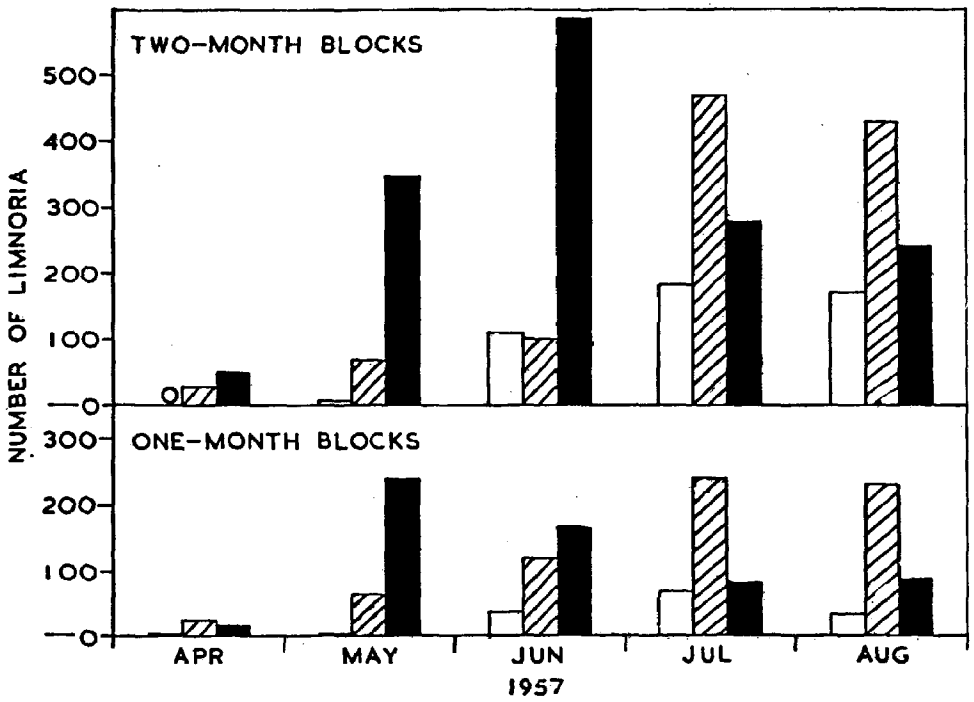
Fig. 8. Seasonal pulse of borers in southern California (Johnson and Menzies, 1956)



Limnoria lignorum. Percentage of seasonal attack in bimonthly periods in various localities. A and B data from William F. Clapp Laboratories; C from Black and Elsey, 1948; D from Johnson, 1935; E and F from Sömme, 1940. N = Yearly sum of animals on test blocks.

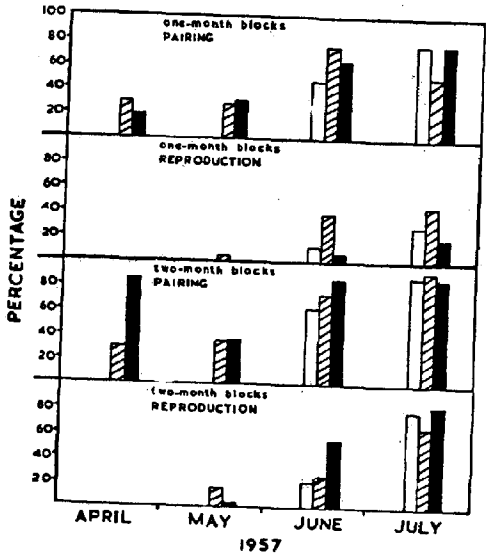
Fig. 9. Seasonal patterns of Limnoria in different locations (Johnson and Menzies, 1956).

A



The migration of the three species of *Limnoria* at Southampton (Site A) for the months April to August 1957. The data are from blocks of one and two months' exposure. Unfilled columns—*L. tripunctata*; hatched columns—*L. quadripunctata*; filled columns—*L. lignorum*. O signifies the absence of bores of *L. tripunctata* from the April sample.

B



Pairing and reproduction in the three species of *Limnoria* in test blocks at Southampton (Site A) for the months April to July 1957. Unfilled columns—*L. tripunctata*; hatched columns—*L. quadripunctata*; filled columns—*L. lignorum*.

Fig. 10. A, migration of the three species of Limnoria at Southampton, England, for the months April to August 1957. The data are from blocks of one and two months' exposure. B, pairing and reproduction in the three species of Limnoria in test blocks at Southampton, England, for the months April to July 1957. Unfilled columns--L. tripunctata; hatched columns--L. quadripunctata; filled columns--L. lignorum (from Eltringham and Hockley, 1961).

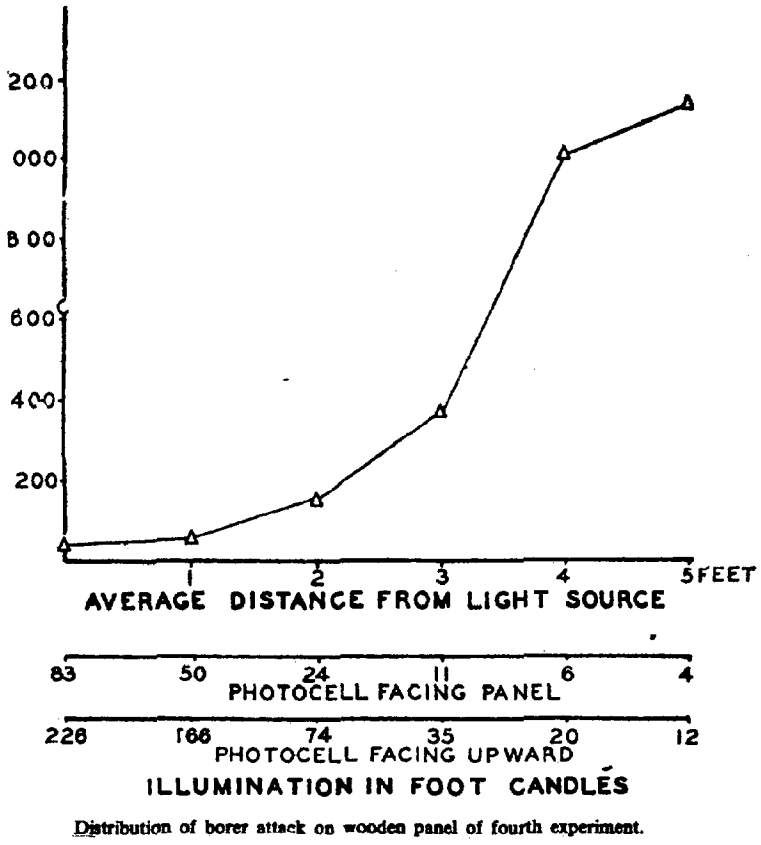


Fig. 11. Distribution of Limnoria attack on wooden panels (modified from Isham, Smith and Springer, 1951).

tide and the mudline increases with depth, resulting erosion may be more critical in intertidal areas due to physical wave action. Piling with heavy Limnoria infestation often show a characteristic hour-glass shape (Fig. 4) due to this erosion. Migration at night (Menzies, 1961) may account for distribution throughout the piling despite high daylight intensity.

HORIZONTAL DISTRIBUTION

Low density Limnoria burrows (Fig. 2) are characteristically within a few mm of a well aerated surface and incorporate numerous fine openings through which water is circulated. In heavier infestations this pattern is more complex and varies with prevailing currents and oxygen tensions. The burrows usually traverse the soft layers between the annual rings of hard wood, and the surface layer is soon reduced to a porous mass subject to erosion.

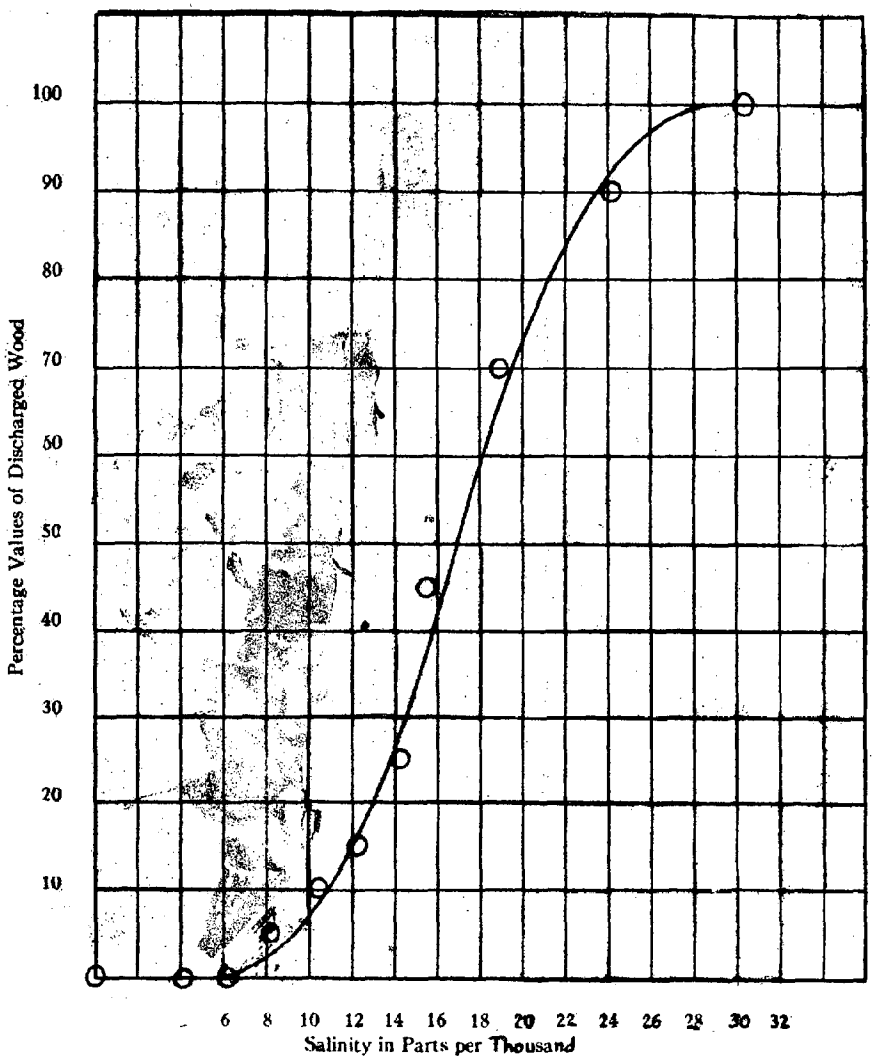
Horizontal distribution within the estuarine environment is defined by physical limits for settlement, boring activity and reproduction which are reviewed by Eltringham (1966). Low salinity between 15 and 16 °/oo usually limits survival (Fig. 12) while the reproductive optimum ranges between 25 and 35 °/oo. Little is known of the lethal effect of temperature; however, Beckman and Menzies (1960) found that L. tripunctata will feed from 10° to 30° C. The reproductive temperature ranges from about 15° to 30° and the greatest population increase is near 25° C. In Fig. 6 the experimental results are correlated with the known geographic range of this species. Laboratory experiments (Eltringham, 1961; Anderson and Reish, 1967) and field data (Menzies, 1957; Menzies, Mohr and Wakeman, 1963) suggest a limiting oxygen concentration from 0.12 to 3.0 ppm. Absence of borer attack in heavily polluted waters (Menzies, Mohr and Wakeman, 1963) may be due to such low oxygen concentrations. Distribution is also related to water current (Doochin and Smith, 1952) with maximum incidence at intermediate velocities of about 0.15 knots as in Fig. 13.

CHEMICAL TREATMENT

The status of knowledge on piling treatment is given by Colley (1967) and Hochman (1967). Fig. 14 shows the results of survival tests of gribbles fed creosoted splinters from different locations in old piling. Aging apparently changes the chemical nature of the creosote and the usability of the wood by the animals. Whereas simple creosote treatment is effective in cold regions in the presence of the additional stresses of the environment, the chemical treatment is not enough in stable tropical waters of steady high salinity and temperature. Some of the differences in preservation observed by Hochman (1967) are given in Tables 1 - 3. Much greater resistance was introduced when a second chemical agent such as an insecticide was included. The high temperatures accelerate the decomposition of chemicals, but other factors are apparently also important since some tropical piling last decades whereas those in high salinity regions with little stress may last only a year or two.

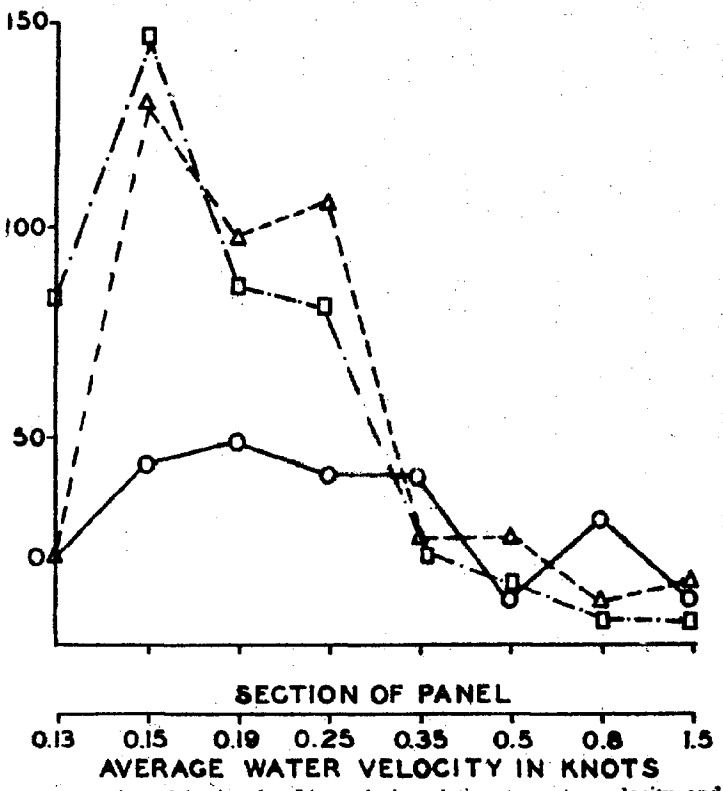
ENERGY DIAGRAM

The general energy flow for the piling system is shown in Fig. 15. The dominant pathway is man-made treated piling eaten by L. tripunctata and the



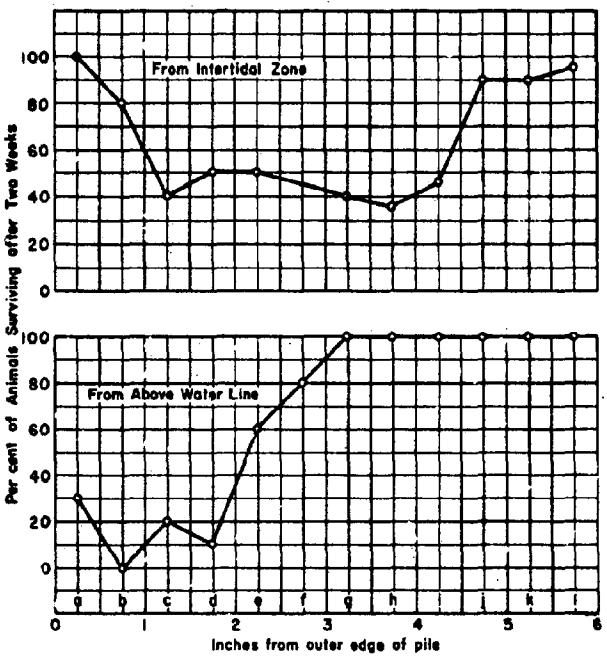
Graph of the activity of *Limnoria lignorum* in water of various salinities.

Fig. 12. Graph of the activity of Limnoria in water of various salinities (from Kofoid and Miller, 1927).



Intensity of boring by *Limnoria* in relation to water velocity and orientation of wooden surfaces. First experiment. Square conventions, lower horizontal panel; triangle, upper horizontal panel; circle, vertical panel.

Fig. 13. Intensity of boring by Limnoria in relation to water velocity and orientation of wooden surfaces. Squares represent lower horizontal panel, triangles represent upper horizontal panel, and circles represent vertical panel (from Doochin and Smith, 1952).



Per cent of *Limnoria* Surviving on Splinters from Various Depths into the Interior of Leached Creosoted Piling.

Fig. 14. Per cent of Limnoria surviving on splinters from various depths into the interior of leached creosoted piling (from Vind, Hochman, Muraoka and Casey, 1957).

—Preservative Values of Creosote Solutions at Pearl Harbor

Additive[b]	Exposure to Destruction at Pearl Harbor	Major[a] Attacking Organisms	Factor of Improvement
None	13	L & M	
5% Chlordane	35	M	2.7
1% Dieldrin	12	M	2.5
1% Endrin	>60	M	>4.6
1% Toxaphene	>54	L	>4.0
2% Malachite Green	16	L	1.2
1% Tributyltin Oxide	49	L	3.8
1% Phenylmercuric Oleate	24	T	1.8
3% Copper Naphthenate	>70	L & M	>5.3

(a) L = Limnoria, T = Teredo, M = Martesia.
(b) Basic treatment is creosote, 15 pcf.

—Data on Piles in Areas Where *Limnoria* Were Present[1]

Pile No.	Location	Length of Service (yr)	Condition
Pine			
P1-5	San Juan, P. R.	15 to 16	Excellent
P-11	Freeport, Tex.	9	Excellent
P-12	Freeport, Tex.	9	Slight Limnoria attack
P-13	Freeport, Tex.	9	Moderate Limnoria attack
P-14	Pensacola, Fla.	4	Slight Limnoria attack
P-15	Pensacola, Fla.	20	Almost complete erosion, in intertidal zone
P-46-7	Teteron Bay, Trinidad	18	Fair
P-52-57	Lake Charles, La.	32 to 33	Excellent

1/ Revision of Table 1, Reference 23.

—Data on Piles in Areas of Severe *Limnoria* Hazard[1]

Pile No.	Location	Length of Service (yr)	Condition
Pine			
P-16-19	Miami, Fla.	14	Complete erosion in intertidal zone
P-20-25	St. Thomas Is., V. I.	5	Heavy Limnoria attack in intertidal zone
P-26	Wrightsville Beach, N. C.	7	Heavy Limnoria attack in intertidal zone
P-27-32	San Juan, P. R.	5	Heavy Limnoria attack in intertidal zone
P-33	San Juan, P. R.	5	Complete erosion in intertidal zone
P-42-45	Guantanamo Bay, Cuba	4	Heavy attack in intertidal zone
P-46-50	Galveston, Tex.	12	Moderate Limnoria attack
P-58	Flagler Beach, Fla.	33	Heavy Limnoria attack
P-59	Flagler Beach, Fla.	11	Heavy Limnoria attack
P-60	Flagler Beach, Fla.	13	Heavy Limnoria attack
Douglas-fir			
F-1	Inner Basin, Guam	14	Slight Limnoria attack
F-13	Naval Supply Center Fuel Dock, Pearl Harbor, Hawaii	10	Moderate Limnoria attack
F-14	Naval Supply Center Fuel Dock, Pearl Harbor, Hawaii	10	Heavy Limnoria attack
F-15	Apra Harbor, Guam	12	Excellent
F-16	Apra Harbor, Guam	12	Slight Limnoria attack
F-17	Apra Harbor, Guam	12	Excellent

1/ Revision of Table 1, Reference 23.

Tables 1-3. Comparison of piling service and resistance in different estuaries (Hochman, 1967).

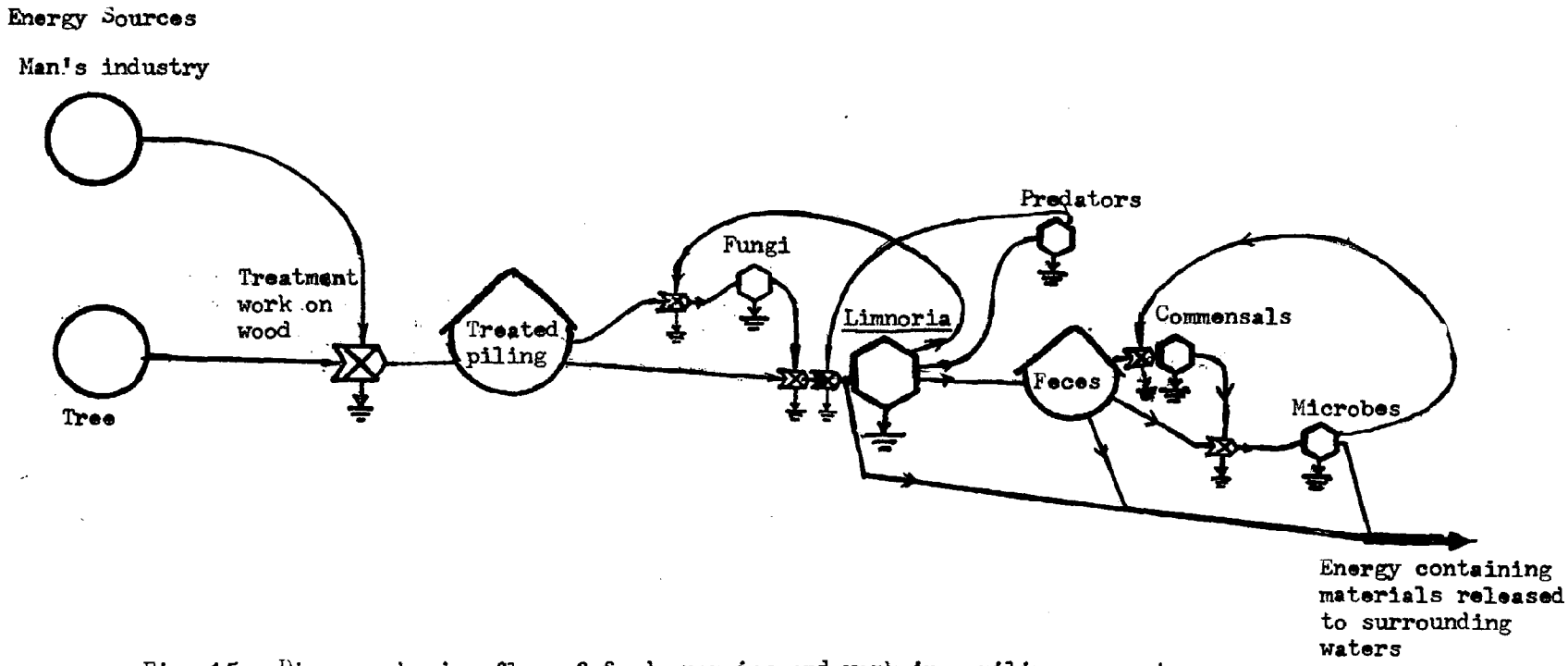


Fig. 15. Diagram showing flow of food energies and work in a piling ecosystem

Symbols: energy source; energy storage; work done by one unit on another process;

Self maintaining organisms.

resulting energy is metabolized into heat or loss as feces to the outside estuary. Microbial decomposition undoubtedly results in some energy loss from all components and is especially important in feces decay.

Several living components other than the gribbles have been described in this system. Reish (1954) has demonstrated remains of Limnoria in several polychaete annelids. The commensal amphipod Chelura does not feed on wood but is dependent upon the feces of Limnoria (Barnard, 1950; Bourdillon, 1958; Kuhne and Becker, 1964; Kuhne, 1966). Other invertebrate components may be present but their roles in the system are not well defined.

The role of fungal infestation and possible nutritional enhancement of "conditioning" piling prior to ingestion by L. tripunctata is still controversial and investigators report conflicting results (Becker, Kampf and Kohlmeyer, 1957; Lane, 1959; Ray, 1959b; Ray and Stuntz, 1959; Schafer and Lane, 1957). Fungi may be important as a source of protein and vitamins for Limnoria but they are not essential for survival under all conditions.

The major food of Limnoria, if not the only diet, is wood. The cellulase for digestion is a product of the midgut diverticula (Fahrenbach, 1959) and is produced by Limnoria itself rather than symbiotic microorganisms (Ray, 1959b).

A piling system is always in a state of deterioration and needs constant renewal of substrates from man for continued existence. Interactions or infections of other piling systems occur chiefly by floating wood or Limnoria migrations. The energy flow is simple with no primary production.

Chapter E-13

SALINA SYSTEMS

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INTRODUCTION

A salina system is the sequence of shallow ponds used in an age old technology for the solar evaporation of sea water to produce table salt. The stages in the sequence of increasing salinity are accompanied by characteristic organisms that appear wherever these systems are found throughout the world. The sequence has three main stages or subsystems: the first with blue-green algae predominating in benthic mats, the second with brine shrimp and characteristic species of the flagellated microscopic phytoplankton Dunaliella, and the third dominated by two types of red and pink bacteria. Although these stages may also develop in disturbed marine environments involving brine wastes, the salina is a domesticated version of the hypersaline lagoon systems that develop in nature (see Ch. A6) and is an example of the management of a whole ecological system for the service that it provides. The three stages of a primitive salina system in Puerto Rico are shown in Fig. 1, including the changes in characteristic organisms and associated sediments. A large French salt works is diagramed in Fig. 2, and a generalized flow chart for the extensive modern salina complexes of California is presented in Fig. 3.

Historical Note

Since the earliest times, man's inescapable physiological need for salt has led him to the edges of the sea. And here, by this limitless supply, he began the ancient business of removing salt from the ocean waters by using the evaporating power of solar energy and dry wind. Slowly he progressed in his knowledge of the art until, some thousands of years ago, a technology emerged that was based on a rather sophisticated though largely intuitive grasp of chemical succession and its consequences for organisms living in concentrated sea water. This technology came to involve the transfer and storage of water at various stages of evaporation, and required an interconnected series of large containers making up what is called in Spanish a salina system as shown in Fig. 1 and Fig. 2. Today, though most of the world's salt comes from other sources, men are still about this antique occupation whose methods have changed little across so much time and learning (Baas-Becking, 1931a; Bloch, 1963). And for all of that time, a small number of living organisms have been with the briner as a part of his work and his tradition. Having evolved the special capacity to survive in this strange environment dominated by multiple chemical and physical stresses that isolate them from severe biological competition and predation, they form integrated living systems whose color contrasts sharply with the stark white of crystals and vast crusts of drying salt.

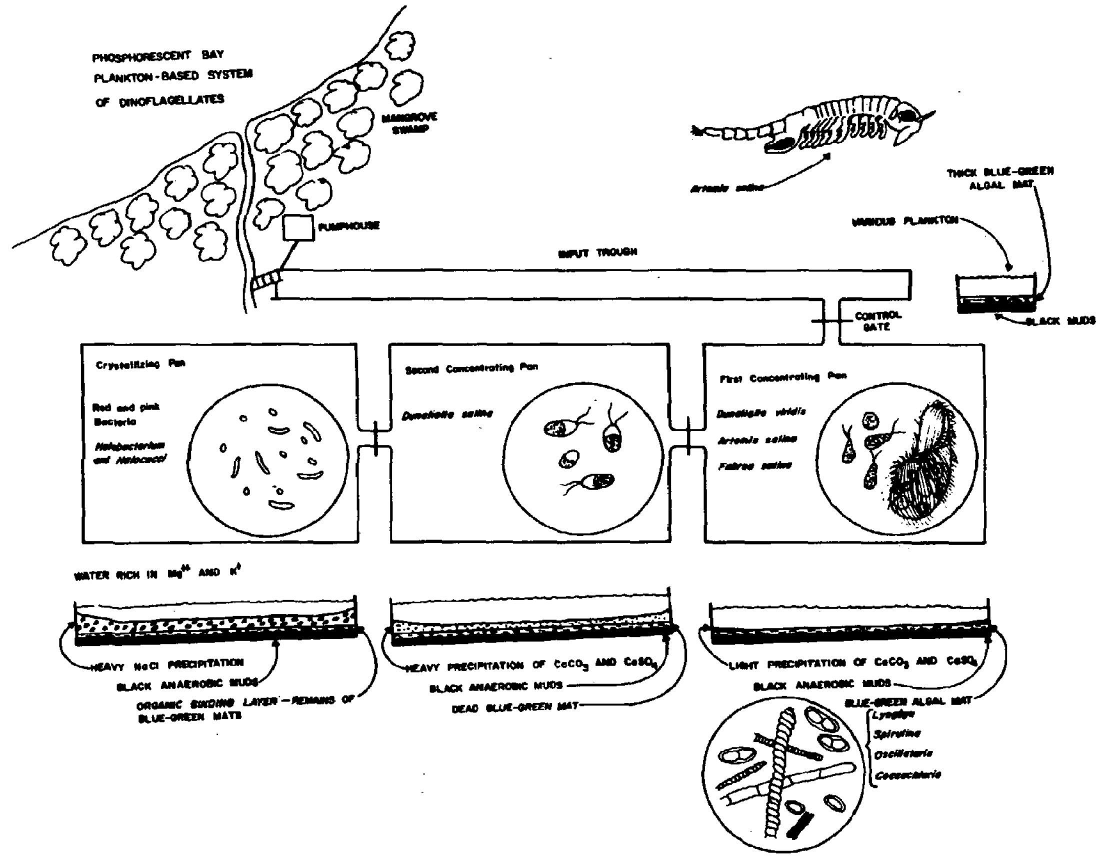

Fig. 1: Simplified schematic drawing of a primitive salina in Puerto Rico. As total salinity increases, chemical and biological succession result.

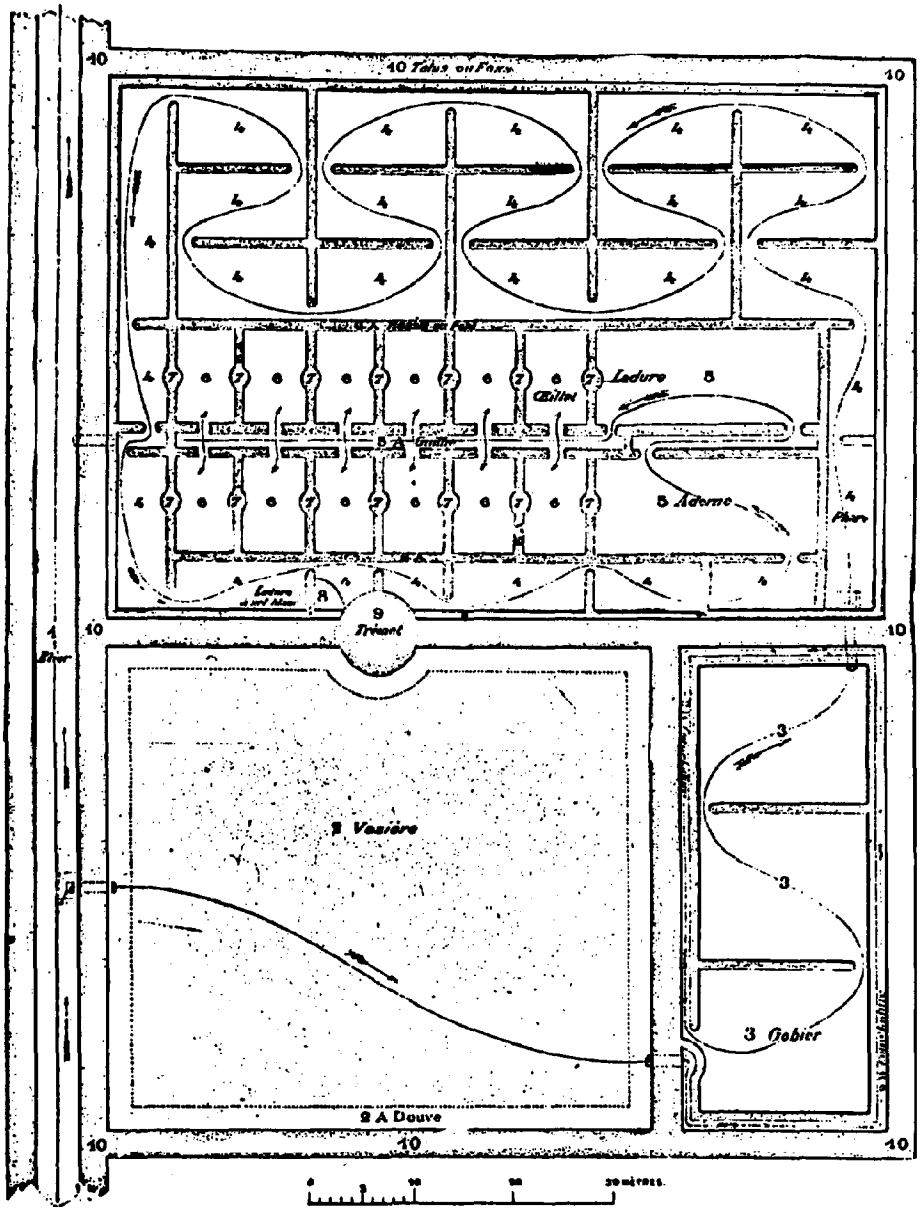

Fig. 2: Plan of a large commercial salina at Croisic-Batz, France, from Labbé (1924). The input trough (1) leads into large concentrating pans (2,3,4,5), and finally into the smaller crystallizing pans (6).

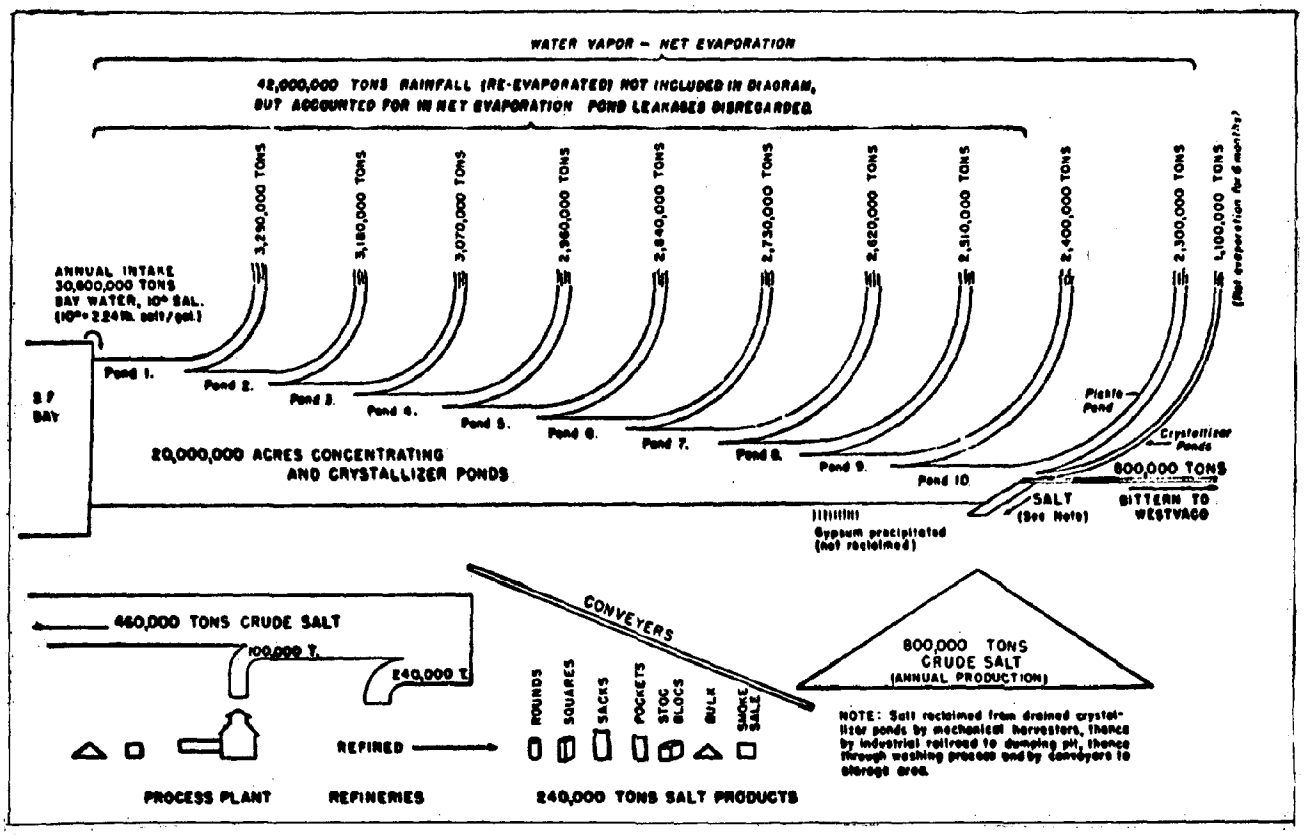

Fig. 3: Production flow chart for the large salina complexes of California, from Ver Planck (1958).

PRINCIPLES OF SALINA MANAGEMENT

To obtain salt from the ocean is easy; one need only let the water evaporate to dryness and harvest the product, there is little technology involved. But for man's taste and needs, this has not been enough. Sea salt is a complex and bitter tasting mixture of many things, including KCl, $MgCl_2$, $CaSO_4$, $CaCO_3$, and $MgSO_4$, as well as NaCl, the common table salt. To obtain the NaCl which makes up about 80% of the weight of the total mixture, and to obtain it in as pure a state as possible, the briner exploits the process of differential ion precipitation. Each of the many compounds dissolved or ionized in sea water has a characteristic solubility. As the water evaporates and salts become increasingly concentrated, those that are least soluble and those present in the greatest concentration relative to their solubility begin to precipitate. In natural, unmanaged systems such as the hypersaline lagoons of Texas and Mexico, this process may continue while many different salts become saturated and the bottom becomes covered with layers of salt arranged according to the solubility series.

In the salina, however, man exerts control and the water that remains after a given material has largely precipitated is moved by pump or by gravity to another section of the salina where the next salts may precipitate relatively free from contamination by the less soluble ions. After a number of precipitations and transfers, the point is reached where NaCl, one of the more soluble salts, becomes saturated and precipitates heavily. The remaining liquid, now called the bittern and rich in Mg^{++} and K^{+}, is removed and the crystals of NaCl are harvested, washed, and dried (see Fig. 3). By this process, so fundamentally simple, the briner may obtain a product that is more than 99% pure (VerPlanck, 1958; Kaufmann, 1960).

As a consequence of the differential solubilities and precipitation, the ionic composition of the water above the precipitate progressively changes and becomes enriched with the more soluble ions. This continuously shifting chemical composition of evaporating sea water is shown in Fig. 4. The "containers" mentioned earlier are actually broad shallow pans scraped in the ground, each some 15 cm. deep and much larger than a foot ball field. The general arrangement of such pans is shown by Fig. 2, a schematic drawing of a large commercial salina in France. The depth-area relationship of the pans is designed to evaporate large quantities of water as quickly as possible through solar heating in the shallow depth and through dry wind moving over a relatively long fetch and large surface, with the area of the first concentrating pans being fixed at about 10-15 times the area of the final crystallizing pans (VerPlanck, 1958). These individual pans are microcosms of arrested biological and chemical succession in which a small number of unusual organisms have adapted to what Baas-Becking termed an environment on "the borderland of physiological possibilities."

The old briners appreciated the intricate coupling of biological and chemical components of these systems, and came to rely on it for the most critical aspect of salina management. Traditionally, they marked the appropriate time to transfer the evaporating water from one pan to the next by the appearance of certain indicator organisms and the death of others (Baas-Becking, 1931b). Now, though more precise measures of the chemical

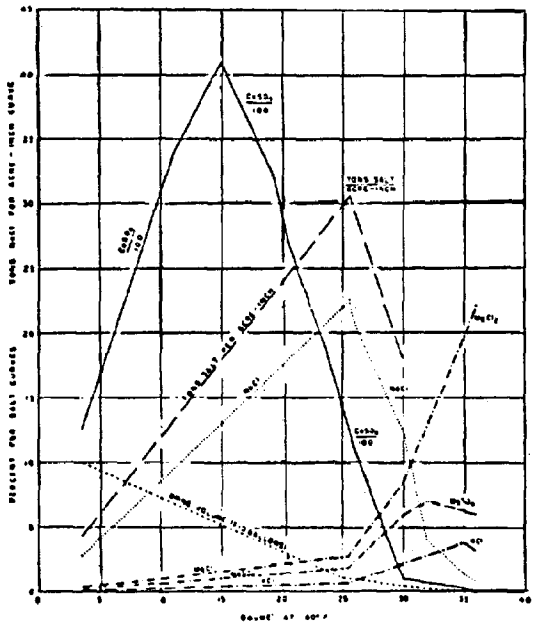

Fig. 4: Ionic succession with increasing salinity as given by VerPlanck (1958).

state of the water are used, such as specific gravity and conductivity, an appreciation is beginning to develop of the good science that was behind the briner's art. Repeated physiological studies of the common brine organisms performed during the last fifty years have shown that these species are very sensitive to the ionic composition of the medium, and that each is confined by this mechanism to a rather characteristic and narrow range in the chemical succession (Baas-Becking, 1931b; Boone and Baas-Becking, 1931; Croghan, 1958a). With this in mind, it would have been possible and perhaps preferable to redraw Fig. 4 in more meaningful biological terms with different species of organisms instead of salinities on the abscissa. Now, after thousands of years, those who design to study these highly stressed salina systems and their compelling questions of chemical and biological coupling may feel a certain kinship with the briners of simpler times, whose intuitive grasp of a fundamental ecological principle begins to emerge from the echoes of their ancient instructions for taking salt from the sea.

CHARACTERISTICS OF THE BRINE LIFE AND MEDIUM

Stress and Diversity in Salina Systems

The characterization of the brine ecosystem as "stressed" implies a certain operational concept of the term and what it means to the organisms involved. Following the ideas developed by Odum (1967b), stress is considered to be the set of energy drains that are required of an organism to remain as part of a particular system. The magnitude of the energy drain or tax is, in effect, the magnitude of the stress. Such a definition reduces many kinds of stress phenomena to a common operational basis which can be evaluated in universal energy units. Special stress drains include not only the obvious metabolic work that may be required, such as ion "pumps" or enzyme systems for active transport, but also the work required to synthesize, maintain, and transport extra cellular organelles or special molecular structures and configurations. Few, if any, such complete evaluations have yet been made for any environment, and certainly not for the brine where several interacting stress factors dominate the biological dimensions of the system. It is only when the energetic drain of environmental stress becomes too large a portion of the organism's total energy budget that its behavior becomes markedly abnormal and the stress becomes obvious. Too great a drain, of course, results in death; but even at lower levels of stress, energy that is switched into doing this kind of work is no longer available for other activities at the species or system level. Thus stressed systems are low in diversity, often inefficient at mineral cycling, and develop other unusual and interesting principles for study. In salina systems, Copeland and Jones (1965) as well as the author (Nixon, 1969) have found less than 6 species per 1000 individuals counted. Finally, energy that is channeled into the work of stress adaptation may be displayed in a variety of morphological, cytological, and biochemical ways, each basically analogous in ultimate function, but each following particular genetic programs that have been produced by the evolutionary histories of the different species.

Osmotic Stress

Organisms living in the brine waters of salina systems are faced with serious problems of osmoregulation. In these hypersaline environments the concentration of dissolved particles is greater in the medium than within the cell cytoplasm of the organisms. Under such conditions, as with the dry air surrounding life in terrestrial deserts, the cellular water tends to diffuse across cell membranes from the higher internal concentration toward the medium. If nothing were done to stop it, this process would continue until the concentration on both sides of the membrane was the same, and there was no free energy difference across the membrane. In order to prevent such cellular dehydration from occurring, the plants and animals of the brine must put a portion of their energy into some kind of osmoregulatory work.

The physical chemistry of the medium acts to modify the energy drain somewhat. As the concentration of total salts increases from the relatively dilute levels of sea water (about 3.5%), the ions of the various salts begin more and more to react together to form ion pairs (Davies, 1962). These ionic pairs then behave osmotically as a single particle of solute, thus lowering the osmotic pressure from the value that would be expected for a complete ionization of the salts. In such highly concentrated solutions of electrolytes, it is this "activity" rather than the total concentration of the salts that is relevant to osmoregulation (Flannery, 1956). The "activity" of NaCl at several concentrations is given in a review by Ingram (1957). This water stress may be met in several ways, as shown in some recent studies of well known brine organisms. In the halophilic bacteria, indications are that requirements for high salt are involved with a biochemical specialization of specific salt requiring enzymes within the bacterial cell (Larsen, 1967). Moreover, the evidence seems to show that in at least several species the salt requirement is specific for Na^+ ions. When these enzymes are isolated from the halophilic bacteria and placed in low salt suspensions, they lose their activity. Results such as these indicate that some brine bacteria may cope with the high osmotic pressure of their environment by adopting it, at least in part, for their own internal composition (Larsen, 1967). Species with more differentiated cytoplasmic structures may develop more involved mechanisms for dealing with the osmoregulatory problem. Marré and his co-workers (Marré et al 1958; Marré and Servettaz, 1958) have postulated an active transport system involving Na^+ for the flagellated brine alga, Dunaliella salina. In this case, Na^+ ions were observed to be taken in across the cell membrane to raise the internal osmotic pressure as much as 25 atmospheres above that of the medium. In such a situation, water will tend to diffuse into the cell and must be pumped out, perhaps through a vacuolar system. It is interesting to note, however, that current work by Johnson et al (1968) on Dunaliella viridis, a sister species of the brine environment that dominates at somewhat lower salinities than D. salina, indicates that the internal salt content of these cells must be below that of the medium at higher salinities, since ATP dependent CO_2 fixation was quickly inhibited in cell extracts at NaCl concentrations above 1.7 molar. The activity of glucose-6-phosphate dehydrogenase extract was completely inhibited at concentrations of 1.28 molar NaCl, while whole intact cells

grew well at 2 molar NaCl. Still more involved means of regulating the internal water balance may be operating in the brine shrimp, Artemia salina. Considerable work by Croghan (1958 a,b,c,d,e) on the osmoregulation and ionic flux in this organism has shown that it maintains a rather constant internal osmotic pressure in the haemolymph and gut fluids across a wide spread of external salt concentrations. This internal osmotic stability is shown in Fig. 5. The haemolymph osmotic pressure is kept below that of the medium, and seems to involve a coupling of active transport mechanisms in the gut and across the first ten pairs of the animal's branchiae, with Na^+ being the ion of major flux. Assuming that active transport was the primary mechanism of osmoregulation in Artemia, Croghan (1958e) calculated a drain of 6% of the organisms' energy budget into this function. However, he points out that even this minimal figure which does not include the energetic cost of synthesizing and maintaining the transport system can not be taken too seriously, since he has presented evidence that other osmoregulatory mechanisms such as the Donnan effect may be operating in ways that are difficult to evaluate. More sophisticated measures of the energetic cost of ionic balance and water transport in these organisms must await more refined data on the various membrane phenomena involved, as well as on the pH of cellular fluids and the isoelectric points of the proteins of hypersaline organisms, and the functional partitioning of cellular molecules and ions. Here, as in so many environments, the rich fields of biochemical ecology and ecological cytology are waiting to be developed. Using the energy flow diagrams developed by Odum (1967c, 1967d), the role of high salt stress as an energy drain can be shown as a general phenomenon as in Fig. 6.

A Strange Chemical Mix - Ionic Stress

As the sea water evaporates, not only does the concentration of total salts increase, but the relative proportions of the ions in solution also change. This constantly shifting chemical composition of the medium places another severe stress on each of the brine organisms. Chemical succession is a consequence of the different solubility characteristics shown in Fig. 4, and can be seen in a different form in Fig. 7, where the amounts of the various major ions in relation to the amount of Cl^- present can be seen changing as water from the Mediterranean Sea evaporates in a costal salina. Redfield (1958) has shown in another context that such ionic ratios are important in selecting and maintaining species whose internal composition is as compatible as possible with the medium, so that as little energy as possible must be put into differential ion uptake, and the use of nutrients is as efficient as possible. In the highly unusual chemical mix of the salina waters, all of the species must do some ionic work to overcome this stress. Data from Pillai (1954, 1955) is shown in Fig. 8, and indicates that blue-green algae from hypersaline lagoons accumulate and maintain an internal chemical composition that is quite different from that in the surrounding water. Studies by Boone and Baas-Becking (1931) and by Baas-Becking (1931b) on Artemia salina, Duneliella viridis, Dunaliella salina, and various blue-green algae, indicate that these organisms, even though they are capable of life in a severely stressed osmotic environment, are very sensitive to the relative proportions of the ions such as K^+, Ca^{++}, Na^+, and Mg^{++} in their medium. This sensitivity may be a function of total

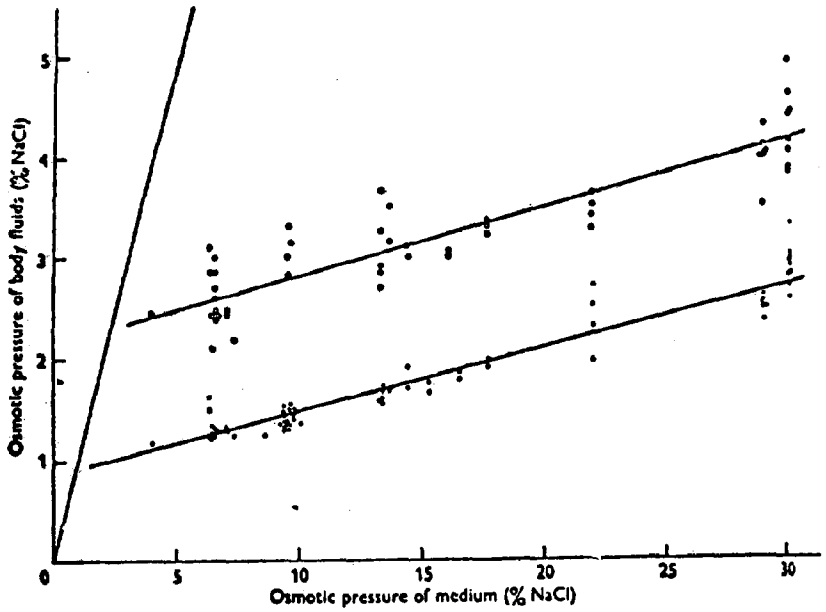

Fig. 5: Relative osmotic stability in Artemia: the graph shows the relation between osmotic pressure of the body fluids and the medium. The open circles represent gut fluids, the solid points represent the haemolymph, and the diagonal line through the origin represents isotonicity. (Croghan, 1958b).

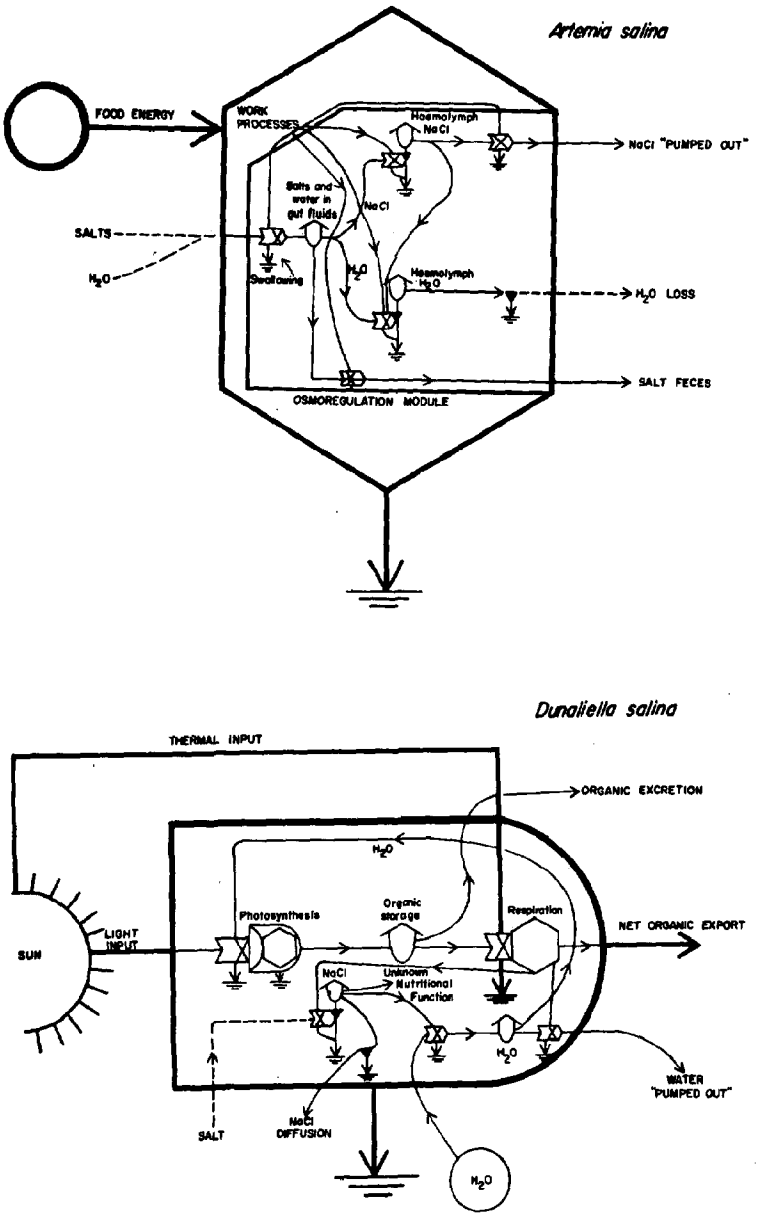

Fig. 6: The energetics of osmoregulation in Artemia salina (top) and Dunaliella salina (bottom). The hypothetical scheme for brine shrimp is based on work by Croghan (1958c,d), and the scheme for Dunaliella salina is based on the data of Marré and Servettaz (1958) and Marré, Servettaz and Albergoni (1958).

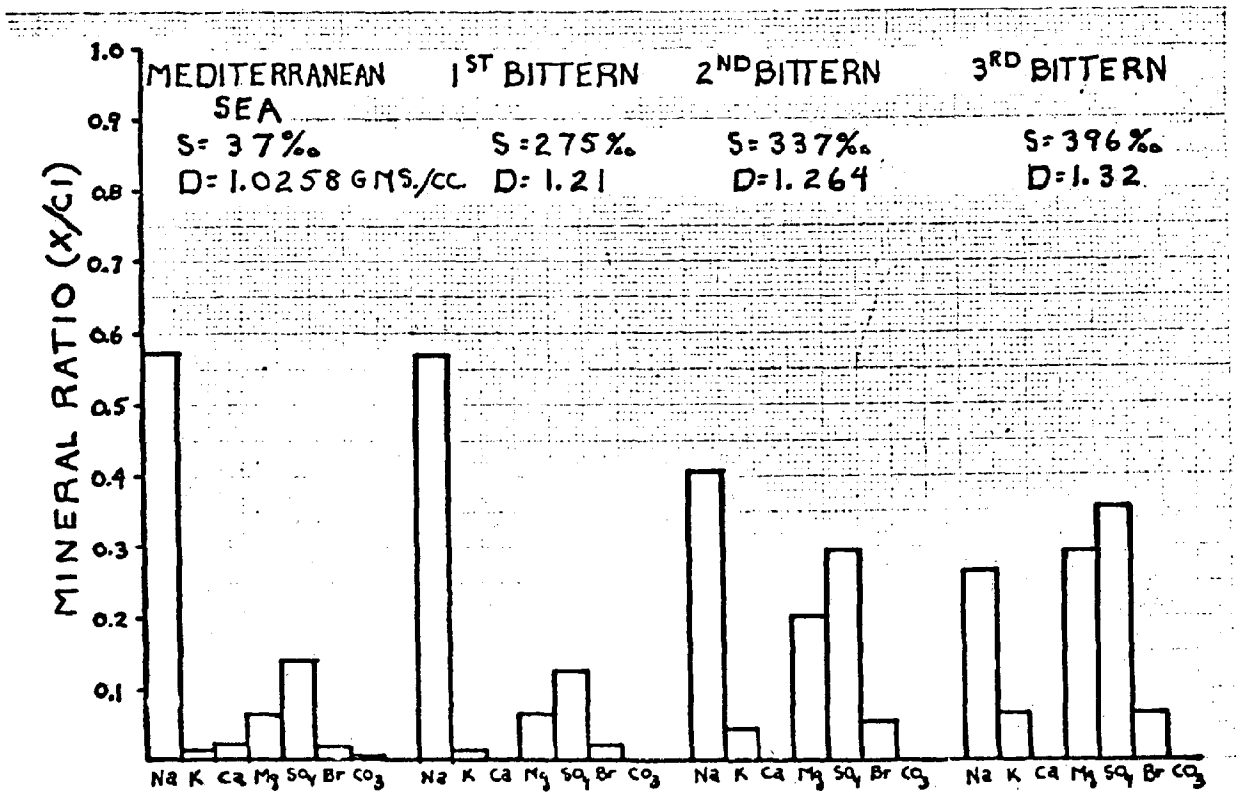

Fig. 7: Changing ion ratios in the waters of a Mediterranean salina. The amount of each mineral (X) has been divided by the amount of chloride to obtain the ratio. Note that calcium is removed early in the process, and that magnesium and potassium become increasingly important, as well as SO_4. Clarke (1924).

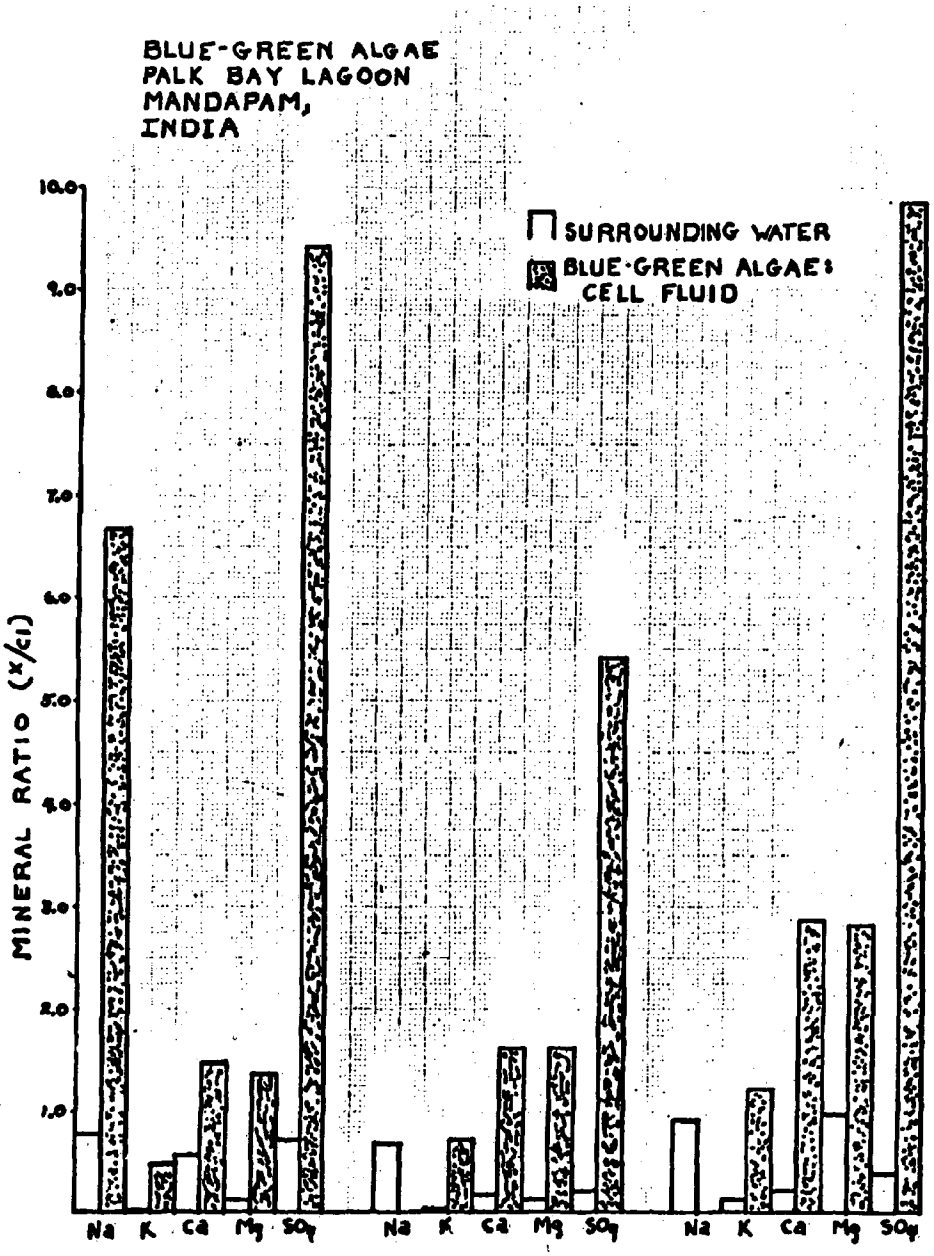

Fig. 8: The ionic composition of some blue-green algae in India contrasted with the ionic composition of their surrounding water at three salt levels increasing from left to right. Ratios were obtained by dividing the amount of each mineral (X) by the amount of chloride present. (Pillai, 1955).

salinity and is further complicated by various sets of antagonistic ion relationships. Baas-Becking and Boone have worked out a chemical matrix of survival in terms of ionic balance and antagonism for some brine organisms. Their results are given in Fig. 9. Baas-Becking relates the toxic effect of Ca^{++} and Mg^{++} on D. viridis to the pronounced negative charge that he reports these cells carry on the outside of their membranes. The Mg^{++} and Ca^{++} are antagonistic, and more Mg^{++} is required to affect the Ca^{++} at higher salinities. Johnson et al (1968) have shown that the Na^{+} requirement in this organism is specific and can not be satisfied by K^{+} or by Mg^{++}. On the other hand, Baas-Becking found that while blue-green algae are tolerant of Ca^{++} at lower salinities, they become sensitive to it at high salt levels, and to Mg^{++} at still higher levels. At 4 molar NaCl, they would grow only when no bivalent metal ions were present. More recent work by Croghan (1958a) has confirmed the sensitivity of Artemia to K^{+} ions, and has shown the mechanism to involve a replacement phenomenon in which K^{+} ions are rapidly substituted for Na^{+} ions in the haemolymph when the potassium concentration in the medium gets high (Croghan, 1958e). His data are also shown in Fig. 9, and indicate that a ratio of Na^{+}/K^{+} must be kept at 10 or above for the long term survival of brine shrimp. As D'Agostino (1965) has demonstrated, however, there are several physiological strains of Artemia, and the particular ionic tolerances may vary in different populations. Cole and Brown (1967), for example, have reported brine shrimp from a few waters having an unusually high K^{+} content for these organisms. The red and pink bacteria of the highest salt pans, those last in the salina series, are also responsive to the ionic composition of the soup in which they must live or die. As pointed out by Ingram (1957) and by Larsen (1967) in recent reviews, these bacteria are particularly sensitive to the amounts of Mg^{++} and K^{+} in the water, as well as Na^{+}, and require relatively large amounts of these ions for good growth.

As one compares these approximate ionic parameters marking the limits of survival for each of the major brine species with the distribution of the species in each of the subsystems along the salinity gradient of the salina, one must agree with Baas-Becking that the concept emerges of a set of coupled microcosms in separate pans, each regulated in species composition by the interaction of salinity and ionic succession. That this concept was a part of the briner's traditional working technology is shown by the fact that he timed the storage and transfer of evaporating water in each pan by the life and death of particular system dominants such as blue-green algal mats, the green alga Dunaliella viridis, the brine shrimp Artemia salina, the red-orange alga Dunaliella salina, and the red and pink halophilic bacteria Halobacterium and Halococci (Baas-Becking, 1931b). In effect, the briner took advantage of the analytical results of a continuous ecological bioassay to control and manage his salina.

Additional Stresses

Oxygen

The high salt levels of the brine systems act to decrease the amount of oxygen that can dissolve in the water, and force different species to channel larger amounts of energy into gas exchange through the synthesis of

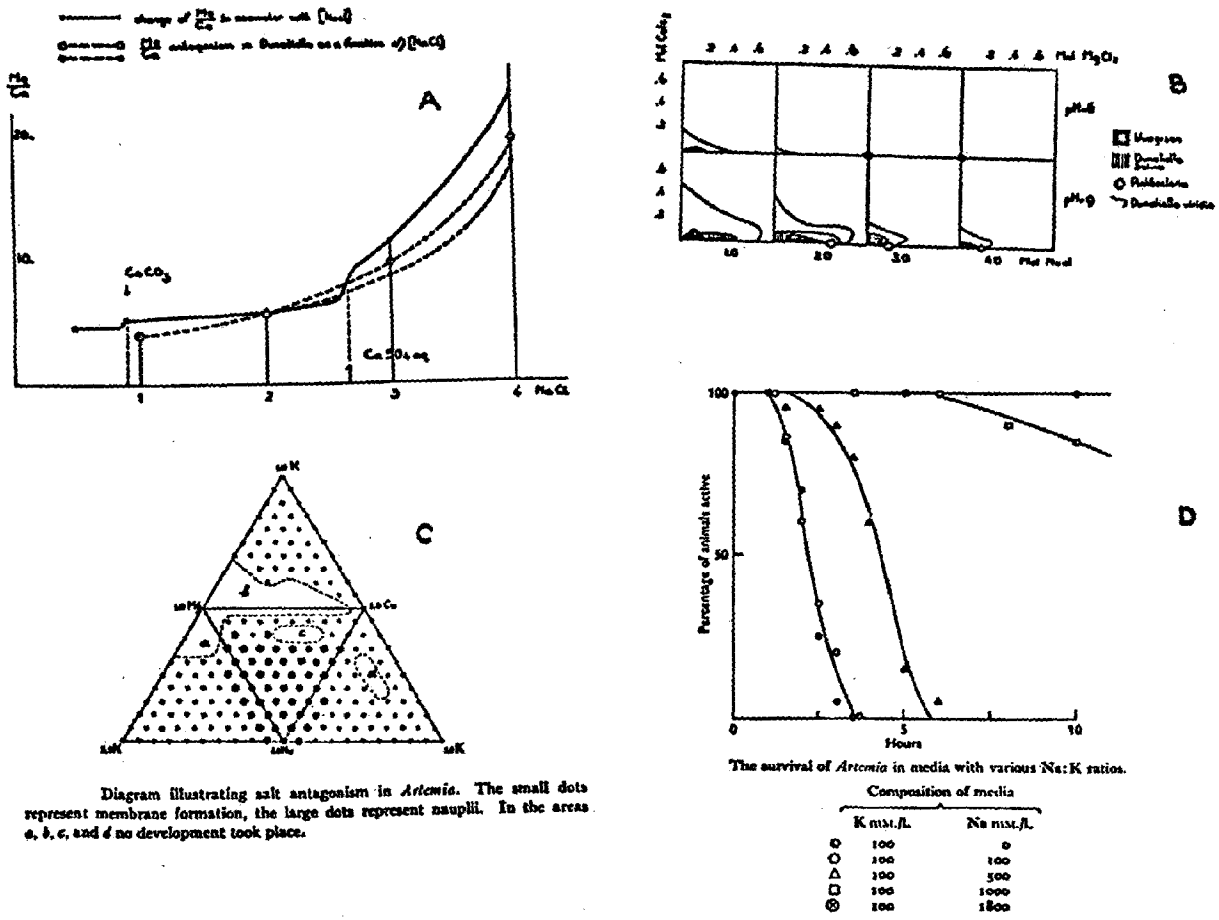

Diagram illustrating salt antagonism in *Artemia*. The small dots represent membrane formation, the large dots represent nauplii. In the areas *a*, *b*, *c*, and *d* no development took place.

The survival of *Artemia* in media with various Na:K ratios.

Fig. 9. Ion antagonisms and salt effects in some common salina species; Figs. 9A and B are from Baas-Becking (1931b), Fig. 9C is from Boone and Baas-Becking (1931), and Fig. 9D is from Croghan (1958a).

increasing amounts of respiratory pigments such as hemoglobin in the brine shrimp (Gilchrist, 1954). The decreasing solubility of oxygen in sea water brine is given by Copeland (1967b) in Fig. 10.

Temperature

As shown by Adams (1934) and by Harbeck (1955), the effect of increasing salinity is to decrease the rate of evaporation and to raise the temperature of the salt solution over that of pure water under the same equilibrium conditions. For this reason, as well as their shallow depth, salina systems undergo relatively wide and rapid diurnal cycles of temperature that are not usually characteristic of aquatic systems. This responsiveness of the water to the high daytime temperatures of areas suitable for large scale evaporation also decreases the solubility of oxygen in water and aggravates the salt effect mentioned above (Copeland, 1967b).

Pulsing Inflows

The pulsing inflows in salina management also place stress on the living systems in the brine pans. By transferring water from pan to pan, each subsystem is faced with changes in total salinity and ion ratios, as well as with sudden changes in population sizes and species composition as the organisms are swept in and out with the transferred water. And after each transfer, each subsystem must again move through a short succession, only to be arrested once more. Of course, even if man did not intervene by his management, each subsystem would be destined to remain only a small span in the overall dominating chemical succession of changing ions and increasing salt.

pH and Alkalinity

Copeland (1967b) has followed the changing pH and alkalinity of evaporating sea water and obtained the results shown in Fig. 11. Such oscillating changes may be of some significance, particularly in the range of pH 8 - 9, where the HCO_3^- and CO_3^{--} concentrations are changing rather rapidly, and the water is shifting from a system dominated by bicarbonate to one richer in carbonate. The actual biological consequences of such changes are still not clear, however, and some work such as that by Gibore (1956) indicates that brine algae may not be particularly sensitive to pH changes in this range. Older work, such as the classic study of a salina in France by Labbé (1924) has emphasized the role of pH in regulating the different species along the salinity gradient, with special emphasis on the brine ciliate, Fabrea salina. Labbé's curve of pH and species distribution in the salina is given in Fig. 12.

Adaptations to Stress

Whole systems as well as organisms must partition their flows of energy in various ways to meet the work and maintenance demands of stress environments. These drains of energy into work functions may take various forms and produce interesting and novel events for study.

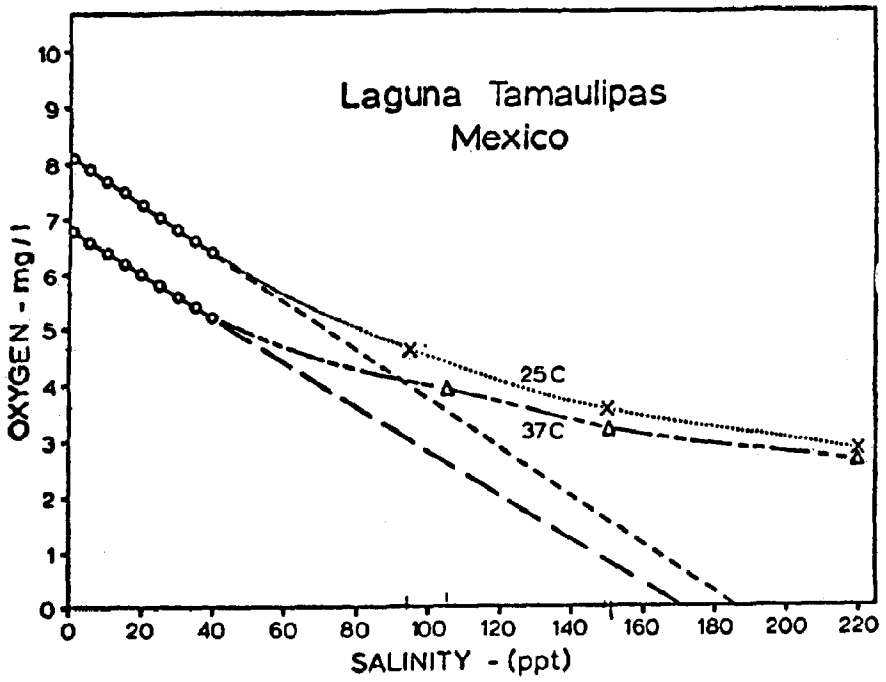

Fig. 10: "Saturation values of dissolved oxygen versus salinity. The open circles indicate 100% saturation in mg./l, using the solubility constants of Truesdale et al. (1955), at 25 and 37C. The straight lines indicate theoretical saturation when the data at lower salinities are extrapolated. The open triangles and crosses indicate 100% saturation as measured in water from the Laguna Tamaulipas, Mexico, at various salinities." (Copeland, 1967b.)

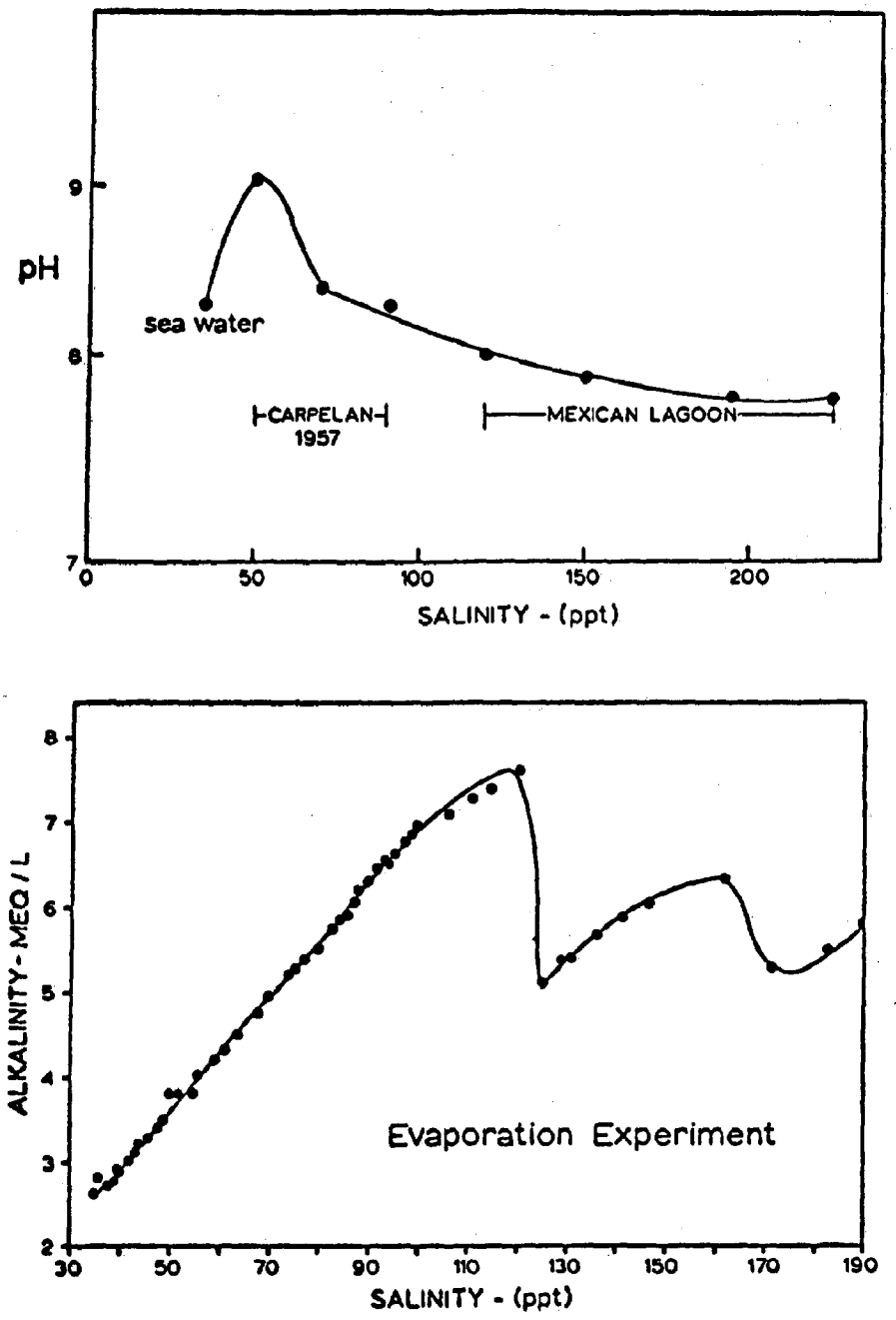

Fig. 11: Changing pH and alkalinity of evaporating sea water. (Copeland, 1967b.)

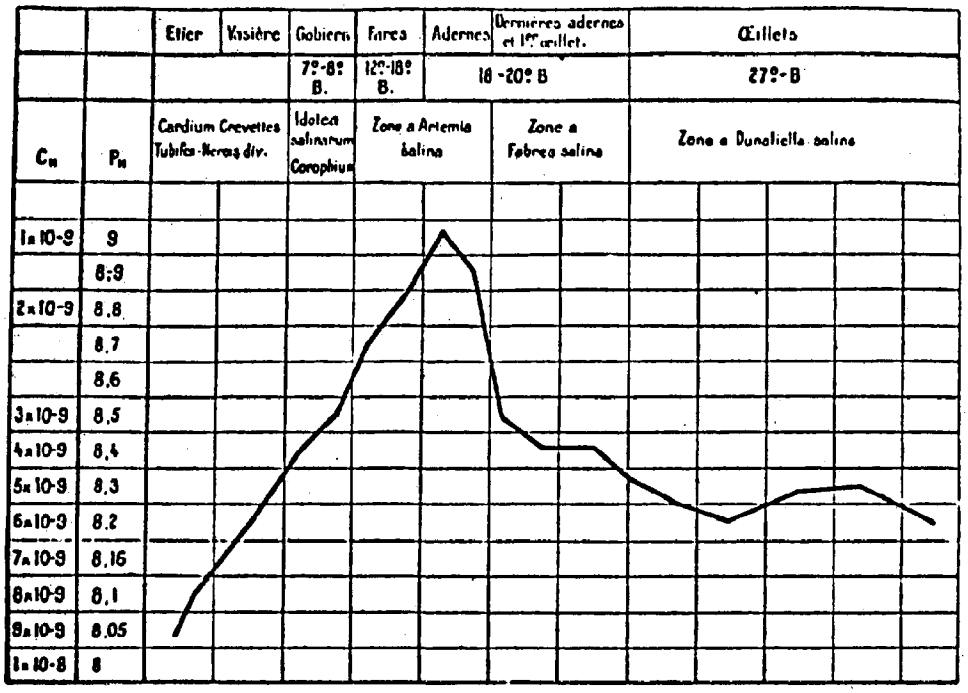

Fig. 12: pH and dominant species along a salinity gradient in a large French salina. (Labbé, 1924).

Some better known and rather striking responses to salt stress at the organismal level have been described by Gilchrist for Artemia, including an increasing amount of hemoglobin synthesis (1954), a decrease in the length of females and in the number of females bearing eggs, and a decrease in the number of eggs carried per female at high salinities (1956). In a similar manner, Villeneuve-Brachon (1940) observed that the red-brown pigment of the brine ciliate Fabrea salina also increased with salinity. Current studies on the biochemical development of Artemia by Warner and McClean (1968) show additional idiosyncrasies that may develop as a consequence of an organism being genetically programmed for stress adaptation. Their results indicate that the brine shrimp is not capable of the de novo synthesis of the purines, adenine and guanine. These nucleotides are usually formed de novo through a biosynthesis of some five simple compounds to form the purine ring structure, with adenine being formed first and then converted to guanine (West et al, 1966). It is interesting, though perhaps of no real significance in this connection, to note that the heat of formation of adenine is considerably higher than that given by Morowitz (1968) for the pyrimidines or some thirty other common low molecular weight constituents of living material. Can this be an example of an energetic luxury being given up under stress? Artemia must obtain at least the basic purine structure through its food chain or, perhaps, through the organic soup of its medium, since it has been shown by Warner and McClean that once the purine ring is provided, the brine shrimp can complete the synthesis.

The reference to the brine medium as an organic soup was prompted by the work of Wilson (1963), who found that the levels of dissolved organic carbon in brines are higher than for any other natural water measured, in some cases exceeding 100 p.p.m. Perhaps, as Odum (1967b) has pointed out, this is indicative of simple, stressed systems where species diversity is low and the heterotrophic consumption of stored organic matter is incomplete. Perhaps it is also of some adaptive value to organisms who must put large amounts of energy into osmoregulation and ionic balance to excrete large amounts of organics into external storage. Such material could always be used later as organic fuel, while sparing the organisms the work of internal storage and transport. One is reminded of the vast differences in the complexity and the energy required for development and maintenance between conventional aircraft that must store and carry only one fuel component, the petroleum based oxidant, and the early rocket engines designed for operation in space that had to carry and store both oxidant and the oxidizer. Perhaps future research will center on the nature of these external biological fuels stored by stressed ecological systems and their role in the chemical control and coordination of living components in the system.

At the highest salt levels, where osmotic stress is greatest and the ionic ratios most aberrant, metabolic energy is put into the synthesis of large amounts of auxiliary pigments, and the red and pink Halobacterium and Halococci, along with the red-orange alga, Dunaliella salina, become the dominant organisms. From above, they color whole evaporating pans a bright red, like squares on a giant's checkerboard. The color is due to an abundance of carotenoid pigments, and one wonders why organisms that must put so much work into meeting other stresses would switch so much of their

remaining energy into carotenoid biosynthesis (Fox and Sargent, 1938; Flannery, 1956). In both the bacteria and the algae, the function of these carotenoids has been attributed to the protection of the bacteriochlorophyll and the chlorophyll from photo-oxidation under very high light intensities (Dundas and Larsen, 1962, 1963). But other functions for the organisms and the system as a whole may be served as well. It is interesting to remember the red color of snow algae (Marré, 1962), and to speculate on the role of these pigments as good receptors in the infared regions of the spectrum. Tiffany (1938) has mentioned that their presence in snow algae greatly accelerates melting, and it is not difficult to imagine that such a mechanism of intracellular heating could be necessary to the metabolic kinetics of an alga growing at very low temperatures. It may also be beneficial to an alga or bacterium living in a medium of low nutrients and high stress, where organic storages are large and metabolic demands for osmotic and biosynthetic work may favor increased respiration rates and more rapid turnovers of important nutrients. The simple explanation of light protection also ignores the bright green colors of Dunaliella viridis and the blue-green algae which grow under the same bright light conditions but at lower salinities and survive quite well without putting energy into the synthesis of large amounts of carotenoids. It does not seem unreasonable to suppose that systems under severe stress will evolve mechanisms to use as much of the energy that is available to the system as possible.

As mentioned earlier, brine systems are characterized by having only a small number of species present. This low species diversity means that the relatively small amount of energy that remains in the system after the adaptive drains for work and maintenance must be partitioned among fewer compartments as it flows through the system. Odum (1967b) has speculated that the species that are included in such stressed systems may be nutritional generalists, and that "without specialization, the total consumer action is inefficient." Such inefficient consumption leads to an organic accumulation, and a series of possible substrates for the largely heterotrophic subsystems or red bacteria at very high salt levels. Lower species diversity may also make these systems less stable and more sensitive to stresses and shocks for which they are not preadapted.

OVERALL METABOLIC PATTERNS OF SALINA SYSTEMS

Questions of the overall productivity and the diurnal patterns of photosynthesis and respiration in the stressed systems of the salina have been studied by Carpelan (1957) and by Copeland and Jones (1965). Beyers (1965) has published data on the metabolic patterns of brine systems in small laboratory microcosms. Carpelan's work emphasized the lower salinity pans early in the evaporating series where large amounts of energy did not have to be put into stress adaptation and productivity was high. His results indicate an average production of about 700 gms. carbon/M^2/Yr. in the lower salinity pans where the major nutrients were concentrated by evaporation, and the systems behaved as if fertilized. At the highest salinity studied, about 15%, he found the production of brine shrimp to be approximately 6.3 gms. dry weight/M^2/Yr. Using a crude estimate of 5.5 Kcal./gm. dry weight of animal

tissue (Morowitz, 1968), this would mean a secondary production of 34.6 Kcal./M^2/Yr. of Artemia. The salina in this study was located in California, and Carpelan found a strong seasonal pulse of algal production following the increasing inputs of solar energy in spring and summer. Copeland and Jones studied a salina in Puerto Rico, and obtained data on diurnal changes in dissolved oxygen and carbon dioxide in systems operating at high concentrations of 15, 19, and 22% salt. They were able to show that productivity at these salinities was very low, with P/R ratios remaining less than 1.0 in each pan. Their curves and data are given in Fig. 13. The author's unpublished data on laboratory brine systems at about 15% total salt is in agreement with these values, though there is evidence that such systems, when enriched with inorganic fertilizer, can become quite productive and maintain themselves at higher metabolic levels for long periods of time. Beyers found P/R ratios of 1.12 in his unfertilized brine microcosms at 18.8% salt. Perhaps in nature the thick blue-green algal mats so characteristic of the first subsystems in the salina act as nutrient sinks by effectively taking up and holding nutrients in their stored organic matter, and leaving the water that is to be transferred on through the system low in nutrients. Such algal mats are cultivated by the briners because they help to bind the earth bottoms of the pans and reduce water loss through seepage. Later, when the pans are used as crystalizing beds, the old algal mat helps to separate the white salt from the black anaerobic bottom muds during harvest (Hof and Fremy, 1933). The dark color of the algae also inhances the thermal absorption in the pans and increases evaporation.

Unfortunately, no clear picture has yet emerged of the details of the metabolic relationships and patterns in salina systems, though the available data suggest that the later subsystems at the highest levels of salt stress are predominately heterotrophic systems based on the infrared photo-stimulated oxidation of organic fuels produced by photosynthesis in the earlier subsystems where energetic flows for stress adaptation remain relatively small. This relationship is shown in the energy flow diagram of Fig. 14, a reflection of the present state of knowledge concerning these systems. Future research will have to be directed toward obtaining quantitative values for the principal flows and storages.

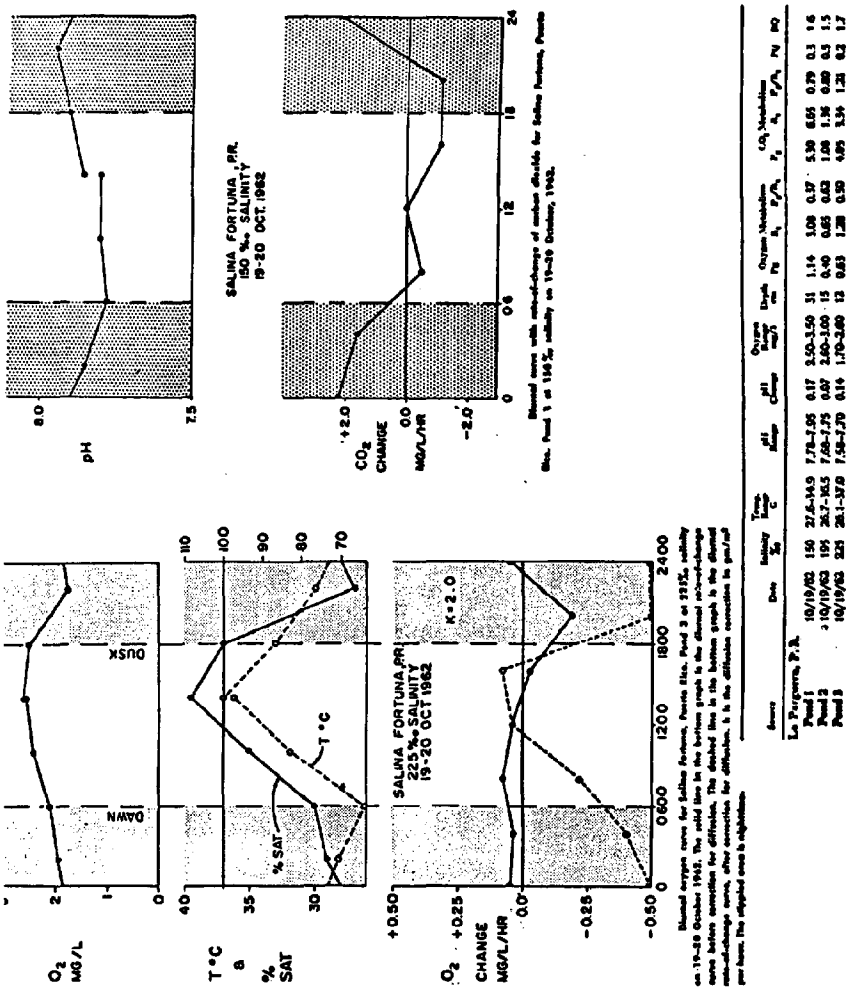

Fig. 13: Diurnal patterns of oxygen and carbon dioxide metabolism in a small saline in Puerto Rico. (Copeland and Jones, 1965).

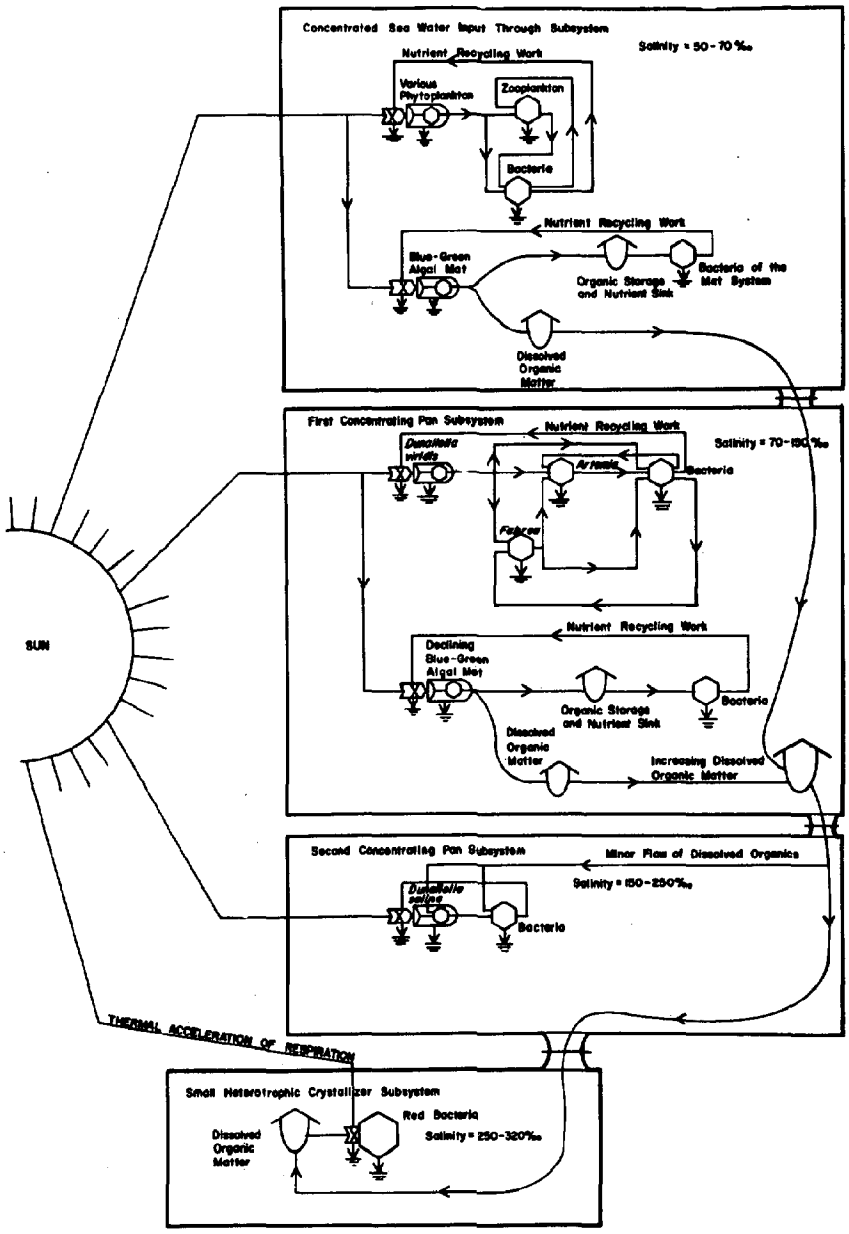

Fig. 14: Major pathways of energy flow through the salina system, showing principal species and storage compartments as well as work loops and flows. Later subsystems at higher salt levels become increasingly heterotrophic and depend on the organic input of carbon fixed in earlier less stressed subsystems. (See Fig. 1).

Chapter E-14

BRINE POLLUTION SYSTEM

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INTRODUCTION

From desalinization plants and oil field operations come large quantities of highly concentrated salt water. Not only is the salt in high concentrations, but the ratio of salts is quite different from that of sea water. The energetics of the receiving systems are modified with the resulting changes in circulation systems, osmotic regulation, stratification, destruction of bottom communities and soil structure, specific heat, hydrogen ion balance, buffer system modification, low species diversity, solubility of oxygen, turbidity and ion balance.

From studies concerning salinas, natural hypersaline lagoons and the cases of brine polluted communities, a picture is developing about the new emerging systems characteristic of brine pollution. Very few organisms are capable of adapting to the high salinities, strange chemical balance and varying inputs of brine waters.

System Example - Chiltipin Creek - Upper Copano Bay, Texas

An example of a system dominated by brine pollution is found in upper Copano Bay on the central coast of Texas. Chiltipin Creek, which serves as a drainage canal for extensive oil field activities, empties into upper Copano Bay and carries considerable brine from oil wells. The type of oil field operations includes the pumping of brine water and oil from shallow wells and allowing the oil to float off in settling ponds and draining the brine to Chiltipin Creek (Teller, 1961). The ratio of brine to oil is about 50 - 100 to 1, resulting in about 8.6 million gallons of brine per day drained into Copano Bay (Teller, 1961). A diagram of the Chiltipin Creek-Upper Copano Bay area is shown in Figure 1, in which the sampling stations are indicated by circled x's. The bay area is shallow (less than one meter in depth) and the wind-driven currents insure almost complete mixing. In addition to the brine being deposited each day, considerable oil floats into Copano Bay from escapes from the settling ponds at oil wells.

Preliminary data not previously published from a years study of the upper Copano Bay has been made available here from an unpublished manuscript (Copeland and Moseley).

Brine Characteristics

Whether the brine water comes from oil field operations or desalinization plants, the chemical characteristics are similar; i.e., they result from the evaporation of sea water. The chemical characteristics of sea water (the source

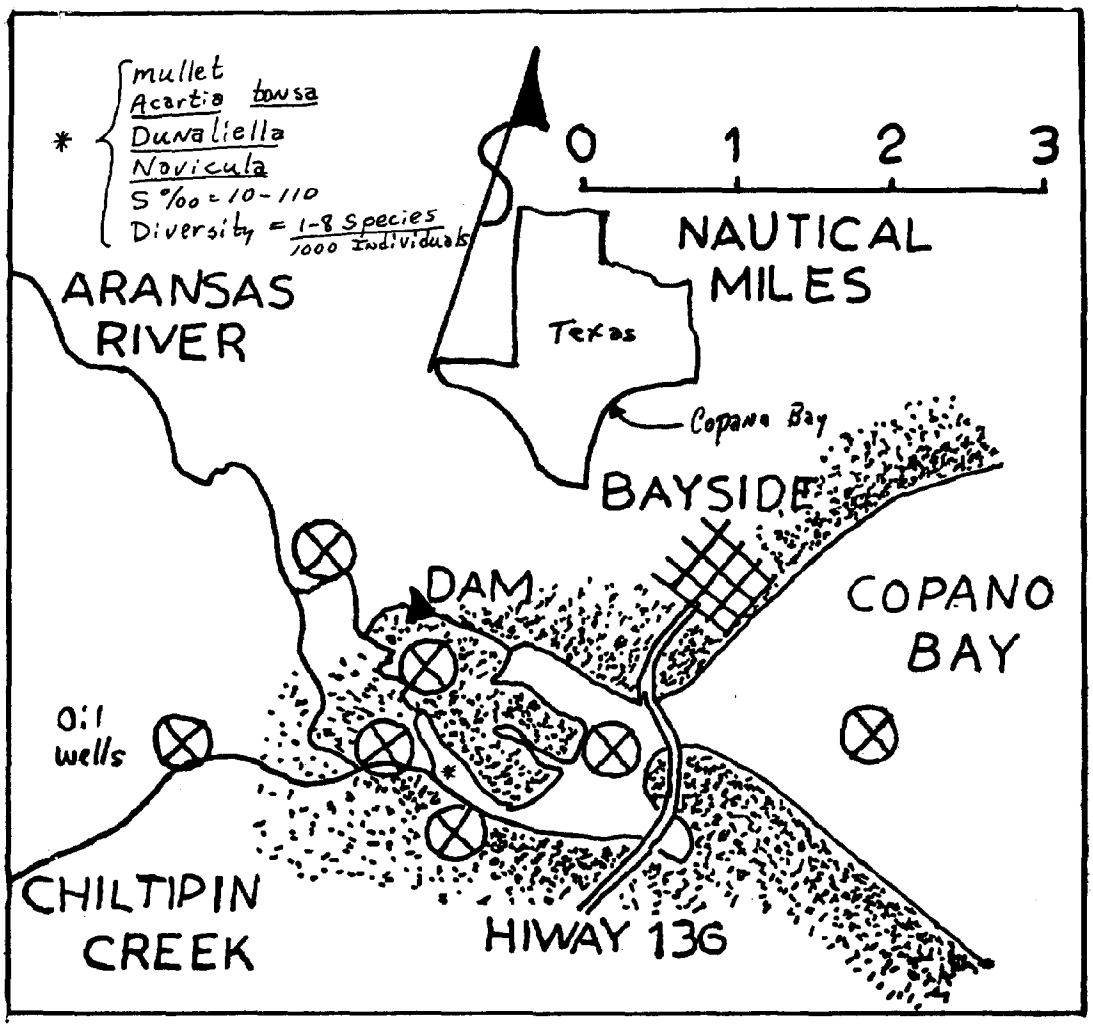

Fig. 1. Upper Copano Bay, Texas and Chiltipin Creek (brine source) with sampling stations indicated. (From Copeland and Moseley). Asterisk (*) indicates representative community location.

of brine) are amazingly constant in all parts of the world (Harvey,1957),but during evaporation there is differential precipitation of salts (Copeland,1967b) so that a chemically unbalanced solution remains. Therefore, the characteristics of a brine waste would be a function of the completeness of evaporation processes.

A. G. Collins (1967) discusses the chemical characteristics of Cretaceous and Tertiary brines in Mississippi and Alabama. The results of a chemical analysis of four types of brine waste and that of normal sea water are shown in Table 1. Some brines from deep formations contain more bromide than can be concentrated during evaporite formation, but it can be accounted for with association with bromide concentrators such as seaweeds, corals, etc. Most brines resulting from oil field operations on the Gulf coast are of types 4 and 5 in Table 1.

The correlation coefficient for several ions was relatively high (Table 2). High correlations were found in the Tertiary waters between Cl and Total Dissolved Solids (T.D.S.), Cl and Na, Na and T.D.S., Br and Ca, and Ca and depth. Cretaceous waters exhibited high correlations between Cl and T.D.S., Ca and Cl, Na and T.D.S., Ca and T.D.S., Br and T.D.S., Na and Cl, Ca and Br, Ca and depth, and Cl and Br.

A. G. Collins (1967) concluded that oil field waters obtained from marine rocks usually are relict sea water. Their salinity ranges from 35 parts per thousand to more than 300 ppt. As sea water evaporates, the carbonates precipitate first, followed by the sulfates (Copeland,1967b; A. G. Collins,1967). Little or no bromide precipitates or is occluded with these. The increased chloride concentration can be attributed to evaporation. Magnesium depletion and calcium enrichment probably result primarily from dolomitization. Bicarbonate depletion is the result of carbonate precipitation. Sulfate depletion is the result of gypsum precipitation and/or bacterial action. Thus, evaporated sea water, whether it is from oil fields or desalinization operations, yields a brine of strange chemical characteristics. A diagram showing the differential precipitation of evaporating sea water (from Copeland, 1967b) is shown in Figure 2. As the sea water evaporates, the certain salts precipitate to the sediments and leave a solution of imbalanced ions.

SYSTEM CHARACTERISTICS

Salinity

Because systems dominated by brine water pollution are receiving large quantities of evaporated sea water, the salinity of the receiving system is usually relatively high. As shown in Figure 3, the average salinity for the Chiltipin Creek-Upper Copano Bay area was generally higher than normal estuarine salinity, ranging from 10 to more than 60 parts per thousand (ppt). Even the tremendous flooding resulting from Hurricane Beulah on the Texas coast during September 1967 did not appreciably lower the salinity in the area after the flooding had subsided, whereas the coastal salinities elsewhere along the central and southern Texas coast were almost fresh (personal observation). It should be noted that individual measurements at the mouth of Chiltipin Creek revealed salinities as high as 110 ppt.

Table 1. Elemental analysis of four types of brine and normal sea water. (From A. G. Collins, 1967; Table 1).

No.	Concentrations in Mg. per Liter							Type
	Na	Ca	Mg	Cl	Br	SO_4	HCO_3	
1	1,600	40	40	820	10	340	2800	HCO_3-Na
2	1,100	280	75	560	8	2200	420	SO_4-Na
3	10,600	900	1300	19,000	65	3900	160	Sea water
4	23,000	1,000	4000	36,000	118	1000	80	Cl-Mg
5	18,700	120,000	9000	270,000	3200	0	0	Cl-Ca

Table 2. Correlation coefficients for all samples between various elements (From A. G. Collins, 1967; Table 2).

	Depth	T.D.S.	Na	Ca	Mg	Cl	Br	I	HCO_3	SO_4
Depth	1.000	0.793	0.473	0.767	0.477	0.763	0.631	0.562	−0.387	0.329
T.D.S.	0.793	1.000	0.771	0.781	0.566	0.965	0.704	0.279	−0.402	0.277
Na	0.473	0.771	1.000	0.219	0.151	0.727	0.187	−0.026	−0.046	0.184
Ca	0.767	0.781	0.219	1.000	0.657	0.760	0.890	0.474	−0.560	0.245
Mg	0.477	0.566	0.151	0.657	1.000	0.571	0.724	0.256	−0.464	0.089
Cl	0.763	0.965	0.727	0.760	0.571	1.000	0.681	0.271	−0.404	0.261
Br	0.631	0.704	0.187	0.890	0.724	0.681	1.000	0.435	−0.568	0.209
I	0.562	0.279	−0.026	0.474	0.256	0.271	0.435	1.000	−0.343	0.217
HCO_3	−0.387	−0.402	−0.046	−0.560	−0.464	−0.404	−0.568	−0.343	1.000	−0.004
SO_4	0.329	0.277	0.184	0.245	0.089	0.261	0.209	0.217	−0.004	1.000

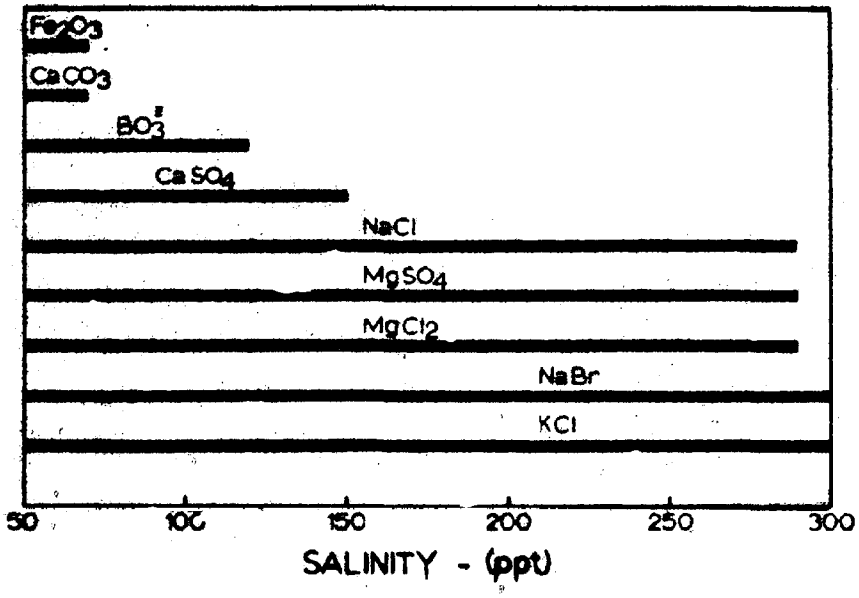

Fig. 2. Precipitation of salts during evaporation of sea water. The end of the bar indicates the salinity at which precipitation begins. (From Copeland 1967b; Fig. 3).

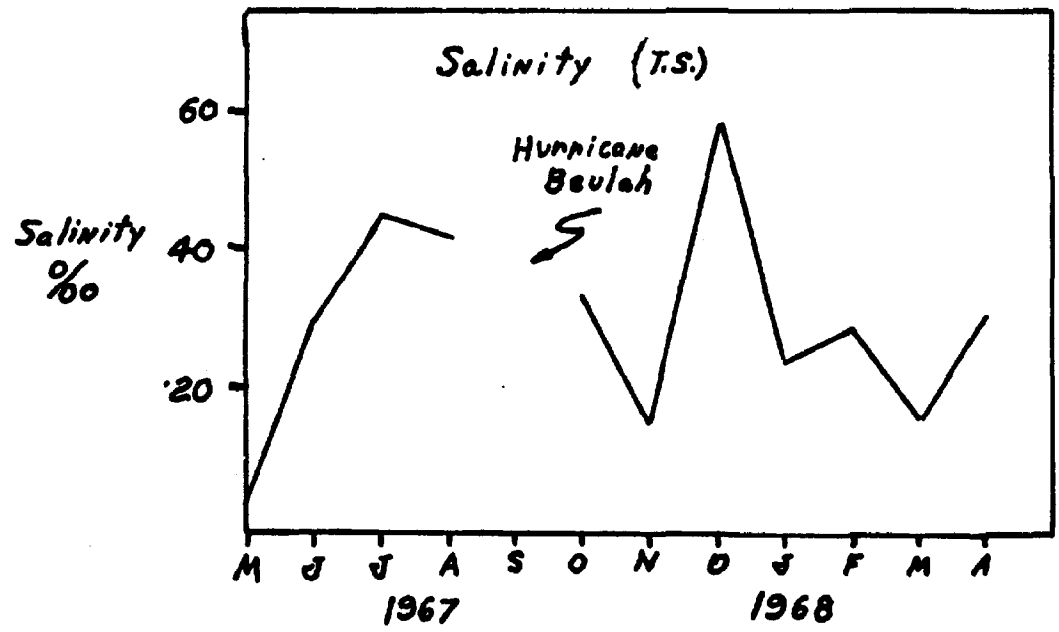

Fig. 3. Average salinity (total solids) for Chiltipin Creek-Upper Copano Bay brine dominated system for 1967-68 (Copeland and Moseley).

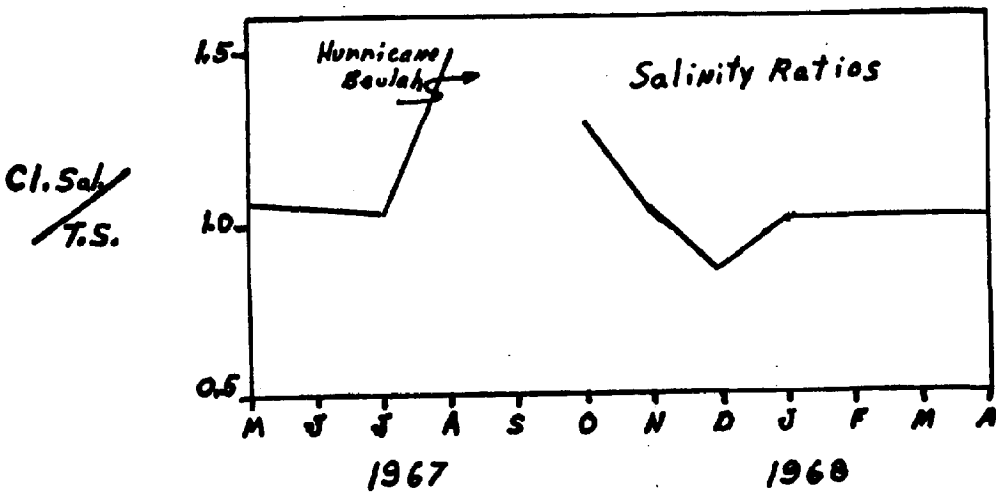

Fig. 4. Ratio of salinity as determined by chloride titration to total solids in Chiltipin Creek-Upper Copano Bay brine dominated system for 1967-68 (Copeland and Moseley).

Renfro (1960) reported salinity as high as 60 ppt in the Aransas River, nine miles from its point of confluence with Chiltipin Creek. He noted that appreciable salt water intrusion occurred in the Aransas River from the Chiltipin Creek brine source during times of normal rainfall on the Aransas River watershed. As indicated in both Figure 3 and by Renfro's (1960) data, the salinity of the system is erratic. Thus, the organisms living in the brine dominated system are subjected to both high salinity stress and high salinity shocks as masses of highly concentrated brine move into the area.

The ratio of salinity as determined by chloride titration to total dissolved solids (T.D.S.) is a means by which normality of the brine solution can be determined. That ratio for Chiltipin Creek - Upper Copano Bay is shown in Figure 4. On two sampling dates the ratio was significantly higher than one, indicating a dominance of chloride in the system. This is in contrast to the findings of A.G. Collins (1967) who found that there was a one-to-one ratio of Cl to T.D.S. in the 132 samples of brine that he measured. The anomalous ratios in Chiltipin Creek mean , however, that the organisms are subjected to anomalous ion ratios in the brine system in addition to the salinity shocks. This substantiates the findings of Copeland (1967b) for evaporating lagoon waters in the Mexican Laguna Madre, where species diversity was extremely low.

Community Structure

Apparently due to the stresses resulting from salinity shocks, anomalous ion ratios, strange buffer systems, high pH and low oxygen solubility, few organisms are capable of adapting to brine dominated systems. The primary producers in the Chiltipin Creek - Upper Copano Bay system are composed mainly of Dunaliella sp. and a diatom Navicula sp. However, as indicated in Figure 5, the chlorophyll concentration in the system indicates fairly large populations of these apparently highly adaptable species.

The zooplankton diversity was relatively low, with Acartia tonsa present in all samples. Diversity and population density of zooplankton are shown in Figure 6, which is an average of four samples for each sampling date. During the June - July 1967 and the June 1968 population peaks the dominant zooplankton form was a rotifer, which made up about 90% of the zooplankton population during these times. The rotifer was not present during the other months.

The number of species and the number of individuals of fishes were also low in the Chiltipin Creek - Upper Copano Bay system (Tables 3 and 4). During most of the annual cycle, only occasional fishes were collected although sampling was practiced during each month. Renfro (1960) collected only four species of fish at the higher salinities in the lower sector of the Aransas River adjacent to the Chiltipin Creek area (Table 3). All of the species collected by Renfro and us were plankton feeders, indicating an adaptation of the system for utilization of the plankton production. Significant populations of fishes existed in Chiltipin Creek area only during July - August in our 1967-68 sampling program (Table 4). All species shown in Table 4 correspond with those caught by Renfro (1960) in the lower Aransas River at the higher salinities (Table 3).

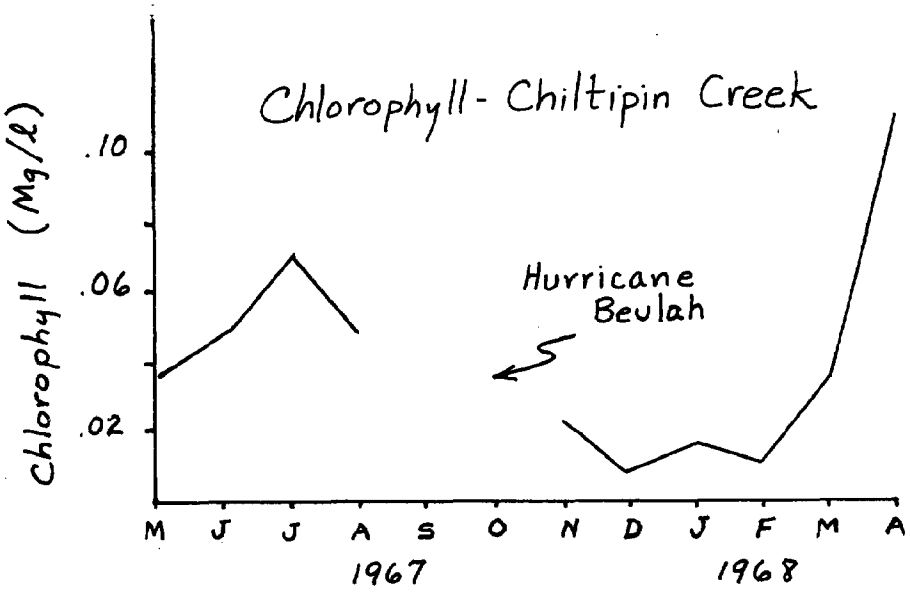

Fig. 5. Chlorophyll a for two stations in Chiltipin Creek-Upper Copano Bay brine dominated system. Each point represents an average of four samples (From Copeland and Moseley, unpublished data).

Table 3. Collections of fishes in the lower Aransas River, Texas by salinity ranges (From Renfro 1960; Table 1).

Salinity ‰ from: to :	.05 1.0	1.1 5.0	5.1 15.0	15.1 25.0	25.1 35.0	35.1 45.0	45.1 55.0	55.1 58.6
Collections in this salinity range	10	2	3	2	1	5	7	2
Lepisosteus spatula	1	—	1	—	—	1	—	—
Brevoortia patronus	3	—	—	—	—	—	2	—
Dorosoma cepedianum	47	5	6	—	—	2	—	—
Dorosoma petenense	32	—	1	—	—	—	—	—
Anchoa mitchilli	114	79	35	—	—	—	—	—
Notropis lutrensis	3	—	—	—	—	—	—	—
Ictalurus furcatus	1	—	1	—	—	—	—	—
Syngnathus scovelli	1	—	—	143	—	21	—	—
Fundulus grandis	8	—	11	2	—	4	17	2
Lucania parva	45	—	—	3	6	19	1	—
Cyprinodon variegatus	144	12	46	28	11	173	161	13
Gambusia affinis	22	6	1	4	—	—	—	—
Mollienisia latipinna	13	—	2	61	1	129	1	—
Menidia beryllina	285	124	75	302	484	137	77	7
Mugil cephalus	10	1	18	57	9	30	31	4
Mugil curema	1	—	—	—	—	—	—	—
Micropterus salmoides	—	—	—	1	—	—	—	—
Chaenobryttus gulosus	—	—	—	2	—	—	—	—
Lepomis macrochirus	41	8	—	6	—	—	—	—
Lepomis megalotis	4	—	—	—	—	—	—	—
Bairdiella chrysura	—	2	—	—	—	—	—	—
Eucinostomus argenteus	—	—	3	—	—	—	—	—
Cichlasoma cyanoguttatum	7	1	—	4	—	—	—	—
Dormitator maculatus	—	—	—	—	—	—	1	—
Gobiosoma bosci	1	—	—	4	—	—	—	—
Trinectes maculatus	1	1	1	—	—	—	—	—

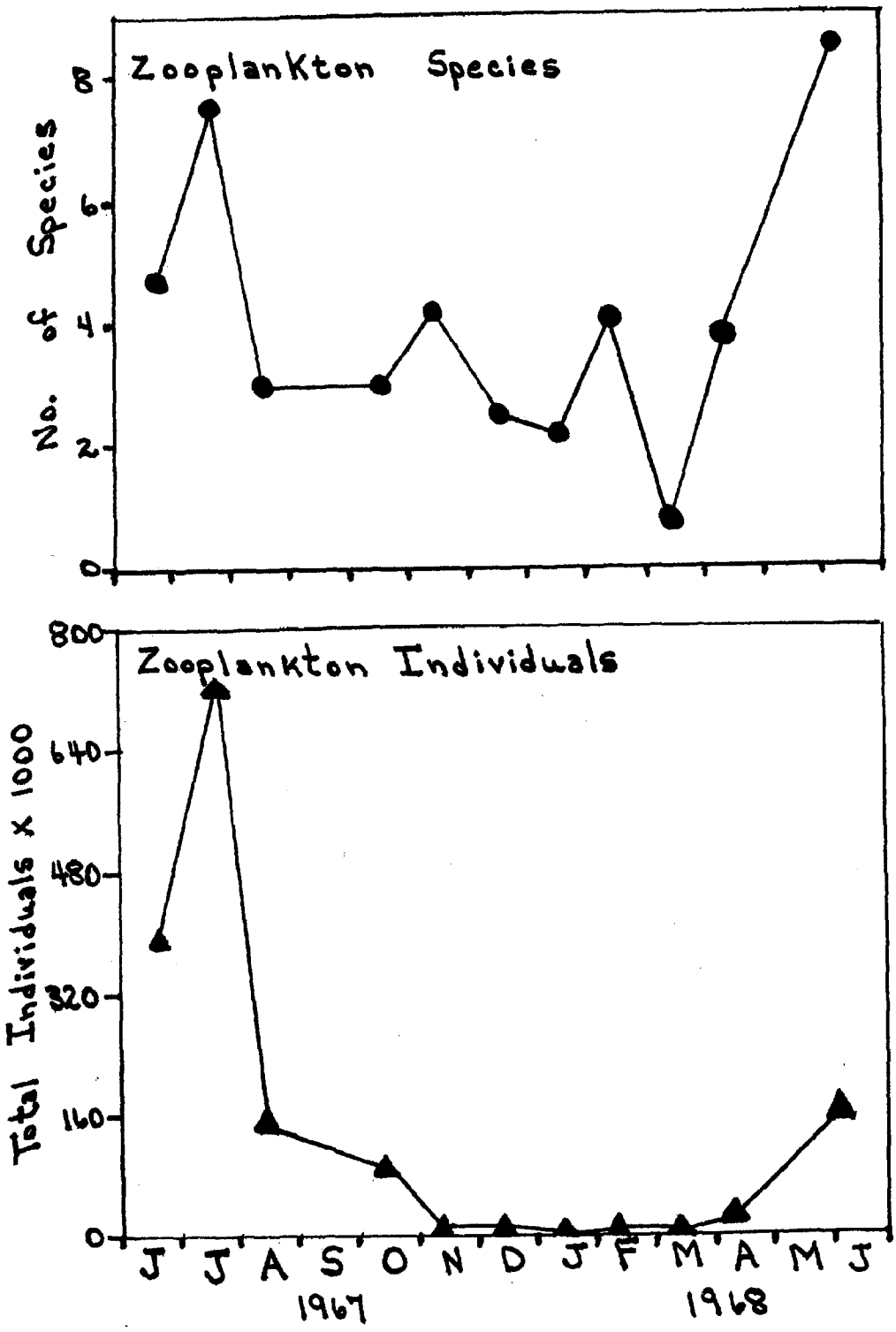

Fig. 6. Zooplankton diversity (upper graph) and total individuals per m^3 (lower graph) for two stations in the Chiltipin Creek-Upper Copano Bay brine dominated system (From Copeland and Moseley, unpublished data).

Table 4. Total species and number of individuals per m^2 at two stations in the Chiltipin Creek-Upper Copano Bay brine dominated system during July 1967 to May 1968 (Copeland and Moseley, unpublished data).

Sampling Period	Total Species	Species	Total No./m^2 (Average)
July 1967	3	Mugil cephalus Dorosoma cepedianum Fundulus grandis	2.0
August 1967	1	Mugil cephalus	1.0
November 1967	0		0
December 1967	0		0
January 1968	0		0
February 1968	0		0
March 1968	1	Cyprinodon variegatus	< 0.1
April 1968	1	Mugil cephalus	< 0.1
May 1968	0		0

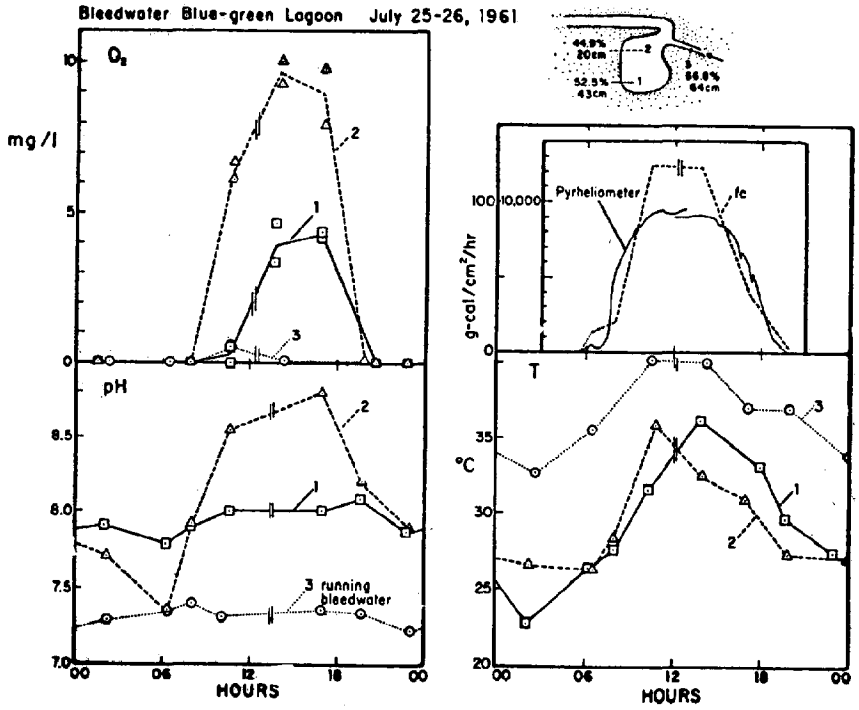

Fig. 7. Diurnal record of oxygen, pH, temperature, and light in a brine disposal lagoon discharging into the Laguna Madre (inset map shows position of stations and inflow of brine). (From Odum et al. 1963; Fig. 8.)

Coupling of Food Availability

Apparently the combination of stresses during winter limits the population to a few organisms and, with the fluctuating brine loads causing shocks, the number of total individuals is also low (see Fig. 6 and Table 4). During spring and early summer, however, the total number of individuals (while the species diversity is still relatively low) increases to bloom characteristics. The chlorophyll measurements (Fig. 5) indicate high populations of phytoplankton during July 1967 and March - April 1968, and the chlorophyll concentrations were higher than those for most stream mouth communities (see chapter on oligohaline systems).

There is a direct coupling of food chains in the brine dominated system to correspond to the sporadic presence of the primary producer. In the Chiltipin Creek area the spring and early summer peaks of phytoplankton (indicated by chlorophyll measurements) are accompanied by peaks in zooplankton (Fig. 6) and abundance of plankton-eating fishes (Table 4). Apparently, the fish, for example, can not energetically afford to compensate for the strange stress of brine and have energy left over for foraging if the food materials are scarce; they can only afford it if the food materials are relatively highly concentrated. Thus, the coupling of trophic levels is necessary in order to provide immediate benefits in a system subjected to the stress of brine pollution.

It is interesting to note that all of the fish captured in the brine dominated system are plankton feeders, thus eliminating the energetic cost of developing a tertiary trophic structure and high diversity system.

Community Metabolism

Localized lagooning of brine wastes from individual oil wells along the Texas coast develops dense blue-green algal mat systems where the water is shallow. Odum *et al.* (1963) reported the community metabolism of such a system on the edge of the Texas Laguna Madre, near Flour Bluff, Texas (Fig. 7). Shown in the figure are a diagram of the brine lagoon and 24-hour measurements of oxygen, pH and temperature for three sampling stations. In the warm brine input ditch, the oxygen reached values above zero only during morning daylight and the pH was low (curve 3 in Fig. 7). In the middle of the lagoon, where the blue-green mat was dense, the oxygen values far exceeded saturation during daylight and were zero during nighttime. Similar findings are shown elsewhere in Odum *et al.* (1963) and in Copeland and Jones (1965) for natural evaporating brines in Texas and Mexico.

MANAGEMENT HINTS

As shown by the preceding data, the food chains in brine dominated systems are short and the trophic levels are depressed. In other words, the highest order consumer level is the plankton-eating fishes. Also, those species of organisms that are capable of living in brine dominated systems are usually there in large numbers, especially when the brine input is at a steady rate (constant high salinity in the system). Man, therefore, should

utilize these basic attributes to manage and benefit from brine dominated systems.

Whitney (1967) has reviewed the practise of introducing commercially important animals into inland mineral waters in various areas of the world, an environment not much different from the one provided by brine pollution. He noted the use of evaporating lagoons for culturing milkfish, mullets and prawns in Asia and eels and mullets in Europe. All of these organisms are plankton eating animals and are capable of surviving salinities up to 100 ppt or more.

It would appear to be economically feasible to pond brine wastes at the upper ends of estuaries and use the ponds to culture highly adaptable plankton eating fishes for commercial uses. Mullet (Mugil cephalus), a major food fish in some parts of the United States, is one of the major inhabitants of the Chiltipin Creek brine dominated system and has been used extensively for culture in high salinity lagoons in Asia and Europe. If the salinity could be controlled by a steady input of brine into these culture ponds, production approaching the levels indicated for July in Table 4 could be maintained during most of the annual cycle.

Chapter E-15

PETROCHEMICAL WASTE SYSTEMS

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INTRODUCTION

The refining of petroleum results in a waste that is not only toxic to most organisms, but also contains organic compounds that are not easily decomposed. These compounds, when injected into the aquatic environment, present the system with a strange chemistry that can not be easily attacked by present-day organisms. Thus, the energies required by organisms to develop capacities for decomposition and/or survival of these chemicals creates a serious energy drain on the system, resulting in altered species diversity, productivity, metabolism and system structure. Even though information concerning petrochemical waste dominated coastal systems is sparse, it appears that the emerging system possesses unique characteristics.

The future of petroleum exploitation seems to be on the coastal areas and the continental shelves, thus the location of petroleum refining complexes (with their related industries) will necessarily be on the coasts. It is probable, therefore, that the disposal of petroleum wastes in the coastal systems of the United States will become a greater problem than it is at the present. Since much of the petroleum refining has been centered in inland areas, much of the research concerning the effects of waste disposal has been carried out in freshwater systems. It may be possible, however, to utilize the findings of research in freshwater systems to construct a picture describing the emerging new systems in coastal waters.

Refinery Waste Characteristics

Beychok (1967), in his book describing petrochemical wastes, gives tables of typical waste characteristics (Tables 1 and 2). In general, the 17 different wastes exhibited high concentrations of phenols, sulfides, ammonia, suspended and dissolved solids, oil, and exerted high oxygen demands. What effects these uniquely characterized wastes have in coastal systems is not understood.

It should be noted at this point that petrochemical wastes, when subjected to biological processing in ponds or other aqueous systems, decrease in toxicity and oxygen demand with time (Dorris et al. 1961; 1963). It is reasonable, therefore, to assume that the effectiveness of such a harsh waste is not only reduced by dilution in coastal systems but by time-degradation as well.

Table 1. Miscellaneous petrochemical effluent analysis (from Beychok, 1967; Table 3a).

Type of plant	Oil refinery	Oil refinery	Oil refineries	Oil refinery	Oil refinery	Oil refinery	Oil refinery	Oil refinery	Oil refinery	Oil refinery	Oil refinery
Location	U.S.A.	U.S.A.	U.S.A.	U.S.A.	U.S.A.	U.S.A.	U.S.A.	Canada	U.S.A.	Canada	U.S.A.
Size		75,000 b.s.d.	Average of 50 refineries	95,000 b.s.d	95,000 b.s.d.	90,000 b.s.d.	25,000 b.s.d.	32,000 b.s.d.	40,000 b.s.d.	20,000 b.s.d.	40,000 b.s.d.
Reference	B-6 (1960)	B-3 (1964)	B-7 (1963)	B-4 (1956).	A-8	B-9 (1952)	C-6	B-8 (1964)	B-14 (1960)	B-16 (1963)[b]	B-27 (1958)
In-plant pretreatment:											
Clean water excluded	yes	no	probably in some	yes	yes	yes	yes	yes	yes	yes	yes
Sour water stripped	?	no	probably in most	yes	yes	yes	yes	yes	yes	yes	yes
Spent caustic neutralized	yes	no	probably in some	–	–	oxidized[c]	yes	no	no	?	no
Spent caustic excluded	no	yes	probably in some	yes	yes	yes (phenolic)	no	yes	yes	?	yes
API separator	no	yes	very probably in most	yes	yes	yes	no	yes	yes	yes	yes
Coagulation or flotation	no	no	probably in some	no	no	no	no	no	coags	no	no
Other (after separator)	no	no		5-day impound basin	5-day impound	no	no	no	no	no	no
Miscellaneous	includes sanitary waste and 'sprung' water	includes sanitary waste	samples were probably *final* treated effluent	c.w. blowdown, boiler blowdown, desalter effluent excluded	c.w. and boiler blowdown, desalter effluent excluded	c.w. blowdown included	'sprung' water included	c.w. and boiler blowdown, sanitary waste included		c.w. and boiler blowdown, sanitary waste included	
Flow (U.S. g.p.m.)	(4,200)	(5,500)		500–950 (625)	(2,500)	(5,000)	(115)	(500)	(1,035)	(300)	(340)
5-day BOD (p.p.m.)	(146) [275]	(150)			(158)					(1,600)	
COD (p.p.m.)[d]	(490) [880]	(450)		172–541 (349)	(327)				(236)	(1,800)	
Oil (p.p.m.)	(268) [1,240]	(220)			(30)	(80)			(25)	(90)	(250)
Dissolved solids (p.p.m.)				386–678 (562)							
Suspended solids (p.p.m.)	(103) [250]			10–36 (24)	(28)						
Phenols (p.p.m.)		(7)		8–70 (24)	(12)	(20)	(228)	(60) [160]	(10)	(56)	(270)
Sulfides (p.p.m.)		(15)		0–12 (3)	(0)	(2.5)		(12) [30]	(25)		
Ammonia (p.p.m.)		(23)									
Others (as noted)		IOD (21)		BSH (1.3)							
lb/1,000 barrels of feed:											
5-day BOD		(132)	(110)		(50)					(237)	
COD		(396)		(28)	(104)				(74)[a]	(325)	
Phenols		(6.2)		(1.9)	(3.8)	(13.3)	(12.6)	(11.3)	(3.1)[a]	(10.1)	(25.5)

Notes: 150–200 Denotes range of values
(175) Average value
[200] Maximum value.

[a] The reference (B-14) indicates that coagulator was only used as a settler.

[b] Same refinery effluent system discussed in reference B-17 also.

[c] Sulfidic caustic was oxidized and phenolic caustic was neutralized.

[d] COD test method not defined in references. (Probably 2-hour Dichromate in most cases for U.S.A. references.)

Table 2. Miscellaneous petrochemical effluent analysis (from Beychok, 1967; Table 3b).

Type of plant	Oil refinery	Refinery, gas processing, chemicals	Synthetic rubber	Naphtha cracker	Naphtha cracker	Naphtha pyrolysis	Catalytic cracker
Location	U.S.A.	U.S.A.	U.S.A., Australia, Africa	U.S.A.	England	U.S.A.	U.S.A.
Size	45,000 b.s.d.	85,000 b.s.d. refinery 100,000 b.s.d. gas liquids Chemical specialities	30,000 tons/year (Typical plant producing styrene–butadiene rubber)	9,000 b.s.d. (Producing ethylene, propylene, butadiene)	12,500 b.s.d. (Producing ethylene, propylene, butadiene)	(Producing primarily acetylene)	(Cracking gas oil)
Reference	B-28 (1961)	B-43 (1966)	B-10, B-35, B-37, B-48 [c]	B-6, C-26, M-14	C-21	Unpublished data	C-6
In-plant pretreatment:							
Clean water excluded	no	†	–	yes	yes	(Test run data of short duration. Phenol data given below are 'reaction yields' as calculated from absorption of phenols in circulating water used to quench reaction effluent to 90°F)	(Conventional refinery cracker. Phenol data given below are 'yields' as calculated from overhead drum gasoline *and* foul water flows and phenol analyses)
Sour water stripped	yes	yes	none	no	no		
Spent caustic neutralized	–	–	–	–	–		
Spent caustic excluded	yes	yes	yes	yes	yes		
API separator	yes	yes	yes	no	no		
Coagulation or flotation	no	[a] flotation (part of flow)	no	no	no		
Other (after separator)	no	no	no	no	no		
Miscellaneous	c.w. and boiler blowdown included. Sanitary waste excluded	about 4% of flow is sanitary waste	includes c.w. blow-down and process water treating effluents	spent caustic and waste C.A.A. pumped into disposal wells	this is *only* foul condensate from the quench tower overhead drum		
Flow (U.S. g.p.m.)	(800)	(3,500)	(250)	(160)[d]	(135)[d]		
5-day BOD (p.p.m.)	(109)	(125)	50–150	(500)			
COD (p.p.m.)	(314)	(429)	1,600[c]	(500)	55–190		
Oil (p.p.m.)	(76)	(108)			(160)		
Dissolved solids (p.p.m.)			(11,000)				
Suspended solids (p.p.m.)		(259)	(150)[c]				
Phenols (p.p.m.)	(5.7)	(14)	5–15	(25)	6–10		
Sulfides (p.p.m.)	(7.2)	(30)					
Ammonia (p.p.m.)	(32)						
Others (as noted)	Cr (0.9) Fl (2.2)		NaCl (8,200) Other salts (2,600) Detergent (115) Sequestrant (85)				
			Dissolved solids (11,000)				
lb/1,000 barrels of feed:			lb/100 *tons of rubber:*				
5-day BOD	(23.3)	(28.4)[b]	165–495	(108)			
COD	(67)	(97)[b]		(105)	7–23	*Phenols yield:*	*Phenols yield:*
Phenols	(1.2)	(3.2)[b]	16.5–49.5	(5.4)	0.8–2.3	lb/1,000-barrels feed = 155 wt. % on feed = 0.07	lb/1,000-barrels feed = 166 wt. % on feed = 0.05

Notes: 150–200 Denotes range of values
(175) Average value
[200] Maximum value

[a] The reference (B-43) indicates that the *refinery* effluent (about 50% of the *total* effluent) has been processed in an air flotation unit.

[b] Based on 85,000 b.s.d. crude oil plus 100,000 b.s.d. gas liquids.

[c] These data are a composite from various plants and calculated to the basis of a typical 30,000 ton/year production of rubber. The COD is the 2-hour Dichromate test for one 'spot' sample. Suspended solids are mostly crumb rubber.

[d] Foul condensate from quenching of cracked reaction products.

SYSTEM EXAMPLES

Most petrochemical plants are located near coastal systems that receive stress from other means; e.g., related industries, sewage, splashing waves or high temperatures. Thus, in very few instances are case histories available for the assessment of the effects of petrochemical wastes on a system. The following are examples of systems dominated by petrochemical wastes.

Los Angeles Harbor, California

Reish (1965) summarizes the effects of petrochemical pollution on the benthos in Los Angeles Harbor, California. Shown in Figure 1, the Harbor area is used as an example. Wastes from oil refineries are discharged into the Los Angeles Harbor area at several points, most of which are partially closed basins or slips with limited water circulation. Bottoms of the discharge areas were essentially devoid of macroscopic animal life, and the substrate was oily, black, and possessed a sulfide odor (Reish 1965).

Dredging activities in 1953 removed the oily substrate from the Consolidated Slip area and the opportunity was presented to observe the time element involved in degradation of the area (Reish 1965). Schematic diagrams of the degradation of the bottom as determined by benthic organisms are shown in Figure 2 and tabulated species diversity of the affected area shown in Table 3. The healthy bottom was characterized by the presence of many animal species, with the polychaetes Tharyx parvus, Cossura candida, and Nereis procera predominating. Fewer species existed on the semi-healthy bottom with the dominant forms being Dorvillea articulata and Polydora paucibranchiata. Capitella capitata was dominant on the polluted bottom and the diversity was very low. No macroscopic animal life was taken from the very polluted bottom.

There was a progression of polluted bottoms in the sampling area after the dredging had cleaned the oil sludge from the bottom (Fig. 2). This progression of events was also indicated by the species diversity (Table 3), which gives the number of animal species collected during five bottom surveys. The effect of removal of the pollutants from the bottom can be visualized by observing that only five species were found in August 1951, before dredging, while a total of ten species were taken during 1954, shortly after completion of dredging operations. A decrease in species present in the sampling area decreased after that until in December 1955 there were only two species present in the area.

Oil Refinery Effluent Microcosms

Copeland (1966b) reported the results of a laboratory study of the community metabolism of micro-ecosystems containing petroleum refinery effluents. As shown in Figure 3, for a system taken some distance from the outfall of a petrochemical refinery, one gets a metabolic pattern of initial P/R less than one, P/R greater than one at intermediate distances and P/R of unity after considerable distance and/or time (see Copeland and Dorris, 1964 for explanation). By the time P/R unity is reached, the level of community metabolism is relatively high. For systems taken from the outfall area near a refinery

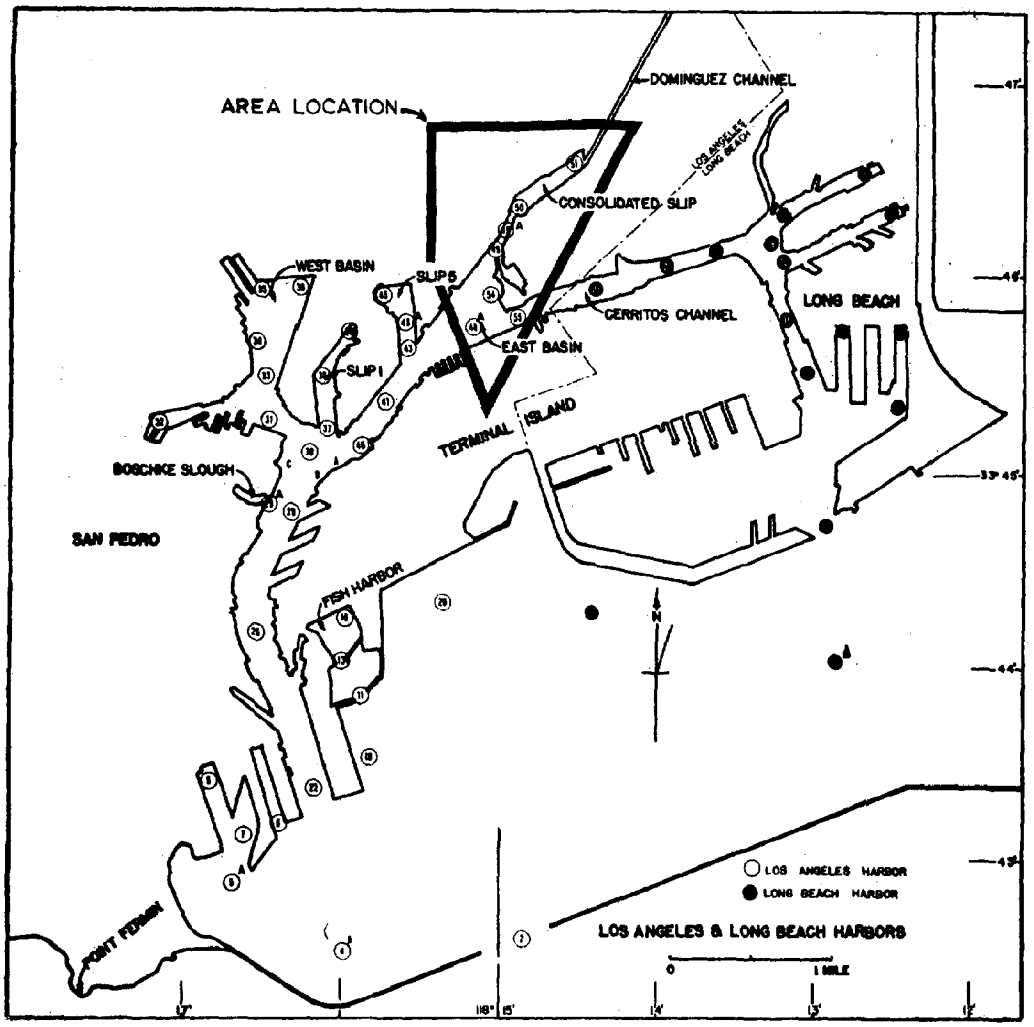

Fig. 1. Map of the Los Angeles-Long Beach Harbors showing location of study area and station location. Petrochemical wastes come in at station 26, West Basin, Consolidated Slip and in Long Beach Harbor (From Reish, 1965; Fig. 1).

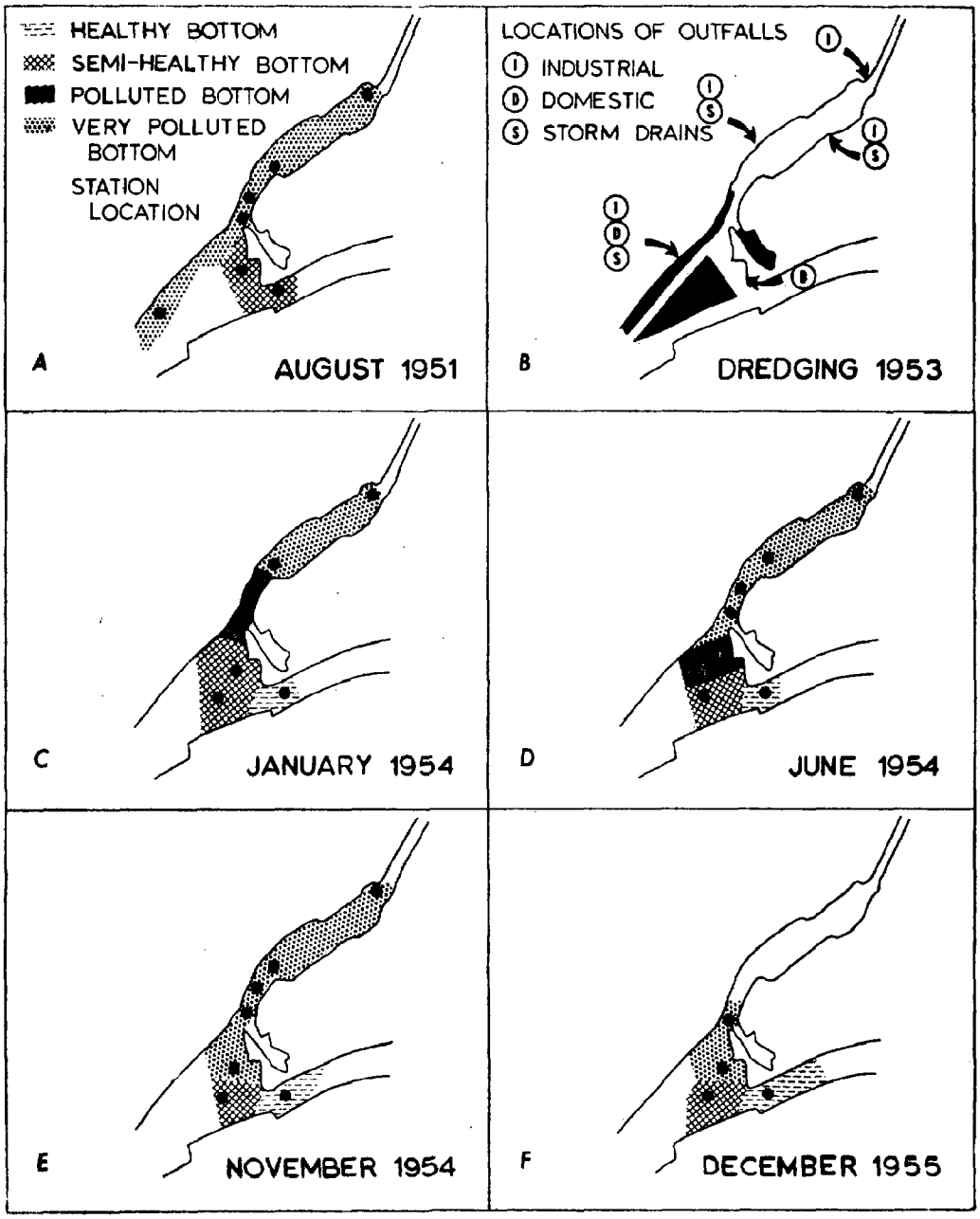

Fig. 2. Animal bottom conditions in the petrochemically polluted area of Los Angeles Harbor (From Reish,1965; Fig. 2).

Table 3. Number of animal species on the bottom of Los Angeles Harbor, 1951-1955 (From Reish, 1965; Table 1).

Station Number	Aug. 1951	Jan. 1954	June 1954	Nov. 1954	Dec. 1955
51	0	0	0	0	—
50	0	0	0	0	—
49 A	—	7	0	0	—
49	1	0	0	0	0
54	5	7	2	0	0
48 A	—	6	8	5	2
Total No of Species	5	10	9	5	2

Table 4. Amount of organic carbon in the sediments of east area of Los Angeles Harbor, 1954 (From Reish, 1965; Table 2).

Station Number	Jan. 1954	June 1954	Nov. 1954
51	8.1 %	10.7 %	9.4 %
50	5.8	9.3	7.5
49 A	2.0	4.8	8.1
49	3.5	3.7	2.2
54	0.6	1.6	2.5
48 A	—	1.4	1.8
55	1.8	2.5	2.1

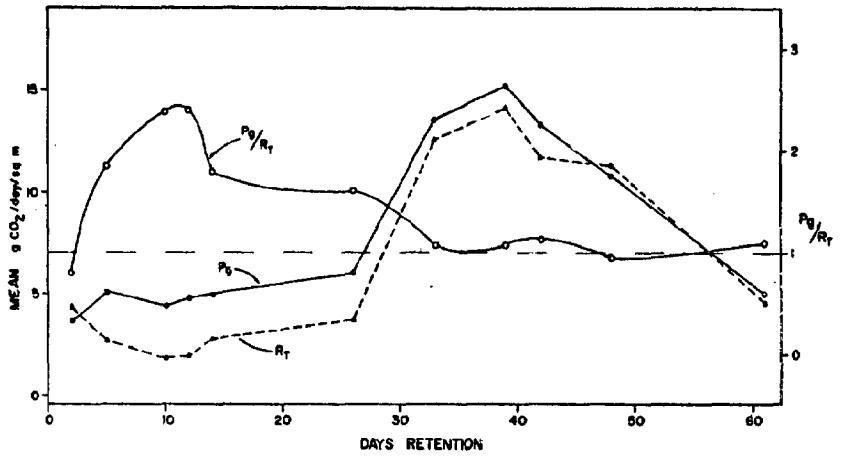

Fig. 3: Community metabolism and P/R ratios for Union Carbide bayou effluent (From Copeland, 1966b; Fig. 6).

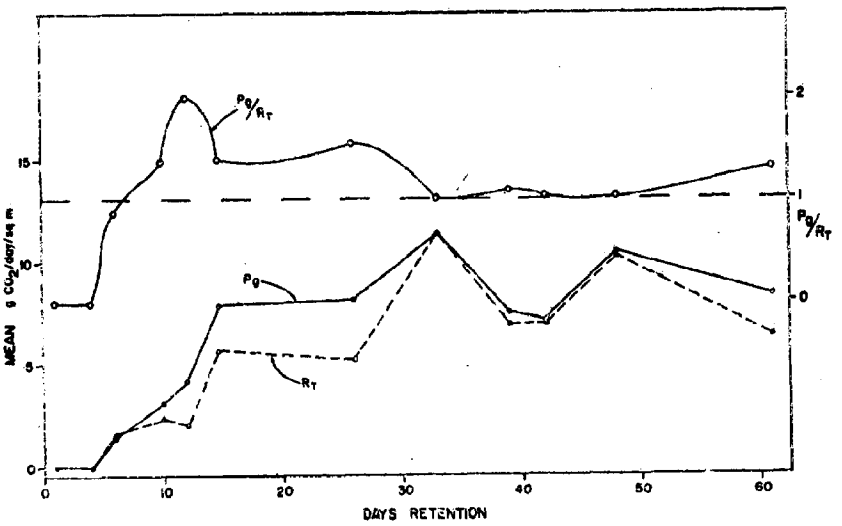

Fig. 4: Community metabolism and P/R ratios for Suntide Oil Refinery effluent (From Copeland, 1966b; Fig. 7).

(Fig. 4), the pattern has one variation; i.e., no metabolic activity for a few days because of the severe toxicity of the waste. In contrast to systems receiving recently evolved organic loading, such as sewage, systems receiving petroleum effluent, with its now-strange compounds, exhibit initial depression in metabolic activity due to the organisms' inability to attack the materials. The pattern shown in Figures 3 and 4 are for holding time. However, the holding time ordinate is the same as distance or area in natural systems, thus making the laboratory studies directly applicable to the real natural system.

Other studies of community metabolism in coastal systems receiving petroleum wastes include Odum (1960a) and Odum et al. (1963). The community metabolism reported by them was based on field measurements. Upper Galveston Bay, Texas which receives the multiple wastes of one of the largest petrochemical processing complexes in the world, exhibits the same metabolic pattern as one advances from the outlet of the Houston Ship Channel into upper Galveston Bay (Odum et al. 1963). Similar results were reported for the Corpus Christi area by Odum et al. (1963). Fig. 5 shows waste distribution from refineries at Bahia Tallaboa, Puerto Rico (Cerame-Vivas et al. 1967).

EFFECTS ON SYSTEM COMPONENTS

While information does not exist to enable characterization of a system dominated by petrochemical wastes, data are available concerning effects on system components. In some cases, information concerning freshwater systems will be utilized to make conclusions pertaining to coastal systems. In the following discussion the criteria of toxicity, metabolism of certain organisms, community metabolism and species diversity will be considered to characterize the petrochemical waste dominated system.

Toxicity

One of the more devastating effects of petrochemical effluents on the coastal environment is toxicity. At the point of effluent entry into a system, many animals and plants are killed or, in the case of animals, driven out of the area. The effects of toxicity are often apparent over a large area, decreasing in intensity with distance from the point of entry, and with dilution and time.

The most dramatic evidence of toxicity to marine organisms by petrochemical wastes is the incidence of fish-kills in coastal systems (U.S. FWPCA, 1960). However, large scale kills are usually confined to areas immediately below waste outfalls and are often associated with application of other stresses (temperature, reduced oxygen tension, etc.) or with the event of spawning migrations into or through an area of high concentrations of toxic substances. These incidences are relatively minor when considered along with the possible long-term effects of very dilute concentrations, more normally occurring in natural systems, upon members of entire food-chains. Bio-assay methods such as described by Doudoroff et al. (1951) and by the American Public Health Association (1960) for estimating acute toxicity to individual organisms are not really meaningful in terms of a natural system. Still, even the use of these methods on salt-water organisms has been sparse.

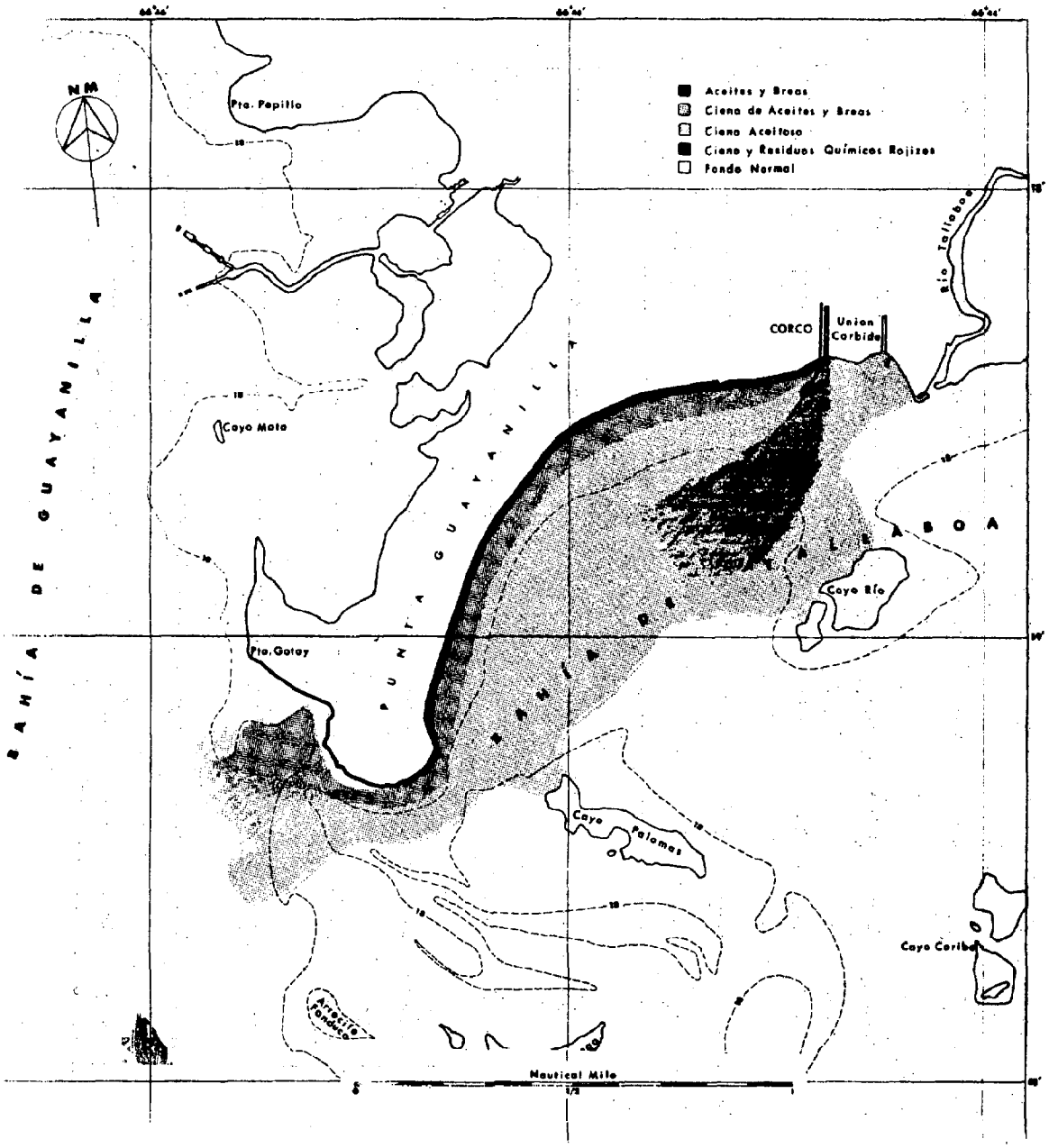

Fig. 5. Map of the Bahia de Tallaboa, Puerto Rico, showing the location of two oil refineries and the distribution of waste from them (From Cerame-Vivas et al. 1967; Plate 2).

Some bio-assay data for marine organisms using petrochemical effluents have been gathered by Hood et al. (1960) with Artemia and Cyprinodon. Copeland (1966b) gives 48-hr TLm for Cyprinodon and Lagodon using effluents from four refineries (Table 5). Marek (1959; 1960) and Spears (1959; 1960; 1961; 1962) performed bio-assays with several estuarine fishes using petrochemical effluents, along with their chemical analyses. Perhaps more realistic are the long term methods described by Graham (1963) in which test organisms are held for several weeks. These means reveal toxicity tolerances to be much lower than indicated by the more usual short-term methods.

The more subtle effects of slight mortality in a population of food fish are discussed at length by Copeland and Wohlschlag (1968). In Table 6, two hypothetical populations of fish are compared, in which 1,000 individuals of fishable size are recruited annually at age III and weighing 1.5 pounds. In both populations, 25% of each year-class are fished annually. The first population has an additional 25% mortality while the second population, due to a slight pollution effect, has an additional 35% mortality. Even though the mortality difference can not be distinguished by standard methods, it can be seen that the total size of the population and total catch are significantly reduced. The most pronounced reduction is among the older and larger individuals. Clearly, survival would be further reduced if the same principle were applied to populations all along the food chain. Bio-assays do not begin to give us a picture of the ultimate effects of petrochemical waste releases upon coastal systems.

Organismal Metabolism

Brett (1958) suggested the use of physiological criteria for assessing the presence of an indiscriminate or population stress such as chronic or sub-lethal toxicity. Because metabolism is an integrative characteristic, any stress condition placed upon an organism should be reflected by changes in metabolic rates. When critically evaluated, a metabolic indicator such as oxygen consumption rates may be useful in assessing the physiological state of an organism. For example, an increase in oxygen consumption rate may reflect greater energy demand for survival and maintenance in presence of petrochemical effluents. Conversely, a decrease in respiration rate to below maintenance level, may indicate an overall decrease in the physiological state of the organism, thereby diminishing its chances for survival and/or reproductive success.

Abnormal metabolism due to loading or inhibiting stresses from petrochemical pollution can have far-reaching effects on individuals, populations and the system. The alternatives are to leave the area or to remain at the expense of reduced ability to compete with other organisms, adequately feed, protect itself from predators and otherwise successfully exploit its present niche (Steed and Copeland, 1967).

Steed and Copeland (1967) subjected several estuarine organisms to low concentrations of petrochemical wastes, which resulted in a depression of metabolic rates at higher concentrations (Fig. 6). In each case, the maximum concentration of effluent did not exceed the median tolerance limit (in other

Table 5. Survival of Lagodon rhomboides and Cyprinodon variegatus in various concentrations of petrochemical effluents (From Copeland, 1966b; Table 1).

Effluent and Organism	Effluent Conc. (%)	Survival after Stated Time Interval (%)				TL_m^{48*} (%)
		6 hr	12 hr	24 hr	48 hr	
Effluent A *Lagodon*	Control	100	100	80	80	
	32	100	100	100	100	
	56	100	100	100	100	65
	75	20	0	0	0	
Effluent A *Cyprinodon*	Control	100	100	80	80	
	32	100	100	80	80	48
	75	100	60	20	20	
	100	100	0	0	0	
Effluent B *Lagodon*	Control	100	100	100	100	
	32	100	100	100	100	
	56	100	100	100	100	65
	75	30	0	0	0	
Effluent B *Cyprinodon*	Control	100	100	100	100	
	56	100	100	100	90	
	75	90	90	80	70	>100
	100	70	60	60	60	
Effluent C *Cyprinodon*	Control	100	100	100	100	
	3.2	100	90	90	80	7.5
	10	90	60	50	40	
Effluent D *Cyprinodon*	Control	100	100	100	100	
	75	100	100	100	90	>100
	100	70	70	70	70	

* 48-hr median tolerance limit (theoretical solution concentration at which 50 percent of the fish survive).

Table 6. Model of a hypothetical fishery, experiencing a slight increase in natural mortality (From Copeland and Wohlschlag, 1968; Table 20).

Age	III	IV	V	VI	VII	VIII	IX	X	XI	XII	XIII	Totals
Wt/Fish (Lbs.)	1.5	3.5	4.5	6.0	7.5	10.5	13.0	16.5	19.0	21.5	25.0	
Stable Population (50% Survival; 25% Fished; 25% Mortality):												
Population	1,000	500	250	125	62	31	16	8	4	2	1	1,999
No. in catch	250	125	62	31	16	8	4	2	1	—	—	499
Wt. of catch	375	438	279	186	120	84	52	31	19	—	—	1,584
10% Additional Mortality												
Stable Population (40% Survival; 25% Fished; 35% Mortality):												
Population	1,000	400	160	64	26	10	4	2	1	—	—	1,667
No. in catch	250	100	40	16	6	3	1	—	—	—	—	416
Wt. of catch	375	350	180	96	45	32	13	—	—	—	—	1,091

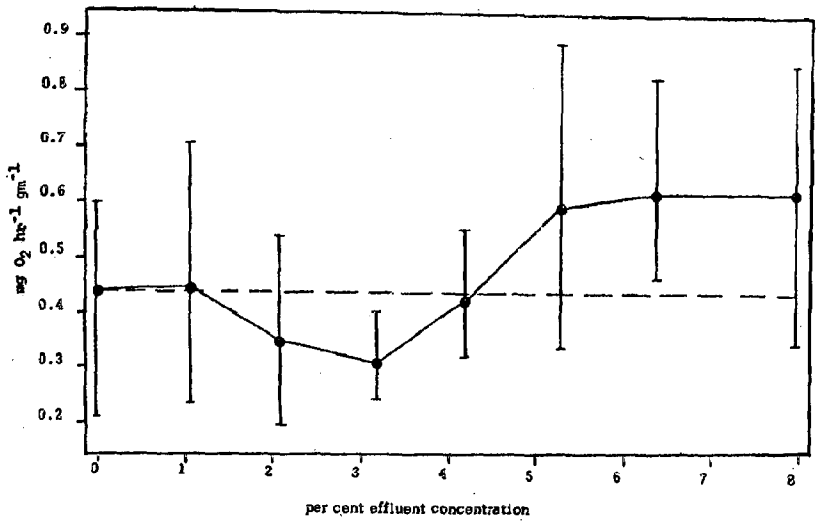

Fig. 6: Oxygen consumption rates of Cyprinodon variegatus (mg O_2/hr/gm) in eight concentrations (percent) of petrochemical effluent (From Steed and Copeland, 1967; Fig. 1).

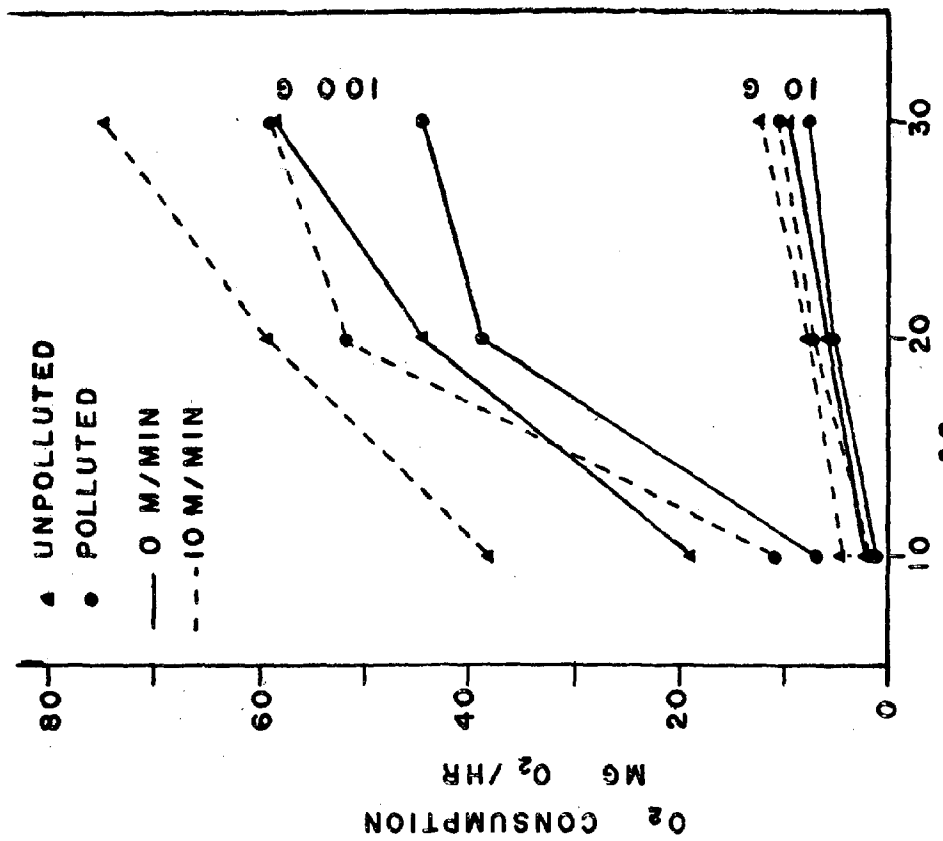

Fig. 7: Oxygen consumption levels of pinfish calculated for 10- and 100-gram fish, for petrochemical waste polluted and unpolluted water and for zero and 10 m/min swimming velocities (From Wohlschlag and Cameron, 1967; Fig. 1).

words, sub-lethal effluent concentrations). Copeland and Wohlschlag (1968), reporting the effects of water from a petrochemical waste dominated system on the metabolism of an estuarine fish, demonstrated a decrease in metabolic rate when the fish were already affected by the stress of cold temperatures. Wohlschlag andCameron (1967) worked out multiple regression equations for oxygen consumption rates incorporating body weights, temperatures, and swimming velocities for fish in polluted and unpolluted systems (Fig. 7). Interpretation of their data indicates metabolic loading under multiple stresses, such that large swimming fish in the polluted system had lower oxygen consumption rates than non-swimming fish in control system, suggesting reduced capacity for growth and survival, especially at temperature extremes.

Community Metabolism

Measurements of the metabolism of the system as a whole give the opportunity to evaluate fertility, appraise disturbance, predict biological events, manage production, develop resource yields and to farm vast systems (Odum and Wilson,1962). The introduction of petrochemical wastes, with its inherent strange chemicals, yields characteristic metabolic patterns in aquatic systems. Extension of knowledge gained from studying freshwater systems receiving petrochemical wastes (Copeland andDorris,1962; 1964) and from studying systems receiving other organic loading (Odum,1956; Odum et al. 1963), when compared to laboratory studies using marine systems (Copeland,1966b), may lead to conclusions characterizing disturbed coastal systems in general.

Results of diurnal oxygen curves for the White River, Indiana receiving organic pollution revealed a characteristic pattern (Odum,1956); i.e., a P/R ratio of less than one near the waste outfall, a P/R ratio of greater than one at intermediate distances below the outfall, and a P/R ratio of near unity even further away. This pattern can be explained by the differential toxic and fertilizing effects of the effluent in the system. The low P/R ratio was the result of toxicity lowering the photosynthetic ability of the system while stimulating the respiration component through decomposition of the organic material. Later, when the toxicity had moderated through time, the effluent material acted as a fertilizer causing the photosynthetic component to increase resulting in a high P/R ratio. The later P/R unity with a high metabolic rate can be attributed to the system's ability to maintain balance at whatever level the organic fuels will allow (Copeland, 1965a).

Patterns similar to the above, for White River, were reported by Copeland and Dorris (1962; 1964) for freshwater systems receiving petroleum refinery effluents. The question now becomes, will this pattern be evident in coastal systems?

The magnitude of photosynthesis and respiration in systems influenced by petrochemical wastes is extremely variable (Odum et al. 1963: Copeland and Dorris 1964). The large variability can be explained by the presence of large amounts of highly toxic materials depressing metabolism in localized areas and the decomposition products enhancing metabolism in larger areas. Odum et al. (1963) reported photosynthesis ranging between 0 and 58 $gm/m^2/day$ and respiration ranging between 0 and 87 $gm/m^2/day$ in petrochemical dominated estuaries of Texas. Copeland and Dorris (1964) reported photosynthesis of 0

to 30 gm/m^2/day and respiration of 2 to 51 gm/m^2/day in freshwater systems. In both cases, the low values occurred at or near outfall areas, with the higher values occurring in well-mixed areas considerably distant from the outfall.

Species Diversity

There are many kinds of systems in which organisms must expend a large amount of energy to adapt to extreme physical and/or chemical conditions. Systems receiving petrochemical wastes represent this type. Because few organisms possess adaptations allowing them to exist under these conditions, reduced biotic interaction (competition, predation, symbiosis, etc.) between species results in a low diversity of resident species and simple food chains. Maintenance costs are so high that if divided between several populations at the same trophic level, energy available for each would not be adequate for continued reproduction. As a result, those organisms able to adapt often abound in large populations as they are more or less limited only by food supply.

The unusual condition of a system being influenced by petrochemical wastes alone was investigated by Copeland (1967a). As may be seen in Fig. 8, diversity of organisms was low near the source of the waste and increased at a greater distance from the source. Although the pattern shown in Fig. 8 was computed from zooplankton samples, other kinds of organisms follow the same relationship (Bechtel et al. 1970).

Phytoplankton

Hohn (1959) reported the diversity of diatom species in the petrochemical waste influenced Galveston and Chocolate Bays, Texas. Near the Houston Ship Channel, the number of diatom species was 24, while in the open Galveston Bay the number of species reached as high as 165. Minter (1964) reported very low diversity of algae in freshwater systems receiving petrochemical wastes.

Zooplankton

In some cases, no zooplankton could be found in waters receiving large amounts of petrochemical wastes (Odum et al. 1963; Minter, 1964). In both reports the instances of no zooplankton occurred during the winter, when the stress of low light intensity and low temperatures was in combination with the stress of petrochemical wastes. Species diversity of zooplankton is always low in petrochemical waste polluted systems (Odum et al. 1963; Minter, 1964) even when environmental conditions are optimal.

Benthos

The diversity of benthic communities are probably more directly affected by petrochemical wastes than are other community components because of their sessile (and thus inescapable) characteristics. Wilhm and Dorris (1968), in their discussion of diversity indexes, demonstrated considerable decrease in benthic species diversity near and below petrochemical waste outfalls in streams and estuaries (Fig. 9). Oil refinery effluents caused a tremendous decrease in benthic diversity in Los Angeles, California harbor (Reish, 1965) and Alamitos Bay, California (Reish and Winter, 1954). The difference between observed species diversity and expected species diversity of benthos in South San Francisco Bay was correlated with toxicity and BOD of petrochemical waste influenced bay waters (Pearson et al. 1967; Engineering-Science Inc. 1968).

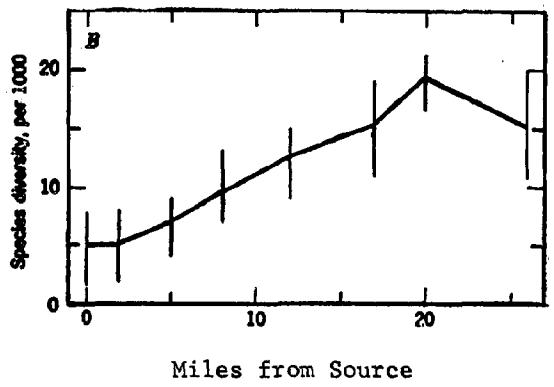

Fig. 8: Zooplankton species diversity per thousand individuals for a system receiving petrochemical wastes (From Copeland, 1967a, Fig. 2).

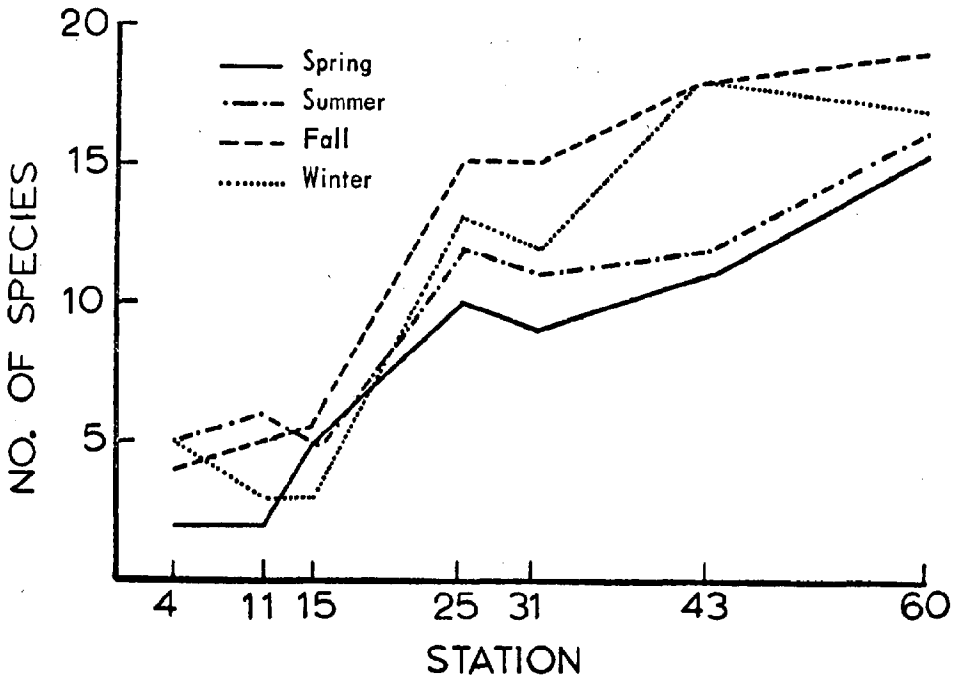

Fig. 9: Variation in numbers of species downstream from a petroleum refinery effluent entrance site (From Wilhm and Dorris, 1966, Fig. 7). Station 4 is nearest outfall with miles downstream for subsequent stations indicated.

Even though the species diversity may be lower, the total number of organisms may be higher in some areas affected by petrochemical wastes. Pimentel (1959) reported such a phenomenon in the Oso Flaco area of California, where the species diversity of benthic clams was lower but the total number of clams in the area was significantly higher. This may be attributed to the fact that toxicity is differentially effective, but those animals that survive do so in the absence of competitors and predators and in the presence of organic foods.

Fish

In many areas affected by petrochemical wastes, the fishes have vacated the zones near outfalls because of the toxicity of the wastes. A motile organism such as a fish will move out rather than tolerate highly toxic materials (Steed and Copeland, 1967). Chambers and Sparks (1959) reported no fish in several collections in an area near a refinery outfall in the upper Galveston Bay, especially during winter when natural stresses were great. Even during optimal growing and nursery seasons, the diversity of fishes in the area was low compared to other areas.

DISCUSSION

Since a single, disturbed system has not been studied in detail, the previously discussed characteristics have been taken from widely separated studies of various aspects of systems influenced by petrochemical wastes. The following discussion will be an attempt to summarize and construct an emerging picture of petrochemical dominated systems. One of the reasons a descriptive analysis can not be obtained for petrochemical waste dominated systems is that refineries and related industries are usually located with other industries, thus creating multiply stressed systems from a varying and complex megalopolis waste.

A petrochemical waste dominated system can be viewed as an adaptive self-organizing system; one in which the energetic costs for adapting against physiological shocks from various toxic materials and from widely fluctuating pH and oxygen leaves little energy for diversification. Thus, the energy input is exceeded by energy demands for maintenance and complexity. This phenomenon is verified by the low species diversity found in petrochemical waste systems, especially near the point of input. These systems are dominated by heterotrophic bacteria, anaerobic forms and blue-green algae. Indeed, the freshwater systems described by Minter (1964) contained blue-green algae as the only phytoplankton during winter when the stresses of cold and low light were added to the stress of pollution.

The low species diversity in petrochemical waste systems provides fewer pathways for biogeochemical cycling and energy flows. Thus, the system processes material at an inefficient rate, leaving much organic material unused (Odum, 1967b).

The strange chemicals contained in petrochemical wastes not only produce toxicity to most organisms, but cause anomalous organismal metabolism.

The physiological stress, as indicated by anomalous metabolic rates, suggests that the organism is unable to successfully exploit its ecological niche. Thus, the productivity of useful products is decreased with the same amount of energy being expended. For example, if the metabolic rate is elevated due to increased maintenance requirements, then this represents a higher energetic cost to the system for the organism to maintain its normal growth and reproduction rates. In other words, without some mechanism for increasing the cycling rates of energy, efficiency and productivity of the system is reduced, especially at the higher trophic levels harvested by man.

Total respiration of the system generally exceeds photosynthetic productivity of the system because of the steady inflow of organic materials in the petrochemical wastes. Respiration is not only a result of the producer-consumer community, but of the large heterotrophic bacterial component. Both photosynthesis and respiration of the system are extremely high, sometimes several times greater than found in most coastal systems. Even though the cycling rate indicated by the high community metabolism is large, the translation through subsequent trophic levels is not large because of the tremendous cost of maintaining the physiological status of the consumers.

Hints For Management

The critical aspects of systems dominated by petrochemical wastes is the consideration of toxic chemicals and the subsequent metabolic anomaly. It is apparent that photosynthetic productivity is enhanced by the addition of petrochemical wastes, but the utilization of the high productivity is limited by the inability of organisms to either survive or maintain physiological equilibrium in the system. Thus any management of systems receiving petrochemical wastes will involve some mechanism for removal or depression of the influences of physiological stress.

The removal of the toxic compounds can be done chemically at great expense to management. Activated charcoal filters or some other mechanism would, at great engineering costs, effect the removal of toxic components before the waste enters the coastal system. A more economical and perhaps more ecologically feasible method would be the use of holding (or "oxidation") ponds for impounding the wastes for a period of time to allow bacterial decomposition of the waste materials. Not only would the toxic components be removed, but the effluents released would stimulate community metabolism in the receiving system (see Dorris et al. 1961; Copeland and Dorris, 1964; Copeland, 1966b, for discussion of holding ponds). If holding ponds are not feasible, perhaps localized areas in natural systems that have already been altered by petrochemical waste disposal could be impounded to serve as holding ponds. Petrochemical industries could be located closely together to avoid threat to extensive areas and allow the common usage of impounded areas. The use of organisms (such as mullet) that are capable of adapting to physiological stresses without great loss of maintenance energies and processing food directly from the producer level for farming the area, would be one way of taking advantage of the high productivity.

Chapter E-16

ECOSYSTEMS STRESSED BY ADDITIONS OF MAN-MADE RADIOACTIVITY

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Ecological systems stressed by high radiation and radioactive substances in the environment were created by the atomic testing at Bikini and Eniwetok in the Marshall Islands, but the effects were in combination with blasting, heat, and the release of large quantities of pulverized rock. Although the nature of marine ecosystems under stress of high radioactivity has been much discussed, there are no good case histories that demonstrate the development of ecological systems dominated or characterized by radiation stress. Inferences on the ecological effects of radioactivity in estuaries must be drawn from laboratory studies of biological accumulation and retention of radionuclides and of radiation effects on organisms. The major pathways through which radioisotopes are cycled can be learned from tracer studies in experimental ponds and from case histories of disposal of low-level radioactivity into natural systems, such as the Columbia River. One indirect effect that may change the nature of an estuarine system results from the changed attitudes of people towards the utilization of fisheries when it is known that low level radioactivity is present.

Estuaries will receive inputs of radioactive wastes from a variety of sources as nuclear energy supplies increasing proportions of man's power requirements. Low-level radiation from natural sources is a normal, perhaps even an essential, component of all estuarine and marine ecosystems. Background radiation represents an input, however small, of energy into the system. We presently do not know the full significance of radiation in the development and maintenance of aquatic ecosystems, nor do we know precisely the sensitivity of an ecosystem to small changes in the background radiation. It is conceivable, therefore, that the intentional addition of radioactive substances to estuarine and other marine environments could stress the existing balance of the system and promote the development of new ecological systems. The nature and results of the potential stresses imposed by elevated radiation in natural ecosystems are difficult to evaluate because highly radioactive systems so far have been associated only with nuclear tests which are accompanied by blasts, disruption of sediments, and other nonradioactive stresses and because other instances of continuous elevated radiation have been regulated at doses thought to have no disruptive influence on the ecosystem receiving the radioactive materials. The disposal of radioactivity into aquatic environments is usually restricted on the basis of estimated radiation dose to man; the effects of radiation on the estuarine ecosystem are not considered directly. It is generally assumed from radiation studies that estuarine and marine populations will not be affected so long as man is protected from excessive radiation.

No disruption or disturbance of a natural estuarine ecosystem by discarded radioactive wastes has yet been substantiated. Such an occurrence is not altogether implausible, however, on the basis of projected uses of nuclear energy and of current information on comparative radiation sensitivities of estuarine organisms. It seems appropriate, therefore, to describe a system receiving considerable quantities of radioactive materials, and to discuss man's future

nuclear activities in terms of how they may affect estuarine and marine ecosystems.

SOURCES OF RADIOACTIVITY IN ESTUARIES

Power Reactors

As of June 30, 1968, 15 nuclear power plants were producing about 2,800 megawatts of electricity in the United States. Thirty-one plants with an output of 22,500 Mwatt were under construction and 56 reactors with a power output of 47,600 Mwatt were planned (USAEC 1968a). Output of civilian power reactors is projected to reach about 140,000 Mwatt by 1980 (USAEC 1968b). Most of these reactors will release into the coolant water small amounts of fission products and neutron-induced radioisotopes. The estimated annual releases from a typical pressurized water power reactor are shown in Table 1.

Ship Reactors

Nuclear-powered ships release approximately the same kinds of radioisotopes as land-based power reactors. Approximately 100 shipboard reactors were operational on U.S. vessels as of June 30, 1968 (USAEC 1968c). The major source of waste radioactivity in estuaries would be the expansion volume of primary coolant which must be released whenever the reactor is started up. The expansion volume for the U.S.S. Savannah (output of 80 Mwatt, thermal) was estimated to contain 0.68 curie, consisting mainly of ^{51}Cr, ^{60}Co, ^{55}Fe, and ^{182}Ta. By contrast the radioactivity contained in the spent fuel elements after 1-year's operation of a 60 megawatt reactor would be over 10^7 curies (NAS-NRC 1959b); whereas routine startup of a nuclear ship in a harbor would introduce only limited amounts of radioactivity into an estuary, a nearshore accident involving a nuclear vessel could be serious--if the integrity of the reactor itself was extensively damaged.

Production Reactors

Except for the precautions necessary to contain the highly toxic ^{239}Pu which is produced by neutron-bombardment of naturally-occurring ^{238}Pu, operation of a production reactor should create no unique problems of waste disposal. The U. S. Atomic Energy Commission operates two fuel production plants - at Aiken, South Carolina, on the Savannah River, and at Richland, Washington, on the Columbia River. The Hanford Works at Richland employs a single pass of water from the Columbia River to cool the reactor core, and the effluent contains more than 60 different radionuclides from leakage of fuel elements and from neutron activation of dissolved materials in the primary coolant water (Tables 2 and 3). During full operation of the plant, some 25,000 curies per month actually have entered the Pacific Ocean from the mouth of the Columbia River. The radioecology of the Columbia River and the ocean waters just off the mouth of the river has received considerable attention, as reviewed in subsequent pages.

Fuel reprocessing plants also release radioisotopes in similar composition to those released by power plants, but reprocessing plants may encounter disposal problems of greater magnitude than those associated with the reactors themselves, simply because of the very high levels of radioactivity in spent fuel. The AEC reprocesses fuel at Hanford, Savannah River, and at Idaho Falls, but

Table 1

Annual release of radionuclides estimated for a pressurized water power reactor of 1050 megawatts electric capacity[1]

Liquid wastes

Isotope	Half-life	μCi /yr	Isotope	Half-life	μCi /yr
^{3}H	12.26 yr	4×10^{15}	^{131}I	8 d	6.61×10^{3}
^{54}Mn	314 d	9.7×10^{-1}	^{132}Te	78 hr	6.99×10^{2}
^{56}Mn	2.58 hr	2.64×10^{1}	^{132}I	2.3 hr	2.8×10^{2}
^{58}Co	71 d	2.95×10^{1}	^{133}I	21 hr	5.13×10^{3}
^{60}Co	5.26 hr	3.48	^{134}I	53 m	2.16×10^{1}
^{89}Sr	50.4 d	9.1	^{135}I	6.7 hr	2.6×10^{3}
^{90}Sr	28 yr	5.76	^{134}Cs	2.1 yr	8.69×10^{2}
^{90}Y	64 hr	1.06	^{136}Cs	13 d	8.36×10^{1}
^{91}Sr	9.7 hr	2.49	^{137}Cs	30 yr	4.58×10^{3}
^{91}Y	59 d	2.11×10^{1}	^{140}Ba	12.8 d	2.28
^{92}Y	3.5 hr	5.13	^{140}La	40.2 hr	2.35
^{99}Mo	66 hr	1.25×10^{4}	^{144}Ce	285 d	7.82

Gaseous wastes

Isotope	Half-life	Ci /yr
^{85}Kr	10.4 yr	5.62×10^{3}
^{133}Xe	5.27 d	1.58×10^{3}

[1] Preliminary Facility Description and Safety Analysis Report, Salem Nuclear Generating Station, Burlington County, New Jersey, Docket No. 50-272.

Table 2

Major isotopes released by the Hanford Facility to the Columbia River as of 1960[1]

Isotope	Half-life	Percent composition (from physical decay) at 4 hr	at 72 hr	at 2 wk
^{56}Mn	2.58 h	27.5	--	--
^{64}Cu	12.9 h	18.8	3.34	--
^{24}Na	14.97 h	13.7	4.04	--
^{51}Cr	27.8 d	8.3	53.14	89.72
^{239}Np	2.35 d	8.2	24.55	2.13
^{76}As	26.8 h	7.3	8.68	--
^{31}Si	2.62 h	4.9	--	--
^{69}Zn	55 m	2.4	--	--
^{72}Ga	14 h	1.3	0.30	--
^{92}Sr	2.7 h	0.8	--	--
^{239}U	23.5 m	0.8	--	--
^{133}I	21 h	0.7	0.51	--
^{92}Y	3.53 h	0.6	--	--
^{97}Nb	72 m	0.6	--	--
^{91}Sr	9.7 h	0.5	0.03	--
^{65}Zn	245 d	0.4	2.67	5.75
^{32}P	14.22 d	0.3	1.76	2.29
^{90}Y	64.4 h	0.3	0.97	0.11
^{135}I	6.7 h	0.3	--	--
^{93}Y	10.1 h	0.3	0.02	--

[1] Modified and expanded from Junkins et al. (1960).

Table 3

Relative abundance of reactor effluent radionuclides, 1964[1]

Major, 90%	Minor, 9%	Trace, 1%		
^{24}Na	^{32}P	^{3}H	^{91}Y	^{143}Ce
^{31}Si	^{69m}Zn	^{14}C	^{93}Y	^{144}Ce
^{51}Cr	^{72}Ga	^{35}S	^{95}Nb	^{142}Pr
^{56}Mn	^{76}As	^{45}Ca	^{99}Mo	^{143}Pr
^{64}Cu	^{92}Sr	^{46}Sc	^{103}Ru	^{147}Nd
	^{132}I	^{54}Mn	^{106}Ru	^{147}Pm
	^{140}La	^{59}Fe	^{122}Sb	^{149}Pm
	^{152m}Eu	^{60}Co	^{124}Sb	^{151}Pm
	^{153}Sm	^{65}Ni	^{131}I	^{152}Eu
	^{165}Dy	^{65}Zn	^{133}I	^{156}Eu
	^{239}Np	^{87m}Sr	^{135}I	^{153}Gd
		^{89}Sr	^{136}Cs	^{159}Gd
		^{90}Sr	^{137}Cs	^{160}Tb
		^{91}Sr	^{140}Ba	^{161}Tb
		^{90}Y	^{141}Ce	^{166}Ho
				^{169}Er
				^{171}Er

[1] Moore and Essig (1966).

civilian plants must be constructed soon to reprocess fuel from nuclear power reactors. The first civilian reprocessing plant began operating in 1967 and construction of up to 15 more is expected within the next 10 years (Eisenbud, 1968).

Fallout

Atmospheric testing of nuclear weapons has resulted in the widespread deposition of radioactive materials. Direct deposition of fallout on the surface of estuaries is supplemented by fallout radioactivity leached from the land mass by freshwater runoff. The radioisotopes of major concern in worldwide fallout are ^{90}Sr, ^{137}Cs, and ^{55}Fe, but many other isotopes have also been detected, including ^{140}Ba-^{140}La, ^{95}Zr-^{95}Nb, ^{131}I, $^{141,144}Ce$, ^{103}, ^{106}Ru, ^{54}Mn, ^{65}Zn, $^{57,60}Co$, and ^{185}W. Since 1966-1967, the rate of fallout deposition has been decreasing, despite the continued testing on a small scale by China and France. Fallout deposition is dependent upon latitude; about 80% of total deposit has been in the northern hemisphere, and about 60% of that in the northern hemisphere is between latitudes 30 and 60. If nuclear devices are used for excavation, as has been proposed for a sea-level canal and certain harbors, local estuaries may also receive intense close-in fallout from the explosions.

EXAMPLES

A Case History of Estuarine Radioactivity - The Columbia River and Environs

More radioactivity has been released to the Columbia River than to any other surface water in the world, because water from the Columbia River serves as the primary coolant for the multiple production reactors at the Hanford Plant near Richland, Washington. Most of the radionuclides released are short-lived and physically decay before they reach the mouth of the river, but during routine operation of the plant at its height during 1955 to 1965, some 900 curies per day or 25,000 curies per month actually were discharged from the mouth of the Columbia into the Pacific Ocean (Seymour and Lewis, 1964, Mauchline and Templeton, 1964, Osterberg, 1965). The principal radioisotope, ^{51}Cr, can be detected easily in the surface waters of the Pacific off the west coast of North America (Figs. 1 - 3). As shown by the maps of the Columbia River mouth, ^{51}Cr decreases as the river plume emerges into the Pacific, but the distribution of the isotope is markedly different during winter (Fig. 1) and summer (Figs. 2 and 3) because of seasonal differences in the prevailing currents. Other isotopes may behave quite differently from ^{51}Cr, however, because of physical-chemical properties or biological phenomena. For example, ^{65}Zn is depleted in surface waters of the Pacific by bioaccumulation and absorption onto particulate matter and thus is not readily measured in the water. Radioactive zinc does appear, however, in offshore plankton and shellfish both north and south of the Columbia River mouth. Despite the relatively large amount of radioactivity flowing through the estuarine environment at the mouth of the river, no disruption or disturbance of the ecosystem has been yet detected. Nor has any disturbance of the ecological balance been noted in the river proper or in the coastal waters just offshore from the Pacific Northwestern United States. The entire system, however,

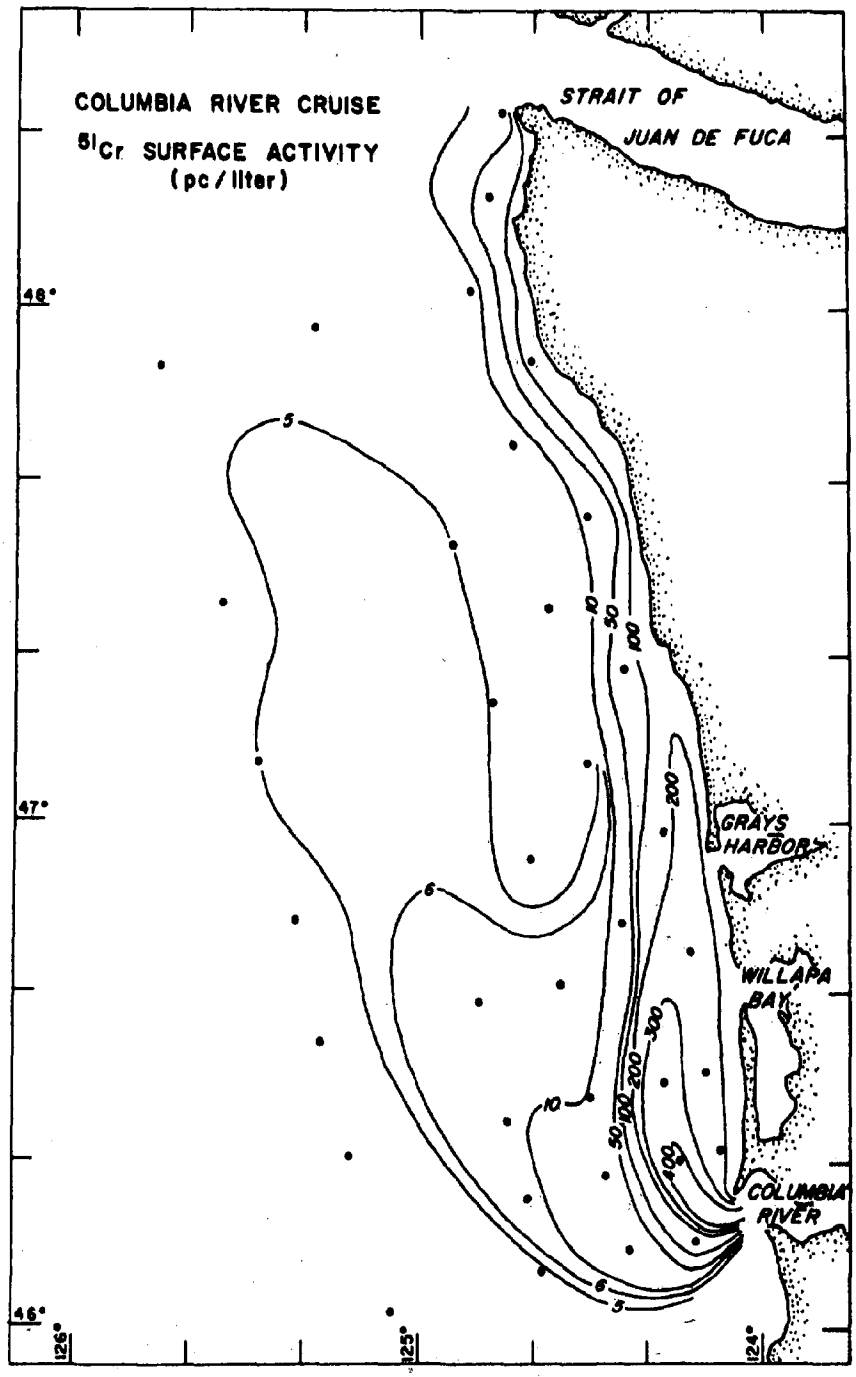


Fig. 1. Dispersal of ^{51}Cr in surface waters off the mouth of the Columbia River, February 1966 (Frederick 1967).

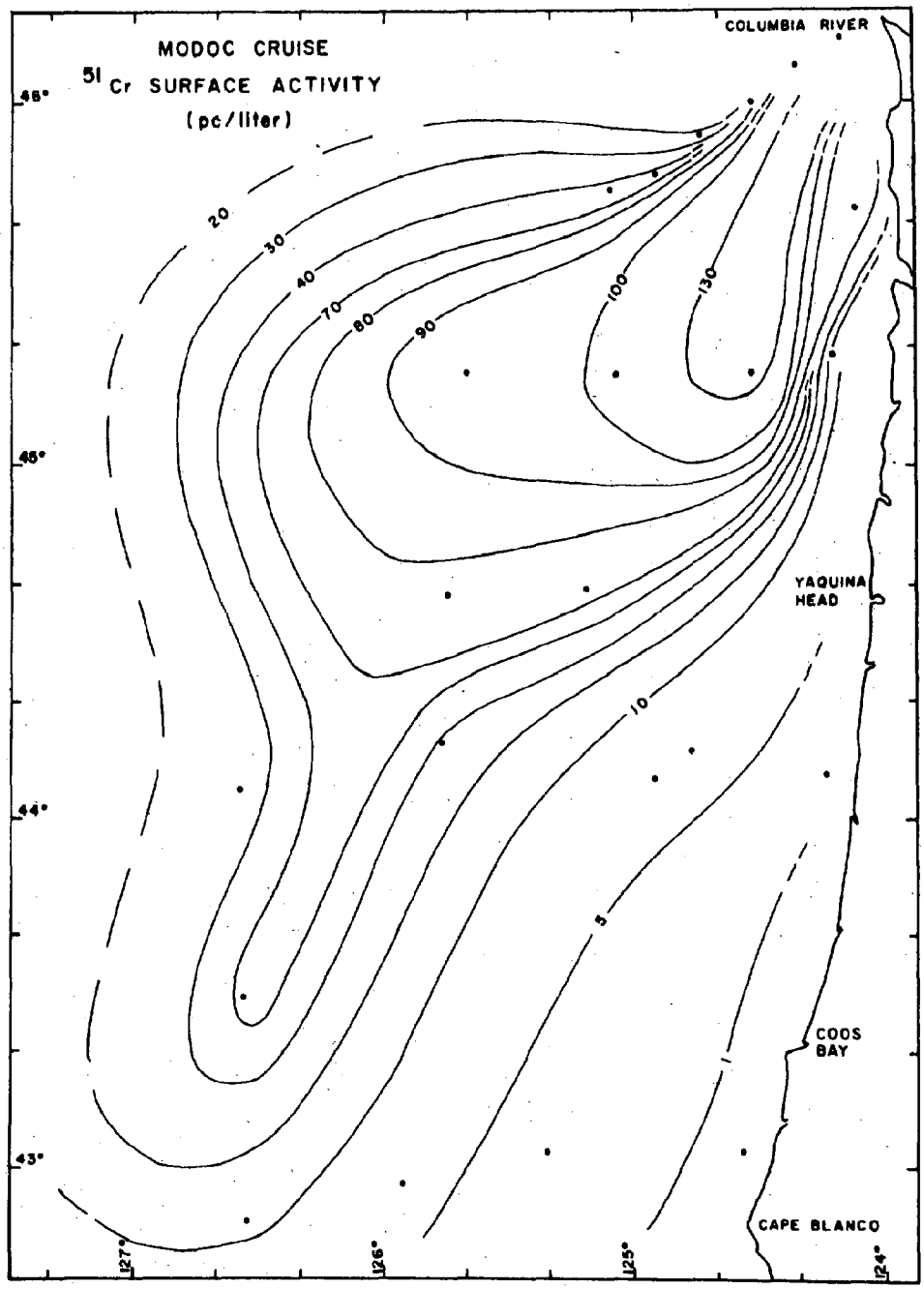


Fig. 2. Dispersal of ^{51}Cr in surface waters off the mouth of the Columbia River, June 1965 (Frederick 1967).

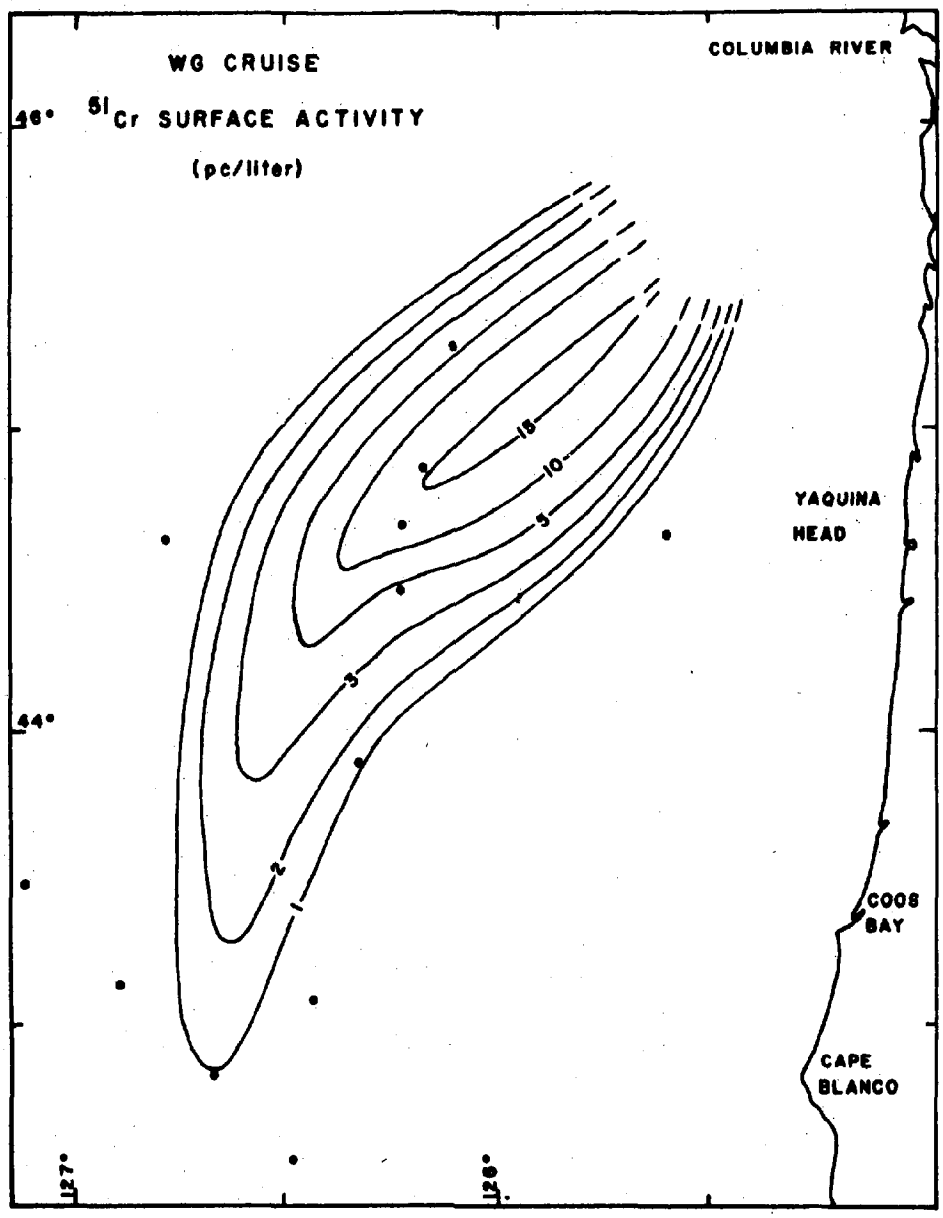


Fig. 3. Dispersal of ^{51}Cr in surface waters off the mouth of the Columbia River, August 1966 (Frederick 1967).

has been under continuous surveillance for several years to ascertain just what the distribution, cycling, and effects of radioactivity would be in the freshwater, estuarine, and marine systems concerned. Some of the observations during those years of surveillance are discussed here.

Introduction of Radionuclides into the Columbia River

In addition to the radioactivity introduced by Hanford, the Columbia River contains typical amounts of naturally-occurring isotopes and fallout isotopes, but these are really insignificant relative to the reactor effluent. At the Hanford facility, the intake water is treated with sodium dichromate to decrease corrosion within the reactor tubes and then undergoes alum flocculation and filtration to remove particulate matter (Silker, 1964). Dissolved materials, including corrosion products from the reactor plumbing and the added dichromate, are subjected to the intense neutron flux in the reactor core as the coolant water passes through, and a wide array of radionuclides are induced. The neutron-induced radionuclides are supplemented in the effluent water by smaller quantities of fission products, either from fuel element leakage through ruptures in the casings or from fission of the very small amounts of uranium occurring naturally in the coolant water. Four hours after release from the plant, 98-99% of the radioactivity is from about 20 radioisotopes (Tables 2 and 3), a mixture of as many as 80 constitutes the remainder (Junkins, et al. 1960, Nielsen, 1963). Most of the radionuclides released into the river are short-lived and decay before the released activity enters the estuary 260 miles downstream some 10-14 days later.

Although the expected composition after 2 weeks of physical decay shows ^{51}Cr to be 89.7% of the radioactivity (Table 3), the actual composition of the radioisotopes measured in the water at Vancouver, Washington, is: ^{51}Cr - 96.5%, ^{65}Zn - 2.2%, and ^{32}P - 1.1% (Wilson, 1964). This apparent enrichment of ^{51}Cr is probably caused by the preferential retention of ^{65}Zn and ^{32}P by sediments and organisms in the river. Zinc 65 is mostly particulate when it enters the estuary at Vancouver, and the maximum concentration of ^{65}Zn in the river at Vancouver occurs during late May when the river is at its peak flow and maximum scouring of sediments is expected (Nelson, Perkins, and Nielsen, 1964).

The continuous flow of ^{65}Zn, ^{51}Cr, and ^{32}P into the Columbia River estuary and adjacent Pacific Ocean results in a reservoir of each isotope in the marine environment. The size of the reservoir is determined by the rate of introduction (Table 4) and the physical half-life of each isotope. The size of the reservoir will remain constant when the loss by decay equals the input from the river. If the average daily inputs for 1966 (Table 4) were sustained indefinitely, the resulting reservoirs would be about 190 Ci of ^{32}P, 17,000 Ci of ^{51}Cr, and 7,400 Ci of ^{65}Zn (Essig and Soldat, 1967).

The reduction in radioactivity released during 1966 was the result of a 1 1/2 month shutdown of all reactors due to a strike and probably also from the permanent shutdown of two production reactors during 1965 (Fig. 4). Additional decreases in the release rate would be anticipated from the permanent shutdown of two more production reactors in 1967 and 1968. At present, only three production reactors continue to operate with Columbia River water as the primary coolant (compared to eight during 1955-1964) (Fig. 4), and this number will likely be decreased soon. The N reactor, which is used for materials production

Table 4

Daily transport of selected radionuclides from Hanford into the Pacific, averaged annually

	1959[1]	1960[2]	1961[2]	1962[3]	1963[3]	1964[4]	1965[4]	1966[4]
	(Ci/day)							
^{32}P	15	17	29	13	12	12	11	9
^{51}Cr	1,030	850	840	650	860	860	800	430
^{65}Zn	21	38	44	29	28	44	49	21
^{239}Np	76	72	67	31	--[5]	--[5]	--[5]	--[5]

[1] Junkins, et al. (1960) Averages from Appendix B-12
[2] Nelson (1962) Table VIII
[3] Wilson (1964) Table III
} Flow rate past Vancouver, Washington

[4] Essig and Soldat (1967) Table III - Flow rate over Bonneville Dam.

[5] No data reported.

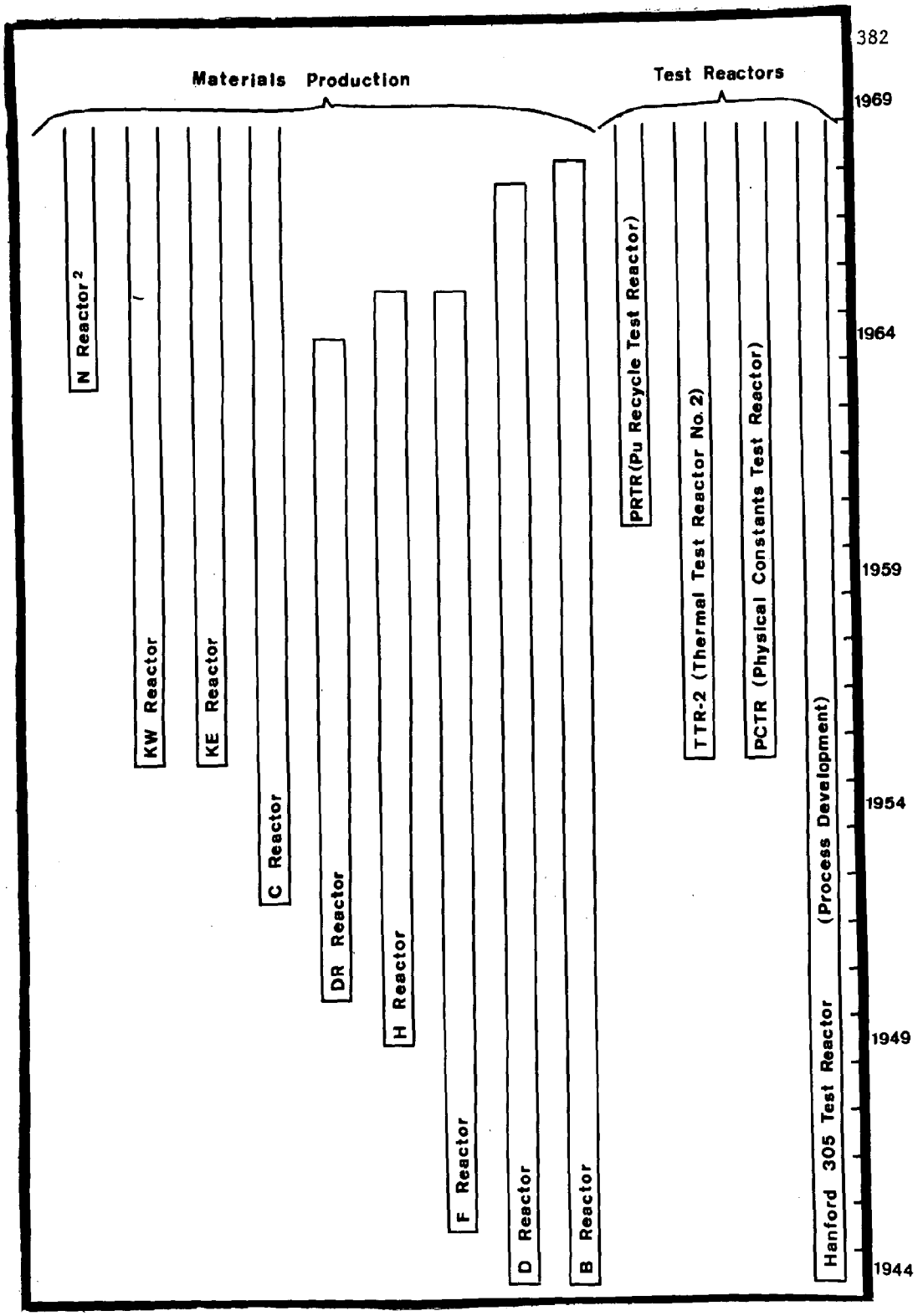


Fig. 4. Summary of reactor operations at Hanford (from AEC 1968a). (1) Test reactors have very low power outputs and radioactivity released is negligible. (2) The N reactor uses recirculating demineralized water for a primary coolant and radioactive wastes resulting from its operation are much less than for other production reactors.

and electrical power generation, is cooled by recirculating demineralized water, so that the amount of radioactive waste released during routine operation is very much lower than that from the older production reactors. The wastes released from the operation of the test reactors at Hanford are negligible.

Distribution of Radioactivity in the Columbia River System

Radionuclides entering the Columbia River at Hanford may remain in solution unchanged as they flow downstream and through the estuary into the Pacific; or they may be absorbed onto suspended particulate matter and sediments, they may undergo changes in chemical or physical state as the river environment changes to estuarine and marine, or they may enter biological food chains at any trophic level. Whether absorbed on sediments or accumulated by organisms, radionuclides are thus retained, or concentrated, in the freshwater and estuarine portions of the river before reaching the ocean. The phenomenon results in an increased exposure to radiation for organisms inhabiting the aquatic environment and also for terrestrial organisms (including man) that rely upon the river and estuary for food, water, or recreation.

The physical states of several effluent radionuclides change before the nuclides enter the estuary at Vancouver, Washington (Table 5, Perkins et al. 1966). Many of the isotopes analyzed enter the river mainly in the soluble form, but are largely particulate when they reach the estuary. In addition, isotopes of some elements, such as iron and ruthenium, probably form insoluble hydrates upon entering the estuarine environment. Since ^{51}Cr, ^{106}Ru, ^{124}Sb, and ^{140}Ba are not strongly associated with particulate matter in the river proper, the input of these nuclides is fairly constant to the estuary. The input of particulate elements, such as ^{65}Zn, however, fluctuates with stream flow. When the flow rate increases, the amount of ^{65}Zn, entering the estuary at Vancouver may equal or exceed the amount released at Hanford, because of the transport of resuspended sediments containing ^{65}Zn (Perkins et al. 1966, Nelson et al. 1964). The distributions of ^{65}Zn and ^{51}Cr absorbed on the sediments of the Columbia River estuary are shown in Figure 5 and 6 for periods before and after peak flow of the river (Jennings, 1966). After the Christmas week flood of 1964, the radioactivity in the sediments decreased at the upstream stations and the ^{65}Zn-labeled sediments were washed farther downstream. The distribution of particulate radioactivity may be an important determinant of the radiation effects in the estuary, because radionuclides associated with particulate matter may be filtered from suspension by filter-feeding invertebrates and fish, as will those accumulated by phytoplankton. See Figs. 7 and 8 for characteristic animals and hydrography.

Freshwater phytoplankton concentrate ^{32}P 5,000-118,000 times and ^{65}Zn 300-19,000 times over the isotopic concentrations in the water, with the highest values occurring during winter when the concentrations of ^{32}P and ^{65}Zn in the water are lowest (Cushing, 1967a). Marine phytoplankton also concentrate ^{65}Zn and ^{32}P, but the zooplankton are usually not separated before analysis. A number of radionuclides have been measured in a wide variety of marine and estuarine organisms near the mouth of the Columbia River. Zinc-65 is usually the dominant isotope. Although ^{51}Cr is the most abundant radionuclide in the lower Columbia River and estuary, it appears infrequently and erratically in the marine biota (Watson et al. 1963, Seymour and Lewis, 1964). Neptunium-239, also present in the water entering the estuary (Table 4), is usually absent in organisms.

Table 5

Physical state of Hanford effluent radionuclides[1]

Radionuclide	Percent soluble	
	In effluent	At Vancouver
^{46}Sc	64	11
^{51}Cr	97.6	92.4
^{54}Mn	97.4	12
^{58}Co	95.7	12
^{59}Fe	36	20
^{60}Co	98.2	9
^{65}Zn	98.2	24
^{95}Zr-^{95}Nb	31	15
^{106}Ru	68	83
^{124}Sb	98.9	94.1
^{140}Ba	97.7	63

[1] Modified from Perkins et al. (1966).

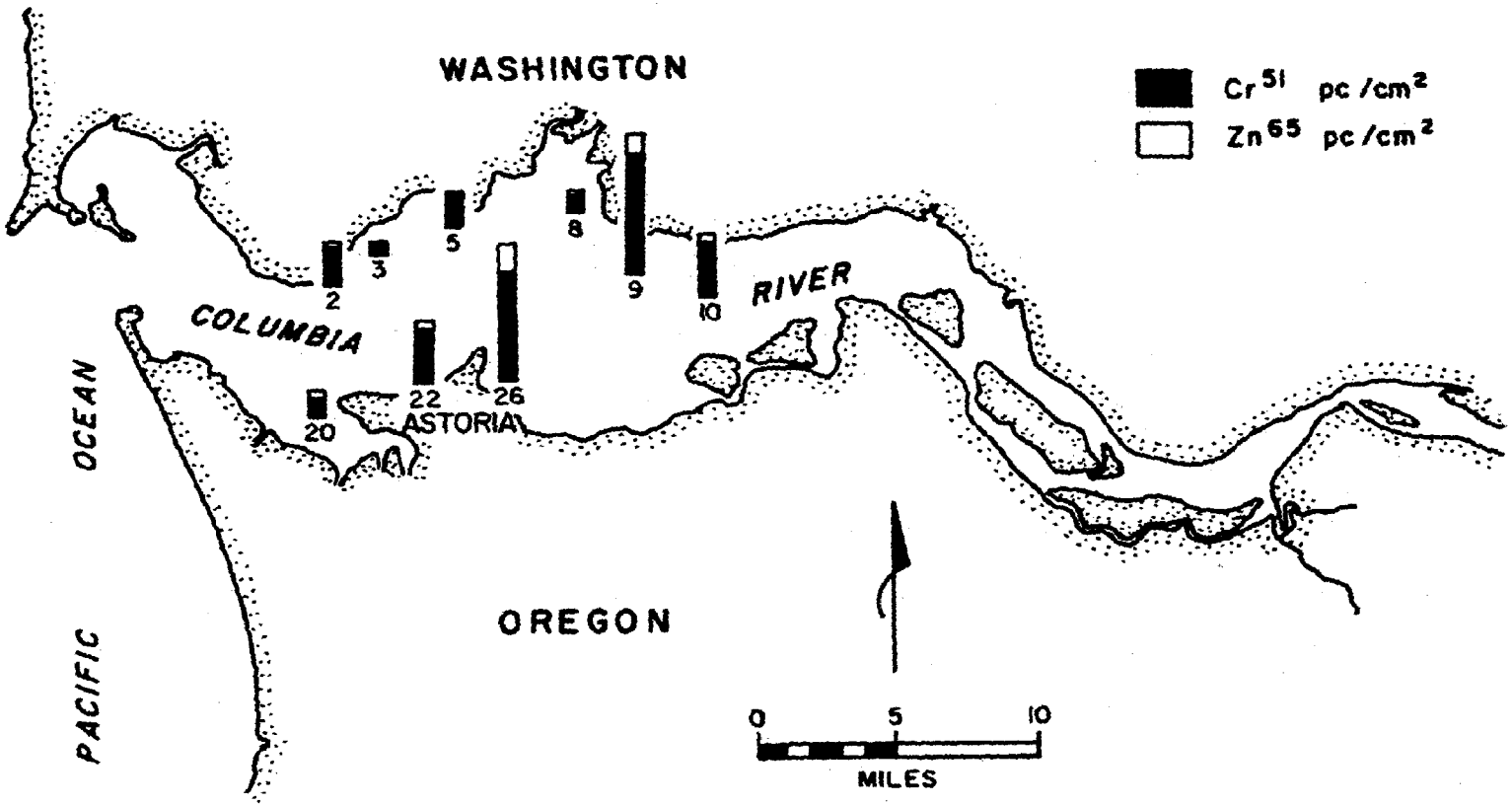


Fig. 5. Radioactivity adsorbed on sediments in the Columbia River estuary before the flood of Christmas week, 1964 (Jennings, 1966).

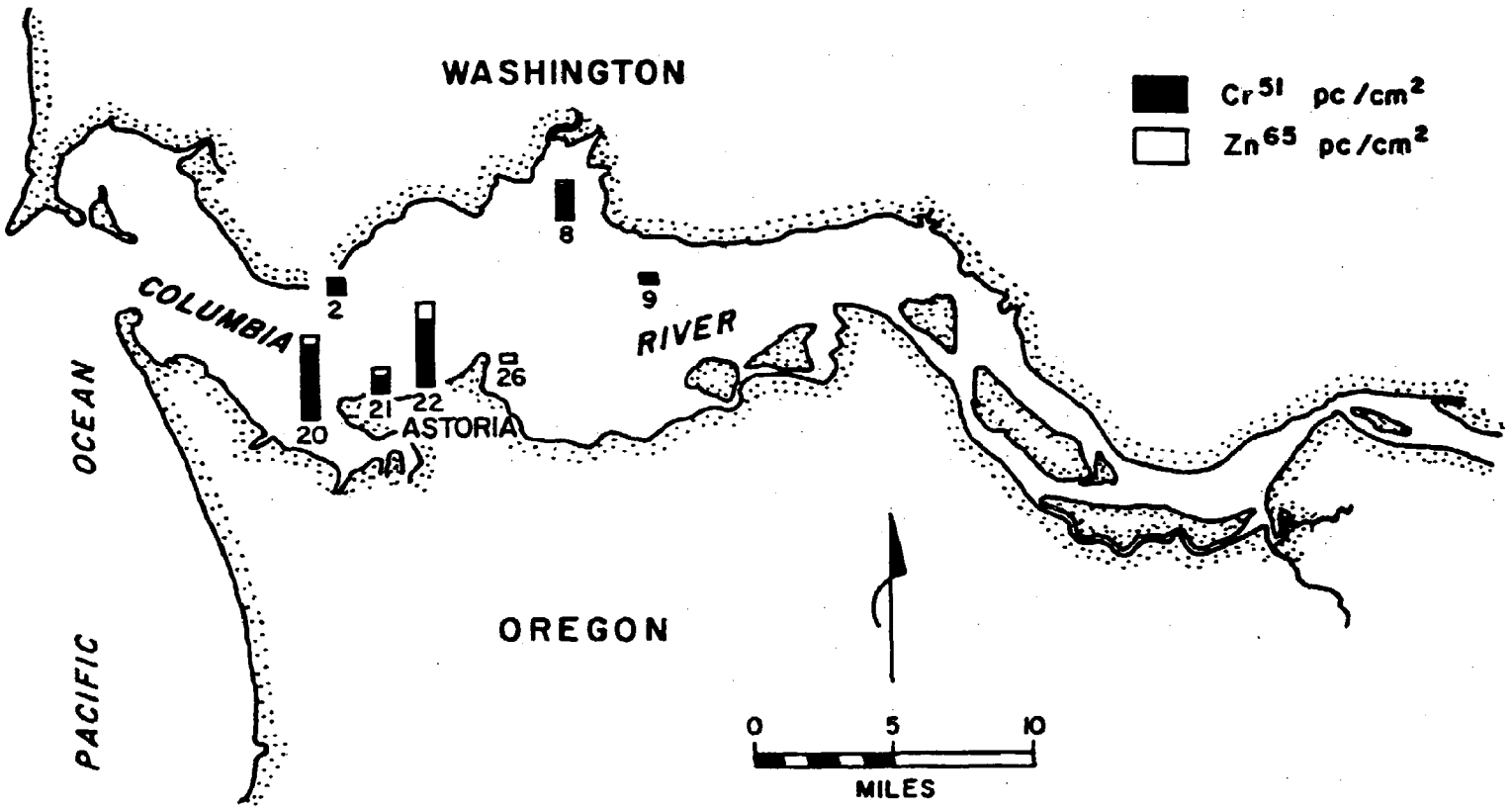
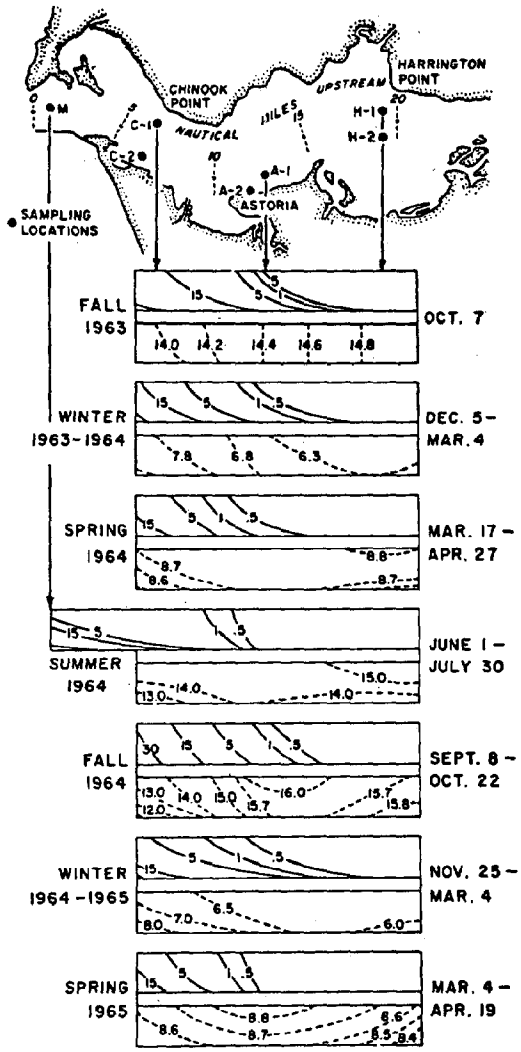
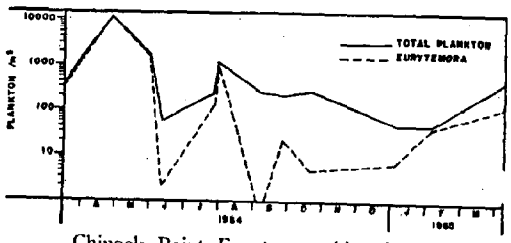


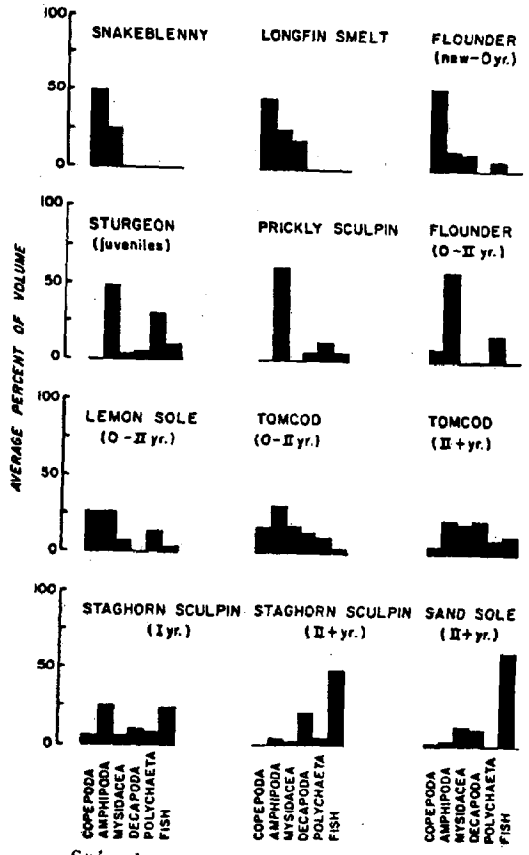
Fig. 6. Radioactivity adsorbed on sediments in the Columbia River estuary after the flood of Christmas week, 1964. Note the decreased ^{65}Zn and ^{51}Cr at the upstream stations (compare with Fig. 5) (Jennings, 1966).



Location of sampling stations, seasonal salinities in ‰ (solid lines), and seasonal temperatures in °C (dotted lines) in the Columbia River estuary.



Chinook Point *Eurytemora hirundoides* populations in relation to total plankton abundance.



Stomach contents of various fishes in the Columbia River estuary.

Fig. 7 and 8. Salinity, zooplankton, and bottom fishes in the mouth of the Columbia River (Haertel and Osterberg, 1967)

Isotopes of cobalt detected in organisms at the mouth of the Columbia probably also arose from Hanford operations, but other nuclides which occur in organisms, including ^{95}Zr-^{95}Nb, ^{103}Ru, ^{106}Ru, ^{141}Ce, ^{144}Ce, ^{54}Mn, ^{140}Ba-^{140}La, and ^{137}Cr, are introduced also by fallout.

Tracers in Experimental Marine Ponds

The interaction of the plants, animals, and micro-organisms in processing radioactive tracers in shallow marine ecosystems is suggested by graphs in Figs. 9 and 10 from Duke et al. (1967). They indicate that radioactive zinc, added to the water, moved into the food chains and was partitioned among the living components. Peak quantities arrived with different time lags in different species according to their size and metabolic activities with respect to zinc.

Although the highest specific activities occurred in filter-feeding shellfish, more than 99 percent of the isotope was found in the sediments 100 days after introduction. This result suggests that estuarine sediments will become a principal reservoir for discarded radioisotopes, and that benthic organisms, especially filter-feeders and deposit feeders, might receive greater doses of radiation than would planktonic or pelagic forms.

Radioactive Waste in the Irish Sea

Figures 11 and 12 show some characteristics of the dispersal of radioactivity in the Irish Sea where about 7,500 curies of fission products are discharged monthly about 2 miles offshore from the Windscale facility near Sellafield, England (Mauchline and Templeton 1964). These wastes arise from the chemical processing of nuclear fuel and consist mainly of longer-lived fission products (^{89}Sr, ^{90}Sr-^{90}Y, ^{95}Zr-^{95}Nb, ^{144}Ce, ^{137}Cs, ^{103}Ru, ^{106}Ru), along with ^{238}U and ^{239}Pu (Longley and Templeton 1965). The rate of discharge of the low-level waste to the sea is governed by the concentration of ^{106}Ru in seaweeds of the genus Porphyra, which are consumed as laverbread in South Wales (Preston and Jeffries 1967). If mans' routine harvest of Porphyra from the shores of the Irish Sea is considered a normal functional component of the ecosystem, then the disposal of radioactive wastes from Windscale must be viewed as a potential stress. By continuous monitoring of the concentrations of ^{106}Ru in Porphyra, however, it is possible to control the amount of effluent released below objectionable levels. No direct effects of radioactivity on the organisms or environment in the Irish Sea have been detected.

Radioactivity in Seaweed Deposited on Beaches

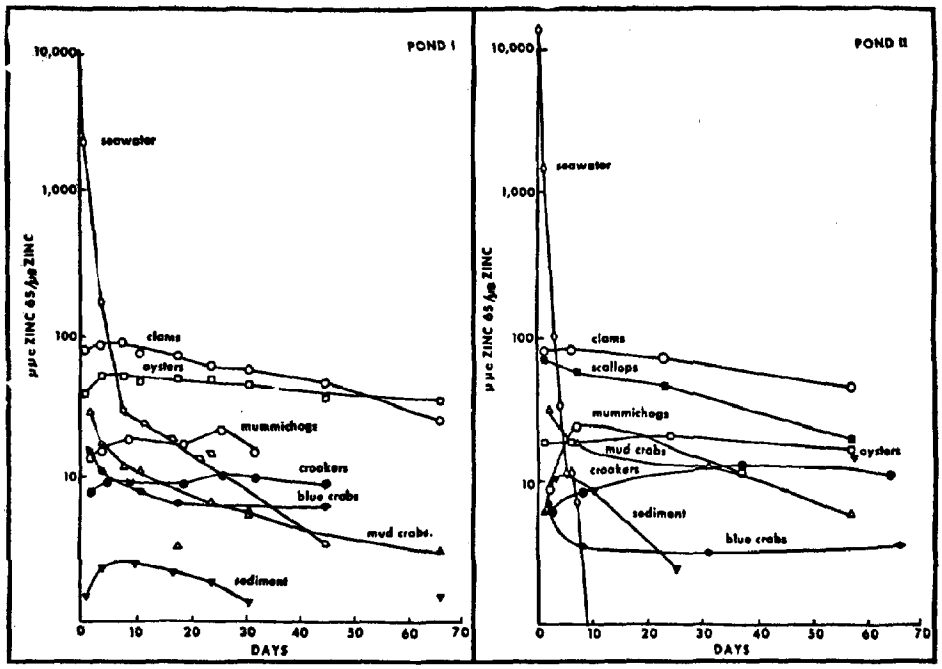
In Figs. 13 and 14 from Shelby (1963) and Angino et al. (1965) are data demonstrating the concentrating action of large weights of Sargassum seaweed deposited over a 100 mile stretch of Texas beach. Several components of radioactive fallout, with low levels of radiation, were deposited over many square miles of the Gulf of Mexico. They were swept from the Gulf and concentrated on the Texas beaches in the Sargassum weed. The example illustrates the kind of problem of stress that might result if large quantities of radioactivity were released near shore.

Estuarine Radioactivity from the Proposed Nuclear Construction of a Sea-level Canal

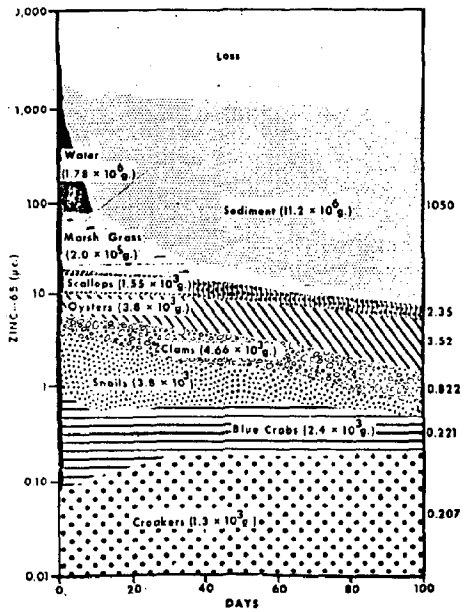
An intensive survey of the distribution of stable elements was undertaken to evaluate the hazards to man from radioactive contamination of the marine and estuarine environments adjacent to the Isthmus of Panama if a sea-level canal were excavated with nuclear devices (Lowman 1967). Radiation effects on the ecosystems per se or on resident organisms were not considered directly in this study, since the criterion of radiation hazard was the maximum specific activity of each radioisotope allowable in man. It was estimated (Lowman 1969a) that 22 radioisotopes would be injected into the hydrosphere at specific activities greater than allowed in man. Most of these would rapidly be diluted by the seawater to tolerable specific activities. The greatest hazards to man would probably arise from ^{3}H, ^{32}P, ^{54}Mn and ^{131}I, but for organisms in the environment, the external radiation dose from these and other isotopes, including ^{24}Na, ^{56}Mn, and ^{203}Pb, would probably be most significant. The potential radiation doses and resultant effects on aquatic populations have not been estimated, but such effects might be overshadowed by the simultaneous effects of siltation from direct close-in fallout and from runoff from freshly excavated spoils.

Effects of Radioactive Contamination in Estuaries

The only conclusive examples to date of effects of radioactive contamination on aquatic ecosystems are associated with test sites, such as Bikini and Eniwetok. Gorbman and James (1958) detected thyroid damage in fish which had accumulated high levels of ^{131}I through a food chain. Under normal circumstances, however, the accumulation of ^{131}I will present no problems, because the isotope has a short half-life (8 days) and there is a substantial amount of stable iodine (about 50 µg/l) available in typical sea water. Other instances involve the modification of human use of fishery resources because of radioactive contamination. The Japanese fishing fleet had to dispose of nearly 500 tons of tuna contaminated with radioactivity after Pacific nuclear tests in 1956 (Polikarpov, 1967). Returning Bikini Islanders also were restricted in their consumption of coconut crabs

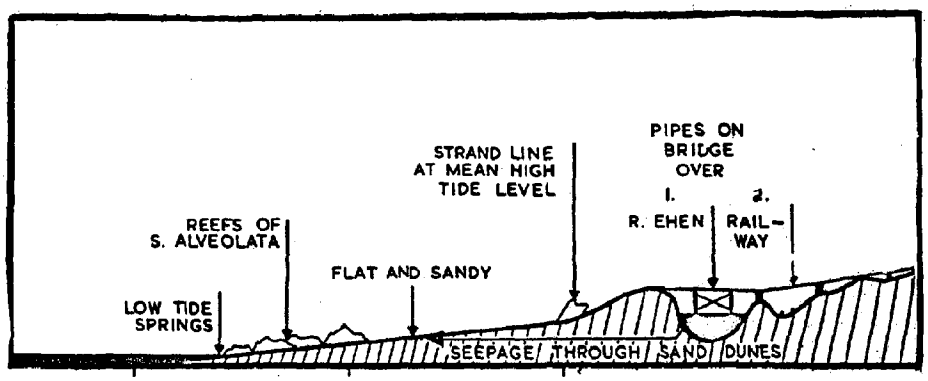


Specific activity of components in experimental ponds. The specific activity of the water at the time zinc 65 was introduced into the pond ("0" days) was 18,100 $\mu\mu$c zinc 65/μg zinc in Pond I and 14,200 in Pond II.

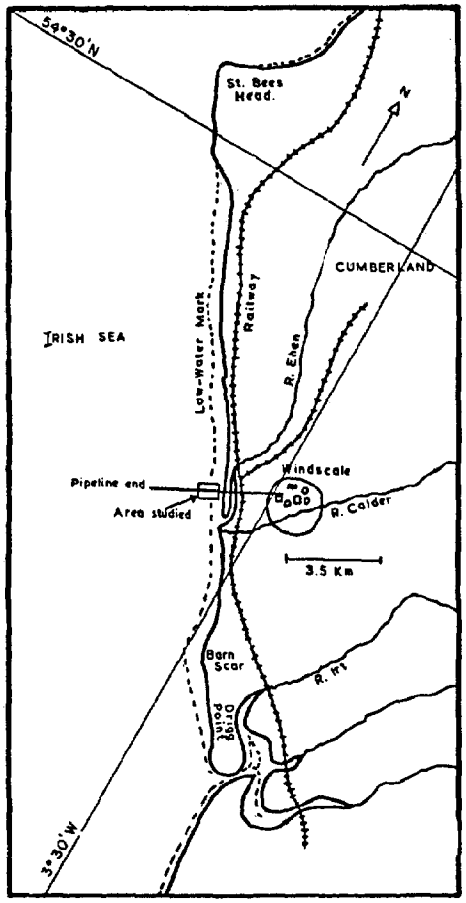


Distribution of zinc 65 in Pond II with time. A total of 10,000 μc is in pond or accounted for by loss from tidal exchange at any specific time. (Amount of zinc 65 in each component after 100 days is shown at right margin.)

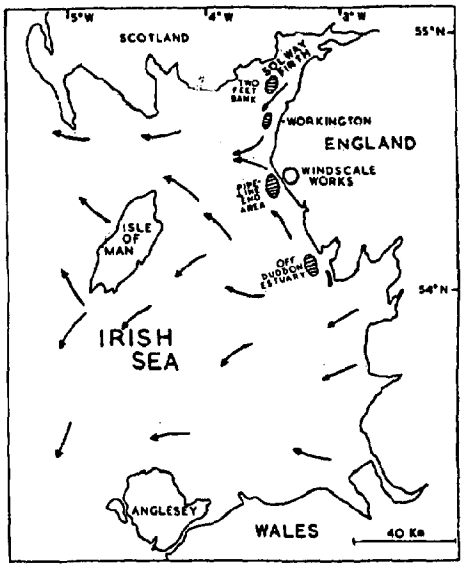
Fig. 9 and 10. Fate of radioactive zinc in marine ponds (Duke, Price, and Willis, 1967).



Section through the beach at Windscale showing the relationship of the River Ehen to beach.

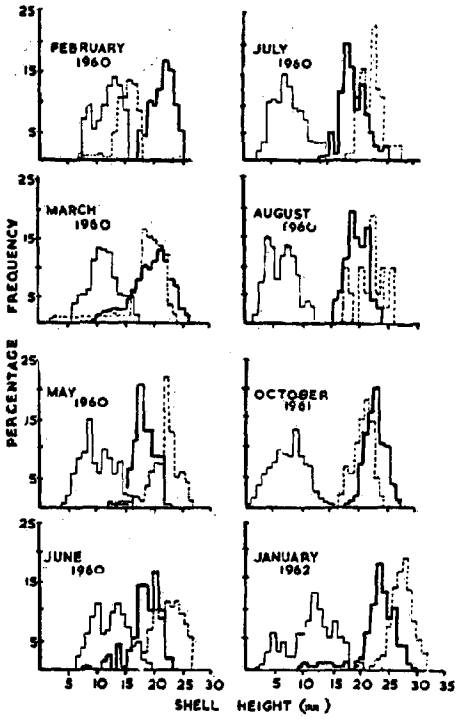


General map of coastline near Windscale showing the area studied (Fig. 1).

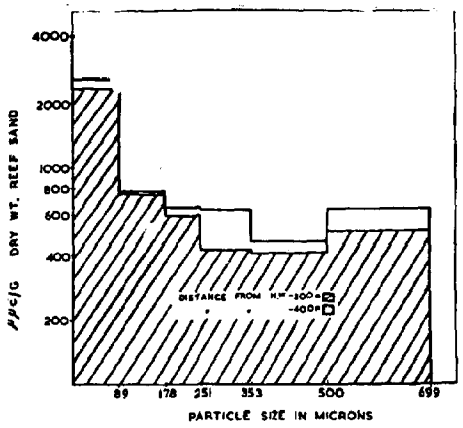


Map of Irish Sea showing areas where rays were sampled. The arrows indicate the direction of maximum ebb streams; direction of maximum flood streams is the opposite.

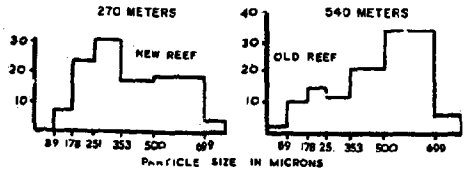
Fig. 11. Release of radioactivity into the Irish Sea across an intertidal zone containing a worm tube reef (Mauchline, Taylor, and Ritson, 1964).



Size distributions of *Littorina littorea* on the upper beach (thin line), middle beach (heavy line), and lower beach (broken line).



Beta activity in the particle grades of *Sabellaria* reef at two distances from mean high-tide level.



Distribution of particle grades in the sand of the reefs of *Sabellaria alveolata*, by percentage weight.

Fig. 12. Characteristics and radioactivity of a worm tube at Windscale, England. Reef builder is Sabellaria alveolata (Mauchline, Taylor, and Ritson, 1964).

Table 6. Chemical content and radioactivity of intertidal life at Windscale, England (Mauchline, Taylor and Ritson, 1964).

Ash content and dry weight (in parentheses), expressed as percentages of wet weight, of algae at three points on the shore at Windscale

Species	Upper shore	Middle shore	Lower shore
Enteromorpha spp.	10.2 (21.2)	10.7 (17.6)	6.1 (12.5)
Ulva lactuca	5.0	4.8	4.6
Fucus vesiculosus	13.0 (30.7)	7.6 (18.0)	3.8 (21.4)
Gigartina stellata	21.7	15.1	13.3
Porphyra umbilicalis	7.1 (19.5)	4.7 (16.2)	5.7 (13.9)

Beta activity, μμc K equivalent/g ash weight, in animals at three points on the shore at Windscale

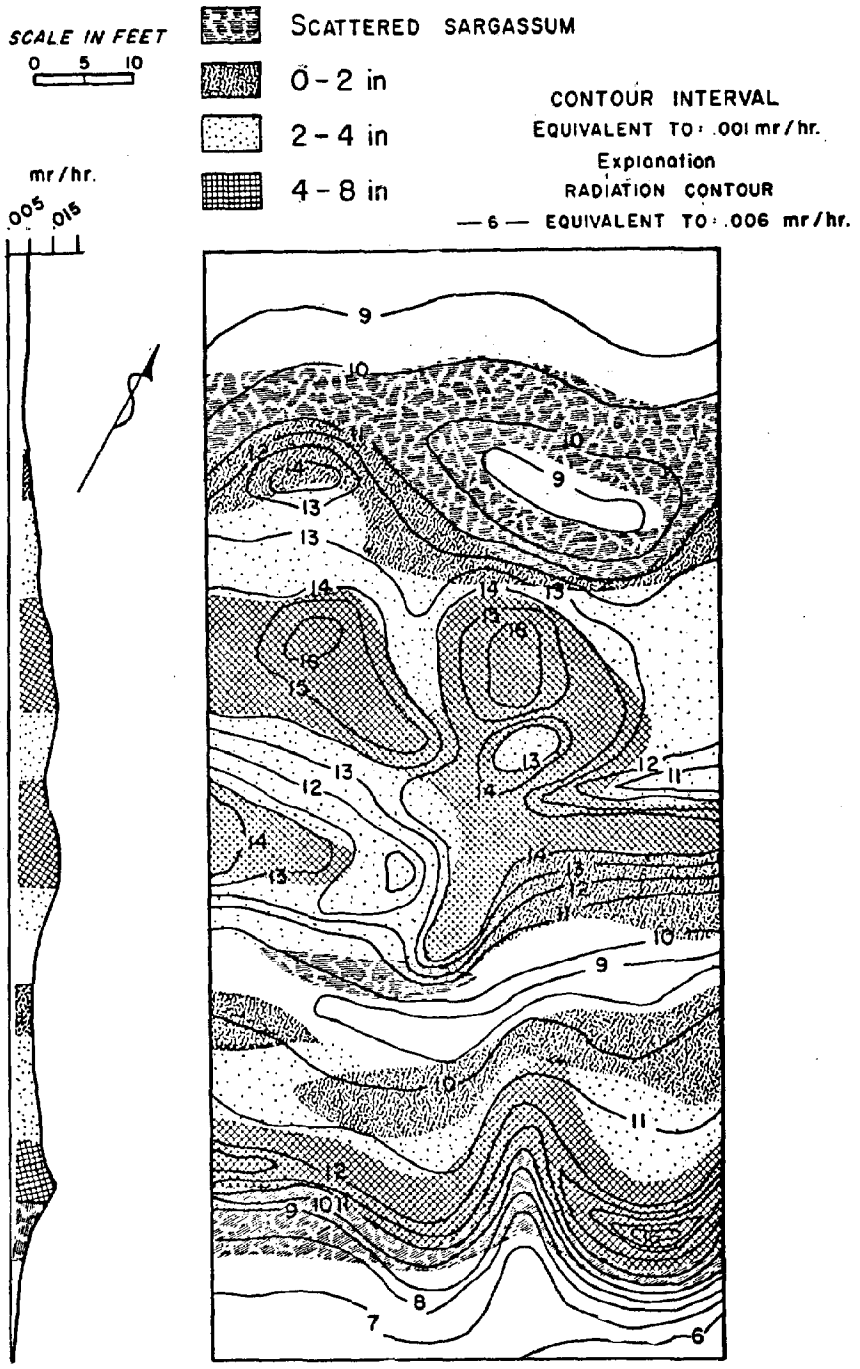
Species	Upper shore	Middle shore	Lower shore
Sabellaria alveolata	4,460	4,980	5,560
Balanus balanoides	260	190	275
Littorina littorea	5,200	6,000	19,000
	540	960	6,000
	520	600	1,900
	500	490	1,010

Beta activity, μμc K equivalent/g ash weight, in Sabellaria alveolata *collected from brackish and seawater pools. Beta activity of reef sand in parentheses*

Meters seaward of mean high-tide level	Salt water	Brackish water
300	4,460 (378)	
320		1,880 (567)
340	4,980 (713)	3,590 (565)
360	5,060 (504)	3,000 (465)
380	5,560 (696)	

Mean levels of β activity in species at different trophic levels

Trophic level	Species	μμc K equivalent/g wet weight
Tertiary consumer trophic level	*Centronotus gunnellus*	11
Secondary consumer trophic level	*Blennius pholis*	19
	Nucella lapillus	29
	Carcinus maenas	98
Primary consumer trophic level	*Patella vulgata*	41
	Littorina littorea	32
	Mytilus edulis	34
	Balanus balanoides	91
	Sabellaria alveolata	88
Producer trophic level	*Ulva lactuca*	119
	Enteromorpha spp.	139
	Fucus vesiculosus	46
	Porphyra umbilicalis	126



Contour map of radiation values in milliroentgens per hour on a section of *Sargassum* covered beach 6 mi SW of Galveston, Texas; July, 1962; shaded areas indicate thickness of *Sargassum*. Profile at left is from the extreme left side of large section. Arrow points north.

Fig. 13. Radioactivity of a section of the heavy drift of Sargassum on Gulf beaches in 1962 (Shelby, 1963).

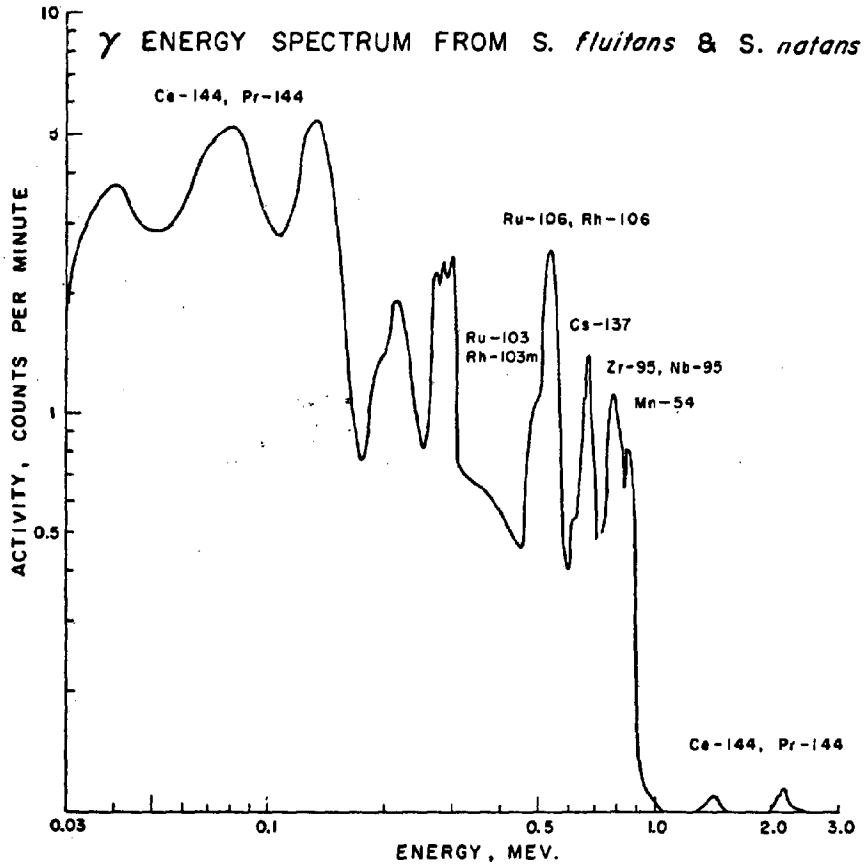


Fig. 14. Gamma spectrum identifying the radioactive elements in the Gulf Sargassum (Angino, Simek and Davis, 1965).

because of the high content of residual ^{90}Sr(USAEC 1968c). Effects of chronic exposure to low levels of radiation from radioisotopes in the environment are unknown, and it is necessary to extrapolate from results of experimental work, which has generally been with acute exposure to relatively high levels of radiation.

In the estuarine ecosystem, the effects of radiation depend on environmental conditions. Thus, the estuarine fish, Fundulus heteroclitus, had an LD-50 (50 days) of nearly 2,000 rads at temperature-salinity combinations of 12° C, 15 °/oo and 22° C, 5 °/oo; 920 rads at 22°, 25 °/oo; and only 300-350 rads at 27° C, 5-25 °/oo (Angelovic et al. 1968). Since estuarine conditions vary over wide ranges, organisms may successfully tolerate various levels of radiation at different times. It is of interest to note that the sensitivity noted above for Fundulus (300-350 rads) is very comparable to the radiation sensitivity of mammals, including man. Tolerance levels for most estuarine invertebrates, however, are considerably higher than for mammals (White and Angelovic,1966). Since the regulation of waste disposal is based indirectly on estimates of radiation exposure to human populations, and since aquatic organisms living continuously beneath radioactive effluents will be subjected to much higher doses of radiation than will man, it is easily conceivable that estuarine populations under certain environmental conditions might experience radiation-induced genetic changes. This premise is subject to considerable argument because it involves extrapolations from mammals to fish and invertebrates and from lethal doses of acute radiation exposure to the subtle genetic effects of continual low-level exposure, but much research effort must be expended before the premise can be proved valid or invalid.

Radioactive contamination of the marine environment will subject marine organisms to a continuous elevated exposure to external radiation. Thus all life stages, including the most sensitive ones, will be irradiated, and populations may be affected by somatic changes in immature forms or by genetic changes in the breeding stock. Results of experimental work to date, however, are difficult to relate to these potential effects, because most experiments have involved relatively short-term doses much higher than will occur in the environment. Any increase in the level of background radiation might produce slow, gradual changes in the population structure of marine ecosystems and in the genetic makeup of the species involved. Such changes may be undetected and it probably will be difficult or impossible to establish a relation between increased radiation and observed changes in natural populations.

EVALUATION OF POTENTIAL DISTURBANCE PATTERNS

The introduction of radioactive materials into estuaries could feasibly alter the natural order of the system in four ways, which vary considerably in probability of occurrence and disruptive capacity:

1. Somatic damage (including death) of estuarine biota.
2. Increase in genetic mutation rates of populations.
3. Increase in growth rate and maximum size of organisms.
4. Reorientation of human uses of estuaries.

These effects are discussed in the ensuing paragraphs.

Somatic damage to estuarine organisms is extremely improbable under the present regulatory procedure for controlling radioactive waste disposal. Although certain life stages, especially from the zygote stage through larval forms, are more susceptible to radiation damage than are adults, radiation exposure from permissible levels of radioactive wastes in aquatic environments probably will never be substantial enough to produce somatic damage. Unfortunately, at this time reliable information on the normal incidence of somatic aberrations, such as tumors and other physical anomalies, are not at all well established for aquatic populations, and it is therefore very difficult to evaluate their sporadic occurrence in systems receiving radioactivity. For example, 15.7% of freshwater mussels Anodonta californiensis, sampled from the Columbia River had abnormal tumors on the foot, mantle, or gills (Pauley, 1967, 1968). The incidence of similar tumors in different species of mussels from other rivers or lakes was lower (4.2-6.5%) but the situation could easily have arisen from species differences or environmental differences other than radiation. No such anomalies have been reported in estuarine organisms exposed to radiation in natural environments, but continued surveillance is certainly prudent.

Aquatic populations are probably much more susceptible to genetic changes than to somatic aberrations because of the high sensitivity of genetic material to ionizing radiation. As was the case with radiation-induced somatic changes, no genetic effects have been traced to an elevated radiation exposure in the natural environment. Establishing that genetic changes have resulted from radiation exposure in estuarine ecosystems will probably be even more difficult than determining causal relationships for somatic changes. Unless the genetic change produces an organism which appears markedly different, a mutation is likely to go unnoticed in the population. Considerable time is required also before a new mutation might be established in a breeding population. If elevated radiation only increases the rate of an already established mutation, many generations may be required before any change is noted in the standing population. Since deleterious mutations are selected against and decreased life expectancy is frequently offset by increased fecundity, changes in the genetic composition of any population may be so subtle that they are undetected until the mutation is firmly established in the population.

Increased growth rates and sizes of various organisms have been noted several times in controlled radiation experiments involving low-level doses (Engel, 1967, Brunst, 1965). If certain organisms are stimulated to faster growth rates with no concomitant decrease in life span, the dynamics of energy transfer in an estuarine ecosystem certainly would be affected, but the nature of the change would depend on so many ecological factors that evaluation and prediction are impossible.

The final type of disruptive influence which radioactive waste disposal may exert on estuaries has received considerably more attention--not because man is concerned about the estuarine ecosystem per se, but because of man's self-protective instincts. Man is the only estuarine-dependent organism with the capability of avoiding excessive radiation, and he can change his recreational and feeding behavior if the need arises. Thus, the potential radiation exposures of persons living near the Columbia River system are continuously evaluated (Foster and Soldat, 1966). The disposal of radioactive materials into the Columbia River so far has not necessitated any modification of human use in the river or estuary; neither has the disposal rate been modified downwards because of excessive human exposure downstream. People living and working in the area, however, do receive radiation significantly above background from several sources, and these sources are monitored frequently to ascertain that no hazard exists. For an individual near Hanford, ^{32}P contained in edible Columbia River fish is the greatest plausible source of radiation. Other potential sources include ^{65}Zn in fish and oysters, short-lived nuclides such as ^{24}Na, ^{76}As, ^{239}Np, and ^{131}I, in drinking water derived from the river, external exposure from swimming in, or fishing from the shore of the river, and inhalation of gaseous radionuclides from the atmosphere. It has been estimated that in 1965 an individual getting the maximum possible dose from all sources in the vicinity of Hanford would have received 14% of the individual dose limit recommended for bone, about 7% of that for the total body, and about 5% of that for the gastro-intestinal tract. By contrast, the typical resident of Richland, Washington, received during 1965 about 1% of the same dose limit for total body and 2.3% of that for the GI tract (Foster and Soldat, 1966). The amount of radioactive material released annually by Hanford has decreased since 1965 (Table 5).

It seems clear that our present system of regulating the release of radioactive materials to freshwater rivers and estuaries is adequate to prevent the sudden disruption of the aquatic ecosystem by radiation. Limits on the environment are now based on the expectancy that the receiving water will be used as a source of drinking water for people and/or that people will consume large quantities of fish, shellfish, or other aquatic or marine foodstuffs produced in the vicinity of the plant discharge. The calculations that lead to the specification of release rates (or maximum permissible concentrations of radionuclides in the effluent) are based on conservative estimates of the behavior of the radionuclides by aquatic and marine food organisms, and on the accumulation and retention of the nuclides in man. Many of the estimates now used are quite tentative and most are probably overly conservative because of a paucity of definitive data. However, the effects of very low levels of radiation are very subtle, and much basic information on the genetic variation in aquatic organisms is also lacking. Continued surveillance and research are therefore required.

To improve the estimates that are now being used, to provide greater assurance of the safety of the limits as now calculated, and to acquire the knowledge needed to cope with unplanned radioactive contamination, additional radioecological research is underway in many laboratories throughout the world. Further research is especially needed on the following: (1) the

fate of radioactive materials discharged into natural waters, (2) the reconcentration of radionuclides by important aquatic organisms, (3) the efficiency of transfer of radionuclides through food webs, (4) the flux of acute releases of radioactive wastes through aquatic communities, and (5) the effects of chronic exposure to low levels of radioactive materials on aquatic organisms, populations, and ecosystems. Continued research in these areas will ensure that we are neither too permissive nor excessively restrictive in our regulation of radioactive waste disposal into aquatic environments, and will safeguard against the inadvertent destruction of natural estuarine systems.

Chapter E-17

MULTIPLE SYSTEM STRESS

Many harbors and estuaries adjacent to urban areas receive wastes and disturbances of more than one type causing waters to develop heterogeneous patches of chemicals, fertilized waters, waters low in oxygen, turbidities, acids, and other conditions alien to normal life of estuarine ecosystems. The mixed pollution situation is possibly the nations most urgent estuarine problem because the condition is a mixture and the causes several. The stress of many different kinds of wastes may be more difficult for an ecosystem to develop adaptations than separate types of wastes acting alone. The continual fluctuations require more kinds of adaptation than there may be food energies to support. Some bays receiving mixed wastes which are primarily nutrients of non-toxic nature may develop extremely high metabolic rates and high rates of photosynthetic production. Such bays are almost micro-organism cultures, but have active larger animal populations too. Potentially such fertile waters are a food producing resource, although we know relatively little about the conditions for management of these mixtures which will channel energies into products of use to man, effectively mineralize the wastes, and stabilize the ecosystem.

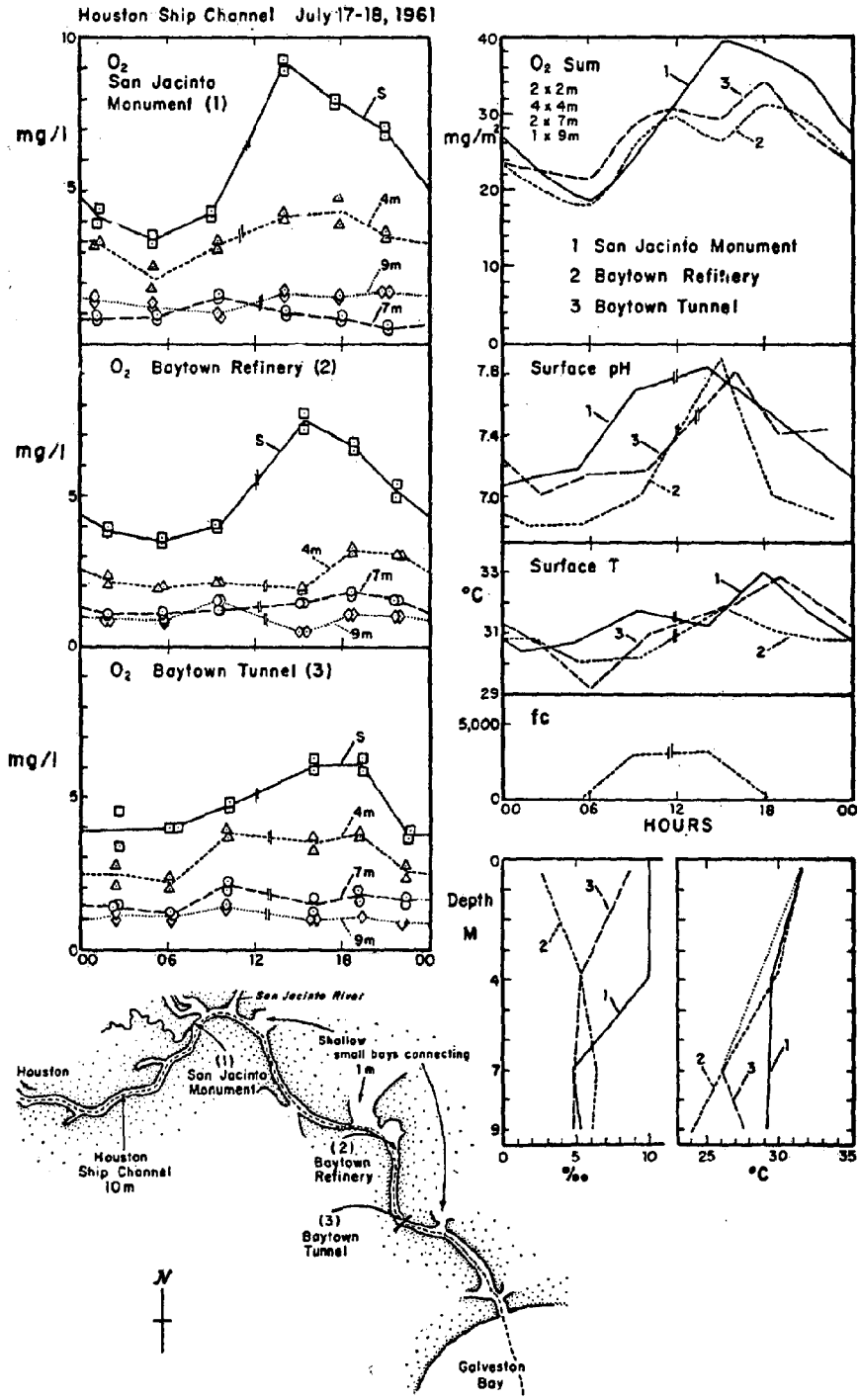
EXAMPLES

Houston Ship Channel

Situated at the upper end of Galveston Bay, Texas is the Houston Ship channel along which are located dozens of major industries that release wastes. Refineries, petrochemicals, sanitary wastes, and many others go into waters that pass out into Galveston Bay. The dredged channel is 40 feet deep, floored with waste sludge and generally black, and sometimes stratified with more saline waters on the bottom. Conditions are patchy, often low in oxygen, and often with high concentrations of oxidants and reducing compounds. Conditions on one summer day are given in Fig. 1 (Odum, Cuzon, Beyers, and Allbaugh, 1963). Chambers and Sparks (1959) found considerable variety of fishes in these waters particularly in zones which were being mixed with San Jacinto River waters. Such concentrations of wastes in small restricted volumes really make these zones oxidation ponds. Decision making about the service such arrangements provide for or against coastal systems of man and nature requires data about the effect of such waters on the broader areas as this stuff is dispersed and mixes over broad areas. See Chap. C-10.

Corpus Christi Harbor, Texas

Corpus Christi, Texas (Fig. 2) receives wastes of two dozen industries. Waters from the harbor are pumped north through cooling systems (see Chap. A-5) into Nueces Bay. The bottom waters and little circulating innermost harbors develop very low oxygen conditions (Fig. 3a). At times photosynthesis is high (Fig. 3b).



Diurnal oxygen, pH, and temperature patterns at 3 stations in the Houston Ship Channel July 17–18, 1961. Four levels are given at each station and a rough calculation of the total oxygen in vertical column is made for the diurnal period.

Fig. 1. Characteristics of a multiple stressed ship harbor in Texas (Odum, Cuzon, Beyers, and Allbaugh, 1963).

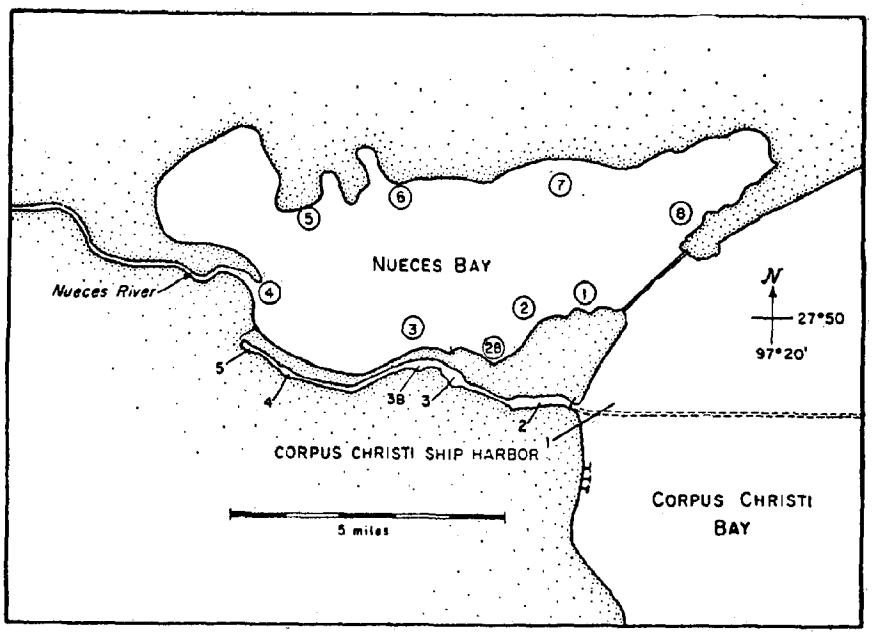
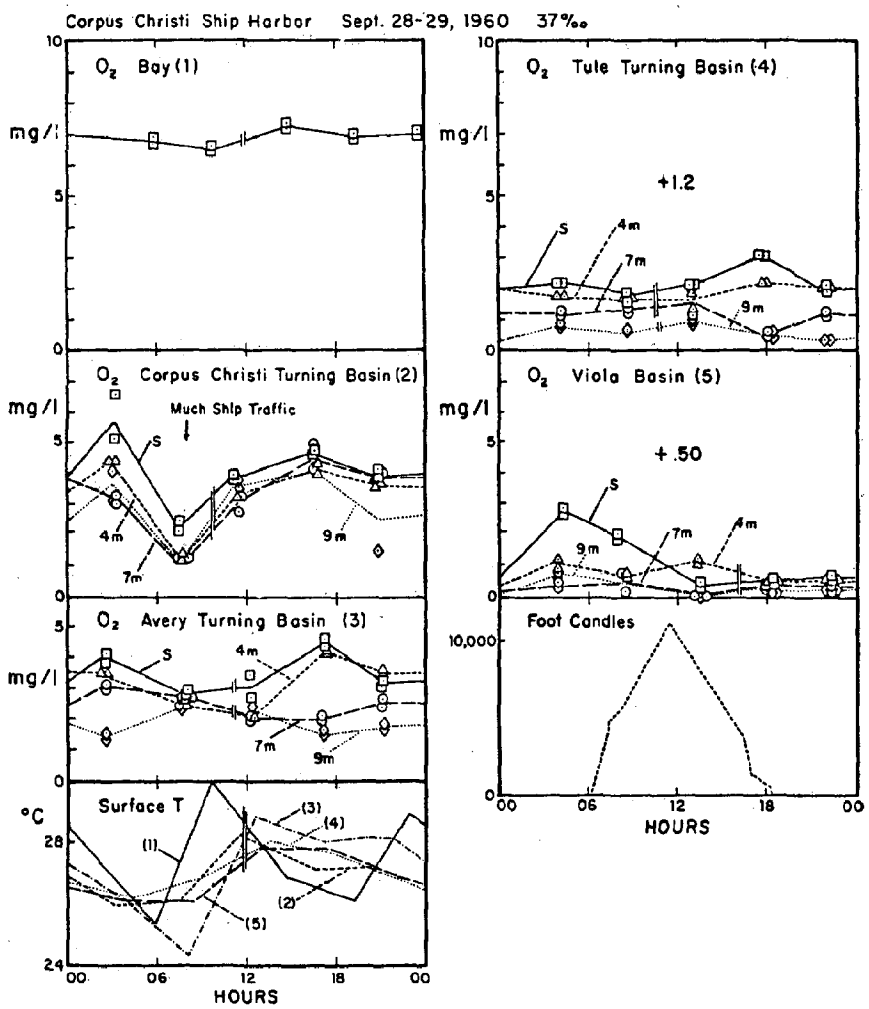


Fig. 2. Corpus Christi Ship Harbor.



Diurnal record of variables for 4 depths and 5 stations in the Corpus Christi Ship Harbor, Sept. 28–29, 1960. Ohle corrections are given (+1.2, +0.5) for stations 4 and 5.

Fig. 3a. Oxygen conditions in Corpus Christi Harbor (Odum, Cuzon, Beyers, and Allbaugh, 1963).

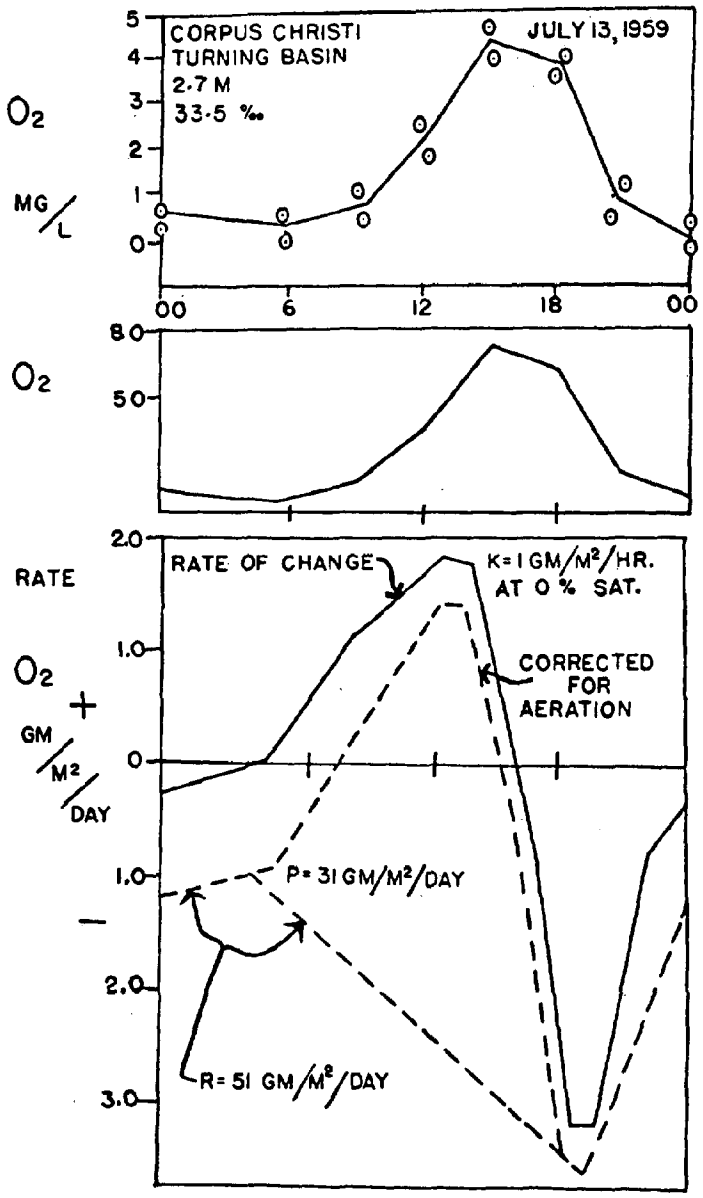


Fig. 3b. Oxygen curves for Corpus Christi Ship Harbor (Odum, 1960a).

Tampa Bay

One of the most fertile estuaries in America is Tampa Bay, Florida (Fig. 4) which receives raw municipal wastes, food processing wastes, the outflows from phosphate district of Florida, and many other wastes. Shallow and with little tidal exchange, the bay seems like one giant bloom. There are high concentrations of cells, nutrients, and other organisms (Figs. 5-8). High fertility persists from low salinities in small headwaters to the full salinities at the mouth under the Skyway bridge (Fig. 5). There are many papers on this bay due to the studies of several laboratories. The Florida red tide is a recurring phytoplankton bloom of a dinoflagellate Gymnodinium breve that is poisonous to fish (Fig. 6). This red waters develops fish-killing blooms in high salinity waters off the west coast of Florida and sometimes within the lower bay. The relationship of the fertile bay culture waters to the red tide outside is still uncertain and under study. The high fertility has not destroyed the general middle salinity characteristics of the ecosystem of the main bay where oysters, copepods, pinfish and young shrimp are abundant to mention some shown in Figs. 9-11. Much of the area has been disturbed in dredging and filling although there are still large areas of shallow ecosystems that serve as fertile nurseries (Fig. 4, 12). The pulse of phytoplankton follows the pulse of light energy to which the release of animal larvae are correlated.

Although the concentrations of nutrients coming into the bay now is much greater than before urban development, the bay was already receiving high nutrients from the phosphate districts, springs and swamp runoff. Tampa bay has small areas of heavy waste stress, but the open question is whether the overall consequent of this eutrophication should be regarded as good or bad for the general ecosystem. See also Chap. E-9.

Boston Harbor

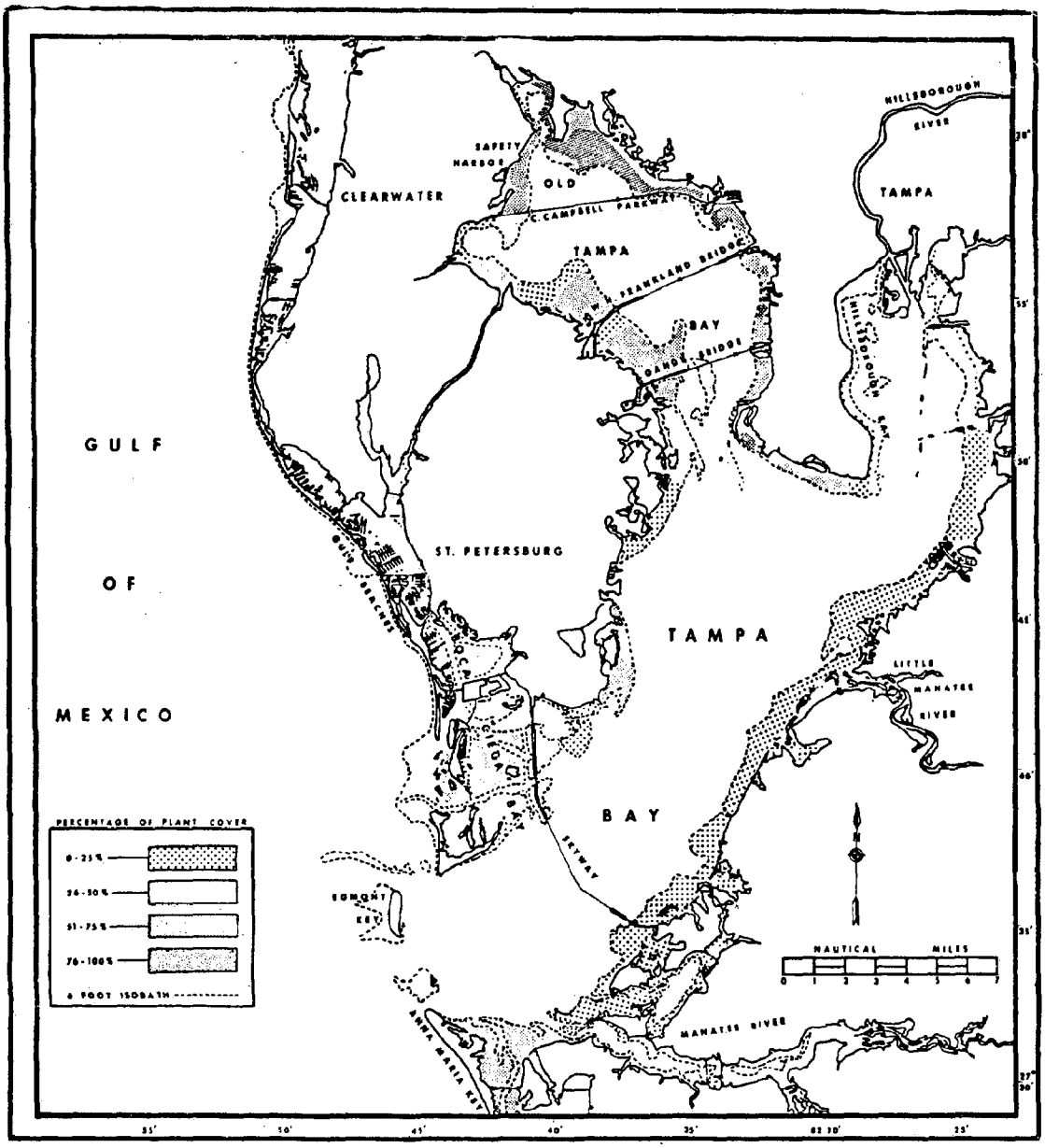
An example of a temperate estuary multiple stressed by man's activity is Boston Harbor (Figs. 14-17) which receives mixed wastes of shipping, municipal wastes, and industrial outputs.

San Francisco Bay

The lower, near-ocean parts of San Francisco Bay are sometimes cited as a pertinent example of multiple waste heterogeneity. See maps in Chapters C-8 and C-9. Fig. 13 from Bain and McCarty (1965) shows the sag in productivity as compared with diluted waters of the Pacific or the zones at the head of the estuary where freshwaters rich in nutrients are inflowing.

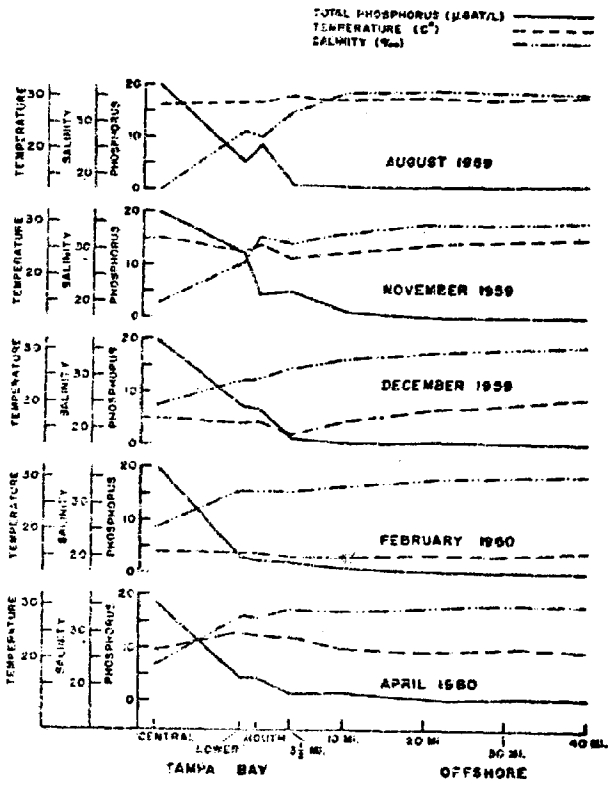
DISCUSSION

Whereas some of the ship channels with multiple wastes are so low in diversity and indices of life that there is no question that stress on ecosystems mainly exceeds the capabilities of living systems, some of the other bays showing more eutrophication than toxicity may be producing more life and yields than before man began introducing wastes. Unfortunately data before are few and technical personnel are divided in their speculation on this.



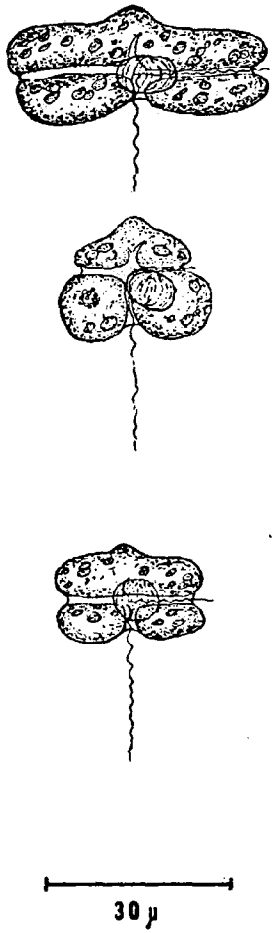
-Location and abundance of submerged vegetation in Tampa Bay.

Fig. 4. Nursery zones in Tampa Bay, Florida (C.M. Fuss in Sykes, 1967a).



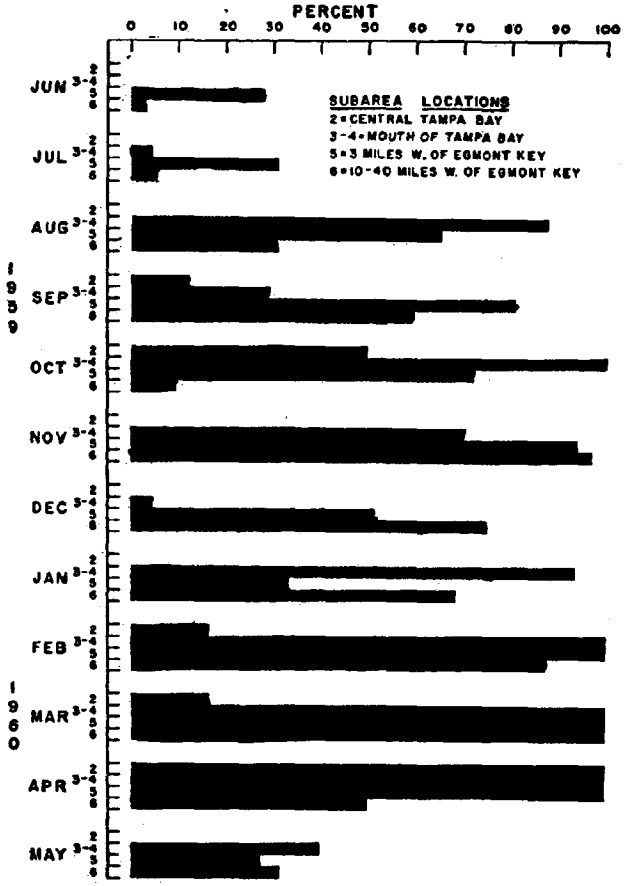
Means for certain months of temperature, salinity, and total phosphorus for Tampa Bay and adjacent neritic waters.

Fig. 5. High phosphorus levels in Tampa Bay (Dragovich, 1960).



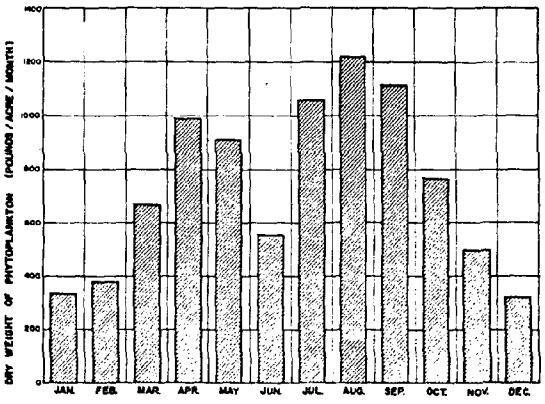
Gymnodinium breve

(Dragovich and Kelley in Sykes, 1967a)



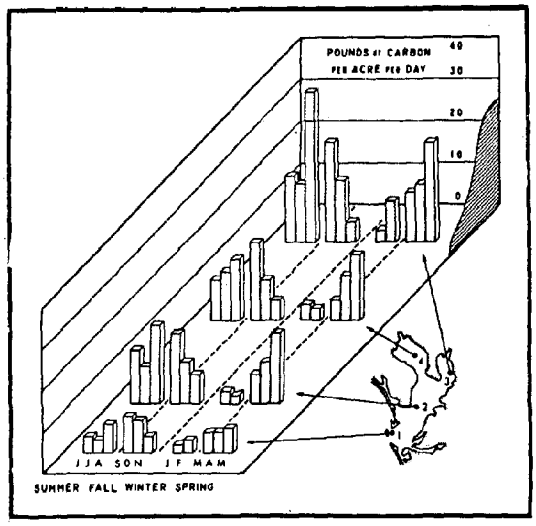
Percentage of samples containing G. breve.

Fig. 6. Red tide organism at the mouth of Tampa Bay, Fla. Gymnodinium breve (Fimucane, 1960).



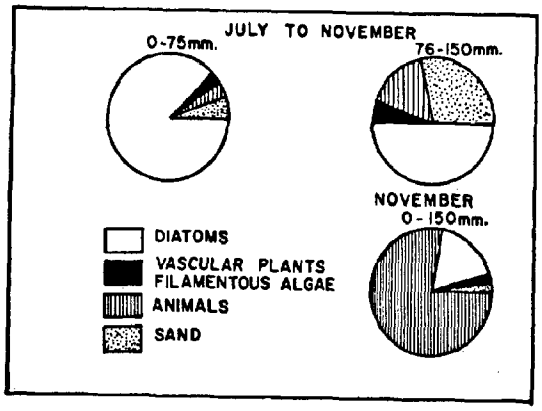
-Average seasonal changes to phytoplankton production in Tampa Bay, 1966-67. To convert pounds per acre per month to kilograms per hectare per month, multiply pounds per acre per month by 0.921.

Fig. 7. Photosynthetic production of plankton (McNulty, 1968).

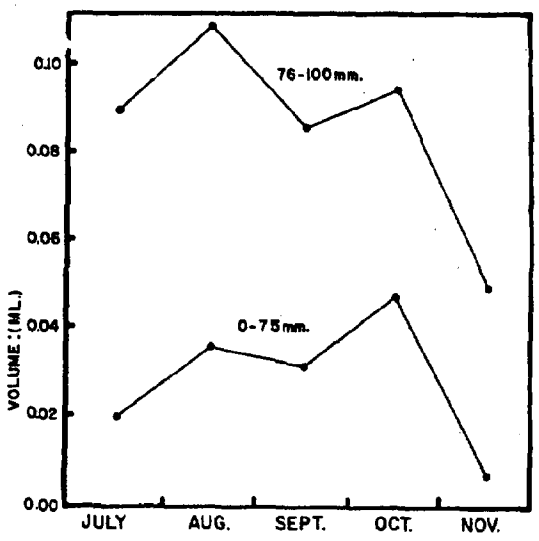


--Estimated production of organic carbon in Tampa Bay, June-November 1965 and January-May 1966. (Station 1, Egmont Key; Station 2, Central Tampa Bay; Station 3, Hillsborough Bay; Station 4, Old Tampa Bay.)

Fig. 8. Photosynthetic productivity of plankton (Kelly and Johnson, 1967).

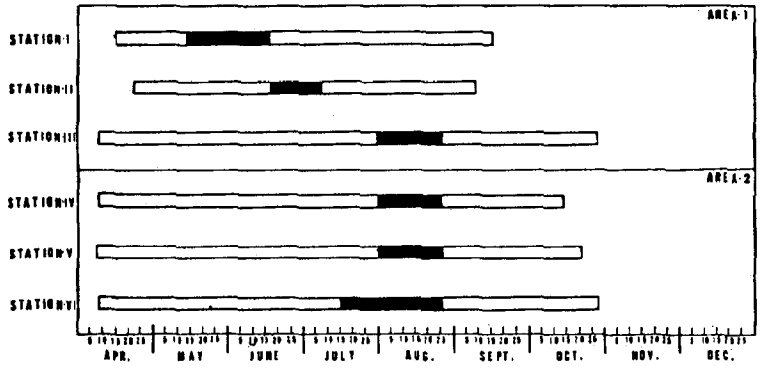


Differences in the foods from stomachs of 1,200 large and small pinfish from July to November 1965 and from over 100 pinfish caught in November.



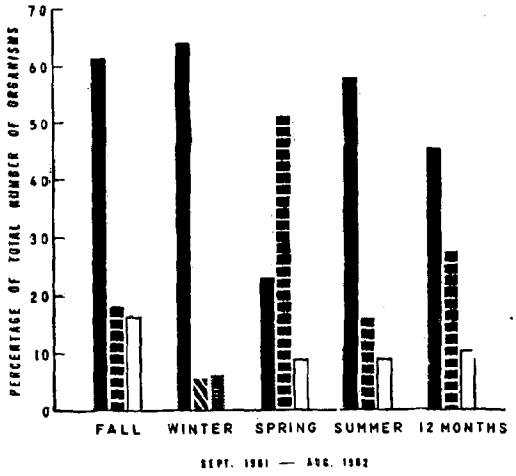
Mean volume (milliliters) of material in 1,300 stomachs from large and small pinfish in the summer and fall, 1965.

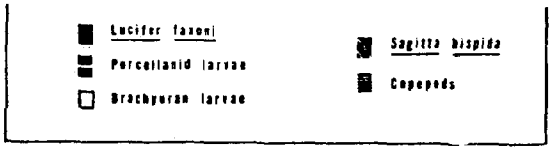
Fig. 9. Herbivorous role of young pinfish in nursery grounds (Hansen, 1967a)



Dates of oyster spatfall (horizontal bars) and periods of peak spatfall (shaded portions of bars) at six stations in Tampa Bay.

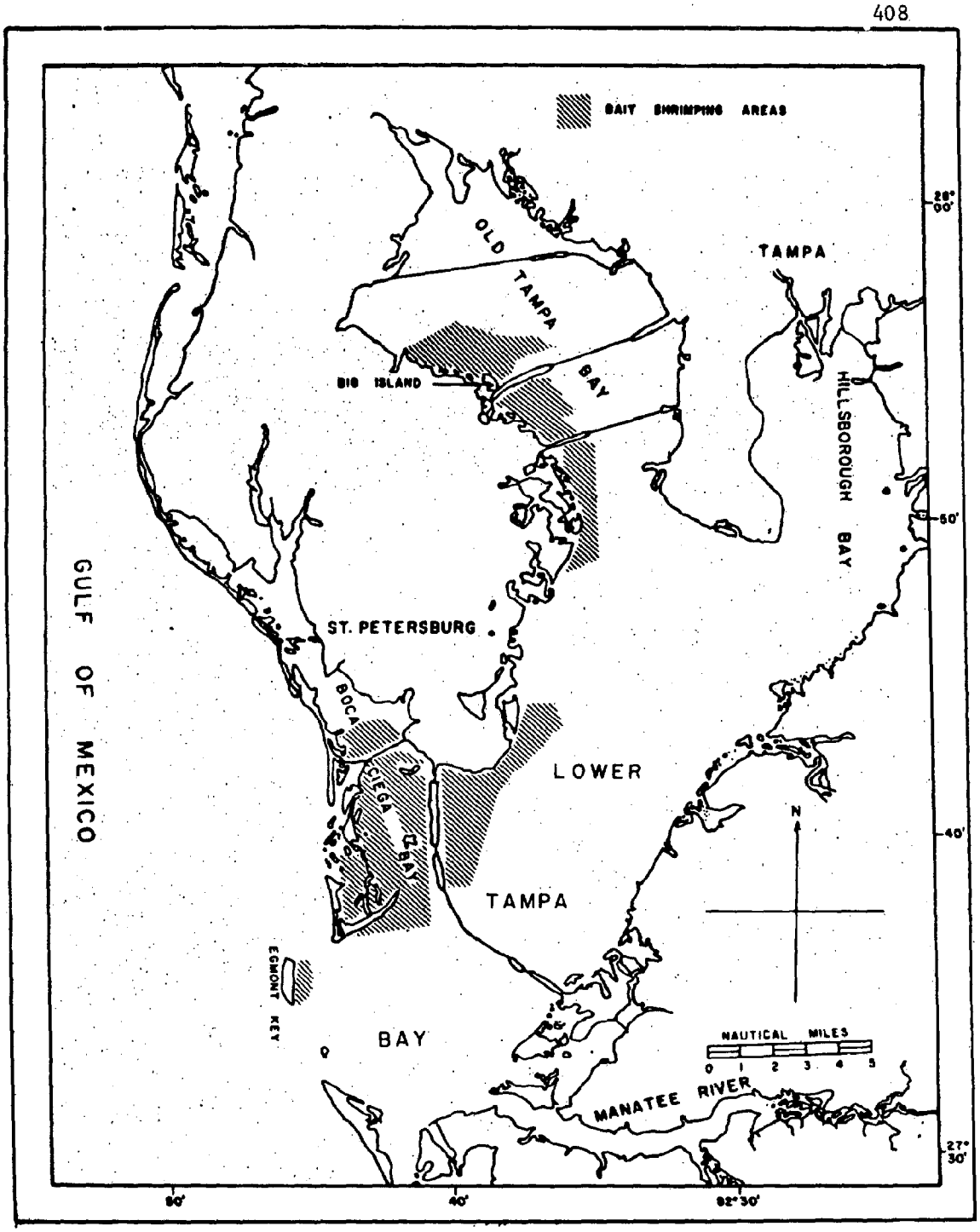
Fig. 10. Pulse of oyster larval release and set in seasons of phytoplankton productivity in Tampa Bay (Finucane, 1967).





-Zooplankton taxa from Tampa Bay and the adjacent Gulf of Mexico that accounted for 5 percent or more of the organisms found during the periods shown, September 1961 through August 1962.

Fig. 11. Larger zooplankton (Kelly and Dragovich, 1967).



--Bait shrimping areas in Tampa Bay.

Fig. 12. Bait shrimp (Penaeus duorarum)--Pink shrimp in Tampa Bay, Fla (Saloman, 1965)

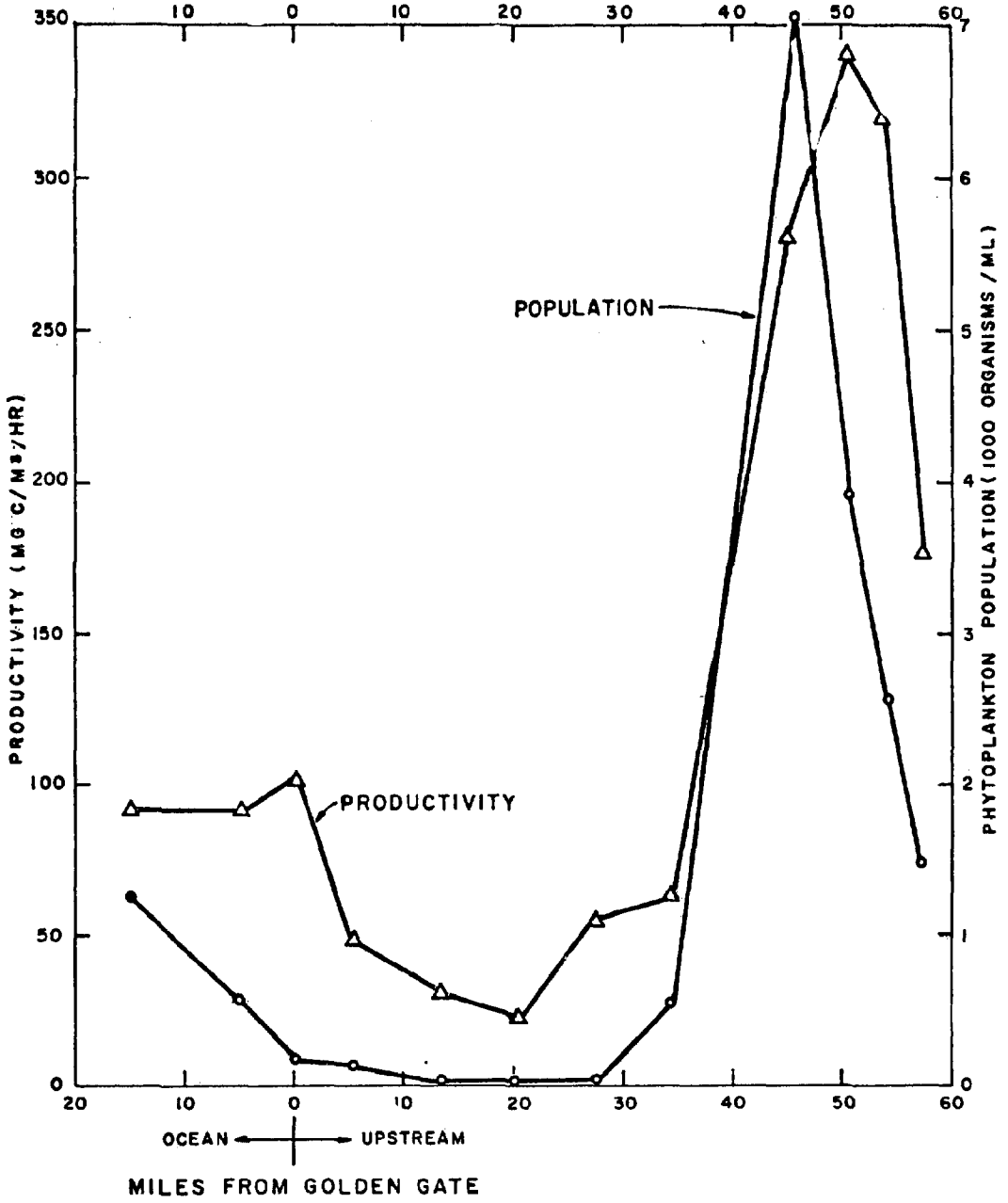
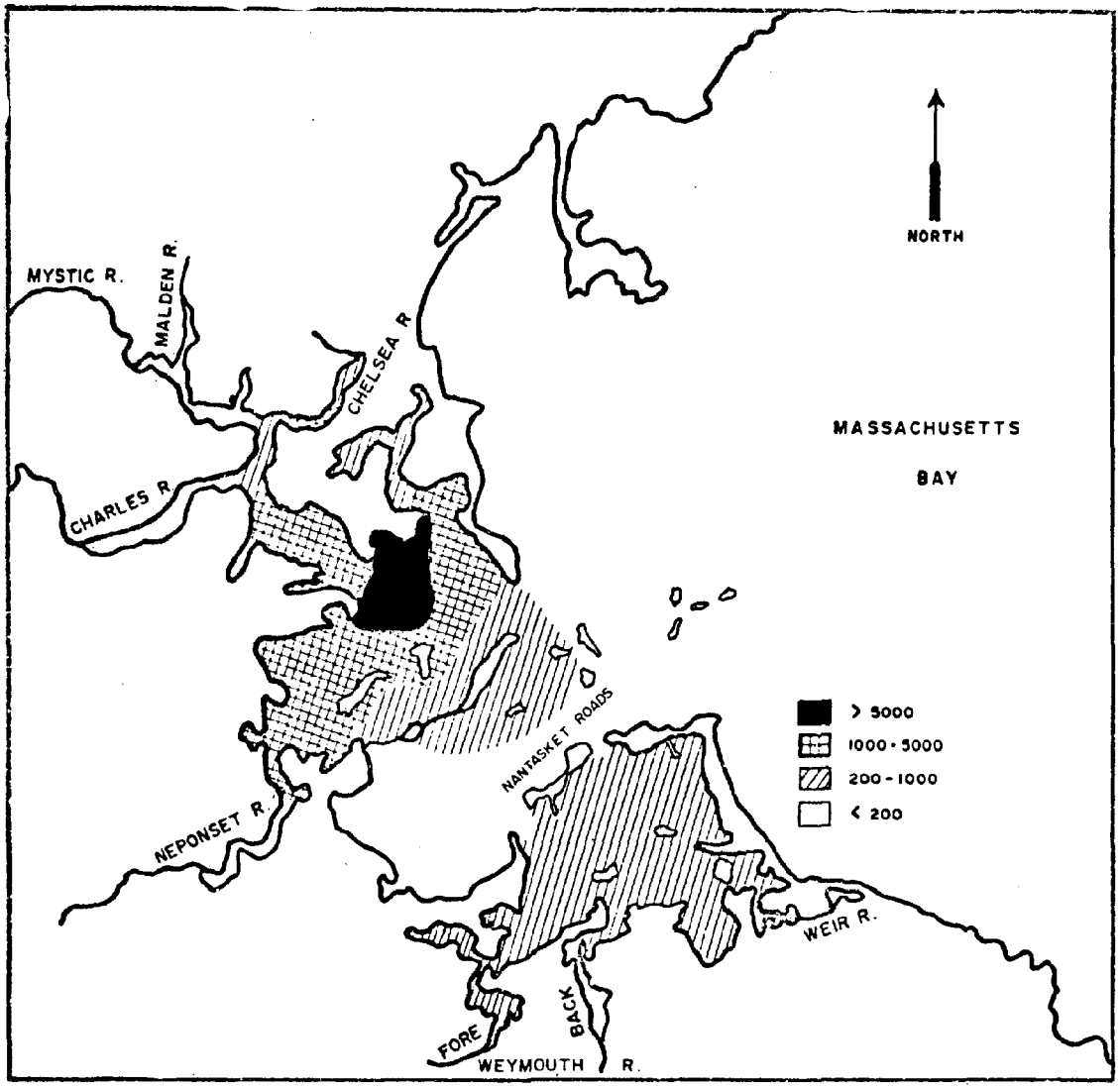
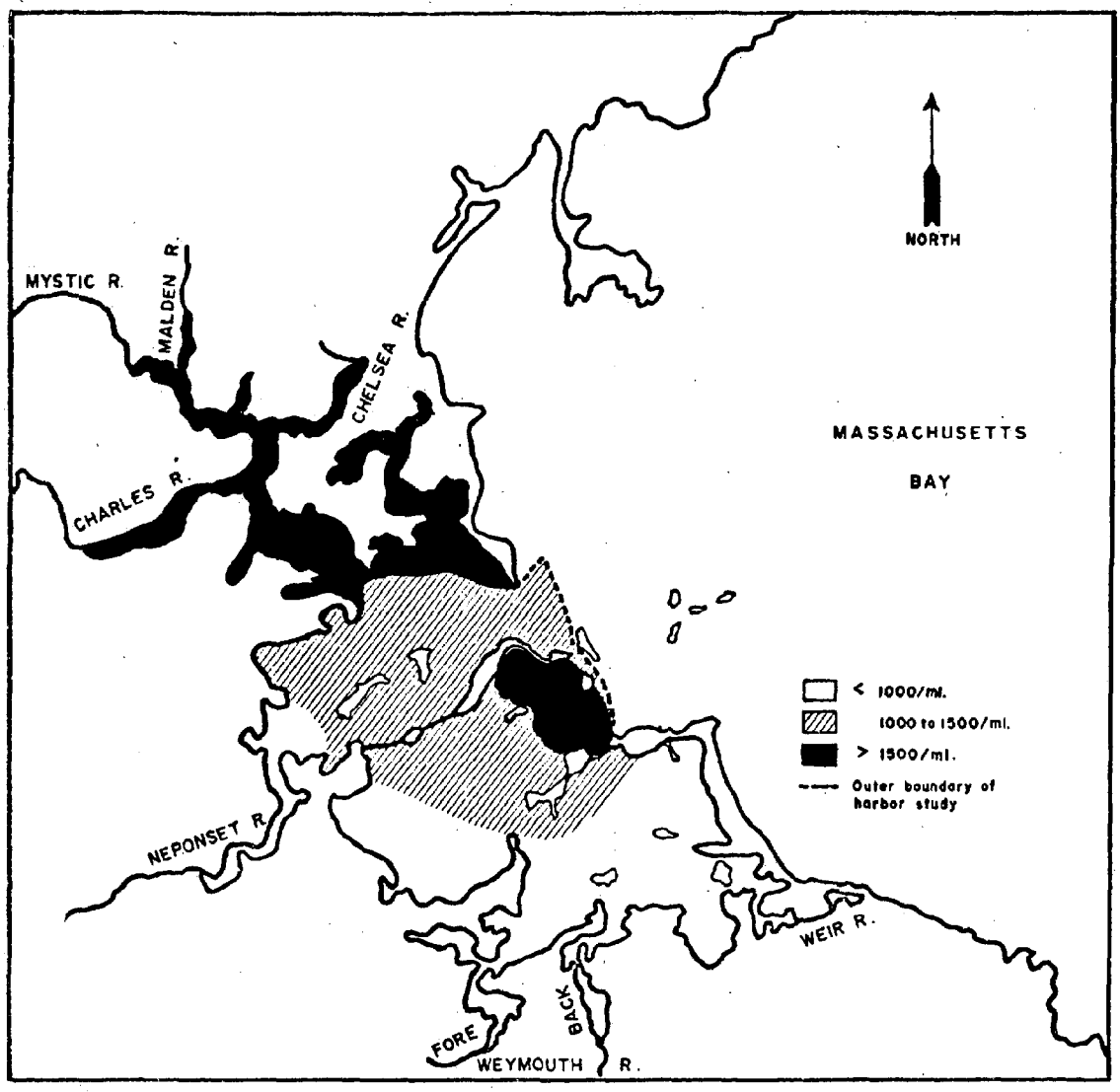


Fig. 13. Photosynthesis rates in San Francisco Bay (Bain and McCarty, 1965).



NUMBER OF POLYCHAETE WORMS PER SQUARE FOOT, BOSTON HARBOR AND TRIBUTARIES, JULY-AUGUST, 1967.

Fig. 14. High concentrations of red worms in multiple waste area. (Steward, 1968).



AVERAGE NUMBER OF PHYTOPLANKTON (number/ml.), IN BOSTON HARBOR, AUGUST, 1967.

Fig. 15. High concentrations of phytoplankton algae in multiple waste areas (Steward, 1968).

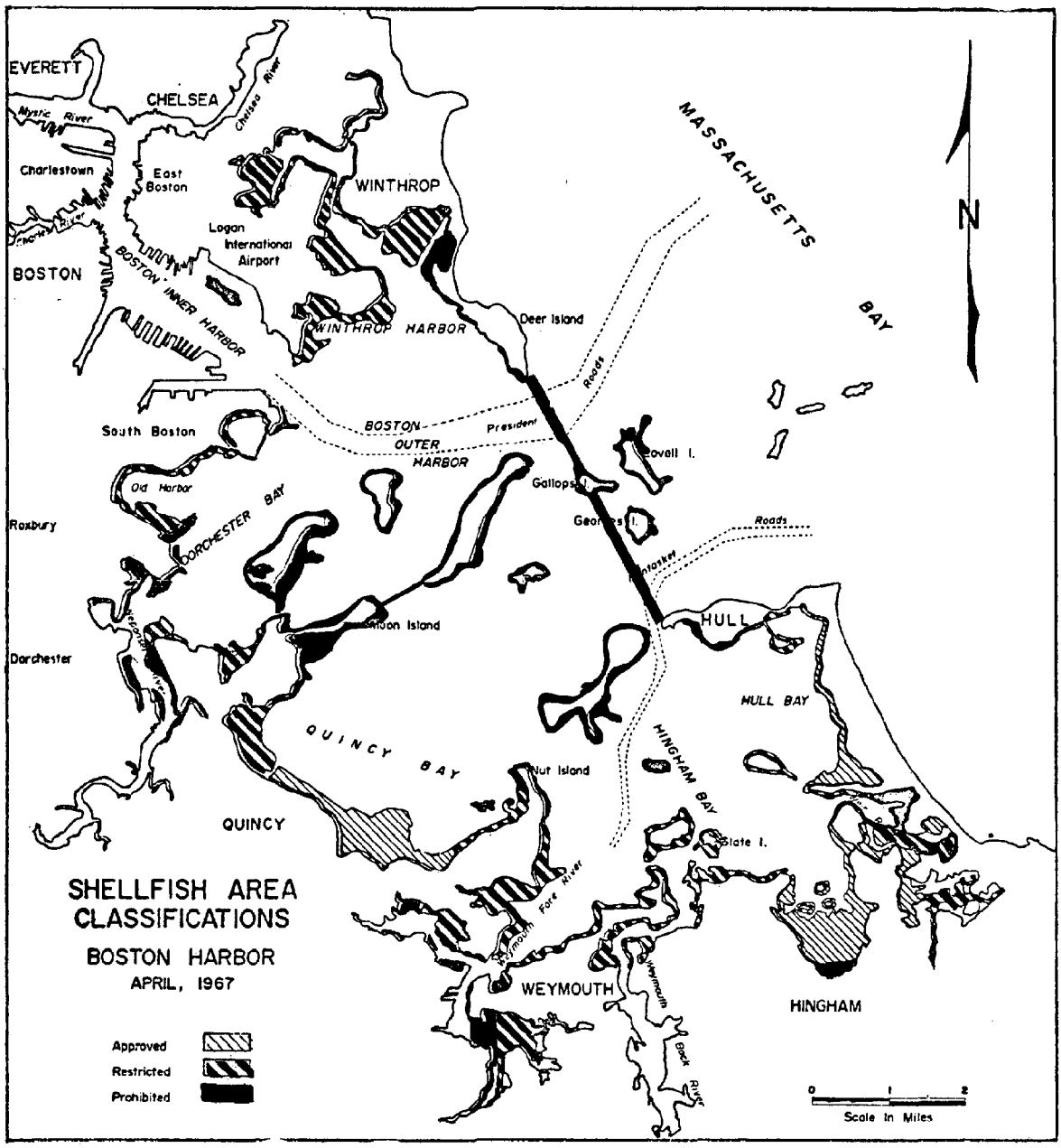


Fig. 16. Areas of Boston Harbor with shellfish containing intestinal bacteria of possible risk to health if eaten (Steward, 1968).

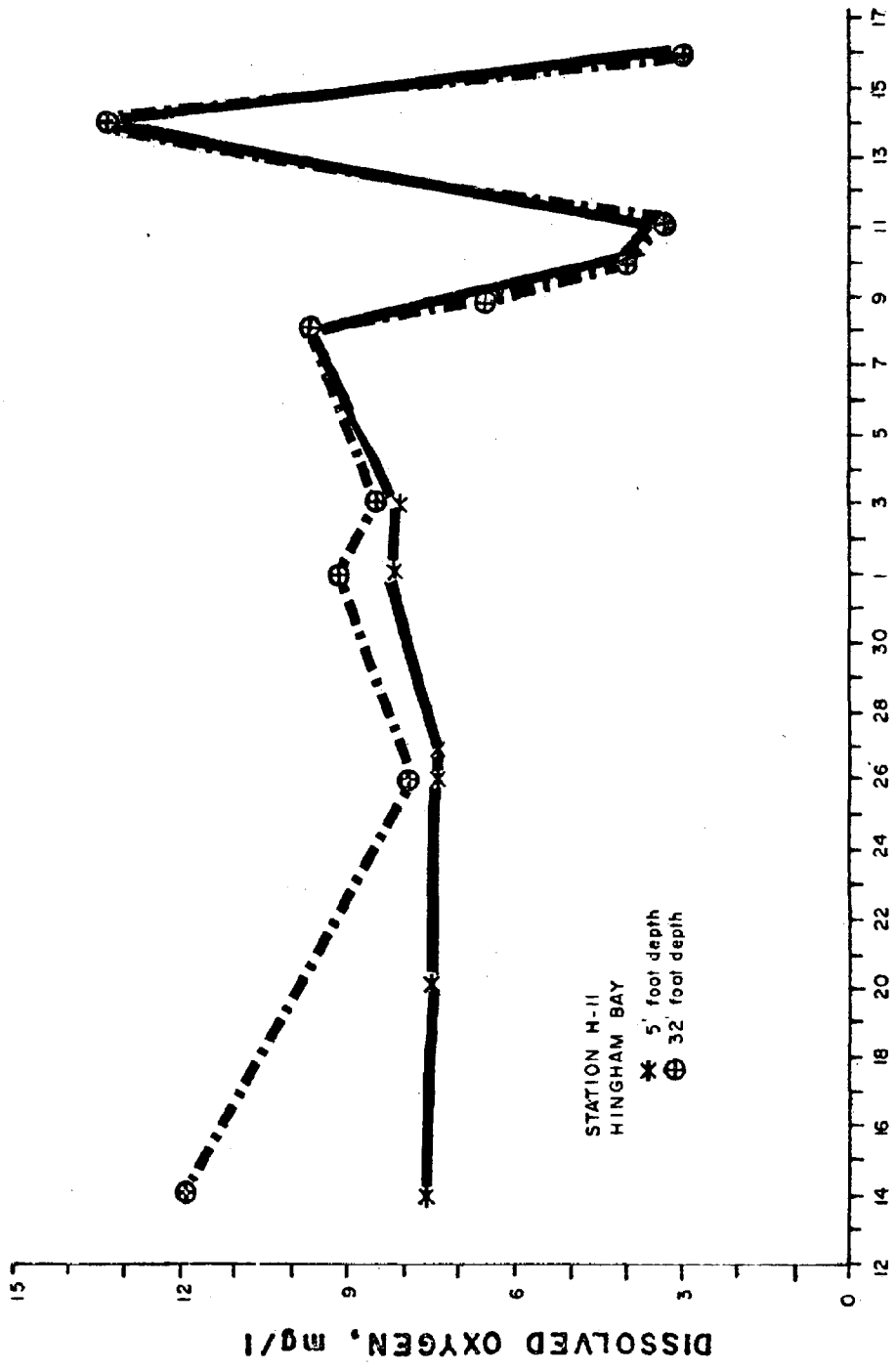


Fig. 17. Moderate oxygen levels in daytime in multiple waste receiving areas (Federal Water Pollution Control Administration, 1968h).

Chapter E-18

ARTIFICIAL REEFS

An artificial reef is an elevated structure of rocks, old cars, pilings, or other rigid materials placed in shallow seas and estuaries. Upon them develop rapid growths of attached organisms because of the fast current renewal that is usually found where a structure protrudes into a water mass well above the main friction layers of the bottom. These structures develop high current ecosystems as already described in Chapter A-3. If they are within the lighted zones, algae are an important part of the system; if deeper, growths are mainly animal. Artificial reefs are being placed into waters because of the good fishing that may result from the attraction of carnivores that then become part of the reef ecosystem. In clear tropical waters, these reefs may become covered with coral. There seem to be few aspects of the artificial reefs that differ from the natural ones except when the material is temporary, such as old car bodies that corrode and collapse.

EXAMPLES

Oil Drilling Platform Supports in the Gulf of Mexico

Gunter and Geyer (1955) described the fouling communities on oil platforms off Louisiana and Texas. Growing in shelf waters, these communities experienced less hydroclimatic stress than is found in many estuaries, and diversities were relatively large. Fig. 1 indicates zonation of two sets of pilings.

Artificial Fishing Reefs and Oil Platforms of California

Carlisle, Turner and Ebert (1964) studied reefs made of streetcar and car bodies and oil drilling platforms along the California coast. Kelp bass and sand bass reached population sizes attractive to fishermen within a year as fouling communities of algae, barnacles, and other invertebrates increased serving as food chains to larger fishes. Kelp became established on some reefs, heavy beds of mussels on others. Representative data are given in Figs. 2-4

An Artificial Reef in Tropical Shallows

Randall (1963a)studied the associations of organisms on a pile of concrete blocks piled in turtle grass in the Virgin Islands. A population of reef carnivores developed and grazed the grass nearby, making a clear area around the reef (Table 1).

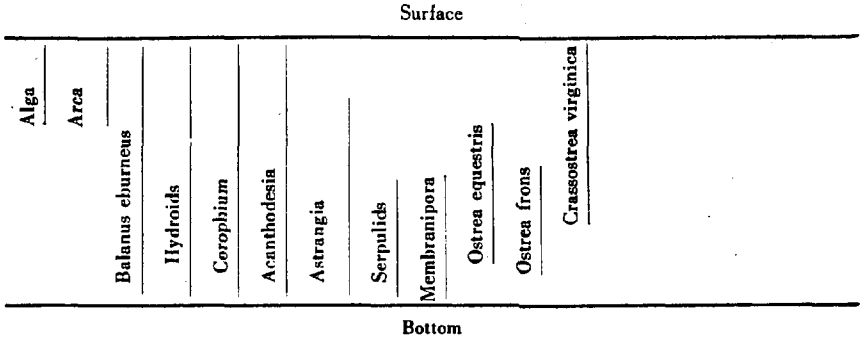
Growth on a Sewer Outfall

In Fig. 5 Turner, Ebert and Given (1966) show the ecosystem associated with a large sewer outfall pipe on the coast of California. Growths of attached consumers were abundant close to the end of the pipe where they were exposed to considerable eddying of sewage waters.

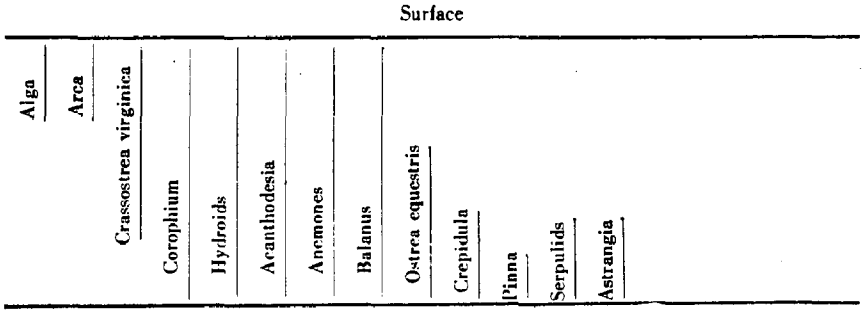
DISCUSSION

These and other cases demonstrate that the artificial reef is a means for concentrating fish for fishing. The basis for the observed concentration of life may be the moving waters which provide some of the life function of food concentrating and bring in nutrients for algae. Theoretically, the number of such reefs that may be developed is restricted by the availability of moving waters not previously stripped of food and nutrient. When the number of reefs becomes large enough to drag the water and reduce current, further reef additions may not be productive. The costs of these reefs has been high because the heavy winter waves disperse and destroy many of the structures that have been arranged. Some car-body reefs placed off Texas could not be located after two seasons.

The development of concentrated marine life on reefs, shores, and beaches is a means for coupling the physical energies of the sea to the service of man. The means for doing this inexpensively needs systematic study.

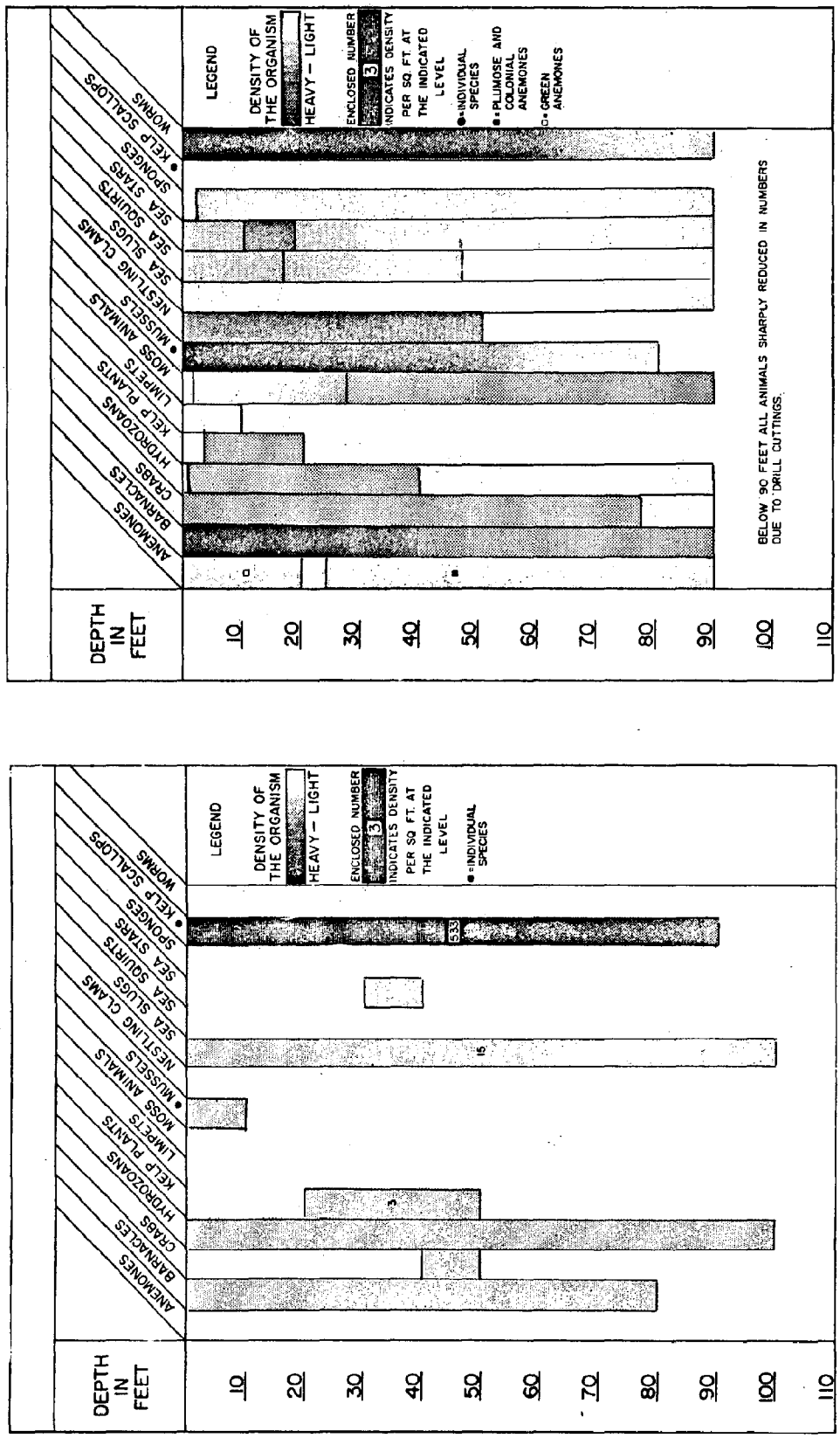


A schematized distribution of the fouling organisms along the two Louisiana oil-well platforms described in the text. The greatest depth was fifty feet.



A generalized graphic distribution of the common fouling organisms along the two Texas oil-well platforms described in the text. The greatest depth was sixty feet. Except for the alga, organisms were not found right at the surface or right at the mudline. Relative abundance at depths are shown very roughly by thickness of the lines.

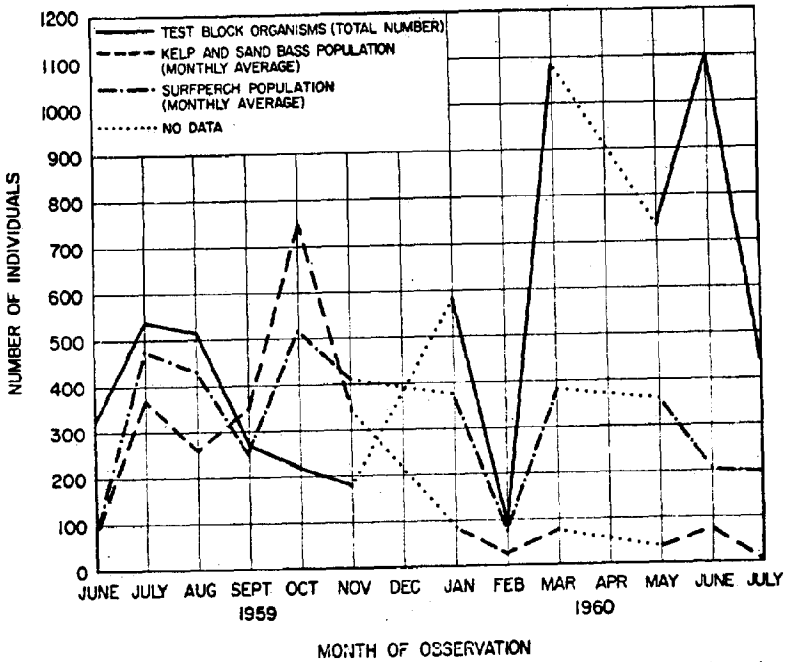
Fig. 1. Zonation on oil platforms off Texas (Gunter and Geyer, 1955).



The vertical distribution of the major biomass on Hazel, July 1959, approximately 12 months after installation.

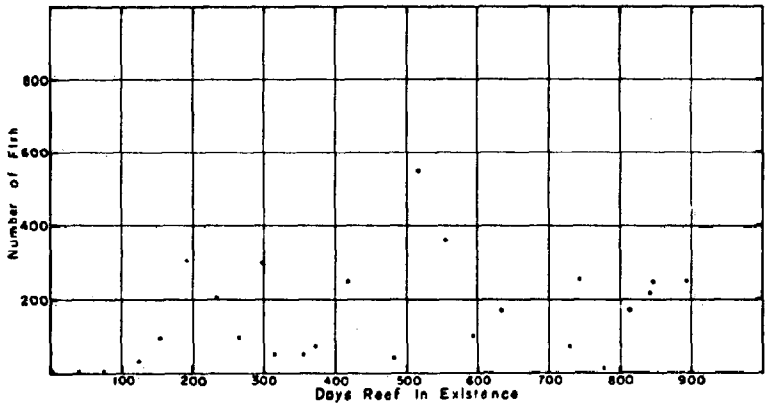
The vertical distribution of the major biomass on Hazel, July 1961, approximately 36 months after installation.

Fig. 2. Vertical distribution of attached animals on a California oil platform at two times after construction (Carlisle, Turner, and Ebert, 1964)

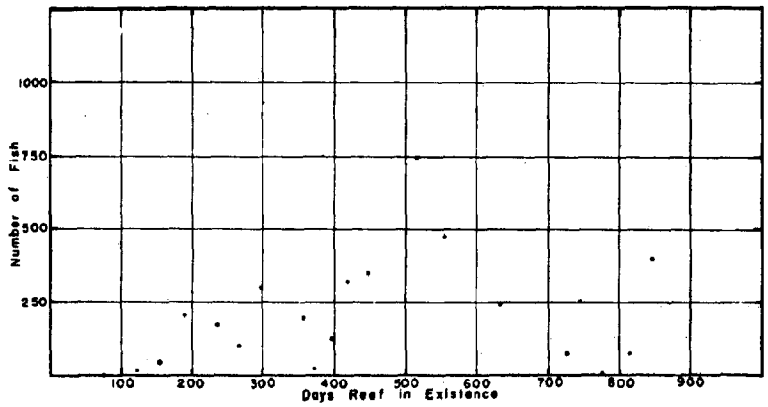


The total number of test block organisms, seaperch, and bass recorded on the Redondo Beach streetcar reef from June 1959 to July 1960.

Fig. 3. Record of animals on a California artificial reef (Carlisle, Turner, and Ebert, 1964).



Increase of kelp bass population with time on car body reef (1,500 fish at 522nd day omitted).

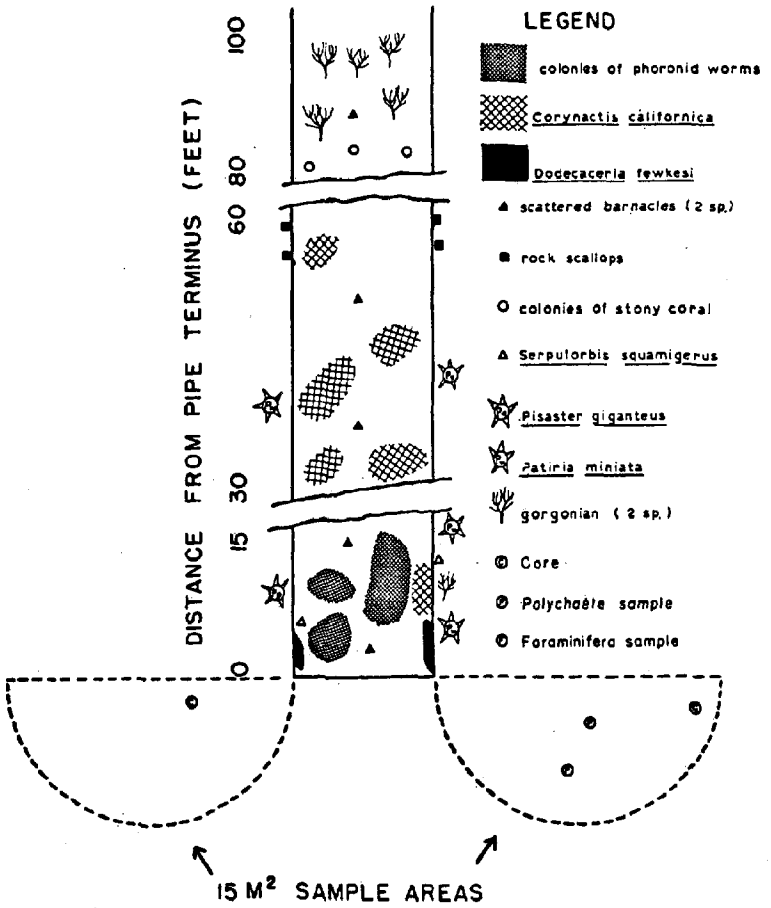


Increase of sand bass population with time on Paradise Cove car body reef (2,000 fish at 525th day omitted).

Fig. 4. Rapid establishment of carnivorous fish stocks on an artificial reef in California (Carlisle, Turner, and Ebert, 1964).

Table 1. Summary of analysis of fishes collected from an artificial reef in St. John, Virgin Islands (column "A") and from a natural reef area of comparable size (column "N") (From Randall 1963a).

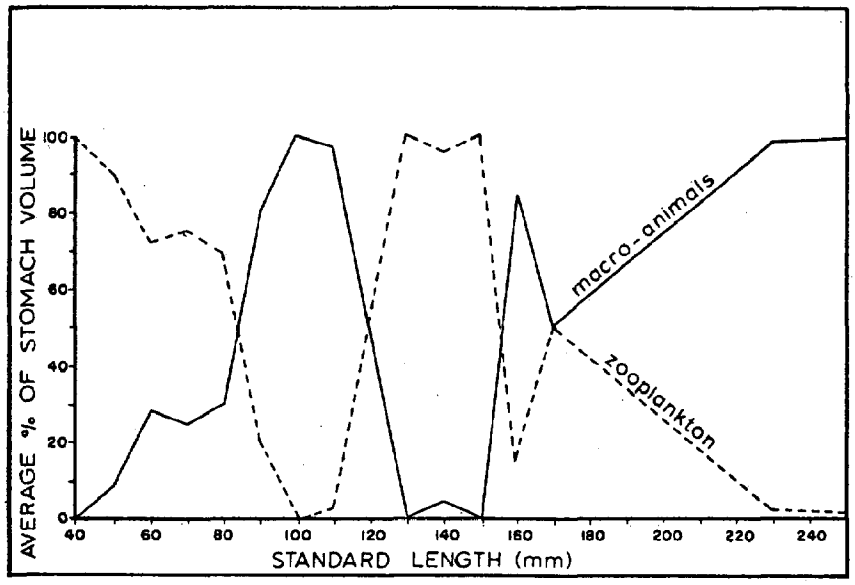
Family	# of Species		Total Weight	
	A	N	A	N
Pomadasidae	4	3	37	7
Serranidae	5	3	16	5
Holocentridae	4	7	11	6
Acanthuridae	3	2	5	2
Scaridae	7	7	4	12
Balistidae	1	—	5	--
Muraenidae	2	5	4	1.5
Mullidae	2	—	2	--
Lutjanidae	1	2	1	5
Diodontidae	1	—	1	--
Scorpaenidae	2	2	1	0.1
Chaetodontidae	3	4	0.2	1.3
Labridae	5	5	0.2	0.5
Pomacentridae	2	7	0.2	4.5
Moringuidae	1	1	0.05	0.1
Tetraodontidae	2	1	0.02	--
Clinidae	1	9	0.01	0.5
Gobiidae	1	7	0.01	0.07
Ophidiidae	1	1	0.01	0.01
Xenocondridae	1	2	—	0.01
Emmelichthyidae	1	—	—	--
Monacanthidae	—	1	—	0.08
Antennariidae	—	1	—	0.02
Grammidae	—	1	—	0.01
Cirrhitidae	—	1	—	0.01
Syngnathidae	—	1	—	---
Microdesmidae	—	1	—	---
Gobiesocidae	—	3	—	0.1
Priacanthidae	—	1	—	0.1
Apononidae	—	3	—	0.1
Aulostomidae	—	1	—	0.5
Grammistidae	—	2	—	0.5
Blennidae	—	3	—	0.7
Totals	55	93	87	47



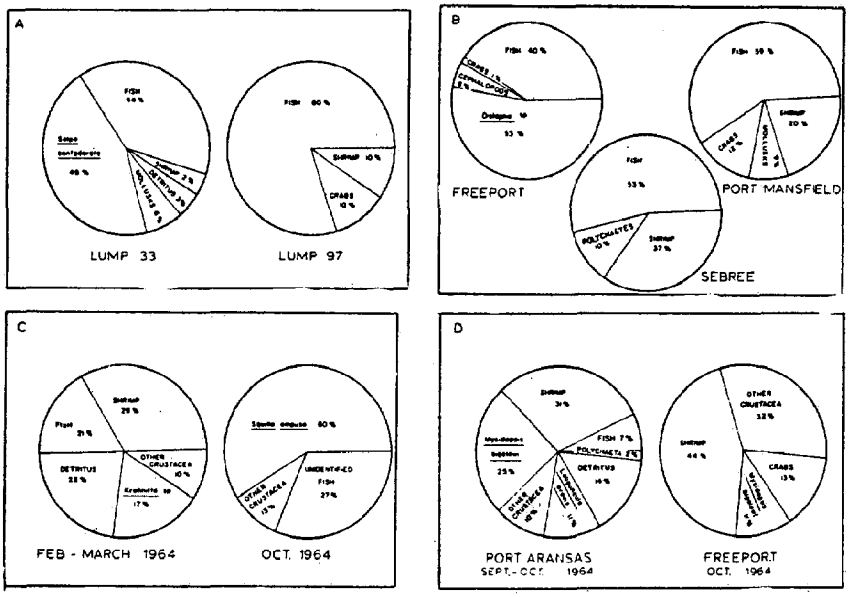
A pictorial representation of the major faunistic assemblages along the last 100 feet of the outfall pipe. No "macroscopic" animals or plants were recorded in the 2 sample areas (each encompassing approximately 135 square feet of bottom area) at the pipe terminus.

Fig. 5. Attached organisms on the aquaduct of a California sewage outfall offshore(Turner, Ebert, and Given, 1966).

The wide diversity of food utilized by the top carnivores of the reef communities is illustrated in Fig 6 for red snappers in the Gulf of Mexico and Table 2 for grouper in the Caribbean.



Ontogenetic Food Progression of Red Snappers.



Food Habits of Red Snappers by Volume. A. Louisiana Adult Red Snappers. (46 Stomachs); B. Texas Adult Red Snappers. (68 Stomachs); C. Louisiana Juvenile Red Snappers. (28 Stomachs); D. Texas Juvenile Red Snappers. (45 Stomachs).

Fig. 6. Food of the reef-dwelling red snapper off Texas (Moseley, 1966).

Table 2

Analysis of the Fishes Found in the Stomachs of Nassau Groupers. (Randall, 1965a)

Family of Fishes	Number of Times Found in Stomachs	Identification
Parrotfishes (Scaridae)	13	Sparisoma aurofrenatum Scarus vetula Scarus sp.
Wrasses (Labridae)	6	Clepticus parrae (3) Halichoeres bivittatus (2) Halichoeres garnoti
Damselfishes (Pomacentridae)	5	Chromis multilineata Chromis cyanea Pomacentrus fuscus Microspathodon chrysurus
Squirrelfishes (Holocentridae)	4	Holocentrus rufus (2) Holocentrus sp. Myripristis jacobus
Snappers (Lutjanidae)	4	Ocyurus chrysurus (3) Lutjanus sp.
Grunts (Pomadasydae)	3	Haemulon aurolineatum (2) Haemulon flavolineatum
Morays (Muraenidae)	3	Gymnothorax moringa Muraena miliaris
Round herrings (Dussumieridae)	2	Jenkinsia lamprotaenia (2)
Lizardfishes (Synodontidae)	2	Synodus intermedius
Sea basses (Serranidae)	1	Hypoplectrus puella
Goatfishes (Mullidae)	1	Pseudupeneus maculatus
Bigeyes (Priacanthidae)	1	Priacanthus cruentatus
Surgeonfishes (Acanthuridae)	1	Acanthurus sp.
Trunkfishes (Ostraciidae)	1	Lactophrys sp.
Silversides (Atherinidae)	1	
Anchovies (Engraulidae)	1	

Chapter F

MIGRATING SUBSYSTEMS

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Corpus Christi, Texas 78411

INTRODUCTION

Characteristic adaptations to the seasonal regime existing in temperate coastal systems that permit the rapid rise of animal populations with energy availability are the migrations of populations of larger animals into the bays from the rivers and the sea at the time of seasonal food pulses. Animals, such as the shrimp, salmon, and shad, make their famous migrations and the population's new young experience their period of most rapid growth rate in the bays at the time of food pulses in that system. This fast growth of young, rather than the actual reproduction, gives rise to the concept of the estuary as a "nursery". Egg production and hatching usually takes place in an area of relative stability far up streams in the case of the salmon or in the stable temperatures and salinity of the sea as in the case of the shrimp . These hatchery areas are connected in such a way as to allow the young to drift into the highly productive and highly pulsing coastal systems.

The migrating stocks are transported from one kind of system (usually the inland stream or the continental shelf of the seas) to couple with the energies of the coastal system. Thus, the migrating stock plugs into the energy cycle of a coastal system, becomes part of the cycle, then unplugs and emigrates to yet another system. In this way, the systems along coasts are inter-connected through the migrating energy couplings. In addition to examples cited here many of the other chapters show graphs documenting seasonal pulses of food and movements of stocks of larger animals into position to use the yields. Fig. 1 is a typical pulse of plankton food availability due to increased energy availability from runoff.

Hemisphere Patterns

Fig. 2 is an overall summary of migrating subsystems for the United States coast from Arctic to Tropics. There is migration northward, landward, and into estuaries with rising pulses of food production in Spring and south and seaward as productive pulse declines in Fall. Foods change as migrating stocks utilize the different kinds of seasonal food accumulations as they pass into different areas. Seals from Alaska, for example, migrating south to waters off California (North Pacific Fur Seal Commission Report of Investigation 1958-1961, 1962) change food in each zone.

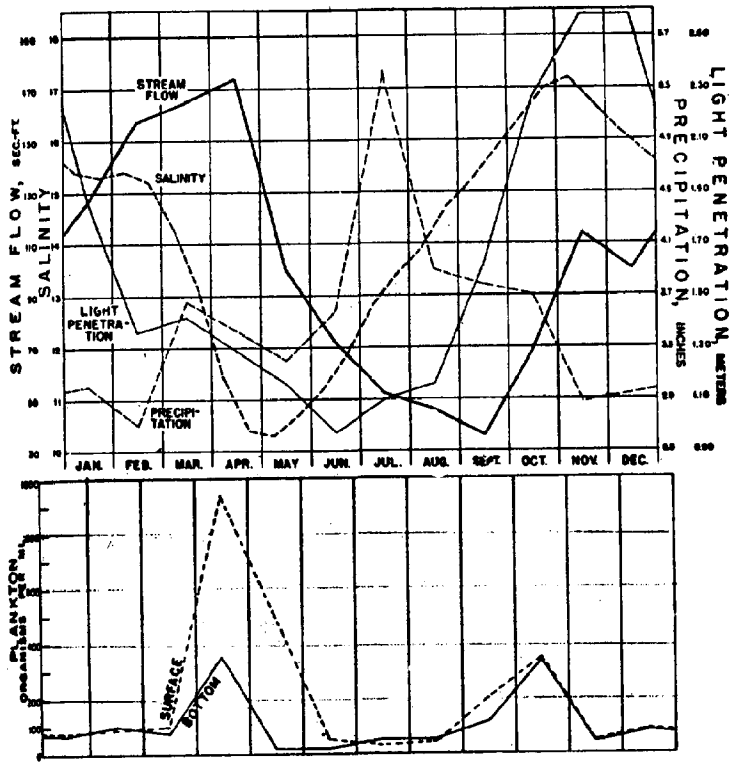


Fig. 1: Seasonal variations in the Patuxent River of stream flow, salinity, light penetration, and precipitation in the upper diagram, and of surface and bottom phytoplankton (From Nash, 1947; Figs. 43 and 44).

Table 2. Salmon catch in Machias River salmon run (Bond, 1967).

MACHIAS RIVER SALMON RUNS

Year	Date of Capture of 10th	Date of Capture of 100th	Rod Catch	Totals Machias	Totals Whitneyville	Remarks
1949	June 21*	July 16	—	362	—	
1950	June 19*	Aug. 2	—	134	—	
1951-52	No records available		—	—	—	
1953	July 10*	July 30	4	258	—	
1954	June 12,	June 23	9	544	65	Start of trapping at Whitneyville
1955	Not available		22	316	—	
1956	Not available		28	396	236	
1957	May 31	June 13	32	451	86	
1958	June 10	June 17	101	617	336	1st fish May 23—earliest to date
1959	June 8	July 7	48	813	637	
1960	June 2	June 22	44	307	182	
1961	June 20*	June 30	130	605	397	Ledge removal in gorge
1962	May 28	July 1	76	373	297	1st fish—May 19th
1963	June 9	June 17	68	347	236	
1964	May 27	June 25	78	373	243	New Denil fishway at Whitneyville 1st fish—May 14th
1965	July 10*	July 28	58	299	199	Incomplete Machias count
1966	June 5	—	92	—	503**	Trapping discontinued at Machias 1st fish—May 13th

* Initial count delayed—trap not in operation at Machias
** Not final count

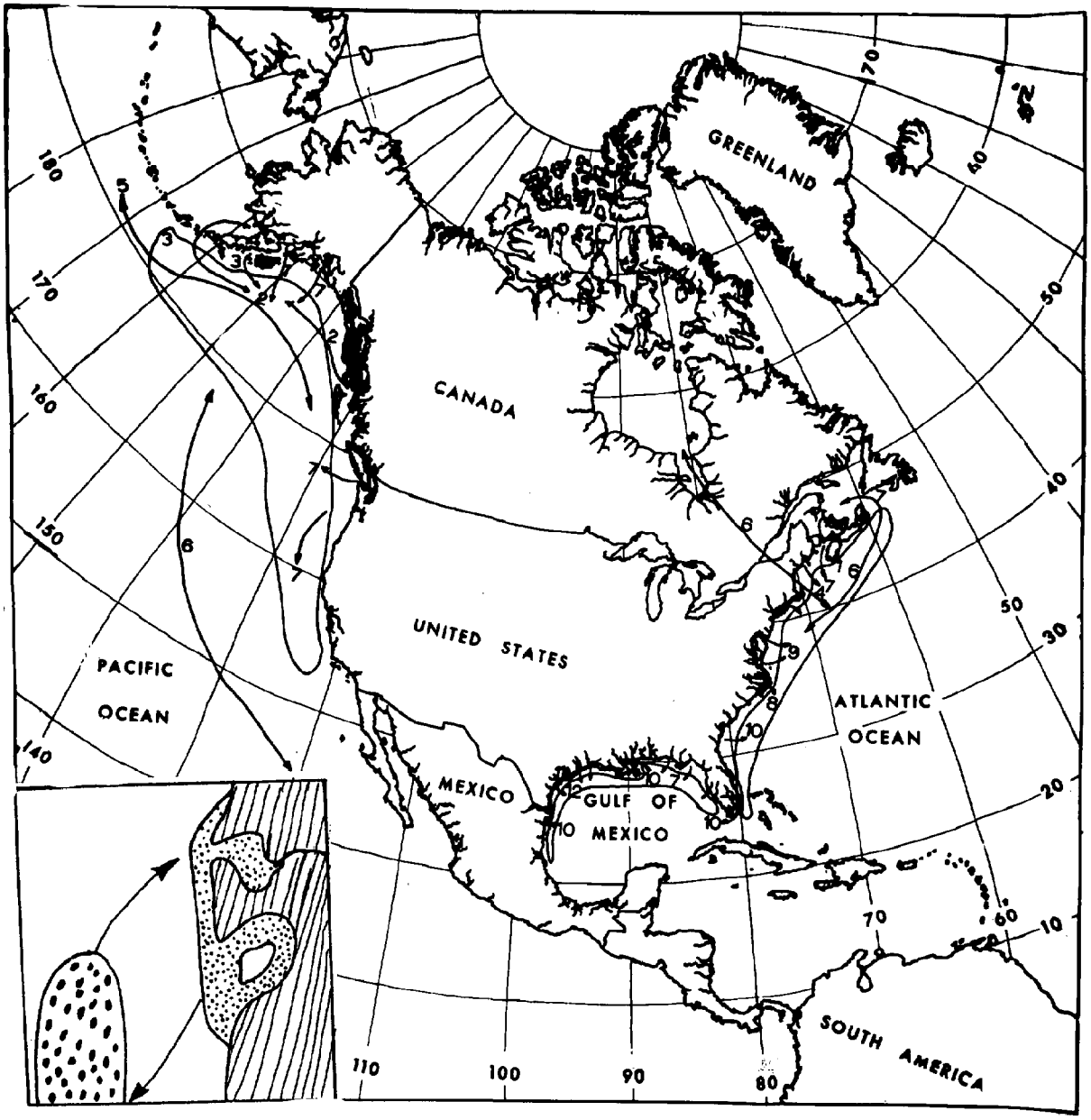


Fig. 2: Summarizing diagram of seasonally shifting stocks of the North American coastal zones. 1, Pandalid Shrimp; 2, Salmon; 3, King Crab; 4, Large Claw Lobster; 5, Sea Mammals; 6, Sea Birds; 7, Herring, Menhaden, alewives, etc.; 8, Shad; 9, Striped Bass; 10, Penaeid Shrimp; 11, Blue Crabs; 12, Mullet; Circles indicate offshore species shift. Inset shows the general pattern: northward, landward to inshore nursery in spring; southward, outward to coastal and offshore wintering areas.

EXAMPLES

Some migrating animals spawn in the sea and the larvae drift into the highly pulsating coastal systems while others spawn far up streams and the larvae are transported downstream to the estuary and still others spawn in the coastal systems which are utilized for nursery areas. A significant percentage of the marine organisms that compose the commercial catch in this country spend at least a portion of their life-history in the coastal systems (Deiner, 1964). Since many of the major migrating forces are of commercial importance, they have received considerable research attention and will, therefore, serve as examples for the following discussion.

Commercial Shrimp

Penaeid Shrimp

According to Guest (1958), the penaeid shrimp off Texas (Figs. 3, 4) lay their eggs on the continental shelf in 10 to 40 fathoms of water. The eggs hatch, in a very short time, and become a part of the drifting plankton of the sea (Fig. 5). After passing through several larval stages while drifting generally toward shore, the post-larvae enter coastal systems through passes or bay-sea interfaces. They experience very fast growth rates (Loesch, 1965) in coastal bays and subsequently emigrate from the system as juveniles, back to the sea (Figs. 6, 7). Few of the adults migrate back to the bay systems, but many are consumed in the sea where they constitute a very substantial part of natural food chains and of the commercial fishery. A diagrammatic summary of the penaeid shrimp life-history in Texas is shown in Figure 3.

Shrimp production in Texas waters is closely correlated with the total annual availability of river input and spring photosynthesis. A plot of shrimp catch and average rainfall for Texas is shown in Figure 8. An increase or decrease in rainfall was followed by similar fluctuations in shrimp catch, generally after a two-year period. Correlation coefficients were significant to the 1% level when rainfall of the two previous years was correlated with an annual shrimp catch (Gunter and Hildebrand, 1954).

Measurements of immigration and emigration have been reported for Aransas Pass Inlet (Figs. 9 and 10) and Cedar Bayou (Figs. 11 and 12). See map in Fig. 4. The migration of postlarval penaeid shrimp through the Aransas Pass Inlet, Texas is shown in Fig. 9 and in other areas (Fig. 13). Brown shrimp (Penaeus aztecus) migrate through the pass into Texas bays during spring and the pink shrimp postlarvae (Penaeus duorarum) migrate into Texas bays during autumn, both during times of higher river input into the bays. Pinks and browns migrate out at end of spring and whites (Penaeus setiferus) go out at end of summer. See Chapter B-3 for more on pink shrimp.

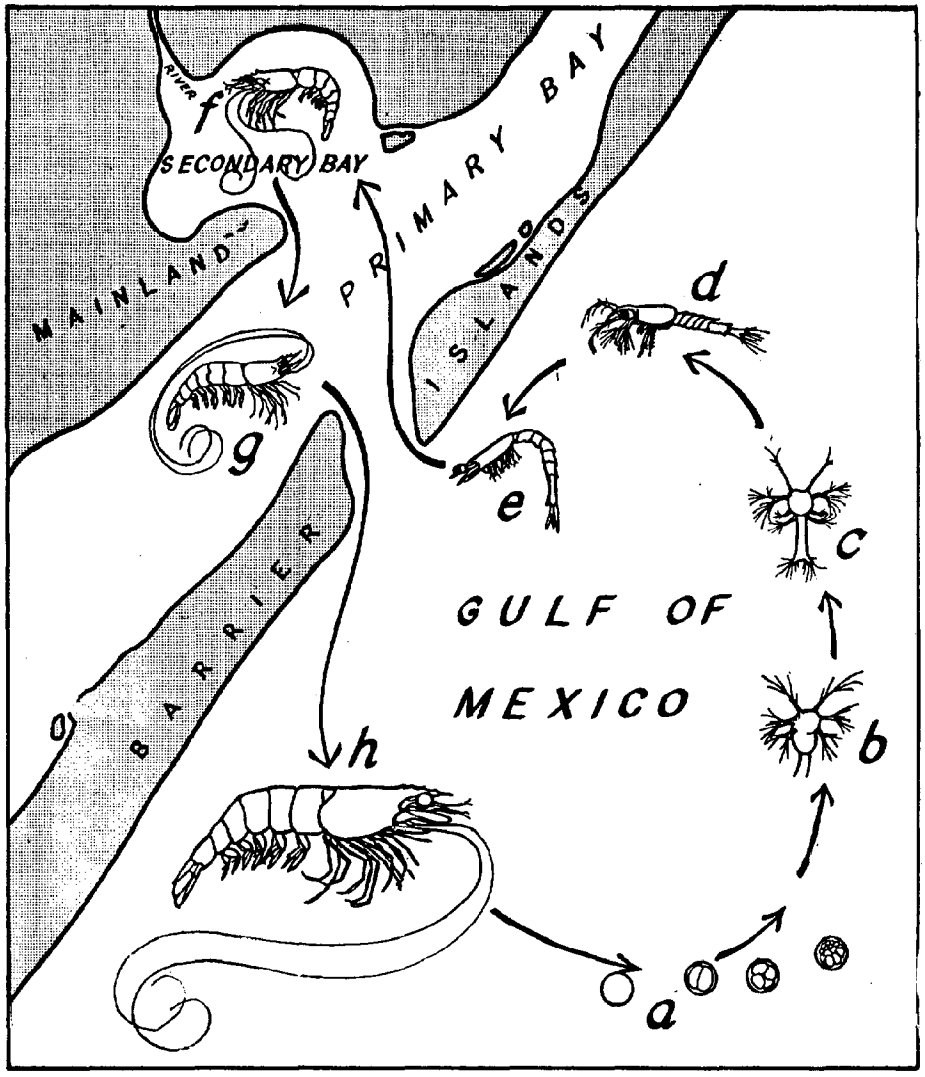


Fig. 3: Diagram of the shrimp life history. a) shrimp eggs; b) nauplius larva; c) protozoea; d) mysis; e) postmysis; f) juvenile shrimp; g) adolescent shrimp; h) mature adult shrimp. (From Guest, 1958; Fig. 4).

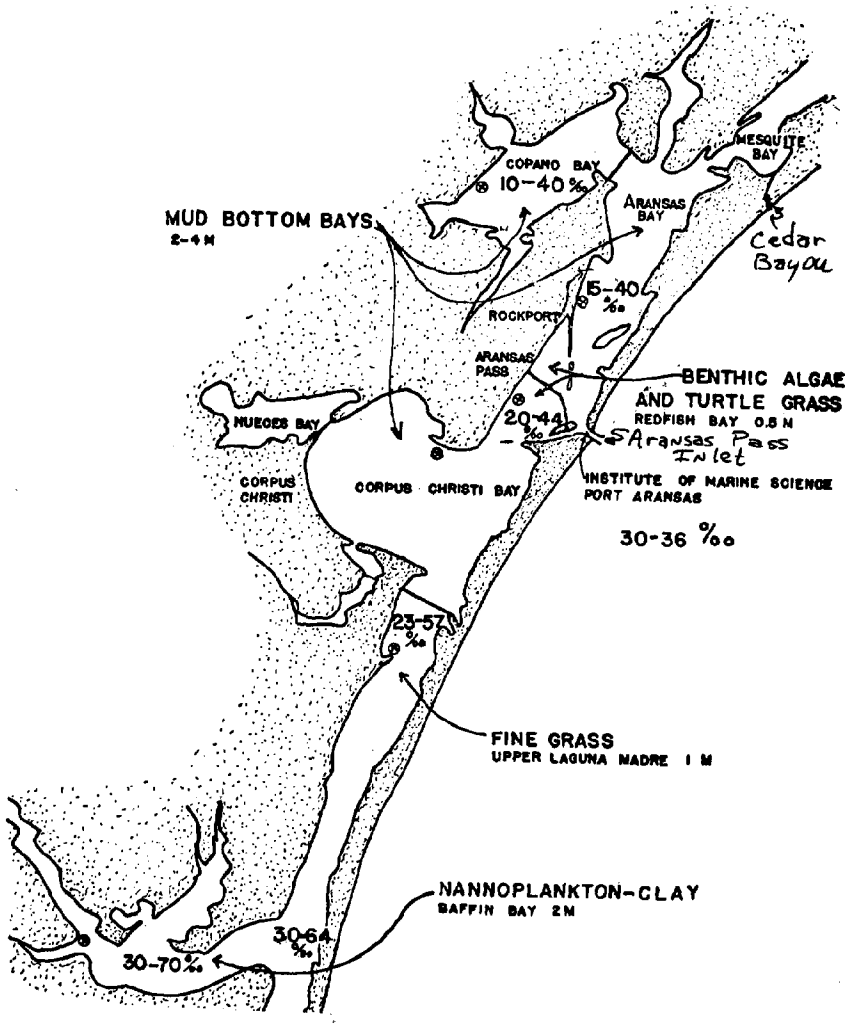
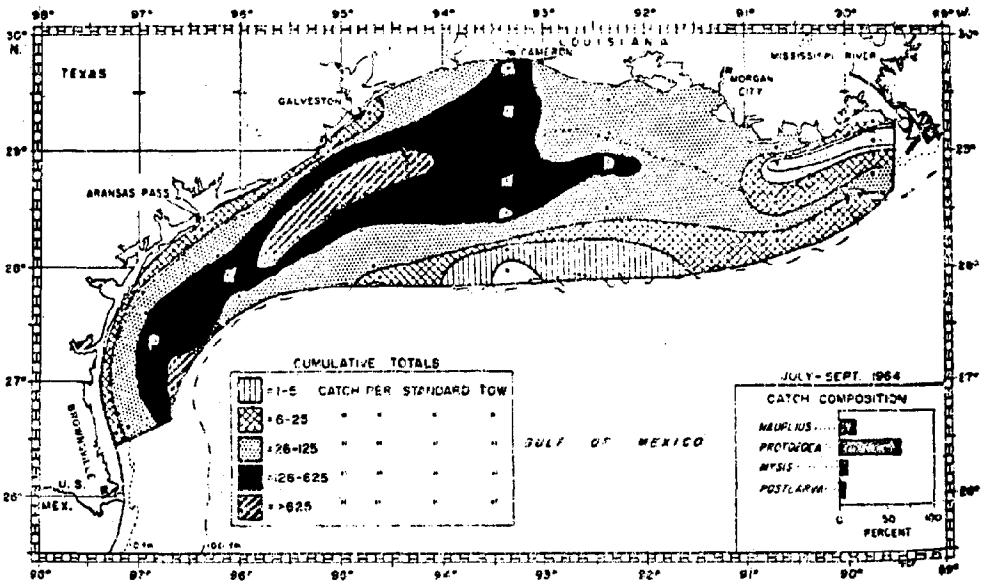


Fig. 4. Central Texas Coast showing Aransas Pass Inlet and Cedar Bayou (inlet) (Odum and Hoskin, 1958).



Distribution of planktonic stages of Penaeus spp. in the northwestern Gulf of Mexico, July-September 1964. Stations occupied are shown as open circles.

Fig. 5. Shrimp larvae off the Texas coast (Fischer in Lindner, 1967).

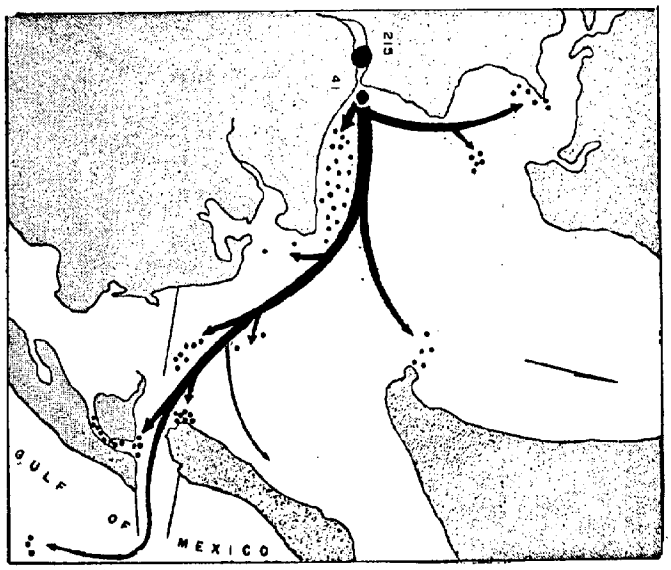


Fig.6. Outward migration of tagged brown shrimp from Galveston Bay (Inglis, 1960).

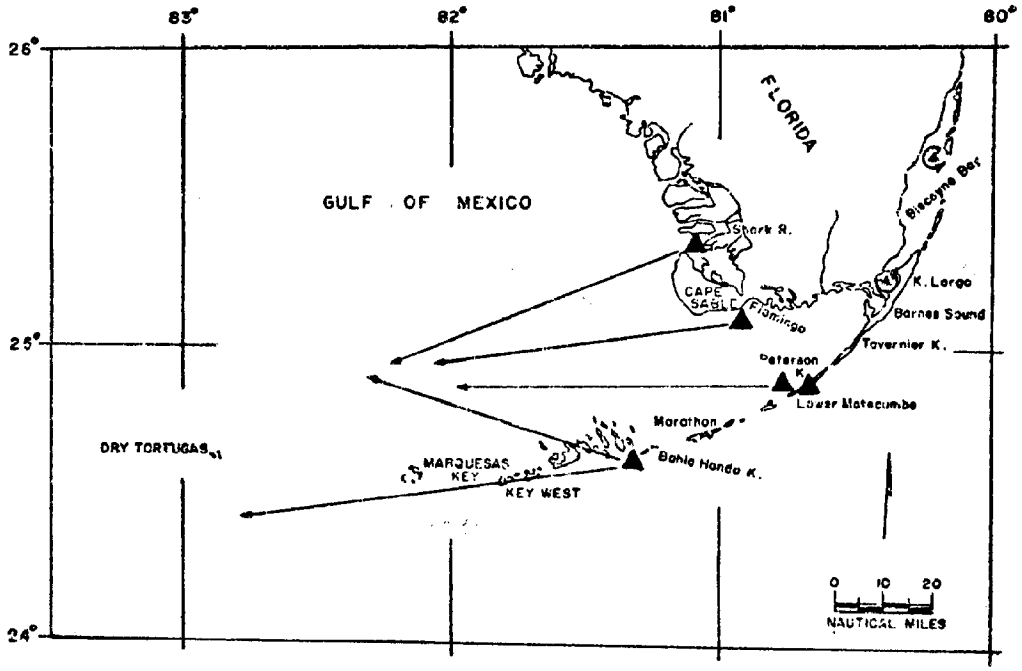


Fig. 7. Migration of tagged pink shrimp from shallow coastal nurseries to shelf waters where they were taken in the Tortugas fishery (Costello and Allen, 1960).

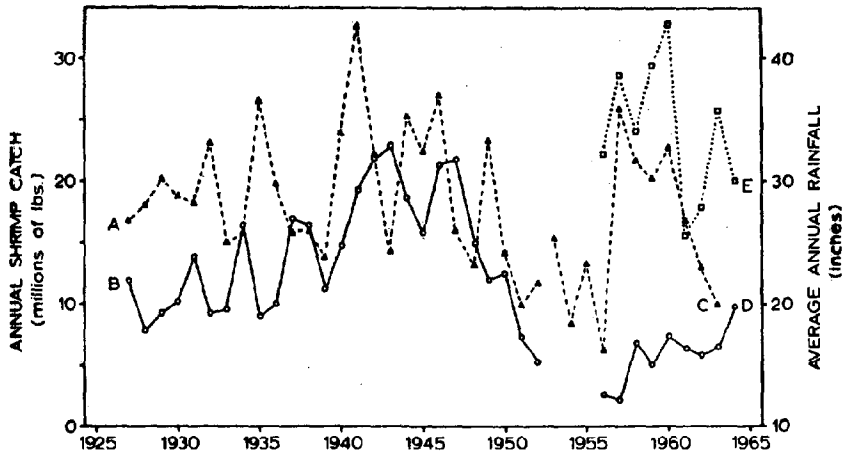


Fig. 8. Annual shrimp catch and average rainfall for Texas, 1927-1964. A and C: Average rainfall for all recording gauges in state of Texas. B and D: Annual catch of white shrimp from Texas waters. E: Annual catch of brown shrimp from Texas waters (From Copeland, 1966 a; Fig. 2).

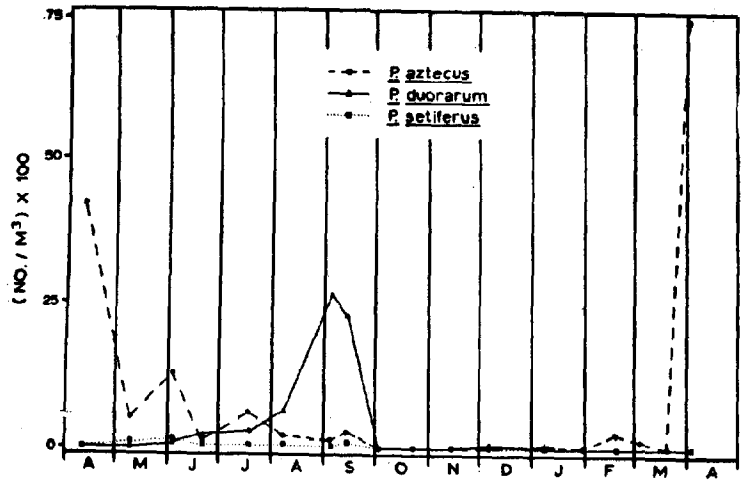


Fig. 9. Number of postlarval shrimp entering the Aransas Pass Inlet, Texas during 1963 and 1964 in number per 100 m^3. Each point represents the mean of 30 samples (From Copeland and Truitt, 1966; Fig. 3).

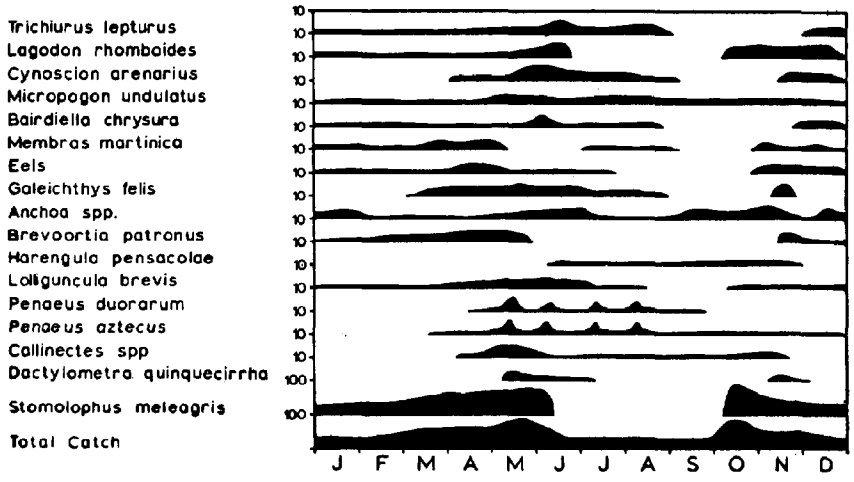


Fig. 10. Biomass in gm/m^3 wet weight of animals moving through Aransas Pass Inlet, Texas, from the bay to the Gulf, 1963 and 1964. The vertical scale is logarithmic (From Copeland, 1965b, Fig. 2).

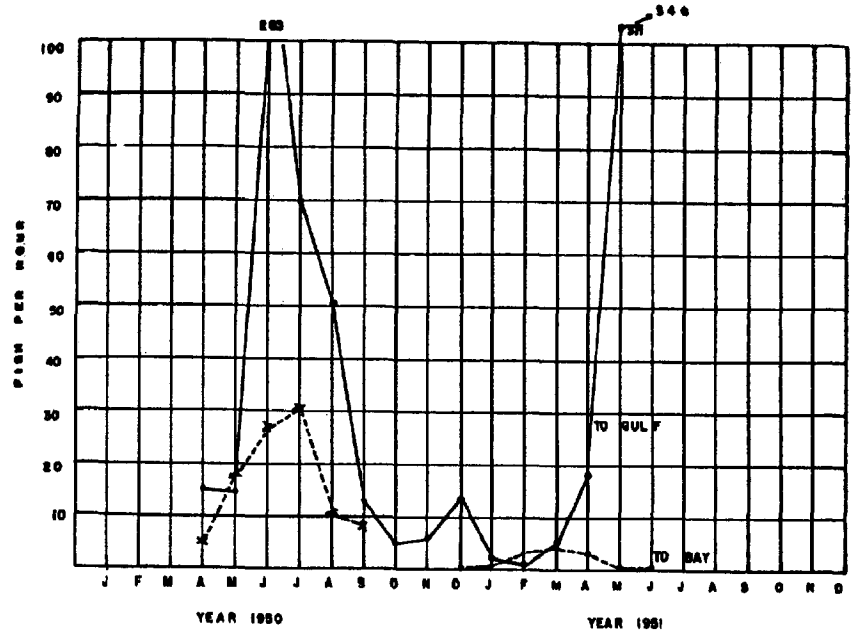


Fig. 11. Number of fish moving through Cedar Bayou, Texas from bay to Gulf and from Gulf to bay (From Simmons and Hoese, 1959; Fig. 12).

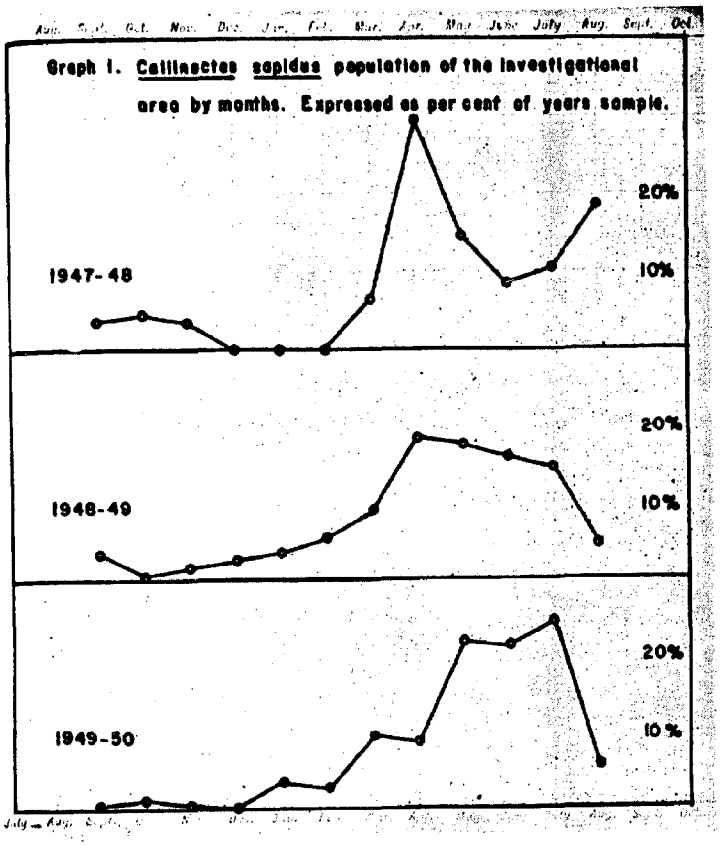


Fig. 12. Blue crab migrations through Cedar Bayou, Texas (From Daugherty, 1952; Fig. 1).

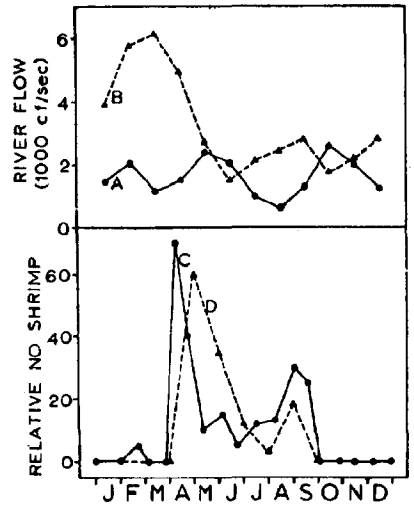


Fig. 13. Relative number of postlarval shrimp entering bays and river flows. A. Guadalupe River, Texas, 1953-1962; B. Neuse River, N.C., 1955-1964; C. Brown shrimp (Penaeus aztecus) entering Aransas Pass Inlet, Texas; D. Peneid shrimp post-larvae entering Brunswick-Onslow Bay, N.C. (Copeland, 1966a).

Pandalid Shrimp

Passing northward on both coasts pandalid shrimps replace penaeid shrimps as principal commercial fishery. In Alaska, Harry (1963) summarizes the patterns of population of Pandalus which develop stored resources during summer food growth periods, develop eggs by fall, and carry them all winter ready for release into the rising bloom of plankton in the Spring. In both Maine and Alaska, these shrimps move to the estuaries to release larvae.

Herring-like Fishes

Some schooling herring-like fish species (herring, alewives, anchovies, sardines, and menhaden) are found as principal members of the plankton-based marine ecosystems from the Arctic to the tropics, inshore and offshore. Although the location of spawning varies with species, data seem to fit the theory that whether spawning is inshore or sometimes offshore, the young ride the pulse of phytoplankton and zooplankton in the spring wherever net gains develop food concentrations. The migrations of the herring-like fishes tie the bays to the open shelf waters where they in turn on outgoing passage are a major food pulse in autumn for the southward moving stocks of larger carnivores, and the peak activity of the trawling of commercial fisheries.

The Gulf menhaden, Brevoortia patronus, exhibit a life-history very similar to that of the penaeid shrimp (Suttkus, 1956). The larvae enter coastal systems on flood tides and grow rapidly to the juvenile stage in the bay before emigrating back to the sea. The adult population constitutes a substantial commercial fishery in the Gulf of Mexico continental shelf. Seasonality of menhaden spawning in the Gulf of Mexico, is shown by Suttkus and Sundararaj (1961) with the volume of the menhaden gonads (Fig. 14). The largest mean volume of gonads occurred during January, which would account for the abundance of young menhaden in San Antonio Bay during March and April (Fig. 16) at time of spring food pulse (from light or rivers).

The Atlantic menhaden, Brevoortia tyrannus, on the other hand, may return to the bay during their adult stage (McHugh et al. 1959). In this way, they constitute a commercial fishery effort in Chesapeake Bay as well as offshore (Fig. 15).

In a study concerning the quantitative populations of organisms in San Antonio Bay, Texas (unpublished), F. N. Moseley and B. J. Copeland, The University of Texas at Port Aransas, reported the seasonal abundance of Gulf menhaden in the upper estuary (oligohaline zone). A summary of one year's data is shown in Figure 16. They reported a large increase in the menhaden biomass (in gm/m^2) during the spring of both 1966 and 1967, with almost negligible amounts during the remainder of the year. It appears, then, that menhaden migrate from the Gulf of Mexico as larvae or postlarvae and utilize the seasonal flux in productivity and food availability during spring before emigration to the Gulf later in the season (summer). This relationship

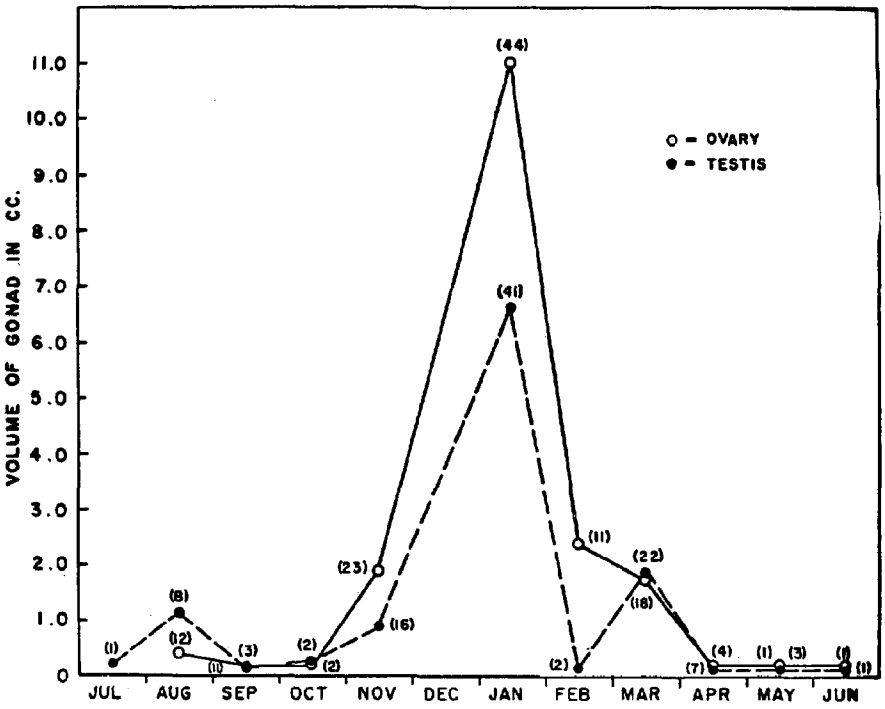
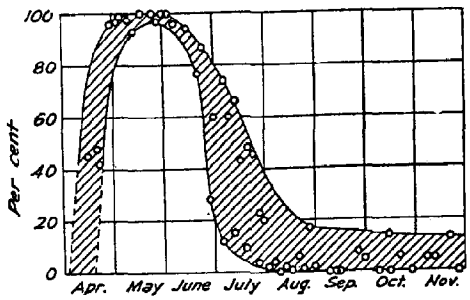


Fig. 14. Mean volume of Brevoortia patronus gonads for various months, using data collected during 1951 through 1958 in Louisiana (From Suttkus and Sundararaj, 1961; Fig. 1).



Seasonal variation in relative numbers of age-group I menhaden from Chesapeake Bay bearing narrow marginal growth increments at the edge of the scale, 1954 and 1955.

Fig. 15. Pulse of juvenile menhaden in Chesapeake Bay at the time of spring production pulse from phytoplankton to zooplankton (McHugh, Oglesby, and Pacheco, 1959).

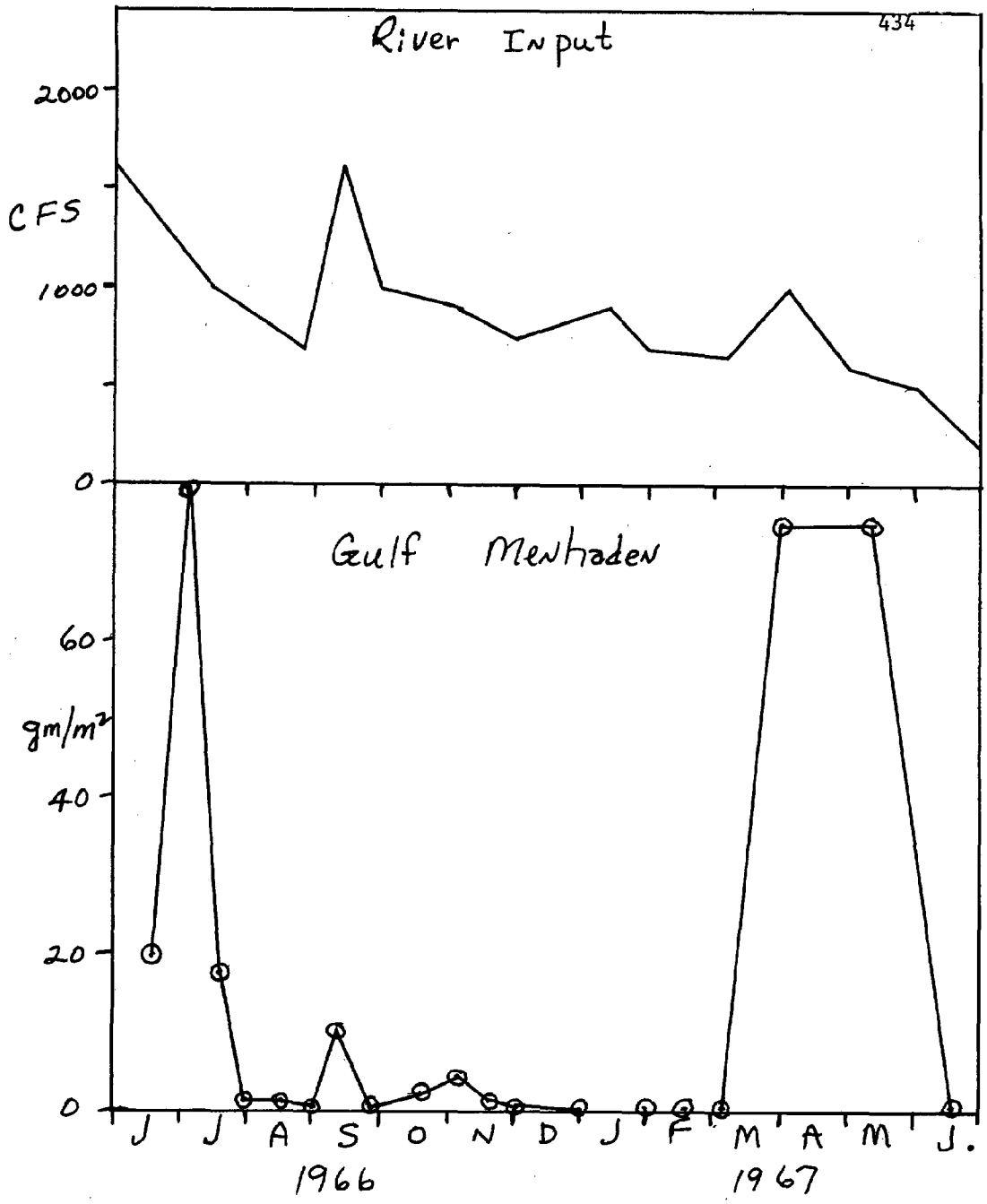


Fig. 16. Seasonal abundance of the Gulf menhaden (Brevoortia patronus) in gm/m^2 (lower curve) and freshwater input from the Guadalupe River in CFS (upper curve) for San Antonio Bay, Texas (From Moseley and Copeland, The University of Texas; unpublished data).

corresponds to the findings of Copeland (1965b) concerning measurements of emigrating juveniles through the Aransas Pass Inlet (see Fig. 10). Gunter and Christmas (1960) reported similar results for other bay systems along the Gulf coast. The peak populations (as high as 80 gm/m^2 in Fig. 16) are composed almost entirely of very small fish (about 20 mm standard length), indicating that a tremendous migrating wave of small menhaden enter Gulf coast estuaries each spring. In New England in Fall southward movement of young Menhaden along the shore (Fig. 17) was shown by Allen et al. (1960). The heavy pulse of spring herring reproduction inshore (Skud, 1959) in Pacific waters is shown in Fig. 18. Skud (1961) shows the dominant year class phenomenon exists in herring at Juneau, Alaska, showing a system stabilizing role for herring in Alaska as in Europe. In Maine there is an offshore gyral to bring offshore larval release into the estuaries for spring growth (Graham and Venno, 1968).

American Shad

The American shad (Alosa sapidissima) is anadromous, spending most of its life in the sea and ascending coastal rivers to spawn (Stevenson, 1899; Talbot and Sykes, 1958; Sykes and Talbot, 1958; Walburg and Nichols, 1967). The young remain in the natal stream until autumn and then enter the ocean, where they may migrate north during summer and south during winter (Walburg and Nichols, 1967). Apparently, the shad will travel great distances up rivers for their spawning activity (Table 1).

Talbot and Sykes (1958) reported studies of the migrations of American shad carried out over a period of 19 years by tagging experiments. Their analysis revealed a consistent migration pattern, which can be summarized as follows:

> "After spawning, adult shad in streams from Chesapeake Bay to the Connecticut River migrate northward and spend the summer and fall in the Gulf of Maine. Canadian shad migrate southward to the Gulf of Maine and also spend the summer and fall there. There is little evidence as to where shad spend the winter months; but it appears that they are scattered along the Middle Atlantic area, for beginning in January or February as the spawning season approaches, they move inshore and are taken in the commercial fisheries from North Carolina to Long Island. They then migrate either north or south to their native streams and spawn, repeating this cycle each year that they escape natural and fishing mortalities. The young shad leave their native streams in the fall, probably spend the winters in the Middle Atlantic area, migrate to the Gulf of Maine each summer with the adults, and when mature return to their native streams to spawn. Those that spawn in streams south of Chesapeake Bay, and particularly south of North Carolina, die after spawning."

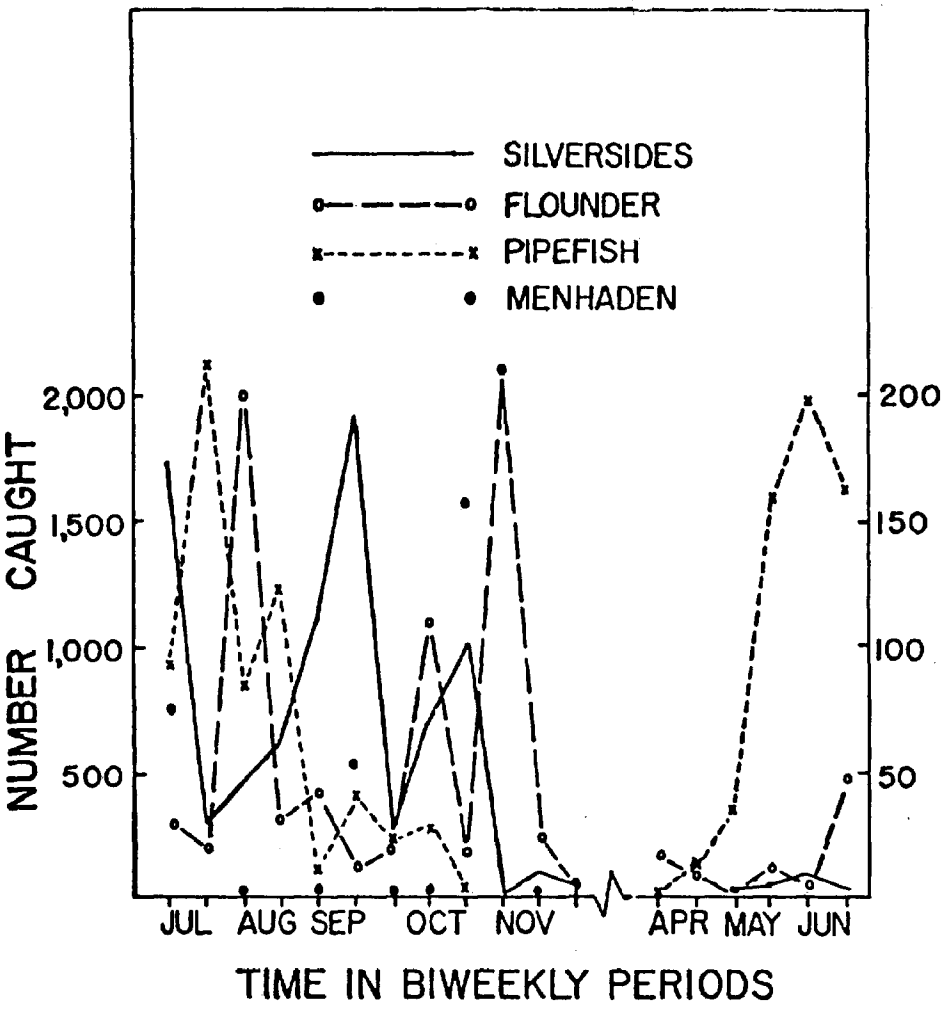


Fig. 17. Fish caught in shallows along the estuarine shore at New Haven, Conn. by Warfel and Merriam, 1944. (Allen, Delacy , and Gotshall, 1960).

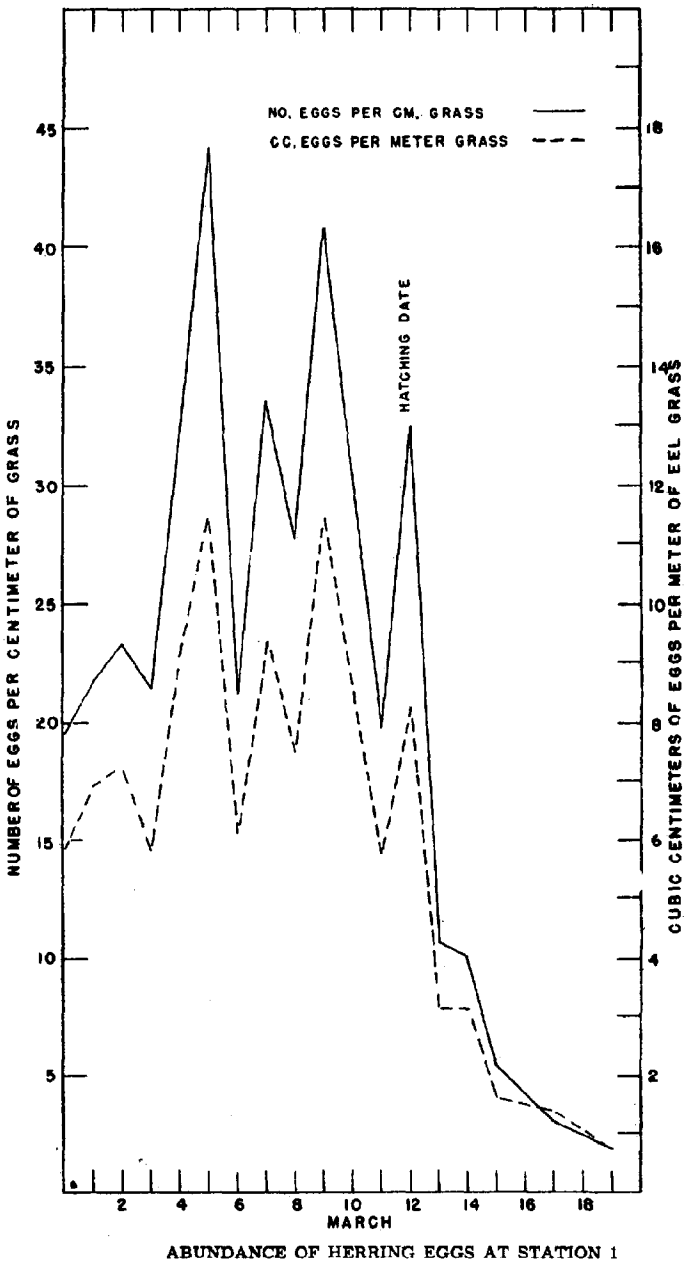


Fig. 18. Massive release of spawned eggs in littoral bottom vegetation by Pacific herring(Clupea pallasii), larvae emerging in phase with coastal spring productivity pulse (Skud, 1959).

Table 1: The orginal, 1896, and 1960 limits of shad range in 23 major rivers of the Atlantic coast (Walburg and Nichols, 1967; Table 1).

River	Distance of source above coastline	Original limit of shad run		1896 limit of shad run		1960 limit of shad run	
		Locality	Distance from coastline	Locality	Distance from coastline	Locality	Distance from coastline
	Miles		Miles		Miles		Miles
St. Johns. . . .	375	Sources.	375	Sources.	375	Lake Washington. . .	250
Altamaha	450	Macon.	370	Hawkinsville . .	300	Hawkinsville	300
Ogeechee	350	Ogeechee Shoals. .	200	Millen	100	Midville	125
Savannah	425	Tallulah Falls . .	384	Augusta Dam. . .	209	Savannah Lock and Dam.	180
Edisto	300	Sources.	300	Jones Bridge . .	281	Norway	120
Santee: Wateree. . . .	350	Great Falls. . . .	272	Great Falls. . .	272	Santee Dam	65
Congaree . . .	410	Green River. . . .	374	Columbia	233	Santee Dam	65
Pee Dee.	497	Wilkesboro	451	Grassy Island. .	242	Blewett Falls Dam. .	242
Cape Fear. . . .	290	Haywood.	210	Smiley Falls . .	181	Lock No. 1	65
Neuse.	340	Sources.	340	Fish Dam	300	Milburnie.	165
Pamlico-Tar. . .	252	Rocky Mount. . . .	157	Rocky Mount. . .	157	Rocky Mount.	157
Roanoke.	457	Weldon	249	Weldon	249	Spring Hill.	215
James.	420	. . . . .	370	Boshers Dam. . .	140	Boshers Dam.	140
Rappahannock . .	248	Falmouth	155	Falmouth Falls .	155	Falmouth Falls . . .	155
Potomac.	400	Great Falls. . . .	190	Great Falls. . .	190	Little Falls Dam . .	180
Susquehanna. . .	617	Binghamton	513	Clarks Ferry . .	279	Conowingo Dam. . . .	205
Delaware	457	Deposit.	256	Burrows Dam. . .	196	Deposit, N. Y. . . .	260
Hudson	314	Glens Falls. . . .	209	Troy	164	Coxsackie.	130
Housatonic . . .	202	Falls Village. . .	150	Birmingham . . .	92	. . . . .	No shad
Connecticut. . .	409	Bellows Falls. . .	204	Windsor Locks. .	89	Turners Falls. . . .	130
Merrimac	140	Winnepesaukee. . .	125	Lawrence	20	Lawrence	20
Kennebec	155	Carritunk Falls. .	108	Augusta.	44	. . . . .	No shad
Penobscot. . . .	255	. . . . .	90	Verona	35	. . . . .	No shad

It appears, then, that there is a strong spring spawning run which brings the shad into coastal systems along the Atlantic coast. The general decline in shad catch is shown in Fig. 19.

Crabs and Lobsters

The blue crab (Callinectes sapidus) makes spawning runs from the estuaries of the Gulf and Atlantic coasts during spring to spawn in the nearshore waters (Churchill,1919; Daugherty,1952). The blue crab, however, spends most of its life in the bay systems, with occasional migrations to the lower parts of rivers. Crabs constitute a substantial commercial fishery in some bay systems along the Atlantic and Gulf coasts.

Daugherty (1952) reported the migrations of the blue crab (Callinectes sapidus) through Cedar Bayou, Texas. The results of his study are shown in Figure 12. The adult blue crab migrates from the bay to the nearshore Gulf to spawn during spring, as indicated by the migration data reported by Daugherty. The megalops larvae of the crab then migrate into the bay systems within a short time, thus arriving during the time of maximum availability of food and optimal environmental conditions.

A top carnivore niche in northern waters is occupied by the big clawed lobster (Homarus) on the east coast and by the king crab (Paralithodes) in Alaska. These large animals when not under fishing pressure that keeps the stocks young develop large animals 10 or more years of age. Their diets switch with availability and both shift to deeper waters offshore in winter and inward into estuaries in summer releasing larvae into the pulse of food productivity in the estuarine nursery. Bright et al.(1960), for example, show release of king crab larvae into the Cook inlet gyral at the time of the main spring plankton blooms. After inshore release of larvae they return to deeper water by summer (Reynolds and Powell, 1964). Sakuda (1966a)and Powell and Nickerson (1965) find maximum content of the adults during the winter suggesting storage role prior to spring spawning and molting cycles. Powell (1967) documents the long seven year growth cycle and the history of some fisheries being depleted in 7 years in small bays on Kodiak Island. The Maine Lobster fishery is summarized by Scattergood (1960a). Sherman and Lewis (1967) show lobster pulse in summer with plankton pulse. Activity of adults and their liklihood of capture is temperature programmed (Dow, 1967).

Striped Bass

The striped bass, Roccus lineatus, spawn in rivers upstream from the bay during spring (Merriman,1937). The larvae are transported downstream into the estuary where they undergo rapid growth rates during the summer. Migrations out of the bay occur during the fall, with the population migration south on the Atlantic continental shelf (Fig. 20). There is a return trip to the north in the spring and the adults may again enter bays. During the migration south in the fall, the migrating population decreases in number as various individuals enter bays along the way. On the other hand, the migration north during spring increases in numbers as outward migrating adults join the northward migrating group.

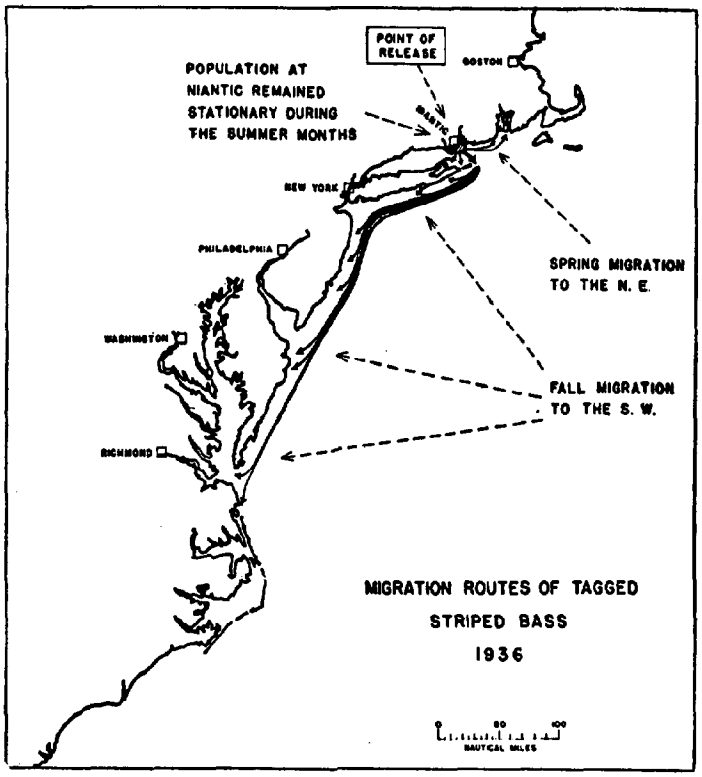


Fig. 20. Chart of the Atlantic coast showing the migrations of striped bass as determined from tagging returns (From Merriman, 1937; Fig. 6).

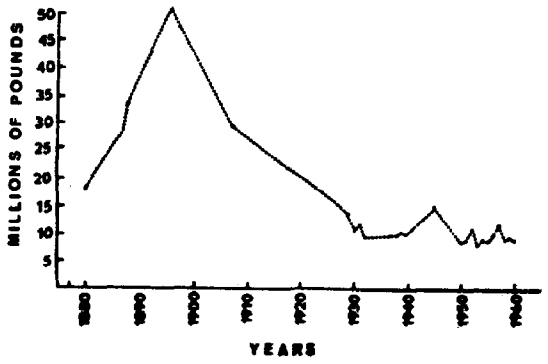


Fig. 19 . Shad catch on the Atlantic coast of the United States (Walburg and Nichols, 1967).

There is a strong seasonal migration of striped bass, with the maximum movement through bay systems during spring and autumn (Merriman, 1937 and others). As shown in Figure 21, catches of striped bass in coastal systems are highly seasonal, indicating a relationship between migrations and food availability.

Salmon

Salmon, including the Pacific species of Oncorhynchus and the Atlantic Salmo, are anadromous; spending adult life at sea, returning to rivers to spawn (Ricker, 1966, and Neave 1966a, 1966b). (See Figure 22). Each species is different in timing and location of its nursery stage. It now seems possible to relate these programs of behavior to the pulse of productivity in the environments that the salmon use for food. Consider the sharp pulse of productivity and migrations in Alaska. The migration of pink, coho, king, sockeye, chum salmon and migrating rainbow trout, and dolly varden char (Revet, 1962) are principle patterns of sharp seasons of Alaska. History of one main salmon system and its disturbance by man is given by Pennoyer, Middleton, and Morris (1965). Bailey (1968) documents young salmon consumption of estuarine zooplankton of the outbound migration. A general summary of the Salmon of Alaska-North Pacific system of migratory exchange is given in a popular pamphlet from Department of Fisheries, 1155 Robson St., Vancouver. Generalities about the separation of the main nursery food niches of the species are : Pinks, intertidal spawning and with estuarine nursery; king-chinooks in rivers; silver-coho in creeks; sockeye-reds in lakes; chums spawn in freshwater but with nursery growth in sea. Meehan (1966) showed importance of food chains of post larval period to growth of salmon moving seaward. See also Thorsteinson (1959).

Sheridan (1966) summarizes workshop with recent studies on Pink salmon in estuaries. One of the included contributions by J. W. Martin indicates timing and growth productivity of fry that enter estuaries in May remaining during maximum period of estuarine productivity among the islands. Then becoming juveniles and after rapid growth they school in August and move out into the open sea. The pinks in their migration place their growth drain on the estuary at the time of its great energy pulse (Tyler, Davis and Bevan, Fig. 24 and 25). They eat herring on their return (Reid, 1961). What happens in alternate years at low pink migration? Who gets the herring that year? See also Gilhausen (1962) and Manzer and Shepard (1962).

Foerster (1968) summarized the life history, migrations and factors affecting the abundance of the sockeye salmon. Spawning occurs in late summer and autumn, usually in streams that originate as lakes. There are runs of adult salmon through the coastal systems during autumn, to enable the adult to reach spawning areas far upstream. The fry, which hatch in April and May, spend about two years in the lakes at the origin of the streams. The young fry start their seaward migration in April to June, at the beginning of their second or third year. Thus, sizeable salmon enter the coastal systems during spring. They remain in the sea feeding areas for about one to four years and, with the onset of maturity, turn back again to coastal waters and to their native streams. Salmon in estuary are in Fig. 23. Lagler and Wright (1962) characterize the fishes in the food chain of outbound salmon especially dolly varden. Though restocking and other steps the salmon migration in Maine still runs in small numbers (Table 2, See page 353).

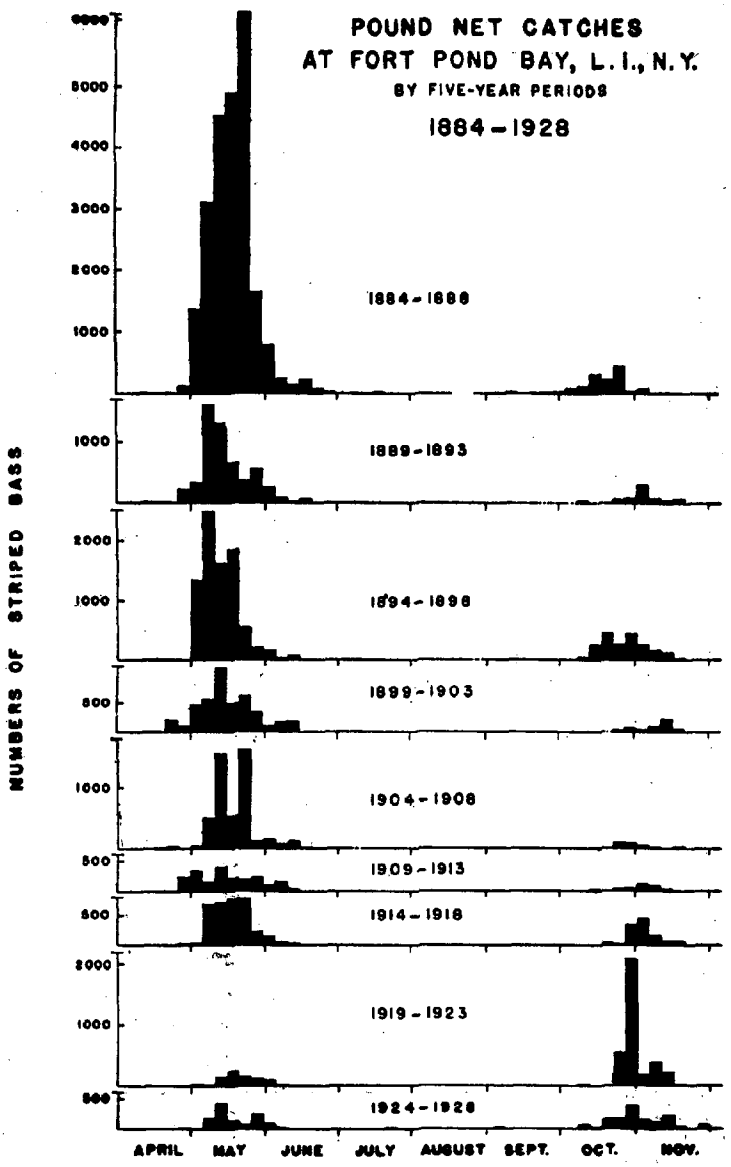


Fig. 21. Number of striped bass caught each 5 days, by 5-year periods in Fort Pond Bay, N. Y. (From Merriman, 1937; Fig. 2).

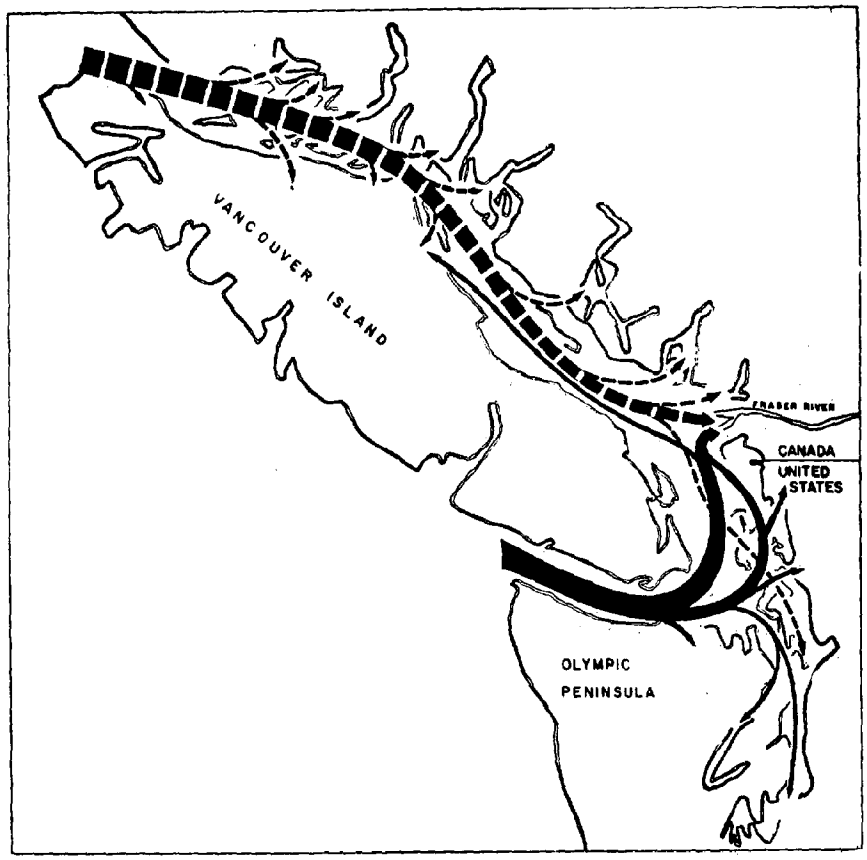
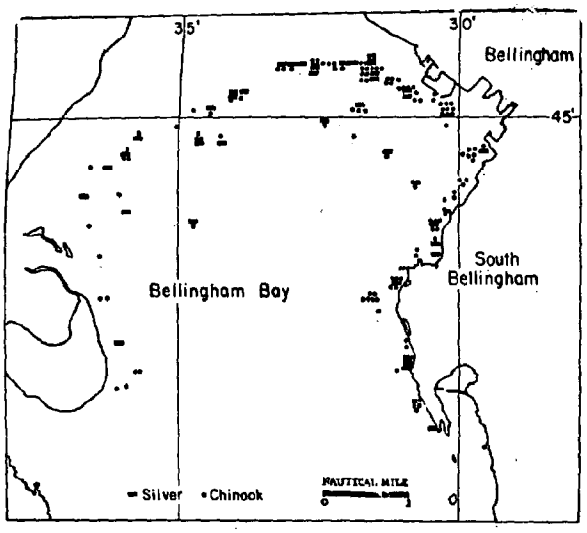
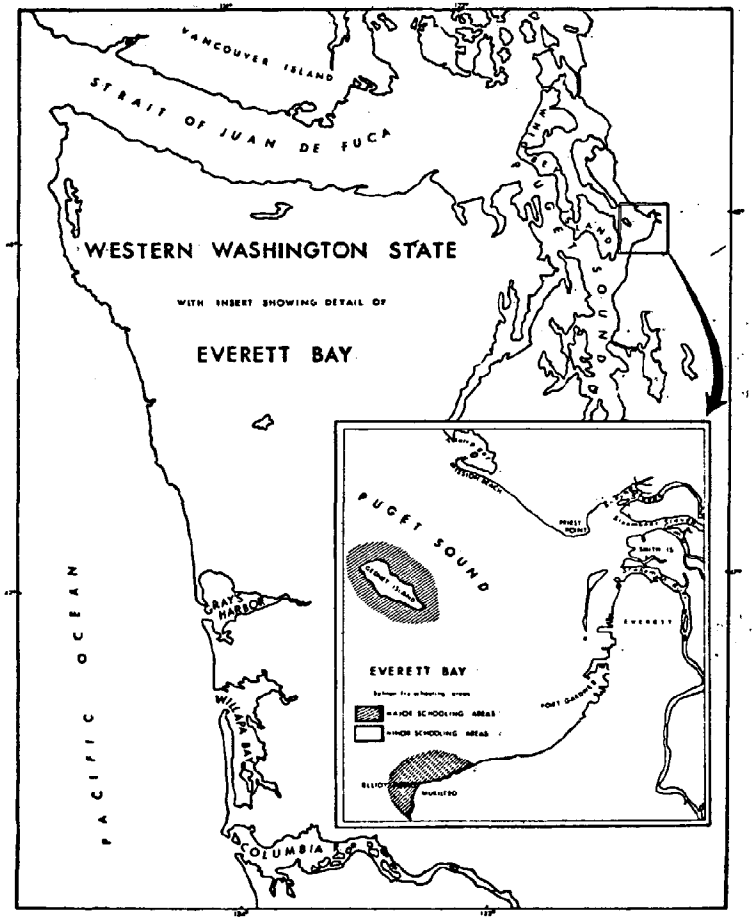


Fig. 22. Schematic presentation of migration routes followed by pink salmon in southern British Columbia and Washington waters (From Vernon et al. 1964; Fig. 1).

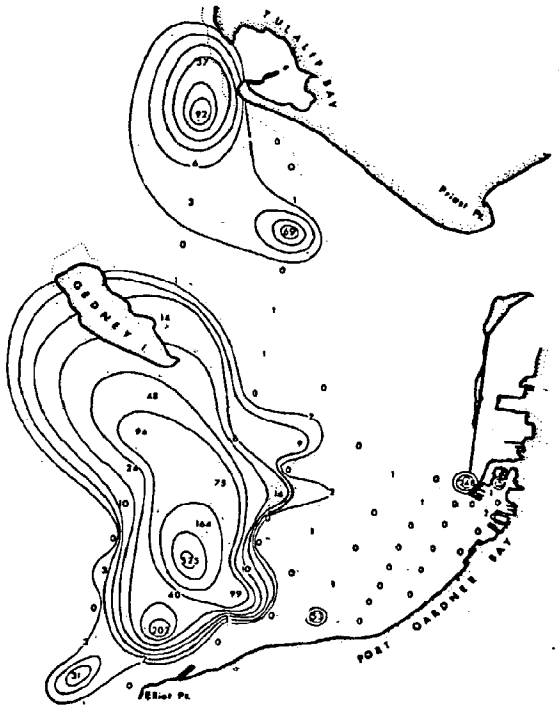


Distribution of marked salmon in Bellingham Bay.

Fig. 23. Salmon in estuaries of Washington (Miller, Wetherall, Zebold Lenarz, Stauffer, Fujioka, Halstead, Salo, and English, 1968).

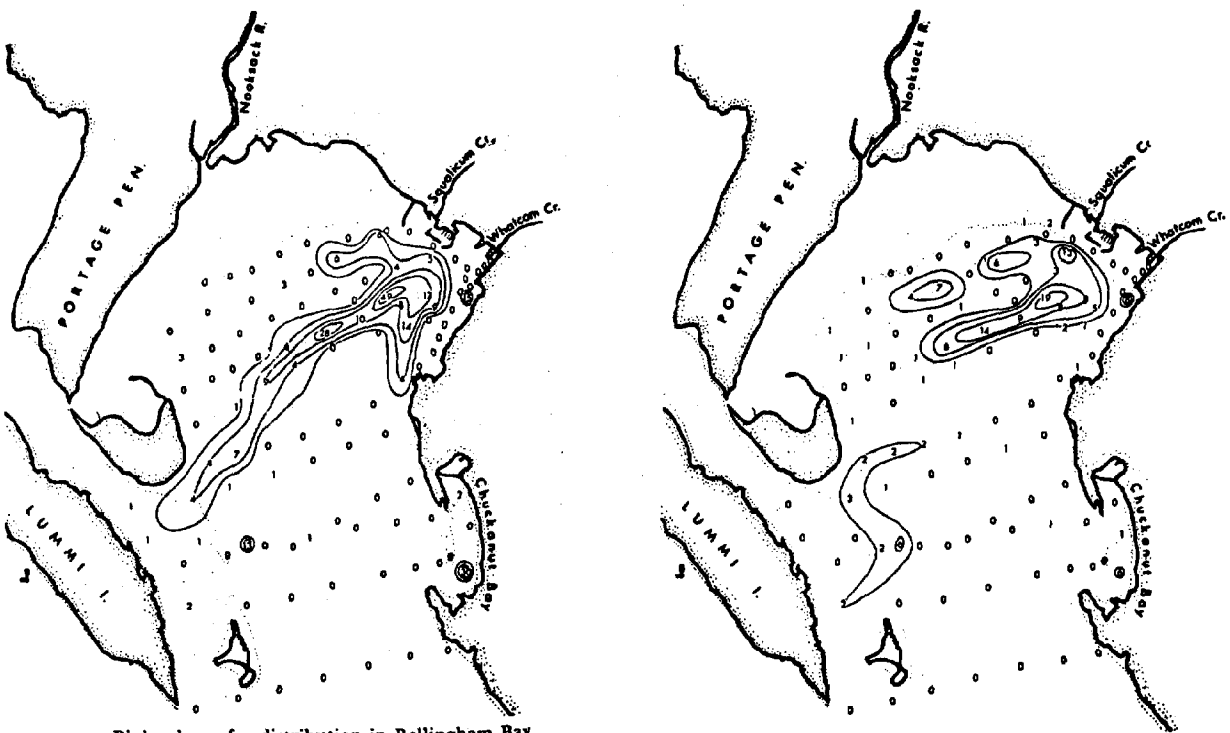


Western Washington with inset of Everett Bay.



—Pink salmon fry distribution in Everett Harbor, April 9-14, 1964.

Fig. 24. Passage of outbound pink salmon through estuaries at time of spring pulse of productivity (Tyler, Davis, and Bevan, 1962; Tyler and Bevan, 1965a).



Pink salmon fry distribution in Bellingham Bay, April 16-23, 1964.

Chum salmon fry distribution in Bellingham Bay, April 16-23, 1964.

Fig. 25. Young salmon outbound in Washington estuaries at time of plankton and river import food pulse(Tyler and Bevan, 1965a).

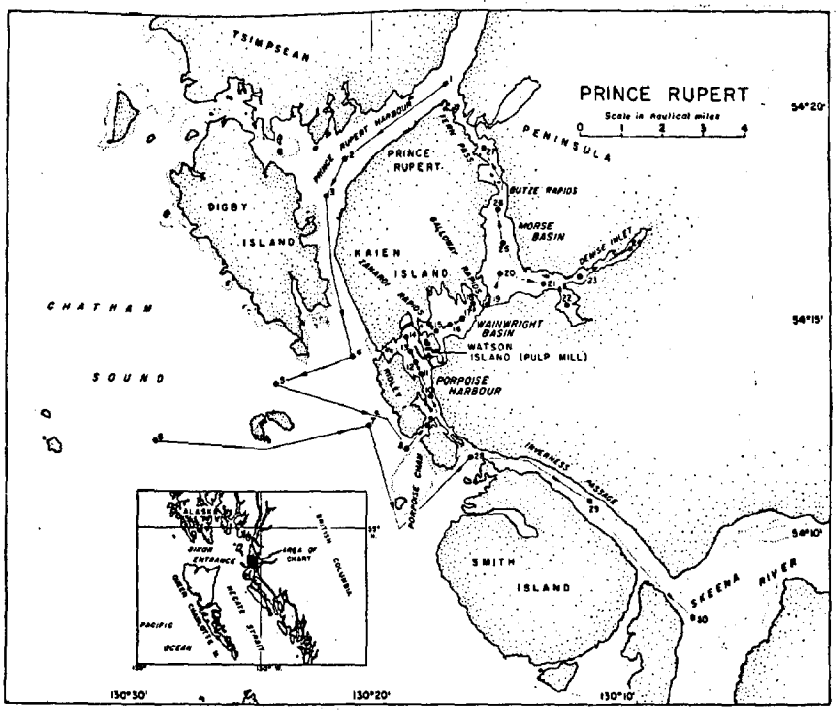


Chart of the Prince Rupert area showing location of stations occupied during September 1961 (No. 1-18), and during April 1962 (No. 1-30).

Fig. 26. Location of low oxygen conditions due to sulfite waste in western Canada (Waldichuk, 1966).

DISCUSSION

The Aransas Pass System

Aransas Pass Inlet in Texas is a tidal pass connecting 500,000 acres of Texas bay systems to the Gulf of Mexico (Fig. 4).which illustrates the magnitude and variability of migrating animals (Copeland, 1965b). The total catch and the catch of 17 more abundant species are shown in Fig. 10. Total average catch was less than one gm/m^3 during Januarv. February, July, August and September; between 1 and 10 gm/m^3 during March, November and December; and between 10 and 100 gm/m^3 during April, May, early June and October. The times of peak migrations correspond with maximum energy flux in the coastal systems of Texas (Odum and Hoskin,1958; Odum and Wilson,1962). As illustrated in Fig. 11, Simmons and Hoese (1959) reported similar results for Cedar Bayou (see position of Cedar Bayou shown in Fig. 4).

The amount of biomass leaving the shallow, highly productive bays of Texas into the Gulf of Mexico is large (Copeland,1965b). Copeland made calculations to illustrate the magnitude of migrating subsystems from the bays to the Gulf. Using one of the larger catches (during October 94.6 gm/m^3) and multiplying by the vertical area of the pass times the speed of the current, it was estimated that 352 kg/sec were passing through the Aransas Pass to the Gulf. To obtain a more realistic value for annual migrations, average biomass movements and current structure were used. From the 150 times during the year that migration movements were sampled, the average ebb tide current speed was 0.65m/sec and the average biomass was 7.2 gm/m^3. Application of the figures to the vertical area of the pass yielded a value of 22.15 kg/sec migratory movement for about four hours each day, the migration biomassbecoming 318,960 kg/day for an annual average. Totaling this value, one obtains the annual movement from the bays to the Gulf as 11.65×10^7 kg/year. Since all the bay area south of San Antonio Bay and north of the Laguna Madre Land Cut are considered to be dependent upon the Aransas Pass Inlet for their connection to the Gulf, one half million acres of bay are involved. Extending the above figures, the net productivity of these bays was computed to be 233 kg/acre (576 kg/hectare) per year (Copeland 1965 b).

The migrating animals coming into the bay systems do not constitute a substantial biomass (Copeland,1965 b). Plankton samples have revealed that almost all species collected in the outgoing biomass are represented by their larvae in the incoming plankton from the sea (Hoese,1965; personal observation). Although the total biomass of these planktonic organisms is relatively small because of their small size (usually a few mm), the number of them is extremely great. The number of shrimp postlarvae migrating through the Aransas Pass Inlet, Texas into the bay system (Fig. 4) is relatively large. Simmons and Hoese (1959) reported a very small number of large organisms migrating through Cedar Bayou into the bay system.

Seasonal Aspects

There is a definite relationship between migration of most organisms and the time of maximum production in the bay systems. Two major sources of foods in bay systems, especially the plankton based temperate systems, are the process of photosynthetic productivity and river-borne input (Nash, 1947; Copeland, 1966a; Odum and Wilson, 1962; unpublished data for Texas Bays). Indeed, the major migration runs occur during spring and/or autumn for most migrating organisms (see Hoese, 1965 for discussion concerning Texas fishes), corresponding to the season(s) of maximum food availability. As shown in Figures 4, 9-13, the migrations of organisms from and into Texas bay systems occur during spring and autumn. Pearcy and Richards (1962) reported a similar relationship for the Mystic River estuary in Connecticut.

The entrance of postlarval penaeid shrimp into bay systems through the Aransas Pass Inlet, Texas and the Brunswick-Onslow bay area, N. C. corresponds to high flow of the rivers of the area during spring and autumn (Fig. 13). Undoubtedly, this coupling of peak migration and increased river flow (accomplished through centuries of natural selection) is essential for the propagation of penaeid shrimp (Copeland, 1966a). Important fluxes occur in bay systems during the high river flows of spring and autumn. These include vitamins and other dissolved organic compounds (Birke, 1968), nutrients (Nash, 1947), lowered salinity (Odum and Wilson, 1962), and flushing and mixing activities (Pritchard, 1967b) In addition to important foods and vitamins transported by rivers into bay systems, the indirect stimulus of the incoming nutrients enhances photosynthetic productivity (Nash, 1947; Odum and Wilson, 1962). The relationship between river input and phytoplankton production in the Patuxent is illustrated in Figure 15. Hoese and Jones (1963) reported populations of fish and invertebrates in Redfish Bay, Texas during spring and autumn, corresponding to periods of maximum productivity and food availability (Chap. B-3).

DISTURBANCES

The popular impression that a general decline in the populations of migrating organisms has occurred over the past several years is by and large correct (Clark, 1966a). Many articles have been written in which much has been said about the effects of civilization upon the abundance or organisms required to enter coastal waters during their life-time. Walburg and Nichols (1967), however, in discussing the decline of the American shad (Fig. 16) over the past several decades, cited multiple causes. Undoubtedly this is true since most coastal systems are the receptors of multiple activities of modern man. We will attempt to discuss several disturbances in coastal systems as they may affect various migrating organisms.

Impoundments

The construction of dams on coastal streams has limited the distance that migrating forms may traverse upstream for spawning or other activities. As a matter of fact, the decline observed in Figure 19 for the American shad was mostly attributed to the fact that available rivers for spawning have decreased (Walburg and Nichols ,1967). In Table 1 are listed several major rivers of the Atlantic coast and the number of miles upstream that are utilized by shad for spawning runs; the number of miles available have decreased markedly over the past century.

As pointed out by Copeland (1966a), the increasing number of dams on rivers has decreased the amount of freshwater and its valuable food materials to coastal systems. Not only have the total amounts of freshwater formerly flowing to coastal systems been decreased, but the present flow is augmented to even the flow rate over an annual cycle thereby decreasing the spring and autumn "floods" that have been shown to be valuable for migrating subsystems. Smith (1966) and Chapman (1966) have reported similar findings. An open question is: How much compensating photosynthetic increase develops in clearer waters when rivers are cut off?

Andrew and Geen (1960) and French and Wohle (1966) discuss the effects of dams on streams used for salmon migrations. The Rock Island Dam on the Columbia River is a notable example and its construction essentially stopped the upstream migration of salmon until fish ladders were installed (which proved to be unsatisfactory).

Hurricane diversion dykes serve as barriers for migrating organisms. Saila (1962) demonstrated the effects of the hurricane diversion structure in Narragansett Bay, R. I. on the migration patterns of the winter flounder. Smith (1966) discussed the hurricane diversion structure on Lake Ponchartrain, La. Talbot (1966) discussed the effects of damming rivers, construction of irrigation diversion canals and salinity control barriers on west coast populations of striped bass.

Dredging

The dredging of canals has upset the current and circulation structure in many coastal systems, which alters the transport route of many larvae of river and sea spawned organisms relying upon certain current patterns to arrive in coastal systems. As pointed out by Smith (1966), the construction of the Mississippi River-Gulf Outlet in south Louisiana altered the current and exchange characteristics in Lake Ponchartrain, increasing its salinity.

Filling

The practice of bulkheading and filling shallow coastal areas to create valuable real estate has resulted in the loss of thousands of acres in some states of valuable nursery area utilized by migrating organisms. Smith (1966) demonstrated the loss of much of the nursery flats in Boca Ciega Bay, Florida. Diener (1964) discussed the loss of nursery area to dredging and filling in Texas estuaries. Talbot (1966) stated that over one-third of the surface area of San Francisco Bay has been filled since 1849, thus partially accounting for the decline in migrating organisms there.

Wastes

Various kinds of pollutants, many of which enter coastal systems, have been shown to either be toxic to migrating organisms or in some way alter their metabolism so that they no longer will inhabit the affected area. If

a pollutant harmful to the organism should be localized at the upper end of an estuary, for example, it would serve as a barrier to organisms migrating upstream or to the larvae drifting downstream to the estuarine nursery area. Some pollutants cause indirect effects that serve as barriers, such as high oxygen demand. (See Figs. 26 and 27.)

Pesticides

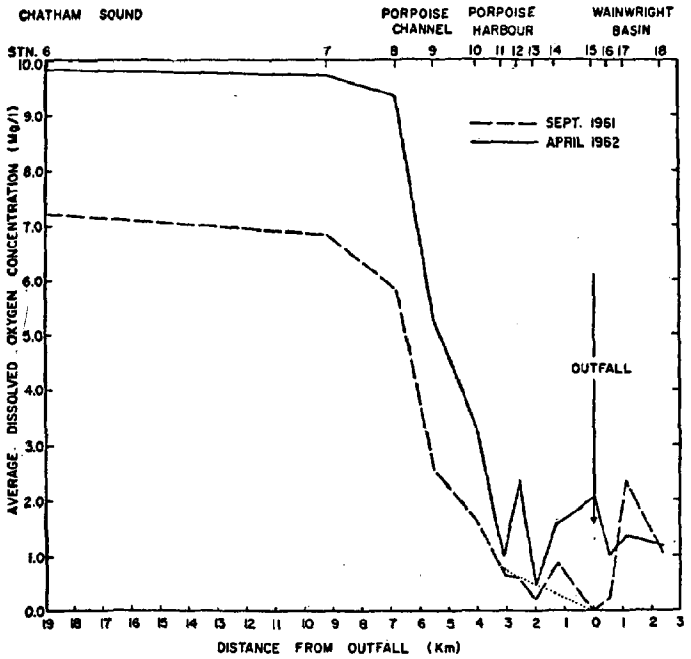
Pesticides may differentially affect different life-cycle stages of migrating organisms, thus either preventing spawning or killing larvae that come in contact with it. Chin and Allen (1957) showed the effects of insecticides on two species of shrimp found in Texas coastal systems. They reported that very small concentrations of insecticides would cause the shrimp not to inhabit waters containing these concentrations. Lowe (1965) reported that blue crabs were rendered sterile and physiologically upset in sublethal concentrations of DDT.

Organic Loading

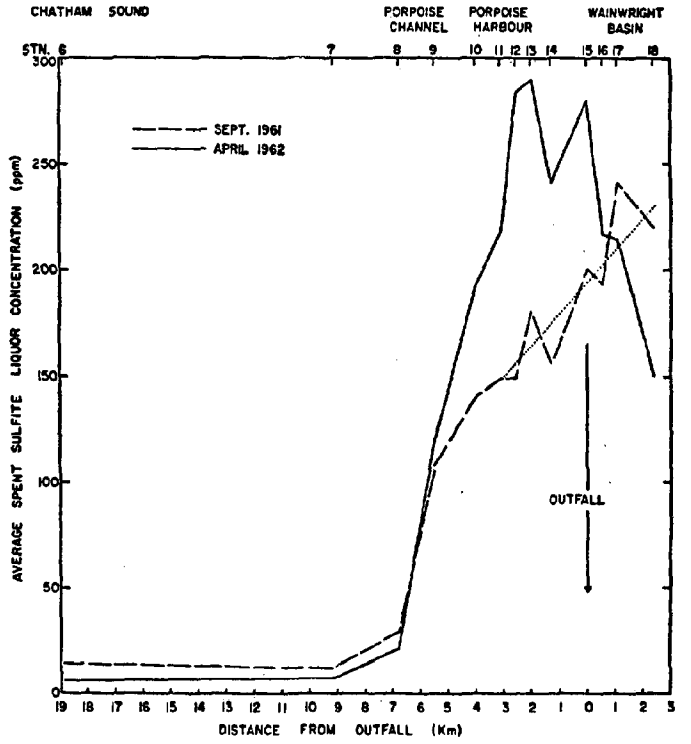
Large concentrations of organic materials from pollution sources usually exert a high oxygen demand upon the system, thus competing with organisms for the available oxygen. Concentrations of these oxygen-demanding wastes in certain areas of coastal systems could cause the oxygen level to drop below critical levels, thus acting as a barrier to migrating forms. Waldichuk (1960a,b) discussed the oxygen demand of paper mill wastes and demonstrated its effect as a barrier to migrating salmon in Bellingham Bay, Washington and in a Canadian estuary (Fig. 8). Bishai (1962) reported the results of tests to show the effects of low oxygen concentrations on salmonid larvae and smolts. He noted that the salmonid young required relatively high oxygen concentrations for normal behavior, thus leading to the conclusion that waters with low oxygen concentrations would act as a barrier to salmon migrations. Herman *et al.* (1962) have shown that coho salmon may grow little or not at all, and even lose weight in the presence of an abundant supply of food at low oxygen levels that could be tolerated for long periods of time.

Complementary and Coordinating Behavior Programs

As described with illustrations in Part I, the total food utilization is made up of many species each one of which may have a pulse of activity in the system at a different time. Particularly as one considers estuaries further south with more species, sequential timing is observed. Theoretical reasoning suggests that a well adapted system of species is one which as a group is using its food energy availabilities as they develop without irregular accumulations or cutting into necessary capital for starting the next year's energy pulse.



-Variation of average DO concentration for the total water column to 20 m depth, with distance from the pulp-mill outfall, for September 1961 and April 1962.



-Variation of average spent sulfite liquor concentration for the total water column to 20 m depth, with distance from the pulp-mill outfall, for September 1961 and April 1962.

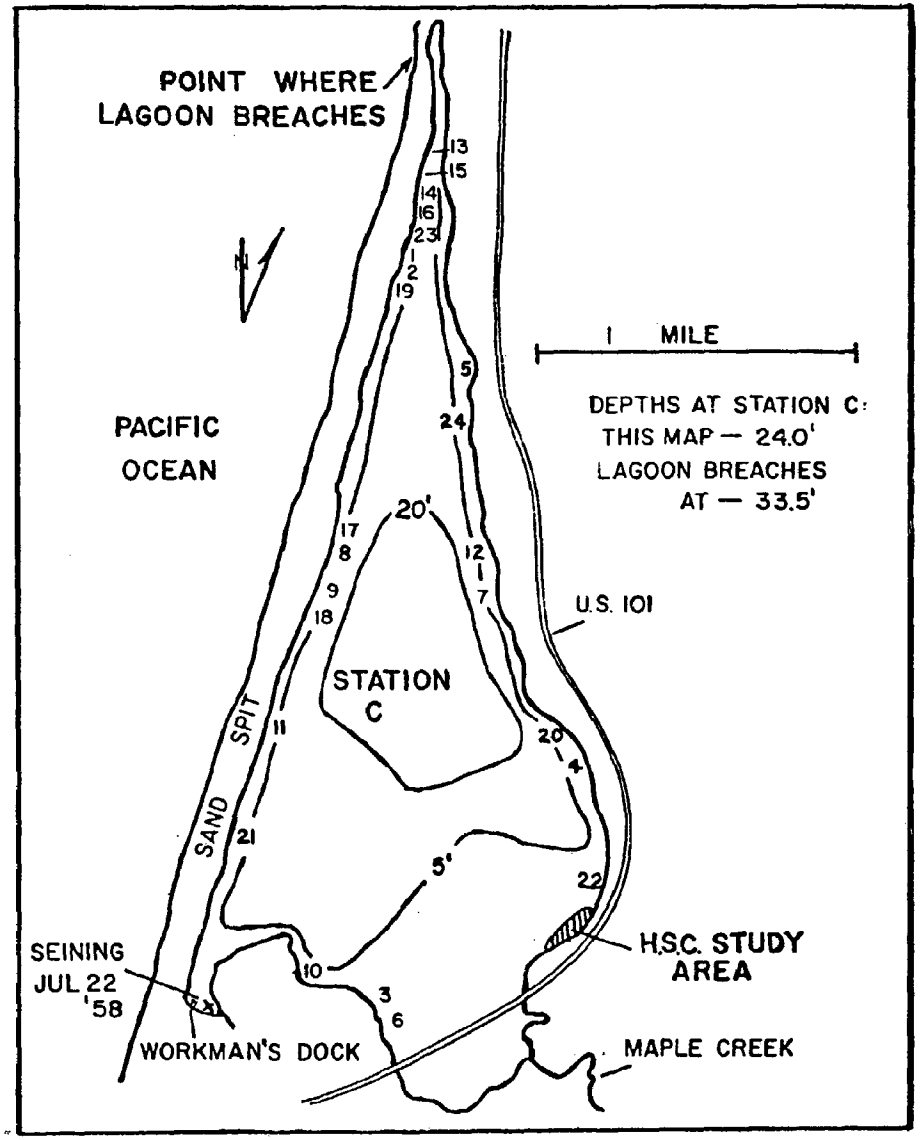
Fig. 27. Oxygen barrier in tidal inlet of Western Canada due to paper mill waste (Waldichuk, 1966). See Fig. 26 for location.

Among the migrants which arrive in the far north to nest in May when there is the brief period of high arctic productivity are the birds such as ducks and sandpipers. These birds start to move out as the energy balance declines after the days start to shorten June 22 moving southward with the same role as the marine fishes and swimming mammals. Recher (1966) shows the relationship of sandpiper migration to stored food availability on the beaches and flats at the end of summer. Habitat utilization and differentiation of niches is suggested by Figs. 28 and 29.

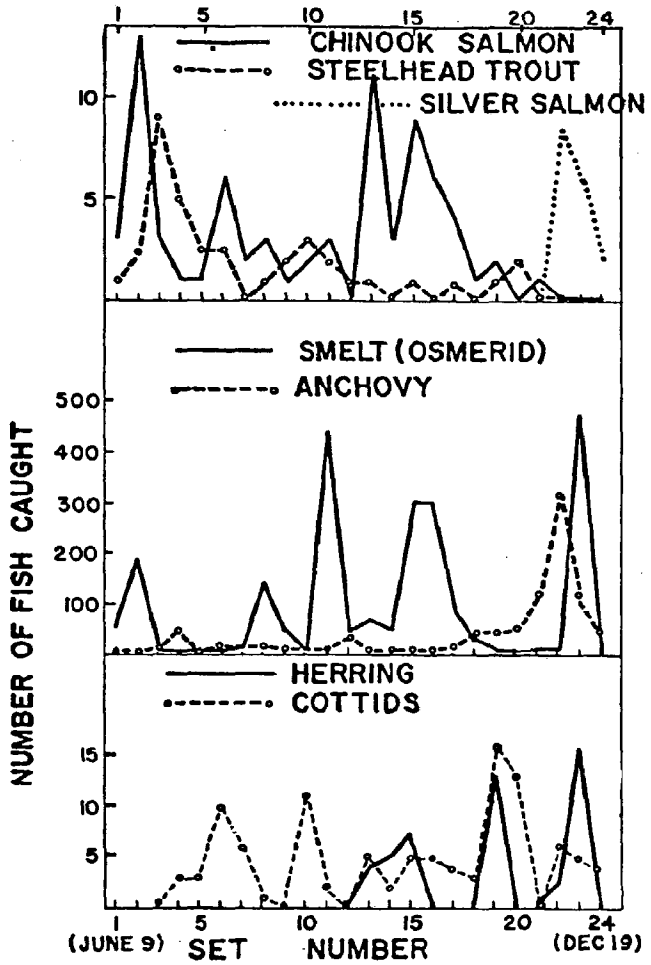
In the migrations flocking and schooling is the rule. In analogy with power transmission in pipelines and electrical power lines, the concentration of potential energy increases the efficiency of long distance tranmission, because the ratio of energies for transmission to energy losses from the channel are less.

Distribution of Fisheries

A very useful document for our attempts to generalize about dominant populations in relation to seasonal programs of energy is the recent set of maps and explanations by Heald (1968) prepared for E. I. DuPont de Nemours Co. For each zone of the Atlantic Coast commercial fisheries are located approximately with shading. The zones of the marine shelf are shown as well as those of the estuaries inshore coastal systems indicating areas where the stocks are located during their seasonal migrations that join the estuarine systems to the economy of the shelf.

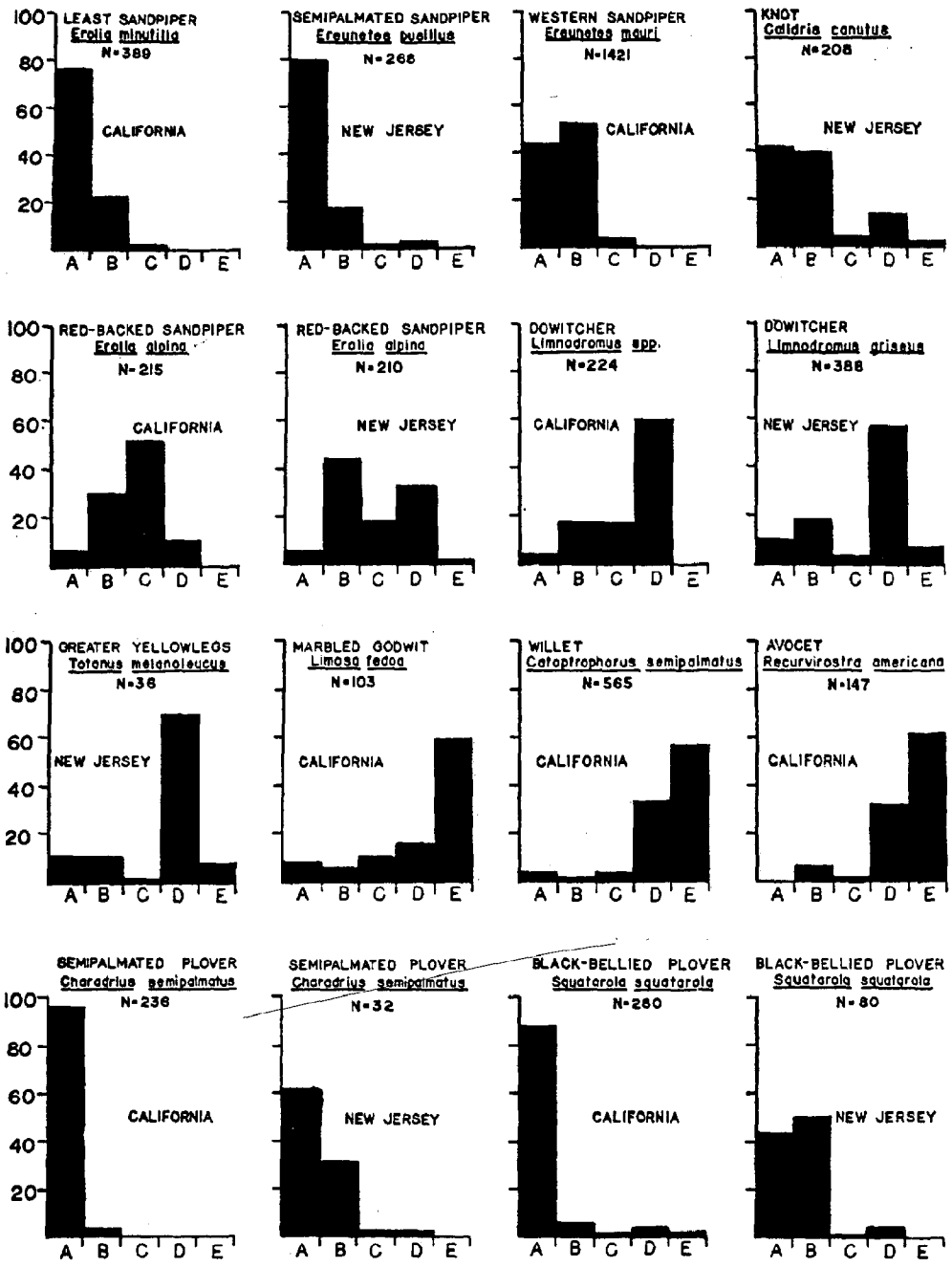


Map of Big Lagoon, Humboldt County, California, showing location of 24 gillnet sets, June-December, 1958.



Catch of seven species of fish in 24 gillnet sets, Big Lagoon, Humboldt County, California, June 19 - December 19, 1958 (from Salo, Hayes, and McGie, 1959).

Fig. 28. Migratory fish in a California estuary (Allen, Delacy, and Gotshall, 1960).



Distribution (% occurrence) of migrant shorebird species on the Palo Alto census area and the New Jersey shore within a series of imaginary zones roughly parallel to the water's edge at low tide. The water's edge is represented by C. Above the water's edge, zone A = areas lacking a surface film of water; and zone B = areas retaining a surface film of water on the substrate. Beyond the water's edge, zone D = the area between the water's edge and a contour line 1 ft beyond and parallel to the water's edge; and zone E = the area beyond the 1-ft contour line.

Fig. 29. Habitat utilization by migratory sandpiper birds feeding on beaches(Recher, 1966).

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