United States
Environmental Protection
Agency

Environmental Research
Laboratory
Corvallis OR 97330

EPA-600/3-80-032
February 1980

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Characteristics of Nonpoint Source Urban Runoff and Its Effects on Stream Ecosystems

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CHARACTERISTICS OF NONPOINT SOURCE URBAN RUNOFF AND ITS EFFECTS ON STREAM ECOSYSTEMS

by

Donald B. Porcella
Darwin L. Sorensen

Utah Water Research Laboratory
Utah State University
Logan, Utah 84322

USEPA Order No. CC81224-J

Project Officer

Kenneth W. Malueg
Freshwater Division
Corvallis Environmental Research Laboratory
Corvallis, Oregon 97330

CORVALLIS ENVIRONMENTAL RESEARCH LABORATORY
OFFICE OF RESEARCH AND DEVELOPMENT
U.S. ENVIRONMENTAL PROTECTION AGENCY
CORVALLIS, OREGON 97330

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FOREWORD

Effective regulatory and enforcement actions by the Environmental Protection Agency would be virtually impossible without sound scientific data on pollutants and their impact on environmental stability and human health. Responsibility for building this data base has been assigned to EPA's Office of Research and Development and its 15 major field installations, one of which is the Corvallis Environmental Research Laboratory.

The primary mission of the Corvallis Laboratory is research on the effects of environmental pollutants on terrestrial, freshwater and marine ecosystems; the behavior, effects and control of pollutants in lakes and streams and the development of predictive models on the movement of pollutants in the biosphere.

This report surveys the literature on urban nonpoint source runoff in order to formulate a basis for evaluating the impacts on benthic stream communities.

Thomas A. Murphy, Director
Corvallis Environmental Research
Laboratory

ABSTRACT

Literature on urban nonpoint source runoff was surveyed to determine the magnitude of the effects of that source of contaminants to stream ecosystems. Very little information was available on ecosystem effects although extensive literature on all aspects associated with contaminant loading was available. However, urban NPS runoff probably exerts unique effects on stream communities because of its random magnitude/impulse loading of contaminants. Because control of NPS runoff is expensive, it is important to determine its actual impacts on stream communities. Ecological literature provided a basis for evaluating such impacts based on benthic invertebrate biomass and diversity, measurement of community primary production and respiration, carbon cycling, and variables related to the contaminant concentrations in the stream. We concluded that a stochastic approach for assessing the impacts would be most feasible for evaluating impacts of urban NPS runoff.

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ACKNOWLEDGMENTS

The following persons contributed their knowledge and skills to this report: Nancy W. Winters (ecologist), Edward F. Cheslak (ecologist), Scott Reger (ecologist), Garry Laughlin (biologist), David S. Bowles (hydrologist), and L. Douglas James (hydrologist). Garry Laughlin helped extensively in preparation of the literature review. Edward Cheslak provided excellent review of the ecological chapters and David Bowles extensively revised and refined our thinking in Section VII. The rough draft typing was done by Sherri Barber, Norma Schiffman, and Mardyne Matthews. The final copy was prepared by Kathleen E. Bayn. Our special thanks to these people and to Kenneth W. Malueg, project officer, whose initiative, sympathy, and patience helped us put this report together.

SECTION I

CONCLUSIONS

Although the literature contains extensive evidence of widespread urban runoff problems in the United States, there are few reports that describe the effects of urban runoff on variables related to stream biota and their interrelationships. Consequently, it is difficult to assess the actual impacts of urban runoff on stream ecosystems. Also, the literature is inadequate to determine the magnitude of the urban nonpoint source (NPS) runoff problem. Such an inventory would require an assessment of river basins in the U. S. and their associated pollutant sources. However, using the number of annual literature citations and specific studies reporting the relative magnitude of pollution sources an an indicator, it is apparent that urban NPS runoff is one of the most serious problems affecting water quality.

Because of the unique characteristic of contaminant input to streams from urban runoff, the effects on stream biota could be extensive. Contaminant inputs are severe and have significant impacts on water quality variables. However, the temporal and spatial patterns of contaminant concentration in the stream could result in differences in community response when compared to steady inputs such as occur with point source waste discharges.

It is necessary to determine the impacts of urban NPS runoff on streams in order to set priorities to ameliorate the problem. The widespread occurrence of water quality degradation by urban runoff, the high cost of treating or instituting best management practices (BMP) and the unclear relationship between water quality variables and community variables are factors which must be considered. Control of urban NPS runoff is directed at the input phases of the hierarchical urban runoff system: watershed, hydrology, and transport. Treatment and BMP would affect the input of contaminants that controls water quality. Ecological variables are affected by water quality. Therefore, the ecological effect of contaminants derived from urban NPS runoff must be determined before deciding on the proper priorities and methods for institution of expensive NPS controls.

The following specific conclusions that relate to these statements were derived from the literature:

1. Stream water quality can vary markedly both temporally and spatially as a function of stream flow and quality and watershed-hydrologic-transport characteristics:

    a. Watershed factors include percent impervious area, traffic and residential densities, industrial activity, street sweeping activity.

b. Hydrologic variables include seasonal patterns of storm intensity and duration, annual precipitation, interval between storms. Also, the receiving stream depends on the same hydrologic variables.

c. Transport factors include the collection and discharge system, presence of detention basins, separate or combined sewers. Transport processes and the hydrologic regime of the stream combine to determine the stream concentration for a given load of a contaminant.

2. Several stream water quality variables are affected significantly by urban runoff: BOD_5, refractory organics, suspended sediments, salinity, Pb, Zn. Although BOD_5 is a commonly measured quality variable, it is not very representative of the total organics load (COD:BOD ratios can approach 10:1) produced by urban runoff. Other water quality variables (nutrients, bacteria, Cu, Cd, ...) are significant in specific cases.

3. The relationship of stream biota response to the highly variant stream quality is unknown. However, effects on biota have been observed in comparative studies. Biomass, diversity, and P/R ratios are affected. Generally benthic invertebrate biomass and diversity decreased. In some cases P/R ratios approached 1.0, indicating greater autotrophic productivity; in other cases where organic loads were decreased, P/R was less than control streams.

4. Response of biota is hypothesized to be a function of stream concentrations of contaminants. Based on an assessment of the literature and evaluation of selected case studies, a set of ecological variables was defined that would permit comparative evaluation of the ecological effects of urban NPS runoff. Comparison with an absolute standard is not feasible at this time and evaluation of effects must be based on upstream/downstream, before/after, natural/affected stream comparisons.

The set of ecological variables would include, but not be restricted to, the following: biomass and diversity of invertebrates, gross primary production (autochthonous and allochthonous) and system respiration, carbon dynamics, and dissolved oxygen dynamics. Habitat description and selected water quality variables would also be determined. Hydrologic variables must be determined. Methods suggested by *Whipple et al.* [1978] should be applied. Stochastic models such as *Padgett et al.* [1977] and *Slatkin* [1978] should be used to provide sampling times, locations, and intervals.

SECTION II

RECOMMENDATIONS

Experimental studies to elucidate the effects of urban NPS runoff impacts on stream communities should include two steps:

1. A synoptic survey of streams throughout several regions of the U.S. The streams should be regionally significant, have comparable reaches or tributaries that are relatively natural and impacted by urban NPS runoff, and adequate historical stream data and hydrologic data should be available or obtained. Analyses of comparable stations during critical (defined by dynamics of natural community) seasons of the year should include benthic invertebrate biomass and diversity, productivity and respiration measurements, carbon turnover times, and characterization of water quality variables. Experimental error of measurements should be determined.

2. An assessment of community response based on the stochastic formulation described in Section VII should be performed and compared to effects of similar contaminant loads produced by steady (point sources) rather than impulse loadings. Some related hypotheses that should be tested include the following:

    a. Impact on communities is a function of water concentrations of contaminants. Such an hypothesis assumes body burdens are a function of water concentration which may not be true in a highly variant system.

    b. Carbon turnover time can be assessed simply and results related to energy flow as well as material cycle.

These hypotheses are the overall assessment of community response and stochastic systems should be evaluated in controlled or laboratory streams to minimize the error and confounding associated with natural stream systems.

SECTION III

INTRODUCTION

Stream ecosystem impacts of materials in urban runoff relate to several specific and unique features that distinguish those impacts from other linked systems. The purpose of this study is to determine if these unique features result in ecological differences that can be distinguished by appropriate ecosystem variables.

THE URBAN NPS/STREAM SYSTEM

By definition, materials in urban runoff originating from nonpoint sources (NPS) and combined sewers are excluded from direct consideration. These NPS materials are transported by water generated as a stochastic (random, but characterized by time of the year) hydrologic event, are affected significantly by temporal and spatial factors that have preceded the hydrologic event, and impact aquatic ecosystems as a discrete, largely receiving-system-independent input. Urban runoff impacts have similarities with other aquatic ecosystem impacts including the relationships between activities and materials produced, the materials themselves, and interactions within the aquatic ecosystem that are not a function of the input. The system watershed can be visualized as consisting of separate but somewhat interrelated phases or processes: (1) the production of waterborne materials in the watershed which can enter the aquatic ecosystem, (2) the hydrologic event, (3) the transport system, and (4) the aqueous environment and its community (Figure 1).

The watershed phase of the system is considered to be the natural system plus a set of activities that produce varying quantities and types of materials, depending on the types of activities. Typical concentrations and mass loadings of materials indicate that there are regional differences in urban runoff. The differences can be ascribed to geography and varying activities taking place in the watershed. These activities include industry type (air pollutants), street sweeping policy, urban density, and traffic.

The hydrologic phase consists of the water-producing storm or precipitation event. The factors that affect runoff quantity are the extent to which runoff from paved areas is collected by a storm drainage network, and antecedent moisture conditions as determined by recent storm activity and specific variations in area and activities. These factors cause variations in flow and pollutant concentration. Concentrations of pollutants vary with type and intensity of the different activities that can and do occur in the watershed. However, the hydrologic event and runoff intensity and duration control the mass of pollutants picked up in the runoff, their concentration, and

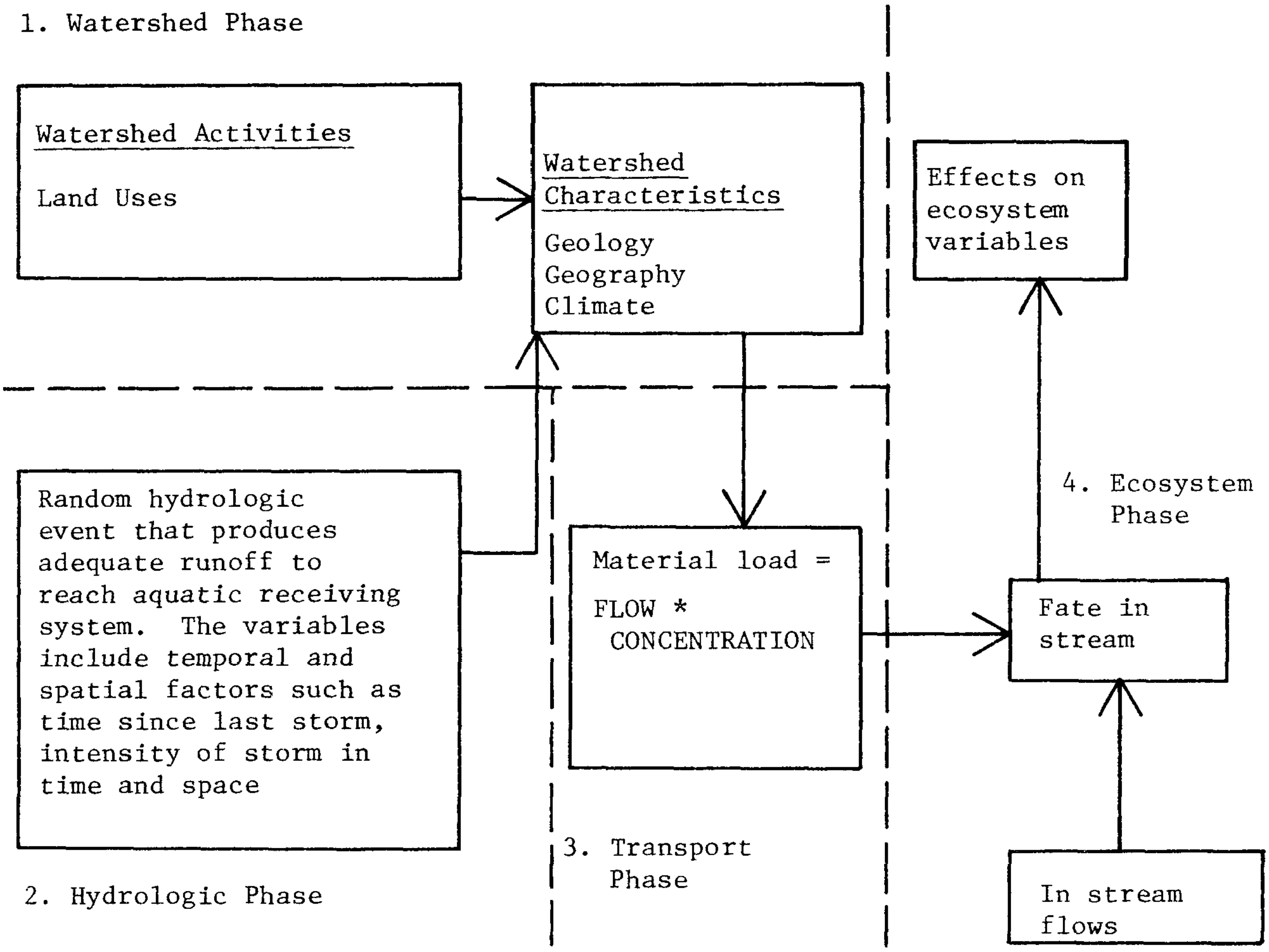

Figure 1. The hierarchical stream/watershed system that produces material flow to aquatic ecosystems.

ultimately the mass flow into the stream. To some extent, depending on relative runoff and stream flows, dilution of materials in the receiving stream depends on the magnitude of the hydrologic event.

The transport system is that phase of the hydrologic event wherein the runoff is transferred to the stream. It is important to distinguish between typical situations such as pipe inflows to streams, retention basins, and recharge basins, because aquatic community response varies considerably with the type of spatial and temporal mechanisms in which materials enter streams.

The stream ecosystem presents many challenges for analysis: stream morphometry, geological differences that result in variations in sedimentation and water quality, flow (hydrological) variations, other cultural impacts (waste discharges and variable land uses and their associated pollutants), and stream community variables. The community variables change with environmental factors such as flow, water chemistry, stream morphometry, and fish and wildlife management. These phases of the urban NPS material input to streams can be visualized as a matrix of specific variables that can be associated with stream community response and pollutant inputs.

URBAN NPS/TIME AND SPACE FACTORS

As a generalization, pollutants from urban NPS are qualitatively the same as pollutants from any other source and impact aquatic ecosystems as the specific contaminant independent of source. However, in a quantitative sense, the intensity and frequency of input varies considerably and randomly on a seasonal basis. Such inputs are described as stochastic and their impact on stream communities is hypothesized to result in a unique community which has the stability, resistance, and resilience to survive such impacts. It is important to define variables that reflect those impacts, the range of intensity of those impacts, and frequency of occurrence.

OBJECTIVES

The purpose of the project was to perform a literature review to determine the state-of-the-art of assessing the ecological impacts of urban nonpoint source runoff on streams. The review would result in a report on a national assessment of urban NPS runoff impacts on biota and ecological system variables. Although there are extensive results on water quality impacts, there is little information that relates to impacts on biota and stream ecosystem variables. Consequently, the literature search was expanded to permit the development of concepts relating to impact assessment that could be evaluated by specific research projects.

The following objectives were established:

1. Provide an overview of current literature on watershed variables, and hydrologic and transport processes of urban NPS runoff.

2. Review the recent literature on effects of urban NPS runoff on water quality in streams.

3. Discuss concepts of ecosystems function and how contaminants affect variables related to ecosystem function.

4. Discuss actual case histories of urban NPS impacts on stream ecosystems.

APPROACH FOR LITERATURE SURVEY

The intent of the literature review was to obtain information on biological/ecological indices of freshwater stream ecosystem impacts of urban runoff. Consequently, several fields of study were identified as relevant: stream ecology, biological/ecological indices, urban runoff and its quality and quantity, and spatial and temporal characteristics. Because of the breadth of the review, some reduction in citations was necessary. For example, a series of annual reviews from 1973 to 1978 of studies of urban runoff and combined sewers contained a total of 907 citations and most of these describe the chemical and hydrological properties of transported contaminants and their sources [*Field and a series of co-authors* 1973, 1974, 1975, 1976, 1977, 1978]. Other studies and reviews indicated a similar situation. Therefore, we concluded that an exhaustive review of the entire literature would be inappropriate and repetitive. The references cited herein were selected to illustrate example processes and properties of urban runoff and their effects on stream ecosystems. Similarly, ecological literature covers a broad array of subjects, not all of which are specifically of value, and specific references were selected to illustrate concepts. We are indebted particularly to an unpublished review by *Hendrix* [1979] and textbooks by *Wanielista* [1978] and by *Linsley and Franzini* [1964] for background information.

The survey of literature was initiated using computerized bibliographic data bases which are commercially accessible (Lockheed Information Services or Systems Development Corporation). In addition, a search on the appropriate subject areas (urban runoff, aquatic ecosystem response to pollutional stress) was performed on the Water Resources Scientific Information Center (WRISC) bibliographic data base. Commercially available data bases searched were COMPENDEX (The Engineering Index - monthly/annual, 1970-current), NTIS (National Technical Information Service, 1964-current), BIOSIS PREVIEWS (*Biological Abstracts, Bioresearch Index*, 1972-current). SCISEARCH was used exclusively to search for papers citing previously published papers with major emphasis on the topics of interest.

In addition to computerized systems, the contents of journals published since 1969 which were throught most likely to publish pertinent papers were searched manually. These included Water Research, Journal of Water Pollution Control Federation, ASCE Index 1970-1974, 1975; Proceedings of the Environmental Engineering Division ASCE; Proceedings of the Hydraulics Division, ASCE; and Proceedings of the Irrigation and Drainage Division, ASCE. The contents of the last three proceedings were searched from 1976 to current since previous publications would have been included in the ASCE Index. Ecological journals searched manually included Limnology and Oceanography, Ecology, Ecological Monographs, Arkiv für Hydrobiology, and Journal Fish. Res. Bd. Canada.

We used the following key words with appropriate key word strings for the computer searches: ecosystem, community, freshwater, reservoir, stream, river, estuary, lake, response, recovery, stress, impact, bioindicators of pollution, land use and urbanization, water quality modeling, urban runoff.

The computer searches provided few useful references. They included 979 citations of which 133 (14 percent) were applicable to the urban runoff system. Of these, only 2 dealt with ecological impacts while 54 were reviews or dealt with models and hydrology/hydraulics. Of the remaining 77 articles, 46 were oriented to urban runoff and 31 dealt with standard water quality analyses. This pattern was similar for reviews of urban runoff. Therefore, we concluded from the results of the computer searches, literature searches, and discussions with workers in the field, that essentially all urban runoff research is oriented toward assessment of watershed, hydrologic, hydraulic and water quality variables, and functional relationships. Except for the few reports discussed in the section on case histories, essentially no research on ecosystem impacts has been conducted. This is despite the unique mode of impact that urban NPS runoff has on streams.

However, there is significant interest in urban runoff subjects. We were able to identify 119 reports that relate to urban runoff, water quality or ecological impacts of urban runoff. Although most U. S. studies were concentrated in the Great Lakes and Middle Atlantic States, reports from all regions of the continental U. S. plus Canada, European and other countries were reviewed (Figure 2). This broad interest indicates the widespread need for assessing ecological impacts of urban runoff. Perhaps ecological impacts were neglected because a testable hypothesis for analyzing urban runoff stream impacts had not been formulated. This report represents a contribution toward the development of such a hypothesis.

THE DEFINITION AND NEED FOR STUDY OF THE EFFECTS OF URBAN NONPOINT SOURCE DISCHARGES TO STREAMS

Urban environments are not typical of the historical geoenvironmental events that occur during the evolution of natural communities that reside in specific ecosystems. Processes occur in urban systems that permit only the most tolerant and resilient species to grow and reproduce. Streams receiving inputs from precipitation-generated runoff that transport a wide variety of materials that have either no effect or are toxic or stimulatory, will develop communities that are unique to the system. Within the context of

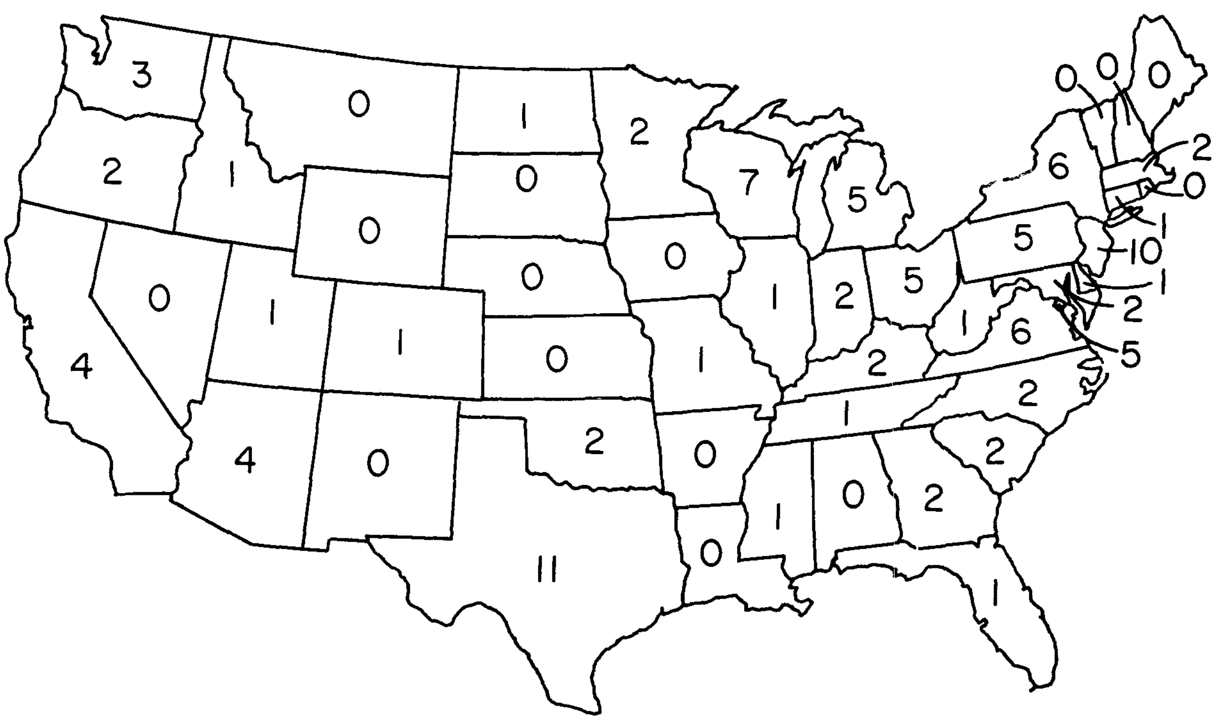

Figure 2. In the United States, 101 references on urban runoff impacts on water quality were identified.

describing this problem, we define the following limits to the urban/stream hydrologic system: (1) Urban environments are systems that are covered by large amounts of impervious surfaces of roads and buildings, and are incorporated and have a population greater than 2500 [*U.S. Department of Commerce* 1975]. Thus, small, as well as large, population centers are included, *i.e.*, cities and towns. (2) Streams are limited to freshwater rivers, creeks, etc. and are small and/or large, varying from first order to the higher stream orders (6-12). (3) The aqueous transport of materials deals with all non-water matter as contaminants, but not necessarily pollutants. A pollutant is any factor that, in the sense of *Mackenthun* [1975], affects the "integrity" of the aquatic environment. Urban contaminants are usually the same as those from wastes and natural sources, but there are specific differences in their impact resulting from the temporal and spatial mode of input.

Urban nonpoint sources are many and varied and their inputs to aquatic systems are characterized by the stochastic nature (random but characterized according to time of the year) of hydrologic events, the carryover effects of previous events, and specific activities in the urban watershed. Consequently, receiving water communities are impacted by a wide variation in concentration (input load and dilution capacity vary) of a broad spectrum of physical factors, chemical toxicants, and biostimulatory compounds. Control of such impacts includes the three major approaches of applying best management practices (BMP), requiring nonsteady state criteria for materials, and biological or ecological criteria for ecosystem response to urban inputs. The last approach deals with the integrity of the aquatic ecosystem; however, the aquatic ecosystem response is least understood and requires considerable research; it has a great effect on the development of BMP and criteria.

Individual organisms and the community of a specific ecosystem will likely respond to urban runoff as the concentration of material (temperature, suspended solids, toxicants, nutrients) rises from a relatively steady level to a peak value and falls off, roughly analogous to a flood peak hydrograph. The occurrence of these hydrologic events is random. The peak value depends on total loading, timing and duration of the loading input and the dilution capacity of the receiving water.

Urban systems have significantly less lag time for hydrologic events than do natural or rural systems because of the relatively large amounts of impervious surfaces. Also, peak flows and concentrations could be greater and the rising and falling rates faster than in natural streams. The response of biota will depend greatly on uptake rates, growth rates, motility (relates to recolonization, attraction and avoidance), and rates of repair. Interpreting the ecological effects of urban nonpoint sources will depend to a large extent on biotic measures which integrate the long term impacts of these stochastic events. A conceptual approach to visualizing the probable effects of such events shows that the wide variety of probable outcomes can be narrowed if better data and understanding of the system processes and components are obtained (Figure 3). Models exist for evaluation of hydrologic inputs in this conceptualized system [for example, *Bovee and Milhous* 1978].

In 1974 the International Hydrological Decade published a report of a study by its subgroup on the effects of urbanization on the hydrological

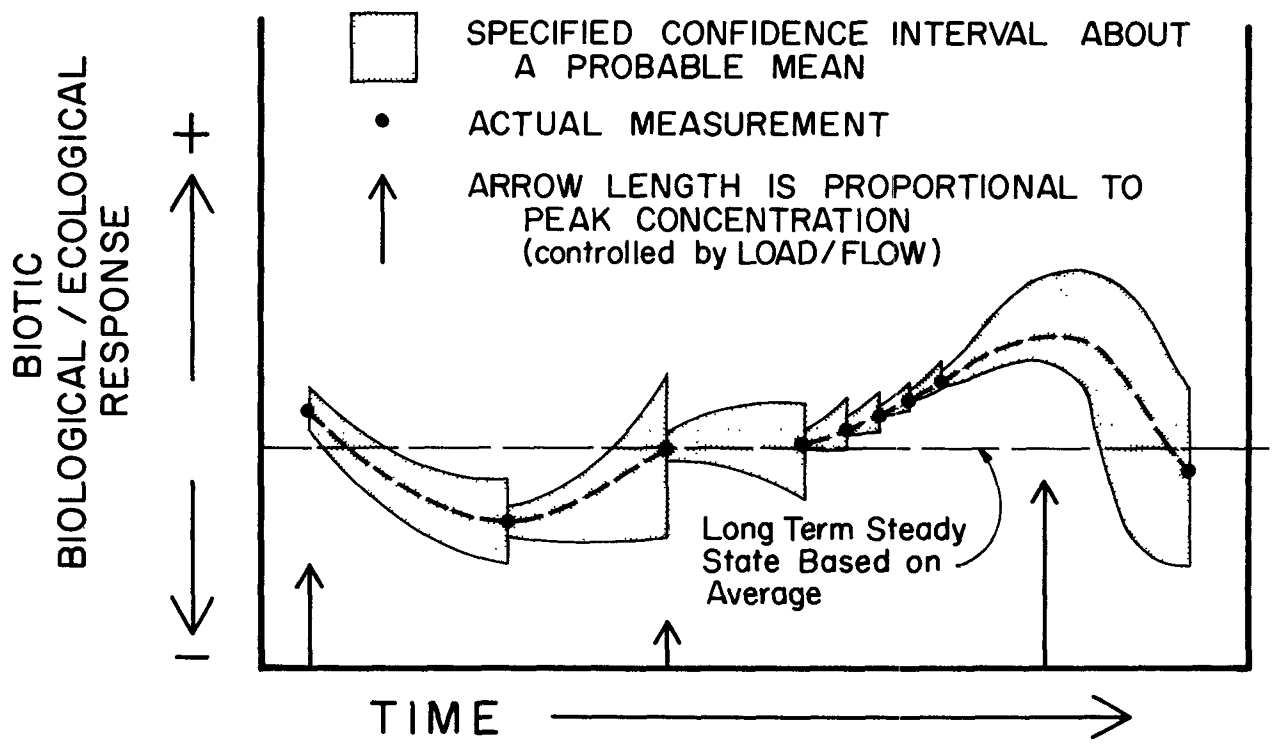

Figure 3. Hypothetical response of stream community to random inputs of toxic/stimulatory materials as exemplified by urban NPS.

environment [*International Hydrological Decade* 1974]. Case studies from the Federal Republic of Germany, the Netherlands, Sweden, USA, and the USSR were presented along with special topic studies of illustrative importance. A summary of findings indicated that the self purification capacity of the lower Rhine River had been reduced by 30 percent due to toxicants. Studies in Sweden showed that, compared to treated wastewater, storm water runoff had more SS, less BOD, and less phosphorus and nitrogen. In addition, flows from separate sewers had as great a pollution potential as combined sewer overflows. It was also concluded that, due to the sudden and brief nature of urban runoff and the large number of outfalls, the reduction in pollution from urban runoff will be very costly, possibly more costly than wastewater treatment.

A questionnaire to all national committees for the IHD identified six priority topics for research on the effects of urbanization on the hydrologic cycle. They were:

1. Changes in surface runoff caused by urbanization, amount and quality (including sediment), for storm water sewerage and urban streams;

2. Research on quantity and quality of runoff in terms of precipitation occurrences in experimental catchment areas;

3. Soil moisture and groundwater, quantity, and quality relationships and effects;

4. Water demand forecasting, including technical, economical, social, and political aspects, and quality of source considerations;

5. Water quality effects related to groundwater arising from direct disposal of wastes and to indirect pollution from disposal of solid waste materials;

6. Effects of wastewater and sludge on the natural purification capacity of receiving waters and their aquatic life, taking into account the processes involved. [*International Hydrological Decade* 1974].

The first five topics relate to the watershed, hydrologic and transport phases of the urban NPS runoff system. Topic 6 is related to the subject of this report and is probably the least studied but most significant at this time.

Our literature review showed that ecological effects of urban NPS runoff have largely been ignored and we believe this has occurred because controls are usually oriented toward contaminant production. However, because of the high cost of controls for urban NPS runoff, it is essential that relationships between urban runoff and ecological impacts be determined. When these relationships are defined, proper priorities for establishing control measures can be instituted.

This report is directed at NPS, although frequently the impact of urban watersheds on stream ecosystems is the result of other categories of pollutant sources. An example of such a major source is combined sewer overflow. Combined sewer overflow behaves very similarly to any urban runoff event but will not be discussed directly herein because this review and analysis is confined to NPS. However, we realize that combined sewer effects may occur in specific instances and may not have been separated in the reports reviewed. Also, ecosystem impacts can be viewed as independent of the source; thus, ecosystem and community concepts relate to combined sewer overflow as well as direct urban runoff. The concepts developed herein apply to any pollutant input that has random, relatively discrete, single event properties.

SECTION IV

THE SIGNIFICANCE OF URBAN RUNOFF

SOURCES AND CONCENTRATIONS OF CONTAMINANTS IN URBAN RUNOFF

Water pollutants from urban watersheds are produced by a variety of activities (sources) that are affected by location variables and activity intensity. Attempts to identify regional characteristics have been largely unsuccessful. For example, *Bradford* [1977] tried to normalize his data to perform this analysis. A statistical summary of data available through 1972 relating land use to materials on urban streets available for removal by storm water, did not show significant patterns or relationships. The author felt that the data were inadequate to show relationships because of different methods of collection and analysis. These differences introduce variance which is not representative of the actual processes taking place.

The types of pollutants are somewhat typical of many point sources (Table 1). However, unlike many point sources such as wastewater effluents, concentrations in NPS waters are extremely variable (Table 2). One way of looking at the effects of urban runoff is to compare the effects of urbanization. For example, *McGriff* [1972] concluded that urbanization increased the volume of runoff and the size of the flood peak, and decreased the lag time. Effects on water quality were threefold: (1) sediment load increased, (2) groundwater recharge was minimized; hence, long term stream flow augmentation and dilution was reduced, and (3) eutrophication resulted with consequent effects on DO and related variables.

Urban watersheds typically consist of transportation surfaces, loading dock surfaces, building surfaces, and vegetated surfaces. These surfaces vary considerably in their permeability. Roads and sidewalks, loading surfaces, and buildings are essentially impervious and all of the precipitation becomes runoff with minimal lag time.

The Soil Conservation Service identifies four types of soils that affect the amount of runoff [*Kent* 1973]. Type A has the lowest runoff and is composed of sand and gravel, type B has moderately low runoff and is mostly sandy, type C has moderately high runoff typically being composed of a shallow soil layer containing some clay, and type D has high runoff and is essentially impermeable because of the high clay content. Bare soil at constructions sites is a major problem in urban areas, particularly as a sediment source.

In most urban situations, bare soil is negligible and permeable land surface is generally a vegetated surface such as gardens, parks, roadsides,

TABLE 1. RANGE OF CONCENTRATION IN URBAN STORMWATER [*WANIELISTA* 1978]†

Variable	Low	High
BOD_5	1	700 mg/ℓ
TOC	1	150 mg/ℓ
COD	5	3,100 mg/ℓ
SS	2	11,300 mg/ℓ
Total Solids	200	14,600 mg/ℓ
Volatile Total Solids	12	1,600 mg/ℓ
Settleable Solids	0.5	5,400 mg/ℓ
Organic N	0.01	16 mg/ℓ
TKN	0.01	4.5 mg/ℓ
NH_3N	0.1	2.5 mg/ℓ
NO_3N	0.01	1.5 mg/ℓ
Soluble PO_4	0.1	10 mg/ℓ
Total PO_4	0.1	125 mg/ℓ
Chlorides	2	25,000 mg/ℓ*
Oils	0	110 mg/ℓ
Phenols	0	0.2 mg/ℓ
Lead	0	1.9 mg/ℓ
Total Coliforms	200	150×10^6/100 mℓ
Fecal Coliforms	55	110×10^6/100 mℓ
Fecal Streptococci	200	1.2×10^6/100 mℓ

† Table adapted and used with permission of publisher, Ann Arbor Science Publ.
* With highway deicing.

TABLE 2. SUMMARY OF NONPOINT SOURCE CHARACTERISTICS [*LOEHR* 1974]

Source	Concentration (mg/l)					Area Yield Rate (kg/yr/ha)					Surface Area of Interest
	COD	BOD	NO_3-N	Total N	Total P	COD	BOD	NO_3-N	Total N	Total P	
Precipitation	9-16	12-13	0.14-1.1	1.2-1.3	0.02-0.04	124	—	1.5-4.1	5.6-10	0.05-0.06	Total land area
Forested land	—	—	0.1 -1.3	0.3-1.8	0.01-0.11	—	—	0.7-8.8	3-13	0.03-0.9	Forest area
Range land	—	—	—	—	—	—	—	0.7	—	0.08	Range land
Agricultural crop land	80	7	0.4	9	0.02-1.7	—	—	—	0.1-13	0.06-2.9	Active crop land
Land receiving manure	—	—	—	—	—	—	—	—	4-13	0.8 -2.9	Crop or unused land used for manure disposal
Irrigation tile drainage, western U. S.											
Surface flow	—	—	0.4-1.5	0.6-2.2	0.2 -0.4	—	—	—	3-27	1.0 -4.4	Irrigated western soils
Subsurface drainage	—	—	1.8 -19	2.1-19	0.1 -0.3	—	—	83	42-186	3-10	Irrigated western soils
Crop land tile drainage	—	—	—	10-25	0.02-0.7	—	—	—	0.3-13	0.01-0.3	Active crop land requiring drainage
Urban land drainage	85-110	12-160	—	3	0.2 -1.1	220-310	30-50	—	7-9	1.1-5.6	Urban land areas
Seepage from stacked manure	25,900-31,500	10,300-13,800	—	1,800-2,350	190-280	—	—	—	3	—	Manure holding area
Feedlot runoff	3,100-41,000	1,000-11,000	10-23	920-2,100	290-360	7,200	1,560	—	100-1,600	10-620	Confined, unenclosed animal holding areas

* Data do not reflect the extreme ranges caused by improper waste management or extreme storm conditions, the data represent the range of average values reported in previous tables.

and vacant lots with ground cover. Thus, the runoff from vegetated surfaces can approach the quantity of precipitation only if the ground is saturated by previous precipitation or if the soil is of an impermeable type. Soils covered with vegetation are quite permeable and surface runoff usually does not occur unless extensive periods of precipitation saturate the soil. Interflow (subsurface flow) and delayed runoff are important effects of such vegetated surfaces. The relative mixture of impermeable and permeable surfaces depends on cultural, socio/economic, geographic, geologic, climatic and urban planning/management variables.

Deposition of materials on urban surfaces is highly dependent on other related activities. For example, vehicles directly deposit (1) tire particles, (2) oil, grease, and fuels, (3) spilled materials from transported goods, and (4) fallout from vehicle exhaust emissions. The amount of these materials depends on the kinds of vehicles (automobiles, light trucking, heavy trucking) and the intensity of use by each category of vehicle. Other materials that can be found on transportation surfaces include wastes from pets and birds, solid wastes (trash collection loss, litter, garbage, leaves, and other vegetation), and industrial and power plant air pollutant fallout. Street sweeping and washing reduce the amounts but may be relatively ineffective in removing some kinds of pollutants [*Sutherland and McCuen* 1978]. The percentage of land surface covered by roads and sidewalks controls the relative impact of these activities on urban runoff.

Barkdoll et al. [1977] concluded that dustfall effects on urban runoff quality were very contaminant specific. In a Knoxville, Tennessee watershed, mass flow of COD, Hg, Cl, As, and PO_4-P in stormwater was closely related to dustfall levels for these contaminants. They also concluded that minerals and solids have relatively constant concentrations from storm to storm while heavy metals, nutrients, and COD have decreasing concentration with increasing runoff. Another significant conclusion was that pollutant removal is primarily a function of total runoff and only slightly affected by runoff intensity (depth of runoff).

Sartor and Boyd [1972] found that the total amount of accumulated contaminant material showed a relationship to elapsed time since the last rain or street sweeping. There was wide variance in the amount of contaminant material observed (290-3,500 lb/curb mile), but it averaged about 14,000 lb/curb mile in the cities tested. The largest portion of polluting substances was associated with fine solids in the street surface contaminants. Rainfall intensity, street surface characteristics, and contaminant particle size were the controlling variables for the rate of rainfall wash-off but street sweeping was primarily aesthetic because it did little to remove contaminants that would be carried in runoff, *i.e.*, fine grained particles. Catch basins were found to remove coarse grained inorganic materials, but were not effective in removing fine particles.

Loading dock areas and buildings can serve as significant sources of pollutants depending on the type of construction material and activities that occur. Spilled chemicals, oils and grease, and other materials at loading docks occasionally provide massive inputs to streams. The density and type of loading activity will control the probability of such events.

Similarly, buildings and the activities associated with buildings are distributed at changing density within an urban area. A comparison of runoff quality from different types of buildings indicates that contaminant load is generally greater in urban than in commercial areas, which is greater than in single family residential areas [*Randall et al.* 1978]. This observation may be more closely associated with traffic than building type and density, although it is difficult to say with complete surely. Traffic volume and contaminant load is greater on highways than city streets, which in turn are greater than for rural road systems [*Wanielista* 1978].

STORMWATER AND POLLUTANT MODELING

The concepts presented in this section, although elementary and somewhat incomplete, are intended only as the introduction to a complex subject. The concepts discussed are those that relate to assessing the impacts of urban runoff on receiving streams. There are detailed models in the literature. Many are evaluated by *Wanielista* [1978], but only two, the U. S. Army Corps of Engineers model (STORM) and the USEPA model (SWMM), are applied by him as illustrated examples. Most models use flow weighted average values and are of some value for management but do not relate to the stream impacts of urban runoff as described herein [*Wu and Ahlert* 1978]. The example publications cited in the following paragraphs illustrate this problem; however, the utility of specific models such as STORM and SWMM for generating management questions and possibly some ecological questions cannot be excluded.

A simulation model for stormwater management (SWMM) was developed by *Lager et al.* [1971]. The model simulates both stormwater quantity and quality and is capable of representing combined wastewater overflow phenomena. The model was validated for the Baker Street combined sewer basin in San Francisco.

Hayden et al. [1972] developed a method using computer modeling techniques of predicting runoff management problems and their effects on a small urbanizing watershed.

Using a modification of EPA's SWMM, *Roesner et al.* [1972] evaluated the effects of land use changes on the quantity and quality of urban runoff in the City of San Francisco. Under the simulation of changing a 305 acre park to multiple residential housing, large increases were predicted for watershed outflow, suspended solids mass emission rate and concentration, and BOD mass emission rate.

Graham et al. [1974] developed equations useful in predictive modeling to estimate imperviousness and specific curb length for urban watersheds. The equations use Census data, including households per acre, population per acre, and employment per acre.

Shih et al. [1975] used a simulation approach to model urban runoff hydrology for small (<20 mi^2) urban watersheds. The modeling method developed estimates of runoff rates and associated confidence limits. Data inputs are obtained from topographical maps (watershed area, slope, channel slope and length), soil survey maps (soil type and approximate minimum infiltration capacity), and National Weather Service publications (rainfall intensity-duration frequency).

McElroy et al. [1976] compiled a handbook of loading functions of pollutants from nonpoint sources which included urban runoff. *Wu and Ahlert* [1978] recommended homogeneous-land-use and statistical synthetic approaches as accurate and practical methods for storm runoff load prediction.

Phamwon and Fok [1977] developed a digital computer model for urban runoff simulation. The model was checked using data for a Hawaiian small urban watershed. Model performance was judged satisfactory. They were criticized by *Golding* [1978] particularly for the overall approach and instrumentation for measuring flow.

Rovey and Woolhiser [1977] developed a computer model for estimating runoff from urban watersheds which relies on a series of planes and channels (kinematic cascade) to simulate a complex watershed. The model was successful in predicting runoff from a watershed near Northglenn, Colorado.

Diskin et al. [1978] developed a two-component parallel cascade model for predicting urban runoff. The model input is the total rainfall hyetograph for the watershed. The two elements of the model calculate runoff from the impervious and pervious area of the watershed in parallel; then the summation of the two elements represents the final output.

Jewell et al. [1978] developed a method for calibrating quantity-quality coupled urban stormwater management models. The method calibrates the quantity model first and then the quality model. Calibration is done primarily by adjusting watershed characteristic parameters for the quantity model, and by adjusting pollutant buildup rates for the quality model. Average conditions measured for several storms are recommended as calibration data.

Work by *Whipple et al.* [1977] indicated that it is not valid to assume that pollution from urban runoff increases with elapsed time since a previous rainfall. Earlier assumptions were erroneously made because there had been insufficient total pollution loading data available for individual storm events. The authors criticize other assumptions (*e.g.*, the effect of street sweeping) in recent modeling efforts (*i.e.*, stormwater management model of EPA) and call for a wider questionning of the relationship between storm characteristics and pollution loadings. In answer to the criticism of *Whipple et al.* [1977], *Field* [1978] pointed out that the USEPA's stormwater management model (SWMM) includes nonlinear storm runoff-washoff functions which were not taken into account by *Whipple et al.* [1977]. He also points out that the SWMM dry-weather pollutant accumulation function is based on real data and is not arbitrary. The SWMM model is not a "complete solution for urban runoff problem assessment... but is an accepted tool in developing solutions to this complex area of environmental management."

Sutherland and McCuen [1978] used a computerized urban nonpoint pollution management model (developed in 1976) that estimates the accumulation, removal by rainfall, and removal by street sweeping of eight pollutants (total solids, volatile solids, BOD_5, COD, TKN, NO_3, PO_4, total heavy metals) to simulate the effects of street sweeping operations in the Washington, D.C. area. They concluded that street sweepers did not ordinarily remove more than 33 percent of the pollutants accumulating on streets because of (1) the inability of sweepers to remove small particles (which contain most of the pollutants) and (2) the less than optimal street sweeping operation procedures.

HYDROLOGY AND TRANSPORT

Storms at a specific location and season tend to occur randomly with respect to time, space, and intensity (Figure 4). Probabilities of occurrence for precipitation events that have a specific intensity (depth of precipitation per hour) for a specific duration are called intensity-duration-frequency curves [*Wanielista* 1978]. Different localities will have different patterns.

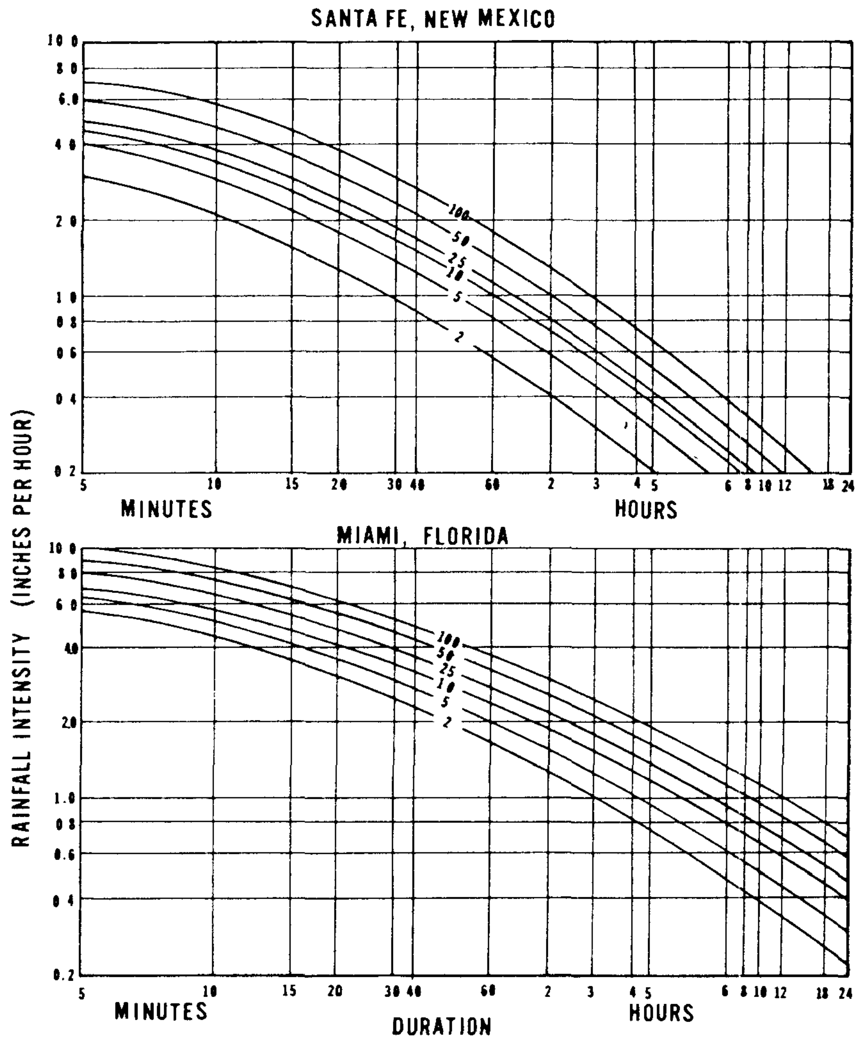

Figure 4. Intensity-duration-frequency curves for rainfall. [Taken from *Wanielista* 1978; used with permission of Ann Arbor Science Publ.].

The precipitation events themselves are probabilistic events; that is, they are random, mutually independent events. One can predict a long term average with a certain probability of success but the occurrence of specific events is not predictable and follows a binomial distribution. Runoff is affected by these time, space, and intensity variables, plus other variables associated with the watershed. These considerations have led to concepts that are specifically related to management [*Wanielista* 1978]: the hydrograph, the pollutograph, and the loadograph (Figure 5). It should be mentioned that this terminology is not universally accepted, although the terms illustrate useful concepts [*Whipple et al.* 1977]. Chemograph has been used in place of pollutograph.

The hydrograph is a relationship between flow (Q) and time (t); flow increases to a peak value and then decreases exponentially depending on storm

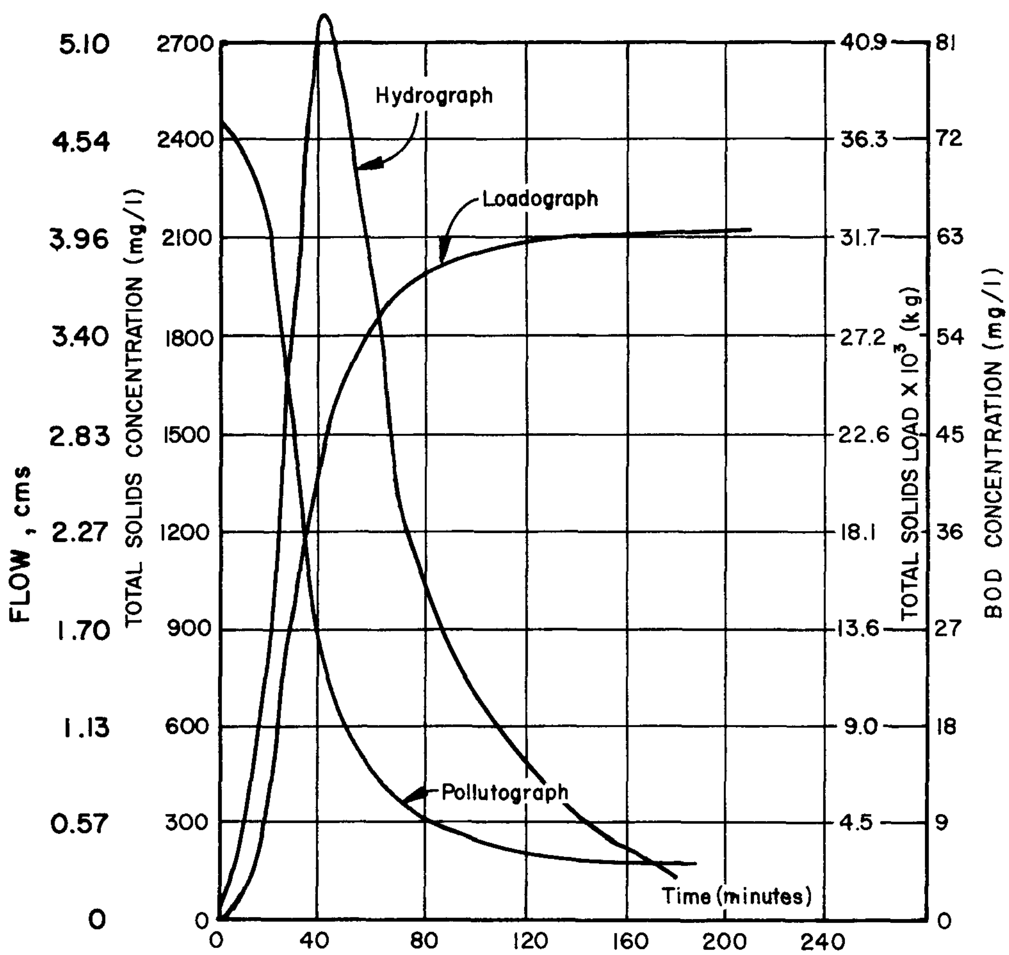

Figure 5. An idealized example of the hydrograph and BOD and SS concentration (pollutograph) and load (loadograph) as a function of time. [Adapted from *Wanielista* 1978; used with permission of Ann Arbor Science Publ.].

and watershed characteristics. The pollutograph is the concentration of the specific contaminant (CC_i) in the runoff; the loadograph is the mass flow ($Q*CC_i = SL$). An example of a pollutograph/hydrograph shows how TOC, SS, and nutrients vary with time and flow (Figure 6) and illustrates several principles:

1. The "first flush effect" where concentrations are higher in the initial runoff because of buildup of materials between storm events and subsequent dilution effects of storm water.

2. The transport of particles requires sufficient velocity to pick up particles and keep them in suspension. This is especially apparent in comparing the N and P compounds where TKN and total P are associated with particulate and the other nutrient forms are ionized.

3. While water quality variables are measured in the runoff, it is the impact of those variables on the aquatic community that requires analysis to interpret impacts on stream ecosystems.

Although many investigators have looked at the storm water hydrologic/quality system, the impacts of the storm water on receiving waters have not been carefully evaluated. Using the data in Figure 6 [*Randall et al.* 1978] for suspended solids in storm water, the suspended solids concentration in a receiving stream can be calculated (assuming behavior as a conservative substance) to illustrate the impacts of a storm event on that stream (Figure 7). The average stormwater flow over the 90 minute calculation interval is 0.117 cms, while the peak flow was 0.150 cms and the minimum flow was 0 cms. The storm runoff characteristics are described by the hydrograph, the loadograph, and pollutograph. However, these events are greatly affected by the relative amount of dilution in the stream itself.

The maximum impact of the storm runoff occurs with a stream flow of 1 cms (approximately 8.5 times the flow of the runoff event) and at approximately 50 minutes after the beginning of the runoff event. Increasing the dilution factor by a factor of 5 (approximately 40 times the runoff flow) reduces the impact by a corresponding factor. Further dilution eventually produces a condition where essentially no measurable impact occurs (streamflow > 100 cms). It is evident that a stochastic approach is needed to assess the impacts of the urban runoff on water quality in the stream. The stream flow varies extensively with time and is greatly affected by storm events. The combination of stream variation with storm runoff variation and pollutant concentration variation requires that the stream impact be assessed using a stochastic approach. The components of the aquatic community are impacted by those variables as events which may have a short term or a long term impact on their growth or survival.

Although the stormwater runoff, as shown in Figure 6, is measured as if it were entering the stream in Figure 7 at a specific and discrete point over the time interval and it is assumed that there is instantaneous and complete mixing in the stream, in reality urban runoff is transported by a wide variety of hydrologic systems. These systems can be categorized as surface or subsurface, point input or lateral input, delayed inflow or direct

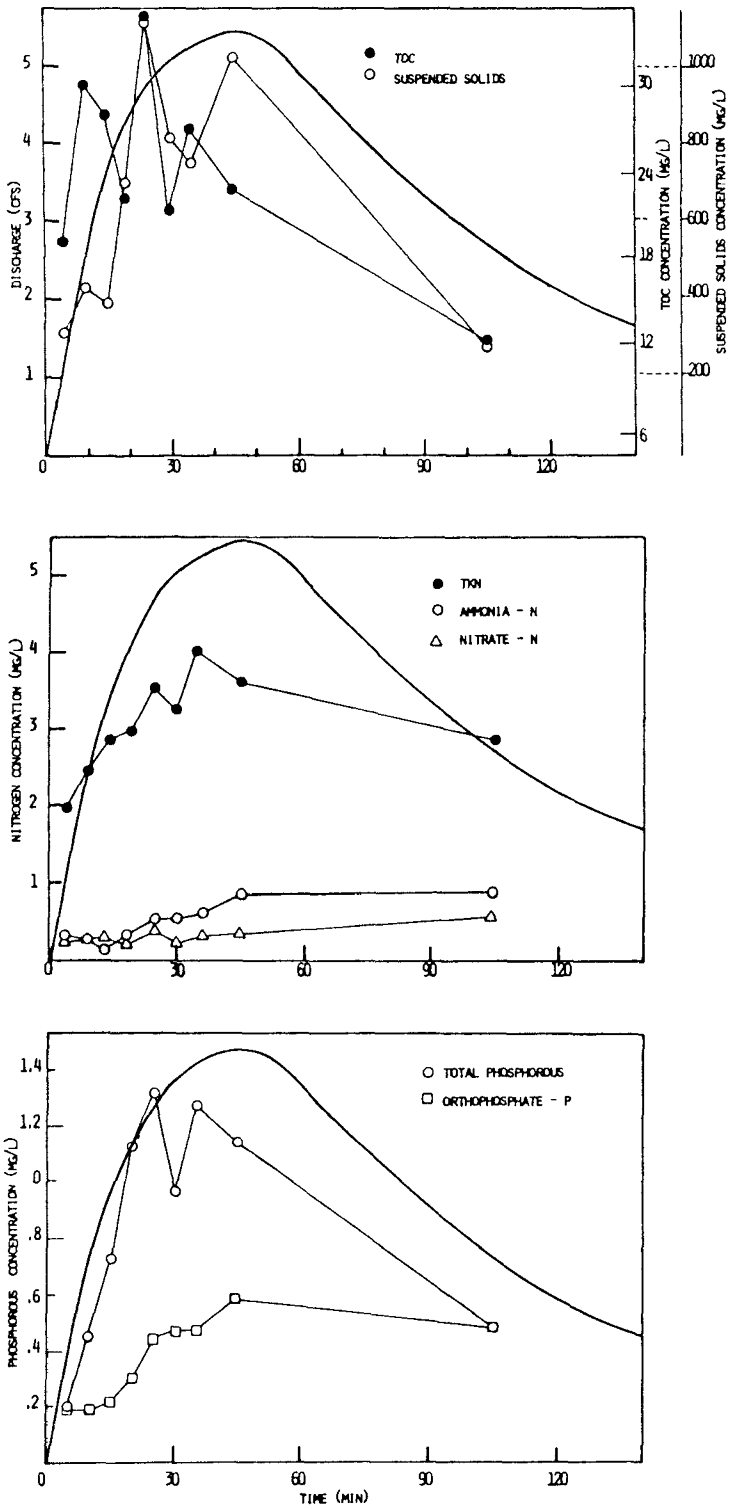

Figure 6. Variation of flow and water quality with time during a runoff event. [Taken from *Randall et al.* 1978; Denver Street Station.]

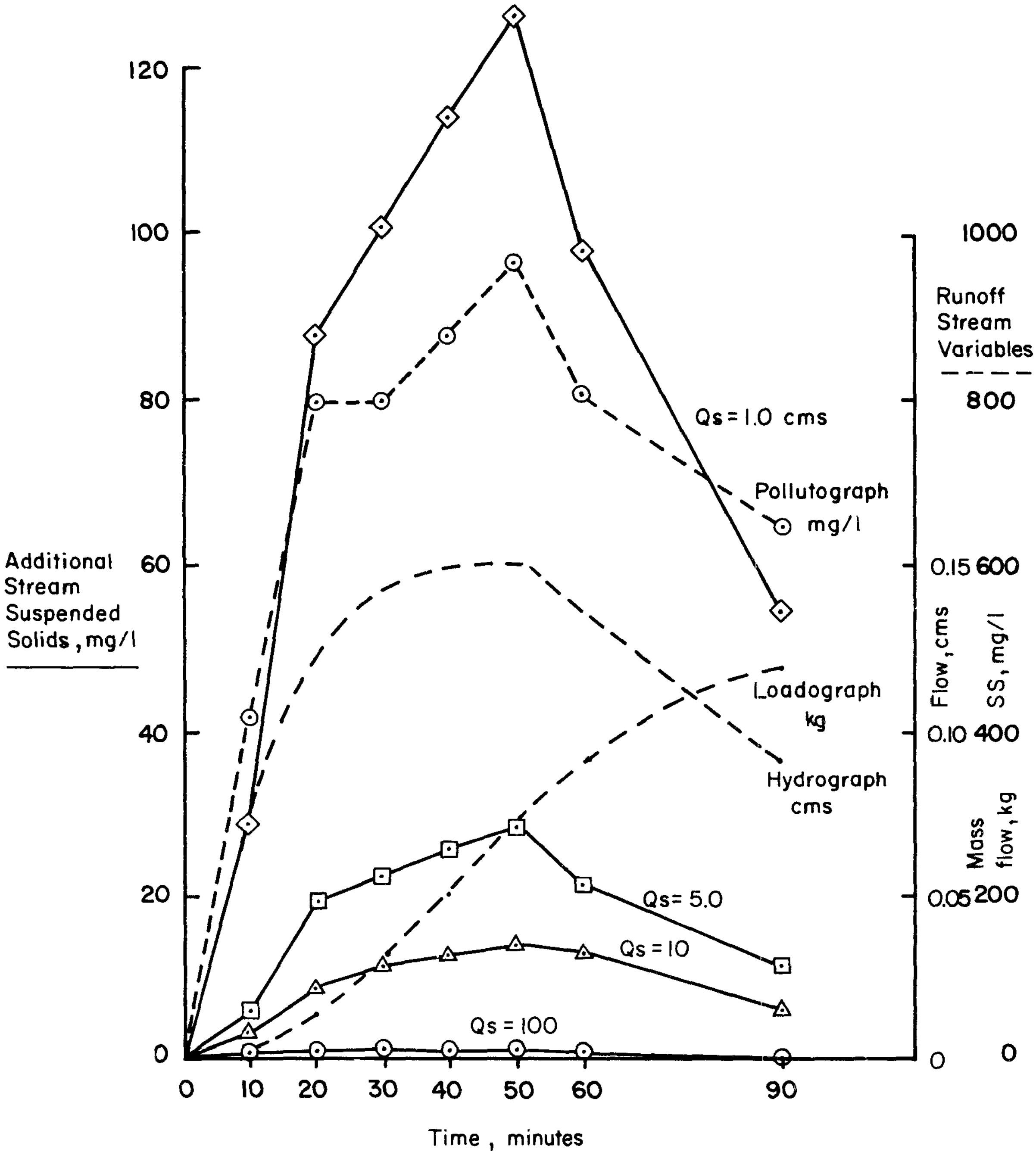

Figure 7. Effects of urban runoff on water quality of an hypothetical stream (runoff data interpolated from Figure 6 data).

inflow (Figure 8). Although many examples occur, the four shown are probably the most typical kinds of inputs to streams receiving urban runoff. Example No. 1 shows a storm drain entering a retention basin. Systems of such basins appear to be a valuable means of managing urban runoff [refer to *Haan and DeVore* 1978].

For example, *Dale* [1978] discussed the efforts of the Metropolitan Sanitary District of Greater Chicago (MSDGC) to control flooding and pollution from urban runoff through the construction of a system of tunnels and reservoirs to hold runoff water until it can be treated. The system, currently under construction, will serve a 971 km^2 (375 mi^2) area in the center of metropolitan Chicago. When completed, the project is projected to save $750 million/year in flood and water pollution damage. The system will help protect Lake Michigan and allow local waterways to meet "fishable and swimmable standards."

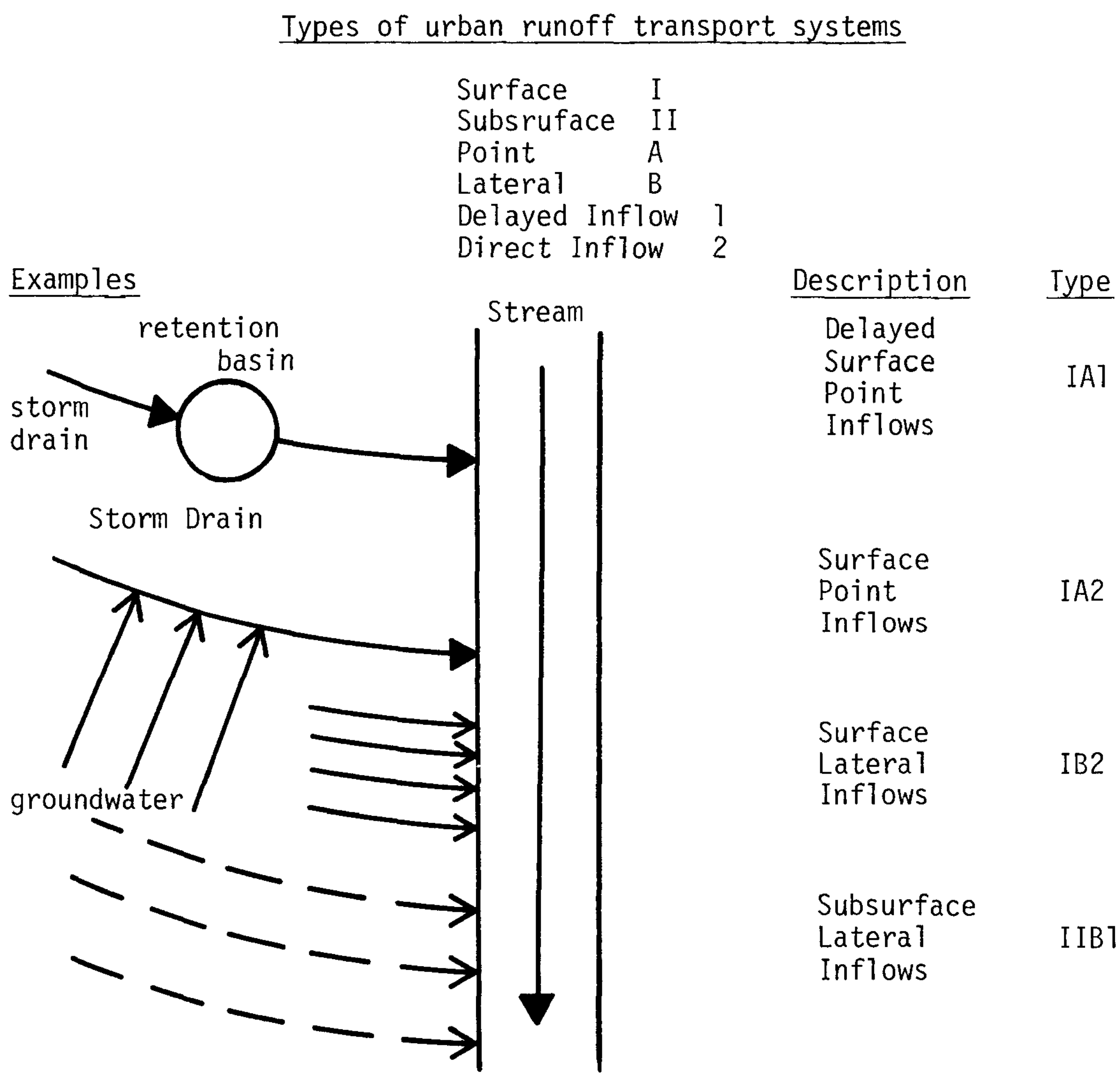

Figure 8. System classification of transport of urban runoff to streams.

The delayed inflow of example 1 could represent a diversion of storm drain water into a treatment plant as might occur for first flush influents. *Cherkauer* [1977] reported on the ability of a series of lakes in a small stream system in the suburbs of Milwaukee, Wisconsin, to modify the intensity and quality of urban runoff. The storage capabilities of these lakes have a positive impact by lowering the peak discharge and "flashiness" of surface runoff and increasing the length of time a stream is above baseflow levels during a storm event. The lakes also even out concentrations of dissolved solids including Na and Cl which are deposited during road salting. Another example of a delayed flow mechanism is the use of wetlands for treatment of urban runoff waters [*Hickock et al.* 1977]. The wetlands were effective in upgrading the quality of urban runoff if application was properly managed, and apparently no impacts to the wildlife or vegetation (type or abundance) occurred as a result of the project.

Thus, the discharge of urban runoff into streams can take a wide variety of inputs, further complicating the idealized picture shown in Figure 7. Impact on aquatic communities is dependent on the temporal aspects as exemplified in Figure 7 and the spatial aspects as exemplified in Figure 8. All of these aspects combine with the watershed characteristics to produce the input to the aquatic ecosystem. The aquatic ecosystem is impacted by those variables associated with the urban runoff as it is diluted in the streams. These variables include flow, various water quality parameters, the character of the stream in relation to the urban runoff and the composition of the aquatic community itself.

For purposes of simplification, we define short term and limited impacts as those areas within the stream where no permanent and/or significant effect on the aquatic community occurs. There are certain types of impacts that might be limited in space but have a marked effect on the aquatic community. An example would be the development of a barrier that prevents movement of higher organisms past a specific point due to water quality problems. DO deficits can function in this manner to prevent fish migration. Another example is the loss of an essential component of an aquatic community, which may affect survival of other components. With these constraints in mind, we can ignore certain minor impacts that result from water quality changes that are limited in area and time. In many respects a "mixing zone" concept can be considered as the zone where these small impacts take place [*USEPA* 1976].

WATER QUALITY VARIABLES

As stated in the Introduction, we describe the flow input phase of the urban runoff process in terms of the watershed and the hydrologic and transport systems. The material input phase of that process involves specific physical, chemical, or biological variables that represent contaminants picked up in the runoff and transported into the stream. A listing of selected variables important in analyzing urban runoff and the activities that produce them indicates that considerable variation in activity control measures would be required to improve water quality across the range of variables selected (Table 3). Although many studies have been performed on these variables, they are not always measured equally well nor at similar

TABLE 3. ESTIMATED EFFECTS OF CHANGES FROM NATURAL CONDITIONS IN WATERSHED USE ON SPECIFIC WATER QUALITY (WQ) VARIABLES

Major Identified Urban Watershed Use	Flow	Alkalinity, Hardness, pH	TDS	Temp.	Sediments (Bed & Suspended)	DO	Pathogenic Bacteria	Detritus	Dissolved and Particulate Organics	Pesticides	Nutrients	Toxic Materials	Impact Totals
1. Drinking Water Source (No flow restriction), aesthetic preservation. Geological/ geographic/climate as is.	0	0	0	0	0	0	0	0	0	0	0	0	0
2. Residential	+	+	+	+	+	+	+	+	+	+	+	0	11
3. Transport and Corridor (auto/truck, streets, highways, railroad, powerlines, airports, waterway/navigation)	+	+	+	+	+	+	0	0	+	0	+	+	9
4. Industrial/Power Plants	+	+	+	+	+	+/-	+	+	+	+	+	+	12
5. Municipal/Commercial	+	+	+	+	+	+	+	+	+	+	+	+	12
6. Parks, Vegetated Land, Gardens	0	0	0	0	+	+	+	+	0	+	+	0	6
7. Recreational (Pets, land related, water related)	0	0	0	0	0	+	+	+	+	0	+	0	5
8. Air Pollutant Fallout	0	+	0	0	0	0	0	0	0	0	+	+	3
WQ Impact TOTALS	4	5	4	4	5	6	5	5	5	4	7	4	58

0 = minimal or no effect of activity on variable
+ = increase in variable
- = decrease in variable
+/- = increase or decrease in variable with increased activity

frequencies.

Concentrations (Table 4) and mass loadings (Table 5) of many of these variables indicate that runoff concentrations can be higher than raw sewage and have significant impact on a receiving stream. The values in Tables 4 and 5 were obtained by arithmetically averaging data summarized in the cited papers. Even with relatively low loadings and high dilution of highly toxic materials (mercury and some of the chlorinated hydrocarbons), environmental and other criteria [abstracted from *USEPA* 1976] can be surpassed for a period of time (Table 6). Assuming an urban area of 10 km^2 and mass input within a single 24-hour period, the average concentration in a stream would follow the relationships shown in Figure 9.

The interaction between watershed activities, water quality variables, and stream ecosystem responses is best understood by separating the different components of the system. An abstract example (Figure 10) shows that each phase of the urban NPS runoff process affecting the stream ecosystem can be separated in terms of its impact on the stream response variables.

Thus flow and flooding and specific water quality variables such as salinity, suspended solids, organics, bacteria, and nutrients interact with specific aquatic physiological and ecological functions. These functions include respiration and photosynthesis and the concepts of structure, such as food webs, succession, and trophic levels. In this section we discuss selected literature examples of chemical, physical, and biological variables traditionally associated with water quality [see *Pennack* 1971, *Gakstatter et al.* 1977, *Ott* 1978a, b, for general review and discussion of water quality and various indices].

Flow and Flooding

Urbanization results in increased imperviousness of the watershed which in turn enhances the intensity and volume of runoff from the watershed [*Cech and Assaf* 1976, *Gundlach* 1976, *Hossain et al.* 1978]. These factors often link urbanization with increased flooding of streams draining these watersheds [*Doehring and Smith* 1978].

Under conditions of uncontrolled development in the Menomonee River watershed in Wisconsin, *Walesh and Videkovich* [1978] found increases of up to 4.5 times in predicted 100 year flood stages.

The hydrologic alterations of urban development in coastal areas were studied by *Cech and Assaf* [1976]. In the Houston, Texas area, the quantity of urban runoff was 3-5 times more than non-urban runoff, and 2-2.5 times higher in the smaller Beaumont area.

Salinity

Most severe salt impacts result from road deicing. Timing is critical due to season, freeze-thaw cycles, storm events [*Scott* 1976]. *Jodie* [1975] reported on the quality of storm water from urban freeways in Milwaukee, Wisconsin, and showed that salt concentrations were generally high with surges

TABLE 4. CONCENTRATIONS OF CONTAMINANTS IN URBAN RUNOFF

Reference	mg/ℓ								
	BOD_5	COD	TS	SS	TDS	Tot. P	Cl^-	NO_3-N	Alkalinity
Max[1]	620	3100	15000	36000					
Min	2	20	260	5					
Mean	46	100	3700	660					
n	6	4	5	4					
Max[2]	280	3100	14000	14000	1100	7.3	430		
Min	1	20	270	5	20	< 0.02	2		
Mean	21	100	1500	5000	190	1.10	10		
n	6	6	2	3	2	3	2		
Max[3]	72		3200	2500	1600			14	270
Min	10		80	52	20			< 1	20
Mean	25		900	530	250			< 3	93
n	4		4	4	4			2	2

	Ag	Cr	Cu	Fe	Mn	Ni	Pb	Zn	Fecal Coliform MPN/ 100 mℓ
Max[2]									240000
Min									< 2
Mean									126000
n									4
Max[4]	0.066	0.178	0.072	4.822	0.842	0.066	0.598	0.694	240000
Min	0	0.017	0.028	1.528	0.132	0	01080	0.035	9
Mean	0.017	0.083	0.056	2.973	0.408	0.018	0.271	0.211	36500
n	7	7	7	7	7	7	7	7	12

[1] *Brownlee et al.* [1970].
[2] *Whipple et al.* [1974a].
[3] *Wells et al.* [1975].
[4] *Randall et al.* [1978].

TABLE 5. LOADINGS OF URBAN RUNOFF

Reference	kg/km^2, day												
	BOD_5	TOC	SS	VSS	Tot. P	Tot. N	Cl^-	NH_3	Pb	Zn	Cu	Ni	Cr
Max[1]	180	350			8.50		150						
Min	5	11			0.27		36						
Mean	43	96			2.76		93						
n	11	7			7		6						
Max[2]	121*							6.7*	2.4	1.22	1.28	1.39	0.47
Min	34							3.9	0.6	0.33	0.07	0.06	0.03
Mean	79							4.9	1.4	0.78	0.47	0.54	0.23
n	3							3	4	4	4	4	4
Max[3]	10	36	940	86	1.65	18							
Min	4	15	270	34	0.63	3							
Mean	7	23	500	63	1.11	8							
n	4	4	4	4	4	4							

* Projected or estimated data.
[1] *DiGiano et al.* [1976].
[2] *Whipple et al.* [1976b].
[3] *Randall et al.* [1978].

TABLE 6. FRESHWATER ENVIRONMENTAL CRITERIA (TOXICITY, IRRIGATION, AND OTHERS). LOWEST VALUE CITED EXCEPT FOR VARIABLES CITED AS DRINKING WATER STANDARDS†,*.

Concentration, *mg/ℓ* as stated material					
Metals		Pesticides & Other Organics		General Quality Variables	
Arsenic	0.05*	Aldrin/dieldrin	3×10^{-6}	Aesthetics	**
Barium	1.	Chlordane	1×10^{-5}	Alkalinity (as $CaCO_3$)	≥20.
Beryllium	0.011	2, 4-D	1×10^{-1}*	Ammonia (as NH_3)	0.02
Cadmium	0.010* (0.0004-0.0012)	2, 4, 5-T	1×10^{-2}*	Boron	0.75
Chromium	0.05*	DDT	1×10^{-6}	Chlorine	0.002
Copper	1.* (0.1•96 hr LC 50)	Demeton	1×10^{-4}	Coliform, Fecal	**
Iron	0.3*	Endosulfan	3×10^{-6}	Color	**
Lead	0.05* (0.01•96 hr LC50)	Endrin	2×10^{-4}* (4×10^{-6})	Cyanide	0.005
Manganese	0.05*	Guthion	1×10^{-5}	Dissolved Oxygen	≥5.
Mercury	0.002* (0.00005)	Heptachlor	1×10^{-6}	Gases	≤110%
Nickel	(0.01•96 hr LC50)**	Lindane	4×10^{-3}* (1×10^{-5})	Hardness	**
Selenium	0.01* (0.01•96 hr LC50)	Malathion	1×10^{-4}	Nitrates/Nitrites	10.*
Silver	0.05* (0.01•96 hr LC50)	Methoxychlor	1×10^{-1}* (3×10^{-5})	Oil and Grease	Virtually Free **
Zinc	5.	Mirex	1×10^{-6}	pH 5-9 (6.5-9.0)	--
	(0.01•96 hr LC50)	Parathion	4×10^{-5}	Phenol	0.001*
		Phthalate esters	3×10^{-3}	Phosphorus	**
		Polychlorinated		Sulfide (H_2S)	0.002
		biphenyl (PCB)	1×10^{-6}	Tainting Substances	**
		Toxaphene	5×10^{-3}* (5×10^{-6})	TDS + Salinity (Cl^-, $SO_4^=$)	250.
				Temperature	**
				Turbidity (SS)	**

† Note: A discussion of mixing zones should be reviewed [*USEPA* 1976; p. 103].
* DWS, Environmental Criteria are shown parenthetically when lower than DWS.
**See *USEPA* [1976].

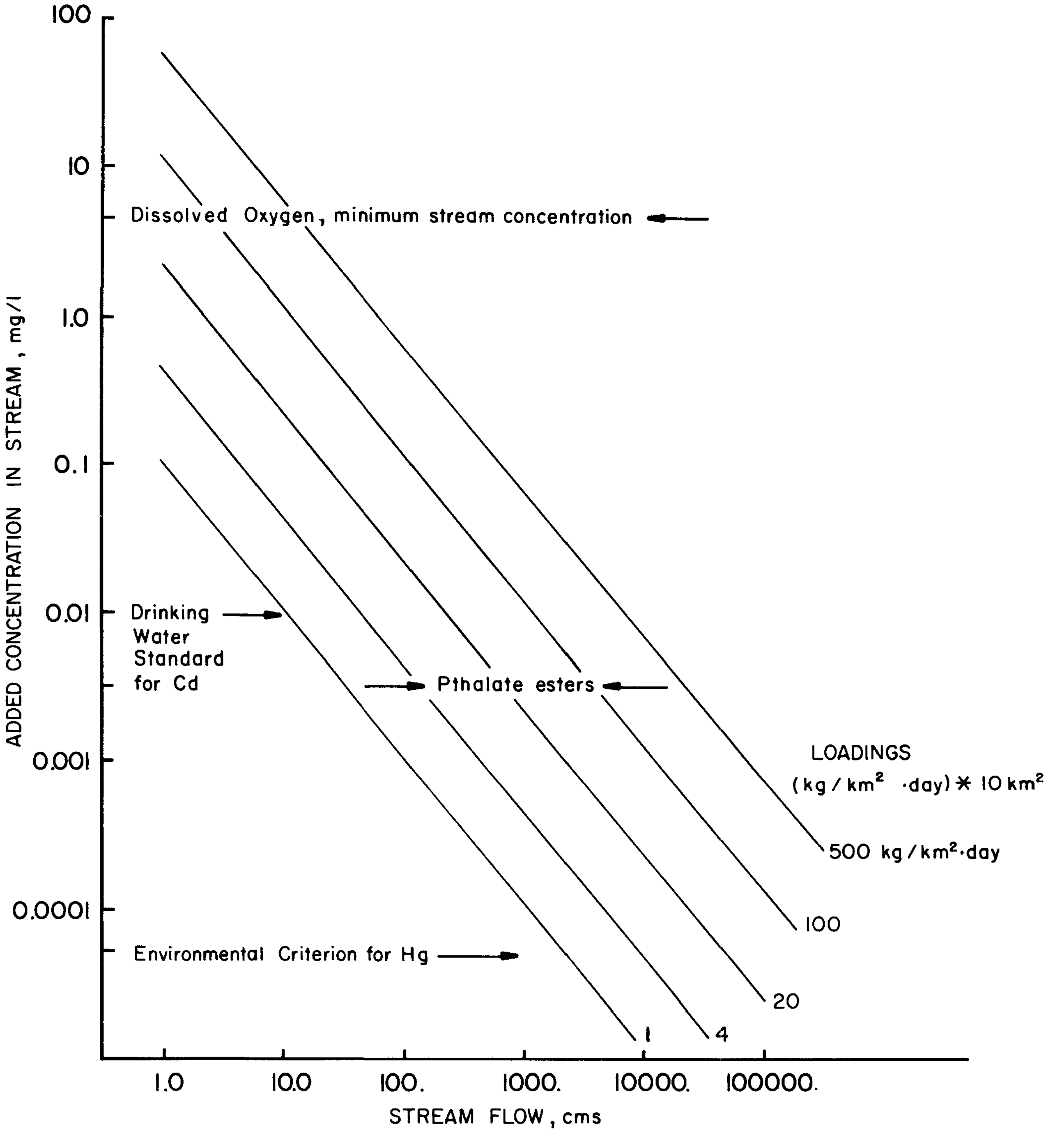

Figure 9. The relationship between streamflow and added concentration of contaminant from loadings. Levels of selected water quality variable criteria added to show significance (see Table 6).

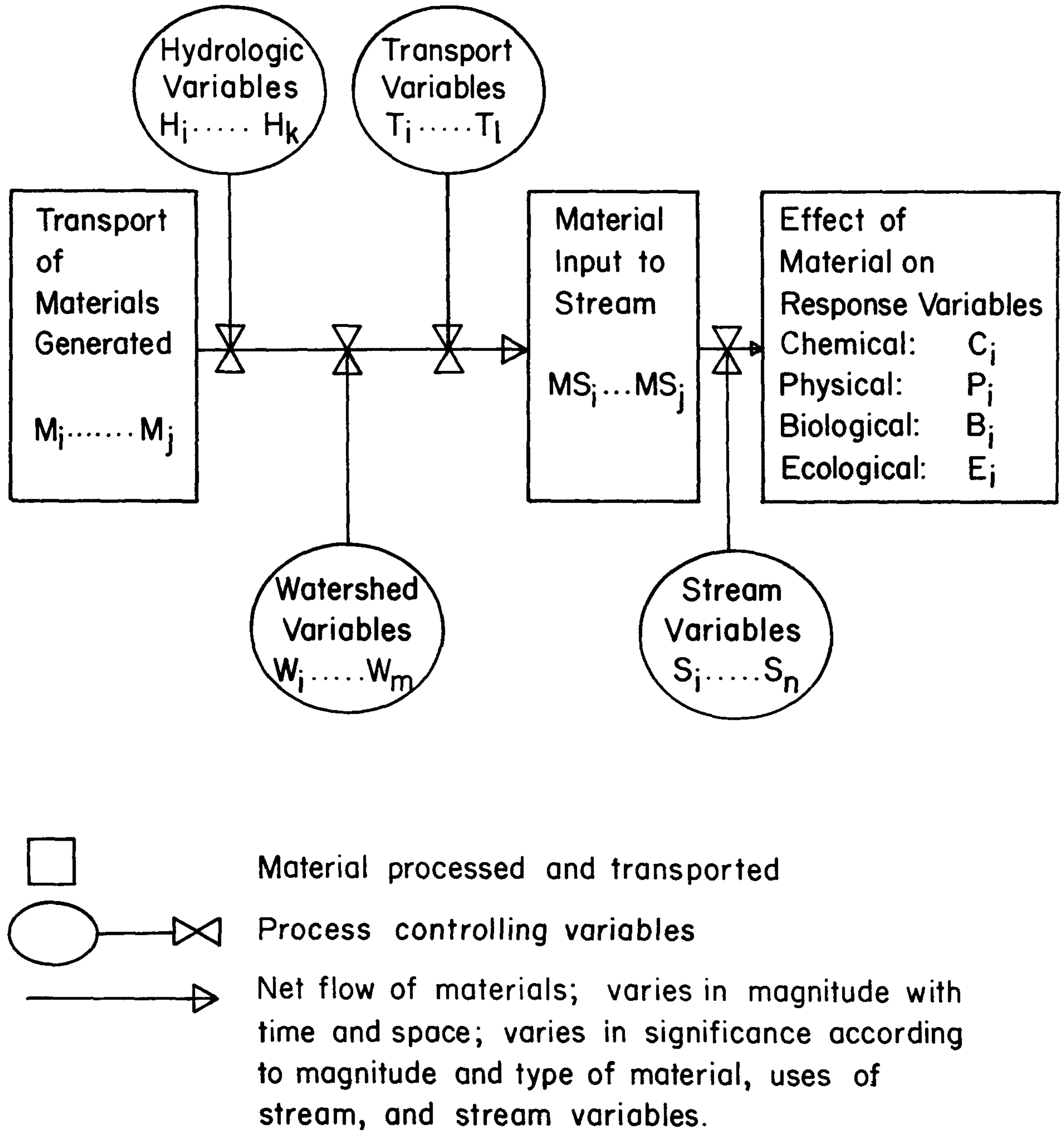

Figure 10. Separation of processes and concepts for analysis of urban NPS impact on stream ecosystem.

during the winter and spring. Although salt input from deicing may enter streams indirectly, it can usually be almost totally accounted for.

Hawkins [1976] reported that some of the salt from street deicing in a New York urban watershed was stored in the groundwater. This would delay input to streams and also continue input after seasonal road salting was completed. Approximately 82 percent of the applied salt could be accounted for in the watershed drainage.

Judd [1970] showed how significant salts could be when studying the effects of runoff on shallow, glacially-derived, First Sister Lake in Michigan. Studies were made from the winter of 1965 through the winter of 1967. There were heavy snows in both 1965 and 1967 and, therefore, road salts were used extensively. Due to input of road salts from the surrounding area, the lake stratified and failed to turn over during the winter and spring thaws of 1965 and 1967. The lake did turn over during the winter of 1966. The effects of the salt for the three years were:

	Sampled in April, µmho/cm at 18°C		
	1965	1966	1967
Surface (0 m deep)	442	339	361
Bottom (6 m deep)	690	366	648

Also, complete depletion of dissolved oxygen in the bottom layer occurred in 1965 and 1967.

Sedimentation

One of the most significant problems associated with urban NPS runoff is suspended solids (SS), inorganic and organic materials that cause turbidity and sedimentation problems. In Tucson, SS in runoff decreased as residential areas became more developed [*Mischi and Dhasmadhikari* 1971]. Frequently, "first flush" concentrations of SS are higher than raw sewage and later stages are equivalent to secondary effluents [*Corberg* 1977]. When combined with the flow produced in runoff, the impact of runoff pollutants is greater than the secondary effluent for the same area in Sydney, Australia [*Corberg* 1977]. One runoff event in Sydney lasting less than an hour that resulted from 13 mm of rain over a 131 km^2 area produced 1150 kg SS, 100 kg BOD_5, 13 kg PO_4, and 11 kg NH_3.

Ellis [1976] studied the water quality of urban runoff in the Silk Stream catchment which includes 3323 km^2 of North London, England. Rainfall-runoff lag times were only 20 to 30 minutes. The first flush carried between 350-3000 mg/ℓ SS, with the peak of solids and flow approximately coinciding. Volatile solids followed a similar pattern. Inorganic sediments make up 45-70 percent of the SS. The bulk of the sediments ranged between 0.1-0.5 mm diameter and showed a log normal distribution with a tendency toward bimodality as the distance of transport increased. The stream had a DO sag where sediments accumulated. Those sediments were high in Cd and Pb.

In presenting some of the findings of the International Reference Group on Great Lakes Pollution from Land Use Activities, *Judd and Carlson* [1978] point out that urban areas contribute high loadings of phosphorus, sediment, heavy metals, toxic organics, and chloride. This pollution is from storm and combined sewers as well as runoff.

The application of statistical procedures by *Haith* [1976] in selecting land use and water quality in New York river basins indicated that river concentrations of suspended solids, nitrogen, but not phosphorus were related to land use. High density residential land use was an important factor affecting suspended solids in rivers. However, the relative contribution to the river of point and nonpoint sources in suspended solids pollution from urban areas could not be estimated in the study.

Guy and Jones [1972] argue that to prevent excessive sediment loads to streams caused by the ongoing process of urbanization, erosion control is an important aspect of design. Proper controls can minimize this input. However, the origins of SS are not always due to erosion but may be related to accumulated particulates that vary with housing density and other factors.

Whipple et al. [1978] provided data illustrating the complexity of studying urban NPS runoff effects:

Date	Suspended Solids from a Modern Multiple Family Housing Area kg/storm
7/23/76	20.1
8/06/76	299.
9/10/76	189.
10/26/76	1.82

To obtain these data, the flow contaminant variable had to be sampled throughout the storm event; single samples did not suffice. Also, the landscaping and impervious surface provided excellent erosion control so the SS resulted from accumulated particles. The authors demonstrated that, in general, modern multiple family units provided more contaminants per unit than did single family housing.

Metals

Although Pb is the principal metal associated with urban runoff because of its use in automotive fuels, Zn and other metals are frequently found at deleterious levels. The association of Pb and Zn in sediment cores with the onset of urbanization signals the potential impacts of urban runoff on aquatic ecosystems [*Christensen et al.* 1978].

Bryan [1974] reported 0.33 kg/km^2 day of lead discharged from the 106.7 acre (432 ha) urban drainage basin in Durham, N.C. In one instance, dissolved lead concentrations were thought to be great enough to interfere with BOD measurements. However, subsequent observations revealed that the lead was associated with suspended solids and had no apparent effect on BOD.

Oliver et al. [1974] in studies of urban Ottawa, reported that amounts of chlorides in snow and runoff from snow melt were related to street deicing practices. However, lead came from leaded gasoline and was found to be tightly bound to fine particulate matter in the snow. Thus, if plowed snow were dumped away from waterways, the vast majority of lead from this source would be retained at the dump site with the particulate matter.

Newton et al. [1974] reported that lead deposited on city streets from automobile exhaust resulted in significant concentrations of lead in runoff. In the Oklahoma City area, the authors predicted an average lead concentration in runoff of 0.23 mg/ℓ. An adjusted average concentration of lead in snow and ice collected from a heavily traveled street in Oklahoma City was 3.2 mg/ℓ.

Wilber and Hunter [1975] reported on heavy metal contributions from urban runoff, and presented data from two drainage areas in Lodi, New Jersey. Seven individual storm hydrographs were analyzed and first flush effects were found to be highly significant. Pb, Zn, and Cu were the major heavy metals, accounting for 90-98 percent of all the metals observed (Pb, Zn, Cu, Ni, Cr). Pb:Zn ratios suggested the sources of these metals in runoff and precipitation may be similar. With regard to the relative contributions of metals in stormwater runoff, direct precipitation, and secondary treatment plant effluent, runoff accounted for as much as 86 percent of the total.

Also, *Wilber and Hunter* [1975] studied urban runoff of heavy metals discharged into the Saddle River near Lodi, New Jersey. Base flow metal concentrations were quite variable, probably due to industrial use variations in the sub-basins. Confirming their previous results, stormwater runoff showed a "first-flush" effect for heavy metals and the distribution in precipitation was similar to that in runoff, *i.e.*, Pb, Zn, were dominant. Bottom scouring of unconsolidated sediment was found to be important in metals' transport by rivers.

The chemical relationships of heavy metals in streams are being developed [*Vuceta and Morgan* 1978]. The effects of temperature, pH, and other ions on the concentrations of metallic ions in water explain the equilibrium concentrations available for biotic uptake or other reactions in stream ecosystems. *Davis and Leckie* [1978] illustrate how these metals could be accumulated in sediments.

Organic Pollution and Dissolved Oxygen

A major question about the significance of NPS pollution emerged with the application of stream quality models for analyzing the effects of point sources on stream dissolved oxygen (DO) resources. Earlier editions of these models showed that relatively good prediction of observed data was obtained by applying the models under well-defined stream conditions. However, results were poor when NPS pollution became important; this indicated that NPS pollutant impacts were significant to the DO resource. Many investigations then showed the impacts of NPS organics as measured by Biochemical Oxygen Demand (BOD) on the stream DO [*Corder* 1977, *Mischi and Dhasmadhikari* 1971, *Whipple et al.* 1974a, 1974b, 1976a].

The models, based on the *Streeter-Phelps* [1925] equations, have been improved and applied to practical situations by a variety of investigators so that the methodology is essentially a textbook procedure [*Nemerow* 1974, *Thomann* 1971]. There are specific processes peculiar to urban runoff, e.g., the sudden impulse of load that requires analysis [*Smith and Eilers* 1978]. A recent review of the water quality modeling technology illustrates some of the refinements in the approach [*Grenney et al.* 1976]. Future refinements include the aspects of randomness and experimental error by applying DO models that involve stochastic techniques [*Malone et al.* 1979].

Application of modeling and statistical approaches for analyzing urban NPS runoff has added insights to the material input phase of the process. *Rickert et al.* [1975] used a model of DO in the Willamette River, Oregon, to evaluate the effects of reducing carbonaceous and nitrogenous oxygen demand. They concluded that nonpoint sources contributed 46 percent of the total oxygen demand during the dry weather season of 1974. Thus, point source treatment alone would be an incomplete method for eliminating severe oxygen demand problems in the Willamette.

Often, however, the question of organic inputs to streams has not been separated from the problem of combined sewer overflow as is frequently observed in the eastern United States. *Gates* [1975] concluded that combined sewer overflows were important point sources of BOD_5, and that significant amounts of BOD_5 were contributed by nonpoint sources in the study area. Also, *Pennino and Perkins* [1978] reported that current overflows contributed a minimum of 0.66 million kg of BOD and 1.97 million kg of suspended solids to the James River (Virginia) each year. *Lindholm* [1976] studied pollution from combined sewers and used mathematical models to analyze the problems of BOD_5 discharge, retention basin design, setting of storm overflow, and the size of the secondary clarifier in the wastewáter treatment plant.

Organic loads include easily degraded materials (measured by BOD) and carbonaceous materials that are not as readily degraded (COD and TOC). For example, *Thompson et al.* [1974] showed that urban runoff in Lubbock, Texas, contained COD/BOD ratios of 9.5:1. Typical raw sewage ratios are 2-4:1 [*Eckenfelder and Ford* 1970]. This reflects the synthetic nature of much of the urban NPS runoff as contrasted to fecal materials in domestic wastewaters. *Mackenzie and Hunter* [1979] showed that aromatic compounds in Delaware River sediment were derived from slow-degrading crank-case oil, a component of urban runoff. *Wakeham* [1977] showed similar sources for Lake Washington (Washington).

However, most NPS organic material loading has been assessed as BOD. In non-urban areas (including single family housing), *Whipple et al.* [1974a] found that during rainy periods, organic (BOD) loading from unrecorded sources increased up to 10 times more than during a typical dry period. They also found that cropped areas contributed as much BOD loading in the New Jersey area they studied as did single family housing. Industrial and non-industrial urban areas contributed significant amounts of BOD. The weighted mean annual BOD loading for an industrial area was about 235 $kg/km^2/dy$ with storm runoff being the controlling influence. An urban residential area contributed an average of 64 $kg/km^2/dy$.

Yu et al. [1975] investigated organic pollution for seven small New Jersey watersheds, three in urban areas. They found substantial organic pollution (BOD_5) from an urban-industrial and an urban-residential area with light industry. This BOD load could be greater than or equivalent to the load from secondary sewage effluent. Loading from storm periods was more than 10 times the dry weather load. Agricultural areas could contribute as much or more organic load as urban-residential areas. Frequency distribution of BOD concentrations were plotted and were projected to be very useful in modeling efforts.

Rimer et al. [1978] applied land use planning concepts to stormwater runoff in the Piedmont Region of North Carolina (Triangle J Council of Governments, 208 Study). Generally, NPS pollution increased with increasing impervious area. Business districts did not follow the pattern because of minimal land disruption and the practice of street sweeping. Authors found that DO was an important water quality variable for stream management and that the other variables (BOD_5, COD, SS, TKN, NO_3-N, TP, TOC, heavy metals at selected sites) were less important. However, these variables were the input form of the toxicant that affected the stream DO.

Whipple et al. [1976] studied stream pollution from urban watersheds in New Jersey. They concluded that for residential areas BOD loads from unrecorded sources would range from 9-13 g/person/day. In heavily industrialized/commercialized areas, the loads would be greater. The unrecorded pollution sources must be considered if pollution control plants are to be cost effective.

Sullivan [1974] of the American Public Works Association denies *Whipple*'s assertion that the APWA report underestimated BOD due to possible runoff prior to street sweeping or catch basin cleaning. He also points out that the information cited by *Whipple et al.* [1974a] is specific to the Chicago area.

Although the sources of organics that contribute to DO depletion in streams are varied (pet and vegetable litter, fuels, oil and grease, combined sewers, garden sources), several unique sources that could on occasion have great and sudden impact on streams have been noted. *Thompson et al.* [1974] observed that discharges from fire fighting runoff were highly polluted with COD concentrations as high as 1740 mg/ℓ and suspended solids concentrations of 670 mg/ℓ. *Schultz and Comerton* [1974] studied the Montreal International Airport and estimated that projected increases in air traffic would increase the organic load in storm sewers from that airport by the year 1985-86, to 8100 kg BOD/day. The use of aircraft deicers (diluted ethylene or propylene glycol) provided most of this load.

Weibel et al. [1966] showed that organo-chlorine pesticides could attain significant levels in urban runoff from a residential-light commercial section of Cincinnati, Ohio.

Bacteria

Indicators of bacterial pathogens have been used to assess the potential danger to human health of urban NPS runoff. *Olivieri et al.* [1977] found high levels of recoverable pathogens and indicator organisms in urban runoff from Baltimore; *Pseudomonas aeruginosa* (10^3-10^5/100 mℓ), *Staphylococcus aureus* (10^0-10^3/100 mℓ); *Salmonella* and enteroviruses were found at levels of 10^0 to 10^4/10 liters of urban runoff. *Shigella*, although not recovered due to culture process interferences, may have been present. The authors recommended that because of relatively low densities of pathogens, and current regulations prohibiting use of water polluted with urban runoff, that it would not be cost effective to disinfect the large quantities of urban runoff. They recommended, however, that combined sewer overflows (10 times higher in pathogens) be disinfected to reduce the load for subsequent culinary water treatment plants and contact recreation use hazard.

Geldreich et al. [1968] compared the bacteriological character of stormwater runoff from city streets, a suburban business district, a storm sewer, and a wooded hillside adjacent to a city park in Cincinnati, Ohio. Highest seasonal mean values of fecal coliforms were measured in street gutters (47,000/100 mℓ) and runoff from a business district (40,000/100 mℓ) in the autumn. They concluded from fecal coliform/fecal streptococci ratio (FC/FS) data that fecal contamination was mostly from cats, dogs, and rodents (FC/FS≃0.7) rather than human (FC/FS≃4.4) sources. Fecal indicators and pathogens survived storage in stormwater longer at winter temperatures (10°C) than at summer temperatures (20°C).

Davis et al. [1977] observed a "first flush" effect in all hydrographs from rural and urban drainages in the Houston area. Fecal coliform and fecal streptococcus concentrations were higher in the more developed areas.

Nutrients

Nutrient problems in aquatic ecosystems are primarily directed toward lakes and impoundments and the problems associated with increased plant productivity as a result of eutrophication. Lake restoration by point source nutrient diversion is ineffective when NPS inputs are not controlled [*Emery et al.* 1973]. The effects of nutrients in streams have not been well documented; however, undoubtedly increased productivity would result from added nutrients [*Ball et al.* 1973]. Such nutrients can be supplied extensively by urban NPS runoff.

Loehr [1974] summarized the characteristics of nonpoint sources of pollution from various land uses; agriculture, forest lands, and urban runoff (Table 7). The relative contributions of urban runoff with municipal/industrial contributions to aquatic ecosystems for nutrients were compared; a range of 0-87 percent for nitrogen and 0-89 percent for phosphorus was obtained in wastewaters while urban runoff contributed 0-48 percent and 0-57 percent, respectively. Thus, urban runoff contributed significant quantities of nutrients in certain locales.

Weibel [1969] discussed the sources and types of urban drainage in the light of eutrophication potential. He pointed out that rainfall from the Cincinnati, Ohio area contained enough inorganic nitrogen (0.69 mg/ℓ) and total phosphorus (0.08 mg/ℓ) to support algal blooms. Urban runoff adds

TABLE 7. PRECIPITATION CHARACTERISTICS [FROM *LOEHR* 1974]

Constituent	Concentration under Given Conditions* (mg/ℓ)							
	1963-64 Urban	1963-64 Rural	Cooper	Northern Europe	1963-69 Forest	Feth	4 yr Ohio	Joyner
Nitrogen								
NH_4-N	-	-	-	0.06	0.16	0.17-1.5	1.1	-
NO_3-N	-	-	0.14	0.31	0.30	0.56	1.15	-
Inorganic N†	0.7	0.9	-	-	-	-	-	-
Total N	1.27	1.17	-	-	-	-	-	0.73
Phosphorus								
Total PO_4-P	-	-	-	-	0.008	-	0.02	0.04
Hydrolyzable PO_4-P	0.24	0.08	-	-	-	-	-	-
Suspended solids	13	11.7	-	-	-	-	-	-
COD	16	9	-	-	-	-	-	-
Major ions								
Ca	-	-	0.65	-	0.21	-	-	-
Cl	-	-	0.57	-	0.42	-	-	-
Na	-	-	0.56	-	0.12	-	-	-
K	-	-	0.11	-	0.19	-	-	-
Mg	-	-	0.14	-	0.16	-	-	-
SO_4	-	-	2.18	-	3.1	-	-	-
HCO_3					0			

* Data are primarily yearly averages.
† Inorganic N = NH_4, NO_2, and NO_3-N.

significant amounts of N and P to the rainwater, and combined sewer overflows may be very high in nutrients (3.0 mg/ℓ total P). Various methods of prevention and treatment are discussed. Most include storage with subsequent use of the water resource, or routing to secondary treatment plants.

Wenster, et al. [1975] showed that success for the eutrophication control program in Swedish lakes by point source phosphorus removal depended significantly on the relative contribution of urban NPS runoff nutrients.

SECTION V

ECOLOGICAL CONCEPTS

PURPOSE OF ECOLOGICAL ANALYSIS OF URBAN NPS RUNOFF

The analysis of the effects of urban NPS runoff on stream ecosystems is a special case of the general problem of the ecological analysis of changes in environment, communities, and specific organisms. *Patrick* [1949] and many others have argued that biotic and ecologic changes are the most accurate for assessing physical and chemical changes on ecosystem health because of their integration and direct response to the complex melange of chemical and physical variables. In this analysis we attempt to develop concepts that are holistic rather than directed at a specific variable such as DO or a critical organism. It is a macroscopic rather than microscopic view of ecological function and structure, thus the defined variable boundaries tend to be inexact and at times lump more than one process, trophic level, species, etc. Although definition of holistic properties of the macroscopic system that we define to be an aquatic ecosystem is a relatively new field of endeavor, it should be possible to define a set of variables useful for interpreting ecological changes in streams. We intend to provide a selected list of variables against which "before/after" or "upstream/downstream" effects can be observed. The defined target variables should aid in explaining and perhaps predicting community response to random, impulse-type, variable magnitude environmental changes in streams.

GENERAL ECOLOGICAL CONSIDERATIONS

"An ecosystem is a set of organisms and inanimate entities connected by exchanges in matter or energy" [*MacMahon et al.* 1978]. Biological relationships of organisms and larger organizational systems can be shown using a hierarchical approach (Figure 11). Perturbations of all types impact ecosystems at the level of growth and reproduction of individual organisms [*Woodwell* 1970]. Interactions in communities or the biosphere consist of either matter/energy impacts on organisms or organism impacts upon organisms viewed in terms of effects on growth and reproduction [*MacMahon et al.* 1978]. Growth and reproduction respond to concentration of energy or material and can be viewed as an uptake process or an enzymatic process [*Reynolds et al.* 1975, *Chen and Selleck* 1968, *Fitzgerald* 1969, *Sprague* 1971, *Eppley and Thomas* 1969, *Michaelis and Menten* 1913, *Monod* 1949]. This can be represented as follows using the symbols of *Monod* [1949] for growth controlling materials ($S_1,\ldots S_n$):

$$\mu_t = f(S_1, S_2, S_3,\ldots, S_n)$$

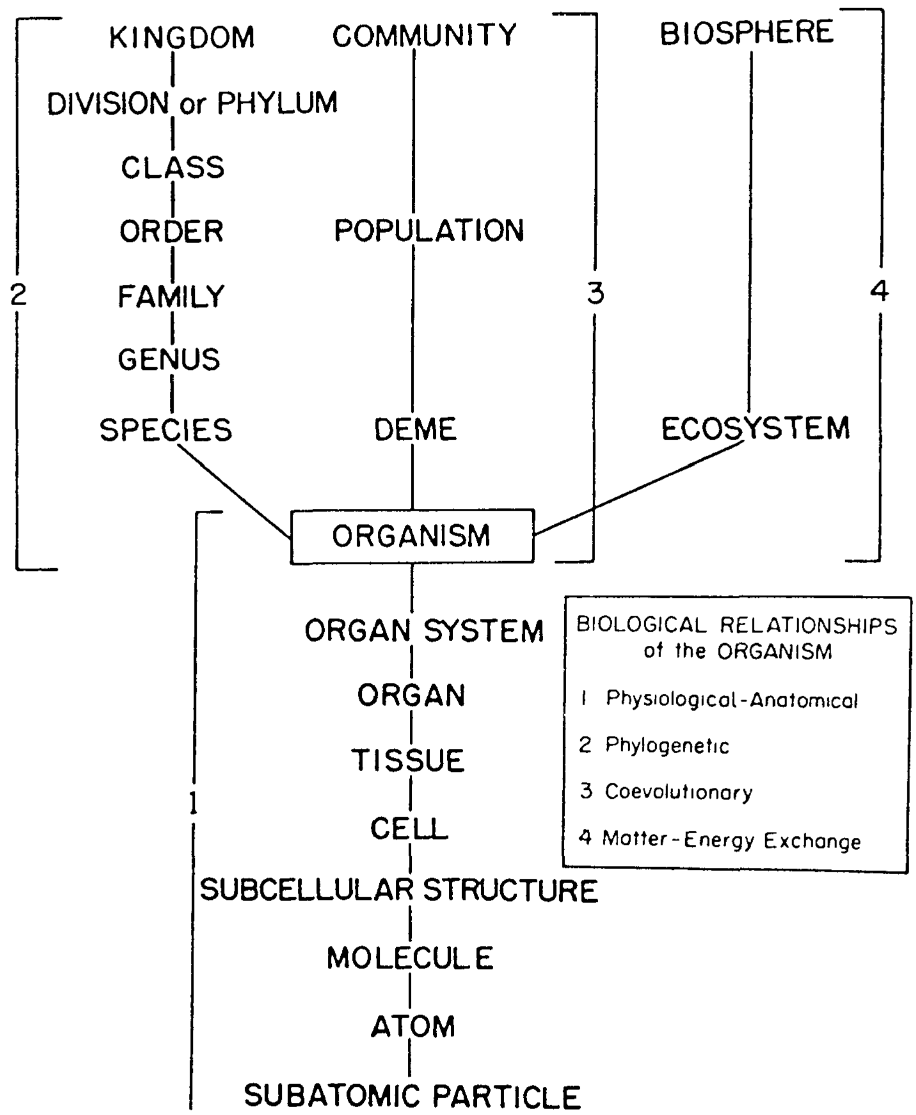

Figure 11. Biological relationships of the organism [from *MacMahon et al.* 1978].

Because communities are made up of a set of populations (all the organisms of a taxon) that in turn are composed of smaller assemblages (deme) and the individuals themselves (organisms), the growth-promoting effects (removal mechanisms would be treated similarly) of materials and energy inputs on the community can be viewed as the resultant (μ_t) of the specific effects on growth generalized to the level of a population (*e.g.*, *Patrick* 1971]:

$$\mu = g(\mu_1, \mu_2, \mu_3, \ldots, \mu_n)$$

The population is the minimal size capable of analysis. Community variables relating to effects on growth of populations are still largely hypothetical relationships. Diversity estimates, biomass, turnover, keystone species, relate to these hypotheses. Probably the most successful such demonstrated relationships for aquatic ecosystems is the summer chlorophyll *a* - springtime total phosphorus relationship in lakes as defined by Vollenweider in 1968 and many others since then.

Chemical and physical laws apply to ecological systems and are useful for evaluating structure and function. For example, in terms of the system role of energy processing, the laws of thermodynamics can be applied with reasonable accuracy at our present level of understanding of ecosystems and of irreversible thermodynamics. Although ecosystems act primarily as energy processing systems, rates of processing are controlled by essential elements and by temperature. Consequently, ecosystems could be very inefficient energy processors in order to maximize rate controlling nutrient concentrations by rapidly cycling nutrients [*Reichle et al.* 1975].

Energy processing and nutrient cycling can be considered fundamental attributes of ecosystems and community function and structure would be optimized to maximize these functions. As an ecological system, there should be a set of variables that allows an essentially complete description of the ecological state of the system. Several key macroscopic variables have been identified as exemplifying important properties of ecosystem structure and function (Table 8). Although the listed variables are quite general, they can be easily applied to specific situations with selected specific measurements. If these variables (V_T) are visualized in terms of growth, they can be related to growth regulating processes as follows:

$$V_T = h(\mu_T) = h(g(\mu_1, \mu_2, \ldots, \mu_n))$$

Unfortunately, very few of these variables have been applied in urban stream studies.

CATEGORIZATION OF ECOLOGICAL VARIABLES

There are several ways and many levels of categorizing the vast number of biological, ecological, and environmental processes and interactions and the variables that measure them. The physical or abiotic environment contains those chemical and physical factors that are affected by or affect organisms (biota). The community is a set of populations of organisms which function in the abiotic environment. There are spatial and temporal factors involved in the interactions between biotic components and the abiotic environment. Although both concepts are confined to a specific and defined space, function in a community is measured over a period of time while structure is measured at a specific time (instantaneous). There is some overlap between these concepts. For example, energy processing is a functional process that takes place in a time scale between structurally defined components. The ecosystem of interest in this report is the fresh water stream ecosystem with its fresh water biota and the abiotic components of the stream and of the stream substrate and watershed. The fresh water biota can be characterized as composed of taxonomic groups or community groups. Examples of taxonomic groups include the various classes of organisms (taxa). Community groups can be classed on the basis of habitat (such as, benthic, planktonic organisms), or function within the system (such

TABLE 8. VARIABLES IDENTIFIED WITH PROPERTIES THAT DESCRIBE THE STATE OR CONDITION OF ECOSYSTEMS

Properties	Variables* (measured per standard unit area)
I. Environmental	Volume, pressure, abiotic mass flow rates, system attributes (homeostasis-feedback, storage, loss)
II. Structure	
Biomass	Diversity, number, mass, chlorophyll *a*
Energy	Calories
Materials	C, N, P, ..., total mass
III. Function	
Biomass	Change in diversity, number, weight, competition, resilience, resistance
Energy	Energy transfer, efficiency, P/R ratios
Materials	Elemental turnover times, P/R ratios, limiting nutrient

* Can be applied at any or all levels: community, trophic, population, organism, dominance, keystone, guild, (functional groups) or habitat.

as groups that have similar energy/matter processing characteristics; for example, producers, consumers, decomposers; or autotrophs and heterotrophs [*Hendrix* 1979].

MacMahon et al. [1979] imply that habitat categorization is not a useful concept for evaluating ecosystem structure or function. In reality habitat is a controller; it defines the community limits and types of organisms but does not relate uniquely to community structure and function. For example, benthos refers to the organisms that inhabit the bottom of an aquatic ecosystem. The structure and function of the benthos is not dependent on where they reside but on their niche. Perhaps changes in habitat would result in different possible niches but community structural and functional relationships would be related to those niches not the habitat. Consequently, it is incorrect to sample a specific habitat to estimate community structure and function. The structural/functional component of interest should be analyzed and if the component occurs in a specific habitat, the habitat sampling would be valid. Benthic macroinvertebrates would be a valid sampling if the processes studied are totally contained therein.

Stalnaker [1979] and others in his group have developed a scenario for describing change in spawning habitat with changes in flow and stream morphometry. This scenario is seasonally affected by hydrology and life cycle. Given enough time, the resultant spawning frequency and fish production that result from change in habitat produce population distributions, community structure and function that are unique to that habitat and that set of niches that result from the changed habitat. If these arguments are valid, response variables should not be habitat related. For the remainder of this report only niche variables (community structure and function) will be considered as valid response variables.

Impacts of materials that enter fresh water ecosystems are of two types. There are direct impacts where direct linkages occur between the factor being studied and a target biota. Indirect impacts are those where the biotic response is due to secondary interference with some target variable. Examples of direct impacts would be direct toxicity to fish or stimulation of algal growth by nutrients. Indirect impact would include the effect on zooplankton populations resulting from removal of zooplankton predators and inhibition of primary productivity due to light transmission interference from suspended sediments.

Contaminants in freshwater ecosystems affect structure and function because of their differential effects on the species that compose the biota. Sensitive species might increase or decrease depending on type, concentration, and phenological relationships between species cycles and contaminant input. Relationships between species might be altered secondarily, with further repercussions among the community depending on the significance of the contaminant event and the affected species.

Organisms would respond to NPS urban runoff contaminants produced according to the scheme outlined previously in Figure 1; a hierarchy of variables could then be defined to allow analysis of their impacts in the stream. Ideally one would assess urban runoff impacts using an ecosystem model that would provide as output those critical response variables identified as controlling some perception of ecological quality. Although examples of ecosystem models have existed for some time [NSF biome models, *e.g.*, *Park* 1978, *Israelsen et al.* 1975, *Orlob* 1975, also see selected articles in *Middlebrooks et al.* 1973], they are probably not suitable for analyzing specific stream ecosystems because they were constructed with other objectives in mind. However, these models with suitable modifications may be useful for analyzing the general case of urban runoff in streams.

Models of streamflow, habitat, water quality variables, and many other chemical and physical attributes of aquatic systems have been used to evaluate the impacts of changes in stream inputs, thruputs, and outputs but so far these models have not been linked to biotic attributes. Quality variables have been related to standards, beneficial uses and organism requirements, but the direct linkage of abiotic and biotic variables has not been accomplished. Usually semi-quantitative comparisons of controlling variables or of "before/after impacts" on biotic response variables have been the limit of efforts in understanding stream ecosystems.

SELECTED ECOSYSTEM VARIABLES FOR STREAM IMPACT ANALYSIS

The management of stream quality depends on the analysis of the effects of water quality changes on stream communities. Historically, this analysis has been oriented toward perception of stream conditions and corresponding biological [*e.g., Thomas et al.* 1973], chemical, and physical indicators. The most noted example of using indicators is the Saprobien system of *Kolkwitz and Marsson* [1908, 1909, *Sladacek* 1973, for recent review]. The approach of linking problem perception to chemical/physical variables has achieved the greatest acceptability in relating productivity in lakes to the nutrients, N and P [*Sawyer* 1947, *Vollenweider* 1968, *Dillon and Rigler* 1975]. Generalized cause/effect relationships have not been developed between biota and chemical/physical variables in streams.

The hypothesis that concentrations of contaminants affect ecosystem response variables is testable and should result in functional relationships. The problems in testing that hypothesis include: (1) selecting appropriate response variables, and (2) defining an appropriate variable that relates to the concentration of the contaminant (discussed in Section VII). Ecosystems process matter and energy and the impact of urban NPS runoff contaminants (inputs of materials and chemical energy) on communities in stream environments is largely a question of the flux of matter and energy in the ecosystem (Table 8). *Bartsch and Ingram* [1966] reviewed and criticized attempts by pollution control agencies to assess communities in stream reaches by indicator organisms and other such environmental indicies. Similarly, the use of chemical/physical variables has been severely criticized. More recently, holistic ecosystem concepts have seemed to offer more value in evaluating biotic and community response to stream perturbations such as urban runoff. These include biomass, diversity, and energy and material turnover rates. In this section we select a set of response variables based on previous, successfully applied variables plus a minimally redundant set from Table 8 to represent matter and energy flow. In this way we expect to minimize the economic dilemma of accounting for every particle of the ecosystem while avoiding omissions caused by the broad brush of generalized variables.

Biomass

The biomass of a stream community is not as apparent as that of terrestrial communities; in fact, except for fishing we generally prefer stream communities to be relatively inconspicuous. However, for every ecosystem, *Reichle et al.* [1975] define a potential maximum biomass based on average, steady state, peak or other parameter that is primarily related to energy, elemental (nutrient) supply and temperature (Figure 12). Perturbation or random fluctuations in climate or nutrient supply can result in an apparent constant average value below that of the potential maximum biomass. *Reichle* and his associates define this level as the maximum persistent biomass. These ideas in different form were presented by *May* [1973].

We measure biomass of the various phases of the aquatic community depending on how the phases are conceptually organized: (1) trophic levels

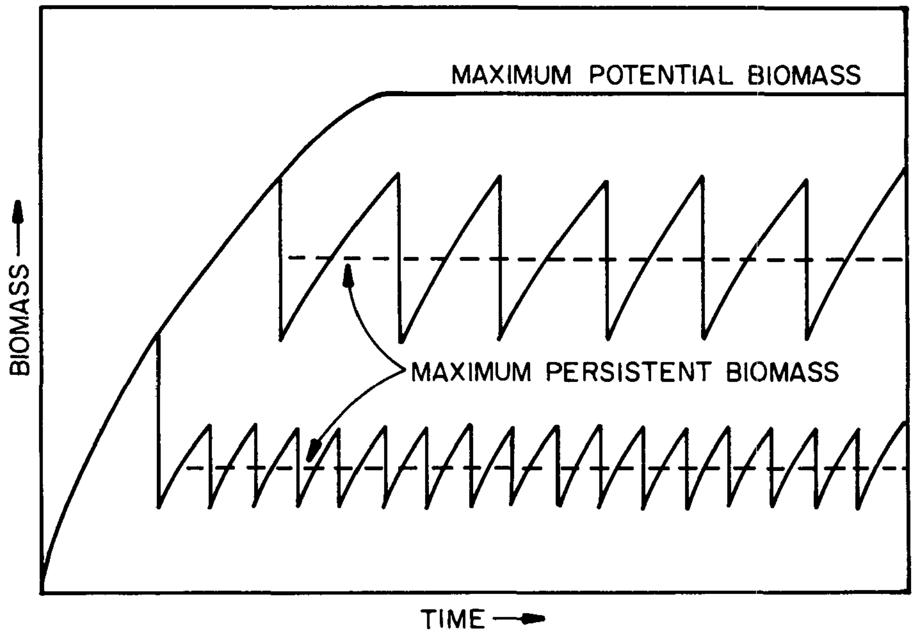

Figure 12. The effects of environmental fluctuations on steady state biomass in an ecosystem. Three levels of persistent biomass are hypothesized based on nonvariant, variant, and a highly variant environment wherein severe perturbations occur [from *Reichle et al.* 1975].

(producer, consumer, decomposer), (2) organism level (as in Figure 11), and (3) photosynthesis/respiration (autotroph/heterotroph). Note that indicator organisms are assessed on the basis of biomass or condition.

Methods of biomass estimation include energy content, dry weight, ash free dry weight, total or particulate organic carbon, chemical oxygen demand, chlorophyll *a*, numbers, volume, and various measurements related to the above. All these methods offer differing levels of accuracy (in relation to "true" ecosystem structure and function), precision, sensitivity and convenience.

Biomass itself has an interpretation; as shown in Figure 11, biomass is most meaningful in an ecosystem context when the individual organisms that make up the community are considered. Food chain and trophic level were among the first concepts applied in attempting to understand communities. It was soon realized that static (one-time) measures of biomass in specific categories were inadequate to understand communities. Phenological and successional biomass changes were important concepts that integrated time with biomass and structure. Concepts related to production and consumption (photosynthesis/respiration; autotrophy/heterotrophy), energy flow, and nutrient cycling similarly integrated time with biomass and function. Therefore, we look at biomass as a measure of community structure and function. Instantaneous biomass provides a statement about structure while change in biomass over an interval is a major aspect of function (productivity/consumption/decomposition/toxicity). Generally, we use biomass to estimate presence/absence, before/after, upstream/downstream, diversity, ..., but rarely in the sense of *Reichle et al.* [1975]. This latter concept depends on carrying capacity (potential maximum biomass) definitions and we do not yet know how to determine this value.

Indicator Organisms

Although *Patrick et al.* [1949] and *Margalef* [1951, 1975] have suggested that community indices would be of more value than single indicator organisms, there are still many stream studies oriented toward individual organisms. Taxonomic difficulties make indicator organisms less useful [*Cairns* 1975, *Bartsch and Ingram* 1966]. *Pielou* [1975] argues convincingly that the ecological value of a specific organism may be minimal or at best difficult to assess. Rare and endangered species are a legal exception (Endangered Species Act. PL 92-205). Broad measures of ecosystem function would probably be unaffected by the presence or absence of rare and endangered species; thus, procedures should consider these populations separately. Bioassays [*APHA* 1976] are one method of developing criteria for indicator organisms and for analyzing observed data [*Sprague* 1971]. Bioassays have not been adequately related to community variables. Physiological indicators have been used to assess stress due to pollution [*e.g., Lynch* 1975] but again these variables are difficult to relate to community variables.

Microbial Indicators--

The system suggested by Kolkwitz and Marsson is based on indicator organisms (both micro and macroorganisms) that define different water quality conditions in streams reflecting self-purification (assimilation capacity): polysaprobic (gross pollution), alpha-mesosaprobic, beta-mesosaprobic, oligosaprobic; also finer divisions are possible [*Sladacek* 1973].

Many microbes have been suggested as useful in the Saprobien Index. In studies of laboratory systems *Bick* [1971] argues that ciliated protozoans are useful for analyzing organic pollution. Their rapid response and worldwide distribution makes them suitable indicators; however, severe taxonomic difficulties limit their use.

Cherry et al. [1977] observed a significant increase (from 15-25 percent) in percent chromogenic bacteria in cultures of bacteria one year after chemical pollution was removed from a fast-flowing stream in Aiken, South Carolina. The stream (Tims Branch) had received 229,900 kg/yr of chemical pollutants including mineral acids, caustics, salts, and organics (terchloethylene, trichlorethylene, and methanol). The recovery level of chromogen (25 percent) was comparable to a nearby control stream (28 percent).

Curtis and Harrington [1971] found BOD and soluble organic carbon to be the most reliable guides to the "slime-promoting properties of a water." Other parameters measured were: suspended solids, PO_4-P, NH_3-N, Kjeldahl N, NO_2^-, NO_3^-, total carbohydrate.

Curtis and Curds [1971] surveyed sewage fungus from 178 sites in

England, Wales, and Scotland. *Sphaerotilus natans* and/or zoogleal bacteria were most often dominant. Cluster analysis showed certain relationships between organisms. Diversity indices showed that slimes with mixed dominant organisms also supported a greater variety of other organisms. *Ruthven and Cairns* [1973] studied effects of phenol, pH, and heavy metals on protozoans in laboratory systems and found very sensitive and specific response to the different metals relative to other organisms and individual protozoans.

Lowe [1974] compiled, standardized, and centralized ecological information concerning the requirements and pollution tolerance of fresh water diatoms. As had *Patrick* over the last 30 years, *Lowe* suggests that diatoms (Bacillariophyta) are particularly useful as indicator organisms of the condition of inland waters because they are relatively easy to identify to the species level, they are the most ubiquitous of aquatic organisms, and each diatom species is thought to occupy a different niche in the aquatic ecosystem with individual responses to chemical and physical parameters.

Three hundred common species and varieties of fresh water diatoms are tabulated with their environmental requirements and pollution tolerance. The tolerance ranges for species are given for pH, nutrients, salinity (holobion spectrum), organic (saprobien), current, general habitat, specific habitat, seasonal distribution, temperature spectrum, and geographical distribution and other information. The data were compiled from 48 references. Tables of ecological profiles make up 297 pages.

Dickman [1969] found that differences in periphyton colonization of agar coated slides -- controls and containing suspected toxicant -- reflected the nature of the toxicant being tested. While interface conditions were uncontrollable, this method was suggested as useful for screening potentially introduced pollutants.

Aquatic Flora--

Ray and White [1976] suggest that because of accumulation processes, vascular plants (*Potamogeton* and *Equisetum*) and the blue green alga, *Oscillatoria*, be used to assess heavy metal pollution in aquatic systems. The alga was found (with one exception) at the more polluted sampling sites where the vascular plants were absent. Some evidence was given indicating that the alga accumulated lead and thus was a good indicator for lead pollution. The authors concluded that analysis of plant tissues, when compared with control streams, is a reliable indicator of heavy metal pollution.

Investigations of ecosystem change in Swedish lakes due to acid precipitation provide insight to the subtler effects on aquatic communities [*Grahn* 1970]. As these lakes have become acid, a succession has taken place in the aquatic macrophyte community. Sphagnum, which needs free CO_2 (no HCO_3^-) for growth, is taking over littoral habitats formerly occupied by vascular plants (Isoetides, *Lobelia dortmanna*, and *Latorella uniflora*). In addition, much of the microbial decomposer population had been destroyed, and decaying plant matter was building up, further restricting higher plant habitat.

Aquatic Fauna--

Howmiller and Scott [1977] discussed several types of indices which are distinguished by the kind of information they summarized. They are critical of diversity indices based on community structure for not being sensitive to physiological characteristics or ecological affinities of the taxa in the communities. The authors also point out that indices based on the presence or abundance of one or more groups of organisms may be insensitive to environmental change unless species-level analysis is done. They suggest an environmental index which considers both composition and ecology of the community fauna as indicated by oligochaetes.

By monitoring sediment selectivity and viability of a Great Lake amphipod, *Pontoporeia affinis*, *Gannon and Beeton* [1971] were able to evaluate the potential impact of disposal of dredged harbor sediments. The methods are simple and inexpensive and can be adapted to other water pollution problems and benthic organisms.

Gaufin [1973] conducted 96 hr TLM's on 20 species of aquatic insects and one amphipod to determine the effects of low DO, elevated temperatures, and low pH. Longer term bioassays were also conducted which indicated increased sensitivity with increased exposure.

Beck [1977] compiled and standardized data on the environmental requirements and pollution tolerance of 230 taxa of freshwater chironomids in order to help in the evaluation of data from aquatic macroinvertebrate collections taken for the assessment of water quality. Chironomids were selected because they can be identified to species level relatively easily [*Beck* 1977]. However, larvae are the most common benthic stage and are not easy to identify. They are among the most ubiquitous aquatic invertebrates, and each taxon is thought to occupy a different niche in the aquatic community. Water quality spectral ranges for pH, salinity, nutrients (trophic status), degradable dissolved organics, oxygen, temperature, turbidity, current, general habitat, specific habitat, seasonal distributions (emergence), feeding behavior, and geographic distribution are summarized for each taxon according to developmental stage. The 28 years of work by the authors provides most of the information.

Rosenberg and Wiens [1976] used artificial substrates to assess the significance of oil pollution on chironomids in Trail River in Northern Canada. *Wilson and McGill* [1977] used the method of collecting pupal skins shed by members of the chironomid fauna which lodge and collect along stream banks and vegetation to evaluate the water quality of an English stream receiving treated sewage effluent. The stream being investigated could not have been sampled by other conventional methods such as kicknetting. Despite the draw-back of only using one taxonomic group, the authors were able to show a definite relationship of species composition to water quality. *Chironomas riparius* was the most responsive, being most dominant near the sewage outfall, and less abundant in cleaner water.

Pearson and Rosenberg [1976] studied the effects of organic matter accumulation (cellulose) on the macrobenthos of fjordic systems in Scotland and Sweden. They considered that more generalized species occurred in the benthic community as depth of aerobic sediment decreased. The process of deoxygenation of sediments is initially accompanied by an increase in biomass and productivity followed by an afaunal condition. No species alone served as a good indicator of community condition at any state of the succession. Diversity indices (mathematical or graphical) were advantageous, within their limits, for comparing faunas at different stages in the succession. The use of indices must be qualified by presentation of specific data.

Crowther and Hynes [1977] studied the effect of road deicing salt on the drift of aquatic macroinvertebrates in Laurel Creek which flows through urban Waterloo, Ontario. This stream had maximum concentrations of Cl (1770 mg/ℓ), Na (9550 mg/ℓ), and Ca (4890 mg/ℓ) during the winter of the study period (1975). Pulses of salt concentrations up to 800 mg/ℓ as Cl had no effect on *Hydropsyche betteni*, *Cheumatopsyche analis*, and *Gammarus Pseudolimnaeus* under laboratory trials. Under field conditions, all organisms drifted when concentrations of artificially added salt exceeded 1000 mg/ℓ as Cl.

Anderson et al. [1978] used the fingernail clam (*Musculium transversum*) as an indicator of water quality in the Mississippi and Illinois Rivers. By using the ciliary beating rate of the gills, they found this organism very sensitive to heavy metals (6×10^{-4} mg Zn/ℓ) and unionized ammonia. Investigations of the effects of light, temperature, DO, sodium nitrate, sodium sulfate, cyanide, Pb, Cu, Zn, suspensions of silica and illite clay, and raw Illinois River water were made and they concluded that heavy metals and unionized ammonia played major roles in removing the fingernail clam from the rivers studied.

A review of the literature dealing with the use of indicator organisms to monitor trace metal pollution has been prepared by *Phillips* [1977] (189 references). The author concludes from his review that despite the increasing amount of information about metals in organisms, more attention must be paid to the condition (weight, reproductive state, etc.) and environment (salinity, temperature, etc.) wherein the analyzed organism dwells. Ignoring these variables can lead to erroneous interpretation of data. Bivalve molluscs and macroalgae were the best studied organisms used as metal pollution indicators. It is recommended that both of these types of organisms be used to give a more complete picture of the trace metal load.

Diversity

In considering community structure and function, steady-state and requisite community stability were defined. With correction to a seasonal time frame, steady state exists only for a climax community. A climax community is stable by definition. Stability exists under relatively constant environmental conditions. Although several properties of ecosystems have been defined as representing stability, diversity and complexity of niches in the community have not been accepted as measuring stability [*Orians* 1975]. *May* [1973] argued that complexity does not confer stability; stability arises from coevolutionary development in ecosystems. *MacMahon et al.* [1978] derive similar arguments. Stability increases with organization of the community. Thus measures of organization (negentropy) are measures of community stability.

Although acceptable measures of community stability are difficult to define [*Hulbert* 1971], the concepts of community diversity have utility in measuring stream ecosystem responses to perturbations. *Slobodkin and Sanders* [1969] assume that there is a connection between species diversity and the physical properties (abiotic) of the environment. They define a high diversity category plus three categories of low diversity (species poor) environments: (1) new environments (succession occurring); (2) severe environments (predictable but with extreme environmental conditions such as arctic seas, high salinity lakes, and hot springs. These possess low abiotic diversity); (3) unpredictable environments (environmental conditions vary extremely and frequently). The authors advance arguments that environmental predictability is a more important factor in controlling diversity than other variables. Thus diversity, by whatever measure [*Pielou* 1975, *Kaesler et al.* 1978], would be an adequate variable to assess impacts on stream communities regardless of its relation to stability.

Slobodkin and Sanders [1969] discussed the contribution of environmental predictability to species diversity. Many of their conclusions spell out in ecological terms why reaches of streams impacted by urban runoff tend to be species poor. They point out that the combined effects of environmental severity and unpredictability are such that a severe and unpredictable environment, such as one might expect in a stream subject to the hydrology and pollution from urban runoff, tended to be poorer in species than either a severe or an unpredictable environment. The authors also concluded that generally different life history stages or organisms show different sensitivity to environmental changes in "unpredictable" environments. The authors felt that competitive interactions are different in predictable and unpredictable environments in such a way that fewer species persist in unpredictable environments. Areas of low environmental predictability would be less likely to be invaded by species from high predictability areas.

Concepts of stability may be extremely dependent on "keystone" species [*Paine* 1969] and/or "foundation" species [*Dayton* 1972]. Keystone refers to a single species that controls community function and structure by its presence. Foundation species is the set of critical species that define the community.

Other concepts related to stability refer to the ability of the community to resist environmental changes (resistance) and the phenomenological inverse of resistance, the ability of a community to return to a stable or specific composition when stress (or change) is relieved (resilience).

Stability, resistance, and resilience are properties of systems that have been used to analyze the effects of environmental changes on aquatic ecosystems. A variety of diversity estimates have been used to estimate the role of these concepts; diversity (the number of niches (species?; probably deme) and individuals per niche in a community), niche richness (number of niches), and evenness (distribution of individuals among the niches) have been the most frequently cited. In streams most of these estimates have been applied to benthic invertebrates.

Several methods of calculating diversity have been devised. *Margalef* [1965] describes several approximating indices based on varying levels of resolution in identifying and counting organisms; also he estimated diversity based on visible light absorption ratios.

Because of the lack of problem definition, recent discussions of diversity and redundancy indices have not clarified the issue of whether diversity measures stability [*Hamilton* 1975, *Zand* 1976, *Logan* 1977]. The question of which is the correct method for estimating community stability is probably unanswerable at this time.

The question of which is the appropriate calculation for determining information content in species counts of samples results from mathematical, sampling, and interpretive differences. According to *Pielou* [1975], the Shannon (Shannon-Wiener) index (H') is used for large communities where sampling is used and the sampling does not reduce the diversity [*Shannon and Weaver* 1949]. It is only an estimate. The Brillouin index (H) is used for collections (exact census of entire community).

Because stream ecosystems are usually sampled, the Shannon index is usually appropriate and the following approximation where all taxa are assumed [*Simberloff* 1978] known is used:

$$H' = - \sum_{i=1}^{i=t} \frac{ni}{N} \log_e \frac{ni}{N}$$

where n = number in taxon i, N = total number, and t = number of taxa. Because of patchiness and similar clustering problems, sampling provides significant error; the larger the sample, the closer H'' approaches the "true diversity" (H'). Hierarchical diversity (H') can be determined depending on whether species, genera, ..., or other taxa are used [*Pielou* 1975, *Kaesler et al.* 1978].

Previously, redundancy was used in an incorrect manner and consequently had little interpretive value for analyzing stream impacts [see *Zand* 1976]. Many investigators have used overlap, percentage similarity, and other formulations that relate to the amount of common information (similar species and/or numbers in each species) contained in different sampling locations (spatial) or at different times (temporal).

Haedrich [1975] tested the hypothesis that the Shannon species diversity and overlap (percentage similarity, PS) can be used together to assess environmental quality. Samples of fish populations in Massachusetts estuaries and embayments had Shannon diversity values ($\log_e$) that ranged annually from 0.4 to 2.4 in close correlation with pollutional stress ($\log_{10}$ population, r = 0.84). Little seasonal change in population composition as indicated by high seasonal PS was found in areas of low annual diversity, while areas of high diversity had lower seasonal PS.

Brock [1977] compared a percentage community similarity (PCS) index and an index based on actual abundance of taxa in a study of zooplankton in Lake

LBJ, a reservoir in central Texas. He concluded that indices such as those studied facilitate comparisons of control areas and those receiving effluents. He also concluded that an index based on actual abundance of taxa was too sensitive to rare species, thus leading to high sensitivity to sampling error. A better result was obtained using a PCS index indicating structural functional differences between communities. *Brock* recommended using several indices that have different areas of sensitivity.

Although *Hocutt* [1975] argues otherwise, evenness, the ratio of observed diversity to maximum diversity for the observed number of species, appears to be a valid concept and mathematical formulation. Evenness should be determined according to the appropriate index, Shannon's or Brillouin's. For Shannon's, assuming that the total number of species (S) are known, evenness (E') is determined as follows [*Pielou* 1975]:

$$E' = H'/\log_e S$$

Pratt and Coler [1976] assume they obtain a complete collection by using artificial substrates (rubble in baskets) and consequently they suggest calculation of Brillouin's diversity and evenness values to describe effects of urban runoff on macroinvertebrates. No data were presented.

Previously, *Wilhm* [1970], *Wilhm and Dorris* [1968], *Reger* (1973], *Gislason* [1971], and many others have used Shannon's diversity to describe the effects of stream pollution. Their results indicate that diversity reflects pollution effects on stream communities.

Stauffer et al. [1974] found decreased diversity indices (Shannon) for fish communities in areas of thermal discharge where temperature intervals exceeded 80-87°F (26.7 - 30.6°C). Also, fish collected from thermally enriched areas were less "healthy" than fish from colder waters.

Stoneburner et al. [1976] compared the diversity estimates derived from the Shannon-Weiner equation and the sequential comparison index (SCI) developed by *Cairns et al.* [1968] and *Cairns and Dickson* [1971]. They concluded through statistical comparison of the two indices that the diversity estimates in terms H' and SCI are similar in their ability to predict the effects of wastewater on aquatic community structure. Also, they concluded that diatoms and macroinvertebrates show corresponding trends of sensitivity to waste loading.

Ghetti and Bonazzi [1977] compared indices of diversity (H'), species richness (d), and evenness (e) with each other and with the Saprobien and several biotic indices for the macroinvertebrate community, developing equations for the conversion of one index to another. Diversity was the most scattered variable. Correlation of variables allowed definition of four classes of water (non-polluted, weakly polluted, polluted, and strongly polluted). Also, their paper reflects the geographic variation in indices used to assess polluted streams in Europe and the USA.

Kaesler et al. [1978] redefine sampling so that the sample is a "universe," *i.e.*, a completely censused community (collection). Therefore, they use Brillouin's diversity estimate (*H*). The advantages are that it is not biased and that it is more sensitive to low diversities. Moreover, they followed *Pielou*'s [1975] argument and showed that no one actually uses the Shannon diversity index (*H'*) because the true species count (taxon or niche) is unknown. In most cases a comparison of events is desired and as long as adequate attention to statistical matters is maintained, the choice of indices is immaterial. *Pielou* [1975] indicates that the use of *H* as an estimator of a large community that can only be sampled would be invalid. *Kaesler et al.* [1978] develop hierarchical diversities of trophic and functional morphological classifications that seem to hold promise.

Other methods of reducing large quantities of species and count data are being developed as ecological tools. *Boesch* [1977] reviewed methods of numerical classification of ecological data. Resemblance measures and clustering methods are considered. Guidelines for interpreting numerical classifications and for applying ecological classifications are included. The author encourages the use of numerical classification techniques, and feels that they can enhance interpretation of the classification data and improve ecological insight.

Smith and Greene [1976] describe numerical methods for describing marine benthic invertebrate communities affected by submarine outfalls. Although they suggest that at our present state of knowledge the methods are principally useful for generating hypotheses, they do conclude that H_2S concentration in sediment porewater has major impact on community composition.

Kaesler et al. [1978] argue that species diversity is unnecessary; generic level provides adequate information. *Moeller et al.* [In press] are using a stream classification system for assessing "community quality" (diversity?) that relates to macroinvertebrate functional groups as defined by the stream continuum theory [*Cummins et al.* 1973]. *Heck* [1976] argues that only detailed community analysis will allow the interpretation of impacts on aquatic communities. However, the complexity of the system he studied plus its incomplete physical description may be the reason for the lack of demonstrated value of other community oriented measures.

Based on the foregoing results, measurement of diversity as a means of describing impacts of contaminants on stream communities is a valid method. Generally, diversity should be interpreted as an operationally useful measurement and no conclusions on its ecological significance should be made. Analysis of benthic invertebrate communities are preferred but other groups (fish, periphyton) can provide useful data. Generic diversity would generally be adequate and measurements should be comparative: upstream/downstream, before/after. Calculation of related indices can provide additional information.

FUNCTIONAL GROUPS AND MATERIALS CYCLING

This section deals with those macroscopic properties that reflect the fundamental processes of ecosystems: (1) groups of organisms that manipulate and transform significant amounts ($\geq$ 10 percent) of energy and materials; and (2) the turnover of those materials. Perturbations of aquatic ecosystems should be massive enough to affect these properties. The perturbation does not have to be massive enough to cause a permanent change but should provide a measurable change that can be statistically demonstrated. Moreover, the change should last for a measurable period.

We have discussed indicator organisms and diversity, both of which are used frequently and with some success. Although valuable concepts, the disadvantages and complexities of taxonomy and definition make them less useful. Also the level of response is not quantitatively related to impact, nor, at the present state of knowledge, is it possible to develop quantitative relationships between impacts and the properties of functional groups and material turnover. However, it seems more feasible to develop such quantitative relationships for these properties than previously were developed for indicators and for diversity estimates [*e.g., Maguire* 1971].

The concept of functional groups began to evolve in earnest with *Odum*'s [1969, 1971, *Holling* 1973] continued development of the trophic levels concept (producer, consumer, decomposer) and energy flow through trophic levels. At the present time we use functional groups in the development of concepts about material processing. The foremost example of this approach for freshwater stream ecosystems is the stream continuum theory which describes the biogeochemistry of organic matter [*Minshall* 1967, 1968, *Cummins et al.* 1973, *Boling et al.* 1975]. The role of functional groups of invertebrates (chiefly aquatic insects) in processing organic matter depends on the sources of energy and materials (natural or cultural allochthonous organic matter; autochthonous organic matter), and the status of the environment itself (tree canopy, riparian vegetation, geology, etc.). Generally, changes in P/R ratios and types of functional groups occur in relation to each other and in relation to inputs of material and energy. Natural undisturbed communities in a given environment might have $P/R < 1.0$ while after disturbance, the same environment might shift to $P/R > 1.0$. The consequences would be to increase the proportion of specific functional groups (grazers) relative to the natural condition (shredders, ...).

Further work by *Moeller et al.* [In press] traces dissolved organic carbon (DOC) transport through streams of differing morphology. Although they experienced difficulty in relating dissolved organic carbon to particulate organic carbon (POC), they were able to explain essentially all of the DOC variation by discharge (flowrate), watershed area, and link magnitude (a measure of stream order). Low DOC watershed output came from watersheds relatively undisturbed by human activities while higher values often were associated with culturally influenced larger streams. However, the relationship of DOC (or POC) to stream impacts of urban NPS runoff is not definable at this time.

Richey et al. [1978] argue that one means of assimilating these specific processes into a simpler broad scope hypothesis is to consider carbon flow. By determining the total carbon flow in a lake ecosystem (C_T) and the portion of the flow that cycles (C_C), they define a cycling index (C_I) that could be related to lake condition ($C_I = C_C/C_T$) with further information and analyses. The method of determining turnover time depends on estimating biomass in elemental equivalents and then estimating the elemental flow through the biota using isotopic tracers or appropriate chemical analysis. The overall concept is related to that of *Reichle et al.* [1975] who summarize turnover times for terrestrial systems (initial mass/change in mass per time) for C, N, and Ca (Table 9) and energy dynamics (which they term ecosystem metabolism) (Table 10). *Stumm and Baccini* [1978] show that environmental disruption in lakes results in less energy efficiency, more rapid biogeochemical cycling of elements, increased productivity, and loss of diversity. All of these methods of analysis tend to reduce ecosystems to simpler terms, masking those properties that make them unique from each other. This is a necessary step in developing an encompassing theory of ecosystem operation. For the purposes of assessing impacts such as urban NPS runoff impacts on aquatic communities, this approach seems to be the most valid. Unfortunately, there are not data that allow assessment of this hypothesis.

Other approaches have included the use of microecosystems (and of models as previously discussed in this chapter). Microecosystems permit the study of communities under defined and controlled conditions wherein material turnover and energy flow can be assessed relative to specific perturbations [*Porcella et al.* 1975, *Mitchell and Buzzell* 1971, *Cooke* 1971, *Witt et al.* 1979, *Geisy* 1978]. It is possible that these systems present the best approach for identifying those processes and hypotheses that would be testable under field conditions where random input, varying magnitude impulse loadings cause changes in stream community function and structure.

Cornell et al. [1976] have developed some concepts of change in diversity, biomass, and evenness over time that seem to show promise as a sensitive analytical means of assessing community structural response. The problem with applying diversity or biomass directly to assessing environmental problems is that those indices have not been related to changes in time except comparatively. *Cornell et al.* argue that their concepts integrate temporal patterns.

Summary

A minimal set of response variables that relate to assessing impacts of contaminants on aquatic communities would include habitat description, generic diversity of aquatic macroinvertebrates, an assessment of carbon turnover rates and a minimal set of water quality variables that relate to the major contaminant inputs (DO, nutrients, metals, suspended solids, salts). Carbon is the major ecological element and stream macroinvertebrates process the major quantities of materials and energy. The turnover of materials such as carbon would likely be affected by variations in water quality caused by contaminant input. Some inventory studies should be made to assess populations of rare and endangered or other significant species where appropriate.

Sampling in time and space should be adequate for comparative purposes (before/after; upstream/downstream). If an understanding of ecosystems were adequate to predict potential energy and material flow in communities, such comparisons would be unnecessary.

TABLE 9. MATERIAL TURNOVER TIMES IN TERRESTRIAL ECOSYSTEMS [TAKEN FROM *REICHLE ET AL.* 1975][1]

Temperate Deciduous Forest Component	Turnover time, years		
	C	N	Ca
Soil	107	109	32
Forest Biomass	155	88	8
Litter	1.12	< 5	< 5
Total	54	1815	445
Decomposers	0.01	0.02	?

[1] Comparison based on assessment of many variables; see *Reichle et al.* [1975] for explanation and footnotes.

TABLE 10. VARIATION IN TERRESTRIAL ECOSYSTEM VARIABLES [TAKEN FROM *REICHLE ET AL.* 1975][2]

Variable	g Carbon/m^2 yr			
	Mesic Forest	Xeric Forest	Prairie	Tundra
Gross primary production (GPP)	1620	1320	635	240
Autotrophic respiration (RA)	940	680	215	120
Net primary production (NPP)	680	600	420	120
Heterotrophic respiration (RH)	520	370	271	108
Net ecosystem production (NEP)	160	280	149	12
Ecosystem respiration (RE)	1470	1050	486	228
Production efficiency (RA/GPP)	0.58	0.52	0.34	0.50
Effective production (NPP/GPP)	0.42	0.45	0.66	0.50
Maintenance efficiency (RA/NPP)	1.38	1.13	0.51	1.00
Respiration allocation (RH/RA)	0.55	0.54	1.26	0.90
Ecosystem productivity (NEP/GPP)	0.10	0.20	0.23	0.05

[2] Comparison based on assessment of many variables; see *Reichle et al.* [1975] for explanation and footnotes.

SECTION VI

CASE HISTORIES OF RANDOM EVENT AND MAGNITUDE IMPULSE IMPACTS ON STREAMS

The selection of case studies was not intended to be exhaustive nor restricted to urban runoff events. Generally, the approach used was to look for stream impact analyses and use those which were illustrative of the principles outlined previously. Major reasons for this approach were the lack of studies related to urban runoff impacts on, and the very few related to, impulse inputs to stream ecosystems.

FLOODING EFFECTS ON FISH COMMUNITIES

Harrell [1978] analyzed the diversity effects of flooding on the Devil's River in Texas. A fortuitous flooding event (ninth largest flood on record) occurred midway during *Harrell*'s study to assess the fish community of the river. Marked changes in habitat occurred as a result of the flooding; distinctness of spring, riffle, pool, channel, and intermediate habitats generally were blurred and riffle-like habitat-types were more frequent. Fish community diversity declined. *Harrell* suggests that the ecological plasticity of the communities allowed shifts in the roles and associations of fish species and thus allowed the community to be maintained. In this respect, the community is stable; naturally stressful environments [*Slobodkin and Sanders* 1969, *Orians* 1975] would require dominance and maintenance of a few species.

EFFECTS OF TOXICANT SPILLS

Although many oil spill impact studies on marine ecosystems have been done [*e.g., Foster and Holmes* 1977, *Nelson-Smith* 1977, *Dicks* 1977], very few similar types of studies have been done in freshwater systems. However, spill impacts are random impulse type events and therefore are analogous to urban NPS runoff impacts on streams, although they occur less frequently.

Cairns and Dickson [1977] describe two stream systems that were impacted in this manner, the Clinch River and the Roanoke River. On June 10, 1967, caustic wastewaters (pH = 12) from a fly ash pond associated with a power plant adjacent to the Clinch River were released through a collapsed dike and caused extensive damage to the aquatic community. The inflow was 40 percent of the normal Clinch River flow at that time [*Cairns et al.* 1970]. In 1971 an acid spill occurred. Meanwhile during this period, day-to-day operations of the power plant and periodic flooding had short term effects on the aquatic biota. The authors and co-workers assessed benthic macroinvertebrates at 21 ecologically similar stations during the years 1969, 1970, and 1971 and analyzed that data based on numbers, biomass, diversity, and presence/absence (cluster analysis).

Approximately 200,000 fish were killed by the spill; however, most of the analysis centered on benthic invertebrates because invertebrates:

1. are relatively sessile and cannot avoid stress;
2. have long and complex life cycles;
3. are important members of the food web and affect related organisms;
4. have sampling techniques that are more reliable than for other organisms;
5. have more biological information obtained per dollar invested.

The initial measurements (June 20, 1967) were made by State of Virginia employees and *Cairns* and associates analyzed these data to show that the number of taxa and organisms per unit area were severely impacted by the spill:

Station No.	Description	RM	Taxa	Total Benthos/ft^2
1	Upstream (reference)	+21.0	24	520
2	Upstream but backwater effects	+ 0.5	9	30
3	Downstream	- 0.3	0	0
4	Downstream	3.5	5	270
5	Downstream	9.5	11	275
6	Downstream	12.5	15	650
7	Downstream	17.2	8	640
8	Downstream	30.5	11	175
9	Downstream	40.2	22	750
10	Downstream	56.5	18	80
11	Downstream	65.5	22	260
12	Downstream	77.5	32	1070

The authors noted that the benthic community was 8 percent midges at Station 1 but increased to more than 60 percent at damaged stations, illustrating their tolerance to stress. By Station 10, the midges were less than 30 percent. The pattern at these lower stations was that of lesser impact (due to dilution and physico-chemical reactions).

Recovery was not rapid and still was an ongoing process for the sampling collected in July-December, 1969, for the same and additional stations:

Station No.	Description	RM	Taxa	Total Benthos/ft^2	Diversity (H'', log 2)
1-4	Above the spill (=4) range	+ 1.5 to +26.5	48-54	24.6-145.2	2.97-4.03
7		- 0.2	43	47.0	2.92-4.21
8		- 0.9	33	2.4	3.64-4.35
9		- 3.9	36	4.0	2.88-4.01
10		- 9.5	42	8.0	3.19-4.06
11		-18.5	46	18.8	3.61-4.17

The biotic density was considerably depressed but diversity had reached control levels. Unfortunately no data on diversity were presented for the 1967 data so it was not possible to analyze diversity as a function of time according to *Cornell et al.* [1976]. The longer life cycle and non-motile organisms were those that had not returned; thus, recovery is a function of recolonization time which is controlled by life cycle and motility.

STREAM COMPARISONS - MICHIGAN

Ball et al. [1973] analyzed three streams in Michigan that illustrate the impacts of pollutants on stream communities: (1) the Jordan River was least developed with the downstream site differing from the upstream site because of a fish hatchery effluent; (2) the AuSable River was affected chiefly by recreational use but also by a previously thriving lumber industry and forest fires; (3) the Red Cedar River was in a more urban/industrial region and, although having received waste effluents directly in past years, urban runoff and some industrial inputs were more common. Generally upstream stations were better quality than downstream.

The authors and their colleagues had performed a variety of measurements that they related to the rivers as affected by material inputs. Although they concluded that the rivers were geochemically similar, the slope of the Red Cedar River was not as steep as the other two. Generally, total organic carbon, non-carbonate hardness and total phosphorus were greater at the lower station of the Red Cedar River than the other stations. Suspended matter was considerably greater for both stations on the Red Cedar River than for the other rivers. Generally toxins (metals, pesticides, and PCB), inorganic nitrogen compounds and chloride were greater on the Red Cedar. Although data on macrophytes and fish were discussed, those data were not directly comparable to other measures because species and their biomass were not always present at all the stations.

The most important variables of comparison are listed in Tables 11 and 12. The AuSable River seemed most productive based on diurnal DO fluctuations although the gross productivity of the Red Cedar River was greater. A comparison of average P/R ratios indicated that the better quality rivers (Jordan and AuSable) had fairly typical values for clean midwestern rivers while the Red Cedar River was the most variable and had values outside of the normal range of P/R ratios. This may have been associated with the considerable input of organic matter in the lower Red Cedar, possibly from urban runoff.

The effects on the community structure were considerable as measured by diversity. Although diatom diversity was greater for the Red Cedar River than the AuSable, and for invertebrates the AuSable was greater than those of both the other rivers. Also, the error associated with the estimated diversity for the AuSable and the Red Cedar Rivers was greater than the Jordan. This observation relates closely to the theoretical development of $\Delta H'$ as proposed by *Cornell et al.* [1976]. Based on these data (Tables 11 and 12), one would conclude that any of the variables, DO, diversity (periphyton, invertebrates), and productivity would be of value in assessing urban NPS runoff impacts in freshwater streams.

TABLE 11. COMPARISON OF STREAM QUALITY VARIABLES FOR DIFFERENT RIVERS IN MICHIGAN [*BALL ET AL.* 1973]

Variables	Upper Jordan	Lower Jordan	Upper Au Sable	Lower Au Sable	Upper Red Cedar	Lower Red Cedar
DO difference, mg/ℓ (maximum observed)	2.5	2.5	4.5	9.0	5.5	5.2
DO maximum, mg/ℓ	9.5	13.	12.	13.	10.5	9.0
Mean Flow, m^3/sec	1.8	5.5	1.7	2.2	1.9	5.4
CV, percent	13	16	27	27	53	110
Diatoms (artificial substrates)						
Maximum Observed Production, mg C/cm^2·day	120	230	70	80	100	440
Genus Diversity (H", log 2)	2.10	2.15	1.78	1.35	1.71	1.92
Invertebrates						
Benthic Species Diversity	4.33	3.37	-	2.76	-	1.70
CV, percent	6	18		31		27
Drift, gC/m (width)·day	8	7	17	12	10	9
Average P/R ratios	0.9	0.8	0.9	0.9	1.3	0.6

TABLE 12. AN ENERGY BUDGET FOR THE THREE RIVERS [*BALL ET AL.* 1973]

Energy Source and Units	Upper Jordan	Lower Jordan	Upper Au Sable	Lower Au Sable	Upper Red Cedar	Lower Red Cedar
Periphyton Production ($g\ C/m^2/d$)	0.03	0.08	0.03	0.06	0.06	0.08
Macrophyte Production ($g\ C/m^2/d$)	0.03	0.01	-	0.29	-	0.01
Sum of Periphyton and Macrophyte	0.06	0.09	0.03*	0.35	0.06*	0.09
Gross Photosynthesis ($g\ C/m^2/d$)	0.6	1.2	1.2	2.5	4.0	2.8
Community Respiration ($g\ C/m^2/d$)	0.7	1.6	1.3	2.9	3.1	5.0
Terrestrial Input (g C/m/day)	323	762	267	268	797	236
Fine Particulate Organic Matter (g C/m/day)	3,900	13,200	10,700	10,900	15,900	39,600
Dissolved Organic Matter (g C/m/day)	18,200	22,800	32,200	22,200	65,800	216,400

* Value is low because macrophyte production was not estimated.

URBAN NPS RUNOFF IMPACTS ON STREAM COMMUNITIES

A variety of studies of urban NPS runoff impacts on stream communities have been performed and some will be summarized herein but only one will be discussed in detail as examplifying the processes involved.

Chisholm and Downs [1978] studied the benthic invertebrate ecology of a small West Virginia stream which received sediment from construction of a super-highway. In comparison with a nearby control stream, the diversity index, generic count, and total count of invertebrates indicated severe reduction or destruction of the benthos in the impacted stream. The greatest degradation occurred in areas of highest sediment movement. When construction ceased, the benthic population of the impacted stream recovered to a comparable level with the control stream. Colonization was from upstream, unimpacted tributaries.

Except in streams subject to combined sewer and sanitary overflow discharges, *Ragan and Dieteman* [1975] found streams in areas of Maryland, urbanized during and after the 1950's, to be of good quality. This was not expected but the authors attributed this to the lack of sanitary discharges in the area, and to the large (>10 square miles) size of the drainage basins being studied. Even though traditional water quality parameters did not show dramatic changes, pollution sensitive fish species (*e.g.*, Rosyside Dace) were no longer found in urbanized areas. A nearby, unurbanized stream had the same 21 species of fish that were present in 1912.

DiGiano et al. [1975, 1976] directed their study in Greenfield, Massachusetts, toward both short term impacts on water quality and the longer term disruptions of the benthic macroinvertebrate population caused by urban runoff. Using weekly grab samples from the Green River, they found that total P, turbidity, chloride, and total coliform concentrations increased with runoff events. They were unable to show conclusively that BOD, TOC, and oil and grease concentrations increased with runoff. It was suggested that more detailed analyses of both discharge and concentration be used in determining the mass loading for relatively short (3.2 mile) river reaches. Distinguishing between combined sewer overflows and stormwater runoff events is also important, particularly for bacteriological studies. In an analysis of three specific storm events sampled at 15 or 30 minute intervals, they found considerable variations from storm to storm of total pollutant loading (Table 13).

The study of the benthic diversity of the macroinvertebrate community (Figure 13) of the stream showed constancy in diversity (Brillouin's) and composition of central collections. However, downstream collection, in urbanized areas of the stream, showed reduced diversity and a community shift to more pollution tolerant (polysaprobic) species. In addition, the seasonal variation in diversity increased at downstream stations. The physical character of the stream bottom and availability of substrate deteriorated due to sedimentation. Sedimentation did not explain all of the changes in benthos. The contribution of contaminants from natural and man-made sources remains to be determined. Bioaccumulation of metals may result through the detritus-based food chain. The authors stated that stress from

TABLE 13. RANGE OF LOADING RATES RECORDED FOR STORM EVENTS IN 1975

Parameters	March 24, 1975	June 12, 1975	July 9, 1975
BOD (lb/day)	-95.0 - 6100	1180 - 2550	22.5 - 380
Total P (lb/day)	-	34.6 - 83.6	1.0 - 21.5
TOC (lb/day)	-1040 - 8040	1160 - 9900	95 - 2570
Chloride (lb/day)	10400 - 12500	5240 - 7830	2130 - 5210

these heavy metals on benthic invertebrates needs more investigation.

CASE HISTORY SUMMARY

The major conclusions about these case histories are:

1. macroinvertebrates and periphyton seem to be excellent for monitoring impulse impacts on streams;
2. diversity and biomass are often used for the analysis of monitoring but they are not always adequately sensitive;
3. indicator organisms seem not to be useful;
4. water quality variables do not always reflect stream impacts directly. The biotic community seems to be more important in reflecting impulse inputs of contaminants.

A scenario to illustrate the hydrologic/physico-chemical variables and biological/ecological variables for determining urban NPS runoff impacts on stream ecosystems would help illustrate some of the problems inherent in analyzing these kinds of systems. A storm occurs at a specific time for a specific interval. The precipitation begins to run off depending on antecedent storms, soil moisture and impervious surfaces. The pickup of contaminants varies with antecedent "washings," velocity of runoff and natural and cultural activities in the runoff basin. Transport of the water and contaminant materials enters the stream diffusely or at a point or several points and over an interval and pattern controlled by the transport system. This pattern provides the random event/magnitude impulse of contaminants.

The impulse of contaminants is diluted within the stream and has variable impact depending on peak concentration of contaminants, duration of exposure and life cycle period of the organisms making up the community. Survival of specific organisms/taxa/niche is controlled by such events. Eliminated community components can be replaced by recolonization but the interval depends on out-of-stream or upstream sources and physiological variables (length of reproductive cycle, motility). Other niches may be opened by such inputs but generally communities become simpler, nutrient cycling speeds up (shorter turnover times), and efficiency decreases. The variables that seem most useful in reflecting these processes are not necessarily the classical biotic density and diversity measurements used

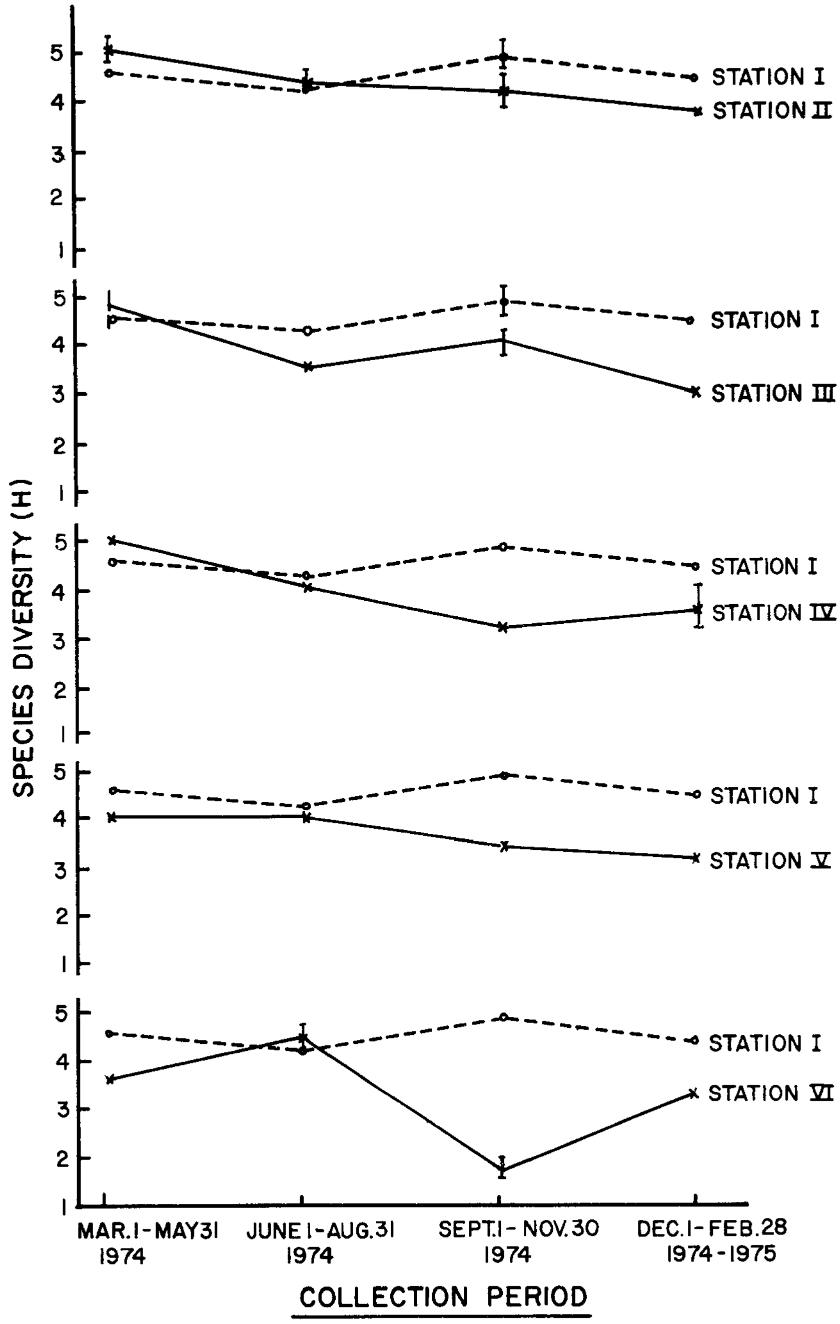

Figure 13. Comparison of the species diversities of the benthic macroinvertebrates between Stations 1 and 6, Green River, Greenfield, Massachusetts [*DiGiano et al.* 1976].

previously but those processes associated with community function: energy flow and material cycling.

SECTION VII

THE ANALYSIS OF URBAN NPS RUNOFF IMPACTS ON STREAM ECOSYSTEMS

Throughout this report, we have asserted that random event and magnitude impulses to stream ecosystems would result in changes in ecological variables related to community structure and function. In this section, we propose an approach for assessing such impacts. Because the spatial distribution of the individuals of a species frequently follows a log-normal distribution [*Slocomb and Dickson* 1978] and the probability distribution of runoff events can be determined, a stochastic approach suggests itself. A stochastic model is a probabilistic model that includes a time dependent component. Some techniques that illustrate the application of stochastic models to water quality data indicate the feasibility of this approach. *Krishman et al.* [1974], *Padgett et al.* [1977], and *Padmanabhan and Delleur* [1978] apply this approach to BOD modeling in streams and thus are able to calculate the mean DO in a stream, the error about the mean, and the probability of violating a specific criterion or standard. All three generated data using storm runoff models because actual data were insufficient.

Smith [1978], in a recent review of optimization in ecological systems, discusses the concept of attempting to explain ecosystem functional variables using optimization models while listing recent attempts to model and optimize ecological processes. An example of how temperature modeling can be used for assessing catastrophic events such as climatic and flow variations that result in extreme water temperatures illustrates a reasonable duplication of actual data [*Morse* 1978]. It is simple to see how such events could result in an ecological catastrophe but it is difficult to assess the magnitude of the ecological response and even more difficult at this time to develop the probabilistic relationships between the impulse and the response. For example, *Slatkin* [1978] used a model study to show that stability existed only when the time scale for environmental change is roughly comparable to the average response time of those changes. Means and variances did not determine extinction tendencies.

A PROBABILITY HYPOTHESIS FOR ECOLOGICAL RESPONSE VARIABLES

Impulse loadings can vary seasonally with material (nutrient, toxicant, oxygen demand) or energy (allochthonous reduced carbon, light, temperature) inputs. Such random impulses are exemplified for a hypothetical seasonal cycle for fish in Figure 14. Two sensitive stages are illustrated: (1) egg

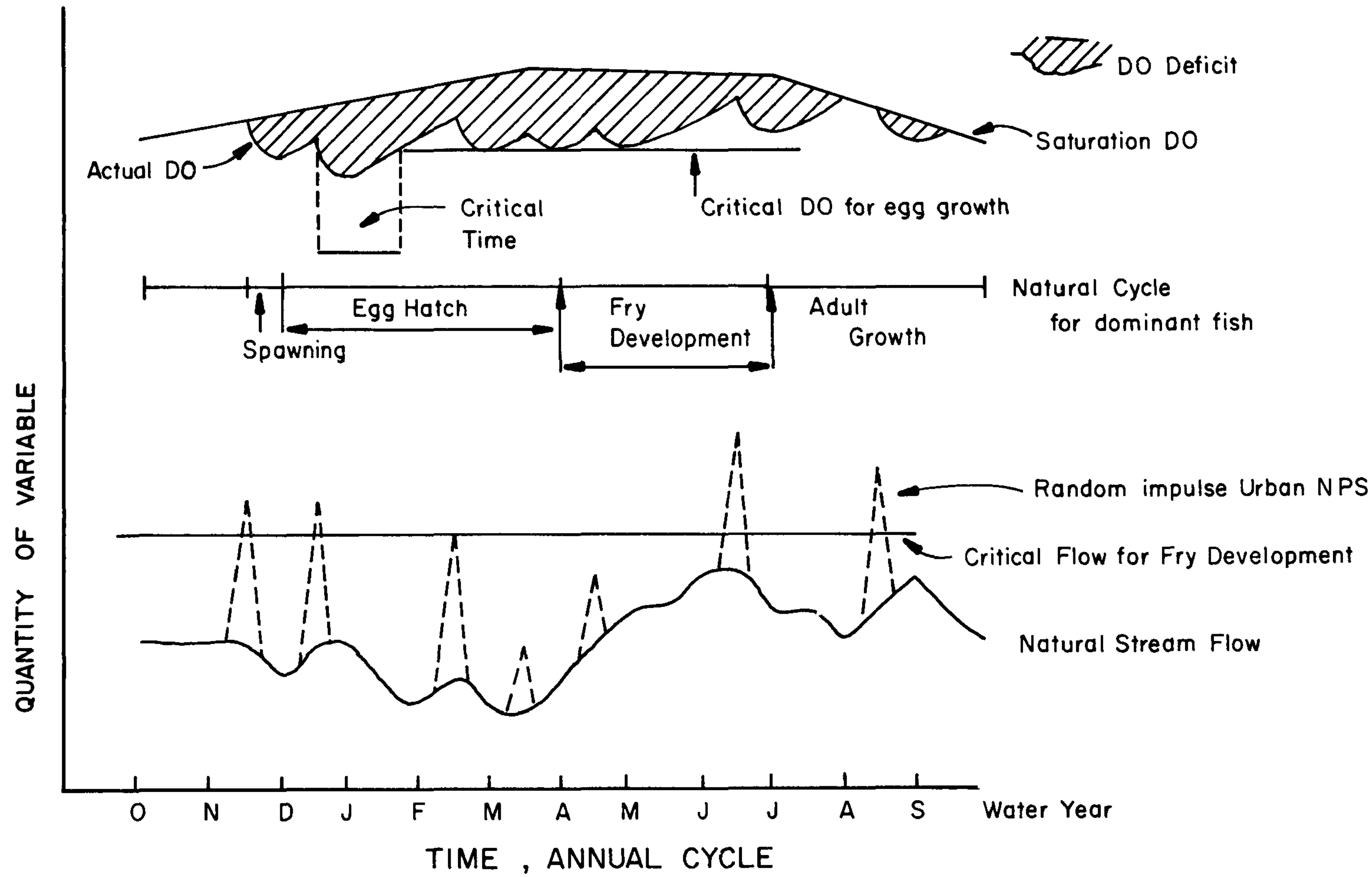

Figure 14. Hypothetical stream ecosystem effects of random impulse (urban NPS) loadings on ecological variables.

growth requirement for dissolved oxygen (the DO is utilized by degradation of organic matter in urban NPS runoff); (2) fry development requires lower streamflow to catch food and maintain safety from predation by large fish. If events cause changes in DO and flow variables to violate the requirement for a sufficient time and a sufficient magnitude, that year class could be eliminated. Continued annual cycles where violations occur could eliminate the species.

Thus the probability that a species or individual will not survive is equal to the probability that at least one event will occur that will eliminate the species or individual. For example, where flow is the event that threatens the species:

$$\text{prob (species eliminated)} = 1 - \text{prob } (Q \leq q_c) = 1 - F_Q(q_c)$$

in which

Q = flow

q_c = critical value of flow at which species is eliminated

$F_Q(q_c)$ = cumulative probability distribution of flow evaluated at q_c

However, flow has a deterministic seasonal component and therefore, F_Q and Q can be written as functions of time. Also, the critical value of flow varies with the stage of development of the species and is also a function of time. Therefore, Equation 1 can be rewritten as follows:

$$\begin{aligned}\text{prob (species eliminated)} &= 1 - \text{prob } (Q(t) \leq (q_c(t)) \\ &= 1 - F_Q(q_c(t);\ t)\end{aligned}$$

Similar relationships can be derived for other response variables. This approach is only a flow magnitude event, but this basic procedure can also be extended to the case of a flow magnitude-duration event system. With the relationship developed above, survival versus probability curves could be obtained that relate to inputs having a certain probability (Figure 15).

The effects of a certain urban NPS runoff input on a specific ecological variable would not be a specific answer but would vary within the probability of a specific event occurring and a specific magnitude with a certain probability of response depending on the space wherein the impact occurs (Figure 16). Because response is concentration related and high concentrations have a low probability of occurring, the probability of a species surviving or not surviving may be difficult to assess by modeling and calculation. One means of assessing this approach is to determine with careful measurement, the pattern of urban NPS inputs to a stream, determine the immediate response, relate the response to average, peak, range of concentration, and determine if the community structure and function varies with time (before/after) or space (upstream/downstream) as would be expected. The hypothesis for testing is: the community structure and function is simpler (or less) than would be expected or than a control for a chronic input of material of the same total loading; *i.e.*, random event and magnitude impulse loading is more deleterious than a chronic, steady loading.

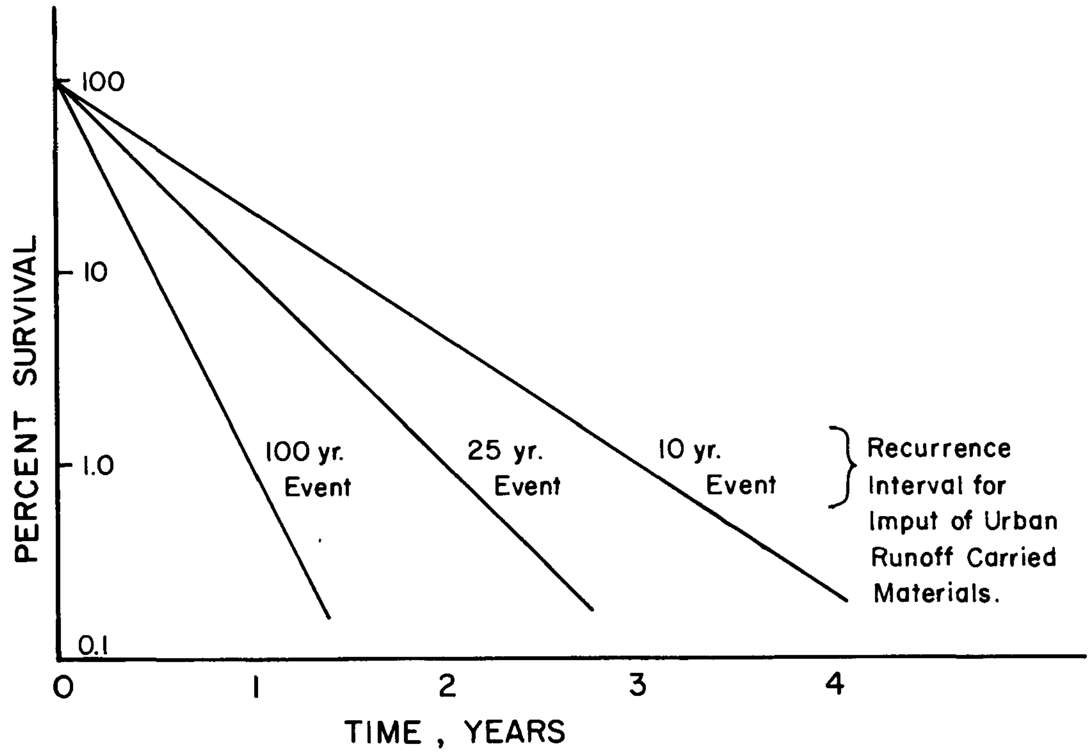

Figure 15. Percent survival decreases with more severe urban runoff events.

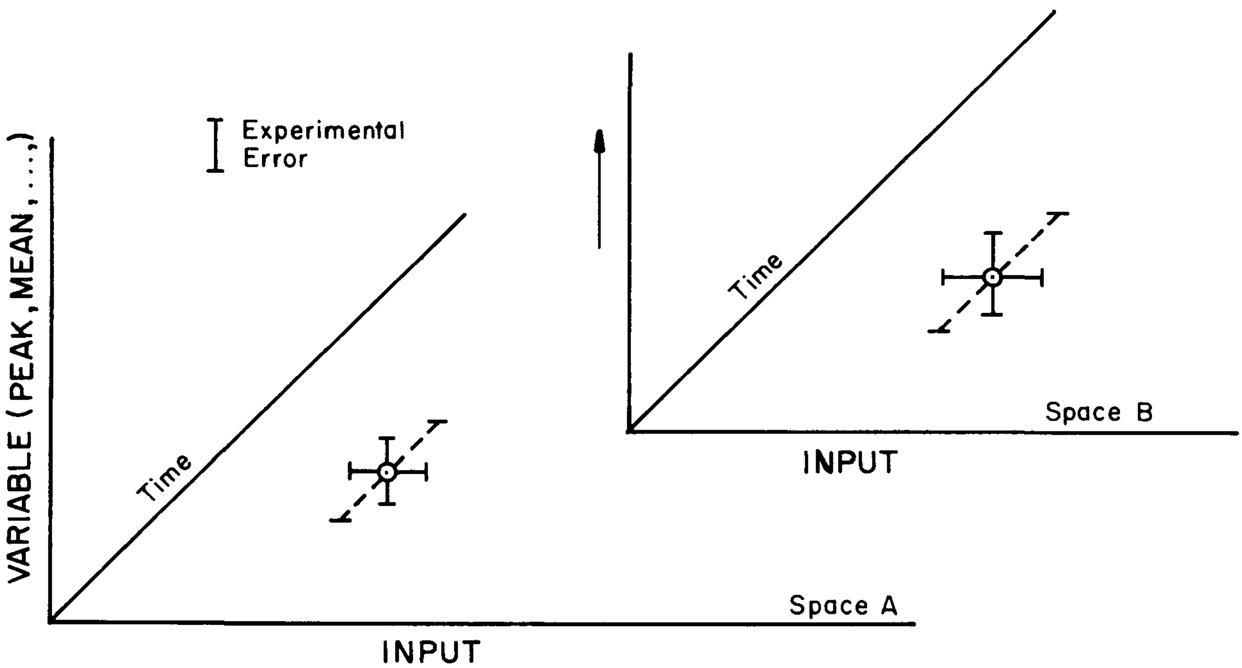

Figure 16. There is experimental error for system variables which adds to the variance caused by response to stochastic inputs.

Ecological variables for assessing these impacts would be measurements of structurally and functionally defined biomass, photosynthesis, respiration, diversity, in time and space. The materials, carbon and oxygen, would be measured. Values would be estimated using these units and energy units. Water quality and material input measurements should be made, especially throughout the flood hydrograph. The number of required measurements could be determined by using rarefaction procedures [*Simberloff* 1978]. Eventually, based on a probability defined by policy, management or law, the stream sampling in time and space could be defined on the basis of the ecological resource being protected.

SECTION VIII

REFERENCES

Anderson, K. B., R. E. Sparks, and A. A. Paparo. 1978. *Rapid assessment of water quality using the fingernail clam* Musculium transversum. UILU-WRC-78-0133. Wat. Res. Cen., Univ. of Illinois at Urbana-Champaign, Urbana, IL. 115 p.

APHA. 1976. *Standard methods*. APHA. New York, NY. pp. 685-882.

Ball, R. C., N. R. Kevern, and T. A. Haines. 1973. *An ecological evaluation of stream eutrophication.* Tech. Rpt. #36. Inst. Wat. Res., Mich. St. Univ. East Lansing, MI.

Barkdoll, M. P., D. E. Overton, and R. P. Betson. 1977. Some effects of dustfall on urban stormwater quality. *Journal WPCF* 49:1976-1984.

Bartsch, A. F., and W. M. Ingram. 1966. Biological analysis of water pollution in North America. *Verh. Internat. Verein. Limnol.* 16:786-800.

Beck, W., Jr. 1977. *Environmental requirements and pollution tolerance of common freshwater chironomidae.* EPA-600/4-77-024. U. S. EPA, Cincinnati, OH. 261 p.

Bick, H. 1971. The potentials of ciliated protozoa in the biological assessment of water pollution levels. *In: Proc. Int. Symp. on Identification and Measurement of Environment Pollutants.* Ottawa, Ont., Can. pp. 305-309.

Boesch, D. F. 1977. *Application of numerical classifications in ecological investigation of water pollution.* EPA-600/3-77-033. U. S. EPA, Corvallis, OR. 115 p.

Boling, R. H., Jr., E. D. Goodman, J. A. Van Sickle, J. O. Zimmer, K. W. Cummins, R. O. Petersen, and S. R. Reiçe. 1975. Toward a model of detritus processing in a woodland stream. *Ecol.* 56:141-151.

Bovee, K. D., and R. Milhouse. 1978. *Hydraulic simulation in instream flow studies: Theory and technique.* IFI Paper No. 5, FWS/OBS-78/33. U. S. FWS, Ft. Collins, CO 80526. 131 p.

Bradford, W. L. 1977. Urban stormwater pollutant loadings: A statistical summary through 1972. *Journal WPCF* 49:613-622.

Brock, D. A. 1977. Comparison of community similarity indexes. *Journal WPCF* 49:2488-2493.

Brownlee, R. C., T. A. Austin, and D. M. Wells. 1970. *Variation of urban runoff with duration and intensity of storms.* Interim report. Wat. Res. Cen., Tex. Tech Univ. Lubbock, TX. 67 p.

Bryan, E. H. 1974. Concentrations of lead in urban stormwater. *Journal WPCF* 46:2419-2421.

Cairns, J., Jr. 1975. Quantification of biological integrity. *In: The Integrity of Water.* U. S. EPA. Sup. of Doc. 055-001-01068-1. Wash. D. C. 20402. pp. 171-188.

Cairns, J., Jr., D. W. Albaugh, F. Busey, and M. D. Chanay. 1968. The sequential comparison index: A simplified method for non-biologists to estimate relative differences in biological diversity in stream pollution studies. *Journal WPCF* 40:1607-1613.

Cairns, J., Jr., and K. L. Dickson. 1971. A simple method for the biological assessment of the effects of waste discharges on aquatic bottom-dwelling organisms. *Journal WPCF* 43:755-772.

Cairns, J., Jr., and K. L. Dickson. 1977. Recovery of streams from spills of hazardous materials. *In: Recovery and Restoration of Damaged Ecosystems.* J. Cairns, Jr., K. L. Dickson, and E. E. Herricks, eds. Charlottesville, VA. pp. 24-42.

Cairns, J., Jr., K. L. Dickson, and J. S. Crossman. 1970. The biological recovery of the Clinch River following a fly ash pond spill. *25th Industrial Waste Conference.* Part One. pp. 182-192.

Cech, I., and K. Assaf. 1976. Quantitative assessment of changes in urban runoff. *ASCE J. Irrig. Drain. Div.* 102:119-125.

Chen, C. W., and R. F. Selleck. 1969. A kinetic model for fish toxicity threshold. *Journal WPCF* 41:294-308.

Cherkauer, D. S. 1977. Effects of urban lakes on surface runoff and water quality. *Wat. Res. Bull.* 13:1057-1067.

Cherry, D. S., R. K. Guthrie, F. L. Singleton, and R. S. Harvey. 1977. Recovery of aquatic bacterial populations in a stream after cessation of chemical pollution. *Water, Air, and Soil Poll.* 7:95-101.

Chisholm, J. L., and S. C. Downs. 1978. Stress and recovery of aquatic organisms as related to highway construction along Turtle Creek, Boone County, West Virginia. *Geol. Sur. Wat.-Supply Paper 2055.* U. S. GPO, Wash., D. C. 20402. 40 p.

Christensen, E. K., J. Scherfig, and M. Koide. 1978. Metals from urban runoff in dated sediment of a very shallow estuary. *Environ. Sci. Tech.* 12:1168-1173.

Cooke, G. D. 1971. Aquatic laboratory microsystems and communities. *In: The structure and function of fresh-water microbial communities.* John Cairns, Jr. ed. Amer. Micro. Soc. Symp. pp. 47-85.

Cordery, I. 1977. Quality characterization of urban storm water in Sydney, Australia. *Wat. Res. Res.* 13:197-202.

Cornell, H., L. E. Hurd, and V. A. Lotrich. 1976. A measure of response to perturbation used to assess structural change in some polluted and unpolluted stream fish communities. *Oecologia (Berl.)* 23:335-342.

Crowther, R. A., and H. B. N. Hynes. 1977. The effect of road deicing salt on the drift of stream benthos. *Environ. Pollut.* 14:113-126.

Cummins, K. W., R. C. Petersen, R. O. Howard, J. B. Wuycheck, and V. I. Holt. 1973. The utilization of leaf litter by stream detritivores. *Ecol.* 54:336-345.

Curtis, E. J. C., and C. R. Curds. 1971. Sewage fungus in rivers in the United Kingdom: The slime community and its constituent organisms. *Wat. Res.* 5:1147-1159.

Curtis, E. J. C., and D. W. Harrington. 1971. The occurrence of sewage fungus in rivers in the United Kingdom. *Wat. Res.* 5:281-290.

Dale, J. T. 1978. Bottling rainstorms--Chicago's tunnel and reservoir plan. *Journal WPCF* 50:1888-1892.

Davis, E. M., D. M. Casserly, and J. D. Moore. 1977. Bacterial relationships in stormwaters. *Wat. Res. Bull.* 13:895-905.

Davis, J. A., and J. O. Leckie. 1978. Effects of adsorbed complexing liquids on trace metal uptake by hydrous oxides. *Env. Sci. and Tech.* 12:1309-1315.

Dayton, P. K. 1972. Toward an understanding of community resilience and the potential effects of enrichments to the benthos of McMurdo sound, Antarctica. *In: Conservation Problems in Antarctica.* pp. 81-95.

Dickman, M. 1969. A quantitative method for assessing the toxic effects of some water soluble substances based on changes in periphyton community structure. *Wat. Res.* 3:963-972.

Dicks, Brian. 1977. Changes in the vegetation of an oiled Southampton water salt marsh. *In: Recovery and Restoration of Damaged Ecosystems.* J. Cairns, Jr., K. L. Dickson, and E. E. Herricks, eds. Univ. Press of Virginia. Charlottesville, VA. pp. 208-240.

DiGiano, F. A., R. A. Coler, R. C. Dahiga, and B. B. Berger. 1975. A projection of pollutional effects of urban runoff in the Green River, Massachusetts. *In: Urbanization and Water Quality Control*. Annual Symp. of the Amer. Wat. Res. Assoc. Proc. 20. pp. 28-37.

DiGiano, F. A., R. A. Coler, R. Dahiga, and B. B. Berger. 1976. *Characterization of urban runoff - Greenfield, Massachusetts. Phase II.* Wat. Res. Res. Cent., Univ. of Mass. Amherst, MA 137 p.

Dillon, P. J., and F. H. Rigler. 1975. A simple method for predicting the capacity of a lake for development based on lake trophic status. *J. Fish. Res. Board. Canada* 32:1519-1531.

Diskin, M. H., S. Ince, and K. Oben-Hyarko. 1978. Parallel cascades model for urban watersheds. *ASCE J. Hydraul. Div.* 104:261-276.

Doehring, D. O., and M. E. Smith. 1978. *Modeling the dynamic response of floodplains to urbanization in eastern New England.* Environ. Res. Cen. CSU, Ft. Collins, CO 80523. 95 p.

Eckenfelder, W. W., and D. L. Ford. 1970. *Water pollution control: Experimental procedures for process design.* Pemberton Press. Austin and New York. 269 p.

Ellis, J. B. 1976. Sediments and water quality of urban storm water. *Wat. Serv.* 80:730-734.

Emery, R. M., C. E. Moon, and E. B. Welch. 1973. Enriching effects of urban runoff on the productivity of a mesotrophic lake. *Wat. Res.* 7:1505-1516.

Eppley, R. W., and W. H. Thomas. 1969. Comparison of half-saturation constants for growth and nitrate uptake of marine phytoplankton. *J. Phycol.* 5:375-379.

Field, R. 1978. Discussion: Effects of storm frequency on pollution from urban runoff. *Journal WPCF* 50:974-975.

Field, R., and R. Bowden. 1976. Urban runoff and combined sewer overflow. *Journal WPCF* 48:1191-1206.

Field, R., R. Bowden, and K. Roygonyi. 1977. Urban runoff and combined sewer overflow. *Journal WPCF* 49:1095-1104.

Field, R., and B. B. Gardner. 1978. Urban runoff and combined sewer overflow. *Journal WPCF* 50:1170-1185.

Field, R., and D. Knowles. 1975. Urban runoff and combined sewer overflow. *Journal WPCF* 47:1352.

Field, R., and P. J. Szeeley. 1974. Urban runoff and combined sewer overflow. *Journal WPCF* 46:1209-1226.

Field, R., and P. Weigel. 1973. Urban runoff and combined sewer overflow. *Journal WPCF* 45:1108-1115.

Fitzgerald, G. P. 1969. Some factors in the competition or antagonism among bacteria, algae, and aquatic weeds. *Phycol.* 5:351-359.

Foster, M. S., and R. W. Holmes. 1977. The Santa Barbara oil spill: An ecological disaster? *In: Recovery and Restoration of Damaged Ecosystems.* J. Cairns, Jr., K. L. Dickson, and E. E. Herricks, eds. Univ. Press of Virginia. Charlottesville, VA. pp. 166-190.

Gakstatter, J. H., T. E. Maloney, and F. B. Lotspeich. 1977. *Guidelines for assessing the benefits of best management practices to stream ecosystems.* EPA CERL-039. Corvallis, OR 97330. 19 p.

Gannon, J. E., and A. M. Beeton. 1971. Procedures for determining the effects of dredged sediments on biota--Benthos viability and sediment selectivity tests. *Journal WPCF* 43:392-398.

Gates, C. D. 1975. Urban runoff in Binghampton, New York. *In: Urbanization and Water Quality Control.* Annual Symp. of the Amer. Wat. Res Assoc., Proc. 20. pp. 38-44.

Gaufin, A. R. 1973. *Water quality requirements of aquatic insects.* Office of Research and Monitoring. EPA-600/3-73-004.

Geisy, J. P. 1978. *Microcosms in ecological research.* SREL. 6th Annual Symp., Aiken, SC. In preparation.

Geldreich, E. E., L. C. Best, B. A. Kenner, and D. J. Van Donsel. 1968. The bacteriological aspects of stormwater pollution. *Journal WPCF* 40:1861-1872.

Ghetti, P. F., and G. Bonazzi. 1977. A comparison between various criteria for the interpretation of biological data in the analysis of the quality of running waters. *Wat. Res.* 11:819-832.

Gislason, J. C. 1971. *Species diversity of benthic macroinvertebrates in three Michigan streams.* Tech. Rept. #20. IWR. Mich. St. Univ.

Golding, B. L. 1978. Discussion of Proc. Paper 13068. *ASCE J. Hydraul. Div.* 103:458-459.

Graham, P. H., C. S. Lawrence, and H. J. Mallon. 1974. Estimation of imperviousness and specific curb length for forecasting stormwater quality and quantity. *Journal WPCF* 46:717-725.

Grahn, O. 1970. Macrophyte succession in Swedish lakes caused by deposition of airborne acid substances. *Water, Air, and Soil Poll.* 7:295-305.

Grenney, W. J., D. B. Porcella, and M. L. Cleave. 1976. Water quality relationships to flow-streams and estuaries. *In: Methodologies for the Determination of Stream Resource Flow Requirements: An Assessment.* USFWS. Ft. Collins, CO 80526. pp. 35-88.

Gundlach, D. L. 1976. Unit hydrograph parameters versus urbanization. *ASCE J. Irrig. Drain. Div.* 102:388-392.

Guy, H. P., and D. E. Jones, Jr. 1972. Urban sedimentation -- In perspective. *ASCE J. Hydraul. Div.* 98:2099-2116.

Haan, C. T., and R. W. DeVore. 1978. *Proceedings of International Symposium on Urban Storm Water Management.* Univ. of Kentucky, Lexington, KY. 348 p.

Haedrich, R. L. 1975. Diversity and overlap as measure of environmental quality. *Wat. Res.* 9:945-952.

Haith, D. A. 1976. Land use and water quality in New York rivers. *ASCE J. Env. Eng. Div.* 102:1-15.

Hamilton, M. A. 1975. Indexes of diversity and redundancy. *Journal WPCF* 47:630-632.

Harrell, H. L. 1978. Response of the Devil's River (Texas) fish community to flooding. *Copia.* 1:60-68.

Hawkins, R. H. 1976. Salt storage and runoff in urban watershed. *ASCE J. Env. Eng. Div.* 102:737-743.

Hayden, J. W., J. Dickson, and D. Wisnuwski. 1972. Water resources management for a small watershed in an urbanizing area. *In: Watersheds in Transition.* A. C. Csallany *et al.*, eds. Amer. Wat. Res. Assoc. Urbana, IL. pp. 380-386.

Heck, K. L., Jr. 1976. Community structure and the effects of pollution in sea-grass meadows and adjacent habitats. *Marine Biol.* 35:345-357.

Hendrix, P. 1979. *Review of biological and ecological methods for assessing stream ecosystems.* Draft Report. Univ. of Georgia, Inst. of Ecol. Athens, GA.

Hickock, E. A., M. C. Hannaman, and N. C. Wenk. 1977. *Urban runoff treatment methods. Volume I - Non-structural wetland treatment.* EPA-600/2-77-217. U. S. Environmental Protection Agency. Cincinnati, OH. 121 p.

Hocutt, C. H. 1975. Assessment of a stressed macro-invertebrate community. *Wat. Res. Bull.* 11:820-821.

Holling, C. S. 1973. Resilience and stability of ecological systems. *Ann. Rev. Syst. & Ecol.* 4:1-23.

Hossain, A., A. R. Rao, and J. W. Delleur. 1978. Estimation of direct runoff from urban watersheds. *ASCE J. Hydraul. Div.* 104:169-188.

Howmiller, R. P., and M. A. Scott. 1977. An environmental index based on relative abundance of oligochaete species. *Journal WPCF* 49:809-815.

Hulbert, S. H. 1971. The non-concept of species diversity: A critique and alternative parameters. *Ecol.* 52:577-586.

International Hydrological Decade. 1974. Hydrological effects of urbanization. *Report of the Sub-group on the Effects of Urbanization on the Hydrological Environment, of the Co-ordinating Council of the International Hydrological Decade.* The Unesco Press, Paris, France. 280 p.

Israelsen, E. K., D. R. Bernard, R. M. Twedt, and H. M. Runke. 1975. *A technique for predicting the aquatic ecosystem response to weather modification.* PRWG138-1, Utah Water Res. Lab., Utah State Univ., Logan, UT 84322. 157 p.

Jewell, T. K., T. J. Nunno, and D. D. Adrian. 1978. Methodology for calibrating stormwater models. *ASCE J. Env. Eng. Div.* 104:485-501.

Jodie, J. B. 1975. Quality of urban freeway storm water. *Trans. Res. Record.* 556:1-5.

Judd, J. H. 1970. Lake stratification caused by runoff from street deicing. *Water Res.* 4:521-532.

Judd, J, and G. A. Carlson. 1978. Land use and water quality. *Clearwaters* 8:8-10.

Kaesler, R. L., E. E. Herricks, and J. S. Crossman. 1978. Use of indices of diversity and hierarchical diversity in stream surveys. *In: Biological Data in Water Pollution Assessment: Quantitative and Statistical Analyses.* ASTM STP 652. K. L. Dickson, J. Cairns, Jr., and R. J. Livingston, eds. pp. 92-112.

Kent, K. M. 1973. *A method for estimating volume and rate of runoff in small watersheds.* [Quoted from *Wanielista* 1978]. USDA, SCS. TP-149.

Kolkwitz, R., and M. Marsson. 1908. Okologie der pflanzlicken Saprobien. *Berichte der deutschen Bot. Ges.* 26a:505-519.

Kolkwitz, R., and M. Marsson. 1909. Okologie der tierischen Saprobien. *Int. Rev. ges. Hydrobiol. u. Hydrogr.* 2:126-152.

Krishnan, K., P. Radha, J. J. Lizcano, L. E. Erickson, and L. T. Fan. 1974. Evaluation of methods for estimating stream water quality parameters in a transient model for stochastic data. *Wat. Res. Bull.* 10:899-913.

Lager, J. A., R. P. Shubinski, and L. W. Russell. 1971. Development of a simulation model for stormwater management. *Journal WPCF* 43:2424-2435.

Lindholm, O. G. 1976. Pollutional analysis of combined sewer systems. *ASCE J. Env. Eng. Div.* 102:301-312.

Linsley, R. K., and J. B. Franzini. 1964. *Water resources engineering.* McGraw-Hill Book Co., San Francisco, CA 654 p.

Loehr, R. C. 1974. Characteristics and comparative magnitude of nonpoint sources. *Journal WPCF* 46:1849-1872.

Logan, D. T. 1977. Discussion: Indexes associated with information theory in water quality. S. M. Zand, Journal WPCF 48:2026 [1976]. *Journal WPCF* 49:879-880.

Lowe, R. L. 1974. *Environmental requirements and pollution tolerance of freshwater diatoms.* EPA-670/4-74-005. U. S. EPA, Cincinnati, OH 334 p.

Lynch, M. P. 1975. The use of physiological indicators of stress in marine invertebrates as a tool for marine pollution monitoring. *In: Proceedings of Marine Technology Society, Tenth Annual Conference.* NOAA 75051202. NTIS Springfield, VA. 22161. pp. 881-890.

MacKenthun, J. L. 1975. Legislative requirements. *In: The Integrity of Water.* U.S. EPA. Sup. of Docu. 053-001-01068-1, Wash. DC 20402. pp. 5-8.

Mackenzie, M. J., and J. V. Hunter. 1979. Sources and fates of aromatic compounds in urban stormwater runoff. *Env. Sci. and Tech.* 13:179-183.

MacMahon, J. A., D. L. Phillips, J. V. Robinson, and D. J. Schimpf. 1978. Levels of biological organization: An organism-centered approach. *BioScience* 28:700-704.

MacMahon, J. A., D. J. Schimpf, D. C. Anderson, K. G. Smith, and R. L. Bayn, Jr. 1979. *An organism-centered approach to some community and ecosystem concepts.* Unpublished manuscript. Utah State Univ., Logan, UT 84322. 30 p.

Maguire, B., Jr. 1971. Community structure of protozoans and algae with particular emphasis on recently colonized bodies of water. *In: The Structure and Function of Freshwater Microbial Communities.* J. Cairns, Jr., ed. Amer. Micros. Sco. Symp. pp. 121-149.

Malone, R. F., D. S. Bowles, W. J. Grenney, and M. P. Windham. 1979. *Stochastic analysis for water quality.* Utah Water Res. Lab. UWRL/Q-79/01, Utah State Univ., Logan, UT 84322. 75 p.

Margalef, R. 1951. Diversidas de especies en las communidades naturales. *P. Inst. Biol. Appl.* 9:5-27.

Margalef, R. 1965. *Perspectives in ecological theory.* The Univ. of Chicago Press. 111 p.

Margalef, R. 1975. External factors and ecosystem stability. *Schweiz. Z. Hydrol.* 37:102-117.

May, R. M. 1973. *Stability and complexity in model ecosystems.* Princeton Univ. Press, Princeton, NJ. 235 p.

McElroy, A. D., S. Y. Chiu, J. W. Nebgen, A. Aleti, and F. W. Bennett. 1976. *Loading functions for assessment of water pollution from nonpoint sources.* EPA-600/2-76-151. U.S. EPA, Wash. DC 20460. 445 p.

McGriff, E. C., Jr. 1972. The effects of urbanization on water quality. *J. Environ. Qual.* 1:86-89.

Michaelis, L., and M. L. Menten. 1913. Die kinetik der invertinwirkung. *Biochem. Z.* 49:333.

Middlebrooks, E. J., D. H. Falkenborg, and T. E. Maloney, eds. 1973. *Modeling the eutrophication process.* Proc. Utah Water Res. Lab., PRWG136-1, Utah State Univ., Logan, UT 84322. 228 p.

Minshall, G. W. 1967. Role of allochthonous detritus in the trophic structure of a woodland springbrook community. *Ecol.* 48:139-149.

Minshall, G. W. 1968. Community dynamics of the benthic fauna in a woodland springbrook. *Hydrobiologia* 32:305-339.

Mischi, E. F., and V. V. Dhasmadhikari. 1971. Runoff--A potential resource. *Water and Wastes Eng.* 8:28-31.

Mitchell, D., and J. C. Buzzell. 1971. Estimating eutrophic potential of pollutants. *ASCE J. Sanit. Eng. Div.* 97:453-465.

Moeller, J. R., G. W. Minshall, K. W. Cummins, R. C. Peterson, C. E. Cushing, J. R. Sedell, R. A. Larson, and R. L. Vannote. In Press. Transport of dissolved organic carbon in streams of differing physiographic characteristics.

Monod, J. 1949. The growth of bacterial cultures. *Annual Review of Microbiology* 3:371-373.

Morse, W. L. 1978. The dishonest method in stream temperature modeling. *Wat. Res. Res.* 14:45-51.

Nelson-Smith, A. 1977. Recovery of some British rocky seashores from oil spills and cleanup operations. *In: Recovery and Restoration of Damaged Ecosystems.* J. Cairns, Jr., K. L. Dickson, and E. E. Herricks, eds. Univ. Press of Virginia. Charlottesville, VA. pp. 191-207.

Nemerow, N. L. 1974. *Scientific stream pollution analysis.* McGraw-Hill. 358 p.

Newton, C. D., W. W. Shephard, and M. S. Coleman. 1974. Street runoff as a source of lead pollution. *Journal WPCF* 46:999-1000.

Odum, E. P. 1969. The strategy of ecosystem development. *Sci.* 164:262-270.

Odum, E. P. 1971. *The principles of ecology.* 3rd Ed. Wiley. New York, NY.

Oliver, B. G., J. B. Milne, and N. LaBarre. 1974. Chloride and lead in urban snow. *Journal WPCF* 46:766-771.

Olivieri, V. P., C. W. Kruse, and K. Kawata. 1977. *Microorganisms in urban stormwater.* EPA-600/2-77-087. U.S. EPA, Cincinnati, OH. 182 p.

Orians, G. H. 1975. Diversity, stability and maturity in natural ecosystems. *Unifying Concepts in Ecology.* W. H. Van Dobben and R. H. Lowe-McConnell, Eds. pp. 139-150.

Orlob, G. T. 1975. Modeling of aquatic ecosystems. *In: The Integrity of Water.* U.S. EPA, Sup. of Docu. 055-001-01068-1. Wash. DC 20402.

Ott, W. R. 1978a. *Water quality indices: a survey of indices used in the United States.* EPA-600/4-78-005. 138 p.

Ott, W. R. 1978b. *Environmental indices: theory and practice.* Ann Arbor Science Publ., Inc. Ann Arbor, MI.

Padmanabhan, G., and J. W. Delleur. 1978. *Statistical and stochastic analyses of synthetically generated urban drainage quantity and quality data.* Tech. Rept. No. 108. Wat. Res. Cen., Purdue Univ. West Lafayette, IN. 111 p.

Padgett, W. J., G. Schultz, and C. P. Tsokos. 1977. A stochastic model for BOD and DO in streams when pollutants are discharged over a continuous stretch. *Intern. J. Environ. Studies.* 11:45-55.

Paine, R. T. 1969. A note on trophic complexity and community stability. *Amer. Natur.* 103:91-93.

Park, R. A. 1978. *A model for simulating lake ecosystems.* CEM Rept. 3. RPI, Troy, NY 12181. 19 p.

Patrick, R. 1949. A proposed biological measure of stream conditions, based on a survey of the Conestoga Basin, Lancaster County, Pennsylvania. *Proc. Acad. Nat. Sci. Phila., PA.* 101:277-341.

Patrick, R. 1971. Diatom communities. *In: The structure and function of freshwater microbial communities.* J. Cairns, Jr., Ed. Amer. Micros. Soc. Symp. pp. 151-164.

Pearson, T. H., and R. Rosenberg. 1976. A comparative study of the effects on the marine environment of wastes from cellulose industries in Scotland and Sweden. *Ambio* 5:77-79.

Pennack, R. W. 1971. Toward a classification of lotic habitats. *Hydrobiologia* 38:321-334.

Pennino, B. J., and R. A. Perkins. 1978. Hydraulic design for combined sewer diversion. *ASCE J. Hydraul. Div.* 104:595-606.

Phamwon, S., and Y. Fok. 1977. Urban runoff digital computer model. *ASCE J. Hydraul. Div.* 103:723-735.

Phillips, D. J. 1977. The use of biological indicator organisms to monitor trace metal pollution in marine and estuarine environments -- A review. *Env. Poll.* 13:281-317.

Pielou, E. C. 1975. *Ecological diversity.* Wiley Intersci. Pub. 165 p.

Porcella, D. B., V. D. Adams, P. A. Cowan, S. Austrheim-Smith, W. F. Holmes, J. Hill, IV, W. J. Grenney, and E. J. Middlebrooks. 1975. *Nutrient dynamics and gas production in aquatic ecosystems: The effects and utilization of mercury and nitrogen in sediment-water microcosms.* Utah Water Res. Lab., Utah State Univ., Logan, UT 84322.

Pratt, J. M., and R. A. Coler. 1976. A procedure of the routine biological evaluation of urban runoff in small rivers. *Wat. Res.* 10:1019-1025.

Ragan, R. M., and A. J. Dieteman. 1975. Impact of urban stormwater runoff on stream quality. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc., Proc. 20.* pp. 55-61.

Randall, C. W., T. J. Grizzard, and R. C. Hoehn. 1978. *Impact of urban runoff on water quality in the Occoquan watershed.* Bull. 80. Vir. Water Res. Res. Cent. Blacksburg, VA 77 p.

Ray, S., and W. White. 1976. Selected aquatic plants as indicator species for heavy metal pollution. *J. Env. Sci. Health.* A11:717-725.

Reger, S. J. 1973. *Benthic macroinvertebrate diversity in three differentially perturbed Michigan streams.* M.S. Thesis. Dept. Fish and Wildlife, Mich. State Univ.

Reichle, D. E., R. V. O'Neill, and W. F. Harris. 1975. Principles of energy and material exchange in ecosystems. *In: Unifying Concepts in Ecology.* W. H. Van Dobben and R. H. Lowe-McConnell, Eds. Dr. W. Junk B. Publishers. The Hague. pp. 27-43.

Reynolds, J. H., E. J. Middlebrooks, D. B. Porcella, and W. J. Grenney. 1975. Effects of temperature on oil refinery waste toxicity. *Journal WPCF* 47:2674-2693.

Richey, J. E., R. C. Wissmar, A. H. Devol, G. E. Likens, J. S. Eaton, R. G. Wetzel, W. E. Odum, N. M. Johnson, O. L. Loucks, R. T. Prentki, and P. H. Rich. 1978. Carbon flow in lake ecosystems: A structural approach. *Sci.* 202:1183-1186.

Rickert, D. A., W. G. Hines, and S. W. Mckenzie. 1975. Planning implication of dissolved oxygen depletion in the Willamette River, Oregon. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc. Proc. 20.* pp. 70-84.

Rimer, A. E., J. A. Nissen, and D. E. Reynolds. 1978. Characterization and import of stormwater runoff from various land covers types. *Journal WPCF* 50:252-264.

Roesner, L. A., D. F. Kibler, and J. R. Monser. 1972. Use of storm drainage models in urban planning. *In: Watersheds in Transition.* A. C. Csallany *et al.*, Eds. Amer. Wat. Res. Assoc. Urbana, IL pp. 400-405.

Rosenberg, D. M., and A. P. Wiens. 1976. Community and species responses of Chironomidae (Diptera) to contamination of fresh waters by crude oil and petroleum products, with special reference to the Trail River, Northwest Territories. *J. Fish. Res. Board Can.* 33:1955-1963.

Rovey, E. E., and D. A. Woolhiser. 1977. Urban storm runoff model. *ASCE J. Hydraul. Div.* 103:1339-1351.

Ruthven, J. A., and J. Cairns, Jr. 1973. Response of fresh-water protozoan artificial communities to metals. *J. Protozool.* 20:127-135.

Sartor, J. D., and G. B. Boyd. 1972. *Water pollution aspects of street surface contaminants.* EPA-R2-72-081. U.S. EPA, Wash. DC 237 p.

Sawyer, C. N. 1947. Fertilization of lakes by agricultural and urban drainage. *J. of the New England Water Works Assoc.* 61:109-127.

Schulz, M., and L. J. Comerton. 1974. Effect of aircraft deicer on airport storm runoff. *Journal WPCF* 46:173-180.

Scott, W. S. 1976. Effect of road deicing salts in sodium concentration in an urban watercourse. *Env. Poll.* 10:141-153.

Shannon, C. E., and W. Weaver. 1949. *The mathematical theory of communication.* Univ. of Illinois. Urbana, IL.

Shih, G. B., J. P. Riley, and E. K. Israelsen. 1975. *Regional analyses of runoff characteristics of small urban watersheds.* Utah Water Res. Lab., PRWG87-1, Utah State Univ., Logan, UT 84322. 69 p.

Simberloff, D. 1978. Use of rarefaction and related methods in ecology. *In: Biological Data in Water Pollution Assessment: Quantitative and Statistical Analyses.* K. L. Dickson, J. Cairns, Jr., and R. J. Livingston, Eds. ASTM. Philadelphia, PA 19103. pp. 150-165.

Sladacek, V. 1973. System of water quality from the biological point of view. *Arch. Hydrobiol. Beith.* 7:1-218.

Slatkin, M. 1978. The dynamics of a population in a Markovian environment. *Ecol.* 59:249-256.

Slobodkin, L. B., and H. L. Sanders. 1969. On the contribution of environmental predictability to species diversity. *Brookhaven Symp. Biol.* 22:82-95.

Slocomb, J., and K. L. Dickson. 1978. Estimating the total number of species in a biological community. *In: Biological Data in Water Pollution Assessment: Quantitative and Statistical Analyses.* K. L. Dickson, J. Cairns, Jr., and R. J. Livingston, Eds. STP 652. ASTM, Philadelphia, PA 19103 pp. 38-52.

Smith, J. M. 1978. Optimization theory in evolution. *Ann. Rev. Ecol. and Syst.* 9:31-56.

Smith, R., and R. G. Eilers. 1978. Effect of stormwater on stream dissolved oxygen. *ASCE J. Env. Eng. Div.* 104:549-559.

Smith, R. W., and C. S. Greene. 1976. Biological communities near submarine outfalls. *Journal WPCF* 48:1894-1912.

Sprague, J. B. 1971. Measurement of pollutant toxicity to fish. III. Sub-lethal effects and "safe" concentrations. *Wat. Res.* 5:245-266.

Stalnaker, C. B. 1979. *Stream habitat and its physical attributes as influenced by water and land management.* Utah Water Res. Lab. UWRL/G-79/01. L. D. James, Ed. Utah State Univ., Logan, UT 84322. pp. 148-170.

Stauffer, J. R., Jr., K. L. Dickson, and J. Cairns, Jr. 1974. Field evaluation of the effects of heated discharges on fish distribution. *Wat. Res. Bull.* 10:860-876.

Stoneburner, D. L., L. A. Smock, and H. C. Eichhorn. 1976. A comparison of two diversity indexes used in wastewater impact assessment. *Journal WPCF* 48:736-741.

Streeter, H., and E. Phelps. 1925. *A study of the purification of the Ohio River.* U.S. PHS. Bull. No. 146. Wash., DC.

Stumm, W., and P. Baccini. 1978. *Lakes.* A. Lerman, Ed. Springer-Verlag, New York. pp. 91-123.

Sullivan, R. H. 1974. Letter: Street runoff. *Journal WPCF* 46:537a.

Sutherland, R. C., and R. H. McCuen. 1978. Simulation of urban non-point source pollution. *Wat. Res. Bull.* 14:409-428.

Thomann, R. V. 1971. *Systems analysis and water quality management.* Env. Res. and Appl., Inc. New York, NY 10017. 286 p.

Thomas, W. A., G. Goldstein, and W. H. Wilcox. 1973. *Biological indicators of environmental quality.* Ann Arbor Sci. Publ. Inc. Ann Arbor, MI. 254 p.

Thompson, G. B., D. M. Wells, R. M. Sweazy, and B. J. Claborn. 1974. *Variation of urban runoff quality and quantity with duration and intensity of storms, phase III.* Texas Tech Univ., Wat. Res. Cen. Lubbock, TX. Interim Report. 56 p.

U. S. Department of Commerce. 1975. *Statistical abstract of the United States: National data book and guide to sources.* 1050 p.

U. S. Environmental Protection Agency. 1976. *Quality criteria for water.* Wash., DC 20460. 256 p.

Vollenweider, R. A. 1968. *The scientific basis of lake and stream eutrophication with particular reference to phosphorus and nitrogen as eutrophication factors.* Tech. Rept. to OECD, Paris, DAS/CSI/68. 27:1-182. (mimeo)

Vuceta, J., and J. J. Morgan. 1978. Chemical modeling of trace metals in fresh waters: Role of complexation and adsorption. *Env. Sci. and Tech.* 12:1302-1309.

Wakeham, S. G. 1977. A characterization of the sources of petroleum hydrocarbons in Lake Washington. *Journal WPCF* 49:1680-1687.

Walesh, S. G., and R. M. Videkovich. 1978. Urbanization: Hydrologic-hydraulic-damage effects. *ASCE J. Hydraul. Div.* 104:141-155.

Wanielista, M. P. 1978. *Stormwater management: Quantity and quality.* Ann Arbor Sci. Publ., Ann Arbor, MI. 383 p.

Weibel, S. R. 1969. Urban drainage as a factor in eutrophication. *In: Eutrophication: Causes, Consequences, Correctives.* Nat. Acad. of Sci. Wash., DC. pp. 383-403.

Weibel, S. R., R. B. Weidner, F. M. Cohen, and A. G. Christianson. 1966. Pesticides and other contaminants in rainfall and runoff. *Journal AWWA.* 58:1075-1084.

Wells, D. M., R. M. Swezy, B. J. Claborn, and R. H. Ramsey. 1975. *Variation of urban runoff quality and quantity with duration and intensity of storms--Phase III.* Final Rept. Vol. 4. OWRT Project B-177-TX. Wat. Res. Cen., Texas Tech Univ., Lubbock, TX. 47 p.

Wenster, P., S. O. Ryding, and C. Forsberg. 1975. Naturvårdsverkets RR-undersökning. 7. Diffus kommunal förorenings-belastning. Exempel från Motala och Uppsala. (Recovery of polluted lakes. A Swedish research program on the effects of advanced wastewater treatment and sewage diversion. 7. Diffuse municipal pollution. Examples from the towns of Motala and Uppsala.) *Vatten* 3:274-285.

Whipple, W., Jr., B. B. Berger, G. D. Gates, R. M. Ragan, and C. W. Randall. 1976a. *Characterization of urban runoff.* Wat. Res. Res. Inst. Rutgers Univ. New Brunswick, NY 08903.

Whipple, W., Jr., J. V. Hunter, and L. Y. Shaw. 1978. Runoff from multiple family housing. *Water Res. Bull.* 14:288-301.

Whipple, W., Jr., J. V. Hunter, and S. L. Yu. 1974a. Unrecorded pollution from urban runoff. *Journal WPCF* 46:873-885.

Whipple, W., Jr., J. V. Hunter, and S. L. Yu. 1976b. *Characterization of urban runoff--New Jersey.* Wat. Res. Inst., Rutgers Univ. New Brunswick, NJ 08903. 89 p.

Whipple, W., Jr., J. V. Hunter, and S. L. Yu. 1977. Effects of storm frequency on pollution from urban runoff. *Journal WPCF* 49:2243-2248.

Whipple, W., Jr., J. V. Hunter, S. L. Yu, J. P. Hewitt, and J. Cirello. 1974b. *Unrecorded pollution and dynamics of biochemical oxygen demand.* New Jersey Wat. Res. Res. Insti. Rutgers Univ. New Brunswick, NJ 08903.

Wilber, W. G., and J. V. Hunter. 1975. Contributions of metals resulting from stormwater runoff and precipitation in Lodi, New Jersey. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc. Proc. 20.* pp. 45-54.

Wilhm, J. L. 1970. Range of diversity index in benthic macroinvertebrate population. *Journal WPCF* 42:R221-224.

Wilhm, J. L., and T. C. Dorris. 1968. Biological parameters for water quality criteria. *Biosci.* 18:477-480.

Wilson, R. S., and J. D. McGill. 1977. A new method of monitoring water quality in a stream receiving sewage effluent, using chironomid papal exuviae. *Wat. Res.* 11:959-962.

Witt, J. M., J. W. Gillette, and J. Wyatt (Ed.). 1979. *Terrestrial microcosms and environmental chemistry.* Proc. of Symp. on Terrestrial Microcosms and Environmental Chemistry. Nat. Sci. Found. RA 79-0026. Washington, DC 20550. 147 p.

Woodwell, G. M. 1970. Effects of pollution on the structure and physiology of ecosystems. *Sci.* 168:429-433.

Wu, J. S., and R. C. Ahlert. 1978. Assessment of methods for computing storm runoff loads. *Wat. Res. Bull.* 14:429-439.

Yu, S. L., W. Whipple, Jr., and J. V. Hunter. 1975. Assessing unrecorded organic pollution from agricultural, urban, and wooded lands. *Wat. Res.* 9:849-852.

Zand, S. M. 1976. Indexes associated with information theory in water quality. *Journal WPCF* 48:2026-2031.

SECTION IX

BIBLIOGRAPHY

American Public Works Association. 1969. *Water pollution aspects of urban runoff*. FWPCA. WP-20-15. 272 p.

AWRA Research Committee. 1978. Some research and information needs in surface water quality. *Wat. Res. Bull.* 14:517-523.

Bailey, B. H., R. H. Ramsey, B. J. Claborn, R. M. Sweazy, and D. M. Wells. 1975. *Variation of urban runoff quality and quantity with duration and intensity of storms -- phase III: final report; Vol. 3.* OWRT Project No. B-177-TEX. Water Resources Center; Texas Tech Univ., Lubbock, TX. 24 p.

Bayne, B. L. 1973. Aspects of physiological stress in estuarine conditions. *Biol. J. Linnean Soc.* 6:360.

Beeton, A. M., P. K. Kovacic, and A. S. Brooks. 1976. *Effects of chlorine and sulfate reduction on Lake Michigan invertebrates.* Center for Great Lakes Studies, University of Wisconsin-Milwaukee. EPA 600/3-76-036. U. S. EPA Duluth, MN 55804. 132 p.

Bell, D. E., B. J. Claborn, R. M. Sweazy, R. E. Peterson, R. H. Ramsey, and D. M. Wells. 1975. *Variation of urban runoff quality and quantity with duration and intensity of storms -- phase III: final report; Vol. 2.* OWRT Project No. B-177-TEX. Water Resources Center; Texas Tech Univ. Lubbock, TX. 50 p.

Bergersen, E. P., and D. L. Galat. 1975. Coniferous tree bark: a light weight substitute for limestone rock in barbeque basket microinvertebrate samplers. *Water Res.* 9:729-731.

Betson, R. P., and W. McMaster. 1975. Nonpoint source mineral water quality model. *Journal WPCF* 47:2461-2473.

Birge, W. J., J. A. Black, A. G. Westerman, P. C. Francis and J. E. Hudson. 1977. *Embryopathic effects of waterborne and sediment accumulated cadmium, mercury and zinc on reproduction and survival of fish and amphibian populations in Kentucky.* Water Resources Research Institute. University of Kentucky, Lexington, KY. 28 p.

Birge, W. J., J. J. Just, A. G. Westerman, J. A. Black, and O. W. Roberts. 1975. *Sensitivity of vertebrate embryos to heavy metals as a criterion* *of water quality. Phase II. Bioassay procedures using developmental stages as test organisms.* Water Resources Institute. University of Kentucky. Lexington, KY. 36 p.

Bloomfield, J. A., R. A. Park, D. Scavia, and C. S. Zahorcak. 1973. Aquatic modeling in the eastern deciduous forest biome, U. S. International Biological Program. *In: Modeling the Eutrophication Process.* (E. J. Middlebrooks, *et al.*, eds.) Ann Arbor Science Publ. Ann Arbor, MI. pp. 139-158.

Bowdre, J. H., and N. R. Krieg. 1974. *Water quality monotoring: bacteria as indicators.* Bulletin 69. Virginia Water Resources Research Center, Virginia Polytechnic Institute and State University. Blacksburg, VA. 20 p.

Brockway, C. E., and K. P. Grover. 1978. *Evaluation of urbanization and changes in land use on the water resources of mountain valleys.* Idaho Water Resources Research Institute, Univ. of Idaho, Moscow, ID. 105 p.

Brownlee, R. C., T. A. Austin, and D. M. Wells. 1970. *Variation of urban runoff with duration and intensity of storms.* Interim report. Water Resources Center; Texas Tech Univ. Lubbock, TX. 67 p.

Bucci, S. A., C. A. Carlozzi, and B. G. Oakland. 1976. *A modeling approach for selecting and measuring the ecological impacts of natural and cultural nutrient influences on lakes and their watersheds.* OWRT Pro. A-071-MASS. Water Resources Research Center. Univ. of Massachusetts. Amherst, MA 01002. 32 p.

Buikema, A. L., Jr., C. L. See, and J. Cairns, Jr. 1977. *Rotifer sensitivity to combinations of inorganic water pollutants.* Bulletin 92. Water Resources Research Center, Virginia Polytechnic Inst. and State Univ., Blacksburg, VA. 42 p.

Buikema, A. L. Jr., J. Cairns, Jr., and G. W. Sullivan. (Undated) *Rotifers as monitors of heavy metal pollution in water.* Bulletin 71. Water Resources Research Center, Virginia Polytechnic Inst. and State Univ., Blacksburg, VA. 74 p.

Bureau of Water Management. 1971. *Chlorinated municipal waste toxicities to rainbow trout and fathead minnows.* EPA Grant No. 18050 GZZ. Michigan Dept. of Natural Resources. Lansing, MI 48926. 49 p.

Cairns, J., Jr. ed. 1971. *The structure and function of freshwater microbial communities.* Amer. Micros., ASTM, Philadelphia, PA. 301 p.

Cairns, J., Jr. 1974. Environmental quality indicators for aquatic ecosystems. *In: Environ. Plan. Sem., Proc., Drexel Univ., Philadelphia, PA.* pp. 47-61.

Cairns, J., Jr. 1974. Indicator species *vs.* the concept of community structure as an index of pollution. *Wat. Res. Bull.* 10:338-347.

Cairns, J., J. S. Crossman, K. L. Dickson, and E. E. Herricks. 1971. The effect of major industrial spills upon stream organisms. *26th Industrial Waste Conference*. Purdue Univ. Part One. pp. 156-170.

Cairns, J., Jr., and K. L. Dickson. 1972. *An ecosystematic study of the South River, Virginia*. Bulletin 54. Water Resources Research Center. Virginia Polytechnic Inst. and State Univ., Blacksburg, VA. 104 p.

Cairns, J., Jr., K. L. Dickson, and E. E. Herricks. eds. 1975. *Recovery and restoration of damaged ecosystems*. University Press of Virginia. Charlottesville, VA. 531 p.

Cairns, J., Jr., G. R. Lanza, and B. C. Parker. 1972. Pollution related structural and functional changes in aquatic communities with emphasis on freshwater algae and protozoa. *Proc. Acad. Nat. Sci. Phila.* 124: 79-127.

Canale, R. P. 1976. *Modeling biochemical processes in aquatic ecosystems*. Ann Arbor Science Publ., Ann Arbor, MI. 390 p.

Carey, G. W., L. Zobler, M. R. Greenberg, and R. M. Hordon. 1972. *Urbanization, water pollution, and public policy*. Center for Urban Policy Research, Rutgers, Univ., The State Univ. of New Jersey, New Brunswick, NJ. 214 p.

Carlsson, L., and J. Falk. 1978. *Urban Hydrology in Sweden - an inventory of the problems and their costs*. Report no. 3017. Department of Water Resources Engineering, Lund Institute of Technology, University of Lund.

Chang, S. L. 1972. *Zoomicrobial indicators of water pollution*. National Environmental Research Center. PCA021. MF-A01. Cincinnati, OH. 21 p.

Cherkauer, D. S. 1975. The hydrologic response of small watersheds to suburban development: observations and modeling. *In: Urbanization and Water Quality Control*. Annual Symp. Amer. Wat. Res. Assoc., Proc. No. 20. pp. 110-119.

Chutter, F. M. 1978. Discussion: Application of a new coefficient of similarity to pollution surveys. C. F. Pinkham and J. G. Pearson. *Journal WPCF* 50:791-792.

Cochran, G. F., D. L. Fitzsimmons, D. D. Fowler, J. R. Garrett, J. Leland, and L. Reichert. 1977. *Management of water quality in an arid urban environment: A study of Las Vegas Valley, Nevada*. Water Resources Center, Desert Research Institute, Univ. of Nevada, Reno, NV. 327 p.

Colston, N. V., and A. N. Tafuri. 1975. Urban land runoff considerations. *In: Urbanization and Water Quality Control*. Annual Symp. Wat. Res. Assoc., Proc. No. 20. pp. 120-128.

Cox, W. E., and W. R. Walker. 1975. Water quality rights in an urbanizing environment. *In: Urbanization and Water Quality Control*. Annual Symp.

Amer. Wat. Res. Assoc., Proc. No. 20. pp. 252-258.

Crossman, J. S., J. Cairns, Jr., and R. L. Kaesler. 1973. *Aquatic invertebrate recovery in the Clinch River following hazardous spills and floods.* Bulletin 63. Water Resources Research Center, Virginia Polytechnic Inst. and State Univ., Blacksburg, VA. 66 p.

Cummins, K. W., K. W. Cummins, J. J. Klug, R. G. Wetzel, R. C. Petersen, K. F. Saburkropp, B. A. Manny, J. C. Wuycheck, and F. O. Howard. 1973. Organic enrichment with leaf leachate in experimental lotic ecosystems. *BioSci.* 22:719-722.

Cvancara, A. M. 1975. *Distribution and ecology of river mollusks in central and western North Dakota.* Water Resources Research Institute. North Dakota State Univ., Fargo, ND. 5 p.

Daniel, T. C., P. E. McGuire, G. D. Bubenger, F. W. Madison, and J. G. Kourad. 1978. Assessing the pollutional load from nonpoint sources: Planning considerations and a description of an automated water quality monitoring program. *Environ. Manag.* 2:55-65.

DeFilippi, J. A., and C. S. Shih. 1971. Characteristics of separated storm and combined sewer flows. *Journal WPCF* 43:2033-2058.

Dickson, J. D. 1972. Changing priorities in urban water resource management. *In: Watersheds in Transition.* A.C. Csallany, T. G. McLaughlin, and W. D. Strifflex, eds. Amer. Wat. Res. Assoc., Urbana, IL. pp. 377-379.

Dickson, K. L., John Cairns, Jr., and R. J. Livingston, eds. 1977. *Biological data in water pollution assessment: quantitative and statistical analyses.* Amer. Soc. for Testing and Materials. Philadelphia, PA 19103. 184 p.

Dills, G. G., and D. T. Rogers, Jr. 1972. *Aquatic biotic community structure as an indicator of pollution.* Alabama University. Geological Survey of Alabama. Tuscaloosa, AL. 25 p.

Donigian, A. S., Jr., and N. H. Crawford. 1976. *Modeling pesticides and nutrients on agricultural lands.* EPA-600/2-76-043. U. S. EPA. Athens, GA 30601. 318 p.

Donigian, A. S., Jr., and N. H. Crawford. 1977. *Simulation of nutrient loadings in surface runoff with the NPS model.* EPA-600/3-77-065. U. S. EPA. Athens, GA 30601. 110 p.

Duckstein, L., and C. C. Kisiel. 1976. Water quality control under uncertainty: optimal stopping rules for sampling. *J. Hydrol.* 28:331-342.

Edwards, R. W., K. Benson-Evans, and M. A. Learner. 1972. *Biological survey of the River Taff.* Water Pollution Control (London) 71:144-185.

Egloff, D. A., and W. H. Brakel. 1973. Stream pollution and a simplified diversity index. *Journal WPCF* 45:2269-2275.

Ferris, V. R., J. M. Ferris, and C. A. Callahan. 1972. *Nematode community structure; a tool for evaluating water resource environments.* Water Resources Research Institute. Purdue University. Lafayette, IN. 40 p.

Feuerstein, D. L., and A. O. Friedland. 1975. Pollution in San Francisco's urban runoff. *In: Urbanization and Water Quality Control, Annual Symposium of the Amer. Wat. Res. Assoc., Proc. 10.* pp. 85-94.

Gates, C. D. 1976. *Characterization of urban runoff - New York.* Cornell University; Holister Hall. Ithaca, NY 14853. 65 p.

Gevirtz, J. L., and P. G. Rowe. 1977. Natural environmental impact assessment: A rational approach. *Environ. Manag.* 1:213:226.

Gibson, M., R. H. Ramsey, B. J. Claborn, R. M. Sweazy, and D. M. Wells. 1975. *Variation of urban runoff quality and quantity with duration and intensity of storms - phase III: final report vol. 1.* OWRT Project No. B-177-TEX. Water Resources Center; Texas Tech University. Lubbock, TX. 71 p.

Graham, P. H. 1978. *Report to Congress on control of combined sewer overflow in the United States.* EPA-430/9-78-006. U. S. EPA. 226 p.

Guthrie, R. K., R. N. Ferebee, and D. S. Cherry. 1974. *Biologic detection and control of water pollution.* Water Resources Research Institute. Clemson University, Clemson, SC 29631. 39 p.

Hardt, R. A., and S. J. Burges. 1976. *Some consequences of area wide runoff control strategies in urban watersheds.* Technical Report No. 48. Charles W. Harris Hydraulics Laboratory, University of Washington, Seattle, WA 98195. 81 p.

Hardy, R. W., *et al.* 1977. *The impact of urbanization on New England lakes.* U.S. Department of the Interior. Office of Water Research and Technology. Washington, DC. 167 p.

Hazel, C. 1976. *Assessment of effects of altered stream flow characteristics on fish and wildlife. Part B: California.* Final Report. Jones and Stokes, Inc. Sacramento, CA 95814.

Heany, J. P., and R. H. Sullivan. 1971. Source control of urban water pollution. *Journal WPCF* 43:571-579.

Hendrix, P. 1979. (Draft) *Assessment of impacts on freshwater biota.* EPA report in preparation. E. G. Farnworth. University of Georgia, Institute of Ecology. Athens, GA. 82 p.

Herricks, E. E., and J. Cairns, Jr. (Undated) *Rehabilitation of streams receiving acid mine drainage.* Bulletin 66. Water Resources Research Center, Virginia Polytechnic Inst. and State Univ., Blacksburg, VA. 284 p.

Hinton, D. E., M. W. Kendall, and J. C. Koenig. 1973. *Enzyme and tissue alterations in fishes: a measure of water quality.* Water Resources Research Institute. University of Kentucky. Lexington, KY. 87 p.

Hordon, R. M. 1972. Changing watersheds in metropolitan areas: A statistical analysis of selected basins in New Jersey. *In: Watersheds in Transition.* A. C. Csallany, *et al.*, eds. Amer. Wat. Res. Assoc., Urbana, IL. pp. 394-399.

Howard, W. T., and R. D. Buller. 1975. Unanswered development and water quality. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Res. Assoc. Proc. No. 20. pp. 129-139.

Huff, F. A. 1976. Relation between atmospheric pollutions, precipitation, and streamwater quality near a large urban-industrial complex. *Water Res.* 10:945-953.

Huff, D. D., J. F. Koonce, W. R. Ivanson, P. R. Weiler, E. H. Dettman, and R. F. Harris. 1973. Simulation of urban runoff, nutrient loading, and biotic response of a shallow eutrophic lake. *In: Modeling the Eutrophication Process.* (E. J. Middlebrooks, *et al.*, eds.) Ann Arbor Science Publ., Ann Arbor, MI. pp. 33-56.

Hughes, B. D. 1978. The influence of factors other than pollution on the value of Shannon's diversity index for benthic macro-invertebrates in streams. *Wat. Res.* 12:359-364.

Hunter, J. V., S. L. Yu, and W. Whipple, Jr. 1975. Measurement of urban runoff petroleum. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 162-168.

Hurlburt, S. H. 1978. The measurement of niche overlap and some relatives. *Ecol.* 59:67-77.

Inhaber, H. 1975. International conference on environmental sensing and assessment. *In: International Conference on Environmental Sensing and Assessment, Volume 2.* Institute of Electrical and Electronics Engineers, Inc., New York, NY pap. 31-1, 3 p.

James, L. D., A. C. Benke, and H. L. Ragsdale. 1975. *Integration of hydrologic, economic, ecologic, social and well-being factors in planning flood control measures for urban streams.* Georgia Institute of Technology. NTIS. PB-241-385. 203 p.

Johnston, R. F., P. W. Frank, and C. D. Michener, eds. 1978. *Ann. Rev. Ecol. Syst.* Vol. 9. 618 p.

Kaesler, R. L. 1978. *Diversity of stream communities under conditions of pollutional stress.* Pro. C-6007 Kansas Water Resources Research Institute, University of Kansas, Lawrence, KS 66045.

Krisek, R. J., and E. F. Mosonyi, eds. 1974. *Water resources instrumentation: Vol. 1. measuring and sensing methods.* Ann Arbor Science Publ., Ann Arbor, MI. 596 p.

Krisek, R. J., and E. F. Mosonyi, eds. 1974. *Water resources instrumentation: Vol 2. data acquisition and analysis.* Ann Arbor Science Publ., Ann Arbor, MI. 593 p.

Kuhner, J., and M. Shapiro. 1976. *Urban runoff pollution control - state-of-the-art.* EEI p. 221.

Kupryano, V. V. V., and B. G. Skakal'skiy. 1973. Urbanization and its effects on regimen and quality of surface waters. *Vodnyye Resursy* 2:172-182.

LaGrega, M. D., and J. D. Keenan. 1975. Characterization of water quality from combined sewer discharges. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 153-161.

Larson, G. L., C. E. Warren, F. E. Hutchins, L. P. Lamperti, D. A. Schlesinger, and W. K. Seim. 1978. *Toxicity of residual chlorine compounds to aquatic organisms.* EPA-600/78-023. U. S. EPA, Duluth, MN 55804. 117 p.

Leopold, L. B., F. E. Clarke, B. B. Hanshaw, and J. R. Balsley. 1971. *A procedure for evaluating environmental impact.* Geological Survey Circular 645. 13 p.

Litwin, Y. J., and A. S. Donigan, Jr. 1978. Continuous simulation of nonpoint pollution. *Journal WPCF* 50:2348-2361.

Lubinski, K. S., R. E. Sparks, and L. A. John. 1974. *The development of toxicity indices for assessing the quality of the Illinois River.* Water Resources Center. University of Illinois at Urbana-Champaign. 46 p.

Maier, W. J., and W. R. Swain. 1978. Organic carbon - a nonspecific water quality indicator for Lake Superior. *Wat. Res.* 12:523-529.

Martel, G. F., and R. T. Lackey. 1978. *A computerized method for abstracting and evaluating environmental impact statements.* Virginia Water Resources Research Center. Bulletin #105. Virginia Polytech Inst. and State Univ., Blackburg, VA. 93 p.

McAniff, R. J., and C. E. Willis. 1976. *The value of data acquisition from water quality monitoring systems.* OWRT Pro. A-086-MASS. Water Resources Research Center and Dept. of Food and Economics. Univ. of

Massachusetts, Amherst, MA 01002. 104 p.

McCuen, R. H. 1975. *Flood runoff from urban areas.* Tech. Report No. 33. University of Maryland, College Park, MD. 70 p.

McGraw, W. 1973. The HUD viewpoint of runoff and sediment management. *In: Planning and Design for Urban Runoff and Sediment Management.* C. T. Haan and B. J. Barfield, eds. College of Engineering, University of Kentucky, Lexington, KY.

McPherson, M. B. 1978. Urban runoff control master planning. *ASCE J. Wat. Res. Plan. & Mgmt. Div.* 104:223-234.

Meyers, C. D. 1977. *Energetics: systems analysis with application to water resources planning and decision making.* U. S. Army Engineer Inst. for Wat. Res., Kingman Bldg. Fort Belvoir, VA 22060. 42 p.

Moore, S. F., and D. B. McLaughlin. 1978. Design of field experiments to determine the ecological effects of petroleum in intertidal ecosystems. *Wat. Res.* 12:1091-1099.

Mount, D. I., and G. J. Putnicki. 1966. *Summary report of the 1963 Mississippi fish kill.* Thirty-first North American Wildlife and Natural Resources Conference. 8 p.

Mulkey, L. 1977. *Non-point water quality modeling in wildland management: a state-of-the-art assessment.* U. S. EPA. EPA-600/3-77-036. Vol. 1. Athens, GA 30601.

Muncy, R. J., G. J. Atchinson, R. V. Bulkley, B. W. Menzel, L. G. Perry, and R. C. Summerfelt. 1979. *Effects of suspended solids and sediment on reproduction and early life of warm water fishes: a review.* Draft. U. S. EPA. Contract No. CC80741-J. Corvallis, OR 97330. 100 p.

National Academy of Sciences. 1969. *Eutrophication: causes, consequences, correctives.* Proceedings. Washington, DC. 661 p.

Novotny, V. 1975. Effluent variability estimation for complete-mix activated sludge treatment systems. *Wat. Res.* 10:699-709.

NTIS. 1976. *Water quality modeling-hydrological and limnological systems. Vol. 1: 1964-1974.* National Technical Information Service. Springfield, VA 22161. 183 p.

NTIS. 1977. *Water quality modeling-hydrological and limnological systems. Vol. 2: 1975-1977.* National Technical Information Service. Springfield, VA 22161. 185 p.

NTIS. 1978. *Bioindicators of pollution. Vol. 2. November 1977 - October 1978.* NTIS/PS-78/1143. National Technical Information Service. Springfield, VA 22161. 43 p.

NTIS. 1978. *Effects of land use and urbanization on water resources and water quality. Vol. 1. 1971-1976.* NTIS/PS-78/0487. National Technical Information Service. Springfield, VA 22161. 184 p.

NTIS. 1978. *Effects of land use and urbanization on water resources and water quality. Vol. 2. 1977- May 1978.* NTIS/PS-78/0488. National Technical Information Service. Springfield, VA 22161. 79 p.

Oglesby, R. T., C. A. Carlson, and J. A. McCann. 1973. *River ecology and man.* Academic Press. NY. 465 p.

Olson, T. A., and T. O. Oolaug. 1972. *Lake Superior periphyton in relation to water quality.* Project No. 18050 DBM. University of Minnesota. School of Public Health, Minneapolis and Dept. of Biology. Duluth, MN 253 p.

Orsborn, J. F., and C. H. Allman, eds. 1976. *Proceedings of the symposium and specialty conference on instream flow needs; Vol. I, II.* American Fisheries Society. Bethesda, MD 20014. 657 p.

Pennsylvania State University. 1971. *Histochemical and cytophotometric assay of acid stress in freshwater fish.* EPA Grant No. DXJ 18050. Pennsylvania State University. University Park, PA 16802. 113 p.

Pesch, C. E., and D. Morgan. 1978. Influence of sediment in copper toxicity tests with the polychaete *Neanthes arenaceodentata. Wat. Res.* 12:747-751.

Pickett, E. B., and B. J. Brown. 1977. *Guidelines for monitoring and reporting demonstration projects, section 32, program streambank erosion control evaluation and demonstration act of 1974.* Instruction report H-77-1. Hydraulics Laboratory, U. S. Army Engineers, Waterways Experiment Station, P. O. Box 631, Vicksburg, MS 39180. 40 p.

Pierce, R. H., Jr., C. R. Brent, H. P. Williams, and S. G. Reeves. 1977. Pentachlorophenol distribution in a fresh water ecosystem. *Bull. Environ. Cont. Toxocol.* 18:251-258.

Pinkham, C. F. A., and J. G. Pearson. 1976. Applications of a new coefficient of similarity to pollution surveys. *Journal WPCF* 48:717-723.

Pisano, M. A. 1976. Nonpoint sources of pollution. A federal perspective. *ASCE J. Env. Eng. Div.* 102:555-565.

Pisano, W. C. 1975. Cost-effective approach for combined and storm sewer cleanup. *In: Urbanization and Water Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 219-230.

Poole, R. W. 1978. The statistical prediction of population fluctuations. *Annual Rev. Ecol. Syst.* 9:427-448.

Preul, H. C. 1974. Assessment of urban runoff quantity and quality. *In: Water Resources Instrumentation, Vol. 1. Measuring and Sensing Methods.* R. J. Krizek and E. F. Mosouyi, eds. Ann Arbor Science Publ., Ann Arbor, MI. pp. 517-537.

Radziul, J. V., D. D. Blair, P. R. Cairo, J. T. Coyle, and M. M. Pence. 1975. Effects of non-point discharge on urban stream quality. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 201-210.

Ragan, R. M., and A. J. Dietemann. 1976. *Characterization of urban runoff in the Maryland suburbs of Washington, DC.* Water Resources Center. Univ. of Maryland. College Park, MD 20742. 137 p.

Randall, C. W., F. A. Garland, T. J. Grizzard, and R. C. Hoehn. 1975. Characterization of urban runoff in the Occoquan watershed of Virginia. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 62-69.

Reed, C. 1978. Species diversity in aquatic microecosystems. *Ecol.* 59: 481-488.

Reynolds, J. H., E. J. Middlebrooks, and D. B. Porcella. 1974. Temperature toxicity model for oil refinery waste. *ASCE J. Env. Eng. Div.* 100: 557-576.

Reynolds, J. H., E. J. Middlebrooks, D. B. Porcella, and W. J. Grenney. 1975. Effects of temperature on growth constants of *Selenastrum capricornutum. Journal WPCF* 47:2420-2436.

Rice, Ramona, G. 1978. *Interrelationships of seasonal algal assemblages to the distribution of iron, manganese, cobalt, zinc, copper, and lead.* Masters Thesis. Arkansas Water Resources Research Center, Univ. of Arkansas. Fayetteville, AR. 133 p.

Roberts, M. 1972. Watersheds in the rural-urban fringe. *In: Watersheds in Transition.* Csallany, A. C., *et al.*, eds. Amer. Wat. Res. Assoc. Urbana, IL. pp. 388-393.

Rosenberg, D. M., and A. P. Wiens. 1978. Effects of sediment addition on macrobenthic invertebrates in a northern Canadian river. *Wat. Res.* 12:753-763.

Saunders, G. W. 1971. Carbon flow in the aquatic system. *In: The Structure and Function of Fresh-water Microbial Communities.* Amer. Micros. Soc. Symp. p. 31.

Schneider, R. R. 1978. Planning diffuse pollution control: an analytical framework. *Wat. Res. Bull.* 14:322-336.

Segeren, W. A. 1978. *Urbanization and water management.* Proc. of Technical Meeting 34. Committee for Hydrological Research. Tno, 24. The Hague, Netherlands. p. 7.

Sheldon, A. L. 1972. A Quantitative approach to the classification of inland waters. *In: Natural Environments.* J. V. Krutilla, ed. The John Hopkins Univ. Press. Baltimore, MD. pp. 205-261.

Shelton, S. P. 1978. Water quality modeling in a low flow stream. *Journal WPCF* 50:2289-2306.

Singh, R. 1974. Water quality of urban storm runoff. *In: Water Resources Instrumentation, Volume I. Measuring and Sensing Methods.* R. J. Krizek and E. F. Mosonyi, eds. Ann Arbor Science Publ., Inc., Ann Arbor, MI. pp. 577-589.

Sliter, J. T. 1976. Focusing on nonpoint sources. *Journal WPCF* 48:3-6.

Smith, J. H., W. R. Mabey, N. Bohonos, B. R. Holt, S. S. Lee, T.-W. Chou, D. C. Bomberger, and T. Mill. 1977. *Environmental pathways of selected chemicals in freshwater systems. Part 1: Background and experimental procedures.* EPA 600/7-77-113. Athens, GA 30605. 81 p.

Sorensen, D. L., M. M. McCarthy, E. J. Middlebrooks, and D. B. Porcella. 1977. *Suspended and dissolved solids effects on freshwater biota: a review.* U. S. EPA. EPA-600/3-77-042. 64 p.

Sparks, R. E. 1977. *Effects of sediment on aquatic life.* Illinois Natural History Survey. River Research Laboratory. Havana, IL 62644. Manuscript.

Sprague, J. B. 1969. Measurement of pollutant toxicity to fish. I. Bioassay methods for acute toxicity. *Wat. Res.* 3:793-821.

Stoner, J. D. 1978. *Water quality indices for specific water uses.* U. S. Geological Survey Circular 770. Arlington, VA. 12 p.

Sullivan, R. H., and M. J. Manning. 1976. Impact and control of urban stormwater runoff. *In: Proc. Conf. on Nonpoint Sources of Water Pollution: Problems, Policies, and Prospects.* Purdue Univ., Water Resources Research Center, West Lafayette, IN. 159 p.

Sutherland, R., and R. McCuen. 1976. A mathematical model for estimating pollution loadings in runoff from urban streets. *In: Mathematical models for environmental problems.* C. A. Brebbia, ed. Pentech Press Limited, London. 537 p.

Talling, J. F., and J. Rzoslca. 1967. The development of plankton in relation to hydrological regime in the Blue Nile. *J. Ecol.* 55:637-662.

Talsma, A. R. 1972. *The characterization and influence of domestic drain effluents on the Red Cedar River.* M. S. Thesis. Dept. Fisheries and Wildlife, Mich. St. Univ. 119 p.

Texas Christian University. 1971. *Industrial wastes: effects on Trinity River ecology, Fort Worth, Texas.* Project No. 18050 DBB. Texas Christian Univ., Fort Worth, TX 76129. 163 p.

Thelen, E., and L. F. Howe. 1978. *Porous pavement; including the principles of development and a porous pavement design manual.* Franklin Institute Press. Philadelphia, PA. 98 p.

Trama, F. B., and R. C. Ahlert. 1978. *Phosphorus source management for eutrophic lakes. Phase 1: Tributary phosphorus loading.* Rutgers Univ., New Brunswick, NJ.

Trotta, P. D., J. W. Labadie, and N. S. Grigg. 1977. Automatic control strategies for urban stormwater. *ASCE J. Hydr. Div.* 103:1443-1459.

Tsai, C., and J. A. Tompkins. 1974. *Survival time and lethal exposure time for the Blacknose Dace exposed to free chlorine and chloramine solutions.* Water Resources Research Center. Univ. of Maryland. College Park, MD. 22 p.

USEPA. 1977. *Nonpoint source program, status report.* Corvallis Environmental Research Laboratory. Corvallis, OR 97330. 8 p.

Uttormark, P. D., J. D. Chapin, and K. M. Green. 1974. *Estimating nutrient loadings of lakes from non-point sources.* U. S. EPA. EPA-660/3-74-020. 112 p.

Verdi, D. A., C. C. Clarkson, and D. D. Adrian. 1976. *River basin water quality sampling.* OWRT Pro. A-083-MASS Water Resources Research Center. Univ. of Massachusetts. Amherst, MA 01002. 110 p.

Vernberg, W. B., and P. J. DeCoursey. 1977. *Effect of sublethal metal pollutants on the fiddler crab* Uca pugilator. U. S. EPA. EPA-600/3-77-024. 140 p.

Welch, P. C. 1978. *Research needs relating to water/land problems in urbanizing areas.* Special report No. 8. Water Resources Center, Univ. of Illinois, Urbana-Champaign, IL. 78 p.

Welch, P. C., ed. 1978. *Water land problems in urbanizing areas.* Proceedings of the annual meeting of the Water Resources Centers. Univ. of Illinois at Urbana-Champaign. 78 p.

Wells, R. 1973. Impact zoning: A framework for innovative design. *In: Planning and design for urban runoff and sediment management.* C. T. Haan and B. J. Barfield, eds. UKY TR72-73-CEEDS College of Engineering, Univ. of Kentucky, Lexington, KY.

Wells, D. M., T. A. Austin, and B. C. Cook. 1971. *Variation of urban runoff with duration and intensity of storms.* Water Resources Center. Texas Tech Univ., Lubbock, TX. 139 p.

Whipple, W., Jr. 1975. Water quality control: a re-evaluation. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 286-291.

Whipple, W., Jr., J. V. Hunter, R. C. Ahlert, and S. L. Yu. 1978. *Estimating runoff pollution from large urban areas - the Delaware Estuary.* Water Resources Research Institute, Rutgers Univ., New Brunswick, NY 08903. 73 p.

Whitton, B. A., ed. 1975. *River ecology.* University of California Press. Berkeley. 725 p.

Widmer, H. M., *et al.* 1976. *Effects of an urban road system on lead content of an urban water supply source.* Water Resources Center, Univ. of Massachusetts. Amherst, MA. 18 p.

Wilber, W. G., and J. V. Hunter. 1977. Aquatic transport of heavy metals in the urban environment. *Wat. Res. Bull.* 13:721-734.

Zogorski, J. S., W. G. Characklis, R. H. Schumer, and F. Gaudet. 1975. Temporal characteristics of stormwater runoff: an overview. *In: Urbanization and Water Quality Control.* Annual Symp. Amer. Wat. Res. Assoc. Proc. No. 20. pp. 100-109.

TECHNICAL REPORT DATA
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1. REPORT NO. EPA-600/3-80-032	2.	3. RECIPIENT'S ACCESSION NO.
4. TITLE AND SUBTITLE CHARACTERISTICS OF NONPOINT SOURCE URBAN RUNOFF AND ITS EFFECTS ON STREAM ECOSYSTEMS		5. REPORT DATE February 1980 issuing date 6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) Donald B. Porcella Darwin L. Sorensen		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS Utah Water Research Laboratory Utah State University Logan Utah 84322		10. PROGRAM ELEMENT NO. C32B/A 11. CONTRACT/GRANT NO. Req. CC81224-J
12. SPONSORING AGENCY NAME AND ADDRESS Environmental Research Laboratory--Corvallis Office of Research and Development Environmental Protection Agency Corvallis Oregon 97330		13. TYPE OF REPORT AND PERIOD COVERED Final - Literature review 14. SPONSORING AGENCY CODE EPA/600/02

15. SUPPLEMENTARY NOTES

Project Officer: Kenneth W. Malueg, Environmental Research Laboratory
Corvallis, Oregon 97330 (503)757-4761 (FTS) 420-4761

16. ABSTRACT

Literature on urban nonpoint source runoff was surveyed to determine the magnitude of the effects of that source of contaminants to stream ecosystems. Very little information was available on ecosystem effects although extensive literature on all aspects associated with contaminant loading was found. However, urban NPS runoff probably exerts unique effects on stream communities because of its random magnitude/impulse loading of contaminants. Because control of NPS runoff is expensive, it is important to determine its actual impacts on stream communities. Ecological literature provided a basis for evaluating such impacts based on benthic invertebrate biomass and diversity, measurement of community primary production and respiration, carbon cycling, and variables related to the contaminant concentrations in the stream. We concluded that a stochastic approach for assessing the impacts would be most feasible for evaluating impacts of urban NPS runoff.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
Water quality Urbanization Stream ecosystems	Urban nonpoint source runoff	06/F

18. DISTRIBUTION STATEMENT Release to public	19. SECURITY CLASS (This Report) Unclassified	21. NO. OF PAGES 110
	20. SECURITY CLASS (This page) Unclassified	22. PRICE

EPA Form 2220-1 (Rev. 4-77)