Assessing the Environmental Impacts of Beach Nourishment

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With sea levels rising under global warming, dredge-and-fill programs are increasingly employed to protect coastal development from shoreline erosion. Such beach "nourishment" can bury shallow reefs and degrade other beach habitats, depressing nesting in sea turtles and reducing the densities of invertebrate prey for shorebirds, surf fishes, and crabs. Despite decades of agency-mandated monitoring at great expense, much uncertainty about the biological impacts of beach nourishment nonetheless exists. A review of 46 beach monitoring studies shows that (a) only 11 percent of the studies controlled for both natural spatial and temporal variation in their analyses, (b) 56 percent reached conclusions that were not adequately supported, and (c) 49 percent failed to meet publication standards for citation and synthesis of related work. Monitoring is typically conducted through project promoters, with no independent peer review, and the permitting agencies exhibit inadequate expertise to review biostatistical designs. Monitoring results are rarely used to scale mitigation to compensate for injured resources. Reform of agency practices is urgently needed as the risk of cumulative impacts grows.

Keywords: beach nourishment, biological impacts, monitoring, public trust, sampling design

Beaches are in a constant state of flux, accreting and eroding in response to waves, currents, winds, storms, and sea-level change. As a consequence, development along ocean shorelines entails substantial risk of property loss. In recognition of the vulnerability of coastal development to shoreline erosion and flooding, and in response to the value of fish and wildlife habitat, the US Congress passed the Coastal Barrier Resources Act in 1982 to discourage overdevelopment of largely undeveloped coastal barriers along the Atlantic and Gulf coasts (Wells and Peterson 1986). Under incentives from the federal Coastal Zone Management Act, individual states have also developed coastal management programs that establish setbacks and impose other restrictions on development along ocean beaches. Nevertheless, development on coastal barriers has burgeoned dramatically.

As escalating rates of global warming lead to more rapid rise in sea level and greater frequency and intensity of storms, demand for engineered solutions to shoreline erosion is intensifying (Barth and Titus 1984). Massive dredge-and-fill projects have become a common method of combating shoreline retreat. Between 1922 and 2003, beginning with the first beach nourishment at Coney Island, New York, at least 970 projects have "nourished" more than 6050 kilometers (km) of US shoreline along the Atlantic and Gulf coasts, using 430 million cubic meters (m^3) of fill (*www.nicholas.duke.edu/psds/nourishment.htm*). During nourishment, sediments from a dredge site or terrestrial source are added to the beach to elevate it and extend it seaward. Unlike seawalls and groins that act only to harden the shoreline or redistribute sediment, nourishment temporarily adds sediment to the beach system (Bush et al. 2004). State resource agencies' preference for beach nourishment to combat shoreline retreat is motivated by a well-founded desire to avoid the negative impacts of hardened structures on the recreational and biological habitat values of ocean beaches; however, any presumption that nourishment projects are ecologically benign is derived from an incomplete and flawed body of science.

The sand beach represents a productive and unique habitat supporting the seasonal nesting of threatened and endangered sea turtles and dense concentrations of benthic invertebrates that feed surf fishes, resident and migrating shorebirds, and crabs (Brown and McLachlan 1990). The beach and nearshore coastal habitats are substantially disturbed by and can be functionally degraded through the process of nourishment. Permits for beach nourishment projects in the United States have routinely required monitoring of biological resources on the beach and at the dredging site. Despite decades of monitoring and scores of reports (reviewed by Nelson 1993), much uncertainty persists about the ecological impacts of nourishment and how to minimize and mitigate them. Here we conduct a synthesis of the

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sampling designs, statistical analyses, and bases for interpretations across 46 studies done to assess the ecological impacts of beach nourishment. Using this synthesis, and an overview of agency practices in permitting beach nourishment, we help to explain why so much effort at such high cost has led to so little progress toward understanding and predicting ecological impacts, and we suggest some remedies.

Methods of assessing study designs

We searched for all available reports, publications, and theses evaluating biological impacts of beach nourishment. The search, which was restricted to the United States so as to include only studies affected by the same federal framework of environmental policy, was facilitated by Nelson's (1993) identification of early unpublished reports, the category that still constitutes the large majority of this literature. Assuming that the peer-reviewed, university-examined, and more widely cited gray literature tends to be of higher quality than unpublished reports that are only locally available and not readily accessible, the statistics based on our sample most likely understate the frequencies of study deficiencies. Each of us independently reviewed every study and answered the same set of questions about its subject matter and the scientific basis for its conclusions. The few (< 5 percent) disagreements between our findings proved to be caused by misinterpretation, which was resolved by reexamining the documents. Where the same study was produced in multiple forms (e.g., as an unpublished report and as a refereed paper), we considered only the most critically reviewed version.

Studies were characterized by decade of initiation, type (gray literature report, thesis, or published paper), process of interest (dredging or filling), geographic location, target biota (soft-bottom or hard-bottom macroinvertebrates, fish, sea turtles, or shorebirds), and approach (observational monitoring, controlled experimentation, or modeling). For each type of target biota, we computed how frequently each of a series of physical habitat variables and potential biological responses was assessed. Finally, we evaluated the sampling designs, statistical analyses, and bases for interpretations and conclusions in each of the studies by applying fundamental principles of statistical inference (as exemplified by Schmitt and Osenberg 1996 and Underwood 1997) to answer a series of questions (see box 1). The standard applied to the last question on scholarship was that of *Marine Ecology Progress Series,* an international journal appropriate for the topic and for which one of us (C. H. P.) has served as editor for two decades.

Characterization of studies

Our sample of available studies (table 1) was dominated by unpublished reports (59 percent). Although anonymous scientific peer review has been widely endorsed as the most reliable means of ensuring rigor (NRC 2000), this process is not applied to environmental monitoring proposals or final reports that are mandated by permitting agencies on behalf of public trust resources. Our review illustrates a tendency of temporally increasing publication of the impact assessment studies from the 1970s, when they first appeared (7 percent published in peer-reviewed journals), to the 1990s (33 percent). These absolute percentages are most likely biased upward because gray-literature reports of limited circulation are more difficult to find and therefore underrepresented in our review.

Of the 46 studies assessed, 83 percent were conducted along the Atlantic coast and 13 percent along the Gulf coast of the United States. Most came from Florida (29 studies), reflecting a concentration of nourishment projects along its 200-km southeastern coast, where at least 50 million m^3 of sediment were deposited on beaches between 1960 and 2000 (Bush et al. 2004). Other states represented are North Carolina (9 studies), South Carolina (4), New Jersey (2), California (1), and Michigan (1).

Benthic invertebrates were the most frequently targeted organisms (78 percent of all studies), reflecting their suitability as ecological indicators. Benthic invertebrates are relatively sessile (therefore allowing spatial patterns to imply causation), can be sampled quantitatively without high cost, are well described taxonomically, and reveal ecologically meaningful and important patterns, even at coarse levels of taxonomic discrimination (Warwick 1988). Few assessments of beach nourishment have considered its impacts on demersal fishes (33 percent), and even fewer have considered impacts on shorebirds (4 percent), although both these groups of organisms have value to humans and provide ecosystem services.

Only one assessment (Manning 2003) employed experimental manipulations, widely acknowledged as the most rigorous means of inferring causation in ecology (Paine 1977), and none employed modeling (tables 2, 3), the most widely accepted tool for evaluating the dynamics of fish populations (Hilborn and Walters 1992). Thus, two of the most powerful scientific tools are routinely overlooked in favor of purely observational monitoring. Monitoring can be a seriously flawed means of testing impacts, because of uncontrolled, confounded factors that often taint inferences (see Connell's 1974 discussion of "natural experiments"). Inferences reached by comparing results of separate monitoring studies are particularly tenuous, because in none of these contrasts does only a single factor differ among studies.

The physical habitat and biological response variables commonly monitored in beach nourishment projects (tables 2, 3) include many of relevance. Among those that monitored habitat condition, varying percentages of studies measured turbidity; mean grain size; sediment grain-size distribution; surface cover by hard substrata (shells, limestone, etc.); sediment mineralogy, organic content, and compaction; surface topography; and habitat damage from gear contact. The biological responses assessed included total abundance of the entire biotic assemblage, abundance of component taxa, total biomass, biomass of selected taxa, size–frequency distribution of selected taxa, various species diversity indices, community composition, and some measure of physiological status of an important species (table 4). Nevertheless,

Table 1. Characterization of 46 studies monitoring ecological impacts of beach nourishment.

Authors	Date of document	Decade of initiation	Type	Process	Coast (state)	Target biota[a]	Study type
Blair SM, Flynn BS, Markley S	1990	1980s	Paper (nonrefereed)	Dredge	Atlantic (FL)	Macroinvertebrates (hard)	O
Blair S, Flynn B, McIntosh T, Hefty L	1990	1990s	Report (final)	Dredge	Atlantic (FL)	Macroinvertebrates (hard)	O
Bowen PR, Marsh GA	1988	1970s	Report (final)	Dredge	Atlantic (FL)	Macroinvertebrates (soft)	O
Broadwell AL	1991	1980s	Master's thesis	Fill	Atlantic (FL)	Sea turtles	O
Burlas M, Ray G, Clarke D	2001	1990s	Report (final)	Both	Atlantic (NJ)	Fish, macroinvertebrates (soft)	O
Charvat DL	1987	1980s	Master's thesis	Fill	Atlantic (FL)	Macroinvertebrates (soft)	O
Coastal Science Associates Inc	2002	2000s	Report (interim)	Both	Atlantic (NC)	Fish, macroinvertebrates (soft)	O
Courtenay WR Jr, Hartig BC, Loisel GR	1980	1970s	Report (final)	Dredge	Atlantic (FL)	Fish	O
Culter JK, Mahadevan S	1982	1970s	Report (final)	Both	Gulf (FL)	Macroinvertebrates (soft)	O
Davis RA Jr, FizGerald MV, Terry J	1999	1990s	Paper (refereed)	Fill	Atlantic (FL)	Sea turtles	O
Deis DR, Spring KD, Hart AD	1992	1980s	Report (final)	Both	Gulf (FL)	Macroinvertebrates (soft)	O
Dodge RE, Hess S, Messing C	1991	1980s	Paper (nonrefereed)	Both	Atlantic (FL)	Macroinvertebrates (soft, hard)	O
Donoghue CR	1999	1990s	PhD thesis	Fill	Atlantic (NC)	Macroinvertebrates (soft)	O
Fisher LE, Dodge RE, Messing CG, Goldberg WM, Hess S	1992	1990s	Paper (nonrefereed)	Both	Atlantic (FL)	Macroinvertebrates (soft, hard)	O
Goldberg WM	1985	1980s	Report (final)	Both	Atlantic (FL)	Macroinvertebrates (soft, hard)	O
Gorzelany JF, Nelson WG	1983	1980s	Paper (refereed)	Fill	Atlantic (FL)	Macroinvertebrates (soft)	O
Hayden B, Dolan R	1974	1970s	Paper (refereed)	Fill	Atlantic (FL)	Macroinvertebrates (soft)	O
Holland HT, Chambers JR, Blackman RR	1986	1970s	Report (final)	Both	Gulf (FL)	Fish	O
Johnson RO, Nelson WG	1985	1980s	Paper (refereed)	Dredge	Atlantic (FL)	Macroinvertebrates (soft)	O
Jutte PC, Van Dolah RF, Levisen MV	1999	1990s	Report (final)	Dredge	Atlantic (SC)	Macroinvertebrates (soft)	O
Jutte PC, Van Dolah RF, Levisen MV	1999	1990s	Report (final)	Fill	Atlantic (SC)	Macroinvertebrates (soft)	O
Lindeman KC, Snyder DB	1999	1990s	Paper (refereed)	Fill	Atlantic (FL)	Fish	O
Manning LM	2003	1990s	PhD thesis	Fill	Atlantic (NC)	Fish, macroinvertebrates (soft)	O, E
Marsh GA, Bowen PR, Deis DR, Turbeville DB, Courtenay WR Jr	1980	1970s	Report (final)	Fill	Atlantic (FL)	Macroinvertebrates (soft, hard)	O
Nelson WG, Collins G	1987	1980s	Report (final)	Fill	Atlantic (FL)	Fish, macroinvertebrates (soft)	O
Nelson DA, Mauck K, Fletemeyer J	1987	1980s	Report (final)	Fill	Atlantic (FL)	Sea turtles	O
Nester RT, Poe TP	1982	1980s	Report (final)	Fill	Great Lakes (MI)	Fish, macroinvertebrates (soft)	O
Parr T, Diener D, Lacy S	1978	1970s	Report (final)	Fill	Pacific (CA)	Macroinvertebrates (soft)	O
Peterson CH, Hickerson DHM, Johnson GG	2000	1990s	Paper (refereed)	Fill	Atlantic (NC)	Macroinvertebrates (soft)	O
Posey M, Alphin T	2002	1990s	Paper (refereed)	Dredge	Atlantic (NC)	Macroinvertebrates (soft, hard)	O
Putt RE, Spring KD, Graham BD, Deis DR, Walensky RE, Rudolph HD	1984	1980s	Report (final)	Both	Atlantic (FL)	Macroinvertebrates (soft, hard)	O
Rackocinski CF, Heard RW, LeCroy SE, McLelland JA, Simons T,	1996	1980s	Paper (refereed)	Fill	Gulf (FL)	Macroinvertebrates (soft)	O
Raymond PW	1984	1980s	Master's thesis	Fill	Atlantic (FL)	Sea turtles	O
Raymond B, Antonius A	1977	1970s	Report (final)	Dredge	Atlantic (FL)	Macroinvertebrates (hard)	O
Reilly F Jr, Bellis VJ	1983	1970s	Report (final)	Fill	Atlantic (NC)	Fish, macroinvertebrates (soft)	O
Rumbold DG, Davis PW, Perretta C	2001	1990s	Paper (refereed)	Fill	Atlantic (FL)	Sea turtles	O
Ryder CE	1993	1990s	Master's thesis	Fill	Atlantic (FL)	Sea turtles	O
Salomon CH, Naughton SP, Taylor JL	1982	1970s	Report (final)	Dredge	Gulf (FL)	Macroinvertebrates (soft)	O
Salomon CH, Naughton SP	1984	1970s	Report (final)	Dredge	Gulf (FL)	Macroinvertebrates (soft)	O

(*continued*)

(Table 1, continued)

Authors	Date of document	Decade of initiation	Type	Process	Coast (state)	Target biota[a]	Study type
Spadoni RH, Campbell TJ	1981	1970s	Report (final)	Dredge	Atlantic (FL)	Fish, macroinvertebrates (soft)	O
Turbeville DB, Marsh GA	1982	1970s	Report (final)	Dredge	Atlantic (FL)	Macroinvertebrates (soft)	O
Van Dolah RF, Wendt PH, Martore RM, Levisen MV, Roumillat WA	1992	1990s	Report (final)	Both	Atlantic (SC)	Fish, macroinvertebrates (soft)	O
Van Dolah RF, Martore RM, Lynch AE, Wendt PH, Levisen MV, Whitaker DJ, Anderson WD	1994	1990s	Report (final)	Both	Atlantic (SC)	Fish, macroinvertebrates (soft)	O
Versar Inc.	2003	2000s	Report (final)	Fill	Atlantic (NC)	Fish, macroinvertebrates (soft), shorebirds	O
Versar Inc.	2004	2000s	Proposal	Both	Atlantic (NC)	Fish, macroinvertebrates (soft), shorebirds	O
Wilber DH, Clarke DG, Ray GL, Burlas M	2003	1990s	Paper (refereed)	Fill	Atlantic (NJ)	Fish	O

E, experimental study with controlled manipulations; O, observational monitoring.

a. "Soft" or "hard" refers to whether the bottom organisms assessed occurred on sediment (soft) or limestone reef (hard) bottom.

disturbingly high percentages of assessment studies (25 to 38 percent for dredging and 17 to 80 percent for filling) failed to measure any habitat variable (tables 2, 3). Despite the need for dredging contractors to monitor topography as a permit condition and as a measure of performance, this important habitat characterization was not routinely reported in the impact assessment documents. Its complete omission from dredge sites (table 2) is especially critical because creation of deeper pits induces fine sedimentation, which can inhibit recovery of the natural benthic invertebrate community for years (Rakocinski et al. 1996).

Although studies frequently measured relevant aspects of physical habitat condition, only sedimentation rate, out of many potentially important physical processes, was estimated with any appreciable frequency (tables 2, 3). Sediment transport, erosion of fine sediments off the beach face, dynamics of turbidity plumes, concentration of large shells, and other physical processes likely to influence the biota and affect recovery went without evaluation in any impact study. Few studies measured changes in body size within species, which can indicate mode of recolonization (larval transport and settlement versus migration of older stages; table 4). Measurements of biological processes such as burrowing and predation rates have been reported from only one study (Manning 2003) and recruitment rate from one other (Lindeman and Snyder 1999). Gut contents of fish were only occasionally measured as an indication of feeding success (included in physiological status; table 4). Despite the scientifically compelling advice of Nelson (1993) to avoid use and risky interpretation of diversity indices, this practice is still common in beach nourishment studies (table 4) and still without rigorous conceptual support (Hurlbert 1971). When the simple, more readily interpretable species richness is measured (matching current usage in basic ecology), the necessary adjustments for statistical dependency on abundance (Hurlbert 1971) are missing.

Sampling design

Our synthesis of sampling designs reveals numerous inadequacies that seriously compromise the studies' results and conclusions (box 1). Researchers engaged in field sampling to estimate biotic abundances usually used appropriate devices, but the 39 percent incidence of failure to employ the least biased gear would be viewed as unacceptably high for any scientific granting agency. The most frequent violation came from the use of grab samplers instead of cores to sample soft-sediment invertebrates. This results in failure to sample to the full depth of occupation of the sediment column. Furthermore, those sedimentary strata that are included in a grab sample are not sampled equally (in contrast to a core, which projects its surface area downward), making it impossible to estimate density accurately. Grab samples also include varying amounts of sediments per sample, depending on bottom hardness and on obstructions such as shells.

A serious shortcoming in the sampling designs of most studies was the failure to consider both natural spatial variation and natural temporal variation on multiple scales so as to craft a sampling design that minimized unexplained error variance and prevented confounding of sources of variation (Green 1979, Underwood 1997). Although 26 percent of the studies included sampling in a spatially nested and 30 percent in a temporally nested design, not one incorporated this information into a nested analysis so as to partition out scales of natural variability.

Few of the monitoring studies of beach nourishment employed a priori power analyses of any sort (9 of 46 contained inappropriate power analyses, and only 1 contained an appropriate analysis) to help determine how much replication was required to detect an effect of biologically meaningful magnitude. Only one study (Wilber et al. 2003) employed post hoc power analysis to quantify the magnitude of the effect that could be detected and thereby provide insight into how to interpret an absence of a statistically significant difference. This power analysis showed that the design could detect only threefold or greater differences in surf fish abundance, which obviously did not provide much resolution. A large fraction (62 percent) of past assessments of ecological impacts of beach nourishment possessed sampling designs without adequate power to detect effects of importance (defined as an ability to detect with 80 percent probability a decline of approximately 50 percent or an increase of approximately 100 percent).

Most, but not all, studies included sampling of control sites, but there was a relatively high incidence of potential violations of the basic principle of independent controls (box 1). Few beach nourishment studies followed the sound statistical advice (Hurlbert 1984) of interspersing treatments and controls to avoid spatial interdependence. This is due, in part, to the lack of replication of fill sites in most projects, because sediments are typically deposited along a continuous stretch of the shoreline (Nelson 1993). Despite the common practice of extracting sediments for nourishment from multiple dredge sites, interspersion of treatments and controls to determine the impacts of sediment mining is also rare. Where the lack of replication of disturbed sites prevents a fully interspersed design, bracketing of the disturbed site with controls on both sides is the next best option. This was done in 11 of 35 controlled studies examining the impacts of filling. Often, however, the putative control site was located too near the fill site, so that impacts transported by physical along-shore processes probably modified the control at least at one end of the beach (Hayden and Dolan 1974). Absent a gradient design that spaces sites at varying distances away from the fill site, rigorously identifying when a putative control has been compromised and quantifying the spatial extent of impact is difficult or even impossible. A gradient design has been employed in only one assessment of fill impacts (Hayden and Dolan 1974). In some studies, sites that had been recently nourished were then used to represent controls for subsequent nourishment (Burlas et al. 2001). Such a design violates the concept of a control and should be avoided to prevent the bias of underestimation of impacts of nourishment.

Table 2. Physical habitat variables most frequently measured in studies of the impacts of dredging practices as part of beach nourishment.

(percent)

Environmental variable	Macroinvertebrates Soft bottom (*n* = 16)	 Hard bottom (*n* = 8)	Fish (*n* = 6)
Turbidity	13	25	33
Sedimentation	13	50	0
Mean grain size	56	13	17
Sediment grain-size distribution	56	38	33
Surface cover by hard substrata	0	0	0
Sediment mineralogy	25	0	17
Organic content of sediment	44	13	33
Sediment compaction	0	0	0
Topography	0	0	0
Habitat damage from gear contact	0	25	0
No habitat variables measured	25	38	33

The duration of monitoring in these studies was frequently insufficient to characterize the biota before nourishment or to demonstrate the duration of habitat and biological impacts afterward. Sampling before the disturbance occurs should be sufficient to characterize natural preexisting differences between treatment and control sites in physical habitat and biotic systems (Stewart-Oaten et al. 1986). Frequently, studies did not adequately anticipate the nourishment project, and permit-granting agencies failed to delay the project to allow initial biotic characterization during relevant productive seasons. Eighty-seven percent of monitoring studies, with

Table 3. Physical habitat variables most frequently measured in studies of the impacts of filling practices as part of beach nourishment.

(percent)

Environmental variable	Macroinvertebrates Soft bottom (*n* = 27)	 Hard bottom (*n* = 5)	Fish (*n* = 9)	Sea turtles (*n* = 6)	Shorebirds (*n* = 2)
Turbidity	33	0	44	0	0
Sedimentation	4	20	0	0	0
Mean grain size	59	20	11	17	100
Sediment grain-size distribution	52	20	22	33	0
Surface cover by hard substrata	7	0	22	0	100
Sediment mineralogy	26	0	0	0	0
Organic content of sediment	30	0	0	0	0
Sediment compaction	4	0	0	67	0
Topography	52	0	33	0	100
Habitat damage from gear contact	0	0	0	0	0
No habitat variables measured	11	80	22	17	0

Table 4. Biological response variables most frequently assessed in monitoring studies of beach nourishment.

(percent)

	Macroinvertebrates				
Biological response variable	**Soft bottom (*n* = 32)**	**Hard bottom (*n* = 9)**	**Fish (*n* = 15)**	**Sea turtles (*n* = 6)**	**Shorebirds (*n* = 2)**
Total abundance	88	78	87	17	100
Taxon-specific abundance	84	89	100	17	100
Total biomass	6	0	13	0	0
Taxon-specific biomass	6	0	27	0	0
Size-frequency distribution	6	11	40	0	0
Diversity index	78	22	60	0	0
Community composition	41	0	33	0	0
Physiological status	6	11	40	67	0

an average duration of 1.5 years, were terminated before recovery of the affected biological resource was demonstrated.

Statistical analyses

The statistical analyses done on the monitoring data to test for biological impacts suffer from critical flaws in the basic sampling design; from improper analyses that do not match the sampling design; from failure to fully explain, justify, and report on the analyses; and from unjustified interpretations (box 1). A disappointing 27 percent of the beach nourishment studies conducted no formal statistical tests of significance.

Many studies (70 percent of the 33 studies that employed formal statistical testing) failed to include all appropriate independent factors or did not test for significance of all the measured response variables. The most serious analytic deficiency was the almost universal failure (in 41 of 46 studies) to isolate estimates of impact from confounding contributions of natural spatial and temporal variation by using a BACI (before–after, control–impact) type of analysis (Green 1979, Stewart-Oaten et al. 1986). The tests for biological impacts generally used either a spatial contrast among control and disturbed sites or a temporal contrast across time at sites before and after dredging or filling. Use of only spatial contrasts requires that no natural spatial variation exists between control and disturbed sites, a generally flawed assumption, since the structure of macrobenthic assemblages varies according to the morphology of beaches at scales of tens to hundreds of meters (Barros et al. 2002). Using a temporal contrast to evaluate the impacts of nourishment requires the assumption that the response variable (typically organism density) would remain constant over time in the absence of any impact. This assumption is violated by benthic invertebrate populations, whose seasonal variation is quite dramatic on sandy beaches (Manning 2003). Even if season is controlled, interannual differences can be large, confounding the ability to use temporal contrasts to infer impacts. Interestingly, 50 percent of the studies of beach nourishment were designed with the potential for BACI-type analysis, yet failed to conduct this most appropriate and rigorous analysis of variance to reach conclusions unbiased by natural spatial and temporal change.

Despite the emergence of powerful methods of multivariate statistical analysis of community responses to perturbations (Clarke 1993), few studies of impacts of beach nourishment went beyond tests on separate taxa or totals of taxa. Those few studies that did conduct assessments of community response employed similarity indices to compare nourished and control biotas (typically restricted to the soft-sediment invertebrates). No study applied the gold standard of multivariate analysis, nonmetric multidimensional scaling (n-MDS), an ordination procedure that has been demonstrated to discriminate ecological patterns with far greater resolution than univariate responses (Clarke 1993). The software package for this analysis, PRIMER 5 (*www.pml.ac.uk/primer/*), also includes other routines that permit analysis of how well physical habitat variables explain biotic response patterns.

Most (84 percent) studies overlooked formal statistical analysis of how changes in a physical factor or process may have caused a biological response (box 1). This oversight is particularly important in the case of the benthic invertebrates, for which much basic biological research demonstrates that sedimentology can dictate community composition (Gray 1974). Often the physical factors are monitored by a consultant separate from the one who conducts the biological studies, thereby inhibiting coordinated sampling and joint analyses of sampling results. Partly as a consequence, no monitoring study of beach nourishment has critically assessed how mismatched sediments continue to serve as a press disturbance (Bender et al. 1984) after completion of the beach filling activity. (A “press disturbance” is one that continues to affect the biological system for some relatively long period of time, as contrasted with a “pulse disturbance,” which is a discrete event.) Beach nourishment is universally considered a short-term pulse disturbance, inappropriately viewed as analogous to natural sediment movements during a major storm. Uncharacteristically coarse sediments can be expected to remain for years on intertidal beaches and to become concentrated by wave action in the biologically most important zone, the swash zone, where they can continue to modify natural invertebrate abundance and community composition for years. Enhancement of the fraction of fine sediment during beach nourishment also has the potential to represent a press disturbance long after filling is completed, as wave energy over time erodes and exposes fill materials into which mud has been embedded and thereby continues to inject biologically deleterious turbidity into the surf zone. For both methodological and biological reasons, the biological impacts of elevating turbidity during and after beach nourishment are never properly assessed (Telesnicki and Goldberg 1995). The persistence of a veneer of sediments over a coral reef or hard-bottom habitat constitutes a press perturbation that can last at least as long as the typical 3- to 10-year interval between repeated nourishment projects (Lindeman and Snyder 1999).

Conclusions and interpretations

The conclusions of beach nourishment studies are often flawed by lack of compelling support from adequate evidence, analysis, or interpretation (box 1). In our sample, the authors of 73 percent of the studies misinterpreted at least some of their results. Few studies (22 percent) included attempts to interpret observed biological responses by appeal to mechanistic processes. The conclusions of 56 percent of studies lacked rigorous support from evidence and analysis, most often because the sampling design, the analyses, or both failed to control for both natural spatial and temporal variation. The failure to address the power of the study design also frequently led to unjustified conclusions of absence of impacts, when capacity to detect even large impacts was compromised by high natural variability and low replication. The scholarship of the science in these studies was poor. A large fraction (49 percent) of beach nourishment studies failed to do more than a superficial job of citing (0 to 10 citations) and synthesizing relevant scientific literature.

Agency practice and policy implications

Our review of studies of impacts of beach nourishment, mostly monitoring studies conducted as a condition for permits, reveals serious deficiencies. The widespread flaws in design, analysis, and interpretation help explain why so much uncertainty still persists over the ecological consequences of beach nourishment despite four decades of monitoring at substantial expense. Substandard biological monitoring of beach nourishment persists despite the publication of reviews that provide explicit guidelines for the variables that should be monitored and the spatial and temporal scales to consider (Nelson 1993, NRC 1995, Schmitt and Osenberg 1996, Greene 2002). Further detailed guidance required to produce a model study design to assess impacts of beach nourishment with rigor is implicit in our descriptions of study flaws (box 1).

Box 1. Frequency of flaws in sampling design, statistical analyses, and interpretations and conclusions of 46 studies monitoring biological impacts of beach nourishment.

Flaws in sampling design

- Failure to employ the least-biased device available: 39 percent
- Failure to incorporate both spatial and temporal variation: 48 percent
- No nested sampling of spatial patterns: 74 percent
- No nested sampling of temporal patterns: 70 percent
- Failure to consider full consequences of seasonal variation: 39 percent
- Lack of appropriate a priori power analyses: 98 percent
- Lack of appropriate post hoc power analyses: 98 percent[a]
- Inadequate power in design to detect large impacts: 62 percent[a]
- Absence of controls: 15 percent
- Controls not independent of treatment or each other: 36 percent[b]
- Insufficient duration of sampling to demonstrate recovery: 87 percent[a]

Flaws in statistical analyses

- Absence of formal statistical tests: 27 percent[a]
- Multiple factors confounded in tests: 70 percent[c]
- Absence of BACI (before–after, control–impact) analysis: 89 percent[a]
- No inclusion of multivariate testing of community composition: 67 percent[d]
- Failure to test linkages between physical habitat and biological responses: 84 percent[a]
- No testing to discriminate between a pulse and press disturbance: 100 percent[a]

Flaws in interpretations and conclusions

- Misinterpretation of statistical test results: 73 percent[c]
- Lack of credible mechanistic explanation for biological responses: 22 percent[a]
- Conclusions not properly supported by observations and statistical test results: 56 percent[a]
- Citation and synthesis of literature fails to meet minimal publication standards: 49 percent[a]

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a. $n = 45$ studies that, at the time of review, included results.
b. $n = 39$ studies with controls.
c. $n = 33$ studies that did statistical tests.
d. $n = 39$ studies that sampled multiple species.

Inadequate funding of basic process-oriented science in the beach ecosystem contributes to the prevailing high uncertainty involved in predicting biological impacts of beach nourishment projects. To extrapolate from the demonstration of any given nourishment project's impacts and make reliable predictions about future projects requires a basic understanding of the processes that drive the dynamics of the natural system at 1- to 10-km resolution, a typical length scale of nourishment. Although many monitoring studies are adequately funded for the narrow goal of assessing impacts, funding for interdisciplinary studies of fundamental processes in the natural beach system, such as coupled physical and biological consequences of relative sea-level rise, waves, currents, and storms, either has not been sufficient or has not been pursued by the basic science community. The US Army Corps of Engineers recently invested $8.6 million in an 8-year program monitoring the impacts of a New Jersey project (Burlas et al. 2001), a monitoring effort that, despite its cost, advanced basic understanding relatively little (Greene 2002). Funding at about twice this level and for the same duration could have created a model study of fundamental beach processes that would improve the generic capacity to predict impacts. Funding agencies for basic science and the scientists who apply for their research grants bear partial responsibility for this poverty of understanding. The National Science Foundation and NOAA (National Oceanic and Atmospheric Administration), through their coastal ocean programs, could provide targeted funding for physical–geological–biological process studies of beaches to enhance the necessary fundamental understanding that is now lacking. Such interdisciplinary studies, using observation, experimentation, and modeling of the nearshore ecosystem, also provide potential for incorporating the consequences of multiple stressors on a landscape scale and thereby approach the ideal of ecosystem-based management endorsed by the US Commission on Ocean Policy (US COP 2004). There is a need for more basic and holistic research on process as well as more rigor in project-specific impact analyses.

So why do the federal (US Army Corps of Engineers) and state permit-granting agencies that are responsible for carrying out the mandates of the National Environmental Policy Act (NEPA) fail to ensure prevailing standards of scientific rigor and thus to discharge their obligation to protect public trust resources? Partial answers to this critical question come from a consideration of the process by which the monitoring components of beach nourishment permits are developed. Neither the US Army Corps of Engineers nor the state permitting agency employs the anonymous scientific peer-review process that is central to insuring high standards of excellence in basic scientific research (NRC 2000). Unfortunately, the federal and state permit offices, in their approval of monitoring designs, do not demonstrate adequate expertise in the critical discipline of biostatistics to ensure that the studies meet high standards of scientific rigor. The absence of expert review and rereview in the approval process to achieve acceptable designs is made more serious by the recognition that the monitoring is typically designed and conducted by private contractors, usually associated with the proponents of the nourishment project, rather than by independent research organizations. Anonymous peer review is needed for environmental impact statements (EISs), environmental assessments (EAs), monitoring proposals, and final reports to induce consulting agencies to employ their expertise to elevate beach nourishment science to prevailing standards of scientific rigor. Towns, counties, and other local units of government cannot be expected to possess the technical expertise to ensure scientific and statistical analytic rigor in monitoring studies for beach nourishment: they trust the state and federal governments to perform that function, a trust that is misplaced.

Not only can environmental permitting agencies be criticized for failure to ensure that studies of environmental impacts of beach nourishment meet basic standards of rigor in science, but the justification for the permit decision and required monitoring can also be challenged on occasion. Permitting agencies often yield to political pressure for nourishment permits and justify allowing high or uncertain risks by arguing that the agency can improve future permit decisions by learning from monitoring this risky aspect of the project. Such an argument is disingenuous if there is insufficient biostatistical expertise on the agency's staff and no independent scientific peer-review process to guarantee the rigor needed to assess impacts effectively. Furthermore, if this were an honest motivation and not just a rationalization, then funding would be in order for directed research on whether the very aspect of the study that is under question is adequately tested. This would often involve funding well-designed experiments and population modeling to complement the monitoring.

The most important scientific challenge in meeting the obligations of NEPA is evaluating the potential cumulative impacts of multiple projects in the context of the growing impacts of other human activities on coastal ecosystems. With rising sea levels and enhanced storminess driving increased demand for beach nourishment and washing away the fill even faster, cumulative impact is a critical concern that is not adequately evaluated through the current process of simply attaching scientifically flawed monitoring requirements to individual permits. Not only must rigorous analysis of cumulative effects address the expanding scope of beach nourishment, but it must also include the consequences of multiple escalating stressors in this coastal zone. This is the essence of ecosystem-based management for coastal resources, an overarching recommendation of the US Commission on Ocean Policy (US COP 2004). Absent legitimate assessment of cumulative impacts, EISs and EAs done for beach nourishment projects will also continue to be chronically deficient (Lindeman 1997).

Federal and state permitting agencies also often allow the required assessment studies to evade evaluation of important and highly uncertain potential impacts on the grounds of intrinsic difficulty of monitoring. Impacts to fish populations fall in this category, because their mobility and high natural variation in space and time prevent direct or indirect impacts on population size from being detected by empirical sampling of any individual beach nourishment project. Full evaluation of potential impacts on fish populations would require population modeling based on rigorous observations of process, probably including experimental tests of mechanisms. Such modeling should be done on the relevant large spatial and long temporal scales that define population processes, an impossibility for empirical monitoring. No permit condition for beach nourishment has required such modeling, despite the central role that this approach plays in fisheries science, the great importance of surf zone and nearshore habitats to many valuable fish populations (Hackney et al. 1996), and the documented extreme damage that beach nourishment inflicts on invertebrate prey on beaches (Rakocinski et al. 1996, Peterson et al. 2000) and on reef habitat (Lindeman and Snyder 1999). Consequently, beach nourishment threatens essential fish habitats to an undetermined degree.

Environmental monitoring requires explicit goals. Without a defensible goal, monitoring becomes a tax on those who are paying for the project, functioning merely to sustain employment in consulting companies. Monitoring of beach nourishment should have two goals: first, to answer open questions about environmental impacts, and second, to quantify injury to public trust resources so as to allow compensatory mitigation. If the rigor of the science assessing impacts of beach nourishment were elevated through changes in agency process and through improved basic understanding of beach processes, the first of these motivations could disappear over time as the critical questions about environmental impact are answered. The second motivation should persist, except that only rarely now is mitigation ever required for habitat degradation, and never for injury to living resources arising from beach nourishment. Habitat mitigations that are now applied (e.g., rock removals from beaches by heavy equipment) are generally ineffective and typically involve intense disturbances likely to cause even more biological injury. Because restoring the natural granulometry of beach sands after filling with incompatible sediments may be impossible and retaining natural sediments is of such great biological significance (Nelson 1993), monitoring sediment size composition would best be done during the project. Then ongoing application of fill that fails to meet strict compatibility standards could be halted, and coarse components could be sieved out or fine ones winnowed out before completing the project.

Permitting any, let alone unlimited, filling and bulldozing of beach habitat without providing effective mitigation is inconsistent with regulatory treatment of other important habitats, such as salt marshes, seagrass beds, and coral reefs. NOAA requires compensatory mitigation for loss of ecosystem services to be funded by the party responsible for damage in other coastal habitats (Fonseca et al. 2000), a requirement somehow forgotten when beach ecosystem services are lost through nourishment. Restoring each injured species may not be feasible, and may require indirect measures such as protection of shorebird nests. However, some restorations could be achieved using aquaculture methods to reseed nourished beaches with lab-raised bivalves and those amphipods that lack pelagic dispersal to aid recolonization.

We suggest one solution to the challenge of how to make fundamental changes in the permitting process at federal and state levels so as to ensure compliance with NEPA and protection of public trust resources. The piecemeal project-by-project approach to assessing impacts and (rarely) providing mitigation for impacts should be replaced by a centralized program analogous to the wetland mitigation banking programs present in many states. Appropriate levels of monitoring and mitigation charges could be assessed to each project and paid into a single fund. The money could be used to fund research proposals addressing impacts of beach nourishment that are reviewed by qualified biostatistical and interdisciplinary scientific experts. Funded studies could include modeling at appropriately broad spatial and temporal scales to assess cumulative impacts and to evaluate fish population impacts. Studies could also involve experimental and observational tests of coupled physical–biological processes critical to understanding, modeling, and predicting biological impacts of beach nourishment.

Our review demonstrates that much uncertainty surrounding biological impacts of beach nourishments can be attributed to the poor quality of monitoring studies. Because neither federal and state permit-granting agencies nor consulting companies ensure sufficient rigor in beach monitoring done as a permit condition, and because the agencies rarely require compensatory mitigation of even egregious injuries, the required monitoring now serves little public purpose. Enhancing understanding of the impacts of beach nourishment consequently requires changes in agency process so that (a) monitoring studies are designed by adequately qualified scientists and required to meet prevailing standards of scientific rigor, (b) studies have clear goals that will advance knowledge of environmental impacts and be used to mitigate injuries, and (c) the process-oriented science required to fully understand the ecological impacts of beach nourishment is funded. Unless agency practices change, environmental uncertainty over impacts of beach nourishment will persist, and projects will continue to externalize significant costs by passing on natural resource injuries to the public at large without due avoidance, minimization, and mitigation.

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References cited

Barros F, Underwood AJ, Lindegarth M. 2002. A preliminary analysis of the structure of benthic assemblages of surf zones on two morphodynamic types of beach. Journal of the Marine Biological Association of the United Kingdom 82: 353–357.

Barth MC, Titus JG. 1984. Greenhouse Effect and Sea Level Rise: A Challenge for This Generation. New York: Van Nostrand Reinhold.

Bender EA, Case TJ, Gilpin ME. 1984. Perturbation experiments in community ecology: Theory and practice. Ecology 65: 1–13.

Brown AC, McLachlan A. 1990. Ecology of Sandy Shores. Amsterdam: Elsevier.

Burlas M, Ray GL, Clarke D. 2001. The New York District's Biological Monitoring Program for the Atlantic Coast of New Jersey, Asbury Park to Manasquan Section Beach Erosion Control Project: Final Report. Vicksburg (MS): US Army Corps of Engineers.

Bush DM, Neal WJ, Longo NJ, Lindeman KC, Pilkey DF, Esteves LS, Congleton JD, Pilkey OH. 2004. Living with Florida's Atlantic Beaches: Coastal Hazards from Amelia Island to Key West. Durham (NC): Duke University Press.

Clarke KR. 1993. Non-parametric multivariate analyses of change in community structure. Australian Journal of Ecology 18: 117–143.

Connell JH. 1974. Field experiments in marine ecology. Pages 21–54 in Mariscal R, ed. Experimental Marine Biology. New York: Academic Press.

Fonseca MS, Julius BE, Kenworthy WJ. 2000. Integrating biology and economics in seagrass restoration: How much is enough and why? Ecological Engineering 15: 227–237.

Gray JS. 1974. Animal–sediment relationships. Oceanography and Marine Biology Annual Review 12: 223–261.

Green RH. 1979. Sampling Design and Statistical Methods for Environmental Biologists. New York: Wiley.

Greene K. 2002. Beach Nourishment: A Review of the Biological and Physical Impacts. Washington (DC): Atlantic States Marine Fisheries Commission. ASMFC Habitat Management Series no. 7.

Hackney CT, Posey MH, Ross SW, Norris AR. 1996. A Review and Synthesis of Data on Surf Zone Fishes and Invertebrates in the South Atlantic Bight and the Potential Impacts from Beach Renourishment. Wilmington (NC): US Army Corps of Engineers.

Hayden B, Dolan R. 1974. Impact of beach nourishment on distribution of *Emerita talpoida,* the common mole crab. Journal of the Waterways, Harbors and Coastal Engineering Division 100: 123–132.

Hilborn R, Walters C. 1992. Quantitative Fisheries Stock Assessment and Management: Choice, Dynamics, and Uncertainty. New York: Chapman and Hall.

Hurlbert SH. 1971. The non-concept of species diversity: A critique and alternative parameters. Ecology 52: 577–586.

———. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54: 187–211.

Lindeman KC. 1997. Comparative assessment of beach systems of Florida and the Antilles: Applications using ecological assessment and decision support procedures. Pages 134–164 in Chambers G, ed. Managing Beach Resources in Smaller Caribbean Islands. Paris: UNESCO. UNESCO Coastal Region and Small Island Reports no. 1.

Lindeman KC, Snyder DB. 1999. Nearshore hardbottom fishes of southeast Florida and effects of habitat burial caused by dredging. Fishery Bulletin 97: 508–525.

Manning LM. 2003. Ecology of ocean beaches: The importance of human disturbances and complex biological interactions within a physically rigorous environment. PhD dissertation. University of North Carolina at Chapel Hill.

Nelson WG. 1993. Beach restoration in the southeastern US: Environmental effects and biological monitoring. Ocean and Coastal Management 19: 157–182.

[NRC] National Research Council. 1995. Beach Nourishment and Protection. Washington (DC): National Academy Press.

———. 2000. Strengthening Science at the Environmental Protection Agency: Research Management and Peer-Review Practices. Washington (DC): National Academy Press.

Paine RT. 1977. Size-limited predation: An observation and experimental approach with the *Mytilus–Pisaster* interaction. Ecology 57: 858–873.

Peterson CH, Hickerson DHM, Grissom Johnson G. 2000. Short-term consequences of nourishment and bulldozing on the dominant large invertebrates of a sandy beach. Journal of Coastal Research 16: 368–378.

Rakocinski CF, Heard RW, Lecroy SE, McLelland JA, Simons T. 1996. Responses by macrobenthic assemblages to extensive beach restoration at Perdido Key, Florida, U.S.A. Journal of Coastal Research 12: 326–353.

Schmitt R, Osenberg C, eds. 1996. Detecting Ecological Impacts: Concepts and Applications in Coastal Habitats. New York: Academic Press.

Stewart-Oaten A, Murdoch WW, Parker KR. 1986. Environmental impact assessment: “Pseudoreplication in time”? Ecology 67: 929–940.

Telesnicki GJ, Goldberg WM. 1995. Comparison of turbidity measurement by nephelometry and transmissometry and its relevance to water quality standards. Bulletin of Marine Science 57: 540–547.

Underwood AJ. 1997. Experiments in Ecology: Their Logical Design and Interpretation Using Analysis of Variance. Cambridge (United Kingdom): Cambridge University Press.

[US COP] US Commission on Ocean Policy. 2004. An Ocean Blueprint for the 21st Century: Final Report. Washington (DC): US COP.

Warwick RM. 1988. Analysis of community attributes of the macrobenthos of Frierfjord/Langesundfjord at taxonomic levels higher than species. Marine Ecology Progress Series 46: 167–170.

Wells JT, Peterson CH. 1986. Restless Ribbons of Sand: Atlantic and Gulf Coastal Barriers. Slidell (LA): US Fish and Wildlife Service, National Wetlands Research Center.

Wilber DH, Clarke DG, Ray GL, Burlas M. 2003. Response of surf zone fish to beach nourishment operations on the northern coast of New Jersey, USA. Marine Ecology Progress Series 250: 231–246.