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Environmental Assessment and Monitoring of Artificial Habitats

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I. Introduction

Monitoring and assessment of artificial habitats to determine their characteristics and effectiveness has been of interest from the time that humans first introduced objects into the aquatic environment. While the first assessments of artificial habitats may have been only qualitative impressions of a curious observer, they established a basis and interest for periodic evaluation.

In recent times, the planning and construction of artificial habitats has been directed toward more specific objectives (Bohnsack and Sutherland, 1985; Nakamura, 1985; Sato, 1985; Hueckel *et al.*, 1989; Relini and Relini, 1989). Accordingly, a need has arisen in the biological sciences for more specialized methods to quantitatively assess and monitor habitats to determine if objectives are being met. There also has developed a general acceptance that a quantitative, scientific approach to environmental evaluation will yield the most useful understanding of the influences that affect the performance of man-made habitats (e.g., Bohnsack, 1989; Seaman *et al.*, 1989a). The development of a considerable body of ecological theory has led to a more specific investigation of assumptions that underlie various hypotheses of interest. Within the aquatic sciences, innovative assessment

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methods have been and continue to be tested and improved (Andrew and Mapstone, 1987). Concurrently, scientists are developing more useful methods of data analysis.

This chapter presents a review of the assessment and monitoring methodology for artificial habitats. The literature referenced herein, although extensive, is not complete. One should be aware of the vast amount of previously published materials that are available. The answers to many questions may be found there. The literature should always be consulted before any project is initiated. A good place to begin is the annotated bibliography on artificial reefs prepared by Stanton *et al.* (1985). To fully understand the application of various techniques it is necessary to appreciate biotic and abiotic factors and their interaction. Thus, we briefly review characteristics of freshwater and marine environments and problems unique to sampling them. It is essential to know the limitations or special considerations that accompany each technique. Part of the intent of the discussion is to permit more rational decisions about what to assess or monitor and which methods are "best" given the study situation and objectives. Where studies or information on artificial habitats are lacking we draw from the literature on natural reef and related ecosystems.

II. Purpose of Ecological Assessment and Monitoring

The design, implementation, and analysis phases of assessment studies on artificial habitats must maintain a focus on the question(s) that motivated the research. A common concern, for example, is whether the purpose for which the reef was constructed has been achieved. Clear objectives contribute to a rational and scientific approach to habitat evaluation. Assessments that are not goal-oriented can be costly, in view of expenses for personnel and ship operating costs. Further, poorly designed studies can provide misleading and irrelevant information that may lead to inaccurate conclusions.

It is preferable to formulate the objective of any study with reference to a testable hypothesis (Green, 1979; Stewart-Oaten *et al.*, 1986). This means making a generalized statement (based on previous observations) about the topic and then conducting experiments or additional observations that are designed to reject the statement. If a given hypothesis is found totally or partially false, the next step is to identify those parts that are false and formulate a new hypothesis. It may be necessary to modify the questions and redefine the hypotheses until they can no longer be rejected. For example, Spanier *et al.* (1985) posed the hypothesis that recruitment of fishes to an artificial reef might be enhanced by "baiting" it and then compared rates of

recruitment to baited and nonbaited sites. Their conclusion was that their hypothesis was not rejected. But had the recruitment rate been equal to or less than that of a nonbaited reef, they would have had to reject their hypothesis and offer a new, modified hypothesis.

Forming a series of hierarchial questions or hypotheses to establish the purpose of an assessment program also determines which methodology is most appropriate for data collection and analysis. In the following sections, we address four principal purposes for assessment and monitoring of artificial habitats, which in turn dictate the selection of methodology to obtain pertinent information.

A. Status of the Artificial Reef

Knowledge of the status of an artificial reef is essential before appropriate questions can be formulated. To determine a reef's "status" some *a priori* knowledge of the reef is needed (e.g., background literature or data from pilot studies). The status should reflect an accurate "picture" of the reef. It may consist only of data from one small place during a brief time. Because of the suite of possible hypotheses to be tested and the natural variability of data, it is often better to have several "pictures" over an extended period. Without knowledge of reef status it would be difficult to formulate hypotheses about habitat size, complexity, carrying capacity, or effects on biomass (e.g., Prince *et al.*, 1985).

The status of an artificial reef is relative to some standard, such as, to itself over time (Hastings, 1979); another artificial reef (Bortone and Van Orman, 1985a); a nonreef area (Alevizon and Gorham, 1989); or a natural reef (Fast and Pagan, 1974). The status of an artificial reef could include information ranging from many variables, measured continuously to only one observation about one variable. Clearly the term "status" is quite broad and the description possible from any reef can vary considerably in complexity and detail.

1. Duration of Study

Various time frames can be used to establish reef status, depending on the study objectives. It is not possible to monitor all habitats all the time but the intervals between sampling surveys can affect whether or not an impact can be detected by the sampling strategy. For example, a short-term survey on a newly established reef (e.g., once a month for a year) may not be long enough to record conditions that may take years to become established. Likewise, a survey done once a year but always on the same tidal cycle may not be able to detect the influence of tides on the faunal composition of a reef inhabited chiefly by tidally influenced species. The temporal basis for studies can be hourly, to account for reproductive or other activity (Colton and Alevizon, 1981; Harmelin-Vivien *et al.*, 1985); night to day, for monitoring fish population changes (Rutecki *et al.*, 1985; Zahary and Hartman, 1985; Bortone *et al.*, 1986; Moring *et al.*, 1989); a month or less, for evaluation of tidal and lunar effects on the biota (Prince *et al.*, 1985; Ardizzone *et al.*, 1989; Bailey-Brock, 1989; Buckley and Hueckel, 1989; Moring *et al.*, 1989; Rountree, 1989); from three months to a year, to evaluate seasonal influences (Bailey-Brock, 1989; Bell *et al.*, 1989); or many years, to account for long-term changes (Buckley and Hueckel, 1989).

Monitoring may need to be conducted over several years to adequately assess colonization and succession. Faunal and floral elements may only gradually colonize new habitat structure (Sutherland, 1974; Sutherland and Karlson, 1977). Turner *et al.* (1969) found no mature fouling communities on an artificial reef in southern California, after more than five years of study. For a general reference to colonization and succession see Begon *et al.* (1986). Readers should be aware that succession is not universally accepted by all contemporary ecologists (see Connell, 1978). (See also related discussion in Chapter 3.)

The inevitable but unpredictable nature of many natural events demonstrates the importance of long-term monitoring. However, it is equally important to note that it may be necessary to adapt the time frame of some surveys to accommodate the short-term impacts of environmental perturbations, caused by natural or artificial influences such as storms or pollution events.

2. Comparative Studies

Comparative studies of artificial reefs are often required by government regulation. Superficially, they seem an easy and straightforward way to determine the effectiveness of a reef. However, environmental assessment conducted over a brief time at a limited locality may not give a representative picture of mechanisms controlling resource availability. Too often, decisions are made based on human biases toward constructing reefs with features that may not be important. This limited view has led to the rapid (and not necessarily justified) expansion of reef construction (Meier *et al.*, 1989). We do not suggest avoiding interreef assessment studies, but they must be planned and the data interpreted carefully.

Comparing the fauna and flora of an artificial reef to a natural reef is an inevitable consequence of the human comparative process and is a worthwhile endeavor. The question of whether or not artificial reefs increase productivity or merely act as resource attractants is entirely in order (Bohnsack and Sutherland, 1985; Bohnsack, 1989). Studies need to examine movement of organisms between natural and man-made habitat to measure the impact of artificial reefs on natural reefs (Brock *et al.*, 1985; Fast and Pagan, 1974; Matthews, 1985). Care should be taken to assure that the assessments are made in an unbiased, scientific, and responsible manner, and are absolutely nondisruptive to the habitat.

B. Relation to Management Objectives

Artificial reefs have had, and will continue to have, an application to fisheries management (reviewed in Chapter 5). Sampling strategy must address the reef's principal management objective (Ahr, 1974). For example, reef placement and design may target a particular species or group of species (Sato, 1985), such as the attraction and habitat improvement of midwater fishes (Stephan and Lindquist, 1989; Buckley *et al.*, 1989), or providing benthic habitat for crustaceans (Davis, 1985). Other questions include, Was the rationale for providing the artificial reef justified? Can the resource base support a potential increase in exploitation rate provided for by the placement of an artificial reef? Was the management strategy met? If the reef was built to increase the maximum sustainable yield (MSY) or optimum yield (OY) for a fishery (Roedel, 1975), then data collection should be designed and scheduled to measure the appropriate factors to monitor MSY over time (Buchanon, 1974; Feigenbaum *et al.*, 1985; Gannon *et al.*, 1985; Solonsky, 1985; Polovina and Sakai, 1989). In another situation, artificial reefs can serve as a refuge for a species and are "off limits" to certain fishing methods, or times of year. Or, they can be used as a refuge to aid in the recovery or maintenance of an existing natural reef community (Brock *et al.*, 1985; Seaman *et al.*, 1989a; Feigenbaum *et al.*, 1989a), or as a means to avoid user conflicts among fishermen. A reef also could provide a safe haven for juveniles, preferred food for adults, a reproductive sanctuary, or enhance survival of a precarious life stage needed to assure a critical level of productivity (Anderson *et al.*, 1989; Campos and Gamboa, 1989).

Nonbiological factors also motivate deployment of artificial habitats. For example, increased local tourism oriented toward SCUBA diving or reduction in user conflicts (Samples, 1989) may be the reason for establishing a reef (Fig. 6.1). When an artificial reef is emplaced to broaden user participation, a much different type of sampling design may have to be considered than that used when the objective of a reef is to increase biomass. Economic and social assessment of habitats is addressed in Chapter 7.

C. Determining the Influencing Factors

To understand and ultimately manage an environmental system it is important to know the specific controlling or influencing factors (Gauch, 1982;

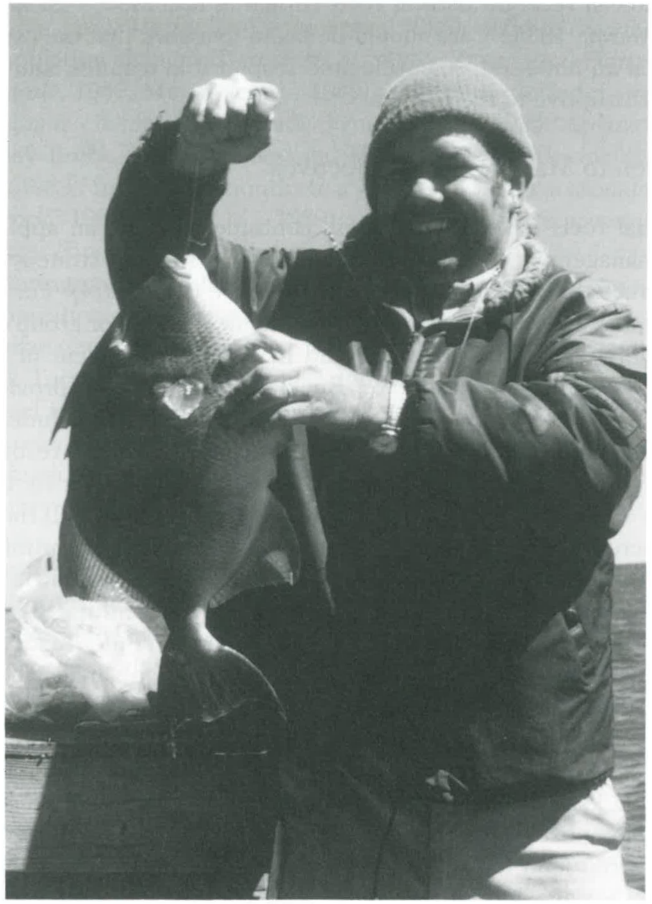

Figure 6.1 Scientific assessment of artificial habitats is often motivated by various fishery interests and concerned with achievement of management objectives. In the northern Gulf of Mexico the gray triggerfish (*Balistes capriscus*) is designated as an "underexploited" species that can provide a new resource to meet increased recreational fishing pressure in the coastal United States.

Bortone and Van Orman, 1985a; Patton *et al.*, 1985). It may be possible to manipulate key environmental variables to improve a habitat's performance in meeting management objectives. Through experimental manipulation we may gain insight into the role of various factors in the basic ecology of the entire aquatic ecosystem (Bohnsack and Sutherland, 1985). To simplify the research approach, we will review briefly two categories of influencing factors and their potential significance to artificial habitats. (See Chapter 3.) While not always mutually exclusive, their effects are different enough to warrant separate consideration.

1. Density-Dependent Factors

Density-dependent factors control the growth of a population in such a way that as the population size changes, so does the degree of control of the factor. The relationship can be positive, negative, or in some combination, depending on the interaction of other factors. Density-dependent factors have a quantitative influence on the population with regard to the quantity of something needed by the individuals in the population, such as space or food. For example, a species may respond to the amount of food available as a density-dependent factor (Pitcher and Hart, 1982): the more food present, the greater the potential food consumption, and, therefore, the greater the population. The response by the community may not always be so dramatic or immediate. Concomitantly, there are natural limits to density-dependent factors as population growth rate operates.

Biotic features are often thought of as being density-dependent (e.g., predator–prey relationships). Meanwhile, some nonliving (i.e., abiotic) factors can have a direct (positive, negative, or combinational) impact on an organism's life history as well. If artificial reefs contribute to needed factors in the aquatic environment (e.g., reef size [Grove and Sonu, 1985], number of hiding places or rugosity [Chandler *et al.*, 1985]), then by making those factors more available the reef may be physically enhanced from a human perspective. Ultimately, this is the most significant concept justifying research associated with artificial reefs. The presumption has been that if a species is limited by its habitat, then providing more of the habitat should result in an increase in the fishery resource.

2. Density-Independent Factors

Density-independent factors are conceptually more difficult to understand than density-dependent factors. Density-independent factors are those whose severity and influence is not dependent on the density of the population (e.g., Pennak, 1964). For example, artificial reef assemblages have been impacted by storms (Bortone, 1976; Matthews, 1985) and red tides (Smith, 1975).

Generally, most density-independent factors are abiotic environmental factors. Tides, storms, currents, waves, and substrate are among the density-independent factors that are most often examined and recorded in artificial reef studies (e.g., Lukens *et al.*, 1989). Sometimes it is nearly impossible to separate their effects from those usually considered density-dependent. The phenomenon of upwelling may be an example of a density-independent factor that impacts the amount of available oxygen, but oxygen is a density-dependent factor in most aquatic communities.

Features of reef design, such as surface materials, position, geographic location, configuration, area, and volume, are often significant aspects of the artificial reef environment (Brock and Norris, 1989; Sheehy, 1985). These can be density-dependent if the organisms require them as part of their life history and if abundance is limited or directly controlled by the relative amount of their presence (e.g., number or size of crevices). Similarly, reef design features relative to position (e.g., substrate, distance from shore, depth, amount of light) can act as density-independent factors.

D. Testing Scientific Hypotheses

Artificial habitats often are used as models to experimentally test various hypotheses about the environment (Bohnsack, 1989). Because many of their physical aspects can be manipulated or controlled, they present an opportunity to better understand the aquatic environment (e.g., Hixon and Beets, 1989), especially in conjunction with studies on natural habitats such as coral reefs. Study topics include the species–area hypothesis (Molles, 1978); colonization theory (Fast and Pagan, 1974; Sale and Dybdahl, 1975; Smith and Tyler, 1975; Talbot *et al.*, 1978; Lukens, 1981); species diversity (Slobodkin and Fishelson, 1974; Helfman, 1978); and whether a fauna associated with a reef is present by chance (stochastic processes) or is predictable (deterministic processes) owing to the intimate interaction of environmental factors (e.g., Dale, 1978; Helfman, 1978; Sale, 1978, 1980; Talbot *et al.*, 1978; Sale and Dybdahl, 1975; Smith and Tyler, 1975).

Some studies have tried to manipulate the artificial reef environment to determine which factors are the most limiting to certain species (Hixon and Beets, 1989). By adding or subtracting habitat, competitors, food, parasites, or other factors suspected of controlling life history parameters, it is likely that answers to questions involving these factors will be found while causing little or no damage to natural reefs. Moreover, questions about the utility of artificial reefs for increasing biological productivity can be tested by careful experimental manipulation of the various environmental components of an artificial reef. (See Fig. 6.2.)

III. Problems in Assessing Environments

Monitoring and assessment of artificial habitats must account for certain problems and be cognizant of environmental variation, sampling error, and

Figure 6.2 Much of the experimentation with artificial habitats has involved relatively small structures. Concrete blocks in various configurations commonly are used; see also Figure 1.10. (Photograph courtesy of J. Bohnsack, U.S. National Marine Fisheries Service.)

biotic and abiotic interactions. It is essential to understand what data are to be collected and how they will be statistically analyzed, before beginning actual study. The reader should refer to general texts on environmental sampling and data analysis such as Gauch (1982), Green (1979), and Ludwig and Reynolds (1988).

Some variation is inherent to any factor in the environment, whether biotic or abiotic. Interaction with other variables or features in the environment can reduce or exaggerate the inherent variation that exists (Green, 1979; Poole, 1974; Zar, 1984). Variation also can result from the sampling technique used to conduct an assessment (i.e., the error due to our inability to measure anything with absolute accuracy). This so-called "sampling error" can be reduced with further refinement of technique and careful study.

Environmental research concerning artificial habitats must consider the potential effect of intra- and interspecific competition for available resources. If competition for resources increases to a level that limits individual or species survival, it may influence species abundance or the intensity with which that organism interacts in the life zone. Species interactions usually have a positive or negative impact on the well-being of the species involved (Paine and Levin, 1981). However, determining causative factors and measuring the intensity of the impact is not always straightforward.

A. Assessment Problems in Aquatic Environments

The physical proximity, climate, structural composition, and natural and human-induced perturbations associated with terrestrial environments adjacent to the shore can have direct and significant influence on the aquatic environment. Submerged substrates are often derived from terrigenous sources. Suspended materials including solids, as well as living materials, can greatly impact water clarity and other features. For example, light penetration, water column productivity, and dissolved oxygen content can be interconnected. In addition, gradient (especially in freshwater and nearshore coastal environments) and slope can have the most profound influence on the aquatic community. Variables such as current flow, tidal height and frequency, sediments and deposition rates, and mixing are particularly affected by gradient.

Aquatic ecosystems not only are multidimensional but also present a near weightless environment to organisms. Since water is approximately 800 times more dense than air, a greater variety of materials might be suspended for greater periods of time than is possible in the terrestrial environment. Further, time takes on additional significance, because water column fea-

tures at one time and place may eventually move to another place. This feature should be considered in any attempt to assess the environment and its resources. A general perspective of aquatic environments is provided by Tait (1981) and Nybakken (1982) for oceans and estuaries, and Cole (1983) for freshwater rivers, lakes, and streams.

1. Assessment Problems Unique to Reef Communities

Assessment techniques must account for the spatial irregularity that reefs present. Reefs are structures composed of a variety of natural and artificial materials. Natural reefs can be composed of living materials such as hard coral, soft coral, sponges, oysters, or vegetation. They also can be composed of dead materials such as submerged logs, sipunculid worm casings and "drowned" coral reefs, or nonliving materials such as rocks, natural deposits, hardened volcanic extrusions, and other geologic formations. All provide substrate heterogeneity and physical relief.

Reef communities are characterized by the very different activities that organisms display. Activity can shift dramatically from day to night (Starck and Davis, 1966; Zahary and Hartman, 1985) or with changes in the aquatic environment such as turbidity or season (Harmelin-Vivien *et al.*, 1985). As the activity of organisms changes, so does the ability of scientists to detect them.

2. Assessment Problems Unique to Artificial Reefs

Human intervention via creation of artificial habitats adds complexity to the aquatic environment. Artificial reefs can be and often are significant modifications to the natural environment. After deployment, an artificial reef becomes interactive with the surrounding habitat. As described earlier both positive and negative interactions can occur. The placement of the artificial habitat, therefore, introduces a source of variation in the natural habitat.

The types of materials from which artificial habitats are constructed may present the assessor with special conditions. They may contribute to water column clarity and purity. Chemicals from the materials may become incorporated into the tissues of reef-colonizing organisms and affect their behavior or other aspects of their life history. The spatial complexity of the reef material may render some assessment methodologies useless, and special techniques may have to be developed for each set of assessment circumstances. Above all, the presence of an artificial reef is likely to alter the species–species, species–environment, and environment–environment interactions found in an area (Fast and Pagan, 1974; Russell *et al.*, 1974; Molles, 1978; Chandler *et al.*, 1985; Alevizon and Gorham, 1989).

3. Assessment Problems Unique to Artificial Reef Organisms

A fascinating aspect of studying the organisms associated with artificial habitats is the unique role each plays within the biotic community. Unique species mixes and communities (or assemblages, see Chapter 3) may occur. Altered trophic level interactions can result from the altered species mix. Occasionally the presence of a species at a site where it was not found before the deployment of habitat structure indicates that colonization was influenced by the availability of some limiting factor (e.g., space, shelter, or food) not available previously. Although we might argue that a species occurring at an artificial reef is preadapted to the features provided by the reef, the species may have modified its "natural" life history features to be able to exist in its new environment. The assessment problem that this presents is that in the new array of circumstances a species may take on a new or altered role or niche. Sometimes it may be necessary to alter the plan of assessment to accommodate the altered life-style of a species in its new circumstance.

The interactions of organisms and substrate influence other members of the artificial reef community. Bailey-Brock (1989) showed that encrusting species on an artificial reef make the reef surface even more complex with time. It is important to be aware of such factors when designing any study.

IV. Data Needs

The data to be obtained in an assessment program can be identified after the objectives of the artificial habitat have been established, and the limitations that the environment can place on a study are recognized (Caddy and Bozigos, 1985). One approach is to list the features both of an organism that we expect to be affected by habitat deployment, and of the environment expected to have an effect on that habitat. Virtually all variables can be assigned to one of these two groups. These then become the dependent and independent variables. Usually it is the dependent variables we are interested in evaluating as they are impacted by the independent variables. Since artificial reefs represent such an interactive circumstance, several variables may be interdependent or codependent.

Faced with a seemingly unlimited number of variables and types of data, the scientist initially needs to identify those variables best suited to answer the specific questions that have been posed. Some studies will determine the effect of one independent variable on one dependent variable (e.g., the relationship of hole or hiding place size with regard to fish size; Hixon and Beets, 1989). Others incorporate many variables (e.g., ecology of artificial and natural reefs; Bortone and Van Orman, 1985a; Duval and

Duclerc, in Harmelin-Vivien *et al.*, 1985). While a research program may seek to assess the greatest number of variables to the highest degree of precision and accuracy, clearly there are both physical and fiscal limitations to research. The following discussion addresses the types of data that are generally more useful regarding questions concerning artificial reefs. These are more or less "standard" variables that are obtained in many studies. Their utility lies in not only their usefulness in a description, but also their value in comparing data from other studies. The following outline lists variables commonly measured and initial references to methods of assessing and monitoring artificial habitats.

I. Biotic Variables
  A. Fishing
    1. CPUE: Brock, 1985; Crumpton and Wilbur, 1974; Feigenbaum *et al.*, 1985; Matthews, 1985; Myatt *et al.*, 1989; Polovina and Sakai, 1989; Relini and Relini, 1989; Solonsky, 1985
    2. Survey: Crumpton and Wilbur, 1974; Milon, 1989b
    3. Recruitment: Buckley and Hueckel, 1989
  B. Abundance
    1. Number of individuals: Alevizon and Gorham, 1989; Anderson *et al.*, 1989; Bailey-Brock, 1989; DeMartini *et al.*, 1989; Moffit *et al.*, 1989; Thorne *et al.*, 1989
    2. Cover: Bailey-Brock, 1989; Bohnsack, 1979; Fitzhardinge and Bailey-Brock, 1989
    3. Density: Ambrose and Swarbrick, 1989; Anderson *et al.*, 1989; Ardizzone *et al.*, 1989; DeMartini *et al.*, 1989; Harmelin-Vivien *et al.*, 1985; Jones and Chase, 1975; Thresher and Gunn, 1986
    4. Diversity: Baynes and Szmant, 1989; Moffit *et al.*, 1989
    5. Dominance: Jones and Chase, 1975
    6. Community similarity: Jones and Chase, 1975
    7. Occurrence: Anderson *et al.*, 1989; Fitzhardinge and Bailey-Brock, 1989; Laufle and Pauly, 1985
    8. Richness: Moffit *et al.*, 1989
    9. Relative importance: Fitzhardinge and Bailey-Brock, 1989; Jones and Chase, 1975; Laufle and Pauly, 1985
    10. Biomass: Anderson *et al.*, 1989; Brock and Norris, 1989; Jones and Chase, 1975; Moring *et al.*, 1989; Moffit *et al.*, 1989
    11. Colonization: Ardizzone *et al.*, 1989
  C. Life history
    1. Age: Gannon *et al.*, 1985; Moring *et al.*, 1989
    2. Growth: Fabi *et al.*, 1989; Moring *et al.*, 1989; Prince *et al.*, 1985
    3. Size

a. Length: Ambrose and Swarbrick, 1989; Anderson *et al.*, 1989; Bell *et al.*, 1985; Brock, 1985; Brock and Norris, 1989; DeMartini *et al.*, 1989; Fabi *et al.*, 1989; Feigenbaum *et al.*, 1985; Gannon *et al.*, 1985; Moring *et al.*, 1989
b. Weight: Ambrose and Swarbrick, 1989; Brock, 1985; Brock and Norris, 1989; Fabi *et al.*, 1989; Feigenbaum *et al.*, 1985; Gannon *et al.*, 1985; Moring *et al.*, 1989

4. Condition factor: Prince *et al.*, 1985
5. Feeding: Brock, 1985; Buckley and Hueckel, 1985; Gannon *et al.*, 1985; Jessee *et al.*, 1985; Moring *et al.*, 1989
6. Habits: Brock, 1954
7. Preferred habitat: Jessee *et al.*, 1985; Moring *et al.*, 1989
8. Predation: Jessee *et al.*, 1985
9. Larval development: Jessee *et al.*, 1985
10. Reproduction: Gannon *et al.*, 1985; Jessee *et al.*, 1985; Moring *et al.*, 1989
11. Migration: Buckley and Hueckel, 1985; Davis, 1985; Myatt *et al.*, 1989

D. Abiotic Variables

1. Substrate: Ahr, 1974; Ambrose and Swarbrick, 1989; Baynes and Szmant, 1989; Chandler *et al.*, 1985; Mathews, 1985
2. Local conditions
   a. Location: DeMartini *et al.*, 1989; Lukens *et al.*, 1989; Stanley and Wilson, 1989
   b. Temperature: Sanders *et al.*, 1985
   c. Visibility: Ahr, 1974; Carter *et al.*, 1985; Kevern *et al.*, 1985; Sanders *et al.*, 1985; Stephan and Lindquist, 1989
   d. Pollution: Ahr, 1974; Mathews, 1985
   e. Waves and sea state: Mathews, 1985
   f. Weather: Lukens *et al.*, 1989
3. Reef attributes: Alevizon *et al.*, 1985; Ambrose and Swarbrick, 1989; Grove and Sonu, 1985; Helvey and Smith, 1985; Hixon and Brostoff, 1985; Lukens *et al.*, 1989; Molles, 1978; Sanders *et al.*, 1985
4. Earth-Tuned features
   a. Lunar cycles: Sanders *et al.*, 1985
   b. Currents: Baynes and Szmant, 1989; Lindquist and Pietrafesa, 1989
5. Season: Sanders *et al.*, 1985
6. Time of day: Sanders *et al.*, 1985

A. Biotic Variables

Biotic variables are usually dependent variables. Most often there is some aspect of organism life history, population, or community ecology that is affected by some other biotic or abiotic variable that we can define as important. The complexity of the interactions increases with each higher level of organization.

The species-specific life history parameters can be many and may include life stages (larvae, juvenile, or adult; Grove and Sonu, 1985) or sex (Gannon *et al.*, 1985). Growth data include rate of growth, maximum size, and size at life stage (Gannon *et al.*, 1985; Prince *et al.*, 1985; Ambrose and Swarbrick, 1989; Fabi *et al.*, 1989). Reproductive parameters include age (or size) at sexual maturity, fecundity, reproductive season, reproductive mode (many artificial reef-associated marine species are hermaphroditic) and behavior, and spawning conditions (Jessee *et al.*, 1985). Food and feeding habits as well as inter- and intraspecific behavioral interactions are also important (Jessee *et al.*, 1985; Sutherland, 1974; Fitzhardinge and Bailey-Brock, 1989). Immigration of organisms to the habitat, and the activities of resident or migratory species also are important behaviorial information (Hixon and Beets, 1989). Studies of the habitat requirements of species can be oriented toward the details of microhabitat (fine-scale) or macrohabitat (gross-scale) (Jessee *et al.*, 1985).

Population parameters include abundance (Anderson *et al.*, 1989), sex ratio, and life stage (e.g., Grove and Sonu, 1985). Sometimes more important to our understanding of a population than sheer numbers is the size or biomass of the specific life stages of the population. Biomass present at one time is most often referred to as "standing biomass" or "standing crop" (Bardach, 1959; Brock and Norris, 1989) and is used as a measure of the population (Anderson *et al.*, 1989). A more exacting study would determine biomass in terms of productivity (rate of biomass change per unit time per unit of area or volume). However, such rates are difficult, if not impossible, to measure because of the highly mobile, and thus elusive, nature of the highest proportion of the organisms associated with artificial reefs. However, productivity has been generally inferred from standing biomass (Bohnsack and Sutherland, 1985). Additional population parameters might include migration (immigration and emigration) as well as a host of fishery parameters such as recruitment (i.e., age or size of first capture) and mortality (fishing, natural, and total).

Fisheries associated with artificial reefs do not usually target a single species but often target an assemblage of organisms. This is because of the nonselective fishing methods usually employed and the occurrence of many species susceptible to the gear. Numerous studies have developed stock assessment estimates from catch data (e.g., Grove and Sonu, 1985; Feigenbaum *et al.*, 1989a,b). Ecologically, it is difficult to study individual species without reference to the others with which it associates. Community or species assemblage parameters, therefore, include species composition (species presence or absence), abundance, and diversity (as measured by H′, species richness and species evenness; Pielou, 1966). Interspecific associations include commensal, symbiotic, parasitic, and predator–prey interactions.

All biotic components may have potentially significant impacts on the desired target species of the artificial habitat, although thus far research has neglected some of these components (Luckhurst and Luckhurst, 1978). For example, only limited assessment of the attached community, i.e., sessile animals and attached algae, has occurred (e.g., Fitzhardinge and Bailey-Brock, 1989). The water mass flowing past the reef and the near-reef community also contains important communities (Glynn, 1973) and should be an integral part of artificial reef research. (See Fig. 6.3.)

The impact of human activity such as fishing pressure or boat traffic (Harmelin-Vivien *et al.*, 1985; Mathews, 1985) should be examined if it is likely to be an important influence on the dependent variables. For additional discussion of economic and social variables see Milon (1989a,b) and also Chapter 7 of this volume.

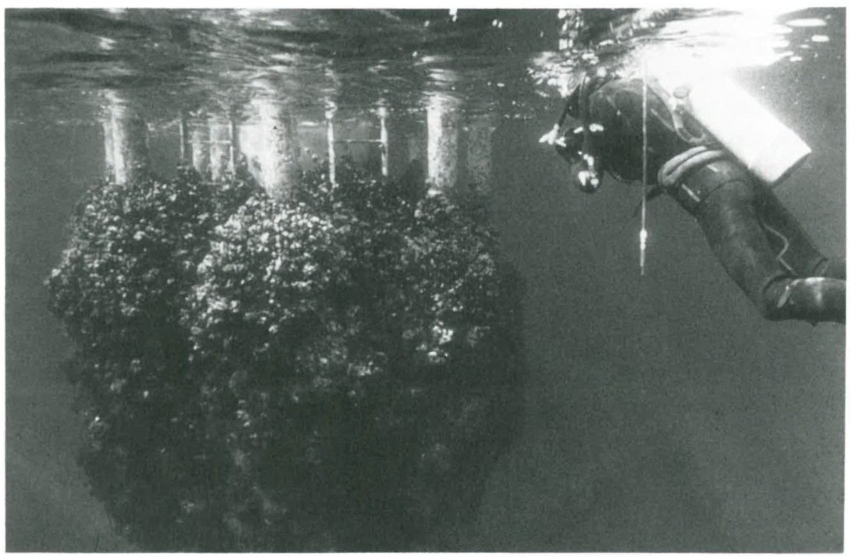

Figure 6.3 The potential for development of a large population of sessile organisms attached to an artificial habitat is demonstrated by these mussels "fouling" the legs of a petroleum platform. (Photograph courtesy of American Petroleum Institute and Sport Fishing Institute.)

B. Abiotic Variables

Many nonliving parameters are studied as part of artificial habitat assessments and are usually considered independent variables. They are essential to examining the impact of the habitat and its surrounding abiotic environment on the biological attributes of the artificial habitat. However, if one is concerned about the biological portion of the artificial reef environment and how it affects the reef itself (e.g., the surface materials after having been encrusted), then the measurements of the abiotic community could be treated as dependent variables. These most often include the more familiar and easily measured water condition parameters, but other nonliving habitat parameters should be considered as well.

1. Habitat Materials, Construction, and Design

The materials from which the habitat is constructed are often important in affecting the local environment of its community. The chemical composition of materials can influence a reef (Woodhead *et al.*, 1985; Fitzhardinge and Bailey-Brock, 1989; Fabi *et al.*, 1989). Many areas have laws regarding the proper treatment of reef materials. For example, automobiles must have oils and other petroleum products removed. Tires are known to leach toxic substances for considerable periods after deployment (Anonymous, 1974). Certain materials such as fly ash blocks and plastics may prove hazardous (although see Woodhead *et al.*, 1985). While leached materials may become diluted in a vast water body, there is the potential of a local negative impact via accumulation of these products in the food chain. Direct interference with the ability of some attached or settling organisms to take-up residence can occur as well (Cairns *et al.*, 1976).

Some material may enhance the attachment of settling organisms and thus increase potential food items available for predators. Some literature describing colonization rates includes discussion of the presumed beneficial feature they have at helping a reef to "mature" and become a "real" ecological community (e.g., Bailey-Brock, 1989; Fitzhardinge and Bailey-Brock, 1989). Although there is little evidence that preferred target fish species consume these attached items (but see Prince *et al.*, 1985), nevertheless, their presence does seem to enhance the aesthetics of the structure and create a "climate" that enhances the fishery. These "nonconsumable" habitat components may, however, provide the necessary structure or features for the consumable prey items of preferred fishes and macroinvertebrates. Surface texture can have a profound influence on the biofouling community and may be just as important as composition material and should, therefore, be recorded in any reef study.

Grove and Sonu (1983) defined and diagrammed the various artificial reefs components as follows: reef unit, a single module; reef set, a cluster of reef units; reef group, an assemblage of reef sets; and reef complex, an arrangement of the reef groups. (See also Chapter 4, including Fig. 4.5.) Each level of complexity can be important in how it influences the biota.

The reef unit is the basic deployed structure of the reef. It can consist of surplus materials such as tires (or clumps of tires if they are deployed as a unit), or a prefabricated module. Several features of the reef unit are important to consider when gathering data. The surface materials should be identified for composition and texture (Hixon and Beets, 1989). The number and size of the holes, openings, and crevices referred to as habitat complexity (Alevizon *et al.*, 1985; Gorham and Alevizon, 1989) or rugosity (Luckhurst and Luckhurst, 1977; Molles, 1978), may be especially significant in creating hiding places for some species or in facilitating the recruitment of certain species or sizes of individuals to the reef (Hixon and Beets, 1989). The population abundance may be directly dependent on the structural complexity (Harmelin-Vivien *et al.*, 1985). Other obvious parameters of the reef unit are unit height, width, area, and volume (Alevizon *et al.*, 1985).

The data-gathering process described for the reef unit is additive and can be expanded to each component of the reef complex. We must, therefore, have data, not only on the reef units, but also on the overall reef configuration. This includes total reef height (the actual height of the reef above the substrate and closest surface depth); total reef volume; interunit distance or scatter (Lukens *et al.*, 1989); proximity to previously existing structures (Ahr, 1974); general dimensions of the total deployment, including shape, orientation to currents, and slope or gradient of the substrate (Baynes and Szmant, 1989). Reef configuration and orientation information, when coupled with data concerning tides, current direction, sediment drift, and topographic orientation may prove useful in predicting optimal deployment of future structures (Bortone and Van Orman, 1985a).

2. Local Conditions

Information about local habitat must include detailed data on water quality and conditions, sediments, and weather. The "standard" water quality measurements include temperature and salinity. With these it is possible to characterize water masses by calculating density. Water clarity, visibility, or turbidity all relate to the amount of light that is able to penetrate to depth. This parameter is important biologically, but it also influences the ability of divers to make observations. The number of organisms potentially observed when using a visual survey is directly related to water visibility (Sanders *et al.*, 1985). Although visibility is most often measured vertically in the water column from the surface, when related to visual surveys it is better measured horizontally at the depth of the sampling survey (Stephan and Lindquist, 1989).

Other water quality parameters important to water column productivity are nutrients (nitrates and phosphates), oxygen (dissolved oxygen), and particulate carbon. Additional parameters could include specific pesticides, heavy metals, or other compounds, according to the objectives of the assessment.

Sediment parameters are useful when assessing or monitoring artificial reefs (Ahr, 1974). These include the size of sediment particles (measured as grain size), type of sediment (based on percent composition of shell, quartz, or other specific materials), depth, and area. Sediment size and type is significant because it can directly impact the decision where to place an artificial structure (Mathews, 1985). Substrate properties will influence the type and abundance of benthic organisms available as food sources. The amount and rates of siltation can impact filter feeding animals and may be especially important when monitoring nearshore artificial reefs (Mathews, 1985). Artificial reef materials may influence local currents and other aspects of flow (i.e., eddies, turbulence) which, in turn, affect substrate characteristics and thus the longevity of the reef itself.

The position of the structure relative to sources of terrestrial water discharge, such as estuaries, bays, and larger effluent discharges should be recorded (Bortone and Van Orman, 1985a). Freshwater river discharge and runoff can influence artificial habitats closest to terrestrial areas (Hastings, 1979). Many studies have shown that there is a species-specific orientation that can be observed on artificial reefs (e.g., Klima and Wickham, 1971; Lindquist and Pietrafesa, 1989). Other studies indicate that currents are important for settling organisms, larval transport, and therefore, juvenile recruitment. Similarly, distances to major currents, the continental shelf edge, a canyon, or some other structure might have a significant impact on the reef community.

Proximity to potential sources of colonizing organisms is important to note, such as the closest seagrass beds, marshes or other nursery areas (Bortone *et al.*, 1988). However, researchers should be aware that "closest" in spatial distance may not necessarily mean "closest" in practical or effective distance, as water currents may make transport from some distant area more likely than transport from a nearby area.

3. Earth-Tuned Conditions

The stage and intensity of predictable currents or tides should be noted because of their influence on the colonization of available substrates (Sanders *et al.*, 1985; Baynes and Szmant, 1989). Variable current patterns, temperatures, and salinities may prove significant. Although the results of such an influence may not be immediately detected by most assessment studies, the comprehensive long-term impact (i.e., over a few decades or more) may profoundly affect local conditions.

C. Number of Samples

Due to the complex nature of environmental parameters, their inherent natural variability, and the large number of potential interactions, the data obtained in ecological studies should be interpreted with caution. The important questions, What can this variation tell us? and How do we account for variation? must be considered. The answer to the first is rather simple. We can examine the relationship between parameters and compare variability. This usually is done graphically and statistically, as an initial step in understanding the causes of variation and their influence.

Accounting for the sources of variation is another matter. The first step is to acknowledge that variation can and does occur, due to sampling technique, the inherent variation in all things, or some other factor. To describe and account for variation a large number of samples measuring the variables of interest must be taken. The natural variation associated with reef organisms is compounded by the spatial heterogeneity of reefs (Harmelin-Vivien *et al.*, 1985).

The number of samples required varies for each situation (Andrew and Mapstone, 1987). For samples in which the factors are slightly variable relative to their mean value, as few as three to ten replicates may be required. Conversely, more than 50 replicates may be necessary if variance greatly exceeds the mean value of the parameter. The reader may wish to refer to texts such as Green (1979) or Zar (1984), which outline the statistical methodology needed to determine the number of replicates.

Generally, it is reasonable to expect accountability in variation of ecological parameters to occur with 20 to 40 replicates. Ideally more should be taken, but the expense and time usually cannot be afforded. In fact, few artificial reef studies have been conducted using more than five samples to account for variation; of those studies only a few have actually calculated the number of samples necessary. Sale and Douglas (1981) determined that an optimal sample number of four would provide enough information on the variability on reef fish assemblages. Harmelin-Vivien *et al.* (1985) thought a replicate number of at least 12 samples was necessary. Bortone *et al.* (1989) indicated that values of attributes such as species diversity and cumulative number of species stabilized after 16 to 32 samples, depending on the methodology. Bortone *et al.* (1986, 1989), Jones and Thompson (1978), and

Kimmel (1985b) used a minimum of eight samples to account for 90% of the inherent variability using the species–time random count visual survey method based on sample size estimates established by Gaufin *et al.* (1956).

In our view, most studies merely make a guess as to what seems a reasonable (or affordable!) number of replicates. It is not that most scientists are not aware of the importance of having a complete understanding of the nature of variation in artificial reef environments, it is more a limitation placed on the study design by the available equipment, personnel, and other resources. Consequently, the conclusions attained by an assessment and monitoring study should not always be considered as totally accurate and should be interpreted with caution. The inability to adequately account for variability may render some statistical analyses useless.

D. Frequency of Sampling

Frequency of sampling also should reflect the objectives of the study. To assess daily activity of a species on an artificial reef, sampling may have to be done continuously or at least hourly. Day–night activity (e.g., Hobson, 1965; Samples, 1989) in the environment should be monitored at least four times a day (day time, night time, and during the dusk and dawn crepuscular periods). Lunar influences on the artificial reef community are also well known (Hastings, 1979). To account for this variable, sampling would have to occur at least every seven days. A monthly sampling regime generally misses lunar influence. Seasonal variation on the reef should be monitored at least every three months in temperate areas. In some parts of the world, such as the tropics, seasons are more often related to the amount of rainfall and, therefore, should coincide with those regional conditions.

If the prime consideration in assessing an artificial reef is the colonization or defaunation rates, then the sampling design should be at a frequency to evaluate these rates in a meaningful way (Ardizzone *et al.*, 1989; Lukens, 1981). Studies of succession, as indicated above, should be of shorter intervals initially (e.g., weekly, monthly) and later reduced to a seasonal or even yearly monitoring regime, as appropriate. For example, Ardizzone *et al.* (1989) indicated that the succession of a benthic community associated with an artificial reef in the Mediterranean Sea had not become fully stabilized at the climax level after five years. Turner *et al.* (1969) suggested a similar situation in fouling communities off California.

The sampling schedule is especially critical if the objective includes time-related trends. Here sampling should occur at regular intervals and not be missed (Witzig, 1983). Unfortunately, this is almost never possible due to problems with equipment, personnel, weather, or disruption in funding. Additionally, it is important to be flexible in the sampling regime to examine the impact of unusual influences. Bortone (1976) was able to sample an artificial reef a few days before and after a hurricane to describe the storm's impact on it.

V. Assessment Methods

This section reviews the methods used to assess and monitor an artificial reef. There are advantages and disadvantages to each technique, under preferred and nonpreferred situations. Also, methods may have to be modified to accommodate the local situation or the specific study objective. Where most of the methods have been applied to artificial reefs, some (or special features of them) have been employed only on natural reefs. Since faunistically and topographically (see Chapter 3) these habitats present nearly equivalent sampling situations, there should be little difficulty in transferring any method used on natural reefs to artificial reefs. Sampling techniques that are least disruptive to existing conditions reduce the impact of variability induced by sampling and enhance the probability of reliable replicate samples.

Regardless of the environment, each method must be described clearly and referenced so that it can be duplicated. This will aid future research and permit better evaluation of the appropriateness, extent, and quality of the data. Assumptions associated with data collection should be stated (Andrew and Mapstone, 1987). Also, the perceived beneficial and detrimental features of the methods used, and recommendations for their modification should be identified.

A. Assessing Biotic Variables

Special care should be taken not to stress organisms when gathering data. This risks adding a new source of variation to the natural variability already present. The biological adage that an organism's response to stress can be to "adapt, move, or die" should be heeded.

1. Individual Parameters

Sampling design is often aimed at gathering information from individuals first, and later the whole population. Below we discuss several methods which, although aimed at the population and community level of organization, begin by gathering data from the individual organism. These data can be combined easily to form larger units. However, to gather data at a higher level and then reduce them to a smaller unit is nearly impossible. (See Ricker [1968] for a general introduction to methods for assessing life history features of fishes, which also can be applied to macroinvertebrates.)

Data may be gathered through capture of individual organisms. Capture techniques should be nondestructive to the organisms and nondisruptive to the habitat. It is necessary to know if the capture method will bias the data. For example, we have noticed that trap-caught specimens can give an unrealistic picture of feeding habits of some fishes because they tend to feed in the trap, sometimes on organisms that they rarely encounter. If the focus of the study is species specific (e.g., the growth features of a species on two different reefs), it may be necessary to use a collecting technique best suited for that particular species.

It is desirable to return captured organisms to the water quickly, with as little trauma as possible. The degree of handling and amount of time to avoid trauma varies with each species and life stage. Clearly, there is some depth, and rate of retrieval from depth, beyond which survival for some organisms is considerably diminished, owing to rapid dissolution and expansion of gases (Gotshall, 1964). If handling is not excessive, depth is not unreasonable, and retrieval rate is slow, the specimens should survive tagging and release for migration studies (Davis, 1985; Matthews, 1985; Solonsky, 1985; Hixon and Beets, 1989). Survival is unlikely for fish in which the branchial arteries have been ruptured ("gill-hooked"), or that have been dragged by bottom trawls, or captured by other actively fished gears.

Live specimens can provide much information about size, species composition, feeding (regurgitation of stomach contents from live organisms is quite successful), growth, and reproductive condition. If the animals are to be sacrificed, it is essential to optimize the data obtained from each individual. Possible uses include otoliths or bones for growth studies; parasites for levels of intensity and infestation, as well as stock identification; food and feeding habits from stomach contents (also indicates the habitat in which they have been); gonads, for fecundity, maturation, and sex ratios; and various tissues for stock identification.

Information such as reproductive and feeding behavior, local movements, territory size, competitive interaction, and habitat requirements can be gathered through remote or surface-tended collection methods. However, many of these data are best gathered through *in situ* inspection by a diver, through some remote video camera facility, or even a research submersible (Fig. 6.4).

2. Population Parameters

For the purposes of this chapter the terms stock, unit stock, and population are used synonymously. A population (i.e., ". . . a number of genetically similar individuals living in a limited framework of time and space," Emmel, 1976:89) is the sampling unit to which many biological studies are oriented. The term stock is generally considered synonymous with the term

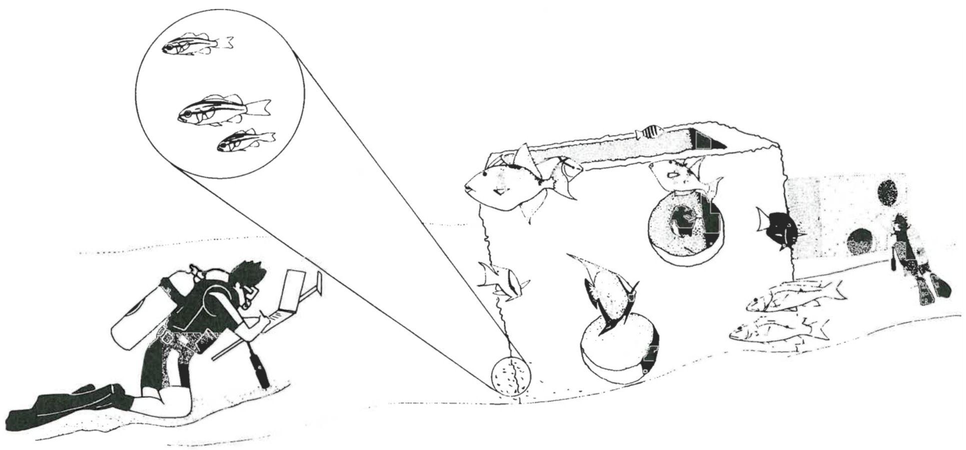

Figure 6.4 The scientist's ability to make observations and record data underwater has greatly enhanced environmental assessment programs concerned with artificial habitats. (Drawing courtesy of J. Bohnsack.)

population (Pitcher and Hart, 1982). During the past two decades there has been an emphasis on "unit stock," an operational concept used by fishery biologists and considered to be a population that can be managed as a single group (i.e., their life history features are similar enough with regard to how they interact with the fishery). Sometimes a unit stock may be a local group of several species that can be managed as a unit because the fishing techniques used to capture them do not permit a separate fishery for each species. For example, in warmer Gulf of Mexico and Caribbean waters, reef fish are generally considered by U.S. fishery managers as a single unit stock even though ten or more species from at least three families (generally snapper [Lutjanidae], grouper [Serranidae], and grunts [Haemulidae]) constitute the fishery. For a general discussion of unit stock see Gulland (1983).

Sampling from a population should be unbiased, or at least the bias should be recognized. In other words, data should be gathered from representatives of the population that were not selected for any particular predetermined attributes (e.g., size). Although it may never be possible to obtain a truly unbiased sample from a population, due to gear selectivity or behavior of the organisms, it is still our goal. Most statistical analyses have a precondition that the data are unbiased and were collected "randomly" from a larger population.

Although specific population analyses presented in fishery textbooks are beyond the scope of this chapter, we summarize methods used to measure

or assess relative abundance of the population. The purpose of most population studies is to determine population density, either by species or species group, stated in terms of size or mass per unit area. It is important to calculate the mean value of these parameters as well as their variance (Thresher and Gunn, 1986).

B. Nonvisual Methods for Fishes and Invertebrates

A broad variety of nonvisual techniques are useful when sampling in areas of low visibility (Bardach, 1959; Moring *et al.*, 1989) or when other conditions such as local weather and sea state hinder or prevent visual surveys. This is generally the situation in many freshwater lakes and streams (Crumpton and Wilbur, 1974). Techniques for sampling fishes include hook-and-line, nets (gill, trammel, trawls, etc), electrofishing, traps, creel surveys, and expert angler evaluation (Crumpton and Wilbur, 1974), as well as hydroacoustic techniques (Thorne *et al.*, 1989). Except for trawling, these methods are relatively nondestructive to habitat. Except for creel survey techniques, these have the advantage in artificial reef studies of providing individual specimens for further analysis.

Perhaps the most often used nonvisual sampling method is hook-and-line. It provides specimens for life history analysis, is relatively nondisruptive to the habitat, and often reflects the usage a reef receives (Moring *et al.*, 1989). This is especially important since artificial reefs are frequently built to improve fishing. Obviously, an excellent way to evaluate reef effectiveness is to fish the reef with gear typical to the area.

One effective way of monitoring the fishing status of a reef is to record the catch data in terms of the effort employed, as catch per unit effort (CPUE). Catch can be recorded as number of fish (often as the preferred species sought; Buckley and Hueckel, 1989) or biomass (Brock, 1985). Effort units vary according to the fishery (e.g., numbers of hooks fished per hour, fishermen or boats). CPUE has been recorded, for example, as number of fish per rod hour (Feigenbaum *et al.*, 1985), number of fish and number of strikes (by species) per unit time (Beets, 1989), and biomass per line per unit time (using a fixed number of hooks or lures; Buckley *et al.*, 1989). It is important to consistently define both variables being measured (Matthews, 1985).

Gill and trammel nets are used to assess the standing stock of fishes associated with artificial reefs (Gannon *et al.*, 1985; Relini and Relini, 1989). Usually the same standard gill net used in a commercial fishery is used in scientific sampling, but often an "experimental" net (with varied mesh sizes) is used to capture fish of different sizes (Liston *et al.*, 1985; Moring *et al.*, 1989). Vertically fished gill nets sample the entire water column (Moring *et al.*, 1989). Effort data from nets also can be recorded to make CPUE estimates (Gannon *et al.*, 1985).

Baited fish traps have been used somewhat effectively (Bardach, 1959; Miller and Hunte, 1987; Bortone *et al.*, 1988). Traps also can provide accurate CPUE data, but only if they are standardized and records are maintained on the amount of time each trap is effectively fished. Moreover, they should be calibrated with an independent survey method such as a visual inspection with SCUBA (Miller and Hunte, 1987). When using unbaited traps, Stott (1970) indicated that estimates on fish abundance may be unreliable unless similar refugia habitats exist throughout the area being sampled.

Data on catch and effort may not always be from a research effort or a fishery-independent study as described previously. Sometimes CPUE data can be from the fishery through a creel census, survey, questionnaire (Buchanon, 1974), ship logs, or fishing tournament catch records (Stanley and Wilson, 1989). Biases can result from poor recall or memory.

Mark-and-recapture techniques are often used to estimate fish and macroinvertebrate populations (Robson and Regier, 1968). Their value is limited in most studies on artificial reefs because of the migratory nature of many species and the difficulty in obtaining large numbers of organisms to mark or tag to obtain a reliable estimate of the total population. Also the low probability of survival of fishes collected at depth, handled at the surface, and returned to depth, creates a strong bias toward overestimatating population size. High variability of data makes this technique unattractive for most artificial reef situations.

In some areas, the historical use of explosives to capture fish in artisanal fisheries has been adopted in scientific studies (Bardach, 1959; Goldman and Talbot, 1976). This practice, too, has bias. For example, sharks have been attracted to the noise and also to dead and dying organisms; they not only remove important specimens from the water column but also may affect the recording performance of the researchers (Russell *et al.*, 1978). Explosives are clearly detrimental to the habitat and some species are more susceptible to them. Explosives, however, overcome the limitations caused by irregular substrates, which plague most net and some hook-and-line sampling methods. Ichthyocides such as rotenone have been used in both artisanal fisheries and scientific sampling (e.g., Smith, 1973). Poisons, however, are not considered an optimal sampling method due to their interference with recolonization as well as the lack of control over their potency and impact on nontarget organisms.

The disadvantage of nonvisual sampling is that some species, or age groups within a species, are biased for or against capture. For example, some species notably seek out traps (Bardach, 1959; Miller and Hunte, 1987), and some can be attracted to areas with various baits or baited hooks (Somerton *et al.*, 1988).

C. Visual Methods for Fishes and Macroinvertebrates

Reliable data on species identification, abundance, size, and distribution may be obtained by an observer, either operating as a diver (either skin or SCUBA) or using a submersible (Moffit *et al.*, 1989; Shipp *et al.*, 1986) or ROV (remotely operated vehicle; Greene and Alevizon, 1989; Van Dolah, 1983). Assumptions that apply include the following: species or species groups can be accurately identified (Brock, 1954; Harmelin-Vivien *et al.*, 1985), numbers of individual organisms (and schools) and their body sizes can be accurately estimated, and the proportions of these organisms counted are representative of the entire habitat (Brock, 1954).

Visual techniques do not disturb the habitat (Bardach, 1959; Harmelin-Vivien *et al.*, 1985) and are minimally disruptive to the organisms. Since they do not remove organisms from the environment they can be repeated to achieve replicate sampling. When SCUBA or skin diving are used, field gear requirements are few. Finally, visual techniques are less selective when compared to most other sampling methods (Brock, 1954).

Visual techniques are flexible and can be adapted to a variety of different situations and habitats, such as benthic and midwater situations (Rountree, 1989). They can provide qualitative data on the condition of an artificial reef (Prince and Brouha, 1974; Bortone, 1976; Myatt *et al.*, 1989), presence of organisms and community structure (Helvey and Smith, 1985), and quantitative data on the density and relative abundance of species or the entire community (e.g., Kimmel, 1985b; Dennis and Bright, 1988).

A special feature of visual assessments involves the ability to accurately record *in situ* observations for various organisms. This ability may be limited by organism mobility; their secretive, sedentary nature; or their coloration pattern (cryptic to conspicuous). Some species vary temporally in color pattern and activity, creating a problem of unequal detection (Brock, 1982; Reese, 1975). These behaviors may violate the assumption that each species and individual has an equal probability of being detected (Harmelin-Vivien *et al.*, 1985). The presence of the observer can also influence the behavior of some species (i.e., attraction or dispersal) thereby affecting detection (Collette and Earle, 1972). Our perception is that diver influences on the fish fauna are minimal because the few species that are affected seem to acclimate quickly to an observer's presence (except where spearfishing occurs). Further study of this aspect may allow the data to be adjusted for bias created by unequal detection between species, sexes, or life history stages.

Disadvantages of visual surveys should be recognized and, if possible, compensations applied to enhance the utility and validity of observations. Inadequate visibility is one of the most common limitations (Fig. 6.5). DeMartini *et al.* (1989) indicated that visibilities below 3 m drastically compromise results. Some methods require greater visibility (e.g., 5.6 m in Bortone *et al.*, 1989). An inverse relationship between object–observer distance and observability was reported by Harmelin-Vivien *et al.* (1985). It has been noted (e.g., Sale and Sharp, 1983; Sanders *et al.*, 1985) that the volume of water that can be effectively surveyed is a function of water visibility. Two common and contrasting protocols exist. In one, observers visually search a predefined area or volume of water (Fig. 6.6). In the second, a variable volume, extending to the limits of underwater visibility is searched. Where visibility fluctuates, error in detection of objects can be magnified and limit data utility. Depth also limits the amount of time spent surveying. It is essential to refer to appropriate dive tables or a diving officer to avoid physio-

Figure 6.5 Much of the methodology employed in visual assessment of artificial habitats has been developed in natural reef systems featuring relatively high water clarity.

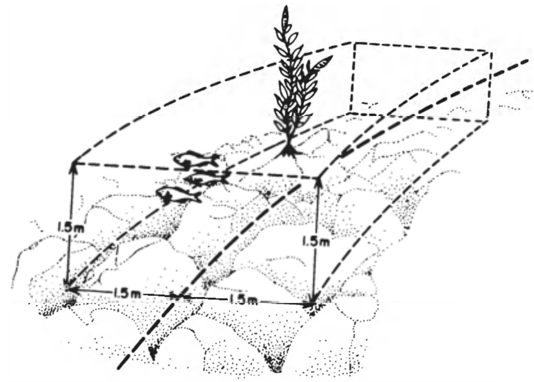

Figure 6.6 A representative diver transect, denoted by heavy line, over benthic habitat surveys a predetermined, three-dimensional horizontal and vertical distance. The boundaries are denoted by dashed lines. (From Jessee *et al.*, 1985.)

logical problems associated with SCUBA diving [National Oceanic and Atmospheric Administration (NOAA), 1975; Lang and Hamilton, 1989]. Strong surges and waves can hinder diver surveys (Spanier *et al.*, 1985). Nocturnal studies obviously require illumination but may bias the fauna recorded due to species being attracted or repelled by lights (Zahary and Hartman, 1985). Use of a red filter between the lens and the light source can reduce the impact of the typical incandescent light spectrum on organism behavior.

Perhaps the most severe limitation of visual faunal estimates is their underestimation of real abundance (Wickham and Russell, 1974; Harmelin-Vivien *et al.*, 1985; Buckley and Hueckel, 1989). This is related to the combination of spatial heterogeneity of the substrate of artificial habitats and the cryptic nature of many organisms (Brock, 1982; DeMartini *et al.*, 1989), which can reduce any individual organism's probability of being observed and result in underestimates of population density (DeMartini *et al.*, 1989).

Conceptually there are differences between the method used to gather data and the method used to record them. Data should be recorded as observations occur, especially in the case of underwater assessments, unless safety is compromised. Rarely is there any advantage to delay recording diver-observation data. However, Rutecki *et al.* (1985) indicated a preference for recording data upon completion of the survey. Obviously, increased delay (even minutes) between making an observation and recording it increases the probability of error.

Plastic slates of various types (Helfman, 1983) are used for most data recording. Typically, data are written directly on the slate, usually with

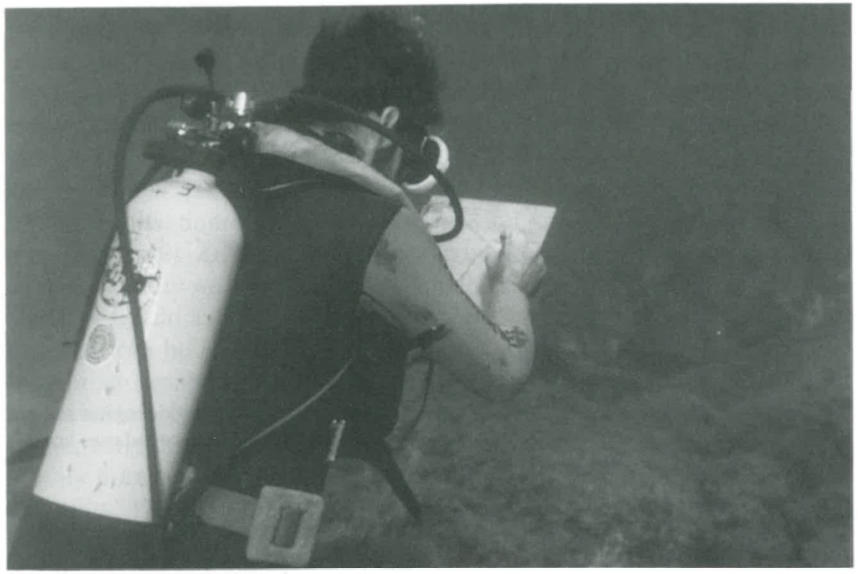

Figure 6.7 A pencil and a plastic slate, roughened with sandpaper, is a commonly used data-recording method. This diver has prelisted the species likely to be seen on the slate, but many researchers think this biases the sampling survey.

pencil (Fig. 6.7). Plastic (e.g., Mylar) or waterproof paper is sometimes affixed to the slate. Slates are inexpensive, and adaptable to each survey. A disadvantage is that valuable observation time may be lost if the divers must look away from their observations to see what they are recording. Photocopies of original data sheets can serve as a "back-up" and should be kept separate from the originals until the data are stored in a computer data base. There also should be a duplicate of the computer data base.

The best recording methods are those in which data are recorded only once, in a format that is immediately archivable. This reduces the chance of error due to transcription. Some type of data recorder, such as that found on various computer input devices (i.e., diskette or tape media), is probably the best overall method. There have been only a few such studies that have used the technique to date (Hixon, 1980). Most certainly, direct computer input of data will be the mode of *in situ* data recording in the future (see Williams and Briton, 1986).

Alternatives to direct computer input are audio and video recording devices that make a permanent copy of the records in a format that is readily transcribed to an archive. These techniques are used both above and below

the water (Alevizon and Brooks, 1975; Jones and Chase, 1975; Larson and DeMartini, 1984; Bortone *et al.*, 1986, 1989; Ebeling *et al.*, 1980; Greene and Alevizon, 1989). Audio- and videotapes produce a reasonably permanent record, allow recording of many data items rapidly, can be reexamined to verify an observation, and gather data initially not considered important. Moreover, they represent a relatively inexpensive way to inform nontechnically oriented individuals about a specific study or a data collection method.

The disadvantage to this system is that all data must be transcribed from the film or magnetic tape. Transcription time is usually equal to or greater than the time required to make the recording. Also, these devices are expensive, of varied quality, and require some special training (such as the use of a full-face mask for audio recording) and handling (audio and video recorders are easily damaged by moisture). The limited resolution and field of view of video recorders (not only but especially in murky or poorly lit waters) may preclude accurate species identification (Seaman *et al.*, 1989b) and abundance estimates.

Still photography has been used successfully in data recording (see review by Weinberg, 1981). It has the advantage of being relatively inexpensive and easy to operate (Fig. 6.8). Photography has been used to record relatively sophisticated data on populations and geographic conditions (Lundälv, 1971), but in some instances it produces notably unreliable data, especially under poor visibility. Fishes are extremely difficult to identify or even see from a still photograph, as their movement against a background is often essential for recognition (Bortone *et al.*, 1986). Photographic techniques are discussed in the section on sessile communities below.

Remote recording devices have a definite advantage at depths greater than those safely accessible by SCUBA for some types of data. ROVs and manned submersibles are usually equipped with audio and video recording, either directed to the surface or recorded *in situ*. Automatic depth recorders and positioning systems are invaluable for determining the exact position and extent of some reefs. Recent use of transponders to produce a three-dimensional-like image of submerged objects shows tremendous potential for locating, identifying, and assessing the form of benthic artificial structures (Lukens *et al.*, 1989).

In the following sections we present a more specific discussion of the attributes of visual methods. Readers also are advised to consult Harmelin-Vivien *et al.* (1985) for a comprehensive introduction.

1. Observer Limitations

Most visual assessment methods use a diver as the observer and several limitations and factors can affect performance. Diver ability is an obvious

Figure 6.8 Still photographs are a valuable way to document the reef area for later comparisons, as well as for showing the reef to interested individuals, but they do not permit precise quantitative data acquisition on the nektonic community.

variable that must be considered. Divers unfamiliar with diving and, therefore, preoccupied with diving techniques will be diverted from their study objective. Limited visibility, turbulence, or extreme depth may exacerbate problems among even experienced divers. A potential source of error is the observer's ability (usually related to experience) to accurately count, measure, record, and identify species (Stephan and Lindquist, 1989; Harmelin-Vivien *et al.*, 1985). Narcosis at depths greater than 30 m can occur and lead to inattention, but this can be mediated to some extent by using Nitrox (a mixture of nitogen and oxygen) as a breathing gas (Seaman *et al.*, 1989b). Exposure to cold and diver fatigue can cause disorientation and lack of concentration (Harmelin-Vivien *et al.*, 1985). Participation of an individual assigned the role of "dive safety officer" is essential.

Divers can influence the behavior of organisms. For example, Harmelin-Vivien *et al.* (1985) noted lower fish abundance when successive visual surveys were conducted over the same area. Kimmel (1985b), however, could detect no difference in abundance of fishes when replicate surveys were conducted along a transect. Some organisms are attracted or repelled by features such as lights and bubbles (Smith and Tyler, 1972). However, the

use of a dive light (even diurnally) can facilitate the identification of organisms in dark recesses, at depth, or under other dim light conditions.

Often data are taken by both divers operating in pairs for safety purposes. Divers should have the same skill levels with regard to species identification, enumeration estimation, and size judgement. Several studies have evaluated variation among observers. Beets (1989) observed that divers were similar in most aspects of data recording except when surveying uncommon or cryptic species. Bortone *et al.* (1989) indicated disagreement between two divers with regard to some parameters, such as number of species and species diversity, when using some visual census methods but not with other methods. Epperly (1983) found that even though divers agreed on identification and number of species, individual observer variation was considerably less than between-observer variation for enumeration of individuals.

Another source of variation among observers is attributable to their visual perception ability. Morrison (1980) indicated considerable individual difference for perceiving objects and ability to see moving or stationary objects. Individuals also differ in ability to see objects of different contrast (Ginsburg and Conner, 1983) and to perceive dimensions of objects (Enns and Rensink, 1990).

Clearly when multiple observers are recording visual census data it is necessary to measure their comparability, especially since some observers may produce comparable results using one visual survey method but not others. In any case, diver training seems to improve agreement both within and between observers (Galzin, 1985, in Harmelin-Vivien *et al.*, 1985).

2. Species Identification

Correct species identification may be the most important part of a visual census data base (Harmelin-Vivien *et al.*, 1985). An error here may render even the simplest of species presence or absence analyses completely useless. This procedure may require much training, particularly in areas of high species diversity (e.g., coral reef communities in the Caribbean may potentially have over 400 fish species in residence). Skill is necessary to be able to accurately record the identity of many species of different sizes, swimming behavior, and color patterns. Some species may have separate adult male and female color patterns with different patterns as juveniles, and at night (Starck and Davis, 1966). There may be similarity of color patterns among species, also, due to adaptation or genetic similarity.

To avoid identification mistakes, some studies have eliminated small, cryptic species that have a high probability of being misidentified or missed completely (e.g., Greene and Alevizon, 1989). This is especially practical when the objective is to assess a target species or group of species (e.g., Buckley and Hueckel, 1985; Jessee *et al.*, 1985). Species may be included as members of a group such as a genus or family, or given a numerical designation that at least separates one species from another (e.g., Bortone *et al.*, 1986). A photograph or capture of a voucher specimen can facilitate identification. Spanier *et al.* (1985) used a trammel net to collect specimens to verify species and sizes. One of us (J.J.K.) has used quinaldine (an anesthetic) and a "micro-barb" spear to collect voucher specimens.

Humans are thought to rely on a "search image" for sight identification (Enns and Rensink, 1990; Corbetta *et al.*, 1990). During a survey it may be preferable to make a visual sweep looking for only species with a similar body profile, size, life stage, behavior, color, position in the water, or relationship to the substrate (Brock, 1954; Molles, 1978; Grove and Sonu, 1985; Bohnsack and Bannerot, 1986; Moffit *et al.*, 1989). Greene and Alevizon (1989) referred to these search image groups as discrete census groups. This strategy may affect the overall sampling if additional time is required for repetitive visual sweeps for each census group.

3. Enumeration

The enumeration of species (or species groups) also requires some special attention, particularly in view of the extreme range in abundance and patchy distribution that may occur on both natural and artificial reefs. It is relatively easy to accurately count a few individuals of a few (target) species in relatively clear water. Usually, however, conditions are less favorable.

To reduce enumeration errors several things should be considered. (See Fig. 6.9.) Species enumerations generally underestimate the actual number present because of the cryptic and secretive nature of many species (Sale and Douglas, 1981; Sale and Sharp, 1983; Harmelin-Vivien *et al.*, 1985). Wickham and Russell (1974) found that for pelagic species around midwater structures there was a tendency for divers to overestimate small populations and underestimate larger populations (see also Bevan *et al.*, 1963; DeMartini and Roberts, 1982). Harmelin-Vivien *et al.* (1985) found that divers had little difficulty in correctly enumerating schools of fish of six or seven individuals but the error of enumeration increased at higher school densities. To deal with this problem several studies have used abundance groups. Observers form a search image of a group comprised of a certain number of individuals and then determine how many groups they see. The group size varies with each study but generally ranges from 20 to 1000 in size (Harmelin-Vivien *et al.*, 1985). Instead of counting individuals, Workman *et al.* (1985) counted in abundance units (i.e., 1, 1–20 individuals; 2, 21–50; 3, 51–100); Russ (1984) and Kingsford (1989) used base 3 logarithms; Sanders *et al.* (1985) enumerated particularly abundant schools by orders of magnitude (i.e., 1×10^2, 1×10^3); Gladfelter *et al.* (1980) used doublings for abundance categories. Other researchers have used words to describe abundance units (i.e., rare, 1; occasional, 2–5; frequent, 6–10; common, 11–25; abundant, > 25; Bortone, 1976). Some studies have used abundance categories but have not clearly defined them (e.g., Hastings *et al.*, 1976; Smith *et al.*, 1975).

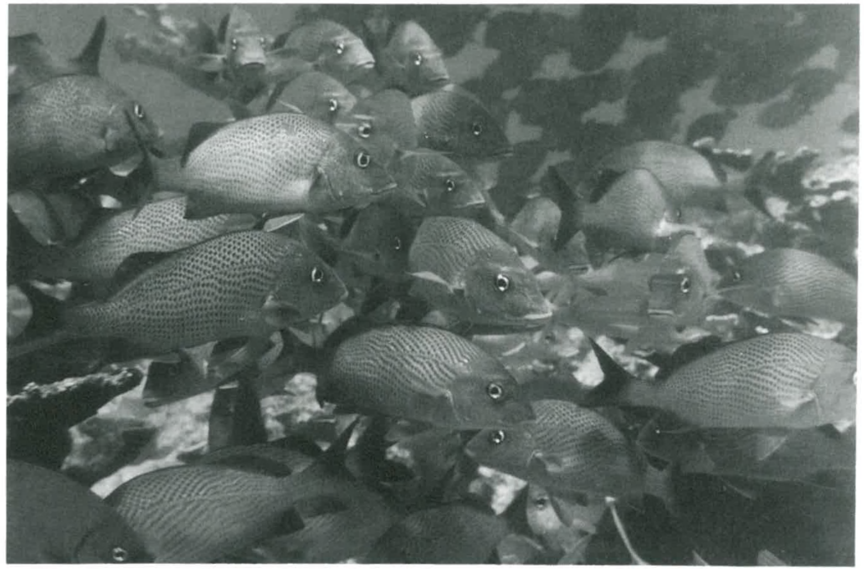

Figure 6.9 In areas of high species diversity and abundance, enumerations can become somewhat complex. For example, an artificial reef in the Bahamas held large populations of sailor's choice, *Haemulon parrai* (foreground), and three surgeonfish species, *Acanthurus* spp. (background).

Counting by abundance units may considerably facilitate the enumeration process and may lessen the chance of error. Since the species abundance data are often transformed for statistical analyses, recording data in this fashion approaches a log transformation. The chance for error when counting by abundance units comes from incorrectly assigning a group to an abundance class when the group is close to the boundary of a unit (e.g., assigning a group of 25 individuals to an abundance unit of 26–50 individuals; Harmelin-Vivien *et al.*, 1985).

When counting a school of mixed species it is preferable to estimate the number of the entire school and then estimate the proportion or percent composition that each species makes to the school. We have found that mixed-species schools are usually composed of similar appearing species of similar size making species specific enumeration difficult, especially in conditions of poor visibility.

4. Size Estimation

Size estimates are often recorded to evaluate the importance of a species or species group at an artificial habitat. (Additional discussion of the rationale for such data is provided in Chapter 3.) Generally, size is estimated as length and later converted to weight or biomass (Brock and Norris, 1989) using data from studies on length–weight relationships (e.g., Dawson, 1965; Bohnsack and Harper, 1988). Bohnsack and Bannerot (1986) used a rule mounted on a stick and held at a distance to aid in estimating fish body length. Others have used rulers, calibrated slates, surveyor's tape, or some other object of known size as a reference (McKaye, 1977). It is important to have a reference of known length due to the magnification effect of water. Bell *et al.* (1985) and DeMartini *et al.* (1989) found that with practice observers could reliably estimate lengths underwater.

To avoid making small errors in estimating length, some studies have estimated fish size and assigned them to size groups. Harmelin-Vivien *et al.* (1985) classed fish as being < 12 cm long, 12–18 cm, or > 18 cm; Matthews (1985) used a slightly different size grouping: < 6 cm, 6–12 cm; 12–20 cm, > 20 cm. Beets (1989) assigned fish to 2.5-cm units; McCormick and Choat (1987) used 5-cm length classes, and Bell *et al.* (1985) assigned fish to 10-cm length units. Our experience indicates that even inexperienced divers have little difficulty in correctly assigning fish to 5 cm size groups.

Schooling fish do not present a major problem when estimating lengths since fish of equal size tend to school together so that it is not necessary to estimate the size of all individuals in the school but, only one or a few members. Upon occasion different sized individuals may occur within a school. The human eye has the ability to recognize odd individuals (with regard to size or species) in a school, making them relatively easy to detect (Theodorakis, 1989).

5. Visual Census Methods

A plethora of census methods have been employed, with variations peculiar to each investigator or modified to accommodate special circumstances including environmental conditions, study objectives, ability of the investigators, and availability of resources. While flexibility is the greatest advantage of visual census methods, directly comparing data obtained by different methods is difficult if not impossible. Below, the methods are presented by major type with regard to whether or not the observer moves in a predetermined direction (transect), remains stationary (point count or quadrat), or swims haphazardly (species–time random count). Some methods do not fit easily into any category.

a. Transect This is the oldest (Brock, 1954; Bardach, 1959) and the most frequently used method of visually surveying fishes and macroinvertebrates. It also has been the most widely applied to natural and artificial reef surveys, probably due to its simplicity and well-defined protocol. However, its application varies considerably among users. The transect or strip transect is censused by a diver, or sometimes by submarine or ROV, moving in a direction of predetermined distance or time. Timed transects are usually employed when movie or video cameras are used, as the exposure time of the film or videotape often predetermines transect "length" (Alevizon and Brooks, 1975; Bortone *et al.*, 1986; Seaman *et al.*, 1989b). The physical distance of transects can vary considerably. Thresher and Gunn (1986) surveyed a 500 m transect while being towed from a boat. Prince *et al.* (1985) used a transect length of 180 m. Anderson *et al.* (1989) used transects as short as 5 m. Other studies have employed transects of varying lengths (Jessee *et al.*, 1985; Anderson *et al.*, 1989; DeMartini *et al.*, 1989).

When the bottom is heterogeneous with regard to habitat it is especially important to keep the transects short. A long transect can traverse many habitats (and may include ecotonal areas that are noted to concentrate or combine faunas; Anderson *et al.*, 1989) and provide data that do not allow the variation in population parameters to be attributed to habitat association (Jones and Chase, 1975). Harmelin-Vivien *et al.* (1985) suggested using long transects for homogeneous areas and shorter transects for heterogeneous zones. Transects perpendicular to the depth contour should be avoided as habitat and species mix often change with depth (Harmelin-Vivien *et al.*, 1985).

Divers generally swim along the transect, recording species abundance and size in a three-dimensional corridor that can vary considerably in width and height (Fig. 6.6). Brock (1954) used a transect width of 6 m; Ambrose and Swarbrick (1989) and Anderson *et al.* (1989) used a 1-m-wide transect to study recruitment. Most studies have employed a width of either 1 or 2 m (e.g., McCain and Peck, 1973; Kimmel, 1985a; Lindquist *et al.*, 1985). The rationale for narrow width is that the diver can observe smaller fishes (Harmelin-Vivien *et al.*, 1985; Sale and Sharp, 1983) and easily estimate it underwater (i.e., 1 m is about one arm length from the transect).

Width appears to be a critical component of transect methodology. Sale and Sharp (1983) found that transects of various widths returned different population density estimates in the same survey area. Much of this difference derives from the differential detection of some species due to their size and behavior, hence observability. Harmelin-Vivien *et al.* (1985) suggested using different widths for different species (e.g., 1 m or less for smaller, cryptic species such as gobies (Gobiidae), and 3–5 m for larger species such as parrotfish (Scaridae). Kevern *et al.* (1985) used a transect width equal to one-half the estimated visibility on either side of the transect line. While this is a practical modification of the method, Sale and Sharp (1983) do not recommend using variable width transects within a study as this may make density comparisons unreliable.

Because most fishes and some macroinvertebrates are nektonic and move about considerably during a visual survey, it is important to maintain constant area and time parameters between surveys or samples within a study. Comparison of two transects of identical length but sampled for different lengths of time can produce incomparable data. Caughley *et al.* (1976) indicated that the probability of missing species on aerial transect surveys increased with the speed of the observer. Some studies have tried to maintain constant swimming speeds to reduce this problem (DeMartini *et al.*, 1989; Lincoln-Smith, 1988), but this can be difficult to do under varying current conditions.

Often a preplaced line is used as a guide for defining the census area and a reference for estimating transect length and width. Setting a transect line requires effort to place and remove the line (Kimmel, 1985b), and habitat disturbance; impact on the observability of organisms potentially can occur.

Brock (1954) developed a sampling protocol for transects that outlines criteria for the inclusion or exclusion of organisms from the count. A diver includes fishes within his/her forward view and within the zone of predetermined width. If an individual organism is nearby but does not enter the zone, it is not counted. An individual, once counted, is not recounted if it reenters the area. This may be difficult to determine under some circumstances because of the multitude of individuals and their movement. Fishes behind the diver are never counted. If one member of the school enters the area the entire school with which it is associated is included in the enumeration. Species found to be attracted to the area due to the activities of the diver-observers should not be included in the enumeration (Harmelin-Vivien *et al.*, 1985). In addition, the distance and angle of the observable field should be kept constant to maintain a consistent level of detectability for all organisms (Keast and Harker, 1977).

The vertical dimension of transects has not been consistently reported (Harmelin-Vivien *et al.*, 1985). Heights from 1.5 m (Jessee *et al.*, 1985) to the vertical limits of visibility have been proposed (Hollacher and Roberts, 1985). Since a three-dimensional area defines the living space of the aquatic community, the definition of the vertical component of the space is critical to the accurate determination of density estimates (DeMartini *et al.*, 1989).

b. Point Count (Quadrat) While the transect method has the observer moving through a zone to enumerate fauna, in a point count or quadrat technique the observer remains at a fixed point or place and conducts the enumeration within a prescribed area or volume. This general method is particularly useful when movement by the observer is difficult. This can occur when viewing from a submersible, a ROV, or when a video recording device is used (Smith and Tyler, 1973; Bortone *et al.*, 1986, 1989; Shinn and Wicklund, 1989).

Bohnsack and Bannerot (1986) suggested using a random series of kicks and randomly selected compass headings to choose sample points. However, if a random method of site selection is chosen, it may be necessary to avoid ecotonal areas. Most artificial reefs, however, are not large enough to allow the random selection of observation points, thus systematic surveys are probably more appropriate. Once the sampling point is chosen, the observer determines distance to the circumference of the sample area (or lateral dimensions in the case of a quadrat; Slobodkin and Fishelson, 1974). This is facilitated by use of a weighted line of fixed length.

Quadrat area dimensions vary from study to study. Bohnsack and Bannerot (1986) determined that in the clear waters of the Caribbean a radius of 7.5 m was effective. Bortone *et al.* (1989) used a radius of 5.6 m (area, 100 m^2) where water clarity was more limiting. Stephan and Lindquist (1989) used a radius of 4 m on a shipwreck reef where visibility was generally low. Laufle and Pauley (1985) used a 6 m × 6 m square quadrat to estimate the fauna of an artificial reef. Luckhurst and Luckhurst (1977) used a smaller quadrat (3 m × 3 m) to study fish recruitment.

A problem in selecting radius length for a point count census is that the longer the radius, the more likely one will miss observing a smaller, cryptic individual at the perimeter. This might be countered somewhat by closely inspecting the survey area after the prescribed time of observation. While this increases the probability of observing rare, cryptic, or diminutive organisms for inclusion in the total faunal list, such additions should not be included in the data set for quantitative analysis since sample time is inconsistent between surveys.

In a point count survey the observer turns at the central point for a fixed interval of time (usually five minutes; Bohnsack and Bannerot, 1986). Zahary and Hartman (1985) used a survey time of 15 min or until no new individuals were recorded for 3–5 min. One should be aware of the need to keep the variables of time and area in any protocol constant. Bortone *et al.* (1986, 1989) determined that the amounts of time and area allotted to a method were perhaps more significant in determining the community than the type of method used. Thresher and Gunn (1986) used an "instantaneous" time interval in their point counts and concluded that the density estimates obtained were relative to amount of observation time.

c. Species–Time Random Count Perhaps the most innovative visual survey technique is the species–time random count method developed by Jones and Chase (1975), Thompson and Schmidt (1977), and Jones and Thompson (1978). It is referred to in later studies as the rapid visual technique (RVT) (e.g., DeMartini and Roberts, 1982) or the rapid visual count (RVC) (e.g., Kimmel, 1985b). It has the unique feature of accounting for species by merely listing them and scoring them according to the time interval in which they were first observed during the survey.

Typically a diver swims in a random (i.e., haphazard) fashion over the survey area and lists species in the time interval in which they are initially observed. A species is given a score of 5 if it is first seen within the first 10 min of the 50-min observation period; a 4 if it is first seen in the next 10 min, and so forth. The method is based on the principle used in bird faunal surveys in which abundant species are likely to be observed first and rare species are likely to be seen last (Beals, 1960). Jones and Thompson (1978), following the recommendation of Gaufin *et al.* (1956), determined that eight, 50-min surveys would account for 93.5% of the species in highly diverse coral reef areas. Replication reduces the problem of the overweighting influence of observing rare species during the first time interval. In this example the maximum achievable score of relative abundance is 40 (e.g., score of 5 for each of eight surveys), and the least possible score is 1 (e.g., a species seen last during only one of the surveys).

The method produces faunal lists that include more species than other methods and is particularly useful for recording rare species (Kimmel, 1985b; Bortone *et al.*, 1986, 1989). Its disadvantage is that abundance is really a score of probability of encounter and is complicated by the varied spatial distribution patterns of species in the assemblage (DeMartini and Roberts, 1982). The species–time random count method produces data, therefore, that are not comparable with data from most other faunal surveys. The other visual methods, described above, can report data in terms of density per area or volume, while the species–time random count technique cannot.

To overcome this problem Kimmel (1985a,b) developed the visual fast count (VFC) method in which the diver records the actual number of individuals of any species seen within a preselected biotope. Expected frequencies were also used in place of the arbitrary interval scores of Jones and Thompson (1978). This modification of the species–time random count technique may prove to be a useful compromise of the best features of several techniques and could be adjusted to incorporate species size and the area surveyed.

d. Special Methods Other visual assessment methods differ from the preceding categories and are unique to a particular study. In some studies, individuals are enumerated until they are all counted (Downing *et al.*, 1985; Alevizon *et al.*, 1985; Sale and Douglas, 1981; Moffit *et al.*, 1989; Hixon and Beets, 1989). This method has been used, usually without regard to time or area, under the premise that the area is small enough for the observer to count every organism that is the target of the survey. It has been applied most often to natural and artificial reefs that are discrete and limited in size.

Bortone *et al.* (1986) surveyed a fish fauna by using strobe-illuminated, still photographs. A color transparency was exposed at 10-m intervals in each of the four major compass headings along a 100-m transect. Although a permanent record of the habitat and fauna was provided, the temporal variability and the resolution of the photographs inhibited accurate species identification and enumeration.

Boland (1983) used two video cameras to record the fauna in stereo. Organism size and transect width could be determined from the tapes. This method, while expensive, shows promise where remotely operated visual surveys are required.

The spot-mapping technique of Thresher and Gunn (1986) requires that observers make detailed maps indicating the location of individuals relative to habitat features. Nonmoving organisms, such as corals, plants, and sponges, or sedentary, territorial species are better served by this technique. The technique requires considerable time and effort but may provide detailed information about home range and habitat associations.

e. Visual Assessment Comparisons Comparison of results of visual census techniques has occurred because of the recognized need to rigorously examine sources of bias in the data (Barans and Bortone, 1983; Thresher and Gunn, 1986). Brock (1982) compared visual transect with rotenone survey data and concluded that visual census of fishes should be restricted to diurnal species and that transect surveys underestimated cryptic and abundant species. Christensen and Winterbottom (1981) established coefficients to make visual survey results comparable to surveys conducted with rotenone, so as to adjust the abundance of cryptic and secretive species. Sale and Douglas (1981) noted that visual surveys of fish assemblages are comparable to ichthyocide collections in restricted areas, but these methods each sampled slightly different components of the fauna.

Sanderson and Solonsky (1986) compared the species–time random count (RVT) with a strip transect (STT) and found that both methods produced a similar qualitative description of the fauna and that the RVT was more cost-effective. The STT is preferable, however, when quantitative data are required (Sanderson and Solonsky, 1986). DeMartini and Roberts (1982) found good qualitative agreement between the species–time random count method and a total count method, but the RVT overemphasized the importance of widespread, rare fishes and underestimated clumped but abundant species. Comparison of the RVC, VFC, and transect methods indicated reasonably good qualitative similarity (i.e., presence or absence) for the 80 species common to all three methods; however fewer species were encountered using the transect technique. Quantitatively, VFC results were similar to the transect method (Kimmel, 1985b).

Thresher and Gunn (1986) compared the spot mapping technique to six other methods to estimate the density of highly pelagic fishes. Transect and point count methods gave different estimates of density, but the point count estimate was less variable.

Day and night surveys using six methods (i.e., transect, point count, species–time random count, cinetransect, cineturret, and still photography) at two different reef habitats produced comparable overall qualitative (i.e., species presence or absence) faunal descriptions (Bortone *et al.*, 1986). Methods that produced more "information" (i.e., greater numbers of species and individuals) were better at categorizing faunas in terms of their habitats. The conclusion from this analysis was that observation time and size of the survey area, and not the particular census method, were the most important aspects of any method for describing a community (Bortone *et al.*, 1986). Another study (Bortone *et al.*, 1989) compared transect, point count, and species–time random count techniques but standardized the data for survey time and area. Before standardization, the point count and transect methods produced statistically similar faunal parameters. The point count method provided a faunal description that was less variable and had a higher and less variable species diversity parameter for the assemblage. After standardization, however, the transect method was more efficient in sampling the number of individuals. This was probably due to the closer proximity of the observer to the faunal elements during the transect.

The use of multiple methods to make accurate and precise determinations of the density estimates of reef fish assemblages may seem redundant but will probably continue. From a technological view, it offers a way of comparing and refining the assessment methods. In addition, using multiple methods for assessments is thought by some to be the only valid way to evaluate the reliability of density estimates and to determine sources of bias in the data (Thresher and Gunn, 1986).

D. Nonvisual Methods for Studying the Sessile Community

The benthic, sessile component of many aquatic communities can be effectively sampled with various surface-tended devices such as trawls, dredges, and grab samplers (Thorson, 1957; Needham and Needham, 1962; Lebo *et al.*, 1973). A portion of the habitat is damaged or removed by these gears, however. Further, they are inefficient for assessing the irregular substrate typical of artificial habitats. Other methods have been developed.

Most studies on artificial reefs use some type of settling plate that is positioned temporarily on the reef, then removed and later examined in the laboratory (Fig. 6.10). Harriott and Fisk (1987) compared types of settlement plates in experiments on coral recruitment off Australia. The advantage of using removable sampling surfaces is that they can be experimentally manipulated to test various hypotheses (Hixon and Brostoff, 1985; Schoener and Schoener, 1981). Russ (1980), for example, isolated several plates with wire mesh to investigate the effect of grazing animals on the sessile community.

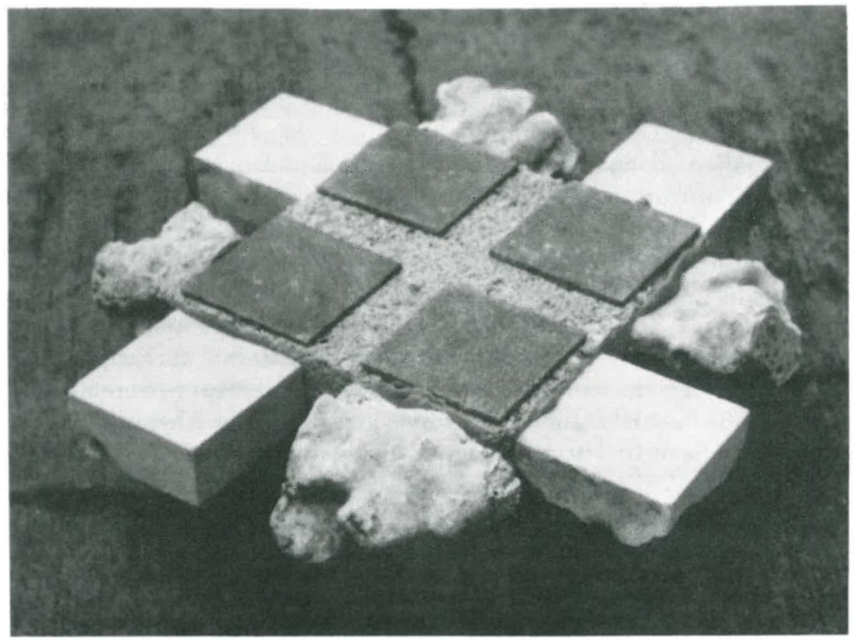

Figure 6.10 These four settling plates of PVC plastic are mounted on a concrete block for study of epibenthos on artificial habitat (from Hixon and Brostoff, 1985).

Plate size has varied among studies (e.g., 14.5 and 103 cm^2, Osman, 1977; 50 cm^2, Hixon and Brostoff, 1985; 15 cm^2, Fitzhardinge and Bailey-Brock, 1989). Materials used as settling plates (Fig. 6.10) include dead *Acropora* coral (Rogers *et al.*, 1984); cement blocks, terra cotta tiles, and plexiglas plates (Birkeland, 1977); red bricks (Woodhead and Jacobson, 1985); polyvinyl chloride (PVC) plastic pipes (Birkeland *et al.*, 1981); and corrugated quarry tiles (Baggett and Bright, 1985).

A suction or air-lift sampler is effective for removing members of the benthic community from the reef surface (Gale and Thompson, 1975; Gannon *et al.*, 1985; Benson, 1989; Hueckel *et al.*, 1989). Meanwhile, infauna can be sampled adjacent to the reef either by pushing a tubular device into the the bottom to obtain a core of predetermined depth (Hueckel *et al.*, 1989), or with various grab samplers. These methods may be more effective when operated by divers rather than by remote.

In the laboratory, sample analysis typically includes the number of species, number of individuals, and biomass (Bailey-Brock, 1989; Hixon and Brostoff, 1985; Osman, 1977; Fitzhardinge and Bailey-Brock, 1989). Percent cover has been estimated for sessile invertebrates from photographs of plates or by measuring the actual surface (Schoener and Schoener, 1981; Bailey-Brock, 1989). Carter *et al.* (1985) examined the relative importance of sessile organisms both on the surface of the settling plates and observed those that grew over others. Rutecki *et al.* (1985) directly measured the length of the periphyton, *in situ*, to determine the growth of plant material attached to the reef surface. The time and duration of immersion of the settling plates is critical to the understanding of the colonization–defaunation aspects of succession (Fitzhardinge and Bailey-Brock, 1989).

E. Visual Methods for Sampling the Sessile Community

Underwater field observations of organisms started as early as 1785 when Calvolini collected specimens from submarine caves in Sorento, Italy (Riedl, 1966). Since that time many survey techniques have been developed, primarily by terrestrial plant ecologists. Qualitative studies establishing species lists and zonation patterns remain popular (e.g., Goreau, 1959; Goreau and Wells, 1967; Goreau and Goreau, 1973; Scatterday, 1974; Bak, 1975; Colin, 1977; Taylor, 1978; Wheaton and Jaap, 1988; Jaap *et al.*, 1989). Quantitative benthic research began with minimal area concepts in submarine ecology (Gislen, 1930). A popular method of quantifying macroinvertebrates and plants has been through the use of quadrat sampling along transects, as developed by phytosociologists and marine ecologists (Stoddart, 1969; Scheer, 1978). Transects have been of great utility to document species of

invertebrates and algae (Hueckel and Buckley, 1989). Diver or remotely operated photographic techniques (both still and video) have been used successfully (Lundälv, 1971; Bohnsack, 1979; Weinberg, 1981; Van Dolah, 1983; Baynes and Szmant, 1989; Bergstedt and Anderson, 1990).

Macroinvertebrates and macroalgae (i.e., those easily seen with the naked eye; Hueckel and Buckley, 1989) are the life forms that are generally surveyed using still photographs or quadrat sampling. The still photographic technique has been used to estimate species composition, number of organisms and even percent cover and relative density of the sessile community (Bohnsack, 1979; Baynes and Szmant, 1989). Lundälv (1971) used stereo photography for the *in situ* assessment of size and growth of sessile organisms.

F. Assessing Abiotic Variables

Many aspects of assessing and monitoring biotic variables also apply to the methods used to measure abiotic variables, including surface- and SCUBA-tended methods (Loder *et al.*, 1974). Information on variables such as date of deployment, distance from shore, and reef construction materials can be obtained from the permit application forms that must be filed for most reef construction projects. Bortone and Van Orman (1985a) were able to obtain information on the abiotic aspects of the reefs they studied from a previously prepared artificial reef atlas for Florida waters (Pybas, 1988). In this section we review some of the more widely proposed methods of measuring the most frequently studied abiotic features of artificial habitats.

1. Measuring the Habitat

It is much easier to determine the three-dimensional size parameters of artificial habitat before it is deployed (Moffit *et al.*, 1989). This description includes the area and volume of each material used, as well as surface texture. This, however, is rarely done. Usually a dive team makes an *in situ* survey to determine the amount of each type of material used (Bortone and Van Orman, 1985a) and their surface texture. Rugosity (i.e., the total amount of surface area available for habitation; Chandler *et al.*, 1985) can be used to examine the ratio of living biomass to surface area (Luckhurst and Luckhurst, 1977). Such evaluations must be done consistently and based on quantitative criteria.

To be able to describe the role of reef building materials and surface types in community succession, the history of deployment must be known. This is often determined from permit records or published information, including newspaper accounts or navigation charts for shipwrecks (Ambrose and Swarbrick, 1989; DeMartini *et al.*, 1989). The periodic addition of materials to reefs should be documented.

Divers often use devices as simple as a tape measure for the *in situ* measurement of artificial reefs (Bortone and Van Orman, 1985a; Alevizon *et al.*, 1985). From these measurements it is possible to calculate cross-sectional area and reef volume (Grove and Sonu, 1985).

In addition to using diver measurements (Carter *et al.*, 1985; Feigenbaum *et al.*, 1985), some studies have relied on depth recorders and positioning relative to buoys or LORAN coordinates to determine habitat configuration (Gannon *et al.*, 1985). Recently sidescan sonar has been employed to determine position, size, and extent of submerged artificial reef materials (Lukens *et al.*, 1989; Matthias, 1990).

Although overall reef configuration may be carefully planned before deployment, its actual orientation in place is often quite different because of problems during deployment. Reefs should be carefully monitored after deployment, as well, to detect movement.

2. Sediments and Substrate

The composition of bottom materials on which artificial habitats are deployed can be studied by remote, surface-tended operations. Most studies have used divers, however (Mathews, 1985). Sediments can be retrieved by hand (Bortone and Van Orman, 1985a) or with coring devices similar to those used to measure the infauna (Mathews, 1985). Baynes and Szmant (1989) determined the erosional potential and impact of water flow on the substrate by using "clod cards."

Knowledge of sediment composition and texture is important in reef placement (Chandler *et al.*, 1985). This can be accomplished visually for coarse sand or gravel substrates (Carter *et al.*, 1985), or by laboratory techniques for smaller grain sizes (Folk, 1968; Baynes and Szmant, 1989). To determine the load bearing suitability of a site, a penetrometer (Jones, 1980) or an even more elaborate vane shear apparatus (Dill, 1965) could be used.

Charts and maps, if accurate and of adequate detail, are good sources of information on bottom topography and profile. DeMartini *et al.* (1989) and Ambrose and Swarbrick (1989) used maps and a planimeter to determine the area covered by habitat types such as kelp cover.

3. Water Column

From an engineering perspective, as discussed in Chapter 4, and from a biological point of view water currents are clearly an important variable to measure when studying artificial habitats. Lindquist and Pietrafesa (1989) used firmly anchored current meters at different depths. Diver-implaced current meters and hand-held compasses also have been used (Baynes and Szmant, 1989). Bray (1981) used a stopwatch to measure the time that waterborne particles moved a fixed distance. Sanders *et al.* (1985) utilized a lunar phase index to quantitatively characterize this variable.

Most studies of light penetration use a Secchi disk in both a horizontal and vertical mode (Lindquist and Pietrafesa, 1989; Baynes and Szmant, 1989). For greater precision other studies have measured the amount of light penetrating to a specified depth using a measure of *in situ* irradiance both at the surface and at depth (Carter *et al.*, 1985). Although not strictly considered an abiotic feature, chlorophyll *a* is clearly an important consideration in evaluating aquatic productivity (Ardizzone *et al.*, 1989).

VI. Summary

Aquatic ecology is a rapidly changing field, and one should not expect the theory or methods presented herein to remain static. When designing an assessment or monitoring study the reader is encouraged to contact personnel at research institutions for the latest information regarding implementation of these methods. Moreover, individuals interested in conducting this type of research for the first time should solicit the aid of those with experience in the field. This concluding section is organized as a checklist of major considerations for assessment of artificial habitats.

- The purpose or main goal for constructing the reef should be determined.
- A research or study plan should be formulated to determine if the reason for constructing a reef has been met. This is best done by posing a series of questions to be answered about the reef. The questions should be designed so that collected data could reject the hypotheses or statements about the reef or its circumstances.
- Studies comparing artificial reefs before and after deployment and comparing artificial habitats to natural habitats are important to evaluate the variables associated with a reef.
- The data gathered should be of the appropriate kind and degree of accuracy to answer the questions posed.
- A list of the abiotic and biotic variables likely to help answer the questions posed should be made.
- The variables examined should be clearly defined as being either dependent or independent.
- Measured variables should be described statistically. Attempts should be made to determine how the variation in the dependent variables are related to the variation in the independent variables.

- Assessment studies should consider general problems associated with sampling the aquatic environment.
- The number of samples should be large enough to attribute the sources of variation found among the variables.
- The frequency of sampling should be more frequent than the anticipated frequency of the change in the variables suspected of influencing the reef parameters.
- Much data can be gleaned from individual specimens, especially by monitoring the changes in their life history features relative to environmental changes.
- If the rationale for building a reef is to improve fishing then it is important to design a study that is able to detect appropriate changes.
- Visual assessment and monitoring methods can provide an effective way to evaluate an artificial reef, but researchers should be aware of their limitations. Water condition has an especially significant effect on the quality of data obtained.
- Observers using visual assessments require adequate training in diving and use of the specific technique.
- Voucher specimens should be obtained whenever species identification is questionable.
- Population enumeration can be facilitated by using a search image for species that are similar in some life history features.
- For larger populations, counting groups of individuals rather than single individuals can facilitate enumerations.
- Size of organisms should be estimated relative to some known size or distance.
- Transect methods for visual assessment should be of constant length, width, and height dimensions to permit reliable comparisons of samples within a study.
- In heterogeneous environments transects should be of a shorter length. Care should be taken to make sure one transect does not include more than one type of habitat.
- Point count surveys or total faunal counts are more appropriate to areas that are limited in size or those that are heterogeneous.
- Visual techniques are easily modified to the special circumstances of each situation. Care must be taken to assure that the sampling protocol is well-defined and closely followed.
- The amount of survey time and area are critical variables to visual survey methods.
- Removable sampling plates are effective for studying the sessile flora and fauna on artificial reefs.

- Some physical features of an artificial habitat are best measured prior to deployment.
- Sediment and substrate type should be monitored as they can indicate the impact that the reef structure has on local currents and water movement.
- Those unfamiliar with sampling design and protocol should consult with specialists in these areas before attempting to design an assessment or monitoring study of artificial habitats.

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