Volume 4-Chapter 1

AQUATIC BIOLOGY RESOURCES

Minnesota Environmental Quality Board
Regional Copper-Nickel Study

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Regional Copper-Nickel Study
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PRELIMINARY
SUBJECT TO REVIEW

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REGIONAL COPPER-NICKEL STUDY REPORT OUTLINE

A NOTE ABOUT UNITS

This report, which in total covers some 36 chapters in 5 volumes, is both international and interdisciplinary in scope. As a result, the problem of an appropriate and consistent choice of units of measure for use throughout the entire report proved insurmountable. Instead, most sections use the system of units judged most common in the science or profession under discussion. However, interdisciplinary tie-ins complicated this simple objective, and resulted in the use of a mix of units in many sections. A few specific comments will hopefully aid the reader in coping with the resulting melange (which is a reflection of the international multiplicity of measurement systems):

1) Where reasonable, an effort has been made to use the metric system (meters, kilograms, kilowatt-hours, etc.) of units which is widely used in the physical and biological sciences, and is slowly becoming accepted in the United States.

2) In several areas, notably engineering discussions, the use of many English units (feet, pounds, BTU's, etc.) is retained in the belief that this will better serve most readers.

3) Notable among the units used to promote the metric system is the metric ton, which consists of 2205 pounds and is abbreviated as mt. The metric ton (1000 kilograms) is roughly 10% larger (10.25%) than the common or short ton (st) of 2000 pounds. The metric ton is quite comparable to the long ton (2240 pounds) commonly used in the iron ore industry. (Strictly speaking, pounds and kilograms are totally different animals, but since this report is not concerned with mining in outer space away from the earth's surface, the distinction is purely academic and of no practical importance here).

4) The hectare is a unit of area in the metric system which will be encountered throughout this report. It represents the area of a square, 100 meters on a side (10,000 m^2), and is roughly equivalent to 2½ acres (actually 2.4710 acres). Thus, one square mile, which consists of 640 acres, contains some 259 hectares.

The attached table includes conversion factors for some common units used in this report. Hopefully, with these aids and a bit of patience, the reader will succeed in mastering the transitions between measurement systems that a full reading of this report requires. Be comforted by the fact that measurements of time are the same in all systems, and that all economic units are expressed in terms of United States dollars, eliminating the need to convert from British Pounds, Rands, Yen, Kawachas, Rubles, and so forth!

Conversions for Common Metric Units Used in the Copper-Nickel Reports

1 meter (m)	=	3.28 feet = 1.094 yards
1 centimeter (cm)	=	0.3937 inches
1 kilometer (km)	=	0.621 miles
1 hectare (ha)	=	10,000 sq. meters = 2.471 acres
1 square meter (m^2)	=	10.764 sq. feet = 1.196 sq. yards
1 square kilometer (km^2)	=	100 hectares = 0.386 sq. miles
1 gram (g)	=	0.037 oz. (avoir.) = 0.0322 Troy oz.
1 kilogram (kg)	=	2.205 pounds
1 metric ton (mt)	=	1,000 kilograms = 0.984 long tons = 1.1025 short tons
1 cubic meter (m^3)	=	1.308 yd^3 = 35.315 ft^3
1 liter (l)	=	0.264 U.S. gallons
1 liter/minute (l/min)	=	0.264 U.S. gallons/minute = 0.00117 acre-feet/day
1 kilometer/hour (km/hr)	=	0.621 miles/hour
degrees Celsius (°C)	=	(5/9)(degrees Fahrenheit -32)

Volume 4-Chapter 1 AQUATIC BIOLOGY RESOURCES

1.1 INTRODUCTION AND SUMMARY OF FINDINGS

As discussed in the Water Chapter, (Volume 3-Chapter 4) northeastern Minnesota and the Regional Copper-Nickel Study Area (Study Area) have numerous lakes and streams. The Study Area contains approximately 310 lakes over 10 acres in size or approximately 2 percent of the lakes in the state of Minnesota. The 2623 kilometers of streams in the Study Area range in size from small creeks, less than 1/2 meter across, to large rivers over 50 meters wide. Approximately 9 percent of the 5223 KM^2 Study Area is covered by water and provides diverse habitat for a variety of aquatic organisms.

These streams and lakes constitute a valuable resource and serve several important functions. They are habitats for fishes and other valuable aquatic organisms, which are utilized for both recreational and commercial purposes. Lakes and streams are also important because they function as natural "sinks" for "wastes" from the terrestrial system.

Copper-nickel development may place "new" stresses (factors that cause a change in the development or continued existence of aquatic ecosystems or their components) on these valuable aquatic resources. These stresses may have both direct and indirect impacts (measurable changes in the ecosystems) on the biota of the lakes and streams in the Study Area. For example, increased sedimentation from mine sites may result in decreased spawning in the Area's streams and lakes and thereby decrease the number of fish found in the Area (see section 1.7.8). Air emissions from processing operations or smelters may reach the water and result in changes in water chemistry and/or heavy metals content (see Volume 3-Chapters 3 and 4) which may be toxic to aquatic life.

The Study Area comprises a large region with several major watersheds. The lakes and streams in these watersheds are variable in both physical and chemical characteristics. This variability indicates possible variability in susceptibility to impacts from the potential stresses of copper-nickel development.

Copper-nickel development could result in several significant impacts on the aquatic resources of the Study Area. The leaching of heavy metals from waste rock/lean ore piles may, if not mitigated, could result in serious damage to large portions of the Study Area's rich aquatic resources. Potential heavy metals loadings as projected in Volume 3-Chapter 4 would exceed by a great amount the tolerance levels of aquatic organisms in the Study Area. The relatively high total organic carbon content of the waters in the Study Area appears to have some ameliorating affects on the toxicity of some of the heavy metals (copper in particular) while the toxicity of others appears to more directly affected by hardness, and other parameters of water chemistry.

The alkalinity and pH of some of the waters in the Study Area appear to be decreasing as a result of acidified precipitation. If these trends continue it is likely that severe damage will result in the aquatic ecosystem. Although not especially related to the development of copper-nickel in the region decreases in the pH of precipitation in the region may result in increased rates of release of heavy metals from any copper-nickel waste rock/lean ore stockpiles which are established. Losses of fisheries and other aquatic organisms can be expected to occur if pH in lakes or stream decrease below values of 6.0. In particulare some lakes which already have calcite saturation indices above 3.0 may be most susceptible to acidic precipitation. Several general types of physical impacts can be expected in relation to the development of copper-nickel resources including: changes in stream flow, channelization/diversion, temperature changes, increased suspended solids loadings and loss of terrestrial vegetation. If planning efforts are not directed toward mitigating the potential impacts from these physical impacts it is likely that there would be significant losses of fisheries and organisms lower in the trophic system. These impacts would be localized in comparison to the impacts which could potentially result from heavy metals pollution. Secondary development may impact the Study Area's aquatic resources if it is not planned in a coordinated fashin. Increased human populations in the region would increase nutrient loads in lakes and streams. Since the majority of lakes and stream in the Study Area are already mesoeuthrophic it is likely that secondary development, which is expected to be concentrated near water amenities, would increase nutrient loads to the point where some lakes could become euthrophic. These euthrophied lakes would probably experience nuisance algal blooms and their use as fishing and swimming lakes would probably have to be decreased significantly.

Many of the potential impacts of copper-nickel development on the Study Area's aquatic resources could possibly be mitigated. Mitigation of physical impacts and the problems associated with secondary development is fairly well understood.

Mitigation of the problems associated with heavy metals releases are only poorly understood at this time. There are several possible approaches to the problem of mitigating the heavy metals pollution stresses the best one would probably be to eliminate the potential discharge. Although it may be possible to eliminate this discharge during the life of the mine further studies are necessary to determine the possibilities for eliminating such a discharge after the operation ceases. In some situations revegetation of waste rock/lean ore piles may be a useful method for decreasing runoff and potential leaching problems. This method however, may not be effective in this situation, in fact it may result in increased leaching problems. Vegetation placed on the surface of one of these stockpiles may release organic acids which could increase the leaching potential of the stockpile. Further studies of test stockpiles, such as those which are currently being conducted by the Minnesota Department of Natural Resources, will be necessary before an appropriate solution to the long term potential of heavy metals leaching can be adequately established.

1.2 PHILOSOPHY AND METHOD OF APPROACH TO REGIONAL CHARACTERIZATION

The physical and chemical variability in the Study Area coupled with historical factors has produced a large number of complex ecosystems which contain a wide variety of aquatic species. These ecosystems and species are likely to vary in their responses to the potential stresses resulting from copper-nickel development. It is necessary to characterize (semi-quantitatively describe the important components and processes comprising aquatic ecosystems and their functional relationships) these ecosytems and species to estimate their susceptibility to these stresses and predict the impacts which may occur in conjunction with copper-nickel development. This study was directed toward characterizing the ecosystems in a semi-quantitative fashion because there are currently no adequate quantitative methods for predicting the responses of a given ecosystem to the stresses of mining development. This characterization is different from a baseline study where many years of sampling are necessary to quantitatively establish "normal" conditions and the natural variability in the systems. Site-specific quantitative baseline data can be used to compare pre-operational, operational, and post-operational conditions to determine what if any impacts have occurred. If copper-nickel development occurs, and if the establishment of baseline conditions is desired, then additional monitoring will be necessary.

In order to assess the impacts of copper-nickel development on the aquatic biota of the Study Area it was necessary to develop a classification system for aquatic communities. This classification serves to "simplify" or group these ecosystems into a relatively smaller number of distinct types which can be considered in more detail than if each species or "sub-community" were evaluated and it allows for the extrapolation of the prediction of impacts to portions of the Study Area and northeastern Minnesota where sampling could not be conducted.

Field studies were undertaken during the spring, summer, and fall of 1976 and 1977 because few historical data on the biological components of the aquatic ecosystems in the Study Area are available. These studies were designed to identify the dominant organisms and communities, describe their spatial and temporal distribution in the Study Area, and determine the similarities among the various ecosystems for the purpose of developing this classification system.

The field studies were established in conjunction with water quality studies (see Volume 3-Chapter 4) to provide data on the relationship of biological communities (i.e. the aquatic classification) to physical and chemical factors. In addition, literature reviews were undertaken to gather information on the environmental requirements, life histories, and trophic relationships of the dominant organisms identified during the field studies. The combination of field studies, literature information, and data from related studies in the region, provides the basis for a regional characterization of the aquatic ecosystems in the Study Area.

The characterization of the aquatic ecosystems serves as a basis for identifying sensitive components or systems within the Study Area. This information is then employed as the foundation for the assessment of impacts from copper-nickel development on the aquatic ecosystems of northeast Minnesota.

Aquatic ecosystems have traditionally been characterized by the indigenous species in the system and their abundance. Species and their population sizes have been used as an indication of the prevailing environmental conditions in streams and lakes. Species diversity provides an index of the relation of the number of taxa and the number of individuals in a population. Diversity is highest when there is an even distribution of individuals among taxa. Generally, it is thought that greater diversity indicates a healthier community and therefore cleaner environment. In recent years diversity has gained prominence as a measure of the condition of aquatic ecosystems. However, reliance on diversity indices can be misleading, as low diversity has been reported in clean environments when clean water species dominate the community (Archibald 1972).

Recently it has been suggested that the analysis of the trophic or functional relationships within aquatic ecosystems using parameters such as chlorophyll a, biomass, and relative population size of trophic levels may be more meaningful than the study of indigenous species and diversity in interpreting the health and nature of aquatic ecosystems. In this study primary emphasis was placed on the examination of functional relationships and secondary emphasis on the indigenous species and diversity.

Life in the aquatic biosphere is unified by the water medium in which it occurs. Resources which are utilized by aquatic organisms are all transported through water although they may reach the water in a variety of ways. Studies of the aquatic biosphere have been divided on the basis of trophic (food chain) levels to show the patterns of flow for stresses which may result from copper-nickel development. These trophic levels are: primary producer (aquatic vegetation), herbivore (aquatic invertebrates, small fish and moose), and carnivore (some invertebrates, most fish and, in some instances, birds or mammals from the terrestrial biosphere).

The differences between the biota of lakes and streams require that these systems be studied separately, although the connection between them will be eva-luated. Stream communities are composed of three major groups of organisms:

1) Periphyton (attached algae) are the major primary producers although macrophytes (large aquatic plants) do exist in streams. In addition to the production of oxygen periphyton provide food and habitat for the aquatic invertebrate fauna. Because periphyton are stationary, they integrate the effects of changing physical and chemical factors in a given area and may act as biological monitors of water quality;

2) Invertebrates (primarily insects, molluscs, and crustaceans) are the next link in the food chain and are a primary source of food for fish. For this reason it is important to monitor changes in these communities. Invertebrates are relati-vely immobile within a stream and like periphyton, tend to reflect stream con-ditions at a given point. Changes in community structure and species composition may indicate changes in environmental conditions which are not detected through direct sampling of water quality or other physical parameters;

3) Fish are the last step in stream food chains. Some streams are spawning areas for fish from lakes and many serve as habitat for young fish and minnow species that prefer flowing water.

Lake communities have a trophic structure similar to that of stream communities, however, the proportion of free floating organisms in lakes is much larger than in streams. In the case of lakes the groups of organisms include:

1) Phytoplankton (algae) and macrophytes are the primary producers in lakes. Phytoplankton are a food source for lake herbivores (zooplankton, fish, etc.).

2) The zooplankton (microscopic animals), and other herbivores are also important food sources for fish.

3) Bottom dwelling invertebrates (primarily molluscs and insects), feed on decomposing organic matter and are in turn preyed upon by fish. As in streams, most fish in lake communities are carnivores.

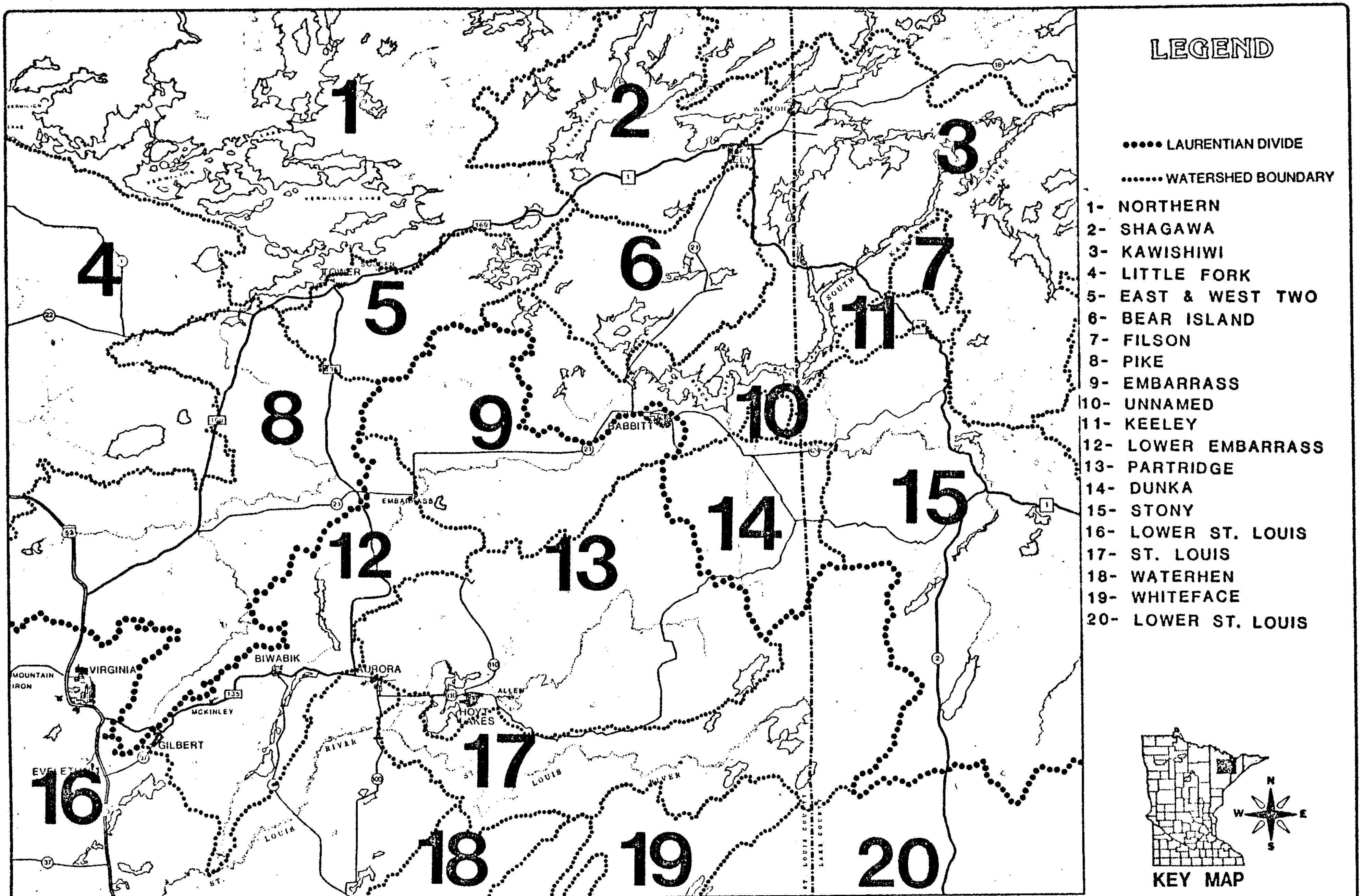
1.3 REGIONAL OVERVIEW

The streams and lakes of the Study Area form two distinct ecosystems. Characteristic species or groups of species are generally different in lakes and streams. These differences are primarily the result of adaptation by individual species or groups of species to survival in flowing or standing water. Because these differences exist, streams and lakes will be discussed separately.

The Study Area is divided into two major watersheds by the Laurentian Divide. Water north of the Divide flows through the Rainy River Watershed to Hudson Bay. South of the Divide, water flows through the St. Louis River Watershed system to Lake Superior.

The Rainy River Watershed has been divided into 12 sub-watersheds and the St. Louis River Watershed into seven sub-watersheds (Figure 1a). These sub-watersheds are hydrologic entities which were determined by major rivers, streams, and lake basins. The water quality within these sub-watersheds is similar except in those areas where streams are affected by taconite mining (see Volume 3 Chapter 4 for further details). Figure 1b also indicates the distribution of streams of various orders in the Study Area. The importance of stream order is discussed in section 1.3.1.

Figures 1a and 1b



MEQB REGIONAL COPPER-NICKEL STUDY

FIGURE 1A WATERSHEDS

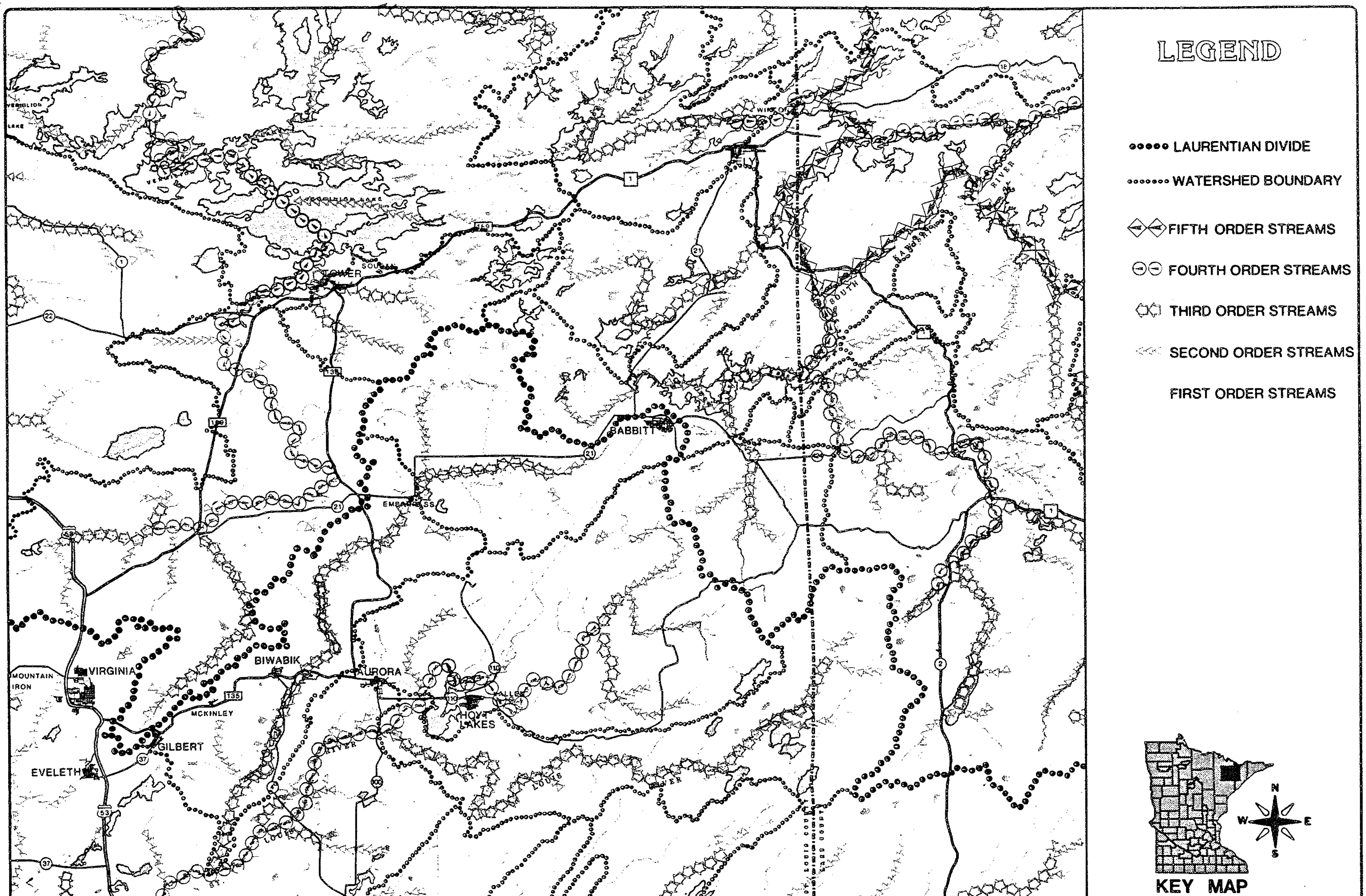


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FIGURE 1B

STREAM ORDERS

1.3.1 Streams

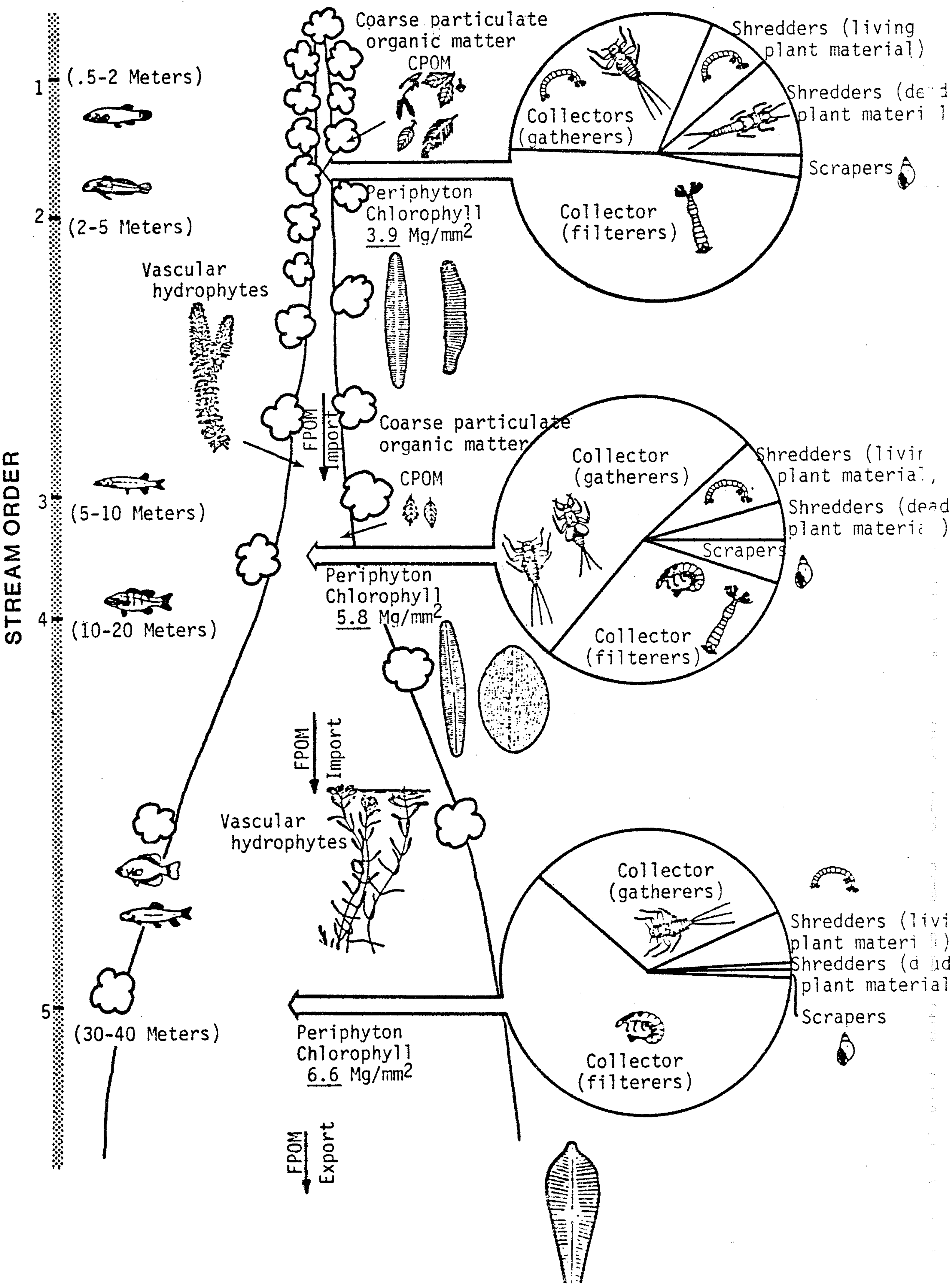
Aquatic life in Study Area streams is diverse. Sampling during 1976 and 1977 identified 433 diatom taxa, 39 macrophyte taxa, 424 benthic invertebrate taxa at 54 sampling sites, and 40 fish species at 98 sampling stations. These organisms were distributed among their preferred habitats throughout the area. The taxa found in Study Area streams are similar to those found in other northern woodland streams in Minnesota, Wisconsin, Michigan, and Ontario (see first level reports, e.g. Lager et al. 1978, for further details).

The aquatic communities in Study Area watersheds are similar among watersheds of equal stream order. The order of a stream is dependent upon its position in the watershed. The headwater streams are first order; second order streams are formed when two first order streams meet; and, further increases in order occur as streams of equal order join lower in the watershed (Horton 1945, Strahler 1957). Within watersheds, communities change as a continuum from the headwaters to downstream areas. These changes are generally related to stream order, which proved to be a useful classification scheme for aquatic communities in the area and allows description of communities in unsampled areas. Figure 2 summarizes the general relationships between stream order and the biological communities.

Figure 2

Within Study Area watersheds there is a longitudinal succession of aquatic communities. Changes in community function and size of functional groups are related to changes in stream order. In general, the amount of autochthonous material (organic matter produced within the stream) increases and the amount of allochthonous material (organic matter derived from the terrestrial environment)

FIGURE 2 RELATIONSHIP OF COMMUNITY TO STREAM ORDER (FROM CUMMINS 1976)



decreases with increasing stream order. These changes are reflected in the relative abundance of invertebrates which utilize these food sources. Changes in fish communities are also related to changes in stream order. As stream order increases, there is an increase in the number of fish species present and in the size of individuals. This reflects the availability of more habitat, greater habitat diversity, and more food downstream (Williams et al. 1978).

The Kawishiwi River, the Little Isabella watershed including Snake River and Snake Creek, and the headwaters of the Cloquet River watershed are distinct from other Study Area watersheds (Figure 1). The Kawishiwi River is a series of river-lakes connected by short riffles. Except in these riffles, where filter-feeding insects predominate, fish and invertebrate communities in the Kawishiwi River more closely resemble those in Study Area lakes than those in other streams. The Little Isabella and Cloquet watersheds are unique because they contain the highest percentage of trout streams in the Study Area. Other cold-water species associated with trout also occur in greatest abundance in these watersheds. These species are all relatively uncommon and are found only in the few widely separated cold-water habitats in the Study Area (Williams et al. 1978).

1.3.2 Lakes

Lakes in the Study Area have fewer species than the streams. The sampling program identified several hundred species of phytoplankton, 76 zooplankton taxa, 18 macrophyte taxa, 119 benthic invertebrate taxa at 44 sampling sites, and 42 fish species in 112 lakes previously sampled by the MDNR. The species identified in Study Area lakes are similar to the species found in other Minnesota and Wisconsin lakes, and in the Experimental Lakes Area (ELA) of Ontario. Lakes in the Study Area are moderately productive (chloropyll a concentrations 5-20 ug/1, phosphorus 20-50 ug/1), soft-water lakes. Most of the 310 lakes in the Study Area are highly colored because of bog drainage. The dominant taxa are generally the same in all lakes.

Based on their trophic status, two types of lakes are present in the Study Area: oligotrophic and meso-eutrophic (see Volume 3-Chapter 4 for further details). The majority of lakes in the Study Area are meso-eutrophic. These lakes are not as deep or clear as oligotrophic lakes but have higher biological productivity. Within this group of lakes there is a wide range of physical and chemical conditions so that differences are not obvious. None of these lakes are capable of supporting trout but otherwise the biological communities are similar to those found in oligotrophic lakes (Seisennop et al. 1978).

Oligotrophic lakes are clear, deep lakes and are found in the northern portion of the Study Area. Trout populations can be maintained in the oligotrophic lakes because the hypolimnion in these lakes remains cold (less than 20°C) and well oxygenated (dissolved oxygen greater than 6 mg/l)(see Figure 21 regarding lake strata throughout the year.

1.4 REGIONAL CHARACTERIZATION

1.4.1 Stream Ecosystems

The geology, topography, and atmospheric conditions in a watershed are the principal determinants of the physical and chemical conditions of a stream ecosystem. These in turn determine the characteristics of the stream community. The physical and chemical factors considered to be most important to stream organisms include: stream size, temperature, substrate, current velocity, hardness, pH, total dissolved inorganic matter, total dissolved organic matter, particulate organic matter, and streamside vegetation. Most aquatic organisms have wide tolerance ranges for these physical and chemical factors, and can be found under greatly varying conditions. The optimal conditions for a given species are reflected in its abundance and growth rate.

Woodland stream ecosystems in the temperate zone are dependent on the surrounding watershed for the majority of their energy in the form of terrestrial vegetation (allochthonous material). The importance of allochtho-nous material in the stream ecosystem diminishes as stream order increases and the input of allochthonous materials relative to autochthonous primary production decreases. Primary production in streams increases with increasing stream size and decreasing shading by terrestrial vegetation.

Stream macroinvertebrates are adapted to process and utilize the available food sources, thus transfering energy through the food web. These macroinvertebrates can be placed into functional groups which describe their feeding habits and thus their role in the ecosystem (Table 1). Shredders of dead plant material, which feed on coarse particles of organic terrestrial matter (CPOM) (greater than 1mm in size), function to reduce organic particle size by their feeding activities. The actual food source for the shredders of dead plant material may be the fungi and bacteria which are decomposing the organic matter. As the allochthonous material is broken into finer particles (FPOM) (less than 1mm in size), it becomes the food of collectors that either filter the water or gather the particles from the substrate. Periphyton growing on stream bottoms are consumed by scrapers which are adapted to remove attached algae from rocks, logs, and other submerged substrates. Macroalgae and vascular hydrophytes are fed on by shredders of living plants and by invertebrates adapted to feed by piercing cellular plant tissues. Invertebrate predators feed on all invertebrates in the streams.

Table 1

Fish act as primary, secondary, and tertiary consumers by feeding on detritus, algae, invertebrates, and fish. Figure 3 demonstrates the trophic relationships which are characteristic of streams.

Figure 3

Within watersheds there are physical and chemical gradients from headwater streams to the mouth (Hynes 1970, Pennak 1971). These physical-chemical trends have been related to changes in stream order, as defined by the branching of streams within watersheds (Horton 1945, Strahler 1957, Harrel and Dorris 1968). Because physical-chemical conditions determine the characteristics of stream communities, attempts have been made to relate biological parameters to stream order (Kuehne 1962, Harrel et al. 1967, Harrel and Dorris 1968). Stream orders within the Study Area are indicated on Figure 1b.

Cummins (1975) suggested that stream order provides a simple method of classifying stream ecosystems. Stream order classification is based on the relationship between stream order and the structure and function of stream ecosystems (Cummins 1974; 1975; 1976). Cummins recommended that comparisons between streams be made on the basis of functional relationships rather than on distinctions among the indigenous species which may be taxonomically different but functionally similar.

Table 1. Dominant genera of invertebrate functional groups in Study Area stream orders one through five in 1977 drift samples. The primary food source of each functional groups is indicated.

FUNCTIONAL GROUP	FOOD SOURCE	DOMINANT GENERA — STREAM ORDER 1	2	3	4	5
Shredders of dead plant material	Detritus 1-4 mm; mainly leaf litter	Amphinemura Leuctra Platycentropus Lemnephilus Nemotaulius	Leuctra Paracapnia Endochironomus Grammotaulius Lepidostoma	Endochironomus Leuctra Paracapnia Platycentropus Pycnopsyche	Endochironomus Paracapnia Lepidostoma Brillia Shipsa	Endochironomus Brillia Lepidostoma
Shredders of living plant material	Living vascular hydrophytes and macroalgae	Cricotopus Helophorus (adult) Haliplidae Polypedilum	Cricotopus Polypedilum	Cricotopus Polypedilum Haliplidae Ptilostomis	Polypedilum Cricotopus Ptilostomis Triaenodes	Cricotopus Polypedilum
Collector-gatherers	Detritus less than 1 mm; on or within the substrate	Paraleptophlebia group Baetis group Eukiefferiella Ephemerella Chironomus	Baetis group Paraleptophlebia group Eukiefferiella Hyalella Ephemerella	Paraleptophlebia group Baetis group Ephemerella Hexagenia Eukiefferiella	Paraleptophlebia group Ephemerella Hyalella Tricorythodes Baetis group	Paraleptophlebia group Baetis group Hexagenia Eukiefferiella Ephemerella
Collector-filterers	Detritus less than 1 mm; suspended in the water	Simuliidae	Simuliidae	Simuliidae Hydropsyche group	Simuliidae Hydropsyche group Chimarra	Hydropsyche group
Scrapers	Periphyton	Gastropoda Glossosoma Hydraenidae (adult) Heptagenia	Gastropoda Chloroperlidae Pseudocloeon Epeorus Heptagenia	Pseudocloeon Gastropoda Heptagenia Choroterpes Epeorus	Choroterpes Pseudocloeon Gastropoda Heptagenia	Pseudocloeon Heptagenia Choroterpes Gastropoda

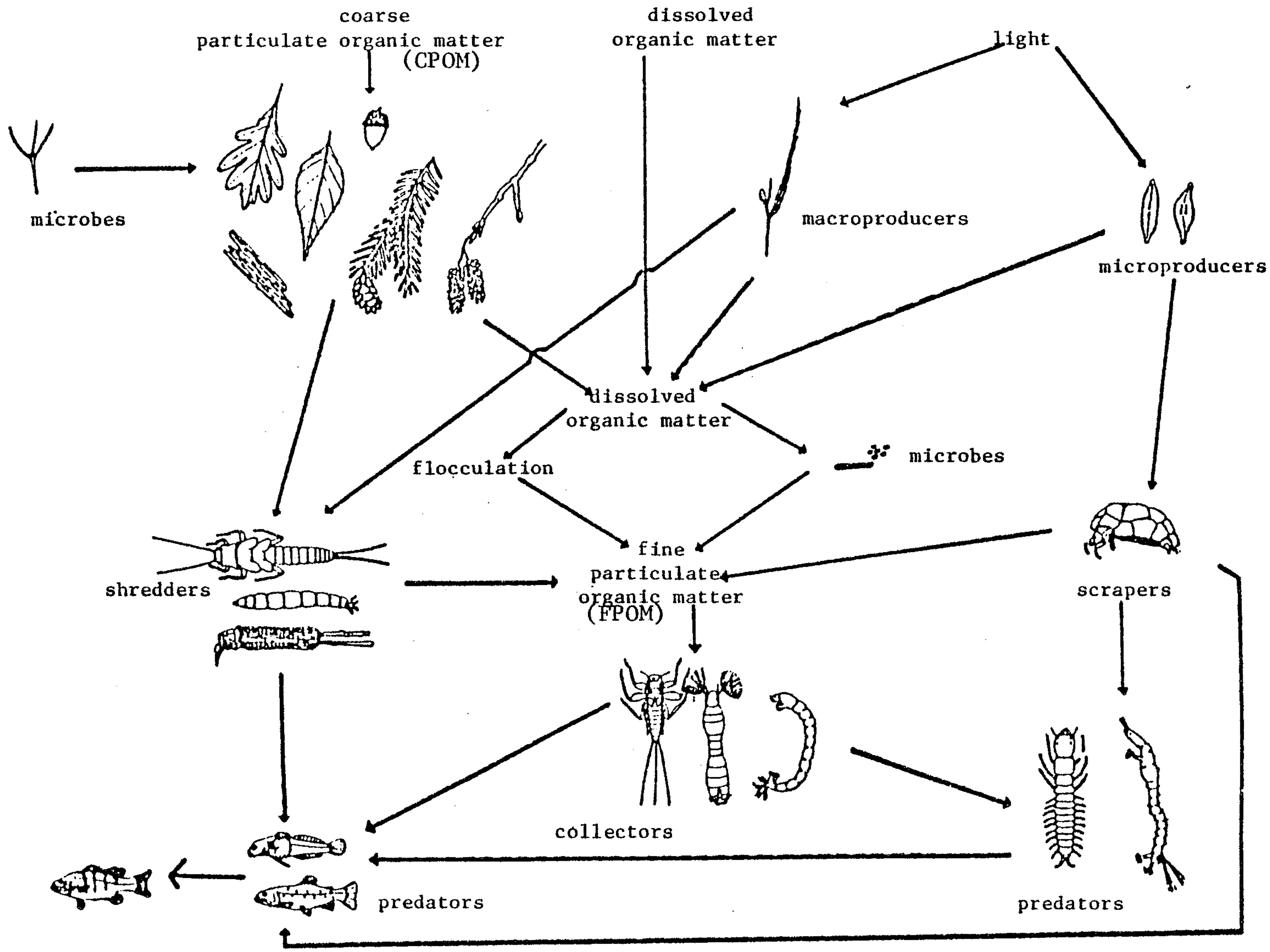


FIGURE 3 TROPHIC RELATIONSHIPS IN STREAMS (FROM CUMMINS 1976)

The classification of streams by order does have several drawbacks. First, streams with similar drainage areas may be of different order (Hynes 1970). Second, an adventitious stream (one which enters a stream of higher order) may resemble the stream it enters more closely than it resembles streams of the same order (Harrel and Doris 1968, Whiteside and McNatt 1972). Finally, aquatic ecosystems change along a steady gradient, and it is not possible to define exact zones (Hynes 1970). This third factor is a problem inherent in any attempt to classify aquatic ecosystems.

The stream order system of classification was chosen for this study because it provides a simple basis for describing stream ecosystems without on-site inspection. Stream order classification also has merit because it synthesizes many factors biologists previously used separately for stream classification.

Field studies of stream ecosystems conducted by Regional Study staff were designed to determine the relationship between stream order and stream communities. During 1976 and 1977, 50 stations were sampled qualitatively for periphyton and benthic invertebrates; fish were sampled semi-quantitatively at 98 stations (Figures 3 and 4). This sampling provided information on the distribution of species in the Study Area. Quantitative invertebrate drift sampling at 30 stations was used to relate stream order to invertebrate functional groups while quantitative periphyton samples were collected at 25 stations to relate productivity and species composition to stream order. The decomposition rates of aspen and red pine leaves were studied at 8 stations. Quantitative periphyton and invertebrate sampling was done intensively at 17 stations to determine the seasonal variations in aquatic communities. In addition to the Study's fish survey, existing MDNR stream survey data were compiled. All MDNR classifications for streams in the Study Area were reviewed and updated

(Williams et al. 1978). Figure 6 is a diagram of the mineral resource zones in the Copper-Nickel Study Area, sampling efforts were directed towards this portion of the region.

Figures 4, 5, and 6

1.4.1.1 Physical and Chemical Conditions in Study Area Streams--Measurements from topographic maps reveals some 1,625 km of streams in the Regional Copper-Nickel Study Area. Generally, these streams are soft-water and slightly acid. Alkalinities range from 10 to 140 mg/l as $CaCO_3$ but are generally less than 100 mg/l. The pH values range from 6.1 to 7.6 and are generally below 7.2. Total organic carbon in Study Area streams ranges from 5 to 30 ug/l, because most of these streams flow through bogs at points along their courses (see Volume 3 Chapter 4 for further details).

Most of the streams consist of long flat stretches connected by short riffles. The substrates along the flat stretches consist of silt, sand, and detritus and, in the riffles, range from gravel through bedrock. Peak flows of Study Area streams occur during the spring snowmelt with subsequent smaller peaks following heavy summer rains (see Volume 3-Chapter 4).

Physical and chemical properties are generally related to stream order. The number of kilometers recorded for the various stream orders in each watershed is presented in Table 2 with the gradients of streams which were sampled. As stream order increases, the average gradient decreases (Figure 7), the number of streams decreases, and their average length increases (Figure 8). Further, the size of streams (i.e. width, depth, discharge) increases with increasing stream order. Increased stream width results in a diminished terrestrial canopy cover and a

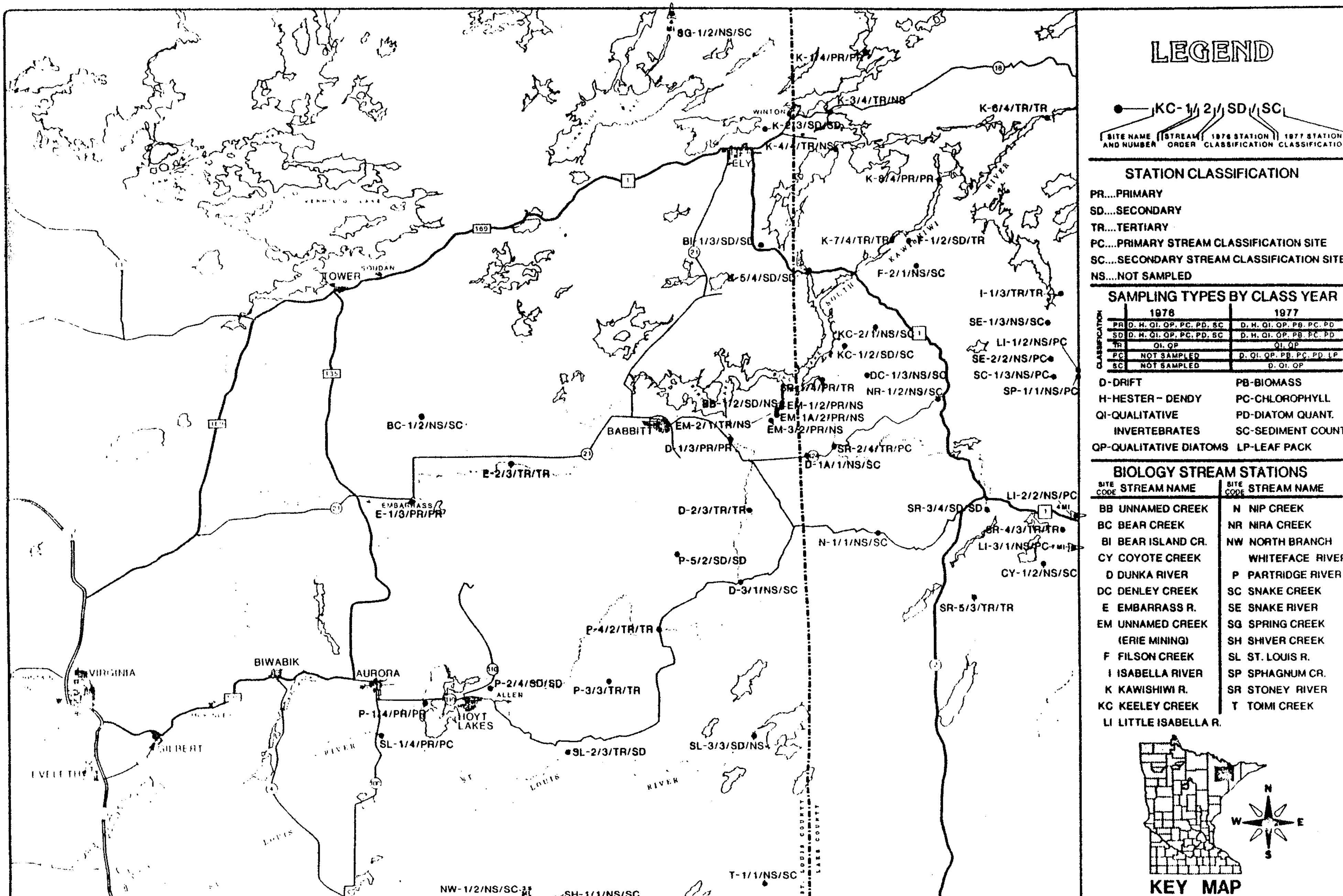


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FIGURE 4 STREAM STUDY STATIONS

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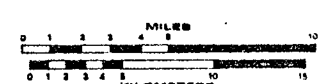

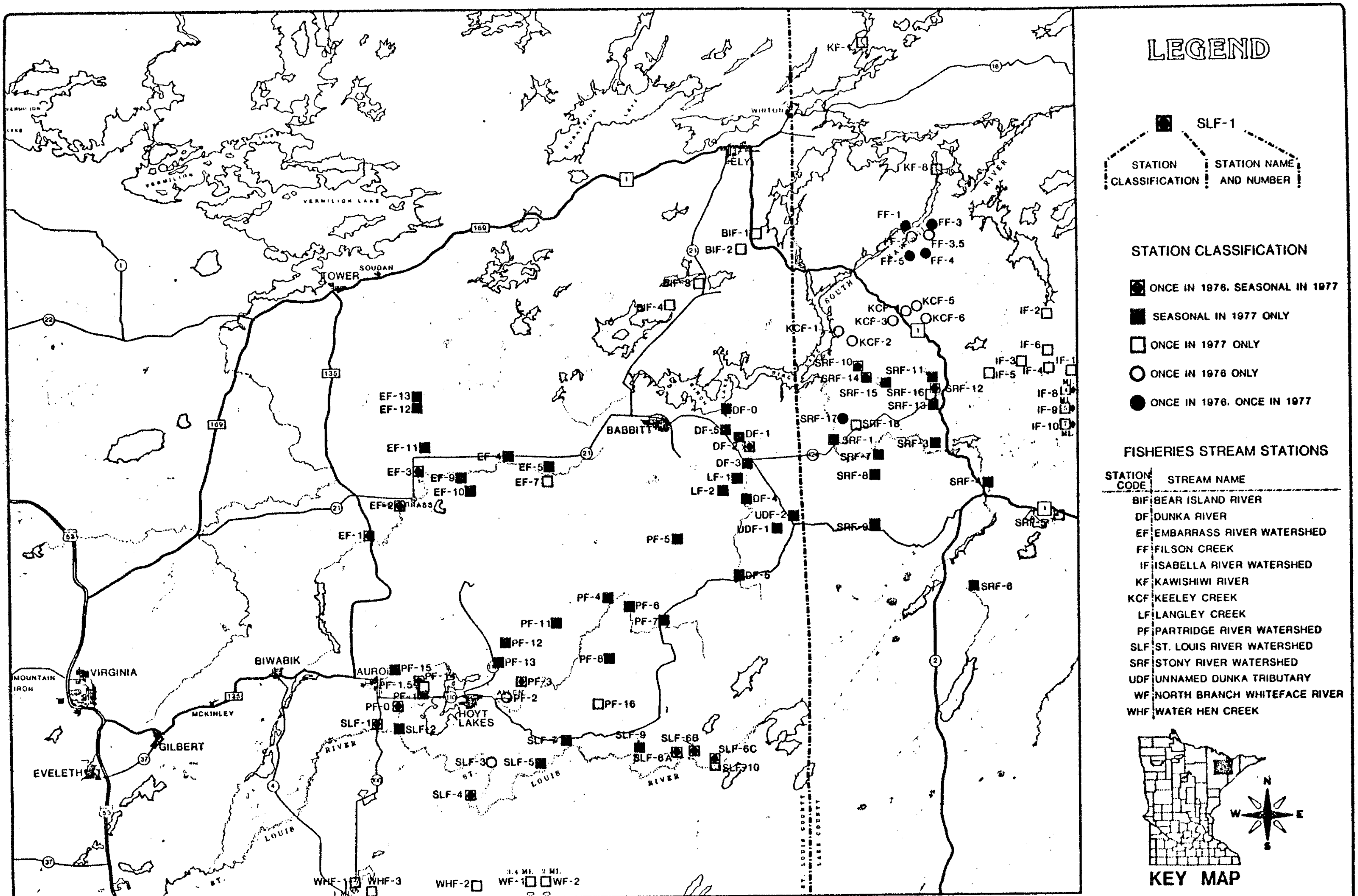

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FIGURE 5 FISHERIES STREAM STUDY STATIONS

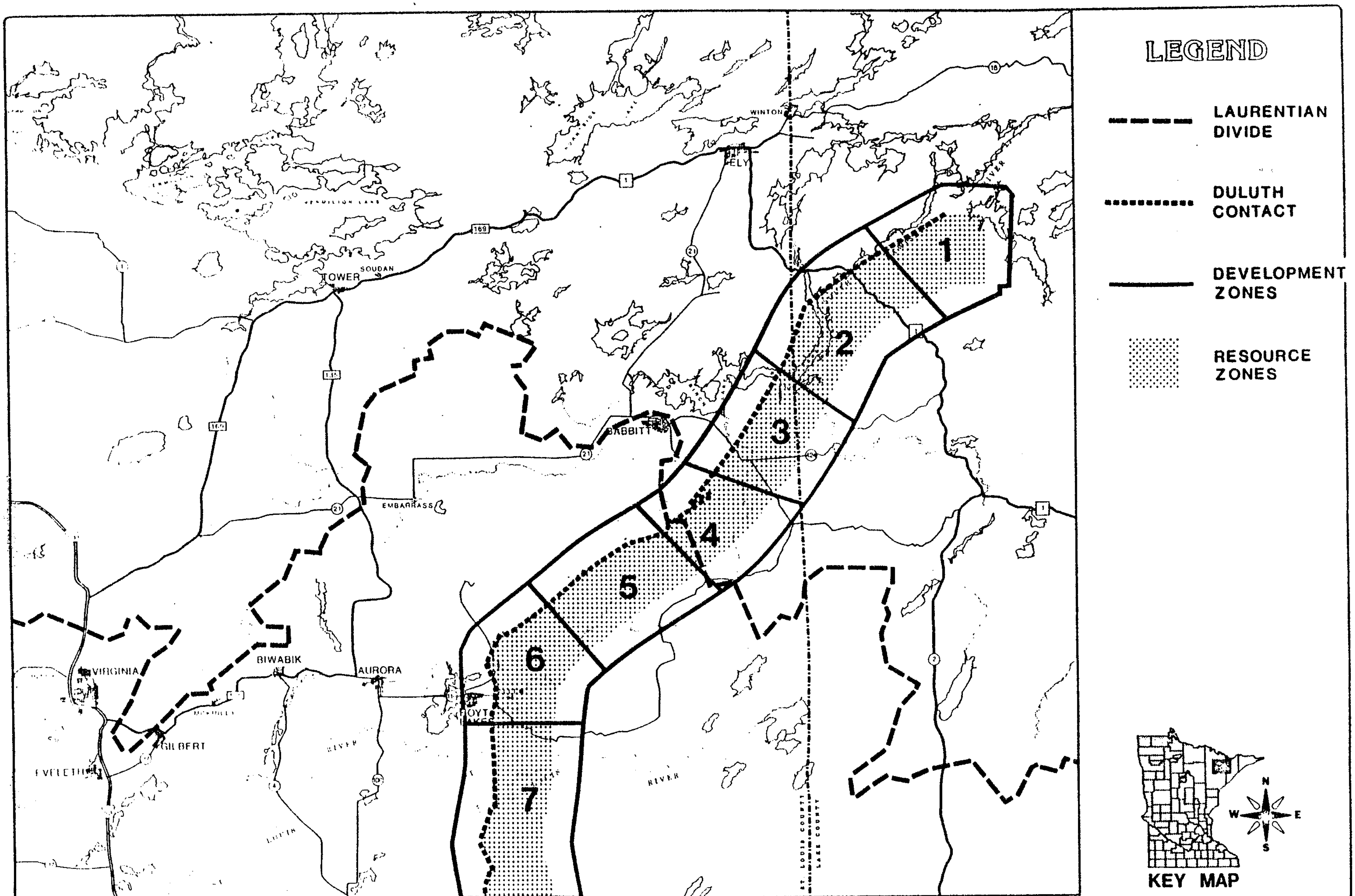

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FIGURE 6 MN CU-NI DEVELOPMENT AND RESOURCE ZONES (VOLUME 3-CHAPTER 2) reduction in the amount of allochthonous material entering the stream relative to stream volume and therefore a reduction in the amount of dissolved organics. Preliminary analyses indicate increases in pH and in the concentration of anions and cations with increasing stream order (see Volume 3-Chapter 4 for further details).

Table 2, Figures 7 and 8

1.4.1.2 Biological Characteristics of Study Area Streams--The physical and chemical changes described above are important to aquatic organisms. Changes in biological communities in Study Area streams are related to stream order (Lager et al. 1978)(Figures 9 through 15) and appear to be similar to the changes discussed by Cummins (1974; 1975; 1976). Although the trends indicated in these figures are related to changes in stream order, organisms do not recognize stream order boundaries; therefore, communities change gradually with increasing stream order. Because of the gradual change in community function with increasing stream order and the variability of data among replicates within sites and among sites within stream orders, three stream order groups were formed and will be discussed separately. First and second order streams (headwater), third and fourth order (midreaches), and fifth order streams (Kawishiwi River) are considered as the major stream types in the Study Area.

There are two major types of headwater streams in the Study Area: streams draining upland forest areas, and streams which drain lowland bog areas. The upland forest headwater streams have higher average gradients (4 m/km) than the lowland streams (3 m/km) and a canopy cover of approximately 50-100 percent. The riparian vegetation is primarily alder, but aspen, birch, pine, spruce, and fir also contribute to the canopy cover and provide allochthonous material to the

Table 2. Stream miles, number of streams and average stream length by order measured in Study Area watersheds.

	Water-shed #	1st ORDER Miles	1st ORDER # Streams	1st ORDER $\bar{X}$ Length	2nd ORDER Miles	2nd ORDER # Streams	2nd ORDER $\bar{X}$ Length	3rd ORDER Miles	3rd ORDER # Streams	3rd ORDER $\bar{X}$ Length	4th ORDER Miles	4th ORDER # Streams	4th ORDER $\bar{X}$ Length	5th ORDER Miles	5th ORDER # Streams	5th ORDER $\bar{X}$ Length	Total Miles
North of Divide																	
Bear Island	1	20.3	16	1.27	13.0	5	2.6	14.45	1	14.45							47.75
Dunka	2	21.45	14	1.5	9.8	4	2.45	7.95	1	7.95							39.2
East and West Two Rivers	3	19.4	13	1.49	11.35	4	2.84	12.7	2**	8.35							43.45
Filson	4	8.6	2	4.3	0.15	1*	0.15										8.75
Isabella	5	1.6	1	1.6	0.35	1	0.35	5.3	1	5.3				1.0	1*	1.0	8.25
Kawishiwi	6	77.05	58	1.33	41.55	15	2.77	17.2	3	5.73	28.2	3	9.4	43.75	1	43.75	207.75
Keeley	7	7.1	3	2.4	4.6	1*	4.6										11.7
Little Fork	8	44.05	28	1.57	11.55	7	1.65	19.0	2	9.5							74.6
Northern	9	108.45	71	1.53	76.3	17	4.49	20.3	5	4.06	26.7	1*	26.7	3.0	1*	3.0	234.75
Pike	10	52.2	51	1.02	20.05	13	1.54	26.5	3	8.83	32.1	1*	32.1				130.85
Shagawa	11	31.3	26	1.20	30.3	9	3.4	14.65	2	7.3	1.9	1	1.9				78.15
Stony	12	56.35	66	0.85	43.6	16	2.73	26.55	5	5.71	24.65	1*	24.65				153.15
Unnamed Creek	13	1.75	1*	1.75													1.75
*split streams			1			2			4			3			2		
TOTAL -North		447.6	349	1.29	262.6	91	2.89	166.6	21	7.93	113.55	4	28.39	47.75	1	47.75	1070.1
South of Divide																	
Embarrass	14	24.6	13	1.89	8.1	3	2.7	14.2	1*	14.2							46.9
Lower Embarrass	15	22.75	21	1.08	4.55	2	2.28	34.5	1	34.5							61.9
Partridge	16	50.35	29	1.7	35.6	7	5.1	8.4	2	4.2	19.1	1*	19.1				113.45
St. Louis	17	23.8	21	1.1	8.95	4	2.2	39.75	1	39.75	4.15	1*	4.15				76.65
Lower St. Louis	18	142.5	92		56.4	24		15.75	5		28.95	1	28.95				243.6
Water Hen	19	12.95	12	1.08	5.05	2	2.53										18.0
Whiteface	20	23.35	7	3.34	2.65	2	1.3	5.3	1	5.3							31.3
*split streams			0			0			1			2					
TOTAL -South		30.3	1.95	1.54	121.3	44	2.76	117.9	10	11.79	522	1	52.2				591.7
														TOTAL MILES WITHIN STUDY AREA			1631.8

*Split streams--ordered segments which cross a watershed boundary; although found in two watersheds, they are counted as one stream.

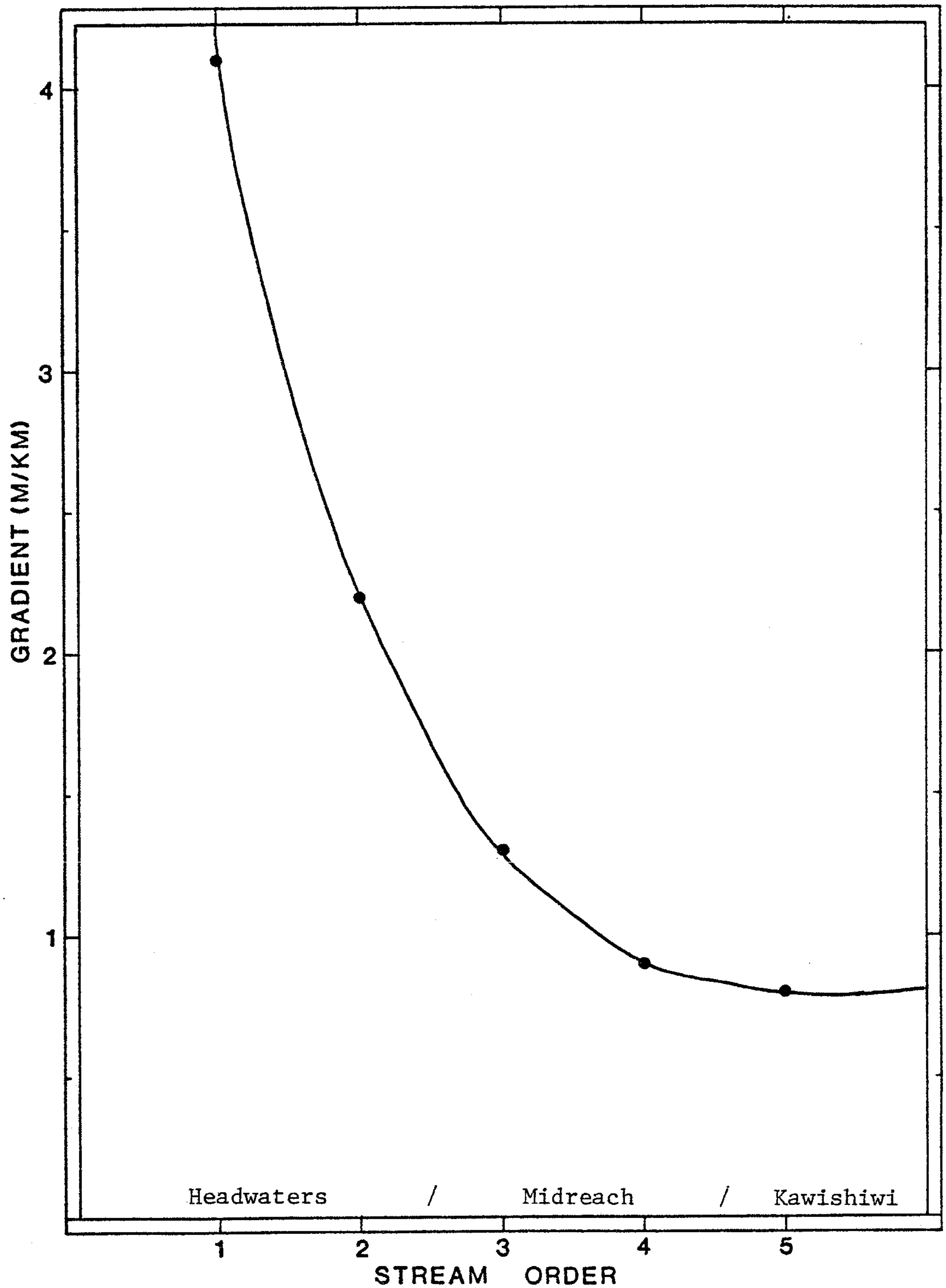


FIGURE 7 RELATIONSHIP OF GRADIENT TO STREAM ORDER

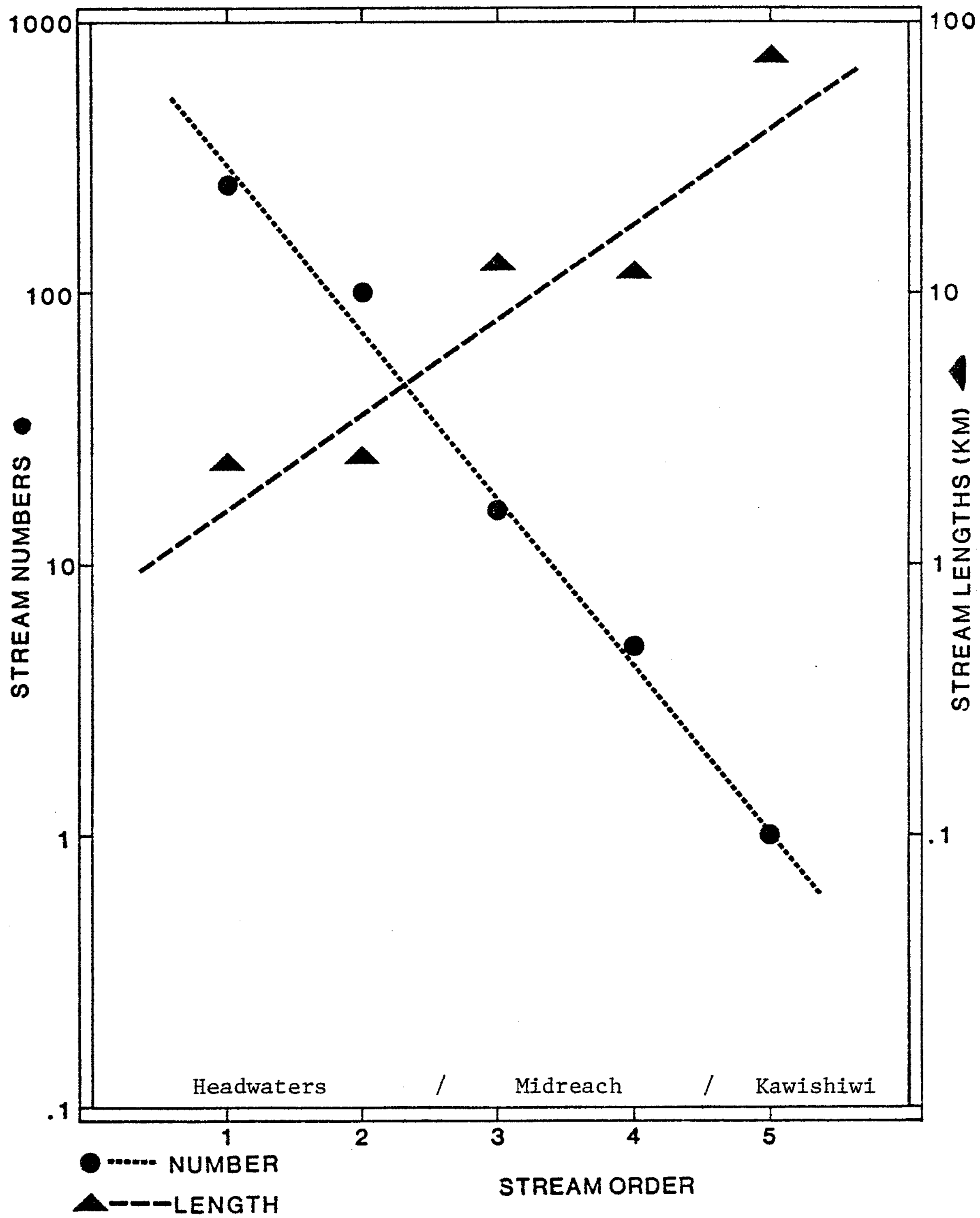


FIGURE 8 RELATIONSHIP OF NUMBER OF STREAMS AND LENGTH OF STREAMS TO ORDER

upland streams. Lowland streams have minimal canopy cover. Sedges, grasses, and scattered spruce, tamarack, and alder are the most common riparian vegetation along lowland streams.

Headwater streams are characterized by low primary production as measured by Chlorophyll a (Figure 9). Periphyton are responsible for most of the primary production as macrophytes are found only in scattered locations in headwater streams. Periphyton communities throughout the Study Area are dominated by the diatoms, which represent 87 percent of the periphyton cells (Johnson 1978a). The acidophilous diatoms such as Eunotia spp. and Tabellaria spp. comprise 20-25 percent of the diatom populations in headwater streams. This percentage drops to less than 15 percent in mid-reach streams and the Kawishiwi River (Figure 10).

Figures 9 and 10

Because most of the energy for consumer organisms is derived from the large amounts of allochthonous material present in headwater streams (Figure 11), the largest populations of dead plant shredders were found in headwater streams where they comprised from 12-22 percent of the invertebrate populations on an annual basis (Figure 12). The shredders were present throughout the year in headwater streams, although the highest relative abundance is from fall through early spring when the largest amounts of allochthonous material are present in the Study Area streams (Figure 13). Fall populations of shredders in headwater streams exceeded 45 percent of the invertebrates present. The relative abundance of shredders varies between the two types of headwater streams. Shredders are approximately 4-9 times more abundant in upland forest streams than in lowland bog streams (Lager et al. 1978)(Table 3).

Table 3, Figures 11, 12 and 13

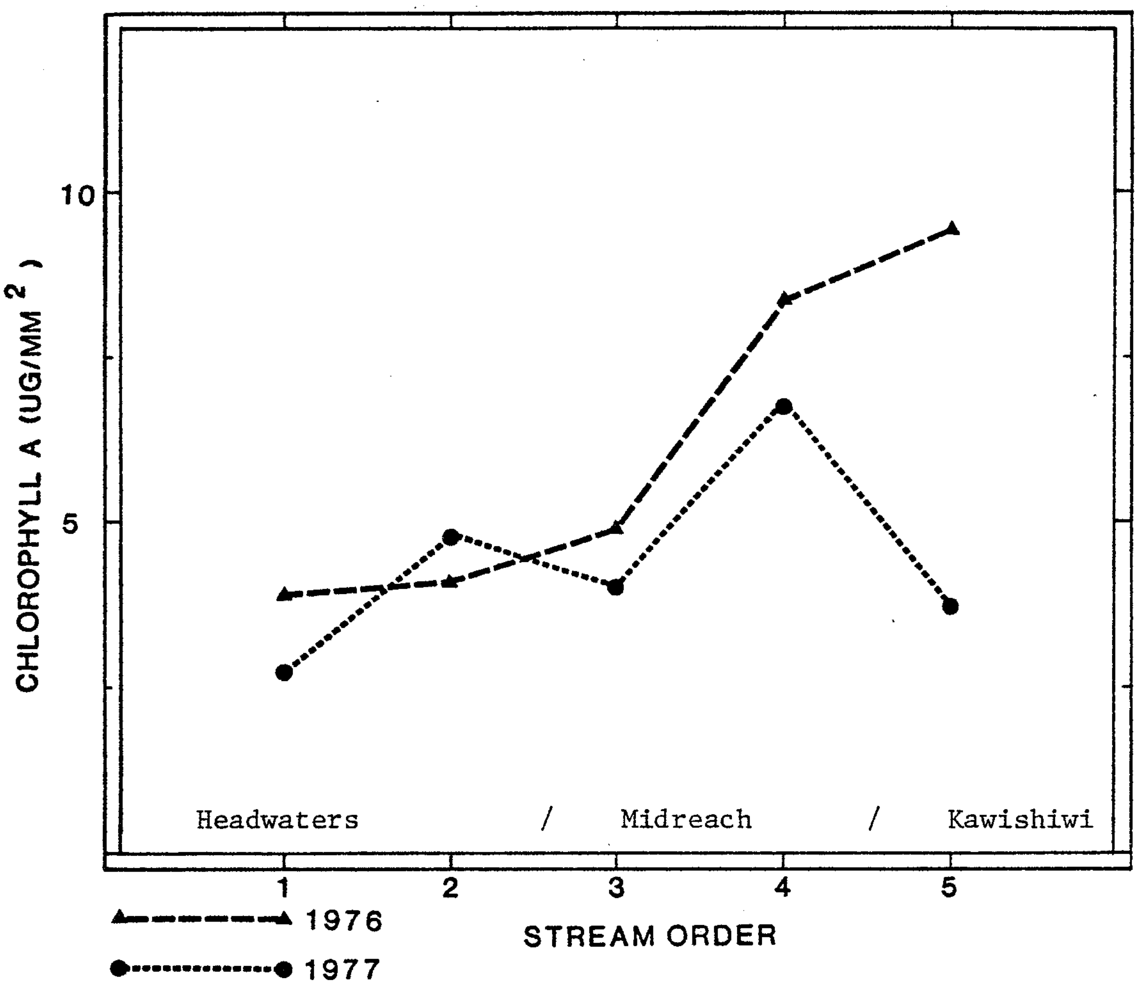


FIGURE 9 RELATIONSHIP BETWEEN MEASURED MEAN ANNUAL CHLOROPHYLL A AND STREAM ORDER.

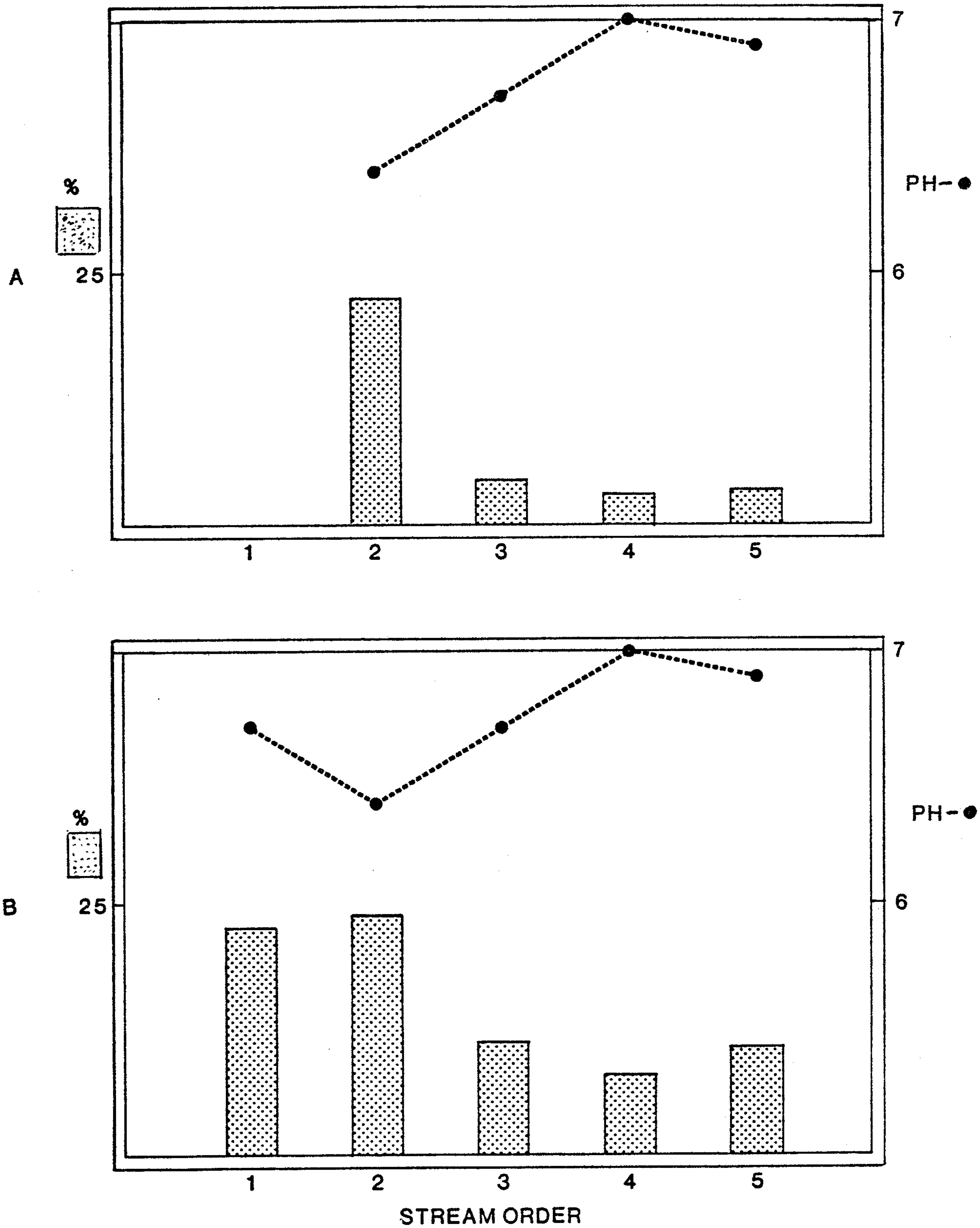


FIGURE 10 ACIDOPHILOUS DIATOM ABUNDANCE IN QUANTITATIVE (A) AND QUALITATIVE (B) SAMPLES VERSUS STREAM ORDER AND PH (PH VALUES ARE GEOMETRIC MEANS BY STREAM ORDER)

TABLE 3. Measured mean relative abundance* of shredders of dead plant material and collector-gatherers in shaded and open streams in the Study Area during 1977.

STREAM ORDER	SHREDDERS				COLLECTOR-GATHERERS			
	Spring		Fall		Spring		Fall	
	Shaded[a]	Open[b]	Shaded	Open	Shaded	Open	Shaded	Open
1	26.8	6.4	0.8	0	23.5	26.7	31.6	90.0
2	20.7	2.7	11.2	0.6	5.2	34.5	39.9	37.6
3	8.7	3.0	4.5	1.1	81.5	86.2	31.3	36.2
4	---	4.8	---	0.7	---	82.2	---	32.0
5	---	.6	---	0.2	---	40.9	---	15.3

[a]Canopy cover 25% to 100%.
[b]Canopy cover less than 25%.
*Relative abundance is defined as the percent of each group in relation to the total number of organisms.

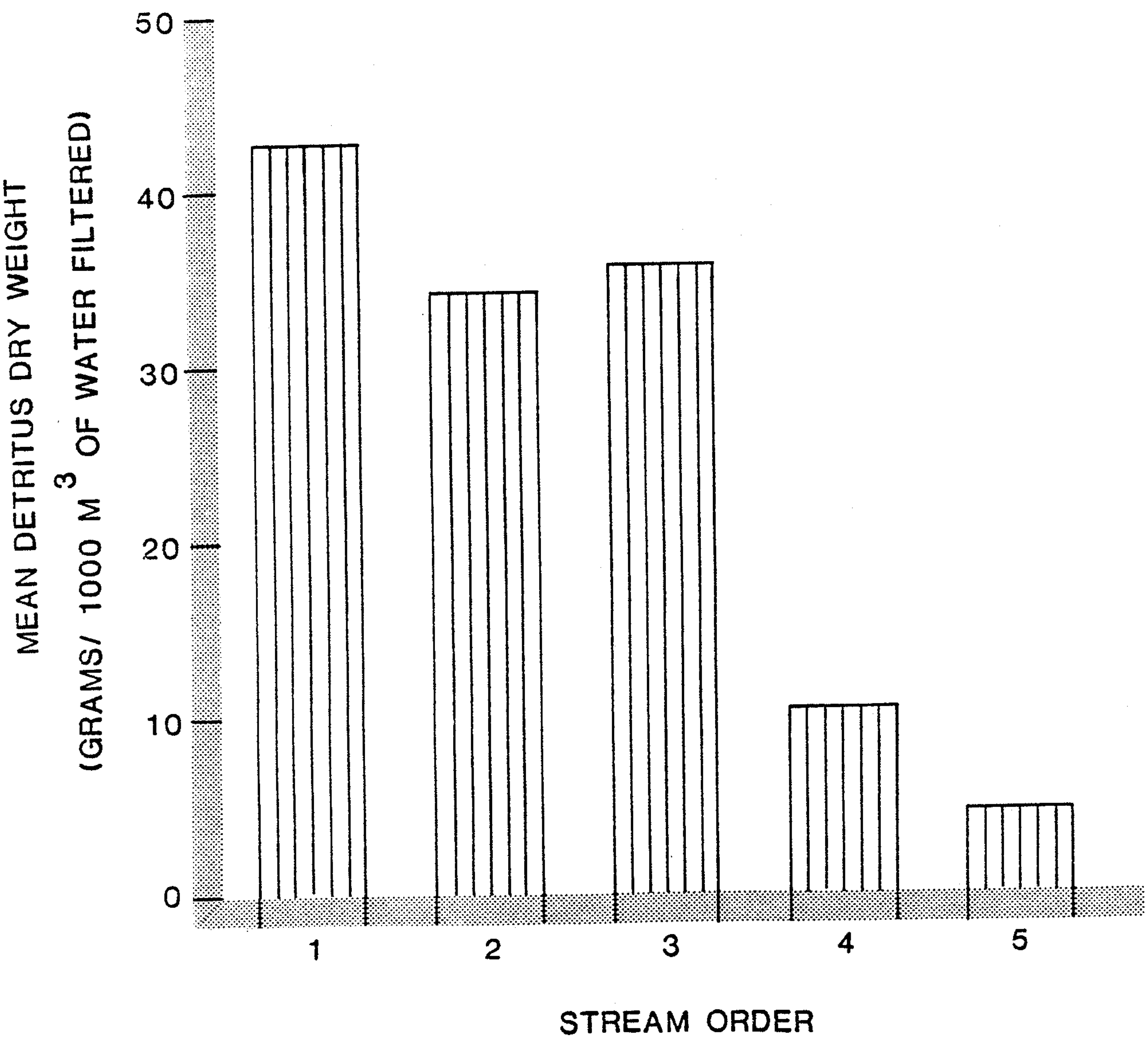


FIGURE 11 RELATIONSHIP OF MEAN DETRITUS DRY WEIGHT TO STREAM ORDER (MESH SIZE 0.44 MM)

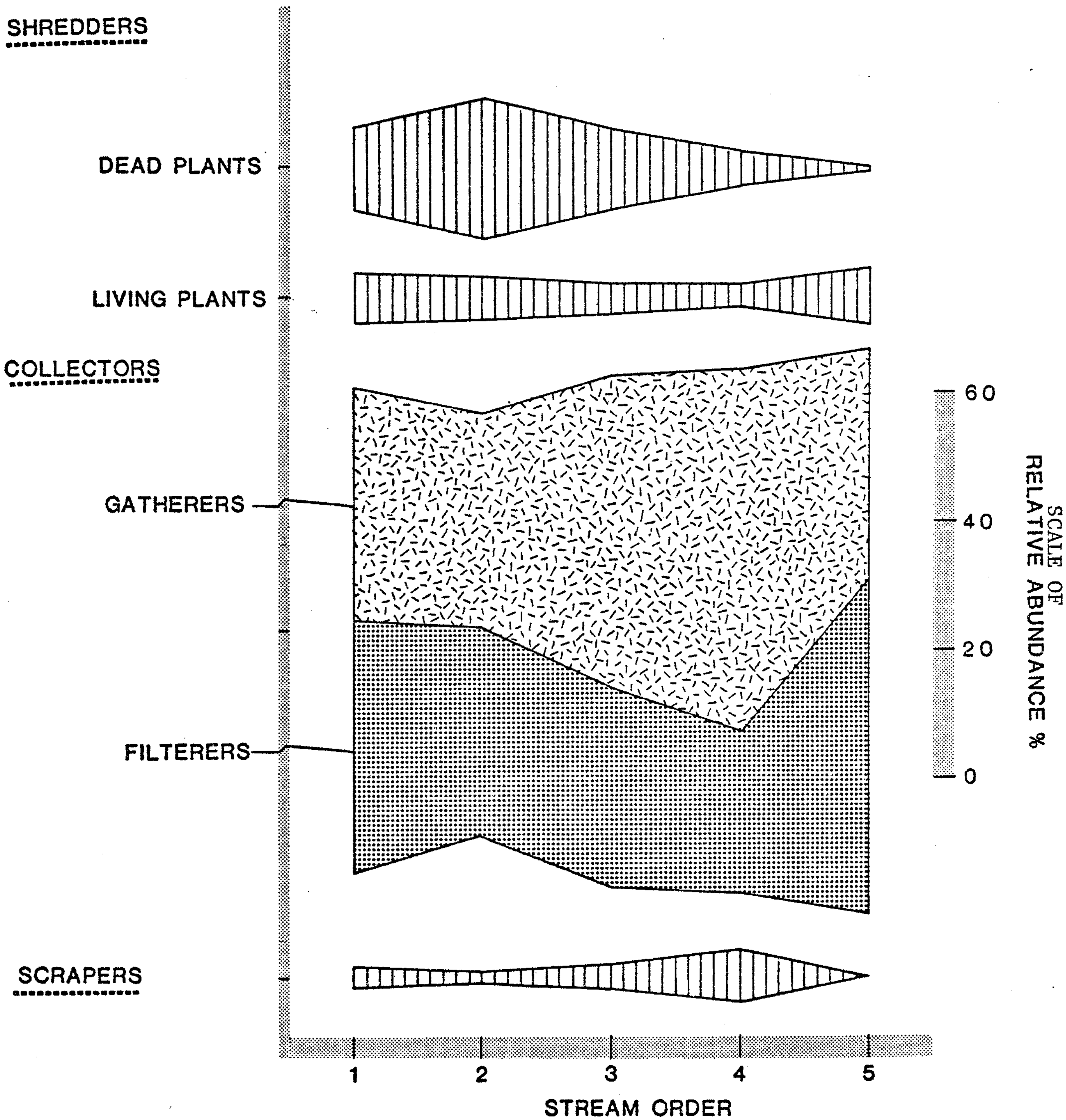


FIGURE 12 MEAN FUNCTIONAL GROUP ABUNDANCE IN VARIOUS STREAM ORDERS (1977 DATA)

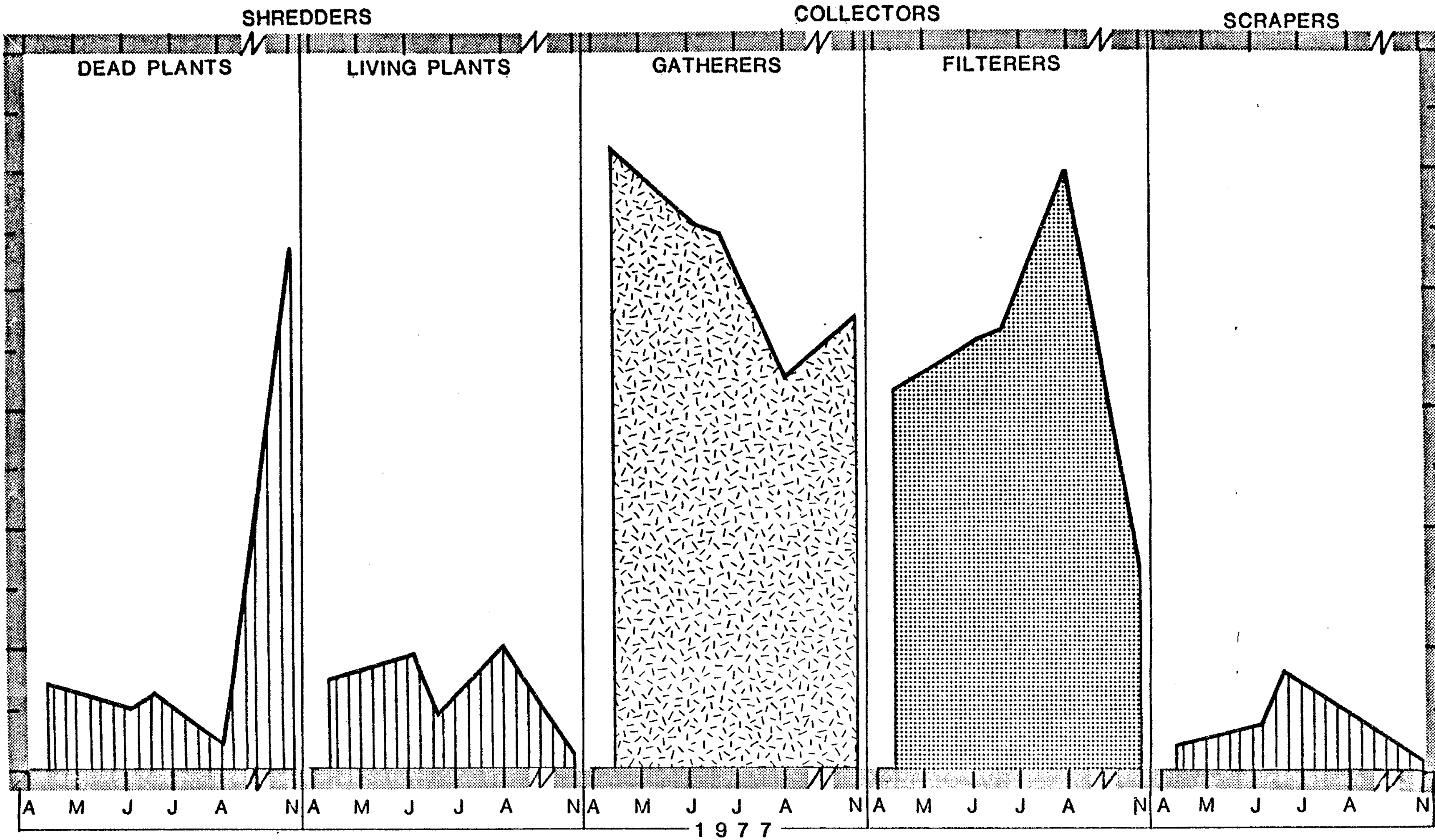


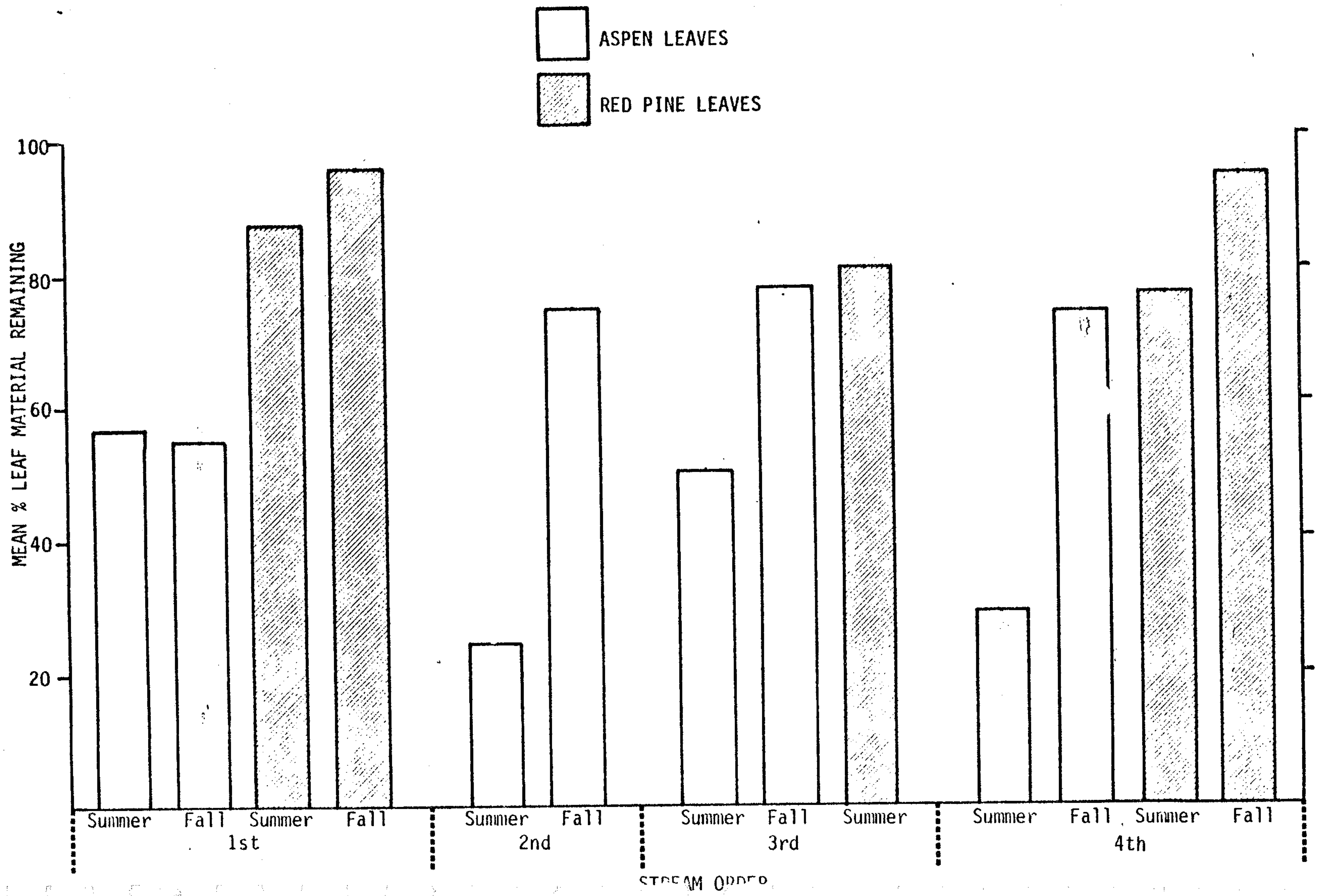
FIGURE [illegible]3. MEAN FUNCTIONAL GROUP RELATIVE ABUNDANCE (ALL 1977 STATIONS)

The primary shredders in headwater streams are stoneflies (Plecoptera) and caddisflies (Trichoptera)(Table 1). In mid-reach streams and the Kawishiwi River, chironomids (Diptera) became increasingly important members of the dead plant shredder group (Lager et al. 1978). These organisms feed on the allochthonous material and also contribute to the breakdown of this material for use by other invertebrates. A measure of the contribution and use of allochthonous energy sources can be obtained by comparing the processing (decomposition) rates for allochthonous placed in a stream as artificial leaf packs of known biomass. Although larger numbers of shredders were found in artificial leaf packs placed in headwater streams than in those placed in mid-reach streams, no difference in the processing rates of aspen or red pine leaves was detected in first through fourth order streams in the Study Area (Williams 1978)(Figure 14). Aspen leaves are processed significantly faster than red pine needles so that more energy is available to stream communities in aspen covered watersheds than in pine covered watersheds.

Figure 14

The collector group is the dominant group of invertebrates in headwater streams as it is in all Study Area streams (Lager et al. 1978). Collectors comprise more than 66 percent of the invertebrates on an annual basis in all stream orders. These organisms utilize the fine organic matter found in the headwater streams. Collector-gatherers are slightly more abundant in lowland streams where much of the organic material drifting out of bogs is finer than the allochthonous material found in upland streams (Table 3). Low populations of scraper invertebrates (5%) are found in headwater streams because of the low periphyton production in these streams.

FIGURE 14 MEAN % OF LEAF MATERIAL REMAINING AFTER 8 WEEKS EXPOSURE FOR ASPEN AND RED PINE LEAVES



The predominant fish communities found in headwater streams are composed of trout and minnows (Williams et al. 1978)(Figure 15). Brook trout (Salvelinus fontinalis) inhabit a small number of first and second order streams where maximum summer temperatures do not exceed 20°C. Thirteen streams in the Study Area are designated trout streams and are generally managed as such by the Minnesota Department of Natural Resources (MDNR)(Figure 6). The Regional Study's survey showed that trout are rarely found in mid-reach streams in the Study Area. Other fish species which are abundant in headwater areas are the central mudminnow (Umbra limi), blacknose dace (Rhinichthys atratulus), finescale dace (Chrosomus neogaeus), and brook stickleback (Culaea inconstans). A number of headwater streams have been classified by MDNR as northern pike streams or cosmopolitan streams. Because of the size of these streams and their susceptibility to drought it is doubtful that many headwater streams sustain populations of these types of fish. Many headwater streams do serve as spring spawning habitat for northern pike (Esox lucius), white suckers (Catostomus commersoni), and the small indigenous headwater species. The relative abundance of the fish species in headwater streams is presented in Table 4.

Table 4, Figure 15

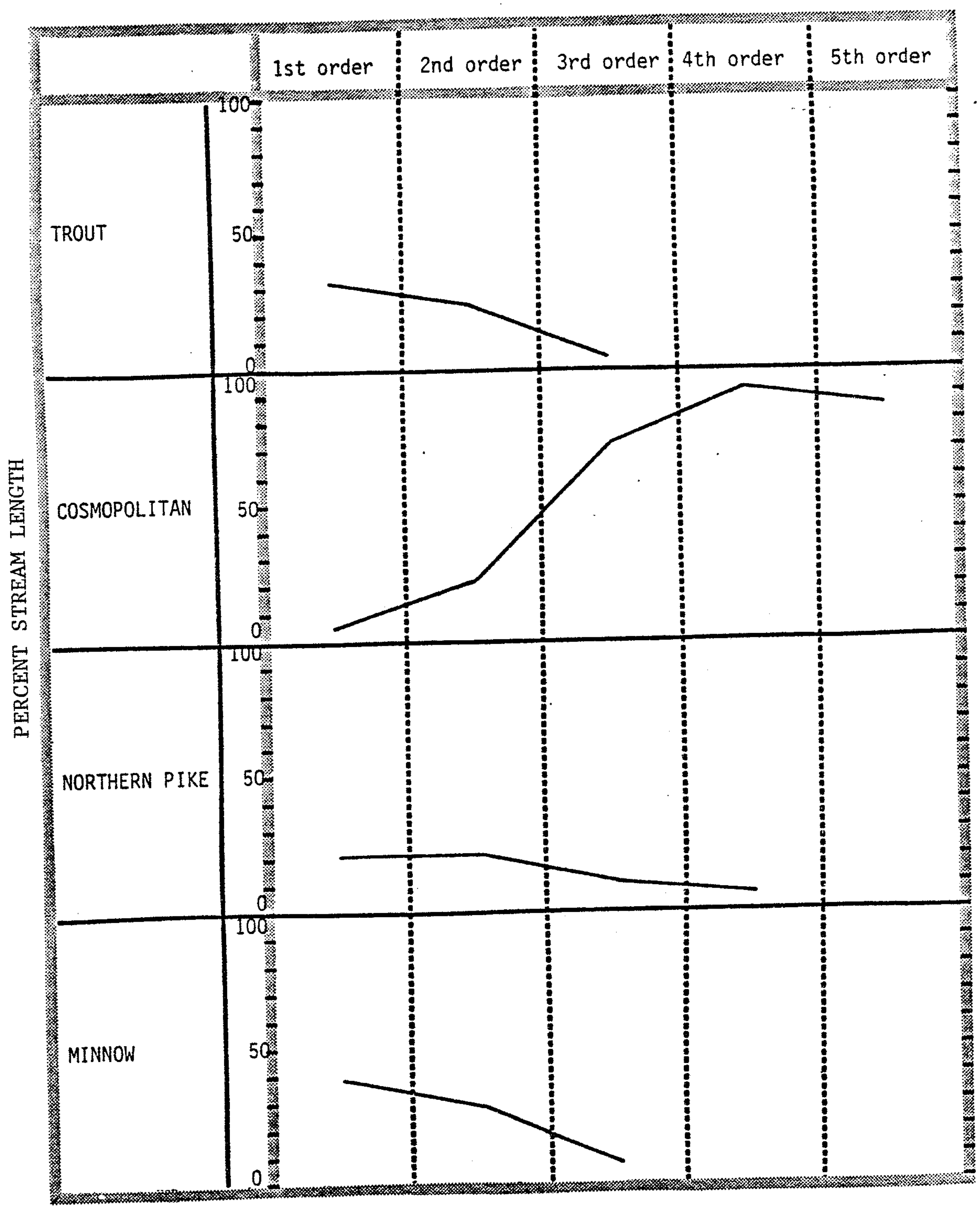
Headwater streams are subjected to large daily, seasonal, and yearly fluctuations in environmental conditions such as flow, pH, and temperature (see Volume 3-Chapter 4). This occurs because of the rapid response of the watersheds of these streams to snowmelt and storms, and because of the lack of dilution water available to temper these fluctuations. For long-term survival in Study Area headwater streams aquatic species must be adapted to these fluctuations and must be able to rapidly recolonize after natural events such as the 1976 drought,

Table 4. Mean relative abundance*of fish species in stream orders one through four in the Study Area during 1977.

SPECIES	STREAM ORDER 1	2	3	4
Brook trout	8.28	2.45	.11	0
Central mudminnow	14.41	2.37	.79	0
Northern pike	1.48	.65	2.50	3.51
Blacknose dace	1.63	16.91	8.	.91
Longnose dace	0	5.20	7.72	1.05
Creel chub	2.07	4.59	4.83	.05
Pearl dace	6.06	9.95	2.01	.18
No. redbelly dace	5.54	1.38	.71	0
Finescale dace	14.71	6.27	2.14	0
Golden shiner	0	.34	1.24	.73
Bluntnose minnow	0	.08	0	0
Fathead minnow	.44	4.06	1.48	.09
Emerald shiner	0	0	0	3.42
Common shiner	1.11	11.02	7.27	7.25
Spottail shiner	0	0	0	.09
Blacknose shiner	1.48	2.30	2.44	22.21
Mimic shiner	0	1.72	0	0
Brassy minnow	.07	.23	.02	0
Shorthead redhorse	0	0	.38	.96
White sucker	4.58	14.38	19.38	7.94
Channel catfish	0	0	.34	0
Brown bullhead	0	0	.15	.27
Black bullhead	0	0	5.97	.18
Yellow bullhead	0	0	.11	.05
Tadpole madtom	0	.04	1.39	1.14
Trout-perch	0	.08	1.24	1.73
Burbot	.30	1.64	2.82	6.89
Brook stickleback	32.30	3.29	2.54	0
Largemouth bass	0	.54	.36	1.78
Smallmouth bass	0	0	0	.09
Rock bass	1.26	.08	1.35	7.62
Bluegill	0	0	.06	.46
Pumpkinseed	0	0	.06	0
Black crappie	0	0	0	.59
Yellow perch	0	.57	10.18	19.07
Walleye	.07	.15	.32	1.60
Log perch	0	.84	1.48	3.24
Johnny darter	0	5.93	3.78	2.69
Iowa darter	.44	.27	.26	0
Mottled sculpin	3.25	2.49	1.88	.73
Unidentified	.52	.19	4.77	3.47
Number of Taxa	19	29	34	29

* Relative abundance is defined as the percentage of the population represented by a given species.

FIGURE 15 RELATIONSHIP BETWEEN PERCENT STREAM LENGTH* OF MAJOR MDNR STREAM FISHERIES CLASSIFICATIONS AND STREAM ORDER



* Percent of total stream length in a given stream order which is represented by each of the various fisheries classifications.

which was nearly a one hundred year drought (Johnson and McCullough 1978)(see Volume 3-Chapter 4). Sampling in 1977 seemed to indicate that stream communities adversely affected by the drought had recovered (see Volume 3-Chapter 4 for further details on low flow conditions).

In mid-reach streams (3rd and 4th order) the forest canopy opens to between 10 and 50 percent cover. As a result of decreasing shading of the streams, light increases and periphyton production increases (Figure 9). Periphyton production as measured by chlorophyll *a* is highest in late spring and late summer (Figure 16) when temperature and light are optimum for diatoms. In mid-reach streams some heavy growths of macrophytes can be found but these areas are isolated (Johnson 1978b). Where these beds occur macrophytes make a contribution to the autochthonous energy pool in the stream. Possibly more significant is the use of macrophytes as a substrate by aquatic insects, spawning fish, and as cover by young fish.

Figure 16

The composition of the diatom community changes between the headwater and mid-reach sections (Johnson 1978). Acidophilous diatom species are still present but only in small numbers (approximately 15%). Alkaliphilous and indifferent species such as *Achnanthes linearis*, *A. minutissima*, *Cocconeis placentula* var. *lineata* and *Synedra* spp. are the dominant species in these streams. *A. minutissima*, the most abundant single diatom species observed in Study Area streams of any order, comprised up to 76 percent of the diatom population in a given stream on an annual basis. The average relative abundance of *A. minutissima* observed in mid-reach streams was 45 percent.

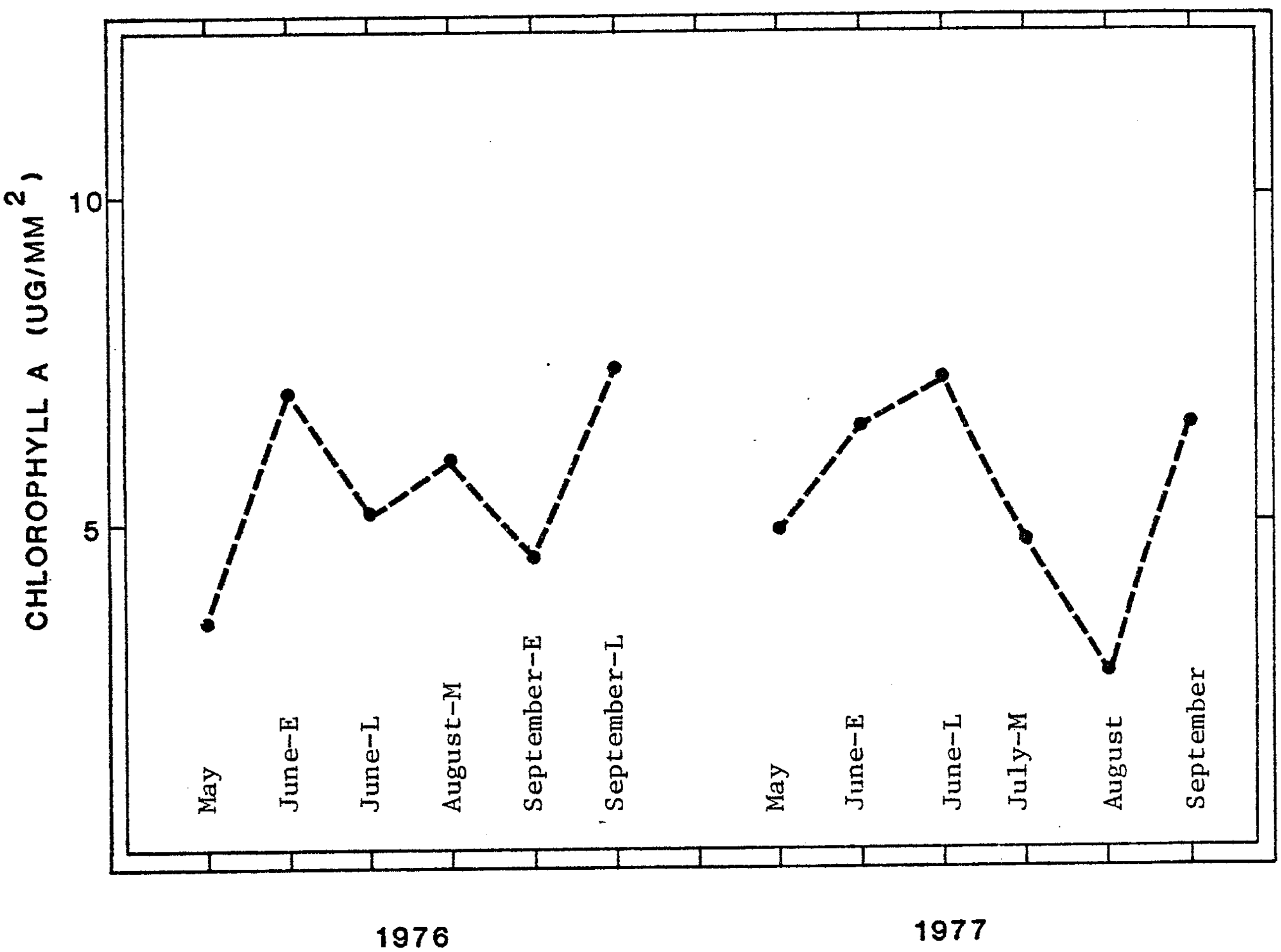


FIGURE 16 SEASONAL VARIATION OF MEAN CHLOROPHYLL A AT ALL STATIONS: 1976 and 1977

E-EARLY M-MIDDLE L-LATE

The proportion of dead plant shredders declines, whereas the proportion of collector and scraper groups increases in mid-reach streams (Lager et al. 1978). The dominant taxa in these functional groups are listed in Table 1. There are few changes in dominant taxa in each functional group between headwater and mid-reach streams. Changes in the relative abundance of the dead plant shredder, collector, and scraper functional groups are related to the decrease in allochthonous inputs and the increase in periphyton production. The increased size of the collector group may indicate that fine particulate organic matter is present in greater quantities in these areas than upstream. The collector group is present in large numbers throughout the year and in all types of streams (Figures 12 and 13), although the dominant taxa change seasonally. Various mayfly (Ephemeroptera) species such as *Paraleptophlebia*, *Baetis* and *Ephemerella* dominate the collector-gatherer group in all stream orders. A major shift in dominant taxa occurs in the filter-feeding portion of this group; in headwater areas Simuliidae (Diptera) are dominant, while in mid-reaches *Hydropsyche* spp. (Trichoptera) is a codominant with Simuliidae. In the Kawishiwi River *Hydropsyche* spp. becomes the only dominant filter-feeder.

Most mid-reach streams support a varied population of fish. The Study's survey indicated that more species and larger individuals are found in mid-reach streams than in headwaters. Predatory species such as walleye (*Stizostedion vitreum*), northern pike (*Esox lucius*), and yellow perch (*Perca flavescens*) commonly occur in mid-reach streams and reach a larger size there than in headwater areas. Brook trout are rarely found in mid-reach streams except at the mouth of the Dunka River, in Snake Creek, and in the Snake River. The relative abundance of many of the forage fish listed in Table 4 is probably an underestimate for mid-reach streams because the collection equipment used in these streams did not adequately sample small species (see Williams et al. 1978).

Mid-reach streams consist of Long, flat sections connected by short riffles. These flat sections store enough water so that during drought periods the entire stream does not dry up. These areas provide havens for aquatic organisms, particularly fish, during adverse conditions such as the drought of 1976. There are also fewer and less severe environmental fluctuations in these streams than in headwater streams because the larger volume of water normally in the streams tempers daily, seasonal, and yearly fluctuations in water chemistry and physical parameters.

The Kawishiwi River is the only fifth order stream in the Study Area except for a small section of the Isabella River (Figure 2). The morphology of the Kawishiwi River does not fit into the stream order continuum for the Study Area because it consists of a series of river-lakes connected by short riffles.

Periphyton productivity of the riffles was higher than in mid-reach streams in 1976 but appeared lower in 1977 probably as a result of sampling difficulties when because high flows scoured the glass slide samplers (Johnson 1978a)(Figure 9). In the Kawishiwi periphyton production is probably less important than phytoplankton production because of the limited amount of riffle area. It appears that the slightly lower pH (pH=6.9) in the Kawishiwi than in mid-reach streams (pH=7.1), may result in greater abundance of acidophilous (acid "loving") diatoms such as *Tabellaria flocculosa*. Other diatoms dominant in the Kawishiwi River included *Gomphonema* spp., *Achnanthes linearis*, and *Cocconeis placentula* var. *lineata*.

Drifting particulate organic matter was low in the Kawishiwi River and as a result the shredder populations were low (Figures 11 and 12). The expected high relative abundance of the scraper group was not observed perhaps because of the interspersion of lakes among the river segments. The relative abundance of scrapers was lower in Kawishiwi River riffles (less than 1%) than in mid-reach streams (6%). The dominant invertebrate group was the collector group. The filter-feeding portion of the collector group reached its maximum abundance (52%) in the Kawishiwi. Hydropsyche spp. was the dominant organism in this group and its abundance was the most distinctive feature of the Kawishiwi River riffles. These filter-feeding caddisflies comprised approximately 50 percent of the invertebrates in Kawishiwi riffles because of the large amounts of plankton flowing out of the lake portions of the Kawishiwi River.

In the lake portions of the Kawishiwi River, observed populations of phytoplankton, zooplankton, and benthic invertebrates were similar to those found in Gabbro, Birch, and White Iron lakes which are drained by the Kawishiwi River. The fish populations of the Kawishiwi River also resemble the fish populations of Gabbro, Birch, and White Iron lakes more than they resemble the fish populations of other streams in the Study Area. The number and size of northern pike, walleye, white sucker, and yellow perch was greater in the Kawishiwi River than in other streams, but slightly less than in Study Area lakes (Table 5). The cisco (Coregonus artedi) and whitefish (Coregonus clupeaformis) are two major fish species found in the Kawishiwi River but not in other streams. Table 6 lists the fish species of the Kawishiwi River.

Tables 5 and 6

1.4.1.3 Streams Currently Affected by Mining--Several streams in the Study Area are currently affected by taconite mining or copper-nickel exploration. Current taconite operations are pumping mine water into the upper Partridge River, First and Stephens creeks (Partridge tributaries), Dunka River, and Unnamed Creek

Table 5. Gill net catch indices for the South Kawishiwi River, Study Area rivers and Study Area lakes.

	SOUTH KAWISHIWI RIVER[a]	STUDY AREA RIVERS[b]	STUDY AREA LAKES[c]
Walleye			
Mean number/net	3.73	.93	8.70
Mean weight/fish (kg)	.32	.21	.52
Northern Pike			
Mean number/net	3.20	6.80	3.45
Mean weight/fish (kg)	.60	.29	.81
White Sucker			
Mean number/net	4.77	7.90	4.75
Mean weight/fish (kg)	.45	.57	.76

[a]Data collected in 1976.
[b]Data collected in 1976 and 1977.
[c]Data collected in 1960-1977.

See Williams et al. 1978 for further details.

Table 6. Fish species collected in the South Kawishiwi River. Data from 1976 sampling and 1967 MDNR fisheries survey.

COMMON NAME	SCIENTIFIC NAME	STREAM SECTOR* AS	BS	CS	DS	ES	FS	GS	HS
Northern cisco	Coregonus artedii	X	X		X	X	X	X	X
Lake whitefish	Coregonis clupeaformis		X		X			X	X-76
Northern pike	Esox lucius	X	X	X	X	X	X	X	X
Golden shiner	Notemigonus crysoleucus		X-67				X-67		X-67
Common shiner	Notropis cornutus		X-67	X-67	X-67	X-67	X-67	X-67	X-67
Blackchin shiner	Notropis heterodon			X-67					
Blacknose shiner	Notropis heterlepis	X-67	X-67		X-67	X-67	X-67		
Spottail shiner	Notropis hudsonius	X-67	X-67			X-67		X-67	X-67
Bluntnose minnow	Pimephales notatus	X-67			X-67	X-67		X-67	X-67
White sucker	Catostomus commersoni	X	X	X	X	X	X	X	X
Tadpole madtom	Noturus gyrinus	X-67	X-67	X-67	X-67	X-67	X-67	X-67	
Burbot	Lota lota				X	X-67			
Bluegill	Lepomis macrochirus		X	X-67	X	X-67	X-67	X	X
Pumpkinseed	Lepomis gibbosus			X-67	X-67	X			X-76
Rock bass	Ambloplites rupestris	X	X	X	X	X	X	X	X
Largemouth bass	Micropterus salmoides	X-67						X-67	X-67
Black crappie	Pomoxis nitromaculatus	X	X	X-67	X-67	X-67	X-67	X	X
Iowa darter	Etheostoma exile					X-67		X-67	X-67
Johnny darter	Etheostoma nigrum	X-67	X-67	X-67	X-67	X-67	X-67	X-67	X-67
Logperch	Percina caprodes	X-67	X-67		X-67			X-67	X-67
Yellow perch	Perca flavescens	X	X	X	X	X	X	X	X
Walleye	Stizostedum vitreum	X	X	X	X	X	X	X	X
Slimy sculpin	Cottus cognatus	X-67		X-67		X-67	X-67		

[a]X=fishes captured 1967 and 1976
X-67=fishes taken in 1967 (gill netting, trap netting, seining)
X-76=fishes captured in 1976 only (gill netting, trap netting)
* Stream sector names are presented in Williams et al. 1978. Stations AS through HS represent a downstream progression.

adjacent to the Dunka Pit (Figure 17). Other mining discharges occur in the lower Partridge below Colby Lake, the St. Louis River below the confluence with the Partridge River, and in the Embarrass River. The resulting chemical changes in these streams include increased pH (.5 units), conductivity (200 umhos), and alkalinity (50 mg/l $CaCO_3$) with concomitant physical changes including increased suspended solids and altered flow regimes (see Volume 3-Chapter 4 for further details).

Figure 17

No significant differences in periphyton diversity, primary production, invertebrate diversity, or invertebrate functional group composition, that could be attributed to mining activities, were observed between affected and unaffected sites within the Study Area except in Unnamed Creek (Table 7 and Figures 18 and 19). It would appear that the amount of change in water quality parameters is not large enough to cause a measurable change in biological communities. There was some evidence that higher total invertebrate populations were present at the affected sites (Figure 18). There were also some shifts in dominant periphyton species (Figure 19). In affected headwater streams, acidophilous species such as *Tabellaria flocculosa* were less abundant than in unaffected headwater streams. There was also a shift to greater dominance by *Achnanthes minutissima* at affected headwater sites. In mid-reach streams the shift was less dramatic as *A. linearis* was reduced in abundance and replaced by *A. minutissima*. The significance of a shift in relative abundance of species when overall productivity does not change is not known.

Table 7, Figures 18 and 19

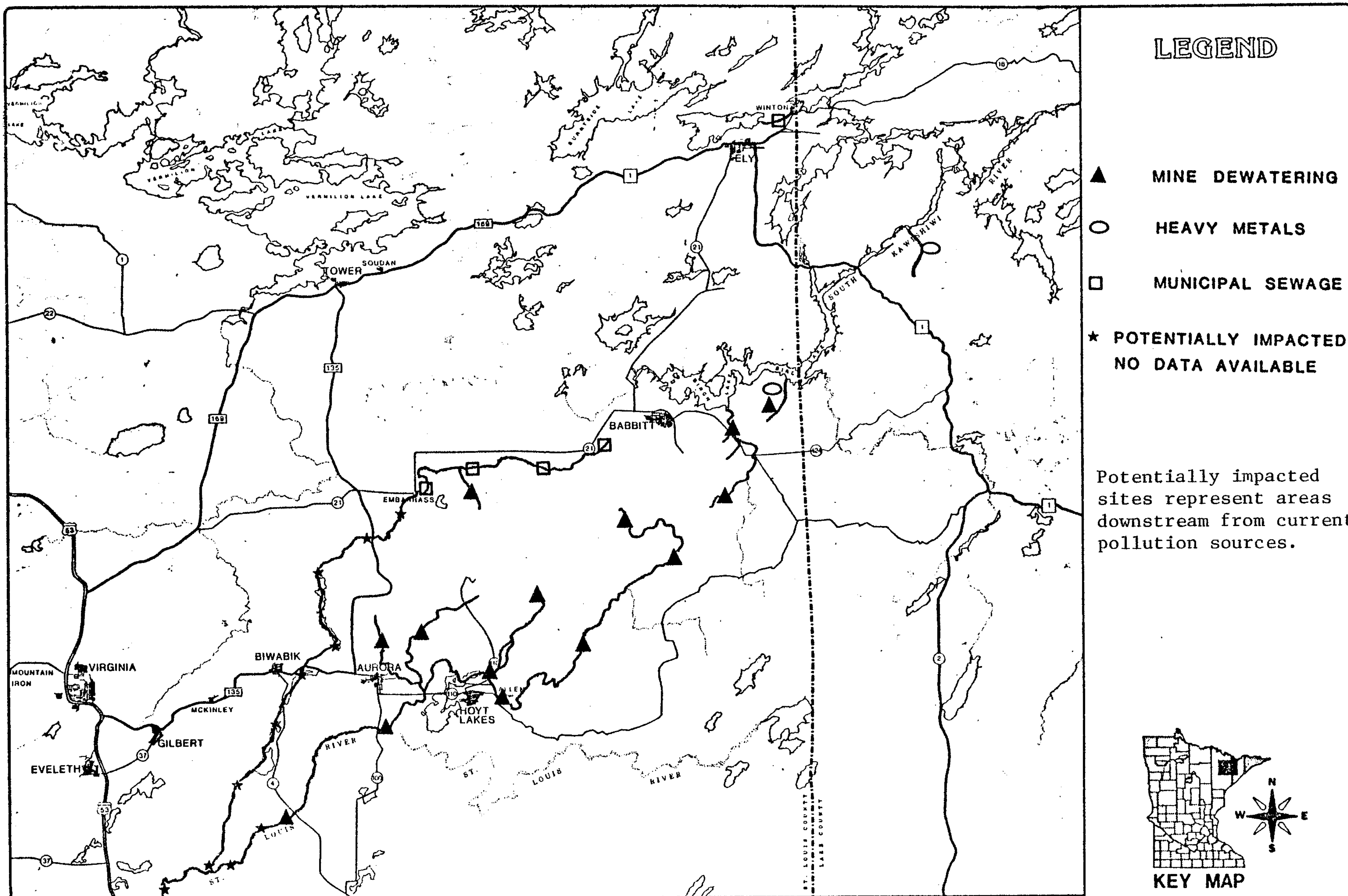


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FIGURE 17 POINTS AT WHICH STREAM CURRENTLY RECEIVE IMPACTED WATER

Table 7. Comparison of biological parameters at stations receiving taconite mine dewatering and unaffected stations.

PARAMETER	STATIONS Affected[a]	Unaffected[b]
Mean periphyton cells/mm^2	1,673	1,551
Mean periphyton chlorophyll *a* (ug/mm^2)	6.42	5.48
Mean periphyton diversity (Shannon-Weiner)	2.27	3.09
Mean number of periphyton taxa per sampling date	23	24
Mean number of drifting invertebrates	622	657
Mean invertebrate diversity	3.44	3.30
Mean number of drifting invertebrate taxa per sampling date	26	23

[a]Stations SL-1, P-1, D-1, P-5, BB-1.
[b]Stations SR-1, BI-1, SL-3, KC-1, F-1.

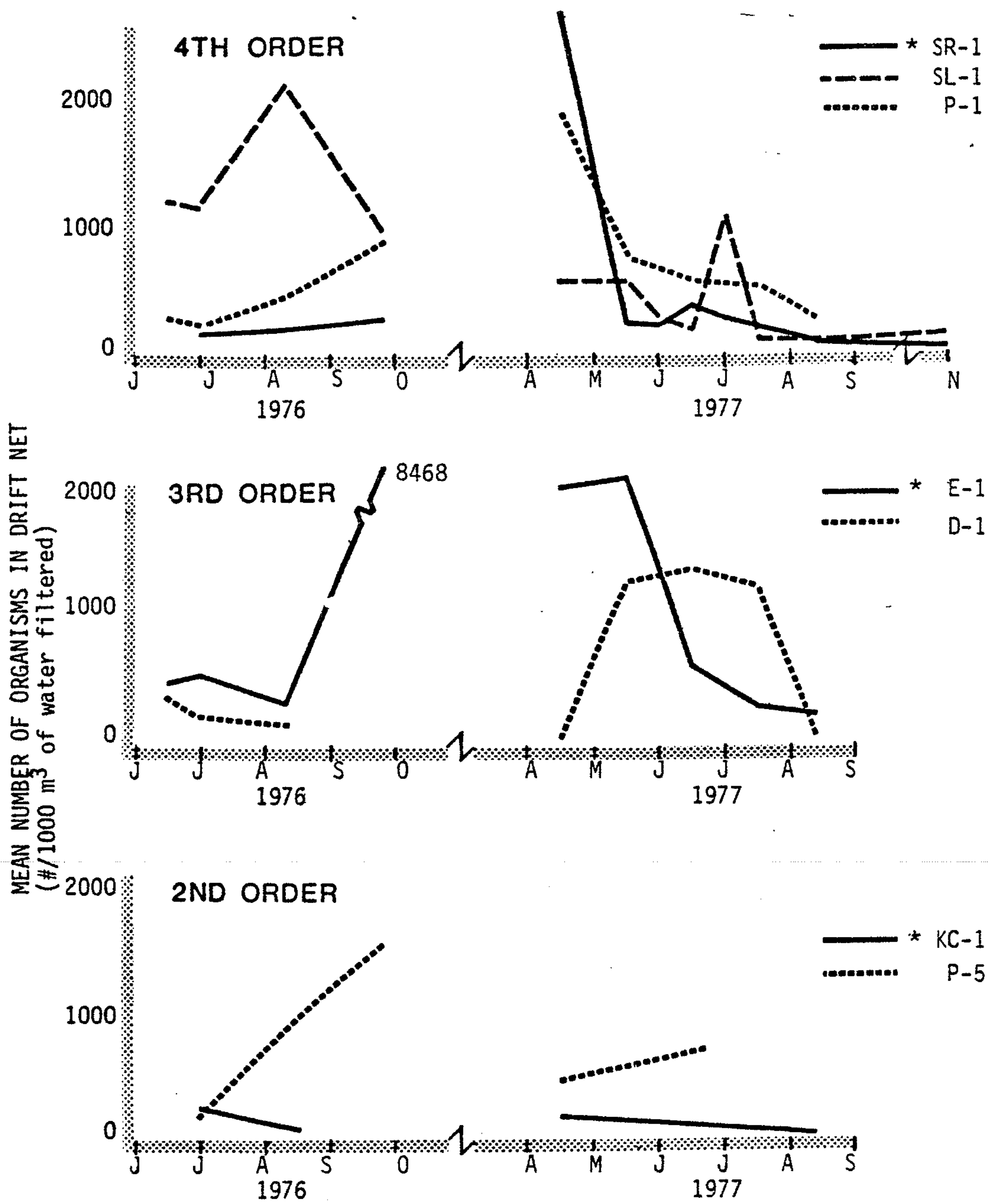

FIGURE 18 COMPARISON OF TOTAL NUMBER OF INVERTEBRATES DRIFTING AT STATIONS AFFECTED BY TACONITE MINE DEWATERING AND BACKGROUND STATIONS OF EQUAL STREAM ORDER

*BACKGROUND STATIONS

** See Figure 4 for station locations.

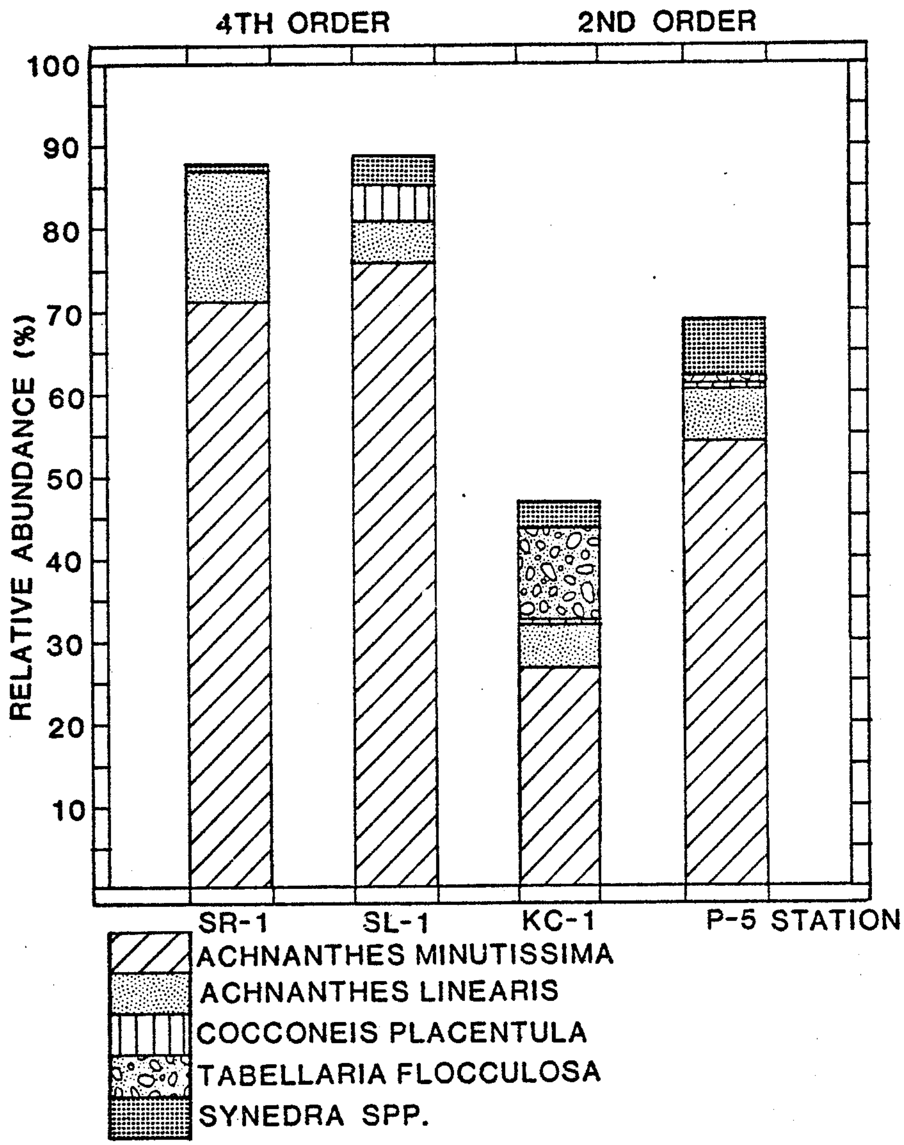

FIGURE 19 DOMINANT DIATOM TAXA AT UNIMPACTED (KC-1) AND IMPACTED (P-5) SECOND AND UNIMPACTED (SR-1) AND IMPACTED (SL-1) FOURTH ORDER STREAMS

In Unnamed Creek, a first order stream which receives leachate from stockpiled Duluth Gabbro and mine waste rock piles, elevated levels of copper (4 ug/l) and nickel (125 ug/l) are present in addition to mine water (see Chapter 3-Volume 4). It is not possible to distinguish the effects of these metals from the effects of the poor natural substrate and the irregular pumping of mine water, which causes fluctuating rates of stream flow. Figure 20 shows the hydrograph of Unnamed Creek for 1976. Primary productivity in Unnamed Creek was similar to other unaffected headwater streams. _A. minutissima_ and _Diatoma tenue_ var. _elongatum_ were the dominant diatom species as in other impacted streams. Acidophilous diatoms were reduced in abundance when compared to other headwater areas. Periphyton diversity was lower in Unnamed Creek (3.1) than in Keeley Creek (3.7), which is a similar headwater stream. However, the number of species was similar (25 in Unnamed Creek and 24 in Keeley Creek in September, 1976)(Johnson and Williams 1978).

Figure 20

The relative abundance of invertebrate functional groups was similar in Unnamed Creek and other headwater streams, but invertebrate standing crop and diversity were lower in Unnamed Creek when compared to other headwater streams. This reduction is more likely to be the result of the fluctuating flow rates rather than of elevated metal levels. Fluctuating flow rates have been shown to have devastating effects on stream invertebrate populations. The fluctuating flows observed in Unnamed Creek are also probably responsible for the shifting of substrate materials which may have further impact on the invertebrate communities (Johnson and Williams 1978).

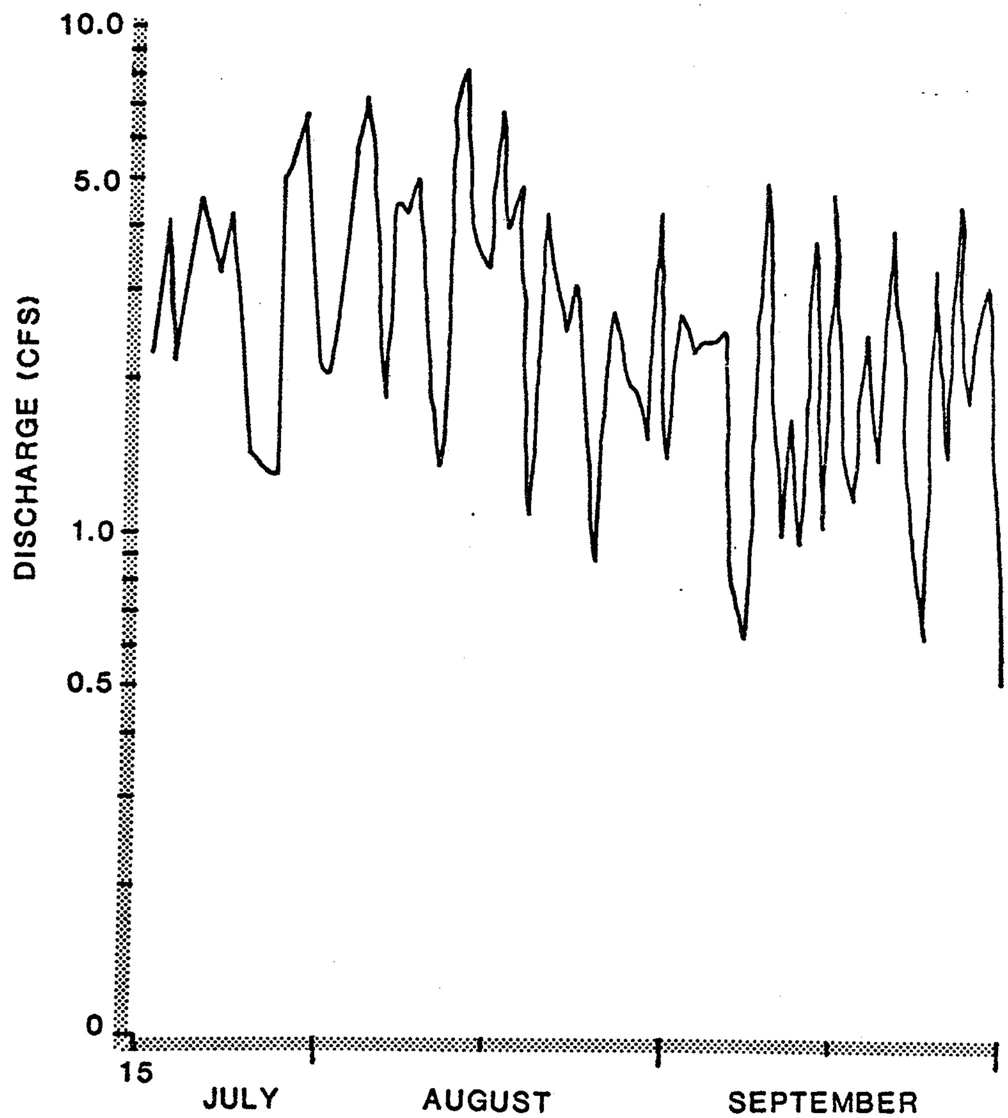

FIGURE 20 MEAN DAILY FLOW AT EM-1 ON UNNAMED CREEK 1976

**See Volume 3-Chapter 4 of this report for further details.

Elevated levels of copper have been found in Filson Creek, which may result from copper-nickel exploration and/or from the close proximity of the Duluth Gabbro Contact and out-cropping of mineralized gabbro in the area (see Volume 3-Chapter 4 for further details). No difference in any biological parameters could be detected when Filson Creek was compared to Keeley Creek, a similar headwater stream with low heavy metal levels (Table 8).

Table 8

Stephans and First creeks currently receive mine dewatering from taconite operations. According to MDNR stream surveys in 1968 the habitatt in both streams has been degraded by siltation. The probable source of the silt is either the taconite mine water and/or surface runoff and erosion from the mine and waste rock piles (MDNR 1968). Stephans Creek was a designated trout stream and was stocked with brook trout until 1951. The management classification of this stream was changed to Warm-water Feeder following the 1968 survey. Trout stocking in First Creek was discontinued in 1977 although it retains a trout management classification. The bottom substrates of these two streams are covered with silt. Because trout require clean gravel for spawning it is unlikely that naturally reproducing populations will again become established in these streams.

1.4.2 Lake Ecosystems

The physical, chemical, and biological characteristics of lakes are different from streams. The cycling of nutrients is one of the more important characteristics of lakes because the input of "new" materials is relatively low compared to that in streams. Lake organisms are dependent on the cycling of

Table 8. Comparison of biological parameters measured at F-1 (impacted) and KC-1 (unimpacted); data from 1976 sampling season.

PARAMETER	F-1	KC-1
Mean number of periphyton cells/mm^2	380	385
Mean periphyton chlorophyll a (ug/mm^2)	4.21	2.70
Mean periphyton diversity (Shannon-Weiner)	3.62	3.73
Mean number of periphyton taxa per sampling date	30	30
Mean number of drifting invertebrates	980	165
Mean invertebrate diversity	3.06	2.91
Mean number of drifting invertebrate taxa per sampling date	19	12

** See first level reports for further details.

nutrients from water and sediments and often may affect the quality of water in a lake. Lake communities can be significantly affected by stratification but are generally not subject to the extreme variation in water flow rates experienced by stream organisms.

Lakes are influenced by the morphometry of their basins and by the geology and morphometry of the surrounding watershed. Nutrients are supplied to the lakes by the watershed through weathering and atmospheric deposition. The inflows and outflows of the lakes and the flushing time for the basin influence the distribution and availability of nutrients and dissolved gases, and the distribution of organisms. The shape of the basin influences patterns of heating and thermal stratification, thereby affecting transport within the lake (see Volume 3-Chapter 4 for further details).

In a lake, nutrients (e.g. carbon, phosphorus, nitrogen) and minerals (e.g. iron, phosphorus, and silica) are cycled internally in complex patterns involving many chemical and biological transformations. The rate at which these substances enter and leave the cycles may dramatically affect the biological communities of a lake.

Lake zones are described in terms of the distribution of primary producers (Figure 21). The area along the shore where sufficient light penetrates to allow macrophyte and periphyton growth along the bottom is called the littoral zone. The profundal zone is the area where insufficient light penetrates for benthic primary producers. The limnetic zone is the open water area above the profundal zone where most primary production occurs.

<u>Figure 21</u>

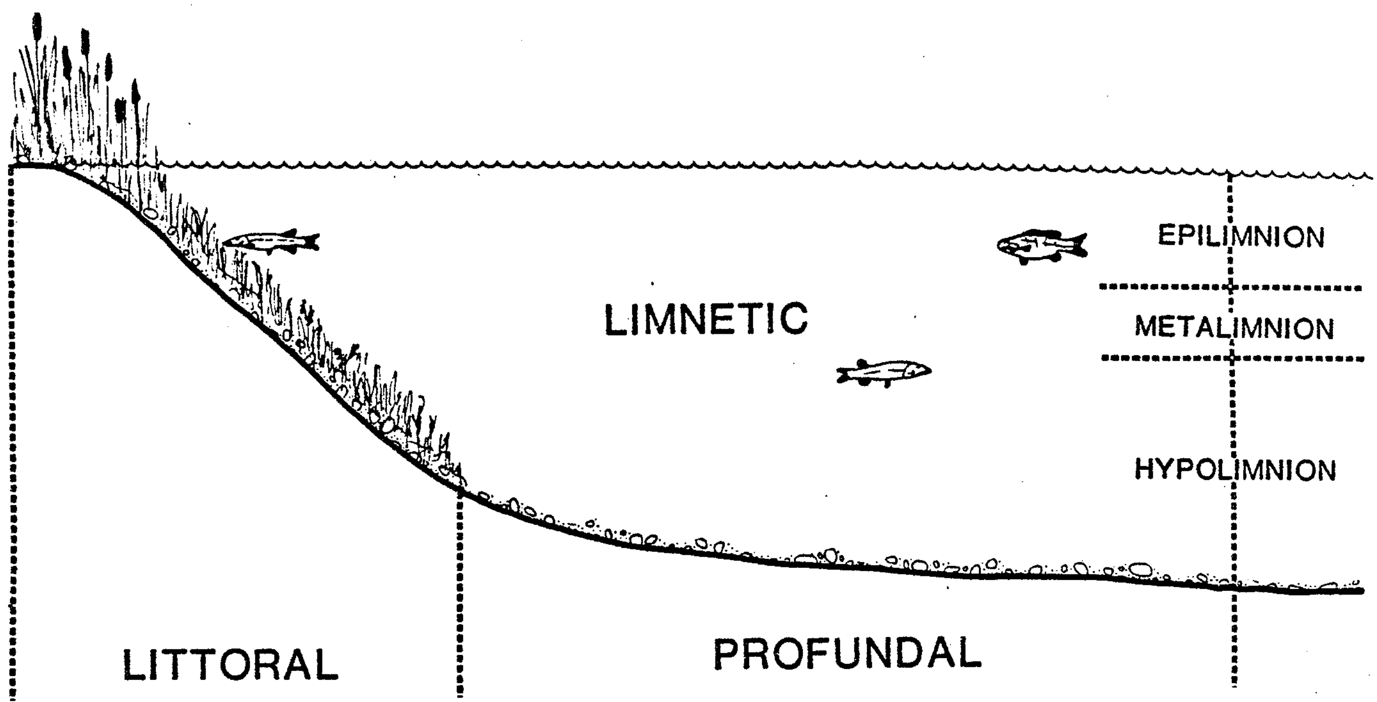

FIGURE 21 LAKE ZONES IN STRATIFIED LAKES

Deep lakes in the north temperate zone usually become stratified in summer, as surface waters warm. In these lakes differences in water density produce layers of warm circulating water (epilimnion), and colder less turbulent water (hypolimnion). The region between these layers, the metalimnion or thermocline, is the zone of rapid temperature decrease with increased depth. During periods of stratification no exchange of water and nutrients occurs across the metalimnion. Primary production is higher in the epilimnion than in the hypolimnion because of higher levels of light.

Shallow lakes are generally well mixed. These lakes have more surface area per unit volume than deep lakes, and winds circulate the water more easily. The productivity of shallow lakes is generally higher than that of deep lakes. Phosphorus, which is often a limiting nutrient for algae, is more available to algae in mixed lakes because it is not trapped in the hypolimnetic water by a metalimnion.

Lakes are often classified according to their productivity or trophic status (Johnson and Vallentyne 1971, Wetzel 1976). Lakes that receive large nutrient supplies and are characterized by high rates of productivity are called eutrophic. Productivity in these lakes often exceeds decomposition, so that large amounts of organic matter accumulate in the sediments. During stratification, the decomposition of this material may eliminate dissolved oxygen in the hypolimnion (see Volume 3-Chapter 4 for further details).

Oligotrophic lakes are those characterized by low nutrient and productivity levels. These lakes are generally cold and deep, and only small amounts of organic material accumulate in the sediments; therefore dissolved oxygen is maintained in the hypolimnion during stratification because little decomposition occurs. Mesotrophic lakes fall between oligotrophic and eutrophic lakes in their range of chemical, physical, and biological characteristics.

Characteristic assemblages of aquatic organisms are associated with each type of lake. The distribution of these assemblages is based on physical and chemical parameters such as temperature, light penetration, pH, nutrients, and alkalinity, which are in turn related to the three lake types described above. These organisms interact similarly to the stream organisms illustrated in Figure 3.

Phytoplankton (algae) are the major source of energy in lakes where allochthonous organic matter is of only minor importance. Phytoplankton capture the energy of sunlight by photosynthesis and transform it into a variety of organic compounds. Although microscopic in size, phytoplankton form the base of the lake food chain upon which fish and other aquatic animals depend for their survival. The productivity of phytoplankton determines many characteristics of a lake including its appearance, the species and numbers of animals present, and even some chemical characteristics.

Larger aquatic plants (macrophytes) also contribute to primary production in lakes. The macrophytes and benthic algae that grow on the bottom substrates can be important contributors to the energy flow in aquatic ecosystems, and are particularly important in lakes with large littoral areas. Macrophytes also appear to be important for the cycling of phosphorus from sediments.

Zooplankton occupy several levels at the bottom of the animal food chain and act primarily as herbivores and detritivores with secondary importance as carnivores. Grazing zooplankton remove organic particles from the water by filtration, and feed on both phytoplankton and particulate detritus. Detritus can be the most important food source for crustacean zooplankton (Wetzel 1976).

Macroinvertebrates and fish occupy the secondary and tertiary consumer levels in lakes. They feed on phytoplankton, zooplankton and other invertebrates and are a food source for decomposers.

Decomposers (bacteria and fungi) play an extremely important role in utilizing dead organic matter and releasing nutrients from the sediments to the water. Bacterial populations are most dense on the surface of the sediments, where most decomposition takes place.

The lake community characterization presented here is based on the sampling of a relatively small number of lakes in the Study Area. Specific lakes were chosen for sampling to provide a range of values for those characteristics known to influence the chemistry and biology of lakes, such as size and depth, watershed soil types, and presence of inlets and outlets. In 1976, 25 lakes were sampled; 5 "primary" and 20 "secondary". The term "primary" lake denotes those which were sampled more frequently in 1976 than those termed "secondary" lakes, in order to examine patterns of temporal variability (Johnson, Lager and Baxter 1978)(Figure 22).

Figure 22

In 1977, 14 of the original 25 lakes were chosen for repeated sampling. These lakes represented a range of values of pH, alkalinity, and total organic carbon (all factors affecting susceptibility to impacts) and types of morphometry.

Plankton samples were collected in primary lakes five times in 1976 and twice in 1977 during the ice-free season. Secondary lakes were sampled twice in both 1976 and 1977. These samples were analyzed for chlorophyll content, phytoplank-ton, and zooplankton. Littoral and profundal benthic invertebrates were collected in

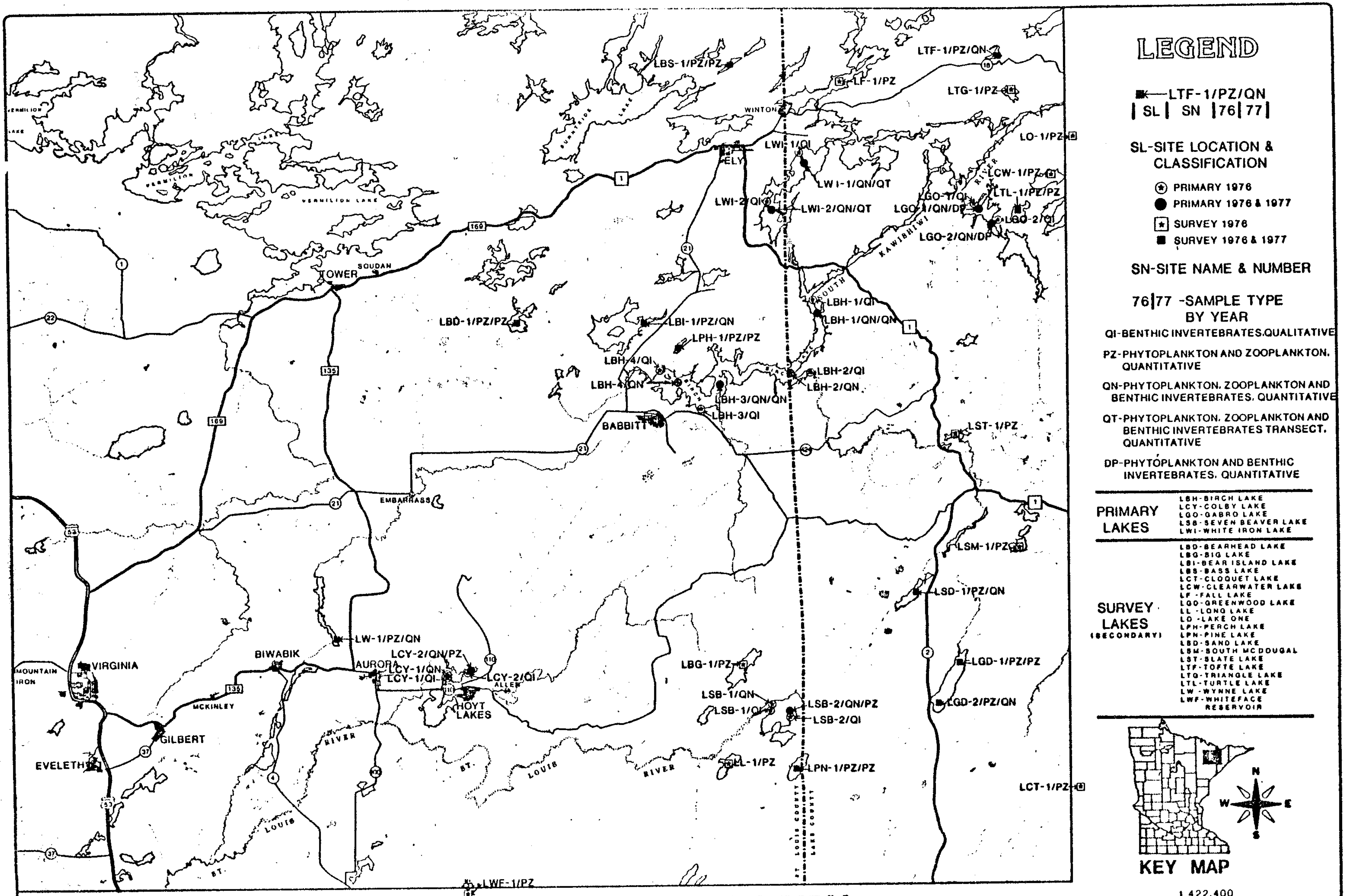

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FIGURE 22 LAKE SAMPLING STATIONS

the five primary lakes in May and October, 1976, and in seven lakes in May, 1977. These were times when maximum densities were expected. Divers collected aquatic plants and clams on transects in primary lakes in the summer of 1976. Data on fish in Study Area lakes were compiled from MDNR Fisheries surveys.

All data were collected on a few dates (two to four for most of the lakes). Since many of the organisms in lakes have very short life spans and population levels can shift dramatically within a few days, the data should be used only to obtain a general picture of the organisms in Study Area lakes. This particularly applies to judgements about relative productivity and nutrient levels among lakes, and the presence or absence of rare species. Similarities within the groups of lakes discussed are general, as each lake is unique and behaves differently in detail from the other lakes.

1.4.2.1 Physical and Chemical Conditions in Study Area Lakes--There are over 310 lakes greater than 10 acres in the Study Area. The majority of these lakes have soft water and are slightly acid. Alkalinity ranges from 3 to 93 mg/l as $CaCO_3$. In comparison, lakes of the Twin Cities metropolitan area range in alkalinity from 78 to 231 mg/l, and alkalinities of prairie lakes in southwestern Minnesota range from 200 to 600 mg/l. The pH in Study Area lakes generally ranges from 6.2 to 8.1. These values are similar to those reported for lakes of the Experimental Lakes Area (ELA) in Ontario (Schindler 1971), but more acidic than those for lakes of the Twin Cities metropolitan area (7.3-9.1) or southwestern Minnesota prairie lakes (8.1-8.5)(see Volume 3-Chapter 4 for further details). Many of the Study Area lakes have high total organic carbon (median 14.0 mg/l, range 4.6 to 38.0 mg/l) concentrations and are often tea colored.

The majority of Study Area lakes have similar physical characteristics. Depths generally range between 3 and 5 meters, with the exception of a few like

Burntside and Tofte which may exceed 10 meters. These lakes vary in size from a few hectares to several square kilometers, but generally are in the range of 2 to 4 square kilometers. Catchment basins may be as small as 2 or as large as 3500 square kilometers. Lakes in the Study Area generally have flushing rates of 3 to 10 times per year, but in Slate Lake the rate may be as high as 82 times per year.

Trophic Status in a lake can be based on chlorophyll concentrations and total phosphorus concentrations, both of which are related to the level of primary production. Because summer is generally the time of maximum productivity for lakes, the summer data were used to characterize lake trophic status. On the basis of maximum summer chlorophyll a concentrations and mean summer total phosphorus concentrations, the Study Area lakes appear to fall into two groups, oligotrophic and meso-eutrophic (mesotrophic/eutrophic). However, because so few samples were collected, any classification is tentative (see Volume 3-Chapter 4 for further details).

Oligotrophic lakes in the Study Area (Table 9) have low chlorophyll a and phosphorus concentrations. In the lakes sampled, summer chlorophyll a concentrations never exceeded 4 ug/l and mean total phosphorus concentrations over the summer were less than 11 ug/l. These lakes were clear, generally deep (greater than 5 meters) lakes.

Table 9

The majority of lakes in the Study Area are meso-eutrophic. In general, these lakes are less than 5 meters deep and highly colored. Several of the lakes sampled appeared to be meso-eutrophic on the basis of phosphorus concentrations and eutrophic on the basis of chlorophyll a concentrations, or vice-versa.

Table 9 Physical and chemical characteristics of study area lakes. Lakes are arranged in order of increasing values for Schindler's (1971) ratio $A_d + A_o/V$. (A_d = area of terrestrial portion of lake's drainage; A_o = surface area of lake; V = volume of lake.) Water chemistry values are averages for summertime samples, 1976 and 1977. For survey lakes only one or two measurements were made.

Lake	$\frac{A_d + A_o}{V}$ (rel.)	A_o (km^2)	Max. Depth (m)	Color (Pt-Co units)	Secchi Disk (m)	pH	Cond. (µmhos/cm)	Total Alkalinity (mg/l $CaCO_3$)	Total Hardness (mg/l)	TOC (mg/l)	Total P (µg/l)	NO_3-N + NO_2-N (mg/l)	NH_4-N (mg/l)	SO_4 (mg/l)	Dissolved Silica (mg/l)
Tofte	0.4	0.47	22	3	5.3	8.6	148	71	31	6	9	0.009	0.01	8.4	0.8
Clearwater	0.6	2.61	14	2	4.0	6.7	39	16	24	7	50**	0.050	0.03	4.0	0.5
Bear Island	1.2	8.64	22	40	2.9	7.4	45	16	24	12	18	0.027	0.04	6.2	3.6
Bearhead	1.3	2.74	14	26	1.8	7.9	68	24	31	11	17	0.030	0.03	8.9	5.8
Triangle	1.3	1.32	12	2	3.8	7.7	65	34	81	7	20	0.010	0.03	4.3	0.4
Big	2.5	3.21	5	14	3.0	7.6	62	25	22	11	30	0.010	0.01	8.1	2.7
Perch	3.6	0.44	9	83	1.3	6.5	29	8	12	16	33	0.017	0.01	4.2	1.9
Bass	3.6	0.68	11	7	5.0	8.2	80	32	41	6	20	0.010	0.02	11.0	3.4
Pine	4.0	1.77	4	103	1.4	7.8	60	18	25	29	29	0.050	0.01	7.8	1.6
Turtle	5.0	1.36	3	30	1.8	6.9	26	9	11	13	22	0.010	0.01	3.1	1.0
Whiteface	6.5	17.22	9	138	1.0	7.1	58	20	42	30	33	0.089	0.09	9.1	6.2
Cloquet	10.4	0.74	2	90	-	7.2	54	21	32	22	40	0.020	0.02	11.0	8.7
Sand	14.4	2.05	12	80	1.4	7.3	64	22	-	28	37	0.009	0.01	8.0	6.0
Greenwood	17.2	5.06	2	170	1.1	6.7	50	8	-	31	38	0.024	0.01	9.8	2.5
One	19.3	3.55	17	27	2.4	6.7	27	15	10	11	40	0.010	0.04	4.2	3.9
Seven Beaver	19.6	5.63	2	168	0.8	6.5	49	13	23	28	43	0.042	0.06	5.4	1.6
Birch	23.6	25.62	8	55	1.9	7.1	69	23	37	14	30	0.073	0.10	8.9	4.9
Long	26.0	1.79	2	30	2.6	7.1	46	14	18	14	11	0.008	-	-	1.6
Wynne	29.4	1.15	16	110	1.9	7.2	139	43	-	25	24	0.130	0.02	29.0	8.6
White Iron	32.6	13.85	15	72	1.7	7.0	52	17	22	14	26	0.031	0.10	8.6	5.2
Colby	47.7	2.24	11	136	2.4	7.1	153	33	60	22	23	0.022	0.09	29.8	6.5
So. McDougal	67.6	1.12	2	260	0.9	6.7	36	11	17	23	40	0.010	-	6.0	5.2
Gabbro	78.2	3.63	15	100	1.4	7.3	48	18	25	15	27	0.037	0.05	4.3	7.3
Fall	106.4	8.93	10	45	1.8	6.7	43	16	24	13	30	0.030	0.06	5.4	4.5
Slate	322.8	0.96	3	180	-	6.8	51	21	22	27	50**	0.060	0.02	5.6	5.8

**See Mustalish et al. 1978 for further details.

These lakes are moderately productive, with maximum summer chlorophyll a concentrations between 5 and 20 ug/l, and mean phosphorus concentrations between 20 and 50 ug/l (Table 9). In a few of these lakes, either chlorophyll a or phosphorus concentrations were in the range typical of the oligotrophic lakes.

Shagawa Lake is probably the most productive lake in northeastern Minnesota. Shagawa is very different from meso-eutrophic lakes in the Study Area because for many years it has received effluent from a sewage treatment plant in Ely. In 1971 and 1972, summer chlorophyll a concentrations in Shagawa Lake were often between 25 and 57 ug/l, and total phosphorus concentrations ranged from 300 to 800 ug/l.

Carlson (1977) attempted to relate the productivity of lakes to chlorophyll a, phosphorus, and secchi disc values in lakes. This index was used in the water quality program (see Volume 3 Chapter 4) and generally indicates that lakes in the Study Area are more productive than indicated in the discussion above. For example, Tofte Lake was classified as oligotrophic in the discussion above, but according to Carlson's index would be considered meso-trophic.

The factors controlling the productivity of lakes in the Study Area cannot be determined from the limited data collected by the Copper-Nickel Study. However, lake morphometry and watershed size may be significant factors. Schindler (1971) proposed a hypothesis to explain the differences and similarities among the lakes of the Experimental Lakes Area (ELA) in northwestern Ontario, where lakes receive most of their nitrogen and phosphorus from atmospheric deposition. Schindler assumed that the vegetation and soils of the watersheds are in equilibrium with the nutrients supplied by atmospheric deposition. Therefore, nutrient inputs to a lake should vary as the ratio of the catchment area of the lake basin to the volume of the lake (Schindler's ratio). This model applies only to headwater lakes (i.e. lakes not receiving inputs from other lakes).

When Schindler's ratio is calculated for Copper-Nickel Study headwater lakes (Table 9), it appears that transparency, maximum summer chlorophyll concentrations, and average summer phosphorus concentrations are related to nutrient inputs from the atmosphere. Concentrations of chlorophyll and total phosphorus tend to increase as Schindler's ratio increases, and secchi disk readings (transparency) decrease. Thus, it appears that nutrients supplied by the atmosphere are important in determining the trophic status of Study Area lakes (see Volume 3-Chapter 4).

1.4.2.2 Biological Characteristics of Study Area Lakes--The observed differences in trophic status among the lakes of the Study Area are apparently not large enough to cause significant differences in biological communities. The dominant species and the structure of the aquatic communities varied only slightly among all of the lakes. The variation was so limited that the biological characteristics of all Study Area lakes can be combined for purposes of discussion.

Summer phytoplankton samples were almost always dominated by the Cyanophyta, Chrysophyta, or Bacillariophyta. Even the most oligotrophic of the lakes were occasionally dominated by the Cyanophyta. Seasonal patterns of dominance were quite variable not only from lake to lake, but within the same lake from year to year. With few exceptions, the Bacillariophyta was the most diverse group, followed by the Chlorophyta and then the Cyanophyta. As many as 40 to 50 species of diatoms were identified in one lake on one date. The Cryptophyta, Pyrrhophyta, and Euglenophyta were the least diverse of the algal groups and were never numerically dominant. Based on dominant species, there are clear differences between phytoplankton communities of the Study Area and those of the Experimental Lakes Area of northwestern Ontario.

Several species of phytoplankton from each taxonomic group were widespread throughout the Study Area. Table 10 lists the most frequently occurring species from each of the major algal groups found in Study Area lakes. This list could be lengthened or shortened depending on the criteria used for inclusion of species. A number of these species are considered by other authors to be oligotrophic indicators. However, a comparison of the existing diatom flora with the diatoms of the sedimentary record from other Minnesota lakes supports the notion that these lakes are generally mesotrophic or eutrophic. Attempts to separate the lakes into trophic subgroups using indicator species were unsuccessful, although *Melosira granulata* may have some value as an indicator of trophic status. The lakes contain a number of algae characteristic of acid water (pH 5-7).

Table 10

Analysis of samples of diatoms in sediment cores from four lakes indicated that the dominant species were similar to those found in the surface water samples and suggested that the lakes have not changed dramatically in trophic status since human settlement in northeast Minnesota (1890)(Gerhart et al. 1978).

Aquatic macrophytes were not important in most Study Area lakes because the littoral zones are typically narrow and rocky. Dense beds of macrophytes are found in shallow bays. The macrophytes were generally not very diverse in Study Area lakes. The aquatic plants of the primary lakes were sampled only in 1976. Less than eight taxa were found at any of the sites sampled. *Sparganium* spp.,

Table 10 Characteristic phytoplankton algae of study lakes. Algae included in the table are those which were present in more than two-thirds of the lakes during at least two sampling seasons and which were present as dominants on at least one occasion. For the Bacillariophyta, Chlorophyta, Cyanophyta, and Chrysophyta dominant species are defined as the 5 most abundant species in each group in a given sample. For the Cryptophyta and Pyrrhophyta dominant species are defined as the 2 most abundant species. The number of units (not cells) of a species was taken as the measure of its abundance.

Species	Frequency of presence or dominance (mean units/slide *)							
	Summer 1976 (24 lakes sampled)		Summer 1977 (15 lakes sampled)		Fall 1976 (23 lakes sampled)		Fall 1977 (12 lakes sampled)	
	Dominant	Present	Dominant	Present	Dominant	Present	Dominant	Presen
Bacillariophyta								
Asterionella formosa	75	96	53	80	96	100	83	92
Cyclotella bodanica	33	75	13	60	13	78	0	17
Fragilaria crotonensis	71	96	67	93	48	87	33	67
Melosira ambigua	29	79	27	53	74	91	58	83
Melosira distans	50	71	27	67	39	70	8	25
Nitzschia sp.	46	88	27	67	17	87	33	58
Tabellaria fenestrata	67	96	47	100	61	83	75	100
Chlorophyta								
Ankistrodesmus falcatus including varieties acicularis & mirabilis	75	96	40	87	61	91	50	75
Botryococcus Braunii	21	88	27	40	4	87	8	8
Oocystis sp.	54	88	20	87	43	87	8	42
Cyanophyta								
Agmenellum quadrupli-catum (= Merismopedia glauca)	71	79	67	87	43	70	17	17
Aphanocapsa delicatissima	88	92	60	80	78	83	58	67
Coelosphaerium Kuetzing-ianum	50	88	20	27	43	91	8	17
Chrysophyta								
Dinobryon bavaricum	92	92	60	73	70	87	33	33
Dinobryon divergens	75	75	60	60	70	87	50	58
Dinobryon sertularia var. protuberans	58	71	60	67	30	65	17	25
Cryptophyta								
Cryptomonas erosa	79	83	60*	67*	74	78	92	92
Pyrrhophyta								
Ceratium hirundinella	50	88	7	47	17	74	8	8

*Cryptomonas spp.

*See Gerhart et al. 1978 for a description of counting methodologies.

Potamogeton epihydrus, Sagittaria spp., Eleocharis spp., Nuphar variegatum, and Potamogeton spp. were found in at least three of the five lakes sampled. Among these, Potamogeton epihydrus was found by Moyle (1945) to be most common in soft waters with alkalinities less than 40 ppm, sulfate concentrations less than 5 ppm and a pH less than 7.5, but was also found in harder waters where the pH remains low. However, Nuphar variegatum was found by Moyle (1945) to be common in hard water lakes on mucky bottoms. The Potamogeton and Sparganium species collected may have been those Moyle found at his softwater sites. He mentions the comparative scarcity of aquatic plants in soft waters, which is consistent with data from this study.

Study Area lakes are characterized by similar zooplankton communities. Some pattern does exist as deep, clear lakes are generally less productive than shallow, stained lakes. Zooplankton productivity also appears to be related to the Trophic State Index of a lake (Table 11). Zooplankton densities in Study Area lakes appear to reach their maxima in late summer and early fall.

Table 11

A small group of zooplankton taxa are common in Study Area lakes. These taxa are usually the dominants and are present throughout the ice-free season. The rare taxa, which are never seasonally dominant, are present in only a small number of lakes. All lakes have the most common taxa, the rotifers, Keratella cochlearis and Polyarthra vulgaris and cladocerans, Bosmina longirostris and Daphnia galeata mendotae, which are the most abundant and widespread taxa in the Study Area.

The benthic invertebrates most frequently collected and most abundant in the profundal zone were Chaoborus spp., Hexagenia limbata, Procladius spp., and

Table 11. Mean rotifer, cladoceran, and copepod density for Group A lakes* averaged over four sampling periods in 1976 and 1977. Units presented represent mean numbers of individuals in standard plankton net sample.*

LAKE	MEAN TOTAL ROTIFERS	MEAN TOTAL CLADOCERANS	MEAN TOTAL COPEPODS (excl. nauplii)	COLOR/DEPTH	TSI
Tofte	5,177	2,215	3,312	clear/deep	low
Bear Island	7,678	1,280	2,125		low
Wynne	7,812	2,323	2,287		med
Bass	9,951	1,850	3,384	clear/deep	low
Perch	19,994	1,562	3,003		med
Bear Head	25,474	1,691	2,634		low
Turtle	28,393	5,155	4,158		med
White Iron	32,741	6,326	7,703		med
Colby	35,816	4,301	4,327		med
Birch	44,028	5,010	10,019		med
Pine	51,793	6,505	5,178		med
Sand	63,500	2,386	6,670	colored/shallow	high
Seven Beaver**	124,169	43,635	14,829	colored/shallow	high

*Group A lakes were sampled in June/July, 1976, October, 1976, April, 1977, and October, 1977.
**Three dates only.

Levels of significance for t-test comparisons of Group A lakes. Units presented are p values of significance tests.

CONTRAST	ROTIFERS	CLADOCERANS	COPEPODS
Low TSI lakes vs. all others	.001	.003	.019
Low TSI lakes vs. high TSI lakes	.000	.000	.004
Clear, deep lakes vs. shallow, colored lakes	.000	.000	.020

***See Piragis et al. 1978 for further details of techniques and analysis.

Chironomus spp. These taxa comprised 89 and 87 percent of all individuals collected in 1976 and 1977, respectively (Figure 23).

Figure 23

These organisms appear to be characteristic of the meso-eutrophic lakes of the region, with their relative abundances in individual lakes depending primarily on substrate composition. Hexagenia limbata was not an important organism in Seven Beaver and Bear Island lakes because the coarse detritus on the bottom is not suitable for it. In Wynne Lake, which has a hard clay substrate, Chaoborus accounted for more than 90 percent of the benthic invertebrates. The dominance of these characteristic taxa is similar to the findings of Hamilton (1971) and Hilsenhoff and Narf (1969) for lakes in the Upper Midwest and Ontario.

Data for oligotrophic Burntside Lake (Shults et al. 1976) may be typical of oligotrophic lakes in the region; there the benthos were dominated by amphipods, Chaoborus and pelecypods. The only oligotrophic lake sampled by the Copper-Nickel Study for benthic invertebrates was Tofte Lake, which appears to be an atypical oligotrophic lake for the region. In July, 1976, dissolved oxy-gen dropped to 2 ppm in the hypolimnion of Tofte Lake; a condition which may account for the restricted invertebrate fauna. Only Chironomus was found in Tofte Lake.

Qualitative sampling in the littoral zone showed a much more diverse fauna than did the profundal sampling. Nearly all the organisms collected in the littoral zone of lakes were also collected in streams. Gastropods (snails) were collected from all the primary lakes. Large pelecypods (clams) were also collected from the profundal zone of the primary lakes. Since clam populations often decrease rapidly in response to disruptions of aquatic systems (Fuller 1974), their presence suggests that these systems have not been heavily disturbed.

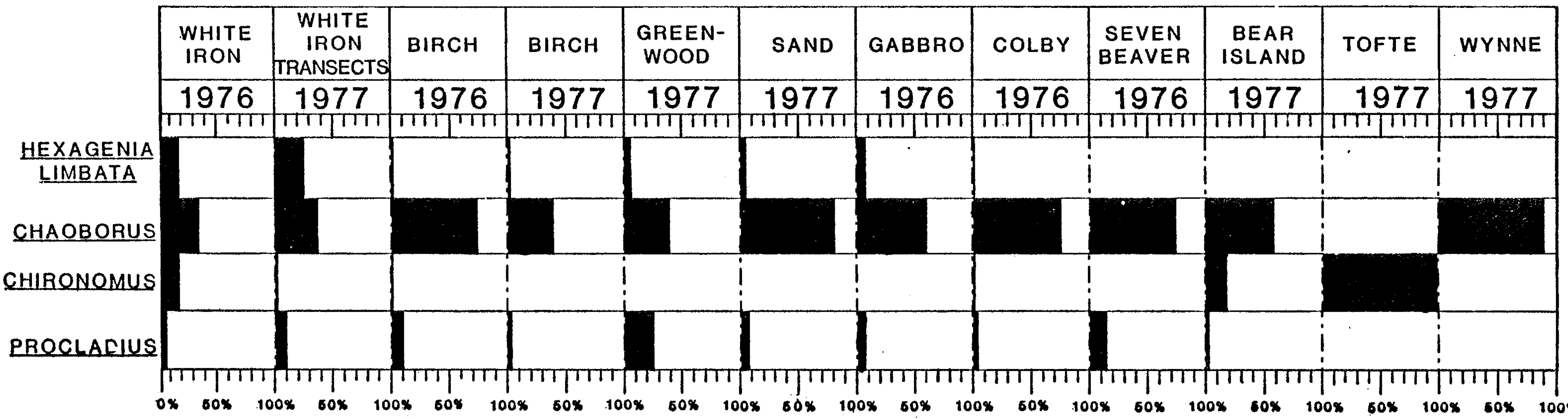


FIGURE 23 MEAN PERCENT COMPOSITION OF DOMINANT TAXA IN LAKES

PERCENTS ARE MEANS OF ALL SAMPLES IN A LAKE

MDNR Fisheries Surveys classify lakes according to a general ecological classification based on fish species in the lakes before management, and according to a management classification which denotes key game species toward which management (stocking, harvest regulations, or no activity) is directed. Because some management practices change the fish populations in lakes considerably, the management classification is good reflection of the present status of a lake.

MDNR data were compiled for 112 lakes in the Study Area; 42 (37.5%) of which are primarily small, shallow lakes of marginal fish value (Table 13). The remaining 70 lakes are classified as walleye (60%), northern pike (15.8%), centrarchid or walleye-centrarchid (11.4%), trout (7.1%), and regular winterkill lakes (5.7%).

The walleye and walleye-centrarchid lakes most often support populations of walleye, northern pike, white suckers, and yellow perch, and frequently support populations of ciscoes and rock bass. Northern pike-sucker-perch lakes in the Study Area are similar to walleye lakes, but lack suitable spawning areas for walleye and therefore do not support walleye populations. Lakes managed for trout tend to support the same fish species as walleye lakes, but are deep and cool and maintain high enough oxygen levels (D.O. greater than 6 mg/l) in summer to maintain trout populations.

The frequency of occurrence of fish species in Study Area lakes is shown in Table 12. Northern pike and white suckers are present in 88.5% and 90.0% of the lakes, respectively, followed by yellow perch (82.8%) and walleye (67.1%). Species lists for Study Area lakes are presented in Table 13. The MDNR fisheries classifications of Study Area Lakes are presented in Figure 24.

Tables 12 and 13, Figure 24

Table 12. Frequency of occurrence of fish species in Study Area lakes. Units presented represent numbers of lakes and percent of Study Area lakes where a given species is found.

	ALL LAKES (70)[a]		WALLEYE LAKES (40)	
SPECIES	Number	Percent	Number	Percent
Northern pike (*Esor lucius*)	62	88.5	39	97.5
Yellow perch (*Perca flavescens*)	58	82.8	37	92.5
White sucker (*Catostomus commersoni*)	63	90.0	38	95
Walleye (*Stizostedion vitreum*)	47	67.1	39	97.5
Rock bass (*Ambloplites rupestris*)	24	34.3	20	50
Tullibee (*Coregonus artedi*)	20	28.5	14	35
Bluegill (*Lepomis macrochirus*)	24	34.2	16	40
Black crappie (*Pomoxis nigromaculatus*)	23	32.9	19	47.5
Burbot (*Lota lota*)	4	5.7	4	10
Tadpole madtom (*Noturus gyrinus*)	5	7.1	3	7.5
Whitefish (*Coregonus clupeaformis*)	6	8.5	3	7.5
Largemouth bass (*Micropterus salmoides*)	10	14.2	4	10
Smallmouth bass (*Micropterus dolomieui*)	6	8.6	4	10
Shorthead redhorse (*Moxostoma breviceps*)	3	4.3	3	7.5
Pumpkinseed (*Lepomis gibbosus*)	9	12.9	6	15
Hybrid sunfish (*Lepomis* sp.)	3	4.3	0	--

Table 12 continued.

SPECIES	ALL LAKES (70)[a] Number	Percent	WALLEYE LAKES (40) Number	Percent
Black bullhead (*Ictalurus melas*)	2	2.9	2	5
Brown bullhead (*Ictalurus nebulosa*)	1	1.4	1	2.2
Channel catfish (*Ictalurus punctatus*)	2	2.9	2	5
Rainbow trout (*Salmo gaidneri*)	1	1.4		
Brook trout (*Salvelinus fontinalis*)	1	1.4		
Muskellunge (*Esox masquinongy*)	2	2.9	1	2.5
Yellow bullhead (*Ictalurus natulis*)	1	1.4	1	2.5
Sculpin (*Cottus* spp.)	3	4.3	2	5
Iowa darter (*Etheostoma exile*)	11	15.7	3	7.5
Johnny darter (*Etheostoma nigrum*)	17	24.3	13	32.5
Log perch (*Percina caprodes*)	5	7.1	5	12.5
Trout-perch (*Percopsis omiscomaycus*)	2	2.9	2	5
Spottail shiner (*Notropis hudsonius*)	7	10.0	6	15
Blacknose shiner (*Notropis heterolepis*)	11	15.7	6	15
Common shiner (*Notropis cornutus*)	5	7.1	2	5
Mimic shiner (*Notropis volucellus*)	4	5.7	1	2.5

Table 12 continued.

SPECIES	ALL LAKES (70)[a] Number	Percent	WALLEYE LAKES (40) Number	Percent
Golden shiner (*Notemigonus crysoleucas*)	7	10.0	4	5
Hornyhead chub (*Hybopsis higuttata*)	1	1.4	1	--
Bluntnose minnow (*Pimephales notatus*)	6	8.6	2	--
Lake trout (*Salvelinus naymaycush*)	3	4.3		
Finescale dace (*Chrosomus neogreus*)	3	4.3		
Brook stickleback (*Culea inconstans*)	3	4.3		
Blacknose dace (*Rhinichthys atratulus*)	1	1.4		
Northern redbelly (*Chrosomus eos*)	2	2.9		
Fathead minnow (*Rimephales promelas*)	2	2.9		
Central mudminnow (*Ambra limi*)	2	2.9	1	2.5

[a]Lakes which are managed for game fish; this groups includes walleye lakes.

Table 13 Fish species distributions in Study Area Lakes. Data from MDNR lake surveys 1950-1977.

Hudson Bay Drainage	Ecological Classification	Management Classification	Northern pike	Walleye	Smallmouth bass	Largemouth bass	Lake trout	Rainbow trout	Brook trout	Tullibee	Whitefish	Bluegill	Pumpkinseed	Hybrid sunfish	Green sunfish	Rock bass	Black crappie	White sucker	Shorthead redhors	Yellow perch	Channel catfish	Black bullhead	Brown bullhead	Tadpole madtom	Burbot	Muskellunge	Sculpin spp.	Iowa darter	Johnny darter	Log perch	Trout-perch	Spottail shiner	Blacknose shiner	Common shiner	Mimic shiner	Golden shiner	Hornyhead chub	Bluntnose minnow	Yellow bullhead	Finescale dace	Brook stickleback	Blacknose dace	Northern redbelly dace	Fathead minnow	Central mudminnow
Tofte*	NP-SP	Tr	+	+				+		+		+				+		+		+															+			+							
Triangle	C-W	C-W	+		+	+				+		+						+		+									+																
Bass	C	C-W	+									+					+	+		+																									
Fall	SW-W	W	+	+						+	+	+				+	+	+	+	+				+	+			+	+	+		+													
Lake One	SW-W	W	+	+						+	+	+				+		+		+									+							+		+							
Clearwater	NP-S-P	Tr	+						+		+							+		+							+		+						+			+							
White Iron	SW-W	W	+	+						+		+				+	+	+		+					+				+	+															
Bear Island	SW-W	W	+	+						+						+		+		+																									
Perch	NP-C	NP,S,P	+									+		+						+																									
Gabbro	SW-W	W	+	+						+						+	+	+		+								+	+	+				+		+									
Turtle	NP-S-P	NP,S,P	+															+		+																+		+							
August	SW-W	W	+	+												+		+		+				+																					
Birch	SW-W	W	+	+		+				+		+				+	+	+		+					+	+			+	+	+	+					+	+							
Greenwood	—	W	+	+														+		+									+			+													
Sand	SW-W	W	+	+														+		+								+	+			+													

*Tofte Lake was reclaimed and subsequently converted to a stream trout lake.

**Abbreviations used: G=Gillnet; MT=Minnow trap; T=Trapnet; S=Seine; Tr=Trout; HW=Hardwater; SW=Softwater; W=Walleye; NP=Northern pike; S=White sucker; P=Yellow perch; C=Centrarchid.

** See Figure 22 for location of lakes in Study Area.

Table 13 continued

	Ecological Classification	Management Classification	Northern pike	Walleye	Smallmouth bass	Largemouth bass	Lake trout	Rainbow trout	Brook trout	Tullibee	Whitefish	Bluegill	Pumpkinseed	Hybrid sunfish	Green sunfish	Rock bass	Black crappie	White sucker	Shorthead redhorse	Yellow perch	Channel catfish	Black bullhead	Brown bullhead	Tadpole madtom	Burbot	Muskellunge	Sculpin spp.	Iowa darter	Johnny darter	Log perch	Trout-perch	Spottail shiner	Blacknose shiner	Common shiner	Mimic shiner	Golden shiner	Hornyhead chub	Bluntnose minnow	Yellow bullhead	Finescale dace	Brook stickleback	Blacknose dace	Northern redlly dace	Fathead minnow	Central mudminnow
Johnson	SW-W,C	W	+	+								+				+		+		+																									
Ojibway	Tr	Tr					+			+								+																											
Basswood 1	SW-W	Tr	+	+			+			+	+							+		+																									
Newton	SW-W	W	+	+														+		+																									
Burntside	Tr	Tr	+	+	+		+			+	+		+			+	+	+		+																									
Shagawa	SW-W	W	+	+	+					+	+						+	+	+	+																									
South Farm	SW-W	W	+	+						+		+					+	+		+																									
Farm	SW-W	W	+	+	+					+		+	+				+	+		+																									
Eskwagama	NP,S,P	NP, S,P	+															+																											
Clear	SW-W	W	+	+								+				+		+		+																									
Garden	SW-W	W	+	+						+		+	+				+	+		+																									

Abbreviations used: G=Gillnet; MT=Minnow trap; T=Trapnet; S=Seine; ST=Stream trout; HW=Hardwater; SW=Softwater; W=Walleye; NP=Northern pike

Table 13 continued

Lake	Ecological Classification	Management Classification	Northern pike	Walleye	Smallmouth bass	Largemouth bass	Lake trout	Rainbow trout	Brook trout	Tullibee	Whitefish	Bluegill	Pumpkinseed	Hybrid sunfish	Green sunfish	Rock bass	Black crappie	White sucker	Shorthead redhorse	Yellow perch	Channel catfish	Black bullhead	Brown bullhead	Tadpole madtom	Burbot	Muskellunge	Sculpin spp.	Iowa darter	Johnny darter	Log perch	Trout-perch	Spottail shiner	Blacknose shiner	Common shiner	Mimic shiner	Golden shiner	Hornyhead chub	Bluntnose minnow	Yellow bullhead	Finescale dace	Brook stickleback	Blacknose dace	Northern redbel' dace	Fathead minnow	Central mudminnow
North McDougal	SW-W	W	+	+												+		+		+																				+	+				
Gypsy	M	RW	+																																										
Two Deer	NP,S,P	NP,S,P	+															+		+								+						+				+							
Fran	NP,S,P	NP,S,P	+															+		+																									
Chow	NP,S,P	NP,S,P	+																	+																									
Shallow	C,HW-W	W,C	+	+		+										+		+		+								+	+	+			+	+	+	+						+			
East Chub	SW-W	W	+	+												+	+	+		+													+			+									+
West Chub	NP,S,P	W	+			+										+		+		+								+																	
Middle McDougal	SW-W	W	+	+														+		+																									
Harris	C	C				+						+	+	+			+	+		+						+		+						+		+	+								

**Abbreviations used: C=Gillnet; MT=Minnow trap; T=Trapnet; S=Seine; Tr=Trout; HW=Hardwater; SW=Softwater; W=Walleye; NP=Northern pike; S=White sucker; P=Yellow perch; C=Centrarchid.

Table 13 continued

	Ecological Classification	Management Classification	Northern pike	Walleye	Smallmouth bass	Largemouth bass	Lake trout	Rainbow trout	Brook trout	Tullibee	Whitefish	Bluegill	Pumpkinseed	Hybrid sunfish	Green sunfish	Rock bass	Black crappie	White sucker	Shorthead redhorse	Yellow perch	Channel catfish	Black bullhead	Brown bullhead	Tadpole madtom	Burbot	Muskellunge	Sculpin spp.	Iowa darter	Johnny darter	Log perch	Trout-perch	Spottail shiner	Blacknose shiner	Common shiner	Mimic shiner	Golden shiner	Hornyhead chub	Bluntnose minnow	Yellow bullhead	Finescale dace	Brook stickleback	Blacknose dace	Northern redbelly dace	Fathead minnow	Central mudminnow
South McDougal	SW-W	W	+	+														+		+				+					+		+	+				+									
Slate	SW-W	W	+	+												+		+		+									+			+													
Bearhead	SW-W	W	+	+		+								+				+		+																									
Lake Superior Drainage																																													
Whitewater	C-W	W	+	+								+						+		+																									
Iron				+														+																											
Cranberry	RG	NP-S-P	+															+		+				+																					
Round	SW-N	W	+	+														+		+																									
Cedar Island	C-W	W	+	+								+					+	+		+		+	+																						
Sabin	SW-W	W	+	+						+							+	+																											
Esquagama	SW-W	W	+	+						+		+	+				+			+																									
Embarrass	CN	W	+	+								+					+	+			+																								
Bassett		W	+	+		+							+			+	+	+		+							+		+																
Cadotte	SW-W	W	+	+		+						+						+		+							+	+	+																

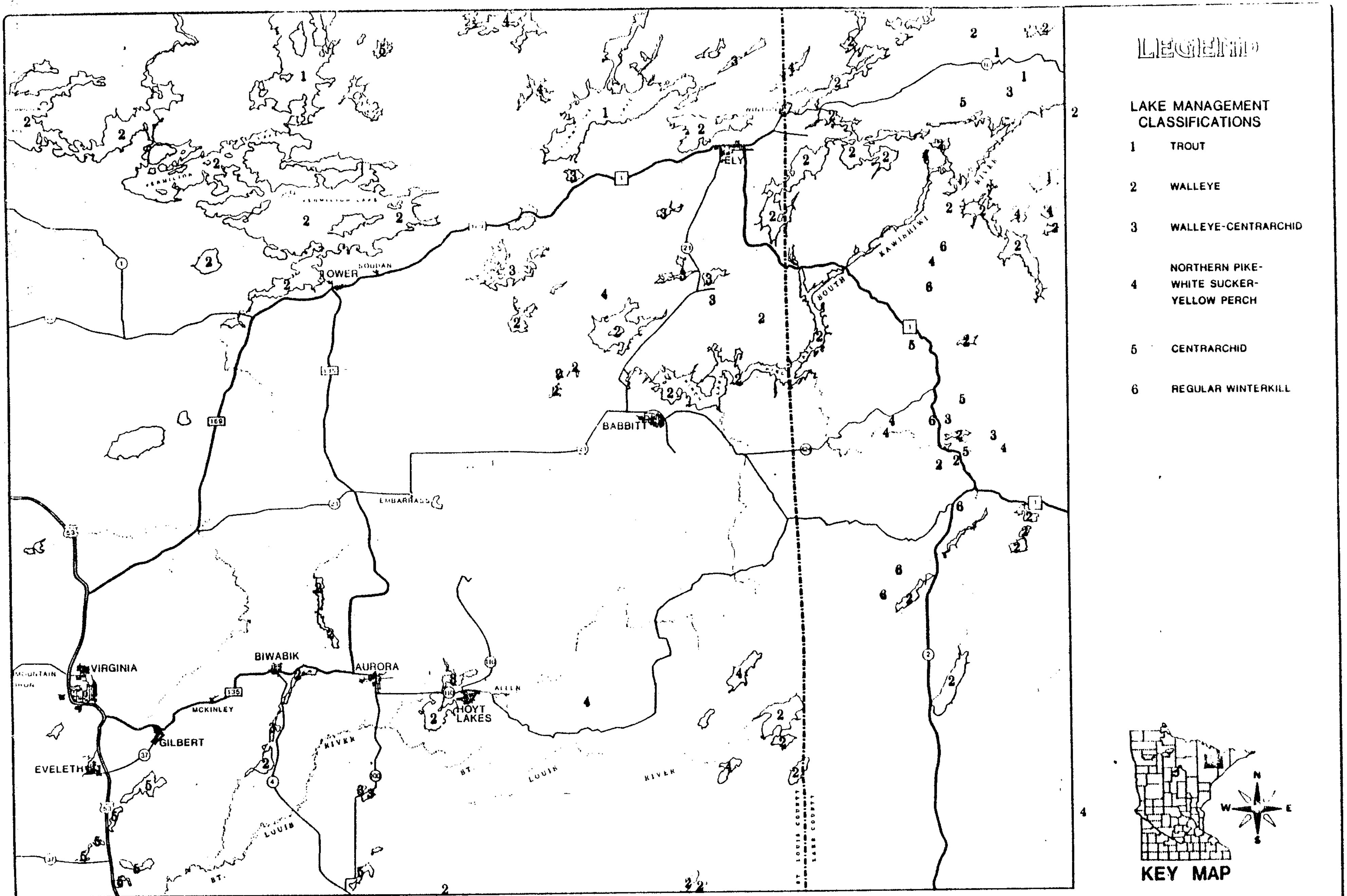
**Abbreviations used: G=Gillnet; MT=Minnow trap; T=Trapnet; S=Seine; Tr=Trout; HW=Hardwater; SW=Softwater; W=Walleye; NP=Northern pike; S=White sucker; P=Yellow perch; C=Centrarchid; M=Minnow; Lake trout; WG=Warm water game fish; LS=Lake survey; TN=Test netting; FT=fish tissue col

Table 13 continued

	Ecological Classification	Management Classification	Northern pike	Walleye	Smallmouth bass	Largemouth bass	Lake trout	Rainbow trout	Brook trout	Tullibee	Whitefish	Bluegill	Pumpkinseed	Hybrid sunfish	Green sunfish	Rock bass	Black crappie	White sucker	Shorthead redhorse	Yellow perch	Channel catfish
Highlife	C	C				+														+	
Dunnigan	SW-W,C	W		+	+							+				+		+			
Little	SW-W	W	+	+														+		+	
Omaday	NP,S,P	NP,S,P	+															+			
Bogberry	M	RW																+			
Bald Eagle	SW-W	W	+	+						+							+	+		+	
Gull	SW-W	W	+	+	+											+		+		+	
Little Gabbro	SW-W	W	+	+						+							+	+		+	
Nickel	M	RW	+															+			
Pietro	NP,S,P	NP,S,P	+										+			+		+			
Shammrock	C	C				+						+						+			
Blueberry	W-C	W-C	+	+								+					+	+		+	
One Pine	SW-WC	W	+	+														+		+	

	Black bullhead	Brown bullhead	Tadpole madtom	Burbot	Muskellunge	Sculpin spp.	Iowa darter	Johnny darter	Log perch	Trout-perch	Spottail shiner	Blacknose shiner	Common shiner	Mimic shiner	Golden shiner	Hornyhead chub	Bluntnose minnow	Yellow bullhead	Finescale dace	Brook stickleback	Blacknose dace	Northern redbelly dace	Fathead minnow	Central mudminnow
Highlife																								
Dunnigan							+																	
Little																								
Omaday																								
Bogberry							+	+				+							+	+		+	+	+
Bald Eagle																								
Gull																								
Little Gabbro																								
Nickel							+					+							+	+		+	+	
Pietro																								
Shammrock																								
Blueberry																								
One Pine																								

**Abbreviations used: G=Gillnet; MT=Minnow trap; T=Trapnet; S=Seine; Tr=Trout; HW=Hardwater; SW=Softwater; W=Walleye; NP=Northern pike; S=White sucker; P=Yellow perch; C=Centrarchid.



MEQB REGIONAL COPPER-NICKEL STUDY

FIGURE 24 FISH MANAGEMENT CLASSIFICATION-LAKES

Walleyes occur more frequently in lakes south of the Laurentian Divide, whereas ciscoes and whitefish are found more frequently north of the Divide. Bullheads and catfish are not known to occur north of the Divide but are found in lakes south of the Divide. There are no known lake trout or stream trout lakes in the Study Area south of the Divide because most lakes south of the Divide are too shallow and warm in the summer to support trout.

Sixty percent of the lakes managed for gamefish in the Study Area are managed specifically for walleyes, and all of these lakes are soft water lakes. Data from 40 lakes managed for walleyes (Tables 12 and 14) were compiled to summarize the characteristics of walleye lakes in the Study Area for comparison with walleye lakes throughout the State of Minnesota. The fish species most frequently collected were walleye, northern pike (found in 97.5 percent of the Study Area walleye lakes), white sucker (95%), and yellow perch (92.5%).

Table 14

Walleye production in Study Area walleye lakes is apparently greatest in Shagawa Lake. It had the greatest number of walleyes per net (37.9 using a standard size gill net for 24 hours), the greatest total weight per net (22.6 kg), and the second largest value for average weight per fish (0.59 kg) for lakes north of the Laurentian Divide. Gabbro, White Iron, Fall, Birch, and Bald Eagle lakes also lie north of the Divide and produce large numbers of walleyes relative to other lakes in the Study Area. South of the Laurentian Divide, Bassett and Cadotte lakes had high numbers of walleyes relative to other lakes south of the Divide, as well as all walleye lakes in the Study Area. Wynne Lake had the largest average fish (.72 kg/fish) of all walleye lakes studied but fewer walleyes per net.

Table 14 Number and weight of walleye, northern pike, and white sucker in managed walleye lakes in the Study Area. Data from MDNR lake surveys 1950-1977.

Hudson Bay Drainage

LAKE	DOW #	WALLEYE Mean # Per Net	WALLEYE Mean Wt Per Net (Kg)	WALLEYE Mean Wt Per Fish (Kg)	NORTHERN PIKE Mean # Per Net	NORTHERN PIKE Mean Wt Per Net (Kg)	NORTHERN PIKE Mean Wt Per Fish (Kg)	WHITE SUCKER Mean # Per Net	WHITE SUCKER Mean Wt Per Net (KG)	WHITE SUCKER Mean Wt Per Fish (Kg)
Fall	38-811	14.3	3.77	0.27	5.3	3.43	0.68	3.5	2.16	0.63
Newton	38-784	9.5	-	-	2.3	-	-	5.8	-	-
Shagawa	69-69	37.9	22.6	0.59	3.7	2.59	0.72	11.0	9.45	0.86
White Iron	69-4	12.6	4.30	0.36	3.8	3.40	0.90	7.6	5.54	0.72
Lake One	38-605	4.7	1.22	0.27	3.1	1.97	0.63	8.7	4.73	0.54
Clear	38-722	2.8	-	-	2.3	-	-	4.3	-	-
Farm	38-779	3.6	-	-	0.1	-	-	3.5	-	-
South Farm	38-778	8.0	-	-	0.8	-	-	6.3	-	-
Garden	38-738	6.2	-	-	0.2	-	-	3.5	-	-
Bear Island	69-115	8.3	5.04	0.59	1.5	1.34	0.90	2.4	1.51	0.63
One Pine	69-61	6.1	-	-	5.4	-	-	5.4	-	-
Johnson	69-117	0.5	-	-	3.0	-	-	5.5	-	-
Gabbro	38-701	7.5	4.05	0.54	5.3	5.45	1.04	4.9	0.77	0.18
August	38-691	4.9	-	-	3.0	-	-	6.3	-	-
Bald Eagle	38-637	14.3	3.77	0.27	7.4	6.43	0.86	5.6	4.35	0.77
Little Gabbro	38-703	4.3	1.88	0.45	3.3	2.48	0.77	2.3	1.74	0.77
Gull	38-590	6.9	-	-	1.6	-	-	2.1	-	-
Birch	69-3	8.0	5.22	0.68	2.4	3.38	1.40	5.7	3.85	0.68
Little	69-56	9.0	-	-	3.7	-	-	4.3	-	-
Greenwood	38-656	6.9	6.26	0.32	4.9	4.59	0.45	4.4	12.29	0.77
North McDougal	38-686	7.6	3.56	0.45	3.3	2.25	0.68	8.7	5.95	0.68
South McDougal	38-659	9.0	-	-	11.0	-	-	9.0	-	-
Sand	38-735	21.0	-	-	6.3	-	-	5.0	-	-
Slate	38-666	5.0	-	-	2.0	-	-	15.0	-	-
East Chub	38-674	3.0	-	-	6.0	-	-	7.0	-	-
West Chub	38-675	-	-	-	9.4	-	-	4.2	-	-
Dunnigan	38-664	14.5	-	-	-	-	-	15.0	-	-
Mean		9.09	5.61	0.62	3.89	3.39	0.87	6.19	4.76	.77

Lake Superior Drainage

LAKE	DOW #	WALLEYE Mean # Per Net	WALLEYE Mean Wt Per Net (Kg)	WALLEYE Mean Wt Per Fish (Kg)	NORTHERN PIKE Mean # Per Net	NORTHERN PIKE Mean Wt Per Net (Kg)	NORTHERN PIKE Mean Wt Per Fish (Kg)	WHITE SUCKER Mean # Per Net	WHITE SUCKER Mean Wt Per Net (KG)	WHITE SUCKER Mean Wt Per Fish (Kg)
Seven Beaver	69-2	9.8	3,86	0.41	4.0	2.19	0.54	3.0	3.09	1.04
Pine	69-1	3.0	1.16	0.41	2.0	3.12	1.58	6.7	-	-
Round	69-48	10.0	5.22	0.54	1.5	1.55	1.04	9.0	9.52	1.26
Colby	69-249	1.3	0.59	0.45	0.67	0.57	0.81	3.0	3.27	1.08
Whitewater	69-376	3.67	1.36	0.36	2.0	1.40	0.72	1.7	2.07	1.26
Wynne	69-434	4.7	3.45	0.72	3.5	2.65	0.77	0.8	0.86	1.04
Embarrass	69-496	3.2	0.95	0.32	2.3	1.89	0.81	-	-	-
Cedar Island	69-568	1.33	0.53	0.41	12.0	6.22	0.54	2.3	5.70	0.86
Esquagama	69-565	4.7	1.27	0.27	3.7	1.65	1.67	2.0	0.90	0.90
Sabin	69-429	1.7	0.91	0.54	6.0	4.60	0.77	2.7	2.66	0.99
Whiteface Reservoir	69-375	5.7	1.85	0.32	1.5	0.85	0.59	3.3	3.14	0.95
Cadotte	69-114	15.9	6.33	0.41	0.2	0.33	1.53	1.4	0.82	0.59
Bassett	69-41	28.3	16.12	0.59	0.5	0.23	0.45	4.1	2.47	0.81
Mean		7.1.	[illegible]	.44	06	1.09	[illegible]8	3 [illegible]	[illegible]	.88

The largest number of northern pike per net was found in South McDougal Lake (11.0) and West Chub Lake (9.4) north of the Divide, and in Cedar Island Lake (12.0) and Sabin Lake (6.0) south of the Divide (Table 14). Birch and Gabbro lakes north of the Divide and Pine, Round, Esquagama, and Cadotte lakes south of the Divide have the largest northern pike (Table 14).

White suckers were most abundant in Slate and Dunnigan lakes, each with an average of 15.0 suckers per net. Shagawa Lake had the second largest number (11.0 fish/net) and the largest average size (.86 kg/fish) of white suckers north of the Divide. South of the Divide the largest white sucker populations recorded were from Round (9.0 fish/net) and Pine (6.7 fish/net) lakes.

Trout are the only species of fish whose presence in Study Area lakes is related to the trophic status of the lakes. Among Study Area lakes, all lakes classified as oligotrophic are managed as trout lakes, while the meso-eutrophic lakes are primarily walleye and centrarchid-walleye lakes.

In comparing the fish of Study Area lakes to those from the rest of Minnesota, it should be noted that 95.7 percent of the lakes managed for lake trout are found in Cook, Lake, and St. Louis counties, while only 48 percent of the walleye lakes are found in those counties (Peterson 1971).

Generally, the abundance and weight of walleye, northern pike, and white suckers in the Study Area lakes are higher than the statewide medians, but the median abundance of northern pike and white suckers is lower than in MDNR Region II (includes Cook, Lake, St. Louis, Carlton, Koochiching, and Itasca counties) medians for soft-water walleye lakes.

Based on biological characteristics in conjunction with physical and chemical parameters, Study Area lakes can be classified, for the purposes of impact assessment, into oligotrophic and meso-eutrophic lakes (see section 1.4.2.1 of this chapter). The distribution of these lake types based on chlorophyll, zooplankton, phytoplankton, benthic invertebrates, and water chemistry is directly related to MDNR fisheries management classifications of trout lakes and walleye/northern pike lakes, respectively. Therefore, based on MDNR fisheries classifications it is possible to generalize from the 25 lakes which were studied intensively to other lakes throughout the Study Area (Table 15).

Table 15

The meso-eutrophic walleye and northern pike lakes are the dominant type in the Study Area and comprise approximately 95 percent of the lakes in the region. Most of these lakes are shallow (less than 5 meters deep), highly colored (generally greater than 10 color units), high in total organic carbon (greater than 15 ug/l), and may receive drainage from bogs. These lakes tend to be moderately productive with summer chlorophyll values in the range of 5 to 15 ug/l. Lakes of this type which are larger than 20 hectares support walleye/northern pike fisheries that are generally more productive than those lakes on a statewide basis.

Approximately 96 percent of the oligotrophic trout lakes in Minnesota occur in Lake, Cook, and St. Louis counties, yet they represent only about 5 percent of the lakes in the Study Area. These lakes are usually deep (greater than 5 meters), clear (less than 10 color units), and have low concentrations of organic carbon (less than 15 ug/l). The organisms found in the oligotrophic lakes are similar to those in meso-eutrophic lakes with the exception of trout, which are restricted to the oligotrophic lakes.

Table 15. Comparison of relative abundance of fisheries management types in lakes sampled by RCNS and EPA (Burntside) to compilation of MDNR data from 70 lakes in the Study Area.

TYPE	LAKES SAMPLED BY RCNS AND EPA	70 STUDY AREA LAKES WITH MDNR CLASSIFICATION
Walleye	62%	60%
Northern pike	15%	16%
Centrarchid or Walleye centrarchid	11.5%	11%
Trout	11.5%	7%
Winterkill	--	6%

1.4.2.3 Lakes Currently Affected by Mining--Dunka and Bob Bays in Birch Lake receive mine dewatering drainage from current taconite operations. In addition, leachate enters Bob Bay from stockpiles of sulfide mineralized gabbro material located adjacent to Erie Mining Company's Dunka Pit. The biological, chemical, and physical characteristics of these bays were studied to determine the effects on Bob Bay of heavy metals from the leachate. There are significant differences between Bob and Dunka bays in the concentration of copper and nickel in sediments and nickel in water. Median copper and nickel concentrations in the water of Bob Bay presently range from 1.9 to 8.0 ug Cu/l and 3 to 79 ug Ni/l at the various stations, whereas the median concentrations in Dunka Bay water range from 1.5 to 2.1 ug Cu/l and 1.9 to 2.0 ug Ni/l. The highest copper and nickel concentrations in Bob Bay are found at the mouth of Unnamed Creek, while concentrations of copper and nickel are similar throughout Dunka Bay. Sediments in Bob Bay contain 52 to 92 ppm and 266 to 724 ppm copper and nickel, respectively, while the sediments in Dunka Bay contain 24 to 35 ppm and 29 to 39 ppm copper and nickel, respectively (see Volume 3-Chapter 4 for further details). Chronic effects on aquatic organisms have been observed and reported in the literature for copper concentrations as low as 5 ug/l and nickel concentrations as low as 95 ug/l. No data are available on the chronic effects of copper and nickel in sediments on aquatic organisms (see section 1.6 of this chapter for further details).

A comparison of the benthic invertebrates from Bob and Dunka bays indicated very few differences. The list of species and the densities of organisms were similar for both bays. The most significant exception was the insect *Tanytarsus* spp. *Tanytarsus dissimilis* has previously been found to be sensitive to copper and nickel (Anderson et al. 1977). This insect was consistently more common in Dunka Bay and in November, 1976, was approximately 20 times more abundant in Dunka Bay than Bob Bay. It is not possible to determine if these population differences result from the differences between copper and nickel concentrations in water and sediment or whether they may be the result of other subtle differences in the substrate or other physical and chemical characteristics of the two bays.

The concentrations of copper and nickel in clams and water lilies from the two bays were examined (Table 16). Copper levels were significantly higher in clams and lilies from Bob Bay. Nickel concentrations in clams from the two bays were similar; however, the water lilies in Bob Bay contained significantly greater concentrations of nickel than those in Dunka Bay.

Table 16

These studies also indicated a direct relationship between the concentration of copper found in sediments and that found in the tissues of both species, as well as a direct relationship between the concentrations of nickel in the sediments and water and those in the water lilies. These results indicate that copper and nickel may be biologically available from both sediment and water. At this time it is not possible to definitively establish the source of these metals (i.e. water and/or sediments) or the short or long term effects of these increased tissue levels on the aquatic organisms of Bob Bay.

1.4.3 Relationship Between Lakes and Streams

Although the communities in lakes and streams are quite different, there often is an important interaction between them. The interactions between lakes and streams range from alterations in various water quality conditions to the provision of food and spawning habitat.

Table 16. Mean copper* and nickel* values from Bob Bay, Dunka Bay, and Birch Lake.

	LBB-1	LBB-2	LBB-5	LDB-2	LDB-3	LB-2	LB-4
Nuphar (ppm)							
Copper	4.08	6.96	—	1.12	—	—	—
Nickel	18.26	8.23	—	.09	—	—	—
Clams (ppm)							
Copper	—	—	2.08	.79	.80	.78	1.39
Nickel	—	—	.41	.51	.68	.33	.69
Water (ppb)							
Copper	8.15	5.75	1.93	2.37	1.77	2.60[a]	2.57
Nickel	67.2	60.91	8.55	2.36	2.43	3.20	3.46
Sediments (ppm)							
Copper	82.0	92.0	91.7	34.0	35.0	22.0	33.0
Nickel	1100.0	496.0	461.0	24.0	38.0	46.0	39.0

[a]Median value.

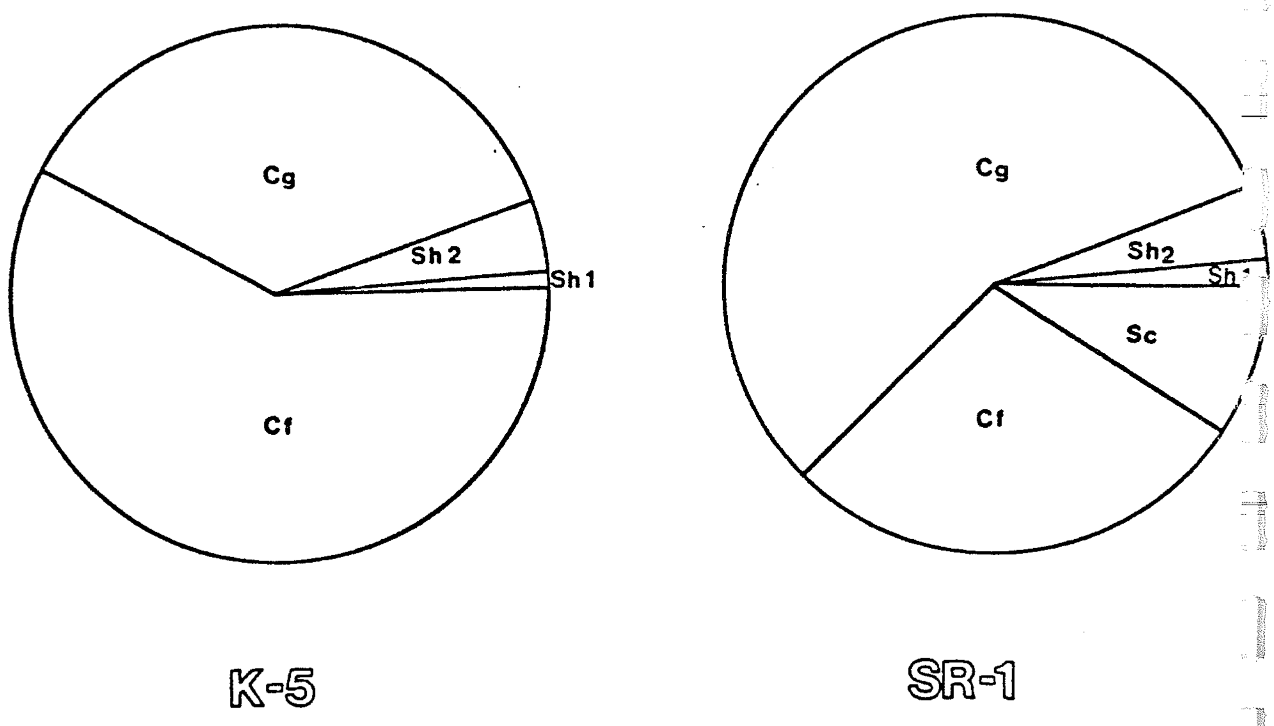
*Nuphar data and sediment data were obtained by means of plasma source analysis, clams and water were analyzed by means of atomic absorption.

Streams transport organic debris, nutrients, and other materials from the watershed into a lake. Once these materials enter a lake they can be trapped in the lake sediments, released unchanged at lake outlets, or cycled through the food chain. The resulting changes in the water quality between incoming and outgoing water can cause the biological characteristics of inlet and outlet streams to be significantly different. For example, if organic debris is removed from a stream system by a lake, the macroinvertebrate fauna may change from a shredder/collector-gatherer community in the inlet to a collector/filter-feeder community in the outlet. The filter feeding insects would thrive on the rich supply of lake plankton drifting out of the lake. This change was observed in Birch Lake where the Stony River flows in and the Kawishiwi River flows out. The Stony River is dominated by large populations of collector-gatherer insects (56.4%), whereas the Kawishiwi River is dominated by collector-filter feeding insects (57.8%)(Figure 25).

Figure 25

Lakes can also change the temperature, dissolved oxygen, pH, and alkalinity of streams, factors which could influence the indigenous organisms of the stream. The removal of pollutants by lakes could be a major positive effect of lakes on stream systems. An example of this phenomenon is the effect of Shagawa Lake on the Shagawa River. High concentrations of phosphorus and nitrogen would be expected in Shagawa River, because Shagawa Lake received sewage effluent from Ely which was high in phosphorus and nitrogen until 1973. The high phosphorus and nitrogen concentrations, however, were not found in the Shagawa River apparently because these nutrients became trapped in Shagawa Lake.

FIGURE 25 RELATIVE ABUNDANCE OF INVERTEBRATE FUNCTIONAL GROUPS AT SITES ABOVE (SR-1) AND BELOW (K-5) BIRCH LAKE IN 1976.



Sh_1= Shredders of dead plant material

Sh_2= Shredders of living plant material

Cg = Collector-gatherer

Cf = Collector-filterer

Sc = Scraper

Streams provide spawning habitat for many species of lake fish and also provide habitat for young game and forage fish. Northern pike, white suckers, and several other species commonly use streams of various orders as spawning habitat. For example, northern pike from Birch Lake spawn in the Dunka and Stony rivers as well as in Keeley Creek. Riffles at the inlets of lakes can also provide an abundant supply of macroinvertebrates for fish. Northern pike and smallmouth bass from lakes were often observed feeding below stream riffles in the Study Area.

1.4.4 Susceptibility of Region to Generic Impacts

Several major stresses may arise from copper-nickel development and affect the aquatic ecosystems of the Study Area. These include increased concentrations of heavy metals, pH changes, and physical alteration of aquatic habitats, including direct loss of habitat. The magnitude of the effects of these stresses in the Study Area is determined by the sensitivity and distribution patterns of the indigenous organisms, by physical and chemical factors and by spatial and temporal variation in these factors. These factors vary among watersheds, stream orders, lakes and seasons and result in a range of sensitivities among the various systems and through the seasons.

Sensitivities were determined on the basis of a modification of a system proposed by Cairns (1976) to rate the assimilative capacity of ecosystems. In this system, various characteristics of ecosystems important in their reactions to stress are rated as good, moderate, or poor. Factors which were examined included: 1) sensitivity of indigenous organisms to variable conditions; 2) structural and functional redundancy; 3) stream or lake flushing rate; 4) water quality which mitigates stress; 5) nearness to a major ecological transition zone; 6) trophic status; 7) location of areas to provide organisms for recolonization once impacted; and 8) ability of indigenous organisms to recolonize. Ratings are professional judgements based on available data.

1.4.4.1 Watershed Sensitivity--Few differences are apparent in the capacities of biological communities in the various Study Area watersheds to endure and recover from either recurring acute or chronic stresses except in the case of the Little Isabella watershed. The Little Isabella watershed including Snake Creek and Snake River, was considered the most sensitive watershed because 56 percent of this stream system is classified as trout streams by MDNR. Trout and other associated cold water species are sensitive to most potential stresses and natural recolonization would be slow because areas which could provide organisms to repopulate an affected area are remote.

1.4.4.2 Stream Sensitivity--Because watersheds are approximately equal in their sensitivity, stream orders were rated on the basis of their sensitivity. As in the discussion of stream communities, stream orders were combined as follows: headwaters (1st and 2nd order), mid-reaches (3rd and 4th order) and Kawishiwi River (5th order). Headwater streams were then divided into cold water and warm water streams.

Headwater streams are highly sensitive to stress because they are totally dependent upon the terrestrial ecosystem for nutrients, unlike higher order streams which receive inputs from many tributaries draining different areas of the watershed. Other factors which cause headwater streams to be highly sensitive include: the lack of consistent flow, low dilutional capabilities, and low buffering capacity when compared to that of higher order streams.

Cold headwater streams are probably the most susceptible streams in the region. This susceptibility is based on evidence that trout and other cold water species are more sensitive organisms than warm water species (see section 1.6 of this chapter for further details). In addition, the ability of these streams to recover after an impact is lower because of the small number of areas which could supply new inhabitants and possible problems for new inhabitants moving into a stream following stress. For example, if the trout in Nira Creek were destroyed, recolonization would have to originate from Nip Creek, which is approximately 41 stream kilometers away, unless Nira Creek were artificially restocked.

Warm headwater streams are also very sensitive to stress, although slightly less so than their cold water counterparts. Greater diversity and functional redundancy is found in these streams. Furthermore, warm water species are generally more tolerant of stress than are cold water species.

Mid-reach streams and the Kawishiwi River are approximately equal in resistance to stress. If third and fourth order streams were separated, third order streams might be considered more sensitive than fourth order streams, on the basis of the lower expected discharge in third order streams. All mid-reach streams and the Kawishiwi River have high functional redundancy. In other words, within each trophic level there are many species available to fill the void if the most sensitive species were lost because of a low level stress. There are also many areas available to supply organisms for recolonization of affected areas once a stress has been removed. Greater dilution and/or flushing is available in the higher order streams, and they are less severely affected by drought.

1.4.4.3 Lake Sensitivity--There are few differences in sensitivity of lakes among the various watersheds. However, there is a difference in sensitivity between the two types of lakes: meso-eutrophic lakes and oligotrophic lakes (Table 17). The difference is primarily the result of the relative amounts of TOC and buffering capacity, since the indigenous species are generally similar in both types of lakes.

The meso-eutrophic lakes are less sensitive to stress, overall, than the oligotrophic lakes. The high TOC content of the meso-eutrophic lakes (10-40 mg/1) is likely to reduce the effect of some heavy metals by complexation (see Volume 3-Chapter 4 for further details). Because of the large number of these lakes in the region, there is a large reservoir of organisms available for recolonization.

Oligotrophic lakes have lower levels of TOC (less than 10 ug/1) and are therefore more susceptible to stress from heavy metals, because the metals cannot be complexed as readily in these waters. Furthermore, the presence of trout in most oligotrophic lakes causes them to be more sensitive. Trout are generally more sensitive to stress than warm-water fish and, once eliminated, could only be restored by artificial stocking.

The buffering capacity and the hardness of lakes is not related to the two lake types. Rather, these factors are related to the lake's position in the watershed. Headwater lakes are low in buffering capacity and hardness but these parameters increase in value downstream. Therefore, the organisms in headwater lakes would be more likely to be subject to pH changes and to metals such as nickel, whose toxicity is affected by hardness.

1.4.4.4 <u>Seasonal Variations in Sensitivity</u> —The sensitivity of aquatic ecosystems varies seasonally. Fish are most sensitive to stress as larvae; they are also sensitive to stress during spawning, and may not spawn if stressful conditions exist.

Aquatic insects are most sensitive to stress during emergence. Other critical periods for insects are during molting and at the time eggs hatch. Spawning times for fish and emergence times for Study Area insects (Tables 18 and 19) indicate that aquatic ecosystems are most sensitive to stress during spring, as the majority of fish spawning and emergence of insects occurs at this time. Sensitivity of the ecosystems may decrease somewhat in early summer, although larval fish are still present and some insect emergence is occurring. Ecosystems are less sensitive between early summer and fall as insect emergence decreases and fish mature. Salmonids are the only fish in the Study Area which spawn in the fall. In addition, some insects emerge in late summer and early fall, so fall is probably the third most critical season.

Tables 18 and 19

In winter, aquatic ecosystems are probably least susceptible to stress, assuming that problems of oxygen depletion do not occur as a result of stresses. Some stoneflies emerge in winter, but aquatic invertebrates, as well as fish, tend to be inactive in winter, and are therefore less likely to be affected by acute stresses.

1.5 PHILOSOPHY AND METHOD OF APPROACH TO IMPACT ASSESSMENT

The responses of aquatic ecosystems to stresses as a result of mineral development are poorly understood. Most of the data available are the result of field and laboratory bioassay experiments on individual species under controlled conditions. The application of these data to the assessment of impacts is difficult because any mining development will entail multiple impacts on a large group of species. Where effects of actual mining operations have been observed, the

Table 18 Life history, habitat preference, pollution tolerance and frequency of occurrence by stream order of the dominant invertebrate taxa of functional groups 1-5 in the Study Area. Shaded areas indicate preferences, partially shaded areas indicate differences within genera.

1. SHREDDERS (dead plants)	EMERGENCE				HABITAT						POLLUTION TOLLERANCE: ORGANICS			POLLUTION TOLLERANCE: pH					POLLUTION TOLLERANCE: ALKALINITY			STREAM ORDER				
	WINTER	SPRING	SUMMER	FALL	RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	TOLERANT	FACULATIVE	INTOLERANT	<5	5-6	6-7	7-8	>8	<50	50-100	>100	1	2	3	4	5
Amphinemura spp.																						▬	▬	▬		
A. linda		■	■			■							■													
A. delosa		■	■										■													
Shipsa spp.																							▬	▬	▬	▬
S. rotunda		■	■		■								■													
Leuctra spp.																						▬	▬	▬		
L. ferruginea		■	■		■				■				■													
L. tennuis		■	■		■				■				■													
Allocapnia spp.	■	■																					▬	▬	▬	
A. minima	■																									
A. pygmaea	■	■			■																					
Paracapnia spp.																						▬	▬	▬	▬	▬
P. angulata		■											■													
P. opis		■	■																							
Taeniopteryx spp.													■											▬	▬	
T. burksi		■			■																					
T. nivalis	■	■											■													
Grammotaulius spp.			■		■	■																▬	▬	▬	▬	
Hydatophylax spp.																							▬	▬		
H. argus		■			■	■																				
Limnephilus spp.		■	■	■	■	■							■										▬			▬
Nemotaulius spp.																										
N. hostilis				■	■																	▬	▬	▬		
Neophylax spp.				■	■																			▬		
N. nacatus					■																					
Platycentropus spp.				■								■				■	■		■			▬	▬		▬	
Pycnopsyche spp.			■	■												■			■			▬		▬	▬	▬
P. guttifer					■																					
P. scabripennis					■											■			■	■						
Glyphopsyche spp.																■			■			▬	▬	▬	▬	
G. irrorata	■	■		■	■																					
Pseudostenophylax spp.			■		■																					
Frensia spp.				■	■																					
Goera spp.		■	■		■								■													
Lepidostoma spp.		■	■		■	■						■				■	■		■			▬	▬	▬	▬	▬
Tipula spp.		■	■	■	■								■	■	■	■			■	■		▬	▬	▬	▬	
Pedicia spp.		■	■		■																					
Erioptera spp.		■	■		■																					
Endochironomus spp.			■		■	■																▬	▬	▬	▬	▬

Table 18 continued

2. SHREDDERS (live plants)	EMERGENCE				HABITAT						POLLUTION TOLLERANCE: ORGANICS			POLLUTION TOLLERANCE: pH					POLLUTION TOLLERANCE: ALKALINITY			STREAM ORDER				
	WINTER	SPRING	SUMMER	FALL	RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	TOLERANT	FACULATIVE	INTOLERANT	<5	5-6	6-7	7-8	>8	<50	50-100	>100	1	2	3	4	5
Ptilistomis spp.		■	■			■								■	■	■	■		■	■		■	■	■	■	
Triaenodes spp.					■	■							■			■	■		■			■		■	■	
T. marginata			■																							
T. tarda		■	■																							
T. injusta		■	■													■	■	■	■	■	■					
Cricotopus spp.						■										■	■	■	■	■		■	■	■	■	■
Polypedilum spp.			■		■	■								■	■	■	■	■	■	■	■	■	■	■	■	■
Nymphula spp.			■									■	■													

Table 18 continued

3. GATHERERS	EMERGENCE: WINTER	SPRING	SUMMER	FALL	HABITAT: RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	POLLUTION TOLLERANCE: ORGANICS: TOLERANT	FACULATIVE	INTOLERANT	pH: <5	5-6	6-7	7-8	>8	ALKALINITY: <50	50-100	>100	STREAM ORDER: 1	2	3	4	5
E. deficiens			■																							
E. invaria		■			■								■													
E. needhami		■	■										■													
E. rotunda		■	■		■																					
E. serrata																										
E. simplex			■										■													
E. sordida			■										■													
E. subvaria		■	■		■								■													
E. temporalis			■			■					■					■	■	■	■							
E. versimilis																										
E. minimella																										
E. frisoni			■																							
~~E. robusta~~																										
Tricorythodes spp.			■			■						■					■	■	■	■	■			■	■	
~~Caenis spp.~~			■			■					■	■	■	■	■	■	■	■	■	■	■		■	■	■	■
Hexagenia spp.																						■	■	■	■	
H. limbata			■			■	■					■	■			■	■		■							
Antocha spp.		■	■		■												■	■		■	■					■
Lasiodiamesa spp.																						■		■	■	■
Eukiefferiella spp.	■	■			■							■				■	■	■	■	■		■	■	■	■	■
Parametrecnemus spp.			■		■	■																■	■	■	■	■
Rheocricotopus spp.	■	■	■	■	■								■										■	■	■	
Heterotrissocladius spp.	■	■	■	■	■																	■	■	■	■	
Micropsectra spp.			■	■	■	■						■	■									■	■	■	■	■
Zevrelia spp.			■	■	■											■			■			■	■			
Chironomus spp.	■	■	■	■	■	■					■	■	■	■	■	■	■	■	■	■	■	■	■	■	■	■
Dicrotendipes spp.	■	■	■	■	■	■					■	■	■			■	■	■	■	■	■	■	■	■	■	■
Glyptotendipes spp.	■	■	■	■	■	■					■	■	■			■	■	■	■	■	■	■	■	■	■	■
Macronychus spp.																						■	■	■	■	■
M. glabratus					■	■							■		■	■	■	■	■	■	■	■	■	■	■	■
~~Optioservus spp.~~		■	■	■	■							■														
O. fastiditus												■						■		■	■					
~~O. trivittatus~~													■													
O. ovalis					■												■	■	■	■	■					
Stenelmis spp.																								■	■	■
S. crenata												■	■		■	■	■	■	■	■	■					

Table 18 continued

	EMERGENCE				HABITAT						POLLUTION TOLLERANCE: ORGANICS			POLLUTION TOLLERANCE: pH					POLLUTION TOLLERANCE: ALKALINITY			STREAM ORDER				
3. GATHERERS	WINTER	SPRING	SUMMER	FALL	RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	TOLERANT	FACULATIVE	INTOLERANT	<5	5-6	6-7	7-8	>8	<50	50-100	>100	1	2	3	4	5
Siphlonurus spp.						■	■					■										■		■	■	■
S. alternatus			■																							
S. marshalli		■														■			■							
Arthroplea spp.																										
A. bipunctata		■				■																				
Rhithrogena spp.		■	■		■				■				■													
Stenacron spp.																										
S. candidum		■	■		■	■																■	■	■	■	■
S. interpunctatum			■		■	■						■	■		■	■	■	■	■	■	■					
S. minnetonka		■	■		■	■						■														
Stenonema spp.													■									■	■	■	■	■
S. annexum		■			■																					
S. exiguum			■		■			■																		
S. fuscum			■		■							■	■													
S. pulchellum			■		■								■													
S. quinquespinum		■										■														
S. Rubrum			■		■							■	■													
S. smithae			■		■							■	■													
S. terminatum		■	■									■	■													
S. tripunctatum		■	■			■						■	■													
S. femoratum		■	■									■	■													
S. integrum		■	■		■								■													
Baetis gr. spp.			■		■	■									■	■	■	■	■	■	■	■	■	■	■	■
B. brunneicolor			■		■																					
B. hageni		■			■																					
B. phyllis		■																								
B. pygmaes		■	■	■	■	■																				
B. vagans		■	■		■																					
B. flavistriga					■																					
B. frondalis		■	■																							
Paraleptophlebia gr. spp.		■	■	■	■								■		■				■			■	■	■	■	■
P. debilis			■	■																						
P. mollis			■	■																						
P. volitans		■	■	■												■			■							
P. praepidita		■	■	■													■		■							
P. guttata		■	■	■													■				■					
Ephemerella spp.		■	■	■																		■	■	■	■	■
E. attenuata			■										■													
E. bicolor			■		■								■				■			■						

Table 18 continued

	EMERGENCE				HABITAT						POLLUTION TOLLERANCE: ORGANICS			POLLUTION TOLLERANCE: pH					POLLUTION TOLLERANCE: ALKALINITY			STREAM ORDER				
4. FILTER-FEEDERS	WINTER	SPRING	SUMMER	FALL	RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	TOLERANT	FACULATIVE	INTOLERANT	<5	5-6	6-7	7-8	>8	<50	50-100	>100	1	2	3	4	5
Chimarra spp.		■	■		■								■									■	■	■	■	■
C. feria		■	■		■								■				■		■							
C. obscura		■	■	■	■							■					■	■		■	■					
C. socia		■			■								■			■	■	■	■	■	■					
C. aterrima		■			■								■			■			■							
Dolophiloides spp.																							■	■		
D. distinctus	■				■								■													
Lype spp.					■																					
L. diversa		■	■																							
Psychomyia spp.																										
P. flavida		■	■		■																					
Neureclipsis spp.		■	■		■						■		■		■	■	■	■	■	■	■			■	■	■
Hydropsyche gr. spp.												■										■	■	■	■	■
H. betteni		■	■	■	■										■	■	■	■	■	■	■					
H. cuanis		■	■		■													■		■						
H. orris		■	■	■	■							■				■	■		■	■	■					
H. simulans		■	■	■									■			■	■	■	■	■						
H. slossonae		■	■		■																					
H. bifida		■	■	■								■					■	■		■	■					
Prosimulium spp.		■	■		■								■													
Eusimilium spp.		■	■		■								■													
Simulium spp.		■	■		■						■	■	■													
Cnephia spp.		■	■		■								■													

Table 18 continued

5. SCRAPERS	EMERGENCE: WINTER	SPRING	SUMMER	FALL	HABITAT: RIFFLE	POOL	MUD, SILT	SAND	GRAVEL	ROCK	POLLUTION TOLLERANCE – ORGANICS: TOLERANT	FACULATIVE	INTOLERANT	pH: <5	5-6	6-7	7-8	>8	ALKALINITY: <50	50-100	>100	STREAM ORDER: 1	2	3	4	5
Hastaperla spp.																										
H. brevis		■											■													
Epeorus spp.																								■	■	
Heptagenia spp.																						■	■	■	■	■
H. hebe			■	■	■								■													
H. flavenscens		■	■		■							■														
Pseudocloeon spp.		■	■		■	■										■	■	■	■	■		■	■	■	■	■
P. anoka			■	■	■																					
P. carolina			■		■							■														
P. cingulatum			■		■							■														
P. dubium			■	■	■							■														
P. parvulum			■		■							■														
Choroterpes spp.																								■	■	■
C. basalis			■	■	■	■																				
Agapetus spp.		■	■		■												■		■			■	■	■	■	■
Glossosoma spp.		■	■										■									■		■		

Table 19 Food habits, spawning behavior, and movement of the fish species found in the Study Area. Preferred conditions shaded.

	FOOD				SPAWNING: TIME				SPAWNING: SUBSTRATE								SPAWNING: CURRENT					SPAWNING: TEMPERATURE °F					SPAWNING: HABIT					MOTION		
SPECIE	ALGAE	ZOOPLANKTON	AQUATIC INSECTS	FISH	SPRING	SUMMER	FALL	WINTER	CLAY, MARL, MUD, SILT	SAND	GRAVEL	RUBBLE, BOULDER	MIDWATER	ORGANIC DEBRIS	VEGETATION, ALGAE	UNDERSIDE-ROCKS, ETC.	INDIFFERENT	STILL	WEAK	MODERATE	FAST	35-45	45-55	55-65	65-75	75-85	HIDERS	SCATTERERS-GRAVEL	SCATTERERS-VEGETATION	SUBSTRATE NESTERS	BURROW NESTERS	SEDENTARY	SEMI-MOBILE	MOBILE
N. redbelly dace	X				X	X									X			X											X			X		
Brassy minnow	X				X	X			X									X											X			X		
Fathead minnow	X					X										X		X						X							X	X		
Pugnose shiner		X			X																											X		
Blackchin shiner		X			X	X												X														X		
Pearl dace		X			X					X	X								X	X				X			X					X		
Rainbow smelt			X		X						X										X		X					X						X
Central mudminnow			X		X										X				X					X					X			X		
Finescale dace			X		X																											X		
Longnose dace			X		X	X					X								X	X			X					X				X		
Creek chub			X		X	X					X									X	X			X			X					X		
Lake chub			X		X							X							X	X								X					X	
Common shiner			X		X	X				X	X									X				X						X		X		
Spottail shiner			X		X	X									X			X	X										X			X		
Pumpkinseed			X		X	X			X	X	X	X							X						X	X				X		X		
White sucker			X		X	X					X									X			X					X					X	
Tadpole madtom			X			X												X													X	X		
Iowa darter			X		X									X				X											X			X		
Johnny darter			X		X											X		X	X	X											X	X		
Log perch			X			X				X								X	X	X									X			X		
Blacknose dace			X		X	X					X										X				X			X				X		
Brook trout			X	X			X				X								X	X	X	X					X							X
Lake herring			X	X			X	X			X	X						X	X	X		X						X						X
Lake whitefish			X	X			X	X		X	X	X						X		X		X						X						X
Trout-perch			X	X	X						X	X								X													X	
Rock bass			X	X	X	X					X	X						X	X					X	X					X		X		
Smallmouth bass			X	X	X	X					X								X					X						X		X		
Black crappie			X	X	X	X				X	X							X							X			X					X	
Yellow perch			X	X	X						X							X					X									X		
Mottled sculpin			X	X	X											X			X				X								X	X		
Slimy sculpin			X	X	X											X			X			X									X	X		
Lake trout				X			X					X								X		X					X							X
Northern pike				X	X										X			X	X			X	X						X			X		
Burbot				X				X		X	X							X	X	X		X						X				X		
Largemouth bass				X	X	X				X	X							X						X						X			X	
Walleye				X	X						X	X			X				X	X			X					X					X	
Golden shiner	X	X	X	X		X									X			X								X			X				X	
Emerald shiner	X	X	X	X	X	X							X																				X	
Shorthead redhorse	X	X	X	X	X						X									X			X					X					X	
Blacknose shiner	X	X	X	X	X	X				X								X											X			X		
Bluntnose minnow	X	X	X	X	X	X										X		X							X						X	X		
Brook stickleback	X	X	X	X	X										X			X					X	X						X		X		
Bluegill	X	X	X	X	X	X				X	X							X							X					X		X		
Black bullhead	X	X	X	X	X	X				X								X							X							X		
Yellow bullhead	X	X	X	X	X	X				X					X			X							X							X		
Brown bullhead	X	X	X	X	X	X				X					X			X							X							X		
Channel catfish	X	X	X	X	X	X				X					X			X									X						X	

impacts have been difficult to assess because of the lack of control areas, poor baseline data on natural variability and varying physical and chemical conditions. Only in situations where impact has been severe have the effects been clearly documented.

In order to assess the impact of copper-nickel development on aquatic ecosystems in northeast Minnesota several studies were undertaken. First field and laboratory bioassay studies were conducted to determine the effects of water quality in the Study Area on heavy metal toxicity. In addition, literature on the response of aquatic organisms to mining-related stresses was reviewed. This literature review included bioassay studies and field surveys. Portions of the Copper-Nickel aquatic biology data from sites impacted by taconite mining, reported in sections 1.4.1.3 and 1.4.2.3 of this chapter, are also useful in the assessment of mining impact. Therefore, the approach was to apply results of Regional Study analyses (Johnson and Williams 1978) and in other cases to rely on literature data in order to predict the impact of copper-nickel development on the different aquatic ecosystems in the Study Area. Because information is not available on the response of many of the aquatic organisms found in the Study Area, the prediction of impacts must be general. Also, because the effect of changes in one trophic level on another trophic level are not yet understood, prediction of indirect impacts through the food chain are based on judgemental evaluations.

Aquatic ecosystems demonstrate their sensitivity to stress at the species and community levels. Stress may cause modification in the reproductive success, growth rates and mortality rates of individual species. Changes which occur to individual species are further reflected in community level changes.

Aquatic communities respond to stresses of any kind through changes in their structure and function. These changes are of three general types: 1) changes in the species composition (species present); 2) changes in species diversity; and 3) changes in productivity. The occurrence of one of these changes may or may not cause another kind of change depending on the type of stress, magnitude of the stress, and the interactions in the system. The significance of each of these changes must be analyzed in terms of the ability to measure changes, the interactions of components within the system, and the unique qualities of the ecosystem in relation to other proximate ecosystems. Assessment of impact also must be considered in relation to natural variability in the ecosystem. Changes in the species present or their relative abundances may result from stresses which decrease the ability of a species to survive under a new set of environmental conditions. These new conditions may affect a species directly as mentioned earlier or indirectly by favoring a competitor or predator species.

The diversity of aquatic communities is a measure of both the number of species in the community and the number of individuals of each species. Changes in diversity may result from several causes. Decreases in diversity result from the loss of species or increases in the relative abundances of one or more species in the community. Increases in diversity result from the addition of species to the community or changes which cause organisms in the community to become more evenly distributed among the species.

Some authors have interpreted decreases in diversity as an indication of pollution and stress on aquatic systems (Cairns et al. 1972). Others see high diversity as reflecting community stability and ability to resist stress (Margulef 1963). However, many question this interpretation (Archibald 1972) and cite instances of low diversity in natural, undisturbed situations. In the Copper-Nickel Study diversity indices were calculated as one parameter that can reflect functional changes in the system. However, the dynamics of ecosystems are not understood well enough to predict the direction of change of diversity as a result of a particular stress. For example, diatoms in a Study Area stream affected by taconite mining and copper-nickel leachate are more diverse than those in other streams in the Study Area.

Productivity changes may occur at any trophic level because of environmental stress. Decreases in productivity at one level ultimately affect the productivity at every higher trophic level. These changes also affect the rates and amount of allochthonous and autochthonous material which is cycled within or through the ecosystem.

The recovery of an aquatic ecosystem following stress is related to several factors: 1) the degree of initial impact; 2) the presence of residual toxicants; 3) the presence of adjacent undamaged areas to provide recolonizing organisms; 4) the ability of these organisms to recolonize; and 5) the season. In general, when no residual toxicants remain, aquatic ecosystems recover rapidly. The lowest trophic levels such as periphyton or phytoplankton recover most rapidly, followed by organisms in higher trophic levels.

As aquatic ecoystems are subjected to increasing levels of stress, they begin to respond in one or more of the ways described above. When stress is at a low level, changes in aquatic communities are subtle and easily masked by natural variability. As the level of stress increases, the effects become qualitatively more obvious, and finally they become measurable. The question of when biological change becomes "significant" has been a controversial subject among biologists for some time. Various definitions of "significant impact" have been proposed. Eberhardt (1976) describes significant impacts as follows: "significant ecological damage ensues when there is a reduction of the productivity of an ecosystem in terms of qualities perceived as desirable by man." Others such as Cummins (1976) consider any change in the structure and/or function of aquatic communities to be significant. In an attempt to quantify "significant," Cairns (1967) considered that an aquatic community which does not vary more than 20 percent from the empirically estimated maximum steady state diversity has not changed significantly. Many aquatic biology sampling programs are designed to detect a 25 percent change in the parameter which is being studied (e.g. Dickson et al. 1971). Although a 25 percent change may be significant, it is rarely stated why the investigator chose this detection level.

The conclusion of participants in a workshop entitled "The Biological Significance of Environmental Impacts" (Sharma et al. 1976) was that: "An impact is significant if it results in a change that is measurable in a well designed sampling program, and if it persists or is expected to persist more than several years." This definition assumes that ecosystems have high natural variability and, therefore, only large changes can be detected. Data collected by the Regional Copper-Nickel Study demonstrated this type of high variability in all biological parameters measured. Therefore, any change over and above this variability which could be detected in the future would indicate a significant change had occurred. As a result, the above definition of significant impact will be used in all further discussions of aquatic biological impact.

The second criteria of "significant impact" is that any change persists for several years. In general, aquatic ecosystems recover rapidly following the removal of a stress of no residual stress remains. Recovery is also dependent on the availability for recolonization of organisms in adjacent unstressed areas and on the ability of these organisms to move between habitats.

The mechanisms of recolonization include the upstream migration of stream invertebrates (instream and adult flights), downstream drift of stream invertebrates, flights of adult insects between water bodies and the movement of fish between systems. Reproduction by organisms surviving a stress is also a method for the recovery of aquatic communities and potentially may result in a population less susceptible to the given stress.

In most cases, recovery from localized degradation occurs quickly and completely following mitigationof stress. Cairns et al. (1971) found that recovery was rapid when no residual toxicants remain in the ecosystem and there are areas to provide organimsms for recolonization. However, recovery is slow and rarely complete while pollutants at a sufficient concentration remain in the system. Jones (1940, 1958) reported little recovery of aquatic biota 35 years after lead mining in Wales ceased, because of continued runoff from the mine site. Because lakes often trap pollutants and recycle them, recovery of lake ecosystems can be much slower than streams which tend to flush themselves during high flow periods. Shagawa Lake is an example of a lake where a sewage effluent input has been reduced, but the continual recvycling of excess nutrients has slowed the recovery of the original biological communities. Lakes also tend to be more isolated from each other than streams, which causes difficulty in the movement of recolonizing organisms. For instance, Kennedy (1955) observed recolonization in less than three months in a stream which had been dry for three years. larimore et al. (1959) felt that the versatility of stream organisms, because of their adaptations to fluctuating environmental conditions and life cycles, accounted for rapid stream recovery.

Within the Study Area the aquatic ecosystems are largely inter-connected or in close proximity to one another. Therefore, there are many areas which can provide organisms for recolonization and the organisms should be able to easily move between systems. As one moves upstream in any watershed, recolonization becomes slower both because there are fewer organisms upstream and there are fewer equivalent areas. In the coldwater headwater areas, recolonization becomes a problem because these areas are rare and isolated (see section 1.4.1.2 of this chapter).

Several types of stress are likely to affect aquatic ecosystems if copper-nickel development proceeds. Potential stresses include: 1) heavy metals; 2) pH changes; 3) physical alteration of aquatic habitats including direct loss of habitat; and 4) stress from associated secondary development (see Volume 3-Chapter 4). The following sections discuss each of these streases by first discussing results from literature and experimental studies, followed by discussions of sources, impacts and mitigation potential.

The effect of each of these stresses has been studied in the field and laboratory. While the general effects of these stresses are well known, it is difficult to determine levels where measurable changes will begin to occur.

For the purposes of discussion, available data on all stresses were organized in a similar manner. The data compiled from the toxicity bioassays and literature reviews are summarized (Figures 27 to 36) to provide a means of assessing the potential impact of copper-nickel development on the aquatic ecosystems in the Study Area. The summaries of stress response provide a method to determine levels at which impact may begin to occur although detection could be difficult because of the natural variability in aquatic systems.

In summarizing these data it was evident that a great deal of variablility exists in the critical levels from various stress studies. This variability is the result of the different species studied, variable physical and chemical test conditions, lack of control stations in field studies, and various combinations of stress. Also in some cases, a lack of data on levels of hardness, alkalinity, pH, and total organic carbon TOC in waters used in experiments makes it difficult to apply the available data to the assessment of impacts.

For these reasons, the stress charts were constructed to indicate ranges where measured effects have been recorded. For metals (Figures 27 to 34) and ore beneficiation reagents (Figure 35), orders of magnitude were chosen. The entire order of magnitude was shaded whenever an observation was made within that range. The type of organism affected within each magnitude range is indicated on each figure.

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For these reasons, the stress charts were constructed to indicate ranges where measured effects have been recorded. For metals (Figures 27 to 34) and ore beneficiation reagents (Figure 35), orders of magnitude were chosen. The entire order of magnitude was shaded whenever an observation was made within that range. The type of organism affected within each magnitude range is indicated on each figure.

The effects of pH changes were summarized to indicate levels where biological changes have been observed in the field or laboratory (Figure 36). To be conservative where contradictory data exist, the lowest reported level is indicated on the figure.

Because few data exist on the levels of physical stress which produce changes, the response of aquatic ecosystems to physical stresses have been summarized qualitatively (Figure 37). Measurable changes followed by slow recovery (recovery time greater than one year) were considered to be significant impacts.

To determine the level of stress at which significant impact might occur within the Study Area, further summarization was necessary (Figure 26). Again, ranges of stress are indicated as well as the levels where: 1) no change is expected; 2) changes may occur but would not be measurable; 3) measurable changes would be expected only with a long term exposure to the stress; 4) measurable changes would be expected with either long or short term exposure. As any stress moves into a range, the indicated effect is expected to occur.

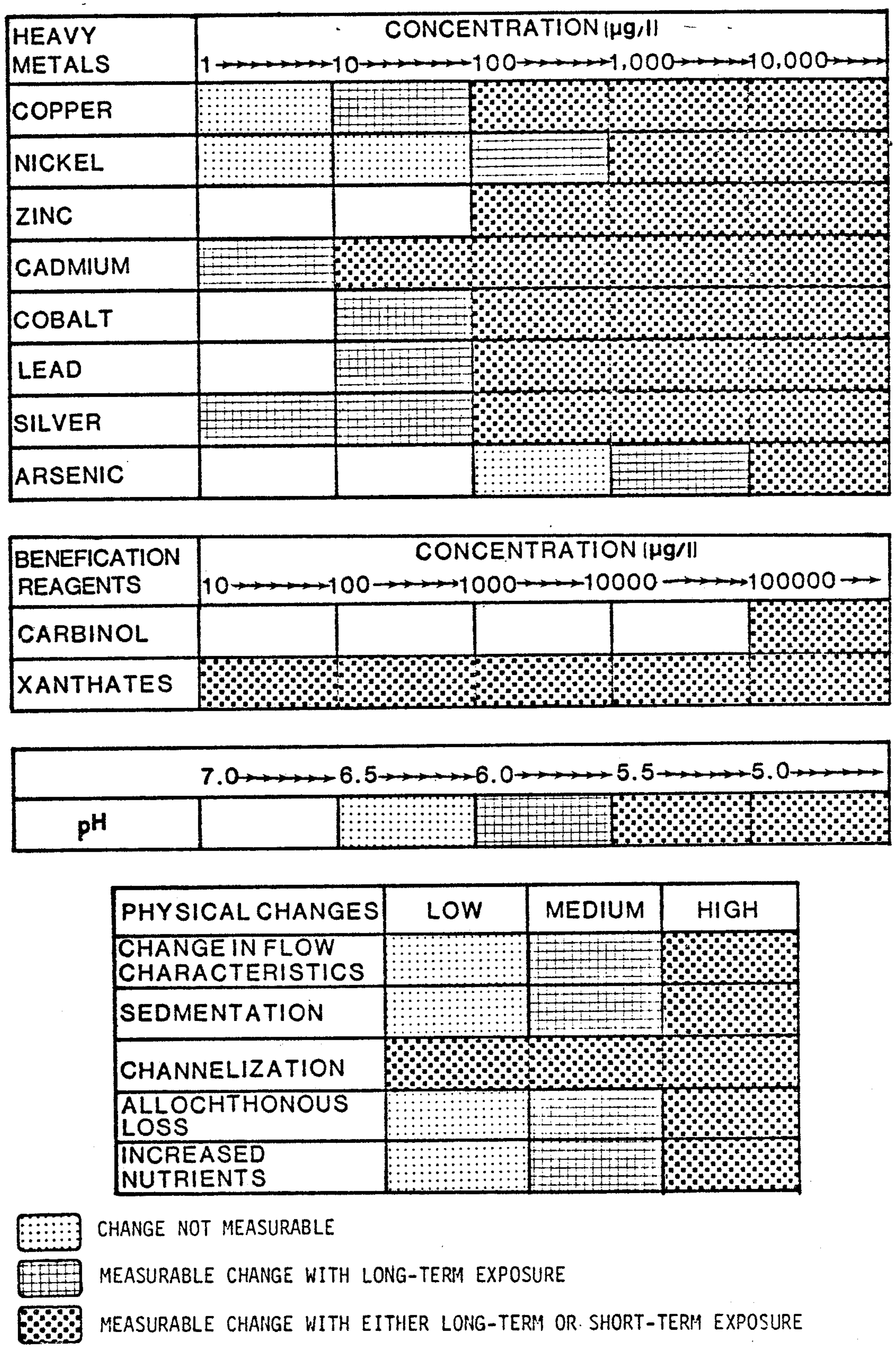
Figure 26

The categories of effects were assigned on the basis of the summary stress charts (Figures 27 to 37). Unmeasurable effects were assigned to ranges where one or two observations of chronic effects have been made and are clearly lower than the majority of observations. Long or short term measurable effects were assigned to those ranges where acute effects have been observed. Long term measurable effects have been assigned to those ranges where no acute effects have been observed but chronic effects have been observed.

Where low level stress exists, changes in fecundity, growth rate, and mortality rate would go undetected because of the high variability in the system. As levels increase, measurable long-term effects within any trophic level may occur if the stress remains long enough. Again, reductions in reproduction, growth, and mortality would occur and could be detected after several years. Changes of this type within one trophic level would, given enough time, affect organisms in other trophic levels. In a high stress situation, direct mortality can be expected in relatively short time periods (e.g. less than one month).

Figure 38 summarizes the various stress factors which could cause significant biological impact and indicates those ecosystems (e.g. headwater streams) where the various stresses would cause significant biological impact. Two categories are presented: short term impact and long term impact. Short term impact occurs when significant biological impact occurs but recivery is rapid (generally within 1-2 years) once the stress is removed. Long term impact occurs when significant biological impact occurs and recovery is slow (greater than 2 years) after the

FIGURE 26 SUMMARY OF STRESS RESPONSES OF AQUATIC ECOSYSTEMS FROM LITERATURE AND BIOASSAY EXPERIMENTS.



stress is removed. In some cases where impact is indicated it is probably unlikely because the particular stress would probably not occur in the ecosystem. For example, it is unlikely that the Kawishiwi River would be channelized for any reason although if it were, significant impact would occur.

Figure 38

The following discussions describe in more detail the areas of impact indicated on Figure 38.

1.6 HEAVY METALS AND BENEFICIATION REAGENTS

1.6.1 Potential Sources of Heavy Metals

The impacts of copper-nickel mining are expected to be greater than those experienced with taconite and iron ore mining. There are a variety of potential sources of heavy metals which could introduce significant quantities of metals into the aquatic environment over a number of years. These metals loadings could result in impacts on the aquatic organisms found in the lakes and streams of the Study Area.

There are four potential major sources of heavy metals from a copper-nickel mining development: waste rock/lean ore stockpile runoff, tailing basins, mine dewatering and a smelter. The models for these different sources are presented in detail in Volume 3-Chapter 4, section 4.6 of this report.

Mine water may be discharged from open pit or underground mines. The quality of this water is influenced by a variety of factors including: source of water, time in contact with mineralized material, etc. Mine dewatering water is, depending upon its quality, likely to be pumped directly into the tailing basin of the fresh water makeup system for the mill.

■ LONG TERM IMPACT
♦ SHORT TERM IMPACT
NO MEASUREABLE IMPACT

			HEAVY METALS	PROCESSING REAGENTS	PH DECREASE	CHANNELIZATION	SEDIMENTATION	FLOW CHANGES	NUTRIENTS	LOSS OF TERR.VEGETATION	TEMPERATURE INCREASE	ENTRAINMENT
RIVERS	HEADWATER	CONSTRUCTION				■	♦	■	■	■		
		OPERATION	■			■	♦	■	■	■	■	■
		POST OPERATION	■			■	♦	■	■	■		
	MID-REACH	CONSTRUCTION				■	♦	■	■	■		
		OPERATION	■			■	♦	■	■	■	■	■
		POST-OPERATION	■			■	♦	■	■	■		
	KAWISHIWI	CONSTRUCTION					♦		■			
		OPERATION	♦				♦		■		■	■
		POST-OPERATION	♦				♦		■			
LAKES	OLIGOTROPHIC	CONSTRUCTION	♦				♦		♦			
		OPERATION	♦				♦	♦	♦			
		POST-OPERATION	♦				♦	♦	♦			
	MESO-EUTROPHIC	CONSTRUCTION	♦				♦		♦			
		OPERATION	♦				♦	♦	♦			
		POST-OPERATION	♦				♦	♦	♦			

FIGURE 38 SUMMARY OF POTENTIAL IMPACTS (ASSUMPTIONS DISCUSSED IN TEXT)

Runoff from waste rock and lean ore stockpiles appears to be the mose important source of potentially toxic waters. Leaching of the sulphide materials in the lean ore pile may release large quantities of heavy metals during the operation. If and when the lean ore pile is processed before the operations shuts down the waste rock pile may be the most important source of heavy metals. The leaching of metals from these two surces if unmitigated could damage the aquatic ecosystem over a large portion of the Study Area. The leaching potential of these stockpiles is discussed in detail in Volume 3-Chapter 4 of this report and in first level reports.

Tailing basins have historically presented significant water quality problems in mining regions and in general had major effects on surrounding streams. The pH of the wter from many of these basins is in the acid range and large quantities of heavy metals are commonly present in the seepage and runoff from the basins. The tailing basin from the copper-nickel operation in northeastern Minnesota is not expected to present problems on the scale of those observed in other parts of the country. The relatively high buffering capacity of the expected tailing material is expected to maintain the tailing basin at a pH near neutral values. The neutral pH of the water in the tailing basin is expected to result in relatively limited metals leaching in the basin (see Volume 3-Chapter 4). If these conditions are present in a copper-nickel tailing basin it is likely that leachate from the waste rock/lean ore piles will be much more important than tailing basin seepage.

The models used in this study (Volume 2-Chapter 4) indicate that the quality which wuld be internally cycled in the smelting operation could be very poor. The quantities of this water are very small in comparison to the quantities represented by the other portions of the operation. The influence of the smelting operation on regional water quality is very dependent on the proportion of water recycled in the system and the potential uses of this water in other parts of the operation. The fate of the smelter waters require further study in conjunction with site specific assessments.

Of all the potential sources the water from the smelting operation is the only one which would definitely stop when the operation ceases. The waters from the open pit mine may not present a problem for many years, in fact until the pit fills or the water level in the pit reaches the level of local ground water tables. The waters from the lean ore stockpile may or may not be eliminated prior to shutdown of the operation, whereas the waste rock stockpile will remain in tact and could potentially present perpetual problems unless effective mitigation procedures are established.

1.6.2 Responses of Aquatic Organisms to Heavy Metals

The toxicity of heavy metals to aquatic organisms is affected by various water quality paramters. Increased toxicity generally occurs as hardness, total organic carbon, (TOC), and dissolved oxygen (DO) decrease and as temperature increases (e.g. EIFAC 1976, and Mount 1968). Exceptions to these trends occur. For instance, Clubb et al. (1975) found that cadmium toxicity to insects was decreased at low DO presumably because of low metablic rates. The effect of pH on heavy metal toxicity is also unclear. Spraque (1964) found that zinc toxicity decreased with increased pH while Mount (1966) observed greater toxicity at high pH.

Within the Study Area, surface waters are generally soft but contain large amounts of organic carbon (TOC). Since these are important factors in determining the toxicity of heavy metals, field and laboratory bioassays were undertaken to determine the effect of surface water quality on the toxicity of copper, nickel and cobalt, the metals of primary concern. The data generated through these tests can therefore be used to evaluate the significance of stress levels presented in Figure 26.

Field bioassays of copper and nickel toxicity were conducted with fish, zooplankton and phytoplankton. Various lakes and streams were chosen to represent different levels of TOC, hardness, alkalinity and pH, factors which influence heavy metal toxicity. Acute tests were run on fathead minnows (*Pimephales promelas*) and *Daphnia pulicaria* on copper, nickel and cobalt individually and in combination. A flow-through apparatus was employed for the fish tests while static tests were performed on zooplankton. Both types of tests were performed in a mobile bioassay laboratory. Chronic tests of the effects of copper and nickel on phytoplankton were conducted in bottles containing lake water suspended in several Study Area lakes.

Laboratory tests were performed to determine the chronic toxicity of copper, nickel, cobalt and beneficiation reagents to fathead minnows and to test the acute toxicity of copper-nickel leachates. Further testing of phytoplankton productivity in laboratory chemostats was done to substantiate the results of the insitu phytoplankton bioassays.

1.6.2.1 Toxicity of Metals in Study Area Waters--Based on the results of the bioassay experiments on fish, zooplankton and phytoplankton it appears that the acute toxicity of copper is lower in Study Area waters than has been reported in the literature and nickel toxicity is about the same as reported in the literature. The toxicity of copper was affected by TOC and the toxicity of nickel was affected by hardness and to a lesser extent by TOC. Predictive models relating copper and nickel toxicity to these parameters were developed (Lind et al. 1978).

Phytoplankton productivity was affected at 50 to 100 ug Cu/l and 100 ug Ni/l in both Birch and Greenwood Lakes (Gerhart and Davis 1978). The effects observed were sometimes stimulatory and sometimes toxic. At concentrations above these levels toxic effects always occurred. Individual taxa varied in their responses to the metals. Laboratory bioassays indicated that phytoplankton in clear lakes with low TOC are usually more sensitive to copper and slightly more sensitive to nickel than in lakes with higher TOC. In Clearwater Lake, where median TOC was 7.5 mg/l, copper was severely toxic at 50 ug/l Cu.

Concentrations as low as 5-10 ug/l copper have adversely affected algae (Horne and Goldman 1974). Steeman-Nielson and Laursen (1976) reported that 25 ug/l could be either toxic or stimulatory depending on the receiving water. Nickel concentrations of 4-9 ug/l were found to affect diatom diversity and cause shifts in species composition in natural populations (Patrick et al. 1975) whereas concentrations of 100 ug/l were toxic in laboratory tests.

Table 20 presents the results of acute bioassays of copper toxicity to the fathead minnow. Toxic concentrations (96 hour LC-50) ranged from 88.5 ug Cu/l in Lake Superior water to 2,336 ug Cu/l in Embarrass River water. Copper toxicity was found to be highly correlated with TOC as a result of these tests. Chronic effects of copper began at 13.1 ug Cu/l in Lake Superior water. Chronic tests also indicated that TOC had some affect on long term copper toxicity. Acute copper toxicity to fathead minnows has been reported at 84 ug/l whereas chronic toxicity effects have been reported at 18 ug Cu/l for fathead minnows (Mount and Stephan 1969). Brook trout have been reported sensitive to copper at 5 ug/l.

Table 20

Acute nickel toxicity to fathead minnows was most highly correlated to water hardness according to the results of the Study's bioassays. Toxic concentrations (96 hr LC-50) ranged from 2,916 ug Ni/l in Kawishiwi River water (hardness = 28 ug/l) to 17,678 ug Ni/l in the St. Louis River (hardness = 91 ug/l) (Table 21).

Table 21

Chronic effects were observed at 433.5 ug Ni/l in Lake Superior water. Pickering and Henderson (1966) reported that 4,600 to 9,800 ug Ni/l were acutely toxic to fish in soft water. Chronic effects of nickel have been observed at nickel levels as low as 730 ug/l in water approximately four times as hard as the Lake Superior water used in these studies (Pickering 1974).

Acute copper and nickel toxicity to zooplankton was also inversely correlated to the TOC content of the test water. In addition, hardness was inversely correlated with nickel toxicity. Zooplankton were found to be more sensitive to copper than were fathead minnows: toxic concentrations (48 hr LC-50) ranged from 7.24 ug Cu/l in Lake Superior water to 627 ug Cu/l in the St. Louis River. Nickel was also more toxic (48 hour LC-50) to *Daphnia pulicaria* than to fathead minnows; concentrations ranged from 697 ug Ni/l in South Kawishiwi River water to 3,757 ug Ni/l in St. Louis River water. These values are considerably higher than the 5 ug Ni/l (Baudoin and Scoppa 1974) and 95 ug Ni/l (Biesinger and Christenson 1972) reported as acutely toxic to other species of zooplankton.

Based on these data, it appears that copper is less toxic in the Study Area than reported in the literature on an acute basis because of the high TOC (4.6-38

Table 20 Acute toxicity of copper to the fathead minnow in different surface waters.

Test water	Test date	96-hr LC50 (μg/l)	95% confidence interval on LC 50 (μg/l)	Hardness (mg/l as $CaCO_3$)	Alkalinity (mg/l as $CaCO_3$)	pH	TOC (mg/
Lake Superior	2/14/77	114	33.3	48	44	8.03	3.
Lake Superior	3/7/77	121	33.5	45	44	8.04	3.
Lake Superior	3/21/77	88.5	27.5	46	41	7.98	3.
S. Kawishiwi R.	7/12/77	436	103	30	21	6.82	12
S. Kawishiwi R.	8/1/77	516	111	37	21	7.28	13
St. Louis R.	9/26/77	1586	302	87	20	7.11	36
St. Louis R.	10/17/77	1129	269	73	18	6.94	28
Colby Lake	5/15/78	550	142	84	12	7.07	15
Colby Lake	6/5/78	1001	234	66	12	6.97	34
Embarrass R.	7/3/78	2050	535	117	41	7.29	30
Embarrass R.	7/10/78	2336	548	121	36	7.28	36

Table 21 Acute toxicity of nickel to the fathead minnow in different surface water

Test Water	Test Date	96-hr LC50 (μg/l)	95% confidence interval on LC50 (μg/l)	Hardness (mg/l as $CaCO_3$)	Alkalinity (mg/l as $CaCO_3$)	pH	TOC (mg/l
Lake Superior	2/28/77	5209	1521	45	43	8.05	4.2
Lake Superior	3/14/77	5163	1491	44	42	8.01	3.7
S. Kawishiwi R.	7/18/77	2916	685	29	20	6.50	12
S. Kawishiwi R.	7/25/77	2923	631	28	21	7.00	14
St. Louis R.	10/3/77	12356	2893	77	19	6.99	32*
St. Louis R.	10/10/77	17678	5459	89	20	7.09	33
St. Louis R.	10/25/77	8617	2398	91	19	7.04	30
Colby Lake	5/22/78	5383	1186	86	18	7.16	15

*mean of known values substituted for missing datum.

mg/l) found throughout the Study Area. As TOC increases, the toxicity of copper can be expected to decrease.

The acute toxicity of nickel in the Study Area, on the other hand, may be greater than generally reported in the literature because of the soft water (less than 100 mg/l $CaCO_3$) in the Study Area.

The acute toxicity of cobalt, in Lake Superior water, to fathead minnows and Daphnia pulicaria was 530 ug Co/l and 2,025 ug Co/l respectively. Fathead minnows were affected in chronic tests at 112.5 ug Co/l or approximately one-fifth the acute level. Very few data are available for comparison. Effects have been observed at cobalt levels between 50 and 100 ug/l by Shabolina (1964) and Tabata (1969).

1.6.2.2 Protection Limits--Based on the summarization of available data on the toxicity of heavy metals (Lind et al. 1978) and the Study's bioassay results (Lind and Chatterton 1978), protection limits which should assure continued health of the aquatic communities in the Study Area were developed for impact assessment purposes. Two levels are discussed. First a level which would protect the integrity of the aquatic community during either long (chronic-greater than 1 week) or short term (less than 1 week) exposures. The second level would not provide protection during long term exposures but would be adequate for infrequent short term (acute) events.

As in other discussions the limits have generally been placed at the lower end of the order of magnitude where reported effects have occurred. Where possible the results of the Study'a bioassays have been used to modify published data in arriving at these levels.

Copper--A level of 10 ug Cu/l should assure that no long or short term degradation of aquatic communities should occur. This is the level on Figure 27 at which no measured change would occur. This level was chosen because very few effects on aquatic organisms have been recorded below this level. Also, EPA (1976) has recommended that 0.1 times the 96-hour LC-50 be used as a criteria to protect aquatic life. Applying this factor to the Study's bioassays would result in critical levels greater than 44 ug Cu/l except in Lake Superior where 9 ug Cu/l would be critical. Therefore, 10 ug Cu/l should be adequate to protect aquatic organisms in the Study Area where TOC is generally higher than found in Lake Superior.

Figure 27

For the protection of aquatic life during short term exposures a level of 100 ug Cu/l should not be exceeded. This is below the 96 hr LC-50s recorded in Study Area water and approximately equal to the 96 hr LC-50s recorded in Lake Superior water. Therefore in waters with adequate TOC (greater than 10 mg/l) a level of 100 ug Cu/l should be more than adequate while giving protection to those surface waters with low TOC (less than 5 mg/l). Levels of copper above 100 ug/l can be expected to cause severe long term effects.

Nickel--Nickel concentrations of 100 ug/l should provide long and short term protection to aquatic communities in the Study Area (Figure 28). Only one laboratory study has indicated any effects on aquatic organisms below 100 ug Ni/l (Biesinger and Christenson 1972). The EPA (1976) has recommended 0.01 times the 96 hr LC-50 for protection of aquatic life. Applying this factor to the Study's bioassays results in critical levels of 29 ug/l in the South Kawishiwi River and approximately 100 ug/l in the St. Louis River. However, chronic tests of fathead

FIGURE 27 AQUATIC TOXICOLOGY OF COPPER

STRESS EFFECTS	CONCENTRATION (μg/l) 0.1–1	1–10	10–100	100–1,000	1,000–10,000	10,000–100,000	100,000–
TISSUE ACCUMULATION (EFFECTS UNKNOWN)			AF				
DECREASED EGG LAYING SUCCESS			F	F			
DECREASED HATCHABILITY OF EGGS			F				
DECREASED EGG PRODUCTION			F	F			
DECREASED EMBRYO SURVIVAL			F				
DECREASED WEIGHT / GROWTH RATE		F	ACFIM	A			
DECREASED STANDING CROP / POPULATION		ACF	F				
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)		C	CFM				
LC50 -- 15-25 DAYS			C				
LC50 -- 10-15 DAYS			I	I		I	
LC50 -- 4-10 DAYS			M	FP	FIP		
LC50 -- 1--4 DAYS		C	C	CI	C		

A- ALGAE **C**- CRUSTACEANS **F**- FISH **I**- INSECTS **M**- MOLLUSCS **P**- PROTOZOANS

[shaded box] LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

[blank box] INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE minnows in Lake Superior water indicated that significant effects did not occur until 433.5 ug Ni/l. Also EPA (1976) states that 100 ug Ni/l should protect all aquatic life.

Figure 28

For short-term protection 1,000 ug Ni/l should not be exceeded. The 96 hr LC-50s reported in the literature and from the Study's bioassays all exceed 1000 ug Ni/l with the exception of a South Kawishiwi test with Daphnia pulicaria and a 48 hr LC-50 of 510 ug Ni/l for Daphnia magna (Biesinger and Christenson 1972).

Cobalt--A level of 10 ug Co/l should prevent long and short term changes in aquatic communities in the Study Area (Figure 29). This level was selected as the lower end of the order of magnitude at which chronic effects were observed during the Study's bioassay program. Because few published data are available no EPA recommendations have been made.

Figure 29

Short term protection would be provided if cobalt levels remained below 100 ug Co/l. Mean acute cobalt toxicity to the fathead minnow was 531 ug/l in Lake Superior water.

Zinc--The aquatic community should not be affected at zinc concentrations of 100 ug/l during either short term or long term exposures (Figure 30). Chronic effects on aquatic organisms have generally been reported at concentrations exceeding 100 ug Zn/l. No bioassays using zinc were conducted during this study so the affects of Study Area water quality are unknown.

Figure 30

FIGURE 28 AQUATIC TOXICOLOGY OF NICKEL

STRESS EFFECTS	CONCENTRATION (μg/l) 0.1	1	10	100	1,000	10,000	100,000
TISSUE ACCUMULATION (EFFECTS UNKNOWN)		C					
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS							
DECREASED EGG PRODUCTION			C	I			
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE				A			
DECREASED STANDING CROP / POPULATION		A					
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)						I	
LC50 -- 15-25 DAYS				C			
LC50 -- 10-15 DAYS							
LC50 -- 4-10 DAYS					FIP	FIP	
LC50 -- 1--4 DAYS				C	C P	C	

A- ALGAE **C**- CRUSTACEANS **F**- FISH **I**- INSECTS **M**- MOLLUSCS **P**- PROTOZOANS

(shaded box) LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

(blank box) INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

FIGURE 29 AQUATIC TOXICOLOGY OF COBALT

STRESS EFFECTS	CONCENTRATION (µg/l) 0.1	1	10	100	1,000	10,000	100,000
TISSUE ACCUMULATION (EFFECTS UNKNOWN)							
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS							
DECREASED EGG PRODUCTION			C				
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE		F		A			
DECREASED STANDING CROP / POPULATION							
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)						I	
LC50 -- 15-25 DAYS			C				
LC50 -- 10-15 DAYS							
LC50 -- 4-10 DAYS						IP	
LC50 -- 1--4 DAYS			F		C		

A- ALGAE C- CRUSTACEANS F- FISH I- INSECTS M- MOLLUSCS P- PROTOZOANS

[shaded box] LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

[empty box] INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

FIGURE 30 AQUATIC TOXICOLOGY OF ZINC

STRESS EFFECTS	CONCENTRATION (µg/l) 0.1 → 1	1 → 10	10 → 100	100 → 1,000	1,000 → 10,000	10,000 → 100,000	100,000 →
TISSUE ACCUMULATION (EFFECTS UNKNOWN)		A			C		
DECREASED EGG LAYING SUCCESS				F	F		
DECREASED HATCHABILITY OF EGGS				F	F		
DECREASED EGG PRODUCTION				C			
DECREASED EMBRYO SURVIVAL					F		
DECREASED WEIGHT / GROWTH RATE				A			
DECREASED STANDING CROP / POPULATION				A			
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)				F	F I	I	
LC50 -- 15-25 DAYS				CF	F		
LC50 -- 10-15 DAYS					F		
LC50 -- 4-10 DAYS				M	FMP		
LC50 -- 1--4 DAYS			C	C	C		

A- ALGAE C- CRUSTACEANS F- FISH I- INSECTS M- MOLLUSCS P- PROTOZOANS

LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

The short term exposure protection limit has also been set at 100 ug Zn/1 because various tests have shown zinc to be acutely toxic at levels between 100 and 1,000 ug Zn/1 (Bandouin and Scappa 1974).

Cadmium--Short and long term protection of the aquatic community will be achieved at concentrations not exceeding 1 ug Cd/1 (Figure 31). Short term protection would be provided at 10 ug Cd/1. The EPA (1976) has set a limit of .4 ug/1 for cladocerans and salmonids in soft water and 4.0 ug/1 for less sensitive species. In hard water the limits were raised to 1.2 ug/1 and 12.0 ug/1 respectively.

Figure 31

Lead--A level of lead not exceeding 10 ug/1 should assure protection of aquatic organisms in either long or short term exposures (Figure 32). The EPA (1976) recommends .01 times the 96 hr LC-50. If this factor were applied to data published by Pickering and Henderson (1966) for soft water and fathead minnows, the lower limit would be approximately 56 ug Pb/1, slightly higher than the 10 ug Pb/1 currently recommended.

Figure 32

Short term protection from lead would be provided if lead does not exceed 100 ug/1. Numerous reports of acute effects on aquatic organisms have been noted between 100-1,000 ug Pb/1.

Silver--A lack of data and high variability between published studies does not allow any protection limits to be set for silver at this time (Figure 33). Silver levels as low as 1 ug/1 have been toxic during long term exposures (Nehring 1976).

FIGURE 31 AQUATIC TOXICOLOGY OF CADMIUM

STRESS EFFECTS	CONCENTRATION (µg/1)						
	0.1	1	10	100	1,000	10,000	100,000
TISSUE ACCUMULATION (EFFECTS UNKNOWN)		C	A				
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS			F				
DECREASED EGG PRODUCTION		CF	F	F			
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE		F	AF				
DECREASED STANDING CROP / POPULATION		CF	F				
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)		F	F	AF		I	
LC50 -- 15-25 DAYS		C					
LC50 -- 10-15 DAYS							
LC50 -- 4-10 DAYS				P	I	I	
LC50 -- 1--4 DAYS			C	C	C		

A- ALGAE C- CRUSTACEANS F- FISH I- INSECTS M- MOLLUSCS P- PROTOZOANS

LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

FIGURE 32 AQUATIC TOXICOLOGY OF LEAD

STRESS EFFECTS	CONCENTRATION (μg/l) 0.1	1	10	100	1,000	10,000	100,000
TISSUE ACCUMULATION (EFFECTS UNKNOWN)		C					
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS				F			
DECREASED EGG PRODUCTION				CF			
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE			F	A			
DECREASED STANDING CROP / POPULATION							
MORPHOLOGICAL DEFECTS			F	F			
DECREASED SURVIVAL (VARYING DEGREES)			F	F		I	
LC50 -- 15-25 DAYS				C			
LC50 -- 10-15 DAYS					I		
LC50 -- 4-10 DAYS					FI	FIP	
LC50 -- 1--4 DAYS				C	C		

A- ALGAE C- CRUSTACEANS F- FISH I- INSECTS M- MOLLUSCS P- PROTOZOANS

(shaded box) LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

(empty box) INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

Figure 33

Arsenic--The EPA (1976) has recommended 50 ug As/1 as the maximum arsenic concentration allowable in domestic water (Figure 34). However they admit that not enough data are available to make recommendations to protect aquatic life. Data available in the literature indicate that no changes in the aquatic community occur if levels do not exceed 100 ug As/1.

Figure 34

Metal Combinations--Results of bioassays using two metal combinations of copper, nickel and cobalt indicate that the effect of these metals are additive to slightly more than additive (Table 22). An exception was the toxicity of copper and nickel to fathead minnows which was slightly less than additive. Potency ratios (e.g. ration between copper LC-50 and nickel LC-50) were also calculated using these data (Table 22). Therefore, protection limits for these combinations can be calculated by converting the metals to equivalent copper (for copper-nickel or copper-cobalt combination) or nickel (for nickel-cobalt combination) concentrations and applying the appropriate limits discussed above. This approach was utilized to assess the impacts of metl dishcarges on water quality and is discussed further in the following discussion of heavy metal impacts. It is not clear whether a combination of these three metals could be converted to copper units.

It should also be noted that other heavy metals are probably additive but only limited data are available on this subject.

Table 22

FIGURE 33 AQUATIC TOXICOLOGY OF SILVER

STRESS EFFECTS	CONCENTRATION (µg/l) 0.1→1	1→10	10→100	100→1,000	1,000→10,000	10,000→100,000	100,000→
TISSUE ACCUMULATION (EFFECTS UNKNOWN)	F						
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS							
DECREASED EGG PRODUCTION							
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE			A	A			
DECREASED STANDING CROP / POPULATION							
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)		I	F				
LC50 -- 15-25 DAYS							
LC50 -- 10-15 DAYS		I					
LC50 -- 4-10 DAYS							
LC50 -- 1--4 DAYS					P		

A- ALGAE **C**- CRUSTACEANS **F**- FISH **I**- INSECTS **M**- MOLLUSCS **P**- PROTOZOANS

[shaded box] LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

[empty box] INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE

FIGURE 34 AQUATIC TOXICOLOGY OF ARSENIC

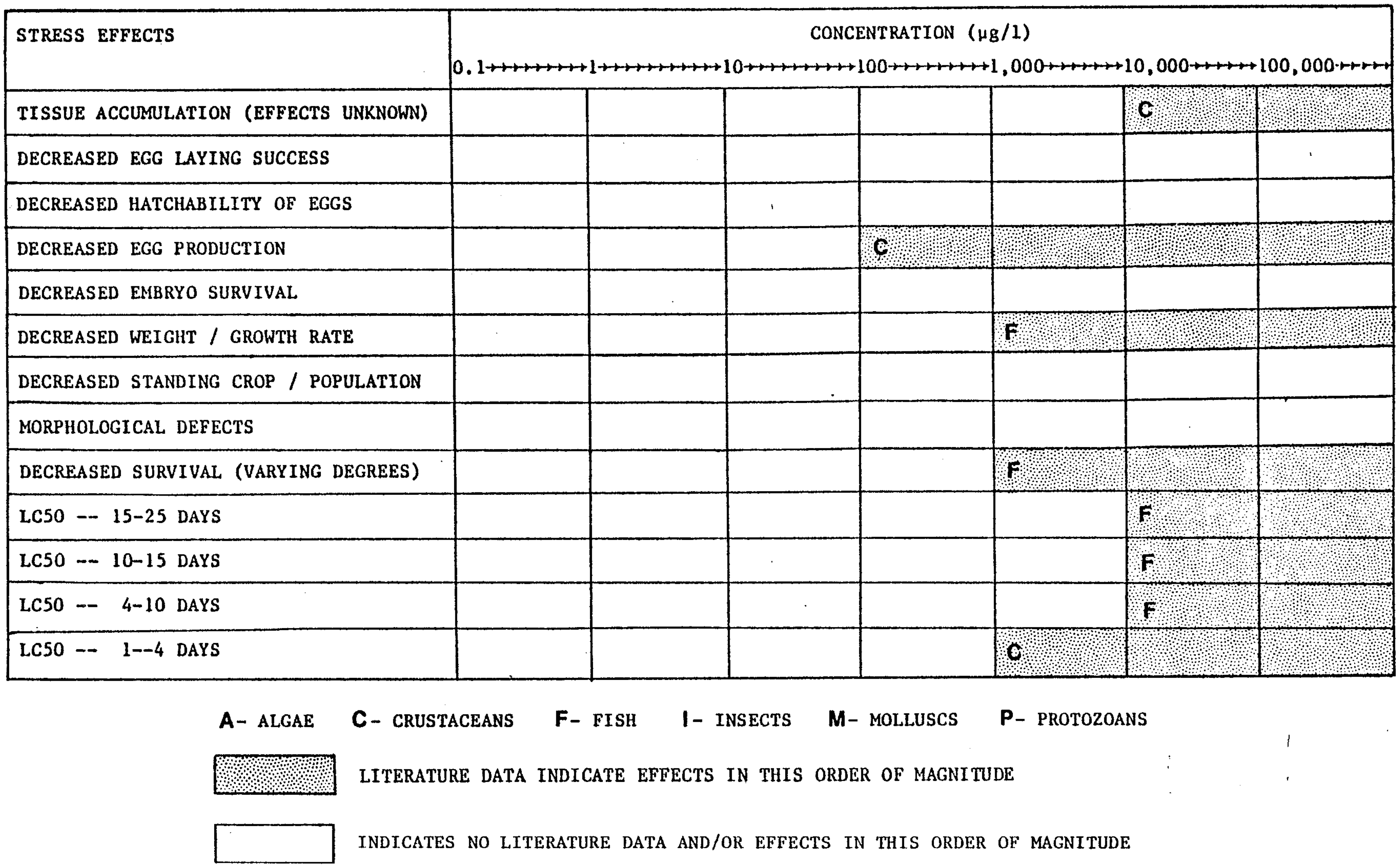


Table 22 Toxicity of Copper-Nickel Mixtures to the Fathead Minnow in Water from Lake Superior and the South Kawishiwi River.

Test Water	Test Date	Cu conc.[a] / Ni conc.	96-hr LC50 as Cu(μg/l)	Mixture LC50 as Cu[c] / weighted mean Cu LC5
Lake Superior	4/4/77	.0255	66.0 (59.4,73.4)[b]	.610
Lake Superior	4/21/77	.0313	46.3 (41.1,52.2)	.428
Lake Superior	5/9/77	.0280	45.5 (38.5,53.9)	.421
So. Kawishiwi R.	8/29.77	.156	229 (165,316)	.478
So. Kawishiwi R.	9/5/77	.124	241 (223,260)	.503

[a]potency ratio = .0209 (Lake Superior) Potency ratio is the ratio of LC-50 for copper to LC-50 for nickel.
.164 (So. Kawishiwi R.)

[b]95% confidence limits on LC50.

c Values of less than 1.0 indicate synergistic effects, i.e. the mixture is more toxic than each metal individually, values of 1.0 indicate direct additivity of toxic effects, values greater than 1.0 indicate that the are less toxic when combined than when presented individually.

** See Lind et al. 1978 for further details of toxicology studies.

Table 22 continued. Toxicity of Copper-Cobalt Mixtures to the Fathead Minnow in Lake Superior Water

Test Date	Cu conc.[a] / Co conc.	96-hr LC50 as Cu(μg/l)	Mixture LC50 as Cu / Weighted Mean Cu LC50
1/30/78	.218	128 (116,141)[b]	1.18
2/6/78	.219	163 (144,186)	1.51
2/27/78	.152	149 (135,165)	1.38

[a]potency ratio = .204

[b]95% confidence limits on LC50.

Toxicity of Nickel-Cobalt Mixtures to the Fathead Minnow in Lake Superior Water

Test Date	Co conc.[a] / Ni conc.	96-hr LC50 as Co (μg/l)	Mixture LC50 as Co / Weighted Mean Co LC50
2/13/78	.125	619(563,680)[b]	1.17
2/20/78	.120	797(696,912)	1.50
3/6/78	.110	602(539,671)	1.13

[a]potency ratio = .102

[b]95% confidence limits on LC50

Table 22 continued Toxicity of Copper-Nickel Mixtures to *Daphnia pulicaria* in Water from Lake Superior and the South Kawishiwi River.

Test Water	Test Date	Cu conc.[a] / Ni conc.	48-hr LC50 as Cu (μg/l)	Mixture LC50 as Cu / weighted mean Cu LC50
Lake Superior	3/30/77	.00697	7.77 (6.96,8.67)[b]	.836
Lake Superior	4/5/77	.00634	8.12 (7.39,8.93)	.874
Lake Superior	4/13/77	.00652	7.18 (6.70,7.69)	.773
S. Kawishiwi R.	8/2/77	.0883	65.0 (61.6,68.5)	1.193
S. Kawishiwi R.	8/9/77	.0628	58.6 (c)	1.077
S. Kawishiwi R.	8/9/77	.0617	60.0 (55.9,64.4)	1.101
S. Kawishiwi R.	8/16/77	.0245	67.0 (c)	1.230
S. Kawishiwi R.	8/23/77	.0308	57.1 (50.5,64.7)	1.049
S. Kawishiwi R.	8/23/77	.0256	68.2 (62.1,75.0)	1.252
S. Kawishiwi R.	8/30/77	.0284	74.7 (66.8,83.6)	1.372

[a] potency ration = .00489 (Lake Superior)
.05521 (S. Kawishiwi R.)

[b] 95% confidence limits on LC50.

[c] confidence limits could not be calculated.

Table 22 continued Toxicity of Copper-Cobalt Mixtures to *Daphnia pulicaria* in Lake Superior Water.

Test Date	Cu conc.[a] / Co conc.	48-hr LC50 as Cu (μg/l)	Mixture LC50 as Cu / Weighted Mean Cu LC50
1/31/78	.00615	8.74 (7.88,9.68)[b]	.940 (.848,1.041)
2/7/78	.00642	13.0 (11.51.4,14.717)	1.40 (1.24,1.58)

[a]potency ratio = .00459

[b]95% confidence limits on LC50

Toxicity of Nickel-Cobalt Mixtures to *Daphnia pulicaria* in Lake Superior Water

Test Date	Ni conc.[a] / Co conc.	48-hr LC50 as Ni(μg/l)	Mixture LC50 as Ni / Weighted Mean Ni LC50
4/12/78	.477	1962 (1760,2187)[b]	1.03
4/17/78	.954	1096 (858,1399)	.577
4/25/78	.949	1648 (961,2828)	.867

[a]potency ratio = .939

[b]95% confidence limits on LC50

Leachate Bioassays--To further deal with the question of the toxicity of heavy metals in combination, leachate bioassays were undertaken using *Daphnia pulicaria* as a test species. These tests also allowed the testing of metal combinations which would be approximations of actual effluents which might occur during or after copper-nickel mining. Leaching of heavy metals from lean ore stockpiles and waste rock piles will be the most significant source of heavy metals from copper-nickel operations. (See Volume 3-Chapter 4 for further details). Currently there are four sources of leachate in the Study Area. These are found at the U.S. Steel exploration pit, INCO exploration site, AMAX exploration shaft site and at Erie Mining's Dunka Pit. These leachates contain elevated levels of several metals, particularly nickel, copper, cobalt and zinc, (see Volume 3-Chapter 4).

The results of the leachate bioassays are presented in Table 23. This table also presents predicted toxicity levels based on the results of metal combination bioassays, and the models relating water quality to toxicity. Only the copper-nickel combination was used in these predictions. In none of these tests was a leachate shown to be more toxic than predicted using the models developed from the field and laboratory bioassays discussed previously. The leachates from Erie Mining and U.S. Steel were extremely toxic. A solution containing 2.2% leachate from Dunka Pit and mixtures containing .1% to 9.95% U.S. Steel leachate were acutely toxic to *Daphnia pulicaria*. In contrast, 100% INCO leachate caused no mortality and 100% AMAX leachate caused 20% or less mortality. This occurred even though the cobalt and zinc concentrations, in the leachate, were not used to derive the predicted toxicity.

Variations in the toxic levels of the various leachates were related to differences in metal levels in the leachates and the different water quality of the dilution water (Table 23).

Table 23

1.6.2.3 Beneficiation Reagents--These reagents are used as collectors and allow the copper-nickel containing particles to attach to bubbles and be removed during the flotation process (see Volume 2-Chapter 3).

Protection Limits-- For various reasons discussed below it is not possible to determine protection limits for benefication reagents.

Carbinols - Only a single test has been conducted on carbinols (Hawley 1972). In this test the 96-hour LC50 for fathead minnows was between 100-1,000 mg/l. More data would be necessary to set protection limits for these chemicals.

Xanthates - Wide ranging results have been obtained during testing of various xanthates (Figure 35). Levels as low as 10 ug/l have been acutely toxic to aquatic organisms (Hawley 1972). The Study's tests of sodium isopropylxanthate were in contrast to Hawley's results. The mean 96-hour LC-50 for fathead minnows was 37.7 whereas for *Daphnia pulicaria* the mean 48-hour LC-50 was 22.0. Because of these discrepancies and lack of chronic tests it is difficult to set any protection limits for these chemicals.

Figure 35

1.6.3 Impacts of Heavy Metals

In streams heavily polluted with heavy metals, periphyton (Wood 1975), macrophytes (Lande 1977), invertebrates (Wood 1975, McIntosh and Bishop 1976), and fish (Hitch and Etnier 1974) have been completely eliminated. However, in most cases,

Table 23 Acute toxicity of copper-nickel leachates to *Daphnia pulicaria*

Leachate	Dilution Water	Test Date	Effect	% Leachate Having Effect	Chemical Conditions in Listed Percentage of Leachate: Cu (µg/l)	Ni (µg/l)	Zn (µg/l)	Co (µg/l)	Mn (µg/l)	Hardness (mg/l as $CaCO_3$)	Organic Carbon (mg/l)	pH	Predicted Cu LC50	Predicted Ni LC50	Cu conc. / pred. Cu LC50 + Ni co[...] / Pred N[...]
Seep 3	----	4/6/77	no mortality	100%	T270[a] F52[b]	T3200 F930	T73	T440	T2740 F840	446	T20 F18	8.30	98	8866	3.1 (total metal) 0.64 (filtered metal)
Seep 3	Oil Discharge	9/14/77	48-hr LC50	2.4% (1.5,3.7)[c]	T46	T720	T17	T40	T238	267	F7.3	8.09	27	6159	1.8
Seep 3	Oil Dishcarge	6/24/78	48-hr LC50	8.0% (5.9,10.9)	T68	T1438	T45	T34	T511	309	T8.1	8.23	31	6833	2.4
U.S. Steel	EM-6	4/6/77	complete mortality	1.7%	T210	T294	T5.3	T13	T171	164	T7.9	8.32	30	4360	7.1
U.S. Steel	S. Kawishiwi R.	4/6/77	complete mortality	1.7%	T210	T294	T3.8	T12	T109	31	T11	7.45	46	1320	4.8
U.S. Steel	Lake Superior	4/26/77	complete mortality	0.1%	T11	T22	T1.3	T1.4	T6.0	45	T3.0	7.99	8.9	1751	1.2
U.S. Steel	EM-6	10/26/77	48-hr LC50	1.0%	T150	T130	T6.7	T11	T67	118	F9.4	8.20	38	3439	4.0
U.S. Steel	S. Kawishiwi R.	10/27/77	48-hr LC50	0.7% (0.6,0.8)	T114	T101	T4.1	T7.7	T77	23	T13	7.04	57	1064	2.1
U.S. Steel	Partridge R	7/25/78	48-hr LC50	19.9% (7.2,13.7)	T408	T643	T12	T33	T244	86	T32	7.80	178	2756	2.5
INCO	----	7/28/77	no mortality	100%	T96	T65	T2.7	T17	--	765	F19	8.31	92	13,002	1.0
INCO	----	11/29/77	no mortality	100%	T23 F9.9	T4500 F3500	T13	T250 F201	--	490	F12	8.35	51	9478	0.93 (total metal) 0.56 (filtered metal)
EM-8	Oil Discharge	6/25/78	48-hr LC50	84% (71,100	T54	T7408	T135	T152	T1439	831	T9.0	8.04	36	13,791	2.0
Minnamax	----	7/26/78	20% mortality	100%	T21	T310	T100	T22	T220	699	T8.3	7.89	32	12,197	0.68

[a]Total concentration [b]Filtered concentration [c]95% confidence limits on LC50

** See Lind et al. 1978 for further details.

FIGURE 35 AQUATIC TOXICOLOGY OF XANTHATES

STRESS EFFECTS	CONCENTRATION (µg/l) 0.1→1	1→10	10→100	100→1,000	1,000→10,000	10,000→100,000	100,000→
TISSUE ACCUMULATION (EFFECTS UNKNOWN)							
DECREASED EGG LAYING SUCCESS							
DECREASED HATCHABILITY OF EGGS							
DECREASED EGG PRODUCTION							
DECREASED EMBRYO SURVIVAL							
DECREASED WEIGHT / GROWTH RATE							
DECREASED STANDING CROP / POPULATION							
MORPHOLOGICAL DEFECTS							
DECREASED SURVIVAL (VARYING DEGREES)					F	F	F
LC50 -- 15-25 DAYS							
LC50 -- 10-15 DAYS							
LC50 -- 4-10 DAYS					CF	CFM	M
LC50 -- 1--4 DAYS							

A- ALGAE C- CRUSTACEANS F- FISH I- INSECTS M- MOLLUSCS P- PROTOZOANS

(shaded box) LITERATURE DATA INDICATE EFFECTS IN THIS ORDER OF MAGNITUDE

(empty box) INDICATES NO LITERATURE DATA AND/OR EFFECTS IN THIS ORDER OF MAGNITUDE heavy metal pollution has been responsible for reduced productivity and diversity and shifts in the dominant organisms at all trophic levels of aquatic ecosystems. For example, studies in the Northwest Miramichi River system of New Brunswick, have shown shifts in diatom species composition, reduced macrophyte growth, reduced invertebrate abundance and diversity, and reduced fish populations and the avoidance of spawning tributaries in stream sections affected by copper and zinc (Besch et al. 1972, Besch and Roberts-Pichette 1970, Cook et al. 1971, Saunders and Sprague 1967, Elson 1974, Pippy and Hare 1969).

Sensitive groups or species are present within each major trophic level and shifts to more tolerant organisms can be expected in the presence of heavy metals. Diatoms are the most sensitive algal group and shfts to blue-green and green algae have been observed (Patrick et al. 1975, Patrick 1978, Wright et al. 1973, Stokes et al. 1973). Among the macrophytes, submerged plants and dicotyledons seem to be the most sensitive to metal pollution (Besch and Roberts-Pichette 1970).

Shifts to benthic invertebrate communities dominated by chironomids and/or oligochaetes have been observed in several studies of heavy metal pollution (Winner et al. 1975, German 1972, EIFAC 1976, Cook et al. 1971, Falk et al. 1973, Yost and Atchison 1972). Molluscs are considered the most sensitive invertebrate group and are quickly eliminated by heavy metal pollution. Field studies have indicated that molluscs rarely occur if heavy metals are present at elevated levels (Cook et al. 1971, Carpenter 1924, 1925, Jones 1940a, 1940b). Among the aquatic insects, mayflies (Ephemeroptera) are the most sensitive (Sprague et al. 1965).

Among the fishes coldwater species (e.g. salmonidae) are considered more sensitive to heavy metals than warm water species although some contradictory data exist. Fishes are able to detect heavy metal gradients and will follow them to avoid lethal conditions.

The toxicity of heavy metals also varies with different life stages of aquatic organisms. For example Bell (1970) reported that aquatic insects are most sensitive at times of molting and emergence. The most sensitive life stages of fish are during spawning and larval stages (McKim 1971).

Some adaptation of aquatic organisms to heavy metals has been reported. Stokes et al. (1972), Harding and Whitton (1976), Say et al. (1977) have reported genetic adaptation to copper, nickel and zinc by algae. Isopods also appear to adapt genetically to copper and lead (Brown 1976). Physiological resistance to zinc but not to cadmium and lead has been observed in fish (Lloyd 1960, Sinley et al. 1974, Sephar 1976, McIntosh and Bishop 1976, Benoit et al. 1976, Holcombe et al. 1976 and 1977).

1.6.3.1 <u>Approach to Assessment</u>--In order to assess the impacts of heavy metals on the aquatic environment a simple system was developed in an attempt to quantify the combined impacts of what are believed to be the fur metals which may be released in the largest quantities, copper, nickel, cobalt and zinc. Utilizing the toxicity data presented in section 1.6.2 relative toxicities were estimated for each of these metals in relation to copper toxicity. Because of the great variability among toxicity data and the differential influences of water chemistry on the toxicities of these metals their relative toxicities were fixed within orders of magnitude. In effect this system normalizes the concentrations of the four metals to toxicologically equivalent copper concentrations. The formula for "copper equivalent units" (CEU) was established, for this purpose, to be:

(copper) + 0.1 (nickel) + (cobalt) + 0.1 (zinc) = CEU

Using this methodology allows one to translate all potential impacts into those which would be expected at a copper concentration which equals the CEU value. In this system the toxicities are assumed to be additive, although it is possible that in reality some may be slightly more or less than additive. This method also assumes equivalent mobility of the various metals which is in fact not the case, therefore, this system represents a relatively conservative picture of the transport and toxicities of potential heavy metal releases.

1.6.3.2 Significance of CEU Values--Table 23a summarizes the impacts of various CEU concentrations on algae, invertebrates and fish. This table was developed using all available data from the literature, and the Study's experiments with Study Area waters and potential leachate mixtures (see section 1.6.2 for details). This table indicates CEU concentrations which would result in impacts on the aquatic ecosystem in the case where one either does or does not assume the influence of water chemistry to have significant effects on toxicity. The influence of water chemistry on toxicity could be an important factor in determining the levels of impact which would result from the release of metals laden waters. Because of the variability of present water quality throughout the region it is important to consider the toxicity of the potential metals releases both with and without the influence of water chemistry.

When evaluating impacts based on information in Table 23a, it is important to consider the "state of art" of toxicology knowledge. The data indicate the CEU concentrations between 5 and 30 ug/1 may result in chronic impacts on aquatic organisms, however, it is not possible to project definite impacts because of the influences of water chemistry on toxicity. CEU levels between 30 and 100 are expected to cause some measureable impact on algal populations, but until levels

COPPER EQUIVALENT UNITS (CEU in ug/liter)	ALGAE	INVERTEBRATES	FISH
10,000-			
2,500-			-Major fish kills and species losses irrespective of WQ
1,000-			
600-		-Major decrease of population size and diversity, indirect loss of fish due to lack of food, irrespective of WQ	
300-	-Acute population and density losses irrespective of WQ		-chronic impact,loss of population irrespective of WQ, acute impacts dependent on WQ
100-	-Chronic impact on species diversity irrespective of WQ	-Acute losses of species and population size, dependant on WQ	
30-	-Potential chronic impact on species diversity dependent on WQ	-Chronic impact, loss of diversity, limited acute effects	-Chronic impacts dependent on WQ
10-			
5-	-No measureable impact irrespective of WQ	-No measureable impact irrespective of WQ	-No measureable impacts irrespective of WQ
2.3-	- - - - - - - - - - - - - - - -	- Background WQ - - - - - -	- - - - - - - - - - - - - - - - - - -
1-			

Table 23a Impacts of copper and/or copper equivalent units on aquatic organisms

WQ- indicates quality of receiving waters near 100 CEU some tolerant species may be uneffected. Concentrations between 100 and 300 CEU may have acute effects on algal populations but it is not possible to project these with present knowledge, whereas above 300 CEU serous acute impacts could be expected irrespectively of water chemistry. Some species of invertebrates are acutely sensitive to CEU levels as low as 100 however impacts can not be projected with certainty unless values exceed 600 when dramatic damage to the invertebrate community would be unavoidable. In these same terms one can not be certain of major damage to fisheries less CEU concentrations exceed 2,500 ug/l however some acute and chronic impacts have been observed at lower concentrations.

It may be useful, at this point, to evaluate two cases of different CEU concentrations to consider both the state of present knowledge and the types of impacts which may be involved. Models presented in Volume 3-Chapter 4 indicate that the CEU concentrations in a direct discharge from a lean ore pile (Model II) would be 8,300 ug/l. A discharge of this quality without any dilution would be acutely toxic to all aquatic organisms. All organisms downstream of this quality of discharge would die or move to other areas to the point in the watershed where the discharge is diluted by a factor of 3 to 4. At that point some species of fish would still survive but a long term exposure of this type would probably eliminate all reproduction in the fish population and few if nay invertebrates would be available as a food source. Further downstream when the dilution factor appraches 14 some species of invertebrates could be expected to survive a short term exposure with a greater number of fish remaining alive and having some supply of food. By the point in the watershed where the lean ore pile runoff is diluted by a factor of 25-30 (300 CEUs) a greater number of fish and invertebrate species would be observed although little algal growth could be expected.

Further downstream as the dilution factor approaches 83 a relatively large number of fish species could be found with several species of invertebrates. Dilution by a watershed approximately 250) would be necessary before it would become difficult or impossible to measure the impacts of a chronic discharge of the quality represented by lean ore pile runoff.

A discharge with the quality indicated in Volume 3-Chapter 4 for the tailing basin would present much less of a problem than the lean ore pile runoff. The CEU concentration in the basin is modeled to be 54 ug/l. In this case a chronic discharge could possible have some impacts on the aquatic ecosystem but it is not possible to project with current knowledge.

1.6.3.3 Impact Assessment--The range of possible metals concentrations in Study Area waters is quite large because of the great variety of water management and mitigation techniques which could be utilized for a given copper-nickel development. The water quality source and impact models presented in Volume 3-Chapter 4 therefore present a large range of possible water quality conditions for the Study Area with development. Because of this broad range of potential conditions it is not possible to project the impact on aquatic resources which may occur as a result of a particular copper-nickel development. It therefore appears that the most useful approach to impact assessment may be to project the types of impacts which could be expected if heavy metals pollution occurs in several different ranges.

Areas which are subjected to CEU concentrations of 2,500 or more ug/l can be expected to experience dramatic losses of fisheries, invertebrates and all other aquatic organisms. These losses would occur in the case of either acute or chronic exposures. Damaged areas could be expected to recover relatively slowly, depending upon location in the watershed, because populations downstream of such a low quality discharge would also be damaged.

Metals pollution at CEU levels in the range of <u>600-2,500 ug/l</u> would probably result in impacts which would be similar to those just mentioned with the only possible difference being the survival of a small population of fish. It is likely that an acute exposure to sech CEU levels would have similar long term impacts to those described above because the long term survival of the fisheries would be limited by a shortage of algal and invertebrate food supplies. A stream or lake with metals concentrations in this range or greater would in effect be dead until such time as the pollution source is eliminated.

Concentrations of metals in the CEU range of <u>300-600 ug/l</u> are likely to have severe impacts on a stream or lake ecosystem. Pollution at these levels are less likely to damage fisheries directly although there would still be indirect effects from the losses of food supplies. Toward the lower end of this range aquatic invertebrates and fish would be able to survive acute exposures but long term exposures would probably result in decreased fish and invertebrate populations as well as the total loss of some of the intolerant species. The long term stability of an ecosystem subject to these levels of pollution as a chronic condition, is probably quite limited as the loss of the less tolerant species may result in changes in a variety of trophic relationships.

In the CEU range of <u>100-300 ug/l</u> it is probably that there will be chronic impacts on the aquatic ecosystem, however short term exposure may or may not result in impacts. It is not possible to project impacts of acute exposures on aqutic organisms because water chemistry can have significant influence on toxicity in this range of concentrations. It is likely that some of the more sensitive species will be lost during acute exposures however these losses would be less noticeable than those which would occur with higher CEU concentrations. Chronic exposures in this range would probably result in decreases in reproductive success of the fish population with eventually noticeable decreases in fishing success.

Acute exposures to metals at CEU concentrations between 30-100 ug/l are unlikely to result in noticeable (measureable in relation to natural variability) changes in the aquatic ecosystem. Chronic exposures in this range may or may not have some impacts on the aquatic system. Although some species do appear to be sensitive to chronic exposures at these levels impacts can not be projected with certainty because of the potential influences of water chemistry on the metals toxicity.

Below 30 ug/l it is possible that some chronic exposures would impact the more sensitive aquatic organisms but it may be quite difficult to measure the changes in the ecosystems because of the great natural variability of populations of these organisms. In Lake Superior water chronic effects on fathead minnows were observed at copper concentrations of 13 ug/l. The importance of such changes in a natural ecosystem are unknown. In some cases these impacts could be significant; however, it is likely that natural variability would mask any "changes" that actually would occur. Again, water chemistry may play such an important role in the toxicity of metals that no impacts would occur in some areas while other areas may experience measurable damage.

Figure 35a is an example of the water quality impacts modeled in section 4.7 of Volume 3-Chapter 4. It is apparent from this graph that in the case of an unmitigated discharge significant impacts could be expected over a large portion of

20 x 10^6 MTPY Open Pit Mine/Mill (Smelter not included)
Operational Phase---Discharge Proportional to Stream Flow

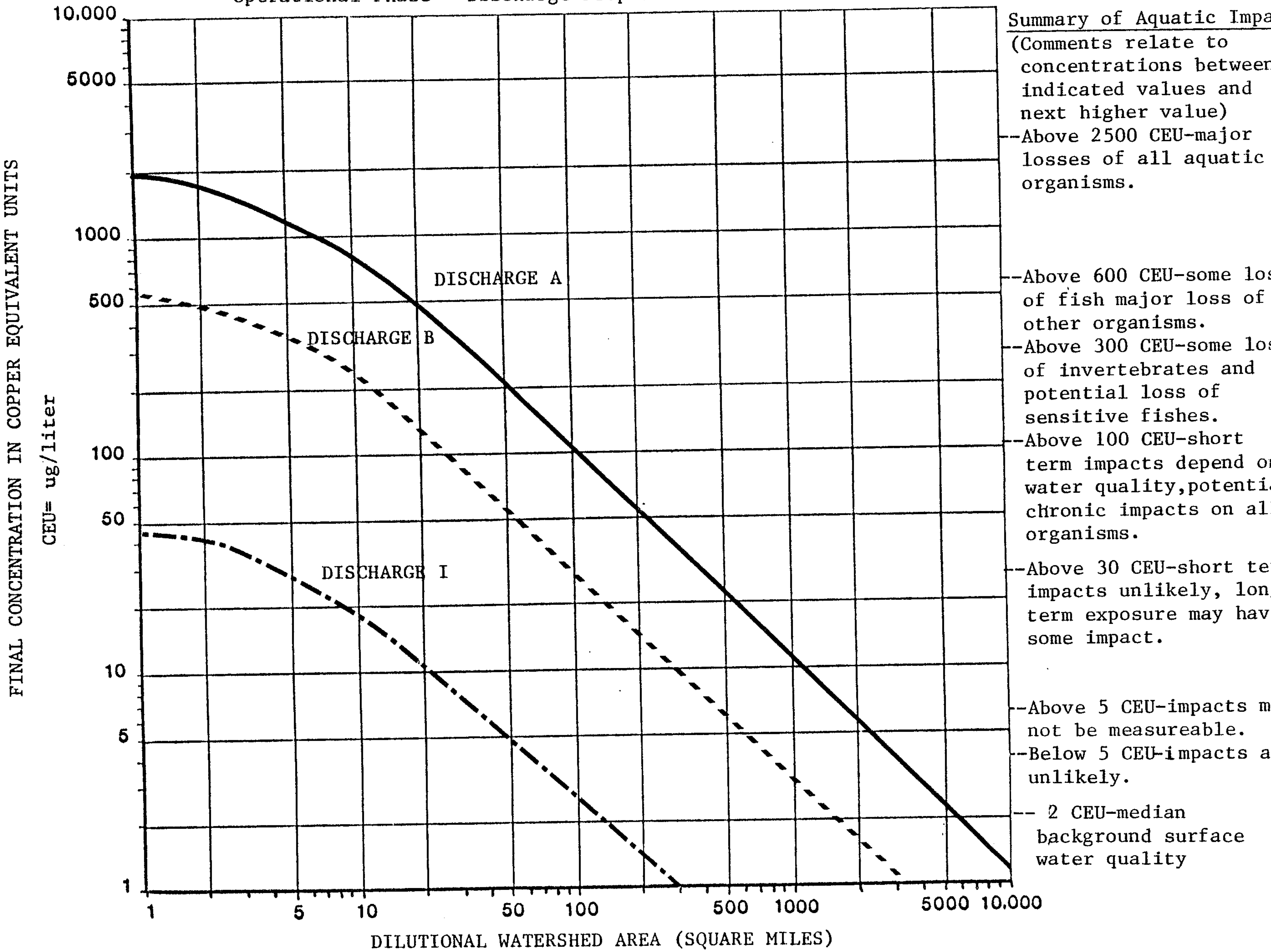


FIGURE 35a. Sample water quality impact model from section 4.7 of Volume 3-Chapter 4.

NOTE: The results presented in this model may vary significantly depending upon hydrological, climatological and other environmental fac... de... an ...ade ... ial ...lace... in st...kpil... will also directly influence water quality. See section 4.7 of Volume 3-Chapter 4 for details of the assumptions and analysis.

the Study Area. The potential for these impacts could be decreased depending on the location of the discharge in the watershed and the types and efficiencies of the mitigation techniques which are utilized.

Section 4.7 of Volume 3-Chapter 4 should be consulted for a detailed discussion of the potential impacts of copper-nickel development of water quality in the Study Area. The changes in water quality and metals loadings can then be considered in light of the above discussions.

1.6.3.4 Potential Mitigation of Heavy Metals--Several techniques are available to decrease the loadings of heavy metals in waters released from a copper-nickel development. As previously mentioned the magnitude of the problems involved will be directly related to the water management scheme which is utilized by a particular operation. The best possible mitigation is of course to totally eliminate the potential discharge. It may be possible to accomplish this by managing a water shed smaller than the ones utilized in the hydrological models presented in Volume 3-Chapter 4 (see sections 4.3, 4.4, and 4.5 of that chapter for detailed discussions).

Other possibilities for mitigation include collecting the runoff waters from the operation and passing them through a treatment plant. During the life of the operation the processing plant may be able to provide some of this treatment if all runoff waters are collected and transported to the tailing basin where they would enter the mill cycle. Following the operation other mitigation procedures would be necessary. At this point in time very little information is available on the long term mitigation of this type of pollution problem. Presumably it would be most desireable to establish a passive system which would require little or not perpetual maintainance, this however, is not possible with the current state of knowledge.

1.6.3.5 Regional Sensitivity--There are no distinct regional patterns for sensitivity to heavy metals pollution. Lakes and streams in the Study Area with relatively high total organic carbon (TOC) content or high hardness values will be relatively less sensitive to heavy metals stresses. It is most important to avoid areas which have low TOC or hardness because the organisms in these areas would suffer greater damage if exposed to equivalent concentrations of heavy metals. Site specific aquatic toxicology studies should be conducted in conjunction with the development of site specific proposals.

1.7 pH CHANGES

1.7.1 Potential Causes of pH Changes

Three factors govern the pH of surface waters: (1) the rate of strong acid input (H^+ ions); (2) the location of lakes and streams relative to prevailing winds, which in relation to atmospheric deposition controls factor (1); and (3) the geochemistry of the surficial sediments and the bedrock of the watersheds of the receiving lakes and streams. Kramer (1976) suggests that factor (3) is most important because it determines the buffering capacity (resistance to acid input) of the surface waters and the watershed runoff which enters the surface waters (see Volume 3-Chapter 4 for further details).

There are three potentially major sources of H^+ ion input into the surface waters of the Study Area. It is possible that the leachate from wasterock/lean ore piles may be acidified however available evidence is conflicting. Experimental laboratory studies have indicated that low grade material which would be placed in these piles has more than sufficient buffering capacity to maintain the pH of the leachate at neutral values (pH=7). Recent data from the gabbro stockpiles at Erie Mining Company's Dunka Pit have indicated that pH in leachate may in fact drop below 7.0. pH values at Erie's Seep 3 were 7.3 in 1976, 7.1 in 1977 and 6.7 in 1978. The pH of the leachate from some test piles at Amax's exploration site have also droped below neutral values. At the present time the cause of these drops in pH are not understood and further studies must be completed before the pH values of potential leachates can be better established.

Another potentially important source of acid input into surface waters is atmospheric inputs. The presence of acids in the atmosphere can cause precipitation to be acidic. Strong acids have been found to be the most important contributors to acid precipitation (Likens 1976, Dovland et al. 1976, Galloway et al. 1976a, Summers and Whelpdale 1975, Gorham 1975, Krupa et al. 1975), although weak acids may also contribute (Galloway et al. 1975). The precursors of acid precipitation are chloride, which forms hydrochloric acid; sulfur dioxide, which is converted to sulfate and then to sulfuric acid; and nitrogen oxides, which form nitric acid. These compounds are released to the atmosphere by various natural and human activities.

Finally, the pH of water from the smelter may be as low as 2.6 (see Volume 2-Chapter 4). If water with a pH this low is released without treatment it could cause serious damage to the aquatic environment. Sulfur dioxide (SO_2) emissions from various combustion sources, such as coal fired power plants, and from various other sources, such as ore smelting, combine with rain to form sulfuric acid, a strong mineral acid. This acid contributes hydrogen ions (H^+), which lowers the pH value of rain. It was once believed that acidic rain was primarily a localized problem that occurred only near the SO_2 source, evidence now indicates that SO_2 can travel long distances and that the problem of acid precipitation is world-wide.

Evidence will be presented in Volume 3-Chapter 4 indicates that some lake and streams in the Study Area are more than likely now being affected by acidic precipitation. The major sources for the SO_2 which causes the acidic precipitation is believed to be out-state, i.e. sources located in the industrialized areas to the east and south of Minnesota. This contention is, in part, speculative and further study would be required to delineate the major sources (see Volume 3-Chapter 4 for further details). Either source of H^+ ions to the surface waters of the Study Area will result in the same types of impacts to the aquatic environment.

1.7.2 Responses of Organisms

Patrick (1968) observed major changes in the density and number of species of diatoms at pH 5.0 to 5.2. Since diatoms are the primary algal type in streams and generally bloom in the spring, a major change in autochthonous production would result if pH drops to these levels. Shifts to green algae which are less sensitive would also be expected.

Direct mortality of aquatic insects may occur as a result of pH decreases below 5.5. Four-day LC-50 values for some species of mayflies, stoneflies, and caddisflies range between 4.5 and 4.7 (Bell and Nebeker 1969; Gaufin 1973). Insects are particularly sensitive to low pH at emergence, which occurs in spring and early summer for most species. Bell and Nebeker (1969) and Butcher et al. (1973) found that pH one-half to two units above survival levels were necessary for successful molting and emergence.

Significant effects on spawning success of fishes have been recorded in the laboratory at pH levels between 4.5 and 6.1. Effects on growth and survival of adults probably would not be observed at these pH values under short-term conditions. As decreases continue, fish populations become dominated by a few large fish with very little recruitment (i.e. replacement of dead) because of the poor spawning success.

Figure 36 summarizes available literature on the impacts of pH decreases on aquatic organisms. See Lind et al. (1978) for a detailed discussion of the impacts of pH changes on aquatic organisms.

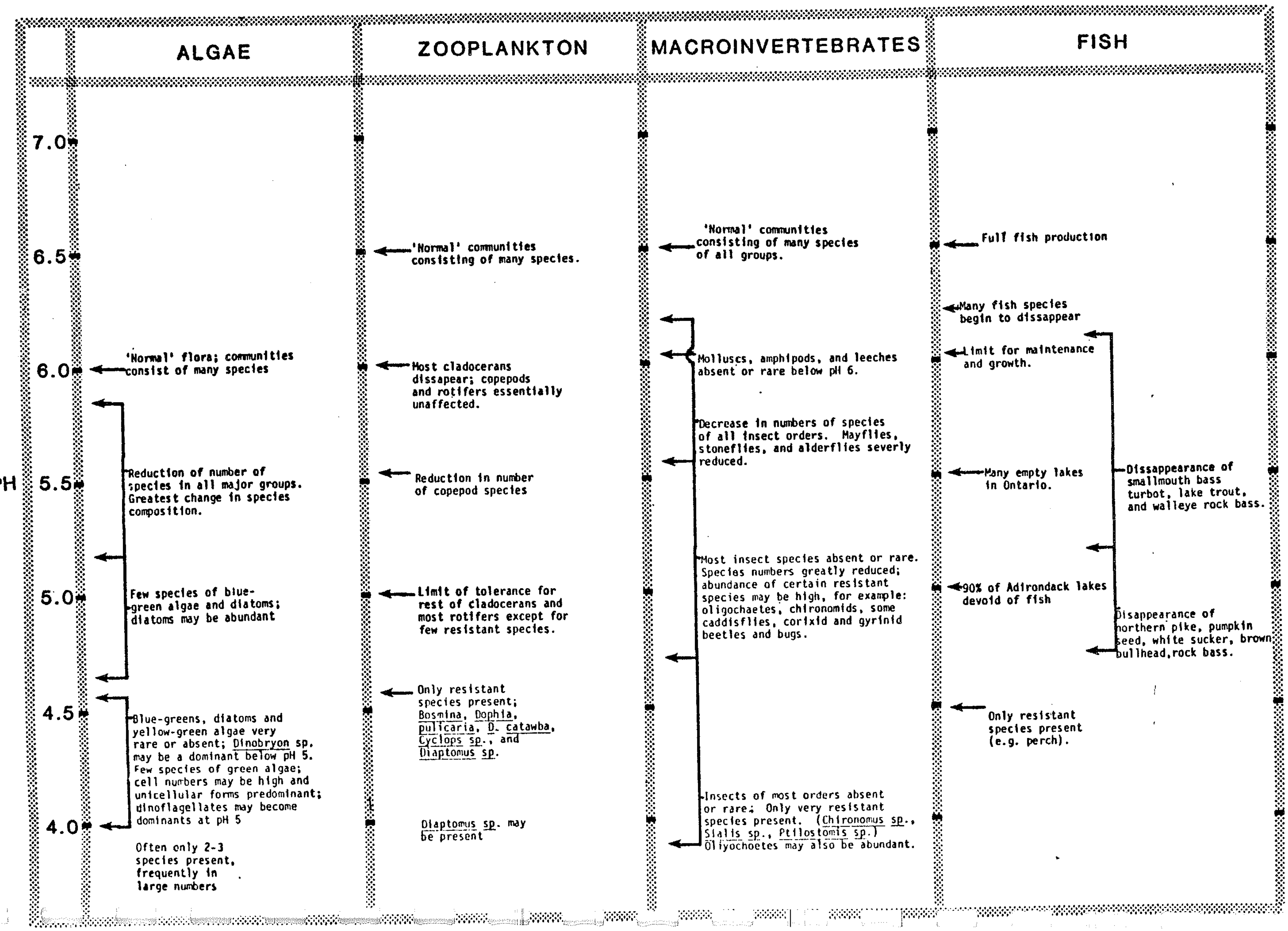
Figure 36

1.7.3 Impacts of pH Changes

As mentioned before, it is believed that there are two potential sources of H^+ ions to the surface waters of the Study Area: acidified leachate and acidic precipitation. At the present time, evidence is equivocal on the question of whether leachates from waste rock/lean ore piles will be acidified or neutral in pH. It does appear, however, that the aquatic ecosystems of the Study Area may be subjected to acid stress from acidic precipitation.

The pH of spring snowmelt waters in some parts of the Area appear to be significantly below neutral (e.g. Filson Creek where the pH was 4.5 in 1977 samples). Acidic precipitation alone or in combination with acidifier leachates appear likely to place an acid stress on the aquatic ecosystems of some of the lakes and streams of the Study Area and the BWCA. The surface waters in these areas have a very low buffering capacity and the pH levels which have already been observed are at levels which are known to cause short-term toxicity problems for aquatic organisms ranging from algae through fish. In particular, the lakes which are of most concern are those with the lowest buffering capacities. Stream systems are very sensitive because the flush of water from spring snowmelt can represent a

FIGURE 36 EFFECTS OF DECREASING PH ON AQUATIC ORGANISMS



majority of the water that the stream may carry throughout the whole year (see Volume 3-Chapter 4 for further details).

If the patterns of acidic precipitation continue the way they are, it is likely that many of the poorly small buffered stream will have noticeable decreases in fish populations during and following spring melt. Recovery from these episodes may be expected to be fairly rapid (i.e. within months) unless or until the sources of recolonizing organisms are themselves affected (i.e. well buffered lakes or large unaffected streams). Once the source areas are impacted, recovery would be very slow.

Some of the lakes which may be the earliest to be impacted by acidic precipitation include: Clearwater, August, Turtle, One, Greenwood, Perch, and Long lakes. These lakes are most susceptible because they have Calcite Saturation Indices above 3.0, an indication of very limited buffering capacity. At the present time, it is not possible to relate levels of sulfate loading to pH decreases in these lakes (see Volume 3-Chapter 4).

Decreases of pH in the waters of the Study Area may increase the toxicity and availability of heavy metals in the aquatic environment. The leaching of the metals from waste rock/lean ore stockpiles would increase as a result of acidified leachate of acid input to the piles via acidic precipitation. At the present time, there is not a consensus as to the relationship between pH and metals toxicity (Mount 1966); therefore, at least qualitatively one must assume that the affects are at least additive.

Although it is not known whether copper-nickel mining per se is likely to cause pH related impacts when considered alone, the significance of long-term pH changes to the lakes and stream of the Study Area and BWCA is of major concern.

Long-term changes in the aquatic communities are probably already underway due to the general decrease in the pH precipitation and thereby surface waters in the Study Area. Because the decrease in pH will be slow, measurement of the biological effects would require intensive long-term monitoring. During this period of decreasing pH, the overall productivity and diversity of the aquatic communities can be expected to decrease.

1.7.4 Potential Mitigation of pH Changes

Because of the as of yet unanswered questions about the acidification of leachates from waste rock/lean ore piles, it is difficult to consider the mitigation measures that would be necessary and/or effective. One basic method of diminishing the problems would be to decrease the potential for input of H^+ ions. The acid input could be limited by controlling the grades of sulfides allowed in the stockpiles and by maintaining or increasing the buffering capacity of the piles. The effectiveness of such techniques will only be known after further studies have been completed. Methods of mitigating the impacts of acidic precipitation on aquatic ecosystems are under investigation at the present time in many parts of the world, consideration of this matter is beyond the scope of this report.

1.7.5 Regional Sensitivity

Headwater lakes and streams are probably the most sensitive water bodies in the Study Area and BWCA. These waters are sensitive because they generally have relatively small drainage areas and thus pH values more closely reflect the pH of precipitation with watershed contributions of buffering capacity and lake chemistry assuming a secondary role. Utilizing data from the United States Forest Service (USFS), it was determined that of 30 lakes studied (in the BWCA) had Calcite Saturation Indices which indicated that they are poorly buffered. See Volume 3-Chapter 4 for a detailed discussion of the regional variability of buffering capacities.

1.8 PHYSICAL CHANGES

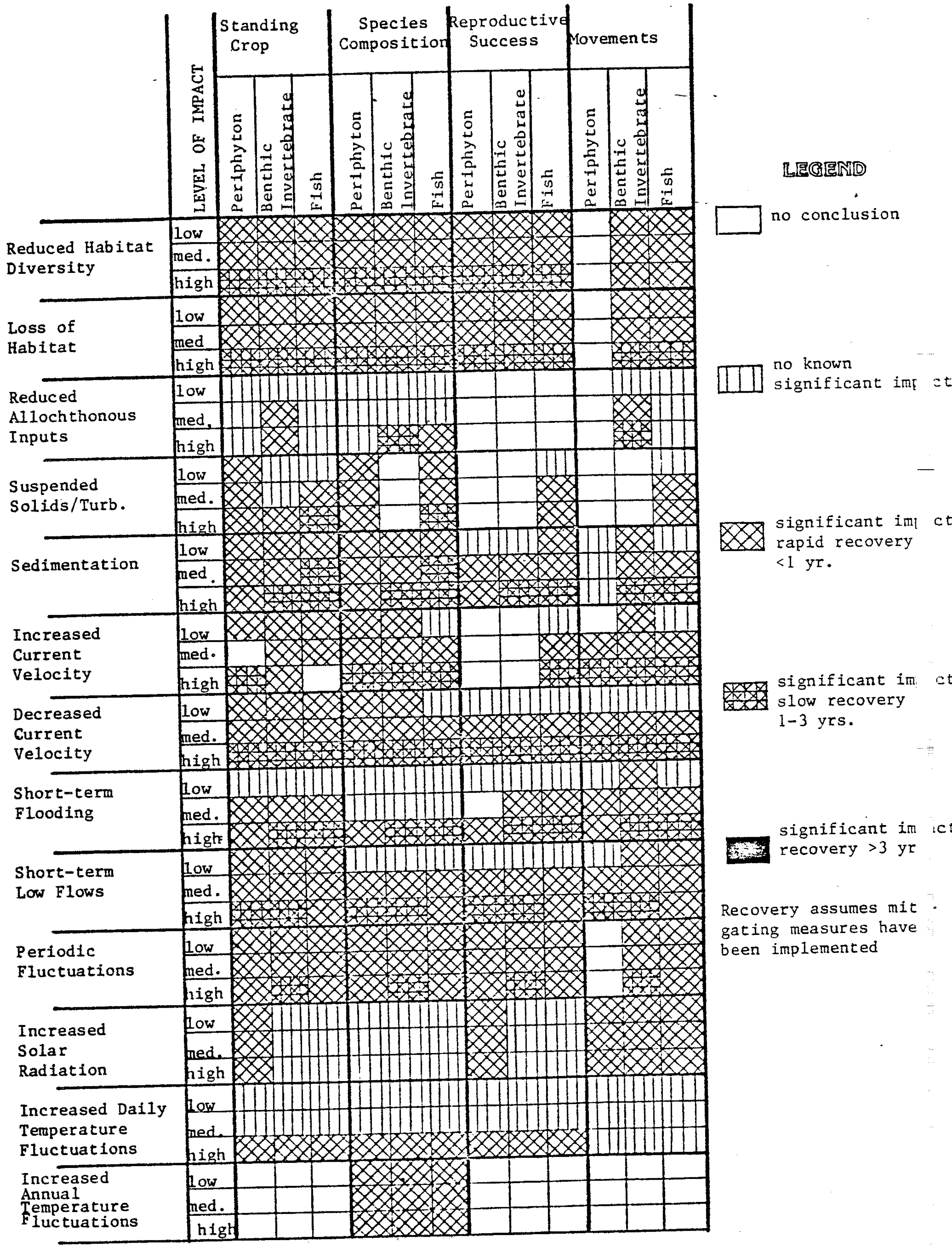
Physical changes to an area such as channelization or removal of terrestrial vegetation can have significant affect on aquatic ecosystems. The effects may range from changes in primary production through loss of complete communities. Figure 37 summarizes the data available from published studies, and indicates the significance of impacts from high, medium, and low levels of various physical stresses on the aquatic environment. Unfortunately, no studies are available which quantify the relationships between amounts of physical stress and the impacts on the aquatic community; therefore, this summary must be qualitative. For the purposes of further discussion, physical stresses have been placed in five major categories including: flow changes, channelization/diversion, temperature changes, increased suspended solids, and loss of terrestrial vegetation. Each of these types of physical stress is discussed individually in the following sections.

Figure 37

1.8.1 Flow Changes

1.8.1.1 Potential Causes of Flow Changes--The development of a copper-nickel mining and processing operation may involve large changes in the flow regime of streams in the Study Area (see Volume 3-Chapter 4, Hydrological Impacts section for further details). Stream flows may be affected by a mining development in several different ways. The containment of the mining site for the purposes of

FIGURE 37 EFFECTS OF PHYSICAL STRESSES



** See Johnson and McCullough 1978 for further details.

gathering runoff may, for example, reduce the flows of adjacent streams. During dry years, such a reduction of runoff could potentially result in low or no flow conditions of a duration which is longer than the aquatic organisms naturally encounter or can survive. If a situation is encountered where runoff is contained and gathered from the watershed of one stream and any excess water which is collected is discharged to another, smaller stream, it would be possible to increase the flow of the small stream significantly. Such a situation may be observed at Erie Mining Company's Dunka Pit, where mine water is collected (the apparent source of this water is the Dunka River) and subsequently excess water is discharged into Unnamed Creek. In this case, the discharged water causes significant fluctuations in the hydrograph of the creek (see section 1.4.1.3 of this chapter for a discussion of the currently observed impacts on Unnamed Creek). The significance of potential hydrological changes on the flow regimes of streams in the Study Area is discussed in detail in Volume 3-Chapter 4.

1.8.1.2 <u>Responses of Organisms</u>--Alterations in flow are known to cause major impacts in aquatic communities. Periphyton, benthic invertebrates, and fishes all have specific current velocity preferences (Hooper 1973, Williams and Hynes 1977, Ambuhl 1959, Fraser 1972, Peterson 1977). Many organisms are adapted to high current velocities and changes in current velocity either cause mortality among the organisms or force them to move to more favorable habitat. Periods of low flow also cause loss of riffle habitats and stagnation in the remaining pools. Surviving organisms will be forced into the remaining pools during low or no flow periods where oxygen and food will become limiting (Larimore et al. 1959, Armitage 1977, Waters 1964, Thorup 1970, Fraser 1972, Peterson 1977).

Reduced natural discharges may have adverse effects on reproduction in fishes (Hooper 1973). Barriers and resultant altered water velocity may decrease reproductive success of fish in two ways: they may delay or disrupt the normal routing or speed of migration of spawning fishes or fill spawning sites with sediments because of the water's reduced transport capacities (Fraser 1972, Peters 1962, 1967). Flows conducive to redd (nest) construction and egg incubation may be eliminated and spawning areas adjacent to main channels may be exposed.

Although the aquatic biota in small headwater streams is currently subjected to periodic zero flow conditions, significant biological impact can be expected in these small streams. Decreased stream flow may cause many problems for invertebrates including: dessication or stagnation, reduction in available living spaces, elimination of food producing areas, disruption of life cycles and changes in remaining habitat. As noted, one of the most critical changes which can occur in the habitat is a reduction in current velocity. Many species of invertebrates and periphyton are adapted to areas of high current velocity and cannot survive under conditions of low current velocity. Because periods of low flow do occur with some regularity in headwater areas, the fauna of these streams is somewhat adapted to these conditions. During these periods, invertebrates may become dormant, move to more suitable habitat, or move deep into the stream substrate to avoid dessication.

Losses of high food production areas (riffles), lower current velocity, and stagnation in pools during extreme conditions will all adversely affect the fish populations of headwater streams. Overall, the stream-carrying capacity will be reduced. Spawning by trout would be disrupted since low flow periods would generally occur during the fall when brook trout spawn. These fish require high current velocity through the spawning beds to provide oxygen to the eggs and larvae.

Further effects on spawning will result from the altered spring flows. High flows and flooding necessary for northern pike to spawn may not occur in the headwater areas. Some effect may also be seen on walleye spawning in streams, although their use of Study Area streams for spawning is not as well documented as is use by northern pike.

1.8.1.3 Impacts of Flow Changes--As mentioned earlier, at this time it is not possible to quantify the impacts of flow changes on aquatic organisms. The potential for impacts, from flow changes, on aquatic organisms is dependent on the location of the operation in the watershed. The influence of an operation on the flow regime of a first order stream is likely to be much greater than it would be on any higher order stream.

Although flow changes from an operation which impacts a first order stream may be relatively large, it appears from the hydrological analysis that such flow changes would occur infrequently. Evidence from the literature indicates that aquatic organisms recover from flow changes relatively rapidly (less than one year). If recovery in the Study Area occurs as rapidly as may be expected, it is unlikely that there will be any long-term impacts resulting from flow changes. If, however, extreme low flows occur with a frequency greater than the recovery rate for the aquatic communities, it would be possible to have observable impacts on the organisms during the life of the operation.

In watersheds containing tailing basins, desirable impacts may occur. Although large amounts of land would be used for the tailing basins, the seepage from these basins would contribute significant amounts of water to the baseflow of small streams. For example, a 1,625 hectare tailing basin would cover approximately two-thirds of the Keeley Creek (2nd order) watershed. Seepage from that basin in semi-permeable soil if not collected and recycled would be approximately 31 l/sec, which is 55 times greater than August, 1976, baseflow of 0.6 l/sec. Therefore, the effect of naturally occurring low flow periods would be diminished in headwater streams. Also, overall productivity would be increased in these streams because there would be more continuously available habitat available, but this may be offset by the lower quality water resulting from seepage.

Similar, but less severe, effects should be expected in third order streams with watershed areas less than 40 km^2. In fourth order streams such as the St. Louis or lower Partridge with watershed areas greater than 40 km^2, the biological effects of flow changes would probably be insignificant. Tailing basin seepage could increase discharge during low flow periods as much as 25%; however, the increase would probably not cause any increase in productivity, nor would it reduce the impact of natural low flows significantly. Increased frequency of low flow periods may cause some impact, but low flow events in fourth or higher order streams would not be great enough to devastate the aquatic biota.

Overall the increased frequency of low flows and the possible loss of spring flows could cause a reduction in stream productivity and spawning success in the smaller watersheds. The addition of tailing basins within the affected watersheds would tend to mitigate the effects of lost watershed area if seepage was allowed. If flows return to normal, recovery of the stream communities should proceed rapidly. Invertebrate and algae communities should return to normal within two years after flows return to normal. Recovery of fish populations may take longer depending on the extent and severity of the impact.

1.8.1.4 Potential Mitigation of Flow Changes--The influences of water appropriation on the flow regime of a stream and aquatic organisms can be diminished by carefully planning the appropriation scheme. Alternative appropriation methods may include: appropriating only during peak flows, appropriating proportionately to flow, and constant appropriation. An appropriation plan which decreases the potential for extreme low flows would significantly diminish the likelihood of obsrvable impacts on aquatic communities.

1.8.1.5 Regional Sensitivity--First and second order streams are likely to be most sensitive to changes in flow. The organisms in these streams are somewhat better adapted to low flow conditions than the organisms in higher order streams. There appear to be no portions of the Study Area which would be especially more sensitive to flow changes.

1.8.2 Channelization/Diversion

1.8.2.1 Potential Causes of Channelization/Diversion--The development of copper-nickel mines in northeastern Minnesota may result in the channelization and/or diversion of streams at the mine and mill sites or at the tailing basin. These modifications may be necessary to either move a stream around the development site or to move water away from a site faster to prevent flooding. Further, channelization of streams may be necessary to facilitate the construction of roads and railroads. The extent and design of the channelization will be dependent upon the specific sites chosen for the various mining phases.

1.8.2.2 Responses of Organisms--Physically, channelization/diversion projects generally result in the straightening of streams so that the channel morphology becomes homogeneous. High current velocities, the loss of the pool/riffle interspersion and the removal of natural obstructions result. Discharge fluctuations are generally more rapid and average low flows are decreased.

Because of these physical changes the number and variety of ecològical niches will be reduced. This will reduce the overall carrying capacity of the affected system. There may be little effect on the periphyton communities because the removal of riparian vegetation could increase the amount of sunlight available and therefore productivity. On the other hand lack of attachment surfaces might reduce productivity.

The loss of riparian vegetation would reduce the amount of allochthonous material reaching streams which could affect the composition and productivity of invertebrate communities. Other significant changes in the invertebrate community will result from the loss of habitat area and diversity. Significant changes in the invertebrate species composition in channelized streams was observed by Etnier (1972) and Hansen (1972) while Crisp and Crisp (1974), Arner et al. (1976) and Moyle (1976) observed reduced invertebrate productivity.

Significant reductions in fish populations in the affected streams can be expected. Up to 98% reductions in fish standing crops have been observed in channelized streams (Wharton 1970, Tarplee et al. 1971, Moyle 1976). These losses were generally the result of reduced habitat diversity, cover and production of food.

Increased current velocities during spring runoff can inhibit the migration of spawning fish thus affecting the reproductive success of fishes. Also, because channelization reduces the amount of spring flooding the amount of spawning habitat for species such as northern pike would be reduced.

1.8.2.3 Impacts of Channelization/Diversion--Without mitigation the impact of channelization will be long-term. Congdon (1971) observed very little recovery 130 years after channelization had occurred in the Charilton River, Missouri. In another study recovery was noted by Tarplee et al. (1971) 15 years after channelization in the coastal streams of North Carolina.

Within the Study Area, significant biological impact would be expected as a result of any major channelization. It is most likely that only small headwater streams would be subjected to channelization or diversion; thus, impact is most likely to occur in these streams. Lower diversity and productivity within all trophic levels will occur. Also spawning within these streams will be disrupted. The effect of these changes in higher order downstream areas would probably not be measurable unless all headwater areas of the watershed were impacted. For example, extensive channelization in Filson Creek (2nd order) would not significantly alter the aquatic biota in the Kawishiwi. It is unlikely that a channelized stream would be used for spawning by Kawishiwi River northern pike because of decreased spawning areas. While this lack of spawning would be a major change in Filson Creek, northern pike populations in the Kawishiwi River would not be significantly affected because Filson is only one of many spawning areas.

Channelization projects in mid-reach streams would probably cause decreases in local game fish populations because of the loss of pool areas in streams. In addition, decreases in diversity and productivity of periphyton and invertebrates would occur.

The channelization and/or diversion of streams in the Study Area will cause significant biological impact without mitigation. This impact will probably be limited to those streams directly affected by the projects. It can be expected that any recovery within the affected systems will be extremely slow unless mitigative measures are undertaken to increase the amount and diversity of habitat.

1.8.2.4 Potential Mitigation of Channelization/Diversion--The impact of channelization and diversion could be minimized through use of a variety of mitigation techniques.

Several methods are available to mitigate the impact of channelization and diversion in the Study Area. Mitigation procedures which should be considered include:

1) increase streams sinuousity (curvature)(Zimmer and Bachman 1976, Lund 1976);

2) place substrates such as large rocks, wire gabions and rip rap in the stream where possible (Barton et al. 1972, Lund 1976);

3) revegetate the stream banks (Lund 1976);

4) restrict the length of channelization to less than one kilometer (Bayless and Smith 1967, Buckley et al. 1976).

A further consideration would be to assure that current velocities during spring runoff do not prevent the upstream movement of fish. In addition, spawning areas must be identified and protected during spawning times (mainly spring).

1.8.2.5 Regional Sensitivity--The portions of the Study Area which are most susceptible to impacts from channelization/diversion efforts are fish spawning areas. Available information does not indicate any concentrations of spawning areas within the Study Area; therefore, it would appear that all portions of the area are equally sensitive.

1.8.3 Temperature Changes

1.8.3.1 Potential Causes of Temperature Changes-Operation of a copper-nickel smelter will require the use of water for non-contact cooling. Cooling water released directly from the postulated 100,000 mtpy smelter are modeled as being 8.9°C above ambient while the addition of cooling towers could essentially reduce the release temperature to ambient. Furthermore a maximum temperature increase of 11-17°C could be expected within the condensers. The smelter model making maximum use of cooling towers would require approximately 10,000 l/min make up water for non-contact cooling while blowdown would consist of approximately 8,000 l/min, at ambient temperatures.

1.8.3.2 Responses of Organisms--The effect of increased temperatures on aquatic organisms is well documented (e.g. Coutant 1975, 1976). In general, increased productivity and shifts in the dominant organisms occur throughout the year in the heated water until the temperatures reach levels which are lethal to the majority of organisms. In some cases these shifts may represent a change in the seasonal occurance of species. As temperatures move into the optimum temperature range for individual species, these species flourish. Temperature increases also result in lower diversity. Shifts from algal communities dominated by diatoms to ones dominated by blue-green algae. The life cycles of invertebrates are accelerated by heated discharges so that emergence may be out of synchrony with the normal populations.

Fish have the abiliity to detect temperature gradients and move to their preferred temperature. During the summer fish often avoid heated water zones while during the winter they are often attracted to a thermal effluent. These temperature preferences change seasonally with changes in acclimation temperature. Lethal temperatures also change with changing acclimation temperatures. In general, warmwater species can survive at temperatures of 25-30°C while coldwater fish can survive in water of 18-20°C depending on acclimation temperatures. Unless the heated zone is extensive, the measurement of increased fish productivity would be difficult because of the mobility of fish.

1.8.3.3 Impacts of Temperature Changes--A temperature increase of 5-9°C would cause significant biological impact in localized areas near the effluent in both lakes and streams. Effects would be most severe during low flow periods and in the winter (generally a low flow period). It is unlikely the upper lethal limits for any aquatic species except trout will be exceeded within the mixing zone even during periods of low flow. However, during low flow periods in the summer, the preferred temperatures of some species such as white suckers and walleyes may be exceeded. Because of the ability of fish to follow temperature gradients to their preferred temperatures, no direct effects would be expected on fish populations. However, the placement of the smelter with a large heated discharge on a trout stream (which is unlikely because of the water requirement of such a facility) would eliminate that population although increased primary and secondary production would occur in the heated zone as well as accelerated invertebrate life cycles.

During the winter, fish will congregate within the zone of heated water, again seeking out their preferred temperatures. Potential problems could result if the smelter operation shut down and the rate of temperature decrease was too rapid for fish to acclimate to the cold temperature.

A further problem occurs with the entrainment and passage of organisms through the cooling system. Intake screens generally allow organisms less than 2 cm to pass into the cooling system. These organisms are then subjected to the maximum temperature difference at the condenser for a short period and to mechanical stress in the pumps. While mortality varies among groups and species, algae and invertebrates appear to be relatively tolerant to a once through exposure of this type with mortality reported at a rate of 20%. Fish eggs and larvae on the other hand are very susceptible and high mortality often results with these organisms.

Also, organisms in all groups which are recycled have higher mortality rates than ones which pass through the system once.

Recoveryof the invertebrate and algal communities would be rapid following the shutdown of the smelter. Fish populations, while recovering more slowly due to their long life cycle, would still recover within one to three years depending on the magnitude of change.

1.8.3.4 Potential Mitigation of Temperature Changes--Cooling towers can be utilized to decrease the temperature of the water which will be released from the non-contact cooling water system of the smelter. Making maximum use of cooling towers would decrease the temperature of the discharge water to essentially ambient levels and thus could mitigate these impacts.

1.8.3.5 Regional Sensitivity--The coldwater fishes and their associated communities are most susceptible to temperature increases. There are very limited coldwater fisheries in the Study Area, and if these streams are avoided the impacts of temperature changes should be minimal. In particular, some of the streams in the Study Area with coldwater fisheries were Wyman Creek in the Partridge watershed, Snake Creek, and Little Isabella River in the Isabella watershed, and Nip Creek in the Stony watershed (see Williams et al. 1978 for further details).

1.8.4 Suspended Solids

1.8.4.1 Potential Causes of Increased Suspended Solids--An increase in the concentration of suspended solids is expected to occur as a result of copper-nickel mining development and associated secondary development. Erosion of exposed soils during construction phases will cause the greatest increases in suspended solids. High concentrations can also be expected during spring runoff. After completion of construction activities and site revegetation, sediment loads in adjacent water bodies should decrease. At the mine and mill site revegetation may be limited until the operation is shutdown. Some increases may also occur as a result of mine dewatering and/or tailing basin release although these types of releases are unlikely.

1.8.4.2 Responses of Organisms--If suspended solids concentrations exceed 25 mg/l, significant changes in the aquatic biota can be expected (EIFAC 1965). These changes were outlined by Ritchie (1972) as follows:

1) reduction in primary productivity leading to a decline in the food available for higher trophic levels;

2) reduction in dissolved oxygen if the deposited material is organic;

3) reduction in the survival of fish eggs and larvae;

4) reduction in the number of bottom organisms from a change in habitat;

5) reduction in the feeding efficiency of fish; and

6) loss or change of fish habitat.

The most obvious effect of suspended solids on periphyton is the reduction of light penetration and therefore a reduction in photosynthesis (Ritchie 1972, Rosenberg and Snow 1975, Sorenson et al. 1977). A reduction in diversity may also occur (Samsel 1973).

As suspended solid concentrations increase, reductions in benthic invertebrate populations and changes in the species composition can be expected. These changes occur as a result of sedimentation. The immediate effect of sedimentation on benthic invertebrates is smothering or a forced move to a more favorable habitat (Ellis 1936). After this initial effect, reductions in the standing crop can be expected. Gammon (1970) found that an increase of 20 to 40 mg/l and 80 mg/l of solids caused 25% and 60% reductions in invertebrate populations, respectively.

As the average substrate particle size decreases downstream from a disturbance as a result of sedimentation, changes in the dominant invertebrate organisms will occur. Mayflies, stoneflies and caddisflies will generally be replaced by chironomids and oligochaetes in areas of sedimentation. Neither chironomids nor oligochaetes are preffered fish food as are the invertebrate groups they replace.

Major changes in fish populations can be expected as the concentrations of suspended solids increases. Fish appear to be the most sensitive group to suspended solids. EIFAC (1964) lists the modes of actions of suspended solids on fishery resources as follows:

1) direct mortality or reductions in growth rates and disease resistance;
2) increased mortality of eggs and larvae;
3) reduction or change in fish food organisms; and
4) changes in movements and migration of fish.

Many studies have been conducted on the affect of sedimentation on fish spawning. For example, Hassler (1970) observed 97% mortality in northern pike eggs which had been coated with 1 mm of silt. Destruction of yellow perch eggs by sediment has also been reported by Munch (1962). Brook trout are also severly affected by sedimentation of gravel riffles used for spawning. When sediments clogs the interstices of the gravel, the survival of trout eggs and embryos decreases.

Campbell (1954) reported 100% mortality of trout eggs in a stream with high sedimentation compared with 6% mortality in a clear tributary stream. Peters (1962, 1967) among others also observed low survival of trout embryos in sediment filled riffles.

1.8.4.3 Impacts of Increased Suspended Solids--At the present time, suspended solid concentrations are below 10 ug/l in most streams in the Study Area. Because of the low gradients in Study Area streams, there is little transport of sediment through the systems. Where increases as a result of development occur, only small areas should be affected except during spring runoff when greater sediment transport will occur.

Increases in suspended solids and the resulting sedimentation will have insignificant impact if levels do not exceed 25 mg/l. This quantity of suspended solids would have minimal effects on fishery resources (EIFAC 1965).

Within the Study Area, significant biological impact is expected to occur in areas of increased suspended solids. Any impact would probably be restricted to areas near the source of the suspended solids because of the general lack of sediment transport. Within headwater streams a general decrease in productivity would result. A more serious consequence would be the loss of the naturally reproducing brook trout populations found in a few headwater streams. An increase in suspended solids in these streams would eliminate brook trout spawning habitat and, therefore, the population, unless artificial stocking was begun.

Effects on the spawning of northern pike would also occur in headwater as well as mid-reach streams. Sedimentation in the flooded areas used by northern pike for spawning would reduce the survival of northern pike eggs.

This impact, while significantly affecting the biota in the effected areas, would not be significant to the system as a whole except if isolated trout populations would be lost. Also, recovery following the reduction of suspended solids should be rapid. Gammon and White (1970) observed "immediate" recovery of stream invertebrates following sedimentation, while Barton (1977) and Hamilton (1961) reported the rapid reappearance of fishes following cessation of sedimentation problems.

1.8.4.4 Potential Mitigation of Increased Suspended Solids--Several methods are available to limit the increase in suspended solids that would result from different stages of development. Sediment traps can be used to capture eroding soils before they reach the main stream channel. Revegetation efforts can be initiated at as early a stage as possible to decrease erosion from road cuts, overburden piles, and plant construction sites. During the construction phase, it is likely that erosion would occur and cause some increase in suspended solids loadings. Available techniques could probably eliminate impacts from suspended solids during the life of the operation.

1.8.4.5 Regional Sensitivity--Fish spawning areas are probably the areas which are most susceptible to increased suspended solids and sedimentation. As previously indicated, there are no known concentrations of such spawning areas; therefore, the whole area is consideed to be equally sensitive to sedimentation and suspended solids.

1.8.5 Loss of Terrestrial Vegetation

1.8.5.1 Potential Causes of Terrestrial Vegetation Losses--Site clearing for the mine and plant development will be the major cause of terrestrial vegetation losses as they relate to aquatic ecosystems (see Chapter 2 of this volume for further discussion of other causes of terrestrial vegetation losses). The models presented in Volume 2-Chapter 5 indicate that a land area of 2,300 to 4,150 hectares will be cleared and utilized as the site for the mine processing plant, tailing basin, etc.

1.8.5.2 Responses of Organisms--Leaves dropped into the water by terrestrial vegetation supply a large portion of the energy utilized by primary consumers in woodland streams (Nelson and Scott 1962). In lakes and large streams, the contribution is less, but still important. How far this material travels within the watershed before entering streams is unknown. Most organic material may originate in the riparian vegetation, while at least some of the nutrients are contributed from vegetation in other parts of the watershed. The riparian vegetation also inhibits erosion and the inflow of sediments and provides shade which moderates temperature changes.

Chapman (1962, 1963) reported on the effects of logging, which produced a similar stress on stream ecosystems. He found that the removal of riparian vegetation would increase the sunlight reaching the streams, causing temperature increases. This temperature increase plus the increased light would promote the growth of attached algae if sufficient nutrients are available, (particularly in headwater streams which have been previously heavily shaded). This increase in primary production will tend to offset the energy loss realized from the loss of allochthonous inputs if sufficient nutrients are available. The invertebrate community will change from one which relies on allochthonous material (shredders of dead plants and some collectors) to one which relies principally on autochthonous material (scrapers, some collectors). Secondary production would decrease significantly since approximately two-thirds of the energy in woodland streams is derived from allochthonous sources. This decrease in secondary production would also result in lower fish production.

1.8.5.3 <u>Impacts of Terrestrial Vegetation Losses</u>--Development of copper-nickel mines and processing facilities will result in the use of large amounts of land. Most of this land will be denuded of terrestrial vegetation during the operation of the mine. Therefore, the input of terrestrial organic material (allochthonous material) into streams and lakes draining the mining area would be substantially reduced. Also, increased sedimentation and higher stream temperatures would result from the removal of vegetation adjacent to a stream.

Removal of vegetation from the watersheds within the Study Area may result in significant biological impact in streams and lakes. Greatest changes are expected in headwater streams which at present have heavy canopy cover where impacts would include: increased primary production, and shifts from a shredder/collector invertebrate community to a scraper/collector invertebrate community. Any brook trout populations may be lost due to higher summer temperatures in these streams; warmwater fish populations would not be greatly affected. In larger mid-reach streams, the effects would not be as great as in upstream areas. The nutrient budgets of effected lakes might be changed, but data are not available to address this type of change. It is not possible to predict the affect of reduced organic matter input to lakes.

Overall, major changes would likely occur in headwater streams, but there would probably be little change to downstream areas. Here, excess organic matter would probably be available from other upstream tributaries so that production would not be affected by upstream losses in small portions of the watershed. Headwater lakes with input only from an affected watershed would be the most seriously impacted, while lakes which receive input from other portions of the watersheds would not be seriously altered. Revegetation of the affected watershed would alleviate the stress, but complete recovery would be slow. The impact of vegetation removal could be significantly reduced if the riparian vegetation is undisturbed.

1.8.5.4 Potential Mitigation of Terrestrial Vegetation Losses--Impacts from terrestrial vegetation losses can probably be completely eliminated if a buffer zone is created on the border of any stream that runs through or adjacent to the development site. This buffer zone would provide allochthonous inputs, shading from direct sunlight, and to some extent erosion control for the stream. A buffer zone is probably the most efficient way to mitigate this impact; however, if this is not feasible over the long term, it would be possible to revegetate the impacted area. Revegetation is a rather slow process (see Chapter 2 of this volume), and the impacts on a stream would probably be measurable until a substantial cover of vegetation was re-established over the stream.

1.8.5.5 Regional Sensitivity--Allochthonous inputs are most important for headwater streams. If the net energy flow through a stream is decreased, the overall productivity would decrease and fish populations would be expected to become limited. Unless similar stress occur concurrently on several streams which feed a higher order stream, it is unlikely that any impacts would be observed on the higher order stream. These considerations indicate that headwater streams should be avoided where possible and that buffer zones should be utilized if avoidance is not possible.

1.9 SECONDARY DEVELOPMENT

1.9.1 Potential Causes of Secondary Development

The development of a copper-nickel mining industry in the Study Area would result in the influx of a large number of people to work in these operations as well support services and the families of all of these workers. Analysis of staff requirements indicates that as many as 2070 workers would be required for the copper-nickel development and support services in Development Zones 1 or 2 with an additional influx of 4140 family members (see Volume 5-Chapter 7 for further details). In general, settlement is expected to occur in existing towns, along roads and adjacent to lakes.

Increasing human populations in towns such as Ely and Babbitt will increase the amount of sewage treated in the local sewage treatment plants, thereby increasing the quantities of inorganic nutrients (phosphorus and nitrogen) entering the surface waters. Additional use of lakeshore property for primary housing would also increase the amount of inorganic nutrients reaching the surface waters of the Study Area.

In addition to increased nutrient inputs, increased populations in the area are likely to result in increased pressur on the fisheries resources of the region. At the present time it is not possible to quantitatively determine the impacts that such added pressure will have but it could become an important consideration in lakes that are proximate to areas of population growth.

1.9.2 Responses of Organisms

Increases of inorganic nutrient inputs in lakes and streams may result in the euthrophication of some of the lakes in the Study Area. Euthrophication would lead to increases in the frequency of undesirable algal blooms. These blooms may result in depletion of dissolved oxygen levels during the summer and the concomitant loss of game fish and other aquatic organisms.

If nutrient inputs are not severly increased the changes which occur in the lakes may be more or less "desirable" because the increases in populations and productivity of the algae would be reflected through the whole aquatic ecosystem. In effect the productivity rates for primary consumers would increase and result in increases in fish production rates.

Increased pressure on the fisheries resources of the region from larger numbers of people could result in changes in the characteristics of the fisheries. If pressures are confined to a relatively small number of lakes or streams it is likely that the sizes of the population of fish in these water bodies increase slightly however these increases would be associated with decreased fish sizes.

1.9.3 Impacts of Secondary Development

The majority of the lakes in the Study Area are classified as eutrophic or mesotrophic, based on total phosphorus (see Volume 3-Chapter 4). Additional nutrient loading of some of the Study Area lakes, therefore, could cause them to fall into eutrophic category.

The effect of increasingly eutrophic conditions in Study Area lakes will be conditions similar to those currently found in Shagawa Lake (Shults et al. 1976). Shagawa Lake has been receiving treated sewage effluent since the early 1900s. Prior to this, Shagawa Lake was probably mesotrophic. In 1973 a tertiary treatment plant was placed in operation in an attempt to reduce the input of inorganic nutrients. Biological sampling prior to the operation of this plant indicated that productivity was greater in all trophic levels in Shagawa Lake than in adjacent Burnside lake. This high productivity included game fish species such as the walleye. Unfortunately, while the walleye production was, and continues to be higher in Shagawa Lake than in other Study Area lakes, nuisance blooms of blue-green algae occur during the summer. Levels of total phosphorus in Shagawa Lake ranged from 100-800 ug/l prior to operation of the tertiary plant which is far higher than median level of 23 ug/l for the Study Area lakes which were sampled.

While it is unlikely that further developments will cause phosphorus levels to approach those observed in Shagawa Lake, increases are likely to occur in lakes surrounded by major housing developments. These increases may cause significant long term biological impact in these lakes. measurement of the effects would probably only be possible over long periods. Recovery would be rapid in those lakes with rapid flushing rates such as the lakes in the Kawishiwi River. Other lakes with slow flushing rates would probably recover slowly since nutrients would be retained and recycled through the system.

If the nutrient increases are not severe, the impact may be "desirable" since there will be increased fish production while algal blooms do not reach nuisance levels. If major increases in phosphorus and nitrogen occur, the more severe effects of euthrophication may be observed such as nuisance algal blooms, mid-summer oxygen depletion with resultant loss of game fish populations and general loss of diversity at all trophic levels.

Similar effects can be expected in Study Area streams. Increases in inorganic nutrients would cause significant biological changes to occur. Increased productivity and decreased diversity wuld be the expected impacts although in cases where only small nutrient increases occur diversity may increase.

Stream recovery is expected to be rapid following the reduction of nutrient stress as spring flows would flush residual nutrients out of the systems. Because no physical alteration in the habitat would have occurred, recolonization would proceed rapidly as long as a very large area had not been impacted.

1.9.4 Potential Mitigation of Secondary Development

The best method for limiting the potential impacts from secondary development is to assure the best possible treatment of sewage materials before they are released into the aquatic environment. The conditions in Shagawa Lake have somewhat improved since the operation of the tertiary treatment plant. However, it appears that recovery will take many years because of the continued cycling of the nutrients whch were disposed in the lake before the treatment plant became operational.

To avoid fishing pressures which might decimate the high quality fisheries of the region it would be necessary to study the increased pressures as they develop. It may then prove possible to control the use of lakes or streams which appear to be receiving too much pressure before the impacts become severe.

1.9.5 Regional Sensitivity

As indicated earlier the majority of the lakes in the Study Area are currently classified as euthrophic or mesoeutrophic. The most sensitive portions of the Study Area are those which have the euthrophic lakes as additional nutrient loadings could potentially result in severe euthrophication with its associated algal blooms, etc. There did not appear to be any spatial patterns, in the region, in relation to the distribution of eutrophic lakes (see Volume 3-Chapter 4 for further details of the nutrient conditions in lakes in the Study Area).

1.10 POST SCRIPT

The considerations outlined in this chapter indicate the potential for some very serious long and short term impacts on the aquatic ecosystems of the Study Area. The significance of these potential impacts must be evaluated not only in relation to the ecosystems themselves but also with due consideration to the people who utilize these systems for recreation and other purposes (see Volume 5-Chapter 9 for discussion of outdoor recreation in the Study Area).

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