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**GROUND WATER:  
ALASKA'S HIDDEN RESOURCE**

**PROCEEDINGS**

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GROUND WATER  
ALASKA'S HIDDEN RESOURCE

PROCEEDINGS

Groundwater: Alaska's hidden resource  
Proceedings  
William S. Ashton  
Alaska Ground Water Association

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Technical Chairman

Alaska Section  
American Water Resources Association  
and  
Alaska Ground Water Association

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**SURFACE WATER QUALITY**



# EFFECTS OF RIPARIAN VEGETATION REMOVAL ON AN ALASKAN SUBARCTIC STREAM

by Mark W. Oswood<sup>1</sup>, John G. Irons III<sup>2</sup>, Jerry W. Hilgert<sup>3</sup>, and Charles W. Slaughter<sup>4</sup>.

## ABSTRACT

The effects of riparian vegetation removal on a headwater stream in subarctic Alaska were examined using upstream-downstream and before and after comparisons. The study stream, Little Poker Creek, is located in permafrost-dominated taiga forest at the Caribou-Poker Creeks Research Watershed. Three adjacent study sections were established: an upstream "control," a section ("cut") destined for vegetation removal, and a downstream "recovery" section. Studies in 1982-4 examined pre-removal differences in the three study sections. Riparian vegetation was removed in the 160 m "cut" section in early spring of 1985, with differences among the three study sections examined in 1985 and 1986. Leaf litter input to the "cut" section averaged 0.58 g AFDW/m<sup>2</sup> compared to 37.22 g in the uncut (control and recovery) sections. Temperatures in the "cut" section showed a slight increase compared to the upstream control section. There were significant differences in densities of macroinvertebrates and their functional groups among the three sections (generally higher densities in the control section), and differences among years for some functional groups. However, Analyses of Variance showed no significant section by year interactions, indicating that these differences were not attributable to riparian clearing.

## INTRODUCTION

Streams and rivers traverse a mosaic of terrestrial patches. Some patches are obvious (e.g. agricultural or urban areas) while other patches may be much less obvious (e.g. localized mineral outcrops or groundwater sources). The riparian zone is the interface between the catchment and the stream (Hynes, 1975), and many disturbances (both natural and anthropogenic) to streams involve changes in the riparian zone. A disturbance can be thought of as a patch and characterized by size of patch, frequency of disturbance and other descriptors (Pickett and White, 1985).

In forested regions, disturbances to the riparian zone are frequent natural events. Fire, wind throw, landslide, insect outbreak or disease create patches in riparian corridors and reset successional clocks. Anthropogenic disturbances likewise create patches in riparian areas. Forest clearcuts provide a graphic example of patch disturbance. Riparian vegetation is frequently removed or modified (e.g., planted in monospecific stands), especially in urban areas.

The key role of riparian vegetation in controlling stream energetics has been recognized for some time (Minshall 1978, Vannote et al. 1980). Riparian vegetation provides leaf litter to detrital food webs in streams but intercepts light and hence reduces primary production by stream autotrophs. Loss of riparian vegetation should lead to a decrease in benthic leaf detritus and benthic organisms specializing in processing of leaf detritus. Conversely,

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increased light to the streambed autotrophs should increase the abundance of grazing invertebrates. However, patch size is critical; loss of riparian vegetation over short lengths of streams will presumably have little effect since detritus is transported downstream from areas of intact riparian vegetation. Large patches, e.g. clearcuts, seem clearly associated with changes in stream energetics (e.g. Murphy et al., 1981). However, the minimum patch size (threshold) necessary to produce significant changes in stream energetics is not known. Such threshold considerations may be critical for urban and suburban land use planning where patches are typically numerous and small.

## METHODS

### Site Description

The Caribou-Poker Creeks Research Watershed (CPCRW) is in the Yukon-Tanana uplands of central Alaska at 65°10' N. latitude and 147°30' W. longitude, 47 km north of Fairbanks. The 106 km<sup>2</sup> (40.8 mi<sup>2</sup>) catchment has developed as a mature dendritic drainage system, incised in the Birch Creek schist of the Yukon-Tanana Uplands physiographic province. Predominantly shallow soils on the slopes and retransported deposits of fine-grained, ice-rich sediments with incorporated organics in the valley bottoms are poorly developed silt loams; permafrost-underlain soils are generally found in valleys and on north-facing slopes, underlying open black spruce (*Picea mariana*) forest. The alpine and subalpine ridges, north slopes, and valleys commonly have permafrost at shallow depth. South-facing slopes are generally free of permafrost.

This study was conducted in Little Poker Creek valley, an 11.4-km<sup>2</sup> first-order basin tributary to Caribou Creek. The study stream reach is a narrow (1-2 m width) meandering pool-and-riffle channel in the permafrost-underlain lower valley of Little Poker Creek. The elevation of the study site is ca. 250 m MSL (maximum elevation in the basin is 773 m MSL). Riparian vegetation is comprised of a very open forest overstory of black spruce (*Picea mariana*) with lesser amounts of balsam poplar (*Populus balsamifera*), larch (*Larix laricina*) and aspen (*Populus tremuloides*). The immediate streamside vegetation is dominated by willow (*Salix* sp.), alder (*Alnus crispa*), dwarf birch (*Betula glandulosa*) and blueberry (*Vaccinium uliginosum*), with major ground cover components of *Eriophorum vaginatum* and mosses. Maximum streamside tree heights in the study reach are 12-15 m.

The climate of the study area is continental, with very cold, long winters and brief, warm summers. Less than half of the ca. 300 mm annual precipitation is received as snow, but snow covers the landscape for about 220 days of the year. Mean January temperature is below -25 °C, vs. mean July temperature of ca. 17 °C. During the three summers of this study, monthly precipitation ranged from 1.8 cm to 11.4 cm (Table 1).

Table1. Monthly precipitation (cm), Caribou Creek Valley.

|                            |      |      |      |
|----------------------------|------|------|------|
| June                       | 1.8  | 7.7  | 5.9  |
| July                       | 8.1  | 3.9  | 9.6  |
| August                     | 5.7  | 8.1  | 11.4 |
| September                  | 2.1  | 9.7  | 4.1  |
| Totals -<br>June-September | 17.7 | 29.4 | 31.0 |

The summer streamflow regime of Little Poker Creek, whose catchment is 19% underlain by permafrost, is typical of headwaters streams in the discontinuous-permafrost taiga. The monthly hydrologic response ratio generally varies between 0.4 and 0.8 in summer; the stream is quite "flashy" (responsive to summer storms), and summer streamflow (Figure 1) exhibits a flow regime intermediate between permafrost-dominated and permafrost-free catchments: storm response is sharp, with rapid rises and recessions, but baseflow is maintained at 8-10 L / sec/km<sup>2</sup>.

### Approach

We report here on Phase II of the research (began in 1982) which involved physical, chemical and biological characterization of three study sections: (1) a 160 m long reach subjected to riparian vegetation removal in early spring 1985 (termed the "cut" reach), (2) a 100 m long section immediately upstream of the "cut" reach, termed the "control" reach, and (3) a 100 m long reach downstream of the "cut" reach, termed the "recovery" reach. Phase II (1985-1986) studies at the site involved determination of the consequences of removing riparian cover to the stream biota (e.g., possible shift from allochthonous to autochthonous energy base, changes in community structure) and physical conditions (e.g., water temperature.). Our protocol for examining effects of vegetation removal thus involves two study designs: (1) comparison of upstream, experimental (cleared) and downstream (recovery) stream reaches; and (2) "before and after" studies of all three stream reaches.

### Water Temperature

Twelve thermistors were placed in the water column along a longitudinal transect in Little Poker Creek. Data (in millivolts) were recorded on a Campbell CR5 datalogger (Campbell Scientific) on audio cassette magnetic tapes. These data were then converted to degrees Celsius using a second order polynomial regression equation (each thermistor individually calibrated to an ASTM thermometer). Data were recorded hourly for most of the ice-free season.

### Leaf Litter Input

Leaf litter input was determined with 0.327 m<sup>2</sup> litter trays suspended over the stream surface. Trays consisted of wooden frames with wire mesh (1 mm openings) bottoms. Input was monitored over the ice-free seasons of 1985 and 1986.

### Periphyton Biomass Determinations

Flat rocks were selected from the streambed at each site: periphyton were detached by scraping with a wire brush and preserved in 3% formalin. Samples were placed in pre-washed and tared 47 mm diameter aluminum weighing pans and dried at 105 °C to a constant weight, then ashed at 550 °C for 1 hour for determination of ash-free dry weight. Artificial substrates - glass slides (76 X 25 mm) and ceramic spheres (7.6 cm diameter) - were placed in riffles at each site in 1985 and 1986 for determination of periphyton biomass accumulation. Slides and spheres were removed after six weeks, scraped, and ash-free dry weight determined as above.

## Benthic Sampling

Benthic invertebrate sampling was done with a Surber sampler modified by substitution of a 350  $\mu\text{m}$  mesh catch net and addition of a foam rubber collar to the metal frame contacting the substrate. These modifications decreased the likelihood that early instars of insects would be lost through the net and decreased loss of specimens beneath the sampler when used on irregular substrates.

Quantitative samples prior to riparian vegetation removal were obtained in 1982 on 14 July, 4 August, 23 August, 15 September, and 6 October and in 1983 on 24 May and 6 June. Post cut benthic samples were obtained on 21 June, 17 July, 14 August, 13 September, 30 September and 15 October in 1985. Benthic samples in 1986 were obtained on 6 June, 27 June, 3 August, 3 September, 19 September and 10 October. On each sampling date three random samples were obtained in each study reach. Samples were fixed in Kahle's Fluid, rinsed and transferred to 80% ethanol.

Benthic invertebrates were sorted from debris under a dissecting microscope. Identifications were made to the lowest practical taxonomic level and organisms counted to obtain estimates of numerical abundance. Sorted samples were gently sieved through 1 mm mesh sieves. Ash-free dry weights were obtained for the resulting coarse particulate organic material (CPOM) by drying at 50°C and ashing at 500°C.

## RESULTS

### Periphyton Standing Crop and Biomass Accumulation

Standing crop biomass as ash-free dry weight from samples collected in 1984 through 1986 ranged from 0.37 to 5.51 g/m<sup>2</sup> (Table 2). Variability between samples was high and there was no apparent response of algal standing crop to riparian vegetation removal. Biomass accumulation on glass slides (Table 3) and ceramic spheres (Table 4) exhibited the same high variability. Biomass accumulation appeared higher at the control site during 1985 on ceramic spheres; however, this was not apparent in 1986.

### Water temperatures

Consistent with permafrost presence in the drainage basin, water temperatures in Little Poker Creek are cold. Daily mean temperatures in 1986 did not exceed 4.5 °C (Figure 2). Temperatures in the cut section periodically exceeded temperatures in the unimpacted (control and recovery) sections (Figure 2) but temperature differentials were very small (approximately 0.3 °C differential between the cut and control sections) and very unlikely to have ecological consequences.

### Litter Input

Not surprisingly, removal of riparian vegetation substantially reduced leaf litter input to the stream surface (Figure 3). Average input to the unimpacted sections was 37.22 g/m<sup>2</sup>/year compared to 0.58 g/m<sup>2</sup>/year in the cut section.

Table 2. Periphyton from rocks, mean ash-free dry weight, (N=3).

|                 | Mean Ash-free<br>Dry Weight<br>(grams per square meter) | Standard<br>Deviation |
|-----------------|---|-----------------------|
| June 20, 1984   |   |                       |
| Control         | 0.37  | 0.19                  |
| Cut             | 0.40  | 0.09                  |
| Recovery        | 3.65  | 2.21                  |
| July 31, 1985   |   |                       |
| Control         | 0.80  | 0.29                  |
| Cut             | 2.13  | 0.63                  |
| Recovery        | 1.95  | 0.52                  |
| July 11, 1986   |   |                       |
| Control         | 1.50  | 1.20                  |
| Cut             | 3.64  | 2.35                  |
| Recovery        | 2.28  | 2.69                  |
| August 15, 1986 |   |                       |
| Control         | 4.99  | 2.15                  |
| Cut             | 3.81  | 3.04                  |
| Recovery        | 5.51  | 1.88                  |

Table 3. Periphyton from glass slides, mean ash-free weight (N=3).

|                                     | Mean Ash-free<br>dry weight<br>(grams per square meter) | Standard<br>deviation |
|-------------------------------------|---|-----------------------|
| June 19, 1985 to July 31, 1985      |   |                       |
| Control                             | 0.67  | 0.48                  |
| Cut                                 | 0.55  | 0.10                  |
| Recovery                            | 0.48  | 0.12                  |
| June 19, 1985 to September 12, 1985 |   |                       |
| Control                             | 1.43  | 0.08                  |
| Cut                                 | 1.79  | 0.48                  |
| Recovery                            | 0.42  | 0.20                  |
| July 9, 1986 to August 15, 1986     |   |                       |
| Control                             | 0.79  | 0.05                  |
| Cut                                 | 0.61  | 0.05                  |
| Recovery                            | 0.16  | 0.12                  |

Table 4. Periphyton from spheres, mean ash-free dry weight, (N=3).

|                                     | Mean ash-free<br>dry weight<br>(grams per square meter) | Standard<br>deviation |
|-------------------------------------|---|-----------------------|
| June 19, 1985 to July 31, 1985      |   |                       |
| Control                             | 1.33  | 0.24                  |
| Cut                                 | 0.45  | 0.04                  |
| Recovery                            | 0.40  | 0.04                  |
| July 31, 1985 to September 12, 1985 |   |                       |
| Control                             | 1.67  | 0.06                  |
| Cut                                 | 0.51  | 0.20                  |
| Recovery                            | 0.43  | 0.14                  |
| July 9, 1986 to August 15, 1986     |   |                       |
| Control                             | 0.25  | 0.17                  |
| Cut                                 | 0.24  | 0.02                  |
| Recovery                            | 0.33  | 0.05                  |

TABLE 5. ANOVA tables for macroinvertebrates densities. Site compares densities among control, cut, and recovery sites, and year compares pre-cut (year 1) and post-cut (years 2 and 3).

| TOTAL MACROINVERTEBRATES |                |     |             |         |       |
|--------------------------|----------------|-----|-------------|---------|-------|
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 2180929.119    | 2   | 1090464.560 | 6.654   | 0.002 |
| Year                     | 459226.264     | 2   | 229613.132  | 1.401   | 0.249 |
| Site x Year              | 569836.255     | 4   | 142459.064  | 0.869   | 0.484 |
| Error                    | .265480E+08    | 162 | 163876.250  |         |       |
| SHREDDERS                |                |     |             |         |       |
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 44836.872      | 2   | 22418.436   | 1.125   | 0.327 |
| Year                     | 150290.738     | 2   | 75145.369   | 3.770   | 0.025 |
| Site x Year              | 78651.957      | 4   | 19662.989   | 0.987   | 0.417 |
| Error                    | 3228728.357    | 162 | 19930.422   |         |       |
| PREDATORS                |                |     |             |         |       |
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 7855.715       | 2   | 3927.857    | 5.300   | 0.006 |
| Year                     | 13666.238      | 2   | 6833.119    | 9.220   | 0.000 |
| Site x Year              | 3933.001       | 4   | 983.250     | 1.327   | 0.262 |
| Error                    | 120064.444     | 162 | 741.139     |         |       |
| COLLECTOR-GATHERERS      |                |     |             |         |       |
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 1081262.691    | 2   | 540631.346  | 9.887   | 0.000 |
| Year                     | 34968.914      | 2   | 17484.457   | 0.320   | 0.727 |
| Site x Year              | 144450.869     | 4   | 36112.717   | 0.660   | 0.620 |
| Error                    | 8858132.183    | 162 | 54679.828   |         |       |
| FILTER-FEEDERS           |                |     |             |         |       |
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 28301.196      | 2   | 14150.598   | 2.547   | 0.081 |
| Year                     | 39197.695      | 2   | 19598.847   | 3.528   | 0.032 |
| Site x Year              | 32778.206      | 4   | 8194.552    | 1.475   | 0.212 |
| Error                    | 900066.556     | 162 | 5555.966    |         |       |
| SCRAPERS                 |                |     |             |         |       |
| SOURCE                   | SUM-OF-SQUARES | DF  | MEAN-SQUARE | F-RATIO | P     |
| Site                     | 327.548        | 2   | 163.774     | 2.126   | 0.123 |
| Year                     | 5.779          | 2   | 2.889       | 0.038   | 0.963 |
| Site x Year              | 247.539        | 4   | 61.885      | 0.803   | 0.525 |
| Error                    | 12482.190      | 162 | 77.051      |         |       |

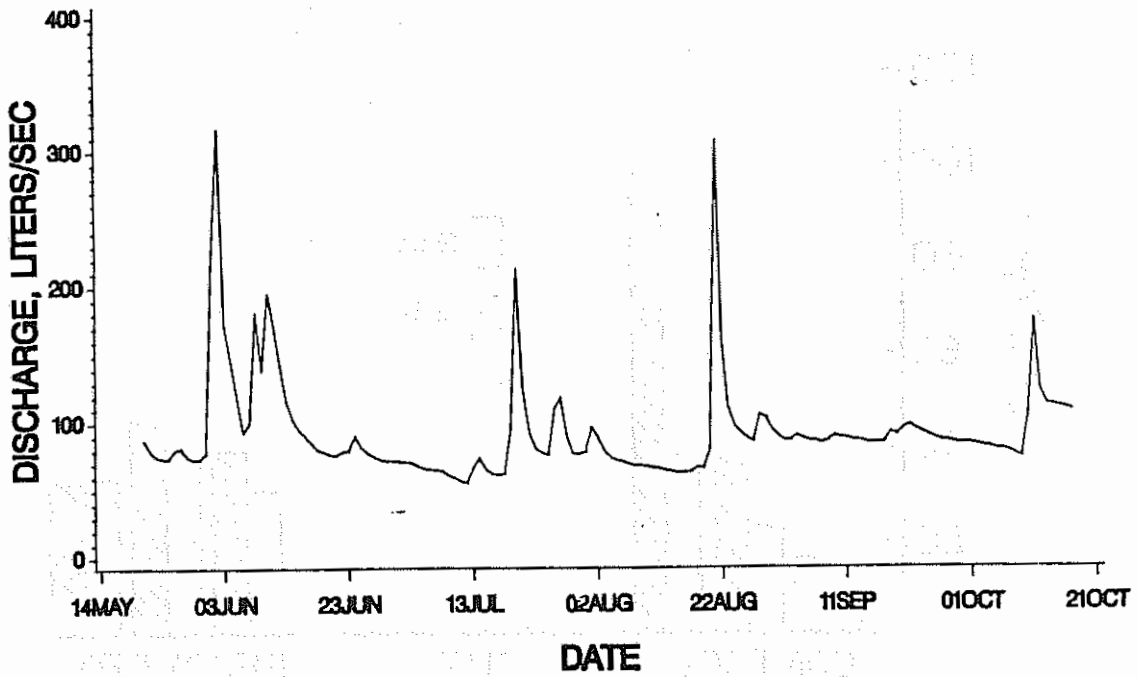


Figure 1. Mean daily discharge for the 1986 ice-free season.

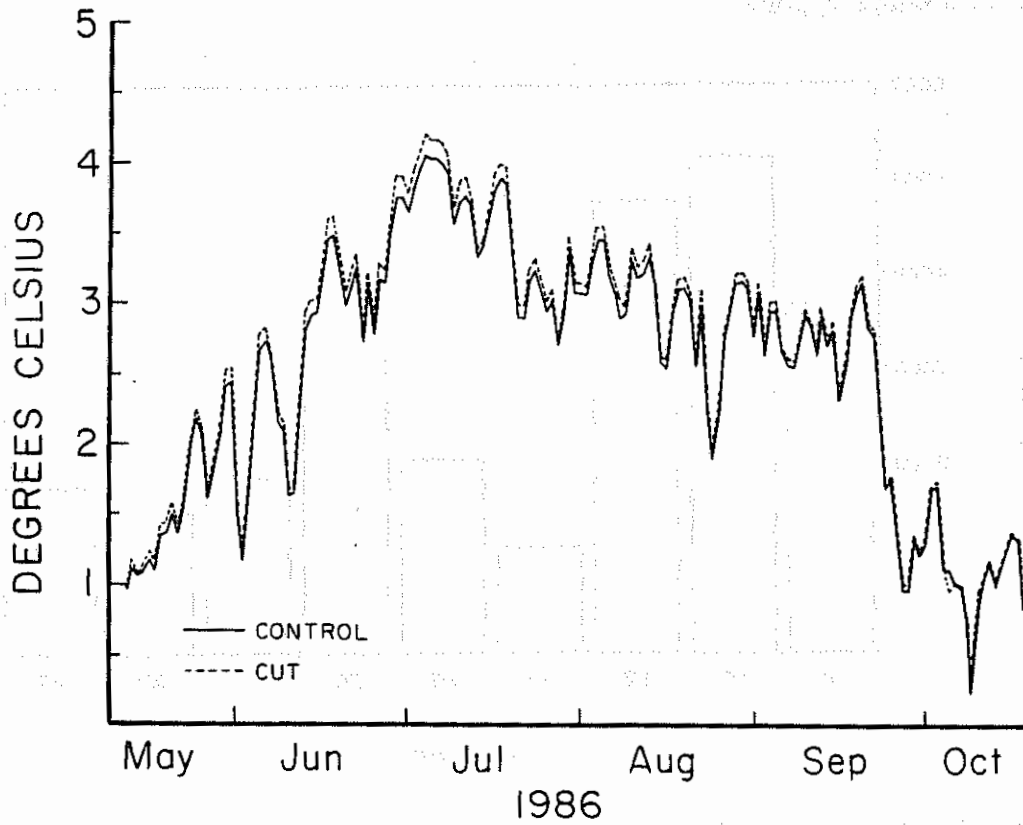


Figure 2. Water temperatures in control and cut sections.

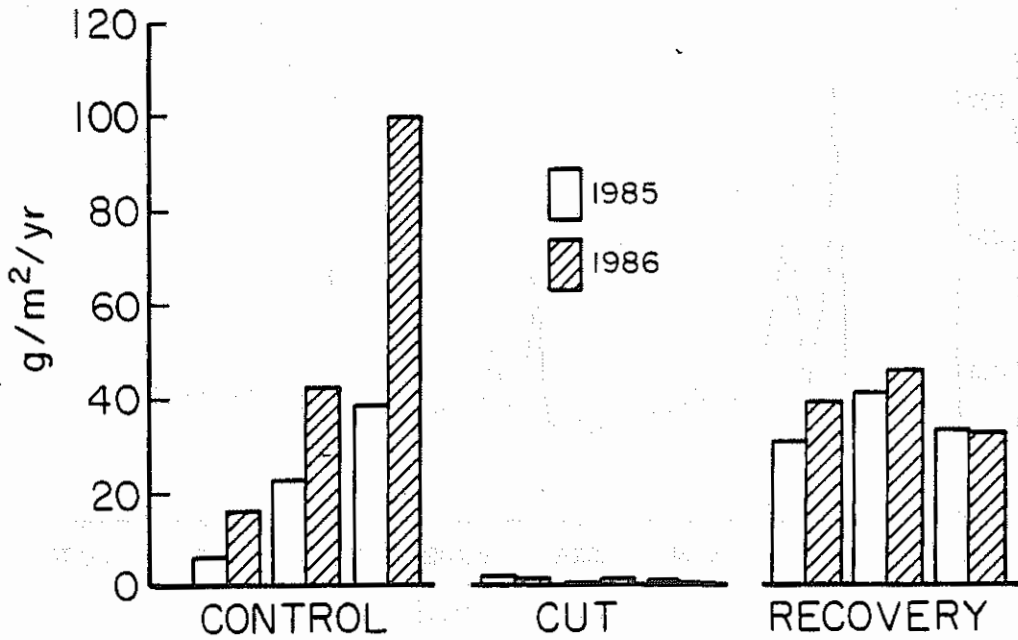


Figure 3. Leaf litter input to the three study sections. Each pair (1985 and 1986) of bars indicates a sampling point.

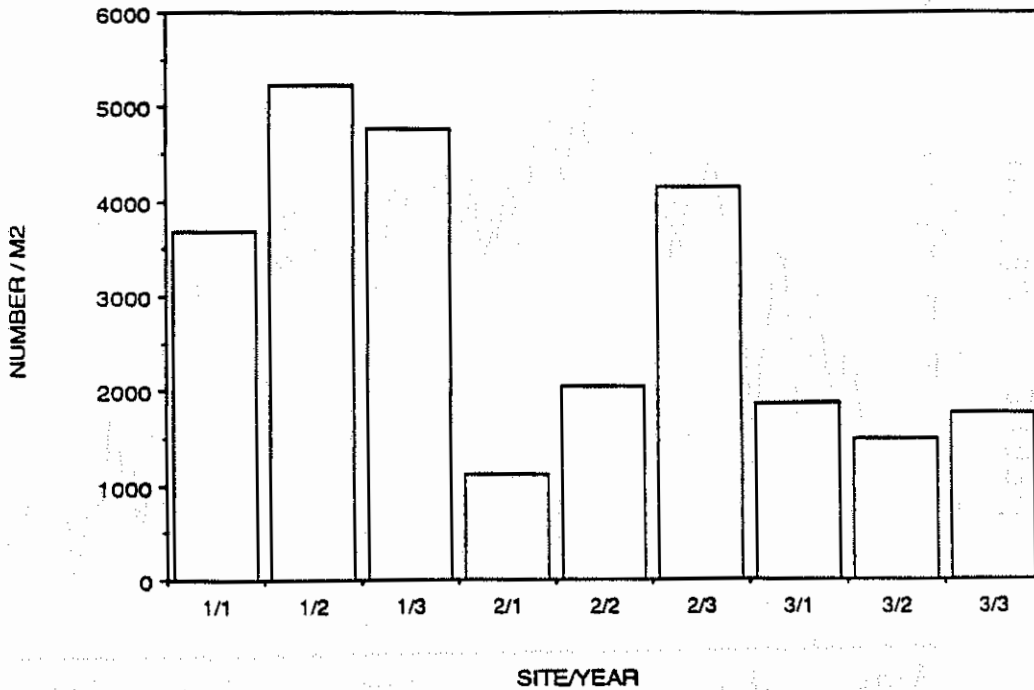


Figure 4. Abundance of benthic invertebrates. Site 1 = upstream control site, site 2 = the cut site and site 3 = the recovery site. Year 1 is prior to riparian vegetation removal; years 2 and 3 are after vegetation removal.



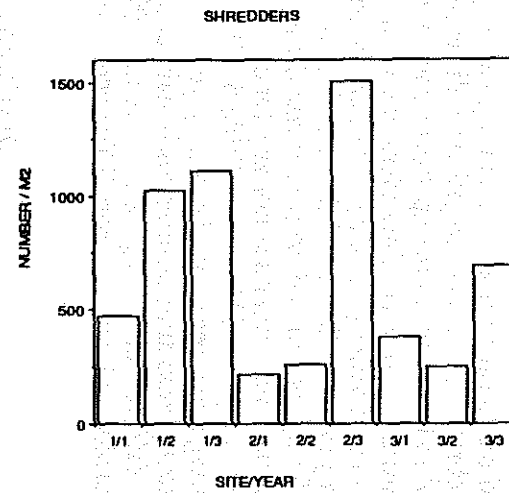
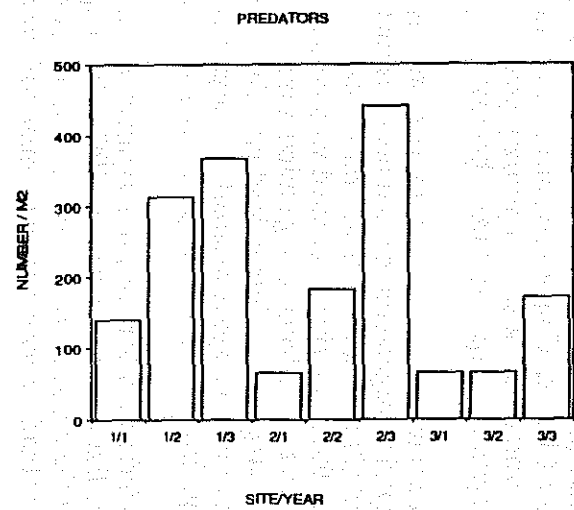
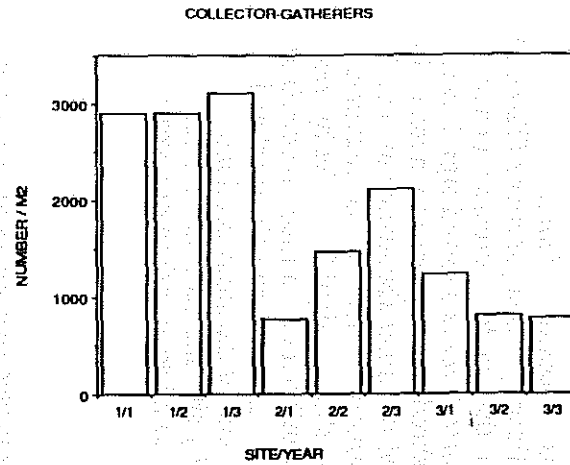
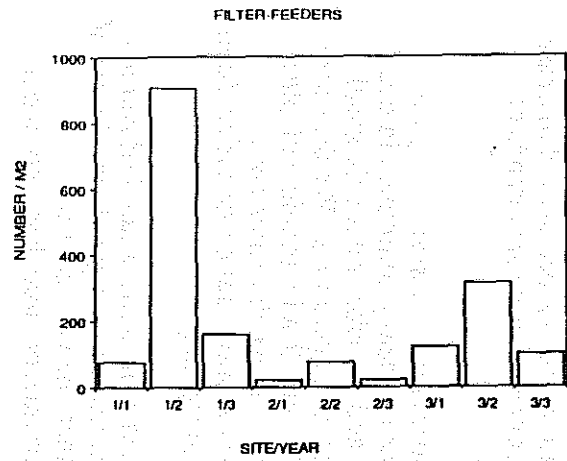


Figure 5. Abundance of functional groups of benthic invertebrates. Site and year designations as in Figure 4.

### Benthic Coarse Particulate Organic Material

Differences in CPOM storage among sections were consistent with decreased leaf litter input to the cut section. Average values of benthic CPOM (g AFDW /m<sup>2</sup>) over the 1985 and 1986 sampling seasons were 2.95 (control), 1.85 (cut) and 3.24 (recovery). However, Analysis of Variance showed no significant difference (P =0.10) in benthic CPOM among sections (two outlier values excluded). The severe reduction in litter input to the cut section was not reflected in benthic storage of CPOM.

### Benthic Macroinvertebrates

Densities of benthic macroinvertebrates showed complex spatial (differences among sections) and temporal (differences among years) patterns. Densities of total macroinvertebrates and collector-gatherers differed significantly among sections (Table 5) and appeared to be higher in the upstream control section than in downstream (cut and control) sections (Figure 4). Shredders, predators and filter-feeders showed significant differences among years (Table 5). Both predators and shredders appeared to increase in density from years 1 to 2 to 3, while filter-feeders showed maximal abundance in year 2 (Figure 5). Scrapers showed no significant differences among either years or sections. Neither total macroinvertebrates nor any functional group showed significant site by year interaction in Analyses of Variance (Table 5). Therefore, differences in macroinvertebrate densities among sites were not affected by year (before and after riparian clearing) and likely represent preexisting differences in habitat quality among the study sections. Conversely, differences among years were not affected by site and probably reflect climatic, hydrological, or energetic differences (common to all sections) among years.

## DISCUSSION

The potential effects of timber harvest on stream ecosystems (see overviews in Geppert et al., 1985, Oswood et al., 1984) include changes in sediment input, water temperatures, and trophic dynamics of the benthic community. Increased sediment loading is generally associated with roads and damage to stream banks, while trophic responses derive from loss of riparian vegetation. Loss of riparian vegetation might be expected to lead to decreased leaf litter input and hence decreases in stream organisms (shredders) dependent upon benthic CPOM. Conversely, increased light input to the stream bed should increase biomass or production of stream primary producers and hence increase abundance of organisms (grazers) dependent upon benthic periphyton.

Our study differed in two ways from most studies of timber harvest impacts : (1) we removed riparian vegetation but did not construct roads or use heavy equipment in tree removal and (2) the cut section was only 160 m in length and therefore relatively small compared to the patch size associated with commercial logging operations. We therefore anticipated that any effects of vegetation removal would be manifest in trophic variables rather than changes in sediment loading. The small size of our patch makes our experiment more analogous to small clearings associated with urbanization or natural forest patches (e.g. fire or herbivore impacts) than to commercial logging.

Our results indicate that removal of riparian vegetation in this subarctic watershed had few effects. It is likely that transport of CPOM from upstream reaches mitigated local losses of leaf litter in the cut section. Likewise, it seems possible that the very cold water temperatures (and perhaps nutrients) may limit primary producers more than light input.

Presumably, removal of riparian vegetation over much longer continuous stream reaches would accentuate the small changes in water temperature and benthic detrital storage, with concomitant changes in functional composition of benthic invertebrates.

Our use of both "before and after" and "upstream-downstream" approaches in study design provided "controls in both time and space" (Green, 1979). We found differences, apparently unrelated to riparian vegetation removal, among our experimental sections. Without "before" sampling, such changes could be erroneously attributed to effects of vegetation removal. Likewise, there were substantial differences in macroinvertebrate densities among years. These were apparently the result of natural variability and not vegetation removal. All studies of ecosystem impacts must contend with spatial heterogeneity among experimental sites and temporal variability. Study designs require both temporal and spatial controls.

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## Water Quality and Macroinvertebrate Distribution in Anchorage Streams

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Surveys of resident biota are an important mechanism for providing insights into the overall pattern of water quality within stream systems. A number of States use instream assessments of benthic macroinvertebrates to determine attainment of aquatic life standards and the maintenance of ecological integrity. Certain groups of macroinvertebrates, particularly the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) are sensitive to impairment in water quality. The community structure of unimpaired streams typically possess a wide diversity of taxa from these sensitive groups which potentially become reduced in number or absent as water quality becomes degraded.

A program to examine macroinvertebrate communities in a number of Anchorage streams was initiated in 1988 in conjunction with the associated study of physical and chemical parameters. A total of thirteen sites were sampled in four stream systems; Fish Creek, Chester Creek, Campbell Creek (including Little Campbell Creek) and Ship Creek. With the exception of Fish Creek, one site in each stream was established so as to be above possible areas of impairment. Sites were also selected so as to minimize substrate size, current velocity and water depth variations between sites. Quantitative samples were taken at three different time periods using a modified Surber sampler, sorted and subsequently identified to the lowest taxa feasible.

Sites were compared according to taxa richness, the ratio of individuals belonging to Ephemeroptera, Plecoptera and Trichoptera (EPT) to total individuals and the presence or absence of taxa with low scores in the Hilsenoff family biotic index. The EPT/total individuals ratios ranged from 0 in Fish Creek to 0.90 in Ship Creek. Chester Creek, Ship Creek, and Little Campbell Creek all showed progressive downstream reductions in taxa richness and the EPT/total individuals ratio when compared to the upstream control site. Campbell Creek displayed insignificant downstream impairment. Based on these approaches sites could be classified into one of three types according to impairment to water quality, (1) insignificant impairment, (2) some impairment and (3) substantial impairment. Seasonal variations were recorded with taxa richness typically lowest in July, but two sites on Little Campbell Creek appeared to show a significant improvement in water quality in September as reflected by the incursion of sensitive taxa into the community structure.

Overall this study has provided baseline information on macroinvertebrate communities in four Anchorage stream systems and shown that the community structure varies significantly according to the water quality regime at the site.

# LIMNOLOGICAL CHARACTERISTICS OF BIG AND LITTLE MINTO LAKES, ALASKA

by Laura L. Jacobs<sup>1</sup>

## ABSTRACT

Basic limnological characteristics (maximum and Secchi disk depth, pH, alkalinity, total and calcium hardnesses, total and soluble reactive phosphorus, total Kjeldahl nitrogen, nitrate, nitrite and ammonia) were measured on Big and Little Minto Lakes on 18 May, 7 July, and 17 August, 1988. Both Big and Little Minto Lakes are relatively shallow (mean maximum depths were 2.5 m and 1.0 m respectively) eutrophic lakes. A shoreline development index of 2.5 for both lakes suggests that they have the potential for developing large littoral communities. Relatively high nitrogen and phosphorus levels were measured throughout the summer on both lakes: Big Minto had a total Kjeldahl N of 2.0-3.6 mg/L; and a total P of 0.17-0.33 mg/L; Little Minto had a total Kjeldahl N of 2.2-5.35 mg/L, and a total P of 0.17-0.48 mg/L. Epiphytic periphyton biomass, as estimated by chlorophyll measurements, increased from July to August. During these two months, total Kjeldahl N and soluble reactive phosphorus increased in both lakes, while total phosphorus levels decreased.

The main focus of my study is to determine the relative importance of periphyton, detritus, water depth, water temperature, vegetation type and plant biomass to macroinvertebrate densities along the Big Minto Lake shoreline. Future plans include these analyses.

## INTRODUCTION

Minto Flats, recognized as one of Alaska's prime wetland areas, was made a state wildlife refuge in April 1988. The important Northern pike fishery and fall waterfowl hunting have brought Minto Flats prominence. This prominence has promoted increased activity by recreational hunters and anglers, and in turn caused concern for the area's ability to sustain excessive taking of waterfowl and fish.

Two lakes within the Minto Flats which are now easily accessed by recreational hunters and anglers, and are traditional hunting areas for Minto villagers, are Big and Little Minto Lakes. These subarctic lakes are located 56 km west of Fairbanks and have supported high densities of fish and waterfowl for

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sometime according to Minto villagers. The Athabaskan name for the lakes reflects this high density of waterfowl. Minto Lake in Athabaskan means "duck feces lake" (Shepherd unpublished, 1987). Although waterfowl and fish populations have been studied, little information exists on other characteristics of the Minto Lakes area.

Studies on Alaskan lakes have documented a wide range of nutrient levels. Hobbie (1960) and Kalff (1967) found arctic lakes to be relatively oligotrophic. Similarly, low nutrient levels were reported for coastal subarctic lakes (Goldman, 1960; Dugdale and Dugdale, 1961; Burgner, 1964) and a few interior subarctic lakes (USGS provisional data, 1975; LaPerriere, Tilsworth, and Casper, 1978). But research on other interior subarctic lakes have reported high nutrient levels relative to other Alaskan lakes (Barsdate, 1967; Alexander and Barsdate, 1971; Barsdate and Alexander, 1971; Alexander and Barsdate, 1974; USGS provisional data, 1975; Vining, 1984; Heglund, 1988).

Currently, studies of the northern pintail ecology and Northern pike population dynamics are being conducted in the Minto Lakes area. Central to these studies is the availability of food. Macroinvertebrates are important food sources to both young pintail (Sugden, 1973) and pike (Solman, 1945), and adult pintail (Krapu, 1974), (adult pike only consume macroinvertebrates opportunistically) and this has been found to be true for pike and pintail of Minto Lakes (F. Burris and B. Murphy, personal communication).

My study was designed to determine limnologic characteristics (morphology and chemistry) of Big and Little Minto Lakes; determine the relative importance of periphyton, detritus, water depth, water temperature, vegetation type and plant biomass to macroinvertebrate abundances along the Big Minto Lake shoreline; and document taxonomic composition and trophic roles of macroinvertebrates. This paper presents the preliminary results of 1987's data and of 1988's May-August field season. Future work includes analyses of benthic, macroinvertebrate, and plant samples. Diel oxygen data were also collected during the 1988 field season and will be analyzed as well.

#### STUDY AREA

Big and Little Minto Lakes are located approximately 56 km west of Fairbanks ( $64^{\circ} 54.4'$ ,  $148^{\circ} 46.3'$  and  $64^{\circ} 52.9'$ ,  $148^{\circ} 50.5'$  respectively). Elevation is at 308 m above mean sea level. Surrounding vegetation includes birch, black and white spruce, cottonwood, aspen and wide expanses of grasses in the upland areas, while low-lying areas and lake margins are covered with willow, alder, sedges and other semi-aquatic vegetation. Both lakes act as collecting basins during high water events on the Chatanika River, Little Goldstream Creek, and the Tanana River (Shepherd and Matthews, 1985). For this reason, lake levels are low during years of low snowfall and precipitation.

Likewise, during years of high snowmelt and precipitation, the flow of water from Goldstream Creek to the Chatanika reverses, and water backs up into the lake system, thereby increasing water levels.

Seasonal fluctuations of this sort may have major effects on wetland productivity (van der Valk and Davis, 1978; Shepherd and Matthews, 1985; Murkin and Kadlec, 1986).

Historically, Goldstream Creek emptied into the inlet of Little Minto Lake and flowed out a well-defined outlet. Sometime in the 1940-50's, the inlet to Little Minto Lake was filled in by sediments. Now groundwater, runoff and high-water events contribute water to the lake. Big Minto Lake also has one obvious channel that serves both as inlet and outlet depending on water levels of the Little Goldstream Creek, Chatanika and Tanana Rivers.

Both lakes are ice-free about 4 months of the year (early to mid-May to late September to mid-October). The Minto Lakes area exhibits typical interior Alaska weather patterns: mean summer temperatures of  $15.7^{\circ}$  C, mean winter temperatures of  $-26^{\circ}$  C, maximum frost-free days equaling 110 days, and mean annual precipitation equaling 29-43 cm (Andrews, 1988).

Origin of the two lakes has not been identified. Surficial studies of the Minto Flats have indicated that most deposits are composed of flood plain alluvium, swamp deposits, and abandoned flood plain alluvium (Shepherd and Matthews, 1985). Along the base of the hills to the west of the lakes lie extensive areas of perennially frozen silts, which are thought to have been deposited in the early Pleistocene.

## METHODS

### Limnology

During July 1987, preliminary sampling of limnological features in both lakes was conducted. Five stations were chosen using a stratified random design: mid-lake, major bays and outlets. Water was collected with a plastic, opaque water column sampler and poured into Nalgene bottles for immediate analysis of pH; preserved with 0.1-mL sulfuric acid for later cation analyses ( $\text{Cl}^-$ ,  $\text{SO}_4^{2-}$ ); bottled unpreserved for later anion analyses ( $\text{Ca}^{+2}$ ,  $\text{Mg}^{+2}$ ,  $\text{K}^+$ ,  $\text{Na}^+$ ); and sealed into glass sample tubes for later total phosphorus analysis. All sample portions were kept cool and dark. Cation, anion, and total phosphorus analyses were completed by the University of Missouri Limnology Laboratory (see Heglund, 1988 for methods). All other characteristics were analyzed within 1-2 hours of our arrival in camp with Hach water chemistry kits (these methods are described in further detail below). Based on preliminary measurements, I determined which of

the limnological characteristics would be sampled more intensively the following summer.

During May 1988, a sampling station was established at the deepest point along the longest axis of each lake and marked with anchor and buoy. Water samples were collected at these stations three times: 18 May, 7 July, and 17 August.

At each sampling station, weather information, water depth, and Secchi disk depth were recorded. Water was collected with a plastic, opaque water column sampler at three depths (surface, mid-depth, and just above the bottom) on Big Minto and at two depths on Little Minto (surface and just above the bottom) due to the shallower water there. Three replicate samples were taken at each depth. Immediately upon pulling the water sampler, water temperature was measured with a calibrated mercury thermometer. A portion of the water sample was then placed in a large Nalgene bottle for immediate analysis (within 1-2 hours of acquisition) of pH, alkalinity, total and calcium hardnesses, color, nitrate/nitrite-N, nitrite, and ammonia with Hach water chemistry kits. Alkalinity, calcium and total hardnesses were determined with the Hach digital titration methods. The results of calcium hardness analysis were then subtracted from total hardness results to calculate magnesium hardness. Water samples were not filtered nor centrifuged, therefore measurements are equal to apparent water color (American Public Health Association, 1985). Ammonia measurements were discontinued after the first sampling trip because of problems with standards, and ammonia was subsequently measured as part of total Kjeldahl nitrogen.

Samples taken for soluble reactive phosphorus, total Kjeldahl nitrogen and total phosphorus were filtered, preserved, stored and analyzed according to Hach Chemical Co. (1987). Soluble reactive phosphorus analysis were usually completed within one week of sample acquisition. Total Kjeldahl nitrogen and total phosphorus analyses were completed within 28 days of collection. Standards were run before samples were analyzed and anytime new chemicals were introduced.

Measures of surface area and shoreline development of each lake were determined from a USGS 1:1354 color infrared aerial photograph (taken July, 1979) read on a Talos digitizing light table. Change in lake levels were monitored from established benchmarks on each lake, with a stadia rod and hand level.

### Habitat Characteristics

Epiphytic periphyton, detritus, water depth, water temperature, vegetation type and plant biomass were sampled along the Big Minto Lake shoreline to determine whether these characteristics affect macroinvertebrate abundances there. Aquatic vegetation was removed from the sampled substrate area



delimited by a 0.05 m<sup>2</sup> cylinder. Epiphytic periphyton biomass (as chlorophyll *a*) samples were derived from rinse water (~10% NaCl) in which aquatic vegetation had been shaken (Cattaneo and Kalff, 1980). The rinse water was subsampled and filtered onto glass fiber filters (Gelman Type A/E), which were immediately placed into a dark can of dessicant until transported to the lab. Chlorophyll samples were then frozen until acetone extraction and spectrophotometric analysis. Procedures described by APHA (1985) were followed for chlorophyll analysis. Chlorophyll concentrations were calculated using the formulas of Golterman, Clymo, and Ohnstad (1969).

After periphyton was removed in the field, aquatic macrophytes were placed in plastic bags and stored in a cool, dark location until transported to the lab. Within one week of collection, plant samples were examined for macroinvertebrates, which were removed and preserved in 80% ethyl alcohol. Plant samples were then blotted with paper, weighed "wet" on a Mettler P163 balance to the nearest 0.001 gram, and frozen for further analyses. Macroinvertebrate, plant, and detritus samples will be processed at a later date.

## RESULTS AND DISCUSSION

Both Big and Little Minto Lakes are relatively shallow eutrophic lakes that can warm to 20° C in the summer. Relatively uniform temperatures throughout the water columns of both lakes during each sampling period in 1988, suggest these lakes were probably mixing from top to bottom during the ice-free period.

Surface area and shoreline development estimates may vary seasonally with fluctuating water levels. Big Minto Lake is approximately 8.5 km<sup>2</sup>. During 1988, maximum water depth at the water sampling station fluctuated between 2.4 m and 2.7 m, with the greatest water depth recorded in July. The surface area of Little Minto Lake is 3.64 km<sup>2</sup>. Maximum water depth measured at Little Minto's water sampling station during 1988 ranged from 0.6 m to 1.4 m with the high recorded in July.

Both lakes have a shoreline development index of 2.5. Håkanson (1981) suggests indices greater than 1 indicate an increasingly irregular shoreline. According to Wetzel (1983), an index of 2.5 indicates great potential for development of littoral communities, true of both Big and Little Minto Lakes.

Color measured in APHA platinum cobalt units was high on both lakes during July 1987, July and August 1988 (Tables 1 and 2). In May 1988, I measured color levels that averaged almost half the August 1987 levels. High levels in late summer may be due to photosynthetically fixed carbon that is excreted by aquatic vegetation (McRoy and Goering, 1974; Wetzel, 1983); thick stands of aquatic

vegetation covered with epiphytic periphyton developed in each of the lakes by July.

The concentration of free hydrogen ions as expressed by pH was relatively high throughout the water column during both years (Tables 1 and 2). Levels of pH increase when photosynthetic activity is great and can approach pH 10 on eutrophic lakes (Goldman and Horne, 1983; Wetzel, 1983).

Alkalinity measurements differed slightly between the two lakes. During the summer of 1988, the alkalinity present in Big Minto Lake was due to bicarbonate. During May, alkalinity in Little Minto Lake was due to bicarbonate; however, during July and August, alkalinity in Little Minto was due largely to carbonate near the surface, while bicarbonate was dominant in water near the bottom. Photosynthesis may have greater influence on carbon chemistry in surface waters of Little Minto Lake where uptake of inorganic carbon is rapid as compared to lower water depths where respiration of organic matter may be more important than photosynthesis. Relatively high alkalinity measurements may be due to both ion-rich groundwater and concentration of salts by evaporation (Wetzel, 1983). The sequence of ranked cations ( $\text{Ca}^{+2} > \text{Mg}^{+2} > \text{Na}^{+} > \text{K}^{+}$ ) is similar to the sequence reported for other Alaskan lakes (cf. Alexander and Barsdate, 1971).

Both Big and Little Minto Lakes are eutrophic. Inorganic phosphorus and nitrogen levels of the Minto lakes during the summer of 1988 were within the range of temperate lakes classified as eutrophic (Table 3) (Goldman and Horne, 1983; Wetzel, 1983). The ratios of total nitrogen to total phosphorus are approximately 12 and 15 for Big Minto and Little Minto Lakes, respectively. According to Smith (1979), ratios that fall between 13 and 21 suggest either nitrogen or phosphorus is limiting primary production.

Total Kjeldahl nitrogen, total phosphorus and soluble reactive phosphorus fluctuated seasonally in the same manner in both Minto lakes. Organic nitrogen and ammonia (measured as total Kjeldahl nitrogen) increased from May to August, while total phosphorus decreased from May to July and then rose in August. Soluble reactive phosphorus levels declined in July from May and August values. I assume that the causes of these same fluctuations are the result of similar nutrient demands of aquatic vegetation, periphyton and phytoplankton. From my observations, it seems that during May when the water is still quite cool and the ice is finally gone, nutrients are still high from the previous year's decomposing plant matter. By July, the water has warmed, production of aquatic vegetation, periphyton and phytoplankton is great. Therefore, demands on nutrients were probably high and nutrients that were mixed into the water column were rapidly depleted. Aquatic macrophytes probably continued to translocate sediment nutrients that are then released through the leaves and into the water, where they are available for periphyton and phytoplankton (McRoy and Goering, 1974). By August, plant growth was so luxurious that light may then have become the

limiting factor in further production. Decomposition began to release nutrients from dying plants and active plants continued to release more nutrients. Water fowl feces may also build up through the season and contribute to nutrient cycling.

The amount of epiphytic chlorophyll *a* measured per m<sup>2</sup> bottom area differed significantly between dates, but not between vegetation types (Tables 4 and 5). The amount of epiphytic chlorophyll *a* measured per gram wet weight of plant biomass differed significantly between dates and was significantly greater on submergent than on emergent vegetation (Tables 6 and 7). Dvorak (1987) and Cattaneo and Kalff (1980) reported that epiphytic periphyton biomass was greatest on submergent vegetation because it provided a larger surface area for periphyton colonization than did emergent vegetation.

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Table 1. Measured limnological characteristics of Big Minto Lake. 1988 data are means of samples throughout the water column during each date. Data from 1987 are means of surface samples from five sites.

| Limnological Parameter                               | July 1987 | May 1988 | July 1988 | August 1988 |
|--|-----------|----------|-----------|-------------|
| Temperature (°C)                                     | nm*       | 11       | 20        | 17          |
| Max. depth (m)                                       | nm        | 2.6      | 2.7       | 2.4         |
| Secchi disk depth (m)                                | nm        | 0.5      | 1.8       | 1.8         |
| pH   | 8.7       | 8.0      | 8.0       | 8.5         |
| Phenolphthalein alkalinity (mg/L CaCO <sub>3</sub> ) | nm        | 0.0      | 0.0       | 0.0         |
| Total alkalinity (mg/L CaCO <sub>3</sub> )           | nm        | 57       | 54        | 59          |
| Ca <sup>2+</sup> hardness (mg/L CaCO <sub>3</sub> )  | nm        | 49       | 50        | 52          |
| Total hardness (mg/L CaCO <sub>3</sub> )             | nm        | 80       | 78        | 78          |
| Mg <sup>2+</sup> hardness (mg/L CaCO <sub>3</sub> )  | nm        | 31       | 28        | 26          |
| Calcium (mg/L)                                       | 17.2      | nm       | nm        | nm          |
| Magnesium (mg/L)                                     | 6.84      | nm       | nm        | nm          |
| Sodium (mg/L)  | 3.38      | nm       | nm        | nm          |
| Potassium (mg/L)                                     | 1.33      | nm       | nm        | nm          |
| Sulfate (mg/L)                                       | 2.2       | nm       | nm        | nm          |
| Chloride (mg/L)                                      | 1.00      | nm       | nm        | nm          |
| Color (PtU)  | 81        | 21       | 54        | 59          |
| Ammonia (mg/L)                                       | nm        | 0.5      | nm        | nm          |
| Nitrate (mg/L)                                       | nm        | 0.0      | 0.0       | 0.0         |
| Total Kjeldahl nitrogen (mg/L)                       | nm        | 2.45     | 3.65      | 3.72        |
| Total phosphorus (mg/L)                              | 0.09      | 0.24     | 0.35      | 0.23        |
| Soluble reactive phosphorus (mg/L)                   | nm        | 0.12     | 0.07      | 0.13        |

\*nm = not measured

Table 2. Measured limnological characteristics of Little Minto Lake. 1988 data are means of water column samples for each date. Data from 1987 are means of surface samples from five sites.

| Limnological Parameter                               | July 1987 | May 1988 | July 1988 | August 1988 |
|--|-----------|----------|-----------|-------------|
| Temperature (°C)                                     | nm*       | 13       | 19        | 19          |
| Max. depth (m)                                       | nm        | 0.8      | 1.4       | 0.6         |
| Secchi disk depth (m)                                | nm        | 0.5      | >1.4      | 0.2         |
| pH   | 9.7       | 8.0      | 10.0      | 10.0        |
| Phenolphthalein alkalinity (mg/L CaCO <sub>3</sub> ) | nm        | 0.0      | 30        | 29          |
| Total alkalinity (mg/L CaCO <sub>3</sub> )           | nm        | 39       | 78        | 85          |
| Ca <sup>+2</sup> hardness (mg/L CaCO <sub>3</sub> )  | nm        | 41       | 55        | 58          |
| Total hardness (mg/L CaCO <sub>3</sub> )             | nm        | 64       | 88        | 100         |
| Mg <sup>+2</sup> hardness (mg/L CaCO <sub>3</sub> )  | nm        | 23       | 33        | 42          |
| Calcium (mg/L)                                       | 22.4      | nm       | nm        | nm          |
| Magnesium (mg/L)                                     | 8.46      | nm       | nm        | nm          |
| Sodium (mg/L)  | 3.51      | nm       | nm        | nm          |
| Potassium (mg/L)                                     | 1.38      | nm       | nm        | nm          |
| Sulfate (mg/L)                                       | 1.40      | nm       | nm        | nm          |
| Chloride (mg/L)                                      | 0.84      | nm       | nm        | nm          |
| Color (PtU)  | 78        | 29       | 81        | 110         |
| Ammonia (mg/L)                                       | nm        | 0.7      | nm        | nm          |
| Nitrate (mg/L)                                       | nm        | 0.0      | 0.0       | 0.0         |
| Total Kjeldahl nitrogen (mg/L)                       | nm        | 2.20     | 3.70      | 6.28        |
| Total phosphorus (mg/L)                              | 0.10      | 0.21     | 0.31      | 0.30        |
| Soluble reactive phosphorus (mg/L)                   | nm        | 0.09     | 0.09      | 0.14        |

\*nm = not measured



Table 3. Mean concentrations (mg/L) of inorganic nitrogen ( $\text{NH}_4^+\text{-N}$ ,  $\text{NO}_2^-\text{-N}$ , and  $\text{NO}_3^-\text{-N}$ ) and phosphorus ( $\text{PO}_4^{3-}\text{-P}$ ) of various lakes. Data taken from literature are either mean concentrations recorded during ice-off to late August, or during what was reported as "summer".

| Lake                                      | Total Inorganic Nitrogen | Total Inorganic Phosphorus | Climatic Zone |
|---|--------------------------|----------------------------|---------------|
| Big Minto Lake, Alaska                    | 0.5                      | 0.11                       | subarctic     |
| Little Minto Lake, Alaska                 | 0.7                      | 0.11                       | subarctic     |
| Ace Lake, Alaska <sup>1</sup>             | 0.19                     | 0.13                       | subarctic     |
| Deuce Lake, Alaska <sup>1</sup>           | 1.08                     | 0.17                       | subarctic     |
| Tangle Lakes, Alaska <sup>2</sup>         | 0.07                     | 0.14                       | subarctic     |
| Scottie Desper Lakes, Alaska <sup>3</sup> | 0.01                     | 0.02                       | subarctic     |
| Smith Lake, Alaska <sup>4</sup>           | 0.07                     | 0.04                       | subarctic     |
| Harding Lake, Alaska <sup>5</sup>         | 0.06                     | 0.03                       | subarctic     |
| Quartz Lake, Alaska <sup>5</sup>          | 0.06                     | 0.00                       | subarctic     |
| Birch Lake, Alaska <sup>5</sup>           | 0.05                     | 0.03                       | subarctic     |
| Barrow Ponds, Alaska <sup>6</sup>         | 0.05                     | 0.002                      | arctic        |
| Lake Tahoe, California <sup>7</sup>       | 0.005                    | 0.002                      | temperate     |
| Clear Lake, California <sup>7</sup>       | 0.4                      | 0.02                       | temperate     |

(1) Alexander and Barsdate, 1974; (2) Barsdate and Alexander, 1971; (3) Vining, 1984; (4) Alexander and Barsdate, 1971; (5) USGS provisional data, 1975; (6) Prentki et al., 1980; (7) Goldman and Horne, 1983.

Table 4. Mean epiphytic periphyton biomass expressed as mg chlorophyll  $a/m^2$ .

|                 | Category     | Mean  |
|-----------------|--------------|-------|
| Vegetation Type | Emergent     | 21.48 |
|                 | Submergent   | 26.37 |
| Date            | 8-10 July    | 20.58 |
|                 | 17-20 August | 27.28 |

Table 5. Results of ANOVA testing the effects of: aquatic vegetation type and date of sampling on epiphytic periphyton biomass (mg chlorophyll  $a/m^2$ ). Means are shown in Table 4. \* $P < 0.05$ .

| Source          | df | MS      | F     |
|-----------------|----|---------|-------|
| Vegetation Type | 1  | 358.63  | 2.30  |
| Date            | 1  | 673.95  | 4.32* |
| Interaction     | 1  | 0.34201 | 0.00  |
| Error           | 56 | 156.11  |       |

Table 6. Mean epiphytic periphyton biomass expressed as mg chlorophyll a/gram wet weight of aquatic vegetation.

|                 | Category     | Mean   |
|-----------------|--------------|--------|
| Vegetation Type | Emergent     | 0.0116 |
|                 | Submergent   | 0.0190 |
| Date            | 8-10 July    | 0.0112 |
|                 | 17-20 August | 0.0194 |

Table 7. Results of ANOVA testing the effects of: aquatic vegetation type and date of sampling on epiphytic periphyton biomass (mg chlorophyll a/gram wet weight of aquatic vegetation). Means are shown in Table 6. \*\*P <0.01, \*\*\*P <0.001.

| Source          | df | MS         | F        |
|-----------------|----|------------|----------|
| Vegetation Type | 1  | 0.00083480 | 10.74**  |
| Date            | 1  | 0.00100821 | 12.97*** |
| Interaction     | 1  | 0.00017396 | 2.24     |
| Error           | 56 | 0.00007773 |          |

## TAKING CHARGE - 4-H WATER QUALITY PROGRAMS

by: Chris Greenfield-Pastro 1/

and

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### ABSTRACT

The Cooperative Extension Service has identified water quality as one of nine National Priority Initiatives. Every American depends on freshwater for life and well being. What man is doing with water is a major concern and the education of youth concerning water issues is critical. It is imperative that today's youth understand the crucial importance of water and the threats to the nation's water supply.

The 4-H Youth Development Program in Alaska is in the process of developing a natural resource youth education program. Water quality will be a part of this program. This presentation includes the youth development goals of the 4-H program and reviews water quality educational resources currently available. Nine states have developed water quality 4-H youth programs and approximately 36 publications are available nationwide to assist 4-H leaders develop water quality programs.

This presentation also suggests that members of the Alaska Section AWRA could assist the Alaska 4-H Program in developing educational materials and activities.

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## INTRODUCTION

4-H is the Cooperative Extension System's dynamic, non-formal, educational program for today's young people. The mission of the 4-H program is to assist youth in acquiring knowledge, developing life skills, and forming attitudes that will enable them to become self directing, productive and contributing members of society.

The Cooperative Extension Service has identified nine national initiatives to generate change for the 1990's and into the 21st century. The nine national initiatives include:

- Alternative Agricultural Opportunities
- Building Human Capital
- Competitiveness and Profitability of American Agriculture
- Conservation and Management of Natural Resources
- Family Economic Well Being
- Improving Nutrition, Diet and Health
- Revitalizing Rural America
- Water Quality
- Youth at Risk

Through identification of these initiatives, the Cooperative Extension Service will focus it's resources on critical issues that impact the economic, social and environmental progress of Americans.

Since water quality has been identified as one of the nine national priority initiatives, extension staff will focus on issue based programs in four main areas: (1) public understanding of the nature and importance of water resources, (2) the impact of chemicals on the water supply, (3) water conservation, and (4) community control of water quality. The attention each state's Cooperative Extension Service gives to each of these four issues will depend upon water quality problems peculiar to that state, as well as staff and funding levels.

Extension education can help people understand the causes and effects of water pollution and can help achieve an atmosphere of

cooperation between the varied interests affected by the water quality question (USDA, 1988a). Every American depends on freshwater for life and well being. It is imperative that today's youth understand the crucial importance of water and the threats to the nation's water supply.

The 4-H Program goal is to create a learning environment for youth that not only develops skills and increases knowledge - but one that helps them deal with stress and assists them in learning to help others. Volunteer leaders, assisted by 4-H agents, deliver 4-H programs to youth. The volunteer leader receives training from extension staff and then passes on skills and knowledge to 4-H members in club or project meetings, workshops, camps, leadership seminars or school enrichment programs. Through these activities, 4-H members "Learn by Doing" and at the same time gain coping and leadership skills that will help them become contributing citizens. Approximately 9,600 boys and girls are enrolled in the Alaska 4-H Program. These youth receive guidance from more than 920 volunteer leaders (USDA 1988b).

#### 4-H PROGRAM EMPHASIS ON NATURAL RESOURCES

In November, 1988, a preliminary survey was sent to seventeen Cooperative Extension Service Agents and volunteer 4-H leaders in Alaska. The purpose of the questionnaire was to determine the degree of interest in expanding a 4-H natural resources program and the direction the program should take.

The results of the questionnaire identified water quality as the highest item of priority to 4-H agents and volunteer 4-H leaders. The results also indicate that only eighty youth are currently enrolled in 4-H natural resource projects statewide.

The Alaska 4-H Program recently has given high priority to developing a sound natural resources program for youth. In 1988 a state team of volunteers and extension staff attended a national volunteer leader forum on natural resource programs for 4-H. Through the efforts of the state team, National 4-H Council and cooperating agencies, Alaska volunteer 4-H leaders will participate in a natural resources training session in the summer of 1989.

Water quality education has not yet been formally introduced to

Alaska 4-H youth. Since it has been identified as the highest natural resource priority area, efforts will be made during the 1989 natural resource training session to train volunteer 4-H leaders to teach youth water quality concepts and how to involve 4-H youth in water quality issues.

Some water quality education activities are happening throughout the state. Approximately forty 4-H members in the Kodiak Aquaculture Club are investigating standards in water quality as they learn about the cleansing effects of tides on crabs and sea urchins. They have also learned how water is contaminated by plastics, drift nets and other pollutants. 4-H members in the Homer/Soldotna area are learning to measure stream flow and water parameters. Last summer fifty 4-H youth in the Tanana District investigated a small lake to identify different plants, animals and insects living in the water habitat. However, these activities are just scratching the surface of the potential 4-H water quality education programs in Alaska.

Extension youth programming which deals specifically with water quality is currently in place in only a few states. Many traditional 4-H programs are in place which deal with issues of conservation of water in both the home and the environment. These are certainly a part of the overall water quality issue and could be the foundation of an updated curriculum on water quality.

Nine states are currently conducting or developing 4-H youth programs to address the water quality initiatives.

Arkansas: Works with SCS staff to conduct water quality programs at all summer camps which reach approximately 1000 4-H members.

Connecticut: Has developed a 4-H program on household hazardous wastes.

Delaware: Conducted water quality workshops in camp settings to reach 200 youth. Members were introduced to water quality concerns and how ground water can be contaminated.

Michigan: One county has developed a groundwater education curriculum for elementary and junior high schools.

Missouri: A science/technology/and society project has been developed which includes water curriculum.

Nebraska: Developed a series of five water quality videotapes with member and leader's guides.

New York: A K-3 and 4-6 curriculum has been developed and introduced in all schools in Nassau County.

Ohio: Developed five activity guides for use in 4-H camp programs.

Vermont: A curriculum for 4-H clubs, camps and school enrichment has been developed and introduced. Three teaching units include: The Water Around Us, Our Ground Water, and Our Surface Water. Video and computer programs are available (Farrell, 1988). Approximately thirty-six publications are available nationwide to help 4-H leaders develop water quality programs (see Appendix A).

Since Alaska does not yet have a 4-H water quality program, there is a particular need for assistance from water scientists and managers in it's initial development. Alaska's environment differs greatly from those in other states where water quality programs have already been developed. Although some of the basic water quality concepts taught elsewhere are applicable, there are undoubtedly special water quality considerations to take account of in Alaska. Local scientific knowledge will be needed in developing our program. We could also use help from local water quality experts who could serve as guest speakers, lead water quality educational activities at camps, or become 4-H leaders and develop 4-H water quality clubs.

## CONCLUSIONS AND IMPLICATIONS

The need for youth to increase their awareness and understanding of water related issues is great. Threats to water quality and environmental deterioration pose serious problems for future generations. In conclusion:



1) We all know there is a need to educate youth about water quality issues to ensure that they will be able to influence water policies in the future.

2) The 4-H club is a very suitable environment to increase youth's understanding of water quality issues and help them become responsible citizens. Because water is a part of everyone's being, it can easily be a part of every 4-H project and could be included in the volunteer leader's plan in working with members.

3) Some excellent teaching materials which deal specifically with water quality have been developed. However, teaching materials are needed which reflect the water quality issues unique to Alaska.

4) The Alaska Section AWRA can be of great assistance in the Alaska 4-H Program's efforts to educate youth about our water supply and its importance. Perhaps the Alaska Section AWRA could establish a committee to develop educational materials and activities for youth that address Alaska's unique water quality needs. Through such a commitment, we can work together to develop a strong education program to address the water quality initiative.

The goal of the Alaska 4-H Program is to provide water quality education through a "Learn By Doing" approach to youth to ensure that tomorrow's citizens have a better understanding of and appreciation for their water supply. Extension educators and the cadre of water scientists and managers must work together to develop 4-H water quality education programs for youth in order to reach the next generation with their message.

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EVALUATION OF THE ENVIRONMENTAL IMPACTS OF  
SNOW DISPOSAL ACTIVITIES WITHIN THE  
MUNICIPALITY OF ANCHORAGE, ALASKA

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ABSTRACT

This report summarizes the results of a study conducted in 1988 by the Water Quality Section of the Municipality of Anchorage's Department of Health and Human Services. The purpose of the study was to evaluate the environmental impacts of 8 snow disposal sites located within the Municipality (see map, pg. app-1). Snow samples were taken throughout the year at all of the disposal sites to determine levels of various chemical constituents in the snow. Monitoring results indicated high levels of oil & grease, suspended solids, lead, iron, copper, and zinc. Volumes of snow at each site were estimated and used to determine the total chemical loadings produced by each of these disposal sites.

Soil sampling was conducted at depths of 1 - 4 feet at several snow disposal sites. Oil & grease, total petroleum hydrocarbons, and lead levels decreased with depth. However, these contaminants were present at depths as great as 4 feet from the surface. Further monitoring of soils is recommended to further define the distribution and fate of these contaminants and to assess their potential impacts on shallow groundwater quality.

Surface water quality monitoring was conducted on water bodies adjacent to the disposal sites. Baseline water samples were collected after extended periods of sub-freezing temperatures to eliminate the influence of snow melt runoff and to provide a basis for comparison. Results were compared to samples taken at regular intervals during the remainder of the year, as the snow which accumulated at these sites slowly melted and either percolated into the soils or flowed overland to nearby surface waters. These comparisons showed general trends and the impacts of snow melt runoff from snow disposal sites on water quality in adjacent surface waters.

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## INTRODUCTION

The Anchorage Airport receives, on average, approximately 70 inches of snowfall each year (National Weather Service, 1989). To maintain the safest driving conditions during the winter months, deicing compounds mixed into sand are applied to road surfaces throughout the Anchorage area. When a snowfall occurs, these compounds are mixed with new fallen snow, removed from the roadways by road maintenance equipment, piled on the roadsides, and later hauled to disposal sites.

Within the Municipality, approximately 3000 lane miles of roadway are maintained by the Municipality and the State of Alaska. Nearly 4000 tons of sodium chloride are applied annually in a salt-sand mixture (3% sodium chloride). During the course of a winter, over 2 million cubic yards of snow are hauled to snow disposal sites.

The magnitude of snow disposal operations has led to concern over the potential impact of deicing compounds, sand and other contaminants deposited on the roadways upon water quality. This investigation focuses on the potential impacts of these constituents on water quality.

## METHODS

Twelve separate testing stations were monitored during this study, and are identified in table 1. Each testing station was monitored either as a snow sample from the disposal site or as a water sample taken from nearby surface waters. The type of sample is indicated in table 1. The table also indicates the estimated snow capacity of the eight separate disposal sites monitored in this study. The disposal sites contained a mixture of snow from residential streets and commercial arterials.

| <u>Station</u> | <u>Sample Type</u> | <u>Location</u>               | <u>Capacity (cu. yd.)</u> |
|----------------|--------------------|-------------------------------|---------------------------|
| 1. CR1         | Snow               | Commercial Dr. near Rampart   | 500,000                   |
| 2. MV1         | Snow               | Mt. View Dr. & Davis Highway  | 300,000                   |
| 3. SK1         | Water              | 15th and Sitka St.            | 200,000                   |
| 4. SK2         | Water              | 15th and Sitka St.            |                           |
| 5. BST1        | Snow               | 13th and "B" St.              | 100,000                   |
| 6. GL1         | Water              | Goose Lake                    | 5,000                     |
| 7. GL2         | Water              | Drainage near Goose Lake      |                           |
| 8. TR0         | Water              | Tudor Rd. near Baxter Drive   | 400,000                   |
| 9. TR1         | Water              | Tudor Rd. near Baxter Drive   |                           |
| 10. TR2        | Water              | Tudor Rd. near Baxter drive   |                           |
| 11. OM1        | Snow               | O'Malley & Old Seward Highway | 100,000                   |
| 12. KL1        | Snow               | Kloep Maintenance Facility    | 500,000                   |

Table 1. Sample Site Locations.

18 parameters were monitored at these sites and each is identified in table 2.

|                           |                   |
|---------------------------|-------------------|
| 1. Hardness               | 10. Iron          |
| 2. Chlorides              | 11. Lead          |
| 3. Oil & Grease           | 12. Zinc          |
| 4. Total Suspended Solids | 13. Cadmium       |
| 5. Total Dissolved Solids | 14. Copper        |
| 6. *Turbidity             | 15. Sodium        |
| 7. Fecal Coliform         | 16. Calcium       |
| 8. Nitrates               | 17. *pH           |
| 9. *Temperature           | 18. *Conductivity |

\* Field Parameters measured for water samples only.

Table 2. Parameter Listing.

Snow samples were gathered in 0.67 cu. ft. containers, sealed and delivered to the laboratory while still in the solid state. Water samples were collected in glass and plastic containers furnished by the laboratories, placed in a cooler with ice, and then delivered to the laboratories. Water and snow analyses were done according to EPA-600, Methods for Chemical Analysis for Water and Waste. Soil samples were obtained by drilling with a two-man auger and placing the samples into 200 ml. glass sample jars furnished by the laboratories. Soil analysis was done according to EPA SW-846, Test Methods for Evaluation of Solid Waste. Laboratory analysis was completed by Northern Testing Laboratories, at 2505 Fairbanks St., Anchorage, Alaska, and by the AWWU laboratory located at the Pt. Woronzof Treatment Facility in Anchorage, Alaska.

For field parameters, pH and temperature were measured with a Beckman model 20 pH meter, conductivity was measured with a YSI model 33 S-C-T conductivity meter, and turbidity was determined using a Hach model 16800 turbidimeter.

## RESULTS

### Snow Samples

The results from sampling snow at the various sites during the course of the melt season were quite consistent.

Parameters which consistently showed amounts in the snow below levels of concern were:

- (1). Hardness, which averaged 32.8 mg/l as CaCO<sub>3</sub>.
- (2). Nitrates, which were at all times below the testing detection limits of 0.10 mg/l.



(3). Fecal coliform, for which all readings except two of 35 were below 10 colonies per 100 ml. The only high samples gave results of 52 and 2000 colonies per 100 ml.

(4). Total dissolved solids had a seasonal mean at all disposal sites of 58.1 mg/l.

(5). Cadmium exceeded its laboratory detection limit of .005 mg/l in three of 31 samples, with a high of .010 mg/l. The remaining 28 samples were below detection limits.

Contaminants in the snow samples which consistently had testing results above the EPA chronic freshwater aquatic life standards (EPA Quality Criteria for Water, 1986) are listed in table 3. Results of special interest included total suspended solids which had a seasonal mean of more than 100 times the average of Goose Lake, a primarily groundwater fed lake. Lead had a mean value of more than 100 times the EPA fresh water chronic level, while copper and zinc averaged 10 times the EPA fresh water chronic level.

| <u>Parameter</u>       | <u>Maximum Level</u> | <u>Minimum Level</u> | <u>Average</u> | <u>Number of Samples</u> | <u>*EPA Level</u> |
|------------------------|----------------------|----------------------|----------------|--------------------------|-------------------|
| Oil & Grease           | 50.0                 | 10.0                 | 17.4           | 35                       | N/A               |
| Total Suspended Solids | 2750.0               | 190.0                | 1114.8         | 21                       | N/A               |
| Chlorides              | 3.9                  | 1.2                  | 2.36           | 31                       | N/A               |
| Iron                   | 127.5                | 10.2                 | 59.34          | 31                       | 1.0               |
| Lead                   | 0.765                | 0.071                | 0.345          | 31                       | 0.003             |
| Zinc                   | 1.28                 | 0.031                | 0.419          | 31                       | 0.047             |
| Copper                 | 0.410                | 0.020                | 0.132          | 31                       | 0.010             |
| Sodium                 | 8.50                 | 1.63                 | 4.76           | 31                       | N/A               |
| Calcium                | 18.17                | 1.02                 | 7.58           | 31                       | N/A               |

All levels are in units of mg/l.  
 \*EPA level is the freshwater chronic limits for aquatic life protection.  
 (EPA Quality Criteria, 1986)

Table 3. Snow Sample Contaminant Levels

A graphical representation of the results of snow sample testing of oil & grease and iron during the melting season at station CR1 are shown in figure 1. Although variations occur, the contaminants show no strong pattern of either concentrating in the snow during the melt season or of being flushed out of the snow by meltwater and rain.

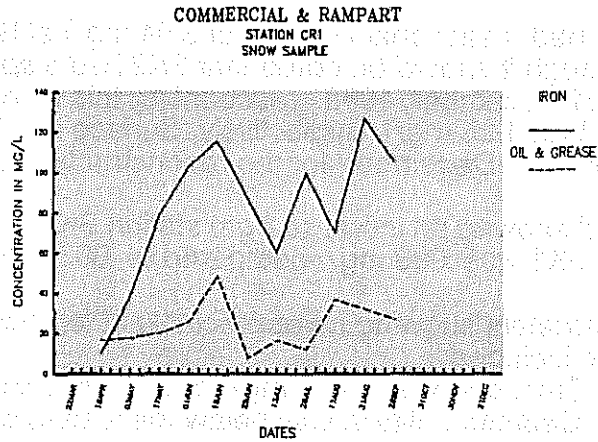


Figure 1.

Water Samples

Surface waters in the vicinity of the snow disposal sites were also sampled. Goose Lake was sampled at the western side by an outflow culvert, about 50 yards from the Goose Lake disposal site. No sampling results could be attributed to the presence of the disposal site, since the surface water flow was in the westerly direction. Total suspended solids, oil and grease, and zinc levels all showed very high readings in the mid July sample only, although the snow had all melted by the 1st of July. Table 4 summarizes the stream sample results.

Another location for monitoring the impact of a snow disposal site on local surface waters was the 15th and Sitka St. site (see map, pg. app-1). Chester Creek flows on the west side of Sitka St., while the snow disposal site is directly to the east of Sitka St. Prior to the reconstruction of this site in the summer of 1988, two storm drain outfalls fed meltwater from this site directly into Chester Creek. Sampling was done upstream from the storm drain outfalls where the creek emerged from a culvert under 15th avenue and Merrill Field (station SK1). A second sampling station was established approximately 300 yards downstream of the outfalls (station SK2).

A seasonal comparison of oil & grease levels at station SK1 vs. station SK2 is shown in figure 2. The seasonal mean at station SK1 was 1.29 mg/l vs. 1.82 mg/l at station SK2. Snow melt contributing to the surface runoff into the creek via the storm drains ended by mid June.

Total suspended solids had a mean value of 7.67 mg/l at station SK1 vs. 21.86 mg/l at station SK2.

Sodium had a seasonal average of 14.9 mg/l at station SK1 vs. a mean of 16.9 at SK2.

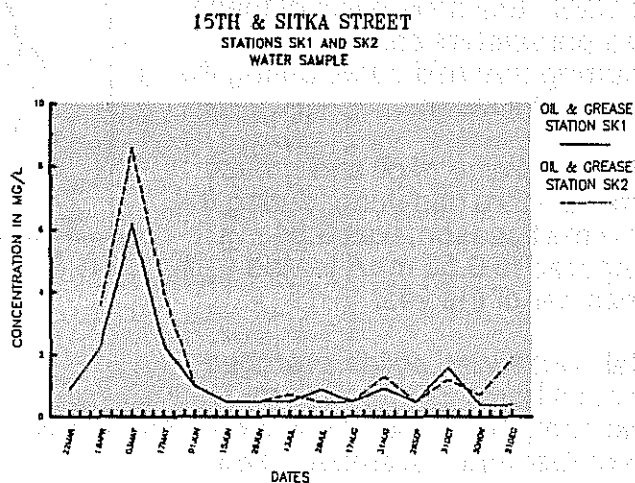


Figure 2.

Iron had a seasonal average of 3.94 mg/l at SK1 compared to 4.73 mg/l at SK2, although it should be noted that SK2 had a spike reading of 11.0 mg/l recorded on 21 December, 1988. Discounting this sampling date, the averages were 3.97 mg/l vs. 4.25 mg/l, respectively. Surface discharge from the disposal area ended prior to 21 December, for the reconstruction of the site had been completed.

Lead showed an increase from station SK1 to station SK2. The average at station SK1 was .003 mg/l compared to .005 mg/l measured at station SK2.

Parameters showing no appreciable difference between the two monitoring stations included:

- (1). Zinc, showing 0.046 mg/l at SK1 vs 0.050 mg/l at SK2.
- (2). Cadmium, which was below the testing detection limit of 0.005 mg/l at both stations.
- (3). Copper, which averaged 0.019 mg/l at both stations.
- (4). Calcium, which averaged 41.2 mg/l at SK1 and 42.1 mg/l at SK2.
- (5). pH, which averaged 6.82 at both stations.
- (6). Hardness, which averaged 178.8 at SK1 vs. 176.6 at SK2.
- (7). Chlorides, which averaged 44.3 mg/l at SK1 vs 44.6 mg/l at SK2.
- (8). Total dissolved solids which averaged 282.4 mg/l at SK1 vs. 285.07 mg/l at SK2.
- (9). Turbidity, which averaged the same amount at both stations of 16 NTU.
- (10). Nitrates, which decreased from 1.84 mg/l at SK1 to 1.6 mg/l at SK2.

Three surface water stations were monitored in the vicinity of the disposal site at Tudor Road and Baxter Drive. Only one of these stations, TR0, applies to this study. Station TR0 is located on the southern side of the disposal site, on a small branch of Chester Creek.

Some contaminant levels were found to be elevated at this site. Figure 3 shows the levels at station TR0 of total suspended solids, sodium and oil & grease throughout 1988. Elevated levels of these three parameters can be seen occurring thru mid June, during the snow melt season.

The mean value of oil and grease from April thru mid June, during the snow melt season, was 3.24 mg/l compared to 0.56 mg/l for the remainder of the year.

Total suspended solids averaged 6.0 mg/l from April through mid June compared to 4.1 mg/l for the months of July, August, September, October, November and December, 1988.

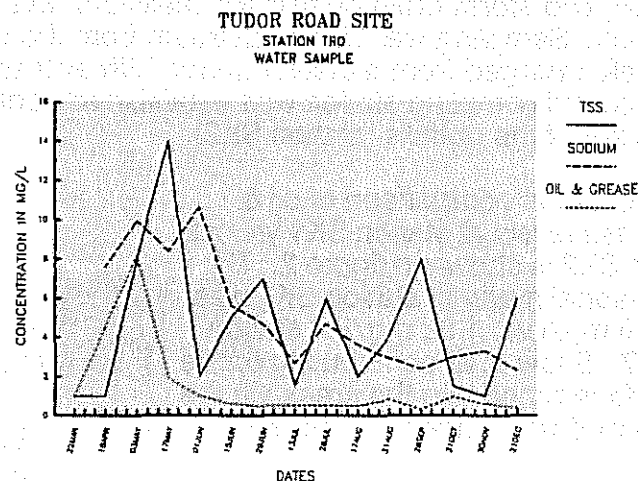


Figure 3.

Sodium levels averaged 19.52 mg/l for the April through mid June period compared to an average of 9.52 mg/l for the remainder of the year. Total dissolved solids also varied over the testing period. The average through mid June was 112.4 mg/l compared to 85.1 mg/l for the remaining months of the study.

Chlorides also showed the same pattern, with 19.0 mg/l as the average for the period through mid June compared to 4.56 mg/l as the average for the remainder of the testing period. Calcium varied from 19.52 mg/l to 12.0 mg/l during the same time spans, and conductivity varied from 105.0 micromhos to 70.56 micromhos during these time periods. The remaining parameters showed no appreciable variation in levels during the course of the testing period.

| <u>Parameter</u>       | <u>Mean Value at Station SK1</u>     | <u>Mean Value at Station SK2</u>     | <u>Statistically Sig. t-Test: <math>\alpha = .05</math></u> |
|------------------------|--------------------------------------|--------------------------------------|---|
| Oil & Grease           | 1.29 mg/l                            | 1.82 mg/l                            | No  |
| Total Suspended Solids | 7.67 mg/l                            | 21.86 mg/l                           | Yes   |
| Sodium                 | 14.9 mg/l                            | 16.9 mg/l                            | No  |
| Iron                   | 3.94 mg/l                            | 4.73 mg/l                            | Yes   |
| Lead                   | .003 mg/l                            | .005 mg/l                            | Yes   |
| <u>Parameter</u>       | <u>Average at TR0 Mar. thru June</u> | <u>Average at TR0 June thru Dec.</u> | <u>Statistically Sig. t-Test: <math>\alpha = .05</math></u> |
| Oil & Grease           | 3.24 mg/l                            | 0.56 mg/l                            | Yes   |
| Total Suspended Solids | 6.0 mg/l                             | 4.1 mg/l                             | Yes   |
| Sodium                 | 19.52 mg/l                           | 9.52 mg/l                            | Yes   |
| Total Dissolved Solids | 112.4 mg/l                           | 85.1 mg/l                            | Yes   |
| Chlorides              | 19.0 mg/l                            | 4.56 mg/l                            | Yes   |
| Calcium                | 19.52 mg/l                           | 12.0 mg/l                            | Yes   |
| Conductivity           | 105.0 umhos                          | 70.56 umho                           | Yes   |

Table 4. Contaminant Level Comparisons

## Soil Samples

Soil samples were taken at each snow dump site after the snow had melted. Samples were taken at depths of 1, 2, 3, and 4 feet. Parameters which showed no appreciable variation with depth included iron, zinc, cadmium and copper. Parameters which did show substantial variation in the soil samples were: oil & grease, total petroleum hydrocarbons, lead, sodium and calcium. The results for stations CR1, SK1, OM1 and KL1 are shown in table 5.

| STATION | SITE                  | SAMPLE DEPTH | DATE OF SAMPLE | OIL & GREASE | TOTAL PET. HYDROCARBONS | LEAD | SODIUM | CALCIUM |
|---------|-----------------------|--------------|----------------|--------------|-------------------------|------|--------|---------|
| CR1-A   | COMMERCIAL & RAMPART  | 0 - 1        | 08/09/88       | 2060.0       | 1670.0                  | 34.0 | 290.0  | 2600    |
| CR1-B   | COMMERCIAL & RAMPART  | 1 - 2        | 08/09/88       | 477.0        | 371.0                   | 92.0 | 240.0  | 1100    |
| CR1-C   | COMMERCIAL & RAMPART  | 2 - 3        | 09/08/88       | 160.0        | 165.0                   | 24.0 | 410.0  | 1027    |
| CR1-D   | COMMERCIAL & RAMPART  | 3 - 4        | 09/08/88       | 152.0        | 117.0                   | 22.0 | 258.0  | 820     |
| SK1-A   | 15TH & SITKA ST.      | 0 - 1        | 08/09/88       | 3630.0       | 3120.0                  | 88.0 | 300.0  | 36      |
| SK1-B   | 15TH & SITKA ST.      | 1 - 2        | 08/09/88       | 3620.0       | 3320.0                  | 42.0 | 210.0  | 2700    |
| SK1-C   | 15TH & SITKA ST.      | 2 - 3        | 09/08/88       | 86.0         | 42.0                    | 15.0 | 179.0  | 14500   |
| SK1-D   | 15TH & SITKA ST.      | 3 - 4        | 09/08/88       | 108.0        | 702.0                   | 11.0 | 80.0   | 11000   |
| OM1-A   | O'MALLEY & OLD SEWARD | 0 - 1        | 08/09/88       | 74.0         | 42.0                    | 10.0 | 210.0  | 530     |
| OM1-B   | O'MALLEY & OLD SEWARD | 1 - 2        | 08/09/88       | 119.0        | 76.0                    | 10.0 | 110.0  | 330     |
| OM1-C   | O'MALLEY & OLD SEWARD | 2 - 3        | 09/08/88       | 161.0        | 55.5                    | 15.0 | 10.0   | 960     |
| OM1-D   | O'MALLEY & OLD SEWARD | 3 - 4        | 09/08/88       | 85.0         | 42.7                    | 14.0 | 12.0   | 958     |
| KL1-A   | KLOEP FACILITY        | 0 - 1        | 08/09/88       | 264.0        | 199.0                   | 35.0 | 180.0  | 7300    |
| KL1-B   | KLOEP FACILITY        | 1 - 2        | 08/09/88       | 64.0         | 45.0                    | 11.0 | 150.0  | 440     |
| KL1-C   | KLOEP FACILITY        | 2 - 3        | 09/08/88       | 807.0        | 40.4                    | 12.0 | 112.0  | 492     |
| KL1-D   | KLOEP FACILITY        | 3 - 4        | 09/08/88       | 111.0        | 47.4                    | 16.0 | 92.0   | 330     |

Table 5. Soil Sample Results

(all units are mg/dry kg)

## DISCUSSION

### Snow Sample Results

One of the main concerns which led to this study was the potential impacts of deicing compounds on the water quality of surface and ground waters. The compound used in the vast majority of street maintenance deicing operations in Anchorage is sodium chloride (NaCl). Studies in northern tier states have shown a significant impact on surface and groundwater qualities from road deicing compounds. (Environmental Science and Technology, Nov. 1987).

The Municipal and state snow removal operations average approximately 1.3 tons of deicing compounds per lane mile per season in the Anchorage area. These deicing compounds are used primarily to keep sand piles friable and frost free. This compares to an average of 15 - 20 tons per lane mile per season used in the Northeastern United States. (Environmental Science and Technology, Nov. 1987). Levels of sodium and chloride found in Anchorage snow samples were relatively low.

Total dissolved solids in snow samples had a seasonal average of 58.1 mg/l, which was less than the average level in Goose Lake, a groundwater fed lake. Sodium levels varied from 1.63 to 8.50 mg/l, with a mean value of 4.76 mg/l. Although there is no EPA aquatic life fresh water chronic level for sodium, this level is approximately 20% of the maximum level for consumption by persons on a restricted sodium intake diet, and relatively low.

The chloride levels varied from 1.20 mg/l as a low to a high of 3.90 mg/l, with an average during the testing period of 2.36 mg/l. This compares favorably with the fresh water chronic level of 11.00 mg/l, or about 21% of the chronic level. Parameters which were of no concern due to their low testing results included nitrates, fecal coliform and cadmium.

Several metals were found in fairly high concentrations in the snow samples and were probably caused by the high vehicular traffic on the streets from which the snow was removed. Lead levels in the snow samples over the season averaged more than 100 times the fresh water chronic level. Iron averaged approximately 60 times the fresh water chronic level. Zinc and copper both averaged more than 10 times the fresh water chronic levels.

The two parameters showing the highest levels in the snow samples were oil & grease and total suspended solids. Oil & grease averaged 17.4 mg/l while total suspended solids averaged 1,114.8 mg/l. The oil & grease source is probably vehicular traffic and snow removal equipment, while the total suspended solids were the result of sanding operations over the winter season.

Seasonal loading rates for oil & grease and suspended solids were calculated as follows:

Total snow collections times the seasonal average for oil & grease content at these disposal sites yields 3150 gallons of oil & grease accumulated in all 8 snow disposal sites. This equates to approximately 400 gallons per site per year. These results were calculated as follows: 2 million cubic yards as snow times the seasonal average of 17.4 mg/l of oil & grease as water times an assumed 0.40 cubic yards of water per cubic yard of snow.

The total accumulation of solids in all 8 snow disposal sites combined equaled (2 million cubic yards X 1114.8 mg/l) 750 tons or nearly 440 cubic yards. This would equate to 94 tons (55 cubic yards) per disposal site per year.

### Water Sample Results

Table 4 summarizes the water sample results for the stream stations SK1, SK2, and TR0 that were monitored in this investigation. Station TR0, located on the south side of the Tudor Road disposal site, illustrates the impacts snow disposal activities can have on stream water quality. This disposal site was redesigned and reconstructed in 1987 to prevent runoff from entering a branch of the South Fork of Chester Creek. The redesign included berm placement around the perimeter of the site to prevent surface runoff of the meltwater.

During the winter of 1987-88, the disposal site was filled to capacity with snow, and some operators pushed snow outside of the bermed containment area. This allowed some overflow, mostly to the south and west of the site, which melted and flowed into the small branch of Chester Creek located just south of the disposal site.

During the period in which the snow overflow melted (thru mid June) oil & grease levels were found to be almost 6 times higher at station TR0 than during the remainder of the testing period (3.24 mg/l vs. 0.56 mg/l). Total suspended solids were 1.5 times higher, sodium levels were more than doubled, and chloride levels were up more than 400%. Calcium and total dissolved solids also showed increased levels during the melt period.

The North Fork of Chester Creek runs parallel to Sitka Street at the 15th & Sitka snow disposal site. Two testing stations were located along this stream. The first was located at the outfall of a culvert coming from under 15th avenue and Merrill Field, prior to any influence from the disposal site. The quality of the water at this testing station (station SK1) was poor due to a large storm drain network in the vicinity of Merrill Field draining into Chester Creek. This stream also runs under the old Merrill Field landfill, so the possibility of leachate from the landfill entering the stream is high. The second station, SK2, was located approximately 300 yards downstream of the snow disposal site and was located to monitor the impact of snow melt runoff from the disposal site.

Some parameter levels in the water at station SK1 were higher than those encountered in snow samples taken from snow disposal sites for the reasons discussed above. Chlorides averaged 44.3 mg/l at SK1 vs. a seasonal mean of 2.36 mg/l in the snow samples taken. Calcium averaged 41.2 mg/l at station SK1 compared to the seasonal average of 7.58 mg/l in the snow samples taken.

Parameters which did show dramatic increases between stations SK1 and SK2 were oil & grease, with levels jumping approximately 50%, and total suspended solids, with levels rising approximately 300%. Also showing increased levels were iron, with a jump of 25%, and lead which had an increase of 67%. Sodium remained relatively constant in the 10-18 mg/l range at both stations.

### Soil Sample Results

Soil sampling was done on August 9, 1988 and September 8, 1988. The first sampling was taken at depths of 1 and 2 feet at each of the snow disposal sites. After review of the initial results, a second sampling round was conducted, with depths of 3 and 4 feet sampled at separate locations within the disposal sites. These results can be seen in table 5. This method caused some difficulties in comparing results, for the identical boring sites could not be identified for the second sampling round. Another weakness of the soil sampling efforts was the inability to obtain any type of baseline samples prior to our sampling program. Some disposal sites were located on native soils, while others were located on fills hauled in from locations throughout the municipality. Disposal sites have also been used to dispose of the tailings from street sweeping operations. These variations in the sites made it difficult to establish levels in the parameters that were directly attributable to the snow disposal operations.

Oil & grease and total petroleum hydrocarbons were found at elevated levels throughout the soil samples. Levels ranged from over 3500 mg/dry kg near the surface to a 100-150 mg/dry kg range at a depth of 4 feet. These results showed that although a filtering action is taking place as the meltwater percolates down through the soil, high levels were still evident at depths of up to 4 feet. This points to the need for further soil sampling at greater depths and groundwater testing to determine impacts on groundwater quality.

Lead, sodium and calcium all gave inconsistent results in the soil samples, with a general trend of decreasing levels with greater depth. With no baseline sampling available, these data were difficult to interpret. Parameters showing no appreciable variation with depth included iron, zinc, cadmium and copper.

## CONCLUSIONS

Snow melt runoff from snow disposal sites has the potential to adversely impact water quality and associated aquatic life. High levels of oil & grease, suspended solids, and heavy metals were the major contaminants found in our analysis of snow melt runoff. These contaminants were the major constituents we found at levels which exceeded the EPA freshwater chronic limits for aquatic life protection.

Our monitoring results accentuate the importance of properly designing snow disposal sites to effectively prevent snow melt runoff from entering adjacent surface water bodies. The retention of snow melt runoff within the disposal site and in detention basins enhances percolation, evaporation, and hopefully minimizes potential adverse impacts on adjacent surface water bodies.

The soil monitoring results were somewhat inconclusive due to the following:



\* Lack of information on naturally occurring levels of constituents present before the site was used as a snow disposal site.

\* Flaws in sampling methodology which made comparisons between different sampling efforts difficult.

However, the soil monitoring results indicated high levels of oil & grease, petroleum hydrocarbons, lead, sodium, and calcium. Levels of contaminants decreased with depth in most cases and illustrated the natural filtering ability of the underlying soils. Further monitoring is needed to determine the potential impacts on groundwater quality.

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## CURRENT AND FUTURE WATER QUALITY STANDARDS FOR ARSENIC

Jacqueline D. LaPerriere<sup>1</sup>

### ABSTRACT

The two major valence states of arsenic in water (+3 and +5) are assumed to have different toxicities to aquatic organisms. The U. S. Environmental Protection Agency (EPA) has set aquatic life criteria for arsenic based on total recoverable arsenite +3, and is collecting data for setting criteria for arsenate +5. Meanwhile there are no standard analytical methods for measuring these chemical species. Also, the current wording of aquatic life criteria sets specific concentrations, as averages over particular periods of time, that are not to be exceeded "more than once every three years on the average." This wording yields standards that are unenforceable, because adequate data sets are not available. Enforcement agents must have multiple measurements during the specific time periods (1 hour or 4 days) to conduct average calculations, and the collection of data must have been sustained for more than 3 years before a violation can be shown. The criterion for ambient drinking water is stated as zero because arsenic is a suspected carcinogen. Cancer risks are associated with specific concentrations of nanogram per liter (parts per trillion); yet, the maximum contaminant level for finished drinking water is set at 50  $\mu\text{g/L}$  (parts per billion). This value is apparently believed to be a practical limit, since 63 municipal supplies nationwide have reported arsenic concentrations higher than 50  $\mu\text{g/L}$ . By EPA's own calculations this concentration could increase cancer risk by slightly over one in a hundred. The State of Alaska should set individual water quality standards for each toxicant, using the concentrations (or some lower fraction) that EPA states in the criteria. The wording of many of the toxicant criteria (especially for most of the heavy metals) should not be used

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in State standards, because the data for enforcement are not and will not be available. Concentrations should therefore be stated as numbers never to be exceeded, so that the standards will be enforceable.

## INTRODUCTION

The setting of water quality criteria and resultant standards for arsenic is made difficult by the different toxicity of its several valence states. Since elemental arsenic has five electrons in its outer shell, ions with +3, +5, and -3 valences are possible--though the -3 state, which occurs in the gas arsine ( $\text{AsH}_3$ ), is unstable and therefore probably not found in natural waters. Elemental arsenic, with a valence of 0, is non-toxic, and also rare and insoluble in water. The +3 ion, or arsenite, occurs in waters where reducing conditions are present (e.g., near decomposing organic sediments). The +5 valence, or arsenate, is believed to be the most common form in well oxygenated natural waters because it is thermodynamically more stable than arsenite (+3) (EPA, 1980).

Arsenic is common in the environment, occurring in minerals combined with sulfur and oxygen from which it can be weathered and freed to enter natural waters. It forms bonds easily with carbon and sulfur in organic compounds. Arsenite (+3) is known to react with the sulfhydryl groups in proteins and therefore can exert its toxicity by poisoning enzymes. Arsenate (+5) is believed to exert its toxicity by interfering with oxidative phosphorylation (EPA, 1984). The Canadian Council of Resource and Environment Ministers (CCREM) has reported methylation of arsenic by bacterial action as well as the formation of various other arsenical organics some of which are very toxic (CCREM, 1987).

### Human Toxicity

Humans who ingest an acutely toxic dose of arsenic may die of cardiac failure. However, victims who ingest chronic doses may "give the appearance of a progressive chronic disease state" (EPA, 1980). This insidious effect of arsenic in subacute doses has made it the poison of choice in numerous mystery novels and in the classic play, "Arsenic and Old Lace."

Arsenicals were used in the past as treatment for asthma (Fowler's Solution) and syphilis (various organic arsenic compounds). Case studies of people who were so treated for long periods show that many developed skin cancers some 6 to 18 years later (EPA, 1980). These studies have also shown an increased incidence of lung and systemic cancers among these patients over time.

Arsenic is also a suspected environmental carcinogen, based on studies of human populations exposed to arsenic. It has been associated with increased incidences of skin and lung cancers, and may affect cancer incidences at other sites. There is good evidence of a dose relation to skin cancer among populations in Taiwan exposed through well water. Incidences of skin cancer rose with age and with the concentration of arsenic in the drinking water (Tseng, 1977).

Recently, the idea has been put forward by the EPA that arsenic in trace amounts may be an essential nutrient to humans (Federal Register, 1985). This possibility is based on the reasoning that "Arsenic at low concentrations in the diet enhances some parameters of growth and development in animals..." However, for any chemical to be considered an essential nutrient, it is required that studies be presented that show a specific disease associated with a lack of that essential nutrient in the diet (including drinking water). It may be almost impossible to gather evidence of such a deficiency disease in humans because of the common occurrence of arsenic in ordinary foods (particularly seafoods).

#### Aquatic Organism Toxicity

Toxicity to aquatic organisms has long been assumed to be higher for arsenite (+3 valence) than for arsenate (+5). Currently, few data are available on the toxicity of arsenate; therefore, not enough studies exist on which an aquatic life water quality criterion for arsenate can be based. Also too few data are available to enable the setting of water quality standards for any of the organic arsenic compounds (EPA, 1986).

Currently, when arsenic is analyzed in water samples, standard methods (EPA, 1983) are available for measuring total arsenic, total recoverable arsenic, and dissolved arsenic, but

valence states are not separable by these methods. Total arsenic is measured after a vigorous treatment that includes refluxing with boiling strong acids ( $\text{HNO}_3$  and  $\text{HCl}$ ) that essentially dissolve most solids in the water. The remaining silicates and other insolubles are filtered off before analysis. Total recoverable arsenic is analyzed after mild digestion with a small amount of  $\text{HCl}$  at a temperature below boiling. The dissolved form is separated from sediments at the time of sampling by immediately filtering. All forms are preserved by adding redistilled nitric acid. Dissolved arsenic is then measured in the laboratory without additional treatment. Arsenic is usually measured by atomic adsorption spectrophotometry in which a graphite furnace is used for atomization. There are no standard methods for quantifying the several valence states of arsenic separately.

### CURRENT CRITERIA AND STANDARDS

The most recent federal criteria (EPA, 1986) for ambient water concentrations (raw water supply) "should be zero based on the non-threshold assumption for this chemical." The "non-threshold assumption" is made because arsenic is a suspected carcinogen.

Realizing that zero concentration may be impossible, EPA has published a criterion that might lead to an additional cancer risk of 1 in a million for ambient water of 2.2 ng/L (0.0022  $\mu\text{g/L}$ ) arsenic; and for the same risk level for ambient water, which would not be used for drinking water supply but from which aquatic organisms might be consumed the criterion is 17.5 ng/L (0.0175  $\mu\text{g/L}$ ; Table 1). Finished drinking water has a maximum contaminant level (MCL) of 50  $\mu\text{g/L}$ , as total arsenic. Extending the rationale behind Table 1, 50  $\mu\text{g/L}$  would be associated with an additional cancer risk of about one in a hundred.

The federal water quality criterion for protection of aquatic life from arsenic toxicity is based on "total recoverable trivalent inorganic arsenic." There is no standard method for the measurement of trivalent inorganic arsenic, and recent work at the University of Alaska Fairbanks shows that the separation of the trivalent and pentavalent forms is difficult unless the water is of low ionic strength (Reed and Stolzberg, 1987).

The current federal criterion (EPA, 1986) for the protection of aquatic life states that "freshwater organisms and their uses should not be affected unacceptably if the 4-day average concentration of arsenic (III) does not exceed 190  $\mu\text{g/L}$  more than once every three years on the average and if the 1-hour average concentration does not exceed 360  $\mu\text{g/L}$  more than once every three years on the average." The total recoverable method is recommended in applying the criterion. However, EPA has proposed requiring that many metals be measured in the "acid soluble" form, but has not yet fully developed standard methods for that form.

The State of Alaska water quality standards are the published State criteria for designated uses as amended. Since arsenic is a water quality variable for which no State criteria and standards are expressed, the EPA criteria (EPA, 1986) or the Alaska Drinking Water Standards (Title 18, Alaska Administrative Code, Chapter 80) whichever are lower, stand as the criteria, since arsenic is a toxic substance. All toxic substances are treated similarly in the State of Alaska Water Quality Standards.

The extremely conservative stance of EPA for the ambient drinking water supply criterion comes from standard practice for setting criteria for toxicants known or suspected to be carcinogens. In the case of arsenic and drinking water, the data base on increased cancer incidences is from studies of populations in Argentina and Taiwan. Studies of populations exposed to arsenic in drinking water, in Lane County, Oregon, and Fairbanks, Alaska, have shown no increased incidences of cancer (EPA, 1980). This difference may be because the exposed U. S. populations are so small, or for various other reasons such as the possibility of exposure to different forms of arsenic among these different populations (EPA, 1980). There are also no supporting data from animal studies showing a clear relation between dietary and drinking water exposure to arsenic and increased incidence of cancer. Nevertheless, EPA has decided to be very conservative and treat arsenic as a suspected, if not a known, carcinogen.

The finished drinking water maximum contaminant level of 50  $\mu\text{g/L}$  is apparently a compromise between the polar stances that arsenic is a carcinogen and (at the other extreme) that it is an essential nutrient. This is probably also a practical limit

considering that 63 community water supplies in the U.S. have reported arsenic concentrations greater than 50 µg/L (Federal Register, 1985).

Canadians have stated the same guideline (0.05 mg/L) for both finished drinking water and the protection of aquatic life. They point out that water treatment methods are available for reducing considerably higher concentrations of arsenic to this level (CCREM, 1987) and therefore do not state a limit that is not to be exceeded in the ambient water supply.

The lower guideline for the protection of aquatic life in Canada than in the U.S. comes from a difference in philosophy. Canadian guidelines, for the most part "are set at such values as to protect all forms of aquatic life and...to protect all life stages during indefinite exposure to the water" (CCREM, 1987). U.S. criteria for the protection of aquatic life are somewhat less conservative, being for toxicants essentially a mean value for the four most sensitive species tested.

The Canadian guidelines also do not make a reference to measurement of a particular valence state and form of arsenic. As regards valence state, they point out that "The trivalent and pentavalent forms of arsenic appear to be similar in toxic potential" (CCREM, 1987). The EPA has not set a pentavalent criterion because data are insufficient to meet its guidelines for setting criteria. Recent data from the National Fisheries Contaminant Research Center-Columbia of the U.S. Fish and Wildlife Service show similar acute toxicity values for arsenite (+3) and arsenate (+5) to Arctic grayling Thymallus arcticus; rainbow trout Oncorhynchus mykiss (formerly Salmo gairdneri); and coho salmon, O. kisutch (S. Hamilton, U. S. Fish and Wildlife Service, Yankton, SD, 1987. Personal communication).

The Canadian guideline is also based on "The analysis of the total element, or substance in an unfiltered sample...when appropriate." Canadians believe, "It is protective of the environment because it measures the total quantity of the substance that is present" (CCREM, 1987).

The averaging periods and recurrence intervals that EPA has established for aquatic life criteria of many toxic substances,



including arsenic, were rejected by the Canadians in setting their recent guidelines. They pointed out that there is a "lack of a generally accepted and scientifically sound basis for the duration of the averaging period." Their decision is based on information (EPA, 1986) that concentrations of toxicants that vary over time may be more toxic than a steady dose at the average concentration. The Canadians also remark that Canada does not monitor frequently enough to provide data from which 24-hour or 4-day average values can be compared to guidelines (CCREM, 1987).

The recurrence interval of 3 years is justified by EPA (Federal Register, 1985) by stating that "most aquatic ecosystems can probably recover from most excess concentrations in about 3 years." This was developed from the idea that the "average fish" first reproduces at 3 years of age (Lowell Keup, Standards and Criteria Branch, USEPA, May 1987, Personal communication). The Canadian guideline document correctly notes that a 3 year recurrence interval "could place a fish population in a perpetual state of recovery rather than allow it to recover."

#### FUTURE CRITERIA AND STANDARDS

As more toxicity data are developed, we can expect EPA criteria to be refined. If data show that the toxicity exerted by various forms is sufficiently different, separate criteria may be developed. Publishing separate criteria should also depend on development of standard analytical methods to measure each form.

The public would benefit if the State of Alaska develops criteria and standards (for all toxicants, not just arsenic) that are based on numbers "not to be exceeded" calculated from the published federal criteria with an appropriately conservative safety factor. Since Alaska is not routinely monitoring any waters for any toxicants, it is impossible to enforce criteria that are set as average values over one or more time intervals.

The EPA should return to criteria based on a number "not to be exceeded at any time" to prevent acute effects, and a 24-hour average number set to prevent chronic effects. Currently, it is difficult to find a set of data from any locality that would be extensive enough to enforce a standard that did not use this older

wording. To avoid stressful fluctuation, the amount of variance that would be allowed in the 24-hour average should be specified. The current wording, that the acute and chronic concentrations can be exceeded once every three years "on the average" is especially troublesome, and should not be used because it is obvious that the data set must be longer than 3 years to demonstrate exceedance of the criteria.

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TABLE 1. Ambient water quality criteria for arsenic (USEPA, 1986).

| Increased cancer risk             | $10^{-7}$          | $10^{-6}$ | $10^{-5}$ |
|-----------------------------------|--------------------|-----------|-----------|
| Exposure Assumptions (per day)    | Concentration (ng) |           |           |
| 2 L drinking water and 6.5 g fish | 0.22               | 2.2       | 22        |
| 6.5 g fish and shellfish only     | 1.75               | 17.5      | 175       |

**SURFACE/GROUND WATER INTERACTIONS**

## MODELING THE THERMAL REGIME OF AN ARCTIC WATERSHED

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In cold regions, the hydrologic and thermal regimes interact to such an extent that neither can be fully understood without considering the other. The consequences of a man made or environmentally induced alteration in the thermal regime can have dramatic and perhaps dire effects on the hydrologic regime and vice versa. In the Arctic, the depth of the active layer affects the amount of soil moisture storage, the magnitude of base flow, even the shape of the recession curve. At a time when much consideration is being given to the possible effects of climatic warming, it is important to remember the ramifications on the hydrologic regime.

In order to predict the dynamics of the active layer as a result of climatic warming, we utilized a mathematical model to calculate the heat transfer in the top 10 m of soil. This program is a two dimensional conduction model with phase change. Solution of the transient heat conduction is by finite element method. This model is capable of solving unsteady state problems with thermal properties which vary with time and/or space. Using mean daily temperatures at the ground surface and a time increment of one day, the model was run to simulate the thermal regime. The model accurately simulated soil temperatures for four years of recorded data. The model was then run to predict the response of the thermal regime given an increasing annual temperature over a period of 50 years.

## MODELING THE HYDROLOGIC REGIME OF AN ARCTIC WATERSHED

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Watersheds in the high Arctic conform to the hydrologic principals accepted for more temperate regions, however extreme climatologic differences cause marked differences in the hydrologic regime. Arctic basins are characterized by snow accumulation throughout the winter and a brief, yet intense snowmelt event in late May or early June. Strong thermal gradients in the active layer tend to desiccate the surface organic soils during the winter, in the spring these soils will absorb up to 1.5 cm of water before downslope runoff occurs. Precipitation is usually light during the winter and early summer, increasing to the maxima in late July or August. Continuous ice-rich permafrost acts as a barrier to subsurface migration of soil water. Evaporation is low during the winter months, but can exceed 2 mm per day during the summer.

All of these important aspects of the hydrologic cycle were modeled using the Swedish HBV model. HBV is a conceptual model design for continuous calculation of runoff. Water from snowmelt or rainfall is partitioned into evaporation, infiltration, soil storage and runoff. The test basin was Imnavait Creek, a 2.2 km<sup>2</sup> zero-order watershed on the North Slope of Alaska. Meteorologic and hydrologic data collected during 1985, 1986, 1987 and 1988 were used to calibrate and verify the model.

HBV has been successfully applied to model streamflow in Sweden, Norway and in subarctic Alaska, but had never been used to model a basin completely underlain by permafrost. The model closely predicted soil moisture levels, evaporation, snow accumulation and ablation. It satisfactorily predicted runoff volumes, however more calibration data is required to improve the estimates of peak flows. Streamflow from rainfall events was modeled very well, however accuracy lapsed when the rainfall was mixed with snow.

## DEVELOPING GROUND WATER FED SPAWNING AND REARING HABITAT FOR ANADROMOUS FISH ON THE CHUGACH NATIONAL FOREST

By Dave Blanchet, Hydrologist, Chugach National Forest<sup>1/</sup>

### ABSTRACT

Since the early 1980's, the Chugach National Forest has initiated a series of salmon habitat improvement projects using ground water as a flow source. Projects have involved the construction of channels and ponds for salmon spawning and rearing. Several different salmon species have been targeted. Much of this fisheries enhancement work has been done in coordination and cooperation with gravel extraction projects on the Forest.

Ground water fed channels and ponds on the Chugach have been modeled in large part after salmon habitat projects first initiated in British Columbia. The Chugach projects, however, have a number of features unique to their own location. Most of the Chugach projects are developed in permeable glacial outwash deposits with a shallow ground water table. Chugach project sites have colder water temperatures and lower salmon productivity rates than found in British Columbia. Rearing pond productivities, however, are showing improvement as the newly developed ponds increase in biologic activity and diversity. The Chugach is currently planning for development of several ground water fed spawning channels in Prince William Sound. Channel benefits will be based on fisheries merits alone, with no coordinated gravel extraction planned.

### INTRODUCTION

Alaska offers a wide spectrum of stream and lake habitats that salmon use for both spawning and rearing. By replicating the most productive of these habitats while using new flow sources, salmon productivity on certain drainages can be significantly increased. Tapping into the shallow ground water table in permeable gravel deposits has been demonstrated as an excellent technique for developing new flow sources (Blanchet, 1984.)

This paper examines habitat needs for various salmon species, and shows how these needs can be applied to design and construction of ground water fed spawning and rearing projects. The paper also looks at four Chugach National Forest projects for enhancement of salmon spawning and/or rearing habitat. The projects and their target salmon species and habitat are listed in Table I.

All four projects have been designed to increase annual returns of adult salmon for sport and/or commercial fisheries. The projects differ from one another in location, design, and target species, however, all use ground water as a flow source. These ground water fed systems display moderated stream flows and relatively steady water temperatures, both of which are desirable conditions for salmon spawning and rearing.

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Table I

| Name/Location                        | Target species                | Target Habitat          | Startup Date |
|--------------------------------------|-------------------------------|-------------------------|--------------|
| Williwaw Ponds<br>Portage Valley     | Cohos, Chums,<br>and Sockeyes | Spawning and<br>Rearing | Spring, 1986 |
| Lyon Creek Ponds<br>Turnagain Pass   | Cohos                         | Rearing                 | Summer, 1986 |
| Mile 25 Channel<br>Copper R. Highway | Cohos<br>(Sockeyes)           | Spawning                | Summer, 1987 |
| Pigot Creek<br>Pigot Bay, PWS        | Pinks & Chums                 | Spawning                | 1990         |

The diversity of the four projects is useful for displaying the wide range of considerations used in designing new habitat for salmon. Project approaches for developing spawning habitat differ considerably from those for rearing habitat. Developed rearing ponds can have the added benefit of creating new habitat for waterfowl and shorebirds.

Critical for new habitat projects is to first gain a clear understanding of existing spawning and rearing habitat use within the selected drainage. This fisheries information can be used to maximize compatibility between newly developed habitat and that already in use. Designing projects in this manner helps greatly in enhancing overall productivity.

### SALMON HABITAT NEEDS FOR SPAWNING AND REARING

Fisheries enhancement projects on the Chugach National Forest have been designed to increase returns of adult salmon. These projects increase salmon availability to commercial and/or sport fishermen, and can enhance salmon viewing opportunities for the public. Several of the Forest habitat improvement projects have been initiated to mitigate impacts from land disturbing activities (e.g. gravel extraction). Other projects, however, have been undertaken strictly for their fisheries benefits.

All five species of Pacific salmon frequent streams of the Forest including: coho, sockeye, chinook, pink and chum salmon. Each species tends to use slightly different segments of lakes and streams for spawning/rearing, however, considerable habitat overlap occurs between some species.

Success of fisheries enhancement projects is dependent in part on whether the pre-existing salmon population has been limited by available spawning habitat or rearing habitat. If a project does not address the limiting habitat factor, then it may have negligible effects on improving salmon productivity (Metzger, 1989). Developing successful ground water fed spawning and rearing projects requires a clear understanding of existing salmon population dynamics. A brief explanation of spawning and rearing needs of the five salmon species is given below.

1. **Pink and chum salmon** often spawn in the lower reaches (including intertidal zones) of small to moderate sized streams. The fry depart from the stream shortly after emerging from the spawning gravels, and use the ocean for rearing. Available rearing habitat is

extremely large, leaving spawning habitat as the key limitation to productivity for these two salmon species. Developing new spawning areas or improving existing areas works to increase adult salmon returns. Pinks return to their stream of origin to spawn and die on a two year cycle. For chums the cycle is three to five years.

2. **Coho and sockeye salmon** generally spawn in gravels on small to moderate sized streams, or in areas of ground water upwelling. Coho fry reside in sloughs, ponds, or quiet stream segments for one to two years after emerging from spawning gravels. Sockeye fry generally reside in lakes or ponds. On the Forest, freshwater residence time for both species is often two years. Residence times are dependent in part on food availability within a stream/lake system, water temperatures, and minimum size requirement for salmon to outmigrate from the system. These factors vary by stream, and therefore, residence times vary as well. By spring or summer of the second year, coho and sockeye smolts migrate to the ocean and remain there for one to two years before returning to their original stream to spawn. Their full life cycle is usually four to five years.

Because small numbers of adult cohos can produce enough fry to utilize large amounts of rearing habit, coho populations *tend* to be limited by rearing habitat. Sockeyes, by contrast, *tend* to be limited by spawning habitat, since they are often using large lakes or ponds for rearing. Depending on existing habitat limitations, enhancement projects for cohos and/or sockeyes can target rearing habitat, spawning habitat, or *both*. Successful enhancement projects can show excellent benefits due to the high sport and commercial values of both coho and sockeye salmon.

3. **Chinook salmon** generally spawn in coarse substrate in swift river segments. Fry tend to rear in sloughs, ponds, and quiet stream segments. Chinook tend to be limited by rearing habitat and this is usually the focus of habitat improvement projects.

### DEVELOPING GROUND WATER FED SPAWNING CHANNELS

Ground water fed spawning channels can potentially be developed for any of the five salmon species. Channels developed for pink and chum salmon have a high probability for success since these species are not rearing limited. Spawning channels may also be developed for coho, sockeye, and chinook salmon, but should be attempted only if these species are not limited by rearing habitat, or if additional rearing habitat is also developed. Chinook salmon's preference of high volume and velocity streams for spawning is generally incompatible with ground water fed channels.

After selecting one or more target species, a series of hydrologic and fisheries criteria need to be evaluated. These include the following:

1. **Flow Volumes.** Ground water fed spawning channels have been successfully developed by excavating in permeable gravel deposits with a shallow ground water table. The flow volume derived through excavation is of primary interest to spawning channel design. Winter low flows are of particular concern since this is usually the time of greatest stress on salmon eggs incubated within the stream gravels.

Ground water flow volumes vary considerably based on location. For spawning channel sites selected within small drainage basins, minimum ground water flows are restricted to something less than the winter baseflows out of the basin. For larger drainages it is sometimes possible to tap into mainstem streamflows, resulting in relatively large ground

water volumes, even during winter months. The Williwaw Creek spawning channel taps water from adjacent Portage Creek. This channel has winter low flows of 5 to 10 CFS, and peak flows of around 50 CFS during summer and fall rainstorms.

2. **Channel Design - Width, Depth, Gradient, and Flow Velocity.** Based on available ground water flow volumes, various spawning channel designs can be considered. Channels need to be designed with enough depth to allow for fish passage and desirable spawning conditions (during spawning periods.) Channels must also provide sufficient water depth and velocity to provide good intergravel flow to eggs and/or fry during low flow periods. For a given flow volume, channel depth can be increased by decreasing channel width or gradient.

Decreasing channel width has the effect of increasing both stream depth and flow velocity. It decreases stream area available for spawning (although this may not be "lost" area if portions of the channel go dry during low flow periods.)

Stream depth can also be increased by lowering the stream gradient. Gradient can be reduced by increasing channel length (i.e. increasing channel sinuosity) or by installing flow control structures (weirs) within the channel. Both techniques have the effect of increasing available spawning area while slowing stream velocity.

Stream velocities can be lowered to the extent that "desirable" habitat will no longer be selected by spawning salmon. Critical base velocities vary by salmon species. Low velocity streams are also more likely to freeze to the bottom during the winter, and have lower dissolved oxygen levels. Both these features can be very detrimental to the survival of over-wintering salmon eggs within the stream gravels.

3. **Channel substrate.** Ideal channel substrate for a spawning channel is based on both the targeted salmon species and channel flow velocities. Smaller salmon use smaller sized substrates for making their spawning redds. Individual salmon, however, can make use of larger substrates with an assist from increased flow velocities. Areas selected for development of ground water fed spawning channels are often composed of coarse, permeable gravels that are ideal for spawning substrate as they are. Sometimes, however, existing substrates are inappropriate for the targeted salmon species. Substrate can be changed by either screening the existing gravels to remove undesirable size fractions, or by bringing in properly sized gravels.

4. **Water Quality.** Major water quality concerns for spawning channel development include: suspended sediment load, turbidity, temperature, and dissolved oxygen. Ground water fed channels developed in permeable gravel deposits generally have desirable water quality for salmon egg survival (i.e. moderated temperatures, low sediment loads, and high dissolved oxygen.) On some occasions, spawning channels can tap turbid waters from a nearby glacial stream through permeable gravel deposits. Turbidity and fine sediments can have detrimental impacts on eggs within the spawning channel. Turbidity can be reduced by increasing the distance separating the ground water fed channel from the main channel. However, increasing separation also reduces flow into the ground water fed channel and consequently reduces available spawning area.

Channels fed by deep ground water flows have little fluctuation in water temperatures and relatively warm winter temperatures. However, ground water fed channels tapping into water from a nearby mainstem stream can take on temperature characteristics of the main stream. Temperature effects from the mainstem are reduced by increasing the

distance between the two channels, but this also reduces the amount of flow in the new spawning channel.

Shallow ground water in permeable gravels deposits generally has a high dissolved oxygen concentration (desirable for fish habitat.) Dissolved oxygen can be slowly depleted on long spawning channels with low velocity flows.

5. **Scour/Channel Dynamics.** Ground water fed channels have very moderated flows. Peak flow conditions do not carry sufficient water to move gravels along the streambed. Scour is usually *not* a problem unless water spills into channel from a surface water source. Berms or dikes can often be constructed to protect these channels from surface flooding.
6. **Bank Protection.** Due to the moderated character of peak flows, bank rip rap is generally not needed. Boulder rip rap is necessary along the shore of the spawning channel to prevent spawning salmon (particularly chums and cohos) from eroding the stream bank. Rip rap also provides some cover for emerging fry. Banks of newly constructed spawning channels should be revegetated to eliminate bank erosion and intrusion of fine sediments into the channel. Planting woody vegetation helps to moderate water temperature fluctuations, and provides additional cover for emerging salmon fry.

Large variability exists for the design and construction of ground water fed spawning channels. An optimum spawning channel design attempts to maximize beneficial spawning habitat features and production capabilities for the targeted salmon species.

#### DEVELOPING GROUND WATER FED REARING PONDS

Ground water fed rearing ponds will focus on only coho, sockeye, and/or chinook salmon. On the Forest, pond development has focused primarily on cohos and to a lesser extent on sockeyes. Rearing ponds should only be developed if: 1) an excess of spawning habitat exists, 2) hatchery stock are used to populate the ponds rather than natural stock, or 3) rearing and spawning habitat are developed together within the same ground water system.

After selecting one or more target salmon species, a series of fish, watershed, and wildlife requirements need to be considered for designing and constructing rearing ponds. Waterfowl and shorebirds can use rearing ponds and the adjacent shorelines for feeding, nesting, brood-rearing, and resting (Lee, 1985.) Pond development considerations include:

1. **Flow Volumes.** Maintaining sustained flow volumes is less important for ponds than for spawning channels. Sufficient flow must be provided to ponds in order to: a) allow for outmigration of smolts and immigration of fry during the spring and summer, b) allow for immigration of adult spawners *if* spawning area exists within or upstream from the ponds, and c) maintain dissolved oxygen concentrations at an acceptable level (particularly during the winter months.)
2. **Pond Depths.** Ponds need to be designed with a combination of deep water zones (12 feet or greater) and shallow zones (ideally around 1.5 to 3 feet.) Deep water is needed to provide overwintering areas for salmon fry and pre-smolts. Deep areas also provide good feeding habitat for diving waterfowl. Shallower ponds can freeze to the extent that the remaining water becomes depleted in dissolved oxygen. High winter mortalities can result to resident fish.

Shallow areas allow for growth of submergent and emergent vegetation. This vegetation provides protective cover for the fry as well as habitat for their macroinvertebrate food source. Shallow areas also provide excellent feeding for dabbling ducks. Shallows have higher summer water temperatures that help to increase fry growth rates. Virtually *all* growth for fry occurs during the summer months.

Rearing ponds on the Forest have been designed with approximately 25% deep areas and the remainder in shallow zones. If two or more deep areas exist within a pond, access between these areas should be available to fish during winter months.

3. **Pond Shape.** Rearing ponds should be designed to maximize the amount of shoreline. Shorelines provides excellent cover and shading for rearing fish, and for waterfowl and shorebirds as well (Lee, 1985.) Shorelines are a source for woody debris, and this debris provides cover and nutrient supplies to the pond. The ideal pond design should then have a strongly undulating shoreline with a series of embayments and/or backwater areas. Islands developed within the pond also increase shoreline and provide excellent nesting habitat for birds. Increasing shoreline length also increases pond excavation and contouring costs, however, it also greatly improves habitat benefits.
4. **Water Quality.** Water quality concerns for rearing ponds are again: suspended sediment load, turbidity, temperature, and dissolved oxygen. Ground water fed ponds generally have desirable water quality for rearing fish. On some occasions, rearing ponds will tap turbid waters from a nearby glacial stream through permeable gravel deposits. This can negatively impact rearing fish in that: a) turbidity reduces light penetration (and productivity) in the pond, b) fine suspended sediments can cause gill damage to rearing salmon fry, and c) fine sediments can build up on the pond bottom and reduce pond depth and volume. Turbidity can be reduced by increasing the distance separating the pond and the main channel.

In Southcentral Alaska, pond temperatures generally do not get high enough to endanger rearing salmon. On the Forest, ponds with warm summer temperatures usually provide more productive rearing habitat than cooler ponds (Schmid, 1989). Summer temperatures for ground water fed ponds can be increased by either increasing pond surface area, or by decreasing inflow into the pond.

Dissolved oxygen concentrations have remained at high levels in rearing ponds developed on the Forest, even during winter months (Schmid, 1989.) Unacceptably low dissolved oxygen concentrations could be improved by increasing inflow to the pond or by providing aeration.

5. **Bank Revegetation.** Developing vegetation around newly constructed ponds is important to future rearing success. Listed below are considerations for revegetation.
  - a) Pond banks should be revegetated immediately after pond construction to avoid erosion of bank sediments into the pond.
  - b) Revegetating with woody species (e.g. willows, cottonwoods) along the pond shoreline increases available cover for fish and birds. Grasses tend to outcompete woody species and are less desirable for use along shorelines.
  - c) Moving back from the immediate shoreline, banks can be either mulched and grass seeded, or planted with woody species. Grass seeding provides better immediate ground cover, while woody species provide better winter browse for moose. Banks will usually convert to woody species after about a decade.

d) Fertilizer applications for revegetation efforts can wash off into adjacent ponds. In limited amounts, this "pond fertilization" can increase pond nutrient supplies and productivity. Fertilizers used for revegetation, however, usually have a higher phosphorous content than is desirable for pond fertilization.

6. **Submergent and Emergent Pond Vegetation.** Newly constructed rearing ponds are generally devoid of bottom vegetation, and years may be required for its growth. Submergent and emergent vegetation increase pond productivity considerably. They provide cover for salmon fry and habitat for macroinvertebrates - a major food source for the rearing (coho) fry. Bottom vegetation also provides feed for waterfowl, and waterfowl help increase nutrient levels within the pond. Plantings of submergent and emergent vegetation have not been tested on the Forest, but might be able to accelerate the development of a diverse, productive pond community.
7. **Predation.** Salmon fry in new rearing ponds are very susceptible to predation due to lack of cover. If Dolly Varden trout have access to the pond, they will prey on young salmon fry and can seriously damage the salmon population. As cover characteristics within the pond improve, survival rates for the salmon fry improve greatly. Cover characteristics within newly constructed ponds on the Forest have been sharply improved by submerging brush bundles into the ponds. Salmon fry use these brush bundles for cover and as a feeding area. For stocked ponds, access of predator fish into the ponds can be blocked at the outlet by using a weir.

Successful development of ground water fed rearing ponds requires the establishment of a diverse ecological community within and adjacent to the pond. Community development may take a number of years, however, some shortcuts are possible, including plantings or placement of brush bundles. Well designed and constructed ponds provide not only rearing habitat for salmon, but also feeding, nesting, and resting habitat for waterfowl and shorebirds.

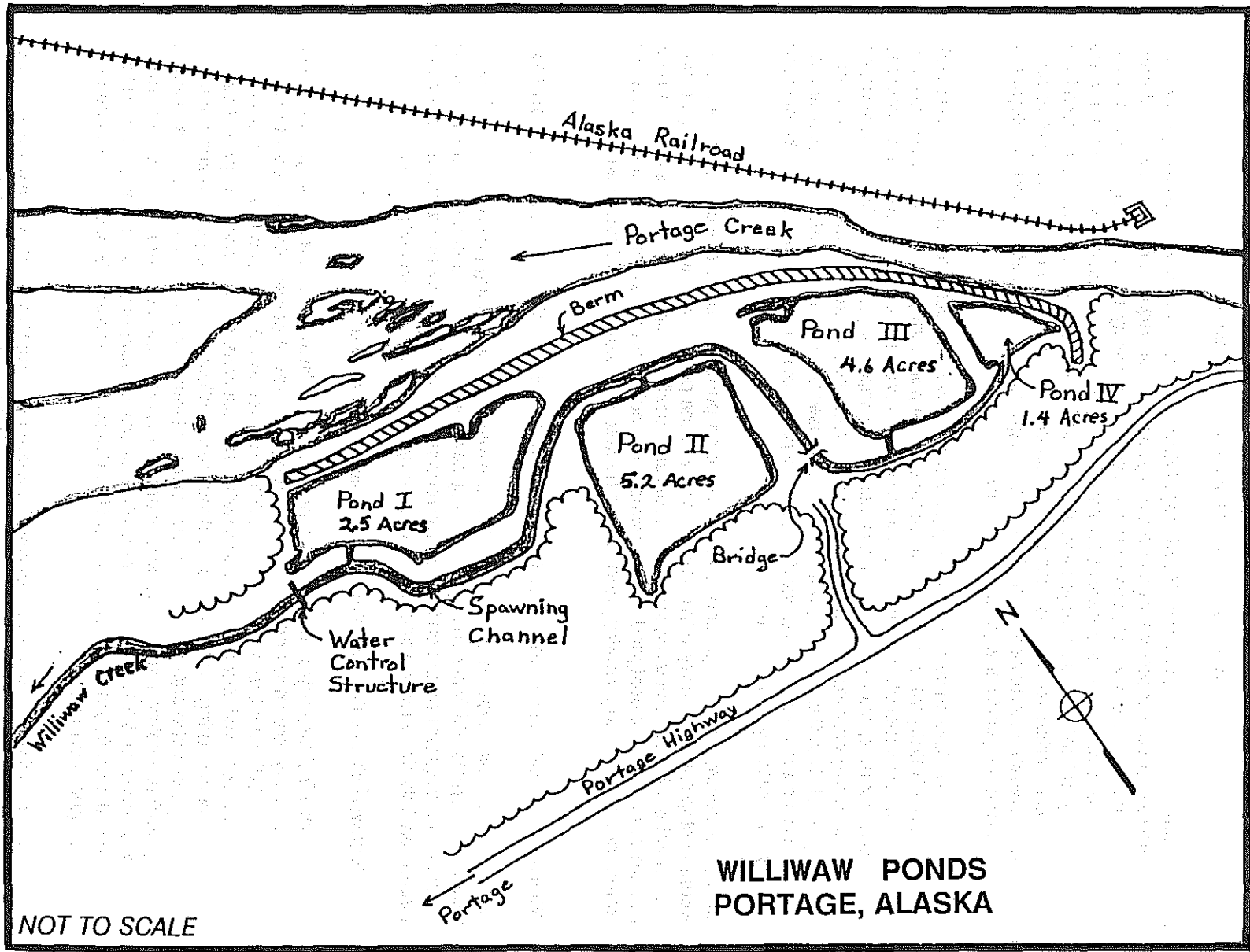
## **GROUND WATER FED SPAWNING AND REARING HABIT PROJECTS ON THE FOREST**

This section examines four ground water fed spawning and/or rearing projects on the Forest. Two of the projects (Williwaw and Lyon Creeks) have rearing ponds, and were constructed in conjunction with gravel extraction for highway reconstruction. The other two projects have been selected on fisheries merits alone, and focus on spawning channel development.

### **A. Williwaw Creek Rearing Ponds and Spawning Channel**

**1. History.** This project is located at the head of Williwaw Creek, a small, ground water fed tributary stream to Portage Creek in Portage Valley. A gravel pit was excavated at the head of Williwaw Creek during the 1950's. Excavation left a shallow pond about one to two feet deep with springs upwelling within and adjacent to the pond. The pond froze to the ground during most winters, and was too shallow to provide good rearing area for sockeye or coho salmon. The stream channel below the pond was used for spawning by sockeye salmon and to a lesser extent by chums. Escapement records have been maintained for Williwaw Creek since 1975. Sockeye escapements vary by year from less than 50 adult fish in the stream to over 500. The average is about 200. Chum escapement averaged around 10.

In 1981, Williwaw Creek was identified as a potential project site for gravel extraction and development of ground water fed spawning and rearing areas. Project feasibility evalua-



NOT TO SCALE

WILLIWAW PONDS  
PORTAGE, ALASKA

tions and data collection were initiated. A conceptual design with four rearing ponds and an interconnecting spawning channel was completed in November 1983. Pond and channel construction commenced in June 1984, and included extraction of 250,000 cubic yards of gravel used for highway reconstruction. The channel and three of the ponds were completed in March 1986, and the final pond was completed in August 1987. Brush bundles were placed in the ponds during the summers of 1986-88. Revegetation done in 1987-8 included grass seeding, fertilizing, and planting willows. The ponds were stocked with approximately 60,000 coho fry in early June 1988.

**2. The Completed Project.** The completed Williwaw Project has four ponds totaling 13.7 acres and a 2,900 foot spawning channel averaging 18 feet wide. This project is targeted to increase harvestable salmon production by 5,600 chum, and 1,400 coho and sockeye (Nelson, 1988.) This new population will take several years to establish. Salmon spawned in the new channel and/or reared in the new ponds will not start returning to Williwaw Creek until the summer of 1989 or 1990.

After completion of the spawning channel, use of the system by chum salmon jumped markedly, with about 40 chums spawning in the new channel each year from 1986 to 1988. Only a very few chums now spawn in Williwaw Creek downstream from the new channel. For sockeye salmon, about a third of each year's escapement spawns in the new channel, and the rest spawn downstream on Williwaw Creek. Only a very few coho adults have been seen using the new spawning channel.

**3. Problems.** Problems that have arisen with the Williwaw Creek project have included: a) lack of cover around and within the ponds, b) predation of salmon fry by resident Dolly Varden, c) loss of planted willow stock due to overcompetition by planted grasses, particularly annual rye grass, d) low water temperatures, and e) intrusion of fine suspended sediments from Portage Creek into the ponds.

Severe predation by Dolly Varden on stocked coho fry occurred during the summer of 1988 (likely up to 50%.) Problems with predation will diminish as more vegetation develops within and adjacent to the ponds. A balance should develop between salmon and Dolly Varden populations. After a number of years, willow and cottonwood plantings should take hold along the pond banks among planted grasses. Rearing habitat should show continued improvements for years, even decades, to come.

Problems with cold water temperatures and fine sediment intrusion, however, will likely continue as set features of the Portage Valley ground water system. These water quality features slow growth rates for both spawned salmon eggs and rearing salmon fry.

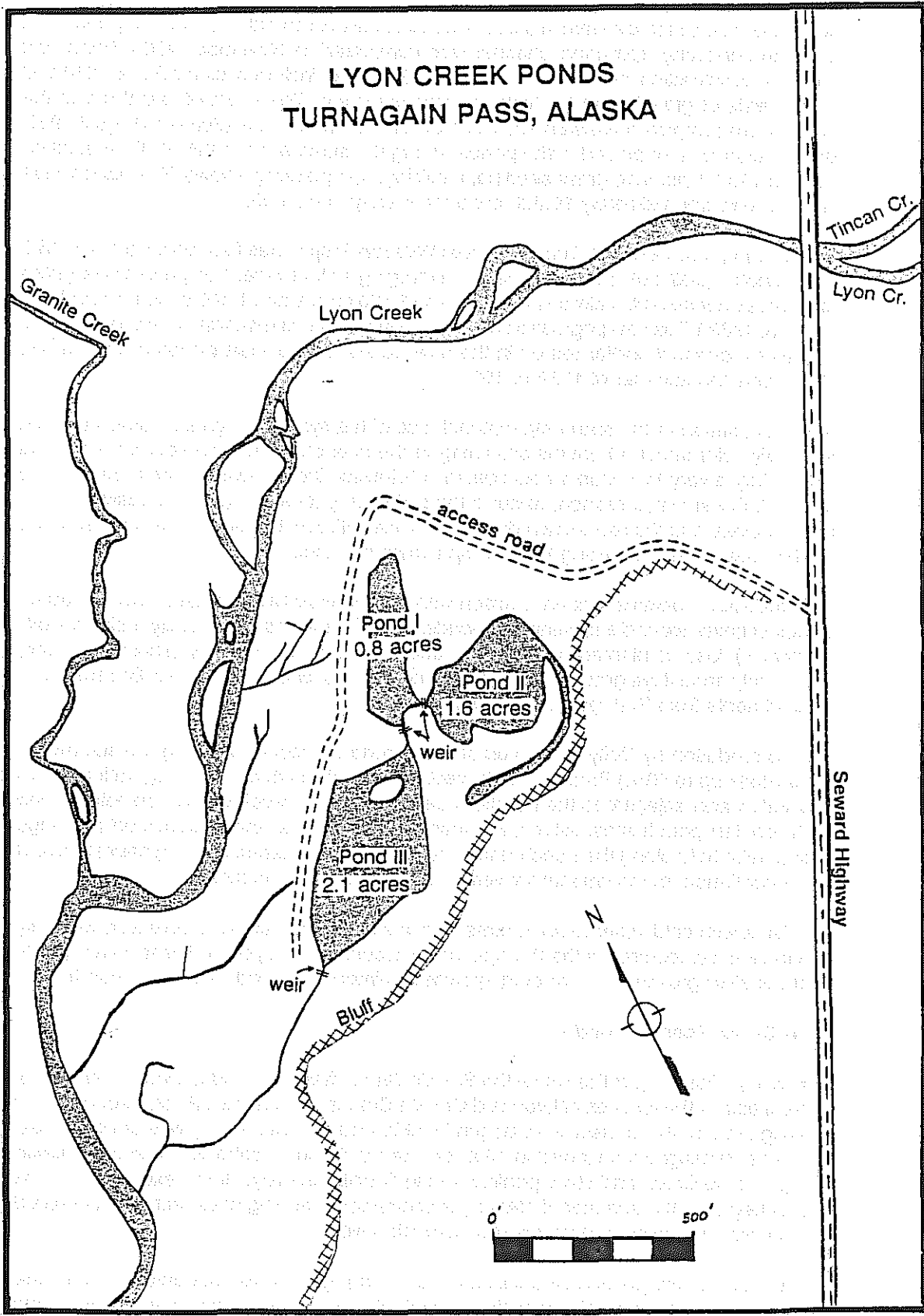
## **B. Lyon Creek Rearing Ponds**

**1. History.** This project lies within the Sixmile Creek drainage, and is situated in alluvial gravels east of the junction of Lyon and Granite Creeks at Turnagain Pass. Ground water investigation on this project were begun in 1981 and material testing was done in 1982. Fisheries investigations started in 1982 on the small run of coho and chinook salmon using Granite Creek and other portions of the Sixmile drainage. Initial excavation of the ponds began in the summer of 1984 by a contractor intending to extract about 300,000 cubic yards of gravel for highway reconstruction work.

During excavation, previously undiscovered silt stringers were encountered within the alluvial gravel deposits. The resulting gravel/silt mix contained an unacceptably high



# LYON CREEK PONDS TURNAGAIN PASS, ALASKA



percentage of fines for road construction. Only 45,000 cubic yards of the material was eventually used. Fisheries habitat design was scaled down to three relatively small rearing ponds, and these were completed in 1985. Plans for development of a spawning channel above the ponds were dropped.

The three ponds were stocked with coho salmon fry in June of 1986 and 1987. Perforated aluminum plates were installed at the pond outlets to keep fry from outmigrating during the summer and fall, and to block access by predators (Dolly Varden) into the ponds. Coho fry have a two year residence time in the pond before outmigrating. In 1988, stocking was switched to October using coho pre-smolts instead of fry (Schmid, 11/3/88.) These presmolts should outmigrate during the spring and summer of 1989. Revegetation work on the pond banks was carried out during the summers of 1987 and 1988, using willow plantings, grass seed and fertilizer. Brush bundles were installed in the ponds in 1987 (40/pond) and 1988 (20/pond) to provide cover for the salmon fry.

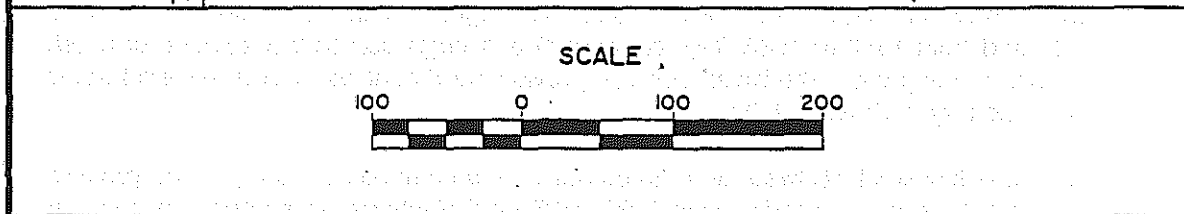
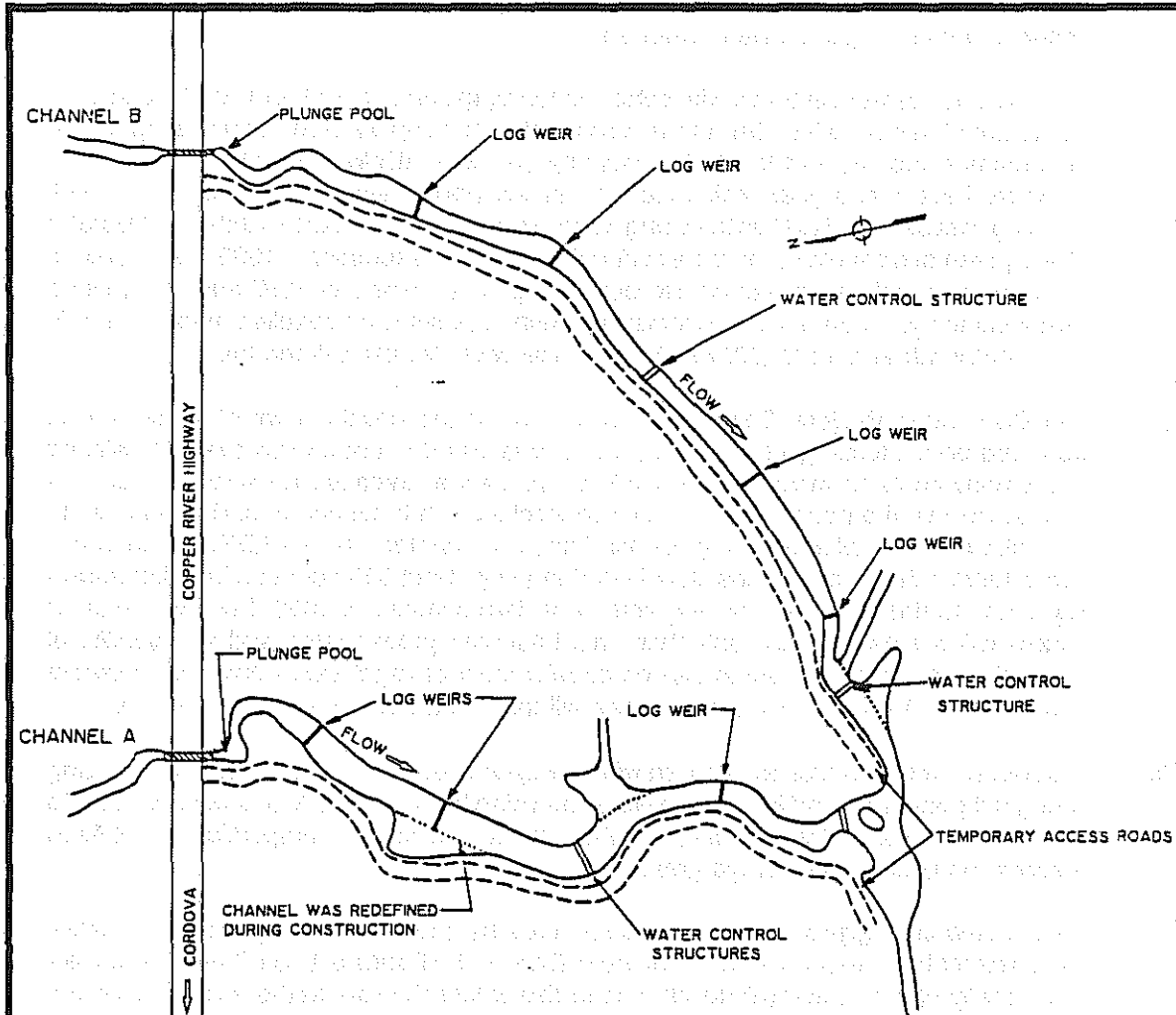
2. **The Completed Project.** This project has involved the development of three ground water fed ponds totaling 4.5 acres in surface area. Ponds were constructed with islands and embayments to maximize the available shoreline. Because no spawning area is connected with the ponds, they have been stocked with hatchery fry and pre-smolts to capitalize on the available rearing habitat. The ponds are targeted by ADF&G to increase harvestable coho salmon in the Sixmile drainage by about 600 per year. The first returning adults to the ponds are not expected until late summer of 1990. Success rates on returns will not be available until that time. Improved growth rates and survivability of stocked cohos is dependent in part on development of vegetation within and adjacent to the ponds. A diverse pond community will take a number of years to develop.
3. **Problems.** Problems that have arisen with the Lyon Creek ponds include: a) downsizing of original pond size, and lack of available spawning habitat, b) lack of cover around and within the ponds, c) loss of planted willow stock due to overcompetition by planted grasses, particularly annual rye grass.

Some research suggests that larger ponds have the potential of being more productive as habitat for both rearing fish and waterfowl (Lee, 1985.) Ponds at Lyon Creek are smaller than initially planned, and will not change in size unless they are further excavated in the future. Growth rates for fry in the ponds showed significant improvement from 1986 to 1987 and from 1987 to 1988. This appears due in large part to the addition of brush bundles to the ponds. The brush bundles provide significant cover and feeding habitat for salmon fry (Schmid, 1989.)

Planted willow stock showed high mortalities due to competition from planted grasses and from dry summer conditions in 1988. Additional plantings are planned for summer 1989 using black cottonwood. Woody species should become dominant at this site, but this will take a number of years.

### C. Mile 25.25 Spawning Channel, Copper River Highway

1. **History.** This project involves two ground water fed tributary streams that originate just north of the Copper River Highway at Mile 25.25. The tributaries flow under the road and join together after about 200 yards. The resulting stream then continues south for over a mile into the tidal slough system of the Copper River Delta. The stream system has over eight acres of rearing habitat along its length, but spawning is limited to the two upper tributary sections. Study of these spawning areas revealed relatively poor spawning



**MILE 25.25, COPPER RIVER HIGHWAY**  
**SPAWNING CHANNELS**  
**CORDOVA RANGER DISTRICT**  
**CHUGACH NATIONAL FOREST**

substrate (with a high percentage of fines.) In late winter, the tributary channels went dry in some locations, leaving spawned salmon eggs susceptible to freezing and/or desiccation. In 1985, 50 sockeyes and about 50 cohos were found using the two channel sections for spawning. Fry sampling in 1986 and 1987 showed low fry populations downstream from the spawning area, indicating poor egg to fry survival rates.

Fisheries investigations of the Mile 25.25 system identified the availability of a large amount of underutilized rearing habitat. The system appeared to have good possibilities for improved productivity if spawning viability could be enhanced. Features lacking from the spawning channels included both consistent winter flow, and good spawning gravels. Winter ground water studies indicated the water table could drop about eight inches below the channel bottom. In February 1987, a preliminary design was developed for improving the two spawning channels. Construction began in late May 1987, and involved excavating the existing channel down several feet with a backhoe. A foot of sorted gravel (3/4" to 3") was then placed into the channel. Flow control weirs were installed for regulation of channel water levels. Rip rap was placed along the channel banks to keep spawning salmon from burrowing into the banks. Channel construction was completed in July 1987 (Metzger, 1988.)

2. **Completed Project.** This project consists of two spawning channel segments with a combined length of 1,200 feet, and an average width of 15 feet. About 2,500 cubic yards of material was excavated from the channels and 730 cubic yards of sorted gravel was then placed back into the channels (Jacobs, 12/88.) The new channels have a slightly reduced gradient, and water flow is maintained throughout the winter.

Improved spawning use occurred almost immediately within the new channel. Over 350 spawning cohos were counted using the channel in 1987 (a 700% increase over the previous year.) The channel produced over 700,000 coho fry indicating an 80% survival rate. Fry then went on to use not only the immediate downstream channel for rearing habitat, but also other areas further out into the Copper River Delta. Studies on tagged fry done in 1987 and 1988 indicate that this portion of the Delta has a large amount of rearing habitat that is currently underutilized due to lack of nearby spawning area. Fry from the Mile 25.25 channels are able to migrate to these rearing areas. As a result, spawning productivity from the Mile 25.25 channel carries a much higher value than initially realized. The harvestable coho return from the 1987 spawning should be about 6,000 adults (Metzger, 1989.)

3. **Problems.** Some channel maintenance will be needed on the spawning channel sections in the future. This is due to deposition of fine sediments within the gravels, and movement of gravels by the spawning salmon. Springs upstream from the spawning channels act as source areas for fine sediments during high flow periods. Displacement of spawning gravels could increase significantly with a very large return of adult cohos to the channel.

Brown bear populations may to be attracted to this site during sockeye spawning season if sockeyes begin using the channel. This could create some human/bear conflicts. Cohos spawn in November and December and bear conflicts are generally not an issue. Timing of adult coho returns to the Mile 25.25 channels may be an additional concern if it is not well synchronized with commercial fishing use in the area. Tagged adults from the system will be studied to determine when, and what percent are commercially harvested.

#### D. Pigot Bay Spawning Channel, Prince William Sound

1. **Background.** Pigot Bay is the site of a proposed project to be constructed in 1990. The project would involve construction of as much as a mile of spawning channel on the raised floodplain on the north side of Pigot Creek. Construction of the spawning channel would involve excavating down to below the level of the existing water table. The floodplain is composed of coarse alluvial gravels and the water table varies from 0 to 6 feet below the surface. If properly situated, the spawning channel can tap into the moderated base flows of Pigot Creek. The proposed spawning channel targets pink and chum salmon, and would not require adjacent rearing habitat.

Pigot Creek itself is glacial in character and has very large flow fluctuations. Much of the creek's bed material becomes mobile during peak flows. Spawning use on Pigot Creek is negligible. Use of base flows from Pigot Creek, however, would allow for continuous, moderated, low volume flows ideal for spawning. The proposed spawning channel would need to be bermed on one side to prevent peak flooding events on Pigot Creek from spilling into the channel and causing damage. A constructed spawning channel could net returns of several thousand harvestable pink and chum salmon to this system.

This proposed spawning channel will be limited in its productivity primarily by its minimum winter flow levels. Minimum flows on Pigot Creek will be evaluated during early spring of 1989. Cold water temperatures from Pigot Creek could also slow salmon growth and survival rates, as has occurred on the Williwaw Creek spawning channel.

#### SUMMARY

A variety of ground water fed spawning and rearing habitat projects have been accomplished on the Chugach National Forest with apparent good success in increasing salmon productivity. Projects have targeted coho, sockeye, chum, and pink salmon. Prior to design and construction, all projects have been carefully evaluated as to the balance between spawning and rearing habitat needs for the targeted species.

New spawning channels have produced rapid results in that salmon are able to exploit the spawning habitat as soon as the channels are completed. Development of good quality rearing habitat (ponds), however, is generally more complex. It requires careful attention be given to the physical design of the pond. A number of years are then needed for the ecology of a new pond to develop to the point of providing productive habitat.

Construction of rearing ponds can have the additional benefit of creating habitat for waterfowl and shorebirds. This includes habitat for feeding, nesting, brood-rearing, and resting. Again, pond ecology needs to develop before high quality habitats are available for waterfowl.

None of the four projects discussed in this paper is yet old enough to show returns of adult salmon spawned and/or reared within the new systems. The Williwaw and Lyon Projects should begin showing returns this summer. Counts taken on outmigrating fry indicate that very good adult returns may be expected on each of the completed projects.

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## DEWATERING THE ST. GEORGE HARBOR

by David C. Lanning<sup>1</sup>

### ABSTRACT

Geological formations and dewatering data provided input for resistance analog flow nets in the horizontal plane and finite element analysis in the vertical plane to determine the transmissivity of the volcanic rock and gravels found at St. George Harbor. This information was used to predict the total dewatering requirements for the hard rock drilling and excavation methods required in the harbor construction.

### INTRODUCTION

In 1984, Brice Inc. of Fairbanks, Alaska was awarded the contract to construct a small boat harbor on St. George Island, the Pribilof Islands, Alaska. This included road and industrial site development work, as well as quarrying rock for the breakwaters, and excavating the harbor basin out of the volcanic rock of the island. The contract price was approximately \$10 million. The project involved approximately 1 million yards of hard rock excavation.

The purpose of this paper is to discuss the construction problems encountered in dewatering and excavation of the harbor. This includes determining the hydraulic conductivity and transmissivity of the basaltic lavas and the total flow that would need to be pumped in order to dewater the harbor for construction purposes.

### ACKNOWLEDGEMENTS

The St. George Harbor was designed by the late Brent Drage and Jeff Gilman, both of Peratrovich, Nottingham and Drage, Inc. of Anchorage, AK. The final plans were significantly influenced by alternative design ideas and other important contributions by Helenka Brice, Al Brice, and Sam Brice of Brice Inc., Fairbanks, AK.

### BACKGROUND INFORMATION

St. George Island is volcanic in nature and along with St. Paul Island sits close to the edge of the continental shelf in about 100 fathoms of water. The location of St. George Island and the new harbor can be seen in Figure 1.

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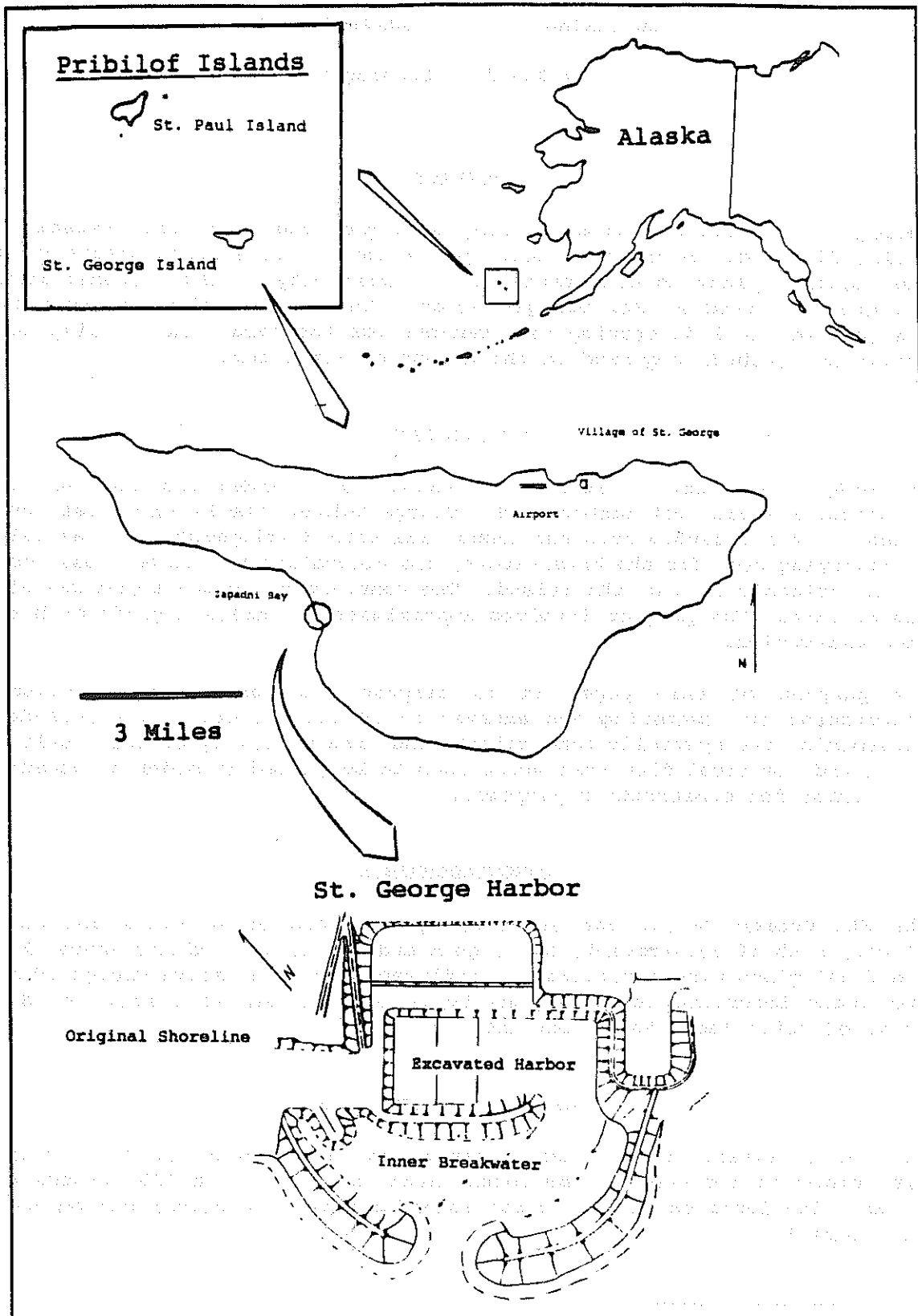


FIGURE 1. Vicinity Map for St. George Harbor.



There are several factors that restricted the solution to the construction problems on this project. Those were:

- 1.) The rock types and gravel layers.
- 2.) The heavy equipment actually available on this very remote site.
- 3.) The very high transmissivity of the rock/gravel bedrock.

These will be discussed individually as follows.

#### Rock Types

Although St. George Island and St. Paul Island are similar in the basic rock types that comprise the islands, St. Paul is a low lying island with sand beaches while St. George has a coast line of high lava cliffs. These coast line cliffs show obvious layering between volcanic ashes and basaltic rock flows. At the tide line of these cliffs, the rock that is usually found is columnar basalt. This is a very dense crystal (specific gravity of 2.7-3.0) having a slightly off vertical and chimney like crystal growth.

The island is comprised of various lava flows with ash like scoriaceous gravels as well as mud and boulder flows between lava layers. The uppermost soils are similar in nature to the hawaiian volcanic soils and are probably very fertile, but are shallow and rocky. The uppermost lava flows are more vesicular (porous) and have a cubic fracture pattern rather than columnar. The larger sizes of armor that Brice Inc. quarried were usually this slightly less dense, more blocky material. Typically, the columnar basalt was just below this large blocky layer.

In general, the rock was fractured, porous and layered. Conversely, the columnar basalt in the lower levels was very competent and impermeable, but was only occasionally usable for armor because of its smaller size.

Because the soils were so discontinuous, when one considered such a large area as the harbor (900 by 400 feet), it is reasonable to treat them as homogeneous. If there had been large or singular discontinuities, perhaps this would not have been a reasonable assumption.

Because of the various layers of rock and gravel and the varying permeability of the rock, it was not always possible to predict how water would flow through the rock. On occasion, it was possible to look down into drill holes and find a piezometric surface down 1 foot while only a few feet away, through solid rock, the cutoff trench had a water surface lower by 4 feet. The drill hole had apparently intercepted a horizontal porous layer from the ocean that ran below the cutoff trench. Fortunately for the dewatering program, the harbor excavation went deep enough to remove those porous layers and reach into the sound, relatively impermeable columnar basalt.

### Available Equipment

The planned method of excavating the harbor out of the solid rock was to first drill a regular pattern of holes. These would then be loaded with the explosives used to blast the rock loose. Excavation with large backhoes or by dragline could then follow. Since no one really knew what problems would be found, the particular details were left for on-site decisions.

Since it was a hard rock quarry job, the holes were drilled with a air rotary drills and down-the-hole hammer bits made by Ingersoll-Rand Corp.. Brice brought 3 of these drills from the mainland; 2 large units and one smaller unit. This drilling method uses air as the drilling fluid to remove cuttings. Air is injected through the stem and button bit to blow the cuttings away from the drill bit and upward out of the hole. Air velocities of about 3000 feet per second are required (Driscoll, 1986). This requires a very large, loud and expensive air compressor. Because of the high pressure and volumes required for drilling, the compressed air is also used to power the entire drill rig for moving and setup.

This drilling method is very efficient in consolidated and semi-consolidated formations but has several disadvantages in the wet environment of the St. George excavation. Even in dry formations higher in the quarry, an occasional rock would fall in from the side of the hole and trap the drill bit and drill stem. It would require considerable time to break up the rock and free the bit. While drilling in the wet environment of near sea level the entire top of the hole would often collapse and trap the drill bit and stem. This was eventually remedied with a short plastic collar at the top of the hole.

Because of the difficulties encountered in the wetter drilling of the harbor itself, it was necessary to dewater in order to drill. Unfortunately it was also necessary to drill in order to blast a sump hole for the pumps. So it was kind of like pulling yourself up by your bootstraps, except down by your pumps. The drilling to place the first dewatering pump was perhaps the longest and toughest on the project and required about 2 weeks. This can be compared to blasting every afternoon in the dry areas for a similar sized shot.

### Excavation

The actual excavation was done with high production backhoes. Two backhoes were brought to the site. One was a CAT 245, the largest backhoe made by the Caterpillar Tractor Company. The other was the Lebherr 961, made by the Lebherr Company of West Germany, and considered the largest backhoe available. Both would excavate to a depth of about 14-15 feet comfortably. Therefore, on our next-to-the-last lift, we excavated down to about +2 feet Mean Low Low Tide (MLLT). We could reach the required -12 feet MLLT comfortably from there.

The situation would have been relatively simple had this been the entire problem. However, the City of St. George (The Owner) had an option in the contract to require that the harbor be excavated to -20 feet MLLT if he found the additional funding required for the excavation. This extra work had been previously negotiated and the City of St. George was vigorously lobbying the State Legislature for the required funds.

We had two options if we were required to excavate to -20 feet MLLT: we could dewater to -7 feet MLLT and excavate with backhoes to the -20 feet MLLT; or excavate with a dragline from the +2 feet MLLT level. The last option was more expensive and physically would require a long lead time to modify the Manitowoc 4600 crane for dragline work. Thus the method and cost depended upon whether we could dewater the harbor to -7 feet MLLT.

#### Dewatering Methods

Brice had expected to find a lot of water, but felt confident that they could dewater the excavation with the pumps that they had brought along to the island. These pumps were manufactured by Gator Pump Inc., of Texas, as irrigation pumps. They were capable of pumping approximately 8,000 - 10,000 Gallons Per Minute (GPM). With 2 pumps, and a third along for a spare, the dewatering capacity was 16,000 - 20,000 GPM, depending on the total head. Since one cubic foot per second (CFS) is equal to about 449 GPM, our dewatering capacity was about 35 to 45 CPS.

One of the pumps was set in the sump as soon as was practical after the excavation, and pumped the level in the sump down to about -4 feet MLLT within minutes. Piping was observed coming from the ocean side, but the pumping was definitely helping dry out the nearby drilling operations. We felt confident that we would be able to hold the water necessary for operations.

The next step was to dig a long cutoff trench parallel to the ocean and behind the inner breakwater to cut off the incoming water and leave the majority of the excavation in a fairly dry state. Unfortunately, by the time we had finished excavating this trench, 600 feet long and 10 feet deep, the pump was running 100 per cent of the time and only holding the depth to about -1 foot MLLT. Since the average elevation of the ocean on the other side of the inner breakwater was +1 foot MLLT, we were holding back a head of about 2 feet across the inner breakwater. Given the height of the breakwater we were pumping across, our rate of pumping was about 25 CFS (11,600 GPM) at this time. Piping was observed at many locations. On the north end of the harbor and behind the cutoff trench, the salt water flow coming up some of the drill holes produced fountains almost 2 feet high.

It was necessary to set the second pump to help hold the water level down so the drilling operations could stay dry enough when we began the blasting and excavating of the harbor itself.

By the time that approximately 1/3 to 1/2 of the harbor itself had been excavated to the required depths of -8 and -12 feet MLLT the owner had not yet exercised his deeper harbor option, and both pumps were operating at their capacities. The water level was being held to the -1 feet MLLT.

## METHODS

### Calculation Of The Transmissivity

No adequate pumping data was taken in order to calculate the conductivity, transmissivity, or storage of the rock by drawdown methods. But the pumping record shows that at two times we are fairly certain of excavation limits and the pumping rates. When Brice had finished excavating the cutoff trench, the one operating pump was at its capacity of 27 CFS (11,600 GPM). The second time was near the end of the excavation for the 1986 season when the pumps were at the limit of their capacity and pumping approximately 25.8 CFS and 21.4 CFS respectively for a total of 46.8 CFS (21,000 GPM).

An analysis of both situations was performed using flow net analysis. The flow nets were drawn using the aid of a constant voltage applied through resistive paper to model the water flow through porous media. Flow net analysis assumes that the soils are homogeneous, and that the flow is laminar and obeys the other assumptions of the Laplace Equation for flow in 2 dimensions. These assumptions are a simplification of the actual situation but are nevertheless not unrealistic and are at least as good quality as the pumping data. Calculation of the Reynold's number yielded a value of 1.0, which indicates a flow on the upper limits of laminar flow in porous media.

When Darcy's Law is applied to flow nets, it is the summed flow from each streamtube for each foot of depth:

$$Q = (m/n) * K * H * D \quad \text{eq. 1}$$

where:

- (m/n) is the ratio of streamtubes to head drops in the flow net
- K is the hydraulic conductivity (in Feet/second)
- H is the total head driving the flow across the media (in Feet)
- D is the depth of the media (in Feet)

(Department of the Army, 1952)

The transmissivity can be found by rearranging equation 1 to:

$$T = Q * (n/m) / H \quad \text{eq. 2}$$

where:

T is the transmissivity (K times D) in Feet<sup>2</sup> per Second and the other variables are defined for equation 1.

### Finite Element Analysis Of The Transmissivity In The Vertical Plane

A finite element heat conduction program written by Dr. John Zarling (Kox.bas) was used to check the reasonableness of the flow net approximations in the horizontal plane. A grid was drawn to model the best known location of the various rock and sand layers in the natural beach-rock fill system that comprised the inner breakwater of the harbor.

This method requires knowing the values of hydraulic conductivities in order to calculate flow. Median conductivities were used for the sands and clays (0.004 Ft per Second and 0.00004 Ft per Second respectively) (Driscoll, 1986), while the conductivity of the basalts was adjusted until the program calculated the actual flow that we were pumping.

### RESULTS

#### Horizontal Analysis Using Flow Nets

The flow net for the first situation can be seen in figure 2 below when the flow was about 25.8 CFS.

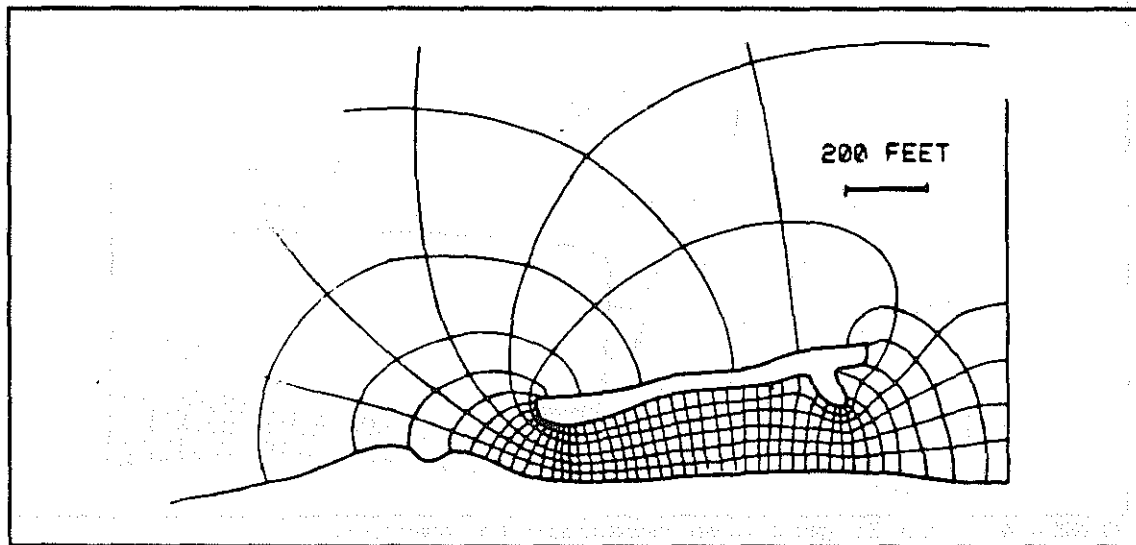


FIGURE 2. The cutoff trench to the sump is finished.

There are 44 streamtubes and 6 head drops in the first situation. Using equation 2 the transmissivity was found to be 1.8 Feet Squared per second.

The flow net for the second situation can be seen in figure 3 below.

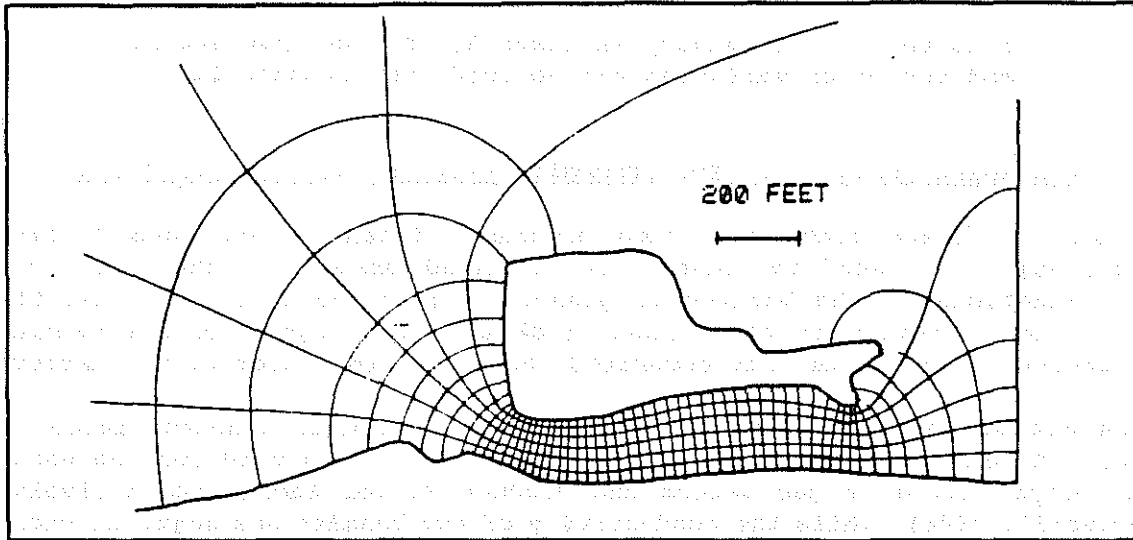


FIGURE 3. The pumps are at their capacities of about 46.8 CFS.

There are 54 streamtubes and 6 head drops in the second situation. The transmissivity is calculated using equation 2, and found to be 2.6 Feet Squared per Second. The difference between these values for the transmissivity is slight when considering the quality of the known data and the method used. Therefore, a range of 2.2 (the average) to 2.6 (the highest) value of transmissivity was used when calculating the expected maximum range of flow during the maximum open harbor work. This situation can be seen in figure 4 below.

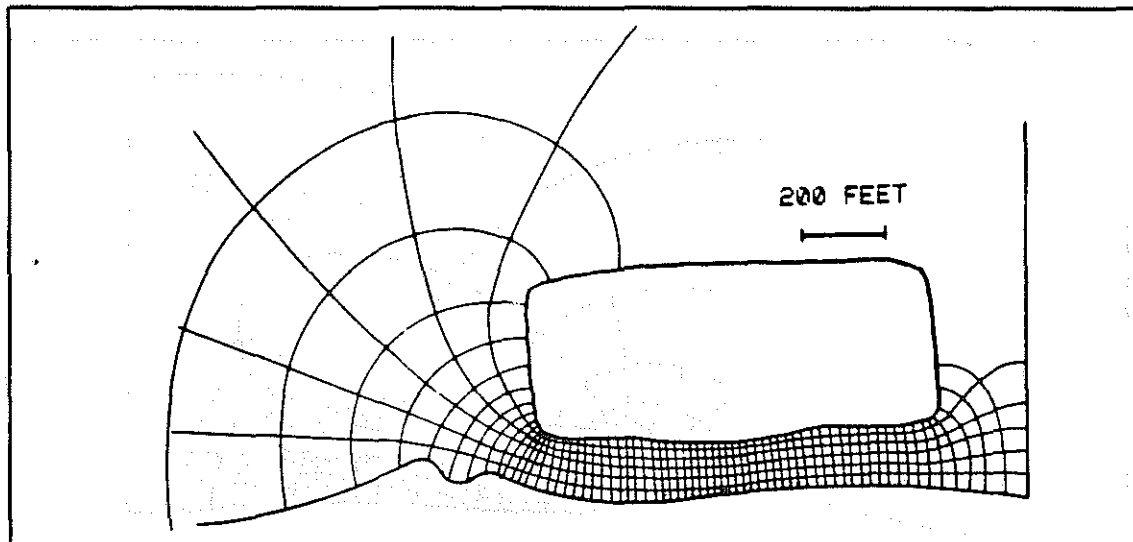


FIGURE 4. The largest area necessary to dewater.

The maximum expected flow at the harbor is calculated using equation 1 and the transmissivities found previously. The number of streamtubes in this situation was 66 and the number of head drops was 6. Using the average transmissivity, the expected flow was 48.4 CFS, and using the maximum transmissivity, the expected flow was 57.2 CFS. This assumed that the same driving head of 2 feet was held across the breakwater separating the harbor from the ocean. If Brice was required to hold the water below -7 feet MLLT during excavation for a deeper harbor, the flow would increase by a factor of the ratio of the driving head gradients. This factor is 8 feet compared to 2 feet or 4. The flow would then be between 195 CFS and 230 CFS or 87,000 to 103,000 GPM).

Vertical Analysis Using Finite Element Analysis

In flow net analysis, every streamtube is assumed to have the same flow. Since about 3/4 of the tubes ran directly through the inner breakwater, and it was about 800 feet long, the flow that the finite element program needed to calculate was:

$$Q = (25.8 \text{ Ft}^3/\text{Sec}) * (3/4) / 800 \text{ Feet}$$

Therefore:  $Q = 0.024 \text{ Ft}^3/\text{Sec}/\text{Ft}$

When a conductivity for the basalts of 0.34 Feet/Second was used in the program calculations, the resulting flow was about right. This flow of 0.024 Feet<sup>3</sup>/Second compared very well with that calculated by flow net analysis of 0.22 -0.26 Ft<sup>3</sup>/Second.

Moreover, the program also calculated the potential at the nodes which enabled the drawing of equipotential lines (head-loss gradients) across the breakwater. These lines are remarkably similar to those vertical lines that are assumed by a horizontal flow net. These head loss gradients can be seen in figure 5 which shows a vertical section through the inner breakwater overlain by the finite element grid.

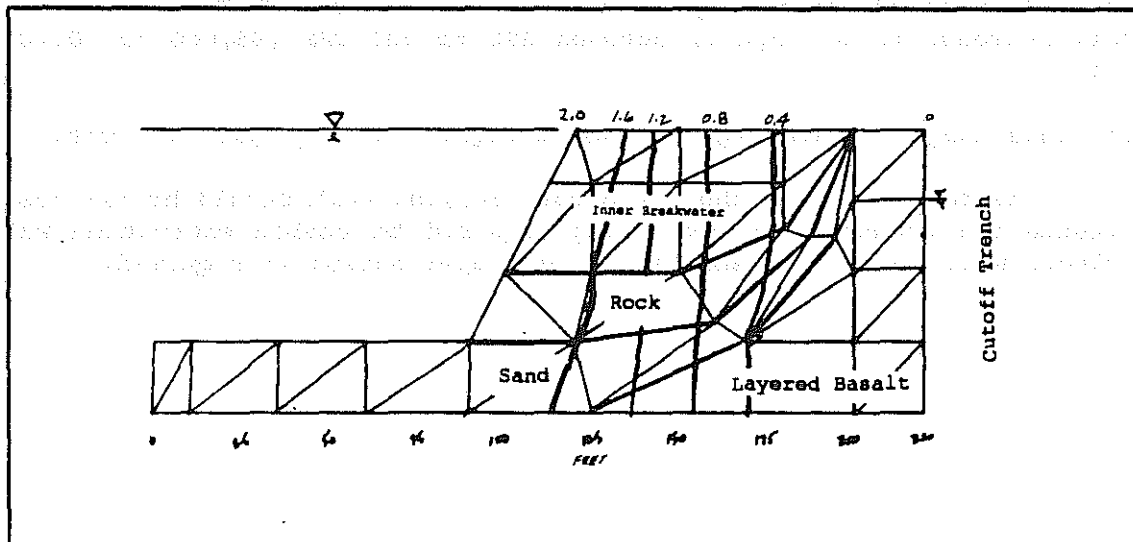


FIGURE 5. Vertical Section through the Inner Breakwater.

## DISCUSSION

The average hydraulic conductivity of the basaltic lavas at the harbor location was found to be approximately 22,500 feet per day. This was determined by reasonable agreement between flow net analysis in the horizontal plane and finite element analysis in the vertical plane using dewatering data at two different stages of excavation. This is a very high value for hydraulic conductivity, and far above the upper value of 300 Feet/Day given for vesicular basalts (Driscoll, 1986). This large flow probably takes place in the gravelly layers of scoriaceous material which separates the layers of vesicular basalt. It would seem more appropriate then to use the transmissivity of the media as the appropriate measure of its permeability rather than the hydraulic conductivity. While the conductivity denotes flow through an average square foot, the transmissivity denotes average flow through a foot wide section down through the breakwater. Since the depth of the porous materials is about 10 feet here, the transmissivity is equal to 10 times the conductivity or about 225,000 feet squared per day.

This analysis does not address several other possible problems associated with the dewatering of this harbor to a -7 feet MLLT. There exists the possibility of liquefaction of the sands and the possibility of a deep seated circular slope failure in the rock fill above the sands (Hirschfeld, no date). Even though the head gradients through the sands and rock are fairly low, these possible failure modes need to be investigated but are not discussed in this paper.

## CONCLUSIONS

- 1.) The anticipated rate of dewatering at the completion of the harbor is estimated to range from 48 to 57 CFS (22,000 to 26,000 GPM). If the required depth of dewatering is to elevation -7 feet MLLT then the flow would increase to a range of between 192 to 228 CFS (86,000 to 103,000 GPM).
- 2.) This larger dewatering rate would require 8 pumps plus backups.
- 3.) Costs to changeover the crane for dragline work should be seriously investigated because the dewatering required to enable excavation with backhoes will be very expensive if the deeper option is required.



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# CHENA HOT SPRINGS AS A NATURAL WATER SOFTENER

by Kyle D. Vaught<sup>1</sup> and Jacqueline D. LaPerriere<sup>2</sup>

## ABSTRACT

Graduate students of a limnological methods course at the University of Alaska Fairbanks collected sufficient water chemistry data at Chena Hot Springs Resort to characterize the domestic and recreational water sources of the resort and a nearby stream. Analyses included temperature ( $^{\circ}\text{C}$ ), specific conductance, pH, alkalinity, total and calcium hardness, total Fe,  $\text{NO}_3^-$ -N,  $\text{NH}_4^+$ -N, and  $\text{SO}_4^{2-}$ . Samples were collected from four sources: a covered hot spring reservoir, two nearby drilled wells, and Monument Creek (a local stream). Analyses were carried out in the field with a Hach DR-EL/4 spectrophotometer and other Hach test kits. Significant differences in conductivity, alkalinity, and total hardness of water from the covered hot springs reservoir and Monument Creek suggested that the aquifer feeding the hot springs is not an important contributor to Monument Creek. On the basis of chemical analyses presented here and in earlier studies we set forth a hypothesis that the hot springs act as a natural water softener, depleting calcium through calcium carbonate precipitation or ion-exchange reactions with rock deep in the thermal spring system. The depletion of calcium, coupled with a high sodium concentration gives rise to a sodium bicarbonate buffering system rather than the usual calcium bicarbonate system.

## INTRODUCTION

A thermal spring consists basically of a deep fault system that allows cold meteoritic water to trickle down to a zone of deep heating from underlying magma; the heated water then rapidly rises to the surface through another fault system (Figure 1). The chemical and physical characteristics of thermal springs vary greatly depending largely on the depth to which the water is circulated and

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the rock type of the deep heated reservoir (Waring, 1917; Biggar, 1974). Chena Hot Springs water is characterized by very high sodium and low calcium concentrations as shown by Waring (1917) and unpublished data (1958) of the U. S. Geological Survey (USGS), a reversal of what is found in most aquifers and surface waters of the area.

Chena Hot Springs is located in interior Alaska, approximately 96 km (60 mi) east of Fairbanks (Figure 2). The hot springs have been the center of a popular resort since about the turn of the century. Investigation of the hot springs' waters began as early as 1912 by P. J. B. LeBlanc (Waring, 1917). The USGS conducted a thorough chemical investigation in 1915, noting that the hot springs had a sodium bicarbonate buffering system (Waring, 1917).

The study discussed here began in spring, 1985 as a field exercise for a graduate level course in "Chemical Methods in Limnology" at the University of Alaska Fairbanks. The original objective of the study was to provide students experience with chemical analyses in the field by comparing the chemical character of Chena Hot Springs waters with that of nearby Monument Creek. In the initial sampling results, very low calcium and total hardness levels in the plywood-covered hot springs reservoir, and a total hardness concentration only slightly higher in a nearby drilled well (well 1) used as a "cold water" source, were of particular interest (Figure 3). To explain the unusual composition of both the hot and cold water supply of the Chena Hot Springs Resort, it was hypothesized that  $\text{Ca}^{+2}$  was being depleted from the hot springs waters by precipitation of  $\text{CaCO}_3$  and that the rising hot springs water mixed with the shallow aquifer of well 1. Thus well 1, about 75 m from the covered hot springs reservoir, had only a slightly higher total hardness than the covered hot springs reservoir water, as a result of rising hot springs water mixing with the shallow aquifer feeding well 1.

Calcium and total hardness were considerably higher in a second drilled well sampled in September, 1987 (well 2, Figure 3) than the covered hot springs pool.

## METHODS AND MATERIALS

Sampling at Chena Hot Springs was conducted on 15 and 22 April 1985, by graduate students in the limnological methods course. Three locations were sampled: the plywood-covered hot springs reservoir, well 1, and Monument Creek approximately 500 m downstream from the resort area (Figure 3). Chemical and physical analyses included temperature ( $^{\circ}\text{C}$ ), specific conductance, pH, alkalinity, total hardness, calcium hardness, total Fe,  $\text{NO}_3^-$ -N,  $\text{NH}_4^+$ -N, and  $\text{SO}_4^{2-}$ . In addition, on 26 September 1987 samples were collected from well 2 for calcium and total hardness analyses. Sediments from the covered hot springs reservoir were qualitatively analyzed for  $\text{CaCO}_3$  content by acidification in September, 1987 as well. All analyses were performed on triplicate samples to enable statistical comparison. Mean concentrations were compared between locations by Student's t-test. Variances were tested for homogeneity by an F test. Analyses were carried out in the field with a Hach DR-EL/4 spectrophotometer and conductivity meter and Model 19000 Hach digital pH meter and temperature probe. Hach digital titrators were used in analyses requiring titration. Standard additions were conducted on each variable to assure accuracy of results.

## RESULTS

The waters of Chena Hot Springs and Monument Creek were clearly very different (Table 1). The characteristics that showed significant differences ( $P < 0.05$ ) between these locations were total hardness, alkalinity,  $\text{NO}_3^-$ -N, specific conductance, and  $\text{SO}_4^{2-}$ . Calcium hardness showed no significant difference because sample variance was large.

Calcium hardness concentration of water from well 2 was much higher than that of the covered hot springs reservoir ( $P < 0.001$ ). Total hardness concentration was much higher in well 2 ( $P < 0.001$ ), than in either well 1 or in the covered hot springs reservoir ( $P < 0.05$ ). Acidification of sediment samples from the covered hot springs reservoir showed that only negligible amounts of  $\text{CaCO}_3$  were precipitated in the surface hot springs water.

TABLE 1. Characteristics of water in the Chena Hot Springs (CHS) area, Alaska. All values are the means of triplicate samples, except as noted otherwise. Standard errors of means are shown in parentheses.

| Characteristic  | Location <sup>a</sup> |                   |             |                      |
|---|-----------------------|-------------------|-------------|----------------------|
|   | CHS                   | Monument<br>Creek | Well 1      | Well 2               |
| Temperature (°C)  | 54.6 (1.4)            | 6.3 (0.2)         | 20.6 (0.6)  | 19.0 <sup>**b</sup>  |
| pH  | 8.7                   | 6.9               | 8.3         | ---                  |
| Specific<br>conductance   | 610.0(26.4)           | 89.0 (1.0)        | 493.0 (6.6) | 169.0 <sup>**b</sup> |
| Total hardness<br>(mg/L as CaCO <sub>3</sub> )  | 18.5 (8.3)            | 45.3 (0.8)        | 21.0 (2.2)  | 55.7 (1.2)           |
| Calcium hardness<br>(mg/L as CaCO <sub>3</sub> )  | 1.7 (1.2)             | 38.2(15.0)        | ---         | 45.0 (2.0)           |
| Iron (mg/L Fe)  | 0.1 (0.0)             | 0.1 (0.0)         | ---         | ---                  |
| Alkalinity<br>(mg/L as CaCO <sub>3</sub> )  | 98.0 (4.4)            | 36.0 (1.3)        | 100.0 (2.3) | ---                  |
| Sulfate<br>(mg/L SO <sub>4</sub> )  | 88.0(16.2)            | 6.3 (1.4)         | 82.0 (4.4)  | ---                  |
| Sodium <sup>c</sup><br>(mg/L Na)  | 110.0                 | 1.5               | ---         | ---                  |
| Nitrite/Nitrate<br>(mg/L NO <sub>2</sub> <sup>-</sup> /NO <sub>3</sub> <sup>-</sup> -N) | 5.4 (0.2)             | 1.4 (0.1)         | 3.3 (0.2)   | ---                  |
| Ammonia<br>(mg/L NH <sub>3</sub> -N)  | 0.9 (0.1)             | 0.4 (0.2)         | 0.6 (0.5)   | ---                  |

<sup>a</sup>See Figure 3 for locations

<sup>b</sup>Single measurement recorded

<sup>c</sup>USGS, unpublished data (1958)

Results of analyses compare well with data reported by the USGS for Monument Creek (unpublished data, 1971) and Chena Hot Springs (unpublished data, 1958).

## DISCUSSION

The distinctly different chemical character of Chena Hot Springs and Monument Creek indicate that the hot springs are not an important contributor to the discharge of Monument Creek.

The much higher concentrations of calcium and total hardness in water of well 2 compared to water of the covered hot springs reservoir indicated that  $\text{Ca}^{+2}$  present in meteoritic water feeding the hot springs system was being depleted in the thermal spring system. This supports our hypothesis that water from well 1 mixed with rising hot springs water, explaining the low total hardness concentration of well 1 water. Negligible amounts of  $\text{CaCO}_3$  in sediment samples from the covered hot springs reservoir indicated that  $\text{Ca}^{+2}$  was depleted deeper in the thermal spring system.

There are at least two explanations for the depletion of  $\text{Ca}^{+2}$  in the thermal spring system: 1) calcium may be precipitated as  $\text{CaCO}_3$  according to the equation  $\text{Ca}^{+2} + 2\text{HCO}_3^- \xrightarrow{\text{heat}} \text{CaCO}_3 + \text{CO}_2 + \text{H}_2\text{O}$  (Brown and LeMay, 1977); and 2) calcium depletion in the thermal spring may also be due to ion exchange reactions of plagioclase and alkali feldspars in the deep heating reservoir with the meteoritic water feeding the thermal spring system. The second mechanism is understood well enough to enable the use of ionic concentrations of  $\text{Na}^+$ ,  $\text{K}^+$  and  $\text{Ca}^{+2}$  in hot spring surface waters as a geothermometer to estimate core temperatures of thermal spring systems (Miller et al., 1975). Biggar (1974), using this geothermometer technique, estimated the core temperature of Chena Hot Springs to be about  $115^\circ\text{C}$ . It is probable that both these mechanisms contribute to the depletion of  $\text{Ca}^{+2}$  in the waters of Chena Hot Springs (D. B. Hawkins, Professor of Geology, University of Alaska Fairbanks; personal communication). This depletion of  $\text{Ca}^{+2}$ , coupled with the introduction of high concentrations of  $\text{Na}^+$  from host rock in the deep heating zone, mimics the ion-exchange effect of a large water-softening unit.

Wescott and Turner (1981) established the existence of a shallow aquifer (at about 8-18 m) through the locality of Chena Hot Springs. We found chemical evidence that this aquifer, at least at the location of well 1, is not isolated from the rising waters of the hot springs.

This study illustrates the level of learning and interest that can be generated in students when laboratory procedures are applied to understanding local "real world" phenomena.

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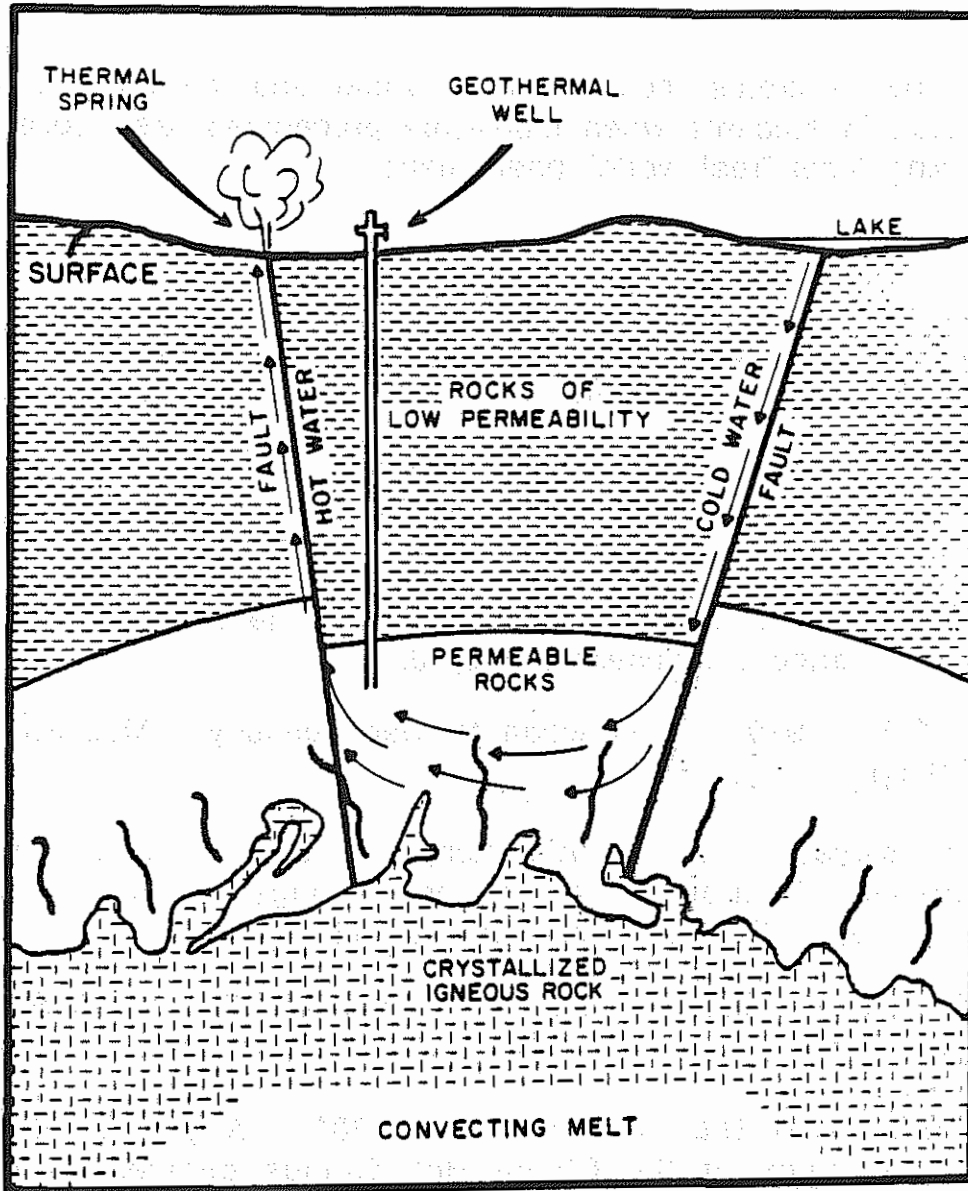


FIGURE 1. SCHEMATIC REPRESENTATION OF A THERMAL HOT SPRING (FROM FORBES AND BIGGAR, 1973).



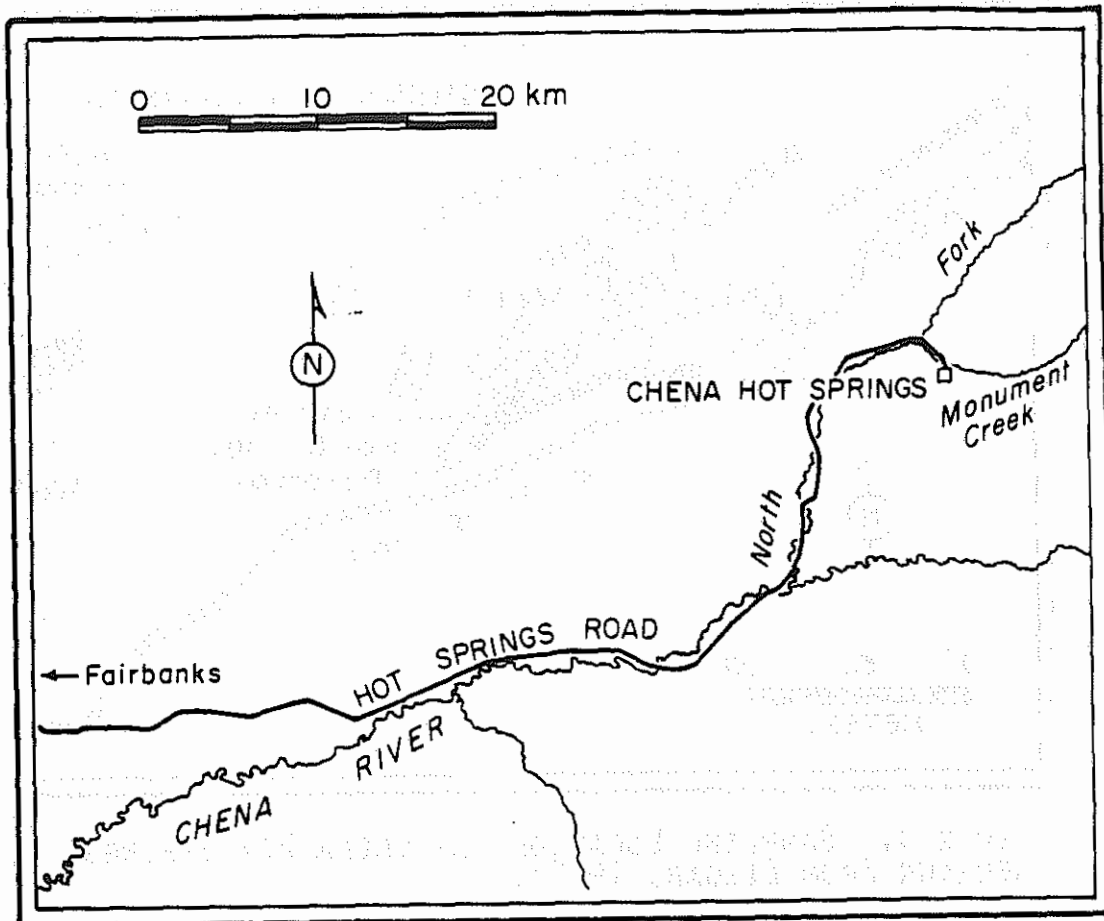


FIGURE 2. LOCATION OF CHENA HOT SPRINGS.

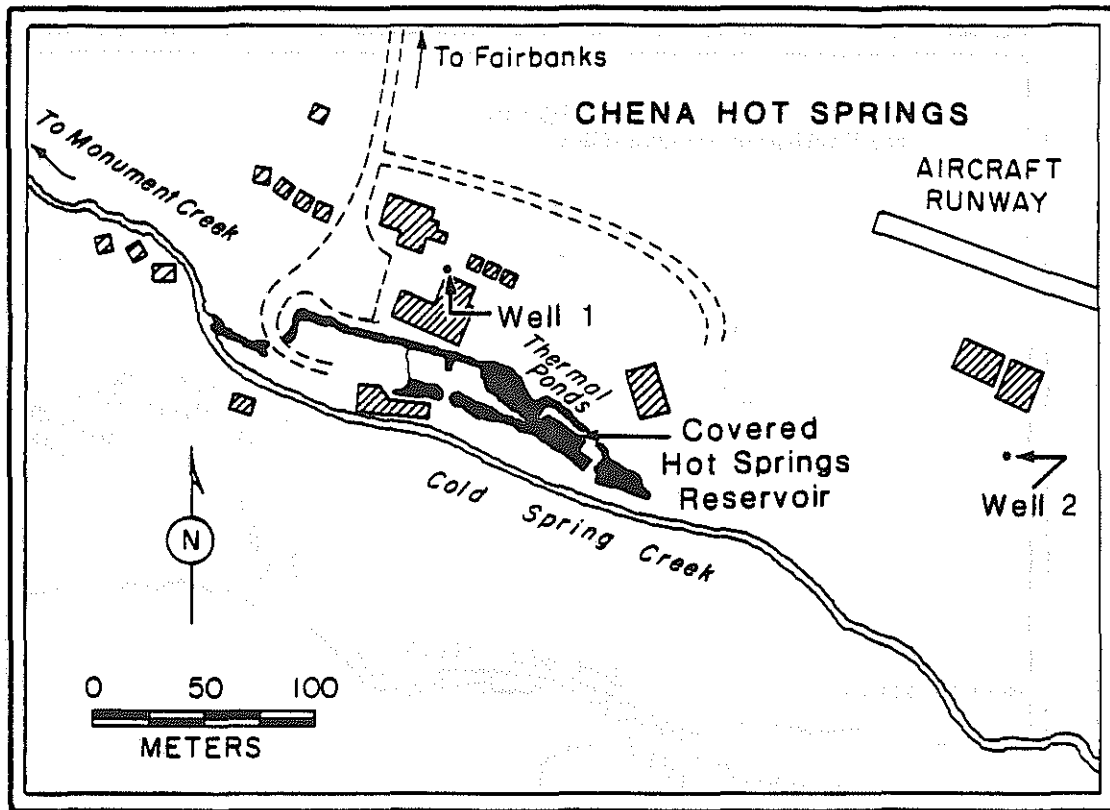


FIGURE 3. SAMPLING LOCATIONS AT CHENA HOT SPRINGS RESORT (REDRAWN FROM BIGGAR, 1972).

**GROUND WATER MONITORING, MODELING, AND  
DATA MANAGEMENT**

GROUNDWATER QUALITY IMPACT  
FROM A LARGE MUNICIPAL DRAINFIELD

by Robert E. Gilfilian 1/

ABSTRACT

In recent years large drainfields have been designed and constructed to meet sewage treatment and disposal needs of municipalities and rural communities. The environmental impact on groundwater aquifers resulting from the operation of large drainfields is not well understood. Very limited information is known about the fate and transport of contaminants discharged from drainfields into the subsurface, particularly for disposal facilities operating in cold regions.

The City of Wasilla is one of the first Alaskan municipalities to construct and operate a large drainfield facility, considered to be one of the nation's largest. The Wasilla drainfield site has a complex geological setting, underlain by several ground water systems and bordered by a wetland area.

The research described here presents the findings of the first 24 months of environmental field study on the drainfield facility undertaken since the start-up of the facility. The field study monitored the characteristics of the wastewater effluent and receiving ground water and surface water systems.

INTRODUCTION

Purpose

The purpose of the field study described in this report was to monitor the performance of the Wasilla Drainfield Facility during the first 24 months of operation. This study was performed under the EPA Project Performance Certification Program. The monitoring program was also conducted for permit compliance purposes required by the Alaska Department of Environmental Conservation Waste Disposal Permit. Wastewater effluent, drainfield beds, vadose (unsaturated soil) zone, ground waters and surface waters were monitored on a monthly basis for several physical and chemical parameters.

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## Description of Drainfield Facility

The Wasilla Drainfield Facility is used as the final treatment and disposal process on the Septic Tank Effluent Pumping (STEP) sewer system serving the City of Wasilla. The wastewater treatment and disposal facilities (FIGURE 1, Project Site Map) were constructed on the 40 acre project site which included flow measurement, clarifier, dosing chamber, 10-acre drainfield with 9 individual beds, septage receiving station, aerobic digester, sludge drying beds and control building.

The drainfield facility consists of 9 drainfield beds designed for a capacity of 440,000 gallons per day (gpd) based on a design soil loading rate of 1.5 gpd/sf and 67 percent redundancy. Each drainfield bed has a total infiltration area of 48,400 square feet, and contains a pressure distribution system designed to achieve uniform application of effluent throughout the bed area.

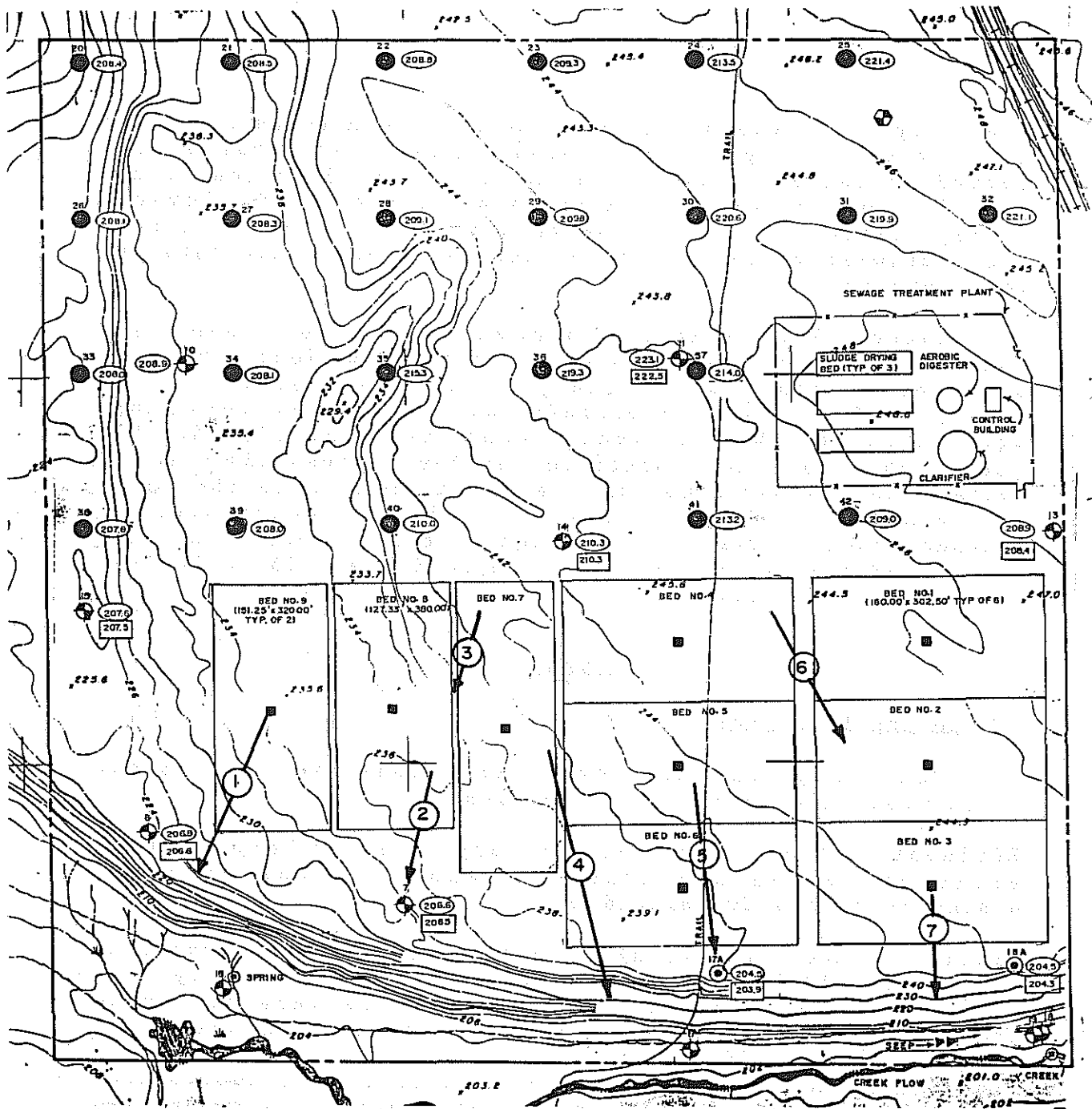
The drainfield distribution piping network consisted of 3-inch diameter polyethylene laterals spaced on 6-foot centers. The laterals have 1 1/64-inch orifices on 5-foot centers. The depth of the drainfield filter rock is 4.5-inches below the bottom of the distribution lateral. The drainfield filter rock is overlain with approximately 5-foot of earth fill. Only Beds No. 7, 8 and 9 had a 12-inch layer of select sand material placed directly below the bed's filter rock.

Each drainfield bed has 5 observation wells installed to the bottom of the drainfield filter rock. An observation well is located in each corner of the bed and one near the center of the bed. These wells are used to observe effluent ponding in the drainfield during a dosing event. In addition, each bed has a casing pressure-vacuum lysimeter and deep monitoring well, both located near the center of the bed as shown in FIGURE 3.

## Description of Hydrogeologic Conditions

The drainfield site is located on a small outwash terrace within a large distinct glacial outwash channel in an area of very complex geology that resulted from glacial deposition. The terrace on which the drainfield system was constructed is composed of outwash material (clean sandy gravels) overlying and interlaid with dense till deposits with possible complex ice contact deposits. FIGURE 2 shows a generalized cross sectional profile view of the subsurface soils and ground water levels underlying the project site.

Following the last period of major outwash deposition, a stream partially eroded the most recent deposit and cut through to underlying tills. This erosion process has left



| LEGEND |  |
|--------|--|
|        | DOMESTIC WATER SUPPLY WELL   |
|        | GROUNDWATER MONITOR WELL (CONSTRUCTED MARCH, 1986)   |
|        | GROUNDWATER MONITOR WELL (CONSTRUCTED OCTOBER, 1986)                                       |
|        | GROUNDWATER MONITOR WELL (CONSTRUCTED NOVEMBER, 1987)                                      |
|        | DRAINFIELD GROUNDWATER MONITOR WELL, PRESSURE-VACUUM LYSIMETERS & SHALLOW OBSERVATION WELL |
|        | SURFACE WATER SAMPLING LOCATION  |
|        | GROUNDWATER SEEP   |
|        | STATIC WATER LEVEL (MEASURED NOV. 1987)  |
|        | STATIC WATER LEVEL (MEASURED NOV. 1988)  |

GROUNDWATER GRADIENT & DIRECTION AS OF NOV. 1988.

- ① → 0.004
- ② → 0.005
- ③ → 0.007
- ④ → 0.010
- ⑤ → 0.010
- ⑥ → 0.008
- ⑦ → 0.009

FIGURE I.  
Project Site Map  
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the steep bluff which exists along the south boundary of the project site. A peat (wetlands) deposit has subsequently formed in the old channel. A small stream flows toward the east in a marshy area along the base of the bluff. Several springs daylight along the western portion of the bluff near the southwest corner of the project site and drain into the small stream.

Three (3) distinct groundwater regimes have been identified beneath the project site. A deep confined aquifer exists approximately 80 to 120 feet below the ground surface over the drainfields. A shallow unconfined aquifer exists 20 to 40 feet beneath the drainfields. The water table for the unconfined aquifer corresponds to the surface water elevations in the swamp located to the south and west of the drainfields. Isolated occurrences of perched ground water exist throughout the project area. The surface of the perched water is approximately 15 to 20 feet below the drainfields.

## METHODS

### Monitoring Locations

The designated monitoring locations used during the field study are described as follows:

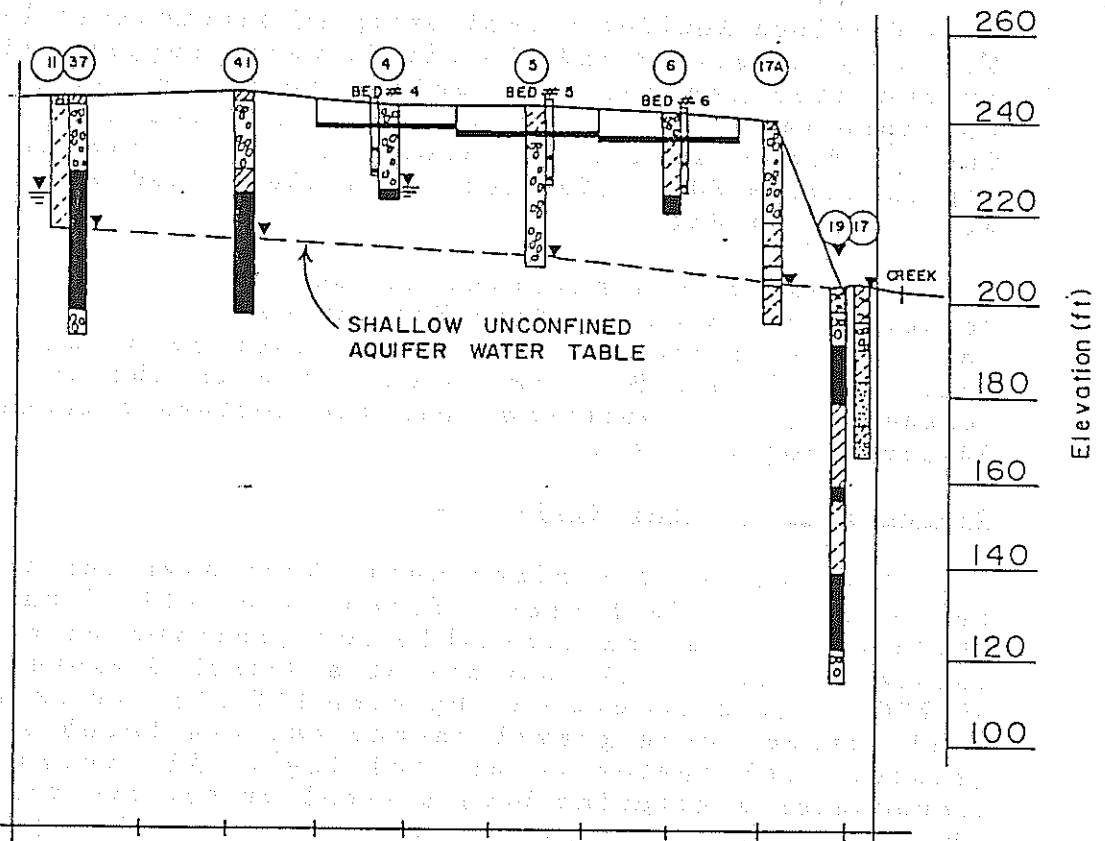
**Effluent** - Monitoring of primary treated septic tank effluent in the clarifier's dosing chamber prior to discharge to the drainfield beds.

**Drainfield Bed** - Monitoring of wastewater effluent ponded in the drainfield bed's filter rock located approximately 5 to 6 feet below the ground surface. Samples were collected from the drainfield's 4-inch diameter observation wells.

**Vadose Zone** - Monitoring of groundwater in the unsaturated soil zone at depths of 5 and 10 feet beneath the drainfield beds. Samples were collected with pressure-vacuum lysimeters located near the center of each drainfield bed.











**Perched Water** - Monitoring of intermittent groundwater condition that occurs on the dense glacial till strata located at a depth of approximately 20 feet below active drainfield Beds No. 1 and 9. Samples were collected from shallow monitor wells completed at the upper surface of the glacial till strata.

**Shallow Unconfined Aquifer** - Monitoring of groundwater condition in Monitoring Wells No. 6, 7, 11, 13, 14, 15, 17A and 18A. This groundwater aquifer is unconfined (water table) with a static water depth of 20 to 40 feet below the site's ground surface. This aquifer daylights in several springs at the base of the bluff adjacent to the stream.



SCALE: HORIZONTAL: 1" = 20'  
 VERTICAL: 1" = 40'

**LEGEND**

-  SILT
-  SANDY GRAVEL/GRAVELLY SAND
-  PEAT
-  SANDY GRAVEL W/TRACE SILT
-  SILTY GRAVEL, SANDY SILT, SILTY SAND
-  DRAIN ROCK IN DRAINFIELD BED
-  STATIC WATER LEVEL (REGIONAL AQUIFER)
-  LYSIMETER
-  MONITOR WELL NUMBER
-  STATIC WATER LEVEL PERCHED GROUNDWATER

**FIGURE 2.**  
**Drainfield Site**  
**Cross Sectional Profile**



Deep Confined Aquifer - Monitoring of groundwater in Monitor Wells No. 16 and 19 and the site's water supply well. This groundwater aquifer lies below the upper aquifer and is confined (artesian) by an overlaying glacial till strata. The confined aquifer's saturated soil zone is located approximately 80 to 120 feet below the ground surface of the drainfield beds.

Surface Water - Monitoring of surface water at two (2) locations shown on the Site Map, Appendix A. The first site consists of a groundwater spring located below the bluff south of Bed No. 9. The second site is the creek at the furthest point downstream near the southeast corner of the 40 acre project site.

#### Groundwater Monitor Wells

A total of 53 monitor wells have been constructed on the 40 acre project site. Forty-five (45) of these wells still exist and are available for groundwater monitoring purposes. Each well consists of a 2-inch diameter Schedule 40 PVC flush threaded casing with PVC slotted screen. The well screens were gravel packed and the borehole annulus grouted with bentonite or "Volclay". All wells used for groundwater monitoring have a steel protective casing above the ground surface with a locking cover.

#### Pressure-Vacuum Lysimeters

Soil pore water samples from the vadose (unsaturated) soil zone were obtained with pressure-vacuum lysimeters. These sampling devices use an in-situ technique that relies upon the creation of a vacuum to induce soil pore water to flow into a collection vessel. The vacuum-pressure casing lysimeters were manufactured by TIMCO Mfg., Inc. The filter media of the lysimeter's porous section is made of virgin teflon (PTFE) with a 70 microns average pore size. Casing lysimeters were installed to collect pore water samples from the unsaturated soil zone at depths of 5 and 10 feet below the bed's filter rock.

#### Sample Collection

Teflon and PVC bailers were used for sampling groundwater from low yield wells. Wells were bailed at least 3 bore volumes, or to dryness, prior to sampling.

Vadose zone sampling was accomplished by attaching an electrically operated vacuum pump to the sampling lines on the lysimeters and developing a vacuum of approximately 18 mm Hg. In most cases, a continuous vacuum was maintained for approximately 24 hours.

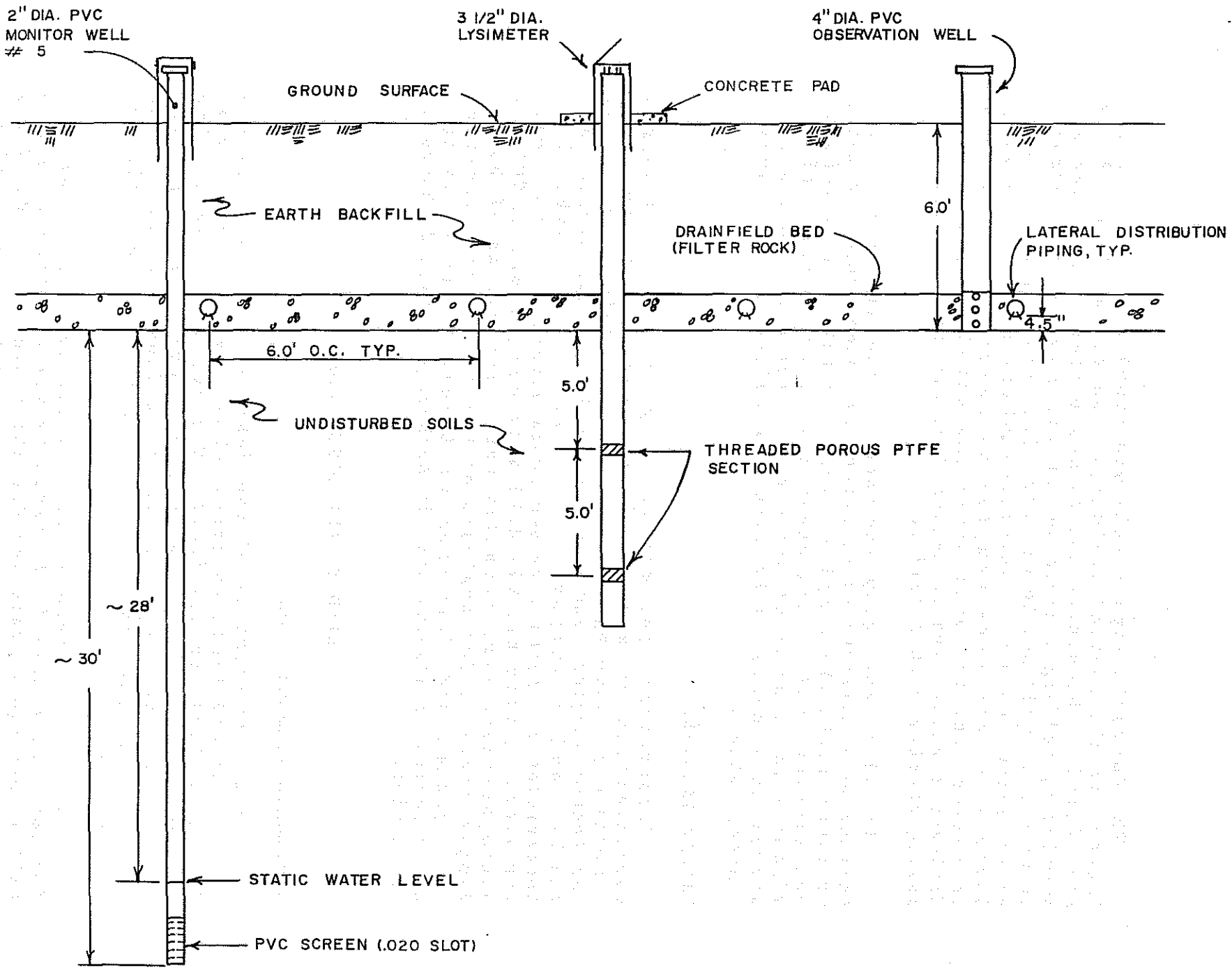


FIGURE 3.

Profile For Drainfield Bed # 5  
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The domestic supply well was sampled from a sink tap located in the control building. Monitor Well No. 19 was free flowing throughout the 24 month study period and was sampled by holding a container at the overflow and allowing it to fill. Well No. 16 was also free flowing but at times experienced low head conditions which required bailing. Surface waters were sampled by immersing the sample container into the open water and withdrawing a sample. Effluent samples were collected directly from the clarifier dosing chamber with a bailer.

In all cases, equipment used in obtaining and storing samples were constructed of materials which would not alter or contaminate the samples. Care was taken in sanitizing and rinsing bailers and other equipment in the field to minimize the chances of cross contamination. Samples were stored in a cooler subsequent to collection and during transportation.

### Sample Analysis

Temperature, pH, conductivity and dissolved oxygen concentration of a sample were measured in the field immediately following sample collection. Additional samples for bacteriological and chemical analyses were stored in a cooler and transported to Mat-Su Test Lab, Inc. Bacteriological tests were performed within 24 hours. Tests for nitrate, nitrite, Kjeldahl nitrogen, chloride, COD and orthophosphate were normally performed within 48 hours. Sample portions for analyses of the remaining parameters were packed and delivered to Chemical and Geological Laboratories of Alaska, Inc. in Anchorage within 24 hours of collection.

## RESULTS AND DISCUSSION

### Hydraulic Loading Rates

Since the start-up of the Wasilla STEP sewer system, the drainfield facility received an average flow rate of 76,302 gallons per day (gpd). The average incoming flow rate was only 17.3% of the facility's original design flow of 440,000 gpd. The daily influent flow measured at the clarifier ranged from a low of 55,613 gpd to a high of 94,866 gpd.

The overall average daily hydraulic application rate for the entire drainfield facility was approximately 0.74 gallons per square foot per day which was nearly 50% of the original design loading rate. In spite of the reduced loading rates, continuous ponding of effluent in the bed's filter rock was typically observed within a few months of the start-up of the bed. Although the operation of the beds did not cause any apparent hydraulic back-up problems, the

beds were technically operating in a state of failure, i.e., inability to completely drain of effluent between dosing events.

Anaerobic conditions were found in ponded effluent samples analyzed for dissolved oxygen. Although the actual loading rate for the drainfield beds was approximately 50% of the original design value, the formation of a clog mat resulted in hydraulic dysfunction and appeared to have developed within a relatively short period of time. The rapid formation of the clog mat may have been the result of the effluent's large BOD characteristic and the low ambient ground temperatures. Continuous ponding conditions of drainfield beds have been shown by other researchers to stimulate clogging of the soil infiltration surface, which eventually results in anoxic soil conditions.

During the month of July, 1988, test holes were excavated in Beds No. 2 and 5. At the time, Bed No. 2 had been shut down for nearly 4 months and Bed No. 5 was still in use. The investigation confirmed the existence of a thin bio-clog mat located at the base of the darkly stained filter rock. Bed No. 2 was dry and Bed No. 5 had effluent ponded above the effluent distribution pipe. Excavation through the clog-mat in Bed No. 5 allowed effluent to drain freely into the underlying unsaturated native gravelly sand soils.

#### Water Temperatures

The ground water temperatures of the shallow and deep aquifers were very similar with an overall average temperature of 37.9 degrees F. Ground water temperatures ranged from a low of 35.8 to a high of 44.6 degrees F, and were noted to fluctuate seasonally. The temperature of the spring was nearly the same as the ground water temperatures, whereas the creek's temperature, as expected, was colder during the winter and warmer during the summer compared to the ground water temperatures.

The temperature of the wastewater effluent was observed to fluctuate seasonally with similar values for 1987 and 1988. Over the 2 year period the mean temperature was 43.4 degrees F, with a range of values from a low of 35.8 to a high of 54.3 degrees F. Wasilla's effluent temperatures were generally several degrees lower than the temperatures of the wastewater entering the City of Palmer Sewage Lagoon which is located approximately 10 miles east of Wasilla. The wastewater temperatures during the summer were nearly the same at both cities; however, during the winter, Wasilla's temperature ranged from 10 to 15 degrees F cooler compared to Palmer's. Although the City of Palmer has over 2 miles of trunk line to their lagoon facility, Wasilla's lower temperature is probably caused by the residence time in the City's STEP system septic tanks.

Significant temperature increases were noted in the groundwater directly below the active drainfield beds. These groundwater temperatures were nearly the same as the wastewater effluent temperatures and typically were approximately 10 degrees F higher than the surrounding ambient ground water temperatures.

#### Groundwater Levels

Since the start-up of the drainfield facility, the groundwater levels remained very stable with minor fluctuations. Hydraulic impact from the discharge of wastewater via the drainfield beds did not cause mounding of the groundwater table beneath the beds. Perched intermittent ground water conditions did occur on the surface of the shallow confining glacial till strata located below Drainfield Beds No. 1, 4 and 9 as a result of operating these beds.

#### Wastewater Effluent Characteristics

A summary of the significant findings on the effluent characteristics during the 24 month study is described as follows:

**Total Suspended Solids** - The total suspended solids were noted to be relatively constant. The mean value over the 24 month period was 58 mg/l which corresponded with the findings of other research projects on septic tank systems. These findings indicated the septic tanks on the Wasilla STEP sewer system were relatively efficient for the removal of suspended solids.

**pH** - The pH value of the effluent was relatively constant with a mean value of 6.7 and ranged from a low of 6.3 to a high of 7.1.

**Chlorides** - The chloride concentration was noted to be similar to values found by other researchers studying septic tank systems. The mean chloride level was 44.7 mg/l and ranged from 30.0 to 58.0 mg/l. Previous research work in the lower 48 states observed typical chloride levels ranging from 37 to 101 mg/l in septic tank effluent.

**Conductivity** - As expected, the conductivity values fluctuated with changes in the effluent chloride levels. Over the 2 year period the mean conductivity level was 838 umhos/cm with a low value of 700 and a high of 960 umhos/cm.

**BOD5** - The biochemical oxygen demand (BOD5) of the effluent had a mean value of 293 mg/l during the 2 year study. This value is relatively high compared to a mean value of 152 mg/l found by other researchers on community septic tank sewer projects in the lower 48 states. Wasilla's effluent

had ranged from a low BOD5 of 224 mg/l to a high of 420 mg/l which is considered to be moderate to high strength wastewater. In comparison to the City of Palmer's wastewater BOD5 level, Wasilla's average BOD5 level is approximately 50% greater than Palmer. Also, the range of the monthly average BOD5 of the wastewater entering the City of Palmer's lagoon are only 76 mg/l compared to Wasilla's range of nearly 200 mg/l.

COD - The effluent's chemical oxygen demand (COD) was noted to correlate with the BOD5 values. Typically the BOD5 value was approximately 60% of the COD value. Similar to the BOD5 values, the 466 mg/l mean COD value of Wasilla's effluent was considerably higher compared to the typical values of 278 mg/l found in the lower 48 states research projects. Wasilla's COD levels ranged from a low of 273 mg/l to a high of 575 mg/l.

Nitrogen - The total nitrogen level had a mean value of 45.6 mg-N/l with a range of 31.7 to 79.0 mg-N/l. These values corresponded very well with the total nitrogen levels of 45 mg-N/l found in the lower 48 states research project on community septic tank systems. The ammonia nitrogen value was approximately 75% of the total nitrogen concentration in the effluent.

Phosphorus - The total phosphorus level was very stable with a mean value of 9.7 mg-P/l and compared very well with the 10 mg-P/l level reported in lower 48 states research projects.

DO - The dissolved oxygen of the effluent value was found to be less than 0.5 mg/l. These values are typical for septic tank effluent; however, it is interesting to note that neither the apparently long transit time in the pressurized STEP sewer main nor the additional time in the clarifier did not appreciably increase the dissolved oxygen level in the effluent.

Inorganic Chemicals - With the exception of a trace amount of arsenic, mercury and silver, the inorganic chemicals in the effluent were below detection limits. However, these results were based on grab samples collected on only 2 occasions, and may not be representative of the effluent characteristics.

Purgeable Aromatic Hydrocarbons - The hydrocarbons were relatively absent in the effluent during the last 12 months of monitoring with the exception of toluene. The same concern expressed about the 2 sampling events for inorganic chemicals applies equally for the concentration of PAH in the effluent. The test results from the initial 12 months of study showed an appreciable amount of ethylbenzene, toluene and xylene in the effluent.

## Vadose Zone

An intensive field monitoring effort was undertaken to obtain performance data on the characteristics of wastewater effluent as it was dosed into the drainfield bed and percolated through the vadose zone (unsaturated soil) to the underlying groundwater table. This monitoring effort involved a coordinated sampling program for the simultaneous collection of samples representative of a dosing event at the following locations:

- Effluent from the clarifier dosing chamber.
- Poned effluent in the drainfield bed's filter rock via the 4-inch diameter observation well.
- Soil pore water-in the vadose zone at depths of 5 and 10-feet beneath the bed's filter rock via the vacuum-pressure lysimeters.
- Groundwater in the saturated soil zone located directly beneath the center of the active drainfield bed via the 2-inch diameter groundwater monitor well.

A summary of the significant findings on the vadose zone characteristics during the 24 month study period is described as follows:

**Dissolved Oxygen** - The dissolved oxygen level found in the effluent, vadose zone and groundwater located beneath the active drainfield bed was typically below 1.0 mg/l. These values indicated anaerobic conditions prevailed during dosing events. The groundwater aquifers surrounding the active beds were found to have higher dissolved oxygen levels which indicated aerobic groundwater conditions.

**Chlorides** - On several occasions the level of chlorides decreased by nearly 50% through the vadose zone. This finding, however, was not consistent as groundwater chloride levels were found at other times to be nearly the same as the effluent chloride levels. As a rule, the vadose zone was ineffective for the removal of chlorides.

**Temperature** - The water temperatures of samples collected during dosing events were nearly the same throughout the vadose zone. These observations indicated rapid infiltration rates since the temperatures of the applied effluent in the bed and underlying receiving ground water system were nearly the same.

**Nitrogen** - Two contrary findings on nitrification processes were noted during the vadose zone monitoring program. The first finding involved the study of nitrification in Bed No. 5 which had been in use for over 1 year and developed a clog mat over the entire bed infiltration area. Ammonia levels in the bed's observation wells, lysimeters and groundwater monitor well were nearly the same, which indicated no

evidence of nitrification. However, as found in Monitor Well 17A and Bed No. 4 the nitrification process occurred as effluent traveled laterally from the bed in the ground water, since appreciable levels of nitrates were found in these wells. The second finding involved Beds No. 1 and 9. At the time of vadose monitoring, Bed No. 1 was a new bed and Bed No. 9 was relatively new with a small area that had developed a clog mat. Ammonia levels in the perched groundwater below these beds were below 0.6 mg/l compared to typical effluent ammonia levels ranging from 16 to 37 mg/l. Total nitrogen levels of the effluent decreased by 80 to 98 percent compared to the groundwater levels. Based on the limited analyses of lysimeter samples, nitrification did occur on certain occasions as indicated by the level of nitrates. Conclusive evidence was not obtained to develop a definitive explanation for the reduced levels of nitrogen in the perched ground water below Beds No. 1 and 9.

Fecal Coliform Bacteria - On nearly all vadose zone sampling events the fecal coliform bacteria was found in the ground water directly below the active beds. The levels of fecal bacteria in the groundwater ranged from a low of 2 to a high of 330,000 colonies per 100 ml. This finding is significant since microorganisms were being transported through unsaturated soil conditions a vertical distance of up to 28 feet. This finding indicated rapid flow conditions during dosing events that resulted in minimal treatment in the vadose zone. Insufficient data was obtained to determine the effectiveness of the sand filter layer placed below Bed No. 9 filter rock compared to the unlined Beds No. 1 and 5.

Biochemical Oxygen Demand - During 2 dosing events the effluent's BOD5 level decreased significantly as the effluent drained from the bed through the vadose zone to the perched groundwater zone. The BOD5 levels of the groundwater were reduced by 93 to 99 percent compared to the applied effluent levels. As found by other researchers of rapid infiltration systems, a large percentage of the BOD5 in primary sewage effluent is contained in the particulate matter. The suspended particles are physically removed by the soil filtration process within the first 150 cm.

#### Shallow Unconfined Aquifer

A summary of the significant findings during the 24 month study on the characteristics of the shallow unconfined aquifer is described as follows:

Chlorides - The chloride levels in monitoring locations surrounding Beds 5 and 9 had increased significantly over the background levels. The flow of effluent from Bed No. 9 appears to have affected the chloride levels in the spring which is located immediately downgradient of the bed.



Monitor Wells No. 17A and 18A had a significant amount of chlorides that were similar to the levels found in the effluent applied to Beds No. 2 and 5. The presence of the high level of chlorides in the monitor wells compared to the background levels of the aquifer indicated the disposal of effluent in the drainfield beds had directly impacted the receiving groundwater system. Chlorides are not absorbed by soil formations and often have been used as tracers in groundwater studies.

Conductivity - The conductivity levels in the spring and Monitor Wells 6, 15, 17A and 18A had risen during 1988 compared to earlier lower measurements. The conductivity value is directly influenced by the amount of chlorides in the water and corresponds with changes in the level of chlorides. These downgradient wells appear to be directly influenced by the discharge of effluent in the drainfield beds.

Chemical Oxygen Demand - The shallow aquifer's COD levels were not consistent although there appeared to be an increase in COD values in nearly all monitoring locations during the months of June and July, 1988. Interpretation of these results were therefore not conclusive. However, COD levels were considerably higher than the background levels indicating an impact from the drainfield activity.

Nitrogen - The total nitrogen level in the downgradient monitor wells had increased over the background levels. Monitor Well No. 17A was the only shallow aquifer well that had an appreciable amount of nitrates, with a high value of 6.5 mg/l. The ammonia-nitrogen level in nearly all of the shallow aquifer wells had increased over the background levels.

Fecal Coliform Bacteria - Except for the monitor well in Drainfield Bed No. 5, only 2 wells representative of the shallow aquifer had fecal bacteria found in them. Well No. 13 had fecal bacteria of 1 and 4 colonies per 100 ml in the months of July and September, 1988. During August, 1988 the clarifier experienced a hydraulic failure that caused wastewater influent to pond on the ground surface located near Well No. 13 which may have influenced the September finding. Well No. 17A had 5 colonies per 100 ml in September 22, 1988.

Purgeable Aromatic Hydrocarbons - Except for the isolated occurrence of toluene in Monitor Wells No. 7, 13 and 15 and benzene in Well No. 15, the shallow aquifer did not have any detectable amounts purgeable aromatic hydrocarbons. The finding of toluene in the above wells indicates the source of these hydrocarbons may have come from the drainfield beds since the influent did have appreciable amounts of toluene. Absorption of toluene in the sandy soils underlying the

drainfield beds would be expected to be minimal due to the low organic content of these soils.

Inorganic Chemicals - Chromium levels above the State's Drinking Water Standards were measured in Monitor Wells No. 6, 7, 11, 15, 17A and 18A. Levels of lead near or above these standards were found in Monitor Wells No. 7, 17A and 18A. The levels of these inorganic chemicals increased over the background levels.

#### Deep Confined Aquifer

A summary of the significant findings on the characteristics of the deep confined aquifer is described as follows:

Chlorides - The level of chlorides in the deep aquifer has not changed since the start-up of the drainfield facility. This finding is significant since it shows the effluent discharge from the drainfield has not caused an impact with respect to chlorides on the deep aquifer as it did to the shallow aquifer.

Conductivity - Except for a one time (September 1988) measurement of a high conductivity level in Monitor Well No. 16, the aquifer's conductivity level has remained near background levels since the start-up of the drainfield facility. As expected, this finding is consistent with the chloride levels reported above.

Chemical Oxygen Demand - The COD level of the deep aquifer had increased in all wells when compared to the background levels. These COD values were considerably lower than the levels found in the shallow aquifer. Also, the upgradient well (water supply well) in the deep aquifer had similar changes in COD value as found in the downgradient Monitor Wells No. 16 and 19. Hence, the COD levels reported for the deep aquifer may be representative of natural conditions.

Nitrogen - During the last 7 months of this study, all of the wells in the deep aquifer had ammonia-nitrogen levels greater than the background levels. Monitor Well No. 19 had the highest ammonia measurement (1.6 mg/l) of all the wells representing the shallow and deep aquifers. Increases in the ammonia level were recorded in both upgradient and downgradient wells. These findings are not consistent with the findings of the chloride levels but are similar to the pH changes. An explanation for these inconsistent findings is not available at this time, and therefore, should be investigated further.

Dissolved Oxygen - It is interesting to note that the dissolved oxygen level of the deep aquifer is several mg/l lower compared to the shallow aquifer. This finding should be expected due to the lack of oxygen transfer available for

the deep aquifer, i.e., confined groundwater aquifers are sealed by overlying nearly impermeable soil stratas that prevent (restrict) oxygen movement. Hence, as indicated by the DO levels in the deep aquifer wells the groundwater is nearly anaerobic with a trace of dissolved oxygen.

### Surface Waters

A summary of the significant findings in the characteristics of the surface waters is described as follows:

**Chlorides** - The chloride level in the spring increased nearly 100% during the months of August, September and October 1988 compared to previous 20 months of measurements. It was 300% higher than the background measurements, whereas, the chloride levels in the creek remained relatively constant for over a year with a value slightly higher than the background measurements.

**Chemical Oxygen Demand** - The COD level of the spring increased dramatically during the months of June and July, 1988, with the July measurement over 100% higher than the background level. The creek's COD level remained relatively constant since the start-up of the drainfield facility.

**Fecal Coliform Bacteria** - The creek's fecal coliform level increased during the winter of 1987-88. During this period of time the moose population increased in this area as noted by their bedding areas located along the creek bank. The spring was free of any fecal coliform bacteria contamination.

### CONCLUSIONS

Wastewater quality of the septic tank effluent and hydraulic loading rates experienced by the Wasilla Drainfield Facility during the first 24 months of operation caused rapid development of soil clogging in the absorption beds. The resultant soil clogging phenomenon significantly affected hydraulic and purification performance. Although the actual hydraulic loading rate was nearly 50% less than the design rate, the strength of the septic tank effluent characterized by BOD5 and total suspended solids coupled with low temperatures may have contributed to premature hydraulic failure of the drainfield beds.

The disposal of septic tank effluent at the Wasilla Drainfield Facility has resulted in some degradation of the ground water under and just downstream of the drainfield beds. Operation of the drainfield facility resulted in violation of the State of Alaska Water Quality Standards which has established a policy of non-degradation.

Anaerobic conditions were detected beneath beds that had developed excessive soil clogging. Despite the existence of a relatively deep unsaturated soil zone beneath the beds, anoxic conditions were noted to prevail as indicated by low dissolved oxygen levels and high levels of ammonia-nitrogen in samples collected from the vadose zone and underlying ground water. Effluent from the drainfield beds infiltrated rapidly through the deep unsaturated (vadose) soil zone into the underlying groundwater table. The lack of appreciable retention time indicated that treatment was minimal and anaerobic conditions prevailed. Microorganisms were being transported to the groundwater system.

#### RECOMMENDATIONS

The results of this field study have raised concerns about the conventional method of designing large drainfields, particularly for those operating in cold regions. A rational design practice should be established to evaluate the quality of the applied wastewater effluent with regard to a hydraulic loading rate that would achieve development of an acceptable soil bio-mat. According to the findings of recent research on septic tanks, a higher degree of quality of the effluent that is applied to the drainfield bed allows for a higher hydraulic loading rate.

The size of the existing Wasilla Drainfield Facility would have to be increased by 800 to 1,000 percent in order to adequately treat and dispose of the original design flow of 440,000 gpd of septic tank effluent having the quality found in this study. Based on current research findings, the size of the existing drainfield facility could remain as is to handle the design flow of 440,000 gpd provided the septic tank effluent is pretreated to reduce the levels of BOD<sub>5</sub> and TSS to 5 and 10 mg/l, respectively.

# **WATER QUALITY MONITORING AT SOLID WASTE LANDFILLS: THE ANCHORAGE EXPERIENCE**

by J. Brett Jokela<sup>1</sup> and W. James Sweeney<sup>2</sup>

## **ABSTRACT**

The Municipality of Anchorage presently operates one solid waste disposal facility, the Anchorage Regional Landfill, located on Hiland Drive near Eagle River. This membrane lined facility replaces four older public landfills, now closed. The ongoing Landfill Water Quality Monitoring Program involves collecting, compiling and evaluating data from monitoring wells, surface water sites and leachate collection points at each of these landfills.

The development of the monitoring program has involved careful planning including consideration of historic data, establishment of monitoring objectives, and recognition of hydrogeologic features and existing monitoring locations for each site. Implementation of the monitoring program has been performed according to procedures manuals developed for sampling, laboratory analysis, and data management. Preliminary results of the initial year of operation under the comprehensive monitoring program are reviewed.

## **INTRODUCTION**

Concerns on the impacts of solid waste landfills on water quality have increased significantly in recent years, both in Alaska and nationally. This is demonstrated by the fact that 21% of the 770 sites now on the Superfund National Priority List (NPL) are closed or active municipal solid waste landfills. Most of these sites are on the NPL due to problems with leachate migrating off-site and contaminating ground or surface waters.

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## Regulatory Concerns

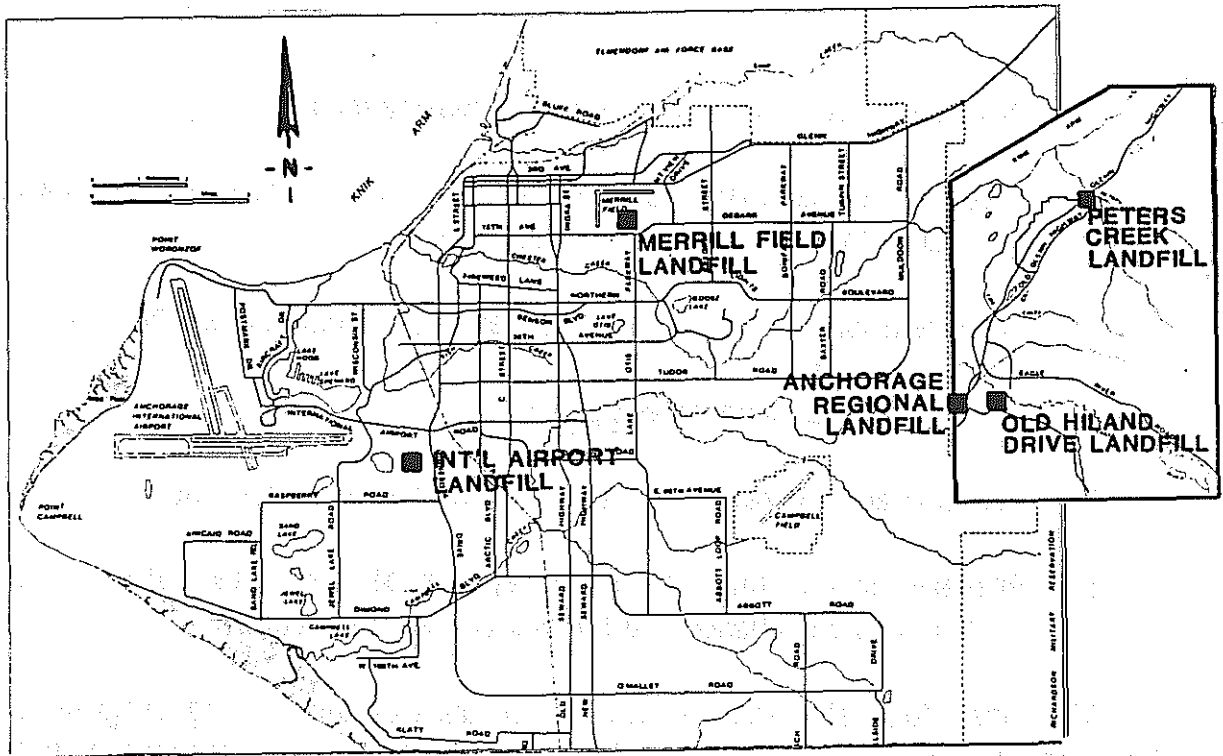
Solid waste landfills are regulated nationally by the U.S. Environmental Protection Agency (EPA). In 1979, EPA promulgated criteria under Subtitle D of RCRA (40 CFR Part 257) that established minimum performance standards for both new and existing solid waste landfills. The existing regulations contain only general performance based criteria on groundwater contamination and monitoring requirements.

The State of Alaska has developed and adopted regulations in Chapter 60 of the Alaska Administrative Code (AAC). The 1987 Alaska Department of Environmental Conservation (ADEC) regulations are significantly more detailed than the EPA Subtitle D regulations, particularly for establishing landfill monitoring requirements. For landfills serving over 2000 people, the regulations require at least one (1) upgradient and two (2) downgradient monitoring wells and additional wells, if determined necessary. In addition, the ADEC regulations establish a list of sample parameters and frequencies for possible monitoring. The actual number and location of monitoring wells and sampling parameters and frequencies are established in the Solid Waste Operating Permits issued by ADEC.

In August 1988, EPA proposed revised criteria for municipal solid waste landfills. The proposed regulations continue to use a general performance based standard, but are significantly more detailed and stringent, particularly in the area of monitoring requirements. The revisions propose a two-phased groundwater monitoring system and corrective action requirements to ensure that groundwater contamination at landfills will be detected and cleaned up. In Phase I, monitoring is required at least semi-annually and the list of parameters is similar to that contained in the current ADEC regulations. If contamination was detected, the Phase II monitoring requirements would include the Phase I parameters, plus more than 200 additional parameters contained in an appendix of hazardous constituents. Corrective actions would be required at a landfill if the Phase II constituent levels were exceeded.

## Anchorage Landfill Concerns

In Anchorage, there are five landfills of concern to the Municipality from a monitoring standpoint. The locations of these sites are shown on the map of Anchorage below. Only the Anchorage Regional Landfill is currently active. The other four (Merrill Field Landfill, International Airport Road Landfill, Peter's Creek Landfill and Old Hilland Road Landfill) are closed.



Water quality monitoring has been conducted at most of these landfills on a periodic or special studies basis for more than 10 years. A considerable amount of water quality data has been collected in the past, particularly for the Merrill Field and International Airport Landfills. However, there has been little documentation of the sampling and laboratory procedures and a wide variety of different people and firms have done the sampling and analysis. In the past, the procedure used to select the firm to do sampling and analysis was low bid. As a result, there is a lack of consistency and documentation of methodologies used and the value of the data collected is questionable.

With the increased emphasis on potential environmental problems that can be caused by solid waste landfills and the huge expenses that can be required for any corrective actions, the Solid Waste Services Department determined that major changes in the landfill water quality monitoring program were necessary. In November, 1986, the Municipality of Anchorage issued a Request for Proposals (RFP) to contract with a firm to help the Solid Waste Services Department establish a consistent and defensible long-term landfill water quality monitoring program. James M. Montgomery, Consulting Engineers, Inc. (JMM) was selected to design and carry out the landfill water quality monitoring program.

### APPROACH TO PROGRAM IMPLEMENTATION

There were six (6) major program elements identified for the landfill water quality monitoring program:

1. Develop a long-term landfill water quality monitoring plan;
2. Prepare a sampling procedures manual, including a quality assurance/quality control (QA/QC) plan;
3. Prepare a laboratory procedures manual, including a QA/QC plan;
4. Develop a data management plan;
5. Perform actual water quality sampling and analysis; and
6. Prepare annual interpretative reports of the data collected and recommend any changes to the water quality program.

The first four elements of the water quality monitoring program are discussed in this section and the last two elements are considered in the sections on Program Implementation and Results.

#### Water Quality Monitoring Plan

The first step in developing a long-term landfill water quality monitoring program was to review and evaluate existing conditions at each of the five landfills. Well log data was collected for all landfills and any available reports on groundwater conditions in the vicinity of each landfill were obtained and reviewed. The location and condition of all existing monitoring wells were evaluated to determine which wells may be acceptable for use in the monitoring program and to determine if modifications to existing wells were necessary or if additional wells were needed. General approximations of the hydrogeologic conditions at each landfill were then made.

The next step in developing the monitoring program was to compile a comprehensive inventory of all water quality data that had been collected in the past. A water quality data catalog was developed that included existing data from 45 water quality monitoring wells, six public and private water wells, and numerous surface and leachate monitoring reports. Approximately 500 separate water quality data reports were examined and categorized. A "quick and dirty" check on the internal consistency and general reliability of the existing data was conducted by applying quality control tests, e.g. major cation and anion balance within 5%. For the most part it was found that the existing water quality data was of questionable value due to insufficient documentation or failure to meet the "quick and dirty" quality control check



criteria. Even though much of the existing data was questionable, some general observations or trends in the results were apparent for several of the landfills.

Based on the review of existing hydrogeological conditions at each landfill and of the existing water quality data, it was then possible to develop recommendations for monitoring at each of the landfills. The objectives of the monitoring were to meet regulatory requirements and to determine if the landfills may be causing any environmental problems. Recommendations were made on the specific monitoring stations to be included in the long-term program, for the parameters to be analyzed at each sampling station, and the sampling frequency for each station.

### Sampling Procedures Manual

A detailed Sampling Procedures Manual [1] was prepared to describe the equipment, methodologies, and analytical procedures to be used in collecting water samples from surface waters, groundwater, and leachate systems at each of the five landfill sites. The purpose of the sampling procedures manual was to provide a "cook book" that could be used by all field personnel involved in the preparation, collection, and subsequent handling of samples. The manual identifies the responsibilities of the sampling personnel and the path for the flow of data from the time of sampling to the return of laboratory reports.

The Sampling Procedures Manual includes a description of each of the landfill sites, the monitoring stations, and procedures for access to each of the sites or stations. Maps showing the location of each site and monitoring station are provided.

Specific detailed sampling methodologies are described for taking samples from monitoring wells, surface waters, and leachate systems. The manual documents procedures that are to be followed in the advance of sampling, during sampling, and after sampling. Sample preservation, transport, and storage procedures and the chain of custody procedures are described. Field note forms, sample label formats, chain of custody forms, field sample checklists, and equipment checklists are included. The quality control sampling requirements are identified, including trip blanks, field equipment blanks, duplicate samples, split samples, and handling procedures for QA/QC samples. Recordkeeping requirements are also documented.

## Laboratory Procedures Manual

The objective of the Laboratory Procedures Manual [2] is to insure consistency and reproducibility of the analytical results obtained in the water quality laboratories. The manual outlines procedures to be utilized in contracting laboratory services, establishes procedures to be used by the laboratory in handling samples and analytical reports, identifies acceptable test methods, defines detection limits and documents QA/QC performance requirements.

An independent laboratory subcontractor is used to perform all laboratory analysis in order to maintain a clear division of responsibilities between the field work (JMM) and the laboratory analysis. The laboratory subcontractor is selected through a competitive bidding process on three criteria: (1) costs; (2) performance record; and (3) local preference. The selection of the laboratory contractor should not be based solely on low dollar bid.

In many cases there are more than one approved test method for the same parameter and it is also possible to have different detection limits for the same test method. To maintain consistency, the Laboratory Procedures Manual specifies test methods and required detection limits. The manual also establishes performance criteria which must be met at any time during the contract term. Failure to demonstrate adequate performance could be cause to terminate the contract.

## Data Management Plan

The purpose of the Data Management Plan [3] is to provide a computerized system for storage and retrieval of water quality data collected at monitoring stations. The system design provides automated retrieval of data for production of user specified reports, tables, or graphics.

Data is entered at a microcomputer workstation, utilizing customized programs developed with dBASEIII+ software. dBASEIII+ programs were compiled to executable versions with "FOXBASE+", which expedites considerably the operation of the database management system. The dBASEIII+ file structure is preserved, so that the data can be shared between various users of this popular software. The database features a very flexible system of report generation, including preparation of files for importing into a graphics software package.

As with any software product, documentation of the database management system was provided. This included step by step instructions for

use, descriptions of each file and record created or used by the system, logic diagrams and complete listings of copyrighted program code.

## PROGRAM IMPLEMENTATION

### Selection of Monitoring Stations

The five landfills coming under the Municipality's jurisdiction vary considerably in age, historic management and available documentation. Various monitoring practices left an array of monitoring wells and stations of varying quality. It was decided to focus on existing monitoring stations wherever possible to minimize startup costs for implementation of the program. A brief review of each landfill and its monitoring sites is outlined below.

Merrill Field. In the 1950's, open dumping of refuse out at the edge of the airstrip at Merrill Field came to be controlled by the City of Anchorage, and sanitary landfilling practices were instituted. In time, fill placement extended into adjacent wetland areas abutting the North Fork of Chester Creek. By the early 1970's it became apparent that the garbage placement would affect the local hydrology. The North Fork of Chester Creek was channelized and placed in corrugated metal conduit pipe so that filling could take place without affecting drainage. Subsequent development upstream of the landfill along the creek has led to enclosure of virtually all of the contributing drainage in underground conduits.

In 1974, a subdrain was added to lower the water table and prevent refuse from saturation as the hydraulic gradient between the airfield and the creek responded to the placement of fill. Two water supply wells operated by the city owned water utility were located in the area near the landfill. Concern for potential contamination of public water supplies from landfill leachate led to establishment of several monitoring wells near the landfill. Data from these wells were utilized in a 1982 USGS report [4] which estimated that vertical migration of leachate from the landfill would be limited by silty "Bootlegger Cove" soils some depth underneath the landfill. Subsequently, a more recent USGS study [5] using geophysical techniques to demonstrate potential leachate migration from the landfill site, into wetlands abutting the creek, downgradient from the site.

Several of the wells established by USGS are being used in the ongoing program. These are summarized below:

Table 1. Monitoring Stations at Merrill Field Landfill

| STATION I.D. | CONDITION      | WATER SCREEN |        |       | MAT'L | SECURITY**    |
|--------------|----------------|--------------|--------|-------|-------|---------------|
|              |                | LEVEL*       | LENGTH | DIAM. |       |               |
| CL8          | Upgradient     | 107'         | 5'     | 2"    | PVC   | Good          |
| CL1          | Pierces fill   | 81'          | None   | 6"    | Steel | Good          |
| CL5          | Downgradient   | 81'          | 3'     | 4"    | PVC   | Good          |
| CL16         | Downgradient   | 76'          | 10'    | 2"    | PVC   | No surf. seal |
| CL18         | Downgradient   | 74'          | 12.5'  | 2"    | PVC   | No surf. seal |
| CL101        | Creek upstream | -            | -      | -     | -     | Manhole       |
| CL110        | Creek downstr. | -            | -      | -     | -     | Open ditch    |
| CL201        | Subdrain       | -            | -      | -     | Steel | Wet well      |

\*Typical elevation of water in well column, ft. above MSL.

\*\* General comments on monitoring well construction.

International Airport Road Landfill. This landfill is located near the intersection of International Airport Road and Minnesota Drive. Like Merrill Field, this landfill grew from an open dump to a controlled sanitary landfill in the 1950's. This facility is located in a wetland upgradient from Connors Lake. The landfill was closed in 1977, and is now shared by a public works maintenance facility and public recreation playfields.

Concern for potential contamination from leachate led to establishment of monitoring wells within the landfill area. A USGS study [6] has demonstrated a regional east to west gradient in the piezometric surface of a deep confined aquifer. The local wetland bog groundwater hydrology is complex, although the water balance appears to be controlled by precipitation and evapotranspiration. Monitoring stations in use include some wells constructed specifically for detection of leachate as well as some relatively poorly protected water level observation wells remaining from USGS studies. The stations in use are listed below:

Table 2. Monitoring Stations at International Airport Road Landfill

| STATION I.D. | CONDITION     | WATER SCREEN |        |       | MAT'L | SECURITY**    |
|--------------|---------------|--------------|--------|-------|-------|---------------|
|              |               | LEVEL*       | LENGTH | DIAM. |       |               |
| BL14         | Upgradient    | 81'          | 2'     | 2"    | PVC   | No seal/guard |
| BL11         | Pierces fill  | 65'          | None   | 6"    | Steel | No seal/guard |
| BL 7         | Downgradient  | 75'          | 8'     | 4"    | Steel | Good          |
| BL17         | Downgradient  | 80'          | 2'     | 2"    | PVC   | No seal/guard |
| BL 9         | Downgradient  | 77'          | 4'     | 4"    | Steel | Good          |
| BL101        | Lake Downgra. | 75'          | -      | -     | -     | Surface grab  |

\*Typical elevation of water in well column, ft. above MSL.

\*\* General comments on monitoring well construction.

Peter's Creek Landfill. Peter's Creek Landfill is located in an abandoned gravel borrow site near the Old Glenn Highway. The landfill was operated from 1981 until 1987. Upgradient monitoring is accomplished through use of a deep water supply well. Several 6" steel wells were constructed in 1984. An attempt was made in 1988 to utilize one of these wells as a downgradient monitoring well, but a number of problems precluded practical utilization of this well. Problems included the large volume of water in the 6" diameter casing, the rusting steel casing, and the lack of a surface or annular seal. A new well was constructed in 1987 utilizing current Municipal specifications, including a locking cap on a steel guard casing, a concrete pad above the ground surface near the well casing, carefully placed screen at the piezometric surface, annular grouting and bentonite sealing, and a 2" PVC well with dedicated teflon and stainless steel positive displacement bladder pump. The monitoring stations are listed below:

Table 3. Monitoring Stations at Peter's Creek Landfill

| WATER SCREEN |              |        |        |       |       |               |
|--------------|--------------|--------|--------|-------|-------|---------------|
| STATION I.D. | CONDITION    | LEVEL* | LENGTH | DIAM. | MAT'L | SECURITY**    |
| PL1          | Upgradient   | 279'   | 11'    | 6"    | Steel | No surf. seal |
| PL5          | Downgradient | 270'   | 20.5'  | 4"    | PVC   | Good          |

\*Typical elevation of water in well column, ft. above MSL.

\*\* General comments on monitoring well construction.

Old Hiland Road Landfill. The Old Hiland Road Landfill was operated from 1964 until 1977. It is located at an old borrow site situated between sandy gravel ridges. The lower edge of the landfill encroaches on a wetland area, among springs tributary to Eagle River some two miles away. The springs occur at the interface between the porous kame deposits and silty depressions derived from glacial impoundments.

Table 4. Monitoring Stations at Old Hiland Road Landfill

| WATER SCREEN |              |        |        |       |       |               |
|--------------|--------------|--------|--------|-------|-------|---------------|
| STATION I.D. | CONDITION    | LEVEL* | LENGTH | DIAM. | MAT'L | SECURITY**    |
| HL2          | Downgradient | 360'   | 15'    | 2"    | PVC   | No surf. seal |
| HL101        | Surface      | 375'   | -      | -     | -     | Spring        |

\*Typical elevation of water in well column, ft. above MSL.

\*\* General comments on monitoring well construction.

Two monitoring wells were installed by the Municipal Department of Health and Human Services. One is in the toe of the landfill itself, while the other lies several hundred feet downgradient, in the wetland area. Each has a very long screen depth, so that water found in the monitoring well is likely to be strongly influenced by surface waters in the wetland. The downgradient

well and the spring waters are being monitored.

Anchorage Regional Landfill. This facility was opened in November of 1987 as the first membrane-lined municipal disposal facility in the State of Alaska. It is located west of the Glenn Highway at Hiland Drive on sandy gravelly soils. Background monitoring wells had been constructed in 1985, but construction of the downgradient wells was delayed until after operations at the landfill began. Each of the wells utilizes a dedicated bladder pump and other details as described for the new well at Peter's Creek.

A spring feeding Fossil Creek, several thousand feet downgradient of the landfill, is also being monitored. Leachate is collected and trucked to a local wastewater treatment plant. Sampling of leachate is done in conjunction with leachate hauling.

Table 5. Monitoring Stations at Anchorage Regional Landfill

| STATION I.D. | CONDITION      | WATER SCREEN |        |       | MAT'L | SECURITY** |
|--------------|----------------|--------------|--------|-------|-------|------------|
|              |                | LEVEL*       | LENGTH | DIAM. |       |            |
| ARL2         | Upgradient     | 384'         | 22'    | 4"    | PVC   | Good       |
| ARL3         | Upgradient     | 359'         | 20'    | 4"    | PVC   | Good       |
| ARL4         | Upgradient     | 336'         | 15.7'  | 4"    | PVC   | Good       |
| ARL5         | Downgradient   | 308'         | 15'    | 4"    | PVC   | Good       |
| ARL6.1       | Downgradient   | 302'         | 20.8'  | 4"    | PVC   | Good       |
| ARL7         | Downgradient   | 300'         | 15.6'  | 4"    | PVC   | Good       |
| ARL8         | Downgradient   | 309'         | 15.6'  | 4"    | PVC   | Good       |
| ARL101       | Spring Downgr. | 270'         | -      | -     | -     | -          |
| ARL201       | Subdrain       | 390'         | -      | -     | -     | -          |

\*Typical elevation of water in well column, ft. above MSL.

\*\* General comments on monitoring well construction.

### Sampling Schedule

As outlined in the Water Quality Monitoring Plan [7] quarterly sampling was selected as the standard sampling frequency for monitoring wells for the first year of implementation of the program. This level of effort was recommended because there was little first hand a priori knowledge of the available monitoring stations. As a database is developed, this frequency of sampling may not be necessary if problems are not encountered.

Surface water monitoring was performed only during the "open water" season, and timed to coincide with other sampling forays. It may be worthwhile to extend this monitoring into winter, if the monitoring sites are accessible. Subdrain sampling was done in accordance with Industrial

Wastewater Discharge permits issued by the Anchorage Water and Wastewater Utility.

Table 6. 1988 Sampling Frequency

| STATION TYPE      | FREQUENCY                                |
|-------------------|--|
| Upgradient Well   | Quarterly (March, May, August, November) |
| Downgradient Well | Quarterly (March, May, August, November) |
| Surface Water     | Quarterly in Summer (May, August)        |
| Subdrain          | Monthly (ARL); Quarterly (MFL)           |

### 1988 Laboratory Analysis

AmTest, Inc., of Redmond, Washington, performed analysis of samples for the following constituents:

|                             |                 |               |
|-----------------------------|-----------------|---------------|
| Total Suspended Solids      | Calcium         | Alkalinity    |
| Turbidity                   | Sodium          | Chloride      |
| Total Kjeldahl Nitrogen     | Potassium       | Sulfate       |
| Nitrate + nitrite           | Magnesium       | Total Phenols |
| Ammonia                     | Arsenic         | Cyanide       |
| Total Dissolved Solids      | Chromium, total | Silver        |
| Chemical Oxygen Demand      | Copper          | Nickel        |
| Biochemical Oxygen Demand   | Manganese       | Mercury       |
| Total Phosphorus            | Lead            | Zinc          |
| Soluble Reactive Phosphorus |                 | Iron          |

Brown and Caldwell Laboratories, of Emeryville, California, performed analysis for Total Organic Carbon, Volatile Organic Acids, and (for ARL201 and ARL2 only) Base/Neutral Acid extractible organics and Pesticide/PCB's.

Laboratory data was checked upon receiving reports of the data from the laboratories. Ion balances of the inorganic analyses were computed, and a comparison was made between calculated and measured dissolved salt content. Performing these checks after receipt of the laboratory data led to revision of approximately 10% of the laboratory reports. Often times the discrepancies were due to typographical errors. In several instances high turbidity, or an extremely complex sample matrix due to the presence of organics led to difficulties with achieving an appropriate balance between anion and cation equivalents.

Duplicate and blank samples were submitted to the laboratories with disguised sample identification. Split samples were sent to Montgomery Laboratories for an independant analysis and comparison to the subcontract

laboratory results. The laboratories also provided data on analysis of sample spikes, in which the laboratory performing the analysis ran a second analysis with a known amount of analyte added to the sample within the laboratory. These Quality Assurance samples were very useful in confirming the results reported by the laboratories.

### 1988 Data Management

Field reports were checked by senior personnel familiar with field operations. Laboratory reports and chain of custody records were logged as they were received and checked. Hardcopy data was compiled and submitted to the Municipality on a quarterly basis. Upon entering data into the computerized data base, each data sheet, or column of a multiple station laboratory report, is marked with a sequential numeric "reference code". This allows direct retrieval of the hardcopy record in case of future need to do so. A "Comments" file within the database allows for notes on the quality of data to be carried with each individual result. For example, if a particular sample had an analytical holding time exceeded, then that fact would be noted and the resulting value marked accordingly.

## **RESULTS**

The following discussion presents some general discussion about findings at each of the landfills. A more complete discussion and tabular presentation of the data is provided in the 1988 Landfill Water Quality Monitoring Program Annual Report [8].

### Merrill Field Landfill

Because of the proximity of the landfill to the north fork of Chester Creek, there has been a longstanding concern that the landfill contributes contaminants to the creek. Because of potential for pollution from a snow disposal site near the landfill, as well as storm drainage upstream and older industrial land use in the area, it has been difficult to pinpoint the source of contamination. 1988 data shown in Table 7, however, demonstrate a consistent tendency toward increasing levels of solutes from the upstream surface water station (CL101) and the downstream station (CL110). This confirms data from earlier USGS work, although data are insufficient to establish this trend statistically. Additionally, it should be noted that no Alaska Water Quality Standards (18 AAC 70) are shown to be violated at the downstream sampling location.

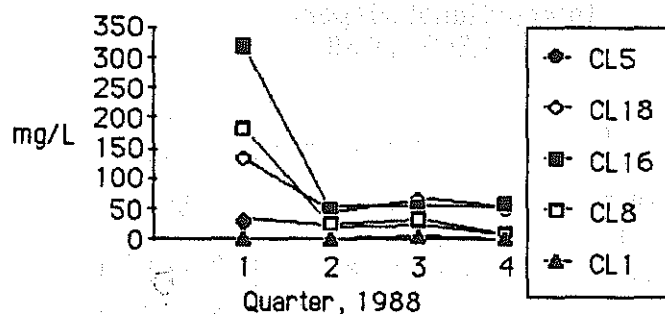


Table 7. North Fork Chester Creek Near Merrill Field

| Parameter (mg/L) | 5/11/88 |       | 8/18/88 |       |
|------------------|---------|-------|---------|-------|
|                  | CL101   | CL110 | CL101   | CL110 |
| C.O.D.           | <5      | 38.3  | 8.84    | 30.6  |
| T.D.S.           | 230     | 252   | 150     | 194   |
| Chloride         | 44.1    | 47.1  | 33.7    | 39.3  |
| Sulfate          | 27.6    | 28.8  | 8.84    | 9.53  |
| Iron             | 0.12    | 4.1   | 0.6     | 3.7   |
| Manganese        | 0.03    | 0.18  | 0.0057  | 0.17  |

A recent USGS study [5] concluded that horizontal transport of pollutants from landfill leachate is occurring in a southwesterly direction from Merrill Field Landfill. This program utilized observation wells from that study for monitoring. Samples from wells constructed in the wetland area (CL5, CL16, and CL18) have been consistently very turbid and may be subject to excessive influence from surface water because of their shallow screen depths. Iron and manganese values were very high in these samples, up to 61 mg/L and 3.1 mg/L respectively for these parameters, which may be indicative of leachate. Chlorides and dissolved solids values in these wells are elevated above the values from CL8, an upgradient well which shows some signs of contamination. The range of values for C.O.D. and T.D.S. are shown below:

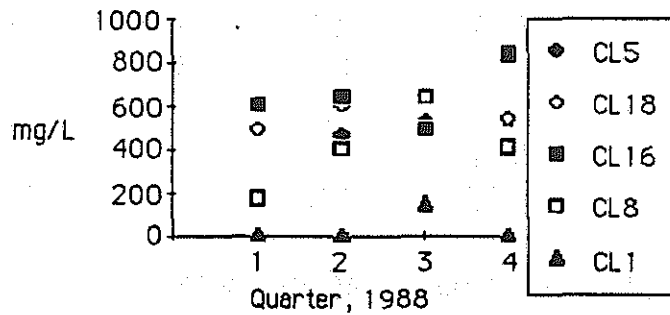
Merrill Field  
C.O.D. 1988



USGS [ 5 ] documented detection of several trace organics in these wells, including dichlorofluormethane, methyl chloride, toluene, vinyl chloride, 1,1-dichloroethane and 1,1,1-trichloroethane. No trace organics were detected in the downgradient wells in 1988.

Data from sampling of the deep well below the landfill (CL1) supports the contention of earlier studies [4,9] that vertical migration of leachate from the landfill is slow, and there is no apparent contamination of deep drinking water aquifer.

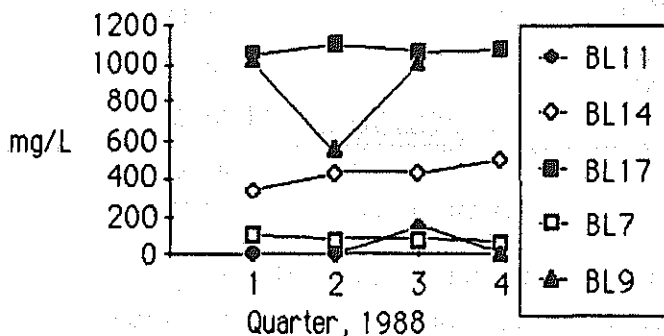
**Merrill Field  
T.D.S. 1988**



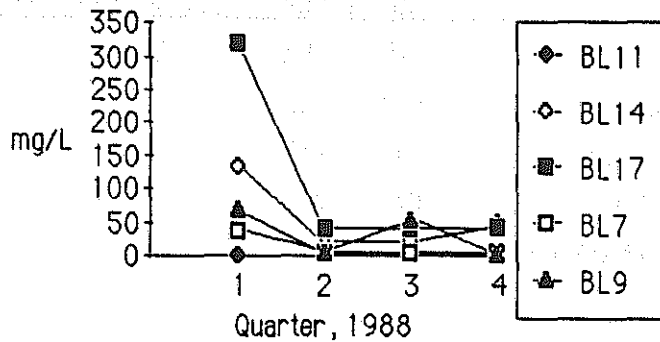
International Airport Road Landfill

Data from this landfill showed seasonal variation in water quality similar to Merrill Field. High COD and lower water levels were found in late winter sampling, but this gave way to increasing concentrations of metals and other dissolved ions in mid-summer, notably in BL14 and BL17. These wells may be subject to surface contamination, as there is no adequate seal for their protection. These two wells show consistently high values of dissolved solids, while BL7, located some distanced downgradient is consistently low in dissolved solids. No contamination from landfill leachate is apparent at Connors Lake.

**International Airport  
T.D.S. 1988**



**International Airport  
C.O.D. 1988**

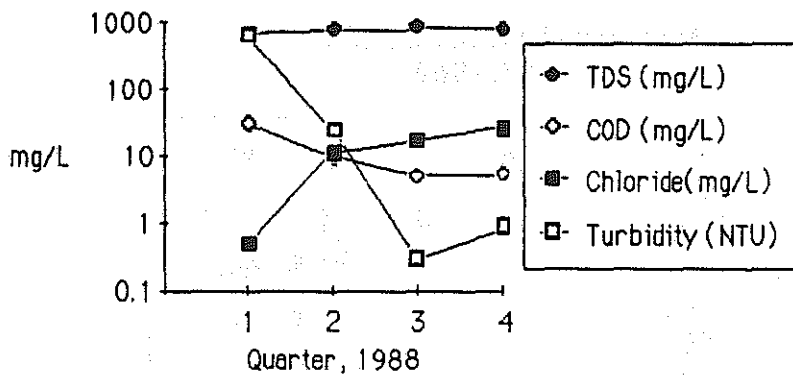


Peter's Creek Landfill

The relatively porous soils in the old gravel borrow site are prone to rapid percolation of any leachate derived from the landfill. The potential for leachate transport is confirmed by monitoring data from the new downgradient well, PL5. This well showed high values of dissolved solutes, but relatively low values of C.O.D.. Compare the values shown below to the values for the upgradient well, which showed a T.D.S. of 162 mg/L and no detectable C.O.D. on the one sampling date for which data is available (9/23)

PL5 is the only well in which trace organics were detected at any of the landfills. These data are shown below in Table 8. It is interesting to note that the concentrations are not showing a steady increase in time, as might be expected in the development of a plume from a constant source. Rather, the concentrations for some parameters have fallen between the September and November samplings.

**PL5 988**



Because of the contamination discovered in PL5, a new monitoring well on the downgradient edge of the landfill is planned for construction this year.

Table 8. Trace Organics found at PL5

| Parameter                    | March | May | September | November |
|------------------------------|-------|-----|-----------|----------|
| Total Organic Carbon(mg/L)   | 3.4   | 4.6 | 4.8       | 4.6      |
| 1,1,1-Trichloroethane(ug/L)  | ND*   | ND  | 11.0      | 12.0     |
| 1,1,-Dichloroethane(ug/L)    | ND    | ND  | 2.0       | 2.0      |
| Methylene Chloride (ug/L)    | ND    | ND  | 7.0       | ND       |
| Trichlorofluoromethane(ug/L) | ND    | ND  | 11.0      | ND       |

\* Not Detected

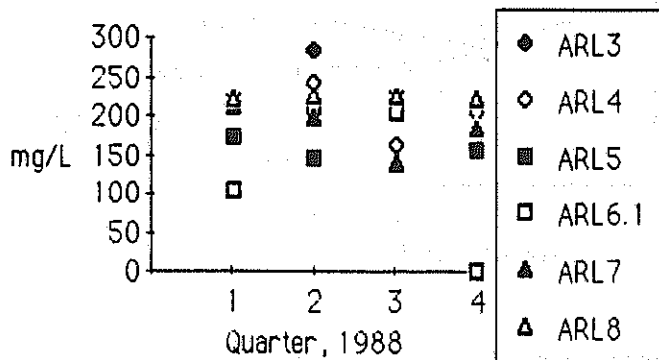
### Old Highland Road Landfill

Strong potential exists for leachate contamination of wetlands at the Old Highland Road site. However, the limited monitoring system presently in use precludes quantitative assessment of the problem. The existing monitoring well is not adequately protected from disturbance, as its guard casing can be lifted out by hand. The long screen length allows dilution of deeper ground waters with surface runoff. This is especially problematic in winter, when an ice crust at the top of the well prevents adequate access for purging and sampling.

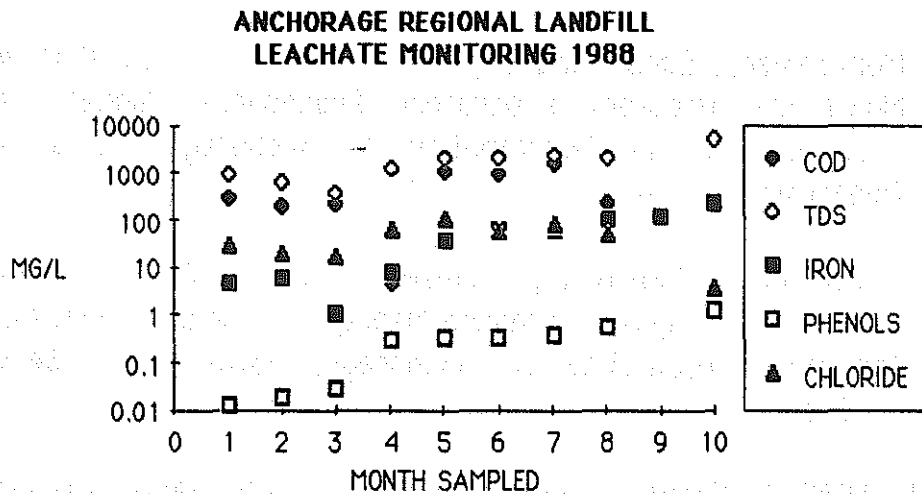
### Anchorage Regional Landfill

Three upgradient wells and four downgradient wells showed some variation in parameter concentrations both seasonally and between wells. However, variation between upgradient and downgradient monitoring well results is insignificant.

**Anchorage Regional Landfill  
T.D.S. 1988**



Data from the seven wells taken together give us a reasonable database for expected variations in background water quality. Leachate strength is shown to be increasing over time. It is probable that this is due to aging of the landfill as well as the increasing ratio of volume of garbage in the landfill to leachate generated.



### CONCLUSIONS

- 1) Consistency in a water quality monitoring program was developed through careful planning and documentation of procedures for sampling, laboratory analysis and data management. The documentation provides benefits for data analysis as the reliability of data is known.
- 2) Anchorage's older sites will continue to have impacts on surrounding environments from leachate migration. Continued monitoring is required to establish the need and scope of corrective action.
- 3) Leachate collected from ARL is within limits posed by permit for discharge into AWWU wastewater treatment system.

### ACKNOWLEDGMENTS

This monitoring program is funded by the Municipality of Anchorage. Dan Crevensten, Matt Tanaka, Curt Conner, and T.C. Wilson performed field sampling. Assistance in planning and evaluation of the program was provided by Harper-Owes, Inc., of Seattle, Washington. Monitoring wells were

constructed at the Anchorage Regional Landfill and Peter's Creek Landfill by MW Drilling under the direction of Golder Associates, Inc.

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## Anchorage Lower Hillside Groundwater Management

By: \* Gary J. Prokosch  
Water Resources Manager

### Abstract

In August of 1988 a strategy to decrease the amount of groundwater withdrawn by the Anchorage Water & Wastewater Utility (AWWU) in South Anchorage was implemented. This strategy was developed by AWWU and DNR/Southcentral Regional Office (SCRO) over a two-year period. Implementation of the strategy was successful due to AWWU's accelerated construction of a number of very important water lines which allowed Eklutna Lake and Ship Creek water to be used in South Anchorage.

The final version of the strategy called for AWWU to abandon the use of 10 wells, some of which were put out of service before full implementation of the strategy, put 8 wells on standby status and reducing their use by 50 percent. In reality these wells are operated once a week for 8 hours to assure proper operation.\* Only one AWWU well is still fully active in the South Anchorage area and its production has been cut back below its past operational levels. See Figure #1.

The history behind this strategy and the events that lead up to its implementation are well known to most Anchorage residents. The water level declines in the Lower Anchorage Hillside area have been recorded by USGS since the late 60s and early 70s. Some areas of the Lower Hillside have experienced decline of between one to four feet per year. Between 1983 and 1985 at least 27 well failures were recorded. (Munter, July 1985 Public Data File 85-13).

The population of the South Anchorage area and specifically the Lower Anchorage Hillside area boomed between 1980 and 1983. AWWU had to keep up with demand for water by their current and future customers and DLWM had to address the situation of well failures. Between 1983 and 1985 a number of factors changed. AWWU water demand increased and new applications were filed with DLWM to increase its production in South Anchorage by 2.7 million gallons per day in addition to an old Central Alaska Utility application which AWWU inherited for 1.5 million gallons per day. In 1986 and 1987 AWWU and DLWM were at odds as to how to address AWWU's demand for water and the DLWM's concern over past well failure.

DLWM would not act on AWWU's new application and felt it proper to hold all pending applications in a 15 1/2 square mile area of the Lower Anchorage Hillside in pending status until the situation was resolved.

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Anchorage Lower Hillside Groundwater Management  
Abstract  
Page 2

In 1987 AWWU recognized the need to cooperate with DLWM to resolve the situation. This action was not directly related to DLWM's insistence but more on the changing economy and the cost of operating both a surface and subsurface public water supply system. With completion of the Eklutna Water Project and the necessary water lines to South Anchorage the need for AWWU's well system as a primary source of water was less important, as a backup system it was mandatory.

As a result of the strategy, AWWU's water withdrawals from its South Anchorage wells has decreased for the 1986 through mid-1988 average of over 6,000,000 gallons per day (gpd) to an average of 437,000 gpd from August through December, 1988.

The USGS monitoring wells located in South Anchorage have shown that the static water levels have been increasing. A well monitored by Roy Glass, USGS, in McMahan Subdivision has shown an increase in the static water level every month since August of 1988. This increase has totalled over 5.0 feet to date (figure #2). In this subdivision between 1983 and 1985 DNR records show that at least 6 wells went dry. It was assumed that more wells went dry that were never reported. This fact was reconfirmed during the adjudication of water rights within this subdivision when six applications showed that an original well had to be deepened.

AWWU has stated that the pumping rates from August through December, 1988, will be maintained barring any emergency need of its wells.

As of this date, SCRO feels that the results are positive and we are optimistic that the short term trend will continue.

As of January 9, 1989 the 900 plus water rights applications that have been held in pending status within the 15-1/2 square mile area of the Anchorage Lower Hillside since November 7, 1985, are again being adjudicated.



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USGS, Monitoring well records, provisional data USGS (Oct. '87 - Feb. 2, 1989).

# ANCHORAGE WATER & WASTEWATER UTILITY

## ANCHORAGE BOWL AREA

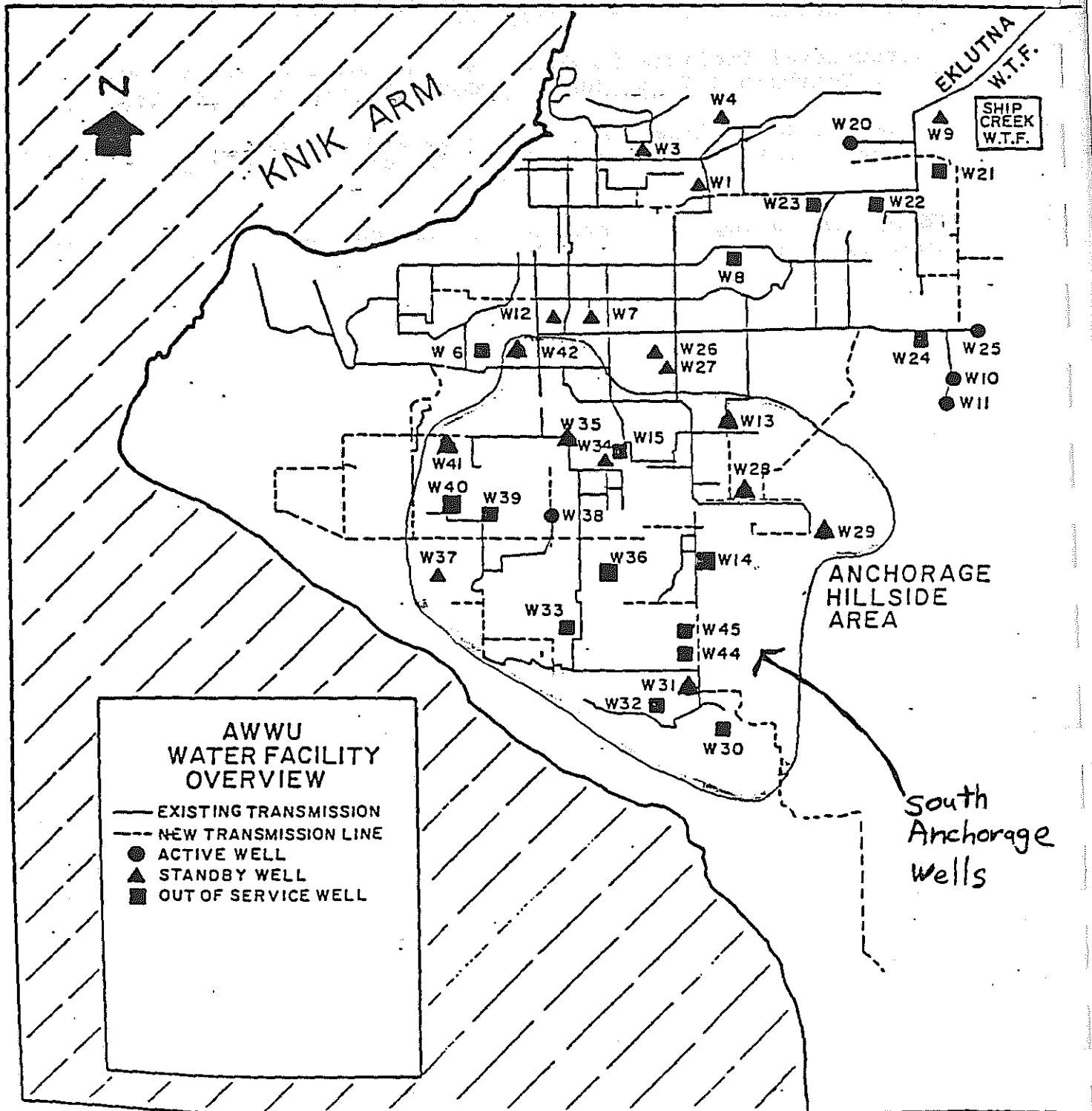
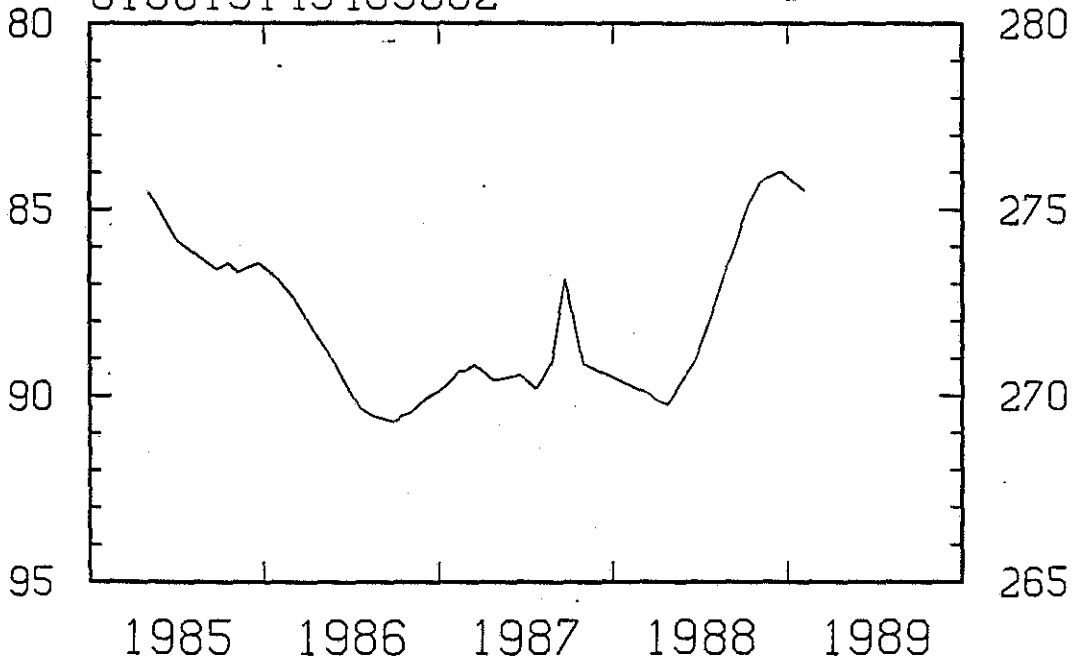


Figure # 1

ROY GLASS WELL

WATER LEVEL, FEET BELOW LAND SURFACE

SB01200328ADBB4 045  
610619149483002



WATER LEVEL, FEET ABOVE NGVD

PREVIOUS DATA, USGS OCT. 1987- FEB. 1989

Figure # 2

# THE USE OF STOCHASTIC METHODS IN GROUNDWATER MODELING

Mark A. Tumeo, Assistant Professor<sup>1</sup>

Adam H. Owen<sup>2</sup>

## ABSTRACT

Since the inception of the concept that the computer could be used to model an ecological system, mathematical modeling of our environment has continuously grown and expanded. Computer models have become increasingly popular as management tools to aid in analysis of the the complex, -interacting factors which must be considered in natural resource and environmental management. This trend has been most apparent in ground water management, where actual data tends to be scarce and the controlling phenomena complex, and in some cases, not well understood.

In addition to the uncertainties associated with limited data and incomplete knowledge, recent advances in our understanding of the environment have proven the need to also include stochastic phenomena, processes are those phenomena which randomly vary in time and/or space within some mathematically definable range. However, most available ground water models are deterministic, that is, they require selection of specific values of input parameters such as reaction rates, permeability or diffusion coefficients and produce single-valued predictions of output variables in time and/or space. Hence, most existing ground water models are of limited value to managers and decision makers because a purely deterministic structure is used to model processes that are intrinsically stochastic. This paper reports on various stochastic methods which are available to accomplish this goal, and provides an example of how one such method, the Monte Carlo method, may be easily incorporated into an existing ground water model.

## INTRODUCTION

Since the inception of the concept that the computer could be used to model an ecological system, mathematical modeling of our environment has continuously grown and expanded. Computer models have become increasingly popular as management tools to aid in analysis of the the complex, interacting factors which must be considered in natural resource and environmental management. Dramatic advances in microcomputer technology, which have placed small, yet high powered, machines in the hands of almost all decision makers, has greatly enhanced this trend (Heidtke, et.al., 1986). This trend has been most apparent in ground water management, where actual data tends to be scarce and the controlling phenomena complex, and in some cases, not well understood.

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Increased use of computer models in management has led to increased concern about the quality and accuracy of information such models can provide. Most available ground water models are deterministic, that is, they require selection of specific values of input parameters such as reaction rates, and produce single-valued predictions of output variables in time and/or space. However, modelers and decision makers have realized that natural systems, and especially ground water systems, are a complex combination of deterministic and stochastic processes. Stochastic processes are those phenomena which randomly vary in time and/or space within some mathematically definable range. Hence, most existing ground water models are of limited value to decision makers because a purely deterministic structure is used to model processes that are intrinsically stochastic. In analyzing ground water movement, contaminant transport, and associated environmental and public health risks, it is essential that the probabilistic aspect of ground water systems be included in the model used. The challenge facing computer modelers today is to incorporate techniques into ground water models which can handle both deterministic and stochastic processes while still providing models which are easily used and understood by the decision maker. This paper reports on various stochastic methods which are available to accomplish this goal, and provides an example of how one such method, the Monte Carlo method, may be easily incorporated into an existing ground water model.

## BACKGROUND OF STOCHASTIC MODELING

Early attempts to include natural variability in environmental models involved statistical extrapolation (LeBosquet and Tsivoglou, 1950; Bulmer, 1957). Statistical approaches however, are limited in capability to handle the complex variations present in ground water systems. A more powerful approach is to encode the current state of knowledge about the variations affecting a process directly in a mathematical expression. Using this approach, the probability density function of the outcome becomes the "fundamental entity in the problem" (Moore and Brewer, 1972). There are currently four commonly used "probabilistic" methods to include stochasticity in mathematical modelling: functional analysis, the Monte Carlo technique, stochastic differential equations, and the moment/probability density function approach.

### Functional Analysis

The title "functional analysis", as used in this paper, refers to any method which involves the use of an assumed function to approximate the mean, variance and/or higher order moments of an output variable (Y) as a function of one or more input variables (X). This grouping encompasses several closely related methods identified in the literature as error analysis, uncertainty analysis, or confidence interval development. A representative example of a functional analysis method is 'First order Uncertainty Analysis' (Benjamin and Cornell, 1970; Cornell 1972). This method involves the use of a Taylor Series Expansion or a perturbation equation to represent the functional relationships of the state variables. It is then assumed that any random component in the system can be completely defined as a normal variation around a zero mean. The Taylor Series is then truncated after the first order term, hence the name "First Order Uncertainty Analysis". This approach has the advantage of only requiring estimates of the mean and variance (first and second moments) of the input parameters. In situations where limited information restricts analysis of the random components, first order analysis is a useful tool.

Functional analysis techniques are in general, limited to generation of only the mean and variance of the output variable. In some cases, such analysis gives answers comparable to those obtained by more complex analysis procedures (Brown, 1986). However, in more complicated situations, there are discrepancies between first order analysis and nonlinearized methods such as Monte Carlo (Scavia, et.al., 1981a and b). Other functional analysis approaches, such as statistical estimation of moment generating equations, point estimation techniques, or numerical analysis are also usually limited to first and second moments. Complete description of a distribution requires definition of all higher order moments. Thus, these methods do not entirely describe the distribution except in the special case where the output is normally distributed. First order error analysis can be extended by truncating the Taylor Series expansion after the second or third order, but carrying higher order terms complicates the mathematics. Some work has been done on relaxing the first order approximation restrictions in other ways (Tung, 1987) but little has yet been reported in the literature.

### Stochastic Differential Equations

The development of stochastic differential equations began around the turn of the century with Einstein's classic solution to the problem of Brownian motion (Einstein, 1905).

$$\frac{\delta f(x,t)}{\delta t} = D * \frac{\delta^2 f(x,t)}{\delta x^2} \quad (1)$$

where  $f(x,t)$  is the number of particles per unit volume and  $D$  is the coefficient of diffusion. Einstein's approach was unique in that he applied a boundary condition which arose from the probabilistic development of the problem. Similar approaches were used independently by von Smoluchowski (1906) and Langevin (1908) to derive the same solution. However, aside from simple extensions to analogous diffusion processes there were few attempts to use the techniques developed by these researchers until the late 1940's. The main reason for the restricted application of stochastic differential equations before this time was the lack of mathematical procedures required to solve the equations. The only mathematics available to early researchers was Riemann Calculus. However, Riemann Calculus is not applicable to stochastic differential equations. Adequate mathematical grounding for work with stochastic differential equations was not available until the work of Ito (1944). Stochastic differential equations continue to receive much attention in stochastic hydrology and precipitation analysis, but application of the technique remains limited due to the complexity of the mathematics. To date Ito Calculus and similar stochastic calculi have been developed only for ordinary differential equations rather than for the partial differential equations employed to represent complex environmental systems. While it is possible to convert partial differential equations into ordinary differential equations, the result is usually a non-tractable set of non-linear equations.

### Moment/Probability Density Function (M/PDF) Technique

The M/PDF technique was developed by researchers at the University of California Davis in an effort to overcome the limitations in existing stochastic methods. (Tumeo, 1988). The technique is based on the derivation of distribution functions for output parameters through a mathematical combination of stochastic variations in the input parameters. A two tiered process is involved: first, mathematical expressions for the moments (i.e. the mean, variance, etc.) of the output variables are derived from the original equations which define the process; and second, these moments are used in conjunction



In natural systems, temperature is a stochastic variable. For this case, assume that the temperature varies normally around a mean ( $T_m$ ) with a standard deviation of  $\sigma_T$ . We wish to know the distribution of possible  $K$  values given this temperature variation. Application of Monte Carlo to this problem involves four steps:

- 1) A random number ( $\xi$ ), normally distributed with a mean of zero and a standard deviation of one, is generated. The random number is normally distributed because the input variable is assumed to be normally distributed.
- 2) The random number is used to select an input temperature ( $T_i$ ) using the following equation:

$$T_i = T_m + \xi \sigma_T \quad (3)$$

- 3) The reaction rate  $K(T)$  which results from temperature  $T_i$  is calculated using equation 1.
- 4) This process is repeated several times and the varying realizations of  $K(T)$  stored. After a sufficient number of repetitions, the probability density function of the resulting  $K(T)$  values can be computed.

The above procedure can easily be extended to several variables simultaneously. Input variables and/or parameters can be linked by predicating the selection of one variable upon the random outcome of another (Tumeo and Orlob, 1986).

Because the Monte Carlo technique provides a straight-forward way of including stochasticity in mathematical models, it is probably the most widely used stochastic method. It has successfully been used to model groundwater pollution (Smith and Schwartz, 1981a and b; Black and Freyburg, 1987) and plume dispersion (Orlob and Tumeo, 1986; Orlob, et al., 1987). Results of a Monte Carlo simulation are also the usual standard to which other methods are compared (Scavia et al., 1981a; Brown, 1986). However, while the Monte Carlo technique is straight forward, it is not without its weaknesses and limitations. Random number generation, the basis of the method, is not an easy process, and can be troublesome and even unreliable, especially on smaller computers. Random number generators usually produce a set of uniformly distributed values between 0 and 1. If some other distribution for the input variable is required, then the distribution of the random number must be transformed. Detailed discussion of the mathematics of random variable transformations may be found in Benjamin and Cornell (1970).

Another central issue in Monte Carlo techniques is determination of the number of times to run the model. Enough runs must be performed to obtain a statistically valid sample. The number required is usually a function of the type and complexity of the output distribution, which is not known a priori. In practice, a certain number of trials are executed (e.g. 300) and the resulting statistics calculated. The process is then repeated with a larger sampling (e.g. 600); new statistics calculated and compared to the previous run. If there is no 'significant' difference, then 300 runs are sufficient. If there is a significant difference, the number of runs are increased and the new statistics compared to the run of 600. This process is repeated until no 'significant' difference is found between successive runs. 'Statistical significance' can be determined by any one of several standard statistical tests for comparison of distributions (Benjamin and Cornell, 1970).



## EXAMPLE APPLICATION

In order to demonstrate how stochastic methods may be applied to existing ground water models, an existing two-dimensional model of solute transport through porous media TWODPLUME, (Slotta, 1987) was modified using the Monte Carlo technique. The model handles either a constant or slug point source of ground water contamination. Advection, dispersion, decay and adsorption processes are modeled in a two-dimension plane. Table 1 shows the a few of the required input parameters and thier default values as included in the deterministic program.

**TABLE 1**  
**EXAMPLE INPUT PARAMETERS FOR**  
**DETERMINISTIC TWODPLUME PROGRAM**

| Variable | Description  | Default Value |
|----------|--|---------------|
| EPSO     | effective porosity   | 0.3           |
| VELX     | average pore<br>water velocity (m/d)                       | 4500          |
| DISPX    | dispersion coefficient,<br>x-direction (m <sup>2</sup> /d) | 6000          |
| DISPY    | dispersion coefficient,<br>y-direction (m <sup>2</sup> /d) | 600           |
| KSAND    | decimal fraction sand                                      | 0.86          |
| KCLAY    | decimal fraction clay                                      | 0.1           |
| KORG     | decimal fraction organics                                  | 0.04          |
| LAMSOL   | first order decay rate (d <sup>-1</sup> )                  | 0.002         |
| FO(I)    | concentration of source (g/m <sup>3</sup> )                | 500.0         |

### Development of the Monte Carlo Version of TWODPLUME

As can be seen from Table 1, the TWODPLUME model requires deterministic values for several input parameters which in reality are stochastic variables. Currently, work is under way at the University of Alaska Fairbanks to add stochastic capability to the model in several variables. Table 2 shows those variables which are currently being examined.

**TABLE 2**  
**INPUT PARAMETERS TO WHICH STOCHASTIC**  
**CABILITIES WILL BE ADDED**

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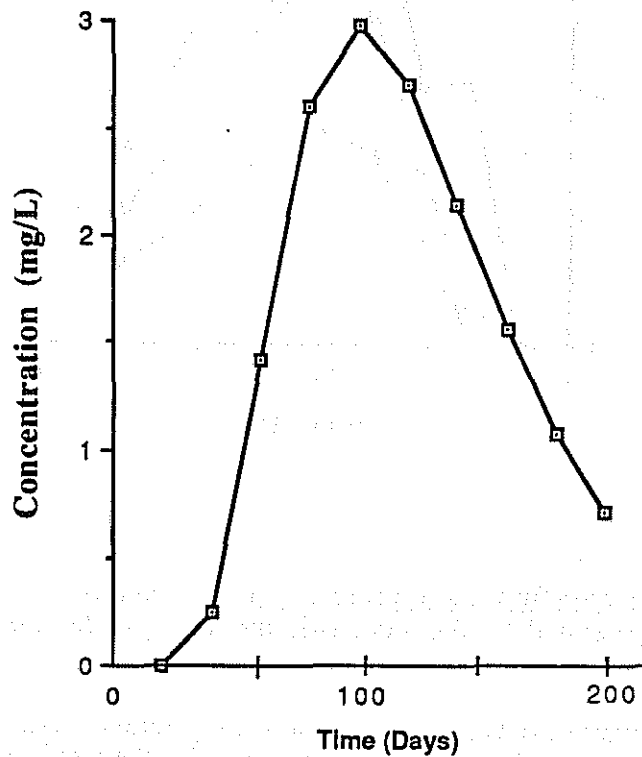
|   |
|---|
| Effective Porosity  |
| Pore Water Velocity                                       |
| Dispersion Coefficients (x and y directions)              |
| Percent Sand, Clay and Organics                           |
| Linear Adsorption Coefficient for Clay, Sand and Organics |
| First Order Loss Coefficient                              |
| Source Strength   |

To date, a preliminary Monte Carlo version of TWODPLUME which allows examination of stochastic variations in source strength has been developed. Stochasticity was introduced into the model by running the deterministic version repeated times, each time randomly selecting a new values for the source strength from a normal distribution. This preliminary model was used to demonstrate the ease of application of the Monte Carlo technique and the increased information which may be gained from using a stochastic model versus a deterministic model.

#### COMPARISON OF DETERMINISTIC AND STOCHASTIC MODELS

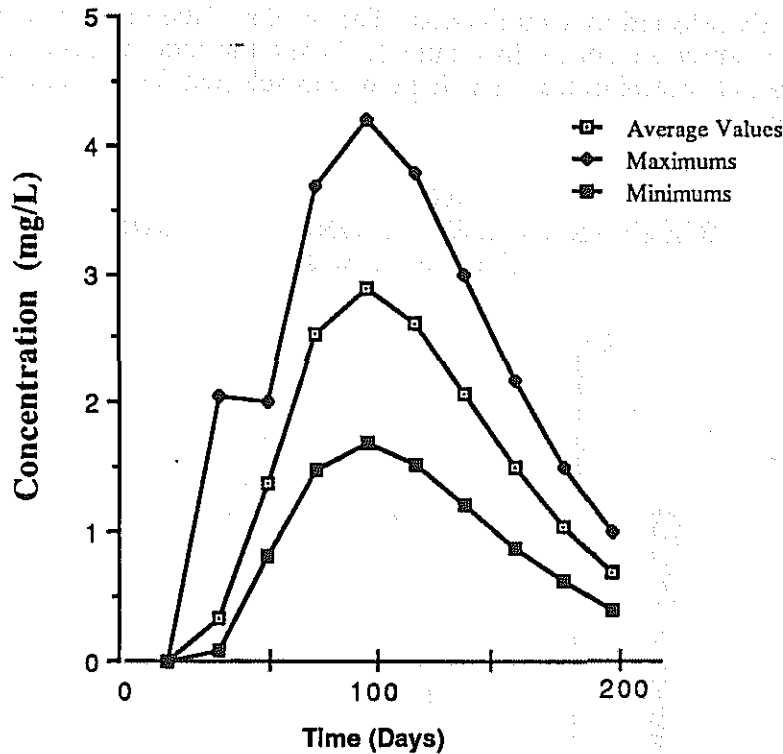
Deterministic version. The deterministic version of the model was executed using the default values for all input variables except source strength. A slug source strength of  $50 \text{ g/m}^3$  was arbitrarily selected for examination. The results of the run at a single X-Y over a 200 day period location are shown in Figure 1. While the model allows examination of several X-Y points at several times, a single point was selected for the sake of simplicity in this example case.

**FIGURE 1**  
**RESULTS OF DETERMINISTIC MODEL**  
**X = 50 Y = 10**



Stochastic version. In the development of the model for this example application, it was assumed that a slug source varied in strength normally with a mean of  $50 \text{ g/m}^3$  and a standard deviation of  $12.5 \text{ g/m}^3$ . The model was run 20 times, each time randomly selecting from the specified distribution. The result is a set of 20 concentrations for each time interval shown in Figure 1 above. With this 'set' of concentrations, the distribution of the concentrations at each time may be calculated. Figure 2 shows the mean concentrations at each time and the range of concentrations calculated. Figure 3 shows the probability density function for the concentration at the peak time of 100 days.

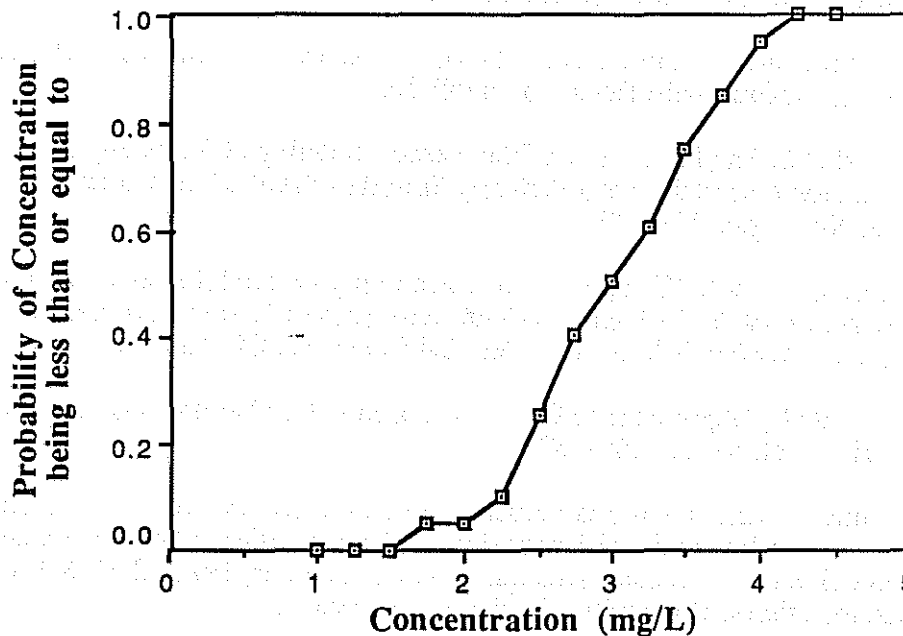
**FIGURE 2**  
**RESULTS OF STOCHASTIC MODEL**  
**X = 50 Y = 10**



Application information from a stochastic model. To demonstrate the advantages of the type of information generated by the stochastic model, consider the following hypothetical management scenario:

A spill of a chemical has been reported up-gradient from a well used as a drinking water source for a small community. The chemical has an EPA limit of  $3 \text{ mg/L}$  in drinking water supplies. You are required to estimate the potential health risks and economic impacts of the spill as the slug passes through the ground water aquifer. When the level reaches above the limit set by the EPA, the community well must be shut down and drinking water trucked into the town.

**FIGURE 3**  
**PROBABILITY DENSITY FUNCTION OF**  
**CONTAMINANT CONCENTRATION AT 100 DAYS**  
**X = 50 Y = 10**



If the purely deterministic model were used in the analysis, Figure 1 indicates that the peak concentration in the well never reaches three. It may therefore be assumed that the community is at minimal risk. However, using the stochastic model (Figure 2), it can be seen that the water well has some measurable probability of experience a concentration over 3 mg/L. From Figure 3, one can calculate the actual probability that the concentration will exceed 3 mg/L on day 100 as 0.50. Hence, there is a 50% chance that the well will exceed the EPA maximum. Using information like that shown in Figure 3, it can be calculated that there is a 30% chance that the level will exceed the EPA minimum as early as 80 days after the spill and may actually remain above the minimum level for up to 140 days after the spill. This kind of probabilistic information is essential for managers, regulators, and decision makers to make accurate and meaningful assessments of impact.

### CONCLUSION

While existing deterministic models have proven useful in the past, recent advances in our understanding of ground water processes have emphasized the need to include the probabilistic phenomena which effect natural systems. As the use of computer modeling for ground water management and impact analysis increases, it will become more and more important that stochastic components be included. Stochastic methods, when applied to existing, deterministic environmental models, expand their capabilities by providing the model user a potentially powerful tool to analyze environmental options based on their probability of impact and the likely magnitude of that impact. Such information is critical to decision makers who are charged with the responsibility of managing environmental quality.

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# RECHARGE, DISCHARGE, AND GROUND-WATER DATA FLOW SYSTEMS IN ALASKA

by James A. Munter<sup>1</sup>

## ABSTRACT

Ground-water data collection occurs in Alaska by a variety of public and private entities and can collectively be considered as a recharge process for a ground-water data flow system. Ground-water data discharge occurs at permanent paper files. Some data follow direct flow paths from recharge to discharge areas, while other data flow through reports or computer systems. Flow paths may be lengthy in order to confirm or enhance data integrity, inform regulatory authorities or the public, conform to legal procedures, or receive approval from funding sources.

As a result of rapid growth in the ground-water industry in Alaska during the 1980's, the total flux of data through the flow system has increased substantially. This increase, coupled with a concurrent increase in computer technology and use, has resulted in the widespread acceptance of computers as appropriate devices for tapping ground-water data systems and increasing storage within the system.

Current capabilities for tapping ground-water data systems with computers vary, and numerous data streams are effectively untapped or are incompletely tapped. These limitations hinder the beneficial use of ground-water data resources and result in the effective loss of data resources for some uses.

## INTRODUCTION

Numerous individuals in Alaska associated with drilling firms, regulatory or data collection agencies, local governments, consulting firms and businesses are part of an overall system of ground-water data collection, transferral, storage, and use. As a result of economic expansion in Alaska during the early 1980's, and environmental legislation at state and federal levels, existing ground-water data are voluminous and are herein viewed as an important resource that merits proper management in order to preserve their value for future generations. Inasmuch as the concept of a ground-water flow system is useful for understanding ground-water resources, the concept of a ground-water data flow system is useful for understanding Alaska's ground-water data resources and their role in Alaska's regulatory, technical and business environment.

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Ground-water flow systems and ground-water data flow systems have many analogous features. Both systems have recharge areas, discharge areas, regional or local scales, flow paths through different environments, stagnation zones, and areas of storage. As shown in figure 1, a ground-water divide separates a local flow system discharging to a lake from a regional flow system flowing under and around the lake. A stagnation zone occurs where three different flow systems abut each other.

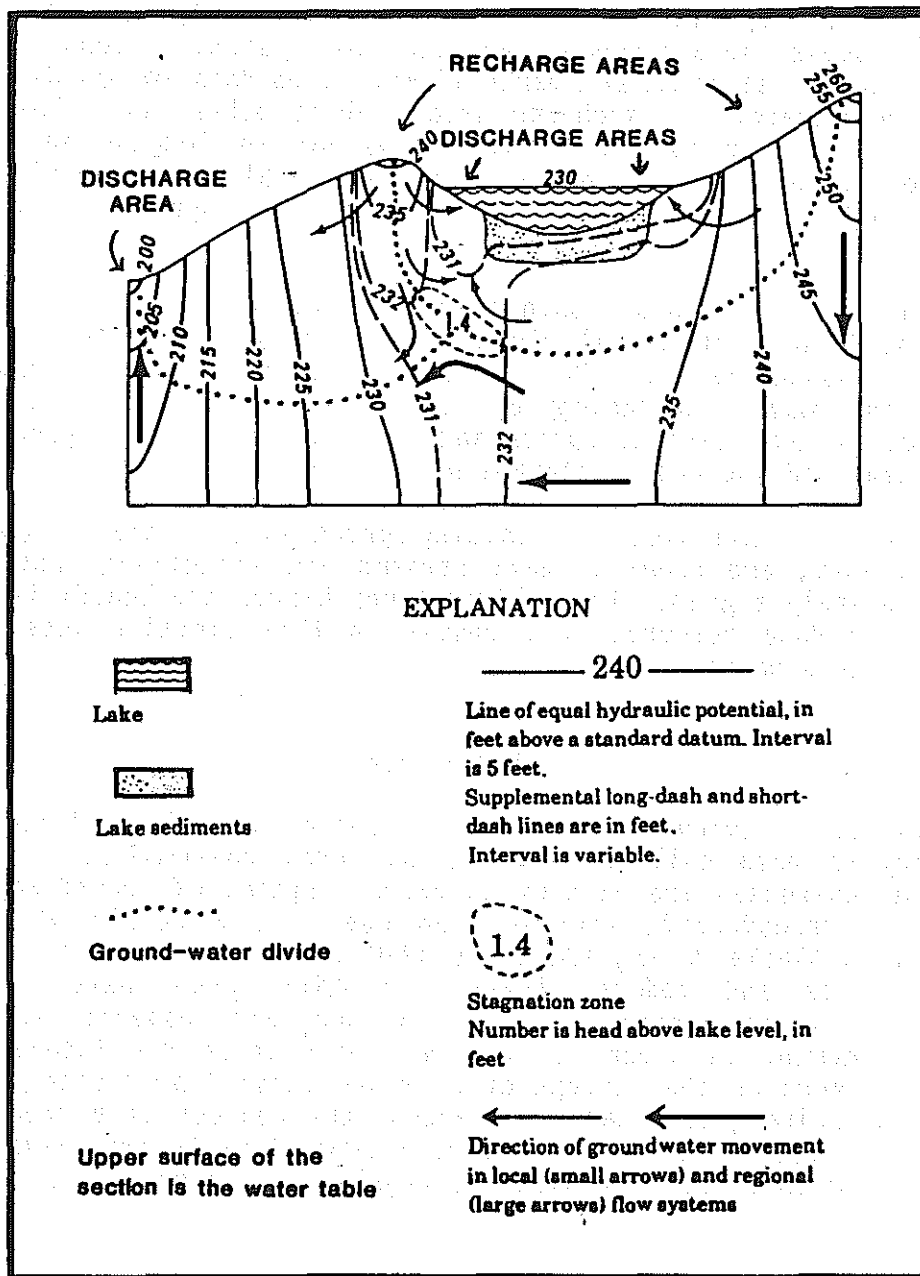


Figure 1. Examples of local and regional ground-water flow systems (modified from Winter, 1978).



## SOURCES OF GROUND-WATER DATA (RECHARGE AREAS)

Sources of ground-water data in Alaska have been described by Munter (1987), Maynard (1988), and Munter (1989). Ground-water data are collected to monitor solid waste and waste water disposal operations, oil spills, coal mining areas, potable water quality, ambient water quality, and potential and actual hazardous waste sites, and to aid water resources development and planning. Water resources development and planning, in turn, commonly precede and facilitate orderly industrial and general economic development.

Types of ground-water data include:

1. Site location information;
2. Hydrogeologic information;
3. Well construction and development information;
4. Periodic water-level data;
5. Aquifer-test data;
6. Field or laboratory determinations of water-quality parameters;
7. Field or laboratory quality assurance and quality control (QA/QC) information; and
8. Water (including wastewater) extraction or injection (water use) information.

Overall, the process of accumulating these types of data is analogous to the accumulation of water in recharge areas of ground-water flow systems. Although rates of accumulation may vary substantially from place to place as a result of local conditions, both processes occur over a wide geographic area and with some regularity. Most importantly, however, both recharge processes are necessary for the continuation of flow in each system. Without recharge, each system would eventually reach a state of static equilibrium.

## SINKS FOR GROUND-WATER DATA (DISCHARGE AREAS)

Ground-water data collection is usually done by generating paper records. Data collection from some sites, in fact, results in the collection of considerable quantities of paper records. Most of these records eventually end up in permanent paper files. Because data typically ceases to move upon entering these files, the ground-water data flow system terminates and the paper files can be considered analogous to discharge areas of ground-water flow systems. As with recharge areas, ground-water data discharge areas are widely distributed geographically and vary substantially in size from place to place.

Primary data discharge areas in Alaska are: the ground-water files at the U.S. Geological Survey (USGS); the public water supply, single family (or on-lot or subdivision), solid waste, oil pollution, wastewater and village files at the Alaska Department of Environmental Conservation; coal mining permit application files of the Alaska Division of Mining; shallow observation well and on-site files of the Anchorage Department of Health and Human Services; and the site studies file and the Alaska Water Use Data System files of the Alaska Division of Geological and Geophysical Surveys (DGGs) (Munter, 1987; W. Petrik, DGGs, oral commun., 1989).

Not all paper files in Alaska are ground-water data discharge areas. DGGs, for example, maintains a Well Log Tracking System (WELTS) file containing approximately 4900 well logs that are designated for transferral to the USGS for computer entry of well-log data and permanent filing in the ground-water files. Similarly, data files maintained by consultants, owners or operators of facilities, or provisional data under technical or administrative review have not reached a final state of repose and are not considered to have discharged from the ground-water data flow system.

#### GROUND-WATER DATA FLOW PATHS

Although some data travel a relatively short route from collection to filing, most data follow more lengthy flow paths. Flow paths may be lengthy in order to confirm or enhance data integrity, convert data to digital or microfiche formats, receive approval from funding sources, conform to legal procedures or inform regulatory agencies or the public.

As an example of these processes, figure 2 illustrates major potential flow paths for well-log data. As can be seen, data may flow through numerous environments prior to being computerized. Although not shown, flow of data can be facilitated by use of reports in which different types of data are compiled.

Once data are computerized or permanently filed, data may be retrieved and used for some purpose other than what was originally intended. This could initiate a new flow path and eventual discharge of data to a different permanent paper file. This is analogous to the extraction of water for use and reinjection into another flow system.

The concept of local and regional flow systems can be applied to ground-water data. An example of a local flow system (or short flow path) would be a USGS field inventory in a project area in which well logs are obtained from homeowners or drillers and promptly computerized and permanently filed.

A regional flow system would be one in which ground-water data follow a long flow path through diverse environments. As an example, consider that well logs prepared by drillers are commonly given to homeowners or builders. The homeowner or builder, in turn, may provide

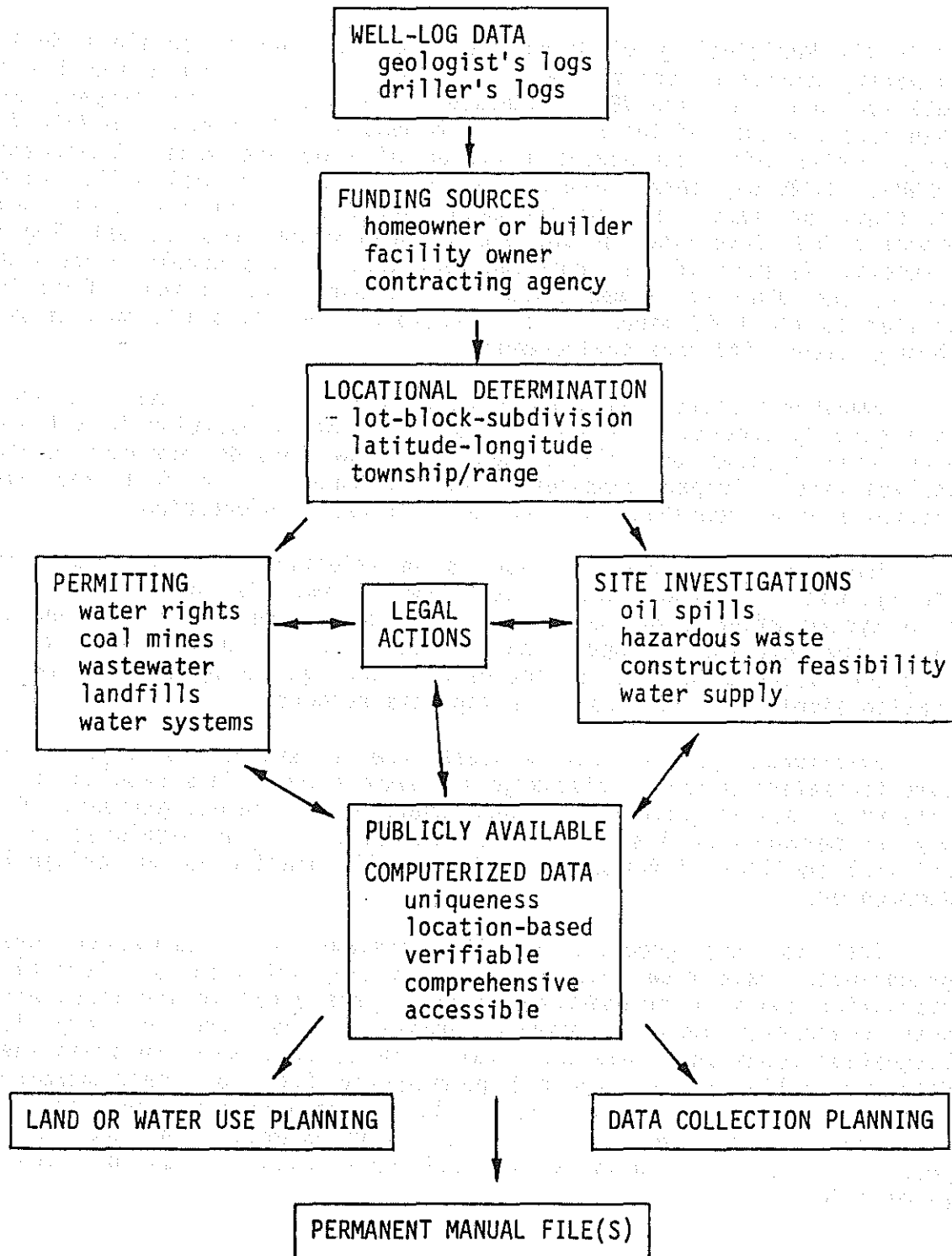


Figure 2. Major potential flowpaths for well-log data.

it to the Municipality of Anchorage (MOA) in order to obtain a Health Authority Approval certificate. The MOA may then forward the log to DGGGS for entry into the WELTS database. A new owner of the property may then obtain a copy of the well log from DGGGS or MOA in order to file for water rights with the Alaska Division of Land and Water Management, (DLWM). DLWM may then forward the log to DGGGS. A fuel spill, water shortage, or septic system contamination problem in the area could prompt a DGGGS investigation and use of information from the well log in a report. As part of the investigation, DGGGS would probably forward the log to the USGS for computerization and permanent filing. From the driller to the USGS manual file, therefore, the log would have passed through seven different environments.

Computer systems play a major role in data flow systems. As shown in figure 2, computerized data occupies a central position in well-log data flow systems and allows data to flow towards numerous useful applications. Proper computerization of data allows fast and easy retrieval of a comprehensive suite of verifiable information.

Functionally, computerization is an effective means of increasing storage within a ground-water data flow system. As in a ground-water flow system after creation of an underground lake (from an abandoned mine, for example), computerization does not materially affect the long-term volumetric rate of flow of data through the system. Computerization simply makes it easier to tap data resources.

Unfortunately, most data in Alaska are not subject to comprehensive computerization prior to discharge to paper files. This results in the effective loss of data for reconnaissance-level investigations. Data may be permanently lost if files are destroyed or misplaced or if critical ancillary information such as well location is not originally documented.

Just as some ground-water flow systems have stagnation zones, ground-water data flow systems also have stagnation zones. Typically, stagnation zones in ground-water flow systems occur in localized areas near boundaries where hydraulic gradients are zero or very low. Stagnation zones in ground-water data flow systems occur in areas where data are collected but are not permanently filed or their permanent filing is delayed by budgetary, legal, or administrative restrictions. Confidential data used in court cases, for example, can stagnate for years prior to being publicly released and allowed to flow to permanent paper files.

## CONCLUSIONS

Ground-water data collection, management and use in Alaska constitutes a system that is analogous in many ways to a ground-water flow system. Both systems have identifiable recharge areas, discharge areas, flow paths, stagnation zones, and areas of storage. Because of the diversity of data users, the flow paths of data can sometimes be lengthy and circuitous. Although computers have a high potential to increase

storage and facilitate extraction of data from ground-water data flow systems, they are underused in Alaska. This limits the beneficial use of ground-water data, and results in the effective loss of data resources for some uses.

#### ACKNOWLEDGEMENT

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**TRANSPORT AND REMOVAL OF CONTAMINANTS IN  
SOIL AND GROUND WATER**

CHARACTERIZATION OF BENZENE TRANSPORT  
IN GROUNDWATER UNDER FREEZING CONDITIONS:  
A REVIEW OF THE CONTROLLING PROCESSES

Michael R. Lilly<sup>1</sup>

ABSTRACT

There has been extensive research into the interactions of solutes in unfrozen soils and groundwater. Research in solute interactions with freezing soils has mainly been limited to salt ions. The chemical and physical processes that are taking place between aromatic organic solutes and water in freezing soils are poorly understood and therefore, will adversely affect the predictions of the transport and fate of these contaminants. A review of solute transport research is presented along with a summary of the important processes to consider when dealing with transport of organic solutes in freezing soil systems. The importance of freezing conditions and solute transport along with natural and artificial freezing situations are also presented. The importance of scale of observation in both data collection and in theory development is also given. This is important in defining controlling processes in solute transport in freezing soils.

INTRODUCTION

The problems associated with groundwater pollution are varied and depend on the contaminants involved and the physical and chemical environment that is contaminated. A great deal of effort has been put into these problems and the interactions involved, however, Kay and Perfect (1988) have reported that relatively little research has been accomplished on how these interactions are changed by freezing soil conditions. The research on solute transport that has been conducted in freezing soils has been focused on salt ion interactions. A review of salt ion interactions will be presented along with the few papers that have studied other contaminants in freezing environments. General discussions of organic solutes and freezing soils will be mainly concerned with saturated, transient, single organic solute systems.

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Pressure and thermal gradients may act as coupled driving forces for groundwater (Freeze and Cherry, 1979). Under most conditions, groundwater flow will follow a direction of decreasing potential head. Thermal gradients will cause convective transport in the direction of heat flow. These flow characteristics are affected by the solute concentration in unfrozen pore water. Increases in salt ions may cause density flows which are opposite to that of the thermal gradient. Effects on the viscosity of the pore fluids, freezing point depressions and precipitation by the solutes should also be considered.

The effects of heavy metals and organic solutes on the interactions taking place in freezing soil are not well defined. Heavy metals may not be excluded from frozen soil zones due to increased adsorption onto the matrix materials as their concentrations increase in the unfrozen pore water. Organic solutes may interact with the formation of the pore ice matrix. These solutes may form separate components due to lower solubility limits. In the case of benzene, it has a freezing point of  $+5.5^{\circ}\text{C}$ , so that if it were to come out of solution, it would form a solid phase mass within the freezing pore water.

Natural and artificial ground freezing effects on contaminant movement in soils may differ from that of non-freezing conditions (Figure 1). Changes in solute migration may be caused by entrapment in frozen soil zones or by exclusion from these zones in which case you would expect solutes to increase in concentration in the unfrozen groundwater and soils. Artificial freezing may be used for control of contaminant migration such as confining a spill location. There also exists the possibility that by ground freezing, contaminated groundwater and soil may have the concentrations of solutes changed if the freezing process can be controlled. This may be effective in containing solutes with very rapid freezing rates or possible excluding them with slow rates of freezing. Techniques of this type would be useful in the disposal of waste pit materials.

The scale of observation is very important in considering solute transport under freezing conditions. Applying observations and theories from laboratory scale experiments to field situations or applying field data into theories of solute interactions at a pore or molecular level should be done with caution. It is important that a complete understanding of solute transport and interactions be made at all scale levels so that a better understanding of natural freezing conditions and planning for artificial freezing conditions may be made.



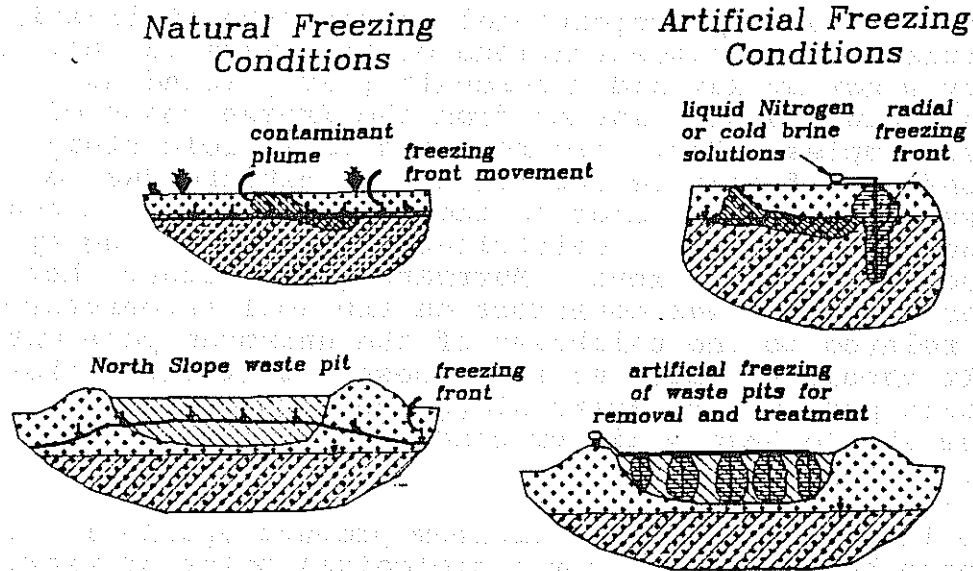


Figure 1. Problems involving groundwater contamination and freezing processes (after Lilly, 1989).

**BACKGROUND**

Alaska is not without groundwater quality problems. Munter and Maynard (1987) reported 72 groundwater contamination sites in Alaska. Many more sites probably exist that are not documented. Petroleum related contamination accounted for 68 percent of the known locations and seven percent were due to salt water intrusion. Even though organic contamination is more prevalent in Alaska, where ground freezing conditions are common, few studies have dealt with the interactions that are taking place under these conditions. Kay and Perfect (1988) reported that research efforts in chemical interactions in freezing ground have mainly dealt with salt ion solutes in soils and the problems of subsea permafrost. The processes that have been evaluated for salt ion migration in freezing soils must be viewed with scrutiny.

Hallet (1978) reviewed the current literature dealing with solute movement in freezing soils. His work indicated that salt ions were the solutes most considered in the available research. Hallet (1978) also reported that cryogenic precipitates resulted from the exclusion of solutes from the formation of pore ice and the consequent concentration increase of solutes in the unfrozen pore water. Baker and Osterkamp (1988) concluded that density flow and exclusion effects of NaCl solutes are dominant processes in freezing soils and that the amount of salt

rejection is inversely proportional to the rate of freezing. Their initial solute concentrations of NaCl were 35 ppt. A laboratory study by Kay and Groenevelt (1983) found no effective exclusion of chlorine from the frozen zones of their test samples. They also reported on a field study which found significant redistribution of nitrate due to convective transport of water to the frozen zones. It could not be concluded that any redistribution was occurring by the advancing freezing front. Murrmann (1973) found that sodium ion diffusion was dependent on the soil temperature which is related to the thickness of the unfrozen pore water films. It should be expected from these observations that the unfrozen pore water layers along the grain surface boundaries should have a higher concentration of ions in solution.

Some investigations of petroleum product spills in cold regions have been studied from a biological point of view. Atlas (1977) showed that the biodegradation of hydrocarbons was greatly reduced during the winter and that volatilization is reduced by ice cover. Horowitz, et al. (1978) showed similar results for a gasoline spill at Barrow, Alaska.

Iskandar and Jenkins (1985) have studied the effects of freeze-thaw cycles on dredge materials with water contents of 160 to 170% by weight. They spiked the dredge material with either heavy metals or a series of organics. Their samples were frozen from the bottom up. The heavy metals tested were cadmium, zinc, copper and nickel in concentrations of 400 to 800 ppb. The organics tested were chloroform, benzene, toluene and tetrachloroethylene in concentrations of 40 to 50 ppb. Their results for the heavy metals were inconclusive. The concentration of organics in the leachate was higher for unfrozen treatments than from freeze-thawed samples. The system which they were analyzing was very complex, a more basic experimental design may have allowed them more conclusive interpretations.

Freezing of contaminated soils may be classified as artificial or natural based upon the controlling thermal regime. Many groundwater contamination locations are effected by seasonal freezing conditions which play a role in the transport and fate of organic solute contaminants. Liquid gas pipelines and commercial cold storage buildings are examples of possible artificial freezing. Excavations, sludge pit removal projects and contaminant control are additional examples of artificial freezing (Iskandar and Houthoofd, 1985, Iskandar and Jenkins, 1985, Sullivan, et al., 1984). Artificial freezing in construction projects has been used successfully for many years. Sludge pits may be frozen so the mechanical properties of the soils increase

and may be removed and disposed of more economically. The use of freezing for contaminant control and possible manipulation is still in the research stages. A good economic analysis of the methods involved is given by Sullivan, et al. (1984), who use freezing basically as a containment process, somewhat like a grout curtain. Liquid nitrogen or freezing brine and glycol solutions are pumped into closed system wells to cause freezing fronts around or through the contaminated areas. The benefits of the ground freezing process are that it can be used for all types of soils, the introduction of environmentally harmful products is limited and the methods allow for flexible boundary conditions. These methods may prove to be particularly useful in cold environments such as the arctic, where naturally occurring freezing temperatures may be used to help in the control of soil and groundwater contamination.

### TRANSPORT PROCESSES

There are many processes that must be considered when considering solute transport. Advection of the bulk unfrozen pore water is driven by pressure and thermal gradients which are affected by the solute concentrations in the pore water. Mechanical dispersion of the solutes is affected by the freezing process by the changes that occur in the unfrozen pore water boundaries, the change in the flow patterns within the pores due to viscosity and bound-water effects and the increase in the tortuosity of the unfrozen pore water passages. Diffusion may play an important role in soils that remain frozen for extended periods of time such as permafrost. Studies on the interactions of freezing and rates of diffusion for organic compounds in soil have not been found.

### FREEZING INTERACTIONS

One major distinction to introduce is that of primary and secondary mass transport and freezing within the soil columns. Primary freezing and mass transport occur when the pore fluids first begin to undergo phase changes. Williams and Wood (1985) show that in the first hours of freezing, the majority of the pore volume may undergo phase change with an anticipated expulsion of pore fluids due to the volume increase of ice over liquid water (Figure 2). As the temperature in the soil drops, the pore ice volume increases and the general flow of water is to the frozen zones, which is defined as the secondary transport of water. Their experiments show that this secondary transport starts to take place some time between twelve and eighteen hours from

the beginning of the test, and is related to the rate of advancement of the freezing front (Figure 3).

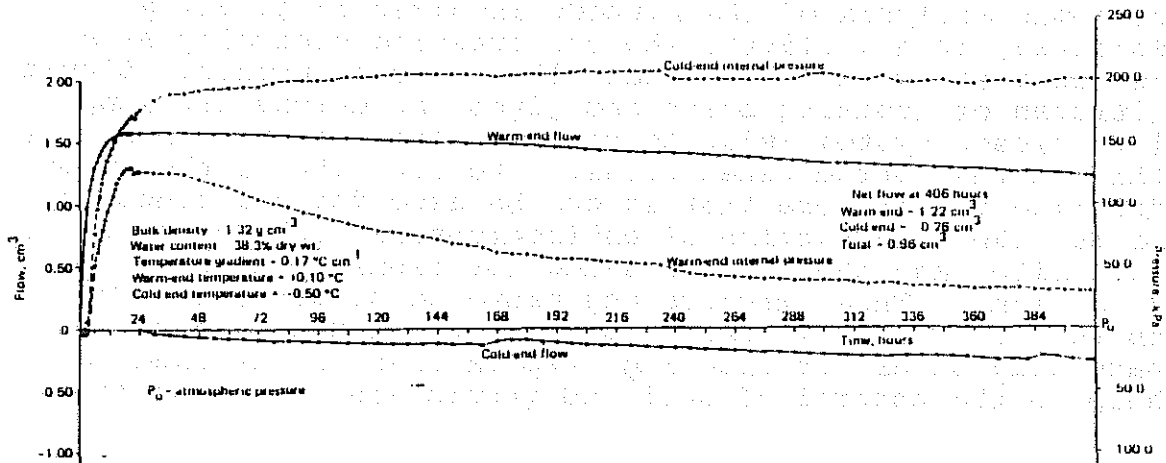


Figure 2. Flows and internal pressures during frost heaving, Caen Silt (from Williams and Wood, 1985).

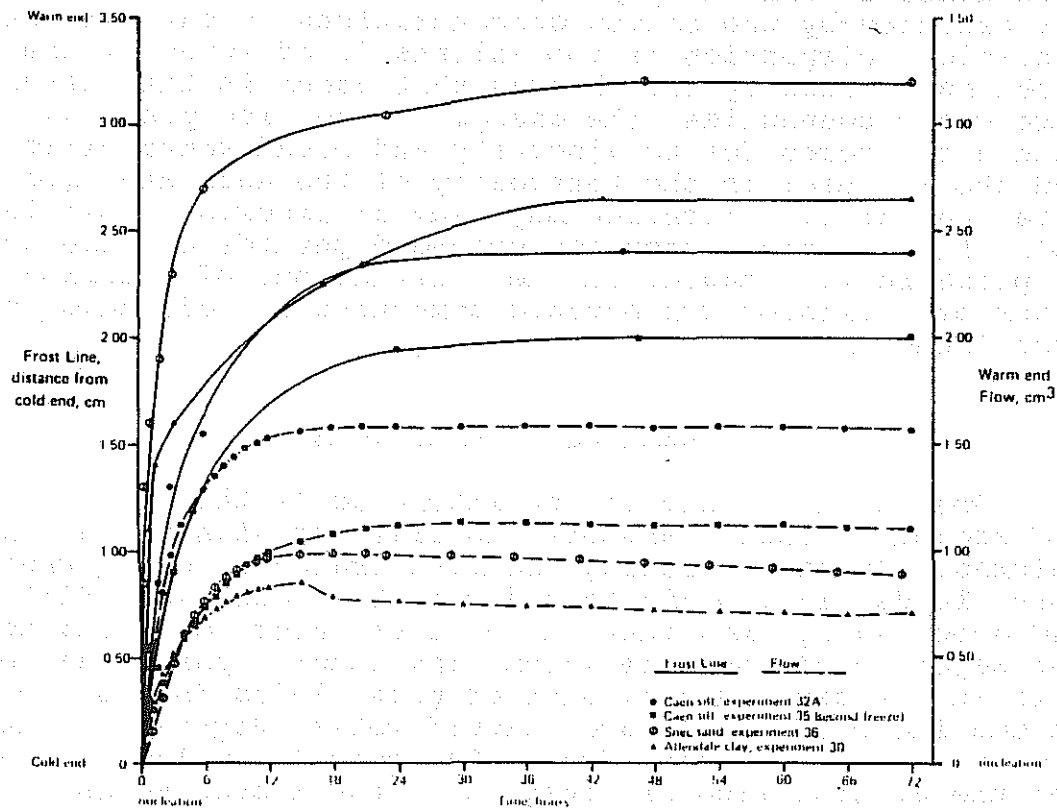


Figure 3. Advance of frost line and accompanying flow of water from warm end of soil (after Williams and Wood, 1985).

## PROBLEM OF SCALE

Any attempt at studying the reported processes involved with solute interactions in freezing soils must be concerned with the scale of reported research. Data is usually collected from field and laboratory scale experimentation. Theories on the processes at work in solute transport are then formed at pore and molecular scales of observation. Some data may be collected from a pore scale, but this has been limited in the past by available detection methods. Information collected on solute transport of organics in freezing soils can be limited in interpretation due to the natural complexity of natural field conditions. This complexity may be reduced and controlled in a laboratory setting. This does not reduce the importance of field data, because most problems that need to be solved are field scale situations.

The freezing of soil at a laboratory scale may be described as a four step process (Tsyrovich, 1975). Figure 4 shows the first step which involves the initial cooling of the sample to the freezing temperature and then further supercooling. The degree of supercooling is affected by the pore size, pore shape distribution, surface area and solute distributions in the pore water (Hallet 1978). Small microparticles may also determine the amount of supercooling by functioning as small nucleation centers. Once crystal formation begins, there is a release of the latent heat of fusion which raises the soil temperatures back up to the freezing temperature of water (step two, Figure 4).

In the third step the temperature does not fall again as pore ice begins to form and fill the pores. This is due to the continued release of latent heat during phase change. When the main volume of the pore has been filled with ice, there is a gradual decrease in the soil temperature. This fourth step is not linear because a small amount of latent heat is still being released and the freezing point of the pore water is dropping due to solute increases and the pressure distribution changes within the pores. However, as the unfrozen film of water around the soil grains gets smaller the temperature decrease does approach linearity. Warming of the soil would be represented in step five.

The assumptions of linear heat flow at the laboratory scale must be viewed with caution. Experimental results may be skewed by radial heat losses in column experiments. Many reports do not indicate the degree of radial or sidewall losses and thus the error involved with the reported data may not be evaluated. Measurements of temperatures in directions parallel and perpendicular to the freezing front are needed in order to evaluate the linearity of the heat

flow. Kay and Perfect (1988) have indicated that few studies of heat and mass transfer in soils have been rigorous in approaching overall equilibrium conditions.

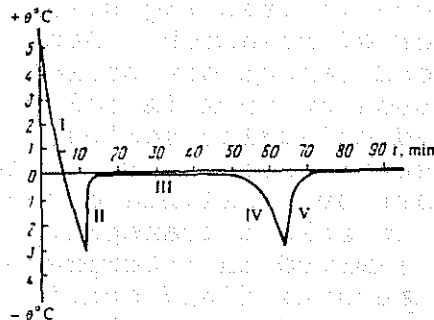


Figure 4. The cooling and freezing curve for a sand (after Tsytovich, 1975).

Chemical data collected at the pore scale is usually representative of a bulk unit volume of material. If solutes are not excluded from the frozen zones, it may be difficult to determine the exact processes involved with containment of the solutes. Questions about relative importances of increased tortuosity of the unfrozen pore water films, entrapment in the pore ice matrix, phase changes in solutes due to solubility boundaries (Figure 5), effects of absorption and adsorption and chemical reactions of solutes must still be approached indirectly. Collection of data at pore scale would greatly increase our ability to interpret the processes controlling solute interactions during freezing.

Considering the problem on a pore scale, organic solute entrapment in the pore ice volume is possible due to the advancement of ice crystal formations around volumes of unfrozen water. Figure 5 shows some of processes that are important at the pore scale. The interactions of the solute with the grain boundaries, advection of unfrozen water out of the pore, exclusion of solutes from the pore ice as it forms are some of the processes that need to be considered. The nucleation characteristics of the initial ice formation also needs to be understood to address this problem (Figure 6). This unfrozen water, if trapped within the bulk pore ice, will decrease in volume as it freezes. This will result in a concentration increase of the solute, which would likely result in a crossing of the solute's solubility boundary. The possibility exists for the formation of solid phase benzene, due to its relatively high melting point of +5.5°C. The rate of freezing is also a process that must be discussed in both a molecular and pore scale level.

The concepts that warrant further consideration at the molecular scale include the size of molecules and how they

may be able to fit into the ice crystal lattice. The character of molecules such as benzene which may interact with free water molecules in such a way as to form weak bonds and become possible nucleation sites during initial ice formation. Further research is needed to define the controlling processes at the molecular level.

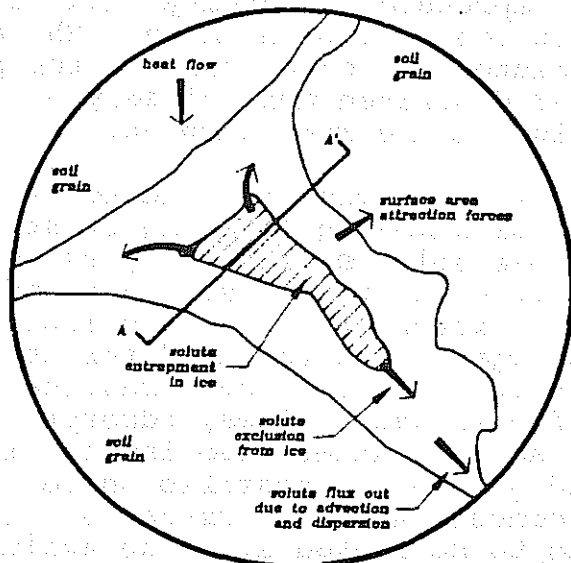


Figure 5. Pore volume diagram of processes involved in the freezing of a saturated media. Area between pore ice and soil grains is unfrozen water (after Lilly, 1989).

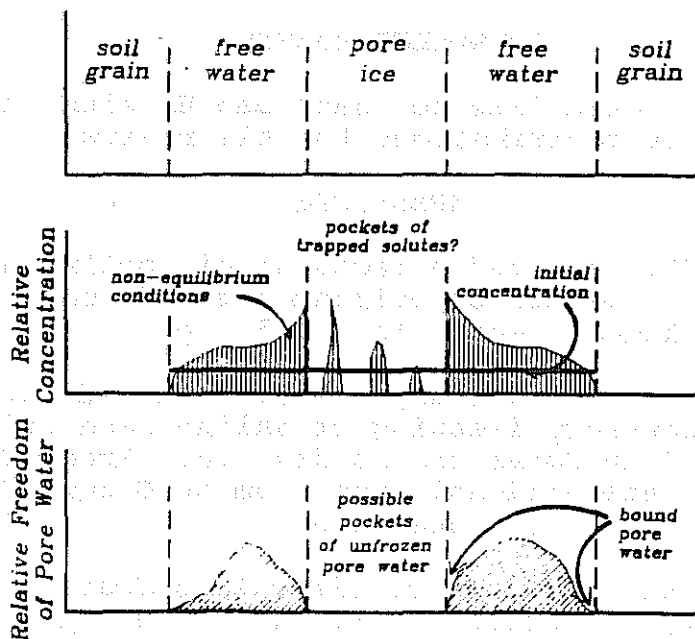


Figure 6. Theoretical concepts along the cross section A-A' as shown in Figure 5.

## CONCLUSIONS

The processes that are involved with saline solutions in freezing soils are becoming more understood. The effects of freezing point depressions, density flow effects, ion exclusion from pore ice have been shown. The contradictions on exclusion of solutes indicates that system parameters must be studied and described more closely as their influences on exclusion are more complex.

There is a lack of conclusive evidence that organics will be excluded from a moving freezing front. The interactions of these solutes with the pore ice matrix is also poorly understood at this time. Rates of freezing, soil chemistry, pore water chemistry and individual chemical properties of the organic solutes need further study. Exclusion of the solutes during ice formation, dispersion of solutes in the unfrozen water films, adsorption and absorption of the solutes under freezing conditions, pore ice nucleation and growth and possible phase changes of the solutes as their concentrations increase are some of the controls that need to be looked at. The exclusion of organic solutes must be described with respect to pore ice formation, freezing front movement and secondary transport to the frozen soil zones.

## ACKNOWLEDGEMENTS

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## SUBSURFACE TRANSPORT OF BTX COMPOUNDS UNDER FREEZING CONDITIONS: A PRELIMINARY MODEL

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Mark A. Tumeo<sup>2</sup>  
Michael R. Lilly<sup>3</sup>

### ABSTRACT

The fate and transport of aromatic hydrocarbons from liquid fuel spills (i.e. benzene, toluene, and xylenes known as the BTX compounds) have recently received much attention from hydrogeologists and engineers. These researchers have presented models which simulate a variety of processes occurring in the subsurface. We have found no models, however, that consider the effects of ground freezing on organic contaminant transport. In this paper, the mathematical development for a two dimensional model which will predict the migration of BTX compounds under freezing conditions is presented. The model is limited to homogeneous saturated porous media under steady, uniform flow conditions.

Some researchers believe that a downward migrating freezing front concentrates contaminants ahead by exclusion: the rejection of solutes from growing ice crystals. This model examines the assumption that the pore scale physics of the freezing front can be simplified on a field scale to solute exclusion and immobilization. Solute exclusion is modeled by a variable exclusion coefficient ( $K_e$ ) which indicates the percentage of solute excluded from the freezing front. If exclusion is assumed not to occur,  $K_e$  is set to zero and the freezing front is modeled as a no-flow boundary behind which solutes are immobilized. If exclusion is assumed to occur, the  $K_e$  value is set to reflect the assumed percentage of exclusion. The non-excluded fraction is immobilized behind the freezing front.

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## INTRODUCTION

It has been estimated that 23% of all underground storage tanks containing hazardous materials are leaking into the subsurface environment (Predpall et al., 1984). Many of these are leaking fuel products which contain potentially toxic solutes and can migrate to the water table. Principle among these potentially toxic solutes are the aromatic hydrocarbons benzene, toluene, and xylenes known as the BTX compounds. BTX compounds pose a particular threat because their maximum solubility (Nyer and Skladany, 1989) exceeds their EPA maximum concentration limit for groundwater used as drinking water (40 CFR, part 141-142) by up to  $5\frac{1}{2}$  orders of magnitude. Because of their large solubility and widespread presence in groundwater, BTX compounds deserve a great deal of study.

The fate and transport of BTX compounds in groundwater has received much study from hydrogeologists and engineers. These researchers have considered a variety of subsurface processes including advection, dispersion, adsorption, biotransformation, filtering, and solution/precipitation. However, apparently none have considered the effects of freezing processes on BTX transport. In this paper, it is hypothesized that the effect of freezing on BTX fate and transport can be modeled by exclusion at and immobilization behind the freezing front. A model based on this hypothesis is developed and illustrated.

The exclusion of impurities from growing crystals has been well documented in several analogous situations. Pfann (1966) describes a process, called zone refining, which can purify a crystalline substance containing impurities in solid solution. In this process, a thawed zone is moved through the material in a carefully controlled manner. Pure substance plus impurity are incorporated into the melt along the thawing front. Along the freezing front, however, less impurity is incorporated into the newly frozen material. In this way, the melt becomes enriched and the solid becomes depleted in impurity.

The theory behind zone refining, called Burton-Prim-Slichter (BPS) theory (see Pfann, 1966), can be used to derive an equation which quantifies the exclusion phenomena. This equation is based on several assumptions; most notable is the assumption that impurities can be included in the solid's lattice in solid solution. However, this equation has also given good results predicting ionic solute exclusion from NaCl ice (Weeks and Ackley, 1982) and freezing soils (Baker, 1987) even though the ionic solute cannot enter the ice lattice in solid solution. The chemistry of BTX compounds is quite different from that of ionic solutes. Hence BPS derived equations that work for predicting ionic solute exclusion at the freezing front may not work for BTX compounds. Even so, the fact that exclusion has been documented in these disparate circumstances suggests that exclusion of BTX compounds may also occur.

Some recent experimental work suggests that aromatic hydrocarbons are re-distributed by freezing processes. Iskandar and Jenkins (1985) froze soil slurries spiked with benzene, toluene, and a number of other contaminants. They froze these slurries from the bottom upwards. It was found that a single freezing re-distributed part of the mass of aromatics upwards. Several freeze-thaw cycles enhanced the loss of these contaminants, presumably through volatilization at the upper boundary. Finally, these authors also found that by keeping the slurries frozen, benzene and toluene were immobilized.

In this paper, a two dimensional model is proposed which predicts the migration of BTX compounds under freezing conditions. The treatment is limited to homogenous saturated porous media experiencing steady, uniform flow. The approach taken here will be quite general; simplifications can be made to suit specific circumstances or situations with limited data. Modifications can easily be made to accommodate new theories of the interactions between BTX compounds, the freezing front, the porous media, the vadose zone, and other solutes.

## MODEL DEVELOPMENT

Typical mass balance based differential equations for solute transport are developed in Freeze and Cherry (1979), de Marsily (1986), and Javandel et al. (1984). A two dimensional form of these equations for steady, uniform flow in the  $x$  direction is:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x} \left( [D]_x \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial y} \left( [D]_y \frac{\partial C}{\partial y} \right) + \frac{\partial C}{\partial x} V_x + \sum_1^r R(C, t) + \sum_1^s S \quad (1)$$

where  $C$  is concentration,  $t$  is time,  $x$  and  $y$  are distance in the  $x$  and  $y$  directions respectively,  $[D]_x$  and  $[D]_y$  are the dispersivity tensor transformed into the  $x$  and  $y$  directions respectively,  $V_x$  is the darcian flow velocity in the  $x$  direction,  $R(C, t)$  represents  $r$  concentration and/or time dependent reactions (i.e. adsorption), and  $S$  is  $s$  source or sink functions at boundaries.

Equation (1) can account for dispersion, advection, adsorption, biotransformation, filtering, and solution/precipitation processes. To account for exclusion, another term must be added to (1). At a moving freezing front, some percentage of dissolved BTX compound may be excluded from the frozen material. The unexcluded solute is left behind and remains immobilized in the frozen material. In this way the freezing front can be modeled as a moving no-flow boundary along which part of the mass of solute is concentrated into the underlying solution.

Consider two small elements of aquifer: the first is just below the present position of the freezing front and the second is just below the first. The volume of each element is  $d_x d_y d_z$  ( $d_z$  was assumed equal to one throughout the model

development). The mass of solute in the first element is  $Cd_x d_y$ . If the freezing front traverses the first element, then the mass of solute excluded ( $M_e$ ) from the first element is given by:

$$M_e = K_e C d_x d_y \quad (2)$$

where  $K_e$  is the exclusion coefficient representing the ratio of excluded solute to initial dissolved solute. Note that  $M_e$  is the mass of solute excluded from the first element and excluded to the second element. Rearranging (2) to find the mass of excluded solute to per unit volume and adding the result to (1) yields:

$$\frac{\partial C}{\partial t} = \frac{\partial}{\partial x} \left( [D]_x \frac{\partial C}{\partial x} \right) + \frac{\partial}{\partial y} \left( [D]_y \frac{\partial C}{\partial y} \right) + \frac{\partial C}{\partial x} v_x + \sum_1^r R(C, t) + \sum_1^s S + \Omega K_e C \quad (3)$$

where  $\Omega = -1$  for the first element (the one traversed by the freezing front),  $\Omega = +1$  for the element below the first, and  $\Omega = 0$  for all other elements in the flow regime.

### MODEL ILLUSTRATION

Explicit central finite difference FORTRAN code for (3) has been written. All processes besides advection, exclusion, and immobilization have been disabled (i.e. the first, second, fourth, and fifth terms on the right side of (3) have been set to zero) so that the effects of freezing might be better illustrated.

To solve (3) several assumptions were made:

1. Aquifer material is homogeneous and isotropic.
2. No co-behavior among solutes exists. In its present form, the model only considers one solute. It must be assumed that other solutes do not affect the behavior of the modeled solute.
3. The modeled solute must remain in solution. No non-aqueous phase liquids are considered by this model.
4. The solute must have negligible affect on solution freezing point, density, viscosity, and latent heat of fusion.
5. Advection upwards toward the freezing front is negligible.
6.  $K_e$  remains constant over the time considered.
7. The hydraulic conductivity in the unfrozen material remains constant over the time considered.

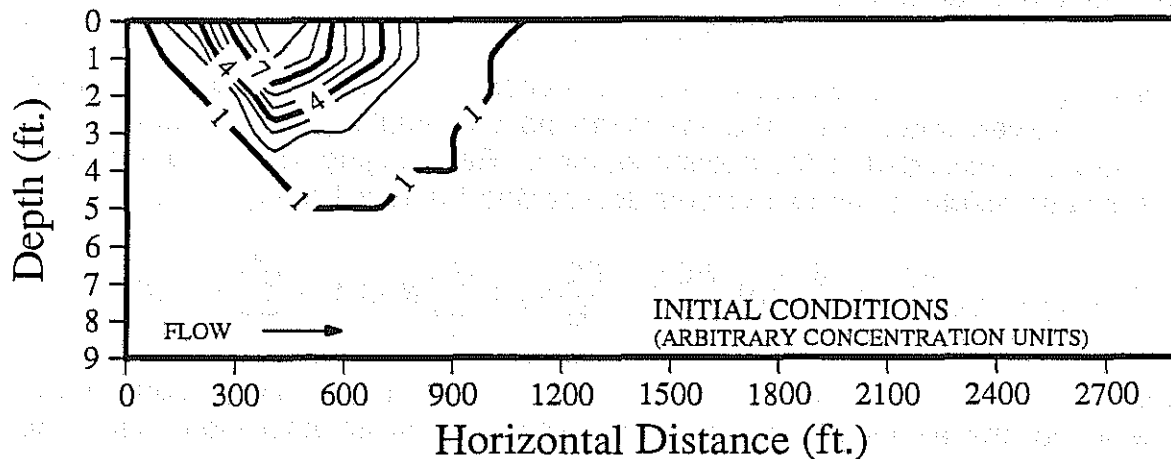


Figure 1. Initial plume used for all three scenarios.

8. Water and solute do not move in the frozen soil.

Figure 1 shows concentration contours (arbitrary concentration units) in a hypothetical contaminant plume. The transport of this plume under three scenarios was examined: 1) no freezing (advection only), 2) freezing but little exclusion, and 3) freezing and much exclusion. The hypothetical aquifer through which this plume migrates has a hydraulic conductivity of 280 ft/day (a typical value for a clean sand) and a water table slope of 0.011 ft/ft. The distance between nodes in the  $x$  and  $y$  directions is 100 feet and 1 foot respectively. The time of transport will be 240 days in 20 day time steps. For the two freezing cases, a hypothetical freezing front penetrated the aquifer to a depth of 4 feet over the 240 day simulation.

Figure 1 and figure 2 (the advection-only case) show that advection merely translates the plume in the direction of flow (i.e. to the right). The plume in figure 2 has spread out relative to the initial plume but this is an artifact of numerical diffusion and is not due to advection or freezing.

A comparison of figure 2 and figure 3 (the freeze-2% exclusion case) shows that freezing with little exclusion retards solute movement in the aquifer. This retardation decreases with depth and is proportional to the time a particular location remains immobilized after passage of the freezing front. This is shown in a more quantitative manner in figure 5. Figure 5 is a residual map obtained by subtracting the values shown in figure 2 from those in figure 3. Note the large negative area in the upper right corner of the plume. This represents the region of solute retardation due to immobilization.

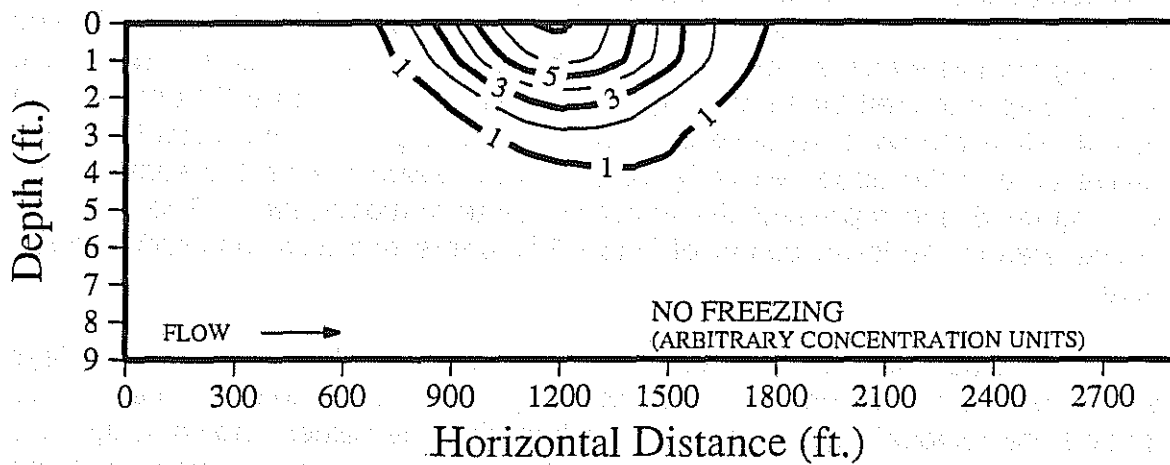


Figure 2. Plume after 240 days transport without freezing.

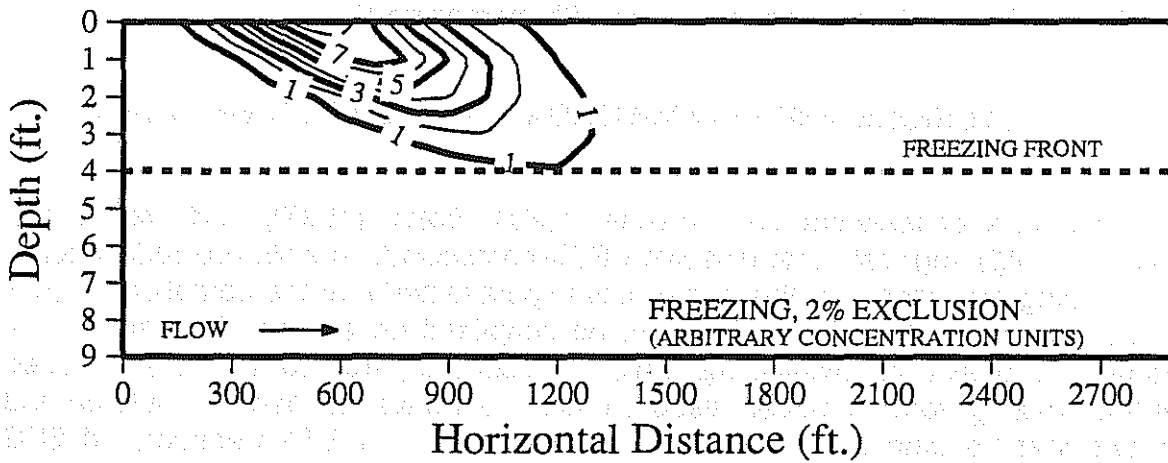


Figure 3. Plume after 240 days transport, freezing, and 2% exclusion.

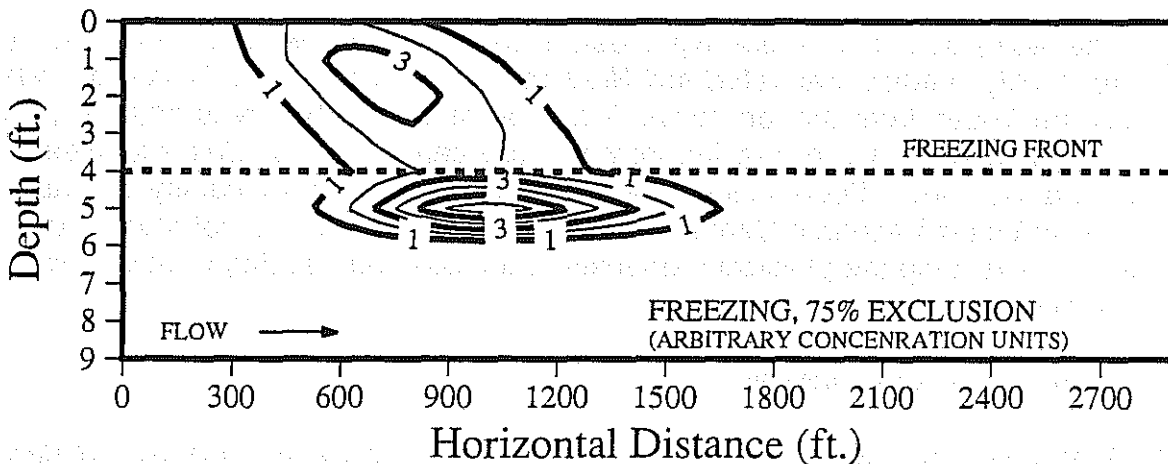


Figure 4. Plume after 240 days transport, freezing, and 75% exclusion.



A comparison of figure 2 and figure 4 (the freeze-75% exclusion case) shows that exclusion redistributes solute downward in the aquifer. Immobilization is also occurring but this effect is masked by the effects of exclusion. Figure 6 shows the residual map obtained by subtracting the values shown in figure 2 from those in figure 4. Note the wide negative area in the upper right and the football shaped positive area in the upper left of figure 6. These features have the same shape as on figure 5 and represent the effect of solute immobilization. The compact positive area in the lower center of figure 6 is where exclusion has redistributed solute.

A comparison of figures 3 and 4 indicate that the exclusion coefficient plays an important role in determining the final shape of a contaminant plume. Figure 7 shows the residual map obtained by subtracting the values shown in figure 3 from those in figure 4. This figure shows only the influence of exclusion on plume shape; the effects of immobilization and advection have been subtracted out. Note that exclusion acts to redistribute solute from the upper left to the lower center of the figure. This redistribution increases with increasing  $K_e$ .

#### CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE WORK

The work of Iskandar and Jenkins (1985), Baker (1987), and Weeks and Ackley (1982) suggests that dissolved BTX compounds in soils are redistributed by freezing processes. In this paper, it is hypothesized that the complicated pore scale physics of this phenomena can be simplified on a larger field scale to a variable exclusion coefficient,  $K_e$ , which reflects the percentage solute excluded at the freezing front. A model based on this hypothesis has been formulated and shows that freezing can have profound implications for BTX transport. If BPS derived equations (Pfann, 1966) also describe BTX exclusion phenomena,  $K_e$  would prove a readily calculable quantity.

The next step in testing our hypothesis is an experiment like that performed by Baker (1983). Vertical saturated soil filled columns spiked with BTX compounds should be frozen from the top down. If a good fit to BPS theory is obtained, an expression relating  $K_e$  to freezing front velocity can be found (Cox and Weeks, 1975; Baker, 1987). Such experiments are underway at the University of Alaska Fairbanks Water Research Center. If a good fit to BPS theory is not found, some new theory relating the physical properties of the soil and freezing front dynamics to  $K_e$  must be found.

Other future work includes:

1. At present this model makes no provision for temperature induced changes in water density ( $\rho$ ) and viscosity ( $\mu$ ). This may prove important for low temperature solute transport because hydraulic conductivity is a function of these

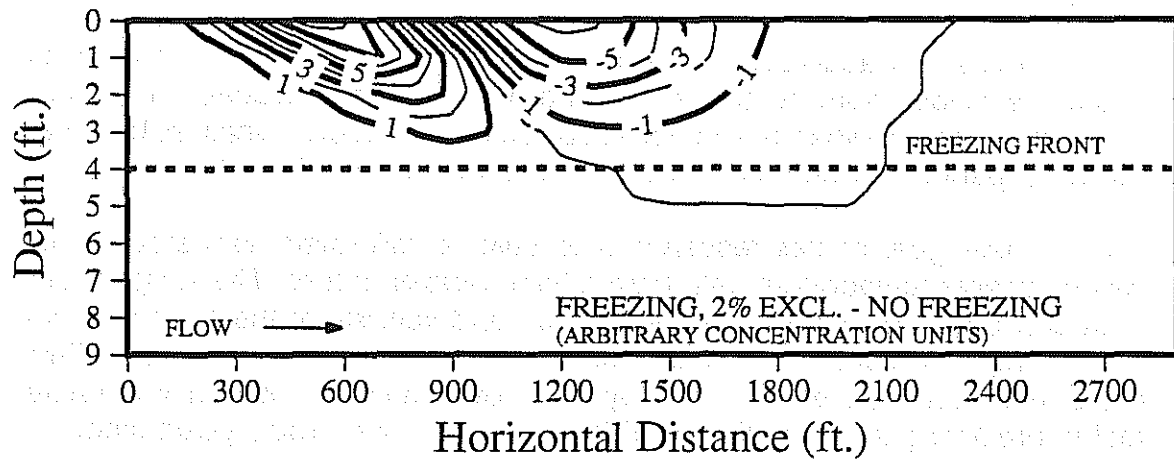


Figure 5. Residual map showing difference between figures 2 and 3.

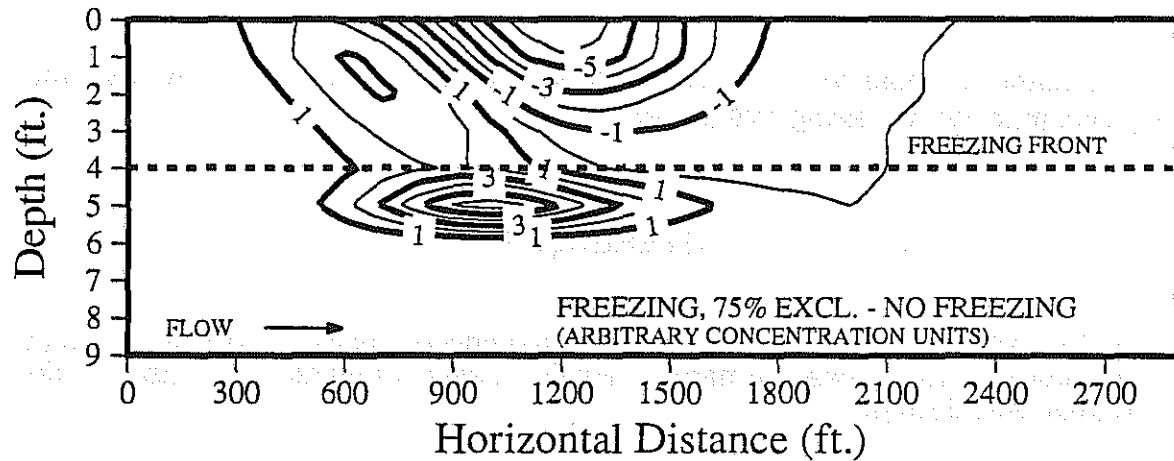


Figure 6. Residual map showing difference between figures 2 and 4.

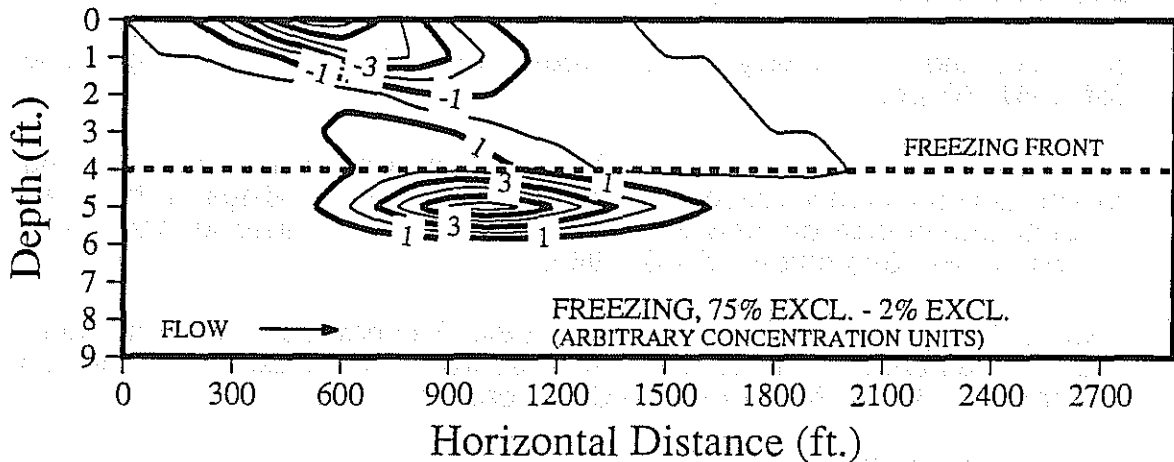


Figure 7. Residual map showing difference between figures 3 and 4.

fluid properties.

2. This analysis has addressed problems concerning freezing only. In order to make the model more complete, thawing must also be included. It is likely that thawing merely liberates immobilized solute without complication, however further experimental work should be done to validate this hypothesis.
3. The ultimate goal of this research is to build a calibrated, validated model which accepts temperature data from a heat transfer model. The temperature data can be used to calculate the position and velocity of the freezing front and changes in fluid viscosity and soil properties at every time step. This combined heat transfer/solute transport model would provide an invaluable tool to predicting the migration of BTX compounds over many years time.

### ACKNOWLEDGEMENTS

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RATE-LIMITED VAPOR EXTRACTION OF DISSOLVED/ADSORBED  
GASOLINE CONSTITUENTS: SOLDOTNA, ALASKA

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ABSTRACT

Vapor extraction of sandy sediments containing no apparent free-phase product has provided effective removal of gasoline hydrocarbons. A 3-well extraction system was designed to withdraw vapor from a 40 foot-thick vadose zone composed of glaciofluvial materials. Initial soil concentrations were generally below 100 mg/kg total petroleum hydrocarbons. Over 128 days of extraction, approximately 2400 lbs. of hydrocarbons were removed. A 2-dimensional compressible flow solution proved adequate for design purposes, yet indicates that fingering and/or non-laminar flow probably occurs. Discharge concentration histories suggest that removal rates are limited by the rate of volatilization from dissolved/adsorbed phases. Increases in the aromatic fraction over time may reflect differential phase change rates. Finally, pulsed and cyclic extraction met with mixed results, although generally favorable.

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