EFFECTS OF EXTERNAL FACTORS ON THE USE 0F BENTHIC MACROINVERTEBRATES AS BIOLOGICAL INDICATORS OF WATER QUALITY

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ABSTRACT:

The use of biological indices in determining water quality is an important part of the assessment of habitat quality. But can external factors such as amphibian interactions or seasonal variations affect the bioassessment? My study tested for the effects of poor water quality on macroinvertebrate fauna of small waterbodies. It also tested for possible effects of interactions among water quality, temporal changes, and the presence or absence of amphibian larvae on the macroinvertebrate fauna. Results suggest a negative effect on the macroinvertebrate fauna of mining effluent water when compared with higher quality water. This indicates that the use of biological monitoring is accurate and useful. There were no significant effects on the macroinvertebrates due to the presence or absence of amphibian larvae, however, there was an effect due to temporal changes. This effect may cause problems in making accurate bioassessments and should be accounted for.

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INTRODUCTION:

The issue of environmental quality is a major one in today's world. Studies and information related to air quality, deforestation, global warming, and water quality are increasingly prevalent. Assessing the quality of water in our streams is essential because of the importance of water in our everyday lives. Conventional chemical analysis, while being important in certain instances, is only one aspect of assessing water quality. Because chemical analysis gives only an indication of quality at one point in time, another analysis should be used in conjunction with chemical analysis.

Biological monitoring addresses the problems with chemical monitoring. Because chemical analysis usually measures water quality at a specific point in time, it may miss episodes such as spring runoff, which can stir up the sediment and potentially introduce heavy metals or other pollutants into the water. Benthic macroinvertebrates are continuously exposed to varying water quality conditions, and therefore they integrate effects of contaminants over time and provide a direct measure of water quality (Carlisle and Clements, 1995). Since the biological components of the water are affected by pollution in a long-term manner, an indication of a problem can be more apparent than results from chemical analysis. "The status of a water body's living systems is the most direct and effective measure of its integrity" (Karr and Chu, 1997).

Biological assessments rely on the use of indices to rate the quality of water based on the macroinvertebrate fauna. There are many different types of indices and most are specific to a pollutant or sample site (Norris and Georges, 1993). The various indices are based on different parameters. For example, some indices are based on indicator species, others on diversity, and some are based only on a biotic component such as abundance

(Cairns and Pratt, 1993). Many of these indices are complex in utilization. Because I am not assessing the actual quality of water, but testing for interactions which may affect a water quality assessment, relatively simple, yet accurate, indices will be used. These include taxa richness (family level), and relative abundance. Across diverse taxa and regions, similar biological attributes (e.g. taxa richness and relative abundance of tolerant organisms) work consistently and reliably as indicators of resource condition (Karr and Chu, 1997).

Because the sample sites are biological systems, there are variable factors that may interact with unpredictable consequences. For example, there may be a predator-prey relationship between amphibian larvae and macroinvertebrate predators. This may affect the types and quantity of macroinvertebrate fauna. When assessing the quality of habitat, these many different factors must be taken into consideration. For this reason, the use of a multimetric approach to the assessment of water quality is now gaining favor over the previous univariate approach. To assess biological conditions accurately, a method that integrates biotic responses through an examination of patterns and processes from individual to the ecosystem level is required (Barbour, et al., 1994; Karr et al., 1986). Figure 1 shows diagrammatically the multimetric approach to bioassessment. Several components are analyzed to give a final biological assessment. Taxa richness and relative abundance are a part of the community level of assessment.

Most often, a sample of the benthic macroinvertebrate fauna of the water body being assessed is used as the biological indicator of water quality. Benthic macroinvertebrates are defined as organisms that inhabit the bottom substrate of freshwater habitats (Rosenberg and Resh, 1993). A macroinvertebrate must meet size specifications in that they must be retained by mesh sizes between 200 and 500 micrometers (Reviewed in Rosenberg and Resh, 1993).

The advantages of this technique over chemical analysis are numerous. First, the macroinvertebrates are found virtually in all water environments (Lenat et al., 1980). Perturbations in many different habitats can affect the macroinvertebrate fauna (Rosenberg and Resh, 1993). Secondly, because of the large number of species of organisms, many responses to environmental stresses can be assessed (Hellawell, 1986; Abel, 1989). Third, spatial analyses of disturbance effects can be done due to the sedentary nature of the benthic macroinvertebrates (Reviewed in Rosenberg and Resh, 1993). Long term analysis of water quality is possible with the use of benthic macroinvertebrates because they act as continuous monitors of the water they inhabit (Rosenberg and Resh, 1993; Hawkes, 1979).

In light of all the advantages, there are disadvantages. (1) Different organisms respond to different stresses in different ways; (2) the processing of the samples can be costly and are, in most cases, time consuming; (3) the distribution and abundance of the fauna can be affected by factors other than water quality (Rosenberg and Resh, 1993). For example, other existing fauna in the sample area may affect the assessment. In addition, temporal variations may also introduce errors in the water quality assessment because of a change in abundance or distribution of macroinvertebrates over time.

One component that may be an integral part of this scheme is that of the interactions such as predation or competition among the organisms in the biological system. Because benthic macroinvertebrates and amphibians both inhabit similar habitats, interactions involving community structure between them are possible. For example, there are predator-prey interactions. Because of predation, amphibian larvae are known to alter activity and shift habitat use (Skelly, 1991). Abundance and taxa richness, as well as other indicators of water quality, may be altered due to the presence or absence of amphibians; especially amphibian larvae, which are the food source for many predacious macroinvertebrates. These macroinvertebrates may or may not be present depending on the presence of a amphibian larvae, not on the presence of pollutants.

Temporal changes may also have an effect on the biological indicators because of seasonal variations in reproduction, metamorphosis, or hatching of the macroinvertebrates. To test if temporal change or amphibian interactions have an effect on these indicators of water quality (however, not on the water quality assessment), I compared two sets of samples collected at different times. The first set was sampled in early summer, and the other sample in middle summer. Of these sample sets, one set consisted of sites with presumably good water quality, and the other set consisted of sites presumably with low water quality due to mining effluent effects.

In the living world a plethora of events and interactions occur that cannot always be accounted for in quantitative analyses. In my study I focused on two variable factors, amphibians and sampling time, each of which may introduce error in water quality assessments. An analysis of these interactions in comparison with water quality will indicate the validity of water quality assessments, and at the same time show that care must be taken in water sampling.

METHODS:

My study sites were located in the Helena National Forest near Helena, Montana. The sites were concentrated around the Park Lake, Chessman Reservoir, Forest Lake, and Frohner Meadows areas. There were also two additional sites sampled from Dog Creek, Northwest of Helena. These study sites consisted of various sizes of pond-like water bodies, termed potholes. From all of the sites surveyed by Hokit et al., the sample set of twenty-four potholes for this study was randomly selected and grouped together based on the following criteria: (1) potholes with mining effluent absent, amphibian larvae absent; (2) potholes with mining effluent present, amphibian larvae absent; (3) potholes with mining effluent absent, amphibian larvae present; (4) potholes with mining effluent present, amphibian larvae present.

For a pothole to be considered as having mining effluent present, there must have been visual as well as chemical evidence for either water entering the site that has passed through mine tailings, or the site must be in direct contact with mine tailings. The chemical evidence consisted of decreased pH and increased conductivity, both of which were sampled for each pothole. Many different approaches can be used to survey the amphibian populations (Olson et al. 1997; Heyer et al. 1994). The amphibian larvae presence/absence qualification in this study was determined by a visual survey of the site for the presence or absence of either amphibian egg masses or amphibian larvae. Sites were sampled two times during the research session, including a sample during early summer and a sample in middle summer. These two sample times were separated by approximately one month. Due to drying of two sites, only twenty-two sites were sampled during the later sampling period.

Stratified random sampling was used as the standard sampling technique for each pothole. The potholes, at the time of sampling, were divided into four quadrants: Northeast, Southeast, Southwest, and Northwest. A sample was then taken from a randomly determined position in each of the four quadrants. By using a random number generator, an approximate distance from the four cardinal directions was determined. This distance extended around the perimeter of the pothole in a clockwise direction. To determine the distance away from shore that the sample was to be taken, again a random number generator was used. This gave an estimated distance to the center of the pothole from the shore that the sample was taken. The sample did not exceed a water depth of one meter.

The samples were collected using a standard D-ring net with a mesh size less than fifty micrometers. The net was swept along the bottom of the pothole for approximately one meter. The contents were then emptied into a bucket. Depending on weather conditions and/or time constraints, either the macroinvertebrates were removed from the sample in the field, or the entire sample, macroinvertebrates and debris, was brought to the lab for separation. In either case, the macroinvertebrates were preserved in 100 percent ethyl alcohol after separation until the identification process could be started.

The identification process involved viewing the macroinvertebrates in the laboratory using a high-power dissecting microscope. The macroinvertebrates were identified to the family level by using several different identification keys. The total number of organisms in each family in the sample were counted and recorded. Statistical analyses were then performed on the data collected.

I used ANOVA to test for the effects of amphibian presence, mining effluent presence, and temporal differences on taxa richness and relative abundance of organisms. Because the ANOVA test assumes the populations have a normal distribution (Triola, 1995), the data was first screened to ensure it met parametric assumptions.

RESULTS:

Taxa Richness Test

ANOVA revealed that there was no significant effect on any of the parameters, except for the presence of mining on taxa richness. The taxa richness in the potholes with mining effluent present significantly decreased in comparison to the potholes with no mining (Table 1, Fig. 2).

Abundance Test

To meet parametric assumptions of normality, the data was log-transformed (Zar, 1984) for the ANOVA test. ANOVA revealed no significant effect of amphibian presence on abundance, nor do the interactions affect the abundance of macroinvertebrates. However, mining had an effect on the abundance (Table 2, Fig. 3). The relative abundance in the potholes with mining was significantly decreased. The sampling time also significantly changed the abundance of macroinvertebrates present in the sample (Table 2, Fig 4).With respect to the temporal change, the later sample set had a significantly higher number of organisms.

DISCUSSION:

Based on the bioassessment indicators, taxa richness and relative abundance, the data clearly show a significant difference due to mining. The taxa richness as well as the relative abundance decreased significantly in the potholes with mining effluent. This indicates that the mining effluent being released into the potholes has a negative effect on the macroinvertebrate fauna of these biological systems. Even from these two limited and simple analyses, a significant effect is apparent. More extensive analyses could be performed with the available data using other indices. However, with the results obtained and with the goal of assessing interactions of the community, not actual water quality, further analyses were not performed at this time. From these results, it is clear that the use of bioassessment protocols is effective in measuring the health of an aquatic system.

With respect to the effect of amphibians on altering the taxa richness and/or relative abundance tests, there was no significant effect. There is no indication of a difference in taxa richness or relative abundance between sites with amphibians and sites without. In addition, there is no indication of an interaction involving the mining effect and an effect from the presence or absence of amphibian larvae. This suggests that when assessing the water quality of an aquatic habitat, the use of biological indicators is not affected by the presence or absence of amphibian larvae. This does not rule out other interactions among the biological components of these communities that were not addressed in this study.

These interactions could include effects of weather on the diversity of fauna, limitations of food resources, or competition among the macroinvertebrate predators. These predators may prey on coexisting species of prey and consequently the structure and diversity of the ecological community depends on the competitive abilities of the predator (Werner, 1991). Since some predacious macroinvertebrates are more susceptible to pollution than others and since some are more competitively successful, the bioassessment may be altered, not because of the water quality alone, but also by competitive pressures. My results did not show these types of effects, however.

There was a significant temporal effect on the relative abundance of organisms in the potholes, however sampling time had no effect on taxa richness. This indicates that due to seasonal variations, there is a possibility of differing results from the bioassessment due to a change in the relative abundance of organisms. Thus, for an accurate assessment, the samples to be analyzed should be taken at approximately the same time, if relative abundance is the sole parameter used to determine water quality. By using other components of the multimetric approach, in addition to using other statistical analyses, such as MANOVA, however, this problem may be corrected for. Future studies may shed light on these results.

The importance of monitoring and determining the quality of our water, and other environmental factors is of immense importance. The accuracy of the assessments is essential to ensure that a non-existing problem is not falsely created by errors in data interpretation. The use of many different metrics in assessments reduces the chances of error, however there are many possible environmental interactions that may not be accounted for by the current assessment protocols. Therefore, care must be taken when assessing water quality using biological indicators. Accurately assessing ecological risks depends on effective biological monitoring (Karr and Chu, 1997).

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Table 1. Results of ANOVA test on taxa richness.

Effect	dF	MS	F	P
Amphibian	1	8.92	0.525	0.473
Mining	1	333.25	19.603	< 0.001
Temporal	1	1.40	0.082	0.775
Amphibian x Mining	1	4.58	0.270	0.607
Amphibian x Temporal	1	0.16	0.009	0.924
Mining x Temporal	1	4.11	0.242	0.626
Amphibian x Mining x Temporal	1	0.46	0.027	0.870

Table 2. Results of ANOVA test on abundance.

Effect	dF	MS	F	P
Amphibian	1	0.101	0.062	0.804
Mining	1	31.547	19.460	< 0.001
Temporal	1	7.668	4.730	0.036
Amphibian x Mining	1	0.444	0.274	0.602
Amphibian x Temporal	1	0.797	0.492	0.487
Mining x Temporal	1	0.991	0.612	0.439
Amphibian x Mining x Temporal	1	0.009	0.005	0.940

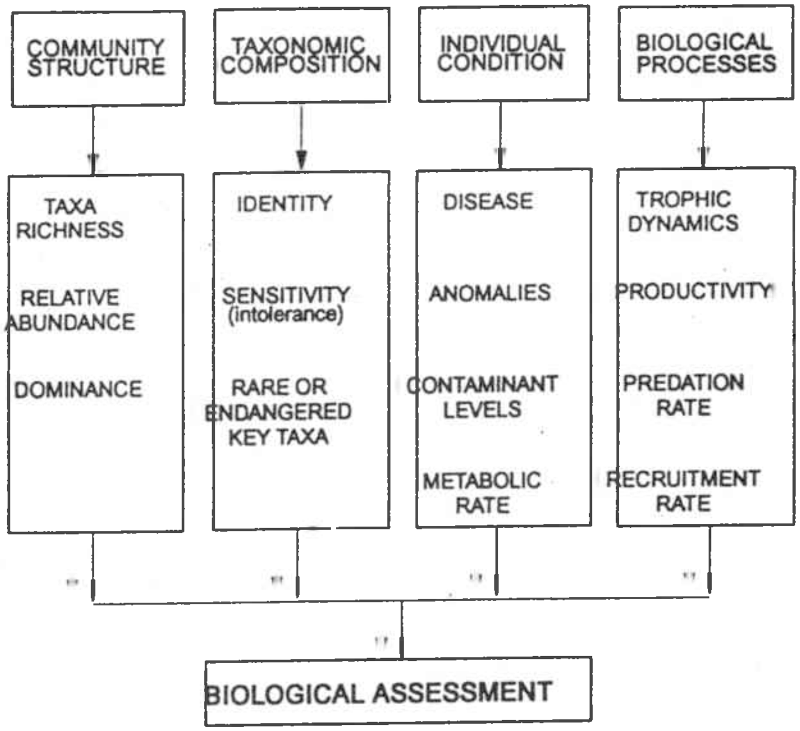
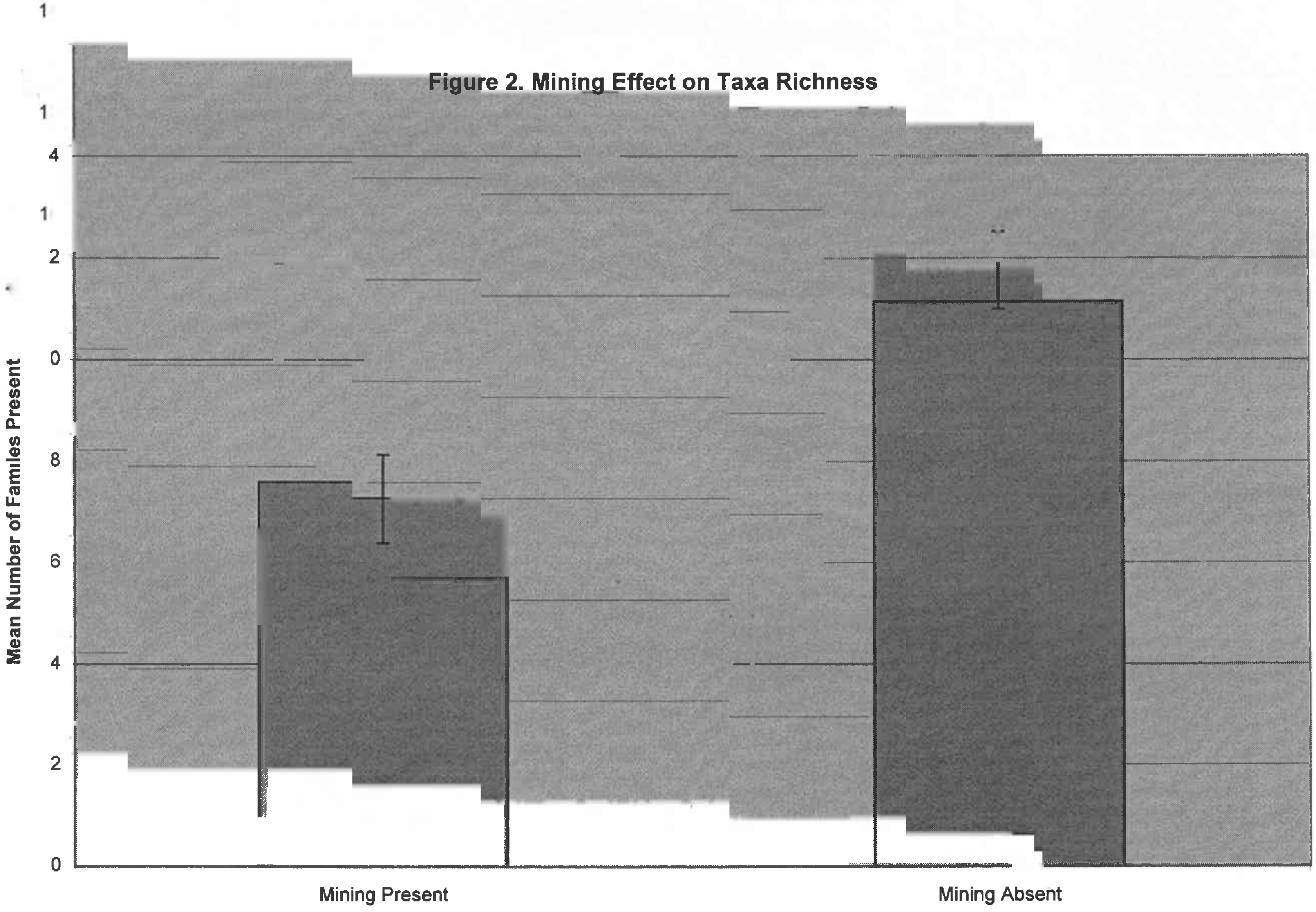


Figure 1. Organizational structure of the types of attributes that should be incorporated into biological assessment (Barbour, M.T., et al. 1995).

Figure 2. Mining Effect on Taxa Richness



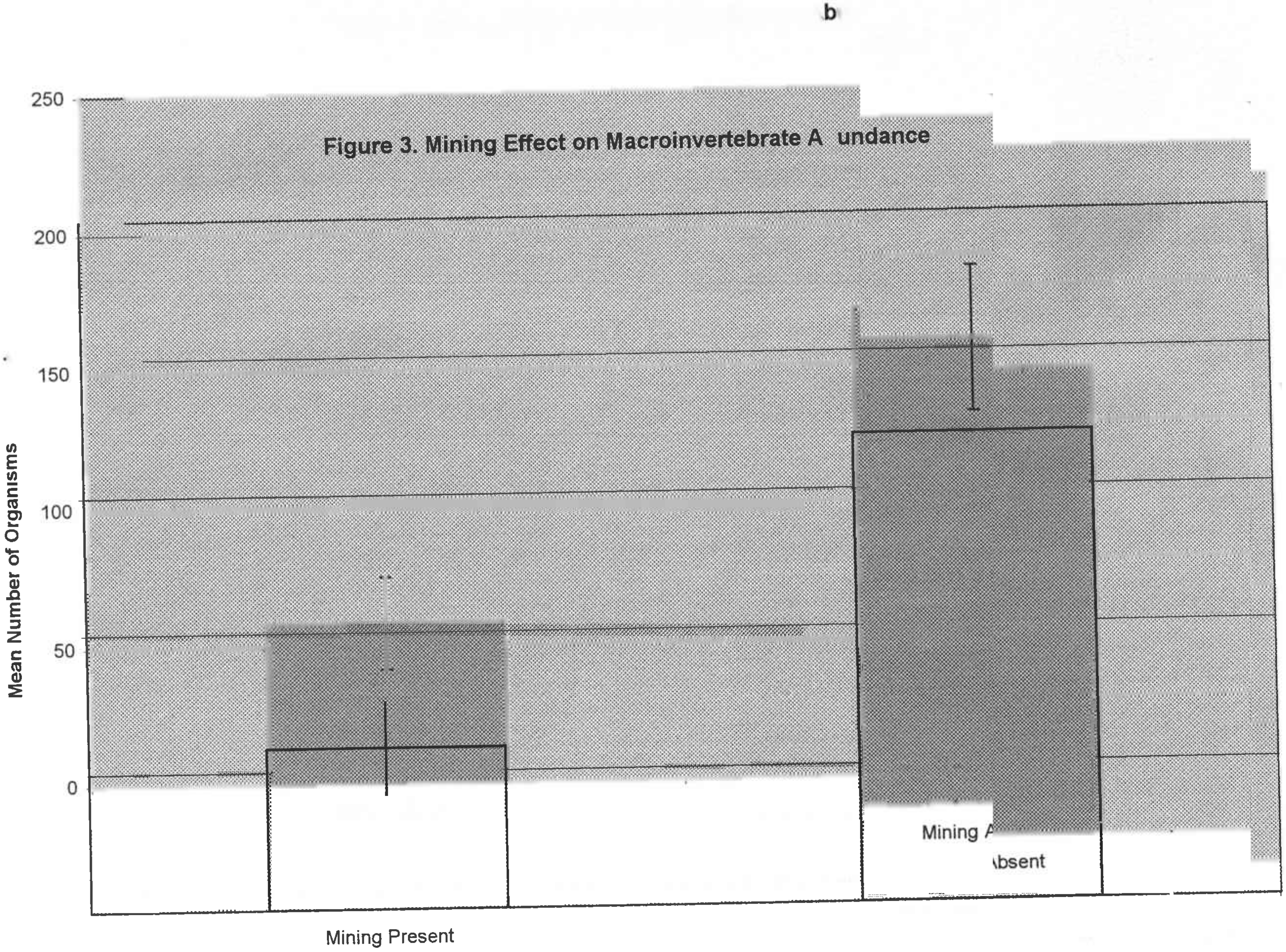


Figure 3. Mining Effect on Macroinvertebrate A undance

Figure 4. Temporal Effects on Macroinvertebrate Abundance