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Vol. III: Resource Document for Environmental Assessments

Final Report

March, 1985

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16. Abstract

This resource document is intended to serve as a user tool to supplement the Procedural Guidelines Manual (Volume IV). State highway agencies can use these resources to more comprehensively address the effects of stormwater runoff in environmental documents (i.e., EIS's and EA's). This document provides a critical summary and review of the technical literature on hydrological, water quality, sediment, and biological impacts of runoff from operating highways. Major pollutant categories include oxygen - consuming materials, nutrients, bacteria, road salt, petroleum hydrocarbons, and metals.

The titles of the other volumes of this report are:

FHWA/RD-	Volume	Title
84/062	I	Executive Summary
84/063	II	Research Report
84/064	III	Resource Document for Environmental Assessments
84/065	IV	Procedural Guidelines for Environmental Assessments
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METRIC CONVERSION FACTORS

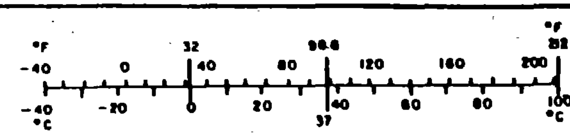
Approximate Conversions to Metric Measures

Symbol	When You Know	Multiply by	To Find	Symbol
		LENGTH		
in	inches	*2.5	centimeters	cm
ft	feet	30	centimeters	cm
yd	yards	0.9	meters	m
mi	miles	1.6	kilometers	km
		AREA		
in^2	square inches	6.5	square centimeters	cm^2
ft^2	square feet	0.09	square meters	m^2
yd^2	square yards	0.8	square meters	m^2
mi^2	square miles	2.6	square kilometers	km^2
	acres	0.4	hectares	ha
		MASS (weight)		
oz	ounces	28	grams	g
lb	pounds	0.45	kilograms	kg
	short tons (2000 lb)	0.9	tonnes	t
		VOLUME		
tsp	teaspoons	5	milliliters	ml
Tbsp	tablespoons	15	milliliters	ml
fl oz	fluid ounces	30	milliliters	ml
c	cups	0.24	liters	l
pt	pints	0.47	liters	l
qt	quarts	0.95	liters	l
gal	gallons	3.8	liters	l
ft^3	cubic feet	0.03	cubic meters	m^3
yd^3	cubic yards	0.76	cubic meters	m^3
		TEMPERATURE (exact)		
°F	Fahrenheit temperature	5/9 (after subtracting 32)	Celsius temperature	°C

*1 in = 2.54 (exactly). For other exact conversions and more detailed tables, see NBS Misc. Publ. 286, Units of Weights and Measures, Price $2.25, SD Catalog No. C13.10:286.

Approximate Conversions from Metric Measures

Symbol	When You Know	Multiply by	To Find	Symbol
		LENGTH		
mm	millimeters	0.04	inches	in
cm	centimeters	0.4	inches	in
m	meters	3.3	feet	ft
m	meters	1.1	yards	yd
km	kilometers	0.6	miles	mi
		AREA		
cm^2	square centimeters	0.16	square inches	in^2
m^2	square meters	1.2	square yards	yd^2
km^2	square kilometers	0.4	square miles	mi^2
ha	hectares (10,000 m^2)	2.5	acres	
		MASS (weight)		
g	grams	0.035	ounces	oz
kg	kilograms	2.2	pounds	lb
t	tonnes (1000 kg)	1.1	short tons	
		VOLUME		
ml	milliliters	0.03	fluid ounces	fl oz
l	liters	2.1	pints	pt
l	liters	1.06	quarts	qt
l	liters	0.26	gallons	gal
m^3	cubic meters	35	cubic feet	ft^3
m^3	cubic meters	1.3	cubic yards	yd^3
		TEMPERATURE (exact)		
°C	Celsius temperature	9/5 (then add 32)	Fahrenheit temperature	°F



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INTRODUCTION

REPORT ORGANIZATION

The purpose of this report volume is to provide the user with a comprehensive resource document to aid in the preparation of environmental documents, especially NEPA related documents such as environmental impact statements (EIS) and environmental assessments (EA). The principal user will in most likelihood be the state highway agency (HA) staff, which in most cases will have primary responsibility for preparation of the documents. Other users could well be state and federal agencies entrusted with EIS review responsibility or direct regulatory authority over certain highway activities. This volume is intended to serve as a complimentary resource to Volume IV (Procedural Guidelines for Environmental Assessments) which is another user oriented manual designed to assist the HA in making the impact projection for a proposed highway project. It includes information on regulatory factors such as water use classifications, standards, and water quality protection programs. Assessment methodologies are also described.

This report is organized into several major areas of potential impact: hydrological, water quality, sediment, and biological effects. The section on hydrological impacts will center on flow and rainfall/runoff effects in lotic systems (streams, rivers, and estuaries), whereas hydrodynamic effects will be emphasized in lentic systems (lakes). Water quality impacts will be considered in terms of both impact type and magnitude as defined below. Documented accumulations of metals, salts, and hydrocarbons in sediments will be presented, since movement of pollutants up the food chain often begins in the sediment. Biological impacts will include documented responses of aquatic biota to stormwater.

Receiving water effects, which will be included in the above impact areas, relate only to the normal operation and maintenance of highway facilities. Effects due to highway construction activities or hazardous material spills will not be considered.

A comprehensive field monitoring program was sponsored by FHWA as part of the same contract from which this resource document is derived. This program involved monitoring of all four impact components, and detailed results are given in Volume II of this report. Since these results represent the most comprehensive analysis of effects of highway stormwater runoff, a brief discussion is also provided in this volume under each pertinent impact category.

Finally, a qualitative summary of all impact areas is given, and a glossary of technical terms is provided.

PROBLEM RECOGNITION

Several investigations have identified operating highways as a potential source of a variety of pollutants to adjacent surface and subsurface waters through mechanisms of the hydrologic cycle (1,2,3). General pollutant categories identified as constituents of highway runoff include particulates, nutrients, oxygen-demanding materials, metals, petroleum hydrocarbons, pesticides/ herbicides, and pathogenic indicator bacteria. These pollutants are summarized in Tables 1 to 3. However, the significance of the intermittent discharge of highway pollutants on the quality and ecology of receiving waters is not well-documented. This information is required in the preparation and review of environmental impact statements (EIS) for proposed highway developments by both federal and state agencies. Together with the assessment is the need to be able to identify with confidence where mitigation is needed and where it is not. The major objectives of the EIS are to

Table 1. Summary of highway runoff quality data for six monitoring sites (1) and typical urban runoff quality based on data from 28 cities (2).

	Highway Runoff (1)				
	Pollutant concentration, mg/l		Pollutant loading, lb/acre/in-runoff		Urban runoff, median EMC,(a) mg/l
	Average	Range	Average	Range	
pH	-	6.5-8.0	-	6.5-8.0	-
Total solids	1,147	145-21,640	260	33-4,910	-
Total volatile solids	242	26-1,522	55	5.9-345	-
Suspended solids	261	4-1,656	59	0.9-375	100
Volatile suspended solids	77	1-837	17	0.2-190	-
Biochemical oxygen demand (5-day)	24	2-133	5.4	0.5-30	9
Total organic carbon	41	5-290	9.3	1.1-66	-
Chemical oxygen demand	14.7	5-1,058	33	1.1-240	65
Total Kjeldahl nitrogen	2.99	0.1-14.0	0.68	0.02-3.17	1.50
Nitrite plus nitrate	1.14	0.01-8.4	0.26	0.002-1.90	0.68
Total phosphorus	0.79	0.05-3.55	0.18	0.011-0.81	0.33
Chloride	386	5-13,300	88	1.1-3,015	-
Lead	0.96	0.02-13.1	0.22	0.005-2.97	0.144
Zinc	0.41	0.01-3.4	0.093	0.002-0.771	0.160
Iron	10.3	0.10-45.0	2.34	0.023-10.2	-
Copper	0.10	0.01-0.88	0.023	0.002-0.199	0.034
Cadmium	0.04	0.01-0.40	0.009	0.002-0.091	-
Chromium	0.04	0.01-0.14	0.009	0.002-0.032	-
Nickel	9.92	0.10-49.0	2.25	0.023-11.1	-
Mercury, $X10^{-3}$	3.22	0.13-67.0	0.73	0.029-15.2	-
Polychlorinated biphenyls, $X10^{-3}$	0.33	0.02-8.89	0.075	0.005-2.02	-
Oil and grease	9.47	1-104	2.15	0.23-23.6	-

a. Median EMC - Median Event Mean Concentration, where the EMC is defined as a flow-weighted average concentration for a discrete storm event. The values shown here represent the overall 28-site median of all site median EMC's.

Metric Units: To convert lb/acre-in to kg/ha-cm multiply by 0.442.

Table 2. Organic priority pollutants in roadside snow and runoff samples, all concentrations in nanograms/L.

Polynuclear aromatic hydrocarbons and other base/neutral organics	Roadside snow[a]		Runoff samples	
	asphalt	concrete	I-94[b]	hwy 50[c]
Naphthalene	195	123	1,800	--
2-Methylnaphthalene	883	--	--	--
1-Methylnaphthalene	668	--	--	--
Biphenyl	382	--	--	--
Acenaphthene	--	--	500	--
Dibenzofuran	810	199	--	--
Fluorene	1237	485	600	--
9-Methylfluorene	767	220	--	--
2-Methylfluorene	763	274	--	--
1-Methylfluorene	1,707	639	--	--
Dibenzothiophene	2,222	803	--	--
Phenanthrene	6,787	4,055	6,900	10,000
Anthracene	246	165	600	10,000
2-Methylanthracene	725	--	--	--
1-Methylphenanthrene	2,117	1,366	--	--
Fluoranthene	3,143	1,820	12,000	--
Pyrene	3,066	1,886	8,000	--
Benzo(a)fluorene	396	179	--	--
Benzo(b)fluorene	192	--	--	--
Benzo(a)anthracene	--	228	--	--
Triphenylene/chrysene	1,070	665	--	--
Benzo(b)fluoranthene	1,501	799	--	--
Benzo(j,k)fluoranthene	207	--	--	--
Benzo(e)pyrene	630	360	--	--
Benzo(a)pyrene	--	250	--	--
O-phenylenepyrene	240	270	--	--
Benzo(g,h,i)perylene	319	391	--	--
1,2-Benzanthracene	--	--	3,400	--
3,4-Benzofluoranthene	--	--	8,600	--
11,12-Benzofluoranthene	--	--	8,600	--
Chrysene	--	--	3,400	--
Acenaphthylene	--	--	200	--
N-Nitrosodiphenylamine	--	--	800	--
Bis (2-ethylhexyl)phthalate	--	--	19,000	15,000
Butyl benzl phthalate	--	--	5,000	--
Di-n-butyl phthalate	--	--	2,800	10,000
Diethyl phthalate	--	--	300	--
Other organics				
Phenols	--	--	1,900	18,000
Trichlorofluoromethane	--	--	200	--

a Snow samples collected at two sites along a highway south of Oslo, Norway with equal traffic volumes but different pavement types (46).

b I-94 in Milwaukee, Wi (7).

c Hwy 50 in Sacramento, Ca (7).

Table 3. Monitored bacterial counts for highway (1) and urban runoff (2) (counts per 100 ml).

Monitoring sites	Total coliform (TC) Maximum	Total coliform (TC) Minimum	Fecal coliform (FC) Maximum	Fecal coliform (FC) Minimum	Fecal streptococci coliform (FS) Maximum	Fecal streptococci coliform (FS) Minimum	Average ratio FC/FS
Milw.-Hwy. 794	600,000	3,000	>100,000	10	4,300	40	0.63
Milw-Hwy. 45	7,900,000	4,500	300,000	490	300,000	1,320	0.76
Milw.			200	>2	360	40	0.04
Harrisburg	175,000	100	>100,000	0	200,000	640	0.33
Nashville	2,900,000	1,700	260,000	150	3,520,000	3,900	0.14
Denver	>100,000	0	2,700	0	>100,000	0	0.33
Urban runoff(2)	–	–	21,000[a]		–		– –

a. This value represents the warm weather median EMC (as defined in Table 2) for 11 NURP sites. The median cold weather EMC for 9 NURP sites was 1,000 FC/100 ml (2).

quantify pollutants emanating from highway operations and to enable sound judgements as to the overall cost-benefit relationship of the facility. Accurate judgements can not be made without knowledge of the impacts of the highway pollutant loadings in receiving water systems and methods for determining these impacts.

In the early 1960's, storm-generated discharges from urbanized areas were first recognized as having potential for significant effects on the quality of surface waters (4,5). In the following years, several investigations were conducted to determine the pollutant characteristics of urban discharges (6). The passing of the 1972 amendments to the Federal Water Pollution Control Act (PL 92-500) greatly accelerated research in this area. Millions of dollars have been expended for research and development efforts on the quality of stormwater discharges, as well as investigations of feasible methods for handling and treatment. Research on the actual impacts of urban runoff in receiving waters did not begin until recently when it became apparent that costs for wet weather flow controls are astronomical. The U.S. Environmental Protection Agency (EPA) 1978 Needs Survey (7) indicated that tens of billions of dollars would be necessary to control stormwater discharges in urban areas. Such high costs have necessitated justification of these controls on the basis of the magnitude of the water quality problem, beneficial uses of the receiving water, control cost versus expected water quality improvement, and several other criteria.

In urban areas, highway runoff contributes only a small fraction of the overall stormwater pollutant loadings to surface waters. Even though pollutant concentrations in highway runoff are comparable to separate urban runoff (also shown in Tables 1 to 3), overall loadings are considerably lower due to lower highway right-of-way surface area compared to total urban watershed area for most surface waters. However, there is considerably more information available in the technical literature on urban runoff effects, especially that generated by the Nationwide Urban Runoff Program. Therefore, selected urban runoff literature will be utilized in this document in areas where specific highway effects are not documented and/or where the results are qualitatively pertinent to the highway situation.

Another significant difference between the urban and highway runoff impact scenarios is that most highways cross through rural areas. The drainage designs for these rural highways are considerably different than in urban areas. Specifically, more extensive use of flush-shoulder and grassy ditch drainage greatly reduces pollutant delivery from highway to receiving water. In contrast, curb-and-gutter drainage and sewer conveyance predominates for urban highways, making pollutant delivery ratios quite high.

DEFINITION OF IMPACT TYPE AND MAGNITUDE

Impacts on receiving water systems due to stormwater runoff can be defined in several different contexts. For the purposes of highway runoff, receiving water impacts can be defined as follows, in order of increasing magnitude:

1. Any measurable increase in pollutant concentration or load compared to background levels,

2. An increase in sediment pollutant concentration, demonstrated bioaccumulation of toxic materials, and/or subtle changes in biological communities,

3. An increase in pollutant concentration(s) sufficient to exceed acute or chronic water quality criteria established by U.S. EPA,

4. An increase in pollutant concentration(s) sufficient to cause contravention of state water quality standards, and

5. Dramatic, highly visible impacts such as fish kills, water supply taste problems, shellfish water closures, and/or severe alteration of the aquatic biological community.

From a regulatory perspective, the impact must be of the highest level of magnitude defined in items 4 and 5 in order to elicit enforcement or mitigative measures. In general, compliance with state water quality standards will provide conservative enough protection to prevent those impacts described in item 5. Such natural forces as severe droughts or floods, or the discharge of presently unregulated toxic compounds can also cause dramatic impacts. Volume IV of this report, "Procedural Guidelines for Environmental Assessments," provides the user with a detailed description of regulatory programs and their relationship to discharges of stormwater runoff from operating highways.

HYDROLOGICAL IMPACTS

Three basic types of hydrological impacts can be associated with the operation of highways. These are:

1. Changes due to watershed or drainage basin areal characteristics, ie., primarily an increase in percent imperviousness due to highway paved surfaces.

2. Changes in the "natural" hydraulic regime of a basin due to highway structural and drainage features.

3. Density stratification effects in lakes receiving deicing salts.

INCREASED BASIN IMPERVIOUSNESS

No specific documented cases of hydraulic impact due to increased basin imperviousness as a result of highway pavement <u>alone</u> were found in the technical literature. In urban, or rapidly urbanizing, areas highway projects may be viewed as contributing to the overall impact scenario, even though viewed independently the project has little effect. The only report in the literature involving highway development was for the Peachtree Creek watershed in Atlanta, GA (4). During the period of analysis the percentage of impervious land cover increased from 17 to 31 percent. The most recent developments, including the construction of a new interstate highway system, were located very near the creek and even within the floodplain. The overall effect of this urbanization was an increase in runoff volume in dry months, decreased base flow in wet months, and a dramatic increase in peak runoff from summer storms. In most rural areas, highway projects will generally not significantly alter the impervious cover of a watershed. For example, the increase in impervious area due to the construction of Wisconsin Highway 15 on the primarily rural Sugar Creek drainage basin in southeastern Wisconsin was insignificant (59 acres [24 ha] of paved highway compared to 28,639 acres [11,600 ha] of total watershed). However, since localized highway/receiving water situations could be more severe, the general effects of urbanization/increased imperviousness are discussed below.

Several typical effects of increased watershed imperviousness are cited in the literature (5,6,7,8,9,10,11,12):

1. Increase in volume of total runoff,

2. Increase in size of flood peak,

3. Decrease in lag time to discharge peak (ie., time of concentration),

4. Increase in channel size due to increased scouring from increased flows,

5. Increase in frequency of flooding,

6. Decrease in baseflow (or groundwater infiltration).

However, Brater and Sangal (9) warn against broad generalizations of urbanization impacts on hydrologic regimes based on numerical values obtained at specific study sites. They cite tremendous spatial and temporal variability in the process and recommend a case-by-case approach to the issue. An ASCE Task Committee (5) suggested that localized modifications of drainage patterns affecting infiltration, percolation, storage and discharge can be more important than increases in paved surfaces.

CHANGES IN WATERSHED HYDRAULIC REGIME

As previously mentioned, localized changes in watershed drainage patterns can have a significant effect on the hydraulic regime of a receiving water (5). For example, if large (long) sections of highway drainage are collected and discharged at a single point, a hydraulic effect could occur. Such effects have not been frequently reported in the technical literature, but are intuitively apparent. A broad-based review of the impact of highways on the hydrogeologic environment was provided by Parizek (13). Although these are impacts that initially occur at the time of highway construction, they will continue to occur during the years of highway operation. Extensive fills and deep ruts can induce the following hydrogeologic transformations:

1. Beheading of aquifers,
2. Development of extensive groundwater drains where cuts extend below water table,
3. Changes in ground and surface water divides and basin areas,
4. Reduction of induced streambed infiltration due to sedimentation,
5. Obstruction of groundwater flows by abutments, retaining walls, and sheet pilings, and,
6. Changes in runoff and recharge characteristics.

A training and design manual entitled "Highways in the River Environment: Hydraulic and Environmental Design Considerations," prepared for the FHWA by Richardson, *et al*. (14) of Colorado State University, should be of benefit to the highway planner. The manual provides much detailed knowledge of open channel hydraulics as well as specific recommendations on planning and design practices to minimize and predict hydraulic effects.

DENSITY STRATIFICATION EFFECTS DUE TO DEICING AGENTS

One notable hydrodynamic effect on receiving waters which can be caused by stormwater runoff is imposition of density stratification due to washoff of deicing agents.

Judd (15) documented this effect for a normally dimictic (two complete overturns per year) lake of glacial origin in Michigan (First Sister Lake). Morphometric data for First Sister Lake include:

1. Surface area - 3.15 ac (12,800 m^2)
2. Volume - 1.96 x 10^6 ft^3 (55,700 m^3)
3. Maximum depth - 23.6 ft (7.2 m)
4. Mean depth (volume/surface area) - 14.1 ft (4.3 m)

The lake received stormwater runoff from an adjacent residential area and portions of a 4-lane highway (Interstate 94). For two of the three years studied, heavy winter snowfalls caused large quantities of deicing salts to be discharged into the lake. For both of these years, this salt input prevented the spring overturn, but not the fall overturn (referred to as either temporary monomixis or meromixis). Judd demonstrated that salt migration into the sediments during the summer allowed the normal temperature-density relationship to predominate in fall. The spring stratification stability during temporary monomixis was calculated to be up to 8.5 times that of the year in which normal dimictic conditions prevailed.

Bubeck, et al. (16) reported a similar occurrence in normally dimictic Irondequoit Bay near Rochester, New York. A winter influx of approximately 21,000 metric tons of deicing salts imposed a vertical density gradient sufficient to prevent complete vertical mixing in spring. The fall overturn did occur, but approximately one month later than would have been predicted based on Bay conditions 30 years prior to the study period. The authors acknowledge that the time at which vertical mixing occurs is contingent upon the specific details of the autumnal cooling, which vary from year to year. Morphometric data for Irondequoit Bay include:

1. Surface area - 1,655 ac (6.7 km^2)

2. Volume - 1.63 x 10^9 ft^3 (0.046 km^3)

3. Maximum depth - 75 ft (23 m)

4. Mean depth (volume/surface area) = 22.6 ft (6.9 m)

Cherkauer and Ostenso (17) also reported winter salinity stratification in a small, shallow, artificial lake in Wisconsin (Northridge Lake) due to input of deicing salts from street runoff. However, the density gradient was not sufficient to prevent the spring overturn. The mean depth of Northridge Lake was only 6.5 ft (2 m), indicating that basin morphometry plays a large role in stratification stability.

Goldman and co-workers (18,19) investigated the aquatic impacts of deicing agent application to roads in the Sierra Nevada mountains of California. Included were sampling data from 27 stream and 8 lake stations. For one of these lakes, Putt's Lake, which is immediately adjacent to I-80, a temporary road-salt induced chemocline occurred in the winter of 1975. It had sufficient stability to overcome the natural temperature density function. Although several other lakes in the area also exhibited a weak chemocline, the temperature inversion seen in Putt's Lake did not occur elsewhere.

In addition to increased density stratification stability effects, stormwater runoff from urban areas and highways can also cause unusual density currents in lakes. Judd (15) also documented this effect for First Sister Lake in Michigan in winter. Surface runoff from asphalt-covered streets had apparently been sufficiently warmed so that it did not immediately sink to the bottom of the lake (in spite of high salt concentrations). Instead, this saline water moved across the lake beneath the ice cover at about the 3 ft (1 m) depth directly to the outflow point. Thus, the temperature of the inflowing water was higher than the lake water above and below the inflow depth.

WATER QUALITY IMPACTS

INTRODUCTION

As discussed previously, impacts on the quality of a receiving water from highway runoff can be defined in relation to either background conditions (prestorm or prior to highway construction) or to a group of criteria and standards set forth as goals by federal, state, or local authorities. Such water quality criteria and standards are typically related to a specified water use. For example, a receiving water intended for drinking water supply might have more stringent criteria for toxic materials than one intended for irrigational use only.

A list of water quality criteria has been developed by the Environmental Protection Agency (21,22). These criteria have no regulatory impact in and of themselves, but rather serve as guidance to state agencies. Table 4 provides a summary of these EPA criteria for pollutants which might be present in highway runoff. These parameters are related to certain intended water uses. Some values were taken from the so-called "Red Book" published in 1976 (22). Most of the toxic parameter values are from a superceding (1980) EPA document (21). Metals levels are often expressed as a logarithmic function with water hardness being the only independent variable. This is because toxicity bioassays have shown that freshwater aquatic organism responses to these metals are related to the hardness of the water used in the experiment. EPA issued new draft criteria (23) for certain metals in 1984 which are listed later in this section.

IMPACTS ON DISSOLVED OXYGEN LEVELS

Many authors have discussed the "potential" dissolved oxygen (DO) depletion of receiving waters as a result of discharges of stormwater.

Table 4. Summary of water quality criteria for possible constituents of highway runoff (21,22).

Constituent	Description of critical level	Year of criteria issuance	Basis for criteria or water use
Suspended solids	Should not reduce the depth of the compensation point for photosynthetic activity by more than 10% from the seasonally established norm for aquatic life	1976	Freshwater fish and other aquatic life
Ammonia	0.02 mg/l (unionized as NH_3-N)	1976	Freshwater aquatic life
Dissolved oxygen	Greater than or equal to 5.0 mg/l	1976	Freshwater aquatic life
Phosphorus	0.10 μg/l (elemental)	1976	Marine or estuarine waters
Nitrate	10 mg/l (as N)	1976	Domestic water supply (health)
Chlorides	250 mg/l	1976	Domestic water supply (welfare)
Sulfates	250 mg/l	1976	Domestic water supply (welfare)
Lead, μg/l (total recoverable)	$_e$(2.35 [ℓn (hardness)] -9.24); 24-hr average $_e$(1.22 [ℓn (hardness)] -0.47); never to exceed	1980	Freshwater aquatic life
	668 μg/l acute toxicity*	1980	Saltwater aquatic life
	25 μg/l chronic toxicity*	1980	Saltwater aquatic life
	50 μg/l	1980	Human health
Cadmium μg/l (total recoverable)	$_e$(1.05 [ℓn (hardness)] -3.73); never to exceed $_e$(1.05 [ℓn (hardness)] -8.53); 24-hr average	1980	Freshwater aquatic life

*Specified as the lowest level that toxicity has been measured for certain organisms, other organisms could be affected at lower levels.

Table 4. Summary of water quality criteria for possible consituents of highway runoff (21,22) (Continued).

Constituent	Description of critical level	Year of criteria issuance	Basis for criteria or water use
Cadmium (continued)	4.5 μg/l; 24-hr average	1980	Saltwater aquatic life
	59 μg/l; never to exceed	1980	Saltwater aquatic life
Chromium (total recoverable hexavalent)	0.29 μg/l; 24-hr average	1980	Freshwater aquatic life
	21 μg/l; never to exceed	1980	Freshwater aquatic life
	18 μg/l; 24-hr average	1980	Saltwater aquatic life
	1,260 μg/l; never to exceed	1980	Saltwater aquatic life
	50 μg/l	1980	Human health
Chromium (total recoverable trivalent)	$e^{(1.08[\ell n\,(\text{hardness})]+3.48)}$; never to exceed	1980	Freshwater aquatic life
	44 μg/l chronic toxicity*	1980	Freshwater aquatic life
	10,300 μg/l acute toxicity*	1980	Saltwater aquatic life
	170 mg/l ingestion of water & contaminated organisms	1980	Human health
	3,433 mg/l ingestion of contaminated organisms alone	1980	Human health
Copper (total recoverable)	5.6 μg/l; 24-hr average	1980	Freshwater aquatic life
	$e^{(0.94[\ell n\,(\text{hardness})]-1.23)}$; not to exceed	1980	Freshwater aquatic life
	4.0 μg/l; 24-hr average	1980	Saltwater aquatic life
	23 μg/l; not to exceed	1980	Saltwater aquatic life
	<1 mg/l	1980	Taste & odor
Nickel μg/l (total recoverable)	$e^{(0.76[\ell n\,(\text{hardness})]+1.06)}$; 24-hr average	1980	Freshwater aquatic life
	$e^{(0.76[\ell n\,(\text{hardness})]+4.02)}$; not to exceed	1980	Freshwater aquatic life

*Specified as the lowest level that toxicity has been measured for certain organisms, other organisms could be affected at lower levels.

Table 4. Summary of water quality criteria for possible constituents of highway runoff (21,22) (continued).

Constituent	Description of critical level	Year of criteria issuance	Basis for criteria or water use
Nickel (continued)	7.1 μg/l; 24-hr average	1980	Saltwater aquatic life
	140 μg/l; not to exceed	1980	Saltwater aquatic life
	13.4 μg/l; ingestion of water & contaminated organisms	1980	Human health
	100 μg/l; ingestion of contaminated organisms alone	1980	Human health
Zinc, μg/l (total recoverable)	47 μg/l; 24-hr average	1980	Freshwater aquatic life
	$e^{(0.83[\ln(\text{hardness})] + 1.95)}$; not to exceed	1980	Freshwater aquatic life
	58 μg/l; 24-hr average	1980	Saltwater aquatic life
	170 μg/l; not to exceed	1980	Saltwater aquatic life
	5 mg/l	1980	Taste & odor
Mercury (total recoverable)	0.00057 μg/l; 24-hr average	1980	Freshwater aquatic life
	0.0017 μg/l; not to exceed	1980	Freshwater aquatic life
	0.025 μg/l; 24-hr average	1980	Saltwater aquatic life
	3.7 μg/l; not to exceed	1980	Saltwater aquatic life
	144 ng/l; ingestion of water & contaminated organisms	1980	Human health
	146 ng/l; ingestion of contaminated organisms alone	1980	Human health
Iron	0.3 mg/l	1976	Domestic water supply (welfare)
	1.0 mg/l	1976	Freshwater aquatic life
Manganese	50 μg/l	1976	Domestic water supply (welfare)
	100 μg/l	1976	Ingestion of marine mollusks

*Specified as the lowest level that toxicity has been measured for certain organisms, other organisms could be affected at lower levels.

Table 4. Summary of water quality criteria for possible constituents of highway runoff (21,22) (continued).

Constituent	Description of critical level	Year of criteria issuance	Basis for criteria or water use
Arsenic	440 μg/l, not to exceed	1980	Freshwater aquatic life
	508 μg/l, acute toxicity*	1980	Saltwater aquatic life
	Zero; cancer risk levels also specified	1980	Human health
Cyanides (free cyanide)	3.5 μg/l; 24-hr average	1980	Freshwater aquatic life
	52 μg/l; not to exceed	1980	Freshwater aquatic life
	30 μg/l; acute toxicity*	1980	Saltwater aquatic life
	2.0 μg/l; chronic toxicity*	1980	Saltwater aquatic life
	200 μg/l; ingestion of water & contaminated organisms	1980	Human health
Asbestos	No toxicity data available for aquatic organisms	1980	Aquatic life
	Zero; cancer risk levels also specified	1980	Human health
PCB	0.014 μg/l; 24-hr average	1980	Freshwater aquatic life
	0.030 μg/l; 24-hr average	1980	Saltwater aquatic life
	Zero; cancer risk levels also specified	1980	Human health
Oil & Grease	0.01 times lowest continuous flow 96-hr LC_{50} for several important freshwater species; minimize sediment levels; surface waters free from floating oils	1976	Freshwater aquatic life

*Specified as the lowest level that toxicity has been measured for certain organisms, other organisms could be affected at lower levels.

Driscoll (24) has listed DO depletion as a major type of receiving water impact from urban runoff. Lager, et al. (25) also mention DO depletion as a major impact of urban stormwater on the basis of an average urban stormwater five-day biochemical oxygen demand (BOD) of 20 mg/l. They also cite two modeling evaluations which predict significant DO sags in rivers downstream of urban areas following precipitation. However, data from the recently completed Nationwide Urban Runoff Program (NURP) showed the flow-weighted mean concentration of BOD_5 to be only 9 mg/l. This mean is reflective of data from 28 cities distributed nationwide (2). Data from Gupta, et al. (1) indicate that 5-day BOD concentrations in highway runoff averaged 24 mg/l with a range from 1 to 133 mg/l. The BOD concentrations in urban stormwater and highway runoff are therefore often compared with the BOD in municipal treatment plant effluents (about 10-30 mg/l). However, actual receiving water impacts from these two BOD sources are not directly comparable for the following reasons:

1. Stormwater loadings are intermittent (acute) in comparison with continuous discharges from sewage treatment facilities.

2. The BOD and COD of stormwater is associated with particulate matter (1,26,27) while the BOD of a secondary treatment effluent can be either soluble or particulate.

3. Transport of BOD in receiving waters from the two sources differ. Stormwater particulates may tend to settle more rapidly in receiving waters.

4. Kinetics of bacterial oxidation of stormwater organic material are considerably different than the kinetics for secondary effluent (28).

Despite all of the comparisons of BOD loadings from stormwater with that from other sources and qualitative evaluations indicating the potential for DO depletion of receiving waters, information documenting the impact in receiving waters is scarce. A major reason for this lack of information is the difficulty in segregating stormwater impacts from sanitary and other sources of pollution. In a review of literature pertaining to receiving water impacts from urban runoff, Heaney et al. (29) cite some examples of DO impact evaluations, however the majority of these studies pertained to combined sewer overflow impacts. None of the 28 NURP sites specifically identified a low DO condition resulting from urban runoff (2). In fact, several studies have shown increases in DO concentration during wet weather. The effects of urban stormwater runoff on two streams near Seattle, Washington likewise demonstrated no DO impact (30). The Miller and Des Moines Creek watersheds are 5,230 and 3,730 acres (2116 and 1509 ha), respectively, and have land use distribution of about 55 percent residential, 10 percent commercial, with the remaining areas open or agricultural. Field data were collected from eight locations on each stream from May, 1973 to May, 1974. Sampling was conducted on a monthly basis and during high and low flow (wet and dry weather) conditions. On the average, 5-day BOD levels increased 4.0 mg/l during wet weather in both creeks; however, no concurrent decrease in DO was identified. In many cases, DO levels increased following precipitation, possibly due to increased reaeration as the result of increased stream flow (i.e., turbulence). The authors also speculated that the decrease in travel time to Puget Sound, as a result of the increase in creek flows caused the BOD material to be flushed from the creeks before there was sufficient time to exert an observable oxygen demand.

In summary, the reports of water quality impacts on streams and rivers from urban runoff found in the literature do not support the likelihood of a DO impact. Most studies were not able to isolate urban stormwater runoff from other pollutant sources such as CSO, sewage treatment plant effluents, and agricultural (or rural) runoff. Some researchers have observed DO depletions during wet weather conditions, and others have found just the opposite. Increases in DO concentrations during wet weather were common and usually attributed to increased reaeration caused by increased turbulent flow or to the runoff waters being saturated with DO. Short-term DO depletion during wet weather was not apparent at either of the two stream sites monitored during the field program to determine highway runoff impacts on receiving waters (See Volume II).

Dissolved oxygen deficits due to long-term accumulation of oxygen-demanding materials from highway or urban runoff in sediments of low velocity receiving waters are difficult to document. There are many potential sources of oxygen-demanding materials to sediments, and this makes it difficult to segregate the demand due to runoff materials. In lake systems, particularly those receiving nutrient inputs from stormwater, decaying organic material from primary producers can be a significant source of oxygen demand to the sediments. Therefore stormwater inputs could result in organic accumulation through direct input of runoff particulates to sediments or through stimulation of primary production.

Although several studies indicate the possibility of DO depletion due to accumulation of stormwater particulates in sediments, there are no studies that define the significance of this mechanism. There has generally been inadequacies in the quantification of the various sources of organic enrichment of sediments to determine the effects attributable to stormwater discharges. A case/control study for determination of sediment oxygen demand rates, in addition to chemical characterization of sediments to trace stormwater particulates, is necessary to properly address this stormwater impact. Limited measurements of sediment oxygen demand (SOD) made with static respirometers at the Wisconsin Highway 15/Sugar Creek site did not reveal an increase in SOD in stream pools downstream from highway runoff input locations compared to a control station (See Volume II).

NUTRIENT IMPACTS AND ACCELERATED EUTROPHICATION

As shown previously in Table 1, nutrient concentrations in highway runoff can be fairly high with a total phosphorus mean of 0.8 mg/l (maximum of

36 mg/l) and a total nitrogen mean of about 3 mg/l. However, the loadings of nutrients from highways to receiving waters is usually quite low because of the relatively small proportion of the watershed used for the ROW (as described in Volume II). As a result of the low loadings, nutrient effects, such as accelerated eutrophication in lakes, are unlikely to be attributable to highways alone. In developing urban areas, however, a highway project could be viewed as part of a cumulative impact on a receiving water (especially water supply sources). For this reason, a qualitative discussion of nutrient impacts from stormwater is presented below. The user is also referred to the section on BIOLOGICAL IMPACTS later in this volume. The results of algal assays which used highway runoff provide additional insight on nutrient effects in terms of potential stimulation or inhibition of algal activity.

To properly understand nutrient impacts on stream and river systems, it is necessary to evaluate the transport mechanism of these nutrients. Verhoff and Melfi (31) evaluated two phosphorus transport theories using a mass balance model. These transport theories can be stated as follows:

1. Continuous flow theory - total phosphorus is washed from land surfaces through tributary systems and into larger receiving systems (larger river or lake) during one storm event.

2. Discontinuous transport theory - total phosphorus is washed from land surfaces into tributary systems and is carried downstream in a series of waves. Each wave consists of deposition as storm flows recede and resuspend during a subsequent storm.

The authors concluded that the predicted hydrograph and total P concentration matched observed field characteristics only when deposition and resuspension terms were included. Therefore, the discontinuous transport theory was considered the most accurate description of total P transport in streams. This result has implications for the transport of phosphorus and nitrogen from highway stormwater since these nutrients are highly associated with particulates. Any instream impacts may be due to nutrient-rich sediment deposits and the availability of these nutrients to rooted aquatic plants (macrophytes).

The impact of stormwater nutrients on streams is much less severe than lakes because of several factors:

1. Critical nutrient levels for nuisance aquatic plant growths are much higher in streams than in lakes.

2. Stormwater nutrients are generally discharged during high flow conditions and therefore are passed through the stream system rapidly. In lake systems, nutrients may tend to accumulate.

3. During runoff, stream turbidity is relatively high (i.e., light penetration is low); primary productivity is inhibited, hence the amount of nutrients which can be assimilated is also limited.

4. Stormwater nutrients are generally associated with particulates. Of these, only a portion may be readily available for aquatic plants. The availability of particulate nutrients may increase with time, therefore having greater effects in water bodies such as lakes with fairly long hydraulic retention times (32).

The effect of urban and highway runoff nutrients on lake and wetland systems is dependent on the nutrient loadings, the availability of the stormwater nutrients, and the removal of the nutrients from the lake or wetland. Removal of nutrients from the lake can be accomplished through settling of stormwater particulates, uptake of nutrients by phytoplankton and subsequent settling, and flushing of the various nutrients from the lake. In deep lakes, settled nutrients may be effectively removed from the system and have little effect on lake productivity. However, release of nutrients, particularly phosphorus, from bottom sediments is possible in extremely productive lakes if the water of the hypolimnion becomes anoxic (during late summer or winter). When the spring or fall turnover (mixing) occurs these nutrients will be available for phytoplankton growth. In shallow lakes where light penetrates to the bottom, nutrients present in the sediments are available for rooted aquatic plants. Macrophytes may be able to utilize nutrients in both the water and sediment (33).

Several articles have been published on the nutrient loadings from an urban area to Lake Wingra in Madison, Wisconsin (34,35,36,37,38). Of these, the work of Cowen and coworkers (36,37) is most pertinent here since the researchers partitioned stormwater nutrient constituents according to algal availability. This was accomplished through the use of mineralization tests (dark incubations) (36). Algal available N (ammonia plus nitrate N) ranged from 4 to 66 percent of the total N in 13 urban runoff samples from Madison, Wisconsin. Total N averaged 0.75 to 3.53 mg/l. After dark incubation of the samples for up to 100 days, algal available N averaged 69 percent of the total N. This increase in algae-available N was due to mineralization of organic N (particulate and soluble). The authors note that these experiments do not indicate the percentage of the total N which could be utilized by algae in a lake. Rather, the value of 69 percent of the total N becoming available should be considered as a maximum. In a related study, Cowen and Lee (37) examined the availability of particulate P using algal assay experiments. It was assumed that all soluble P was available (eventually) to the algae. The algal assays indicated that an average of 30 percent of the particulate P was available to algae. Similarly, but based strictly on chemical and physical analyses of runoff, Browerman, et al. (38) found that the upper limit for potentially available P is the sum of all inorganic soluble P plus 25 percent of the particulate P. Their data are reflective of residential urban areas.

TOXIC MATERIALS

Stormwater-related loadings of toxic materials can be broadly categorized as impacts from deicing agents, toxic metals, hydrocarbons (including those of petroleum origin), pesticides, herbicides, and polychlorinated biphenyls. Each of these categories will be discussed separately.

Deicing Agents

Basic Concepts--

The fate of highway deicing salts (i.e., sodium, calcium and chloride) is conceptually depicted in Figure 1 (39). Salts can be picked up and transported by atmospheric mechanisms such as winds. These salts are eventually discarded or fall-out to land and surface waters. Salts are also dissolved and carried to surface and groundwater components by runoff and percolation, respectively. Salts reaching the groundwater can subsequently be fed back into surface waters through artesian springs. Likewise, surface water salts can be transported to groundwater through recharge mechanisms. Since chloride is more soluble than sodium and calcium, it is more readily transported between the components while sodium and calcium is ultimately adsorbed to soils and sediments, especially clays (8,39).

The fate of salts in streams and rivers can be quite different from lakes. Unlike lakes, where salts can be entrapped and concentrated (17), the impacts of deicing salts on streams and rivers is more transient. This is due primarily to the dynamic flow conditions in rivers and streams. One commonly recorded occurrence (29,40,41,42,43) is that spring snow melt and stormwater runoff exert increased salt loadings (mass per unit time) to streams and rivers. However, this impact is often offset by increased dilution such that peak flow salt concentrations are comparable to or lower than prestorm base flow conditions. But there have been several reports of increased salt

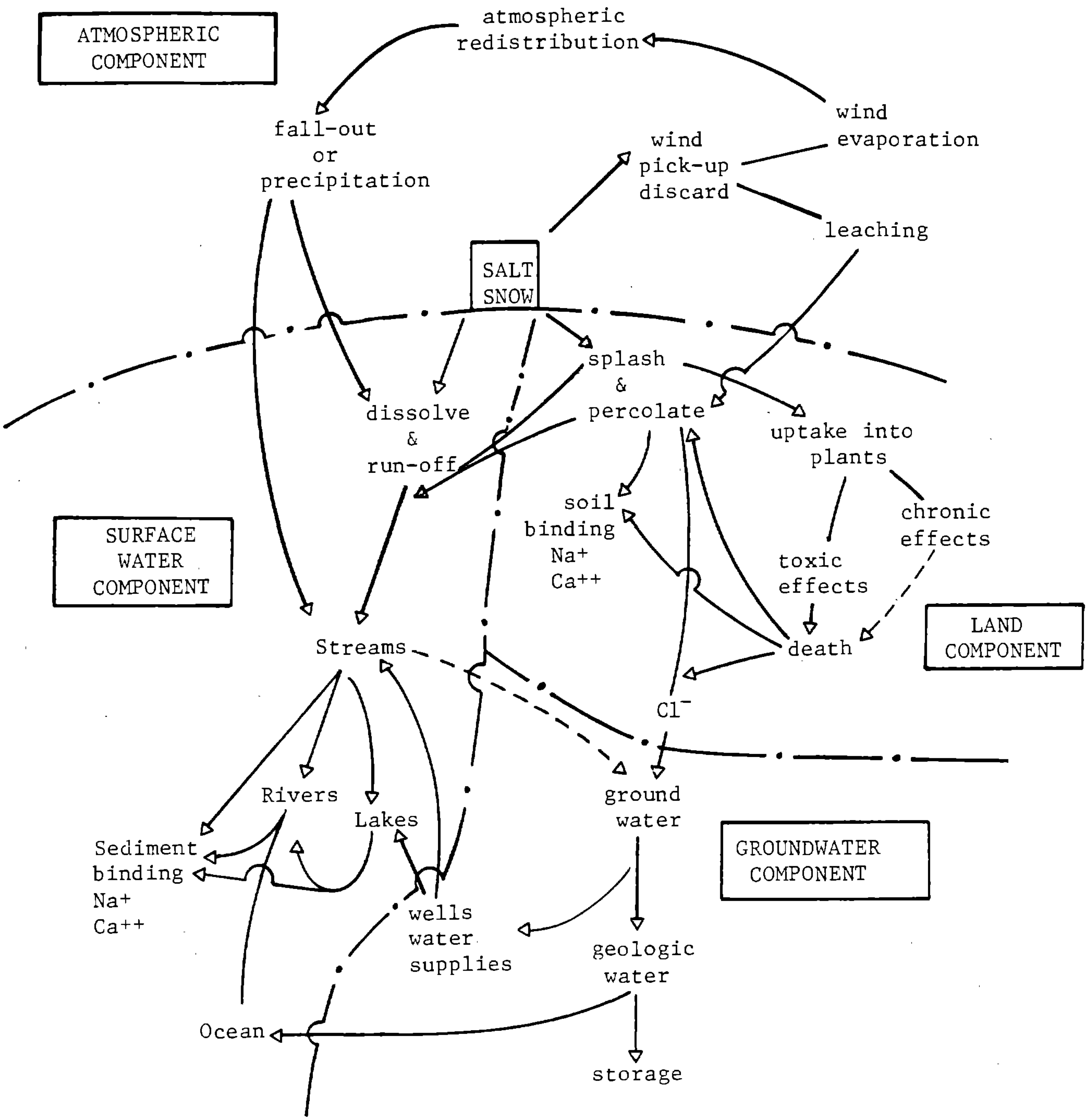

Figure 1. Fate of deicing salts in the environment (39).

concentrations in spring in streams and rivers due to highway and urban runoff when compared to upstream controls (40,41,44). Due to the basic morphometric and hydraulic differences, the impacts of deicing salts on lakes, reservoirs, and wetlands is markedly different than streams and rivers. Salts dissolved in runoff waters can increase the density of the solution so that this discharge sinks to the bottom of lakes. Surface outflows from such a lake will have lower salt levels so that the net effect is an increase in the salt retention time in the lake over the effective hydraulic retention time. In this respect, the lake will act as a storage site for discharged highway salts.

Due to this entrapment and concentration of salts in lakes, a temporary density stratification can be imposed. This means that the input of salts to a lake, which normally mixes completely during seasonal overturns, can increase the density gradient to the point where mixing forces (winds) are no longer sufficient to completely mix the upper mass of water with the stagnant lower mass. This effect has been documented by several investigators (15,16,18,19).

Effects of Highway Salts at Three Field Monitoring Sites--

Winter road salt effects on water quality were determined at all three field sites monitored for this contract (See Volume II). During a winter snowmelt event at the Wisconsin Highway 15/Sugar Creek site the stream exhibited only slightly elevated chloride concentrations at a highway influenced station during the melt (peak of 65 mg/l) compared to the control station (42 mg/l during the melt) and background dry weather concentrations at all stations prior to the melt event (23 - 50 mg/l). Approximately 18.2 tons of salt had been applied to the 3.5 mile (5.6 km) long ROW tributary to the stream during the week prior to the melt event.

A snowmelt/rainfall event at the I-85/Sevenmile Creek site near Efland, North Carolina produced a peak chloride concentration of around 250 mg/l in the stream at a headwater tributary station close to a runoff input. Considerably lower concentrations were evident downstream as a result of dilution with a peak chloride level of 34 mg/l compared to a peak level of 21 mg/l at the control station. Roughly 14.4 tons of salt were applied to this site during the days immediately preceding this event. The I-85 ROW length draining to Sevenmile Creek was about 4.8 miles (7.7 km).

A the I-94/Lower Nemahbin Lake site, also in southeastern Wisconsin, intensive winter monitoring revealed no water quality effects from highway salting. High salt concentrations were measured in runoff and about 22 tons of salt were applied that winter to the 0.75 mile ROW (1.2 km) which drains directly to the 270 acre (110 ha) lake. However, no salt stratification effects were observed and lake outflow salt concentrations were never elevated compared to the inflow.

The Effects of Deicing Agents in Wisconsin--

An ongoing program to monitor road salt content of soil, water, and vegetation in areas adjacent to highways was started in November, 1970 by the Soils Unit of the Materials Section of the Wisconsin Division of Highways and Transportation Facilities (45). The primary objective of this program was to determine if road salt is accumulating along Wisconsin highways, at what rate it might be accumulating, and if accumulation rates differ for different conditions of drainage and soil type. However, this program was also monitoring roadside vegetation and waterbodies (streams, lakes, marshy ponds, and borrow pits) adjacent to highways. Sample analyses include conductivity, chloride, sodium, and calcium. As of the 1979 progress report (Progress Report III), 10,000 soil analyses, 3,300 water analyses, and 240 vegetation analyses had been obtained.

In general, accumulation of sodium and chloride in soil was very slight beyond 50 or 60 ft (18 m) from the pavement. Sodium and chloride in water samples were usually low. Somewhat higher sodium and chloride levels were noted in roadside ditch water. However, the author observed that water in roadside ditches is transitory and generally of such small volume relative to the receiving water body as to have little measurable effect on salt levels of the permanent water feature into which they drain. Higher sodium and chloride values in receiving water bodies were observed during periods of low precipitation and/or freezing conditions. In Wisconsin the amount of calcium chloride applied as a deicing agent represents only a small percentage of the total road salt. It was not surprising that water sample data indicated that there was little contribution of calcium to the monitored receiving waters from road salt.

A small pond in an area receiving surface drainage and supporting cattail growth was monitored. Figure 2 presents the initial water quality data including presalt data (September, 1978) and postsalt data (January and April, 1979). Sodium did not change between presalt and postsalt conditions. Chloride concentrations did increase noticeably with the first year of road salt application, but the values were still quite low (about 12 mg/l) relative to the water quality criterion of 250 mg/l. The author could not explain the high initial values of calcium and conductance, but felt that the additional data which will be collected at this site may help to explain the cause of this unusual condition.

Another of the water quality sites, Murphy Creek, which was monitored from 1970-77 is located in Dane County on USH-14. A control station was located about 700 ft (213 m) upstream of USH-14; while other stations were located just downstream of USH-14 and about 1,300 ft (396 m) downstream through a marsh. The aerial photograph presented in the progress report showed that the creek maintained a well-defined channel through the marsh. There was relatively high upstream concentrations of calcium, sodium, and

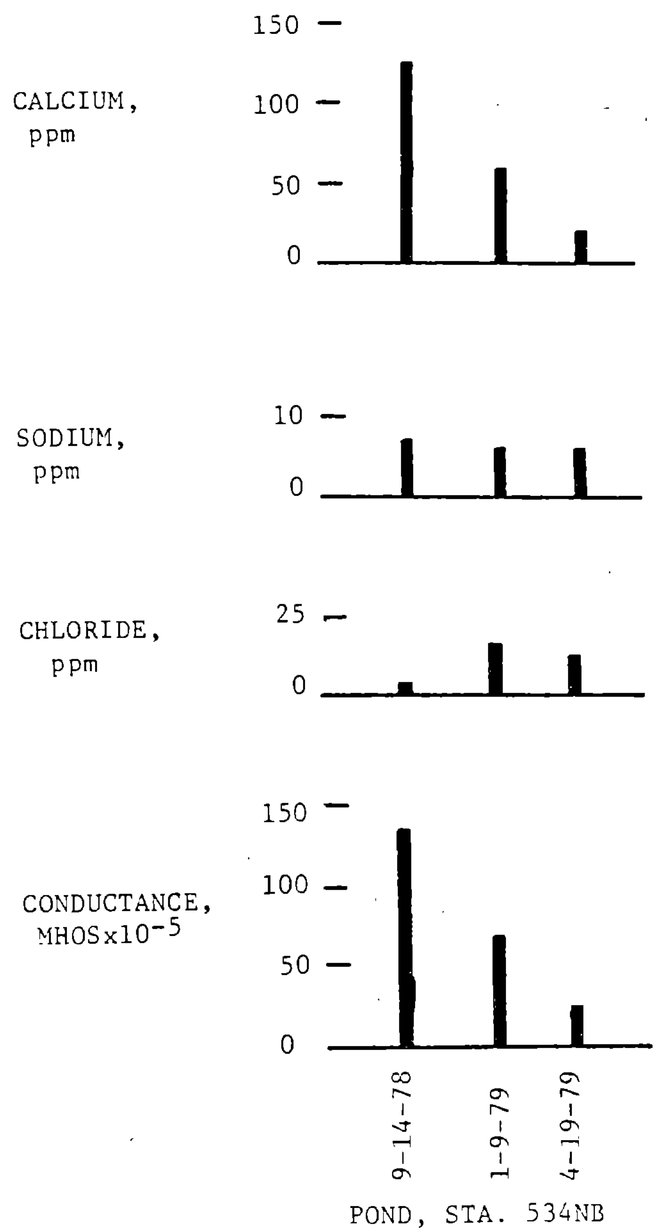

Figure 2. Presalt and postsalt data for a small pond in Manitowoc County, WI (69).

chloride and the author theorized that these may have originated from agricultural sources. The data show that there was generally a slight decrease in chloride concentrations at the downstream marsh sampling point. Sodium and calcium, however, did not generally display this pattern.

Water quality monitoring sites were also located in Portage County on USH-51. This site included three borrow pit lakes; Lake Pacawa, Lake Susan, Island Lake, and a marshy pond. The borrow pit lakes were generally 15 to 20 ft (4.5 to 6.0 m) deep, and the pond was approximately 4 ft (1.2 m) deep. The marshy pond and Lake Pacawa showed relatively high values for calcium, sodium, and chloride with a trend for progressive increase in these constituents. In contrast, Lake Susan and Island Lake exhibit low and generally stable values. The water levels of Lakes Susan, Island, and Pacawa represent the groundwater surface and were therefore indicative of local groundwater chemistry. In contrast, the marshy pond received a greater proportion of its water from surface runoff. Therefore, the probable source of some of the sodium and chloride in the pond water is highway runoff. Because all three lakes have similar hydrology and have similar locations relative to USH-51, and because Lake Susan and Island Lake did not show an increase of chloride and sodium over the period monitored, the source of these constituents in Lake Pacawa could not be attributed solely to highway runoff, but the origin appeared to lie with multiple sources. The author suggested that septic systems, agricultural soil treatment, and wastewater disposal may be a more important source of sodium and chloride than road salt.

<u>The Effects of Deicing Salts in Maine</u>--

A study similar to the one in Wisconsin was undertaken by the State of Maine (46). The objectives of this project were to determine the effect of highway salt applications on the sodium and chloride levels in:

1. Streams and rivers,

2. Private water supplies contiguous to highways, and

3. Soils bordering highways.

The author presented three overall conclusions as a result of the study:

1. The levels of sodium and chloride ions present in the major rivers in Maine have not been influenced significantly by the application of salt to highways for deicing purposes.

2. The levels of sodium and chloride ions present in water and soil along the major highways in Maine have been increased markedly as a result of sodium chloride applications for deicing.

3. Shallow wells drilled in proximity to highways (30 ft or less [9.1 m]) will usually be rendered unsafe for drinking because of excessive chloride concentration.

This study demonstrated pollution of farm ponds (used as a source of water for livestock), in areas contiguous to highways could be a serious problem. Table 5 summarizes the sodium and chloride data collected from randomly selected ponds. Average chloride concentrations of 80 and 110 ppm in August and April, respectively are elevated above the normal levels of 5-10 ppm found in ponds at a higher elevation than the highway, such as site numbers 14 and 19. The data collected as part of this study indicate that continued applications of sodium chloride to highways may eventually cause some farm ponds to be unsuitable for livestock, although such levels are not yet present.

The effects of highway salts on seven rivers in Maine were also determined by Hutchinson (40) in an earlier study. Average insteam sodium and

Table 5. Sodium and chloride concentrations in farm ponds randomly selected along Maine highways to evaluate the effect of salt applications[a] (46).

Site no.	Distance from highway, ft	Town	Sodium, ppm		Chloride, ppm	
			August	April	August	April
1	15	Hermon	77	64	135	200
2	25	Carmel	4	2	1	9
3	15	Etna	65	79	80	128
4	15	Winterport	-	109	-	315
5	35	Littleton	42	25	70	109
6	20	Littleton	7	3	9	64
7	20	Houlton	4	4	6	33
8	15	Houlton	29	11	54	96
9	25	Linneus	71	14	122	99
10	10	E. Corinth	9	13	24	58
11	35	Preque Isle	7	11	31	104
12	50	Mars Hill	2	1	3	25
13	200	Mars Hill	5	4	11	37
14	15	Pittsfield	3	2	6	7
15	20	Pittsfield	4	5	9	16
16	35	Winslow	7	7	20	29
17	20	Augusta	56	65	141	190
18	45	Prospect	121	71	257	315
19	75	Troy	-	1	-	10
20	35	Troy	41	12	60	56
21	5	Mechanic Falls	84	72	160	180
22	15	Gorham	12	10	35	40
23	25	Skowhegan	16	9	76	41
24	-	Dixfield	6	3	25	7
25	60	Waterville	62	50	257	200
26	70	Etna	105	106	318	490
27	82	Topsham	-	43	-	115

[a] Each value is a mean for two annual samplings.

Metric Units: To convert ft to m multiply by 0.3048.

chloride concentrations never exceeded 11 mg/l, in spite of the fact that monitored highway discharges had sodium and chloride concentrations as high as 265 and 845 mg/l, respectively. Again, this was attributed to storm dilutional effects. The upstream control section of Kenduskeag stream had sodium and chloride concentrations only slightly lower than the downstream reach influenced by highway runoff.

The Effects of Deicing Salts in California--

The environmental effects of deicing salts on aquatic ecosystems including streams, lakes, and ponds were also investigated by the State of California (18,19). Analyses were performed on runoff samples, surface water samples, and deicing salt samples. Analysis on deicing salt samples included macronutrients and trace metals. Bioassays were conducted on twenty-one deicing samples for effects on both bacterial and algal metabolic rates. Chloride levels in streams below heavily salted major freeways (e.g., I-80) were found to increase significantly during the winter period. Chloride concentrations measured during periods of actual salting sharply increased and then decreased with time. Small lakes receiving runoff from major highways also showed chloride enrichment compared to other lakes in the state. Road salt-impacted lakes had mean chloride concentrations of 20 mg/l while other "control" lakes had chloride concentrations of only 0.3 mg/l. In addition, several lakes displayed a temporary chemocline which was sufficiently strong to stabilize a temperature inversion in one lake. Bioassays using deicing compounds showed inhibition of bacterial communities by those currently used road salts and detrimental effects of several alternative deicing compounds. Plankton metabolism was not significantly altered by most deicing salt samples. The authors concluded that although road salting results in detectable chloride enrichment in aquatic ecosystems, there is, from a short term investigation, no indication of its detrimental effects on the endemic biota. However, only extensive investigations can detect whether those levels are detrimental on a long term basis.

Other Highway Studies--

The effects of a highway near Oslo Norway on lake water and sediment quality were recently described by Gjessing, et al. (47). Water quality parameters documented included road salt constituents (chloride, conductivity, and calcium), metals and PAH. The effects of the latter pollutants are described later in this document. Table 6 describes the key highway and lake characteristics of interest. The chloride content of drainage waters from the highway averaged 2,300 mg/l (ranging from 35 to 4000 mg/l) while two control tributaries averaged 9 mg/l (ranging from 4.2 to 22 mg/l). The outflow stream from the lake averaged 15 mg/l chloride with a range of only 14.4 to 16 mg/l. In-lake samples were not obtained, but the low outflow values of chloride suggest significant dilution and/or retention of road salts by the lake. Stratification effects were not determined.

Kunkle (41) observed a minor impact from highway salts on Sleepers River in Vermont. Mean chloride concentrations in upstream control reaches were an order of magnitude lower than reaches receiving highway discharges during both dry and wet periods. Chloride levels at the highway stations peaked during summer baseflow and were more than 50 percent lower during storms due to dilution. This is considered a minor impact, even though an order of magnitude increase was apparent, because the mean downstream levels were only 32 mg/l which would not detract from the water's beneficial use.

Sodium concentrations were measured in Black Creek (Toronto, Ontario) which receives both highway and urban runoff (48). During thaw periods, the sodium levels in the creek increased by as much as fifty-fold over baseflow conditions. These concentrations did not return to normal for several days. Estimated chloride concentrations exceeded the standards for domestic water supplies (250 mg/l) for at least three days. Although this appears to contradict the general trend of the lower salt concentrations during runoff period compared to baseflow, it must be emphasized that the sampling periods

Table 6. Highway and Lake characteristics of Norwegian Field Study Site (47).

Lake Padderudvann and Watershed	Highway E-6
• Total catchment area - 593 acres	• 4-lane highway
• Lake surface area - 59 acres	• Row paved area tributary to the lake
• Lake volume - 127 million ft^3	• Traffic volume - 19,400 veh/day
• Mean depth - 49 ft	• Construction completed in 1969

covered snowmelt conditions only, and as such the dilutional capacity was not as great as spring storm conditions. It should also be noted that the upstream control point for Black Creek was not entirely uninfluenced by urban runoff. Several storm sewer outfalls from an industrial area were located upstream of this point.

Three streams (with forested watersheds), affected by road salt from highway runoff near a resort area in the mountains of New Mexico, were studied by Gosz (44). For all ions but sulfate, downstream (below road) ion concentrations were higher than upstream controls. The concentrations were reported as flow-weighted means for the entire water year. Elevated instream salt ion concentrations were observed as far as 1,300 ft (400 m) downstream from the highway inputs. Increases in chloride at downstream stations ranged from double to 20 times higher than the upstream controls.

Road Salt Additives--

Chemical deicing agents used on highways often include a variety of additivies to prevent caking or inhibit corrosion (49). The most common of these are the anti-caking compounds ferric ferrocyanide (or Prussion blue) and sodium ferrocyanide. Ferric ferrocyanide is insoluble and relatively low in toxicity. For example, it has been reported that a concentration of 9,600 mg/l of Prussion blue had no adverse effect on minnows in a 48-hour bioassay (67). Sodium ferrocyanide, conversely is quite soluble and liberates the cyanide ion in the presence of sunlight (75). Threshold toxicity levels for sodium ferrocyanide range from 170 to 600 mg/l depending on the test organism. Schraufnagel (50) found that when exposed to light for 30 minutes or more, 15.5 mg/l of sodium ferrocyanide yields 3.8 mg/l of cyanide. As shown previously in Table 4, the EPA criteria for cyanide for protection of freshwater aquatic life are 0.0035 mg/l and 0.052 mg/l for chronic and acute levels, respectively.

The concentration of sodium ferrocyanide (ten-hydrate) in road salt was around 0.5 pounds per ton of rock salt in 1971, although concentrations may be lower today (49). A sample of road salt applied to the I-85/Sevenmile Creek site in North Carolina (See Volume II) contained 1.8 mg/kg of total cyanide. However, no cyanide could be detected in either highway runoff or stream samples during a melt event. Other studies of effects of anti-caking additives were not found in the technical literature.

Metals

Since highway runoff discharges have been shown to contain metals such as lead, zinc, iron, copper, chromium, cadmium, and nickel, this pollutant group probably represents that of most concern to environmental agencies and the general public. The major sources of these metals for highway systems are automobiles and atmospheric washout (51). Several metals may also be associated with anti-caking agents (50,52), but the concentrations are quite low (3,44). Impacts associated with metals may be of a public health nature if levels exceed those acceptable for water supplies. Also, metals may accumulate in organisms, such as fish, making them unsuitable for consumption. Metals may also exert toxic effects on sensitive aquatic biota resulting in replacement of these organisms with pollution tolerant biota. Water quality criteria for protection of both public health and aquatic biota were previously presented in Table 4.

An important consideration in discussing impacts from metals discharged from highways is the form in which the metals exist. Limited analysis of the soluble fraction of highway runoff performed during the first FHWA characterization study (1) indicated most metals are found associated with the particulate phase. In the detailed source and migration study which followed (3), extensive characterization was made of the particle size classes in which these metals are found, with special emphasis given to determining the "washable" fractions. In recent studies performed by researchers at the

University of Central Florida (53,54), more specific data have been obtained to define not only the relative particulate/soluble fractions, but also the species of metals found in the soluble fraction. This type of data, of course, has a direct bearing on the toxic properties of the highway derived metals.

Depending on the physical/chemical properties of the water, metals can exist as inorganic or organic complexes, chelated, adsorbed onto particulates, and in free ionic form, which is generally the most toxic form (54). Iron and lead are found in mostly insoluble form (5 to 12 percent dissolved), while zinc and nickel are much more soluble, as shown in Table 7. The high soluble fraction percentages in retention basin water samples at the Maitland Interchange demonstrate the sediment deposition of the particulate metals. The speciation of metals in the soluble phase is summarized in Table 8. These data generally show most dissolved metals to be of labile (reactive), inorganic form, with variable distributions between colloidal and non-colloidal forms. Based on both model results and direct observation, the authors also concluded that zinc and cadmium exist largely (70 percent) in ionic form, and are reactive and biologically available. Most of the soluble lead exists as $PbCO_3$ while dissolved copper is associated with humic and fulvic (organic) substances.

Given the high association of highway derived metals with particulates and their rapid removal to the sediments, it is not surprising that field studies of highway runoff impacts have rarely identified significant elevations (i.e., above water quality standards or criteria) of metals in the water column. At the Wisconsin Highway 15/Sugar Creek site (See Volume II), peak wet weather metals concentrations (Pb, Zn, Ni and Cu) were higher at runoff influenced stream stations compared to control stations. For lead, two individual samples had concentrations in excess of the EPA chronic criterion, but not the acute criterion. Other metals did not exceed state standards or EPA criteria, except zinc, which exceeded EPA criteria at both control and

Table 7. Mean Percentage of Soluble Heavy Metals in Highway (I-4) Runnoff from Central Florida (53,54).

Metal	Early Study (53)		Recent Data from Maitland Interchange (54)	
	Bridge runoff at Lake Ivanhoe(n=11)[a]	Runoff from Maitland Interchange(n=15)	Runoff Water(n=16)	Retention Basin water(n=34)
Cadmium	20	50	58	80
Copper	52	46	53	88
Chromium	18	30	33	68
Iron	12	5	4	33
Lead	12	6	6	73
Nickel	92	50	11	78
Zinc	67	17	14	91

[a] n = number of samples.

Table 8. Relative Distribution for Various Dissolved Species of Trace Metals in Water Samples (54).

Metal	Form	Percentage in Water From				
		Maitland Interchange			U.S. 17-92	
		Rainfall	Runoff	Pond	Bridge Runoff	Shingle Creek
Cd	Labile[(a)]	85.9	84.7	86.3	78.1	75.2
	Non-Labile	14.1	15.3	13.7	21.9	24.8
	Organic	1.1	4.3	4.2	3.4	13.3
	Inorganic	98.9	95.7	95.8	96.6	86.6
	Colloidal	19.3	36.4	31.7	38.3	38.8
	Non-Colloidal	80.7	63.6	68.3	61.7	61.2
Zn	Labile	93.7	92.5	96.3	92.5	89.5
	Non-Labile	6.3	7.5	3.7	7.5	10.5
	Organic	0.0	0.7	0.3	0.3	0.2
	Inorganic	100	99.3	99.7	99.7	99.8
	Colloidal	45.8	23.6	83.7	29.2	31.0
	Non-Colloidal	54.2	76.4	16.3	70.8	69.0
Pb	Labile	65.6	72.7	55.4	43.8	67.2
	Non-Labile	34.4	27.3	44.6	56.2	32.8
	Organic	14.6	15.4	17.3	44.0	19.3
	Inorganic	85.4	84.6	82.7	56.0	80.7
	Colloidal	63.3	36.7	54.2	68.7	54.9
	Non-Colloidal	36.7	63.3	45.8	31.3	45.1
Cu	Labile	84.0	45.9	81.0	58.7	49.0
	Non-Labile	16.0	54.1	19.0	41.3	51.0
	Organic	38.3	56.6	53.8	33.4	62.2
	Inorganic	61.7	43.4	46.2	66.6	37.8
	Colloidal	59.8	75.6	72.1	62.0	79.7
	Non-Colloidal	40.2	24.4	27.9	38.0	20.3

[a] Labile - chemically reactive, non-labile - nonreactive.

highway influenced stations. Median wet weather concentrations were not elevated at highway runoff influenced relative to control stations for any of the metals.

At the I-85/Sevenmile Creek site (Volume II), dry weather concentrations of lead, cadmium and chromium generally exceeded wet weather values at both control and influenced stations, suggesting no impact for these metals. Nickel and zinc concentrations were occasionally elevated at influenced stations during storms, but did not exceed standards or EPA criteria. Copper concentrations at influenced stations during storms were frequently elevated compared to control stations, but both control and influenced stations exceeded the EPA criterion during dry and wet weather.

At the I-94/Lower Nemahbin Lake site (Volume II), elevated water column concentrations of metals were not observed, although sediment enrichments were apparent in nearshore areas in proximity to bridge deck scupper discharges. Sediment effects are described in more detail later in this report.

The effects of highway runoff metals on Lake Padderudvann near Oslo Norway were also assessed by Gjessing et al. (47). The highway and lake features were previously described (See Table 6). Metals concentrations were considerably higher in the tributary stream draining the ROW compared to two inflowing control streams, as shown in Table 9. The lake appeared to effectively retain/dilute these metals since the concentrations in the outflow stream were comparable to the inflowing control streams. In-lake water column concentrations were not determined, but two to four-fold sediment enrichments were observed.

Gosz (44) studied metals in several mountain streams in New Mexico which received runoff from ski resort roadways. Analyses indicated extremely low levels of lead, zinc, and copper associated with the deicing agents applied during winter months. Table 10 lists metals inputs and outputs for three watersheds upstream and downstream from the roads. For lead and copper the

Table 9. Average Metals Concentrations in Inflow and Outflow Streams of Lake Padderudvann (47).

Metal	Units	Two Control Inflow Streams Avg.	Two Control Inflow Streams Range	ROW Drainage Inflow Stream Avg.	ROW Drainage Inflow Stream Range	Lake Outflow Avg.	Lake Outflow Range
Fe	mg/l	0.22	0.04-0.48	21	2-40	0.12	0.15-0.26
Cd	μg/l	0.1[(a)]	<0.1-1.3	3	0.2-4.1	<0.1[(a)]	<0.1-0.3
Cu	μg/l	35	5-150	153	60-188	39	1-140
Zn	μg/l	33	<10-63	450	90-610	32	<10-63
Pb	μg/l	1.6	<0.5-3.9	425	64-753	1	0.6-1.1
Ni	μg/l	<5[(a)]	<5-19	45	39-50	<5[(a)]	<5-14
Number of Samples		(12)		(5)		(5)	

Table 10. Road salt input and stream output for heavy metals. Values are grams/ha for the 1976-77 water year (44).

Watershed	Lead		Zinc		Copper	
	Road salt input	Stream output	Road salt input	Stream output	Road salt input	Stream output
S. Fork Tesquue						
Above Road	--	0.2	--	0.4	--	0.3
Below Road	0.3	5.3	0.1	44.9	0.2	3.9
Middle and N. Fork Tesuque						
Above Road	--	0.2	--	0.7	--	0.2
Below Road	0.8	20.5	0.2	144.1	0.4	22.1
Rio en Medio						
Above Road	--	0.2	--	1.7	--	0.3
Below Road	0.7	7.5	0.1	262.2	0.4	13.7

majority of the outputs were associated with particulate matter. The metals output below the road were much higher than those above the road and cannot be explained by the inputs from deicing agents. The authors speculated that the losses (primarily particulate) were due to sodium from the deicing agents causing a breakdown of the granular soil structure by replacement of calcium and magnesium ions. Evidence from the literature was cited to support the contention.

Farris, et al. (55) studied runoff at several interstate highway bridges in the vicinity of Lake Washington. Although total lead ranged from 0.57 to 4.96 mg/l in runoff from the Lacey V. Murrow Bridge (I-90), the researchers failed to identify an increased lead concentration in the lake near the bridge following runoff events. Similar results were obtained for other metals. Metals concentration in the sediments near the bridges were not examined.

Petroleum Hydrocarbons and Other Synthetic Organic Compounds

As shown previously in Table 2, highway runoff contains a variety of petroleum derived hydrocarbons and SOC's, including oil and grease (analytically defined), herbicides, polychlorinated biphenyls (PCB) and polynuclear aromatic hydrocarbons (PAH). However, the receiving water effects of these hydrocarbons have not been extensively studied to date for highway runoff. The field monitoring portion of this report (Volume II) provided oil and grease data for all three receiving water sites and herbicide data for one storm at the I-85/Sevenmile Creek site. The effects of PAH discharges to a lake in Norway were also documented by other researchers (47). Numerical water quality standards and criteria do not exist for "oil and grease," so the magnitude of a water quality effect for this parameter cannot be assessed. Similarly, standards/criteria have generally not yet been promulgated for all PAH's. Standards for some pesticides and PCB are more commonplace, however.

At the Wisconsin Highway 15/Sugar Creek site and I-94/Lower Nemahbin Lake sites, oil and grease concentrations at runoff influenced stations did not exceed those of control stations. At the I-85/Sevenmile Creek site, oil and grease data were obtained for two wet weather surveys. For one survey, limited sampling showed a very high oil and grease concentration (60 mg/l) at one headwater tributary station influenced by highway runoff. A more comprehensive oil and grease survey conducted on a later date (See Table 11) showed a lower, though still discernible effect.

Special sampling was also conducted at the I-85/Sevenmile Creek site to determine the effect of ROW application of pesticides. Baseline samples collected in August 1981 showed that no synthetic organics (i.e., PCB, pesticides) were detectable in Sevenmile Creek. Herbicide (2,4-D) and growth retardants were applied to the ROW on March 23, 1982, with the first runoff event occurring on April 8, 1982. Herbicide (2,4-D) was detected in runoff (19 µg/l) and one stream sample (3.8 µg/l) at a headwater tributary station near the highway. The state water quality standard of 10 µg/l for 2, 4-D was not exceeded. The herbicide was below detection limits at a downstream station. No other pesticides or PCB were detected at any stations during this event.

The PAH content of inflowing and outflowing streams of Lake Padderudvann near Oslo, Norway (the lake and highway features were described previously in the sections on salts and metals, see Table 6) were determined for several storm events (47). Snow samples taken from the ice covered lake nearest the highway (i.e., within 50 m) and water samples from the inflowing stream which drains the ROW were shown to have "high" PAH concentrations. However, the outflowing stream from the small lake exhibited PAH levels comparable to inflowing control streams, suggesting PAH retention in lake sediments and adjacent soils.

Table 11. June 29-30, 1982 - Oil and Grease Survey at I-85/Sevenmile Creek Site.

I-85 Runoff		Stream Stations					
		Control		I-85 Influenced			
				SH1		SH2	
Time	O & G, mg/1	Time	O & G, mg/1	Time	O & G, mg/1	Time	O & G, mg/1
1440	6	1430	3	1600	8	1845	3
1730	6	1730	4	1800	6	2000	3
1900	27	2100	3	2030	3	2100	4
0815	3	0200	8	2330	15	2400	14
		0925	3	0230	9	0400	9
				0530	8	1445	1

MICROORGANISMS IN HIGHWAY RUNOFF

As shown previously in Table 3, highway and urban runoff contains fairly high densities of enteric indicator bacteria (fecal coliforms and streptococci). These have been shown to be of animal origin and therefore suggest a lower public health hazard compared to enteric organisms derived from humans (56,57). However, several studies have demonstrated the presence of human pathogens in low concentrations in highway and urban runoff (1,56,57). This is not surprising since animals have long been known to serve as reservoirs of human pathogens (58). Furthermore, water quality standards are based on the indicator organism only and hence contraventions resulting in shellfish bed or swimming area closures can occur regardless of the actual presence of pathogens.

Although shellfish water closures and public beach water quality violations have been shown to occur as a result of urban stormwater discharges (57,59), little work has been performed to date for highway runoff impacts. Limited fecal coliform and streptococcus data were obtained at all three field monitoring sites studied under this contract (See Volume II). Fecal coliform standards for all three sites were based on monthly geometric means, which makes assessment of short-term stormwater effects difficult. At the Wisconsin Highway 15/Sugar Creek site, the maximum coliform concentration during baseline surveys at a highway influenced station was higher than control stations, but the average fecal coliform densities were lower at the highway influenced station. Average values at all stations were well below the state standard of 200/100 ml (log monthly). Wet weather data were not obtained at this site. At the I-94/Lower Nemahbin Lake site, limited wet weather data for one storm event did not indicate any coliform impacts on lake water. Baseline sampling likewise did not indicate high coliform concentrations at highway influenced stations. Average concentrations were well below the standard at all stations. At the I-85/Sevenmile Creek site, the maximum fecal coliform level at a highway influenced station during wet weather was higher than the control station. Average wet weather coliform densities at highway influenced stream stations were slightly higher than control stations but were still well below the standard of 1000/100 ml (log monthly).

SEDIMENT IMPACTS

Bottom material (sediment) constitutes one of the major substrates for a wide variety of aquatic organisms. The accumulation and cycling of highway stormwater runoff constituents within this substrate is of critical importance. The high particulate association of many pollutant parameters (especially metals and hydrocarbons) implicates the sediments as the primary receiving water sink. It is the purpose of this section to summarize documented accumulation of metals, salts, or petroleum hydrocarbons in sediments. The relative magnitude of impact caused by sediment enrichment is difficult to assess however, because regulatory agencies have yet to provide standards or criteria for undisturbed sediments.

METALS ACCUMULATIONS IN SEDIMENTS

Comprehensive sediment sampling was performed at all three sites monitored under this contract (See Volume II). Core and grab samples were collected to determine possible sediment accumulation of metals, salts and oil and grease. Station locations were established to evaluate spatial effects. Cores were also sliced to investigate the vertical distribution of pollutants. For the two stream sites (Sugar Creek in Wisconsin and Sevenmile Creek in North Carolina), there were no notable or consistent relationships evident for metals accumulations in sediments downstream of runoff inputs compared to control station sediments. Vegetation analysis (cattail and willow) at the Sugar Creek site likewise did not reveal an increase in metals concentration at runoff locations.

Nearshore sediments collected in summer at the Lower Nemahbin Lake site exhibited metals enrichments close to bridge deck scupper drain discharges. Lead, copper and zinc levels at a deep water station influenced by highway runoff were also higher than control stations for a fall survey. Two to eight-fold enrichments in sediment metals were evident at this site. Metals concentrations in cattail samples were also higher at nearshore stations in proximity to the scupper drains. These cattails did not exhibit a toxic response as a result of these metals accumulations, however.

Metals analyses (Pb, Zn, Cd) on sediment samples taken from Lake Padderudvann (47) in Norway (See Table 6 for a description of the highway/lake characteristics) provided the following results:

1. Metals tend to be located in greatest concentration in the surficial (upper 2 cm) sediments,

2. These metals were distributed throughout the lake, with the highest concentrations in deep areas,

3. The metal enrichments relative to normal levels were two to four-fold.

A study of accumulation of metals in sediments of Lake Ivanhoe, a 125-acre (50 ha) freshwater lake near Orlando, Florida, due to stormwater runoff from Interstate 4 bridges, has recently been completed (60). The I-4 bridge consists of two basic sections (each with east and westbound lanes). The two sections are connected on a man-made island in the center of Lake Ivanhoe. Runoff water from the north section (north bridge) drains toward adjacent land on both sides. In contrast, water from the south bridge directly enters Lake Ivanhoe through a series of scupper drains running along the entire length of the bridge. The average daily traffic across Lake Ivanhoe on I-4 for both east and westbound lanes was 93,000 to 108,000 vehicles per day.

Sediment samples were taken under each of the two bridges and in the main body of the lake to serve as a control. In addition, runoff and lake water samples were also taken to document metals inputs from the bridges. Water quality impacts to Lake Ivanhoe were not assessed due to a lack of specific information on other loadings and lake circulation patterns. They did note that the average lake lead concentration of 75 exceeded the Florida state standard of 50. The average lead concentration in scupper drain runoff water was 1,558. Extractable metals concentrations in sediments below each of the north and south bridges of I-4 are shown in Table 12. The difference in levels attributable to the use of direct scupper drainage versus overland flow is apparent. Concentrations in sediments associated with scupper drains are at least twice as high as those under the north bridge (except for cadmium). Unfortunately, metals levels in control areas of Lake Ivanhoe were not presented.

Metals accumulations in a highway runoff detention pond were also evaluated. Table 13 summarizes sediment metals levels in a detention pond in the I-4 Maitland Interchange (West Pond) compared to those in the immediate vicinity of the West Pond outfall into Lake Lucien and in the central area of Lake Lucien (control). Lake Lucien is an essentially undeveloped 57-acre (23-ha) freshwater lake. Traffic volume on Maitland Boulevard was roughly 21,000 vehicles per day while I-4 traffic in that area was 32,000 to 40,000 vehicles per day.

Soil and plant samples were also collected from three floodplain sites which receive direct stormwater runoff from highway bridges either through scupper drains or curb-and-gutter drainage systems. Each site had both influenced and control stations. The results of these analyses are summarized in Tables 14 and 15 for soils and vegetation, respectively. The increase in metals in influenced soils and plants is obvious. Soil enrichments factors for lead ranged from 3.3 to 78 between the three sites. Plant enrichments were lower than those for soils. Although the authors provide no information on vegetation species sampled, they do note that no significant differences in plant diversity were observed between control and influenced stations.

Table 12. Significance of differences in heavy metal concentrations of bottom sediments from Lake Ivanhoe (60).

Element	# of Observations	Mean value (μg/gm dry weight) With scuppers	Mean value (μg/gm dry weight) Without scuppers	Percent Probability
Zn	8/7[a]	96.9	42.0	99.60
Pb	8/7	423.0	132.0	99.99
Cr	8/7	23.9	11.0	97.07
Ni	8/7	7.2	2.8	99.60
Cu	8/7	80.1	29.2	98.71
Fe	8/7	1689.0	643.0	99.85
Cd	8/7	0.5	0.3	91.38

[a]Eight samples were collected from sediments beneath the south bridges with scuppers and seven samples from beneath the north bridges without scruppers.

Table 13. Differences in heavy metal concentrations of bottom sediments from Maitland interchange (60).

Element	Mean values (μg/gm, dry weight)		
	Lake Lucien	Outfall into Lake Lucien	West Pond
Zn	21.1	119.6	35.2
Pb	3.4	75.5	98.4
Cr	2.5	15.3	33.9
Ni	1.2	6.0	10.7
Cu	5.0	40.0	12.0
Fe	421.4	2053.0	3265.0
Cd	0.1	0.6	0.5

Table 14. Statistical analysis of extractable heavy metals in soil samples collected from flood plains receiving highway bridge runoff (60).

Location	Element	Control			Influenced		
		# of Samples	μg/g Oven dry wt. Mean	S.D.	# of Samples	μg/g Oven dry wt. Mean	S.D.
S. R. 17-92 and	Zn	9	2.1	0.8	14	46	40
Shingle Creek	Fe	9	487	108	14	1213	689
	Pb	9	3.5	1.3	14	273	339
	Cr	9	1.3	0.7	14	5.7	4.3
S.R. 192 and	Zn	4	17	14	12	7.7	0.4
Shingle Creek	Fe	4	302	94	12	607	164
	Pb	4	11	7.4	12	36	4.7
	Cr	4	1.7	2	12	3.6	3.1
Interstate 4	Zn	4	50	13	8	253	159
and Gemini Creek	Fe	4	3392	1485	8	5738	1202
	Pb	4	93	40	8	2174	2128
	Cr	4	16	6.8	8	17	4.0

Table 15. Statistical analysis of extractable heavy metals in plant samples collected from flood plains receiving highway bridge runoff (60).

		Control			Influenced		
		# of	µg/g Oven dry wt.		# of	µg/g Oven dry wt.	
Location	Element	Samples	Mean	S.D.	Samples	Mean	S.D.
S.R. 17-92 and	Zn	8	54	55	9	75	43
Shingle Creek	Fe	8	168	151	9	374	546
	Pb	8	14	7	9	41	30
	Cr	8	2.3	1.5	9	3.5	2.4
S.R. 192 and	Zn	4	56	19	10	62	20
Shingle Creek	Fe	4	112	35	10	159	65
	Pb	4	8.2	2.2	10	25	17
	Cr	4	1.7	1.0	10	2.1	1.3
Interstate 4	Zn	6	51	13	9	59	12
and Gemini Creek	Fe	6	155	83	9	174	97
	Pb	6	22	4.6	9	31	17
	Cr	6	3.5	1.8	9	2.7	1.7

Another study was conducted near Miami, Florida to document concentrations of several pollutants found in highway runoff in the water and bottom sediments from ponds near a major highway interchange (confluence of I-95, I-395, and SR836) (61). The ponds receive wet weather discharges from the highway site, as well as dustfall from vehicular traffic and background dustfall. The average daily traffic at this highway site is nearly 200,000. Water and sediment were sampled one time in the two ponds near the interchange during October, 1977. The results of this sampling indicated that the pollutant levels were generally in the range of concentrations found in uncontaminated aquatic environments in south Florida. Chromium in the pond water and lead in the sediments did exceed the reported background levels. The chromium concentration in the pond water averaged 20 μg/l. The average background level is 4.0 μg/l (with a standard deviation of 109 μg/l). Lead concentrations in the sediments were 500 μg/g which is considerably higher than background levels which ranged from 10 to 38 μg/g.

The effect of highway stormwater runoff on Back Creek sediments (near Roanoke, Virginia) was studied by Van Hassel, et al. (62). They measured metal (lead, zinc, nickel and cadmium) concentrations in sediments at three sites. The first site served as a control (upstream of all highway runoff influence). The second site was further downstream and was influenced by runoff from a small highway (Route 220) with a traffic volume of 6,550 vehicles per day. The third site was the furthest downstream and was also influenced by Route 220 which had a traffic volume of 15,000 vehicles per day. Water quality was comparable at all sites. Figure 3 shows the distribution of sediment metals over the entire study period (one year). Lead, nickel, and zinc concentrations were highly correlated with traffic volumes (P less than 0.003). Cadmium concentrations were not (P greater than 0.15). The seasonal distributions of metals for all three sites combined are shown in Figure 4. Lead and zinc concentrations are highest for all areas in spring and decreased with the passing of each subsequent season. Seasonal variations of nickel and cadmium were not significant. The authors attributed elevated spring

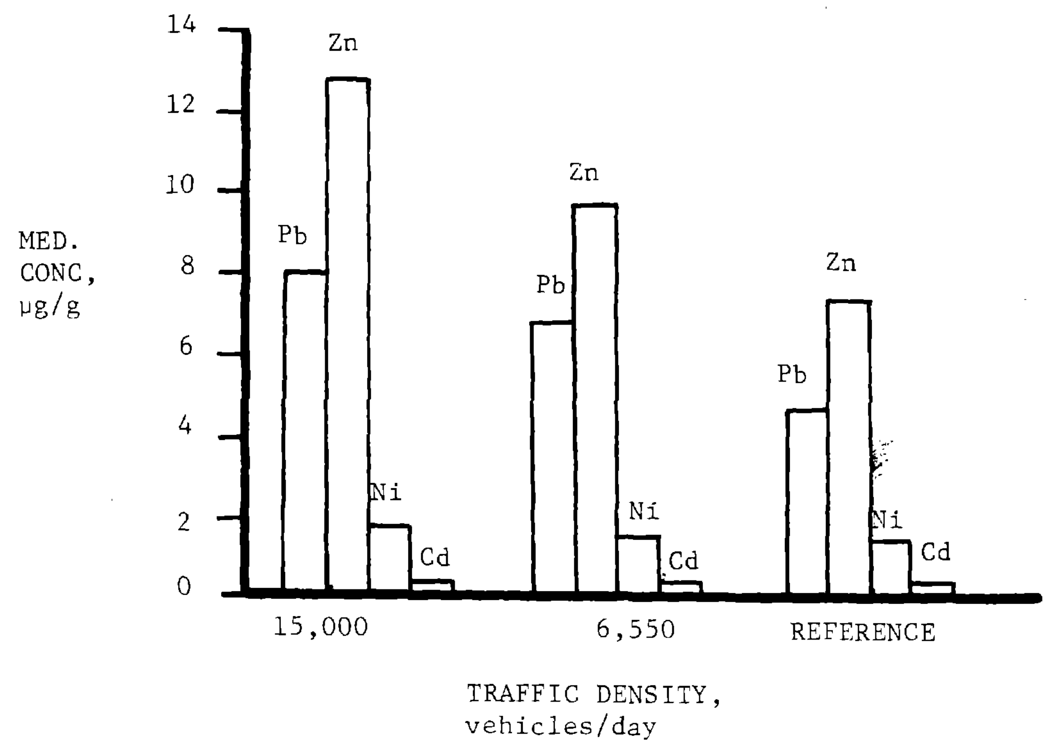

Figure 3. Median concentration of lead, nickel, cadmium, and zinc in sediments at each area over a one-year period (62).

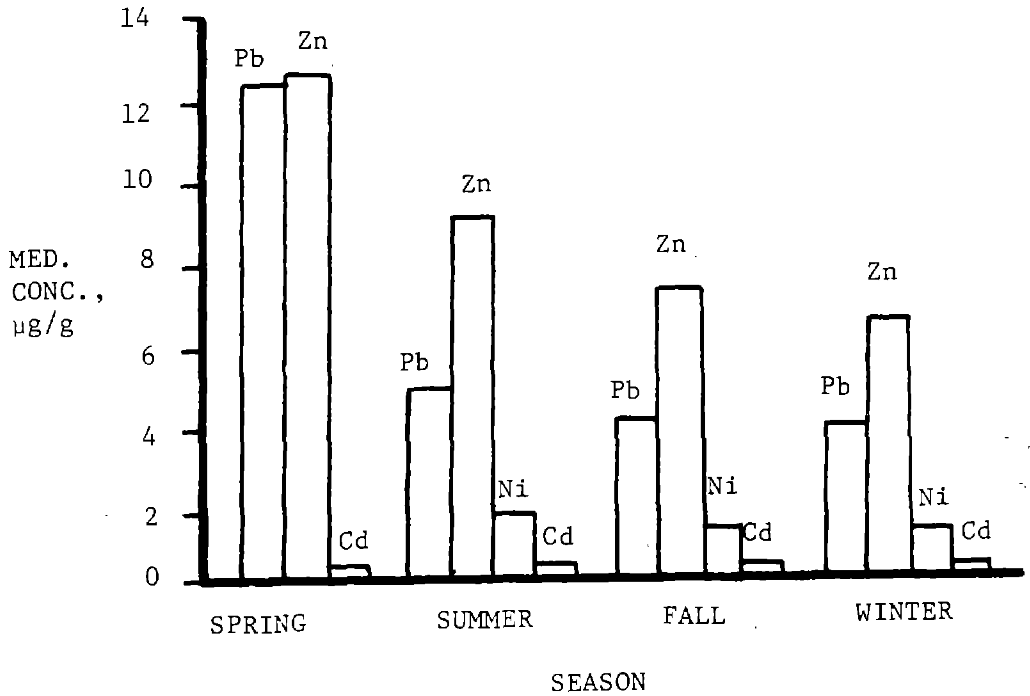

Figure 4. Median concentrations of lead, nickel, cadmium, and zinc in sediments for combined areas over a one-year period (62).

concentrations to snowmelt runoff loads which become dissipated throughout the rest of the year within the receiving water.

Wisseman and Cook (63) documented sediment metals accumulations in Wapato Lake in Tacoma, Washington due to urban stormwater drainage. The watershed studied was 76 percent urban residential, 13 percent undeveloped park land, 8 percent Interstate Highway 5, and 3 percent lake surface. Traffic volume on I-5 was about 80,000 vehicles per day. Much of the watershed is drained into the lake through the north shore storm drain, including 1.7 mi (2.7 km) of I-5 and 11 mi (17.7 km) of city streets. Sediment sample transects indicated strong negative correlations between chromium, lead, zinc, and cadmium and distance from the north shore storm conduit (Figure 5). Sediments on the east shore of the lake (control) exhibited no such trend and contained significantly lower metals concentrations.

DEICING AGENTS

As discussed in Volume II, salt concentrations in sediments at the Wisconsin Highway 15/Sugar Creek site were not higher in areas exposed to highway runoff relative to controls during both summer and winter surveys. Salt concentrations in vegetation samples (cattail and willow) were elevated in runoff influenced areas relative to controls and background values reported in the literature. However, these salt concentrations had no apparent effect on the health or growth of these species.

Only slightly elevated chloride concentrations were observed in most sediment samples in runoff influenced segments of Sevenmile Creek (I-85, North Carolina). At one station, sediment chloride was more than triple that in control station sediments during the September survey. Elevated sodium concentrations were not observed during either sediment survey at this site.

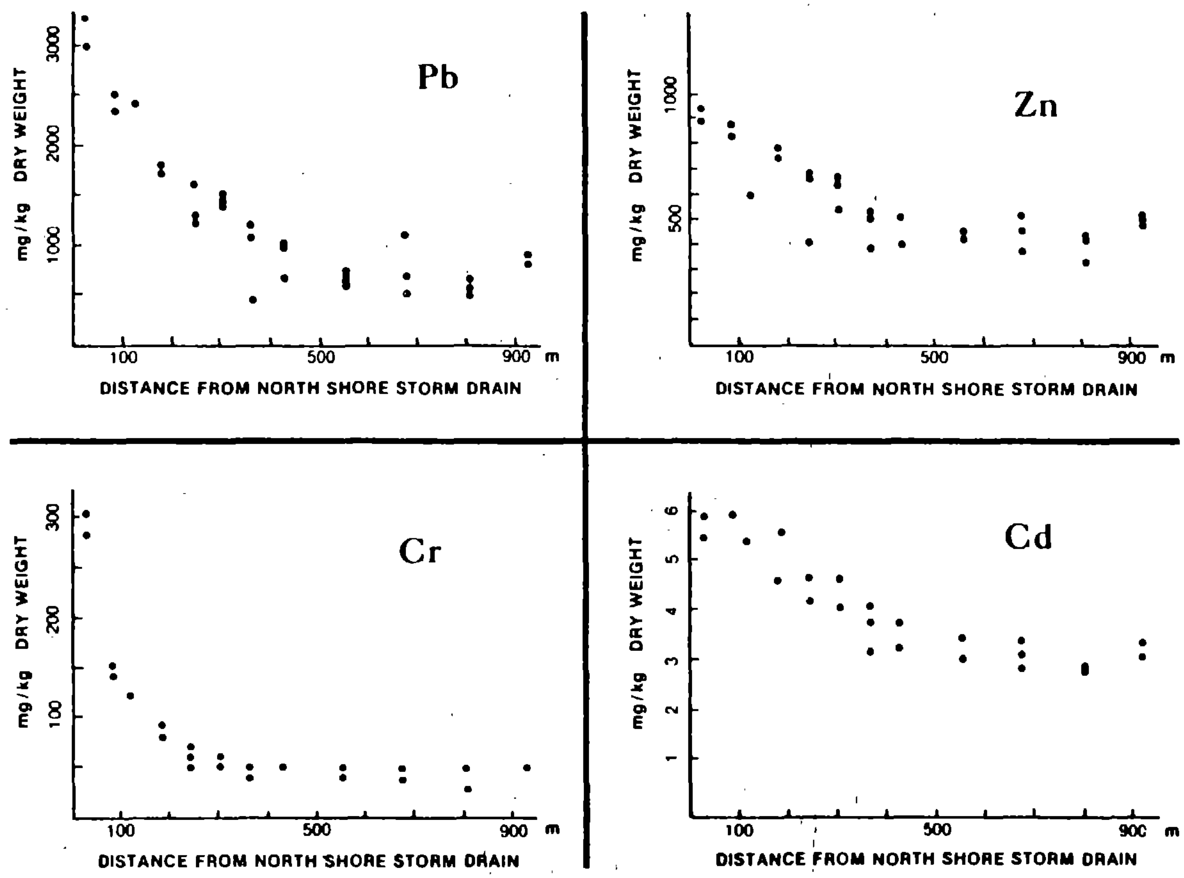

Figure 5. Concentrations of heavy metals in Wapato Lake surface sediments with increasing distance from the north shore storm drain (63).

Salt concentrations in core sample sediments at the I-94/Lower Nemahbin Lake site were not, in general, higher at highway influenced stations relative to control stations. One particular core sample taken near a bridge deck scupper drain did exhibit high chlorides (160 mg/kg compared to control station values less than 20 mg/kg). Salt concentrations in cattails adjacent to bridge deck drains were elevated compared to control stations and reported background literature values. However, as with the Sugar Creek vegetation, the cattails appeared healthy, productive and showed no visible signs of toxicity (chlorosis).

The effect of application of deicing agents to roads on aquatic sediments of First Sister Lake (Michigan) was investigated by Judd (15). This small lake (3.2 acres [1.3 ha]) receives stormwater runoff from both a residential area and I-94. Tables 16 and 17 list the results of these experiments. The interstitial water salt concentrations from a core sample from First Sister Lake indicated that chloride levels in the upper mud layer were equal to the levels in water above the mud; they increased by 70 ppm in the next lowest layer and then decreased. Sodium concentrations continually decreased with mud depth. A laboratory experiment was then conducted in which one mud sample was directly exposed to lake water containing sodium and chloride ions, and another mud sample was enclosed in a plastic bag and placed in the water. Interstitial water from the exposed mud showed elevated sodium and chloride levels compared to the interstitial water from the enclosed mud. This demonstrated the transport of salt ions into the sediments.

PETROLEUM HYDROCARBONS AND SYNTHETIC ORGANIC COMPOUNDS

The accumulation of petroleum hydrocarbons and synthetic organic compounds in receiving water sediments as a result of highway stormwater runoff discharges has not been extensively studied to date. Oil and grease concentrations were measured in sediment samples collected from the three receiving water sites described in Volume II of this report. Although several

Table 16. Core sample interstitial water - First Sister Lake (15).

Sample	Depth into mud (cm)	Sodium (ppm)	Chloride (ppm)	Spec. cond.[a] (μmho/cm)	Total ions (ppm)
Water immediately above bottom	0.0	55	113	468	323
"Soft" muds	8.6	46	113	395	293
"Hard" muds	13.6	40	183	642	444
Blue clay	26.6	35	134	468	323

a. Specific conductance at 18°C

Table 17. Laboratory experiment - salt transport into muds (15).

Column	Water Sample	Sodium (ppm)	Chloride (ppm)	Spec.[a] cond. (μmho/cm)	Total ions (ppm)
A	Enclosed mud interstitial	24	78	290	201
A	Test water	36	157	390	270
B	Exposed mud interstitial	33	133	443	306
B	Test water	39	165	395	273

a. Specific conductance at 18° C.

discrete core samples showed higher oil and grease levels at runoff influenced stations compared to controls, there was no consistent pattern of oil and grease accumulation in sediment at any of the three sites. At the I-94/Lower Nemahbin Lake site, a fuel oil spill occurred on the bridge deck. About 55 gallons of oil was spilled into a marshy area adjacent to the lake. Three months after the spill, oil and grease levels in the sediments at the point of the oil entry to the marsh were an order of magnitude higher than at a sampling location 200 feet (63 m) away, where levels were double those in control areas.

The presence of polynuclear aromatic hydrocarbons in Lake Padderudvann bottom sediments was confirmed by Gjessing, et al. (47) (See Table 6 for a description of the site characteristics). PAH's identified included phenanthrene, anthracene, fluoranthene, pyrene, triphenylene/chrysene, benzofluouranthenes, and benzo (e) pyrene. Only two sediment samples were analyzed for PAH (total PAH concentrations of 920 and 380 mg/l) so little can be said of spatial effects or the relative impact magnitude.

BIOLOGICAL IMPACTS

INTRODUCTION

Field documentation of highway runoff effects on biota of streams, rivers, lakes, and wetlands is not prevalent in the technical literature. The few studies which were found are described in this section. One comprehensive field and laboratory bioassay program was conducted as part of the field monitoring portion of this contract (see Volume II; Results of Field Monitoring Program). These results are briefly described in this volume. Several other bioassay studies have also been performed with highway runoff, by other researchers and these studies are discussed in detail in this section. Limited discussion is also provided on the effects of individual constituents of highway runoff, especially toxics. It must be kept in mind that these effects might not accurately reflect the impacts these constituents exert in complex runoff mixtures in actual receiving waters. These data do not account for synergistic or antagonistic effects, pollutant associations with particulates, or dynamic hydrological conditions. As a result, selected literature on field studies of urban stormwater runoff effects is also included in this section. But again, these effects should be considered in a qualitative sense only, due to differences in the potential magnitude of the impact of urban versus highway loadings of pollutants.

TOXICITY OF INDIVIDUAL RUNOFF CONSTITUENTS

Considerable effort has been expended in the last decade towards laboratory determination of acute and chronic toxicity of a wide range of individual pollutants, including most highway-generated toxicants (especially metals). One concise source for those bioassay data is the series of water quality criteria documents published by EPA in 1980 (21). A separate volume was prepared for each toxicant.

EPA used these bioassay data to derive criteria for protection of assemblages fresh and saltwater aquatic life. These criteria were listed earlier in this report (See Table 3). In 1984, EPA issued new draft criteria for several metals (23). These criteria, along with the 1980 criteria for the other metals (i.e., Ni and Zn) are summarized in Table 18. These EPA criteria, and the bioassays on which they are based, describe cause and effect relationships for the laboratory-grade, soluble, form of a given toxicant. The form and effect of these constituents as found in highway runoff can be quite different. Therefore, more discussion will be devoted later to bioassays with whole runoff and field studies.

Metals

Figure 6 shows the relative toxicities of metals often found in highway runoff in both fresh and saltwaters (64). Although lead is often considered to be the most troublesome pollutant in highway runoff, this perception derives more from its presence and abundance due to use of leaded gasoline rather than its relative toxicity. In order to place the ensuing discussion of metals toxicity in perspective, Table 19 provides a synopsis of metals concentrations likely to be found in highway runoff. Note that these represent undiluted runoff concentrations. The concentrations of highway derived metals in any aquatic environment would be lowered as a result of both sedimentation and dilution. At locations where the highway ROW constitutes the majority of upstream drainage area (i.e., headwater tributaries), the dilution potential would be quite low. At other sites, the dilution potential could easily be orders-of-magnitude.

Comparing the runoff concentrations in Table 19 with the EPA criteria in Table 18, it is apparent that undiluted highway runoff often exceeds aquatic life criteria for certain metals. If dilution ratios (roughly estimated by the ratio of total watershed to paved ROW areas) are greater than 10:1, as they usually are, it appears that EPA acute criteria would generally not be

Table 18. EPA Water Quality Criteria for Metals for Protection of Freshwater Aquatic Life.

Metal	Criteria Issuance Date[a]	EPA Acute Criteria (Maximum), at Water Hardness (mg/1 as $CaCo_3$) of:		
		50	100	200
Hg, μg/1	1984	1.1	1.1	1.1
Pb, mg/1	1984	0.025	0.064	0.160
Zn, mg/1	1980	0.18	0.32	0.57
Cu, mg/1	1984	0.0084	0.016	0.029
Cr, mg/1[b]	1984	0.87	1.50	2.70
Cd, mg/1	1984	0.002	0.0045	0.010
Ni, mg/1	1980	1.1	1.8	3.1

Metal	Criteria Issuance Date[a]	EPA Chronic Criteria (30-day averaging period), at Water Hardness (mg/1 as $CaCo_3$) of:		
		50	100	200
Hg, μg/1	1984	0.2	0.2	0.2
Pb, mg/1	1984	0.001	0.0025	0.064
Zn, mg/1	1980	0.047[c]	0.047[c]	0.047[c]
Cu, mg/1	1984	0.0058	0.011	0.020
Cr, mg/1[b]	1984	0.042	0.074	0.130
Cd, mg/1	1984	0.002	0.0045	0.010
Ni, mg/1	1980	0.056[c]	0.096[c]	0.16[c]

a. 1984 criteria (ref. 23), 1980 criteria (ref. 21).
b. Trivalent chromium
c. 24-hr averaging period.

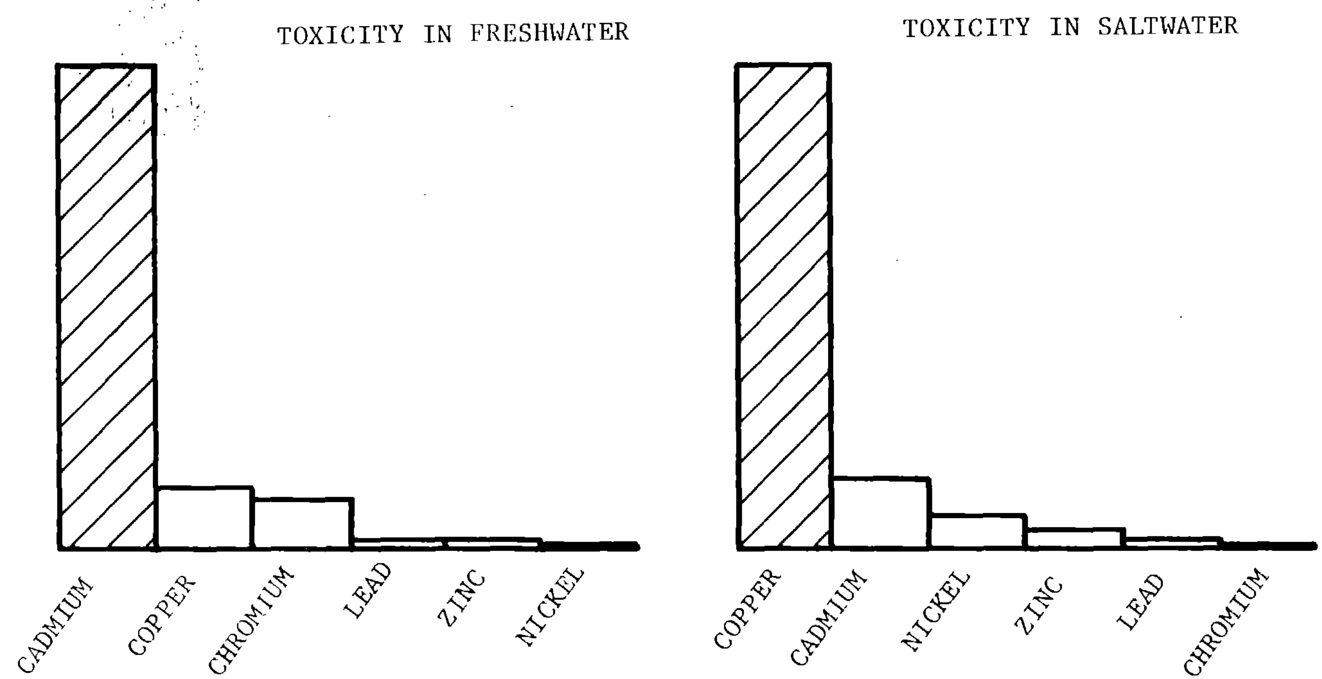

Figure 6. Relative toxicities of heavy metals in freshwater and in saltwater (64).

Table 19. Concentrations of metals in highway runoff (1).

Description of Highway Site	Mercury, µg/l		Lead, mg/l		Zinc, mg/l		Copper, mg/l		Chromium, mg/l		Cadmium, mg/l		Nickel, mg/l	
	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range	Avg.	Range
I-794 (Milwaukee), all paved, ADT = 53,000, urban	2.9	0.13-24	2.90	0.80-13.1	0.69	0.14-3.40	0.16	0.01-0.66	0.057	0.01-0.14	0.068	0.01-0.40	-	-
WI Hwy 45 (Milwaukee), 31% paved, ADT = 85,000, urban	5.2	0.20-67	1.20	0.40-6.6	0.55	0.20-1.90	0.14	0.01-0.88	0.054	0.01-0.14	0.044	0.01-0.09	-	-
WI Hwy 45 (Milwaukee), grassy ditch only, 0% paved, ADT = 85,000, urban	1.5	0.25-11.5	0.21	0.05-0.70	0.18	0.07-0.34	0.083	0.01-0.23	0.04	0.01-0.10	0.047	0.02-0.10	-	-
I-81 (Harrisburg), 27% paved, ADT = 24,000, rural	4.9	0.25-49	0.10	0.05-0.20	0.08	0.01-0.23	0.045	0.01-0.10	0.025	0.01-0.11	0.025	0.01-0.07	-	-
I-40 (Nashville), 37% paved, ADT = 88,000, urban	1.8	0.50-6.7	0.50	0.02-1.70	0.28	0.10-0.61	0.70	0.01-0.20	0.023	0.01-0.05	0.027	0.01-0.06	-	-
I-25 (Denver), 37% paved, ADT = 149,000, urban	1.1	0.25-4.0	0.45	0.30-1.80	0.72	0.33-1.50	0.11	0.03-0.26	0.03	0.01-0.09	0.02	0.01-0.08	-	-
Overall (all sites)	3.2	0.13-67	0.96	0.02-13.1	0.41	0.01-3.4	0.103	0.01-0.88	0.04	0.01-0.14	0.04	0.01-0.40	0.01	0.0001-0.05

exceeded for most highway/receiving water interactions. Regions of the United States with low water hardness, such as the Southeast and Northeast (Figure 7), would be more susceptible to criteria exceedances since allowable metals concentrations increase with increasing water hardness.

Stromgren (65,66,67) tested five fucoid algae for metals toxicity. Copper was more toxic than mercury, and both were far more toxic than zinc, lead, and cadmium. Exposure to mercury at concentrations from 100 to 200 μg/l for 10 days produced a 50 percent reduction of growth rate. A lead concentration of 2600 μg/l caused a 50 percent reduction in growth rate for three species, and cadmium produced the same effects at a similar concentration. A 50 percent reduction in growth rate takes place at 0.06 59 0.06 mg/l of copper and at 5 to 10 mg/l of zinc.

Lead is a cumulative poison, and its chemistry in freshwaters is dominated by its tendency to form complexes with inorganic and organic complexing agents. Lead present in natural waters has the tendency to be adsorbed by clays or complexed with dissolved or suspended organics and with hydroxides of iron, aluminum, and manganese (68). Davies (69) has concluded that soluble lead is probably the primary toxic form in both acute and chronic exposures and that pH affects the relative proportions of soluble, colloidal, and precipitated lead. Exposure to 0.030 mg/l total lead (hardness 45.3 mg/l calcium carbonate) has been shown to cause a 16 percent reduction in the reproductive capacity of Daphnia magna (70). The quantity of lead accumulated by freshwater invertebrates usually reflects that found in the sediments (71,72).

The zinc ion is the principal toxic form of zinc. To a lesser degree, the zinc present as the basic carbonate and hydroxide held in suspension can also be toxic (73). At very high concentrations, zinc-contaminated sediments have been found toxic to chironomid larvae (74,75,76). For fish (73) and invertebrates (77,78,79), the toxicity of zinc decreases with increasing

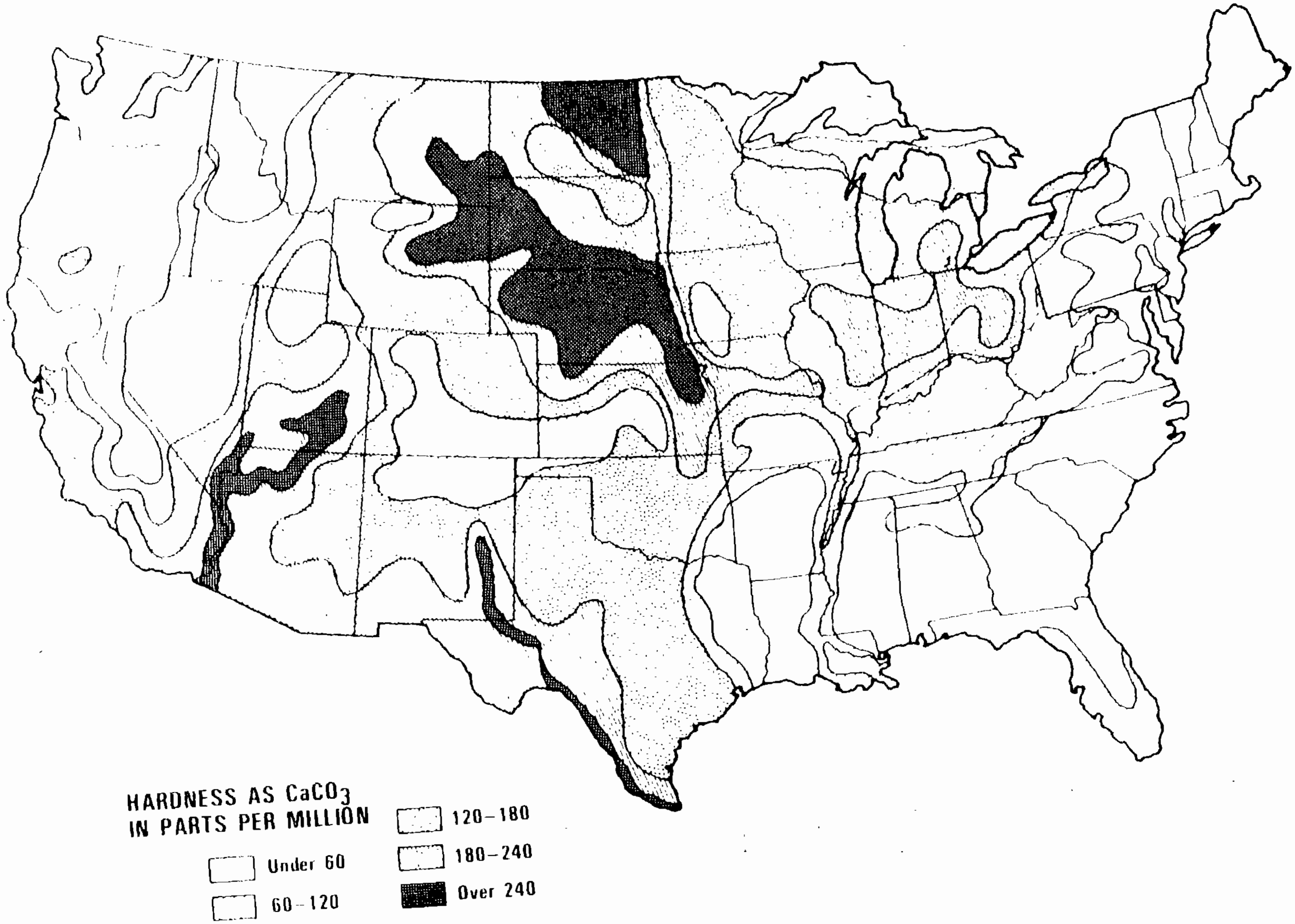

Figure 7. Regional Values for Surface Water Hardness (2).

hardness. Researchers have also found that increasing alkalinity in water with a constant hardness does not reduce zinc toxicity to the oligochaete, Tubifex tubifex (77). Increases in pH above the value of 7 decrease the solubility of zinc salts in the aquatic environment (68). Dissolved oxygen, salinity, and the quantity of organic complexing agents present also affect the toxicity of zinc to fish (73,80). Little bioaccumulation of zinc occurs in deposit-feeding clams when zinc is coprecipitated with amorphic iron oxide or manganese oxide (81).

In surface waters under aerobic conditions the stable oxidation state of copper is the cupric form. The cupric ion forms complexes with inorganic and organic complexing agents such that approximately one percent of the total copper present is the free cupric form (82). Research indicates that the only toxic form of copper to freshwater fish and invertebrates is probably the free cupric ion (83,84,85,86,87). It appears that the total copper concentration producing a lethal threshold approximates the quantity of copper required to satisfy the chelating capacity of a receiving water body (83). Factors affecting copper toxicity to aquatic organisms are temperature, dissolved oxygen, and pH (82,89). The lowest concentration at which sublethal or lethal effects have been reported in the literature are within the range of 0.004 to 0.01 mg/l total copper (68). High quantities of sediment-bound copper have been noted in rivers receiving chronic copper inputs (71,90). However, no evidence exists of biomagnification along the food chain. Klotz (91) found decreased toxicity in algal species at optimal temperatures despite relatively high cellular concentrations of copper.

The three principal oxidation states of chromium in the environment are Cr^{2+}, Cr^{3+}, and Cr^{6+} of which the hexavalent form (Cr^{6+}) is probably the most toxic (68). Chromium toxicity to freshwater invertebrates is dependent on hardness and temperature (77,89). Trivalent chromium concentrations of 0.6 and 0.33 mg/l cause 50 percent and 16 percent reproductive impairment in Daphnia magna (70). Chromium has the potential for bioaccumulation in marine organisms. Tubificid worms have been shown to accumulate significant quantities of chromium when given metal-contaminated bacteria as a food source (92). Marine annelids show toxic effects to hexavalent chromium at levels as low as 0.0125 mg/l; however, concentrations in the range of 0.001 to 0.010 mg/l are believed to produce an effect (93).

Soluble cadmium is a cumulative poison and toxic at extremely low concentrations (68). Studies by Wentsel, et al. (75,74,76,94) have shown high levels of sediment-bound cadmium to affect freshwater macroinvertebrates. Cadmium toxicity to freshwater animals is dependent on hardness (77,95,96), organic complexing (97), pH (96), temperature (96), and dissolved oxygen concentration (96,98). Clubb, et al. (98) observed that a decrease in oxygen concentration reduced both toxicity and bioaccumulation of cadmium in the larvae of freshwater insects. Biesinger and Christiansen (70) found that a total cadmium concentration of 0.0007 mg/l caused a 50 percent reproduction impairment in Daphnia magna during a three-week period, while a concentration of 0.00017 mg/l caused a 16 percent reproductive impairment.

Frank and Robertson (99) studied the influence of salinity on cadmium and chromium toxicity to Blue Crabs (Callinectes sapidus). Cadmium was more toxic than chromium at all salinities. The 96-h LC_{50} at 1-ppt salt was 0.32-mg/l cadmium and 34.2-mg/l chromium. Cadmium and chromium toxicity decreased with increasing salinity.

The chemistry of nickel in freshwater is not well-understood (68). Water bodies receiving nickel inputs appear to accumulate this metal in sediments (90). Temperature, pH, and hardness appear to affect nickel toxicity (68). An increase in water hardness produces a decrease in nickel toxicity to aquatic organisms (77,100). A nickel concentration of 0.03 mg/l causes a 16 percent reproductive impairment in Daphnia magna using a dilution water of 45.3 mg/l calcium carbonate total hardness (70). Fathead minnows displayed no toxic effect at nickel concentrations of 0.380 mg/l (water hardness 200 mg/l calcium carbonate), while a nickel concentration of 0.73 mg/l decreased both the number of eggs produced and the ability of the eggs to hatch (101).

Mercury chemistry in freshwater is complex and poorly understood (102,103,104). Inorganic sediment-bound mercury is biochemically transformed in anaerobic sediments and released to the water column as methyl mercury, which is the more toxic form and that in which mercury is accumulated by aquatic biota (105,106). In contrast to most heavy metals, hardness appears to have only a slight effect on the toxicity of inorganic mercury (68). A mercuric chloride concentration of 0.0034 mg/l produced a 16 percent reproductive impairment in Daphnia magna (70). A major concern with mercury compounds in aquatic environments is the possibility of biomagnification in food chains with high levels in predatory fish (107). The ciliate Uronema nigricans was found to acquire a mercury tolerance within a single generation time (108). Luoma (109) found wide temporal fluctuations of mercury in shrimp because they rapidly concentrated soluble mercury which periodically entered an estuary in stormwater runoff. The shrimp obtained minimal mercury from food (sediment), although sediment-bound mercury was the most concentrated source in the estuary. Grolle and Kuipper (110) found that a concentration of 0.5 μg/l mercury affected species composition of a periphyton community. Concentrations of 5 μg/l mercury produced a small change in the growth pattern and a reduction in the number of species compared to controls. Concentrations of 50 μg/l mercury inhibit the growth of periphyton completely.

Several studies indicate that metals in high enough concentration are toxic to submerged, floating, and emergent plants (111,112,113,114,115,116, 117). Heavy metals in combination with other compounds have long been used as herbicides for the removal of "troublesome" wetland species. The levels that are toxic need to be evaluated on an individual species basis. Ernst and Van der Werff (118) found different resistances to copper toxicity for each of the wetland species they studied. Because of the complex mechanisms which regulate the availability of metals for uptake, few studies have been able to establish recommended levels of exposure or toxicity for wetland plants.

The primary mode of action of heavy metals in plants is inactivation of certain enzymes. These enzymes are invariably important in various metabolic activities of plants, such as the biochemical reactions associated with photosynthesis. The sensitivity of enzymes seems to be metal specific, so no generalizations can be made. Some enzymes appear to be unaffected by heavy metals. Some heavy metals remain in cell walls in the roots, but others are translocated to leaf tissue and are compartmentalized and isolated from the biologically active part of the cell. Other metals complex with metabolized compounds and may be rendered inactive. Other species have evolved naturally resistant strains.

Petroleum Hydrocarbons and Pesticides

Acute lethal toxicity and long term sublethal toxicity of petroleum products are summarized in EPA's publication on quality criteria for water (22). Parker *et al.*, (119) concluded that toxicity test data, especially chronic exposure studies for oil contamination in freshwater species, are exceedingly sparse.

Acute lethal toxicity and sublethal toxicity of common pesticides are summarized in EPA's publications on quality criteria for water (21,22). Direct effects of pesticides on biota are those that affect growth, survival, or reproduction. A summary of acute toxicities of organochlorine and organophosphorus insecticides and herbicides are summarized in Tables 20 and 21. Secondary effects are those that take place in an ecosystem following, and as a result of, direct pesticide effects (120). Secondary effects in aquatic ecosystems were reviewed by Hurlbert (120) and include:

Table 20. Acute toxicities of organochlorine and organophosphorus insecticides to various organisms (120).

	Estuarine[a] phyto- plankton (photo- synthesis reduc- tion, %)	Parts per million										
Insecticide		Daphia[b] (48-hr EC_{50})	Fresh water amphipod (96-hr LC_{50})	Three marine decapods (96-hr LC_{50})	Mosquito[c] larvae (24-hr LC_{50})		Stonefly naiad (96-hr LC_{50})	Fresh- water minnow (96-hr LC_{50})	Twelve freshwater fish (96-hr L_{50})	Seven estuarine fish (90-hr LC_{50})	Tadpoles[d] (96-hr LC_{50})	Mallard duck Single dose LD_{50} (mg/kg body wt)
Organochlorine insecticides												
Aldrin	85	28	9,800	8-33	5.3	--	1.3		--	5-100	150	520
BMC	--	--	--	--	--	--	--		--	--	3,200	--
Chlordane	94	29	26	--	--	--	15	69	--	--	--	1,200
DDT	77	0.36	1.0	0.6-6	70	10	7.0	34	2-21	0.4-89	1,000,800*	>2,240
Dieldrin	85	250	460	7-50	7.9	3	0.50	16	--	0.9-34	150,100*	381
Endrin	46	20	3.0	1.7-12	15	--	0.25	1.3	--	0.05-3.1	120,180*	5.64
Heptachlor	94	42	29	8-440	5.4	--	1.1	56	--	0.8-194	440	≥2,000
Lindane (gamma BMC)	23	460	48	5-10	27	--	4.5	56	2-131	9-66	4,400,2,700*	≥2,000
Methoxychlor	81	0.78	0.8	4-12	67	--	1.4	35	--	12-150	330	≥2,000
Mirex	42	--	--	--	--	--	--	--	--	--	--	≥2,400
Thiodan	87	--	5.8	--	--	--	--	--	--	--	--	33
Toxaphene	91	15	26	--	--	--	2.3	5.1	3-18	--	140,500*	71
Organophosphorus insecticides												
Abate	--	--	82	--	1.6	5	10	--	--	--	--	90
Baytex	7	0.80	8.4	--	4.2	23	4.5	--	930-3,404	--	--	5.94
Chlorthion	--	--	--	--	25	--		3,200	--	--	--	--
Diazinon	7	0.90	200	--	83	--	25	--	--	--	--	3.54
Dibron	56	0.35	110		--	--	8.0	--	--	--	1,700*	52.2
Dichlorvos	--	0.066	0.50	4-45	75	110	0.10	--	--	200-2,680	--	7.78
Dipterex	--	0.18	40	--	--	--	35		--	--	--	--
Dutsban	--	--	0.11	--	2.8	6.3	10	--	--	--	--	75
EPN	--	--	15	--	44	--	--	250	--	--	--	3.08
Fenitrothian	--	--	--	--	5.6	25	--	--	--	--	--	1,190
Guthion	0	--	--	--	--	--	1.5	12,500	4-4,270	--	130	--
Malathion	7	1.8	1.8	33-83	80	320	10	7,500	101-12,900	27-3,250	420,200*	1,485
Methyl parathion	--	--	--	2-7	18	6.5	--	1,600	2,750-9,000	5,200-75,800	--	10
Parathion	--	0.60	0.60	--	3.2	3.1	5.4	--	--	--	1,000*	2
Phosdrin	--	0.16	0.16	11-69	--	--	5.0	--	--	65-800	--	4.63
Phosphamidon	--	8.8	8.8	--	--	--	150	4,200	--	--	--	3.05
Systox	7	--	--	--	--	--	--	1,000	--	--	--	7.19
TEPP	--	--	--	--	--	--	--		--	--	--	3.56

a. Percentage decrease in productivity of a natural estuarine phytoplankton assemblage during a four-hour exposure to 1,000 ppb.
b. Daphnia poles at 15.6° C; EC_{50} = concentration required to immobilize 50% of test organisms.
c. First column: _Culex fatigans_, Second Column: _Anopheles albimanus_.
d. _Bufo woodhousii_ (no asterisk) and _Pseudacris triseriatu_ (asterisk).

Table 21. Acute toxicities of herbicides to various organisms (120).

Herbicide	Estuarine phytoplankton (photosynthesis reduction, %)[a]	Daphnia[b] (EC_{50})	Freshwater amphipod (96-hr LC_{50})	Stonefly naiad (96-hr LC_{50})	Bluegill (49-hr LC_{50})	Tadpoles[d] (96-hr LC_{50})	Mallard duck Single dose LD_{50} (mg/kg body wt)
Amtrole	--	9.8, 0.014	--	--	--	--	>2,000
Dalapon	--	23	--	>100	115	--	--
Dicamba	--	11*	3.9		130	--	--
Dichlobenil	--	--	11	7.0	20	--	>2,000
Dichlone	--	9.8, 3.7*	1.1	--	--	--	>2,000
Dinitrocresol	--	0.014	--	0.32	--	--	22.7
Diquat	--	--	--	--	91*	--	564
Divron	87.4	7.1	0.16	1.2	17*	--	>2,000
2,4 D acid	0	47, 1.4*	--	15	--	--	1,000
2,4 D butoxyethanol ester	--	>100	0.44	1.6	2.1	--	--
2,4 D dimethylamine salt	0	--	>100	--	166	100	--
2,4 D iso-octyl ester	--	--	2.4	--	8.8	--	--
Endothal	--	--	>100	--	0.257	--	--
Eptam	0	46	--	--	--	--	--
Fenac	--	--	4.5*	12	55	--	--
Fenuron	40.9	--	--	--	53**	--	>2,000
IPC	--	10*	10	--	32*	1.2*	--
Hydrothol 191	--	--	0.50	--	--	14*	--
Molinate	--	0.70	4.5	0.34	--	--	--
Monuron	94.1	106	--	--	1.8**	--	--
Naphtha	--	3.7*	0.84	2.3	--		--
Neburon	89.9	--	--	--	0.70	28, 26*	--
Paraquat	--	11, 3.7*	11	>100	400	--	>2,000
Pichloram	--	--	27	48	26.5*	--	--
Potassium azide	--	--	6.4	8.0	1.4	10, 10*	>2,000
Silvex	--	100, 2*	--	0.34	83*	--	--
Simazine	--	--	13	--	118	--	323
Sodium arsenite	--	6.5, 1.8*	--	38	58*	--	--
Sodium azide	--	--	5.0	9.2	0.98	0.10*	>2,000
Trifluralin	--	0.24*	2.2	3.0	8.4	--	--
2,4,5-T	0	--	--	--	0.50	--	--
Vernolate	--	--	0.84	--	9.7*		

a. Percentage decrease in productivity of a natural estuarine phytoplankton assemblage during a four-hr exposure to 1,000 ppb.
b. Values marked with asterisk (*) are 48-hr immobilization concentrations, those without asterisk are 26-hr immobilization concentrations.
c. When 48-hr LC_{50} values are not available, 24-hr (*) or 96-hr (**) values are given.
d. Values marked with asterisk (*) are for Fowler's toad (Bufo woodhousii) and those without asterisk are for Estern chorus frog (Pseudacris triseriata).

1. Reduction of invertebrate populations dependent on aquatic vegetation as a physical substrate or hiding place, increased populations of benthic detritus feeders, and increased abundance of chara and filamentous algae following destruction of aquatic vascular plants with herbicides;

2. Increased abundance of certain invertebrates following reduction of populations of their competitors; increased abundance of invertebrate predators that can utilize the replacing species;

3. Increased growth rates for surviving individuals in food-limited populations whose densities are lowered by pesticide-caused mortality;

4. Population declines of species that cannot sustain the competition of other species that have increased following pesticide-caused mortality of a predator that preyed preferentially on these other species;

5. Changes in oxygen, carbon dioxide, bicarbonate ion, calcium, and other chemical variables as a function of changes in phytoplankton and macrophyte populations.

Cyanides

Cyanides in highway runoff originate from anticaking compounds added to deicing salt. These compounds include yellow prussiate of soda (sodium ferrocyanide) and prussian blue (ferric ferrocyanide). Of these two anticaking additives, sodium ferrocyanide can photodecompose to yield simpler cyanide compounds that are potentially more harmful to aquatic animals than are the original cyanide complex (121). The toxic form of cyanide is not the ionic form, but the undissociated hydrocyanic acid (HCN). Cyanides are toxic due to their ability to inhibit oxygen metabolism; i.e., to render the tissues incapable of exchanging oxygen (22). For free cyanide (sum of cyanide present as HCN and CN^-), the criterion to protect freshwater aquatic life is 3.5 μg/l as a 24-hour average, and the concentration should not exceed 52 μg/l at any time (21). For saltwater aquatic life, acute toxicity occurs as low as 30 μg/l and chronic toxicity as low as 2.0 μg/l for the tested species (21). Cyanide toxicity is regulated by pH, temperature, light, dissolved oxygen, and dissolved metals (22,68). For example, if all other factors are held constant, the percentage of cyanide present in the undissociated form is inversely proportional to pH. More than 99 percent is in this form at pH 7 or below (122).

BIOASSAYS USING HIGHWAY RUNOFF

Bioassay tests performed with actual stormwater runoff from highways provide a much more meaningful evaluation of potential biological impacts. These studies account for the potential for both synergistic and antagonistic effects, the toxicant form (i.e., soluble or particulate-associated), and dilutional effects.

Extensive bioassay work was performed by the University of Wisconsin-Milwaukee Center for Great Lakes Studies as part of this same contract (See Volume II). Runoff was collected during a snowmelt event from a rural highway and during a spring rainfall event from an urban freeway. These runoff waters were assayed with a variety of freshwater organisms including chronic 14-day growth assays with algae (Selenastrum capricornutum) and acute 96-hour assays with a waterflea (Daphnia magna), an amphipod and isopod, a mayfly, and a fathead minnow. For the acute assays, only the gammarid (amphipod) exhibited lethality when exposed to undiluted runoff. Acute toxic effects were not apparent for the waterflea, isopod, mayfly or fathead minnow. Selenastrum did exhibit a significant chronic toxicity response (i.e., reduced growth) to undiluted runoff water.

Other bioassay studies using highway runoff have recently been completed. These include algal assays performed by the California Department of Transportation (123), 96-hour bioassays performed by the University of Washington (124), and a study done by researchers in Norway (125). These three studies are described in detail below. Note that all of these studies were intended to provide data only on acute effects of highway runoff. Further research is needed to address potential chronic or bioaccumulation effects.

California Department of Transportation Algal Assay Studies (123)

This study utilized the 5-day algal bioassay method to investigate the effects of roadway runoff on aquatic biota. Runoff samples for the bioassay tests were obtained at three field sites located throughout California, representing areas of high traffic volumes (Los Angeles with 185,000 average daily traffic [ADT]), medium traffic volumes (Walnut Creek with 66,000 ADT) and low traffic volumes (Placerville with 23,000 ADT). A total of 35 road-runoff samples from 10 storms at the three monitoring sites were tested for the effects of roadway runoff on algal productivity. Two additional field sites were also selected to acquire runoff samples from cut slopes. Two storms were studied at both slope-runoff sites. Bioassays were conducted using water containing an indigenous mixed algal population from Lake Natomas. Roadway runoff concentrations in the assay culture flasks were 0.1, 1.0, 5.0, and 10 percent. Initially, a 0.01 percent concentration was included but this concentration was terminated because it had little effect on bioassay results.

Table 22 is a comparison of selected metals and nutrient concentrations observed in roadway runoff and Lake Natamas waters. Nutrients and metals in the pavement runoff were considerably higher than the ambient levels in Lake Natomas water. The results of the bioassays were related to the contaminant concentrations of roadway runoff which were related to rainfall intensity, length of time between storm events, average daily traffic, and other factors related to the accumulation of roadway pollutants.

Table 22. Lake Natomas/pavement runoff constituents (123).

	Pavement runoff range	Pavement average	Lake Natomas
Metals[a]			
Iron	1000-76,000 μg/l	11,230 μg/l	161.3 μg/l
Total metals - Fe	930-33,200 μg/l	4,880 μg/l	90.1 μg/l
Lead (Pb)	400-9,800 μg/l	2,580 μg/l	10.3 μg/l
Zinc (Zn)	160-22,000 μg/l	2,400 μg/l	10.9 μg/l
Copper	30-320 μg/l	210 μg/l	9.9 μg/l
Nutrients[b]			
Nitrate (nitrogen)	0.35-18.0 mg/l	5.98 mg/l	0.03 mg/l
Kjeldahl nitrogen	1.1-36.0 mg/l	14.4 mg/l	0.13 mg/l
Ammonia	0.3-8.4 mg/l	3.35 mg/l	0.01 mg/l
Total phosphorus	0.13-1.39 mg/l	0.40 mg/l	0.02 mg/l
Ortho phosphorus	0.01-0.81 mg/l	0.12 mg/l	0.01 mg/l

[a] μg/l

[b] mg/l

When concentrations of runoff contaminants were highly, inhibition of algal productivity was evident. For example, runoff samples from a storm in Los Angeles which was preceded by 13 days of dry weather showed extreme inhibition of algal productivity at the 1, 5, and 10 percent levels. Also, for most assays which resulted in significant inhibition, there was a high proportion (85 to 95 percent) of heavy metals such as lead and zinc compared to total metals (excluding iron). In contrast to the relatively high contaminant levels which accumulated on roadways during dry periods, storms sampled within a few days of previous events usually had lower contaminant levels, as well as generally stimulatory algal assay results.

The following conclusions were made from the data collected as part of this study.

1. Highway runoff has the potential to significantly affect the algal component of aquatic communities. These impacts can be inhibitory or stimulatory depending on the chemical composition of the runoff.

2. The concentration of contaminants appears to be the important aspect of road runoff that affects algal growth, whether it is inhibited or stimulated. Heavy metals appear to be the constituent which inhibits algal growth. This study did not determine which metal or metals were toxic, or at what concentrations they become a problem. While the synergistic aspects of the various heavy metals were not investigated, it does appear that elevated levels of zinc and lead in combination are likely candidates for algal inhibition.

3. It appears that an elevated nutrient load in runoff was generally stimulatory, but that the presence of metals dictated the final bioassay results.

4. The removal of particulate materials by physically filtering the roadway runoff did not significantly alter the bioassay response. Cut-slope runoff bioassays were not extensive enough to determine the effects of filtering on algal response.

5. Runoff from suburban (Walnut Creek) and rural (Placerville) highways seems to be stimulatory in nature except when following a significant dry period which resulted in an early temporary inhibition followed by a stimulation phase.

University of Washington Bioassay Studies (124)

The overall objective of this research was to evaluate the effects of highway stormwater runoff on aquatic ecosystems. Evaluation parameters included the test for biochemical oxygen demand (BOD_5) plus bioassays to examine the response to runoff of a green alga, _Selenastrum capricornutum_, a zooplankter, _Daphnia magna_, and a native salmonid, _Salmo gairdneri_ (rainbow trout). These organisms represented three trophic levels and provided an integrated and comprehensive appraisal of the toxic effects of highway runoff water in spite of its chemical complexity and variability. Other project objectives were to:

1. Determine which chemical constituents were primarily responsible for any observed effects,

2. Identify means of mitigating those effects, and

3. Determine the extent to which the results of the study were site-specific.

Highway runoff samples used for most bioassays were collected from the northbound lanes of I-5 at Northeast 158th Street in Seattle, Washington. Average daily traffic (ADT) for the northbound lanes was approximately 50,000 vehicles. Runoff samples were also collected from a site on highway SR-520 (42,000 ADT) and Interstate 90 (7,700 ADT). Proportions of highway runoff used for bioassays included:

1. 100 percent lake water,

2. 75 percent lake water and 25 percent highway runoff,

3. 50 percent lake water and 50 percent highway runoff,

4. 25 percent lake water and 75 percent highway runoff, and

5. 100 percent highway runoff.

A consistent pattern of reduced algal biomass was observed as the proportion of highway runoff from I-5 increased, even though soluble phosphorus and nitrogen levels were equalized in both highway runoff and lake dilution waters. The data also indicated an inverse association between IC_{50} (that concentration which inhibits algal growth by 50 percent) and vehicles traveling during the storm. However, no apparent toxicity was observed in either SR-520 or I-90 samples. The lack of apparent toxicity at the I-90 site was anticipated because of the low ADT (7,700 vehicles/day). However, ADT at the I-5 and SR-520 sites were similar. The authors concluded that higher concentrations of soluble zinc and copper in I-5 runoff were apparently responsible for the observed toxic response.

Table 23 presents a comparison of the range of total metals concentrations found in I-5 runoff with the range of values found to be detrimental to organisms at three trophic levels. The data show considerable overlap between quantities observed in runoff and the levels known to endanger test organisms. The authors indicated that the results of this study suggest that metals are present in a biologically available form at harmful levels in some highway runoff samples.

Rainbow trout exposed to membrane filtered stormwater showed no harmful effects in a 4-day exposure (Table 24). However, bioassays using unfiltered samples resulted in significant mortalities in both 50- and 100-percent dilutions (Table 24). The authors concluded that either the particulates

Table 23. Range of total metals observed in runoff at Interstate-5 site as compared to LC_{50} ranges presented in literature (124).

Metal	Range of concentrations observed at I-5, $mg\ l^{-1}$ [a]	LC_{50} values presented in literature, $mg\ l^{-1}$ [b]	[test organism]
Pb	0.10 - 5.50	1.20 - 542.	[rainbow trout]
		0.45 - 1.90	[daphnial]
		0.50 - 1.00	[algae]
Cu	BDL - 0.50	0.02 - 0.89	[rainbow trout]
		0.01 - 0.50	[daphnia]
		0.006 - 8.00	[algae]
Zn	BDL - 2.50	0.28 - 7.21	[rainbow trout]
		0.10 - 0.66	[daphnia]
		0.10 - 1.20	[algae]

[a] Values obtained from Clark (126) and Asplund (127).
[b] Values obtained from U.S. Environmental Protection Agency (128,129,130).

BDL - Below detectable limit.

Table 24. Summary of bioassays exposing rainbow trout (<u>Salmo gairdner</u>) to I-5 stormwater runoff under various conditions (124).

Date of collection	Sample site	Storm no.	Vehicles during storm	Dates of exposure	Sample treatment prior to exposure	96-hour survival of exposures, %				
						Control	30%	50%	70%	100%
4-16-81	I-5	180	12,250	4-15 to 4-19	L. Washington dilution and storm sample 0.45 Millipore filtered	100	100	-	93.3	93.3
5-10-81	I-5	185	14,650	5-11 to 5-19	Used unfiltered L. Washington dilution and unfiltered storm sample	100	-	30	-	0
5-21-81	I-5	188	62,200	5-21 to 5-25	Same treatment as above, stormwater taken directly off the roadway	100	-	60	-	0
5-21-81	I-5	188	62,200	5-21 to 5-25	Same storm event as above, stormwater allowed to run 60 m through a grassy ditch adjacent to right-of-way	100	-	100	-	93.3

present in the samples or the pollutants associated with the solids were responsible for these deaths. Total suspended solids concentrations for these tests ranged from 35 to 97 mg/l. The following mechanism for mortality was postulated: high rates of respiration result in fish being stressed by the presence of suspended particulates; this stress leads to higher volumes of water passing over the gill and further reductions in pH in the region of the gills; under these conditions metals are more readily desorbed from particulates and can be taken up by the fish.

An experiment was designed to evaluate the proposed management option of allowing highway runoff to travel through grass channels prior to entering a receiving water body. This test involved conducting bioassays using runoff directly from the paved road surface and runoff from the end of the grass channel. Table 24 presents the results of this experiment. Data show that the water which travelled through the grass channel was considerably less toxic than the water directly off the paved surface. Total suspended solids (TSS) concentration for the untreated sample was 97 mg/l, while the TSS level in the end-of-channel sample was reduced to 24 mg/l. Flow diversion through the channel resulted in 61 to 77 percent removal of acid-extractable zinc and lead, respectively.

The authors concluded that since particulates and pollutants associated with particulates were observed to be effectively removed by vegetated areas, provisions should be made to establish or maintain vegetated buffer zones between highways and receiving water bodies. Also, if dilution alone is used to reduce the impact of highway runoff on the receiving waters, a solution of 80 percent (four parts dilution/one part runoff) is needed to avoid an oxygen debt. At sites were ADT exceeds 10,000 vehicles per day, a dilution of 100/1 was recommended to protect biota from heavy metals, a level recommended by the U.S. Environmental Protection Agency (22). Inputs of highway runoff to a receiving water body should not exceed one percent of the system's total volume.

Norwegian Bioassay Studies (125)

A sample from a snowmelt runoff event at highway E6 near Jessheim, Norway (50 km north of Oslo) in November, 1981 was used to evaluate the effects of organic micropollutants (i.e., PAH, PCB, and chlorinated organics) on a variety of aquatic organisms. The concentrations of these pollutants are shown in Table 25.

Although the concentrations of other inorganic pollutants were not given for this particular sample, Table 26, taken from Lygren, et al. (131) for a three-year study at the same site, shows pollutant ranges for these constituents. Average daily traffic at this site was 8,000 vehicles per day. The highway-contained road shoulder barriers of 4 and 28 in (10 and 70 cm) height.

Assays were performed with heterotrophic organisms (i.e., BOD test; bacteria, fungi, and protozoa from municipal WWTP), two algal species, 1-year-old salmon, and salmon eggs hatched on runoff particulates.

The results of the assay with heterotrophic organisms are shown in Figure 8. Note that biological activity increased with increasing proportions of runoff water, demonstrating a clear stimulatory effect from the runoff sample.

The effects of 50 and 100 percent concentrations of snowmelt on two algal species (Selenastrum capricornutum and Synedra acus) are shown in Figure 9. It is apparent that the runoff water exerted no significant acutely toxic effect in terms of numbers of algae. Growth rate calculations over the first three days showed slight stimulatory effects from the runoff.

Although no supporting data were presented in this paper, the authors concluded that acute toxicity effects were also not observed for either the fish eggs experiment or the 1-year-old salmon assays. Note that these were not standardized fish bioassay tests. Two small salmon (1-year-old, 3-4 in

Table 25. Concentration of Organic Micropollutants in Snowmelt Sample from Highway E6 at Jessheim, Norway Used for Bioassay Tests, all units in μg/l (125).

Organic Micropollutant	Filtered Water	Particulate Matter
Haloforms	0.5	(b)
Other compounds (a)	50	similar to filtered sample
HCB, PCB (and other persistent halogenated compounds)	0.05	0.27
Extractable unpolar org. Cl, tot/persistent	1.0/0.5	100/63
Polar compounds	low	low
Extractable polar org. Cl, tot/persistent	9.0(b)	3.0/3.0

[a] Alkanes, phenols, PAH, toluene, benzene, phthalates.

[b] not analyzed

Table 26. Range of Pollutant Concentrations in Snowmelt and Runoff Waters from Highway E6 at Jessheim, Norway (131).

Pollutant	Units	Range in Concentration	
		Snowmelt	Runoff
Total Solids mg/l		370-5,430	228-3,334
Total Suspended Solids	mg/l	230-1,669	174-2,400
pH	units	6.8-8.2	6.7-9.1
COD	mg/l	50-360	55-310
TOC	mg/l	8-58	12-320
Ni	μg/l	48-106	6-436
Cr	μg/l	30-150	13-190
Zn	μg/l	200-740	91-370
Pb	μg/l	81-690	91-360
Cd	μg/l	4-26	2-28
Fe	mg/l	5.5-78.6	4.5-30.1
Hg	μg/l	0.19-13.2	0.6-5.1
Ca	mg/l	3.1-85.1	5.3-18.7
Cu	μg/l	13-430	10-180
Cl	mg/l	2-3,100	3-1,100
SO_4	mg/l	11.1-224.1	4.9-55.2
PAH[a]	mg/l	1,772-11,604	1,403-3,907

[a] PAH - polynuclear aromatic hydrocarbons.

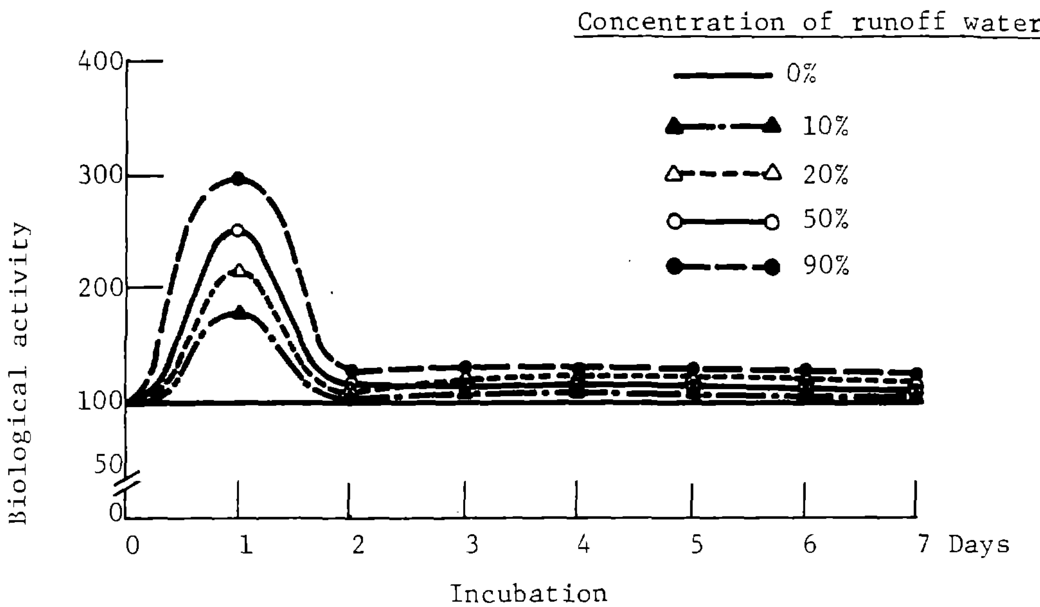

Figure 8. Effect of various concentrations of Highway E6 snow melt runoff on heterotrophic organisms using a standard apparatus (125).

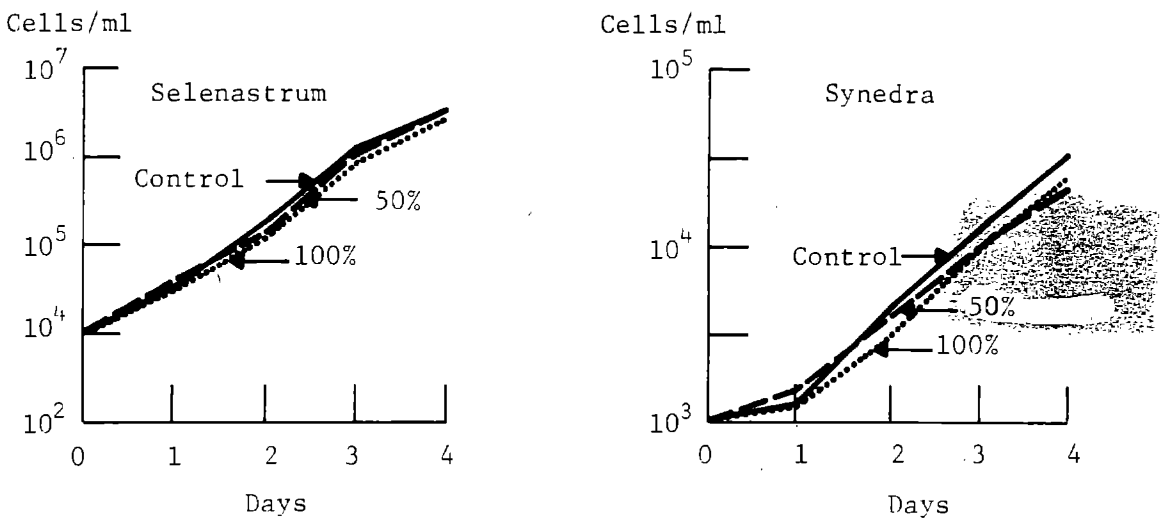

Figure 9. Effects of undiluted and 50% diluted snow melt runoff from Highway E6 at Jessheim, Norway on two algal species compared to controls (125).

long) were placed in a 2-liter beaker filled with undiluted, unfiltered runoff water kept at 6-9 °C (with aeration). The condition and behavior of these fish were compared daily with a control pair of salmon in city of Oslo tap water.

The fish eggs experiment was also somewhat unique. About 50 salmon eggs were placed directly on top of particulates, derived from the snowmelt runoff, in perforated Plexiglas containers. City of Oslo tap water was slowly circulated through the containers over the 53-day experiment. The eggs hatched in 5 weeks. No behavioral or toxic effects were observed as a result of contact with the particulates.

FIELD STUDIES

Streams and Rivers

At the Wisconsin Highway 15/Sugar Creek site (See Volume II), monthly quantitative benthic samplings were performed using both artificial substrate (Hester-Dendy) and Surber samplers. This analysis did not reveal significant differences between samples from control and runoff influenced stations that could be attributed to highway runoff. Similarly, drift and periphyton sampling did not indicate highway runoff impacts at this site. Quarterly benthic sampling at the I-85/Sevenmile Creek site (Efland, NC) likewise did not indicate a runoff impact. Both artificial substrate (Hester-Dendy) and Surber samplers were also used at this site. The Hester-Dendy samples were analyzed qualitatively; obtaining only numbers of organisms as compared to a quantitative analysis which would also include biomass determinations. Surber samples were analyzed quantitatively. Again, see Volume II for details of these field monitoring results.

Reductions in benthic macroinvertebrate numbers and diversity at stations downstream from sources of urban and highway runoff were recently documented for a stream in Northwest London (132). Sampling station locations are shown in Figure 10. Note that Stations 1 and 2 are immediately upstream and downstream, respectively, of the Highway M1 outfall. The mean numbers of invertebrates at each station are shown in Figure 11 and 12. Although the numbers of certain species (i.e., Gammarus pulex, Hydrobia jenkinsi, and Erpobdella octoculata) appeared to be reduced at Station 2, most other species increased in numbers at Station 2, only to be reduced further downstream (reflecting the increasing proportion of other urban inputs).

Species diversity at each station was also computed using three different indices (132). The spatial variation of these biotic indices is shown in Figure 13. Except for Chandler's score, species richness appears to be equivalent at Stations 1 and 2, but reduced by Station 3 (about 2.5 mi [4 km] downstream from the M1 highway). The difference in Chandler's score between Stations 1 and 2 was attributed by the author to the presence of a high scoring organism, Anabolia at Station 1. Chandler's method is a measure of diversity which uses the product of an organisms tolerance rating and its abundance to determine the score. Anabolic (an herbivorous caddisfly), being both intolerant and abundant, would therefore disproportionately elevate the score.

Metals--

A comprehensive field study of metals pollution of a soft water stream (Back Creek) in Virginia influenced by runoff from several highways and roads of gradually increasing traffic volume was done by Van Hassel and co-workers (133). This study included documentation of both water quality and sediment effects, reviewed earlier in this report, and uptake of metals by indigenous benthic insects and fish. Whole body dry weight mean metals concentration are shown in Table 27. The sampling stations for Back Creek are shown in Figure 14. According to the authors, lead concentrations were significantly correlated with traffic density for all three insect families and fish species. Zinc concentrations were also correlated with traffic density for all three fish species. Two fish species (bluehead chub and blacknose dace)

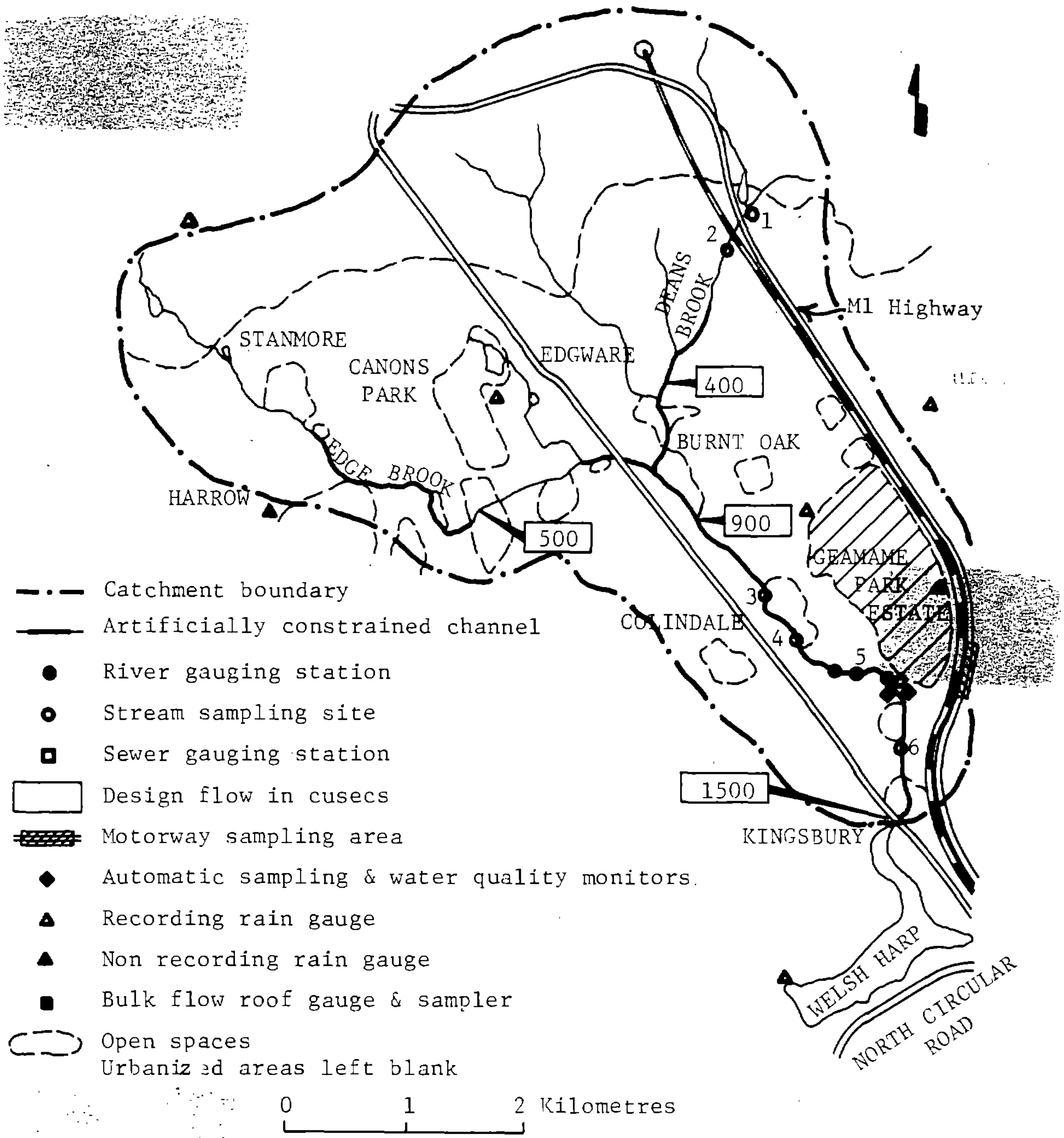

Figure 10. Map of the Silk Stream Catchment area showing the distribution of sampling stations (132).

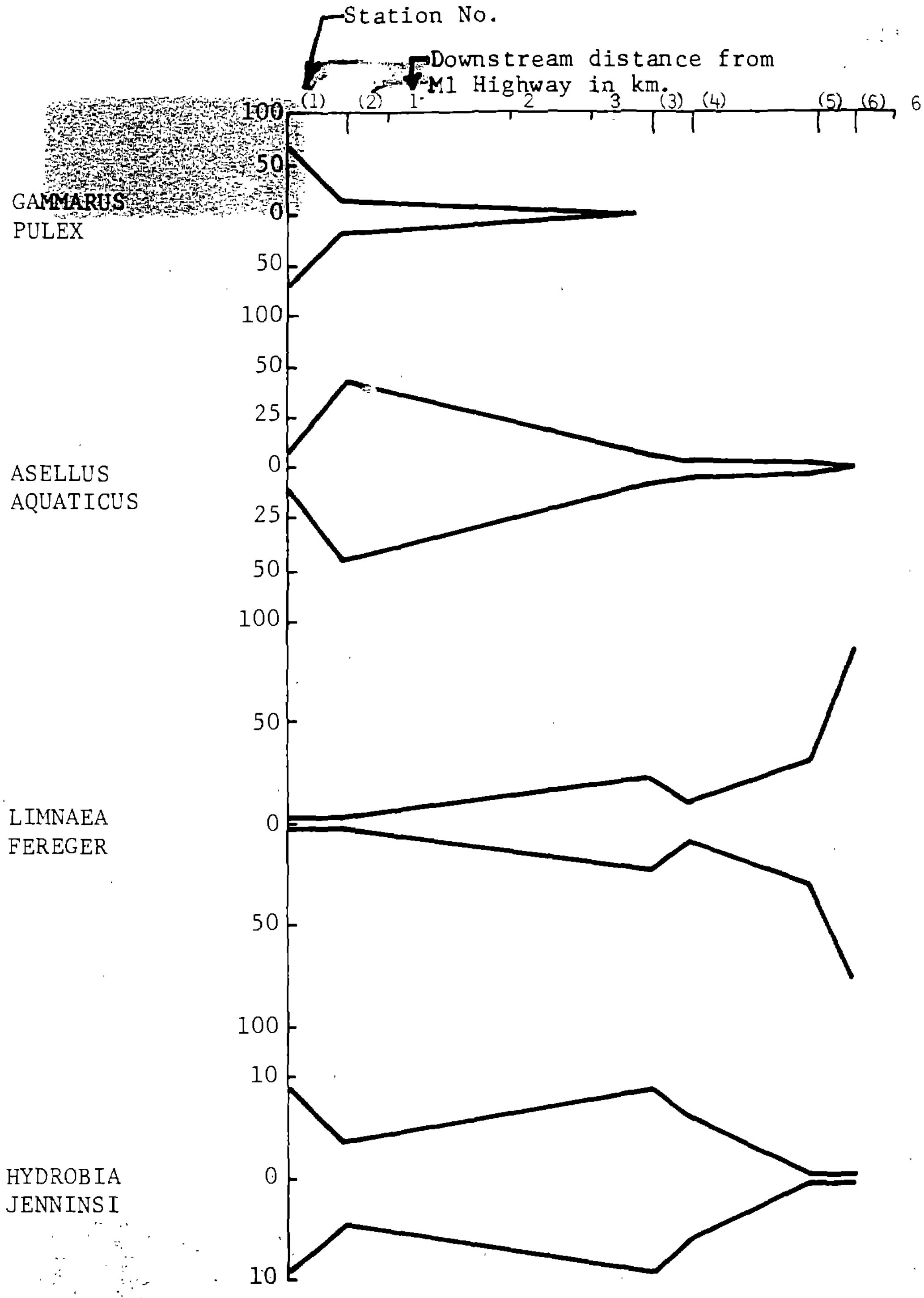

Figure 11. Spatial distribution of macro-invertebrates (132). (Mean numbers)

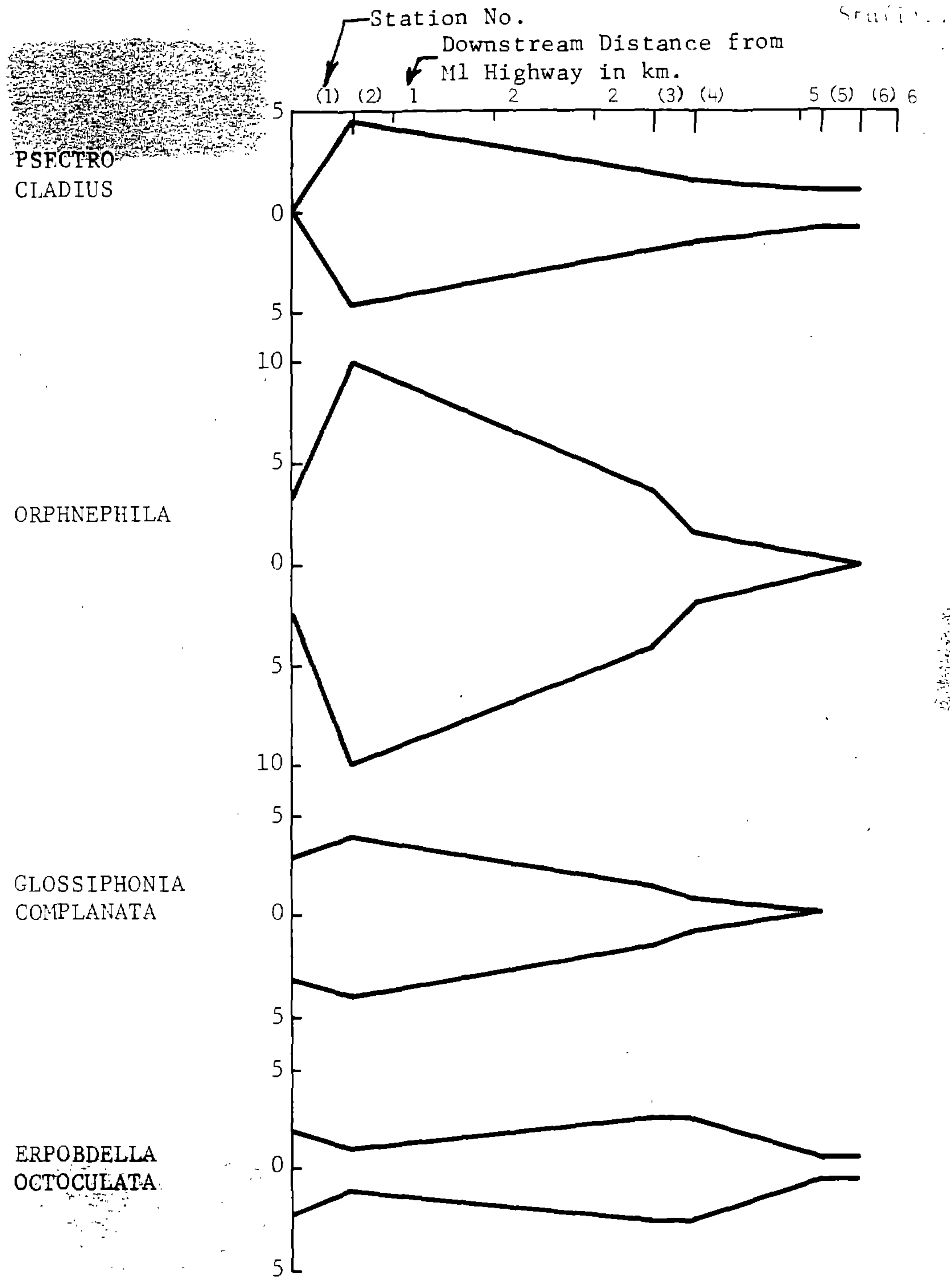

Figure 12. Spatial distribution of macro-invertebrates (132).
(Mean Numbers)

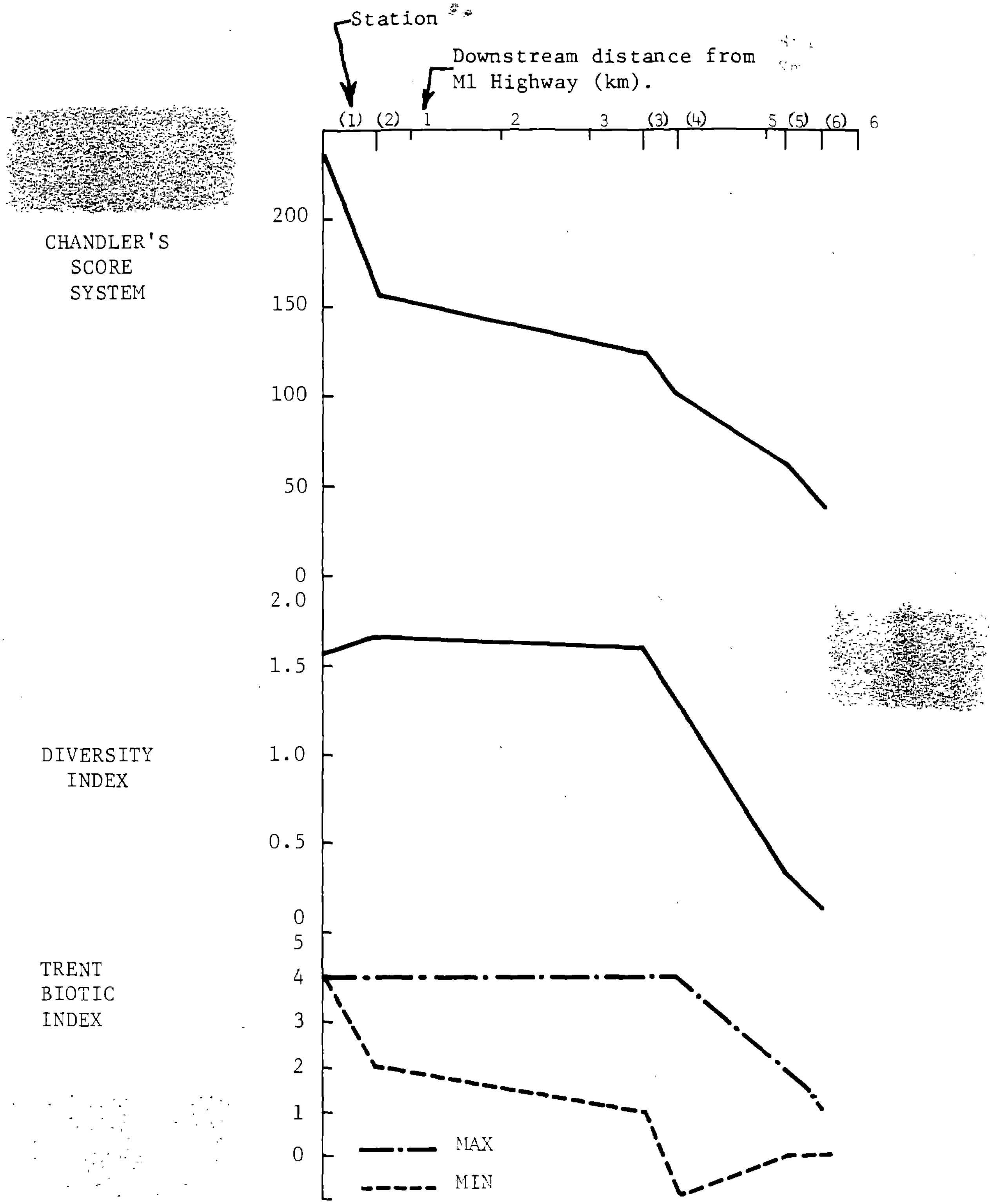

Figure 13. Spatial variation of biotic indices (132).

Table 27. Mean Whole Body Dry Weight Metals Concentrations (μg/g) In Benthic Insects and Fish in Back Creek (Virginia) as Related to Highway Traffic Density (133).

	Benthic Insects			Fish		
	Station (traffic density, vehicles/day)			Station (traffic density, vehicles/day)		
Metal	A(15,000)	B(6,500)	C(<550)	A(15,000)	B(6,500)	C(<550)
	Family Tipulidae			Bluehead Club (Nocomis leptocephalus)		
N	16	29	25	44	45	26
Pb	17.3a	12.0b	7.3b	8.4a	6.6b	5.4c
Zn	106a	92a	83b	98a	94a	76b
Ni	6.7a	3.7a,b	1.8b	2.5a	2.5a	2.0b
Cd	0.97a	0.90a	0.78a	0.42a	0.41a	0.32b
	Family Perlidae			Blacknose Dace (Rhinichthys stratulus)		
N	15	21	50	19	24	44
Pb	27.6a	19.7b	17.8b	15.5a	11.3b	7.2c
Zn	024a	227a	235a	331a	248b	151c
Ni	11.0a	6.4b	2.9c	3.0a	2.9a	2.0b
Cd	0.88a	0.83a	0.69a	0.76a	0.33b	0.66c
	Family Pteronafcidae			Fantail Darter (Etheostoma flabellare)		
N	34	14	79	21	41	24
Pb	20.4a	16.4b	7.1c	19.5a	13.5b	9.6c
Zn	240a	232a	254a	147a	115b	112b
Ni	2.3a	3.4a	2.4a	5.9a	3.4b	2.9b
Cd	1.01a	0.99a	0.63b	0.60a	0.72a	0.65a

Notes: N is sample size.

Within rows, values followed by the same letter are not significantly different (jonckherre ordered alternatives test, P<0.05.)

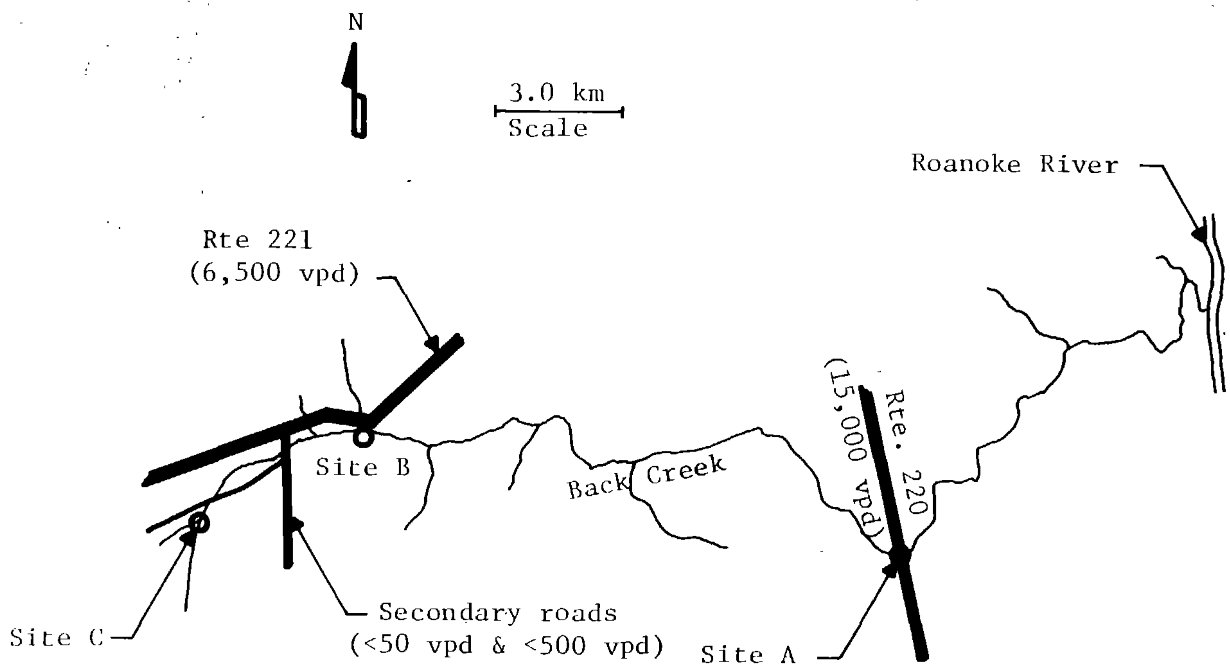

Figure 14. Back Creek (Virginia) sampling stations showing average daily traffic (133).

also showed increased nickel and cadmium concentrations with increasing traffic density. Even through Back Creek receives no industrial or domestic point source discharges, the concentrations of metals in insects and fish are comparable to literature values for animals in polluted waters. Unfortunately, other watershed land uses were not described by the authors.

Trophic level differences in metal concentrations found in this study are also of considerable interest (133). The degree of metals contamination was related to sediment contact. Benthic invertebrates had higher concentrations than fish and the most benthic-oriented fish, the fantail darters, had the highest lead levels. Biomagnification of metals was not apparent since both pteronarcid (detritivores) and perlid (predators) stone flies had similar metals concentrations. Also, the benthophagic fish had lower levels than benthic invertebrates. The tipulids (largely herbivorous detritivores) had lower concentrations than both stonefly families. Tipulids have a lower percentage of hardened exoskeleton. The authors suggest that exoskeletons can serve as a metals sink in invertebrates. One final pertinent conclusion made in this study was that water quality criteria derived from bioassay tests may not provide suitable protection from toxicants which are taken up from sediments by benthic species. Although water column and sediment concentrations were low, concentrations in insects and fish were considerably higher. It should be noted that water-column sampling was not performed during storm events when metals concentrations (or loads) would be highest. As a result, a complete characterization of water quality conditions was not made by the researchers.

Deicing chemicals--

Salt is used extensively in some regions to deice roads and may represent a major constituent of highway runoff during the winter and spring seasons. Both laboratory and in situ studies have been implemented to determine the effect of deicing salt on the drift of stream benthos. Drift is a phenomenon in which organisms dislodge from the substrate and enter the water where they are swept with the current. Several laboratory experiments were conducted on the benthic side swimmer (Amphipoda: Gammarus pseudolimnaeus) and caddisflies (Trichoptera: Hydropsyche betteni and Chaematopsyche analis) (134). Drift was monitored with removable nets housed in five oval fiberglass stream troughs. Laboratory results showed that pulses of 800 mg Cl/l had no effect on the drift of the side swimmer or the caddisflies. In the field, three drift nets collected drifting organisms from a part of the channel which was artificially diverted using corrugated steel sheets. Field experiments indicated that levels as high as 750 mg Cl/l had no significant effect on drift. However, a pulse of 2,165 mg Cl/l increased drift of all organisms. Drift, in general, only occurred at a concentration of 1000 mg Cl/l or more (Figure 15) (134).

Dickman and Gochmauer (135) added sodium chloride to a small stream in Quebec to simulate the loading of road salt. The salt concentration in the stream was maintained at around 1000 ppm (a level higher than would normally be expected due to highway salting alone). Using artificial substrates (grey slate tiles) they quantified the effect of this salt addition on stream algal and bacterial populations during a four-week colonization period. Microsuccession was documented at day 1, 7, 14, 21, and 28. Standing crop of autotrophs was considerably lower at the salt-influenced station compared to the control (ambient stream salt concentrations were 2-3 ppm). Algal diversity was lower at the salt-influenced station after only one week of exposure. Conversely, bacterial density was increased due to salt exposure (see Figure 16). This was interpreted by the authors as being the result of reduced grazing pressure rather than direct stimulation. Diatom parasitism was lower at the salt-influenced station. It was speculated that this was due to greater fungal inhibition at this station.

Sharp (136) reports that recent evidence suggests that sodium and potassium can be limiting nutrients for the growth of blue-green algae.

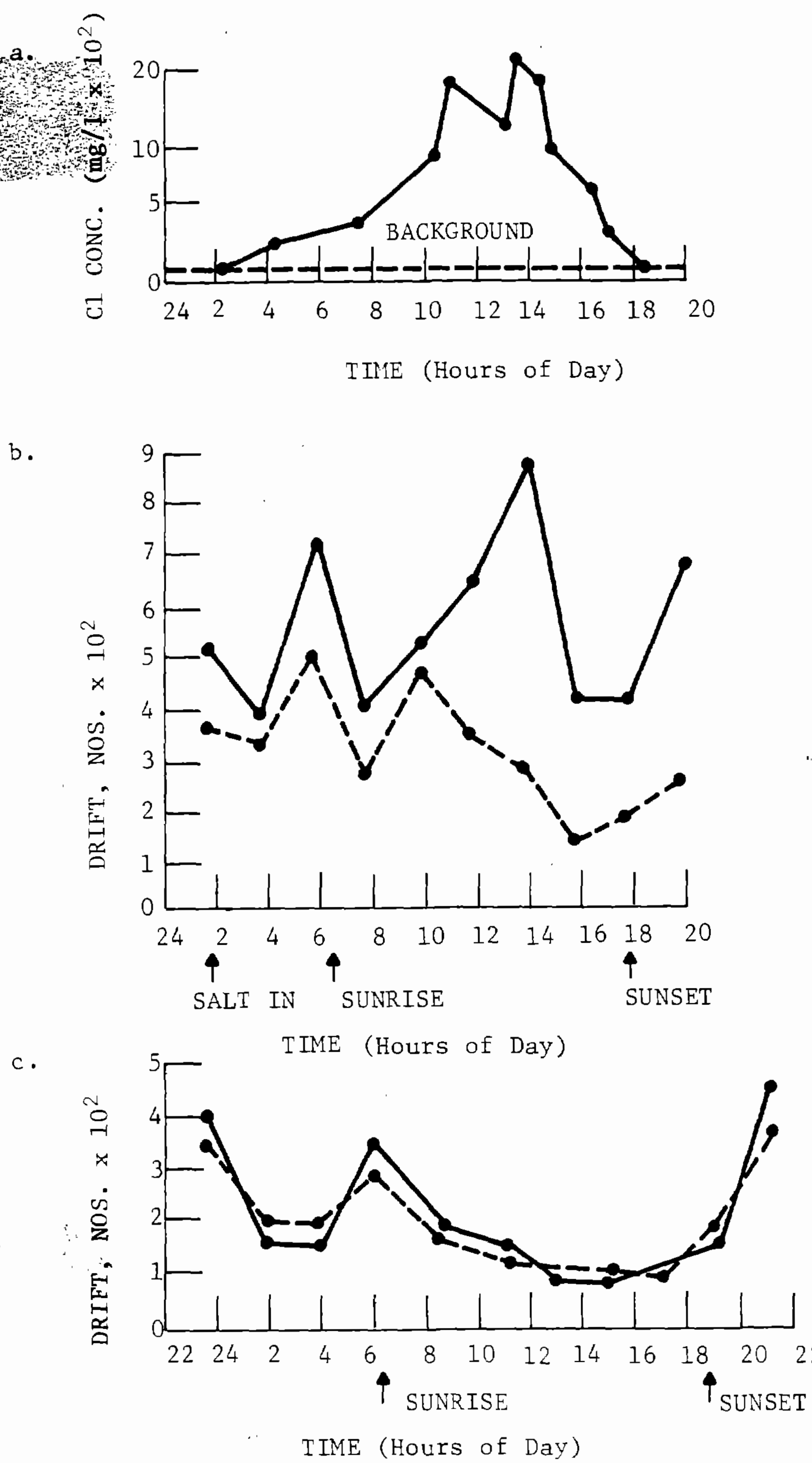

Figure 15. The effect of chloride concentrations on benthic organism drift. a. Chloride concentrations in treated (chloride added) channel (solid line) and untreated channel (dashed line) b. organism drift in treated (solid line) and untreated (dashed line) channels, and c. organism drift in two channels without chloride addition (134).

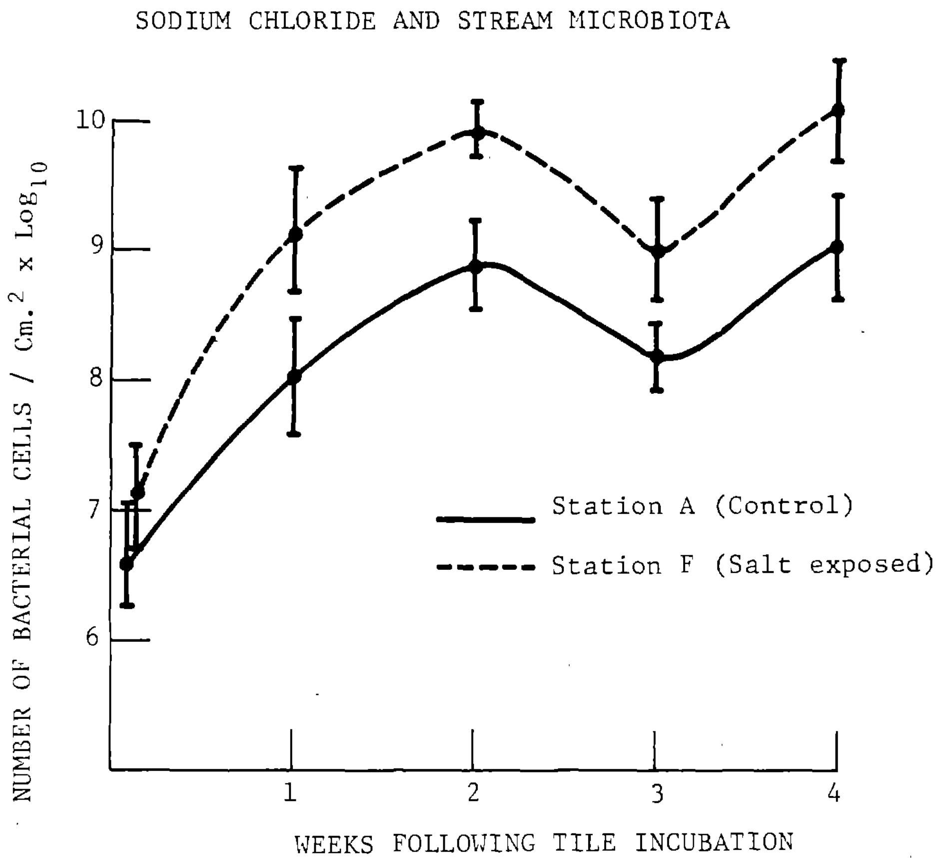

Figure 16. Bacterial density on stream-exposed tiles as estimated from counts of oil-cleared membrane filters. Vertical bars represent 95% confidence intervals (n-3) (135).

Anabaena cylindrica, often present in nuisance algal blooms, require 5 ppm of sodium for optimum growth. Thus, eutrophication problems in some water courses may be related to increasing salt loads.

Nutrients--

Algae growing on submerged surfaces (periphyton) account for most of the primary production occurring in fast flowing streams. Fast or swiftly flowing streams can be defined as those with flow velocities in excess of 50 cm per second. Periphyton are able to colonize and cover an entire stream bottom in 10 days or less under favorable conditions (137,138).

Commercial fertilizer with a composition of 12 percent total nitrogen, 12 percent available phosphoric acid, and 12 percent total potash was applied at a rate of 20 lb/1000 ft^2 along an Ohio highway right-of-way to promote growth of freshly seeded areas (139). A higher density and different taxa of benthic invertebrates in a stream area receiving the nutrient-enriched highway runoff compared to a control site was observed. The difference was attributed to massive growths of the filamentous alga, *Chladophora glomerata*, due to the large inputs of plant nutrients, especially phosphorus. The growth of the alga produced a qualitative and quantitative difference in food supply at the highway-influenced site which led to colonization by different herbivorous and carnivorous benthic invertebrates. The larger quantity of algae at the highway-influenced site provided a larger food source and, therefore, a larger density of benthic organisms. To help alleviate the problem the authors suggested restricting fertilization to springtime only.

An urban canal system in a residential area of Miami, Florida was investigated chemically and ecologically to determine the water quality impact of urban stormwater discharges (140). High levels of macronutrients (nitrogen and phosphorus) as well as other contaminants were observed in the canal following periods of high rainfall. An indicator algal index showed that the canal receiving the stormwater discharges had a higher number of pollution-tolerant algal genera than canals without stormwater.

Lakes

Qualitative sampling of benthic invertebrates at the I-94/Lower Nemahbin Lake site (WI) indicated little effect of highway runoff. Details of monitoring at this site are provided in Volume II. Although several taxa considered facultative to intolerant of pollution (*Fredricella*, *Stenacron*, and *Stenonema*) were found exclusively at control stations, other taxa considered intolerant were found either exclusively at highway influenced stations or at both control and influenced stations (*Enallagma*, *Amnicola*, *Hydracarina*, and *Daphnia*). Limited metals analyses for three indigenous invertebrate species collected from the lake indicated increased burdens for some metals in species from runoff influenced stations compared to controls (i.e., higher Pb, Fe and Cd for an amphipod; higher Pb and Cr for a mayfly, and Fe and Zn for a damselfly). Field microcosm experiments using five indigenous species and one transplanted species (*Daphnia magna*) were also conducted at this site. Organisms were enclosed in flow-through vials and submerged in shallow control and runoff influenced areas of the lake for approximately three-week exposure intervals. Highway runoff was shown to have minimal effect on these species.

Metals--

Researchers at the University of Central Florida recently completed a comprehensive study for the Florida Department of Transportation and the Federal Highway Administration (53). This study investigated the effects of highway bridge deck runoff on the water quality, sediment accumulations, and biological uptake of metals in an adjacent lake (Lake Ivanhoe - 125 acre [50.6 ha]). The effects on water quality and sediments were reviewed earlier in this volume in appropriate sections. One objective of this research effort was to determine relative effects of bridge runoff conveyed directly to the lake through scupper drains compared to runoff from the same bridge discharged indirectly after overland flow.

Biological determinations included metals concentrations in submerged aquatic plants (Hydrilla), emergent macrophytes (Typha), green algae (Spyrogyra), and benthic organisms including crustaceans, mollusks (pelecypods and gastropods), and annelids (tubifex). The concentrations in plants and algae are summarized in Table 28 for those species and metals where significant differences existed between stations with and without scupper drains. Note that direct scupper discharges significantly increase metals concentrations in some biota compared to indirect discharges after overland flow.

Metals concentration factors between water and plants and algae in Lake Ivanhoe were also calculated, as shown in Table 29. Since Spirogyra concentrated more lead, copper, and iron than the other species, the authors suggested the use of Spirogyra as an indicator species for highway runoff contamination.

Although an insufficient number of benthic organism samples were collected to make statistical comparisons between stations with and without scuppers for Lake Ivanhoe, average concentration factors were calculated. For the annelid worms, Tubifex concentrated more lead, zinc, and chromium than all other benthic organisms, while Hirudinea had the highest concentration factor for cadmium. Nickel, copper, and iron were concentrated the most by Arthropodea (crustacea). The authors concluded that Tubifex could be used as an indicator benthic organism for highway runoff contamination.

Table 28. Metals Concentrations in *Hydrilla* and *Spirogyra* from Lake Ivanhoe (53).

Species	Element	No. of Observ.	Mean Values µg/g (dry wt.)		Percent Probability (T-test)
			With Scuppers	Without Scuppers	
Spirogyra	Zn	4/5	188.5	111.7	94.2
	Pb	4/5	368.3	192.9	93.0
	Cr	4/5	46.0	29.0	84.23
	Fe	4/5	1920.0	868.2	95.35
	Cd	4/5	1.5	0.8	99.1
Hydrilla	Zn	5/3	333.0	229.3	87.24
	Pb	5/3	248.0	126.8	81.8
	Ni	5/3	32.8	12.1	93.59
	Cd	5/3	1.8	1.2	83.93

Table 29. Average Concentration Factors of Heavy Metals by Plants and Algae in Lake Ivanhoe (53).

	Concentration Factor (ml/g)						
Plants	Cd	Zn	Ni	Cu	Fe	Pb	Cr
Hydrilla	363	5145	4104	5024	12103	3522	9700
Spirogyra	336	3047	3516	13483	25434	52375	7398
Typha	54	555	255	539	2322	444	610

Wetlands

General--

When discussing biological impacts to wetlands, it is important to consider the hydrological and biological factors which differentiate this surface water ecosystem from streams or lakes. Wetlands can be hydrologically open systems (surface water inputs and/or outputs) or closed systems (no surface water outflow). Wetlands have long been considered by many to be "wastelands" which were of little or no use to man. Recent investigations of wetlands have shown that these ecosystems perform many important functions. Wetlands have been shown to be important hydrological recharge areas and, in the case of a hydrologically open system, can be important in alleviating flood water stress (141). Marshes have also been shown to often act as sinks which trap nutrients from surrounding terrestrial systems (142,143). Wetlands are able to retain and regulate nutrients because these constituents are incorporated into the thick organic substrate which is usually present and into the tissues of the large plant biomass which is characteristic of wetlands. Wetlands are sources of high organic matter production, especially fresh water marshes which have been shown to be among the most productive areas in the world (144,145).

Metals--

Wickliff, *et al.* (146) suggested that heavy metal additions to a Massachusetts wetland caused reduced rates of nitrogen fixation. Mrozek (147) found that Zinc and Lead inhibited the germination of *Spartina alterniflora* seeds. Similarly, both mercury and cadmium reduced the germination rate, total performance, and long-term viability after an initial stimulation of germination, in *Spartina* seeds (148). Increases in detrimental effects were also directly correlated with increases in salinity level and metal concentrations. Medine, *et al.* (149) found that the presence of cadmium, zinc, chromium, lead, and mercury at permissible drinking water levels (500 µg/l Zn, 100 µg/l Cr, 20 µg/l Hg, 20 µg/l Pb, and 10 µg/l Cd) reduced total O_2

production by 27 to 43 percent and total biomass by 32 to 38 percent in artificial freshwater systems.

Because heavy metals do accumulate in wetland plants, the potential for bioaccumulation or biomagnification exists. Guthrie, et al. (150) studied biomagnification of heavy metals by organisms in the marine environment. The results of this study are given in Table 30. The data indicate that the organisms used as food by man (crab, oyster, and clam) were the least likely of the population studied to biomagnify the ten heavy metals studied. Crabs biomagnified less than the attached, filter-feeding oysters and clams. The major biomagnifiers of the metals studied were polychaetes and barnacles.

Some species have shown the ability to acquire a resistance to heavy metals. Phytoplankton from metal-contaminated lakes were found to be much more resistant to the toxic effects of copper and nickel than populations from uncontaminated areas (151,152). In estuaries contaminated by copper, zinc, lead, and cadmium, polychaetes were resistant to copper and occasionally zinc, indicating that lead and cadmium are the contaminants of primary concern (153).

McNaughton, et al. (154) studied heavy metal tolerances in cattails (Typha latifolia). Cattails and soil samples were obtained from a location high in heavy metals and from a control location (Table 31). Cattails obtained from both sites grown on soils high in heavy metals showed similar growth rates and biomass yields, and there was no evidence of chlorosis or any other visible symptoms of toxicity. These data suggest that the species (Typha latifolia) as a whole and not ecotypes (species that have evolved a tolerance) may be resistant to heavy metal toxicity. Although cattails obtained from both sites showed no toxicity symptoms when grown on soils high in heavy metals, the biomass yields for both cattail genotypes were significantly lowered when grown on contaminated soils compared to control soils (Table 32). McNaughton, et al. (154) also observed that Typha is able to occupy industrially degraded habitats of a variety of types: acid-mine

Table 30. Concentrations of metals in environment and organisms forming microcosms in Chocolate and Jones Bays (mg/kg, wet weight) (150).

Sample	Metal									
	Fe	Ba	Zn	Mn	Cd	Cu	Se	Cr	Hg	As
Bay Water	22.14	7.67	4.11	2.37	1.17	1.31	0.11	0.11	0.009	0.005
Sediment	5,196.00	131.00	26.80	113.83	1.88	148.03	1.44	19.88	0.49	2.40
Barnacle	752.33	40.45	648.00	56.63	1.19	15.28	0.77	3.73	0.28	2.00
Crab	12.80	1.50	3.45	0.34	0.14	2.99	0.08	0.05	0.005	0.07
Oyster	31.44	3.51	103.39	1.39	0.48	40.66	0.14	0.43	0.07	0.57
Clam	71.16	1.45	12.83	7.60	1.19	22.75	0.54	0.99	0.11	2.39
Polychaete	21.03	4.70	41.04	7.47	1.75	42.27	0.49	2.02	0.14	1.12

Table 31 Analyses of soils from control (Syracuse, N.Y.) and heavy metal (Palmerton, Penn.) sites (154).

Soil property	Location	
	Syracuse	Palmerton
% organic matter	7.9	4.9
pH	5.8	6.9
Elements (ppm)		
Zn	13	5,000
Pb	27	435
Cd	2	73
N	5.8	0.16
P	10	34
K	210	150
Ca	31,000	·9,500

Table 32. Dry biomass yield (g) of T. latifolia genotypes from heavy metal (Palmerton) and control (Syracuse) locations when grown for 34 days on soils from the two locations: (a) shoot biomass, (b) root biomass (154).

a. Shoot biomass (g)

Cattail Genotype	Soil	
	Palmerton	Syracuse
Palmerton	7.29	15.42
Syracuse	11.64	24.17

b. Root biomass (g)

Cattail Genotype	Soil	
	Palmerton	Syracuse
Palmerton	0.985	2.134
Syracuse	1.505	5.302

drainage in western Pennsylvania, and otherwise devegetated areas near smelters in Sudbury, Ontario.

Deicing Salts--

Bayly and O'Neil (155) studied the fluctuation of sodium in the leaves, rhizomes, and flower stalks of cattail (Typha glauca), and the associated substrate in a marsh located in Ottawa, Canada. The marsh received runoff from a nearby road system that was salted in the winter. Sodium increases in the substrate were attributed to deicing salts. The sodium concentrations in leaves and rhizomes peaked during week 12 (July 21) at the same time sodium peaked in the substrate (Figure 17). The coincident peaking of sodium in the soil substrate suggested that the sodium concentration increase in the tissues of leaves and rhizomes was a passive response to the plant environment. However, based upon growth characteristics (moisture content and leaf length), no biological impact to the cattail community due to increased sodium in the substrate was observed.

Hoese (156), in a review of effects of higher than normal salinity on salt marsh ecosystems, discussed population succession among certain major fisheries organisms such as blue crab, white shrimp, hard clams, and oysters. Rising salinities can damage oyster fisheries by invasion of parasites and predators. Blue crab and white shrimp depend on relatively low salinity for development during juvenile stages. Hard clams and other stenohaline marine organisms typically invade areas of increased salinity providing a new, but often unexploited fishery. The migration of many species inland can also be interrupted if orientation compounds are removed or masked.

A review of the effects of salinity on salt marsh macrophytes was compiled by Smalley and Thien (157). In general, salt concentrations of around seven percent sodium chloride (twice sea water strength) will prohibit

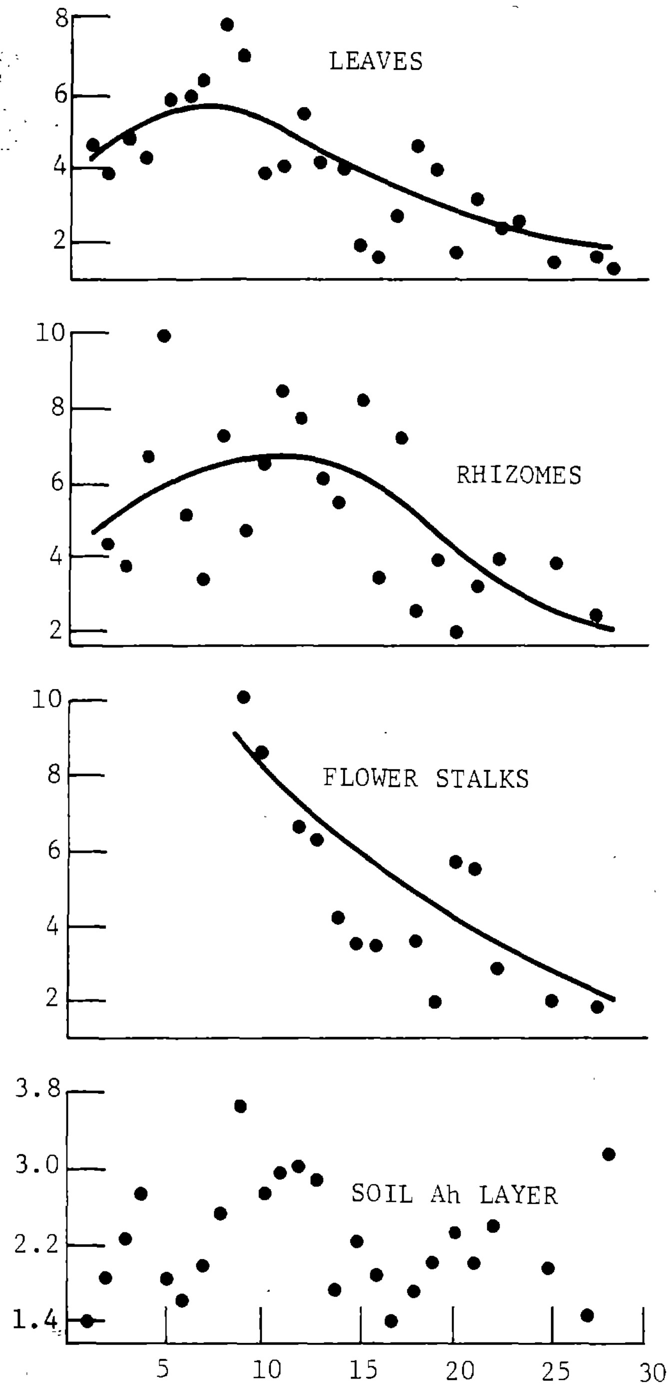

Figure 17. Seasonal variation in sodium content of leaves, rhizomes, and flower stalks of Typha glauca and in sodium content of the soil at the plant site (155).

establishment and survival of most salt marsh species. A change in normal salinity levels will also reduce growth and fertility of many species. Salt tolerances of the predominant Louisiana salt marsh species are summarized in Table 33. The germination of many salt marsh species might be dependant on a reduction of soil surface salinity.

Petroleum Hydrocarbons--

Baker (158) studied the seasonal effects of oil pollution on salt marsh vegetation. Transects (59 x 6.6 ft [18 x 2m]) in the salt marsh were sprayed with 4.8 gal (18 l) of crude oil. The data showed little long-term vegetative damage to most perennial species, but the damage to the annual, *Suaeda maritima*, was severe. A reduction in flowering of many species was noted when flower buds were oiled. Also, oiling of seeds during winter may reduce spring germination.

A 1.25-acre (0.5 ha) saltmarsh in Chesapeake Bay was dosed twice a month with 12 gal (45 l) of No. 2 fuel oil (159). A major portion of the *Spartina alterniflora* (marsh grass) was killed by this treatment; the remaining portion displayed sublethal effects including delayed development in the spring, increased density, and reduced mean weight per stem. Oil which entered the roots and rhizomes of the dead marsh grass was retained in a relatively undegraded state for at least seven months after treatment stopped.

Gebhart and Chabreck (160) studied the effects of various oil concentrations and water levels on Olney Bulrush (*Scirpus olneyi*) and marshhay cordgrass (*Spartina patens*). They found that increased damage occurred with increased oil concentrations and fluctuating water levels. A fluctuating water level (±2 in [±5 cm]) with 80 parts per thousand of heavy crude oil was the treatment most detrimental to *S. olneyi*.

DeLaune, *et al.* (161) studied the effect of crude oil on *Spartina alterniflora* in a Louisiana salt marsh. The data showed that *S. alterniflora* could tolerate a large amount of oil without a short-term decrease in above-ground biomass. Oil placed on the water surface caused a release of iron, manganese, and ammonium from the sediment to overlying water due to the absence of oxygen in the water column.

CONCLUSIONS

HYDROLOGICAL IMPACTS

1. Hydrological effects of urbanization attributed to the increase in impervious area include increased peak runoff rates and total runoff volume, decreased lake retention time, and stream sediment scour. These effects are not likely to result from highway projects since the increase in impervious area is generally minimal compared to the total watershed area.

2. Localized changes in watershed drainage patterns due to highway projects could have significant effects on the hydraulic regimes of small receiving waters with significantly large highway inputs. Such alterations have only been described qualitatively; field documentation was not found in the technical literature.

3. Density stratification has been observed in lakes as a result of deicing salt washoff from urban streets and highways. In several dimictic lakes, this unnatural density stability was sufficient to prevent or delay normal spring overturn. Such an effect was not observed to occur for a 270-acre lake in Wisconsin even though 22 tons of salt had been applied during winter months to the 0.75 mi long I-94 ROW tributary and immediately adjacent to the lake.

WATER QUALITY IMPACTS

1. Water quality effects are usually assessed according to increases in concentration of a particular pollutant over natural or background conditions. The severity of the impact is gaged by comparing pollutant concentrations during or after runoff events with criteria established by

the Federal Government (i.e., U.S. EPA) or, more importantly, with State or local water quality standards.

2. Water quality effects from highway runoff have not been frequently documented. Research on actual water quality effects of urban stormwater runoff are also limited. Furthermore, they provide only qualitative insights into potential highway runoff effects. The nature and concentrations of pollutants are similar, but the relative loadings are often much greater from urban runoff due to relatively larger drainage areas and lower receiving water dilution ratios. Nevertheless, several of these qualitative insights are useful in the assessment of potential highway effects, especially with respect to the particulate nature of both urban and highway-generated pollutants.

3. Dissolved oxygen depletion - Stormwater runoff from both urban and highway areas has been shown to contain biochemical oxygen demand (BOD) (although in relatively low concentrations; generally less than 30 mg/l). Dissolved oxygen depletion in receiving waters due exclusively to highway runoff has not been studied. Several field studies have been conducted for urban runoff, but have failed to identify significant DO depletion in receiving waters. This could be due to the particulate nature of stormwater BOD compared to more soluble forms in other sources such as municipal treatment plant effluents. Other likely reasons for the lack of DO depletion in receiving waters are:

   a. Increased storm flows increase reaeration rates which compensate for BOD loads, and

   b. Increased storm flows decrease the travel and residence times of streams and rivers as they flow to their mouths which minimizes time for bacterial oxidation.

4. Effects of nutrients - Nutrient impacts can be viewed from two perspectives. The first is possible violation of public health or aquatic

life protection standards. In this regard, only the criteria for ammonia and nitrate are pertinent. Few studies documented levels in excess of those recommended by EPA which could be associated primarily with stormwater runoff. The second perspective is that of prevention of accelerated growth of nuisance aquatic plants which often detract from the recreational use of a water body. From this perspective, nutrient loadings from urban runoff sources have been identified by several researchers as being responsible for such accelerated primary productivity. Such effects from highway runoff alone have not been documented.

5. Deicing agents - The application of deicing agents to highways has been frequently shown to increase salt ion concentrations in nearby soils and water bodies. The effects are less pronounced for streams and rivers where increased flows during storm and melt conditions have sufficient dilutive capacity to minimize increases in salt concentration and to keep chloride levels well below the EPA criteria of 250 mg/l. For lakes and wetlands, road salts have been shown to accumulate, in some cases to chloride concentrations an order of magnitude higher than EPA's water quality criteria value. Finally, very little is known of the fate or potential impact of toxic anticaking road salt additives (sodium ferrocyanide and ferric ferrocyanide).

6. Metals - In spite of the prevalence of metals in highway stormwater runoff, few field studies have shown elevated water column concentrations. The high association of metals with particulates and frequently demonstrated sediment enrichments of metals implies a rapid removal by sedimentation. In most cases, it is difficult to compare receiving water concentrations with recent EPA recommended criteria due to the lack of water hardness data.

7. Petroleum hydrocarbons - There are very little data available on receiving water concentrations of petroleum hydrocarbon-related pollutants. Again,

the high affinity of hydrocarbons for adsorption onto particulate matter would suggest sediment enrichment of these materials.

SEDIMENT IMPACTS

1. The high association of vehicle-derived metals and hydrocarbons (and derivatives) with particulate materials suggests that receiving water sediments serve as the primary sink for these pollutants. Data from several highway runoff field studies have demonstrated sediment enrichments of metals, especially in nearshore sediments of lakes.

2. Metals concentrations in sediments receiving highway runoff have been shown to be as much as an order of magnitude higher than control-station sediments.

3. Sediments receiving direct runoff inputs from bridge deck scupper drains can contain metals concentrations which are twice as high as sediments receiving highway runoff from a similar bridge deck but which has first passed overland through vegetated ditches.

4. Metals concentrations in sediments have been significantly correlated with average daily traffic on roads draining to receiving waters.

5. Salt ions delivered to receiving waters as a result of deicing agent application can be transported into sediments (i.e., interstitial water). One laboratory study demonstrated significant increases in mercury release from sediments as a result of salt addition.

6. The long-term ecological effects of sediment enrichments of metals or hydrocarbons cannot be adequately assessed at the present time. Research into the dynamics of sediment/water exchanges, biological uptake, biomagnification, and subsequent toxicity is not sufficiently developed. This is particularly true for stormwater runoff derived pollutants.

Furthermore, regulatory guidance relative to safe levels of pollutants in sediments has not been provided by either Federal or State agencies.

BIOLOGICAL IMPACTS

1. Although considerable information exists on the toxicity of individual constituents in highway runoff (i.e., metals, hydrocarbons, salts), this information is not directly pertinent. The association of these pollutants with particulates in runoff would be generally expected to reduce their acute toxicity.

2. Bioassay experiments which have used actual highway or urban stormwater runoff have provided more meaningful insights.

   a. Algal assays performed by CALTRANS found severe inhibition of mixed algal populations when exposed to low percentages of runoff (1, 5, and 10 percent) from a very high traffic urban freeway (185,000 ADT) after two weeks of antecedent dry weather. However, runoff from lower ADT rural (23,000 ADT) and suburban (16,000 ADT) highways, or from high traffic urban freeways with short antecedent dry days, was generally stimulatory to the algal populations. In addition, filtration of the runoff did not substantially alter assay results, implicating soluble, or subcolloidal, forms of pollutants with both toxic and stimulatory responses.

   b. Algal assays (Selenastrum) conducted by the University of Washington also found toxicity to increase with increasing concentrations of runoff water from a high traffic highway (50,000 ADT). However, toxic effects were not observed with runoff from two other highways (42,000 and 7,700 ADT). The difference in toxic response between the 50,000 and 42,000 ADT sites was attributed to higher soluble copper and zinc concentrations in the 50,000 ADT runoff. Bioassays with rainbow trout and the 50,000 ADT runoff showed no toxic effects with filtered runoff, but unfiltered runoff was toxic with both a 50% dilution and an undiluted sample. Similarly, runoff filtered through a 200-ft (60m) grassy ditch was not toxic to the trout. So unlike the previous algal experiments, pollutants associated with particulates, or the particulates themselves, can be deleterious to rainbow trout.

c. Bioassays conducted by the University of Wisconsin-Milwaukee showed minimal acute toxicity of undiluted runoff from both an urban and rural highway for an amphipod, isopod, water flea, mayfly, and fathead minnows. Two-week exposures with Selenastrum did demonstrate algal inhibition with undiluted runoff (See Volume II of this report).

d. Runoff from a highway in Norway (8,000 ADT) proved generally stimulatory to both heterotrophic organisms (BOD test) and two algal species. The runoff was also not toxic to 1-yr-old salmon and hatching salmon eggs.

3. Field studies with highway and urban stormwater runoff have shown that metals can be taken up by a variety of benthic organisms and fish. However, biomagnification of metals does not seem to be prevalent. Furthermore, the significance of metals uptake is difficult to assess since little is known about long-term effects of various body burdens.

4. Although decreases in abundance and diversity of benthic organisms has been attributed to urban runoff, or urbanization, these effects have not been shown for highway runoff. Similarly, urban runoff discharges have been implicated in accelerated eutrophication of lakes; highway runoff loadings, by themselves, have not been shown to have this effect in field studies. Bioassay experiments have shown that highway runoff can be stimulatory to algae, however.

5. Increased salt concentrations in streams (greater than 1000 mg/l chloride) have been shown to increase the drift of benthic microinvertebrates, decrease the biomass and diversity of algal species, but increase the density of bacteria in natural streams.

GLOSSARY

Acute toxic effect - Effects due to high level (i.e., high concentration) short-term exposure to a toxicant.

Algae - Any of a number of simple plants of the Monera and Protista possessing delorophyll.

Amphipoda - An arthropod group containing the scud or side swimmers.

Aquifer - A stratum of the earth that contains and conducts water; can be unconfined or confined, top and bottom, by impermeable layers.

Baseflow - That flow in a stream, river or estuary produced by groundwater discharge as opposed to surface runoff; also that flow present immediately before runoff from a particular storm event.

Benthal deposits - Bottom sediments, consisting of both inorganic and organic (detritus) components.

Benthic - Pertaining to the bottom habitat or organisms associated with the bottom habitat of an aquatic system.

Bioaccumulation - The long-term retention and build-up of pollutants in the tissue, organs or fluids of living organisms.

Bioassay - A technique using viable organisms to measure the effect of a substance (i.e., pollutant), factor or condition.

Caddisfly - Common name of an insect in the group Trichoptera.

Chlorosis - A yellowing of normally green vegetative structure; often indicative of a toxic response.

<u>Chronic toxic effect</u> - Long-term, low level exposure resulting in a biotic response of bioaccumulation to a toxic level.

<u>Coliform organism</u> - A group of mostly non-pathogenic bacteria originating in the gut of warm-blooded animals, used as <u>indicators</u> of fecal contamination.

<u>Density currents</u> - Water movements, usually in lakes and estuaries, caused by the tendency of denser layers to displace water; can be caused by both thermal and chemical density differences.

<u>Dimictic</u> - Lakes and reservoirs which undergo thermal stratification twice per year (once in summer and inverse stratification in winter) with subsequent overturns in fall and spring; usually occur in cool temperature regions.

<u>Enrichment</u> - Addition of nutrients to an aquatic system.

<u>Eutrophication</u> - The slow aging process of a lake evolving into a marsh and eventually disappearing; often refers to one phase, the process of nutrient enrichment in an aquatic system.

<u>Hypolimnion</u> - The lower layer in a thermally stratified lake or reservoir which is relatively stagnant and isolated from the surface layer.

<u>Infiltration</u> - The seepage of precipitation into the soil.

<u>Lentic</u> - Pertains to standing water, e.g., lakes and wetlands.

<u>Lotic</u> - Pertaining to flowing water, e.g., streams, rivers, and estuaries.

<u>Macrophytes</u> - Rooted aquatic plants.

Mayfly - Common name of an insect in the group Ephemeroptera.

Meromictic - Permanently stratified; usually caused by vertical chemically induced density gradient.

Midges - Common name of an insect in the group Chironomidae.

Monomictic - Lakes and reservoirs which undergo thermal stratification only once per year; cold monomictic lakes never exceed 4 °C and turnover in summer (found in arctic or mountain regions); warm monomictic lakes are never colder than 4 °C and stratify in summer, most lakes in central, eastern, and coastal regions of North America.

Morphometry - Pertaining to the structure, shape, and geometry of a lake, wetland, or stream or river bottom.

Nonpoint pollution - A type of pollution whose source is not readily identifiable or coming from diffuse sources as opposed to a fixed, permitted industrial or municipal effluent pipe or channel.

Overturn - The process of complete vertical mixing of a stratified lake, usually occurs in fall when the relatively stagnant lower colder mass of water mixes with the upper, warmer water mass.

Pathogenic - Capable of causing infection or disease, usually refers to viruses and bacteria.

Periphyton - Aquatic organisms growing around plants.

Phytoplankton - Plant plankton.

Plankton - Microscopic organisms suspended in water without effective locomotion against currents.

Pollutant delivery ratio - Ratio of the quantity of pollutant or sediment actually delivered to a receiving water compared to the total amount generated at the point of erosion or washoff; reflects the deposition of pollutants in runoff conveyances.

Polychaetes - A group of the annelid worm (segmented bodies) phylum characterized by the presence of many locomotion bristles per body segment.

Polychlorinated Biphenyls (PCB) - A group of industrial compounds known to be bioaccumulative; their use is now largely banned in the U.S. but due to their persistence are still widespread in the environment.

Point source pollution - Types of pollution which can be traced to readily identifiable sources, such as industrial or municipal effluents.

Production - The rate of tissue elaboration in an organism.

Radioisotope dating - Determining the age of a material by measuring the rate of decay of a given radioactive substance (isotope) contained naturally in the material.

Rhizome - An underground stem.

Runoff - That portion of a rainfall or snowmelt event, after depression storage and infiltration capacities have been satisfied, which moves overland as sheetflow or in channels and enters a water body, can originate in agricultural, urban and highway areas.

Sedimentation - The physical settling of particulates in water.

Synergistic effect - Cooperative action of discrete agencies, often where the whole effect is greater than sum of the parts; eg., combinations of chemicals which produce highly toxic conditions but are not individually toxic.

Time of concentration - Time required for a drop of water to flow from the most distant part of the watershed to the point of interest; the time required for all portions of the watershed to contribute to the flow at a given point.

Toxicity - The ability of a material to exert a short or long-term negative effect on a living organism by other than mechanical means.

Water quality criteria - The quantity of a pollutant, usually expressed as a concentration, which provides a suitable level of protection in water for a given water use such as fish and wildlife propogation, drinking water, etc.; based upon a compilation of scientific data; not a formal authoritative quantity.

Water quality standard - A formal limit on a pollutant quantity established by a legal authority (usually the state); usually based on criteria or other scientific evidence but also can be more arbitrarily derived as a result of an inadequate data base or by using high safety factors.

Zooplankton - Animal plankton.

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