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THE ROLE OF ECOLOGY IN MARINE POLLUTION MONITORING

ECOLOGY PANEL REPORT

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INTRODUCTION

The report *On the feasibility of effects monitoring* (McIntyre et al., 1978) emphasized the difficulties in monitoring of population and community effects. It was argued that the response of populations and communities to pollutants is non-specific and that it is hard to distinguish pollution-induced from natural changes. However, with the possible exception of some biochemical techniques directed towards specific effects of toxicants, all of the biological monitoring techniques (physiological, behavioural, pathobiological, and genetic) are to varying degrees non-specific. Similarly, in all the above it is imperative to ascertain the natural variability in the response chosen before effects can be categorically regarded as due to pollution. It should be recognized that "from a strictly biological as well as a fisheries point of view it is the population and not the individual that is important and it is argued that unless an effect has consequences at the population level it is insignificant" (McIntyre et al., 1978). Whereas many biological monitoring techniques are capable of suggesting effects of pollutants on populations, ecological monitoring addresses the effects directly. Furthermore, many biological monitoring techniques are rather inflexible, being constrained to the near-shore and often intertidal areas, whilst ecological monitoring can cover all habitats from the intertidal zone to the deep sea. Ecological monitoring does, therefore, provide the only real test of effects on populations. However, early detection of changes on individuals, such as with genetic damage, is clearly preferable since effects on populations may not be shown for many generations.

OBJECTIVES OF BIOLOGICAL MONITORING

The effects of pollutants in populations and communities can be somewhat arbitrarily divided into acute and chronic effects. The detection of acute effects in response to specific pollutants, such as sewage effluent or paper pulp mill waste (Pearson and Rosenberg, 1978) or oil spills (Sanders, 1978) is relatively easy, since changes in populations and communities are usually dramatic. Such studies are usually done over small geographical areas and are relatively short term, seldom longer than 10 years duration. The other and more difficult goal of monitoring is the detection of chronic effects produced by small and often unknown sources over long time periods. Frequently such programmes last for many decades since they try to establish the periodicity of long-term natural cycles. The most comprehensive data on such changes in marine populations come from fisheries statistics and they have been extensively analysed (see Cushing, 1976; Hempel, 1978). Long-term records from the Continuous Plankton Recorder (C.P.R.), (Longhurst et al., 1972) have covered periods of up to 40 years and show that North Atlantic planktonic species have cycles of up to 20 years. In benthic communities both meiofaunal components (Heip, 1979) and *Pontoporeia affinis* in the Baltic Sea (Lassig and Lahdes, 1980) show cycles of at least up to 6—7 years. Indeed many monitoring programmes today are based on earlier projects not originally envisaged as monitoring programmes but subsequently continued because the data cover long time periods (e.g. C.P.R. programme). Ecological monitoring programmes aimed at assessing chronic effects of pollution must cover many decades.

The division of monitoring into two distinct categories is not clear cut and no single set of criteria can be established for ecological monitoring. Rather local aims and objectives must be clearly defined and the appropriate spatial and temporal scale selected.

The unravelling of effects of climatic changes on populations and communities can be greatly aided by broad-scale international monitoring programmes. Under COST project 47 of the European Economic Community four different communities, ***Balanus/Pa-***

tella on hard intertidal substrata, *Ciona* on hard subtidal substrata, and *Amphiura* and *Macoma/Abra* of subtidal sediments are being studied by scientists from ten countries from northern Norway to southern Portugal. By monitoring the communities over entire geographical ranges it is hoped to establish the influence of climatic factors on various population parameters.

BIOTA

One of the key questions in ecological monitoring is which organisms should be used? The following is a brief summary of the advantages and disadvantages of using the various biotic components.

MICRO-ORGANISMS

Microbes are characterized by rapid reproductive rates and, therefore, rapid response at both the population and community levels, but taxonomically they are extremely difficult, needing highly specialized techniques. Some groups may be valuable as markers or indicators for specific types of pollution (e.g., coliform bacteria as indicators of sewage pollution have been used for many years). The specificity of requirements of many species offers considerable potential for monitoring (e.g., the abundance and distribution of hydrocarbon-degrading bacteria might be used in oil-pollution studies). Knowledge of microbial ecology is, as yet, at an early stage of development and this considered together with the taxonomic difficulties, renders micro-organisms currently a limited value for ecological monitoring.

PLANKTON

Both the phyto- and zooplankton are for the most part well described for the North Atlantic and are highly diverse. The plankton is characterized by large amounts of spatial patchiness and seasonal variations in abundance in response to local variations in water masses. Replicability of quantitative plankton sampling is therefore, often poor. Local variability renders plankton, in general, less suitable than benthos for ecological monitoring. However, where large-scale surveys integrate local patchiness — such as C.P.R. tows by ship across the North Sea or North Atlantic, plankton have been effectively used in long-term monitoring (Longhurst et al., 1972). In fact there is no other ecological method which could be used, in practice, to monitor the open North Atlantic.

In certain coastal areas with naturally small numbers of species, as the Baltic Sea, plankton species can be monitored together with benthic species since diversity is so low.

NEKTON

Commercial fisheries statistics provide the most comprehensive long-term monitoring data available in the sea. However, despite the excellent data base no clear effects due to pollution have been established on fin fish except for a few coastal areas (Hempel, 1978). It is probable that commercial fishing has a bigger impact on fish stocks than pollution and, therefore, we have not considered fisheries data. The above cited conference was devoted to this topic and the expertise available was infinitely greater than ours in this field.

HYPERBENTHOS

(Organisms swimming just above the bottom).

This group of organisms (mainly malacostracan crustaceans) has been shown to be sensitive to pollution effects in the Oslo Fjord (Beyer, 1968). However, there is no reliable quantitative sampling gear generally available and knowledge of the fauna is rather poor. Thus, whilst it may be a promising biotic component to monitor, at the present state of knowledge we do not recommend monitoring the hyperbenthos.

BENTHOS

The benthic fauna is more or less sessile and must tolerate pollution or die. It can thus integrate effects of pollution over time and is probably the best all-round biotic component to monitor. In the North Atlantic the taxonomy of the coastal macrobenthos is relatively well known.

Phytobenthos

Macroalgae, both intertidal and shallow subtidal, offer useful ways of monitoring pollution effects. Remote sensing using infrared photography to survey rapidly large areas is a promising tool.

Meiobenthos

The meiobenthos is easy to sample but laborious to sort. Only hard-bodied taxa (especially harpacticoid copepods and nematodes) are recommended in a monitoring context since highly specialized techniques are needed to identify soft-bodied forms such as Turbellaria. Taxonomic problems are large but some species of nematodes are resistant to pollution and anaerobiosis and may be used to indicate polluted conditions. The variability of structural community parameters in space and time is fairly low (Heip, 1979).

Macrobenthos

Compared with the meiobenthos taxonomic problems are few and spatial variability is less. Two categories of fauna should be distinguished i.e., hard-bottom and soft-bottom fauna.

Hard-bottom fauna: Can be counted in a non-destructive manner frequently obtaining actual abundance data. Permanent marked sites can often be used, thus enabling many sites, especially in deep waters to be counted in a short space of time. Subtidally stereophotography can be used to study three-dimensional habitats to good effect (Lundälv, 1971). Fouling communities do, however, vary greatly over short time intervals (3 months in N Carolina, longer in Massachusetts) owing to random variations in larval settlement (Sutherland, 1978; Osman, 1977). If this is a general property, fouling communities may not be appropriate for long-term monitoring.

Soft-bottom fauna: Not restricted to depths permitting diving as are monitoring programmes for hard bottom. Statistical estimates of population abundances must be obtained since sampling is blind and destructive. High diversity of species from many trophic groups giving many possibilities in a comprehensive monitoring programme.

BIRDS

Seabirds are particularly susceptible to oil pollution and some populations have been severely reduced (GESAMP, 1978). Whilst we recognize the need for monitoring seabird populations none of the panel members had any experience in this field and we excluded birds from our considerations.

MARINE MAMMALS

With the elaborate international surveys already being done we did not feel that it was necessary to consider marine mammals.

SAMPLING DESIGN

GENERAL CONSIDERATIONS

The design of the sampling strategy is perhaps the most critical aspect of an ecological monitoring programme. Failure to design a sound strategy will result in inefficiency at best and inconclusive results at worst. The design must be based on careful consideration of the nature of biotic spatial and temporal patterns, the fate and effects of pollutants, and the implications of alterations of the biota. Although these criteria are almost never fully met with sound knowledge at the start of a monitoring programme, some level of understanding is usually attainable even in relatively unexplored seas. For example, bathymetric charts are generally available, often including a gross indication of sediment distribution. Preliminary design may also be based on inferences from general physical laws (e.g., sedimentation of fine grained material) or experience in other regions. If background knowledge is insufficient to allow the development of tentative plans, then initial reconnaissance studies may be required to develop a sound sampling design for monitoring. Monitoring design based on obtaining knowledge of the biotic patterns and fate of pollutants should be an iterative process. As information accrues, sampling design should be modified to accomodate the enhanced perception, whilst considering the need for continuity in approach.

Design considerations are considered in detail below. Most observations and recommendations are relevant to monitoring the majority of marine biotic components, although emphasis is placed on monitoring the benthos.

NATURE AND SCALE OF HABITATS

Sampling sites are often positioned in some regular or systematic fashion, such as transects or grids. On the other hand, the rationale for selecting irregular station locations is frequently not explained. Since most marine environments are to varying degrees heterogeneous and not regular, transects, grids and completely random sampling are generally inefficent and subject to inadequate sampling of limited, but important habitats. An example of this problem, illustrated in Figure 1, is the Marine Ecosystems Analysis programme benthic sampling grid in the New York Bight apex (Pearce et al., 1976; Freeland et al., 1976). When compared with the distribution patterns of sediment types it is apparent that the grid is not intense enough to permit detailed mapping, a common objective of grid sampling. Furthermore, sampling intensity in the muddy habitats at the head of the Hudson Shelf Valley in which sediment-contaminant concentrations are highest was sparse (Walker et al., 1977). The southern-most section of the valley, which may serve as a conduit for sediment-borne pollutants (Hatcher and Keister, 1976) was not directly sampled at all by this grid.

Based on existing knowledge of the nature and scale of habitats, a stratified sampling scheme should be devised. Although the purpose of stratification is to increase precision by reducing within-stratum variance of population or community parameters, the distribution of biotic parameters or even their relationship to environmental parameters is generally not known. Nonetheless, stratification based on known environmental characteristics can usually effect a significant reduction in variance of subsequently measured biotic parameters.

For benthos sampling, data on substrate distribution, depth, and tidal position in intertidal studies may be

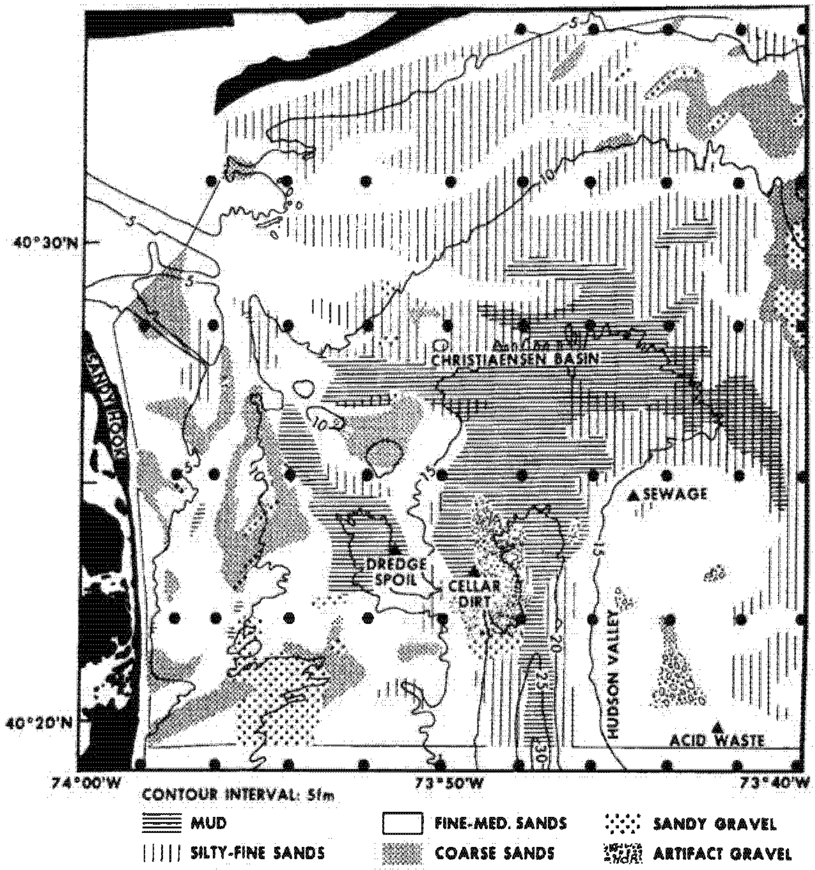

Figure 1. Grid of stations used in sampling benthos in the New York Bight project in reference to distribution of sediment type (Freeland et al., 1976; Pearce et al., 1976).

used to stratify the environment. For plankton sampling the water mass is the relevant habitat unit. Water masses shift with time thus a fixed station location may yield a false picture of temporal variability. Thus it may be reasonable to sample water properties to identify water masses before locating a site for sampling the plankton.

Adequately discerning the distribution of habitats is critical to the selection of control sites for comparative monitoring. Although exactly comparable control habitats never exist, reasonable comparable controls are essential in the separation of broad-scale variation due, for example, to climatic trends, from more localized impacts of pollutants. Occasionally selection of control sites is not feasible either because the potentially impacted area is unique (e.g., the New York Bight apex example cited above) or because the potential effects of pollution may be broadly pervasive. In such cases, down-gradient sampling may be appropriate and trend analyses (correlation and regression) are more appropriate than difference-testing statistical procedures. Saila et al. (in press) present a rational for logarithmically spaced sampling sites along gradients emanating from the source of pollution, based on pollutant concentration.

In a stratified sampling design, samples would ideally be collected randomly within strata. This allows valid extrapolation of the results to the entire habitat or stratum. However, in order to minimize the variance due to the interaction of spatial and temporal pattern as well as for reasons of practical economy, fixed stations may be selected for monitoring. With good relocation this would allow the greatest confidence that the observed effects over time are indeed due to temporal variation. Nonetheless it must be kept in mind that extrapolation of results to the entire habitat represented by the fixed station(s) is technically inappropriate. Obviously fixed station sampling is particularly desirable with those communities which can be sampled non-destructively such as with subtidal or intertidal rocky epibiota. Such communities may be regularly censused visually or photographically.

FATE OF POLLUTANTS

Knowledge of the environmental fate of pollutants should be employed in sampling design to assure that particularly susceptible habitats receive adequate attention. For water-borne and surface-borne pollutants this requires basic understanding of tidal, meteorological and geostrophic currents in the first case with the addition of wind patterns in the second case. Again, regular or systematic sampling would be efficient only if transport were strictly diffusive, when in fact advective transport usually dominates.

For particulate-borne pollutants transport may be more complicated. Whereas current dispersion models may be used to predict the transport of suspended particulates (Csanady, in press), models of the transport of sedimented particulates through resuspension are not generally available. Nonetheless certain inferences can be made concerning the location of sedimentary sites based on fine sediment distribution. An example of application of this rationale is the concept of accumulating and non-accumulating grounds used in British regulation and monitoring of ocean dumping (Standing Committee on the Disposal of Sewage Sludge, 1978). Sewage sludge dumped at the mouth of the Thames does not accumulate near the dumpsite because of swift tidal currents. Similarly, material dumped in Liverpool Bay does not accumulate because of seasonally erosive currents. On the other hand, considerable accumulation of organic material and other anthropogenic contaminants has been found in finer sediments of the Garroch Head dumping ground in the Firth of Clyde (McIntyre, 1977). Parallel situations exist in the Middle Atlantic Bight of eastern United States, where little accumulation of contaminants has been found except in nearby topographic depressions characterized by finer grained sediments (Hatcher and Keister, 1976; Harris, 1976; Lear et al., 1977). Off southern California large sewage outfalls, accumulation of oxygen demanding, and potentially toxic materials are similarly limited to deeper muddy bottoms (Smith and Greene, 1976).

In summary, the susceptibility of the communities, in terms of the degree to which they may be affected by pollutants, is a very important consideration in designing a sampling strategy.

SPATIAL VARIABILITY OF ENVIRONMENT AND BIOTA

Recognition of (the scale of) variation is important in assessing relationship of organisms to their environment and to other organisms. Furthermore, the degree and scale of environmental heterogeneity is a key consideration in determining appropriate sampling schemes and sample sizes (Elliot, 1971). If relatively homogeneous areas exist within an otherwise heterogeneous environment, sampling efficiency may be enhanced by focusing on the homogeneous sub-habitat. For example, Vanderhorst and Wilkinson (in press) were able to reduce variance estimates and thus manpower requirements for sampling the bivalve *Protothaca staminea* in intertidal plots by restricting sampling to portions of the intertidal seasonally covered by *Ulva* sp. and other green algae.

Except in the few rare cases for which direct counting of a total population of interest is feasible, the monitoring of marine ecological variables involves making sample estimates of the parameter(s) of interest. The approach adopted regarding selection of parameters for monitoring, and constraints on sampling in time and space, rests on the idea that one can assign a degree of sensitivity for detection of change in parameters based on the sample estimates with stated probability. The validity of taking this approach rests on how well the designer of monitoring programmes can estimate the true, "population", variance for the parameter of interest. Confidence in this estimate of true variance will depend on many factors, such as patchiness of spatial distribution, temporal variations, and life-history characteristics. Given estimates of the appropriate variances, two alternative methods are presented to answer to the following questions: 1) For a stated sensitivity, how many replicate samples should be taken? 2) For fixed resources (number of samples), what magnitude of change can be detected (i.e., what is the affordable sensitivity)?

Both of the methods presented rest on the commonly known but infrequently applied concept of *hypothesis testing* concerning the equality of two or more gaussian means in an analysis of variance (Snedecor and Cochran, 1967; Kastenbaum et al., 1970). It is assumed that the populations are normally distributed and have equal variance. Although these assumptions are seldom entirely correct for ecological data, analysis of variance is quite robust, and non-normality in primary data can sometimes be improved by logarithmic or other transformation.

There are two types of error in testing hypotheses. Type I error is the rejection of the null hypothesis

(H_0) when it is true and type II error the acceptance of H_0 when it is false. In environmental monitoring studies, the seriousness of committing a type II statistical error may be as important as the commonly reported probability of a type I error. The reason for this is that if one fails to find significant effects and does not compute sensitivity of the methods used, there exists no measure of the probability that real effects did not occur. Failure to detect significant effects may be solely due to sensitivity of method. In attempting to monitor ecological changes this situation should be avoided.

Computation of probability of committing a type II statistical error (B) requires specification of sample size in terms of numbers of replicate samples, specification of the desired significance level (usually=0·05), specification of the magnitude of differences one wishes to detect (sensitivity), and a rather precise estimate of the true variance. The relationship of these variables can be seen by examination of an example provided in Figure 2 (from Vanderhorst and Wilkinson, in press).

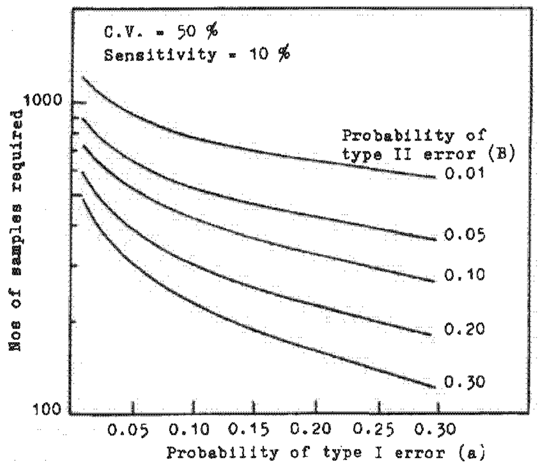

Figure 2. Comparison of type I and type II error probabilities, sample size for detection of a true 10% difference (from Vanderhorst and Wilkinson, in press).

Perhaps the simplest method for obtaining answers to the two questions involving sensitivity posed earlier is to use tabulated values of the maximum standardized range (t) provided by Kastenbaum et al. (1970). Sensitivity in this case can be thought of as the range between a real or expected group of means. For an example of ecological relevance take the density on fixed plots of the littleneck clam (*Protothaca staminea*). The range for two means is from 11 to 5·5 per 0·25 m², based on 36 replicate 0·25 m² quadrats on a fixed plot at a significance level of = 0·05 and with a 10 % risk accepted of committing a type II statistical error (B). The variance common to the means is 27·66 (s.d.=5·26). The maximum standardized range for these data is:

$$\bar{X}_{max} - \bar{X}_{min}/\text{s.d.} = (11 - 5{\cdot}5)/5{\cdot}26 = 1{\cdot}05$$

One can then read directly from a table provided by Kastenbaum et al. (1970) and discover that 20 replicate samples are required to detect a difference of this magnitude with the stated probability.

Using the same data, but constrained by resources to collecting 10 replicate quadrats per mean, one could read directly a value of the maximum standardized range and find it to be 1·706. When multiplied by the standard deviation this gives a sensitivity of about nine clams per 0·25 m² with the stated probability. A somewhat more detailed explanation for doing this as well as useful additional information on the subject of statistical variability in the quantitative sampling of animal populations can be found in Eberhardt (1978).

Ideally, replication will be based on requirements to detect a stated degree of change, for example a 50 % reduction in population density, with a stated level of confidence (see Saila et al., 1976). However, such ideal criteria may not be met because of practical considerations. In such cases, the investigator should attempt to determine which changes can be detected. If the level of ability to discern change is unacceptable, consideration should be given to 1) modifying the sampling design, or 2) choosing alternate parameters for monitoring; and if these would not improve conditions, abandoning the monitoring programme.

TEMPORAL VARIABILITY OF ENVIRONMENT AND BIOTA

The characteristic scale(s) of temporal variability should be estimated and, as data become available, re-evaluated in terms of sampling frequency. Where possible, biological variables which show temporal persistance and lack of short-term variability should be chosen (Heip, 1980). Although there is no universal prescription for sampling frequency in monitoring, as a general rule of thumb a frequency of at least twice that of the characteristic periodicity should be chosen.

For instance, planktonic animals with diurnal vertical migrations should be sampled at least twice a day; when this is not possible, as most often it is not, sampling should either cover the entire vertical dimension or always be performed at the same moment of the cycle. In this way aliasing will be avoided (the use of a sampling interval which is too short to resolve the shortest fluctuations present in the data produces the effect that high-frequency energy appears in the spectral estimates for some low frequencies: see Platt and Denman, 1975, for an ecological example). The same argument can be used for a community which exhibits a pronounced annual cycle. Such a community may be sampled twice a year or even only once a year, when its smallest characteristic period is one year, provided that sampling is always done at the same time of the cycle. An adequate depiction of long-term variation in the benthos of the German Bight has resulted from semi-annual sampling (Ziegelmeier, 1978).

For temporally dynamic communities, sampling should concentrate on periods when biotic flux is minimal. For example, sampling macrobenthos during periods of intense seasonal recruitment should be avoided. If this can be achieved, (annual) sampling during comparable seasonal periods will be sufficient for long-range monitoring.

COMMUNITY PARAMETERS

Communities are defined here as recognizable assemblages of populations which are structured through interactions between the populations themselves and with the physical environment. Although the organization of communities in ecosystems should be approached at that level, at the moment existing ecological theory is incapable of serving as a basis for making accurate predictions of the future behaviour of specific biotic components of the system. Physical theories (e.g. diffusion theory) may be applied to biological phenomena (e.g. patchiness) in certain cases, but we are nearly always ignorant of and therefore unable to take into account the overwhelmingly important biological interactions. Biochemical and physiological studies on single animals, groups or populations in the laboratory ignore these interactions as well and in most cases do not consider the antagonistic and synergistic effects of the many different agents which occur in natural situations.

Moreover, populations in particular environments have been interacting with each other and with the environment for long periods of time. The life cycles of these populations are a result of natural selection over evolutionary time and, within certain limits, are random products of the history of particular environments. In this sense it is impossible to find "normal" or "typical" environments or populations. It should be recognized that there is a lack of a general body of ecological theory allowing accurate predictions to be made about the future behaviour of particular populations and communities in ecosystems.

THE DESCRIPTION OF COMMUNITIES

As the sea is heterogeneous in space and time samples must be taken so that they cover all significant scales on which the variability characterizing the phenomenon exists. One has to identify the highest frequency of this variability (the number of cycles per unit of time). Using the Nyqvist criterion of time series analysis, the minimum number of samples must be twice the highest frequency of the variable under study (Kelley, 1976). It is clear that when a large proportion of the variance in a parameter is found in the high frequencies of the space or time domain, the parameter is, in general, unfit for monitoring purposes aimed at detecting phenomena on large spatial and temporal scales. This is one of the principal reasons why benthic systems are generally more suitable in monitoring programmes since they are largely two-dimensional and do not require the many additional samples needed to characterize the very heterogeneous depth dimension of the water column and, as has been mentioned, they are spatially and temporally less variable. The same criteria can be used to judge whether structural or functional attributes of communities should be measured. By structural attributes we mean the number of individuals and species and kinds of species in a community and their variation in space and time. Functional characters include the type and amount of energy flowing through the community. Structure and function cannot be separated clearly and many structural attributes grade into functional attributes. Structure and function are only aspects of the organization of the community and as such are more or less equivalent as sources of information. The combination of structural and functional measures such as in production/biomass (P/B) ratios offers methods for the comparison of communities which may well be useful in monitoring. All real thermodynamic machines (including ecosystems) capable of maintained behaviour exhibit the dynamic stability characterized by non-linear cyclical processes. Therefore, ecosystems exhibit periodic behaviour which must be identified and characterized. The cycle is a basic element of ecosystem organization. Moreover, since structure relates to organization in space and function to organization in time it will be profitable to add a dimension by looking at functional parameters in space or, with less effort, at structural parameters in time.

Functional parameters which may be used in monitoring programmes are community production and respiration. With micro-organisms measurements are possible by the incorporation of labelled substances, but as yet the methodology is still in the experimental stage and no definite techniques can be recommended for routine monitoring. Measurement of primary production using the ^{14}C method is now standardized but some dispute still exists as to what the results really mean (Sieburth et al., 1977). Respiration by the planktonic community is easy to measure, but depends heavily on species composition and environmental conditions and appears to be unsuitable for general ecological monitoring. Measurement of benthic community respiration is both impractical on a large scale and yields sometimes ambiguous results as anaerobic metabolism cannot be measured and may be important especially in polluted areas (Pamatmat, 1977; Thomas et al., 1976).

Structural parameters can be purely qualitative in that they consist of a list of species which is amenable to statistical analysis. Ideally, density and biomass of the population should be included as well. More sophisticated analysis methods such as trophic structure of the community are advantageous also. Remote sensing may become an important monitoring tool, for example for monitoring plankton biomass but as yet techniques are still in the experimental phase and cannot be recommended.

Biomass

Biomass is a relatively easy community property to measure. It may be quite stable in benthic populations (Warwick, in press; Heip, 1980), but shows considerable spacial variability which renders it less suitable in a monitoring context. Biomass increases under moderate organic enrichment (Pearson and Rosenberg, 1978) and large changes in mean biomass probably indicate stress conditions. Whilst production and respiration can often be inferred from biomass this is not always the case and no general relationships are known to exist.

Abundance

Abundance is a much less useful parameter unless it is restricted to a consideration of size classes, e.g., for zooplankton or macrobenthos. Abundance varies more than biomass, although when restricted to certain size classes overall abundance appears to be less variable than that of individual populations, at least as far as meiofauna is concerned.

Species richness

The number of species obtained in a given taxocene commonly is strongly dependent on the number and size of samples taken. Measurement of diversity is based on the total number of species and individuals and the relative abundance of individuals per species. The Shannon-Wiener index H' estimated from Brillouin's formula $H = 1/N \log N!/N_1!, N_2! \ldots . N_n!$ is less sensitive to rare species than many diversity indices and has been widely used in monitoring programmes (see Pielou, 1975, for a discussion of the Shannon-Wiener diversity index). Eveness indices, which measure the inverse of dominance, rely on the knowledge of the total number of species in the statistical population, which is rarely known. An eveness index based on the Shannon-Wiener index is H/H_{max} where $H_{max} = \log S$, the number of species. The Shannon-Wiener index has been used to indicate long-term changes in community structure (Heip, 1980) and generally has lower values in polluted situations (e.g., Pearson, 1975). Gray (1979) has shown, however, that statistically significant changes in the index are associated with only very gross changes in the community structure; therefore the value of using a diversity index in a monitoring context must be questioned (see, also ACMRR/IABO 1976, Working party on ecological indices of stress to fishery resources). Gray (1979; 1980) used another method of assessing community structure based on the log-normal distribution of individuals among species which he claims can indicate changes caused by organic enrichment over relatively short time intervals.

Number of higher taxa

A simple method of monitoring changes might be by recording the number and identity of taxa higher than the species, i.e., the genus, family or order. For instance, Van Damme and Heip (1977) found a gradual decrease of the number of meiofaunal taxonomic groups from the open sea towards the polluted near-coastal waters of the Southern Bight of the North Sea, where at some stations only one group (nematodes) or two (nematodes and harpacticoid copepods) occurred, contrasting with nearly ten groups in the unpolluted offshore stations. These meiofaunal groups are easily recognized by technicians and their number could serve as a useful parameter for monitoring changes. It is related to total diversity of the community.

Trophic structure

This is important as the trophic position of a species has important consequences in matters such as bioaccumulation and energy flow in general. The relative proportion of primary producers or predators may be related to successional events and to stability of the community; the relative proportion of different feeding types may indicate the predominant type of energy available to the community, etc.

COMPARISON OF COMMUNITIES

Methods to compare communities in space and time are essentially of a statistical nature. Many of the methods are now widely used and are available as standard computer programmes. However, a warning against the indiscriminate use of such analyses is necessary. The use of these sometimes highly sophisticated techniques requires an understanding of the basic assumptions and philosophy of the methods employed and a competent statistician should be consulted in order to interpret correctly the results of such analyses. The following constitutes only a brief overview of existing methods and the reader is referred to the relevant literature for further details.

Analysis of variance

This classical technique is based on linear and factorial models and remains a powerful tool in the analysis of data (see e.g., Sokal and Rohlf, 1969).

Ordination

Methods which fall under the general heading of ordination consist of fitting a set of points with given weights and distances into a subspace of reduced dimensions. Most commonly used are principal component analysis, principal co-ordinate analysis, and analysis of correspondences. These methods are descriptive whereas factor analysis, which is sometimes confused with ordination, is an explanatory model which might be more appropriate when the data include a large proportion of attributes which are only weakly intercorrelated (Clifford and Stephenson, 1975). However, a preliminary investigation of the correlational structure of the set of data by ordination is likely to provide a good overall picture, especially when the first few principal components account for a fairly high percentage of the total variance. It should be kept in mind that if the variances of the variates are greatly inflated by error the component weights may be spuriously large because the first component will have extracted a maximum of the total variance (true and error), the second a maximum of the remaining variance and so on. Error variance is eliminated only in a model which makes specific allowance for it, as is the case in factor analysis (Maxwell, 1977). The technique involves the extraction of eigenvalues and eigenvectors of a matrix of either correlation coefficients or covariance of the total variability described by the original variables. Such techniques are used on quantitative data. For examples of such methods in use see, Boesch (1973); Chardy et al. (1976); Field (1971); Hughes and Thomas (1971); Maxwell (1977); and Moore (1974).

Reciprocal averaging (Hill, 1973).

This technique is essentially a repeated crosscalibration procedure which derives unique ordination of the variates and the species. It is especially useful when some of the variables are qualitative. For examples of this technique with marine data see Warwick and Gage (1975); Fasham (1977).

Numerical classification

The data are grouped into clusters which are relatively independant of one another and which may be classified hierarchically. Can be used on qualitative or quantitative data (see Boesch, 1977; Clifford and Stephenson, 1975).

RATIONALIZATION OF MONITORING PROGRAMMES

In any given marine benthic community there are in general between 70 and 200 species of macrofauna. In devising a monitoring programme which is concerned with benthic community structure an important goal should be the efficient use of resources. The question then arises is it necessary to count and record all the species in a community? The selection of certain key species for monitoring may greatly enhance efficiency but this must be done with care. Criteria which are useful and ecologically important, include species which are 1) important in terms of, a) abundance or productivity, b) physical structure of the community, or c) regulation of community structure (keystone species); 2) susceptible to pollutant stress; and 3) for which there is existing knowledge of the biology of the species (see population studies section). The obvious advantages of selecting certain species are savings in time and expense, allowing more effort to be devoted to investigating the structure and dynamics of their populations. However, difficulties arise when previous knowledge does not allow sufficient confidence in selecting appropriate species. Also, for some communities, such as small macrobenthos and meiobenthos, relatively little time is saved in sampling only a few target species since the laborious sorting process must be performed in any case. Also, focusing only on certain species disallows use of community parameters which have demonstrated efficacy. Based on community parameters, Gray (1979; 1980) suggested an approach that allows the identification of a subset of species that are sensitive to slight organic pollution. Species groups called second order progressive species, have also been identified using subjective methods (Bagge, 1969). Such species are likely to be better indicators of slight and moderate pollution than the extreme opportunists (such as *Capitella capitata;* see Pearson and Rosenberg, 1978); the first order progressive species *sensu* (Bagge, 1969), which may be spontaneously abundant following natural disturbance (Eagle and Rees, 1973). The presence of large numbers of *Capitella* has been suggested by Reish (1960) as a universal indicator of organic pollution; however *Capitella* responds to many forms of disturbance (see Gray, 1979) and only when continuously present in large numbers can it be used to indicate organic pollution.

Use of so-called indicator species is not felt to be a reliable technique for ecological monitoring. Suggested indicators such as *Capitella* merely indicate end-points of pollution effects and it will certainly be more profitable to concentrate on groups of species which respond to slight pollution.

It will be most useful to have hypotheses about changes following pollution which can be tested in the field. Many of the so-called monitoring and baseline surveys have been merely lists of species which occur in the censused area. The biology related to changes in community structure, for example whether tolerance or life-history strategies are important (Gray, 1979) should be emphasized. Ideally, ecological monitoring should parallel studies on the effects of pollutants on populations and communities in order to enhance the predictive capabilities and ascertain the causes of observed changes.

POPULATION STUDIES

Population characteristics are useful for monitoring both lethal and sublethal effects of pollutants. The parameters considered below are useful monitoring tools but the suitability of a species or a parameter is dependent upon both the specific aim of the monitoring programme and the nature of the system being studied. The development of a programme incorporating the population approach is described by Jones (1980).

The populations chosen for study should be selected from those components of the system most at risk from any given pollutant or mixture of pollutants. Two important questions must be considered before a detailed sampling programme can be developed; the criteria influencing these decisions are discussed briefly below.

SELECTION OF SUITABLE SPECIES

Such selection will require a basic knowledge of the biotic assemblages within the study area and may include components from any group ranging from the microbial flora up to the megafauna, including seabirds.

Accessibility and abundance

These factors will exert a marked influence on the types of study which can be adopted, particularly when cost-effectiveness is a significant factor. The littoral zone is the most readily accessible marine habitat and its use is particularly appropriate when surface-borne pollutants such as oil are being studied. Increasing depth causes a general decrease in accessibility and a marked increase in sampling costs but, when these deeper habitats are most at risk, they must be included in the programme design. Limited accessibility may necessitate remote sampling procedures, for instance in deeper sea-bed areas, and this may impose additional statistical considerations, such as sample size and frequency, which may limit the suitability of a potential species for some types of study. Accessibility may also be influenced, even in the littoral zone, by behavioural patterns involving seasonal or tidal migrations or by age-dependent behavioural differences; for instance, young *Nucella lapillus* are found almost entirely within crevices or among algal holdfasts while adults are commonly found on open rock surfaces.

Within any habitat, those species present in low densities are less suitable for the study of characteristics which require destructive sampling techniques.

Distribution

Species which are widely distributed within the study area should be chosen wherever possible; this maximizes the number of potential study sites and populations available for investigation. Ideally, populations should be uniformly distributed at each potential site since patchy distribution will require more elaborate statistical treatment. The selection of relatively homogeneous sites is a vital prerequisite for the subsequent design of appropriate sampling procedures i.e., it is a form of stratification of sampling in which sites are selected for uniformity in order to minimize the natural variability which is always a problem in biological sampling.

Sensitivity

Wherever possible, species known to be sensitive to potential pollutants should be included even when they are considered to be of limited ecological or commercial importance. Such species can provide early indications of potential damage. The nature of the "sensitivity" may vary from an obvious direct influence on the individuals of a population to a more subtle increase in susceptibility to disease of various kinds (see pathobiology panel report).

Ecological role

Species should be chosen to represent a variety of trophic components within the system. Key species (Lewis, 1976) within a community should be included when such a role is known, but these species are often relatively tolerant and may not be the optimum choice if an early indication of change is desired. Species known to be resistant to the potential pollutants should be given a low priority; sensitive species should be chosen wherever possible. The ideal monitoring species is a sensitive key species, e.g. for oil pollution, the limpet, *Patella vulgata,* is an ideal choice.

Mobility

The mobility of the chosen species should be small so that immigration and emigration are not important considerations; the ideal is a sedentary organism. When species mobility is high, studies of population structure and growth will be of limited value since the behavioural response of the organism to the pollutant may result in avoidance reactions and undefined periods of exposure to the pollutant.

State of current knowledge

Wherever possible, species which have been well studied should be used since a basic appreciation of factors such as life cycles, food sources, and physiological tolerances will facilitate the design and interpretation of a monitoring programme. Where species which are relatively poorly known must be used, research into their physiology and ecology should be considered a fundamental part of the programme.

POPULATION PARAMETERS TO BE STUDIED

The applicability of several population parameters will vary both with the species chosen and with the objectives of the monitoring programme. Some of the more valuable approaches are considered below.

Distribution and abundance

Studies of distribution and abundance are particularly appropriate where pollutants are introduced at point sources so that biological effects may be looked for along a gradient from that point source. Sampling considerations are of fundamental importance to such studies and the use of fixed-site, non-destructive sampling techniques (Jones, 1980) may be particularly valuable when the nature of the substratum is suitable, e.g., fairly even bedrock surfaces. The use of artificial substrata for settling and recruitment studies may be appropriate for some organisms such as barnacles. The semi-quantitative approach using species abundance scales (Crapp, 1971) is not recommended for general use because of the lack of measure of variability and the low resolution inherent in this approach. The study of distribution and abundance will detect only lethal effects in sedentary forms while, in mobile forms, it may be difficult to distinguish between mortality and avoidance reactions.

Population structure

Studies of population structure require that year classes or generations be distinguishable and, wherever possible, quantifiable. Species which can be aged are particularly appropriate e.g., bivalves such as *Cerastoderma edule* (Jones, 1979) which carry annual growth-check marks. The combination of length-frequency studies with polymodal analysis (Cassie, 1954) may also be valuable but this is only appropriate when the year classes or generations do not overlap extensively; species with protracted spawning periods or very variable growth rates, fecundity, and larval or adult mortalities are inevitably difficult to study in isolation. Maximum sensitivity can only be achieved when this approach is used in conjunction with studies of recruitment, fecundity and mortality, i.e. population dynamics, and populations exhibiting sporadic recruitment, e.g. many bivalves may be of little value for this approach.

Growth

Growth studies may be particularly valuable as a measure of the sublethal effects of pollutants. There are several approaches to monitoring the growth of a population and the optimal programme should include more than one. The use of scope-for-growth measurements (Widdows, 1978) are considered elsewhere (Gilfillan, 1980) and will not be discussed here.

The allometry of various body characters may be studied (e.g. Jones et al., 1979) and this approach has the advantage of offering a well-established statistical approach. Relationships can frequently be described by linear regression techniques and both temporal and spatial comparisons may be made using covariance analysis. This technique is particularly valuable for detecting changes in the "condition of animals" but the presence of seasonal changes in allometric relationships (Jones et al., 1979) must be taken into account in the design of the programme. This approach requires destructive sampling and is most suitable for studies of vertebrates and invertebrates with hard exoskeletons, e.g. molluscs.

Studies of absolute and relative growth rates may be derived from time series studies of population structure or by the study of specimens of known age. Marking of many organisms is difficult and aging which is best carried out on species that exhibit annual growth-check marks has been established for a number of bivalve species such as *Cerastoderma edule, Pecten maximus* and *Mya arenaria* but such rings may be of dubious value in bivalves such as *Mytilus edulis* or *Protothaca staminea.* The use of rings for aging individuals, therefore, must be used with caution and the validity of such rings *must* be established before

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their use is adopted for any species. This technique offers a unique opportunity for retrospective monitoring also (Jones and Jones, in press), particularly valuable when the available baseline period is short.

The optimum methods for studying growth rates are species dependent but alternatives are reviewed for benthic organisms by Crisp (1971). Both the pattern of growth and the rate of growth are valuable parameters in such studies. These studies should use non-destructive sampling methods wherever possible.

Reproduction

Studies of fecundity may be useful since stress can interfere with reproductive processes at a number of levels. Such studies are practical, however, only for species which produce relatively small numbers of eggs. A more practical approach for many larger species is the use of body component indices as a measure of gonad production (Giese, 1967). For instance, studies of reproductive cycles are of value in investigations of thermal pollution where both fecundity and the timing of the cycle(s) may be modified. There are major problems of interpretation given the current level of understanding of the reproductive processes of most species and such an approach should not be considered in isolation.

Infection and disease

This topic is dealt with in the report of the pathobiology panel, page 135.

Body burdens of pollutants

This topic is considered in several other contributions to this volume.

SELECTION OF SITES

It is strongly urged that, wherever possible, combinations of parameters should be studied for any given species. It is also important that the interpretation of results be made on the basis of seeking persistent trends at groups of sites and *not* on the nature of changes at individual sites.

The selection of sites is an important consideration and should involve the following considerations after the broad sampling pattern has been established:

a) the size of the population at the potential site;
b) uniformity of distribution over that site;
c) the ease of carrying out accurate sampling;
d) the nature of the associated flora and fauna;
e) the stability of the population structure; and
f) the variability of the characters studied.

The aim should be to select suitable sites where variability is both measurable and minimal so that the resolution of the investigation is maximal. Factors a to d can be assessed prior to the operation of a baseline programme while factors e and f can only be evaluated after a suitable baseline period has elapsed. It is recommended that a large number of sites should be used for the development of the initial baselines so that the subsequent rejection of some sites because of excessive instability or variability does not invalidate the programme and its long-term goals.

THE ANALYSIS OF LONG TIME SERIES DATA

Once the monitoring programme has reached a routine phase the data base should extend over long time periods. In the introduction it was stated that plankton cycles can cover 20 years and benthic macro- and meiofauna may have at least 5- to 7-year cycles. Often authorities will try to reduce the number of samples taken on economic grounds without considering the ecological situation. For example, in the Baltic Sea a routine plankton monitoring survey has been reduced from 12 monthly samples per year to 4, on economic considerations. Wolff (1979) has calculated that with the restricted number of samples the changes that one will be able to detect are $\pm 70\%$ of the mean for primary production. Such "efficiency" may render the project worthless. Restriction of the number of samples to be taken must be based on sound statistical criteria for the communities and populations concerned. Often many years of data must be taken to have enough information on which valid reduction in the sampling effort may be based. Advanced forecasting techniques based on time series require up to 30 samples as a minimum for detecting long-term changes (see Chatfield, 1975).

If the objectives are, however, to study long-term year-to-year variations in a given population or community in which the cycles are known from previous studies, as in the case of many plankton and benthic communities in the North Atlantic, it may well be possible to restrict the sampling to one period of the year. Gray (1980) suggests that the variability caused by settlement of larvae in summer in north temperate latitudes is unimportant in a long-term monitoring programme of benthic macrofauna. For this reason sampling could be, reasonably, restricted to winter periods only. Such a programme is already in operation off the east coast of England (Buchanan et al., 1974; 1978). When routine data are available over many years one can begin to analyse long-term trends. An outline of appropriate methods that can be used is given below.

SPECTRAL ANALYSIS

In this technique the variance of a time series is decomposed according to frequency so that the spectral density function or spectrum of the series is obtained. The spectrum is the equivalent in the frequency domain of the autocovariance function in the time domain and they are equivalent ways to describe stochastic processes under certain conditions. Estimation of the spectrum of a time series provides an excellent tool to judge temporal variability in that series at a range of frequencies (see e.g., Platt and Denman, 1975). The technique can be expanded so as to cover two series which are either on an equal basis (equivalent to correlation) or casualty related in that one series is regarded as the input to a linear system while the other is regarded as the output (equivalent to regression) (see, Stephenson, 1978, for an application to benthic data).

PREDICTION

In predictive models the value of one variate is predicted from those of the other variates.

Simple regression analysis

This cannot be recommended as a predictive tool in ecology.

Multiple regression

Multiple regression models sometimes work well but a good fit may be spurious and does not necessarily mean that the model will give good forecasts. The number of explanatory variables should not be too large and it is advisable to fit the model to part of the available data and check it by using the remainder. The use of multiple regression models is in general, not recommended except in those special cases where there are definite reasons why one series of variables should be related to another.

Discriminant analysis

This deals with the problem of discrimination between *a priori* groups perceived to be different, e.g., abundance of several species at two stations.

Canonical variates

These extend the previous analysis to the discrimination between more than two groups (e.g., abundances of several species in several stations).

Canonical correlations

These define the relationships between two or more sets of variables (e.g., species abundances, and physicochemical environmental data).

FORECASTING

In general it can be said that a model to forecast values must contain parameters which can be measured accurately, which is a prerequisite not needed in simulation models. Forecasting in management may be used as a yardstick against which changes can be judged. If forecasts are required for planning or decision-making ideally one should set up a multivariate procedure, like that of Box-Jenkins.

Extrapolation of trend curves

For long-term forecasting it is often useful to fit a trend curve to successive yearly totals. At least seven to ten years of historical data are required and one should not make forecasts for a longer period ahead than about half the number of years for which data are available (Harrison and Pearce, 1972).

Exponential smoothing

An estimate of the future values is obtained as a weighted sum of the past observations in which more weight is given to recent observations and less to observations further in the past (Box and Jenkins, 1970). This procedure can only be used for non-seasonal series showing no trend, but as effects of trend and seasonality are easily removed this poses no real problems.

Holt-Winters forecasting procedure

This procedure is generalization of exponential smoothing in which one can deal with time series containing trend and seasonality. Apart from the fact that seasonal effects can be either multiplicative or additive, which has to be judged from the data, this procedure can be made fully automatic and it is widely used in industry (Coutie et al., 1964).

Box-Jenkins forecasting procedure

This procedure consists basically of fitting a mixed autoregressive integrated moving average model to a given set of data. It is a fairly complex procedure which requires considerable skill on the part of the statistician and at least 50 observations are needed to have success (Box and Jenkins, 1970).

Stepwise autocorrelation

This is also a fully automatic procedure which uses standard multiple regression computer programmes. (Stephenson (1978) has used this on benthic data).

CONCLUSIONS

1) The great advantage of ecological over other monitoring techniques is that it is possible to measure directly population and community changes, over long time periods. Ecological monitoring programmes are flexible and can cover all habitats from the intertidal zone to the deep sea.

2) Ecological monitoring can be divided loosely into two approaches both of which are useful in effects monitoring programmes: (a) Monitoring of acute, local effects which are short term (less than 10 years usually); and (b) monitoring of chronic, broad-scale effects over many decades.

3) The aim of monitoring programmes must be clearly defined since different requirements necessitate different sample designs.

4) Benthic organisms offer advantages in monitoring programmes since they remain *in situ* and usually are comprised of diverse communities. However, in the open North Atlantic, broad-scale monitoring of phyto- and zooplankton (e.g., C.P.R. programme) is clearly preferable. Similarly, in the Baltic Sea, with low benthic diversity, monitoring programmes also include phyto- and zooplankton The biotic component selected for study should depend on relative susceptibility to anticipated impacts, ecological importance, and practical considerations.

5) The design of detailed sampling programmes must be based on preliminary sampling aimed at assessing habitat and community variation in space and time.

6) In general, structural parameters of communities and populations were preferred to functional variables. Growth rate comparisons are, however, highly effective as population parameters of use in detecting environmentally induced change.

7) Whilst initial monitoring programmes cover all species in a given community it is frequently possible to develop a subset of key or indicator species, thus increasing the cost effectiveness of the programme.

8) Experimental approaches to ecological problems should be encouraged since they will help to increase the predictive power of trend analysis.

9) In analysing data, statistical trends should be related to the biology of the organisms concerned considering for example life-history strategies.

10) Time series analysis methods can be used to predict natural fluctuations in long-term data against which the effects of pollution can be assessed.

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