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Guidelines for Beach Restoration Projects

Part I Biological

By Walter G. Nelson

Florida Sea Grant College July 1985

PHYSICAL AND BIOLOGICAL GUIDELINES FOR BEACH RESTORATION PROJECTS

PART I.

BIOLOGICAL GUIDELINES

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GENERAL DESCRIPTION OF THE SAND BEACH FAUNA

Reidl & McMahan (1974) have divided the beach fauna into three groups with names based on the Greek word for sand, psammon. These categories are the epipsammon or epifauna composed of the fishes and birds found on top of the beach, the endopsammon or endofauna composed of the larger animals burrowed in the sand, and the mesopsammon or interstitial fauna composed of microscopic animals living in the interstices between sand grains.

In a classic paper on sandy beach ecology, Erik Dahl (1952) pointed out that there are striking similarities in the types of organisms, particularly crustaceans, inhabiting sandy beaches in various parts of the world. In order to facilitate comparison among beach faunas, Dahl proposed three vertical zones in relation to tide levels across the beach: the subterrestrial fringe, the midlittoral zone and the sublittoral fringe. The recent review by Naqvi and Pullen (1982) divides the beach system into the beach zone (subdivided into upper and lower zones), the surf zone and the nearshore zone. Various other authors have largely followed the general division of the beach into three zones, although terminology has varied (Fig. 1). McLachlan (1983) describes several of the additional classifications proposed and the applicability of such classifications in general to sand beaches.

For the crustaceans, Dahl (1952) suggested that a typical faunal composition would be found within each beach zone in specific geographic areas. The subterrestrial fringe (upper beach zone) he suggests is dominated by talitrid amphipods in temperate areas and by crabs of the genus Ocypode (ghost crabs) in more tropical areas. Dahl (1952) suggested that these two groups were mutually exclusive, but subsequent work has found that they often appear on beaches together (Gauld & Buchanan, 1956; Trevallion et al., 1970). The midlittoral zone (beach face portion of the foreshore) he suggests will be dominated at all latitudes by cirolanid isopods. In the sublittoral fringe (swash zone and surf zone), diversity increases. Dahl suggests that various amphipod groups are dominant in cold or cool temperate areas while in warmer waters they are replaced by mole crabs of the family Hippidae. Shelton & Robertson (1981) have suggested that haustoriid amphipods, which are often dominant on upper and mid beach levels in temperate areas, are replaced by cirolanid isopods on tropical beaches.

Trevallion et al. (1970) have compared the faunal composition of exposed portions of tropical and sub-tropical beaches worldwide while Sheldon and Robertson (1981) have compared temperate and tropical exposed sandy beaches of the Atlantic and Gulf of Mexico. Both reports indicate a high relative similarity among varied geographic locations. In the upper beach zones, in addition to the talitrid amphipods and ocypodid crabs mentioned by Dahl (1952), both haustoriid amphipods and isopods are often found. In the midlittoral zone, polychaetes, isopods and haustoriid amphipods become dominant forms. In the swash zone, the beach fauna is typicaly dominated by coquina clams of the genus Donax, mole crabs (Emerita), and a few species of polychaete worms. The shallow sublittoral zone is typically a region of increased species diversity (Fig. 2). Dominant groups include gastropods (Oliva, Terebra), sand-dollars (Mellita),

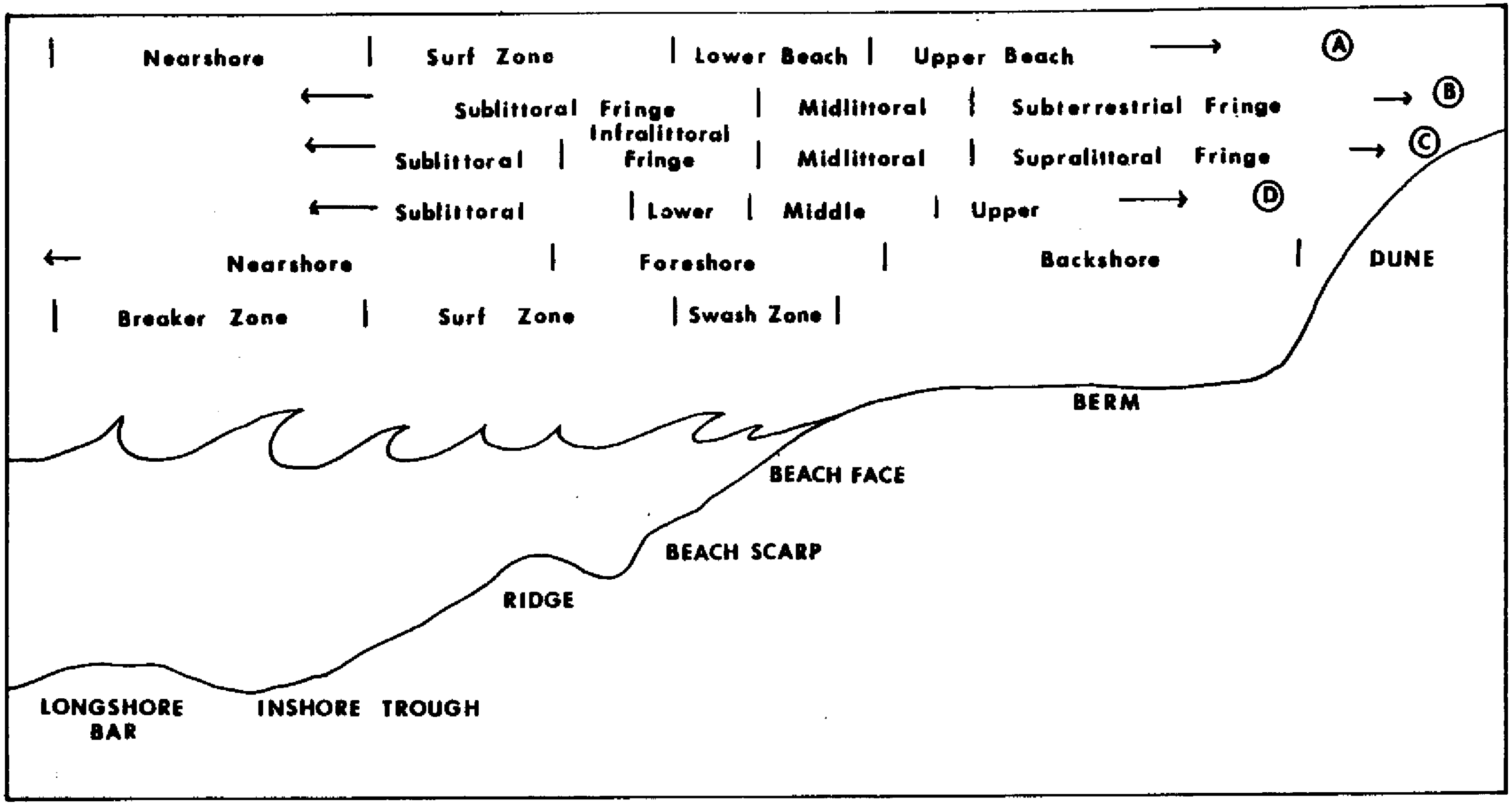

Figure 1. Diagram of vertical zones and major features of the beach and surf zone habitat. Zonation schemes are as follows: A - Naqvi & Pullen, 1982; B - Dahl, 1952; C - Pinchon, 1967; D - Trevallion et al., 1970.

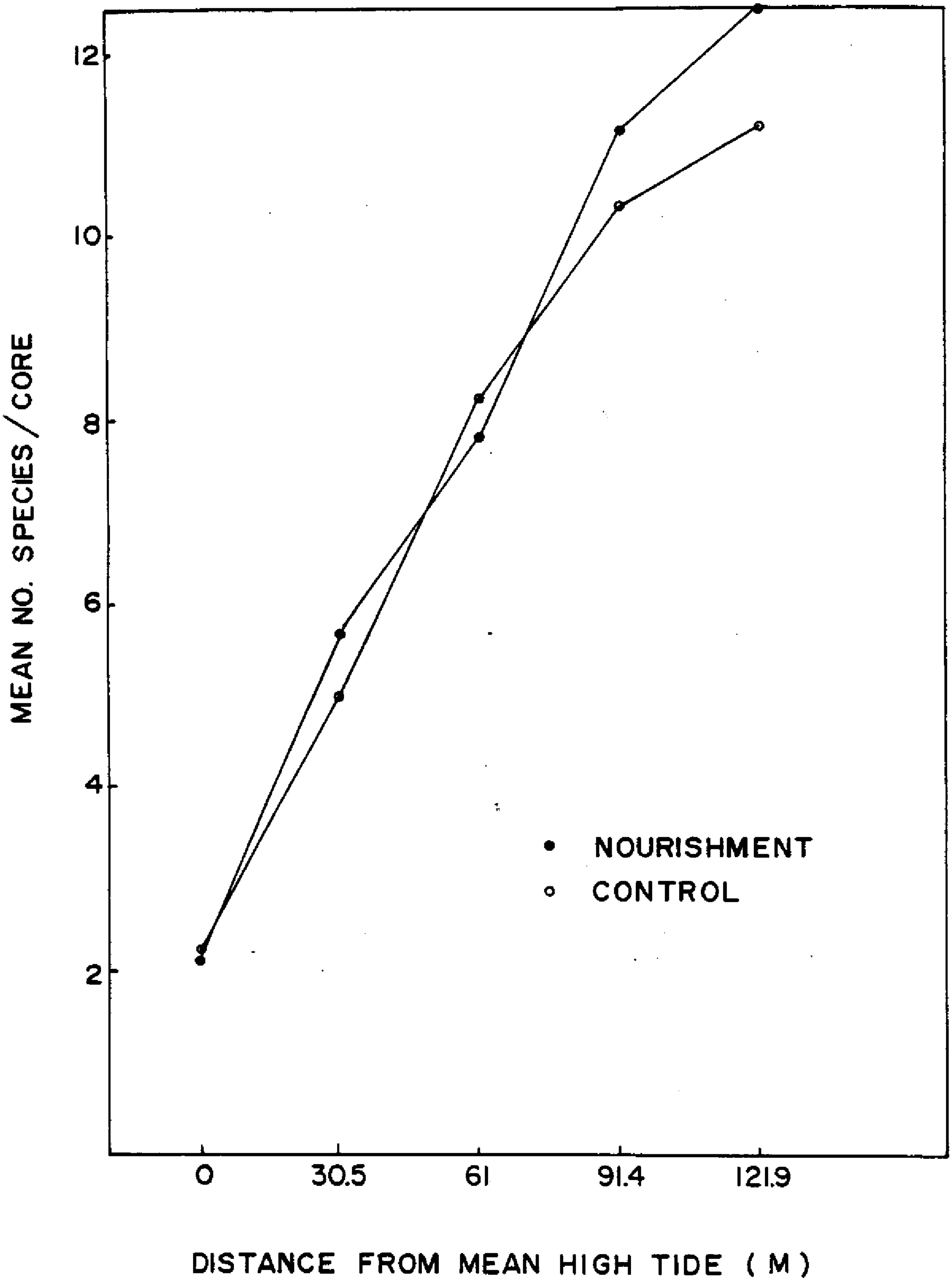

Figure 2. Comparison between control and beach nourishment transects for number of species versus distance offshore. Data are from the Melbourne - Indialantic beach nourishment project. (After Gorzelany, 1983).

portunid crabs (Arenaeus, Callinectes, Ovalipes) and burrowing shrimps (Callianassa). Although the sublittoral zone was not sampled in most of the studies compared by Trevallion et al.(1970) and Shelton & Robertson (1981), other studies indicate a general pattern of increased species diversity in the nearshore zone (Spring, 1981; Gorzelany, 1983). Typically the nearshore region is dominated by polychaetes, haustoriid and other amphipod groups, and bivalves such as Donax and Tellina (Keith & Hulings, 1965, Saloman, 1976; Matta, 1977; Spring, 1981; Gorzelany, 1983; Knott et al., 1983). With respect to the interstitial fauna, Reidl & McMahan (1974) indicate that this group is generally lacking from the dry surface sand of the upper beach but becomes more abundant on the beach face, swash zone and nearshore zone.

Dexter (1972), Croker (1977) and Shelton & Robertson (1981) have all suggested that there is no latitudinal pattern of diversity of intertidal sand beach macrofauna. Much of the tropical work used in these reports has been carried out by Dexter (1972,1974,1976). However, Kaufman (1976) studied a beach in Panama previously studied by Dexter (1972), but, unlike Dexter, sampled while the beach was covered by the tide rather than when the sand was dry. He recorded 197 species as compared to the 41 reported by Dexter (1972), a result which suggests tropical beaches, in contrast to current opinion, may in fact be more species rich than temperate ones when sampled in an appropriate fashion.

Degree of wave exposure appears to be a major factor controlling the diversity of sand beach communities. Comparison of faunal composition on beaches of differing exposure have been reported by Gauld & Buchanan (1956), Dexter (1967, 1976), Morgans (1967), McIntyre (1968, 1970)), Trevallion et al., 1970, Eleftheriou & Nicholson (1975), Croker (1977) and Shelton & Robertson (1981). Typically, more sheltered beaches are often higher in species diversity and density of individuals (Fig. 3). This pattern may be partially explained by the fact that coarser sand grain size and steeper slope are often associated with higher wave energy beaches (Dexter, 1976). Therefore, the sand may drain more rapidly with a resulting increase in dessication stress for beach organisms (Gauld & Buchanan, 1956). In line with this idea, McLachlan (1983) states that it is not wave action but steep beach slopes and coarse sand which may limit beach faunas. Indeed, McLachlan (1983) finds that very exposed beaches with heavy wave action can support more diverse faunas than some less exposed beaches with coarse grain size and steep slope. The key factor is whether the individual beach is reflective or dissipative for wave energy. A flat sloping beach evenly dissipates wave energy in the surf zone and intertidal. High energy dissipative beaches may be optimum habitats for filter feeders, and McLachlan (1983) reports the highest recorded macrofaunal biomass values are from very exposed dissipative beaches.

Table I compares the numerically dominant species characteristic of beaches from both the western Atlantic and Gulf of Mexico. With the exception of New England beaches, the species composition of the sandy beach intertidal is broadly similar. The New England intertidal tends to be dominated by haustoriid amphipods and the isopod Chiridotea caeca. Southern beaches in exposed situations tend to show a lower representation of haustoriids in the intertidal zone.

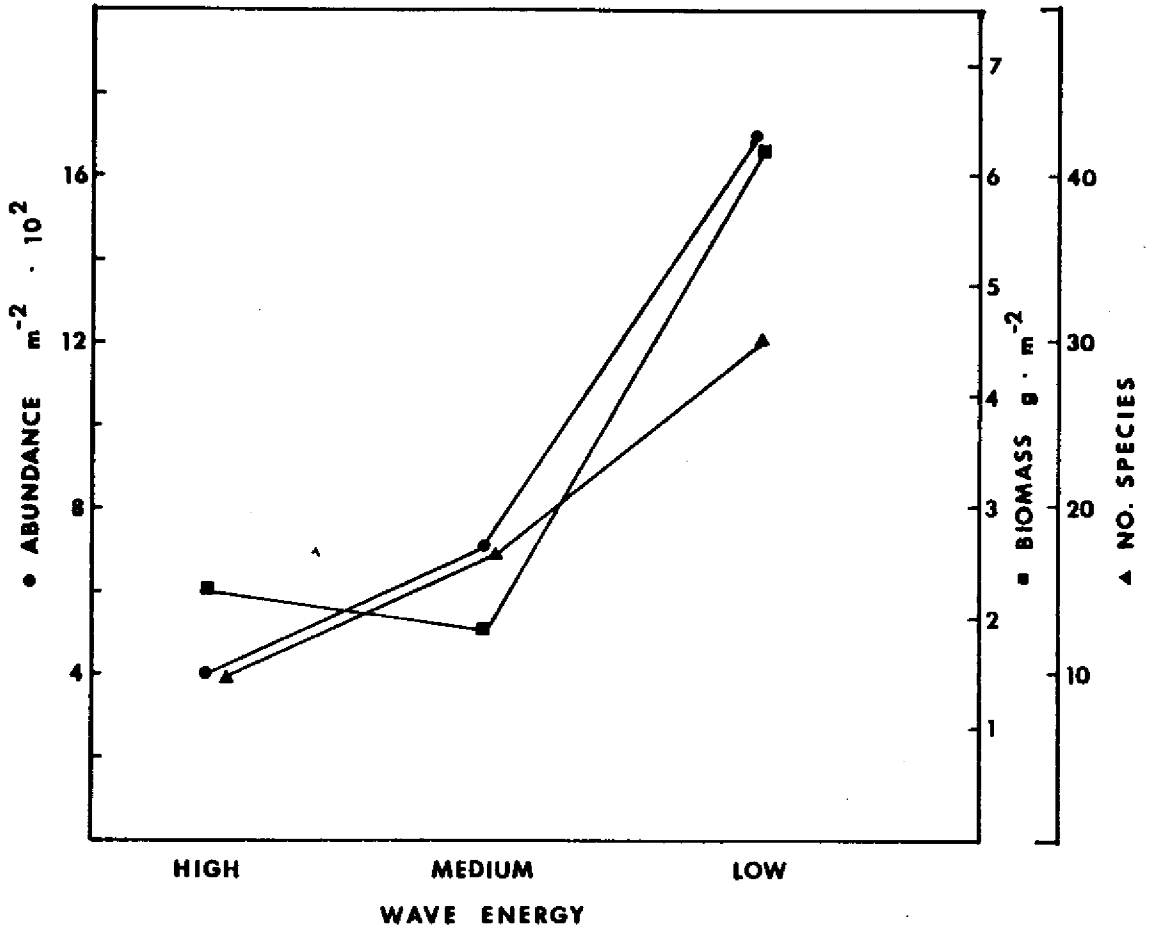

Figure 3. Comparison of mean abundance, number of species and biomass of macrofauna found on beaches of low, moderate and high wave energy. Data from McLachlan, 1983.

Table I. Comparison of the dominant species of exposed sandy beaches of the U. S. Atlantic and Gulf coasts.

	New England (Croker et al.,1975; Croker,1977)	New Jersey (McDermott,1983)	North Carolina (Matta,1977)	North Carolina (Leber,1982a)
Intertidal				
	Acanthohaustorius millsi	A. millsi		
	Amphiporeia virginiana	A. virginiana	A. virginiana	A. virginiana
	Haustorius canadensis	H. canadensis	Emerita talpoida	E. talpoida
	Chiridotea caeca			
	Scololepis squamata	S. squamata	S. squamata	
		Donax variabilis	D. variabilis	D. variabilis
				Donax parvula
				Ocypode quadrata
Subtidal				
	A. millsi	A. millsi		Ovalipes ocellatus
	A. virginiana			Arenaeus cribarius
	Bathyporeia quoddyensis		B. quoddyensis	
	Psammonyx nobilis			
	Manconcuma stillifera			
	S. squamata	S. squamata	S. squamata	
			Parahaustorius longimerus	
			D. variabilis	

Table I. Continued.

	NW Florida (Salomon, 1976)	Texas (Shelton & Robertson, 1981;Keith & Hulings, 1965 Hill & Hunter, 1976)
Intertidal		
	Haustorius n. sp.	*Lepidactylus* sp.
	E. talpoida	*Emerita benedicti*
	S. squamata *Donax texianus*	*S. squamata* *D. texianus* *Donax dorotheae* *Donax roemeri*
Subtidal		
	Acanthohaustorius n. sp. *Protohaustorius* n. sp. *Pseudohaustorius* n. sp.	*Acanthohaustorius* sp. *Protohaustorius bousfieldi* *Parahaustorius obliquus*
	Mancocuma sp. *S. squamata*	
	D. texasianus	*D. variabilis*

Table I. Continued.

	South Carolina (Knott et al.,1983)	Georgia (Dorjes,1977)	NE Florida (Gorzelany.1983)	SE Florida (Marsh et al.,1980)
Intertidal				
	Neohaustorius schmitzi	*Haustorius* sp.		*Eurydice littoralis*
		Nerida agilis		
		E. talpoida	*E. talpoida*	*E. talpoida*
	Spiophanes bombyx	*Callianassa major*		
				Scololepedes viridis
	S. squamata			
	D. variabilis	*D. variabilis*	*D. variabilis*	
		Bathyporeia sp.	*D. parvula*	
		O. quadrata		
Subtidal				Oligochaeta spp.
	A. millsi	*Acanthohaustorius* sp.		*Exogone dispar*
	Protohaustorius deichmannae	*P. longimerus*	*P. longimerus*	
	S. bombyx	*Bathyporeia* sp.	*Bathyporeia parkeri*	Polychaeta spp.
			Paraonis fulgens	
			Haploscoloplos fragilis	
	S. squamata			
	Tellina sp.			
	D. variabilis		*D. variabilis*	
			D. parvula	

The ghost crab Ocypode quadrata is typically found on southern beaches on the upper beach, while the swash zone is dominated by the mole crab (Emerita talpoida) and various species of coquina clams (Donax spp.). The polychaete Scolelepis squamata is the most widely represented of the intertidal fauna, being found from New England to Texas. The subtidal region shows more variability in species composition from location to location (Table I), although typically, haustoriid amphipods and polychaetes are the dominant forms. The polychaete Scolelepis squamata is one species which is broadly distributed in the subtidal as well as the intertidal. In some cases, large numbers of coquina clams are found in the subtidal zone as well as the intertidal swash zone.

THE BIOLOGY OF SELECTED SAND BEACH ORGANISMS

Emerita talpoida (mole crabs)

General Review

The mole crab Emerita talpoida is frequently the most conspicuous and abundant macrofaunal species inhabiting the swash zone on high energy sand beaches along the United States east coast. E. talpoida is a filter feeder specialized for burrowing in the wet sand of the lower foreshore. Unlike Emerita analoga, its west coast analog, E. talpoida requires active wave movement for feeding and does not survive well in quiet water (Pearse et al., 1942). Thoroughly wet sand is also a requirement in that E. talpoida is unable to burrow in either dry or merely moist sand.

Mole crabs are typically observed on the beach distributed in dense aggregations (Pearse et al., 1942; Efford, 1965; Cubit, 1969; Diaz, 1980; Bowman, 1981), which move up and down the beach with the tide. In addition to intertidal migration, Dillery & Knapp (1970) found evidence of longshore migrations in the west coast mole crab E. analoga. Several authors have associated the presence of aggregations with beach cusps (Cubit, 1969; Diaz, 1980; Bowman, 1981). The convergent flow of water in the troughs of beach cusps is the presumed mechanism of formation of aggregations..

A small body of literature is available on various aspects of the biology and ecology of E. talpoida. These aspects include studies of enzyme polymorphism (Corbin, 1977), taxonomy (Snodgrass, 1952), physiology (Edwards & Irving, 1943; Bursey & Bonner, 1977; Schatzlein & Costlow, 1978), larval development (Rees, 1968) and chromatophoric pattern (Shield, 1973). The population dynamics and life history of E. talpoida for the North Carolina region are described in detail by Pearse et al. (1942) and Diaz (1974, 1980). Bowman (1981) has examined in detail the relationship between the spatial and temporal distribution of E. talpoida and physical beach characteristics.

Additional information on mole crabs is available through studies of the west coast species E. analoga. Seasonality has been described by Barnes & Wenner (1968) and Cox & Dudley (1968). Other subjects covered include behavior and aggregation formation (Efford, 1965; Cubit, 1969), mating (MacGinitie, 1938), functional morphology (Knox & Boolotian, 1963), sex reversal (Wenner, 1972), longshore movement (Dillery & Knapp, 1970) and population dynamics (Dudley, 1967). Efford has produced a series of papers which describe feeding (1966), neotony (1967), egg size (1969) and recruitment (1970).

Seasonality and Life Cycle

Natural populations of E. talpoida typically display great variability in seasonal abundance over relatively short time spans. Reilly & Bellis (1978, 1983) report a density shift for this species from approximately 40 m^{-2} to 2920 m^{-2} in a period of little more than a month. A portion of this variability may be spatial patchiness (see below), but much of it relates to periods of strong recruitment to populations. Table II summarizes data comparing the time period and densities for population maxima for a variety of studies of E. talpoida in North Carolina and Florida. Peak densities were found at various times between June and November, although peaks in July and September - October were most common. Density estimates of peak abundance ranged from 700 to 3,750 m^{-2} in North Carolina and 332 to 3067 m^{-2} in Florida. Periods of minimum abundance are typically during February - April in North Carolina and December - February in Florida with minimum densities falling in the range 0 - 504 m^{-2}.

Seasonality of abundance is closely related to the life cycle of E. talpoida. The reproductive period for this species, as defined by the presence of egg-bearing females in the population, is generally May - October although Diaz (1980) reports small numbers of ovigerous females as also being present in January through April in North Carolina. Recruitment of new individuals, either megalopae or small individuals, falls primarily in the period of April - November (Table II) with major peaks in recruitment occurring in June and September.

Diaz (1980) suggests the following life cycle for North Carolina mole crab populations. Females which recruited in September overwinter and produce eggs which hatch the following May (Fig. 4). The May hatch has a planktonic larval stage of approximately 30 days duration, leading to a June recruitment of new individuals on the beach. Females which recruit in June overwinter, producing eggs which are hatched in July. The July hatch has a larval stage of approximately 45 days leading to a September recruitment which closes the cycle (Fig. 4). A portion of the September recruits may overwinter for a second year, again producing eggs for the May hatch. Diaz (1980) estimated that life span was 15 - 18 months for June recruits and 19 - 22 months for September recruits.

Spatial Distribution

Because of the tendency of E. talpoida to be found in aggregations, this species is highly variable in its spatial distribution. Density variation betweeen two transects only 300 m apart may be 800% or even greater (0 vs. 666 m^{-2}, in Matta, 1977). A variety of studies have clearly shown that the bulk of the E. talpoida population is restricted to the swash zone (Table III), although occasional individuals are recorded up to several hundred meters from shore.

Bowman's (1981) study of the spatial distribution of E. talpoida on a North Carolina beach reached several conclusions. This mole crab was found to be typically most abundant in the lower

Table II. Seasonality of occurrence and reproduction of *Emerita talpoida*.

Location	Peak Abundance	Maximum Density m^{-2}	Minimum Abundance	Minimum Density m^{-2}	Reproductive Period	Recruitment Period	Author
Bogue Bank, NC	Sep-Oct	700	Feb	40	Jan-Sep	May-Nov	Diaz (1980)
Bogue Bank, NC	Jun,Jul,Sep	2920	Feb-Mar	0			Reilly & Bellis (1978,1983)
Bogue Bank, NC					Jun-Sep	Aug	Pearse et al. (1942)
Duck, NC	Jul	3239	Mar-Apr	0-43		Apr-May	Matta (1977)
Duck, NC	Nov	3750				Jun	Bowman (1981)
Panama City,FL	Aug-Oct	322	Nov-Jan	32	May-Oct		Saloman & Naughton (1978)
Panama City,FL		540					Saloman (1976)
Melbourne, FL		3067	Dec.	504			Spring (1981)
Melbourne, Fl	Oct.	2226	Feb.	0			Nelson & Gorzelany (1983)

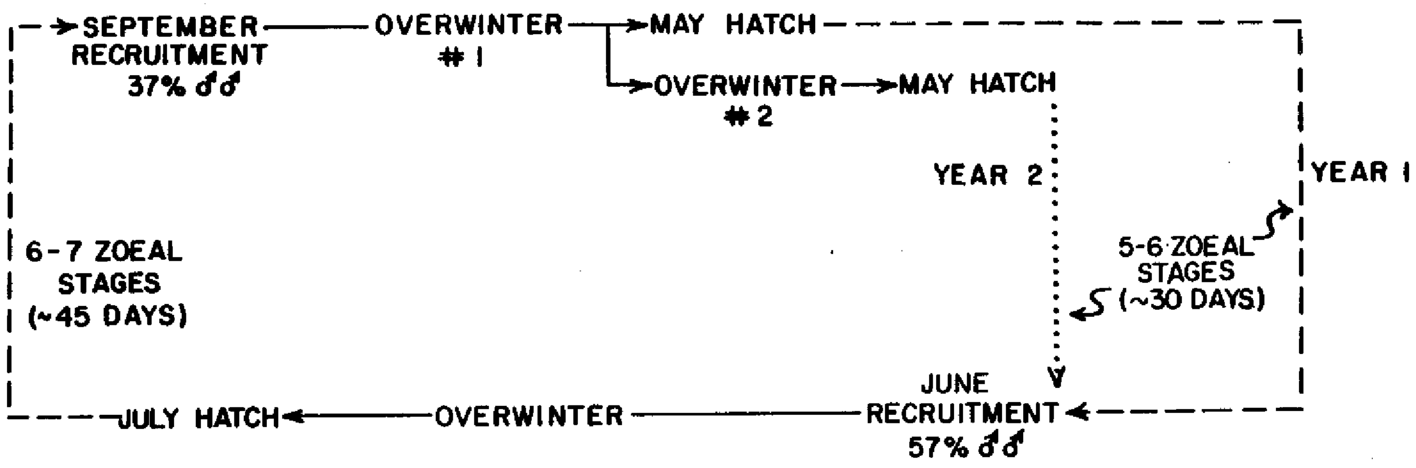

Figure 4. Life cycle of *Emerita talpoida* from North Carolina as described by Diaz (1980).

Table III. Examples of the spatial distribution pattern for the numerical abundance of *Emerita talpoida*.

A) Panama City Beach, Florida

	Salomon (1976)	Culter & Mahadevan (1982)	
Sample Date	1974	1979	1980
Location (Depth m)	Total no.	Total no.	Total no.
Swash zone (0)	1555	39	47
First sandbar (1)	25	2	5
Between sandbars (2.5)	2	0	0
Second sandbar (2)	8	2	0
Beyond second bar (3)	2	0	0

B) Duck, North Carolina (Matta, 1977)

Distance from top of swash zone (m)		Date		
	Oct-Nov	Mar-Apr	Jun	Jul
3.3	541	688	58	2014
7.6	415	435	565	1210
10.6	222	123	492	500
13.6	29	-	-	-
15.2	48	29	22	36
22.7-60.8	0	0	0	0

half of the foreshore in the active swash zone. There apeared to be no selection for region of the beach with a particular degree of slope. Some evidence was found for a preference for sand with a narrow grain size spectrum of 0.5 - 0.6 mm, although co-variables such as spatial location were not factored out. Evidence was found that densities tended to be higher in areas of lower wave energy, particularly troughs of beach cusps.

For the region of Woods Hole, Massachusetts, Edwards & Irving (1943) have stated that adults overwinter offshore in water from 2-4 m in depth. Matta (1977) found no *Emerita* in the periods October - November or March - April at stations of 2 - 3 m depth in North Carolina. No *Emerita* were recorded by Salomon from stations in 10 m depth off the beach at Panama City, Florida, nor were any recorded by Spring (1981) at stations sampled in winter at 2 - 3 m depth off Melbourne, Florida. Nelson & Gorzelany (1983) also recorded only occasional specimens at 2 - 3 m depth off the beach at Melbourne, Florida, at any season. Further extensive sampling of the beaches in the Melbourne vicinity (Nelson, unpub. data) has revealed no evidence of overwintering adults in the nearshore subtidal zone. Whether *E. talpoida* overwinters offshore at southern localities, therefore, *is strongly* doubtful and overwinterig females most probably remain in the swash zone.

Effects of Beach Nourishment

Two studies (Hayden & Dolan, 1974; Reilly & Bellis, 1978, 1983) have examined the impact of beach nourishment on *E. talpoida*, both for projects carried out in North Carolina. Hayden & Dolan (1974) describe the effects of large scale sand deposition on *E. talpoida* populations near Cape Hatteras. This study found evidence for reduced numbers of mole crabs immediately downstream (0 - 66 m) from the discharge site on the beach. However, no evidence of mortality of *E. talpoida* was found since no dead *Emerita* were observed, and 3 m deep cores indicated the mole crabs had not burrowed deeply in these areas. Hayden & Dolan (1974) suggest that *E. talpoida* was not killed by the nourishment but instead moved away from discharge areas into the surf zone where they then can move along the beach, becoming concentrated in shoreline meanders (cusps) on a scale of 180 - 300 m from the discharge site. Hayden & Dolan (1974) concluded that impacted areas recovered in from 2 days to 2 weeks. The Cape Hatteras project added 955,693 m^3 of sand to the Beach (Hayden & Dolan, 1974). The nourishment sand was very similar to the natural beach sand in grain size, and contained relatively little hydrogen sulfide or organic material (Hayden & Dolan, 1974).

Reilly & Bellis (1978, 1983) examined the effects of nourishment on the beach fauna at Fort Macon, North Carolina. The sediment volume was similar to the Cape Hatteras project (904,172 m^3), but the sediment was taken from dredged harbor sediments which had a larger mean grain size, were black in color indicating a reducing environment, contained a great deal of coarse shell hash, frequently included clay materials which formed erosion resistant balls on the beach, and often smelled of hydrogen sulfide. At the Fort Macon site, beach nourishment began in December and caused *E. talpoida* populations to immediately drop to 0 in the region of the project.

The nourished beach showed rapid recolonization several months later during the spring recruitment period. However, the nourished beach showed a one month delay in recruitment compared to the control beach. This was due to the fact that no recruitment of overwintering adults was observed on the nourished beach, whereas these were the first individuals appearing at the control site. Young of the year from pelagic stocks were the only recruits at the nourished beach (Reilly & Bellis, 1978, 1983).

By the summer after nourishment, densities were the same for nourished and control beaches. However, size class composition was radically different. At the control site, an assortment of size classes were present. At the nourishment site, only juvenile mole crabs were present, which resulted in a greatly decreased biomass of *E. talpoida* present as compared to the control site. Reilly & Bellis (1978, 1983) conclude that the failure of the adult mole crabs to recruit was due to their being killed in near-shore overwintering areas due to increased turbidity. Given the description of the nourishment sediments, an alternate mortality source may have been the liberation of hydrogen sulfide into the near shore waters.

Donax spp. (coquina clams)

General Review

Bivalves of the genus *Donax* are often an abundant component of the intertidal fauna on sand beaches in many parts of the world (Coe, 1953, 1955; Loesch, 1957; Wade, 1967a,b,c; Ansell et al., 1972; Ansell & Trueman, 1973; McLusky et al., 1975; McLachlan et al., 1979). The biology of this genus has been recently reviewed by Ansell (1983). Along the Florida east coast, two species are found, *Donax variabilis* and *Donax parvula*, the separation of which has caused considerable confusion in the past (Morrison, 1971). Abbott (1974) considers *D. parvula* to be merely an ecomorph of *D. variabilis*, and ecological information from Melbourne, Florida (Nelson, unpub. data) suggests that this may be correct. Shell morphology of *Donax* species, including *D. variabilis* (Chanley, 1969b), is known to be rather variable (Wade, 1967c). On the Florida panhandle, these two species are replaced by *D. texasianus*.

Considerably more information is available for *D. variabilis* than for *D. parvula*. A number of studies have examined aspects of burrowing and migratory behavior (Pearse et al., 1942; Turner & Belding, 1957; Edgren, 1959; Tiffany, 1971; Mikkelson, 1978, 1981). Most populations of *D. variabilis* are known to migrate up and down the beach with the tides, the migratory behavior being stimulated by the accoustic shock of breaking waves. However, several instances of non-migratory populations have beeen reported (Edgren, 1959; Mikkelson, 1978, 1981). Mikkelson (1981) suggests non-migratory behavior is a local adaptation to deal with a combination of low beach slope and wave energy in an area of irregular semi-diurnal tides and low sand permeability.

Other papers have examined larval development (Chanley, 1969a) and general ecology (Edgren, 1959). The question of what generates the remarkable rainbow of color morphs in *D. variabilis* has been addressed by Mikkelson (1978) and Schneider (1982). The most detailed study of Florida *D. variabilis* populations with respect to intertidal distribution, growth rates and shell color variability is that of Mikkelson (1978).

Almost all available information on *D. parvula* is contained in the taxonomic revision of Morrison (1971). Additional information on abundance, seasonality and distribution is available from a number of relatively recent field studies which have differentiated between *D. parvula* and *D. variabilis* (Reilly & Bellis, 1978, 1983; Spring, 1981; Leber, 1982b).

Seasonality and life cycle

Donax variabilis is distributed from Virginia Beach, Virginia, to the coast of Mississippi (Morrison, 1971). Chanley (1969b) suggests that this species may exhibit summer range extensions as far as Long Island. Morrison (1971) suggests that *D. variabilis* has a two year life span, and in some cases, individuals may survive to a third year. Mikkelson (1978) has provided one of the few measures of growth rate for *D. variabilis* which has not been confused by taking measurements of mixed populations including the *D. parvula* form. He estimates Florida *D. variabilis* grow at a rate of 3-3.7 mm per month in the summer months. Mikkelson (1978) suggests this species spawns in February with a three week larval period resulting in a March settlement. He also suggests Florida *D. variabilis* has a second spawning in June. Leber (1982b) records juvenile recruitment to a North Carolina population as occuring in February and November. The February settlement is indicative of a winter spawn. He suggests 2 year old individuals move into the intertidal swash zone in March after overwintering in the shallow offshore zone. Matta (1977) observed a June settlement of spat of *D. variabilis* in Duck, North Carolina.

Donax parvula is found from Ocracoke, North Carolina to St. Lucie Co., Florida, and is reported to have a two year life span (Morison, 1971). Distinguishing between *D. parvula* and *D. variabilis* is extremely difficult in size classes below approximately 5 mm and has not generally been done. Morrison (1971) suggests that *D. parvula* may spawn somewhat later than *D. variabilis*. However, Leber (1982b) indicates recruitment in a North Carolina population occurs in February as it did for *D. variabilis*. Reilly & Bellis (1978, 1983) observed young of the year for both *Donax* species recruiting during March in North Carolina as well.

Seasonal abundance data (Table IV) indicates that both *D. variabilis* and *D. parvula* tend to achieve maximum densities during the summer period of June and July. Recorded maximum densities for *D. variabilis* range from 166 - 13,114 m^{-2} (Table IV), with a single core yielding an estimate as high as 15,619 m^{-2} (Edgren, 1959). Estimates of *D. parvula* range from 401 - 1425 m^{-2}. The maximum estimated abundance of *Donax texasianus* from Panama City, Florida, was 2050 m^{-2}, with peak periods of abundance occurring in February and May. This later pattern of seasonality appears to differ from that of *D. variabilis* and *D. parvula*.

Table IV. Maximum densities observed for intertidal populations of various Donax species.

Location	Maximum No. per m^2	Period of Max. Abundance	Author
Donax variabilis			
Bogue Banks, NC	1328	June	Reilly & Bellis (1978,1983)
Duck, NC	6326	June	Matta (1977)
Indialantic, FL	166	June[1]	Mikkelson (1981)
Indialantic, FL	791	Sept[2]	Spring (1981)
Indialantic, FL	338	June[2]	Gorzelany (1983)
Indialantic, FL	2001	Feb.	Nelson (unpublished data)
Clearwater, FL	15619[3]	Nov.[4]	Edgren (1959)
Sanibel Island, FL	13144	July	Mikkelson (1981)
Donax parvula			
Bogue Banks, NC	1152	June	Reilly & Bellis (1978,1983)
Indialantic, FL	401	June[2]	Spring (1981)
Indialantic, FL	985	June[2]	Gorzelany (1983)
Indialantic, FL	1425	Feb.	Nelson (unpublished data)
Donax texasianus			
Panama City, FL	2050	Feb,May	Saloman & Naughton (1978)

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1 - Sampled from April - September only.
2 - Sampling was done quarterly only.
3 - Estimate based on a single core.
4 - Only month sampled.

Spatial distribution

Morrison (1971) has observed spatial separation of D. variabilis and D. parvula, with the former being found near the high tide zone and the latter tending to occur at the extreme low tide level and subtidally. Morrison (1971) describes D. variabilis as living intertidally throughout the year, with part of the population often becoming stranded in mid-tide areas during low tide. Leber (1982b) has described seasonal differences in the spatial pattern of these species. D. variabilis and D. parvula migrated together in the swash zone throughout July. In August, D. parvula disappeared from the intertidal zone. No evidence of mortality was found. Instead, D. parvula appeared to have migrated seaward to 1 m depth in the surf zone. D. variabilis ceased tidal migrations in August and remained high on the beach in damp sand. Both forms had disappeared from the intertidal by December. Spring (1981) noted that off Melbourne, Florida, D. variabilis could be found at stations offshore at distances up to 91 m from the high tide line. A more detailed breakdown of Spring's original data indicates that spatial separation of D. variabilis and D. parvula occurs and that, furthermore, the degree of separation changes seasonally (Table V). During fall and spring, D. variabilis is found inshore, and D. parvula is found offshore. During the summer, maxima of both species are found together subtidally at intermediate distances from shore (27-55m from high tide line). During winter, density maxima of both species are found offshore (55-91 m from high tide). G tests of independence (Sokal and Rohlf, 1981) confirm that these patterns are statistically significant ($p < 0.05$). Juveniles of the two forms (combined) tend to have a maximum density offshore at all seasons, with the maximum tending more seawards during the winter (Table V). Matta's (1977) work in North Carolina in an area believed to be north of the distributional limit of the D. parvula form found a Donax sp. (presumably D. variabilis) to be concentrated all year at 30-60 m from the high tide line. Juvenile Donax were observed to be most abundannt at stations farthest offshore.

Effects of Beach Nourishment

Published information on the effects of beach nourishment on Donax is limited. Reilly & Bellis (1978, 1983) report that following December nourishment of a North Carolina beach, no Donax were found until the end of July. The individuals found at this time were young of the year carried in by littoral drift, and not the result of planktonic settlement (Reilly & Bellis, 1978, 1983). Reilly & Bellis (1978, 1983) suggest that adults were killed in the offshore overwintering area by burial by sediment due to their limited mobility. Beach nourishment in this case also apparently prevented larval recruitment by pelagic larvae, although post-metamorphic individuals did drift in.

Table V. Spatial distribution patterns of numerical abundance of *Donax parvula* and *Donax variabilis* at Indialantic, Florida. Data are mean number per core for all three study sites of Spring (1981).

Season	Distance from the High Tide Line (m)					
	0	5	27	55	73	91
Donax parvula						
Summer	0.2	1.3	3.5	7.1	6.0	3.1
Fall	0.0	0.0	0.1	0.2	1.0	2.1
Winter	0.0	0.0	0.0	0.6	0.2	0.8
Spring	0.0	0.0	0.6	3.0	4.1	2.5
Composite	0.5	0.3	1.0	2.7	2.8	2.2
Donax variabilis						
Summer	1.5	2.1	11.3	13.8	4.0	0.2
Fall	14.0	1.3	1.6	2.0	1.5	1.8
Winter	0.1	0.9	0.0	2.2	1.1	1.2
Spring	9.2	8.4	1.0	0.9	0.4	0.6
Composite	6.2	3.2	3.5	4.7	1.8	0.9
Donax spp. juveniles						
Summer	3.6	7.3	48.5	50.6	1.5	25.6
Fall	0.0	0.0	25.6	90.7	48.0	35.1
Winter	0.0	0.0	0.8	15.4	8.9	25.6
Spring	0.0	0.0	11.1	31.7	19.5	2.0
Composite	0.9	1.8	21.5	47.1	19.4	22.0

Haustoriid Amphipods

General Review

Most studies of macroinvertebrate composition on intertidal sand beaches of the east coast have found haustoriid amphipods to constitute a major portion (50-90%) of the fauna (Holland & Polgar, 1967; Dexter, 1969; Croker, 1970,1977; Howard & Dorjes, 1972; Croker et al., 1975; Dorjes, 1977; Matta, 1977; Leber, 1982a). Haustoriids often contributed significantly to total biomass in the above studies as well.

The biology of sand beach amphipods on the east coast has received considerable study. Bousfield (1970) has reviewed the adaptations of burrowing amphipods to the sand environment. Niche diversity and spatial partitioning has been examined both on the southeast (Croker, 1967a,b; Dexter, 1967; Grant, 1981b) and northeast coasts (Croker & Hatfield, 1980). Other areas of study have included life cycles and behavioral ecology (Sameoto, 1969a, b, c), and seasonality and community composition (Croker et al., 1975; Holland & Polgar, 1976; Croker, 1977; Leber, 1982a). Grant has examined factors influencing the presence of haustoriids in reducing sediments (1981a), and has considered the effects of shorebird predation on these amphipods (1981c). He has additionally considered the processes of drift (1980) and sediment transport (1981d) and disturbance as they affect haustoriid populations. General summaries of reproductive characteristics among haustoriid amphipods may be found in Nelson (1980) and Van Dolah & Bird (1980). Feeding of east coast haustoriids has been examined by Croker (1967a).

A variety of papers have examined the biology of individual species in some detail. These include studies of *Neohaustorius schmitzi* (Croker, 1967b), *Amphiporeia virginiana* (Hager & Croker, 1969a,b), *Acanthohaustorius millsi* (Sameoto, 1969b), *Acanthohaustorius* sp. (Croker, 1967a; Dexter, 1967), *Parahaustorius longimerus* and *Lepidactylus dysticus* (Croker, 1967a; Sameoto, 1969b), and *Haustorius canadensis* (Donn & Croker, 1983).

Seasonality and Life Cycle

Available information on the seasonality of east coast haustoriids is presented in Table VI. Almost all species typically reach peak abundances during the late spring and early summer months. Minimum abundances generally occur from late fall through May. There is some tendency indicated for maximum densities to be somewhat greater at the northern locations as compared to haustoriid populations in Georgia and Florida. Reproductive periodicities are relatively variable, with at least a small percentage of reproductive females being present over much of the year for many species (Table VI). Maximum recruitment of juveniles, however, tends to occur during the summer months in most cases.

Life cycle characteristics of haustoriids amphipods have been reviewed in part by Bousfield (1970), Nelson (1980), and Van Dolah & Bird (1980). Haustoriids often have only 1 brood of eggs per female, although females of some species are believed to be multiple brooded

Table VI. Seasonality of occurrence and reproduction of sand beach amphipods.

Site Species	Peak Abundance	Maximum Density m^{-2}	Minimum Abundance	Minimum Density m^{-2}	Reproductive Period	Recruitment Period	Author
Maine							Hager & Croker (1974a,b)
Amphiporeia virginiana	June-Sept	125000	Mar-May	25	Jan-Sept	May-June	
New Hampshire							Croker et al. (1975)
Acanthohaustorius millsi	July	135	Mar	15			
Amphiporeia virginiana	July	4500	Mar	10			
Bathyporeia quoddyensis	July	92	Mar	3			
Haustorius canadensis	July	772	Mar	25			
Massachusetts							Sameoto (1969a,b)
Acanthohaustorius millsi	June-Aug	2475	Apr	15	May-Aug	June-Oct	
Haustorius canadensis	Aug	1035	Winter		May-Oct	June	
Neohaustorius biarticulatus	June-Aug	3045	Winter		May-Oct	June-Aug	
Parahaustorius deichmannae	June-Aug	2145	Feb	0	May-Aug	June-Oct	
Protohaustorius longimerus	July-Aug	630	Feb	0	May-Aug	July-Oct	
North Carolina							Dexter (1971)
Acanthohaustorius millsi					Jan-Dec		
Haustorius sp.					Feb-Apr		
Neohaustorius schmitzi					Feb-Oct	May-Aug	
North Carolina							Matta (1977)
Amphiporeia virginiana	July	347	Oct-June	0			

Table VI. Continued.

Site Species	Peak Abundance	Maximum Density m^{-2}	Minimum Abundance	Minimum Density m^{-2}	Reproductive Period	Recruitment Period	Author
Bathyporeia quoddyensis	June-July	760	Mar-Apr	20			
Parahaustorius longimerus	June	8196	Mar-Apr	20			
Georgia							Croker (1967)
Acanthohaustorius sp.	July-Aug	128	Nov	7	Nov-Apr		
Haustorius sp.	Nov	50	Mar	31	Feb-Apr		
Lepidactylus dysticus	Nov	58	Apr-May	19	Apr-July		
Neohaustorius schmitzi	Mar	443	Nov	185	Apr-July		
Parahaustorius longimerus	Apr-May	266	Nov	33	Oct-Mar		
Florida							Saloman & Naughton (1978), Saloman (1976)
Acanthohaustorius n. sp.	May	1798	Nov	798			
Haustorius n. sp.	Apr	728	Oct	0			
Protohaustorius n. sp.	May	764	Feb	125			
Florida							Spring (1981)
Bathyporeia parkeri	June	1789	Dec	0			
Parahaustorius longimerus	Apr	3212	Dec	56			
Protohaustorius longimerus	June	1610	Dec	0			

(Nelson, 1980). Females tend to mature at small body sizes and to produce small broods of relatively large eggs. Mean brood size is generally less than 10 eggs per female (Nelson, 1980). Van Dolah and Bird (1980) suggested that large egg size is a requirement for the survival of these infaunal species in a course-grained sand environment.

Haustoriids are generally annual type species with maximum life spans of approximately 12 months (Bousfield, 1970), although Sameoto (1969b) suggests Acanthohaustorius millsi may live as long as 17 months. Several studies have indicated that some haustoriids may have 2 generations per year. Dexter (1971) showed that Neohaustorius schmitzi in North Carolina possessed a winter generation which lives about 8 months and a summer generation which lives about 4 months. Females in the summer population were smaller in size and produced fewer eggs both absolutely and as a function of body size (Dexter, 1971). Hager & Croker (1979) describe a similar pattern for Amphiporeia virginiana in Maine. The winter generation was slower growing, lived longer and produced 36% more eggs per female than the rapid growing, short-lived summer population. Other species (Sameoto, 1969a,b; see Table VI) are believed to produce only one generation per year.

Spatial Distribution

Spatial distribution patterns of the haustoriids can be very complex. Dexter (1967) has shown that haustoriid species were broadly separated by wave energy conditions in North Carolina. Amphiporeia virginiana and Haustorius sp. were most abundant on exposed outer coast beaches. Croker et al. (1975) also report exposed locations in Maine as dominated by Amphiporeia virginiana. Muddy-sand beaches of a quiet estuarine sound area were dominated by Lepidactylus dysticus. Beaches of moderate wave exposure located in inlets showed highest desities of Neohaustorius schmitzi, Acanthohaustorius intermedius, A. millsi, Parahaustorius longimerus and Protohaustorius deichmanae.

Further separation arises along the tidal height gradient. Dexter (1971) reports N. schmitzi and Haustorius sp. as being strictly intertidal, while A. intermdius and P. deichmanae are mainly subtidal forms. A. millsi and L. dysticus occur both subtidally and intertidally. Sameoto (1969a,b) concurs with several of these distributions, and adds Parahaustorius longimerus as a mainly subtidal species. Dexter (1971) notes that A. virginiana migrates up and down the beach with the tide. Croker (1967) provides data in general agreement with these divisions, but found P. longimerus in the intertidal zone as well, as did Matta (1977). Transects taken by Matta (1977) from the intertidal out to several meters depth showed that maximum densities of P. longimerus and Bathyporeia quoddyensis occur farthest from the beach. A. virginiana, while found at subtidal sites, was maximally abundant in the swash zone. On the Florida east coast, Spring (1981) found haustoriids occurring almost exclusively in subtidal areas, as has other recent work in this area (Gorzelany, 1983; Allenbaugh, 1984). On the Florida panhandle, Salomon & Naughton (1978) found Acanthohaustorius n. sp., Protohaustorius n. sp., and Pseudohaustorius n. sp. occurring mainly offshore, and Haustorius

n. sp. mainly occurring inshore of the first sand bar.

Croker (1967a) has shown that males of Neohaustorius schmitzi tend to be located higher on the beach than the females. In this study of Georgia haustoriids, there was evidence that different species were found at different sediment depths as well (Croker, 1967). In a detailed study of spatial distribution of haustoriids on a Maine Beach, Croker & Hatfield (1980) have shown that several species partition the beach both horizontally across the beach and vertically within the sand, and demonstrate this partitioning is probably due to competitive interactions. Grant (1981b) has also shown that 2 species of haustoriids (Acanthohaustorius millsi and Pseudohaustorius caroliniensis) are vertically segregrated in an intertidal sand flat in a South Carolina estuary due to competitive interactions. In this case, the 2 species are segragated into the upper oxidized layer and the underlying reducing layer.

Effects of Beach Nourishment

Essentially the only information available on the effect of beach nourishment on haustoriids is that of Reilly & Bellis (1978, 1983). They found that the intertidal species Haustorius canadensis disappeared from the nourished beach following dumping of sand and had not returned to the beach some 3 months later. The strong negative effect on Haustorius is due to the fact that this species lives intertidally where nourishment effects are greatest. Haustoriids in general are not strong swimmers and probably have a limited ability to escape the sand dumping. The slow return of haustoriids would be expected since they brood their young, and brood sizes are relatively small. New recruitment must therefore come from juveniles or adults which migrate into the disturbed area.

Some additional infomation is provided by the field and laboratory work of Grant (1980, 1981d), who showed that small patches of azoic sediment are colonized within one day by adults of a haustoriid species living in surface sediment layers. A deeper dwelling species required a 3 week period, but Grant concludes that complete recovery of azoic patches could take place in 1 month. Laboratory flume studies showed that one mechanism of recruitment could be drift of haustoriids caused by current-induced displacement from surface sediment layers. Such a mechanism might result in more rapid recruitment to large disturbed areas generated by beach nourishment than recruitment of juveniles.

Surf Zone Fishes

General Review

The fishes of the surf zone habitat of exposed sand beaches have not been studied as extensively as those of estuarine habitats. Most available studies are either species listings or seasonal studies. For the United States east coast, surf zone fishes have been described from sites from Connecticut to Indian River, Florida (Pearse et al., 1942; Warfel & Merrriman, 1944; Merriman, 1947; Tagatz & Dudley, 1961; Schaefer, 1967; Dahlberg, 1972; Cupka, 1972; Anderson et al., 1977; Hillman et al., 1977; Applied Biol. Inc., 1981). On the Gulf coast, open coast fish communities have been described at sites from Tampa Bay, Florida to Mustang Island, Texas (Reid, 1955; Gunter, 1959; Springer & Woodburn, 1960; McFarland, 1963; Naughton & Salomon, 1978; Salomon & Naughton, 1979; Modde & Ross, 1981).

Studies of growth, trophic interactions and other basic ecological information are extremely limited. Modde (1980) has reported on the growth and residency of juvenile and surf zone fishes from the Gulf of Mexico. The ecology of the pompano (Trachinotus carolinus and the permit (Trachinotus falcatus) have been studied (Fields, 1962; Finucane, 1969; Bellinger & Avault, 1970). Armitage and Alevizon (1980) have described the diets of juvenile and adult T. carolinus from the east coast of Florida. Among the clupeid fishes, the scaled sardine (Harengula jaguana) has been discussed by Houde (1977) and Omer (1984) while Daly (1970) gives a systematic description of the anchovies of south Florida.

Species Composition

Modde & Ross (1981) point out that surf zone fish composition is typically dominated by relatively few species. However, total species lists of 50 to 70 are common for exposed beach surf zones (Table VII), and Dahlberg records 95 species from a Georgia beach. A typical example of extreme dominance is the community at Horn Island Mississippi studied by Modde & Ross (1981) where 80.2% of total numbers were dusky anchovy (Anchoa lyolepis) and scaled sardine (H. jaguana). Table VII compares the 2 most abundant species of surf zone habitats along the United States Atlantic and Gulf coasts. The table indicates that the major families represented are the Engraulidae (anchovies), Clupeidae (herrings), Carangidae (jacks), and Scianidae (kingfish, spot, croakers). The numerically dominant species were Anchoa mitchelli (bay anchovy), A. lyolepis, A. hepsetus (striped anchovy), H. jaguana, T. carolinus and Menticirrhus littoralis (gulf kingfish). In addition, along the Atlantic coast, the Atlantic silverside (Menidia menidia) is often dominant (Table VII).

The fish making up the inshore surf zone community tend to be either small species or juveniles (Modde, 1980). Modde & Ross (1981) found only 1.1% of the fish collected exceeded 50 mm standard length, and similar results have also been reported by Gunter (1958) and Naughton & Saloman (1978).

Seasonality

The surf zone habitat is believed to contain a few often abundant species which are classified as permanent residents (Modde, 1980) while the habitat is additionally utilized by large numbers of immature migrant fish as a temporary resource (Modde, 1980). Modde (1980) classified only 6 (including H. jaguana, T. carolinus, M. littoralis) of 76 species as permanent surf zone residents. Therefore, seasonal patterns of fish abundance are quite strong. Summer and early fall months possess both higher diversity and higher abundances of surf zone fish than the winter and early spring period (Gunter, 1958; Tagatz & Dudley, 1961; Springer & McErlean, 1962; Cupka, 1972; Hillman et al., 1977; Naughton & Saloman, 1978; Saloman &

Table VII. Comparison of the dominant surf zone fishes among locations on the U. S. Atlantic and Gulf coasts.

Location	Two Most Abundant Species	% Dominance		Total No. of Species	Reference
		Top 2 Species	Top 6 Species		
Atlantic Coast					
Millstone Pt. CO	*Menidia menidia* *Fundulus majalis*	72	94	35	Hillman et al. (1977)
Morris Cove CO	*Menidia menidia* *Brevortia tyranus*	-	-	32	Warfel & Merriman (1944)
Pine Orchard CO	*Menidia menidia* *Sphoeroides maculatus*	63	88	13	Merriman (1947)
Fire Island NY	*Sphoeroides maculatus* *Alosa aestivalis*	66	90	71	Schaefer (1967)
Beaufort NC	*Brevortia tyrannus* *Anchoa hepsetus*	77	86	40	Tagatz & Dudley (1961)
Folly Beach SC	*Menidia menidia* *Anchoa mitchilli*	36	94	41	Anderson et al. (1977)
"Statewide" SC	*Menidia menidia* *Anchoa mitchili*	77	92	39	Cupka (1972)
Sapelo Island GA	*Menidia menidia* *Fundulus majalis*	-	-	95	Dahlberg (1972)
Melbourne FL	*Harengula jaguana* *Anchoa lyolepis*	58	86	68	Peters (1984)
St. Lucie Co. FL	----	-	-	91	Gilmore et al. (1981)
St. Lucie Co. FL	*Anchoa* spp. *Harengula* spp	78	95	60	Applied Biol. (1981)

Table VII. Continued

Location	Two Most Abundant Species	% Dominance Top 2 Species	% Dominance Top 6 Species	Total No. of Species	Reference
Gulf Coast					
Pass-A-Grille FL	*Lagodon rhomboides* *Menticirrhus littoralis*	-	-	48	Springer & Woodburn(1960)
Pinellas Co. FL	*Harengula jaguana* *Anchoa hepsetus*	72	95	62	Saloman & Naughton (1979)
Panama City FL	*Harengula jaguana* *Menidia beryllina*	72	90	45	Naughton & Saloman (1978)
Horn Island MI	*Anchoa lyolepis* *Harengula jaguana*	80	92	76	Modde & Ross (1981)
Gilchrist TX	*Brevortia tyrannus* *Anchoa mitchilli*	81	96	25	Reid (1955)
Mustang Island TX	*Polydactylus octonemus* *Menidia beryllina*	-	-	47	McFarland (1963)
Mustang Island Tx	*Trachinotus carolinus*	69	85	44	Gunter (1958)

Naughton, 1979; Modde & Ross, 1981). One exception to this pattern was described for a South Carolina beach (Anderson et al., 1977) where diversity was highest in summer, but abundance was greatest in winter. This was due solely to the presence of the silverside Menidia menidia during the winter months. Other studies have also indicated this species may be abundant in the surf zone during winter (Cupka, 1972; Hillman et al., 1977).

Modde & Ross (1981) have indicated that not only are there seasonal patterns of abundance of surf zone fishes, there are daily variations as well. Maximum abundances of fish were obtained in the early morning hours mainly due to increased catches of anchovies and herrings. Pompano and kingfish showed less definite daily cycles with a tendency to have increased abundance in the afternoons and early evenings. In sampling at Melbourne, Florida, Peters (1984) found no consistent pattern of size of catch in relation to time of day.

Recruitment patterns of fish in the surf zone varies somewhat among species. Herring (H. jaguana) on the Gulf Coast appear to have 2 main pulses of recruitment, the first in April and the second in late June (Modde, 1980; Gunter, 1958), although some recruitment occurs throughout the period April - October. Peters (1984) found H. jaguana recruited during June off the east central coast of Florida. For other resident species such as T. carolinus and M. littoralis, recruitment appears to be generally continuous from April to October or November (Finucane , 1969; Anderson et al., 1977; Modde, 1980; Peters, 1984). In the case of T. carolinus, Fields (1962) has suggested that the main wave of recruitment to beaches in Georgia occurs in April but that subsequent recruitment occurs at monthly intervals until as late as December. Distributional data collected by Finucane (1969) indicates that juvenile T. carolinus tend to be found only in surf zone areas, whereas, adults are broadly distributed.

Effects of Beach Nourishment

Relatively little quantitative information is available concerning the effects of beach nourishment, as distinguished from borrow area dredging, on the fishes of the near-shore zone. In both cases, the major potential problem for fish will be the clogging of the gill cavities by fine sediments generated by the engineering activity with subsequent death by suffocation (Courtenay et al., 1974). O'Conner et al. (1976) have examined the lethal and sub-lethal effects of suspended solids on estuarine fishes. This work found that mortality can be caused by suspended sediments at concentrations equal to that generated during dredging or dredged material disposal. Benthic feeding fishes were generally affected to a lesser degree than filter feeding, pelagic fishes. Juveniles were more affected than adults. Sub-lethal concentrations of sediment were shown to induce stress responses in estuarine fish which may be biologically significant (O'Conner et al., 1977).

Courtenay et al. (1974) provide qualitative data on the effects of dredging and beach nourishment on 5 beaches in Broward County, Florida. They concluded that there was no evidence of negative effects of dredging on fishes in any area studied. Population levels were similar in disturbed and undisturbed areas. They suggested that motile invertebrates and fishes may have left the borrow area during dredging but that these species returned within 4 months of the end of dredging, although no data are provided. Courtenay et al. (1974) suggest that destruction of suitable habitat may be more significant than turbid conditions since the near-shore fishes may be relatively tolerant of high turbidities. They further suggest that burrowing species such as jawfishes (Opistognathidae), eels (Muraenidae) and gobies (Gobiidae) might be potentially most affected by burial by suspended sediments, yet found no differences in population numbers of these fishes in dredged versus undredged areas. As a result of a resurvey of one of these nourished beaches seven years after disturbance, Courtenay et al. (1980) suggested that populations of a jawfish (<u>Opistognathus whitehursti</u>), a cardinalfish (<u>Apogon townsendi</u>) and a blenny (<u>Acanthemblemaria aspera</u>) had been negatively affected by habitat alteration resulting from beach nourishment. However, alternate explanations were not considered and other factors may have caused these changes during the intervening seven years. Similarly, a study of dredging and beach nourishment at Duval County, Florida (Applied Biology, 1979) found no reductions in fishes, shrimps or crabs in offshore borrow areas as compared with nearby controls.

The only study available which specifically examines the effects of beach nourishment on near-shore fishes is that of Holland et al. (1980). In a survey of fishes before and after beach nourishment at Lido Key, Florida, a temporary increase in fish abundance was noted along the newly filled beach. A similar but persistent increase of fish abundance was observed in the borrow pit throughout the 1 year study.

<u>Ocypode quadrata (ghost crab)</u>

The ghost crab (<u>Ocypode quadrata</u>) is one of the most visible of sand beach residents. This crab is semi-terrestrial and constructs burrows 0.6 - 1.2 m in depth which may be located from near the high tide line to distances up to 0.4 km from the beach (Williams, 1965). The range of the species is from Rhode Island to Brasil. Relatively little work on the ecology of <u>O. quadrata</u> has been done. The basic habits of the species have been described by Cowles (1908) and Milne & Milne (1946) and are summarized by Williams (1965). Feeding habits are examined by Fales (1976), Wolcott (1978) and Robertson and Pfeiffer (1982). Wolcott (1976) demonstrates that <u>O. quadrata</u> is able to meet its water requirements by extracting soil water. Haley (1969) has examined growth and sexual maturity in this species.

The egg laying season for <u>O. quadrata</u> is approximately April to July over the range from Tortugas, Florida to New Jersey (Williams, 1965). Ovigerous females move down the beach and enter the water to lay the eggs. New individuals recruit to the beach following a planktonic larval stage. Younger crabs are found in burrows closer to the water's edge while older crabs tend to be found much farther away. Ghost crabs appear to be relatively long lived (up to 3 years; Haley, 1969).

<u>Ocypode quadrata</u> appears to possess considerable flexibility

in feeding behavior. Ghost crabs in general have been considered scavengers on carrion, feeding on material associated with windrows of material along the high tide line. Wolcott (1978) demonstrated that on a North Carolina beach, O. quadrata obtains over 90% of its diet from live prey, mainly Donax and Emerita taken from the intertidal zone. Wolcott (1978) suggests O. quadrata is an important predator on both species and may consume most of the annual production of Emerita at this location. In contrast, Robertson and Pfeiffer (1982) describe extensive deposit feeding behavior of O. quadrata on a Georgia beach. This beach differs from that studied by Wolcott (1978) in being of much lower wave energy and generally possessing dense patches of microalgae on the sand surface. Apparently ghost crabs are opportunistic feeders that are able to adapt their feeding behavior to local conditions.

Ghost crabs are primarily active at night. They emerge from their burrows at dusk and move toward the water, often entering the edge of the water briefly. Wolcott (1978) observed that the main nocturnal activity was the digging of shallow holes in damp sand to obtain food. Digging on the foreshore ceased at dawn as the crabs moved up the beach towards daytime burrows. Crabs did not return to the same burrows. Burrows are generally shallow on the foreshore and much deeper on the backshore and foredunes (Hill & Hunter, 1976). Hill & Hunter (1976) found that the orientation of the angle of descent of the burrow is controlled by predominant wind direction.

Little quantitative data on the size of ghost crab populations is available. Most estimates of population size are derived from burrow counts (e.g., Wolcott, 1978) rather than direct animal counts and this may introduce significant error. Hill & Hunter (1976) estimated maximum O. quadrata abundance as $4\ m^{-2}$. Wolcott (1978) reports maximum values of 19 crabs per linear m of beach. Leber (1982a) estimates from visual counts of animals along a transect line a maximum density of only 0.05 crabs m^{-2} for a North Carolina beach. Maximum abundance of ghost crabs in North Carolina was found from August until October (Wolcott, 1978; Leber, 1982a; Reilly & Bellis, 1983). Activity of ghost crabs has been found to be temperature dependent with crabs being inactive below 16 C (Reilly & Bellis, 1983).

Only the work of Reilly & Bellis (1978, 1983) provides information on the effects of beach nourishment on O. quadrata. Semi-quantitative data suggested a 50% lower summer population following nourishment. They suggested that since most sediment was deposited below the major concentration of burrows, direct burial was not a major mortality source. They also suggested that ghost crabs may be able to burrow up through overburden material in any case. Therefore, Reilly & Bellis (1978, 1983) concluded that the population decrease was due to emigration by crabs in response to a decreased food supply resulting from the disturbance of intertidal prey populations.

Polychaetes

Polychaetes are often an important component of the sand beach macrofauna (Trevallion et al., 1970; Dexter, 1974,1976,1979; Hill & Hunter, 1976; Saloman, 1976; Matta, 1977; Parr et al., 1978; Saloman & Naughton, 1978; Applied Biology, 1979, Marsh et al., 1980; Spring, 1981; Shelton & Robertson, 1982; Diaz & DeAlteris, 1982; Knott et al., 1983; Gorzelany, 1983; Saloman & Naughton, 1984). For example, Knott et al. (1983) reported that polychaetes were dominant in number of individuals (55%) from intertidal beaches in South Carolina and co-dominant with amphipods in number of species (28%). Subtidally the relative importance of polychaetes tends to increase (Matta, 1977; Spring, 1981; Gorzelany, 1983; Knott et al. 1983).

Members of the family Spionidae are often the dominant elements of the polychaete component found on sand beaches. Of these species, *Scololepis squamata* is one of the most widely distributed (Table 1) and is often quite abundant. *S. squamata* accounted for 80% of all polychaetes at intertidal stations at a South Carolina location (Knott et al., 1983). The species is relatively small, reaching a maximum length of 47 mm and constructs burrows in sandy sediments (Foster, 1971). *S. squamata* is a surface deposit feeder (McDermott, 1983). At a beach site in New Jersey, McDermott (1983) found *S. squamata* dominated the mid-intertidal region (86% of total abundance), forming a 20 m wide, dense band along the shoreline. The density estimate for this species was 40,000 m^{-2}. *S. squamata* showed a distinct offshore movement in the winter months. McDermott (1983) found that *S. squamata* was a dominant food item for several species of surf zone fishes. This species was apparently highly available to fish since it tended to be washed out of the sediment and into the water column.

Aside for the ennumeration of polychaetes in species lists of the papers cited above, there is very little ecological information on the macrofaunal polychaetes inhabiting open coast sand beaches. Meiofaunal polychaetes of this habitat have received considerable taxonomic and some ecological attention (reviewed by Westheide, 1971). A variety of vermiform groups including the Turbellaria, Nemertinea, Archiannelida and Oligochaeta may also be abundant on sand beaches (Gorzelany, 1983). However, taxonomic difficulties remain sufficently imposing such that little specific ecological information is available.

Some information on the effects of sediment deposition on polychaetes is available. Saloman & Naughton (1984) found significant decreases of *S. squamata* occurred on beaches in Panama City, Florida following sediment deposition. Reilly & Bellis (1978, 1983) observed a decrease in *S. squamata* following nourishment, but were unable to determine conclusively if nourishment was the cause of the decline. *S. squamata* was the only organism ever found by Reilly & Bellis in a nourished zone during the nourishment process, suggesting a certain tolerance of this species for sediment disturbance. Mauer et al. (1982) investigated the capability of two polychaetes, *Scoloplos fragilis* and *Nereis succinea*, to burrow upwards through dredge material overburden. They concluded that both species were capable of upward movement through spoil material, and that either species was capable of moving upward through 0.9 m of material. Mortality tended to be higher as the silt-clay fraction of the sediments increased.

Sea Turtles

Although they are not permanent residents of sand beaches, sea turtles are organisms of major concern with respect to possible deleterious effects due to beach nourishment because these endangered or threatened species utilize the upper beach for nesting. The southeast United States and Florida in particular are important nesting areas for the loggerhead turtle, Caretta caretta, and to a much lesser extent, the green turtle, Chelonia mydas and the leatherback turtle, Dermochelys corracea (William-Walls et al., 1983; Harris et al., 1984). Because sea turtles utilize beaches only for nesting activities, no attempt will be made to review their general biology and only information relating to nesting will be presented.

The turtle nesting season in Florida generally ranges from May through August, although some nesting activity may occur in April and September (William-Walls et al. 1983). Initiation of nesting activity may be correlated with increased water temperature in the spring. Mann (1977) observed no correlation of nesting frequency of loggerheads in south Florida with the lunar cycle. Maximum nesting activity typically occurrs in June-July. Loggerhead turtles often produce several nests per season and are estimated to average between 2 and 3 nests per female in the Cape Canaveral area (Erhart, 1979), while estimates from Broward county were 3 to 5 nests per female (Mann, 1977). The interval between nestings for loggerhead turtles on Hutchinson Island, Florida ranged from 11 - 20 days with a mean of approximately 14 days (William-Walls et al., 1983). Average distance between nests for an individual female was on the order of 5 km (William-Walls et al., 1983). The average clutch of eggs produced by loggerhead turtles in the area of Melbourne, Florida was found to be 114 by Erhart & Raymond (1983).

Turtles moving up the beach to nest may return to the ocean without doing so. It has been suggested that false crawl behavior may be due to a number of factors such as sand texture and compactness or characteristics of beach width or slope (William-Walls et al., 1983). Heavy night pedestrian traffic, artificial lights and the presence of seawalls or other such structures have also been implicated (Mann, 1977; Fletmeyer, 1980). Stoneburner (1981) suggested a thermal cue for nest site selection which may also be involved in false crawl behavior.

Both Mann (1977) and Fletmeyer (1980) suggested that sand compaction as a result of beach nourishment might be responsible for an increase in the number of false crawls by loggerheads on certain beaches. However, one natural beach studied had similar numbers of false crawls to the nourished beaches and the causal agent could not be clearly identified in this analysis. Erhart & Raymond (1983) have recently examined in detail the effects of beach nourishment on turtle nesting. Their research showed that false crawls which resulted in unfinished nest pits were twice as frequent in the nourished beach area as compared to control areas. Turtles were often observed to initiate nests 3 - 5 times without depositing eggs, a behavior which was rare on control beaches. Although some turtles successfully nested on the nourished beach, only 36% of successful nests were actually laid in the compact nourishent sand. Among those nests which were laid in the beach nourishment sand, Erhart & Raymond (1983) found no evidence of either decreased hatching success or decreased hatching emergence relative to control beaches. Significantly improved nesting success occurred on the nourished beach on the second year following nourishment, although controls showed no change. Erhart & Raymond (1983) suggest that this improvement was due to removal and recycling of compact beach nourishment sediment which provided a better substrate for turtle nesting.

It therefore appears that beach nourishment may have a significant short-term impact on sea turtle nesting success and that the probable causative factor is compaction of sediments. Fletmeyer (1980) made preliminary studies of sand compaction and found sediment of nourished beaches to be significantly more compact than unnourished beaches. Factors which may contribute to increased compaction include a high silt component, the shape of sand grains and the method of deposition. Fill material transported by truck and distributed by bulldozer may be artificially compacted by the vehicle movement, while deposition by pipeline may result in hydraulic compaction of sediments. The study of methods to prevent a high degree of compaction during beach nourishment would appear to be important in avoiding the decrease in nesting success engendered by nourishment.

Several additional factors relating to beach nourishment activities may have a negative impact on turtle populations by affecting hatchling survivorship. On projects where hydraulic placement of sand is used, the dredge pipe may act as a physical barrier preventing hatchlings from reaching the ocean. Since the pipe may run across beach not being actively nourished, turtle nests away from the immediate nourishment area may be affected. Steps should be taken to prevent this problem. Secondly, where heavy equipment is used on the beach, deep ruts may trap hatchlings and prevent their reaching the ocean. Mann (1977) reports an occurrence of this problem and indicates that loss of view of the horizon and disorientation rather than a physical entrapment of hatchlings may be significant. Both problems have occurred in recent nourishment projects in Florida (R. Witham, pers. comm.). Finally, Mann (1977) reports that external pressure from machinery on the beach can collapse nests, increasing mortality of hatchlings, particularly on beaches with relatively loose sand.

A SUMMARY OF BIOLOGICAL EFFECTS OF BEACH NOURISHMENT

Beach Nourishment Effects

Although a considerable body of information is available on the effects of dredging on benthic communities (Thompson, 1973; Oliver & Slattery, 1976; Mauer et al., 1982; Naqvi & Pullen, 1983), much less is known about the specific environmental consequences of beach nourishment. Effects of dredging and beach nourishment were first reviewed by Thompson (1973), while Naqvi and Pullen (1982) have reviewed recent information. Specific relevant studies include those of Parr et al. (1978), Reilly & Bellis (1978, 1983), Marsh et al. (1980), Culter & Mahadevan (1982), Gorzelany (1983), and Saloman & Naughton (1984).

Parr et al. (1978) analysed the effects of deposition of 585,000 m^3 of dredged sediment from San Diego Bay on the nearshore fauna of Imperial Beach, California. They concluded that direct adverse effects of beach nourishment were few except for the direct burial of some less mobile organisms. In fact, an increase in diversity and abundance of organisms was observed following beach nourishment. However, these increases occurred during the summer when such increases are typical due to the seasonal decrease in physical disturbance from waves. They suggest that nearshore populations are adapted to seasonal sediment movement and will be little affected by receiving high sediment loads. They suggest that offshore organisms might be relatively more susceptible to increased sediment loadings, although no indications of this happening were actually observed.

Reilly & Bellis (1978, 1983) thoroughly studied the effects on the fauna resulting from a deposition of 902,174 m^3 of material which was dredged from an estuary and placed on a beach on Bogue Banks, N. C. Unlike Parr et al. (1978) only intertidal samples were taken. In this project, sediment deposition resulted in complete elimination of intertidal organisms since sediments were added to a depth of 2 m while the intertidal zone was moved 75 m seaward in a single day. Initial recruitment was in 15-30 days while subsequent larval recruitment of the mole crab _Emerita talpoida_ and littoral drift of small _Donax_ into the nourishment area took place within 2 months. No large _Emerita_ or _Donax_ moved onto the nourishment beach following nourishment, and Reilly and Bellis (1978, 1983) concluded that this was due to mortality of larger animals in nearshore overwintering areas although this was not proved. Reilly and Bellis suggest that a delay in larval recruitment to the nourished beach was caused by high turbidity, again without supporting data. Reilly and Bellis (1978, 1983) concluded that the nourished beach recovered slowly. Although _Donax_ densities failed to recover to control densities during the study, density of _Emerita_ was comparable to controls immediately following the end of nourishment. The amphipod _Haustorius canadensis_, which lacks a pelagic larval stage, also had failed to recover as of the end of this study, which was only two months following the termination of nourishment.

Marsh et al. (1980) examined nearshore benthic communities near Hallandale Beach, Florida some 7 years after beach nourishment and offshore dredging. No long term effects of beach nourishment were observed for either the infaunal benthos or for the offshore coral reef biota. Culter & Mahadevan (1982) studied the long term effects of beach nourishment and borrow pit dredging on the benthic infauna of the nearshore zone of Panama City Beach, Florida some 3-4 years following nourishment. For the borrow areas, they found no differences between borrow sites and surrounding areas. They concluded that no long term adverse effects of beach nourishment had occurred either in the borrow pits or on the nourished beaches.

Gorzelany (1983) studied the biological effects of a beach nourishment project involving placement of approximately 413,000 m^3 of sand on the beaches of Indialantic and Melbourne Beach, Florida. There was no evidence from this study that beach nourishment caused any negative effects for any element of the nearshore infaunal community. Natural seasonal variability appeared much greater than any effect of nourishment. This nourishment project was carried out from mid-October through January, a period of low biological standing stock and low recruitment in the nearshore community which Gorzelany concludes may have been ecologically favorable. Additionally, there did not appear to be substantial movement of nourishment sediment into the more diverse offshore areas (Stauble et al., 1983), which may have helped limit negative effects of the nourishment (Nelson & Gorzelany, 1983).

Saloman & Naughton (1984) studied the effects of deposition of an estimated 183,492 m^3 of dredged sand on the macrofauna of the swash zone and first sand bar at 23 sites along the beach at Panama City Beach, Florida. The deposited material was similar to existing beach material at most sites and turbidity associated with deposition appeared relatively low. On the basis of an intensive sampling program, they concluded that sand deposition resulted in decreases of number of species and number of individuals for a 5-6 week period for the swash zone populations. No differences between deposition and non-deposition areas were seen after this period. No effect of sand deposition was observed for the organisms located on the first sand bar.

Comparisons with the Effects of Storm Events

Large scale storm events may generate changes in the physical environment of the nearshore zone which may be related to those which take place during beach nourishment. Storm waves may cause rapid, substantial sediment redistribution and increased turbidities as does beach nourishment. Examination of the recovery of benthic systems following storm events provides an opportunity to evaluate the natural recovery potential of the benthos to sediment disturbance, although it must be remembered that storms may also cause perturbations due to salinity changes.

Keith & Hulings (1965) sampled the shallow sub-littoral (1 m depth) of the Texas coast before and after a hurricane in the fall of 1963. There was little alteration in composition or abundance of species except at one station near an inlet where salinity was drastically lowered.

Croker (1968) considered the effect that passage of two hurricanes had on the abundances of 6 species of intertidal haustoriid amphipods. These storms were of relatively low energy although Croker (1968) reports considerable shifting of sand. No effects on either the distributional patterns or abundances of these amphipods were observed.

Ansell et al. (1972a) observed seasonal changes of the intertidal beach macrofauna on two beaches in south west India in relation to monsoon events. Considerable movement of sand occurred at both beaches during the monsoon period and may have negatively affected polychaetes. Other groups such as the coquina clams (Donax) and mole crabs (Emerita) were not greatly affected. Beach populations recovered after the monsoon except at one site where populations were affected by mining of sand from the beach for construction.

The most thorough study of a storm impact on sand beach fauna is that of Saloman & Naughton (1977). They were able to sample beaches at Panama City Beach, Florida immediately before and after the passage of a major hurricane which caused considerable erosion on the beach. Results indicated that numbers of individuals were little changed following the storm, while number of species was actually increased due to transport of offshore species into the intertidal zone. Species number returned to normal levels within a week. The low rainfall associated with the storm did not greatly alter salinity and may have contributed to the lack of mortality observed in this case.

Apparently the adaptations of intertidal sand beach organisms which allow them to live in a region of high sediment transport also allow these species to survive storm generated wave action, sediment transport and turbidity. There seems to be little evidence that natural sediment transport is a major source of mortality for intertidal beach species. The effects of such disturbance on subtidal organisms which may be adapted to somewhat more stable sediment conditions is not known.

Experimental Sediment Burial Studies

Although experimental tolerances to sediment burial have been determined for several organisms with regard to offshore dredge spoil disposal (Mauer et al., 1978), no information has been available for beach organisms. Experimental burials with different mean sediment grain sizes were therefore carried out on the intertidal organisms which might be expected to be most seriously affected by direct sediment burial. These species were the abundant Donax spp. and Emerita talpoida, and the seasonally abundant snail Terebra dislocata.

Methods

Each experiment was performed in 15 l aquaria which were divided in half with plastic partitions. For Donax and Terebra, beach sand from the swash zone was sieved to remove all macrofaunal organisms and placed in 20.3 cm diameter culture dishes. One culture dish was placed in each half of an aquarium. A section of PVC pipe (15.3 cm diameter by 20 cm tall) was placed on top of the sand such that the outer walls contacted the inner walls of the dish. Animals were added to the sand inside the pipe and allowed to burrow. For Emerita, sieved beach sand was placed to a depth of 5 cm in the bottoms of the aquaria, the tanks filled, and the animals then added. Treatments in each experiment consisted of instantaneous burial by 1, 5 or 10 cm of sediment of a given grain size which was dumped either into the pipe section or into the entire half of the aquarium together with a control treatment where no sediment was added. The sand used for burial had been collected from the beach zone and sieved to provide several size ranges of sediment. Each treatment was replicated 4 times, with treatment being assigned to a given aquarium at random. For each species a separate experiment was run with fine sand (mean grain size = 2.6 phi) and coarse sand (mean grain size = -0.8 phi). Ten Emerita talpoida and 10 Donax spp. were used per replicate. Five Terebra dislocata per replicate were used for the coarse sand experiment, but only 3 per replicate were available for the fine sand experiment. Per cent survivorship was evaluated by removing all animals form each treatment after 24 hrs. Animals were then placed in culture dishes of seawater and examined for signs of life. The percent of animals which were found at the surface 24 hrs after burial was also noted. Water temperature was 28-29° C and salinity was 33-34 ppt.

Results

The results of sediment burial experiments are given in Table VIII. Emerita experienced little mortality in a 24 hr period when buried in up to 10 cm of fine sediment, but experienced 55% mortality when buried by 10 cm of coarse sediment, through which it was unable to burrow. In fine sediments, the sediment becomes sufficiently fluid for the Emerita to move upward through the sediment almost instantly, whereas coarse shell material prevents this. Only 2.5 % of the Emerita reached the surface of 10 cm of coarse sediment as compared with 85% in the fine sediment treatment. Donax experienced relatively greater mortality in fine than in coarse sediments, with maximum mortality occurring for burial under 10 cm of sediment. Terebra dislocata showed little mortality for either fine or coarse sediments at any depth of burial used. These results suggest these organisms can deal with instantaneous burial by sediment at depths of up to 10 cm, although some mortality results. In the case of Donax spp., since the major part of the population is found at some distance from the beach (Nelson and Gorzelany, 1983) where deep burial should not occur, the results suggest that the population should be affected only minorly outside the intertidal zone.

Table VIII. Results of sediment burial experiments for *Emerita talpoida*, *Donax* spp., and *Terebra dislocata*. Numbers are mean % (std. dev.) surviving after 24 hrs burial by 0, 1 5 or 10 cm of either fine (F, mean grain size = 2.6 phi) or coarse (C, mean grain size = -0.8 phi) sediment.

Species (sediment type)	Treatment			
	0 cm	1 cm	5 cm	10 cm
Emerita talpoida (C)	100(0)	97.5(5)	82.5(23.6)	45(17.3)
Emerita talpoida (F)	97.5(5)	100(0)	97.5(5)	95(5.7)
Donax spp. (C)	100(0)	100(0)	82.5(17)	85(5.7)
Donax spp. (F)	100(0)	100(0)	95(5.7)	70(24.5)
Terebra dislocata (C)	100(0)	95(10)	95(10)	90(11.5)
Terebra dislocata (F)	100(0)	100(0)	100(0)	100(0)

GUIDELINES FOR BIOLOGICAL MONITORING OF BEACH NOURISHMENT

Given the conclusion of the preceding section, namely that available evidence points to minimal biological effects of beach nourishment, the question may be raised as to whether biological monitoring of beach nourishment projects is necessary at all. With presently available information, a decision against biological monitoring is decidely premature. Only 4 studies are available which have any data to compare macrobenthic populations before and after beach nourishment (Parr et al., 1978; Reilly & Bellis, 1978, 1983; Nelson & Gorzelany, 1983; Salomon & Naughton, 1984). Of these studies, three have decided deficiencies in terms of temporal or spatial adequacy of sampling (Parr et al., 1978; Nelson & Gorzelany, 1983) or in data analysis (Reilly & Bellis, 1978, 1983). The need for biological monitoring of beach nourishment will not diminish in the future until a more substantial body of well designed and analysed studies is available.

Recommendations for Monitoring Requirements

While every area is ultimately biologically unique, it is doubtful most monitoring sampling designs are constructed specifically to meet the requirements of a given area since detailed baseline studies would be required in each case in order to do so. The presently existing variability in methods suggests that considerable improvement could be made towards increased standardization. Standardization of sampling effort and methodology would appear to offer greater rewards by providing a readily comparable data set from which conclusions concerning general biological impacts of nourishment activities could be drawn.

Despite the desirability of standardization, comparison of several recent project permits given by Florida Dept. of Environmental Regulation indicates that monitoring requirements are broadly variable. Table IX compares the requirements for biological monitoring among five recent beach nourishment projects in Florida. Sediment volumes for these projects varied from approximately 49,000 - 496,900 m^3. Monitoring requirements were even more varied, although intensity of monitoring varied somewhat in proportion to the size of the project. For example, biological monitoring activities required for a Palm Beach County nourishment project were limited to a visual inspection of sabellariid worm reefs, while a Fort Pierce project required no specific monitoring, only a general requirement that turtles be protected from nourishment activities. A second Fort Pierce nourishment project involved extensive biological sampling of the offshore borrow area for 1 year, but no monitoring of the beach area where material was deposited. The Captiva Island permit required extensive borrow area monitoring over a 2 year period, together with monitoring of 2 beach transects. The Melbourne Beach/Indialantic project required monitoring at 5 beach transects. For those projects where quantitative monitoring was required, specific methodology varied widely. For example, borrow areas at Captiva Island were

Table IX. Comparison of biological monitoring requirements for various beach nourishment projects in Florida.

	PROJECT				
	(VOLUME OF SEDIMENT - YD^3)				
BIOLOGICAL MONITORING REQUIREMENTS	FT. PIERCE-INLET (100,000)	FT. PIERCE-BEACH[1] (372,000)	PALM BEACH (65,000)	CAPTIVA ISLAND (650,000)	MELBOURNE BEACH (540,000)
DATE	July 1982	August 1980	Aug 1982	Oct 1981	January 1981
TURTLE PROTECTION (TIMING)	*		*	*	*
VISUAL INSPECTION OF WORM REEFS			*		
SPECIES LIST		*		*	*
SPECIES AREA CURVE				*	
FAUNAL DENSITY		*		*	*
BIOMASS				*	
SPECIES RICHNESS				*	
SPECIES DIVERSITY		*		*	*
FAUNAL PROPORTIONS				*	
FAUNAL SIMILARITY				*	
EQUITABILITY					*

[1] BORROW AREA ONLY

monitored for 2 years at 5 stations with 5 replicate 0.22 m^2 grabs per station. In contrast, the Fort Pierce borrow area was monitored for only 1 year at 10 stations with 3 replicate 0.1 m^2 grabs per station. The section below is an attempt to suggest standardized levels of monitoring effort and standardized methods for carrying out such monitoring activities.

Recommendations for Sampling Program Design and Execution.

Determination of specific biological effects resulting from either removal of sediment from borrow areas or deposition of sediment in the nearshore zone requires careful sampling design and execution. This is due to the fact that natural variability due to seasonality and spatial patchiness of organisms is quite large and may obscure any but the largest of effects potentially caused by coastal engineering activities if sampling design is inappropriate. Although it is desirable to collect as much and as varied data as possible, realities of economic costs and manpower limitations often impose severe restrictions on data acquisition. It is therefore important to concentrate efforts in areas of maximum return.

Appropriate design of an "optimum impact study" (Green, 1979) for organisms on a sandy beach experiencing beach nourishment is far more complex than may be initially apparent. The problems involved in planning such a program relate to sampling methods, sampling design (allocation of sampling effort in space and time), and data analysis. A recent book by R. H. Green, Sampling Design and Statistical Methods for Environmental Biologists (Green, 1979), offers much valuable advice on each of these areas and should be required reading for anyone contemplating an environmental impact study.

The organisms of concern during beach nourishment which are considered here are the sea turtles and the other macrofaunal organisms.

Sea Turtles

Suggested procedures to minimize the impact of beach nourishment have recently been admirably stated by Naqvi & Pullen (1982). These are repeated and in some cases expanded on below.

1) Nourishment sand should match natural sand as closely as possible in grain size distribution and chemical characteristics. Sand compaction appears to be the primary problem for sea turtles arising from beach nourishment. Sand compaction may result from either an excessive amount of fine grain sizes or from the application method. Deposition of sand by truck with movement by bulldozer may mechanically compact sand while deposition by slurry pipeline may hydraulically compact it. A compact beach may be more resistant to erosion and therefore more desireable from an engineering standpoint. Since sea turtles frequently nest at the base of the dune, a possible

solution may be to plow or harrow the backshore area just below the dune after project completion in order to reduce compaction in the turtle nesting area.

2) Avoid covering dune vegetation or depositing material near the base of dunes since turtles appear to prefer nesting near dune vegetation, although this response may be more precisely a response to a dark, broken horizon which contrasts with the seaward horizon (Mann, 1979). Sand deposition on the dunes is also undesirable from the point of view of dune stabilization in that it may kill the vegetation and require a re-vegetation project.

3) Whenever possible, nourishment activities should avoid the turtle nesting season. Maximum safety would limit nourishment to the period October through March. Extension of projects into April might still avoid the main nesting period. Where the need for nourishment of a beach is anticipated, the recording of nesting activity will be extremely valuable in determining optimum scheduling for a nourishment project. The types of information to be recorded are given in Harris et al. (1984). Nesting surveys should not be initiated without consultation and coordination with Florida Department of Natural Resources Bureau of Marine Research, U. S. Fish and Wildlife Service, and the National Marine Fisheries Service.

4) If nourishment activities must be scheduled during turtle nesting season, arrangements for a turtle nest relocation program should be made. Turtle nests should be located in the early morning within areas to be affected by nourishment. Turtle nests located within such areas may be relocated to a nearby undisturbed beach or a secure hatchery facility for hatching. Turtle nest relocation should be carried out only by experienced personnel and is permissable only with the agreement and the granting of permits by the Florida Department of Natural Resources. The United States Fish and Wildlife Service grants permits for this activity outside the state of Florida.

Macrofauna

General Design Considerations

In order to have an "optimum impact study" design, Green (1979) suggests four conditions must be met. Before-impact baseline data must be gathered as a temporal control with which to compare the post-impact data. Secondly, the type of impact and the time and place of its occurrence must be known. Third, all relevant biological and environmental variables must be measured simultaneously with the individual samples collected in order to allow hypothesis testing. Fourth, an area which will not receive the impact of concern must be available to serve as a spatial control. Reliance on a temporal comparison (i. e. before- after) in the absence of a spatial control allows the possibility that a significant change may have resulted from factors other than the factor of prime concern. Similarly, the temporal control is necessary in order to determine that significant differences between areas in terms of the variables to be measured did not exist before the impact occurrred, thus giving a result falsely attributed to the impact. Use of the areas by times design then allows the use of inferential statistics such as analysis of variance (ANOVA) to test for the presence of impact effects.

Hurlbert (1984) has evaluated the use of inferential statistics in ecological analyses and finds that in many cases the underlying assumptions of techniques such as ANOVA have been violated by improper experimental designs. Such violations of assumptions make invalid the conclusions reached by the inferential statistical analyses used. Hurlbert suggests that Green's optimal impact study design, and almost any situation in which analysis of an environmental impact is being done, will result in a statistical sampling design inappropriate for the use of inferential statistics. This is because an environmental impact often falls on only one location, for example a single stretch of beach. It is therefore impossible to randomly assign a particular treatment effect (e.g. nourished beach versus unnourished beach) to particular locations, and more importantly, it is impossible to intersperse the two types of treatments. Interspersion of treatments in a random fashion is necessary to avoid the potential problem of the impingement of chance events on an experiment in progress in such a way that only one type of treatment is affected. Hurlbert considers this problem largely unsolveable and suggests that inferential statistics be avoided in such cases, in contrast to Green.

In the specific case of beach nourishment, interspersion of treatments will be impossible except in the rare case where several areas are being nourished simultaneously. The recent report of Salomon & Naughton (1984) provides one instance where this condition was approximately met and adequate interspersion was indeed possible. For most situations, however, the acceptibility of using inferential statistics to analyse beach nourishment impact must then depend on the investigator's ability to demonstrate that random events are unlikely to impinge solely on the replicates of one treatment. A red tide event affecting only those sample areas within a beach nourishment area would be an example of such an event. Given the nature of the beach environment where physical and chemical parameters are generally similar over the typical scale of a nourishment project, the probability of random events affecting one treatment and not another may be relatively small. The usefullness of inferential statistics in analysing beach nourishment impact would appear to far outweigh the potential for error resulting from lack of interspersion of treatments. However, the investigator should be aware of the potential difficulty and should maintain vigilance for the intrusion of some extraneous factor.

Specific Design Considerations

In order to be able to test for effects of beach nourishment, it will be necessary to take replicate samples within each combination of time and location. Statistically significant differences between nourished and control locations can only be demonstrated by comparison to differences among replicate measurements within treatments. Although not essential, collection of equal numbers of replicate samples for all combinations of time and location will greatly simplify subsequent analysis and is strongly recommended.

Appropriate layout of sample locations will help to avoid the problems associated with the limitations on interspersion of treatments discussed above. Figure 5 compares two possible sampling designs. In the first case both the treatment (i. e. nourishment) and control sample areas are completely non-interspersed, which is a poor design. A better design is to split the control samples to either side of the nourishment area to achieve a better, although incomplete, degree of interspersion.

Design of sampling must be determined by the requirements of the methods to be used for data analysis. It is my recommendation, as it is for Green (1979), that ANOVA offers the best means for analysing the effects of an environmental impact. Meeting the assumptions of the methodology are esential and are discussed below. The preferred design of Fig. 5 is laid out in such a way as to be amenable for analysis of variance. In the following disucusion it will be assumed that the reader has some familiarity with ANOVA and is familiar with the terms "nested" versus "factorial" ANOVA. The excellent summaries of Sokal and Rohlf (1981) and Underwood (1981) are recommended.

In utilizing ANOVA, several design options are available, with the basic contrast being between a single factor nested design and a multifactor design. Both Parr et al. (1978) and Reilly & Bellis (1978, 1983) employed aspects of a nested ANOVA design, although only Reilly & Bellis (1978, 1983) explicitly analysed their data as such. Both papers, however, give insufficient detail about statistical procedures to evaluate whether analyses were correct for the sampling design given. Hurlburt (1984) strongly urges all papers to clearly describe in detail the physical layout of samples and the statistical methods used, a recommendation which is strongly seconded here for beach nourishment work.

The advantage of a nested design is that it allows several levels of the variation inherent on sandy beaches to be included and evaluated in the statistical analysis. For example, Reilly & Bellis (1978, 1983) used 3 parallel transects at each of their sampling locations, with three replicate samples taken from each beach zone examined. The replicate samples are nested within transects, with the transects nested within treatments. Comparisons are still made between the single factor of the treatment received by the beach (i. e. nourishment) and no nourishment (control). In this way spatial variation along the beach is factored out of the main comparison of interest. The disadvantage is that a nested design is a single factor design, so that changes among months can not be effectively analysed simultaneously.

In contrast, a multi-factor design permits the evaluation of differences in area and time simultaneously (2 way ANOVA), or even the anlaysis of area, time and beach zone (3 way ANOVA). Additionally, these analyses permit the evaluation of the interaction between the main factors such as area and time. Green (1979) has even stated that the evidence for an impact effect will be a significant areas-by-times interaction.

As Green (1979) points out, nested designs are extremely useful in preliminary sampling so that a proper apportionment of sampling

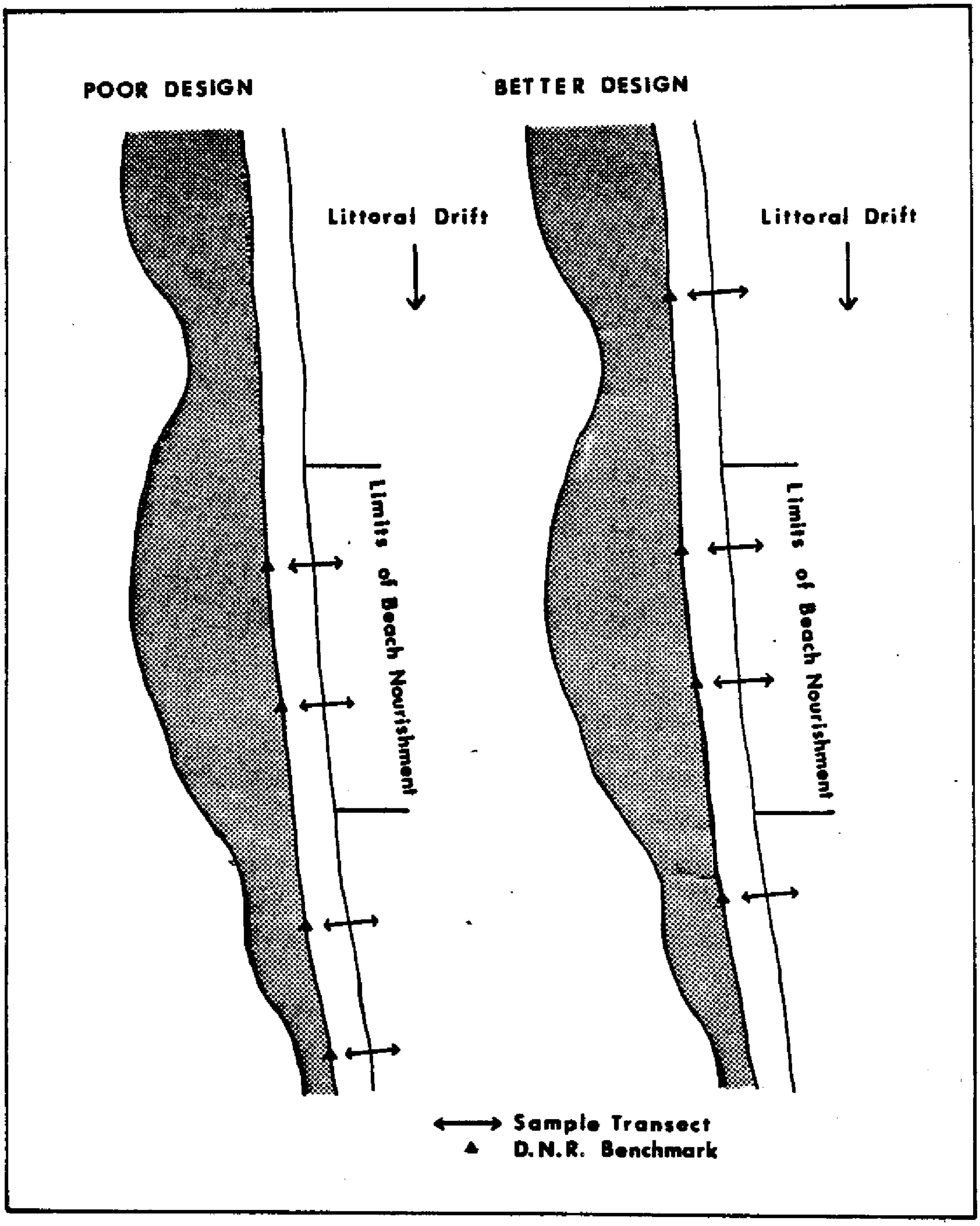

Figure 5. Comparison of two possible sample designs for monitoring of beach nourishment projects.

effort can be made. A major reason for utilizing nested sampling designs which include several parallel transects per sampling location is to minimize the effects caused by aggregation of organisms (mainly intertidal) along the beach. Such aggregations have been described in particular for Emerita (see above) and have been related to beach cusp structures. Specific examination of the relationship between Emerita density and beach cusp structure failed to reveal any patterns in the Melbourne area (Stauble & Nelson, 1985). Additionaly, neither Parr et al. (1978) nor Reilly & Bellis (1978, 1983) reported any significant differences in total density estimates from the parallel transects they established within a sampling location. It would therefore appear that this level of variability may safely be omitted in favor of additional sampling locations to increase replication at this level.

It would appear that a 2-way ANOVA design will be the most efficient method for analysing nourishment effects. It is suggested that the factor "beach zone along a transect" not be included in a single overall analysis to make a 3-way design. This is because it is quite clear that differences between different zones on a beach exist and there seems little point in testing this again. Rather, carry out a 2-way ANOVA at each beach level sampled and discuss the patterns observed at each with respect to time and treatment.

Specific Sampling Considerations

Several detailed papers concerning appropriate methodologies for sampling high-energy sand beaches are presently available (Cox, 1976; Hurme et al., 1979; Gonor & Kemp, 1978), each of which contains information of value. In some cases, recommendations will be made below which contradict some suggestions made in these works.

Given the fact that there is a rapid increase in species diversity and density of organisms as one moves down the beach into the subtidal area, it is necessary that beach sample locations consist of a transect across the intertidal and subtidal zones of the beach. Improved sampling efficiency can be gained by an optimum sampling design on these transects. For example, recent studies (Gorzelany, 1983) indicate very few marine organisms are found in samples from the mean high tide line in Florida. Samples from this point have been required on previous monitoring designs (Nelson & Gorzelany, 1983). These samples necessitate considerable processing time and yet yield little useful information. Their elimination would have resulted in an immediate decrease of approximately 30% in sample processing time in the case of the Melbourne Beach - Indialantic project because of the disproportionate difficulties in sorting organisms from such coarse sediment.

An additional aspect of sampling relates to the use of fixed versus variable sampling sites along an individual sample transect. In the case of the Melbourne - Indialantic project (Nelson & Gorzelany, 1983) sampling design specified that biological samples be taken at fixed intervals of 30.5 m (100 ft) beginning at the mean high tide line and extending 121.9 m (400 ft) offshore. The first sampling increment generally fell within the intertidal zone where the dominant organisms (the coquina clam Donax spp. and the mole crab Emerita talpoida) migrate up and down the beach with the tide. Sampling invariably occurred at slightly different tidal levels between transects and sample dates, leading to the possibility of generating large variability among samples. Figure 6 indicates the variability in total abundance of the coquina clam Donax over a 6 hr period between low and high tides at a fixed point (30.5 m from the high tide line) in the intertidal zone. In this case, variation between minimum and maximum values was almost 2000%. By the simple expedient of sampling at a movable spatial point, fixed only with respect to the swash zone, much of this variability can be removed.

Both Cox (1976) and Hurme et al. (1979) have included diver operated suction (airlift) samplers as useful methods for sampling subtidal beach fauna. Hurme et al. (1979) do point out that this device is only efficient in depths greater than 2 m. Stoner et al. (1983) compared the efficiency of a suction sampler to hand held cores in a bare sand habitat and found the suction method collected 73% fewer individuals per unit surface area of sediment. Given the increased expense, difficulty of operation in the surf zone, and decreased sample efficiency, use of suction samplers should be discouraged in favor of hand held cores.

The use of trenching (Cox, 1976; Gonor & Kemp, 1978) is not recommended. Although suitable for establishing zonation patterns, it is not suitable for density comparisons unless the trenching is replicated, which would require considerable effort.

The size of corers which have been used in beach macrofaunal studies has ranged from 7.6 cm diameter (Marsh et al., 1980; McDermott, 1983) to 20.2 cm diameter (Nelson & Gorzelany, 1983). Corer size is somewhat arbitrary although some recommendations can be made. Lewis & Stoner (1983) tested three core sizes (5.5, 7.6 and 10.5 cm diameter) and found that the smallest sizes sampled more individuals for equivalent sampling area in seagrass habitats. Kajak (1971) reports the opposite for unvegetated bottoms. Whether the use of smaller cores results in lower variances (Gray, 1971) or not will depend on the clump size of the fauna present, with variances being maximal when clump size equals corer size (Cox, 1976). Reilly & Bellis (1978, 1983) tested the relative yield of 5 and 10 cm diameter cores in terms of number of species and individuals. Ten 5 cm cores sampled equal numbers of species and approximately 92% of the number of individuals found with the 10 cm cores, although total area sampled was 24% less. The use of more, smaller cores will increase the degrees of freedom for statistically testing hypotheses and would therefore be preferred over fewer large cores (Green, 1979). This approach has been successfully used by Parr et al. (1970). Additionally, large cores such as used by Nelson & Gorzelany (1983) are more difficult to use effectively and require greater sieving time. Based on the results of Reilly & Bellis (1978, 1983) and Lewis & Stoner (1983), it is recommended that cores of 7.6 cm diameter be used. This core size should also meet the rule of thumb suggested by Green (1979) that the ratio of the area of an organism to that of the sample should be no more than 0.5.

It is recommended that analysis of the beach fauna be restricted to the macrofauna. The sand beach meiofauna is certainly diverse and abundant, but problems with sorting time and taxonomy would be prohibitive for most impact studies. In order to effectively sample the macrofauna, a sieve size of 0.5 mm is recommended. Although use

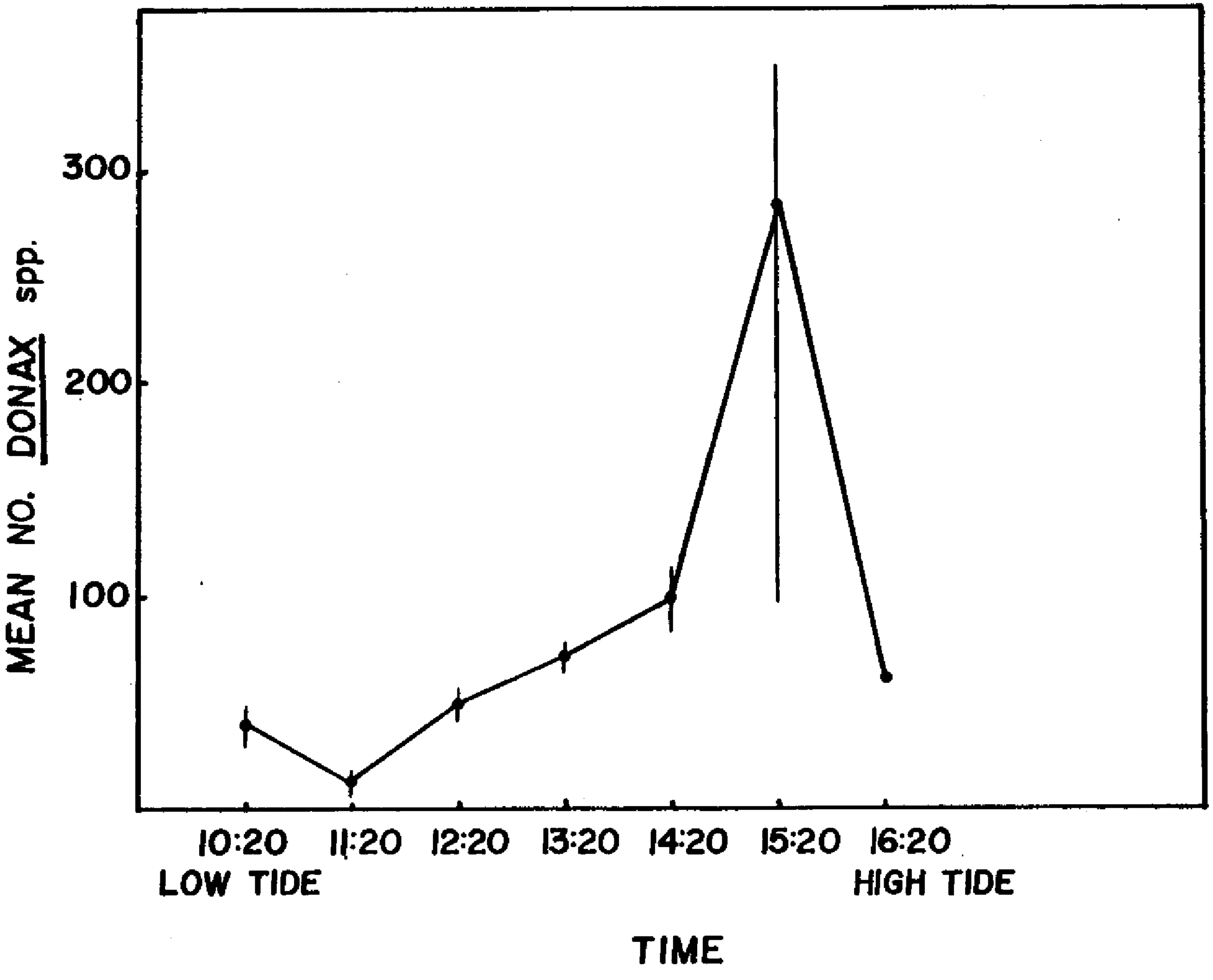

Figure 6. Variation in abundance of the coquina clams *Donax* spp. over 1 tidal cycle at a fixed sampling point 30.5 m from the mean high tide line at a study site at Melbourne Beach, Florida. Vertical lines are ± 1 standard deviation.

of 0.5 mm rather than 1 mm will increase the sorting time due to retention of coarser sediment and increased numbers of animals (2-3 times more animals, Parr et al., 1978; Lewis & Stoner, 1983), it will generally ensure adequate sampling of smaller macrofaunal species and juveniles of the macrofauna.

With respect to the location of sample transects, it is suggested that wherever possible, transects be established to correspond with Florida Department of Natural Resources Coastal Construction Control Line Monuments. This will provide a ready spatial reference system which could be used to allow other investigators to more easily resample previous projects.

In selection of specific sites to be sampled along a given transect, sites have been established based on fixed distances from shore (Matta, 1977; Spring, 1981; Diaz & DeAlteris, 1982; Nelson & Gorzelany, 1983; Knott et al., 1983), a fixed depth (Parr et al., 1978), or at general areas along the shore contour such as the swash zone or first offshore bar (Saloman, 1976; Marsh et al., 1980; Saloman & Naughton, 1984). A combination of these methods is suggested. Sampling an offshore zone rather than a fixed distance may result in considerable variability in the location of samples, yet a fixed interval may be inadvisable for samples in the swash zone. Therefore, as suggested above, swash zone samples should be sampled in the region at the base of wave run-up on the beach. Offshore, subtidal locations should be selected to represent different zones such as the inshore trough and bar, but sample location should be determined from sample time to sample time by a fixed distance measured on a transect line. The actual distances will therefore be determined by the location of physical features of the beach and not by a priori fixed intervals. Because of the typically limited spatial extent of a beach nourishment project, fixed distances established in this way should sample equivalent zones for both nourishment and control areas. If this condition is not met, it will be more important to sample equivalent zones than equal distances and the sampling points must be adjusted accordingly.

The temporal interval of sampling is extremely important, but as with all other study aspects, has varied widely. Also important is the timing of sampling with relation to the nourishment event. Sampling has ranged from one sample period seven years after nourishment (Marsh et al., 1980) to approximately quarterly (Parr et al., 1978; Nelson & Gorzelany, 1983) to monthly (Reilly & Bellis, 1978, 1983) to the superhuman effort of weekly sampling intervals (Saloman & Naughton, 1984). All studies cited except that of Marsh et al. (1980) covered before and after the nourishment event. Although desirable immediately following beach nourishment, weekly sampling will be prohibitive in terms of sample processing time in almost all cases. An indication of the effort involved is the fact that the study of Saloman & Naughton (1984) was published nearly 8 years after the nourishment project it describes. Quarterly sampling is insufficient since sand beach populations can undergo rapid shifts in periods much less than 3 months (Saloman, 1976; Reilly & Bellis, 1978, 1983; Nelson, unpublished data). A recommended sampling scheme is to sample monthly for three months before nourishment, weekly for one month following nourishment and monthly thereafter for 9-12 months. The few studies available suggest that this time interval should adequately cover any important changes (Parr et al, 1978; Reilly &

Bellis, 1978, 1983; Nelson & Gorzelany, 1983; Saloman & Naughton, 1984).

Processing of beach samples is generally well treated in Hurme et al. (1979). A particularly effective method of processing samples in the field is the use of individual 0.5 mm mesh nylon screens for sieving and retention of samples. Contents of a core sample are emptied onto a screen of sufficient size to safely hold the core contents (e.g. 25 x 25 cm). The screen is folded over the sample to form a sack, grasped firmly at the top and sieved either in a 20 l bucket of seawater on the beach or in the surf if conditions permit. After sieving, a label is added to the retained contents, the screen is twisted and securely tied with twine, and the whole mesh sack so made can be preserved. If relaxation of animals prior to preservation is desired, screens may first be immersed in a 20 l plastic bucket with lid containing a relaxant (e.g. 6% magnesium chloride or magnesium sulfate in seawater, Hurme et al., 1979; or 0.15% propylene phenoxetol (2-phenoxyethanol) in seawater, McKay & Hartzband, 1970). After relaxation, screens can be transferred directly into a second bucket of buffered 10% formalin-seawater solution for fixation. Subsequent processing can be carried out as given by Hurme et al. (1979).

One improvement in sample processing may be the use of an air-lift sample sorting device (Allenbaugh & Nelson, in prep.). For sandy sediments, such a device has proved nearly 100% efficient at removing all groups of organisms except those with a heavy shell from most of the sediment retained on a 0.5 mm screen. Unfortunately, bivalves, which are typically dominant on sandy beaches are not effectively removed by this method and must still be sorted almost entirely by hand.

The selection of the appropriate number of replicate samples and the subject of preliminary sampling are closely related. Green (1979) strongly recommended carrying out preliminary sampling for the purposes of 1) evaluating sampling design and statistical options, 2) verifying the sampling method is in fact sampling as expected, 3) checking that the sample size is appropriate to the organisms being studied and 4) determining the number of replicate samples needed to obtain the desired precision in estimates of mean abundances and number of species. The latter point is probably the most difficult of all. As Green (1979) points out, the best sample number is always the largest sample number; however, this is seldom true when cost and processing time are considered. Hartnoll (1983) points out that although sandy beaches present a homgeneous appearance, they are not homogeneous, and even relatively large total sample areas may still give relatively imprecise estimates of mean parameter values. One approach to estimating number of samples has been to plot cumulative number of species recorded for successive addition of samples, using the point where a plateau is reached as an estimate of suitable sample size (e.g. Hurme et al., 1979; Saloman, 1976). Based on analysis of one such set of curves for subtidal benthos, Hurme et al (1979) suggested 8 replicate cores of 15 cm diameter were a reasonable compromise for a beach study. Hartnoll (1983) has pointed out that this type of plot must be carried out by using a computer to consider all possible sequences of samples, since the conclusion will be dependent on the sequence. The utility of this approach which is generally carried out only once, during preliminary sampling, seems questionable since the relationship will certainly change seasonally. The second approach, which appears to be rarely used in practice, is to compute the number of samples required to estimate some variable (e.g., mean abundance) with a given precision, where precision is the closeness of repeated measurements of the same quantity. Methods for this determiation are discussed by Gonor & Kemp (1978) and Green (1979). Parr et al. (1978) provide estimates of the number of samples needed to obtain 50 and 30% precision levels at a 95% confidence level. This gives the number of samples required such that 95% of the time the true mean will fall within ± the precision percentage (e.g. 30%) of the measured mean. To estimate abundance, Parr et al. (1978) calculated that eighty-five 8 cm diameter cores were needed in the intertidal zone of their California study area, but only an average of 12.5 cores were needed subtidally. At the 50% precision level, the number of cores needed dropped to 49 and 6.5, respectively. For estimating mean number of species at the 30% precision level, 44 cores intertidally and 9 cores subtidally were needed, compared with 17 and 5.5 at the 50% precision level. These numbers point out that the smaller the change which one wishes to detect, the larger the number of replicates must be. In the specific case of the study done by Parr et al., (1978), the intertidal samples spanned the entire width of the beach, which would greatly increase the inherent variation in sampling and therefore greatly increase the number of replicates needed to obtain a given level of precision. Both Green (1979) and Sokal & Rohlf (1981) provide methods for estimating the number of replicates needed for detecting a specified percentage difference between means. In both cases, preliminary sampling is necessary in order to estimate the underlying standard deviation for the population being sampled.

It is not yet clear whether or not the use of precision levels to determine replicate number for evaluation of beach nourishment impact is feasible in routine practice. Estimated variation from the preliminary sampling may itself vary considerably from season to season, thus generating different estimates of required replicate number. Given the results of Parr et al.(1978), it would appear that obtaining precision levels better than 50% will be prohibitive in terms of the cost and time involved to process samples. At this level, ten 8 cm diameter cores would appear to offer precision somewhat better than 50% for both estimates of abundance and number of species. The suggestion of Hurme et al. (1979) that eight 15 cm diameter cores would be adequate was based on samples from a station at 20 m depth (Oliver & Slattery, 1976) and appears to be unnecessarily large for shallow water and sandy beach stations. Ten 8 cm diameter cores sample an area of 502 cm^2, which is within the plateau region for species-area curves calculated for sand beaches by Reilly & Bellis (1978, 1983) and Saloman (1976). It is therefore suggested that ten 7.6 cm diameter cores from each combination of location x time may be a sufficient level of replication. If time and budget permit, it would still be wise to take a larger number of samples during preliminary sampling to inspect the species-area curves. Physical pooling of replicate samples should generally be avoided (Green, 1979).

The basic data necessary to monitor biological changes consists of species lists and faunal density estimates taken before and after disturbance at the affected area and an undisturbed adjacent control area. Monitoring requirements for various projects have tended to require a variety of elaborations of these basic data such as:

1) biomass,
2) species richness indices,
3) species equitability indices,
4) species diversity indices, or
5) any of several similarity indices for qualitative or quantitative comparisons of stations.

It is debateable whether such data transformations provide any advantage since the various diversity and richness indicators are highly correlated with the data provided by the simple density estimates and species lists from which they are derived (Hurlbert, 1971; Loya, 1972; Poole, 1974; Green, 1979). Green (1979) has succintly reviewed the use of indices in biological studies and concludes that simple measures such as the number of species are biologically more meaningful, less ambiguous, and often are better indicators of biological change than more complex indices such as the commonly used index of diversity H'. Since calculations of these complex indices also require greater time, their use should be discouraged in beach nourishment monitoring.

When analysing data by parametric statistical methods, it is imperative that assumptions of such tests be met. As Underwood (1981) points out, this step is often omitted by investigators despite the fact that it may have serious consequences for the conclusions reached. The most important test is to insure that homogeneity of variances exists, although assumptions of normality of the data and independence of variances and means are also significant (Green, 1979). Green (1979), Sokal & Rohlf (1981), Underwood (1981) and most basic statistics books clearly discuss the tests of these assumptions and the methods for correcting for violations in assumptions to allow satisfactory statistical analysis.

Summary of Recommendations

1) A general review of data on the effects of beach nourishment on the sand beach macrofauna suggests that minimal biological effects result from beach nourishment. Some mortality of organisms may occur where grain size is a poor match to existing sediments, however recovery of the beach system appears to be rapid. Turtle nesting may be negatively affected due to sand compaction for a period of about one year.

2) Because available studies specifically analysing biological effects of beach nourishment are few and often have design or analysis flaws, further biological monitoring of beach nourishment should be continued until adequate data for deciding whether monitoring is necessary can be made.

3) Nourishment sand should match natural sand as closely as possible in grain size distribution and chemical characteristics.

4) The upper beach should be plowed following nourishment to counteract compaction in turtle nesting areas.

5) Avoid covering dune vegetation with sand, and whenever possible avoid nourishment during turtle nesting season. Relocate turtle nests if nourishment must take place during the nesting season.

6) Biological monitoring techniques should be standardized along the lines given below.

   a) Locate sample stations to provide as much interspersion of treatments as possible.

   b) Two-way analysis of variance appears to be one of the better statistical analysis options. Data must be checked to verify that the assumptions of this test are met.

   c) Hand-held corers offer the optimum combination of ease of use, cost and sampler efficiency.

   d) Use of 7.6 cm diameter cores is recommended.

   e) Restriction of faunal analysis to the macrofauna (animals retained on a 0.5 mm sieve) is recommended.

   f) Use Florida Dept. of Natural Resources coastal construction control line markers to establish sample locations.

g) Sample a transect from the swash zone to the region offshore where the first sand bar is located. Do not sample the high tide line. Sample the swash zone using a moveable frame of reference to make sure samples are consistently taken from the area of wave run-up. Establish stations in the inshore trough and offshore bar areas using a fixed transect line to assist in locating positions.

h) A recommended sampling scheme is to sample monthly for three months before nourishment, weekly for one month following nourishment and monthly thereafter for 9 - 12 months.

i) Ten 7.6 cm diameter cores from each level of location and time are suggested as providing sufficient replication.

j) Use of diversity indices for data analysis is not recommended.

k) If time and budget permit, preliminary sampling is recommended to verify that the above suggestions are adequate for a given location.

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