REGIONAL COPPER-NICKEL STUDY:

BIOLOGICAL EFFECTS OF PHYSICAL IMPACTS TO STREAM ECOSYSTEMS

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MEQB Regional Copper-Nickel Study

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March 13, 1978

INTRODUCTION TO THE REGIONAL COPPER-NICKEL STUDY

The Regional Copper-Nickel Environmental Impact Study is a comprehensive examination of the potential cumulative environmental, social, and economic impacts of copper-nickel mineral development in northeastern Minnesota. This study is being conducted for the Minnesota Legislature and state Executive Branch agencies, under the direction of the Minnesota Environmental Quality Board (MEQB) and with the funding, review, and concurrence of the Legislative Commission on Minnesota Resources.

A region along the surface contact of the Duluth Complex in St. Louis and Lake counties in northeastern Minnesota contains a major domestic resource of copper-nickel sulfide mineralization. This region has been explored by several mineral resource development companies for more than twenty years, and recently two firms, AMAX and International Nickel Company, have considered commercial operations. These exploration and mine planning activities indicate the potential establishment of a new mining and processing industry in Minnesota. In addition, these activities indicate the need for a comprehensive environmental, social, and economic analysis by the state in order to consider the cumulative regional implications of this new industry and to provide adequate information for future state policy review and development. In January, 1976, the MEQB organized and initiated the Regional Copper-Nickel Study.

The major objectives of the Regional Copper-Nickel Study are: 1) to characterize the region in its pre-copper-nickel development state; 2) to identify and describe the probable technologies which may be used to exploit the mineral resource and to convert it into salable commodities; 3) to identify and assess the impacts of primary copper-nickel development and secondary regional growth; 4) to conceptualize alternative degrees of regional copper-nickel development; and 5) to assess the cumulative environmental, social, and economic impacts of such hypothetical developments. The Regional Study is a scientific information gathering and analysis effort and will not present subjective social judgements on whether, where, when, or how copper-nickel development should or should not proceed. In addition, the Study will not make or propose state policy pertaining to copper-nickel development.

The Minnesota Environmental Quality Board is a state agency responsible for the implementation of the Minnesota Environmental Policy Act and promotes cooperation between state agencies on environmental matters. The Regional Copper-Nickel Study is an ad hoc effort of the MEQB and future regulatory and site specific environmental impact studies will most likely be the responsibility of the Minnesota Department of Natural Resources and the Minnesota Pollution Control Agency.

ABSTRACT

Copper-nickel development in northeastern Minnesota may cause physical changes in the areas streams and rivers. These changes, their implications to the biota, and the potential for ecosystem recovery are reviewed.

Channelization, reduced flow, and increased suspended solids result in reduced availability and diversity of habitat. Increased flow may increase the amount of available habitat or render existing habitat unsuitable. Corresponding changes in distribution, abundance and diversity of stream organisms may be expected. Biological recovery is considered rapid and complete when natural conditions are restored. Natural and artificial mitigation is discussed.

BIOLOGICAL EFFECTS OF PHYSICAL IMPACTS TO STREAM ECOSYSTEMS

TABLES (cont.) PAGE

FIGURES

INTRODUCTION

This review discusses the biological implications of physical impacts which may occur in aquatic ecosystems as a result of copper-nickel developments in northeastern Minnesota. The following potential impacts were identified: altered flow regimes, channelization and diversion, increased suspended solids, and temperature changes. Significant temperature changes are not expected to occur and thus are not addressed in detail in this review. The reader is referred to reviews by Coutant (1971; 1973; 1976; 1977) and Blahm (1971) for more information on the effects of temperature changes. Although some of the information included in this review pertains to lakes, the review deals primarily with the effect of these impacts upon streams and rivers.

EFFECTS OF ALTERED FLOW REGIMES

Fluctuations in the flow regime of streams and rivers are a naturally occurring phenomena. Above average precipitation or drought may lead to natural flow variations. Intensive land development and water utilization by man (i.e. dams, reservoirs, diversions, and water abstractions and appropriation) has led to alterations of natural watershed run-off and infiltration characteristics (Stalnaker 1977). Many interacting factors contribute to the run-off patterns in a watershed (Williams and Hynes 1977). Infiltration of precipitation into the ground, the most important factor, apportions water between surface, subsurface, and groundwater flows. Infiltration rates are primarily controlled by soil condition and vegetation cover. Fraser (1972) discusses the biological implications of six components of flow: velocity, depth, width, timing, quality and fluctuations. A number of comprehensive reviews on the physical and biological implications of stream flow fluctuations are available including: Ambuhl (1959); Fraser (1972); Hooper (1973);

Bovee (1975); Tennant (1976); Stalnaker and Arnette (1976a); and Ward (1976).

Alterations in flow may cause a wide variety of physical changes to aquatic habitats. The most obvious effect is the creation of new, or the loss of previously existing aquatic habitat (Briggs 1948; Larimore et al. 1959; Paterson and Fernando 1969; Kroger 1973). Hooper (1973) reported the California Department of Game and Fish criteria for correlating stream flow and cross sectional, surface and food producing areas, water depth, velocity and shelter.

Flow changes may raise or lower water temperatures (Spence and Hynes 1971). Discharges of hypolimnetic water from dams may radically change stream temperatures. Stalnaker (1977) found warm water streams to be cooled by these discharges in summer and Ward (1976) reported a tempering of natural temperature fluctuations and failure of normal ice conditions to develop in winter. The effects of solar radiation springs and tributaries on stream temperature may also be greater or lesser depending on flow (Fraser 1972). Water volume and velocity are primary factors dictating the substrate present in flowing water systems. Altered flow regimes lead to different morphometric conditions, resulting in new equilibria between scouring and deposition of bottom materials (Stehr and Branson 1938; Stalnaker 1977). Hynes (1970) reported changes in bottom substrate movement with varying water velocities; Peterson (1977) stated that all substrates become less stable as velocity increases. The capacity of lotic (flowing) water to carry silt loads may also be increased or decreased by varying flow (Everhart and Dochrow 1970; Stalnaker 1977).

The effects of toxic pollutants and/or chemical enrichment may be intensified or alleviated by concentration or dilution of compounds during periods of fluctuating flows (Fraser 1972; Stalnaker 1977). Development of "blackwater conditions", which result from detritus buildups during periods of low flow, are discussed at length by Larimore et al. (1959).

Periphyton

Relatively few researchers have investigated the impact of changes in flow on periphyton (attached algae) communities. Hooper (1973) found current velocity to be influential in primary production and to be "one parameter that is greatly affected by flows. Whitford and Schumacher (1962) considered replacement of nutrient poor water near cell surfaces an important function of moving water. Under laboratory conditions, McIntire (1966) demonstrated that highest production rates occur in communities with faster current. At the termination of his experiments, however, accumulated organic matter per unit of substrate was the same at both velocities. Kroger (1973) found algae and other primary producers were destroyed during aperiodic exposures from regulated discharges. According to Ward (1976), stabilized flows below a dam provide an environment conducive to development of epilithic algae and other primary producers.

Riparian and Aquatic Vascular Plants

Most aquatic plants are associated with quiet pools and backwater areas (Peterson 1977). Few vascular plants tolerate high water velocities. Peterson (1977) reported that higher plants are generally restricted to velocities below 6 cm/sec but wild celery, river and sago pondweed, white buttercup and mud-plantain occur in fast water. Whitford and Schumacher (1962) found some plants require current velocities in excess of 150 cm/sec, apparently because of their high respiratory and mineral uptake rates. They suggested that the higher velocities provided more potential resources. Reduced flows may create favorable conditions for floating or rooted aquatic plants, and thus contribute to a marked change in a river's ecology (Hynes 1970). Encroachment of terrestrial vegetation into a stream is inhibited by high discharges (Fraser 1972).

Benthic Invertebrates

Water velocity is critical to the distribution, abundance, and productivity of aquatic macroinvertebrates. Radford and Hartland-Rowe (1971) attributed low productivity of an Alberta mountain stream to extreme variations in discharge. The relationship between stream velocity and invertebrate population levels is shown in Table 1. (Sampling techniques were not standard between these studies, therefore, comparisons of numbers of organisms at different velocities should only be made within individual studies.)

As discussed previously, flow characteristics affect the type of substrate found in the bottom of a stream. Needham and Usinger (1956), Hynes (1970), Hynes (1973), Bovee (1975) and Peterson (1977), discussed the importance of substrate composition in determining benthic invertebrate communities (see page Effect of Suspended Solids).

Decreased stream flow may cause many problems for invertebrates including: dessication or stagnation (Larimore et al. 1959), reduction in available living spaces (Armitage 1977), elimination of food producing areas (Waters 1964) disruption of life cycles (Thorup 1970), and changes in remaining habitats. Williams and Hynes (1977) grouped invertebrates into three categories which relate to their ecological requirements:

1) those species not well suited to life in intermittent streams but able to survive short term water level reductions;
2) species which can survive in either lentic or lotic (standing or flowing) waters; and,
3) species adapted to life in intermittent streams.

Groups two and three are characteristic of waters which periodically experience significant reductions in flow. Hynes (1958) reported the decimation of a community that was apparently not adapted to the low flow condition it was subjected to (i.e. apparently a group 1 community).

The mechanisms used by different benthic invertebrates to withstand low or no-flow conditions reflect the particular life cycle of that group and include: dormancy; transfer of activity to other suitable habitats; and retreat to favorable habitat until conditions return to normal (Williams and Hynes 1976, 1977). Figure 1 summarizes the methods employed by ephemeral stream fauna to survive periods of low flow. Stehr and Branson (1938) and Williams and Hynes (1974) also reported burying into the substrate as a means of avoiding dessication at low flows. Harper and Hynes (1970) discussed the diapause adaption of Allocapnia vivipara (Plecoptera) to survive low summer flows. They suggested the significance of this mechanism as a "pre-adaption" which enables it to conquer ephemeral streams as long as it can remain moist within the substrate.

During periods of low flow, stream invertebrates are reported to walk, fly, or migrate to pools (Stehr and Branson 1938) and drift at elevated rates (Armitage 1977) to find suitable habitats. The Chironomidae (Diptera) have been shown to be more resistent to dessication than other insect groups (Hinton 1953, Larimore et al. 1959 and Paterson and Fernando 1969). Fisher and LaVoy (1972) found very few insects in zones of fluctuation, exclusive of the chironomids. Paterson and Fernando (1969) showed an appreciable decline in the invertebrate population with increased exposure time of the substrate. Fisher and LaVoy (1972) demonstrated consistent declines in community diversity under similar conditions. Williams and Hynes (1977) reported accelerated life cycles at low flow conditions as a result of increased production from increased water temperatures.

High discharge levels may cause reductions in benthic invertebrate populations (Hynes 1968, Thorup 1970, Hoopes 1974 and Peterson 1977). Tebo (1955, cited in Hoopes 1974) attributed chronically low benthos levels in a south Appalachian Mountain river to the high frequency of floods. Hoopes (1974) found significant decreases in numbers of taxa, individuals, biomass and diversity per sample after a flood in a small Pennsylvania stream. Thorup (1970) also found reductions in the numbers of species and individuals although few were termed significant. Apparently no chronic effects are common or result from single event high discharges as both Hoopes and Thorup found near complete recovery within four months and termed the community "resilient" and having sustained "no permanent change" from the high discharges. Hynes (1968) observed recovery from a flood by the next breeding season and noted the inhabitants must have survived within the substrate, although *Baetis rhodani* (Ephemeroptera) did not recover.

Fishes

Many lotic fish species have adapted to "very finite qualitative requirements for water velocity," (Ambuhl 1959) and the significance of flow to most is well established (Fraser 1972). Water depth, velocity, and pool/riffle distribution has a profound effect on the species and sizes of fishes (several species of fish) present (Peterson 1977). Table 2 summarizes approximate preferred velocity ranges for some stream fish. High and low discharges are known to have both detrimental and beneficial effects on fish communities.

Reductions in the quality and quantity of suitable habitats for essentially all fish activities is the major problem associated with low flows. Losses of the highest food producing areas (riffles) and proximal feeding areas may occur during low discharges (Fraser 1972; Peterson 1977; Larimore et al. 1959). Further, as flows decrease, territorial requirements increase and the number of optimal feeding stations decreases, thereby reducing the carrying capacity of a stream (Chapman 1966, Fraser 1972). Bovée (1975) suggested that the ultimate carrying capacity of a stream is determined by its ability to supply food.

Tramer (1977) reported smaller benthic fishes, e.g. Etheostoma spp., may bury into the substrate to withstand short-term dessication. Kroger (1973) observed sculpins were killed when stranded on exposed bottom substrate after a rapid decrease in stream level.

Abnormally low temperatures as a result of low flows, may promote the formation of frazil (ice crystals) and anchor ice (Bovee 1975). Somme (1960, cited in Fraser 1972) found direct physical harm to fish by anchor ice, and Stalnaker (1977) reported clogging of fish gill arches by frazil ice.

Depressed discharges may have adverse effects on reproduction in fishes (Hooper 1973). Barriers and altered water velocity may delay or disrupt the normal routing, or change the speed of migration of spawning fishes (Fraser 1972). Spawning beds in the main stream may become too shallow for successful use, or filled with sediments because of the water's reduced transport capacities (Fraser 1972; Peters 1967; Peters 1962). Flows conducive to redd (nest) construction and egg incubation may be destroyed and spawning areas adjacent to main channels may be exposed. Larimore et al. (1959) found that fishes seek out pools and can survive low water periods there if the dissolved oxygen concentration and temperature do not exceed their tolerances. Tramer (1977) showed that fishes having underslung mouths and/or lacking air bladders had the highest mortality rate in this situation because of their inability to gulp the oxygenated surface film.

High water levels may also affect fishes. Food production areas (Larimore et al. 1959) and adjacent cover and spawning areas are expanded (Fraser 1972). Subsurface water flow within the substrate necessary for egg incubation is maintained, and territorial requirements decrease (Pearson 1966; Fraser 1972) thus increasing the carrying capacity of the stream. Although high flows may have additional positive impacts such as providing necessary stimulus for spawning, they may block migration, render spawning areas unsuitable, or cause successfully lain eggs to be swept away and buried (Peterson 1977). Hynes (1970) suggested that for physiological reasons, fishes cannot tolerate extended periods of heavy swimming. Peterson (1977) observed a reduction in diversity of fishes with increasing stream velocities. White (1975) found trout production in a Wisconsin stream to be greater in high water years and poorer in low water years and concluded that the limiting factor in trout abundance is year to year fluctuation in the flow regime.

Ecological Considerations

"Stream organisms have adapted to various special conditions presented by a flowing water environment. Many species have adapted to a rather limited range of velocities or depths and their dependence upon sufficient flow to provide these conditions is usually complete" (Fraser 1972). Extreme flow conditions may cause considerable reduction in the standing crops of aquatic flora and fauna and have an additive effect when other harmful physical conditions occur simutaneously (Peterson 1977).

Fraser (1972) estimated a range of impacts from virtual "elimination of an aquatic environment to an improvement or enlargement of the biota in terms of desired species or total biomass." Kroger (1973) reported that a single water level drawdown had a drastic effect upon productivity and suggested multiple level reductions would have correspondingly greater impacts. Larimore et al.

(1959) observed low water levels to cause increased predation pressure and vulnerability both from within the system and from terrestrial predators.

Mitigation

Many methods for evaluating instream flows and their biological implications have been developed to protect aquatic resources and mitigate existing impacts. Stalnaker and Arnette (1976b) extensively discuss methods for determining the affects of altered flows on aquatic fauna and list preferred velocities for some aquatic insects (Table 3). Hooper (1973) gives suggestions for the collection and analysis of data for stream flow evaluations with emphasis on microhabitat analysis of relationships between flows and ecology, behavior and environmental requirements of fish and invertebrates. Computer analysis of field measurements was used by Waters (1976) to quantitatively express the relationships between streamflow and available food producing, spawning, cover and resting microhabitats for trout.

Peterson (1977) discusses control of surface run-off and creation of reservoirs for controlling stream flows. Impoundment and release of reservoir water can provide reasonably consistent flows that do not fluctuate excessively (Ward 1976) and provide sufficient water for successful spawning in spring (Peterson 1977).

Several authors have recommended minimum flows necessary to protect aquatic life. Tennant (1976) discussed the Montana Method of determining flows and recommended: 10% of the average annual flow as a minimum instantaneous base flow to avoid catastrophic degradation of the inhabitants; 30% as adequate to sustain good survival; and 60% as providing outstanding habitat conditions. Kroger (1973) recommended that flow reductions not exceeding 2.8 m^3/sec/day. Tennant (1976) suggested that flow reductions should not exceed 15 vertical centimeters in six hours. Peterson(1977) suggested maintaining an average depth in riffle zones of 30 centimeters. Pearson et al. (1970) advised "setting an optimum flow to cover the greatest amount of riffle with water flowing at 60 cm/sec". Optimum food producing area will be provided when the maximum surface acreage with depths of 15-90 cm and a velocity of 45-106 cm/sec is maintained (Banks et al. 1974).

EFFECTS OF CHANNELIZATION AND DIVERSION

Channelization of streams and rivers is widespread in the United States. Channelization or channel modification is generally employed to move water downstream faster in order to eliminate flooding problems in upstream areas. Channelization is sometimes necessary for construction of highways located adjacent to stream channels or for the installation of culverts and bridges. The physical effects of channelization include:

1) uniform depth and current velocity;
2) loss of stream length and sinuousity (meandering);
3) loss of pool-riffle interspersion;
4) higher current velocities;
5) increased suspended solid load;
6) increased bank erosion if banks are not stabilized;
7) greater daily temperature fluctuations;
8) abnormally low stream discharge during low flow periods;
9) uniform bottom substrate and reduced habitat diversity; and
10) loss of riparian vegetation and, thus, much of the allocthonous energy input (organic inputs form external sources).

In addition to these effects within the channelized sections, downstream effects such as increased siltation, greater likelihood of flooding, and greater water level fluctuations may result.

Diversions are used to redirect a stream channel so that land-use developments can proceed in the area of the original channel. The physical effects of diversions on streams are basically the same as those described for channelization. No studies on the biological effects of diversion are available but these effects would probably be similar to those of channelization.

Periphyton

Little data is available on the effect of channelization on periphyton communities. Duval et al. (1976) observed no difference in the composition and abundance of periphyton from channelization and unchannelized sections. It might be expected that habitat changes and the swift, even current found in channelized streams would prevent development of abundant periphyton communities. (See page Altered Flow and page , Suspended Solids) Removal of riparian vegetation, however, would increase the amount of sunlight reaching the stream and could increase primary productivity.

Aquatic Vascular Plants

No studies have been conducted on the effect channelization has on aquatic macrophytes. As is the case of periphyton one might expect a reduction in macrophyte production because of habitat changes, habitat loss, higher suspended solids and increased current velocities. Since most stream macrophyte development occurs in quiet areas (Hynes 1970) few if any macrophytes would be expected in channelized streams when these areas are eliminated. (See page Altered Flow and page Suspended Solids for discussions on the effect of these changes on aquatic macrophytes.)

Benthic Invertebrates

Several authors have reported reductions in benthic invertebrate production in channelized streams. A reduction in habitat diversity is generally thought to be the primary factor responsible for this reduction (Arner et al. 1976; Crisp and Crisp 1974; Moyle 1976). Changes in species composition have been observed in other studies. Etnier (1972) observed that chironomid and oligochaete diversity and density were unaffected by channelization while mayfly (Ephemeroptera caddisfly (Trichoptera), and stonefly (Plecoptera) diversity and abundance were reduced. Although Hansen (1972) reported little difference in invertebrate composition from channelized and unchannelized zones, he did note higher chironomid (Diptera) population levels in unchannelized zones and higher hydropsychid (Trichoptera) population levels in channelized zones. Dodge (1976) found slightly higher macroinvertebrate diversities in unchannelized streams than in channelized streams.

A number of studies have reported little effect on the benthic invertebrate community. King and Carlander (1976), Kennedy (1955, cited in King and Carlander 1976), Barton et al. (1972, cited in King and Carlander 1976), and Morris et al. (1968) all found few differences in the benthic community of channelized and unchannelized stream sections. These results may be due to the fact that minimal changes in substrate resulted from channelization in the areas studied by these investigators.

In most studies of channelized steams little regard was given to overall habitat loss. Morris et al. (1968) found little difference in the benthic invertebrate standing crops in channelized and unchannelized stream sections. He did report a 67% reduction in benthic habitat in channelized sections. Other studies have reported up to a 55% reduction in stream length (Hansen 1972; Congdon 1972). A reduction in carrying capacity at all trophic levels is the probable result of this type of loss.

Drift through channelized stream sections has been studied in order to determine the impact of channelization. The results of these studies are inconsistent. Morris et al. (1968) reported drift rates of 8 g/acre-foot in channelized sections and 68 g/acre-foot

in unchannelized sections. Morris found little similarity between the benthic and drifting insects. In contrast, high drift rates through channelized streams were reported by Hansen (1972). He theorized that this was the result of inadequate substrate for colonization in channelized streams. King and Carlander (1976) found very little difference in drift rates through channelized and unchannelized stream sections. These investigators felt that the length of channelized stream sections in their study area was not great enough to affect drift rates.

Two studies in Iowa attempted to relate drift rates to stream morphometry in order to facilitate the prediction of the impacts from channelization (Buckley et al. 1976 and Zimmer and Bachman 1976). A significant positive correlation was found between the number of drift organisms and streams sinuousity in both studies, but no significant correlations were found between drift rates and other morphometric parameters. They also reported that the correlation between sinousity and drift was affected by the amount of debris in the stream. For further information on the effects of habitat alteration, turbidity and flow changes, and problems associated with channelization see page , Altered Flow and page , Suspended Solids.

Fishes

Reduced populations of fishes in channelized streams are reported by various investigators (Beland 1953, Whitney and Bailey 1959, Bayless and Smith 1967, Elser 1968, Irizarry 1969, Wharton 1970 (cited in Congden 1971); Tarplee et al. 1971 (cited in Congden 1971); Etner 1972, Hansen 1972, Arner et al. 1976, Duval et al. 1976, King and Carlander 1976, Lund 1976, and Moyle 1976). In most of these studies, lack of habitat diversity, particularly loss of cover, was responsible for the decrease in fish standing crops.

Specific reductions of the population of fishes in channelized stream sections reported in the studies listed above are as follows:

1) 9% reduction in number and weight of game fishes greater than 6 inches, 85% reduction in number and 76% reduction in weight in game fishes less than 6 inches (Whitney and Bailey 1959);

2) 90% reduction in total weight and number of game fishes greater than 6 inches (Bayless and Smith 1967);

3) 8 times more fish production in natural streams, 7 times more catchable trout, and 10 times more white fish (Irizarry 1969);

4) 77% reduction in standing crop (Tarplee et al. 1971, cited in Congden 1972);

5) 98% reduction in standing crop (Wharton 1970, cited in Congden 1972);

6) 21 species and 304 lbs/acre in unchannelized sections, 13 species and 53 lbs/acre in channelized sections (Congden 1972), and

7) 67% reduction in standing crop (Moyle 1976).

Duval et al. (1976) reported the absence of legal sized trout in channelized streams. Similarly, Arner et al. (1976) found that the average largemouth bass was eight times larger in unchannelized sections than in channelized streams. Loss of stream length was not considered in these studies. If considered, this factor would probably increase the estimated overall loss in fishery production in channelized streams.

Changes in the species present have been observed in many channelized streams. Arner et al. (1976) reported that rough fishes dominated channelized sections while sport fishes were more abundant in unchannelized sections. Trautman and Gartman (1974) tabulated differences in fish species found in channelized and unchannelized stream sections. They found that the following species had been eliminated in channelized areas: central mudminnow, grass pickerel, golden shiner, horny head chub, mimic shiner, tadpole madtom, and pirate perch. Creek chubs, common shiners, and spotfin shiners were tolerant of channelization.

Hansen (1972) observed that swift water areas in channelized sections were not utilized for feeding. He also stated that movements of fishes mask the effect of channelization. Greater fish movements in channelized streams were also reported by Menzel and Fierstine (1976).

Although no studies have dealt directly with channelization effects on spawning, Hynes (1960) does mention elimination of spawning areas as an effect of channelization. Changes in substrate and flow characteristics could be expected to affect spawning behavior and success (see page Altered Flow and page , Suspended Solids).

Daily temperature fluctuations in channelized streams may also present a potential problem to fishes. Temperatures approaching the lethal level for walleyes were found by Hansen (1972). Duval et al. (1976) mention temperature as a major factor in reducing trout populations in channelized Pennsylvania streams.

Several studies have investigated the relationship between stream morphometry and populations of fishes. In rocky streams, populations of fishes increased while in sandy streams there was no relationship (Buckley et al. 1976). Several studies have investigated the relationship between morphometry and populations of fishes. Populations show a positive correlation with stream sinuousity according to studies by Buckley et al. (1976). Menzel and Pierstine (1976) reported no correlation between populations of adult fishes and stream sinuousity but a strong positive correlation between populations of juveniles and sinuousity.

Ecological Considerations

Channelization reduces available ecological niches through direct loss of stream length and a decrease in habitat diversity. This reduces the overall carrying capacity of the stream at all trophic levels. On small streams the loss of riparian vegetation reduces productivity at all trophic levels since allochtonous material is the major energy source in these systems (Peterson and Cummins 1974).

Mitigation

Various mitigating measures are available to reduce the effect of channelization on aquatic ecosystems. Based on their work relating stream morphometry to the biota, Zimmer and Bachman (1976) recommended increasing stream sinuousity as a means of mitigating the effects of channelization. Barton et al. (1972) reported no change in populations of fishes where suitable substrates such as wire gabions, large rocks and riprap were provided in the channelized sections. Lund (1976) made several recommendations for mitigating the effects of channelization including:

1) Alter original stream channels only when absolutely necessary and then keep alterations to a minimum. When a lengthy channelization occurs, meander the new channel as much as possible to correct ditchlike appearance and to retain original stream length;

2) Vegetation (especially trees and shrubs) along new channels should be retained, if possible, to provide bank stability and shade. When topsoil and vegetation are lost, banks should be sloped and topsoil replaced and reseeded down to the high-water mark. Trees and shrubs such as red dogwood, willow, common chokecherry, alder, and birch should be planted along stabilized stream banks;

3) When riprap is needed to hold the stream in a new channel, it should be covered with subsoil and topsoil down to high-water mark and then revegetated with grass, trees and shrubs;

4) Jetties, random rock clusters and other in-stream devices used to create pools, must be properly engineered to withstand the annual high-water and the occasional floods which occur. If riprap material is used to construct mitigating devices, it must be large enough to prevent hydraulic water pressure from spreading it out and burying it;

5) In gravel bottomed streams, jetties and other mitigating devices should be placed 5-7 stream widths apart (alternating from each streambank) to match pool-riffle and meander sequences found in unaltered sections;

6) Mitigating structures must be placed in the currents close to the thalweg to be most effective in providing trout habitat; and

7) Random rock clusters and jetties could be used together (cluster near outer end of jetty) to create larger mid-channel pools.

Bayless and Smith (1967) and Buckley et al. (1976) recommended other procedures for mitigating the effects of channelization which include: 1) no long reach (>1.0 km) channelizations; 2) leaving as much meander in stream as possible; 3) use of proper bank stabilization structures; 4) structures should be added to create pools at low flow but not obstruct water movement at high flow; 5) turning oxbows into ponds; 6) replacing fish producing water acre for acre; and, 7) using flood water retarding measures rather than channelization.

Unless mitigating measures are applied to a channelized stream, there will be major long term impacts on the biota of the stream.

EFFECTS OF SUSPENDED SOLIDS, SEDIMENTATION, AND TURBIDITY

Suspended solids occur naturally in lentic and lotic systems. The amount of sediment present is dependent on the type and condition of the watershed, season, and stream discharge. Distrubance of a watershed, stream, or lake can greatly influence the amount of suspended solids in the system.

The primary source of suspended solids is erosion from the watershed. Man's activities increase the potential for erosion. Agriculture, logging, highway construction, mining activities and industrial construction activities offer high potential for erosion and increased suspended solids. A direct stream disturbance such as dredging or construction of a bridge will also increase the sediment load of a stream.

Particle size and stream velocity are the two critical factors which determine the response of suspended solids in streams. Coarser materials settle rapidly while finer silt particles remain in suspension longer. During spring runoff and other periods of high flows, suspended solids travel further downstream before deposition than during low flow periods. After deposition, sediment may move downstream during freshets and be redeposited. The type of material moved, and the distance it is moved, depends upon the magnitude and duration of the flow.

Suspended solids have three basic effects: 1) inert solids may have a direct effect on organisms while in suspension; 2) they increase the turbidity of the water reducing light penetration; and 3) upon deposition, they may alter the bottom substrate. Suspended solids may also affect the toxicity of other compounds such as heavy metals.

A number of comprehensive reviews on the biological effects of suspended solids are currently available and are listed as follows: Cordone and Kelley (1961); EIFAC (1964); Everhart and Duchrow (1970); Gammon (1970); Alabaster (1972); Ritchie (1972); Hynes (1973); Rosenberg and Snow (1975); and Sorenson et al. (1977).

Standards to protect biological systems from the effects of suspended solids proposed by EIFAC (1964) and EPA (1976). These standards will be reviewed in a later section.

Periphyton

Very few studies have dealt with the effects of suspended solids on the periphyton community. Reduced photosynthesis because of poor light penetration is probably the most obvious effect on periphyton (Sorenson et al. 1977; Rosenberg and Snow 1975; Ritchie 1972; Tarzwell and Gaufin 1953, cited in Cordone and Kelley 1961). Phinney (1959, cited in Cordone and Kelley 1961) felt that there was a two fold effect on primary production; increased turbidity reducing the photosynthetic rate and sediment accumulation preventing free exchange of O_2 and CO_2. Both Swale (1964, cited in Sorenson et al. 1977) and Lund (1969, cited in Sorenson et al. 1977) reported that light penetration was the major factor limiting algal production in the River Lee.

McGaha and Steen (1974) observed that under turbid conditions phytoplankton populations changed from green and blue-green species to diatom species. They concluded that the high silica content of the suspended solids stimulated this increase in diatoms.

Habitat changes also affect the periphyton community (Sorenson et al. 1977, and Cordone and Kelley 1961). Many periphyton species are adapted to attaching to rubble substrates and find sand and silt unsuitable for survival. Smothering of periphyton occurs during deposition of the suspended solids. If the sedimented sand is unstable, no long term periphyton community will develop. Scouring of periphyton can be expected by either the bed load or the suspended materials.

Various observations have been made on algae growing under high sediment conditions. Cordone and Pennoyer (1960, cited Cordone and Kelley 1961) found that sediment in the Truckee River, California virtually eliminated abundant growths of Nostoc spp. A decrease in algal genera from 24 to 16 was observed in a small impoundment with heavy sedimentation (Samsel 1973, cited in Rosenberg and Snow 1975).

Aquatic Vascular Plants

Reduced light penetration may cause a major reduction in macrophyte growth (Hynes 1960; Edwards 1969). As well as lowering macrophyte productivity, suspended sediments cause changes in community composition (Sorenson et al. 1977). Peterson (1977) states that few or no aquatic plants are found in turbid streams. Changes in substrate composition and stability also cause reduced macrophyte development. Edwards (1969) indicated that physical change in the substrate was an important factor causing reduced macrophyte growth during sedimentation. Jones (1949, 1958) concluded that the shifting substrate in the River Rheidol was responsible for the lack of rooted vascular plants. A similar conclusion was made by Nuttal (1972) during an investigation of the River Camel. Substrate movement also may have an abrasive effect upon rooted aquatic plants (Edwards 1969).

Plant succession is affected by suspended solids. Edwards (1969) stated that there is deposition of suspended materials in the lee of macrophyte beds. As this material builds, a change in species and their distribution may occur. If this buildup of sediment continues, smothering of plants may occur. Minckley (1963, cited in Hynes 1970) found the Nitella flexilis colonized the silt bank formed by Potamogeton diversiformis. Myriophyllum heterophyllum colonized Fissidens julranus beds which were eventually smothered by siltation.

Benthic Invertebrates

Very little data exists regarding the direct effect of suspended solids on benthic invertebrates. Hamilton (1961), Nuttal (1972), and Nuttal and Bielby (1973) concluded that abrasion did not adversely affect invertebrates in the systems they studied. Hamilton (1961) observed that mayfly gills were coated with silt in a stream polluted with suspended clay silt. Suspended solids have been found to interfere with feeding and respiration of clams and other filter feeding shellfish (Ellis 1936; Kemp 1949; Brehmer 1965).

The immediate result of sedimentation is either to smother the benthic invertebrates (Ellis 1936) or to force them to move to a more favorable habitat. Several studies have measured the effect of sediment addition on invertebrate drift. Invertebrate drift increased proportionally with sediment additions up to 160 mg/l (Gammon 1970). In this experiment all invertebrate species reacted similarly to sediment additions.

Bjornn et al. (1974) measured an increase in drift as sediment was added to riffles but were unable to correlate this with any decrease in density of bottom fauna. Drift rates returned to normal within one day in this study.

Sediment additions to stream riffles by Rosenberg and Snow (1975) caused significant increases in macroinvertebrate drift. They did not find a linear relationship between drift rates and sediment additions as did Gammon (1970). Further analysis of this data by Rosenberg and Wiens (1975) indicated that chironomid drift rates increased with sediment additions while other groups responded irratically. The invertebrate drift response in this system was studied further by Rosenberg and Snow (1977). They found that drift increased rapidly as sediment was added to experimental channels, tapered off, and then increased again as sediment addition continued. Rosenberg and Snow postulated that sensitive animals are stripped from the substrate immediately while more tolerant animals withstand sedimentation for a longer period before they finally succumb and drift from the area. Herbert et al. (1961, cited in Alabaster 1972) also found higher drift rates in streams with higher suspended solids concentrations even though benthic invertebrate densities were far lower in affected streams.

Significant substrate composition changes may result from sedimentation and as a result, the indigenous fauna changes to one adapted to the new conditions. Substrate preferences of benthic invertebrates are well documented (Hynes 1970; Cummins and Lauff 1969; Brusven and Prather 1974).

Generally, animals in the families Chironomidae (Diptera) and Tubificidae (Oligochaeta) dominate silty conditions, attaining large population size under certain conditions. As the average substrate particle size increases, there is an increase in the number of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera). This change in relative abundances is illustrated in Table 4.

Rubble substrates generally have the most diverse and abundant associated fauna. A silt substrate may have very large standing crops but very low diversity (Percival and Whitehead 1924; Bell 1969; Wene and Wickliff 1940; Hynes 1970; Crisp and Crisp 1974).

In most studies where sedimentation has changed substrate composition, a decrease in secondary productivity has been reported. Nuttal and Bielby (1973) found that control stations supported 36 times more invertebrates than stations subjected to clay pollution. They felt that this effect resulted from covering of the substrate rather than from increased turbidity or an abrasive effect. They also observed an increase in burrowing forms such as turbificids and chironomids in the polluted sections. A small southern Appalachian trout stream affected by silting had significantly larger standing crops at control stations than at affected stations (Tebo 1955). Herbert et al. (1961, cited in Alabaster 1972) observed a large decrease in bottom fauna production in streams with suspended solids of 1030 mg/l and 5800 mg/l. Gammon (1970) correlated the amount of sediment input with bottom fauna changes. An increase of 20-40 mg/l solids resulted in a 25% reduction in benthic invertebrates; 80 mg/l solids caused a 60% reduction in invertebrates. The reason for this change in invertebrate populations was not given. Complete bottom fauna elimination has been recorded in areas of stream sedimentation (Cordone and Kelley 1961).

Studies by Chutter (1969) in South Africa, demonstrated that in areas subjected to small increases in suspended solids there was a decrease in the number of taxa present but there was little effect on the density of invertebrates. Other investigations have found somewhat different results. Forsage and Carter (1974), Gammon (1970), and Casey (1959, cited in Cordone and Kelley 1961) observed a reduction in invertebrate densities but species composition remained relatively unchanged under conditions of sedimentation. Gammon (1970) found an inc: : e in the abundance of the mayfly Tricorythodes; an organism often found in silted stream areas (Edmunds Jr. et al. 1976).

Results from other sedimentation studies such as: Bjornn et al. (1974), Pearson and Jones (1975), Rosenberg and Snow (1975) and Barton (1977) have been inconclusive or have shown no effect on benthic invertebrates.

Fishes

Direct damage to fishes as a result of suspended solids has been documented. Ellis (1937, cited in Ritchie 1972) and Kemp (1949, cited in Ritchie 1972) noted gill clogging by suspended ferric hydroxide was a factor in fish deaths in the River Daha, India (Ray and David 1962). Clogging seemed to contribute to the stress of other toxic conditions. Gill damage and death was also observed in fishes subjected to high suspended solids (810 and 270 mg/l) under laboratory conditions (Herbert and Merkens 1961). Similar gill damage was noted in fishes collected from rivers with high suspended solids (1030 and 5800 mg/l) by Herbert et al. (1961, cited in Alabaster 1972). Damage included thickening and occasional fusion of epithelial cells of the secondary lamellae. Herbert et al. noted that streams with 0-60 mg/l suspended solids supported normal trout populations while those streams with 1030 and 5800 mg/l suspended solids did not.

Wallen (1951) observed opercular cavities and gills clogged with silt in dead fishes subjected to clay silt up to 225,000 mg/l. In this laboratory study of 16 fish species, no distress was noted at silt concentrations less than 20,000 ppm. Death occurred at levels of 50,000 ppm or higher, depending on the species. These results suggested that gill damage and death are produced only at extremely high silt concentrations. This is in conflict with the results of other studies such as Herbert and Merkens (1961), Herbert et al. (1961, cited in Alabaster 1972)

Campbell (1954, cited in Everhart and Duchrow 1970) who found that caged rainbow trout were killed in 20 days at concentrations of 1000-25000 mg/l in Power River, Oregon.

Data cited by Alabaster (1972) indicate that fish will remain in streams which are occasionally subjected to high suspended solids if the average concentration is low (i.e. 0-50 mg/l). Alabaster also stated that streams with higher than 100 mg/l suspended solids are virtually fishless.

Seasonal effects of suspended solids have been noted. Fish are most sensitive to suspended solids during spring months according to Gammon (1970). Bjornn et al. (1974) concluded that winter sediment additions were more detrimental to fish than were summer additions. Larval fish are more susceptible to the effects of suspended solids than adults since they lack the ability to clean their gills by mucus secretions (Everhart and Duchrow 1970). Low level turbidity (1 to 28 ppm) had no observed effect on the growth and survival of larval lake herring although at higher turbidities the larvae remained closer to the surface (Swenson and Mattson 1976). These investigators postulated that this had a positive influence on larval herring survival. No data are presently available on the acute or chronic levels of suspended solids dangerous to larval fish.

Several investigators have examined the effect of high turbidity on fishes. The reason for the observed effects are not always clear in these studies. Buck (1956) studied ponds with turbidity ranging from 25ppm to > 100ppm. Largemouth bass were found to be the most affected by turbidity while flathead catfish appeared to be best adapted to turbid conditions. Clear ponds were 1.7 to 5.5 times more productive than turbid ponds.

A significant reduction in largemouth bass activity in turbid water was reported by Heimstra and Damkot (1969). Green sunfish activity was reduced under similar conditions but not significantly. These investigators also observed more "coughing" and "scraping" in test fish than in control fish. No effect was measured on either attack or feeding behavior although there appeared to be a breakdown of sunfish attacking hierarchies.

Horkel and Pearson (1976) studies green sunfish ventilation rates under various turbidity and temperature combinations. Ventilation rates increased 50-70% at turbidities of 1012 FTU (Formagin Turbidity Units) at 15° and 898 FTU at $25^{\circ}C$. They concluded that this increase in ventilation rates compensated for reduced respiratory efficiency since oxygen consumption had not changed.

Morphological changes in fishes have been found under conditions of high turbidity (Hubbs 1940, cited in Horkel and Pearson 1976). These changes included: reduced eye size, increased size of other sense organs, and changes in body form, contour, fish development and color.

Habitat alterations by sedimentation also affects fish populations. The effect of sedimentation on trout spawning has been extensively studies. Peters (1962; 1967) studied the effect of sediment on rainbow trout eggs. He found that as sediment filled the interstices in gravel, inter-gravel dissolved oxygen concentrations decreased resulting in decreased embryo survival. Hausle and Coble (1976) also found that sand in trout spawning areas reduced the number of emerging trout fry and slowed emergence. These effects were caused by reduced water movement through the redds that resulted in reduced dissolved oxygen and slow removal of metabolic products. Campbell (1954, cited in Everhart and Duchrow 1970) reported 100 percent mortality in eggs placed in a stream with high sedimentation compared with 6% mortality in a clear tributary stream. Other data cited by EIFAC (1964) corroborate these observations.

Fish eggs that are not buried in the bottom are also affected. Silt particles may adhere to the surface of eggs and prevent the exchange of oxygen and carbon dioxide thus causing death (Stuart 1953). Hassler (1970) recorded 97% mortality in northern pike eggs coated with 1 mm of silt. Destruction of yellow perch eggs by high silt concentrations was reported by Muncy (1962).

Bjornn et al. (1974) concluded that sediment in riffles did not produce a population decrease in juvenile steelhead trout or chinook salmon until the pools began filling with sediment. Gammon (1970) observed large decreases in fish densities, except for spotted bass, with increased suspended solids up to 150ppm; no further decreases in fish populations were noted until sediment filled the pools.

As sedimentation increases, habitat diversity decreases resulting in a corresponding decrease in fish populations. Saunders and Smith (1965) found lower standing crops of trout during years of high sedimentation. Sedimentation resulting from highway bridge construction caused a decrease in fish standing crop from 24 to 10 kg/ha in a small stream in Ontario (Barton 1977). Barton observed that many hiding places were filled by silt. In areas of heavy siltation, increases in rough fishes and reductions in sport and forage fishes were recorded by Forshage and Carter (1974) and Peters (1967).

Ecological Considerations

The overall effects of suspended solids on aquatic ecosystems have been discussed by various authors. EIFAC (1964) lists the overall affects of suspended solids on fishery resources as follows: 1) by acting directly on the fish swimming in water in which solids are suspended, and either killing them or reducing their growth rate and resistance to disease; 2) by preventing the successful development of fish eggs and larvae; 3) by modifying natural movements and migrations of fish; 4) by reducing the abundance of food available to fish; and 5) by affecting the efficiency of methods for catching fish.

Ritchie (1972) outlined the overall damage to the aquatic ecosystem as follows: 1) reduction in primary production 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 bottom organisms from a change in habitat; 5) reduction in feeding efficiency of fish; and 6) loss or change in fish habitat.

Mitigation

The literature indicates rapid physical recovery from high suspended solids after levels are reduced (see page). Traditional methods of erosion control which can be applied to reduce suspended solids, include, streambank stabilization, revegetation of upland areas, and leaving buffer strips of vegetation along streams. In cases where erosion precautions do not limit the level of suspended solids other methods are available to mitigate the effects of sedimentation

Luedtke et al. (1973) described the following three methods to reduce the effects of sediments: 1) using gabbion construction to increase the flow velocity in areas of sedimentation to eliminate low gradient sandy stretches; 2) employing log drop structures to increase turbulence and thus the scouring of fine sediment; and 3) removing debris dams that create areas of sedimentation. Log drop structures are not effective in low gradient situations, but can improve stream habitat in flat stream sections. Hanson (1973) recommended construction of sedimentation basins as sink areas to reduce the effects of suspended solids. These basins are constructed by dredging a depression in a quiet section of stream or impounding a similar site.

Based on their review of literature, EIFAC (1964) suggested that inert suspended solids in a range of concentrations, differentially affect fisheries. EIFAC proposed four categories of effects.

1) There is no evidence that concentrations of suspended solids less than 25ppm have any harmful effects on fisheries.

2) It should usually be possible to maintain good or moderate fisheries in waters which normally contain 25 to 80ppm suspended solids. Other factors being equal, however, the yield of fish from such water might be somewhat lower than in category 1.

3) Waters normally containing from 80 to 400ppm suspended solids are unlikely to support good freshwater fisheries although fisheries may sometimes be found at the lower concentrations within this range.

4) At the best, only poor fisheries are likely to be found in waters which normally contain more than 400ppm suspended solids.

> In addition, although several thousand ppm solids may not kill fish during several hours or days exposure, such temporarily high concentrations should be prevented in rivers where good fisheries are to be maintained. The spawning grounds of salmon and trout require special consideration and should be kept as free as possible from finely divided solids.

EPA (1976) recommends the following suspended solids criteria for the protection of aquatic life:

> Settleable and suspended solids should not reduce the depth of the compensation point for photosynthetic activity by more than 10 percent from the seasonably established norm for aquatic life.

Before applying suspended solids standard to an individual stream one must take into account stream gradients and stream flows (Peters 1967). Alabaster (1972) cautions that it is misleading to use standards for suspended solids to predict the impact on spawning and hatching success since additional factors are important in determining how suspended solids react in the system.

RECOVERY OF LOTIC ECOSYSTEMS FROM PHYSICAL IMPACT

Destruction or alteration of aquatic ecosystems may result from many different, but often inter-related, impacts. The biological implication of physical impacts have been discussed in previous sections (see page Altered Flows, Channelization page , Solids page) This section summarizes literature on the rates and processes involved in the recovery of stream ecosystems.

Periphyton and Aquatic Vascular Plants

Little information has been published regarding recovery and recolonization mechanisms of algal communities. Benthic diatom and unicellular algal drift has been reported by Blum (1954, 1956) and Muller-Haeckel (1966; 1967; 1969; 1970a; 1970b; 1970c; 1971; 1973a; 1973b; cited in Muller 1974). Hynes (1970) states "most species are available at all times; they flourish when conditions become suitable, and many are simply opportunists. We know very little, however, about the means of dispersal of many types of algae and cannot at this time explain how species actually get to headwaters and remain there despite the summation of downstream movement to which they must be subject." Based on data presented by the above authors, it appears that recolonization by algae occurs by drift if an upstream source is available and/or by other mechanisms soon after conditions again become suitable.

The narrow range of physical factors which aquatic macrophytes tolerate (see Aquatic Vascular Plants page) directly influences their recovery following physical perturbations. Recovery is expected when current velocity, substrate stability, turbidity and other physical conditions stabilize and are maintained long enough to allow redevelopment.

Benthic Invertebrates

Numerous researchers have discussed the recovery of stream macroinvertebrate communities after alleviation of adverse physical conditions (Wene and Wickliff 1940; Waters 1964; Crisp and Gledhill 1970; Thorup 1970; Hooper 1973). Thorup (1970) studied the influence of a short term freshet, and found that communities adjusted to constant ecological conditions did not sustain permanent damage from short term radical changes in environmental conditions.

Hynes (1960) reported that invertebrates recolonize impacted areas at different rates. Waters (1964) made similar observations and concluded that this may temporarily lead to abnormally high population levels of pioneer species. Community equilibrium is quickly re-established, however, when all species reappear.

The rate at which stream invertebrates return to "background" diversity and populations levels is dependent on the severity and extent of the damages sustained (Cairns 1971). Waters (1964) suggests that the populations of invertebrates that depend on macrophytes for a "significant ecological function" will not return to normal before macrophytes recover.

Short recovery times are reported by the majority of authors. The recovery of invertebrates after flooding (Stehr and Branson 1938) and sedimentation (Gammon and White 1970) was "immediate". Invertebrate recovery was "rapid" following: road construction activities (Barton 1977); channelization (Lund 1976); and substrate exposure and freesing (Paterson & Fernando 1969). "Appreciable" recovery following substrate scouring by flooding was observed by Thorup (1970) and Hooper (1973). Pearson and Jones (1975) noted "fairly complete" restoration within five months after dredging of a British chalk stream. A dredged chironomid/oligocheate/mollusc community recovered within one year (Crisp and Gledhill 1970). Figure 2 summarizes the reappearance of 26 stream insect taxa following five months of zero flow in an Illinois warm water stream.

Recovery of stream invertebrates may result from either reproduction by surviving inhabitants (Stehr and Branson 1938; Larimore et al. 1959), recolonization by new individuals or a combination of these two. Williams and Hynes (1976) studies an Ontario stream and observed the relative importance of four types of invertebrate repopulation: 1) drift (41%); 2) ovipostion (28%);

3) substrate migration (19%); and 4) upstream migration (18%). These results are substantiated by other studies of invertebrate recolonization (Muller 1954; Kennedy 1955; Tebo 1955; Waters 1964, 1966; Brusven 1970; Hooper 1973; Bjornn 1974; Luedtke and Brusven 1976; Williams and Hynes 1976b). Brusven (1970) concluded that, "drift is the most prolific and viable means of recolonization". Waters (1964) regarded drift as a "fully sufficient recolonization mechanism for denuded areas if a good source of drifting organisms is available". Normal drift levels and composition may not recur until the populations of insects with both long (e.g. caddisflies) and short generation times (e.g. midges) return to the carrying capacity of the substrate (Dimond 1967).

Recolonization by upstream ovipository flights of adult aquatic insects may also be important (Stehr and Branson 1938; Muller 1954; Kennedy 1955; Hultin et al. 1974; Williams and Hynes 1976b). Muller (1954) developed the "colonization cycle" concept in which upstream flights compensated for downstream drift of larvae. Muller (1974) cites numerous studies which provide evidence of upstream migrations in the orders Ephemeroptera, Plecoptera, and Trichoptera. This "colonization cycle" is not universally accepted and is considered by some "largely untested and conjectural for most species (Brusven 1970).

Migration within and on the substrate was considered by Williams and Hynes (1976b) to represent 19.1% of total recolization. Luektke and Brusven (1976) listed current velocity, substrate composition and species present as factors controlling instream invertebrate migration.

Various authors report that instream upstream migration is an important invertebrate recolonization mechanism (Bishop and Hynes 1969; Hultin et al. 1969; Brusven 1970; Elliot 1971; Luedtke and Brusven 1976). It has been reported to be 5-30% of downstream drift. Williams and Hynes (1976) reported it accounts for 18.2% of total recolonization. Hultin et al. (1969) regarded underwater upstream movements as "as significant component of the dynamics of amphibiotic insect". In contrast Bishop and Hynes (1969) found upstream movements by invertebrates accounted for only 6.5% by numbers and 4% by weight of the drift. Williams and Hynes (1976b) found that many groups have a preferred method of recolonization. Therefore, if any method is excluded, a different faunal assemblage may result.

Fishes

Few detailed reports on the recovery of stream fishes from physical impacts are available. Some authors have reported recurrance of fishes in general terms. Barton (1977) and Hamilton (1961) reported the reappearance of fishes following cessation of sedimentation problems.

Larimore et al. (1959) detailed the movements of fishes into a drought decimated stream (Figure 2). They reported that twenty-one of twenty-nine previously collected species were found soon after the first instance of high water (week 8) following reinitiation of flow.

Distributional changes of species between pre-drought and post-drought collections indicated that fishes recolonize at different rates. Ten weeks after reinitiation, top minnows, longear sunfish and johnny darters were absent; one third the number and one-half the weight of smallmouth bass were present, and silver jaw minnows and hornyhead chubs occurred in significantly fewer numbers. Seven times as many darters (15 times as many rainbow darters); larger quantities of rock bass, green sunfish, common shiners,bluntnose minnows and stonerollers were collected.

Katz and Gaufin (1953) found the rate of recolonization to be dependant on season, the slowest rates occurring in winter. Larimore et al. (1959) reported that summer also is characterized by slow recolonization rates. Tarplee et al. (1971) found that fifteen years were necessary for full recovery of populations of fishes following channelization. Bayliss and Smith (1967) found "no significant return towards natural stream populations" in 40 years and Congden (1971) reported only 10% recovery in 30 years. In contrast, Lund (1976) reported no long-term effects following sedimentation and turbidity resulting from channelization.

Two recolonization methods are possible for fishes. First, movement from proximal habitats (Hamilton 1961; Hynes 1966; Barton 1977; Tramer 1977) and second, survival in local habitats able to support them (Larrimore et al. 1959; Slack 1955; and Paloumpis 1956 cited in Larimore et al. 1959).

Tramer (1977) noted that headwater areas are repopulated more efficiently by recolonization from downstream than by survival of local drought resistent fish. Colonization of a newly created stream channel by trout was by downstream drift (Kennedy 1955). Larimore et al. (1959) confirmed reports by Slack (1955) and Paloumpis (1956) that pools may serve as "faunal reservoirs" and "faunal havens" for fishes remaining during unfavorable conditions.

Ecological Considerations

Adverse ecological effects resulting from physical changes to stream systems, may be localized, widespread, temporary, or continuously occurring (Larimore et al. 1959).

The response of an aquatic system to stress is dictated by the interactions between ecosystem components and the intensity and duration of stress (Herrick 1977). These complex relationships must also be considered when evaluating the recovery potential of streams. Cairns and Dickson (1977) suggested four biological factors to consider when evaluating this potential:

1) the vulnerability of the system to irreversible damage; 2) its elasticity or ability to recover; 3) its ability to resist structural and functional displacement; and 4) its resiliency, or the number of times a system may overcome displacement.

In most cases, recovery from localized degration, would occur quickly and fairly completely following mitigation of physical stress. Tarplee et al. (1971), suggested that nature can mitigate some adverse ecological effects, but cannot overcome the diminution of habitat and biological production resulting from severe shortening of stream length from channelization. When flows were reinitiated in a dry stream channel, Kennedy (1955) observed rapid colonization by an aquatic community over a period of slightly less than three months. Larimore et al. (1959) considered the "versatility" of stream organisms and their adaptations and movements associated with environment and life cycles to account for rapid stream recovery.

Five factors that control the rate of recolonization are: 1) the extent of the area affected; 2) availability of sources of new organisms; 3) damages to habitat; 4) water levels; and 5) season of the year.

The nature of the stream biota and its response to physical pertrubation is best summarized by Forbes (1883):

> In an aquatic habitat, where wide and violent fluctuations and continual readjustments are the rule, the system of life must be relatively flexible and a species existing in such a system must have within itself recuperative powers to rally against the most destructive recurring attacks.

SUMMARY

Physical changes in streams may occur as a result of copper-nickel development. The important potential physical impacts upon stream ecosystems which may occur in the Regional Copper-Nickel Study area are listed in Table 5. These physical changes are generally caused by channelization, altered stream flows and increased suspended solids and result in an overall reduction in available habitat and habitat diversity. The biological effects are corresponding reductions in distribution, abundance and diversity of aquatic organisms (see Table 5 for a more detailed list of the biological affects).

The response of aquatic organisms to physical stresses is graphically presented in Figure 3. These graphs represent estimates of the response of various biological parameters based on physical impact literature and information on the environmental requirements of aquatic organisms. Because of the interaction between the various physical impacts, it is not possible to assign levels at which significant change occurs.

Based on these response curves, impacts have been rated on their potential for causing measurable biological change (Figure 4). No direct or indirect food chain affects were considered when determining where measurable impacts may occur.

The recovery potential for each group is also indicated in Figure 4. Recovery of aquatic ecosystems from damage is generally rapid and complete if the habitat can be restored to a natural condition. Restoration may occur through natural forces after mitigation of impacts or by artificial habitat alterations which increase the diversity of available stream habitat.

Examination of Figure 4 indicates the following impacts have the greatest potential for causing significant ecological change (i.e. measurable biological change with slow or limited recovery): high suspended solids; medium and high sedimentation rates; increased current velocity; decreased current velocity; frequent short term flooding; rapid and frequent discharge fluctuations; greatly reduced habitat diversity; loss of habitat; and reduced allochtonous inputs.

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TABLE 1: Relationship between water velocity in stream riffles and numbers of bottom organisms in five studies.*

Current velocity **		Number of Organisms				
cm/sec	(ft/sec)	Pearson, et.al (1970) 1/	Surber (1951) 1/	Kennedy (1967) 1/	Arthur (1963) 2/	Needham & Usinger (1956) 3/
0 - 16	(0 - .5)	—	—	—	138	—
17 - 32	(.6 - 1.0)	53	99	444	137	—
33 - 47	(1.1 - 1.5)	90	148	881	532	90
48 - 62	(1.6 - 2.0)	120	115	484	257	—
63 - 78	(2.1 - 2.5)	89	152	289	359	71
89 - 93	(2.6 - 3.0)	105	125	171	352	99
94 - 108	(3.1 - 3.5)	65	339***	—	392	80
109 - 122	(3.6 - 4.0)	62	—	—	365	70

*From Bovee (1975)

**Surber recorded surface velocity, Arthur measured velocity 3 cm from bottom; depth of other velocity measurements not specified.

***Inadequate sample size

1/ Compiled by Giger (1973)
2/ Compiled by Hooper (1973)
3/ Compiled by author from Needham and Usinger (1956)

TABLE 2: Distribution of stream fishes according to stream velocity**

Bottom material	rubble	gravel	sand	silt	mud
Size range (mm)	30	5-30	.5-5	.05-.5	.05
Fall velocity (cm/sec)	150	40-150	5-40	.5-5	.5
(ft/sec)	5	1.3-5	.6-1.3	.06-.6	0-.06
Species					
Stonecat	X*	X			
Flathead chub	X	X	X	X	X
Burbot+	X	X*			
Longnose dace+	X	X			
Shovelnose sturgeon		X*	X		
Sturgeon chub		X			
Shorthead redhorse+		X	X		
Blue sucker		X	X		
Smallmouth bass+		X	X*		
Rock bass+		X	X*		
Rainbow trout		X	X*		
Channel catfish		X	X*	X	X
Longnose sucker		X	X	X	X
White sucker+		X	X	X	X
Brook trout+			X		
Brown trout			X		
Creek chub+			X		
Pearl dace+			X		
Emerald shiner +			X		
Sand shiner+			X		
Plains minnow			X	X	
Brassy minnow+			X	X	
Silvery minnow			X	X	
Northern pike+			X	X	X*
Walleye+			X	X	X*
Black bullhead+				X	X*
Yellow bullhead+				X	X*
Golden shiner+				X	X*
Smallmouth buffalo				X	X*
Yellow perch+				X	X*
Sauger				X	X*
Carp				X	X*
River carpsucker					X
Largemouth bass+					X
Bluegill+					X
White crappie					X
Black crappie+					X

*Indicates preferred range if species found in more than one habitat.

+Indicates species present in Copper-Nickel Study Area.

**From Bovee (1975).

Table 3. Reported aquatic insect depth velocity criteria.*

Species or Group	Preferred Velocity (fps) Mean or Median	Range	Depth (ft)	Reference
Aquatic				
Invertebrates	2.0	0.5-3.5		Surber, 1951
		2.0-3.5		Needham and Usinger, 1956
	2.0	0.5-4.0		Pearson et al., 1970
	1.2	0.5-3.0	0.25-0.5	Kennedy, 1967
		1.5-3.5	0.5 -3.0	Hooper, 1973
Ephemeroptera				
		1.2-2.6	<1.0	Needham and Usinger, 1956
			0.5-1.0	Hooper, 1973
Rhithrogena	4.0		deep	Needham and Usinger, 1956
	2.3	1.23-3.37		Arthur, 1963
Baetis	2.02	1.02-3.02		Arthur, 1963
	3.0			Needham and Usinger, 1956
Ephemerella	1.93	1.03-2.83		Arthur, 1963
Plecoptera				
Arcynopteryx	1.58	0.89-2.27		Arthur, 1963
Tricoptera				
	3.0	1.0-2.0		Hooper, 1973
		1.0		Needham and Usinger, 1956
Hydropsyche	2.35	1.03-3.67		Arthur, 1963
Diptera				
	3.0			Needham and Usinger, 1956
		0.5-1.0		Hooper, 1973
Simulium	2.80	1.91-3.69		Arthur, 1963

*From Stalnaker and Arnette (1976b).

Table 4. The percentage composition of adults of various groups of insects emerging into traps set over different types of substratum in streams in Algonquin Park, Ontario, and comparision of the total numbers emerging from the different types of substratum.*

	Rubble	Gravel	Sand	Muck
Ephemeroptera	35.5	4.6	9.3	20.3
Trichoptera	7.0	1.7	1.7	3.8
Plecoptera	4.1	2.1	0.7	0
Chironomidae	38.2	67.6	83.9	74.8
Simuliidae	10.8	21.4	0.9	0
Miscellaneous	4.4	2.5	3.5	1.0
Ratio of total numbers emerging to total numbers emerging from sand	4.6 in rapids 3.3 in pools	2.1	1.0	1.8

* From Sprules 1947 as reported in Hynes 1970a.

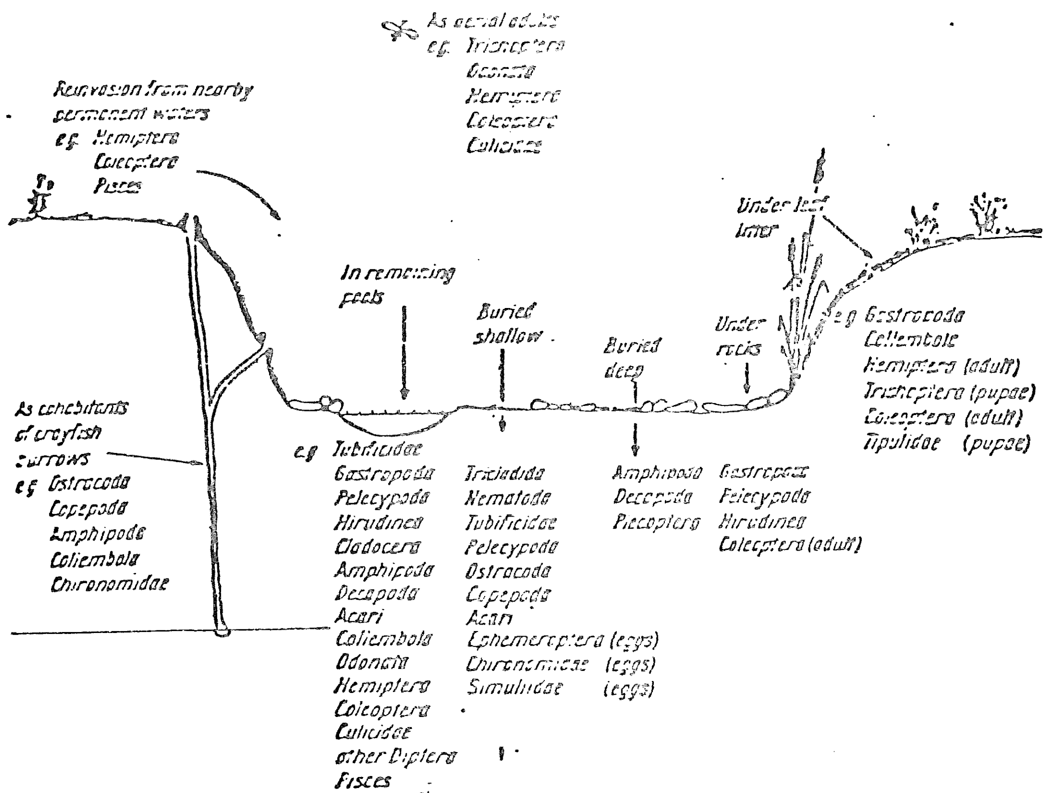
Table 5. Activities which may occur during copper-nickel development, and the physical impacts of these activities on streams and the potential biological changes caused by these physical impacts. (cont.)

ACTIVITIES → CAUSE →	DIRECT PHYSICAL IMPACTS → CAUSE →	PHYSICAL CHANGES WHICH CAUSE BIOLOGICAL EFFECTS → CAUSE →	BIOLOGICAL CHANGES
	11) Short-term fluctuating flows	11) Decreased current	11) Reduced spawning success
	12) Increased stream discharge	12) Short-term flooding	12) Increased primary production
	13) Downstream effects	13) Short-term low discharge	13) Short-term low discharge
	14) Stream shortening		14) Increased fish standing crop
	15) Stream straightening		15) Change in fish behavior
	16) Removal of stream obstructions		
	17) Loss of pool/riffle interspersion		

Table 5. Activities which may occur during copper-nickel development, and the physical impacts of these activities on streams and the potential biological changes caused by these physical impacts.

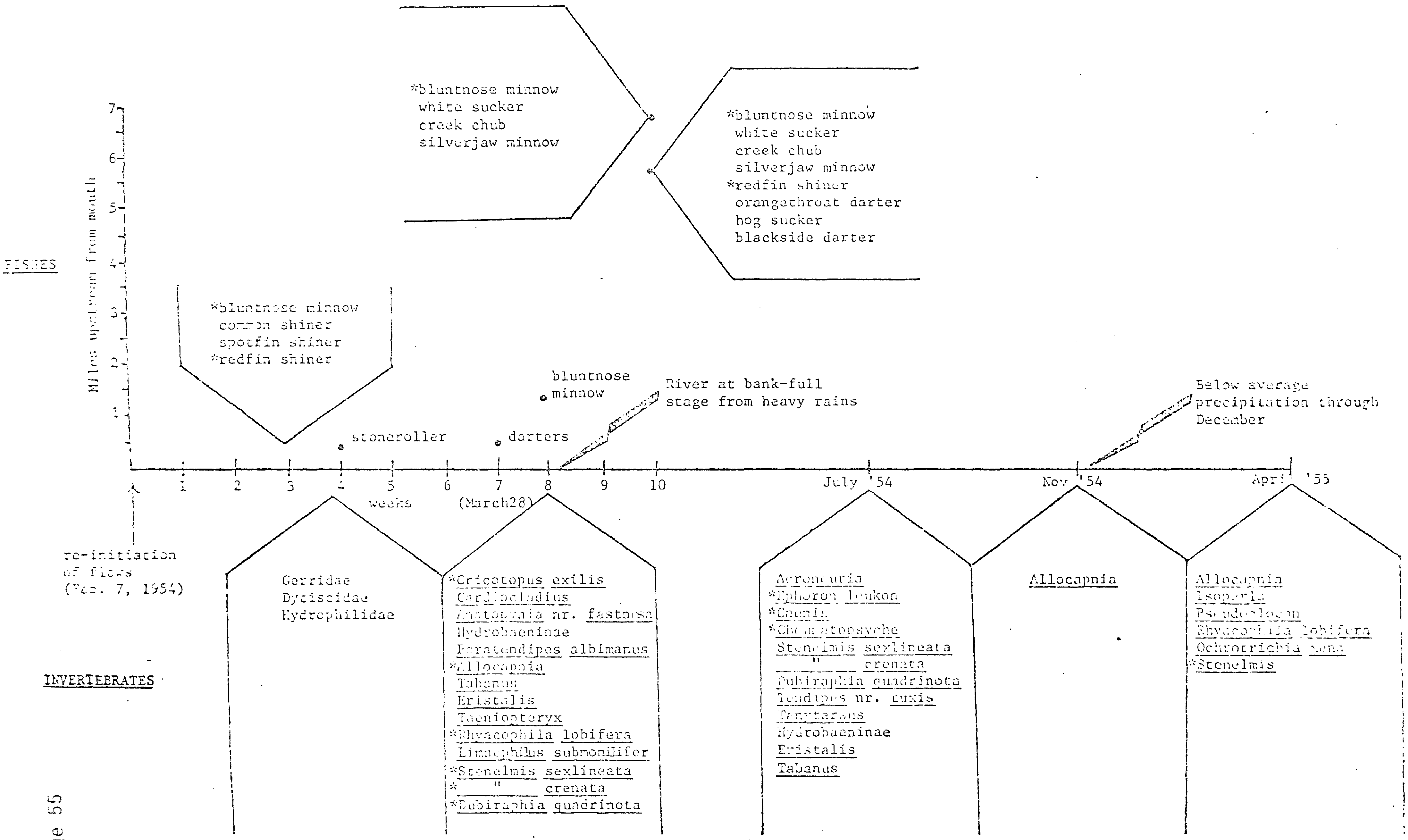
ACTIVITIES → CAUSE →	DIRECT PHYSICAL IMPACTS → CAUSE →	PHYSICAL CHANGES WHICH CAUSE BIOLOGICAL EFFECTS → CAUSE →	BIOLOGICAL CHANGES
1) Land Appropriation	1) Change in watershed runoff characteristics	1) Increased Sedimentation	1) Change in Peripayton species composition
2) Construction (other than roads)	2) Loss of watershed area	2) Increased turbidity	2) Change in invertebrate species composition
3) Road Construction	3) Loss of terrestrial vegetation	3) Reduced habitat diversity	3) Change in fish species composition
4) Water Appropriation	4) Change in terrestrial soils	4) Decreased habitat	4) Reduced primary production
5) Discharge	5) Terrestrial erosion	5) Increased habitat	5) Reduced invertebrate standing crop
6) Channelization	6) Stream erosion	6) Increased solar radiation	6) Reduced fish standing crop
	7) Increased suspended solids	7) Reduced allochtonous inputs	7) Direct periphyton mortality
	8) Loss of riparian vegetation	8) Increased daily temperature fluctuation	8) Direct invertebrate mortality
	9) Decreased stream discharge	9) Increased annual temperature fluctuation	9) Direct fish mortality
	10) Greater variation in annual hydrograph	10) Increased current velocity	10) Blockage of fish movement

Figure 1. Summary of the habits of fauna from a temporary stream which allow survival in low or no-flow conditions.*



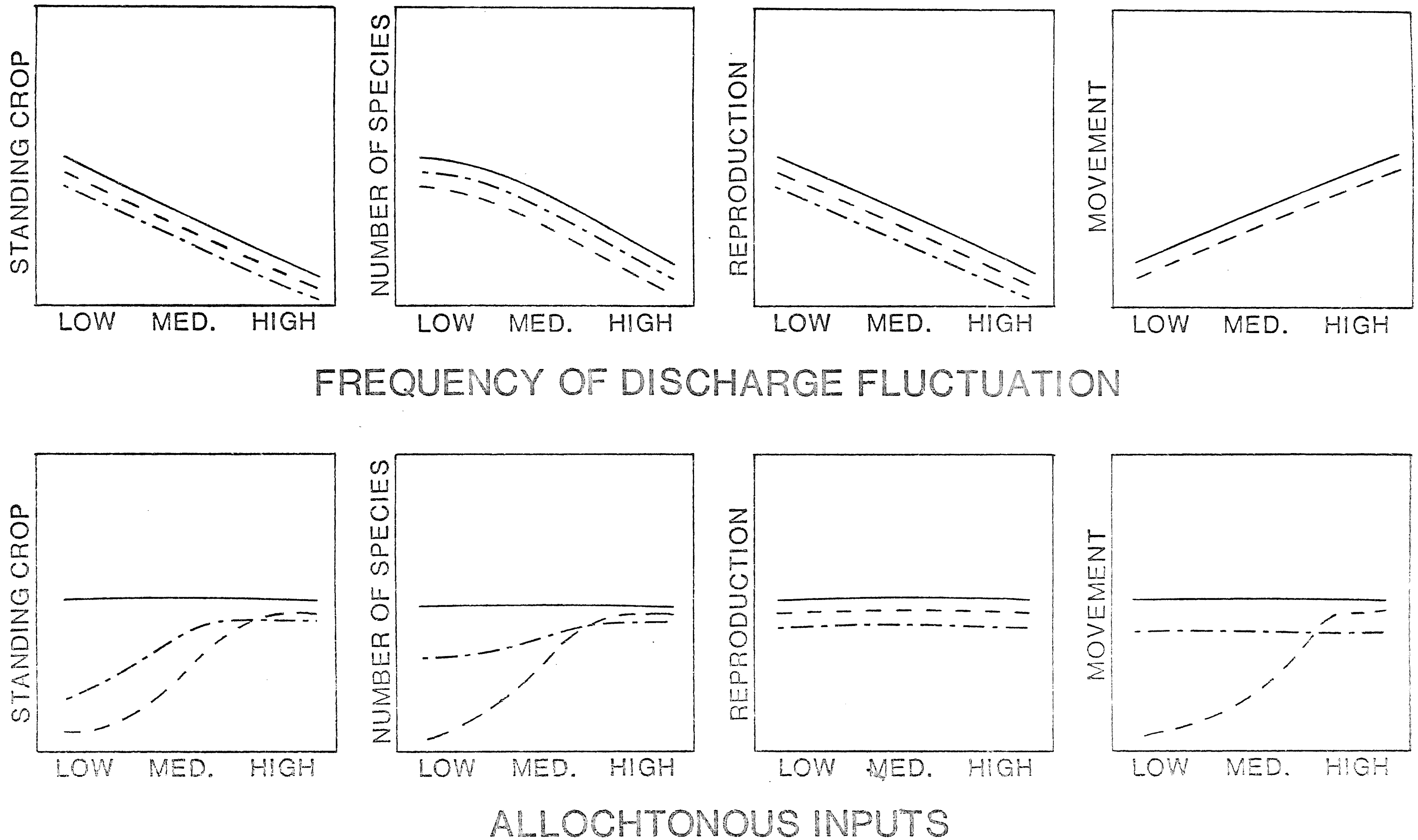
* From Williams and Hynes (1977).

Figure 2. Recolonization of a drought devasted stream community by fish and invertebrates **



* indicates most abundant taxa.

** from Larimore et al. 1959.



——— FISH
– – – – BENTHIC INVERTEBRATES
–·–·– PERIPHYTON

FIGURE 3. RESPONSE OF AQUATIC ORGANISMS TO PHYSICAL IMPACTS.

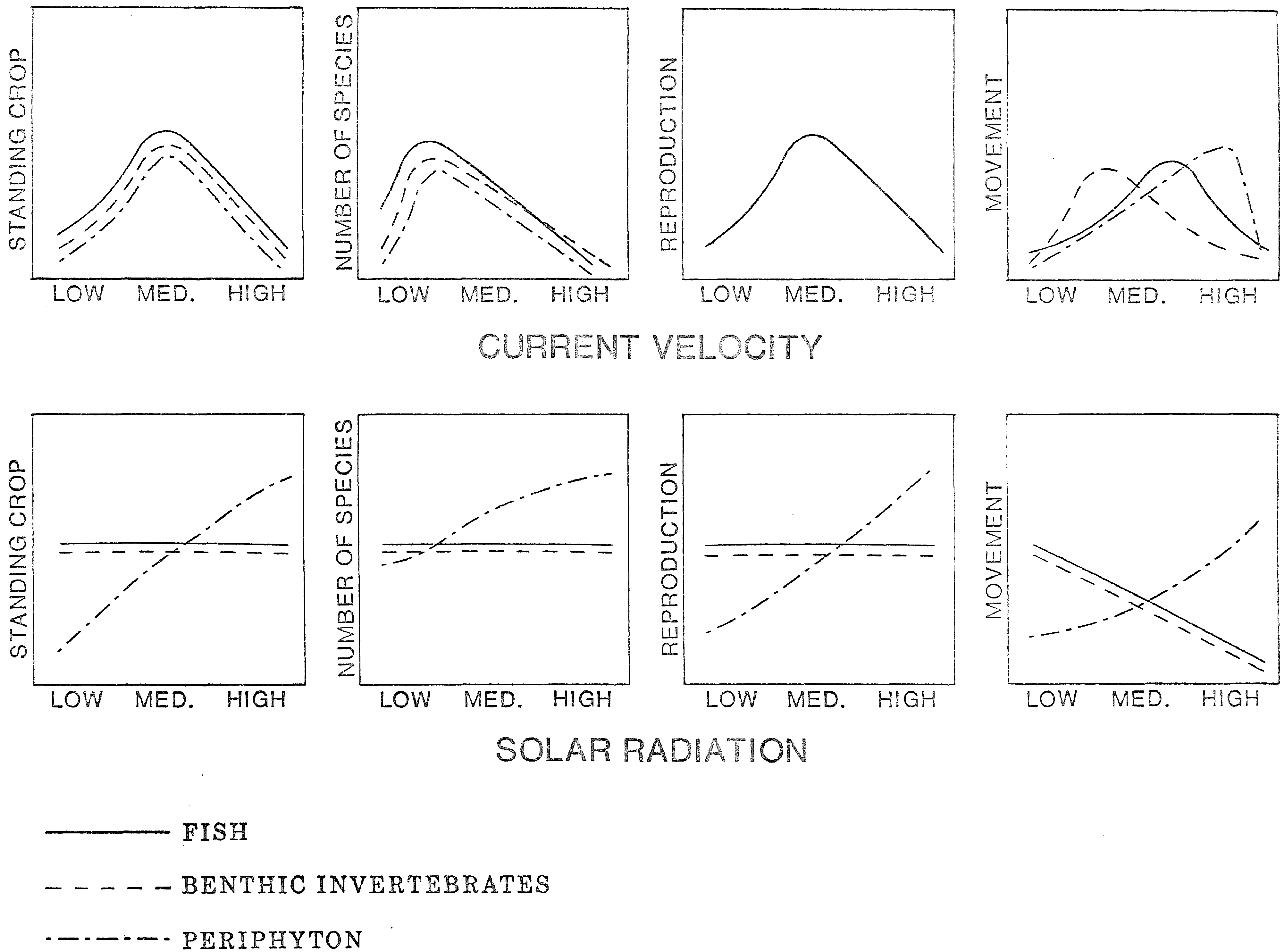


Figure 3 continued

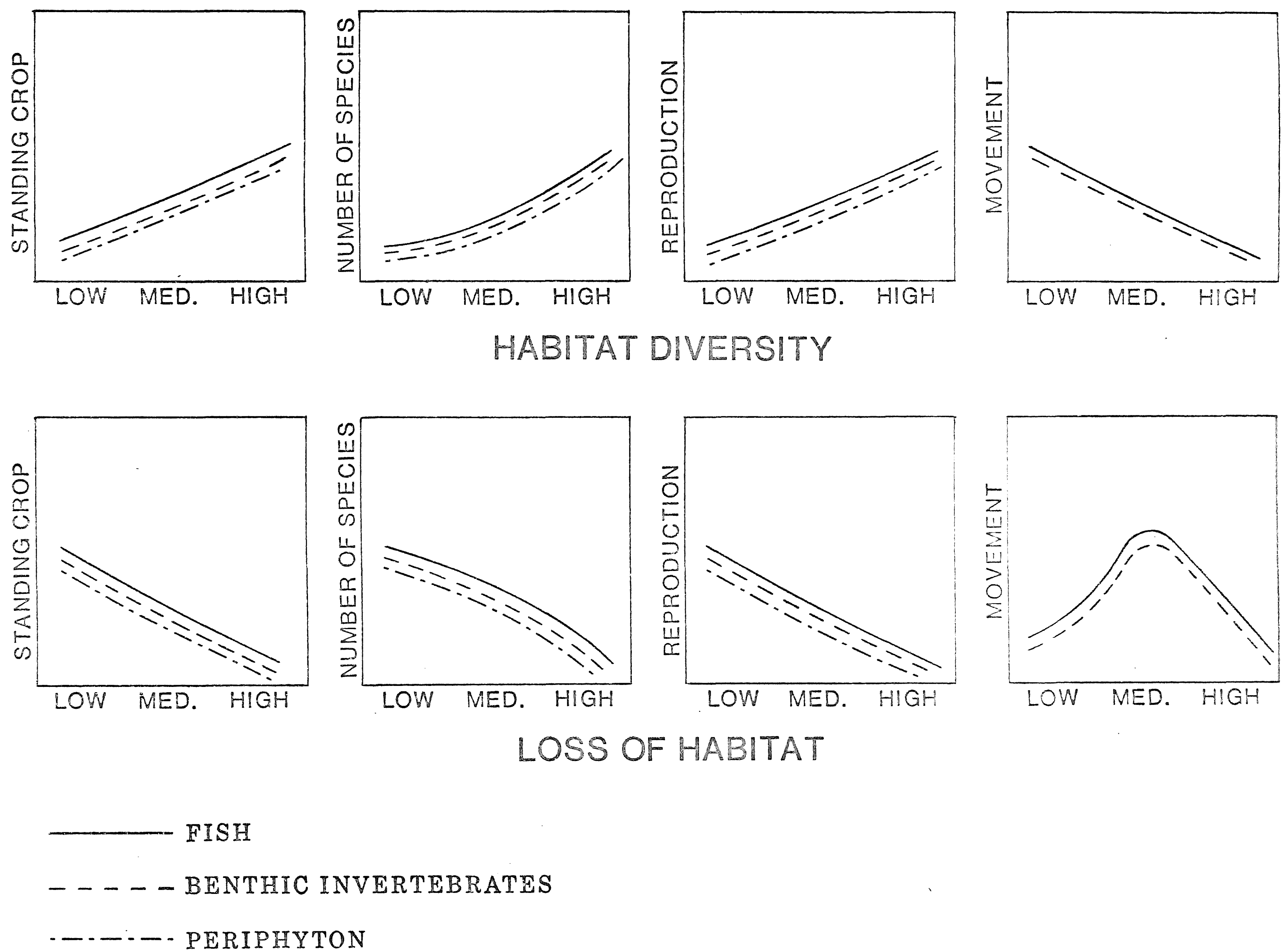


Figure 3 continued

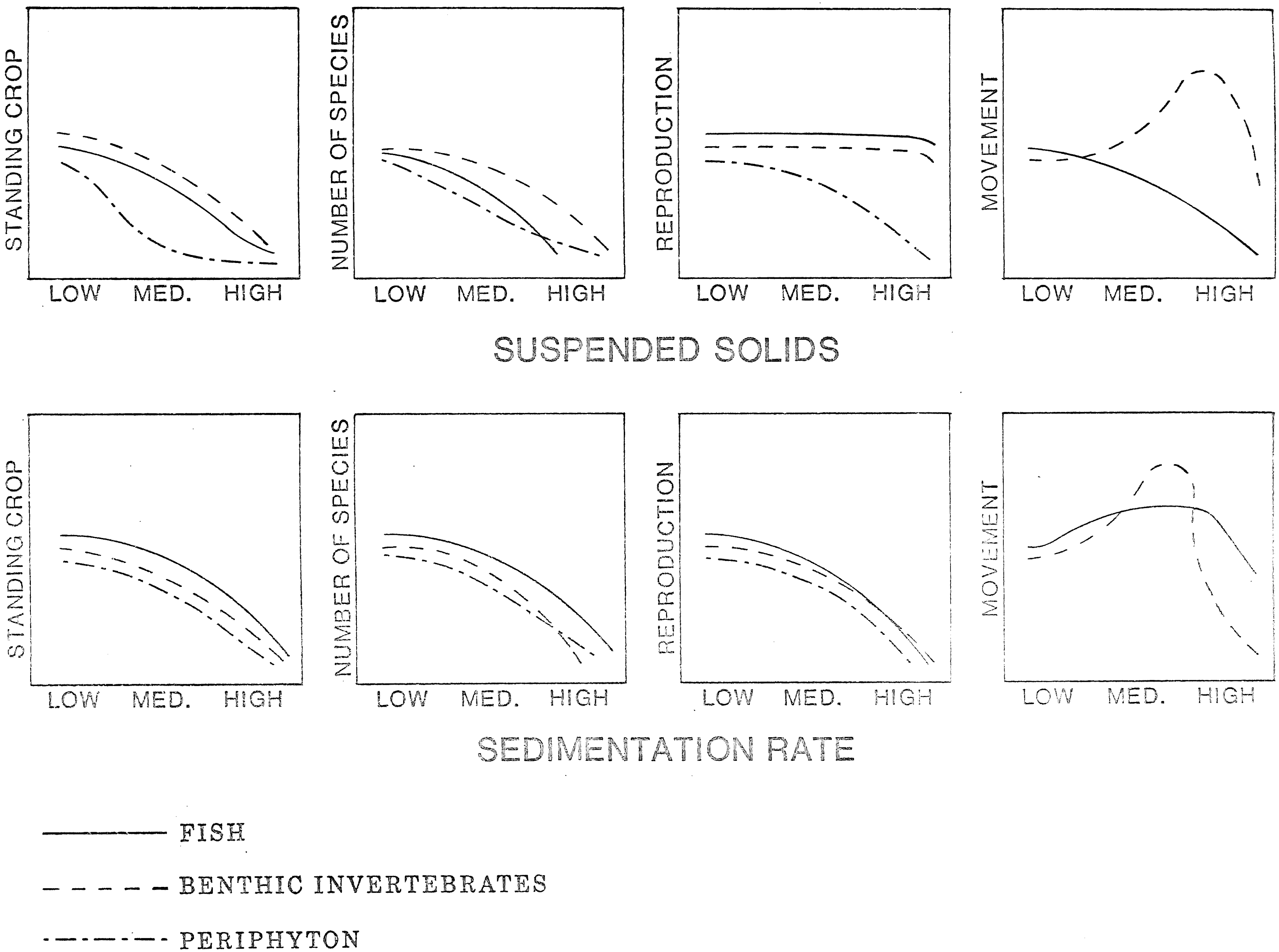
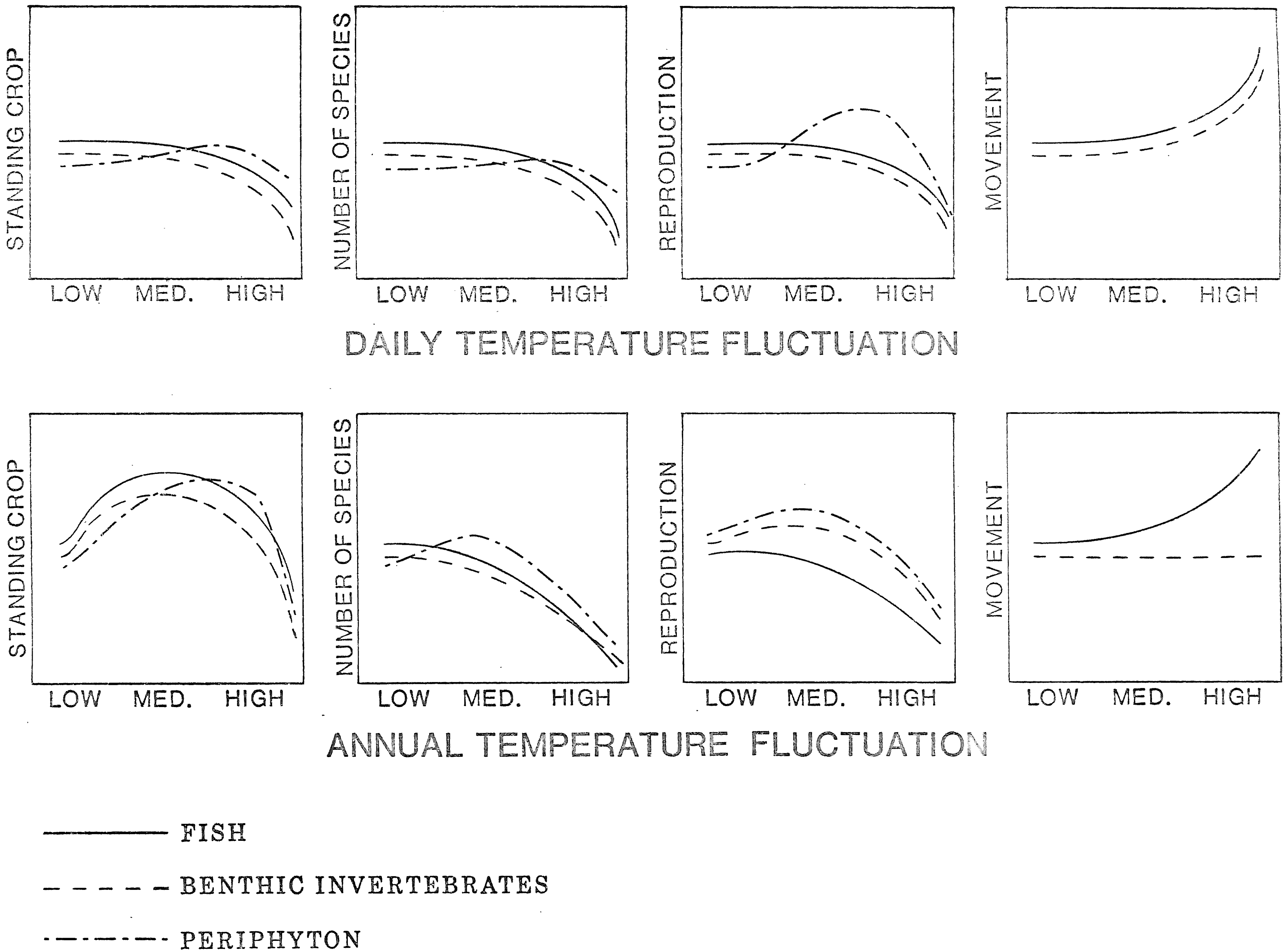


Figure 3 continued



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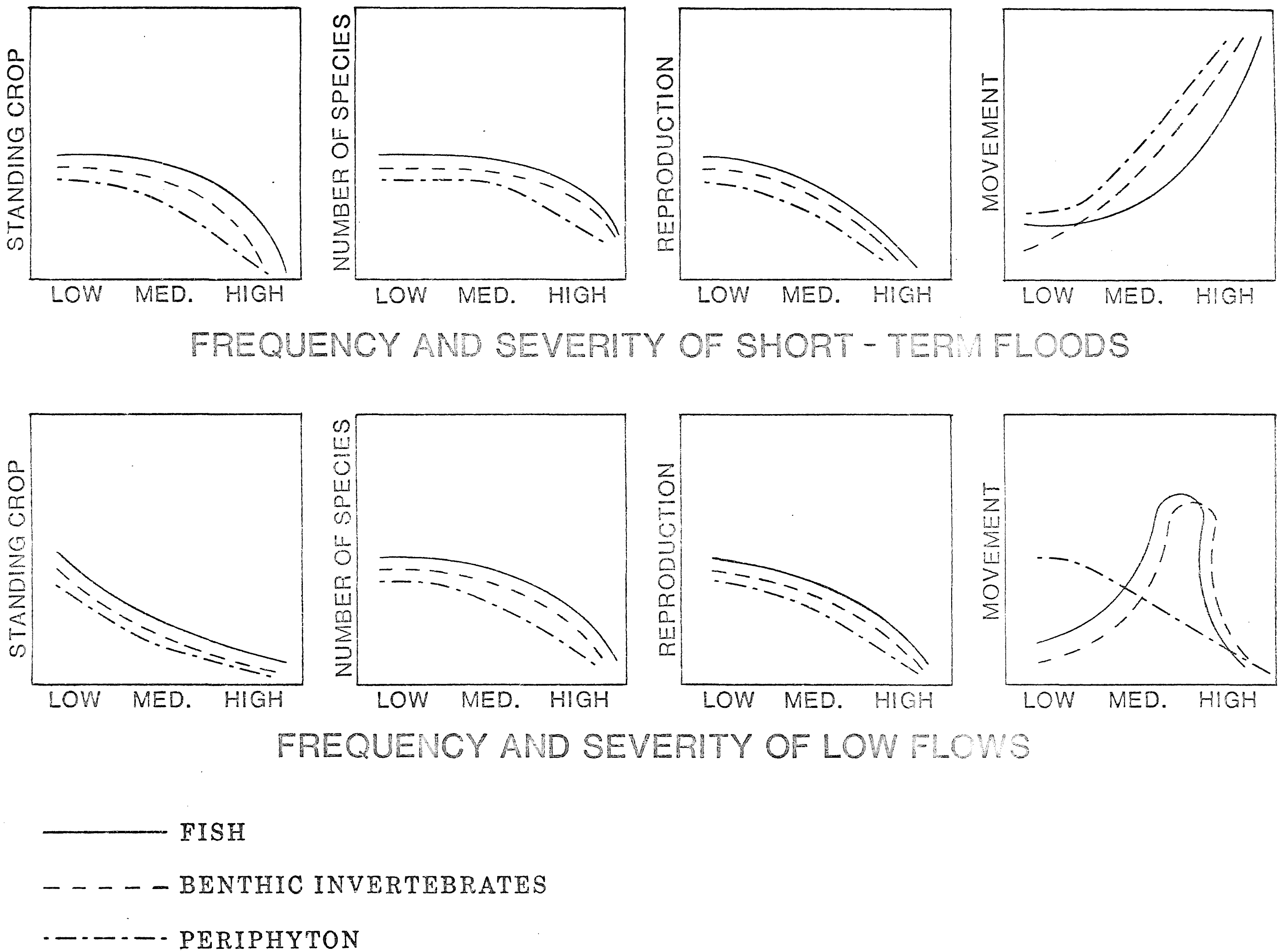

Figure 3 continued