Ecology of an impacted northern Rocky Mountain stream

Darlene Solberg Nardi

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ECOLOGY OF AN IMPACTED, NORTHERN ROCKY MOUNTAIN STREAM

By

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Chairman, Board of Examiners

Dean, Graduate School

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This study was intended to examine, describe and compare the water quality of lower Ashley Creek. Water samples were taken from four sites above the Kalispell Sewage Treatment Plant (KSTP) and two sites below. Community respiration studies were conducted twice.

The upper site of this study was affected by the small lakes and wetlands in the upper drainage. The influx of organic matter lowered pH, and on five of the six sampling dates lowered dissolved oxygen (DO) to levels below the state standards. Total persulfate nitrogen (TPN) was 2-4 times higher at that site and ammonia was 3-12 times higher than the concentrations in other impacted creeks in the area.

Among the many changes, in water quality, caused by the KSTP were: a 300% increase in TPN concentration, a 350% increase in soluble reactive phosphorus (SRP) concentration, and more extended periods of depressed DO concentrations.

Water quality standards or criteria set by the State and/or EPA were exceeded for: turbidity at the lower 3 sites; chlorine in the KSTP effluent; ammonia below the KSTP; dissolved oxygen concentrations below the KSTP and below Smith Lake; and probably pH just above the KSTP.

In order to mitigate the effects, of Ashley Creek, on the Flathead River/Lake, nonpoint sources of pollution must be reduced and riparian areas must be reclaimed. If the newly constructed KSTP also causes the water in Ashley Creek to drop below State and EPA criteria and standards, then land application or wetland treatment should be evaluated.
ACKNOWLEDGEMENTS

Thanks to the sponsors and committee members of the B & B Dawson Scholarship and the Clancy Gordon Environmental Scholarship. The laboratory analyses and the supplies for this study were very costly and the financial assistance was greatly appreciated.

Thanks to my committee members, and particularly to Vicki Watson for her interest in the study. I could always count on her for advice and support.

The faculty, staff and students at the University of Montana, Flathead Lake Biological Station provided expertise and assistance. Their assistance and patience in both the field and the lab was paramount to the success of the study. The contracted analyses at that lab, were carried out with precision and care.

And of course, thanks to my family. As usual, they were always helpful and supportive.
# TABLE OF CONTENTS

**ABSTRACT** ................................................................. ii

**ACKNOWLEDGEMENTS** ................................................ iii

**TABLE OF CONTENTS** .................................................... iv

**LIST OF TABLES** .......................................................... vi

**LIST OF FIGURES** ........................................................ vii

**INTRODUCTION** .......................................................... 1

- FLATHEAD BASIN ......................................................... 1
- ASHLEY CREEK .............................................................. 2

**STUDY AREA** ............................................................... 4

- ASHLEY CREEK DRAINAGE ............................................. 4
- SAMPLE SITES ............................................................ 7
- KALISPELL SEWAGE TREATMENT PLANT ....................... 10

**METHODS AND MATERIALS** .......................................... 14

- WATER CHEMISTRY STUDY ............................................ 14
- RESPIRATION STUDY .................................................... 17
- MACROINVERTEBRATES ............................................... 21
- CHLOROPHYLL a ........................................................... 22
- ASH FREE DRY WEIGHT ................................................. 23
- SURFACE AREA ............................................................ 23
- DARK - LIGHT BOTTLE .................................................. 23
- DIURNAL OXYGEN STUDY ............................................. 24
- FLOW .......................................................... 25

**RESULTS AND DISCUSSION** .......................................... 26

- FLOW VELOCITY AND VOLUME ....................................... 26
- ALKALINITY ................................................................. 29
- TURBIDITY ................................................................. 32
- SPECIFIC CONDUCTANCE ............................................. 35
- pH .......................................................... 37
- TEMPERATURE ........................................................... 41
- CHLORINE ................................................................. 43
- PHOSPHORUS COMPOUNDS .......................................... 45
  - SOLUBLE REACTIVE PHOSPHORUS ................................. 48
  - TOTAL PHOSPHORUS .................................................. 51
LIST OF TABLES

Table 1. Biophysical Variables and Methods Used in Monitoring Water Quality in Ashley Creek, 1990. .............................................................. 16

Table 2. Biophysical Variables Measured at the Six Study Sites on Ashley Creek, 1990. .................................................................................. 28

Table 3. Temperature, pH, Dissolved Oxygen and Specific Conductance Measured at Six Sites on Ashley Creek, 1990. .......................... 38

Table 4. Water Chemistry Analyses - Nutrients, Ashley Creek, 1990. .............................................................................................................. 46

Table 5. Concentrations of Nitrogen and Phosphorus Variables at Various Flathead Drainage Monitoring Sites ................................. 54

Table 6. Macroinvertebrate Totals and Densities Sampled at Six Sites on Ashley Creek, 1990. ................................................................. 97
LIST OF FIGURES

Figure 1. Ashley Creek study area ............................................................... 8
Figure 2. Photosynthesis/respiration chamber ...................................... 19
Figure 3. Estimated flow at three sites on Ashley Creek .................. 30
Figure 4. Alkalinity .................................................................................. 31
Figure 5. Turbidity .................................................................................. 34
Figure 6. Specific conductance ............................................................... 36
Figure 7. pH ................................................................................................. 39
Figure 8. Temperature ............................................................................... 42
Figure 9. Soluble reactive phosphorus load ............................................ 50
Figure 10. Total phosphorus load ............................................................. 52
Figure 11. Ammonia load ......................................................................... 61
Figure 12. Nitrite/nitrate load ................................................................. 64
Figure 13. Nitrification .............................................................................. 66
Figure 14. Total persulfate nitrogen load ................................................... 70
Figure 15. Dissolved organic carbon ....................................................... 73
Figure 16. Nondissolved organic carbon .................................................. 74
Figure 17. Dissolved Oxygen .................................................................. 77
Figure 18. Diurnal oxygen study ............................................................ 79
Figure 19. July respiration study ............................................................. 84
Figure 20. September respiration study .................................................. 85
INTRODUCTION

FLATHEAD BASIN

High quality water is one of the prized assets of the Flathead Valley. The preservation and recreational use values of the Flathead Lake/River ecosystem were estimated in 1983 at more than one hundred million dollars per year (Zackheim and Cooper 1983). Increased degradation of the 495.9 km² oligotrophic lake could have widespread consequences to the Flathead area including transformation of the lake community and negative effects to the local economy.

In 1983 two important documents - the *Limnology of Flathead Lake Final Report* (Stanford et al. 1983) and the *Flathead Basin Environmental Impact Statement* (Zackheim and Cooper 1983) were completed. Both publications indicated that Flathead Lake was suffering from cultural or accelerated eutrophication caused by an influx of nitrogen and phosphorus. (See Appendix A for eutrophication discussion.)

In 1983 Flathead Lake also experienced its first lake-wide algal bloom (Stanford et al. 1983; Bahls 1986). These events attracted the attention of agents of the State Water Quality Bureau who responded by preparing a "Strategy for Limiting Phosphorus in Flathead Lake". The six-point plan included a stipulation that the valley's sewage treatment plants limit the concentration of phosphorus in effluent to 1 mg/l. The Kalispell Sewage Treatment Plant (KSTP) began removing phosphorus April 14, 1989 (L.
Jenkins pers. comm.) in compliance with their discharge permit schedule.
The plan also allowed Flathead and Lake Counties to mandate that no phosphorus containing (> 0.5%) household-cleaning products could be sold within the counties [Montana Department of Health and Environmental Sciences (MDHES) 1985; Flathead County 1986].

ASHLEY CREEK

One of the many tributaries of the Flathead River, Ashley Creek, has not received much concern for its own sake but has received attention because it receives the KSTP effluent and routes it to the Flathead River and ultimately to Flathead Lake. Montana Surface Water Quality Standards indicate that Ashley Creek water quality deteriorates as it travels from its source to the Flathead River (MDHES-WQB 1988a) and at its confluence with the Flathead is of lower quality than that of the Flathead. If its water quality were improved, Ashley Creek has potential for increased beneficial uses and a positive influence on both Flathead River and Flathead Lake (MDHES 1986).

Ashley Creek water quality is a product of many factors including the influx of a variety of materials from both point and nonpoint sources in the drainage and the biotic and abiotic reactions within the creek. Once pollutants have entered the creek, they interact with one another, with the materials already present in the water and with the creek substrate. (See
Appendix B for discussion of pollutants.) Because of all these variables, Ashley Creek, like every stream and river, is a unique system that must be studied by considering its total environment (Connell and Miller 1984; Eschenroeder 1981; Stern and Walker 1978).

Because the creek seems to be heavily impacted by various pollution sources (see Appendix C for loading discussion), this study had three main objectives: 1) to describe the spatial and temporal changes in water chemistry and oxygen demand in lower Ashley Creek and attempt to relate these changes to natural and anthropogenic loading, 2) to show to what degree (if any) the KSTP effluent affects the benthic community as measured by oxygen demand, and 3) to develop tactics and/or strategies for water-resource and pollution management. The first objective was particularly important since the water chemistry and oxygen demand data provide a baseline for any future studies that may be done. During this study the city of Kalispell was planning construction of a new sewage treatment plant which was completed in 1992. The updated plant is adjacent to the old plant and also discharges effluent into Ashley Creek. The new plant is expected to be more efficient at reducing nutrient and organic loading to the creek.
STUDY AREA

ASHLEY CREEK DRAINAGE

Ashley Creek Drainage encompasses approximately 728 km² (Dutton 1987) in Flathead County, northwestern Montana. The Montana Department of Resources and Conservation (DNRC) reported in the *Upper Flathead River Basin Study* (1977) that the annual average flow of Ashley Creek at the outlet of Ashley Lake was 18.8 cubic feet per second (cfs) and the average flow below Smith Lake after the confluence of Truman and Mountain creeks was 30 cfs. Gary Anderson, stated the annual average flow at the outlet of Ashley Lake was 15.8 cfs (pers. comm. 5-26-89). If both of these readings are accurate, then flow in the creek has decreased in the last 15 or so years. Like most western Montana streams and rivers, the flow of Ashley Creek varies tremendously on a seasonal basis. A three year average flow (1984 through 1986) taken near Kalispell (south end of Meridian Road), indicated a range in flow of 10+ cfs in August to 90+ cfs in April and May [Montana Department of Fish Wildlife and Parks (MDFWP) open file].

The creek is about 71 km long, averages 1.5 - 4.5 m wide, and .6 - 1.2 m deep (Dutton, 1987). It has a low gradient and is consequently slow flowing. At times the KSTP effluent comprises 20% of the total flow of the creek (L. Jenkins, pers. comm.).

Ashley Creek originates at and is the outlet of Ashley Lake, located in
northwestern Montana (T28N,R24-23W). An impoundment is located on the southwest side of Ashley lake. This dike, believed to be constructed in 1928 (Ashley Lake Irrigation District), was refurbished in spring 1990. The Ashley Irrigation District stored water rights, along with the dam and headgate, were purchased August 31, 1978 by the MDFWP and 14 other parties for such purposes as irrigating, domestic stock watering, small lake rejuvenation, and waterfowl nesting areas. The MDFWP purchased 9,540 acre feet for fish and wildlife and 1,908 acre feet for dilution of the Kalispell City Sewage Treatment Plant (KSTP) effluent. In 1978 the City of Kalispell denied responsibility for the effect of their effluent on Ashley Creek by refusing to purchase water rights for its dilution. The Montana Fish, Wildlife and Parks Department recognized the hazard and secured 1,908 acre feet (2.6 cfs) for dilution of the effluent.

Between 1931 and 1950 (before the MDFWP assumed responsibility of coordinating flow) a stream gauge 2.5 miles below Smith Lake revealed a maximum discharge of 749 cfs (May 27, 1948) and a minimum of 0 which occurred during summer months when the creek was completely dewatered by irrigators (Dutton, 1987). Because of low water in 1989, irrigators drawing water from Ashley Creek were asked to reduce extraction during low-flow (G. Anderson pers. comm.); instream flow had reached a potentially dangerous level. In this study, flow was found to affect various water quality parameters.
From Ashley Lake, Ashley Creek flows southeast in a broad meandering curve. The upper creek flows through some small marshy lakes, some barrier falls, agricultural fields and pastures and then into a larger, marshy lake, Smith Lake. Below Smith Lake it flows through more rural districts, housing developments, small industries and the south edge of Kalispell. Below Kalispell and the KSTP, a meat packing plant has waste water on the flood plain of, and extremely near Ashley Creek. Below there, and above the confluence with the Flathead River, land use along the creek is mainly agricultural fields and pastures.

The bedrock present in most of the Ashley Creek watershed is dominated by limestone and calcareous argillite (Dutton 1987). These rocks produce soils with silty textures that have low permeability and water yield. During the Tertiary period the mountains uplifted and then eroded, filling the valleys with sediments. Later events reworked the sediments and deposited glacial till and alluvial material in the drainage. During the Ice Age the course of Ashley Creek was shifted around and lakes were dammed up and then released. The deposits again were calcareous with silty textures, but also contained sandy or gravelly layers and organic accumulations from the lakes. The geology of the watershed has important water quality implications including sulfur, ammonia and organic loading. During more recent times volcanic ash has been added to the soils - up to 8 inches in places (Dutton 1987). The series of past
events have resulted in a drainage dominated by silty-calcareous soils with occasional sandy-gravelly soils and occasional organic deposits, each in its own way influencing the chemistry of Ashley Creek.

**SAMPLE SITES**

This study was conducted on the lower portion of Ashley Creek. Since cost would not permit an extensive study on the entire creek, only the lower portion, which was suspected to be most heavily impacted, was included. Six sites were strategically chosen for their location and accessibility (Figure 1). The first and most upstream site was located less than one mile (creek distance) below Smith Lake at the junction with Big Horn Road (R22W,T27N,S32). The area surrounding Smith Lake and the entire east portion of the lake is marsh, covered with organic soils and wetland vegetation.

The second site was near Derns Road (R22W,T28N,S15). This site was in a riffle community with low banks and no shade. Water velocity appeared to be greater than at the other sites. The cobble substratum (64-256 mm diameter rocks) supported that conjecture. The area between these upper two sites is primarily agricultural; parts of the stream banks retain healthy riparian vegetation while others are thinly vegetated or even barren.

The third site was immediately below the railroad bridge west of
Figure 1. Ashley Creek study area, Flathead County, MT.
Meridian Road (R21W,T28N,S13). This site is below Kalispell Wood Products, a small pond that is also fed by streams originating at nearby springs and the inflow of the City of Kalispell's storm drains. Kalispell Wood Products is no longer operated as a sawmill but rather as a "value added" mill. Although Kalispell Wood Products does not "discharge" into Ashley Creek some sources have reported sawdust present in the creek on the property. It is also possible that runoff from the site is contaminated with oil and grease from the equipment on the premises. The substrate at Meridian is mostly silty with only a few large rocks. At the time of the respiration study in September there was a lawn mower, a bicycle, numerous 55 gallon barrels, automobile tires and hubcaps, bones, various unidentifiable pieces of metal and a tremendous amount of lumber lying in the creek. Floating lumber jammed against the bridge supports also detained floating foam and brown scum. Ultimately, during the respiration study in September, a dead chicken came floating down the creek.

The fourth site was immediately above the Kalispell Sewage Treatment Plant [and Stampede (meat) Packing Company] near Airport Road (R21W,T27N,S19). Between sites three and four the creek flows through the southwestern (residential) portion of Kalispell and two more holding ponds from Kalispell storm drains empty untreated water into the creek. Some people believe that many of the older west side Kalispell homes drain their waste directly into the creek in that area.
The fifth site, near Cemetery Road (R21W, T27N, S29) was approximately one and one half miles (creek distance) below the KSTP and also the site at which the Montana State Department of Health and Environmental Sciences Water Quality Bureau considers the KSTP effluent "mixed" with the creek (ARM 16.20.634). Accordingly, this site is where the KSTP collects their water samples for self-monitoring of the creek. This site was the only site in the study that was partially shaded.

The sixth and last site was south of Kalispell at the junction with Snow Line Road (R21W, T27N, S33), a few miles above the site where the creek enters the old Flathead River oxbow. The morphology of the upper and lower creek differs greatly. The habitat resembles a "pool" at both of the most downstream sites. They are considerably deeper (4-5 times) than the uppermost site and the substratum is almost exclusively clay, silt and sand [1/256 - 2 mm diameter (Wentworth, 1922)]. The creek meanders considerably more in the lower reach between Cemetery and Snow Line (4.5 times the straight line distance) than it does in the upper reach between Big Horn and Meridian (1.4 times the straight line distance). The approximate distance traveled (in feet) per drop in elevation is 230:1 in the upper creek and 359:1 in the lower creek.

KALISPELL SEWAGE TREATMENT PLANT

The only permitted point source of effluent discharge into Ashley Creek
is the Kalispell Sewage Treatment Plant (KSTP). The Water Quality Bureau of the Montana Department of Health and Environmental Sciences (MDHES-WQB) is the governing body responsible for regulating point source discharges into Montana’s waterways and groundwater. Along with other agencies, they are also assigned to control nonpoint sources of pollution. The Department monitors the operations of the sewage treatment facility and has set discharge standards based on local and regional water quality objectives. The Department has issued the City of Kalispell permit no. MT-0021938, (dated November 1988), under the Montana Discharge Elimination Systems (section 75-5-101 et seq., MCA, and ARM 16.20.901 et seq. and 16.20.601 et seq.), to "discharge from its domestic wastewater treatment facilities, to receiving waters named Ashley Creek, in accordance with..." the stipulations set forth in the permit. As well as describing the quality of the effluent that may be discharged by the sewage treatment plant, the permit requires some KSTP monitoring of the effluent and some monitoring of two sites in Ashley Creek, one above and one below the discharge (permit no. MT-0021938).

The Kalispell Sewage Treatment Plant (KSTP) first began discharging effluent into Ashley Creek in 1939 at which time the only treatment was primary treatment (L. Jenkins, pers. comm.). Primary treatment was upgraded and secondary treatment was added in 1973. Secondary treatment consisted of a secondary clarifier, an outdoor tower with
redwood scaffolding for biological trickle treatment, an aeration basin and a gravel and sand filtration system. Chlorination treatment was also added in later years. Recently, after concern over the cultural eutrophication of Flathead Lake and legislative approval for the county wide passage of phosphate bans in Flathead and Lake counties, phosphate limits were imposed on the KSTP and an interim tertiary treatment was instituted April 14, 1990. The process was an alum treatment to remove phosphate. [Hydrous oxide floc treatment (alum treatment is of this type) is beneficial because in addition to phosphate it removes many other contaminants (Lee et al. 1978).]

Effluent is chlorinated, beginning May 1 and ending September 30, in accordance with the KSTP effluent discharge permit (MDHES-WQB 1988b). It is only required during this period because it is presumed that human activity on the creek is limited by warmer temperatures.

The KSTP discharges, under permit MT-0021938, into Ashley Creek an average of 1.3 million gallons per day (up to 20% of the total volume of the Creek). The permit was issued by the State of Montana, in accordance with an agreement between USEPA and the State allowing the State to regulate discharges therein. The permit controls the quality of the effluent and requires self monitoring by the City.

Variables addressed in the permit include CBOD$_5$, TSS, fecal coliform, oil and grease, total residual chlorine, pH, total phosphorus, dissolved
oxygen, ammonia, Kjeldahl nitrogen, nitrite + nitrate, temperature, cadmium, chromium, copper, lead, nickel, zinc, and wasted sludge. On occasion, during periods of heavy rains, the plant has not been able to process all of the incoming water and sewage and has discharged barely processed and/or unprocessed effluent - in violation of their permit. This occurred three times in 1989, July 13, August 24, and November 11, but did not occur in 1990.

At the time of this study, Kalispell City was in the process of building a new treatment facility (at an estimated cost of 12 million dollars) on the same grounds as the existing plant (Ashley Creek floodplain). They expected to be operating mid-September 1992. The new plant is a variation of the Bardenpho process. The treatment method, developed in South Africa, has been modified by the University of British Columbia (L. Jenkins, pers. comm.).
METHODS AND MATERIALS

WATER CHEMISTRY STUDY

Chemical analyses of Ashley Creek water at strategic points were used to isolate suspected sources and types of loading to the creek. A spatial change in dissolved or particulate matter in the creek is an important indication of either loading, dilution, or instream retention or transformation. If phosphorus and nitrogen increase downstream of sources, primary productivity may also increase if nutrients limit primary productivity. This is assumed to be indicated by higher peaks and lower valleys in the diurnal oxygen curve and by increased ash-free dry weight and chlorophyll a per unit area of substrate (Stauffer et al. 1979). Although not all possible toxins were tested for, the most likely ones (ammonia and chlorine) were investigated.

Once each month, from April to September, 1990, water samples were collected at six sites (Figure 1) on Ashley Creek. During this same period water samples were collected by the University of Montana, Flathead Lake Biological Station (UMFLBS) at site 6, Snow Line. Because of cost, most of the analyses done by the university were not duplicated for this study and their data were incorporated into these data. The same methods of sample collection and analysis were used for both studies, and an effort was made to collect samples for this study as close to UMFLBS collection dates as possible. Sample containers were acid washed, rinsed 3 times in
deionized water and finally rinsed 3 times with creek water before being filled from approximately 0.6 the depth of the creek (or as near as possible).

The analyses performed required specific treatment of the collected samples for each test. Water samples for nitrite/nitrate, ammonia and soluble reactive phosphorus, were filtered through a 0.45 micron membrane filter, using a Nalgene vacuum hand pump (< 15 inches of Hg). To prevent any chemical transformations from occurring after collection, the filtered samples were immediately placed under dry ice as were unfiltered samples for total phosphorus and total persulfate nitrogen. Unfiltered water was acidified (0.5 ml H$_2$SO$_4$ per 500 ml water) in the field and kept cool (on ice) for particulate and non-particulate organic carbon analyses. Unfiltered water was also kept on ice until reaching the lab where turbidity tests were conducted within 24 hours of collection.

Analytical analyses were done at the University of Montana, Flathead Lake Biological Station, Freshwater Research Laboratory. See Table 1 for biophysical variables and methods used. Quality control criteria for rejection of analyses were: ± 10 percent deviation from known spikes and < 1 standard deviation of the mean of replicated analyses. These checks were run on about every 10th field sample. In addition, laboratory performance evaluations were made about every 6 months using sample standards and unknowns from the United States Environmental Protection
### Table 1. Biophysical Variables and Methods Used in Monitoring Water Quality in Ashley Creek, 1990.

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1 APHA 1985  
2 D’Elia et al. 1977  
3 Menzel and Vaccaro 1964  
4 Perkin-Elmer 1976  
5 Marker et al. 1980  
6 Li-cor 188 integrating quantum meter  
7 measured in situ using Hydrolab Surveyor II  
8 Yellow Springs Dissolved Oxygen Meter  
9 Lind 1974  
10 Amperometric titrater
Agency.

Once each month, April through September, temperature, pH, dissolved oxygen and specific conductivity were measured at each of the six sites using a Hydrolab (Surveyor II, Hydrolab Environmental Data Systems). An effort was made to collect Hydrolab data as close as possible to the date the water samples were collected. Normally, this was within a couple of days.

RESPIRATION STUDY

Respiration and primary production occur in both the water column and the benthos. Fluctuating dissolved oxygen (DO) concentrations within enclosed chambers at six sites, four above and two below the KSTP, were used in this study to attempt to show the location and impact of organic loading on Ashley Creek. The dark- and light-bottle method (APHA et al. 1985) was used to analyze DO demand changes in the water column. Increased loading of organic matter downstream of suspected sources is assumed to be indicated by a decline in dissolved oxygen concentrations as decomposers use DO to metabolize the food source. That process results in a diurnal oxygen curve with lower oxygen levels overall. Evaluation of the oxygen flux should indicate whether the system is being controlled by the external loading of organic matter (respiration of microorganisms) or the internal process of primary production and
decomposition of previously deposited organic matter.

Photosynthesis-respiration chambers are routinely used in aquatic environments to determine the rate and level of change in dissolved oxygen concentrations (McIntire et al. 1964; McIntire and Phinney 1965; Thomas and O'Connell 1966; Lane and Hall 1971; Pfeifer and McDuffet 1975; Rodgers et al. 1978; Kicklighter 1987) and to obtain an index of benthic community metabolism. Using the chamber method for short term productivity, rather than free water, eliminates errors that may be associated with exchange of oxygen through the water-air interface (Owens 1965; Bott et al. 1978). In-situ community respiration studies were conducted on six consecutive days on two occasions, July 18-23 and September 5-10, at the six sites.

At the upper sites, rocks with their associated benthos were collected from a depth of 1-2 feet and sealed in four Plexiglas metabolism chambers (Figure 2). A weight tied to the end of a rope was alternately thrown in opposite directions in the creek and the rock which was closest to where the weight landed was collected and put into the respiration chamber. Because of the different type of substrate at the lower two sites, alternate material was used for both sites in September and for Snow Line in July. For the July study at Snow Line, six 6-inch x 2 5/8-inch PVC (ID) sections were used to collect cores for each chamber. In preparation for the September study, clay tiles were immersed in the creek at both
Figure 2. Photosynthesis/respiration chamber. Plexiglas chamber 7 inches high x 14 inches long x 14 inches wide.
Cemetery and Snow Line on August 9. They were incubated until the respiration study took place at those sites on September 9 and 10.

Two of the four chambers were covered with black garbage bags to eliminate photosynthesis and monitor respiration. Each chamber had eight access ports for water circulation (Figure 2). Water within the chambers was circulated continuously by Teel Model 1P811A (Dayton Elec. Mfg. Co., Chicago 60648) submersible pumps connected to a rheostat. The rheostat was connected to two 12 volt deep cycle RV batteries connected in parallel. The batteries were fully recharged each evening. In a similar study, Pfeifer and McDiffett (1975) used two different sized pumps that created current velocities of 19.6 l/min. and 9.2 l/min. The rheostat in this study was placed at its lowest setting, which delivered a discharge rate of approximately 11.3 l/min.

During the study the chambers were placed in 14-16 inches of water. Because the creek is a deep channel at Snow Line, it was necessary to build scaffolding on which to set the chambers. At Cemetery a site was chosen where rip-rap had been dumped on the bank and into the creek. It provided an elevated stable surface on which to set the chambers and (for the July study) smaller rocks to use inside the respiration chambers. The study ran for four hours at each site. The output tubes on the chambers had port holes to fit the DO probe. Changes in dissolved oxygen and temperature were monitored during the period using a YSI DO meter and
probe. The DO meter and probe were standardized at the beginning of the study, against a value measured by Winkler titration (APHA et al. 1985). The chambers were flushed, when DO concentrations reached high levels or bubbles began to appear on the lid of the chamber, in order to avoid supersaturation of oxygen and to avoid depletion of nutrients.

Insolation was monitored using a Li-Cor 188 light meter and a quantum probe, from sun up to sun down each day that respiration studies were conducted. Light readings were also taken at the beginning, the end and usually once in between, at the site under water.

MACROINVERTEBRATES

After the respiration study was complete, the chamber was disassembled and most of the water escaped. The assembly was transported to the base station and during the July study, the macroinvertebrates were hand-picked from the rocks using forceps and a 2 1/2 power lens. They were preserved in ethyl alcohol and water for later keying (Wiggins 1977; Merritt and Cummins 1984; Pennak 1989) and enumeration. Since there was an absence of rocks at Snow Line, substrate cores were collected and sieved for macroinvertebrate analysis. Since Ford (1962) concluded that 98% of Chironomidaes were found in the top two inches of mud in small streams, the sample cores in this study were taken to insure that depth. PVC sections 3 inches deep x 2 5/8
inches were taken and sieved through a series of U. S. Standard testing sieves, A.S.T.M. E11 numbers 18, 35 and 40. In September, the rocks used in the respiration chambers were not also used for the invertebrate study. At all sites except Cemetery, separate substrate samples were collected, picked clean of macroinvertebrates and measured for surface area. At Cemetery invertebrates were collected from the wooden frame to which the tiles were attached.

**CHLOROPHYLL a**

After completion of the respiration study each day, the rocks used in the chambers were scraped with a nylon spatula and scrubbed with a toothbrush to remove the algae. Minimal water was sprayed to wash the scrapings into ziplock bags. The algae were then frozen. In July, the same rocks were used for the macroinvertebrate study and were thus subjected to light while the fauna was picked off. This may have been deleterious to the chlorophyll a and that procedure was not repeated in September. The frozen algae were later thawed, put into a beaker and brought to volume (usually 200 ml) with distilled water. The beaker was placed on a stirring plate while duplicate 5 ml aliquots were taken and filtered for chlorophyll a (chl a) and ash free dry weight. An acetone extraction and spectrophotometer (Table 1) were employed for the chl a analysis.
ASH FREE DRY WEIGHT

Duplicate 5 ml aliquots were taken from the prefrozen mixture that had been scrubbed off the rocks and brought to volume (see chl a). Each aliquot was vacuum filtered onto a preweighed 47 mm 0.45 um type A/E glass fiber filter and then analyzed for ash-free dry weight (AFDW) according to APHA Standard Methods (1985).

SURFACE AREA

Surface areas of the rocks used in the respiration study were measured by marking off areas with a red pencil, measuring with a cloth tape measure and then calculating surface area. The entire surface area was measured for invertebrate calculations while only the area that appeared to have plant growth was measured for chl a calculations.

DARK - LIGHT BOTTLE

To determine metabolism occurring within the water column and to compare that to benthic community metabolism, a dark - light bottle study was conducted at the six sites concomitantly with the first benthic respiration study, July 18-23. A second dark - light bottle study was conducted September 12-14, immediately following the second respiration study. Three light and three dark 300 ml BOD bottles were filled with creek water (using a delivery tube) at each site and incubated for 4 hours.
at the same depth (14-16 inches) as the respiration chambers. For the July study the YSI DO meter and probe were used to determine DO concentrations in the bottles after 4 hours. The probe (which lacked a stir bar) was inserted into the bottles and the bottles were carefully and gently inverted and swirled. These data are questionable, though, because the pressure sensor on the probe was not under water. (Air pressure is not as great as water pressure and it is not certain that the probe compensated for the difference.) During the second study a YSI DO meter and probe with an attached stir bar were used to determine the DO concentrations. A gas generator was used in the field to provide power for the stir bar.

**DIURNAL OXYGEN STUDY**

On two occasions, May 27-28, and August 29-30, 1990, diurnal oxygen studies were conducted. Three sites were sampled: 1) Airport, the site just above the KSTP; 2) the chlorination tank on the KSTP grounds, which contained the effluent just before it was released into Ashley Creek; and 3) Cemetery, the site immediately below the KSTP. Oxygen and temperature were measured over a 24 hour period using a YSI DO meter and probe. During May, pH was also monitored using an Orion SA 210 pH meter. The DO probe was attached to a five foot long stick in order to reach out into the main flow of the creek. Sampling personnel tested one site, drove to the next site, sampled it, drove to the
last site, sampled it and started over again. This continued for 24 hours.

FLOW

Flow data were obtained from the Montana Department of Fish, Wildlife and Parks for the Meridian and Big Horn sites. The department has a continuous gauge located on Kalispell Wood Products property, just above the sampling site at Meridian, which began operation April 4, 1990. This gauge replaced the old gauge near Meridian that was relinquished to the beavers. A staff gauge is attached to the Big Horn Road bridge and a 622 Gurley meter was used to relate gauge readings to flow.

Flow data for the downstream site, Snow Line, were obtained from the University of Montana, Flathead Lake Biological Station for the period April through August 13, 1990. The pressure transducer on their continuous depth meter was not functioning properly after that period and before October 11, 1990, so the flow was estimated for September, using regression analysis and flow data obtained before and after that date.
RESULTS AND DISCUSSION

FLOW VELOCITY AND VOLUME

Flow separates lotic ecosystems from lentic ecosystems. Flow directly or indirectly affects almost all other parameters that control productivity within the lotic ecosystem. In studies conducted on streams in four distinct geographic areas, Minshall et al. (1983) found that biological processes played a lesser quantitative role in ecosystem dynamics than did current velocity and retention of organic and inorganic matter. Turbulent water is well oxygenated; oxygen not only affects which species will inhabit an area, but is also important chemically. The redox potential is dependent upon oxygen concentration and low oxygen concentrations may facilitate the release of certain chemicals and toxins from some substrates.

Flow distinguishes riffle communities from pool communities. In pools, settled organic matter imposes little biochemical oxygen demand on the water column (Hellawell 1986). There is an interface between the substrate and overlying water that limits interchange between the two matrices. Fast moving water is usually well oxygenated and bedload is increased; more particles are suspended and their entire surface area is exposed where it may react with other suspended materials. Faster moving water can also scour and carry heavier particles, thus increasing turbidity (see turbidity below).
Volume of flow is important for the dilution of many substances. In this study, alkalinity and specific conductance were lowest during the months of highest flow. A relationship was also observed in Ashley Creek between flow volume and alkalinity (Table 2), specific conductance and dissolved organic carbon (DOC). DOC concentration was highest during spring when flow was highest; a result of organic matter within the watershed being transported down gradient. The MDFWP purchased water rights of 2.6 cfs or 1,908 acre feet for instream flow for the sole purpose of dilution of KSTP effluent. Dilution mitigates some deleterious effects of pollutants.

In this study, water quality deteriorated during periods of increased flow. High water eroded and wasted stream banks, increasing turbidity and phosphorus concentrations. Apparently, flooding of organic soils and wetlands depressed dissolved oxygen concentrations and caused an influx of organic matter and nutrients.

Because of record high precipitation in the area during 1989 and 1990, stream flow was high. Precipitation in Kalispell during 1990 was the highest on record (23.93 inches) and precipitation during 1989 was the third highest on record. The second half of 1989 was the same as 1990 (United States Department of Commerce, NOAA 1991). The year with the second highest precipitation on record (22.36 inches) was 1964, the year of extensive flooding in the valley. According to the National
Table 2. Biophysical Variables Measured at the Six Study Sites on Ashley Creek, April - September, 1990.

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Oceanic and Atmospheric Administration records, 1990 precipitation (water equivalent) was substantially higher than the normal (15.93 inches) for the area.

Ashley Creek is a 3rd order stream (Strahler classification, 1957) and during the six months of this study, flow ranged from 182.9 cfs at Big Horn in April, to 14.1 cfs at Meridian (Figure 3) in September (based on data from the MDFWP's continuous recording gauge).

**ALKALINITY**

The two most salient characteristics of Ashley Creek watershed soils are their alkaline pH and their silty texture (Dutton 1987). These characteristics have a high impact on Ashley Creek because they impart a good buffering capacity. This is important because the creek passes through organic soils in a number of places and is able to buffer the acidifying effects.

The alkalinity of Ashley Creek varied from 100.05 to 353.8 mg CaCO$_3$ per liter (Figure 4). Alkalinity was highest during periods of low flow and lowest during periods of high flow (Table 2). The highest concentrations occurred during September. This indicates that snow melt dilutes the conservative ions. Carbon dioxide concentration affects the equilibrium and may either increase or decrease total alkalinity. Carbon dioxide is reduced during photosynthesis, but increased by both plant respiration and
Figure 3.

Estimated Flow at Three Sites on Ashley Creek, 1990

[Graph showing estimated flow at three sites: Big Horn, Meridian, and Snow Line, with flow measurements in cubic feet per second from April to September.]
Figure 4.

Alkalinity
Ashley Creek, 1990

Alkalinity (mg CaCO₃/liter)

- Big Horn
- Derns
- Meridian
- Airport
- Cemetery
- Snow Line

Key:
- Apr-26
- May-22
- Jun-19
- Jul-16
- Aug-16
- Sep-17
the decomposition of organic matter. These influences are most important
in mid and late summer when aquatic vegetation has become established
in large quantities, when flow has decreased and when temperatures are
more favorable for biological activity.

Alkalinity also varied spatially. On each sampling date, alkalinity
generally increased from the upstream sample sites to the downstream
sample sites by 20 to 30 percent. During August there was a decrease in
alkalinity between the two lowest sites that cannot be explained. During
July, August and September there was a substantial increase in alkalinity
from Derns to Meridian. During that same time, nitrite/nitrate increased in
the same area. The inflow of water from Spring Creek between those
two sites could be affecting water chemistry.

**TURBIDITY**

Turbidity is caused by both dissolved and particulate matter suspended
within the water column. High turbidity interferes with many of the
beneficial uses of water. It can affect the normal physiology of aquatic
life, raise the temperature and affect the chemical reactions within the
water (Hellawell 1986). It also interferes with human recreational use and
aesthetic enjoyment of the water.

Turbidity was always lowest at the upper site, just below Smith Lake
(Table 2). Since water is very slow moving in the lake, particulate matter
settles out of the water column and turbidity below the outflow was lowest - between 1.1 and 2.2 NTUs.

Generally, turbidity gradually increased downstream but during low flow it was lower at Snow Line than at Cemetery (Figure 5) indicating some settling of the KSTP effluent. During high flow, turbidity was substantially higher at Snow Line than at Cemetery. Material did not settle out during those months and the material previously settled may have been resuspended and transported downstream. Turbidity was highest in April at Snow Line corresponding with discharge. The creek bottom was never visible at Cemetery (although it was only about 5 feet deep during low flow) and only vaguely visible at Snow Line during the low flow in September. Above the KSTP the water color was a reddish brown - indicative of a high humus load. The marsh areas in Smith Lake and in Mountain and Truman creeks (tributaries) as well as the small lakes on Ashley Creek are responsible for contributing that organic matter (Dutton 1987). Below the KSTP the water was opaque brown and was carrying a heavy suspended load. Other likely causes of turbidity were silt from soils in the drainage (Dutton 1987), agricultural activities, livestock that had access to the stream, denuded streambanks, septic seep from houses built near the creek, sediments from roads, organic-rich soils, construction sites, streambanks and most probably industries such as Stampede Packing and Kalispell Wood Products.
Figure 5.

Turbidity
Ashley Creek, 1990

Turbidity, NTUs

Violation level

KSTP

Big Horn  Derns  Meridian  Airport  Cemetery  Snow Line

Apr-26
May-22
Jun-19
Jul-16
Aug-16
Sep-17
The turbidity criterion (MDHES-WQB 1988a) for cold water aquatic life (10 NTUs) was surpassed at three sites in April and at Snow line in May. In April at both sites downstream of the KSTP and the site immediately above the KSTP turbidity was above 10 NTUs. This is of particular concern because in the Flathead River Basin EIS, Zackheim and Cooper (1983) recommended that land disturbances which increase riverine turbidity be curtailed or eliminated.

**SPECIFIC CONDUCTANCE**

Like alkalinity, conductivity was highest during periods of low flow. During April (high-flow) the minimum conductivity value was observed at Big Horn, 0.184 umhos/cm (Table 3). The maximum value, 0.422 umhos/cm, was observed during September (low-flow) at Snow Line. It increased along a gradient from the upper sampling site to the lower site except on three occasions (Figure 6). On May 3, June 20, and July 14, conductivity was slightly lower at Snow Line than it was immediately below the KSTP. By then, it is likely that many of the ions had adsorbed to clay and silt particles and settled out or otherwise reacted with material present in the lower creek.

The most likely natural contributions to conductivity are drainage from the calcareous soils in the watershed and microbial decomposition of organic matter (sulfur is released largely as hydrogen sulfide) from the
Figure 6.

Specific Conductance
Ashley Creek, 1990
numerous marsh areas in the upper drainage. Suspected anthropogenic subsidies are again decomposing organic matter from livestock and human waste, agricultural practices, and general urbanization, all of which increased downstream.

**pH**

The pH in an aqueous environment is of concern not only because it is critical to the biota which reside there, but also because it affects the equilibria that exist between many compounds, some of which may be toxic. Two reactions of particular interest here are those involving nitrogen and phosphorus. Increased pH leads to the formation of calcium carbonate which coprecipitates phosphate (Otsuki and Wetzel 1972). The rates of nitrification and denitrification decrease in acidic water.

The pH in the study area was slightly alkaline, 7.37 to 8.55 (Table 3). The upper site received an influx of decaying organic matter from the marsh areas upstream causing the most acidic conditions to occur there (Figure 7). The creek was more alkaline at the next three sites, and variation between those three sites was small. Overall, pH values ranged from 7.37 in June to 7.91 in September. Variation was greatest (1.11 units) in July when pH was 7.44 at Big Horn and 8.55 at Airport. The August variation was next highest: 7.45 at Big Horn and 8.36 at Derns (0.91 units). During the other four months, differences were much
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Figure 7.

pH Ashley Creek, 1990
smaller: 0.46 to 0.61 pH units. The KSTP effluent lowered the pH at the
two downstream sites. The State standard does not allow an induced
variation in pH of > 0.5 pH unit. Variation approached this in August and
September when it was 0.42 and 0.43 pH units at 12:08 pm and 11:49
am, respectively. It is not known what pH was later in
the afternoon and it is possible that the variation exceeded the 0.5
standard. Another violation seems to have occurred at Airport in July
when the pH was 8.55. The water quality criterion (MDHES-WQB 1988c)
for cold water aquatic life set a maximum pH at 8.5. The pH exceeded
that criterion by an amount less than the measurement error of the
instrument so the "violation" is questionable.

There are factors other than organic matter that affect pH.
Photosynthesis can greatly affect patterns in pH (APHA 1985; Crumpton
and Isenhart 1988). When CO₂ concentrations are at a minimum, during
periods of highest temperatures and peak photosynthesis, pH usually
reaches a maximum. Ammonia also tends to increase pH values (Bansal
1977) which in turn affects the equilibrium of nitrogen. An increase of
one pH unit (within the range of most freshwater fish) will increase the
NH₃ (toxic) species about 10-fold (Trussell 1972; Thurston and Russo

The pH was also monitored in May during the synoptic oxygen study.
Readings were taken over a 24 hour period, in the KSTP effluent and at
two sites in the creek, one directly above the KSTP, Airport, and one
directly below the plant, Cemetery. During that study the average pH at
Airport was 7.66; at Cemetery it was 7.58. The pH ranged from a low of
7.31 (7:30 am) below the KSTP to a high of 8:02 (8:16 pm) above the
KSTP. The average pH of the effluent before it was released was 7.16.
Effluent pH ranged from 6.64 (5:00 am) to 7.53 (9:01 pm).

TEMPERATURE

Temporally, Ashley Creek temperatures changed with seasonal ambient
temperatures. Creek temperatures increased from a low in April to a high
in July then descended again (Figure 8). Spatially, during April and May,
temperatures were lowest at the upstream site and highest at the
downstream site. In June and July temperatures showed less variation
between sites and during August and September the upper site was the
warmest. The warmer temperatures at Big Horn, later in the season, can
be explained by the fact that Smith Lake is immediately above that sample
site. This shallow lake had warmed up by late summer, as indicated by
the higher temperature of the outflowing water. Below Big Horn and
above Smith Lake many of the stream banks have been denuded by
farmers and ranchers. Without the shading effect of the riparian
vegetation, creek temperatures rise and the community is stressed and/or
modified.
Figure 8.

Temperature
Ashley Creek, 1990

Degrees Centigrade

Big Horn  Derns  Meridian  Airport  Cemetery  Snow Line

Apr-26  May-22  Jun-19  Jul-16  Aug-16  Sep-17
During this study, the KSTP effluent did not substantially affect the temperature of the creek. Maximum variation in temperature above and below the discharge was 0.41°C, which occurred in August. It is probable that during winter months a greater alteration would be evident. During August and September, the water temperature at Meridian was a degree or two cooler than the above and below sites. Inflowing water from Spring Creek just above that site was evidently cooling the creek.

Temperature affects both chemical and biological reactions within the creek. Warmer temperature may stress cool water species and reduce the available oxygen because oxygen is less soluble in warmer water. Chemical reactions speed up at higher temperatures. Increases in temperature up to approximately 30°C speed up both the nitrogenous and carbonaceous degradation processes and thus reduce oxygen concentrations (Zanoni 1969). This probably exacerbates the low DO problem below the KSTP during low flow.

**CHLORINE**

Chlorine was not detected in Ashley Creek (amperometric titration, 0.2 ppm detection level) above the point of KSTP discharge, below the discharge nor even in the KSTP on-site chlorination tank on August 24, 1990. Under their discharge permit, KSTP is required to monitor residual chlorine daily. During the chlorination season, May 2 through September
30, 1989, residual chlorine concentrations, in the effluent, exceeded the permit limit of 0.50 mg/l on three occasions, May 15 and 18 and July 18 (KSTP open file). On two of those occasions it reached 0.82 mg/l. In 1990, the KSTP effluent monthly residual Cl₂ averages ranged from 0.08 mg/l in September to 0.18 mg/l in May. The EPA criterion, for freshwater (USEPA 1986), for chlorine states that aquatic organisms should not be affected unacceptably if the 4-day average concentration of total residual chlorine does not exceed 11 ug/l more than once every 3 years but that it would take 3 years for an unstressed system to recover from a pollution event of that magnitude. It also states that the one hour concentration must not exceed 19 ug/l more than once every 3 years on the average. Ashley Creek dilutes the chlorine concentration of the KSTP effluent. Because both the flow in the creek, and the chlorine concentration in effluent vary, the concentration of chlorine in the creek may fluctuate.

Increasingly larger concentrations of Cl₂ gas must be added at the KSTP as summer progresses (KSTP open files) in order to achieve the same standard. For example, on May 4, 1989, KSTP chlorination was 50 lbs/24 hrs to achieve a fecal count of 70 in 5 mls of effluent. By August 24, chlorination was 100 lbs/24 hrs to achieve a fecal count of 82 in 5 mls. This increased demand could be a result of higher temperatures which stimulated chemical processes. Gases are more volatile at higher temperatures and chemical reactions including the reaction between
nitrogen and Cl\textsubscript{2} accelerate with temperature. During the chlorination of wastewater, because of the chemical reaction forming mono- and dichloramines, no free residual chlorine is obtained until the ammonia has been oxidized (APHA 1985). Chloramines are slightly less toxic than chlorine but fairly stable and may persist within the system for some time (Johnson et al. 1979; Hellawell 1988). Below the KSTP there is evidence of this reaction in the higher ammonia concentration in April before chlorination began (Table 4). Chlorine reacts with fewer materials as pH increases and with those materials which it does react the rate increases with temperature (AWWA 1971). The pH was lower below the KSTP than it was above it. That increase should have improved the effectiveness of the chlorination.

**PHOSPHORUS COMPOUNDS**

Most of the phosphorus in fresh water systems occurs as organic phosphates and cellular constituents in the biota adsorbed to inorganic and dead particulate organic materials (Wetzel 1983; Connell and Miller 1984). But it is the inorganic form that is biologically available - mainly orthophosphate (Krenkel and Novotny 1980). In aquatic environments, phosphorus may undergo several changes in form once it enters a water system. Its fate may include: precipitation (upon chemical reaction with metals, particularly iron and aluminum oxides, calcium and other cations);
Table 4. Water Chemistry Analyses - Nutrients. Ashley Creek. 1990. Parameters are given in ug/liter. NH3 = ammonia. NO23 = nitrite/nitrate. SRP = soluble reactive phosphorus; TP = total phosphorus. TPN = total (persulfate) nitrogen. N/P = nitrogen phosphorus ratio. DOC = dissolved organic carbon. NDOC = non-dissolved organic carbon.

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**August SRP and TP concentrations were reversed from the amount stated by the lab.**
biological assimilation; biological release and recycling; physical uptake by sorption to sediments; release; settling; and deposition within the substrate (Davis and Foster 1958; Otsuki and Wetzel 1972; Wetzel 1983). Some of these reactions make the phosphates unavailable for plant growth.

Adsorption to sediments followed by sedimentation may detain the P or cause it to be bound permanently in the sediments. The rate at which suspended solids settle to the bottom varies seasonally, with flow, from river to river and even within reaches of the same river. Nutrients adsorbed to sediments may become available to macrophytes. This may be occurring below the study site where the creek is slow moving and deep, and thick growths of macrophytes line the edges.

Phosphates are believed to cycle quite rapidly within the aquatic system relative to other nutrients and thus smaller amounts are required to maintain the cycle. Phosphates are transferred from the flora by grazing, decay and decomposition by bacteria. Over time, a portion of the phosphorus assimilated by organisms becomes refractory, and is no longer available for biological growth (Lee et al. 1978).

Benedict et al. (1963) found that as much as 50 percent of the solids traveling downstream were being transported as rolling bottom material. If this is true in Ashley Creek, the contribution of total phosphorus (TP) and total persulfate nitrogen (TPN) to Flathead River may be higher than
the concentrations reported in this study because the maximum depth that
samples were taken was approximately 0.6 the depth of the creek.

Phosphorus may affect other parameters within the creek. The effect
that P has on the abundance of plant growth is controlled by a number of
other variables, including flow rate, depth, turbidity and concentration of
other nutrients. It has been found that lakes with higher phosphorus
concentrations have higher oxygen deficits (Lee et al. 1978; Wetzel
1983). This relationship was also true in Ashley Creek during the May
and August diurnal oxygen studies.

**SOLUBLE REACTIVE PHOSPHORUS**

The quantity of biologically available phosphorus is commonly equated
with soluble reactive phosphorus (SRP) but that value underestimates it
(Lee et al. 1978; Schaffner and Oglesby 1978). Some of the organic P
and loosely adsorbed P is not included in SRP analysis. Ellis and Stanford
(1986) found that during fall, in both the Stillwater River and Ashley
Creek, 45-100% of the particulate phosphorus present above the KSTP
from agricultural and urban drainage was bioavailable. They found 70-
75% of the particulate phosphorus below the KSTP to be available. Their
study also showed that, above the sewage outflow in Ashley Creek,
during spring runoff, a substantial portion (60%) of the soluble phosphorus
was unreactive. In a study done in the Great Lakes area (Young et al.
1982), wastewater particulate phosphorus contained a significantly higher available fraction than that of other sources of particulate phosphorus (55% vs. 30%).

Generally, soluble reactive phosphorus (SRP) was lower at the two upstream sites and highest at the two downstream sites (Figure 9). The large increases at the two downstream sites were caused by the influx of the KSTP effluent. The increase at the two mid sites was a result of nonpoint pollution resulting from agriculture and urbanization.

Urbanization results in a several fold increase in concentrations of phosphorus discharged to surface waters per unit of land surface (Weibel et al. 1966; Keup 1968; Lee et al. 1978). The greatest increase occurs with the transition from forest to urban use.

Soluble reactive phosphorus is high to very high, in Ashley Creek, compared to other surface waters in the basin (Table 5). The concentrations in the upper creek are low compared to other impacted creeks, but over two times higher than the Flathead River below the Ashley Creek inflow. The mid-creek concentrations are considerably higher than other impacted creeks in this area.

Increases in nutrients may induce a change in flora based on their tolerances, efficiencies and competitive abilities (Provasoli 1969). Bothwell (1988) found that SRP concentrations of less than 1 ppb saturate growth of diatom communities and concentrations near 30 ppm
saturate standing crop. Horner et al. (1983) found biomass of filamentous green algae to increase with SRP concentrations up to 25 ppb. Others have suggested that changes in concentrations may affect standing crop in diatom communities when SRP is less than 20 ppb (Watson et al. 1990) or 30 ppb (Bothwell 1988). According to their studies, SRP was not limiting for diatom growth at any site during this study (Table 4), or for filamentous green algae growth at the lower sites. It was, however, probably low enough to limit standing crop at sites above the KSTP (except during April). The lowest observed concentration was 2.7 ppb during September at Derns. Concentrations were always above 30 ppb at the two sites below the KSTP. Flora and algal composition did vary from site to site on Ashley Creek. Variations in nutrients, including SRP, could be part of the explanation.

TOTAL PHOSPHORUS

Total phosphorus (TP) load was generally highest during April, May and June, the months of highest flow (Figure 10). This is a result of loading caused from erosion and runoff associated with spring runoff and large amounts of precipitation. On four of the six sampling dates (high flow months), TP showed a slight decline from Cemetery to Snow Line. Total phosphorus during the study ranged from 12.5 ug/l to 184 ug/l (Table 4). The downstream site had maximum values ranging from 80.3 ug/l to
Figure 10.

Total Phosphorus Load
Ashley Creek, 1990

- Apr-26
- May-22
- Jun-19
- Jul-16
- Aug-16
- Sep-17

Big Horn    Meridian    Cemetery    Snow Line
184.0 ug/l; average concentration was 102.1 ug/l. In contrast, the 1977-1989 average for Ashley Creek reported by the University of Montana, Flathead Lake Biological Station was 421.2 ug/l (Stanford et al. 1990), considerably higher than the Flathead River, Holt station, (mean 20.3 ug/l). The entire period of UMFLBS analysis was before the KSTP began tertiary treatment for P removal.

During this study, total P concentrations were high in Ashley Creek at the upstream sites even during low flow. The six month means for each of the two upstream sites, were comparable to the means of other impacted lotic systems in the drainage, i.e., Stoner Creek and the Stillwater River (Table 5). The means of the two mid sites, however, were considerably higher than other lotic systems in the drainage and the means of the two sites below the KSTP were 4 to 5 times higher than other impacted systems. During May, total phosphorus (TP) was higher at Big Horn than it was at the next three sites. There was no obvious reason for the escalation of TP at that time and SRP did not exhibit the same surge. At Snow Line in both July and September and at Cemetery in August total phosphorus exceeded the 100 ug/l limit suggested by Mackenthun (1973) to prevent nuisance aquatic growth (106, 184 and 146 ug P/L respectively). The August values for SRP, reported by the contracted lab (University of Montana, Flathead Lake Biological Station Fresh Water Lab) at the lower two sites, were greater than the values
Table 5. Concentrations (means and ranges) of nitrogen and phosphorus variables at various Flathead drainage monitoring sites (Stanford et al. 1990) summarized for the period of record, 1977-1989 compared to the six sites on Ashley Creek examined during present study. Data are expressed in ug/l (N and P variables) umhos/cm2 (cond) and n=6 for Ashley Creek and n> 50 for all other variables. TP = total P; SRP soluble reactive P; NO3 = soluble nitrite/nitrate; NH3 = soluble ammonium; TPN = total nitrogen (persulfate oxidation); COND = specific conductance.

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<th>NH3</th>
<th>TPN</th>
<th>Cond</th>
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reported for TP. For this report, these values were assumed to be reversed, but even under that assumption there seemed to be an inconsistency. Soluble reactive phosphorus was substantially lower at Snow Line than it was at the upstream site (Cemetery). During other months it was either similar or higher at the downstream site.

The Kalispell City Sewage Treatment Plant discharge permit included interim (November, 1988 through April 15, 1989) effluent limitations of a minimum 30-day average for total phosphorus (TP) not to exceed 2.0 mg/l. The limitation after April 15, 1989 required the 30-day average for total phosphorus to be 1.0 mg/l or less. In the past, TP concentrations in the effluent have been much higher. During 1988 for example, TP varied from a low of 3.6 mg/l in December to a high of 7.2 mg/l in September (KSTP open file). The average that year was 5.2 mg/l even though the county-wide phosphate ban was in effect at that time.

Generally, large quantities of phosphorus are rapidly removed from streams. Keup (1968) found that phosphorus was removed according to a constant logarithmic rate ($K \pm 0.011$) on a 65-mile reach of the South Platte River receiving two loads of municipal waste. The same study showed that the Sebasticook River, downstream from Dexter, Main, assimilated 29 percent of the phosphorus added as municipal wastes within a four-mile stretch. Biological activity incorporates much of the dissolved P. Some downstream reductions were observed in Ashley
Creek, mostly during low flow. Generally, the longer the transit time of phosphorus, the greater the primary productivity, or the greater the suspended sediment load in the river, the greater the amount of phosphorus that will be converted to unavailable forms (Lee et al. 1978). Ellis & Stanford (1986) found uptake of $^{32}$PO$_4$ in Ashley Creek to be less complete and turnover times longer than other sample areas in the basin. They felt that it was possibly because of higher amounts of PO$_4$ and less phosphorus deficiency.

Unless the standing crop of organisms is increasing or the creek is aggrading, phosphorus storage is temporary (Keup 1968) and the downstream transport of detritus and sediments will increase concentrations elsewhere in the drainage. By controlling the phosphorus load into the Flathead, Henry and Stanford (1983) believe that the trophic state of the lake can be controlled. Ashley Creek is currently adding a substantial P load to Flathead River and controlling that load is paramount to controlling loading in Flathead Lake.

**NITROGEN COMPOUNDS**

Various forms of nitrogen exist in aquatic ecosystems and each has a different implication for water quality: Some are toxic (Russo et al. 1981), some are plant nutrients, and some are neutral compounds that are biologically unavailable. The most salient forms are ammonia, nitrite,
nitrate, gaseous $N_2$ and organic nitrogen (Wetzel 1983). Environmental conditions such as turbulent characteristics, sedimentation, and biophysical and reaeration processes, determine oxidation states (Bansal 1977; Hill 1978). Under anaerobic conditions oxidized nitrate may be reduced to gaseous $N_2$ or to ammonia. Under aerobic conditions, ammonia can be oxidized to nitrite and nitrate, creating a biochemical oxygen demand (BOD). It is generally assumed that chlorophyll-containing organisms metabolize only the inorganic forms, nitrate and ammonia (Krenkel and Novotny 1980), but blue-green algae (Cyanophyceae) are also able to utilize $N_2$.

Aquatic organisms can influence the concentrations of compounds, including nutrients, by metabolic uptake, transformation, storage and release (Lee and Hoadley 1966). This may result in measurements of inorganic N and P that are lower during the growing season because of uptake by plants (Krenkel and Novotny 1980).

**AMMONIA**

In aqueous ammonia solutions, the un-ionized ammonia molecule, $NH_3$, exists in equilibrium with the ammonium ion, $NH_4^+$. The ratio between them is largely a function of pH and temperature (Trussell 1972; Emerson et al. 1975; Crumpton and Isenhart 1988), but $NH_3$ also decreases as a result of increased ionic strength in hard water or dilute saline solutions.
(Emerson et al. 1975). Large fluctuations in ammonia concentrations occur among sites and also within sites according to the season and the time of day (a reflection of temperature and pH fluctuations) even when the total ammonia input is constant (Hermanutz et al. 1987). Manny and Wetzel (1973) found ammonia concentrations in a small hard water stream to be quite variable - but much higher in the morning than later in the day. (In Ashley Creek ammonia input from the KSTP is not constant.) Since photosynthesis and respiration affect pH diurnally, they significantly affect ammonia speciation (Crumpton and Isenhart 1988). The importance of the un-ionized to total ammonia ratio is that the un-ionized form (NH$_3$) is much more toxic (Trussell 1972; Emerson et al. 1975; USEPA 1986). Thurston and Russo (1981) have only recently suggested that the ionized form may also exert some level of toxicity and/or that increased H$^+$ concentration increases the toxicity of NH$_4^+$.

Ammonia toxicity varies between species and according to environmental conditions. Factors that influence the toxicity of ammonia to fauna include pH, dissolved oxygen concentration, temperature, previous acclimation to ammonia, fluctuating or intermittent exposures, carbon dioxide concentration, salinity and the presence of other toxicants (Trussell 1972). The acute toxicity of NH$_3$ has been shown to increase as pH decreases, but data are lacking to show that relationship for chronic toxicity. Data do show the effect of temperature on acute NH$_3$ toxicity,
indicating that toxicity to fish is greater as temperature decreases (Hermanatz et al. 1986). It has been demonstrated (Trussell 1972; Thurston et al. 1981; Crumpton and Isenhart 1988) that for the same total dose of ammonia, trout were significantly more susceptible to fluctuating concentrations of ammonia than to fixed concentrations. Arthur et al. (1987) found that invertebrates were generally more tolerant to ammonia than were fishes, but a notable exception is the fingernail clam (Zischke and Arthur 1987). In this study, fingernail clams were found in Ashley Creek above the KSTP but not below KSTP. Ball (1967) found that coarse fish are as sensitive to ammonia toxicity as salmonid fish. His study pointed out that salmonid species were more sensitive over a short duration (the period of most past studies), but when studies were conducted over longer periods of time all species were equally sensitive.

Ammonia has been reported to be acutely toxic to freshwater organisms at concentrations (uncorrected for pH) ranging from 0.53 to 22.8 mg/l NH$_3$ for 19 invertebrate species representing 14 families and from 0.083 to 4.60 mg/l NH$_3$ for 29 fish species from 9 families (Hermanutz et al. 1986). In that study, salmonids had the lowest tolerance, ranging from 0.083 to 1.09 mg/l. Chronic tests on two daphnids showed effects at concentrations (uncorrected for pH) ranging from 0.304 to 1.2 mg/l NH$_3$. The National Criterion (USEPA 1986) for ammonia is determined using a
formula including variables for the fish species present, (warm water species vs. cold water species), flow and pH. The maximum level of un-ionized ammonia for the indefinite maintenance of fish populations is 0.025 mg liter\(^{-1}\) (Alabaster and Lloyd 1980).

High ammonia levels are associated with organic effluents, either as a result of anaerobic degradation or in sewage effluents (Hellawell 1986). Because ammonia is produced largely by deamination of nitrogen-containing organic compounds and by hydrolysis of urea (APHA 1985), it is present in high concentrations in sewage effluent. Ammonia loading (Figure 11) was exceptionally high at Cemetery as a direct result of the KSTP effluent discharge to the creek immediately above this site. During July and August, ammonia concentrations were 1.567 and 1.068 mg/l at Cemetery. Using the table of Trussell (1972) which is very similar to that of Emerson et al. (1975) at a pH of 8.19 and 23.27°C, the percentage of un-ionized ammonia there in July would be approximately 7.23 or 0.113 mg/l. This concentration is above that found to be acutely toxic to many species. Concentrations at Cemetery during other sample dates were within a more tolerable range (0.006 mg/l to 0.019 mg/l). As a result of nitrification, ammonia concentrations had been reduced to much lower concentrations at the downstream site. At Snow Line un-ionized ammonia ranged from .001 - .008 mg/l.

Ammonia concentrations at Big Horn (mean value 80.0 ug/l) were higher
Figure 11.

Ammonia Load
Ashley Creek, 1990

Ammonia mg/m³ second

Big Horn  Meridian  Cemetery  Snow Line

- Apr-26
- May-22
- Jun-19
- Jul-16
- Aug-16
- Sep-17

KSTP
than at Derns (mean value 14.0 ug/l) for every sample date except during the high flow in May (Table 4). It is most likely that the higher concentrations were caused from anaerobic decay in the large marsh on the eastern portion of Smith Lake and possibly from runoff and from the livestock in the pasture adjacent to this sample site. The surrounding area has been denuded by a high concentration of livestock. The livestock were frequently observed in the creek. They have trampled the stream bank, and removed most of the vegetation in the area.

**NITRITE/NITRATE**

It is generally believed that nitrite is unstable in aquatic environments because it is quickly oxidized to nitrate. For this study nitrate and nitrite were analyzed as one fraction. But nitrite is more toxic than ammonia and has been found to be quite stable in low temperature, low pH environments (Lee and Jones 1986). Nitrite/nitrate concentrations in Ashley Creek samples ranged from 0.0012 mg/l at Big Horn to 1.926 mg/l at Snow Line, both during September (Figure 12). The criteria for nitrite is .05 mg/l for cold water aquatic life (MDHES-WQB 1988c). In this study, it was not known how much of the total nitrite/nitrate was nitrite and if it exceeded the criteria. The combined nitrite/nitrate was much higher than other creeks and rivers within the Flathead Basin (Table 5) and may be adding a substantial load to the Flathead River.
In July and particularly August and September, there was a massive increase in nitrite/nitrate concentrations at Meridian (Table 4). Between Derns and Meridian the creek travels about 4,828 meters (3 miles) through small ranchettes, a pond, and an old lumber mill yard. Also above Meridian is a large cement holding pond for Kalispell storm drains. It was constructed mainly for flood control and at present is not being used. The lumber yard is privately owned and was not accessible for inspection although it has been reported by a number of sources that sawdust piles and debris lie within the creek and in some places line the banks. If the lumber mill is allowing their wood waste to enter the creek or the pond, the microbial breakdown could depress oxygen concentrations and release N. The increased organic matter could even alter the biological community. A reduction in dissolved oxygen did not coincide with the nitrite/nitrate increase which would have been expected if fresh organic waste were elevating N levels. But high O_2 concentrations in the spring water and photosynthesis in the pond could have masked a small decline in O_2.

There is evidence of nitrification in Ashley Creek below the KSTP. In the approximate 2,736 meters between Cemetery and Snow Line there was an 83% reduction in ammonia. There was a concurrent 60% increase in nitrite/nitrate (Figure 13) between these two sites - largely as a result of nitrification (\( \text{NH}_4^+ + 2O \rightarrow \text{NO}_3^- + \text{H}_2\text{O} + 2\text{H}^+ \)). Some input of
Figure 12.

Nitrite/Nitrate Load
Ashley Creek, 1990

Nitrite/Nitrate mg/m³ second

Big Horn  Meridian  Cemetery  Snow Line

Apr-26  May-22  Jun-19  Jul-16  Aug-16  Sep-17
ammonia from anaerobic conditions in the substrate, and subsequent oxidation of ammonia (nitrification), would be expected considering the morphology of the creek downstream of the study area. Even though this input would not be anywhere near the magnitude of the input from the KSTP, it may be substantial. Considering the already high level of nitrite/nitrate, the toxicity of these compounds and nutrient loading to Flathead River/Lake, contributions from the substrate could be important.

**SOLUBLE INORGANIC NITROGEN**

Watson et al. (1990) found diatom community standing crop in the Clark Fork River increased with increasing N up to 200 - 250 ug/l (250 ppb) soluble inorganic nitrogen (NH$_3$ + NO$_2$ + NO$_3$) as long as other factors were not limiting. If the same is true for Ashley Creek, then the KSTP may stimulate higher algal standing crops in the creek. However, lower light levels due to turbidity probably prevent the creek from realizing this potential for growth. All sites above the KSTP had soluble inorganic nitrogen (SIN) concentrations below 250 ug/l (except Meridian in September) while at the sites below KSTP the concentrations were well above 250 ug/l (Table 4). During low flow, there was more contrast in SIN concentrations between the upper and mid creek; high concentrations in the mid creek are likely caused by nonpoint sources of N and possibly nitrogen fixation.
Figure 13. NITRIFICATION
ASHLEY CREEK, 1990

NITRITE/NITRATE  AMMONIA

April

May

June

July

August

September

hp/liter

Airport Cemetery Snow Line

Airport Cemetery Snow Line

Airport Cemetery Snow Line

Airport Cemetery Snow Line
DENITRIFICATION

About 95% of the nitrate nitrogen that disappears from surface waters is transformed into gas; the remainder is immobilized in the sediment (Van Kessel 1977b). Nitrate is lost from stream sediments and overlying water by facultative anaerobic bacteria activity. This denitrification in the sediment and sediment-water interface is a significant sink of nitrogen. It occurs at low redox potentials with the resulting N₂O and N₂ gases escaping from the system (van Kessel 1977b; Hill 1979; Kaushik et al. 1981). Research on the input-output balance of nitrogen (Sain et al. 1977; Van Kessel 1977a; Hill 1979; Kaushik et al. 1981; Hill 1983) has shown that denitrification rates are dependent on variables including season, ratio of water volume and wet bottom area, retention time, temperature, dissolved oxygen concentration in the overlying water, sediment depth, sediment type, diffusion rate of nitrate into the sediment and nitrate concentration in the overlying water. Hill (1979) also found that disturbance of the river bed sediment begins to occur at discharges greater than 2.83 m³ s⁻¹ (99.9 cfs) and under those conditions oxygen was better circulated and denitrification was inhibited. Chatarpaul et al. (1979 and 1980) found the presence of tubificid worms in sediments increased both nitrification and denitrification rates.

Van Kessel (1977a) reported that, during a 19-day period, 56% of the nitrate from sewage effluents (having a 1.7 day retention time) discharged
into canals disappeared over a 800 m long stretch. In Ashley Creek, TPN was lower at Snow Line than it was just below the KSTP, during April, June and July.

**TOTAL PERSULFATE NITROGEN**

Total persulfate nitrogen decreased from Cemetery to Snow Line (Table 4) as a result of denitrification, plant uptake and other sinks or transformations within the creek. Total persulfate nitrogen (TPN) at Snow Line averaged 1606.7 ug/l during this study. This is 12 times as high as the mean TPN concentration in the Flathead River (Holt) even after Ashley creek has emptied into it (Table 5). It is also much higher than other waterways in the drainage (Table 5). Even at the most upstream site, TPN is substantially higher than all of the other UMFLBS monitoring sites within the drainage. This implies that wetlands and nonpoint sources in the upper and middle creek maintain fairly high N levels which the KSTP pushes higher still. Nitrogen export from a watershed has been shown to be significantly correlated to land use (Hill 1978).

During May and September there was an increase rather than a decrease in TPN from Cemetery to Snow Line (Figure 14). In May, possible sources are the application of fertilizer to the agricultural land within that reach and the storage of anhydrous ammonia adjacent to the creek at the Cenex facility. The September net increase between these sites may be partially
due to the fall release of N by primary producers as they became senescent or changes in microbial activity within the creek.

**NITROGEN TO PHOSPHORUS RATIO**

The N:P ratio in lakes, on average, decreases from more than 100 in oligotrophic waters to less than 10 in eutrophic sites (OEDC 1982). The N:P ratios during this study were lowest in the upper creek and considerably higher below the KSTP, ranging from averages of 4 at Derns to 23 at Cemetery (disregarding one aberrant ammonia reading in April). Sewage effluent contains much lower N:P ratios, 2 to 5:1 by weight, than are normally found in natural waters, 15:1 by weight (USEPA 1979). The ratio in Ashley Creek downstream of the KSTP may be higher than that of other sewage effluents because of the phosphorus removal by the KSTP. In the 1988 Flathead Basin Commission Biennial Report, Stanford reported ratios of 10 and < 4 near the mouth of Flathead River (Holt) and near the headwaters of the North Fork of the Flathead River (border), respectively. The N:P ratio in Ashley Creek below the KSTP is relevant because it was substantially higher than that in the Flathead River. In the Clark Fork, Watson (1988) found that the N:P ratio appeared to affect the relative abundance of *Cladophora* and diatoms. The species of dominant aquatic vegetation did vary among study sites on Ashley Creek although any correlations were beyond the scope of this study.
Figure 14.

Total Persulfate Nitrogen Load
Ashley Creek, 1990
ORGANIC CARBON

Carbon is both the initial and end product of organic metabolism and one of the best parameters by which to evaluate productivity and/or organic loading in a system. The organic carbon of natural waters consists of dissolved organic carbon (DOC) and non-dissolved organic carbon (NDOC) (Wetzel 1983). Metabolism of the biota creates a series of reversible fluxes between DOC and NDOC (Wetzel 1983).

Although there are many sources, most of the organic enrichment of waterways comes from soil organic matter (Gjessing 1976) and treated domestic sewage. In addition to KSTP effluent one of the important sources on Ashley Creek is the input of dead plant matter from allochthonous and autochthonous sources. Changes in habitat (e.g., marshes, pasture, deciduous shrubs and trees), watershed disturbance (Meyer and Tate 1983) season of the year, groundwater composition and flux, and residence time all influence the composition and amounts of organic and inorganic carbon (and other elements) in a stream during downstream transport.

Depletion of dissolved oxygen is the principal effect of the discharge of organic matter, as it is the result of heterotrophic utilization by microorganisms (Hellawell 1986). Other possible ecological effects of organic waste loading are the yellowish-brown coloring of the water (Gjessing 1976), toxicity (ammonia and sulfur) caused by anaerobic
metabolism, modification of the substrate by deposition of organic sludges and changes in the biological community composition caused by the addition of nutritive material which may favor some organisms (Hellawell 1986). Some treated effluents may not begin to decompose immediately because of a corollary toxicity that is suppressing microbiological activity.

The world average concentration of DOC in rivers is 5.8 mg l\(^{-1}\) although the range in natural waters is 1 to 30 mg l\(^{-1}\) (Wetzel 1983). For temperate zones the median is 3 mg/l (Meybeck 1982). In Ashley Creek DOC was high: It ranged from a mean of 15.5 mg/l in April to a mean of 8.34 mg/l in September (Table 4). Concentrations were usually highest in the upper creek as a result of snow melt and runoff from the organic soils and wetlands in the upper drainage (Figure 15). The reverse was true with NDOC: Concentrations were highest in the lower creek, as a result of input from the KSTP (Figure 16). During high flow in April, NDOC concentrations reached a maximum of 3.95 mg/l immediately below the KSTP (Cemetery). Both DOC and NDOC move with water, but NDOC will settle to the bottom of static waters. Organic matter accumulated below the KSTP during low flow and was resuspended during high flow in the spring. During this study, the minimum NDOC concentration, 0.92 mg/l, was immediately above the KSTP (Airport). It corresponded with the lowest flow in September (Figure 16).
Figure 15.

Dissolved Organic Carbon
Ashley Creek, 1990

DOC ug/liter

Big Horn  Derns  Meridian  Airport  Cemetery

Apr-26  May-22  Jun-19  Jul-16  Aug-16  Sep-17
Figure 16.

Nondissolved Organic Carbon
Ashley Creek, 1990

![Graph showing nondissolved organic carbon levels at various locations along Ashley Creek in 1990, with specific data points for each month from April to September. The locations include Big Horn, Derns, Meridian, Airport, Cemetery, and Snow Line.](image-url)
CARBON/NITROGEN RATIO

Allochthonous organic matter has a C:N ratio from 45:1 to 50:1 while the ratio of autochthonous organic matter is much lower (Wetzel 1983). Above the KSTP, values ranged from 36.6 (April) to 11.8 (September) with average monthly means ranging from 31.9 (April) to 15.9 (September). Below the KSTP (Cemetery) values ranged from 17.0 in June to 5.1 in July, indicating a higher concentration of nitrogenous material in the effluent. In Ashley Creek the C:N ratios were highest during high flow. Spring runoff loads more organic matter from the watershed into the creek.

DISSOLVED OXYGEN

The emergent vegetation present in the wetlands surrounding Smith Lake generate enormous amounts of decaying organic matter. The oxygen demand of the detritus is the obvious cause of the low dissolved oxygen (DO) concentrations in Ashley Creek at the Big Horn sample site. Dissolved oxygen was lowest at that site (Table 3), on the sample dates April through July. This site is classified by the State as B-2 (MDHES-WQB 1988a) and under this classification "DO concentration must not be reduced below 7.0 milligrams per liter from October 1 through June 1, nor below 6.0 milligrams per liter from June 2 through September 30". During the six months of this study, DO concentration at this site was in
violation every month except September. However, concentrations were
substantially higher at the subsequent sample site, a riffle community.

The EPA water quality criterion for ambient dissolved oxygen
concentration requires an instantaneous concentration of 4.0 ppm for cold
water for life stages other than early life stages (early life stages criterion
is 7 ppm). Levels below 4 ppm are considered to produce "severe
production impairment" for salmonid species and "acute mortality limits"
for invertebrates. A 3 ppm concentration is considered the necessary
"limit to avoid acute mortality" for salmonids. The State standard for
waters classified C-2 is 7.0 milligrams per liter from October 1 through
June 1 and 6.0 milligrams per liter from June 2 through September 30.
Although Ashley Creek is classified as C-2 below the bridge on Airport
Road the State has made an exception to the standard for that portion of
Ashley Creek. The Administrative Rules of Montana (1988) state that in
Ashley Creek, below the bridge crossing on Airport Road, "the dissolved
oxygen concentrations may not be reduced below 5 mg/l from October 1
through June 1, nor below 3 mg/l from June 2 through September 30".

During the warmer low-flow months of August and September, DO was
lowest below the KSTP, dropping to 3.27 ppm (11:44 am) on August 17
and 4.82 ppm (11:19 am) on September 26. This was a drop from 7.6
ppm and 8.9 ppm (respectively) at the sample site immediately above the
discharge (Figure 17). It is very likely that there were additional violations
Figure 17.

Dissolved Oxygen, Ashley Creek, 1990
See Table 3 for Sample Times

![Graph showing dissolved oxygen levels at various locations along Ashley Creek in 1990. The graph includes data for different months and locations, with EPA criteria and state standards indicated.](image-url)
on August 17, and September 26 earlier in the morning. During the
diurnal oxygen study on August 29, DO dropped to 2.0 ppm, and
remained there for one and one half hours (6:34 am - 8:12 am).
Dissolved oxygen was measured on only two dates in August and one
date in September, but if they are representative of the entire month there
were many violations during that time.

Under the KSTP discharge permit, DO is measured between 6 am and 9
am on a weekly basis. During May, DO concentration had begun to
increase by 6 am; in September ascent began about 8 am (Figure 18), so
it is likely that KSTP personnel are aware of violations through their
monitoring.

On two occasions dissolved oxygen was monitored in the KSTP effluent,
and at Airport and Cemetery, over a 24 hour period. On both occasions
the oxygen curves were similar in amplitude at both sites but
concentrations were lower at the downstream site (Figure 17). The 24
hour average on August 29-30 was 3.78 ppm and during certain periods it
was below 3 ppm, the State standard. In August average DO
concentration was 4.81 ppm lower below the KSTP than it was above it.
August was a period of low flow and higher temperatures and the KSTP
effluent constituted a larger percentage of the total flow. One of the
problems related to sewage inputs is the stress put on the receiving
waters because of a high content of reduced nitrogen compounds (Connell
Figure 18.  DIURNAL OXYGEN STUDY
Ashley Creek, 1990

<table>
<thead>
<tr>
<th>Hour (Military Time)</th>
<th>DO Concentration MAY 27-28, 1990</th>
<th>DO Concentration Aug 29-30, 1990</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>10</td>
<td>10</td>
</tr>
<tr>
<td>4</td>
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<td>18</td>
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<td>20</td>
<td>20</td>
<td>20</td>
</tr>
<tr>
<td>24</td>
<td>22</td>
<td>22</td>
</tr>
</tbody>
</table>

- **UPSTREAM**
- **DOWNSTREAM**
- **EPA COLDWATER CRITERIA**
- **STATE STANDARD FOR DO BELOW KSTP C-2 CLASSIFICATION**
- **STATE STANDARD FOR DO ABOVE KSTP B-2 CLASSIFICATION**
Overall lower concentrations of DO below the plant are an indication of nitrogenous and carbonaceous BOD exerted by the effluent. The overall nitrification reaction requires two moles of oxygen for the oxidation of one mole of NH$_4^+$ +. The oxygen demand of nitrogenous matter is about 4.57 times by weight of ammonia nitrogen, 1.14 times by weight of nitrite nitrogen, and 2.67 times by weight of carbonaceous matter (Bansal 1977). When the curves are the same shape above and below, it signals that the influx of nutrients by KSTP did not result in increased instream plant standing crop. This may indicate that nutrients were not limiting at the site above the KSTP. It may also indicate that nutrients were limiting above the plant, but something else (such as light) was even more limiting below the KSTP.

When DO falls to such low levels as it did in August (2 ppm), hydrogen sulfide, ammonia and other toxic substances are released from the substrate. They inhibit plant growth and cause a decrease in benthic invertebrate diversity because of unnatural environmental conditions (Connell & Miller 1984). Ammonia concentrations were higher immediately below Smith Lake during the months June through September (Table 4), than they were at the other sites (excluding below KSTP). Anoxic conditions in the wetlands and in the creek probably caused the increase in ammonia concentration.

Normally, sediments serve as a sink for phosphorus (Lee et al. 1978),
but at low redox potentials (a condition resulting from oxygen concentrations below 2 ppm (2 mg l⁻¹) phosphorus that has adsorbed onto ferric hydroxide and other oxides will dissolve and be released back into the water (Connell & Miller 1984). During July and September, total phosphorus increased from Cemetery to Snow Line. There was no obvious source of phosphorus in that area so possibly the phosphorus was released from the sediments. In the Amazon River Gessner (1960) found that when soluble phosphorus concentrations exceeded 10 µg/l, it was sorbed to finely divided inorganic suspended matter but when the soluble concentrations were less than 10 µg/l, the sorbed phosphorus was released from the sediments.

The rate of re-oxygenation depends on such variable factors as rate of flow, turbulence and depth of the water, the amount of photosynthesis, temperature and, of course, reactions reducing oxygen (Hynes 1963). Below KSTP, the morphology of Ashley Creek is not conducive to re-oxygenation. The creek is slow and deep. The main oxygen supplement comes from photosynthesis, which is patchy.

RESPIRATION STUDY

PHOTOSYNTHESIS AND COMMUNITY RESPIRATION

For polluted streams community respiration is a function of temperature and the concentration of organic matter in the water and on the bottom,
whereas primary production is a function of light and nutrients (Odum 1956; McIntire et al. 1964) and current (Pfeifer et al. 1975). Turbulence influences the amount of nutrients, light, excretion products and CO₂ which conceivably change primary production levels.

In this study the rate of gross photosynthesis (GP) was estimated by adding the net oxygen evolved [increase in DO in the light chambers (see methods, respiration study)] during the period of study to the amount of oxygen consumed by "respiration" (decrease in DO in the dark chambers) during an equivalent period (McIntire et al. 1964; Minshall et al. 1983). A loss of DO, in the chambers, was interpreted as increased biological decay of organic material. Comparison of decreases in DO at each study site, was intended to evaluate the contribution of organic loading in that area.

Plant growth and the production of DO will increase until some parameter within the creek becomes limiting. If nutrients are the limiting factor (which is usually the case), increases in DO may be used to compare nutrient loading between the various sites.

In July, GP at the upper site (Big Horn) was equivalent to that immediately below the KSTP (Table 2). This would lead one to believe that nutrients are not limiting in the upper creek at that time (i.e., the addition of more nutrients did not cause an increase in primary productivity). But underwater light readings were substantially lower at Cemetery and Meridian than at the other sites. Since primary productivity
is a function of light, light may have been limiting at Cemetery and causal for a GP similar to the upstream site. Otherwise, DO generally increased along with solar insolation (Figures 19 and 20).

During July, GP and nutrient concentrations were lower at the three sites between Big Horn and Cemetery, and correlation to GP for all six sites was .723 for SRP, .917 for NO$_2$/NO$_3$ and .754 for NH$_3$ (Brain Power, Stat Vu II - matrix correlation). Gross photosynthesis was also correlated to pH (.852) and to substrate (.719). Sites were ranked from 1 to 10 according to the amount of silt and sand vs. rock present in the substrate (100% silt and sand was ranked 10).

By September, however, conditions in the creek had changed. GP increased at each consecutive downstream site (excluding Snow Line), and doubled below the KSTP (Table 2). Unlike July, when GP was higher at Big Horn, GP was lower there in September. The reason was not determined but possibly flow and/or the presence of humic matter could have been contributing factors. When disturbed, the muddy edges of the creek emitted strong odors that may have been partially due to the emission of hydrogen sulfide during decomposition of organic matter.

Jackson and Hecky (1980) found that primary productivity was depressed by the presence of humic matter in the Boreal Forest Zone. In September correlation was good between GP and substrate (.743), NH$_3$ (.881), NO$_2$/NO$_3$ (.795), SRP (.816), TP (.761), pH (.894) and chl a (.945).
Figure 19. JULY RESPIRATION STUDY
CHANGES IN DISSOLVED OXYGEN vs. SOLAR INSOLATION

<table>
<thead>
<tr>
<th>LIGHT CHAMBERS</th>
<th>DARK CHAMBERS</th>
<th>INSOLATION</th>
</tr>
</thead>
</table>

- **BIG HORN**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA

- **AIRPORT**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA

- **DERRNS**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA

- **CEMETERY**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA

- **MERIDIAN**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA

- **SNOW LINE**
  - Hour (Military Time)
  - PPM DO/LTR vs. QUANTA
Figure 20.
SEPTEMBER RESPIRATION STUDY
CHANGES IN DISSOLVED OXYGEN vs. SOLAR INSOLATION

LIGHT CHAMBERS  DARK CHAMBERS  INSOLATION

BIG HORN

AIRPORT

DERNS

CEMETERY

MERIDIAN

SNOW LINE
Many parameters increased downstream along with GP and it is difficult to determine from this study which ones are causal and which ones are coincident.

Respiration, as measured by decreases in DO concentration in dark chambers, was extremely low at Snow Line in September and extremely high at that site in July causing suspect GP values for both months. It is plausible that the very great decline in DO in the dark chambers at Snow Line during July was influenced by the oxidation of sediments. It was inevitable that some sediments were stuck on the exterior of the PVC cores, some sediments were smeared on the bottom of the chambers as the cores were put in the chambers and some sediments oozed out the bottom of the cores as the scaffolding and other chambers were manipulated.

Low oxygen demand and oxygen output in September at Snow Line was probably attributable to the paucity of colonization on the clay tiles by periphyton and by disturbance of the fragile colony during assembly of the chambers. After incubation of the tiles in the creek from August 9 to September 9 in 14-16 inches of water, a thin layer of very fine sediment was observed but periphyton was not conspicuous. The unstable - shifting silt layer on the tiles was not conducive to plant growth. Perhaps if the tiles had been in deeper water, lying on the bottom, or in faster current there would have been greater oxygen demand due to a more
stable surface (better colonization) or more organic matter settling on the tiles. [Respiration rates have been found to be a function of particle size (Naiman and Sedell 1979)].

In July respiration decreased at each successive downstream site beginning at Big Horn but increased at Airport and then doubled at each site below the KSTP. The decrease at the two sites below Big Horn show processing and/or output of the organic matter that entered the system from the wetland areas upstream. The increase at Airport is most likely due to an influx of organic matter from urbanization near the creek as it flows through the southwest edge of Kalispell. At Meridian minimum respiration was observed. The creek substrate is nearly all fine sediment, velocity is slow there and oxygen consumption was expected to be higher. The lower oxygen consumption may be a result of inhibition of biological processing caused by toxins. There are a few possible sources: treated timbers in the railroad bridge at the site, petroleum products and/or other harmful materials entering the creek from the holding pond for Kalispell City storm drains, Kalispell Wood Products or even individual irrigation pumps.

In September respiration at the upper three sites was similar. There was less input from the upper wetlands because of low flow. So it seems that either a constant influx of organic matter downstream or organic matter detained within the creek from higher flows maintained the respiration
rate. Non-point sources including agriculture (sediments and nutrients) and rural home sites (sediments, nutrients and septic tank drain fields) contribute allochthonous material in that stretch of the creek. At the fourth site, DO in the dark chamber actually increased for a period before it descended. Respiration then escalated below KSTP. The large increase below KSTP may be controlled by current because the difference was not as extreme during July when the current was stronger and there would have been more oxygenation, more dilution and less deposition within the creek. During spring runoff more mineral and organic materials are transported into the creek. The faster moving water is better oxygenated and can carry a heavier suspended load. During low flow, the watershed has less effect on the chemical and biochemical characteristics within the creek.

P/R RATIOS

At the time of the respiration studies, all six sites were autotrophic. The high photosynthesis (P/R) ratios, (Table 2) ranging 1-6, are only indicative of those two time periods when the respiration study was conducted (peak hours of solar radiation and temperature) and are certainly not an indication of what occurs in Ashley Creek over the entire day or year. Data collected during this study were intended to be used to compare sites within Ashley Creek and not to compare Ashley Creek to other
systems. Ratios were highest at the middle two sites during July. They were lower (because respiration was higher) below Smith Lake and, of course, below the KSTP (Table 2). In September respiration was higher at all sites, causing lower P/R ratios.

During July only large rocks (>6 cm) were used in the respiration chambers at the upper 5 sites; but Derns is the only site where the substrate is dominated by large rocks (approximately 80%). At Big Horn, Meridian and Airport about 25%, 10% and 70% (respectively) of the substrate is composed of large rocks. At the lower two sites it is difficult to find any rocks. This study is biased toward what occurs in those micro-environments and is a comparison of those areas rather than a holistic study of the creek.

During the September study artificial substrates were used at Cemetery and Snow Line. The clay tiles supplied an ideal substrate for periphyton growth at Cemetery and plant growth on the tiles was obviously greater than that in the creek. The current was strong enough to free the tiles of silt and provide a stable surface for attachment. Light intensity was also higher since the tiles were incubated at a depth of about 20 inches, significantly closer to incoming light than the bottom sediments. At Snow Line, however, probably because of less current, the tiles were covered with an unstable layer of sediment and plant growth was negligible.
COMPLICATIONS

Some minor complications may have affected the results of the respiration study. When the study was started in the mornings there was, on occasion, a lag period before changes in DO began to occur, particularly in the dark chambers. When a lag period was obvious, that portion of data was not included in the results. Black plastic was used to shade the dark chambers. Because of the port holes on each side of the chambers connecting the pumps and the current, it was difficult to assure total darkness in these chambers at all times.

Although the DO meter was not moved and always kept in the shade and/or covered with reflective material, there were some suspicious readings. During July, the meter was not calibrated before each reading because the instructions and other users stated that it was not necessary. But later it was discovered that performance was improved by frequent calibration. During the September study the meter was recalibrated before each set of readings, usually every 15-20 minutes.

At Snow Line sediments continually settled on the chamber tops and had to be removed so as not to interfere with light penetration. The clay tiles that were incubated in the creek at Snow Line had such a tenuous layer of fine silt that it was difficult to transfer them from the creek to the chambers without disturbing them.

When measuring surface area, it was often difficult to determine exactly
how much of the rock surface was covered with photosynthetically active matter. The portion that was different in color or where periphyton was observed was measured because it seemed more accurate than other reported methods (foil, 25%, etc.).

The Cemetery site was a bend in the creek that had rip-rap to stabilize the stream bank. This site was chosen out of necessity because the creek below the KSTP is a deep channel and otherwise impossible to work in. That site and substrate were not typical of the creek and a variety of circumstances could have affected the results there. The rocks provided a more shallow (greater light intensity) and more stable surface for plant growth, but the abundant duck weed (Lemna spp.), which is common in the lower creek, could have inhibited attached plant growth.

CHLOROPHYLL a

Chlorophyll a data was collected to use in analyzing photosynthesis and respiration data. During July, after the rocks were removed from the chambers, they were picked clean of all invertebrates and then scraped and brushed to remove chl a. Exposure to light causes the breakdown of chl a and so an alternative method was devised for September. A correlation matrix showed no correlation of chl a and any other parameter in July. In September, however, chl a showed correlation to GP ($r^2 = .945$), NH$_3$ ($r^2 = .832$), SRP ($r^2 = .748$), pH ($r^2 = .732$), NO$_2$/NO$_3$ ($r^2 =$
Physiological and environmental factors influence reactions by various species of algae and, therefore, primary production (Bolin et al. 1977). Chlorophyll can have varying photosynthetic activity, depending on age, availability of nutrients, water temperature and season (Hall and Moll 1975). Growth phases (lag, log, stationary or senescent phase) are different for various algal species (Bolin et al. 1977). In undisturbed boreal streams Naiman (1983) found that < 42% of the variation in O₂ production and consumption (excluding macrophytes) was attributed to variations (g/m²) in chlorophyll.

Current velocity affects algal productivity, accumulation, the community structure and also the particular species (Traaen and Lindstrom 1983). Adaptation to light intensity also occurs; at minimal light intensities there is more chlorophyll in cells than at high light intensities (Bolin et al. 1977).

ASH FREE DRY WEIGHT

Aliquots of the same material used for analysis of chlorophyll a were also analyzed for ash free dry weight (AFDW). Ash free dry weight (in g/m²) was low at Big Horn and Cemetery. It was expected to be high at those sites because of the influx of organic matter from the upper marsh and KSTP. On both occasions AFDW was high at Airport. These results were
puzzling, but chances of getting representative rocks were slight especially
where rocks are not the dominant substrate. Again, the data was
collected to use in photosynthesis and respiration analysis.

DARK/LIGHT BOTTLES

In July the differences (t-test) in DO concentrations in the light and the
dark bottles were found to be significant ($p < .01$) at all sites except
Cemetery. The increase in oxygen in the light bottles indicated that
primary productivity exceeded respiration in the water column at that
time. Although these data agree with the P/R study results, there is a
question as to the method of DO analysis during the July study. The
probe (which lacked a stir bar) was inserted into the bottles and the
bottles were carefully and gently inverted and swirled. The reason these
data are questionable is that the pressure sensor on the probe was not
under water. (Air pressure is not as great as water pressure and it is not
certain that the probe compensated for the difference.) If the data are
correct, the lack of a significant difference between $O_2$ concentrations in
the light and dark bottles at Cemetery is not understood. A few possible
explanations are: 1) decomposition is so great that it masks $O_2$
production; 2) light is limiting at that site as was suspected from the
respiration chamber results; 3) a toxin(s) inhibited the growth of the
suspended flora; or 4) during high flow there is a lag period after the
effluent enters the creek, before the flora population reaches its former density.

In September, only Big Horn and Cemetery showed significant (P < .05) differences between the light and dark bottles. Big Horn lies directly below Smith Lake so phytoplankton may be spilling from the lake and causing primary productivity to be higher in the water column at that site. The high concentration of nitrogen and phosphorus in the KSTP effluent are probably the cause of higher primary productivity in the water column at Cemetery. The large difference in results between the July and September studies may be due to temporal changes, i.e., water temperature, volume of flow, life cycle of the phytoplankton or to the method of analyzing O₂ concentration.

MACROINVERTEBRATES

The deviation in distribution and abundance of macroinvertebrates (Appendix E) between the study sites on Ashley Creek was due to site specific environmental conditions. Environmental factors such as particle size of the substrate (Cummins and Lauff 1969), texture of the substrate, total surface area, degree of compaction, current velocity, season, and amount of detritus may act to regulate the species composition and abundance of macroinvertebrates (Ulfstrand 1967; Hynes 1969; Hynes 1970; Cummins 1974b; Minshall and Minshall 1977; Rabeni and Minshall
These factors varied between sample sites on Ashley Creek, thus, macroinvertebrate populations varied. Patchiness is also a result of disturbances, colonizers, colonist sources and species interactions (Townsend 1989). Levels of pollutants and anoxic environments also regulate species composition (Hynes 1969; Connell and Miller 1984; Hellawell 1986; Hellawell 1988). However, because the velocity and channel geomorphology were so different at the various sample sites in this study, it was impossible to conclude whether those factors or other water quality parameters were causal for the presence or absence of particular macroinvertebrate species.

The invertebrate study was biased because only one habitat type (rock in most cases) was investigated at each site. Invertebrates were collected from rocks at the upper 4 sites (and Cemetery in June), although rock was not the dominate substrate at all sites. The species that were dominate in this study are not necessarily the dominate species within the creek.

Higher concentrations per unit area were observed in September than were observed in July (Table 6). This is mainly due to the method of sampling. In July the rocks used in the respiration chambers were also used to collect the invertebrates. Flushing of the chambers removed some of the invertebrates, resulting in a low tally for July. In September, however, separate rocks were used for the two studies and a much higher level of accuracy (and higher total invertebrates) resulted. There are also other important factors that
govern total numbers and diversity (number of families represented) of macroinvertebrates. The abundance of each species varies in time according to its life history stages (Cummins 1962; Williams and Hynes 1973) and natural distribution patterns may be extremely divergent at various times of the year (Cummins 1962).

Another consideration in examining the invertebrate data, is the bias caused by the type of substrate from which the invertebrates were collected. The suspended wood frame and clay tiles used at Cemetery to collect invertebrates from, in September, attracted different types of macroinvertebrates (Crustacea: Isopoda) than did the sediment cores used at Snow Line (Oligochaeta: Haplotaxida, and Insecta: Diptera). Although these sites were similar in many ways (i.e., flow, depth, substrate) the taxa collected were very dissimilar (Appendix E) owing to the two different habitat types sampled.
Table 6. Macroinvertebrates sampled at six sites, in July and September, on Ashley Creek, 1990.

<table>
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<tr>
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<th>Total Families</th>
<th>Organisms/ *m²</th>
<th>September Total Organisms</th>
<th>Total Families</th>
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<td>1221</td>
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*Substrate cores were taken at Snow Line and figures are in cm³.
*Cores were 7.3 cm diameter and in July 15.2 cm deep, in Sept 6.4 cm deep.
**July Cemetery samples were unavailable.
CONCLUSIONS AND RECOMMENDATIONS

1. This study exposed many changes in water quality in Ashley Creek as a result of both natural and anthropogenic loading. Anthropogenic changes were a result of point sources or land use (some parameters increased downstream along with urbanization). Some natural changes were temporal (e.g. temperature) others were influenced by seasonal flow.

The upper site was highly impacted by the natural loading of organic material from the small lakes and marshy areas in the upper drainage. At that site, the mean total persulfate nitrogen (TPN) concentration was 2 - 4 times higher than other impacted creeks in the Flathead drainage and 5 times higher than the Flathead River (Holt) after its confluence with Ashley Creek. The mean ammonia concentration was 3 - 12 times higher than other impacted creeks and 8 times higher than the Flathead River (Holt). Dissolved oxygen (DO) concentration at that site violated State standards on 5 of the 6 sampling dates. Nutrient loading was higher during higher seasonal flows, whereas DO concentrations were more critical during higher temperatures and lower flows.

At the two middle sample sites, there were unexpected findings in the creek. The State criterion (MDHES-WQB 1988c) for pH was surpassed in July, when it exceeded a pH of 8.5 just above the KSTP. An influx of nitrite/nitrate, particularly during low flow, was observed at the sample site just upstream of Kalispell. And, during the September community
respiration study, respiration was lower than expected at that site, indicating the possible presence of a toxic substance(s) in the creek.

The Kalispell Sewage Treatment Plant (KSTP) effluent had substantial affects on Ashley Creek. There was a 300% increase (from the above site) in TPN concentration and a 350% increase in soluble reactive phosphorus (SRP) concentration. Nitrogenous and carbonaceous biochemical oxygen demand (BOD) exerted by the effluent depressed DO concentrations to levels below the State standard for extended periods of time.

Below the KSTP, before its confluence with the Flathead River, Ashley Creek was 12 times higher in TPN, 26 times higher in SRP concentrations and two times higher in specific conductance as the Flathead River below the point where Ashley Creek empties into it.

Surprisingly the KSTP does benefit the creek in two ways:

1) it acted as a fish barrier (G. Anderson pers. comm.) to keep non-game fish from migrating to the upper creek; and 2) it lowered the pH of the creek when it had exceeded the State criterion above the effluent in July.

2. The oxygen demand, measured as a decrease in DO (respiration) in the dark chambers during the community respiration study, varied from July to September. In July respiration rates doubled below the KSTP and in September it was about six times greater than the site above. Increased
oxygen demand indicated that microbial activity was positively influenced by the KSTP effluent. Photosynthesis rates nearly doubled, during both studies, from the site above the KSTP to the site below it. Biologically available nutrients present in the effluent stimulated plant growth.

The data collected from the community respiration study must be evaluated carefully because it is difficult to differentiate the effects of KSTP and of fluvial morphology. Naiman (1983) stated that level and diversity of metabolism in lotic ecosystems are largely functions of channel geomorphology and hydrology. The morphology of Ashley Creek is extremely different above and below the KSTP and from site to site. Many of the major chemical and biological differences within the creek are due to the creek's inherent morphological characteristics and not just a direct result of the influx of organic matter. Also, the material used in the respiration chambers was not identical to the creek's substrate. What occurred in the chambers was not considered characteristic of the creek.

3. The degradation of Ashley Creek is a result of many events, but there are a number of things that could be done to improve the quality of the water. Point sources must not be allowed to have a significant impact on the creek. Best management practices (BMP's) that would reduce or eliminate nonpoint sources of pollution from reaching the creek should be a priority of the Agricultural Stabilization and Conservation Service and the Soil Conservation Service and the Montana Department of Health and
Environmental Sciences Water Quality Bureau and possibly the Montana Environmental Quality Council. These agencies must be given enough power to enforce the BMP's.

Following are a number of specific things that should be done to improve the water quality of Ashley Creek.

Kalispell City must be required to obtain a permit to discharge their storm drain holding ponds into Ashley Creek. Under the permit, regularly scheduled water sampling must be required to ensure that they are not degrading Ashley Creek.

The grounds around Kalispell Wood products should be investigated because of the increase in nitrogen and what appeared to be a decrease in decomposition (microbial activity) at the sample site immediately below that area. If the lumber mill is allowing waste to reach the creek, they are in violation of the Montana anti-degradation policy and the Clean Water Act and should be cited and required to mitigate the problem and use BMP's.

The holding pond gates for Kalispell City storm drains are currently open. This holding pond could be used to treat the storm drain runoff. The storm drain effluent should be skimmed and receive primary treatment before being discharged into the creek.

Another issue is the condition of some of the riparian zones along the creek. Denuding and trampling of the stream banks by livestock, was a
common problem in all sections of the creek. It diminishes water quality and raises water temperatures. Reclamation of the creek banks and fencing a buffer zone to minimize the impact of farming and livestock are needed.

Another study, now that the new KSTP is in operation, should evaluate the impact the new KSTP is making on Ashley Creek. If current water quality criteria and/or standards cannot be met, then land treatment should be considered as an alternative. Holding ponds and wetlands that are properly managed are able to reduce nutrient loads in water. They could be located on property already owned by the City.

The public needs to be educated. Most people do not understand the effects of urbanization on water quality. Education could be done through the media, in schools and through existing agencies such as the Conservation Districts and the Montana Environmental Quality Council.

Hynes (1963) reminds us that even the most satisfactory effluent is not river water and, therefore, produces some alterations to the environment of the river flora and fauna.
Eutrophication is a biological process generated by an abundance of readily available nutrients, the main two being nitrogen and phosphorus (Gloyna and Eckenfelder 1968; Krenkel and Novotny 1980; Hellawell 1986). Actually, any necessary nutrient may be limiting in an ecosystem, but in Flathead Lake both phosphorus and nitrogen are limiting (Dodds and Priscu 1988; Spencer 1988). Eutrophication often interferes with the beneficial uses of a water body. It may cause significant economic loss including reduced recreational and navigational uses and may also render the water less suitable for human or animal consumption. The eutrophication process affects not only algae and other aquatic plants, but also the composition and distribution of the entire aquatic biota (Cummins 1974a; Lee and Hoadley 1966; Wuhrmann 1968; Connell and Miller 1984, Krenkel and Novotny 1980) and reduces stability of the system (Wetzel 1983). The sources of phosphorus and nitrogen problems of greatest concern are domestic wastewater and biological treatment plants (O'Connor 1967; Lee et al. 1978).

Aside from nutrients, other important variables affecting lake trophic status include: light intensity and penetration, temperature, mixing, CO$_2$ content, grazing by predators, average depth, stream velocity, chemistry of water and concentration and types of organisms (Krenkel and Novotny
1980). The rate of eutrophication is influenced by the growth of algae and benthic macrophytes, herbivory and other food web processes such as influx of organic matter and decay and decomposition of organic matter by bacteria. The end result, deterioration of water quality and other symptomatic changes are undesirable and interfere with water uses (Connell and Miller 1984).

Eutrophication is not as well understood in lotic systems as it is in lentic systems (Krenkel and Novotny 1980; Hellawell 1986) because down-gradient movement complicates the nutrient cycling. Spiralling is the interdependent process of cycling and downstream transport of the nutrient within the stream or river (Seki et al. 1980; Vannote et al. 1980; Newbold et al. 1982; Elwood et al. 1983; Newbold et al. 1983; Minshall et al. 1983). Although the nutrients do not usually remain in one area they may still pass through the same trophic level or chemical state many times, providing nourishment to the biota in the system over and over again. Nutrients may be generated upon oxidation of organic matter (O’Conner 1967) - the normal fate of municipal wastes such as KSTP effluent. The breakdown of larger organic particles into smaller more bioavailable forms and the down gradient movement may cause a significant growth of green plants downstream (O’Connor 1967).

Natural systems are able to assimilate and process certain amounts of wastes. The waste assimilation potential of a creek is controlled by the
self-purification process. This process is unique for each creek or river and even for different stretches of the same system. When the concentration of the waste gets so high that it has deleterious effects on the biota of the system and/or interferes with the beneficial uses of the system, the assimilative capacity has been exceeded and remediation is needed.

**APPENDIX B POLLUTION PROCESSING**

The three major types of waste are chemical, bacterial and organic (Velz 1970). All three of these pollutant types are present in Ashley Creek (Dutton 1987). Stable chemical wastes are processed mainly by dilution. Bacteria, such as those found in sewage, are annihilated by exposure to a hostile environment. The most common waste - unstable organic waste (excluding refractory compounds) is processed by a chain of biological processes.

The biological processes are a function of temperature, time and dilution, with more oxygen being consumed at higher ambient temperatures (Velz 1970). In aerobic respiration microorganisms require oxygen as an electron acceptor in order to metabolize organic substrates as sources of energy. The number of bacteria found in the aquatic environment is dependent primarily on the amount of organic matter available to them (Jannasch 1958). Consequently dissolved oxygen (DO) levels are greatly
affected by organic loading to aquatic ecosystems (O'Conner 1967; Hargrave 1969; Zanoni 1969; Fillos and Molof 1972; Krenkel and Novotny 1980; Seki 1982). In areas of organic enrichment, consumption of oxygen during the biochemical processes may cause DO levels to become so low that other life forms are endangered. Another danger associated with depressed DO concentrations are changes in the redox potential which may release nutrients (mainly phosphorus) & toxins (mainly sulfur & ammonia) from the sediments to the overlying water. A major discrepancy could arise from assuming oxygen levels were directly related to the amount of respiration and photosynthesis occurring if some other parameter such as a toxic material were controlling or affecting the viability of the system.

There are other factors that affect oxygen concentrations. Physical characteristics include current velocity (Traaen and Lindstrom 1983; Gloyna and Eckenfelder 1968), temperature (O'Connor 1967; Hargrave 1969; Zanoni 1969) and stream morphology. Additional oxygen sinks include respiration of macroscopic and microscopic plants (O'Connor 1967) and oxidation of inorganic suspensions and sediments (Edwards and Rolley 1965; Hargrave 1969; Wetzel 1983). They may exert an indirect effect on the DO by mixing and by influencing the response of organisms which contribute to the DO flux.

The oxygen consumed during these processes is replenished from the
atmosphere at a rate that is controlled by physical and climatic parameters and from inflow of water and primary productivity (O’Conner 1967; Manny and Wetzel 1973). If reoxygenation is slow and the loading is large the purification process will take longer and there may well be deleterious effects to other biota within the system.

Dissolved oxygen (DO) concentration has been a water quality indicator for many decades. The Royal Commission on Sewage Disposal in Britain published nine reports from 1901 to 1915 (Hellawell 1986) including in its eighth report proposed standards based on the biochemical oxygen demand (BOD) test and the measurement of suspended solids.

**APPENDIX C NUTRIENT LOADING**

There are numerous suspected sources of natural and anthropogenic loading of nutrients and organic matter as well as other impacts on Ashley Creek. Natural influences include watershed climate and geology, e.g. location-specific lithology and geomorphology, riparian conditions, tributaries and lakes and their associated wetlands (Hynes 1975; Minshall et al. 1983; Minshall 1988). Likely sources on Ashley Creek include: Smith, Lone and Monroe Lakes, Truman Creek, the Kalispell Sewage Treatment Plant (KSTP) effluent, irrigation, a lumber mill, a meat packing plant, livestock grazing and farming, Kalispell City storm sewers and low water flows (Dutton 1987). Logging within the drainage may also be
influencing the water quality in Ashley Creek. Hauer and Blum (1991) have shown that timber harvesting has affected water quality in other parts of the Flathead drainage. Currently, it is not known to what extent each of these factors contributes to the degradation of Ashley Creek but the KSTP is assumed to be the major problem on the creek. Problems associated with its effluent include excess nutrients, suspended solids, increased temperature, dissolved oxygen depletion and fecal coliforms (MDHES-WQB 1986). Residual chlorine may also be a pollutant; KSTP chlorinates their effluent May 1 through September 30. At certain times of the year, the 1.3 million gallon per day discharge may comprise up to 20% of the total flow of the creek (L. Jenkins pers. comm.).

APPENDIX D MACROINVERTEBRATE LIST

Macroinvertebrates collected in July and in September from six sites on Ashley Creek: B = Big Horn, D = Derns, M = Meridian, A = Airport, C = Cemetery, S = Snow Line. Cemetery data for July (X) is not available.

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